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Aspects of the sustainability of creel fishing for Norway lobster, *Nephrops norvegicus* (L.), on the west coast of Scotland

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Summary

This thesis describes a study of a number of aspects of the *Nephrops* creel fishery. Various aspects have been studied. In particular, work has focused on how the biology of *Nephrops* affects its catchability in creels, as well as assessing the environmental impacts of the *Nephrops* creel fishery. A smaller study relating to the physiological effects of the capture and transport of live creel-caught *Nephrops*, has also been undertaken.

Fishery-directed sampling of *Nephrops* displayed strong seasonal trends in catch composition and catch rate, reflecting patterns in the *Nephrops* moult and reproductive cycles. Most females were caught in the early spring, following moulting and mating, with catch rates becoming low after spawning in autumn. Catches of males were greatest around February following moulting. Of those animals caught, males were larger than females. These seasonal patterns reflect the exploitation of different components of the stock throughout the year.

Fishery sampling also showed there to be considerable stock heterogeneity both among and within creel fisheries. Global positioning system (GPS)-linked data loggers fitted to selected fishing boats provided a more detailed assessment of spatial variability in catch rate. Results showed both effort and landings per unit effort (LPUE) to be patchily distributed in the area studied, further illustrating considerable stock heterogeneity throughout the area. Both approaches illustrated that catch rate varied with sediment type, with high catch rates of small animals on sandy to firm mud and high catch rates of large animals on softer mud. Additionally, population modelling indicated that, in the areas studied, *Nephrops* are not currently being growth overfished.

Behavioural observations in both the field and laboratory indicated a low probability of entry to creels by approaching *Nephrops*, particularly if other crustacean species were inside the creel. Larger *Nephrops* appeared to have a greater threshold for disturbances such as and the presence of conspecifics or other species as well as reduced gear avoidance, suggesting that *Nephrops* creels select larger animals in the population.

Using sea pens as an indicator of physical disturbance of the sea bed showed that impacts from creeling were low. The sea pens *Virgularia mirabilis* and *Pennatula phosphorea* were caught in moderate quantities in the creel fishery, but were still observed in high densities on creel grounds. Comparison with trawled areas showed some differences in density, the most notable being that the tall sea pen *Funiculina quadrangularis* was less abundant on trawled grounds, indicating an adverse effect of bottom-towed gear.

Bycatch was assessed in two distinct fisheries and varied both between and within areas; however, levels of bycatch (both target and non-target species) in all areas were much lower than previous estimates from the trawl fishery. Additionally, in contrast to the trawl fishery, commercially exploited fish made up a small proportion of the bycatch in the *Nephrops* creel fisheries studied. The survival of *Nephrops* discards is considered to be high, although preliminary results suggest that predation by seabirds may cause significant mortality.

The effects of 'ghost fishing' were investigated experimentally at two sites and it seemed that lost creels do not constitute a serious issue in this fishery. Creels were found to be very selective for their target species, with both target and non-target species being able to escape. Only a few crabs and a small proportion of the captured *Nephrops* died in the creels, indicating no evidence of a 're-baiting' effect from dead animals.

Stress associated with the capture, storage and live transport of creel-caught *Nephrops* was found to be greatest during the transport process, during which *Nephrops* are kept out of water for long periods of time. Temperature was particularly important, suggesting cooling prior to and during transport was essential in order to ensure their survival. Reduced salinity was found to be a major source of stress.

An assessment of the management measures implemented in the Torridon creel fishery show that escape gaps placed in creels are effective in enabling smaller *Nephrops* and other bycatch species to escape. Limiting effort of individual fishermen has been successful, but this has been followed by an increase in the number of boats fishing within the area, leading to an overall increase in effort. Management measures (both statutory and voluntary) currently applied in the Torridon area are discussed and considered in relation to other areas. It is concluded that creel fishing areas need to be considered on a case-by-case basis, accounting for biological characteristics of individual stocks, as well as management issues at a local scale.

The findings of the present study are discussed in the context of providing a detailed description of the characteristics of the *Nephrops* creel fishery on the west coast of

Scotland. Findings are also discussed in relation to other crustacean creel fisheries, enabling issues directly relevant to the specific fishery to be considered in a much broader context. The present study has been particularly valuable in a wider context, as *Nephrops* are found in a habitat which is very different from the majority of other commercially exploited crustaceans, and their life history characteristics are also different. This study provides an ideal platform to assess what general principles from other creel fisheries apply to the *Nephrops* creel fishery, as well as what characteristics are unique to this fishery.

Candidate's Declaration

I declare that the work recorded in this thesis is entirely my own, unless otherwise stated and that it is of my own composition. No part of this work has been submitted for any other degree.

J. Adey

Jonathan Adey

Date 19th March 2007

This thesis I dedicated to the memory of my mum who would have been very proud to see this thesis collecting dust on her shelf. The many achievements in my life including this thesis would not have been possible without her unquestioning support and the drive and determination she instilled in me to follow my dreams and pursue my goals.

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Contents

Chapter 1

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1 GENERAL INTRODUCTION

1.1 Overview

The Norway lobster *Nephrops norvegicus* (hereafter referred to by genus alone) is the most important commercial crustacean in the UK. The greatest proportion of *Nephrops* is landed in Scotland (22,663 t, worth £56.6M in 2004). *Nephrops* are caught in both trawl and creel (baited creel) fisheries. The bulk of the catch is taken by trawlers that catch the animals during periods when they are foraging away from their burrows on the seabed. Creeling accounts for around 7.5% of landings in Scotland by weight, but 20% by value. On the Scottish west coast, creel landings have been increasing since 2000 and constituted around 17% by weight (21% in the North Minch) and 39% by value of the total landings in 2004 (Figure 1.1). This increase in creel landings on the west coast has been accompanied by a decrease in the landings from the trawl fishery (9382 t in 2000 compared with 8235 t in 2004).

Fishing pressure (all methods) is regarded as sustainable at current levels, with the three main west coast stocks (North Minch, South Minch and Firth of Clyde) showing an overall increase in abundance since the 1990s (Keltz & Robb, 2006). However, since *Nephrops* trawlers take a bycatch of white fish, this sector of the industry is currently limited by regulations aimed at conserving whitefish stocks. One expectation of this is an expected move from trawling to creeling, changing the traditional fishing pressures on stocks. With improved infrastructure for live export and comparatively high prices for creel-caught animals, this fishery is increasing in importance. Fisheries managers now consider improved understanding of the *Nephrops* creel fishery to be an important issue within Scottish fisheries. This is particularly pressing as catch rates have declined in recent years in a creel fishery east of the Outer Hebrides, with concern that the stock is being overexploited in this area (Keltz & Robb, 2006). Despite the increasing importance of the fishery, and the fact that there have been *Nephrops* creel fisheries for 30 years or more in some areas, there is a paucity of detailed scientific information on almost all aspects of the *Nephrops* creel fishery.

Aspects of *Nephrops* biology and behaviour may offer them some protection from overexploitation by trawl fisheries. Firstly, the ovigerous females remain largely within their burrows during the long egg incubation period (*ca* 9 mo at this latitude), during which time they rely on stored metabolic reserves supplemented by food acquisition within their burrows or at burrow openings. Secondly, the juveniles remain within their burrows for most of the time until they become sexually mature (Chapman, 1980). Thus the most vulnerable components of the stock are afforded protection from trawling by their behaviour. However, the situation with creeling is not as straightforward. Ovigerous females are enticed from their burrows by creel bait and may comprise a significant proportion of the catch (Watson & Bryson, 2003).

It is known that *Nephrops* creels tend to select the larger animals in the population (Bjordal, 1983; Tuck & Bailey, 2000; Eggert & Ulmestrand, 2000; Bell *et al.*, 2006), but the reasons for this are not fully understood. Much more needs to be known about creel selectivity and the field behaviour of the target organism. Creels are thought to have little impact on the environment. Although this impact is likely to be small, there has been no comprehensive scientific analysis of *Nephrops* creel bycatch, or ghost fishing by lost *Nephrops* creels. Little work has been undertaken to date on the effects of creel arrays on the benthic fauna (sea pens have received some attention: Eno *et al.*, 2001). Such data will help to test the intuitively reasonable hypothesis that creels have a low environmental impact.

1.2 Crustacean creel fisheries

There have been a small number of previous studies investigating aspects of *Nephrops* creel fisheries and a large number on other crustacean species. From these studies some general principles relating to creel fishing are apparent. In particular, catches appear to vary with the moult and reproductive cycle of the target organism (Miller, 1990), but little is known about spatial variation in catch. Environmental impacts are assumed to be low, although this has not been adequately tested, with only ghost fishing thought to be a major problem for the majority of creel fisheries (Jennings & Kaiser, 1998). Furthermore, a number of crustacean creels have been shown to selectively sample their target population (Jury *et al.*, 2001).

As this project is the first of its kind to investigate *Nephrops* creel fisheries, each of these general principles will be investigated in detail; for this reason a detailed review of the literature covering each of these aspects will not be included here, but will be introduced separately in each relevant chapter. To this end, the spatial and temporal variations in creel catch will be reviewed in Chapters 2 and 3 and creel-related behaviour in Chapter 4. The environmental impacts of crustacean creel fisheries will be considered in Chapter 5 with reference to physical impacts, Chapter 6 will address bycatch and Chapter 7, ghost fishing. Finally, the physiological effects of capture, storage and transport of crustaceans will be reviewed in Chapter 8.

1.3 The Nephrops *fishery*

Nephrops is commercially exploited throughout its geographic range, with two main fishing methods employed, trawling and creeling. Trawling is the predominant method, particularly in offshore areas. Creeling usually occurs in areas that are difficult to trawl, owing to, for example, bathymetry, rocky outcrops and sea room, but also takes place in some trawl areas at times when trawling is prohibited. Creeling is important in some inshore areas around Scotland and Sweden, as well as being the only fishing method employed in the Faeroese *Nephrops* fishery.

1.3.1 Trawl fishery

Commercial fishing for *Nephrops* began in the 1950s using specially designed '*Nephrops* trawls' towed by vessels of 10–25m in length. Prior to this, *Nephrops* were discarded as bycatch in a number of European trawl and seine fisheries (Bell *et al.*, 2006). Trawl nets are constructed from single twine synthetic cord, have a mesh size of 70 mm or 80 mm (depending on area and gear type) and use metal otter boards to keep the net open. The ground rope is weighted with either metal rings or rubber discs and a 'bobbin rig' (with larger rubber discs or metal spheres) if fished on rough ground (Howard, 1989). Nets are towed for periods of 2–8 hours, depending on seasonal variations in catch. Trawled animals tend to be smaller than creel-caught animals, on average, and thus command lower prices (particularly when only the abdomens are sold as 'tails'), but are caught in much larger numbers. Trawlers may also catch a large quantity of non-target species, including white fish and other invertebrates (Alverson *et al.*, 1997). The amount of bycatch (all species) from *Nephrops* trawlers in the Clyde Sea Area has been estimated as an average of 9 kg per 1 kg of *Nephrops* landed (Bergmann *et al.*, 2002a).

1.3.2 Creel fishery

Fishing for *Nephrops* using baited creels developed alongside the trawl fishery, to exploit *Nephrops* populations that were not accessible to trawl vessels or in areas where legislation banned trawling. Creel vessels are generally smaller than trawlers (7–12 m in length, most are <10 m), are crewed by one to three men and fish between 400 and 1000 creels per day (Howard, 1989). A typical *Nephrops* creel is made from a steel frame, covered with 25–30-mm mesh netting. Where the mesh is stretched over the frame, it is often protected from abrasion by a binding of polypropylene rope. Each creel has two 75-mm diameter 'hard' eyes, a vertical bait string and a hinged door at one end fastened by a hook on a rubber strip. In total, a typical creel measures approximately 550 × 420 × 300 mm (Figure 1.3).

Creels are usually baited with salted herring or mackerel and fished for a minimum of 24 h. An estimated 1kg of bait is used per kg of *Nephrops* landed (Ziegler, 2006). Creels are often fished in fleets of 50–125, and are attached at intervals of approximately 20 m along a buoyant 'back rope' by 1.8-m long 'strops' (Kinnear *et al.*, 1996). Each end of the back rope is attached directly to buoyed 'tailings', which mark the position of the fleet (Howard, 1989). The majority of creel vessels land large, whole *Nephrops* that are exported live and command a high price in foreign markets. For this reason, care is taken at sea to maintain the catch in good condition, by placing animals in individual 'tubes' (elongate, square- cross-section subdivisions of a box formed by vertical partitions intersecting at right angles: different-sized animals are placed in boxes with 'tubes' of appropriate size).

Following capture most creel caught *Nephrops* are exported to European markets, during which time there are subjected to prolonged periods of emersion out of the water. Processes will vary between exporters but in general the transport process is as follows:

- Following capture *Nephrops* may be exposed on deck (in tubes) for up to an hour.
- They are then placed in seawater tanks on the vessel for up to 10 hours.
- On landing *Nephrops* are either held in chilled seawater tanks on land or in the open sea overnight (12 hours).
- *Nephrops* are then repacked into tubes in polystyrene boxes, which are chilled with ice (ice is not in direct contact with animals).
- Boxes of *Nephrops* are then transported dry, first by chilled van and then by aeroplane to European markets. This transport process takes around 12 hours.

In addition to creels catching larger animals of higher value, there are several advantages of the creel fishery over trawling. Fuel consumption is significantly lower, discard mortality is thought to be almost zero and the negative external effects on the environment are less, when compared with trawling (Eggert & Ulmestrand, 2000; Ziegler, 2006).

1.3.3 Gear conflict

Although some *Nephrops* creel and trawl fisheries are separated by bathymetry, this is not always the case, and where the two fishing methods are used on the same grounds, the potential arises for gear conflicts. In Scotland, this conflict intensified in 1984, when the Inshore Fishing (Scotland) Act 1984 removed the ban on mobile gear within the three-mile limit, opening inshore areas to trawling.

Gear conflict is a growing problem, as competition for resources between sectors increases. This increase in competition is due to more vessels fishing with creels, expanding creel fishing grounds into areas traditionally fished only by trawlers. Financial pressures from a number of sources, in particular increasing fuel costs, can also lead to both sides encroaching on each other's 'boundaries'. A negative side effect of competition is that creeled areas are for the most part constantly occupied by creels, rather than the ground being fished in rotation, as was the traditional practice. This occurs because if creels are removed from an area, trawlers or other creel vessels tend to move in.

Levels of conflict between sectors vary throughout Scotland. In several areas including around Loch Broom and the Summer Isles, there is, in general, a good relationship between mobile and static gear fishermen, whereby creel and trawl areas are designated informally by general consent (Anon, 2005a). This is not the case in areas such as the Clyde, where conflict between fishermen has become a pressing issue. In some areas, such as St Abbs and Eyemouth, moves from mobile to static gear have been attempted but conflict between sectors has led to attempts being abandoned (Fortune, 2005). One solution is the division of areas between the different sectors, which has been shown to be very effective in the Torridon area.

1.3.4 Fishery management

Current exploitation of *Nephrops* is regulated by three main measures under the European Union (EU) Common Fisheries Policy (CFP); minimum landing size (MLS), total allowable catch (TAC) and minimum mesh size (MMS) (Briggs, 1997). Details vary among different fishing grounds. For the ICES Division VIa (west coast of

Scotland), the MLS is 20 mm carapace length (CL) and in 2006 the TAC is 17,257 tonnes. Currently, EU regulations set limits on days at sea, related to mesh size. Technical measures also include square mesh panels in gear. In the UK, quota is allocated to individual vessels through the Producer Organisation (PO) system, in which fishermen pay a percentage of their earnings to their PO for a monthly quota allocation. On the west coast of Scotland, the quota is set at a flat rate per vessel and varies from 4 to 8 tonnes per month (Livingstone, 2001). These regulations are, however, often of little relevance to the creel fishery, as animals as small as 20 mm carapace length are rarely caught and never landed. Catch rates are currently not high enough to be limited by the quota allocation (Livingstone, 2001).

There has been widespread criticism of the CFP since its implementation, with a view that the decision makers were distant from the actual circumstances of the fishery, leading to reactive management. Furthermore, the Council of Ministers, who are advised by the European Commission (EC), set TACs for large ICES sea-areas, whereas the ICES Advisory Committee on Fisheries Management has repeatedly advised that *Nephrops* stocks should be managed on a smaller spatial scale. Establishing a network of inshore fisheries groups around the Scottish coastline may go some way to addressing these problems. These groups will be made up of fishermen and other stakeholders, with expert advice being provided from outside sources. They will be responsible for developing objectives for local fisheries management and formulating appropriate local management plans (Scottish Executive, 2005).

1.3.5 The Loch Torridon *Nephrops* Fishery

The Loch Torridon *Nephrops* creel fishery is of particular interest, owing to the conservation status of the loch, the enlightened attitude of local fishers and the statutory spatial management measures adopted. These factors make this fishery an ideal test bed for management measures, which may in the future, be applied elsewhere.

Loch Torridon is a fjordic sea loch in Wester Ross, Scotland, and contains a high diversity of intertidal and subtidal habitats, supporting a wide range of marine communities of plants and animals. Consequently, the Torridon marine area is considered to be nationally important for marine nature conservation (McHattie, 1999). The loch is not only ecologically important, it is also economically important to the local communities. Torridon is a remote area, with some of the lowest population densities in the EU and the community relies heavily on fishing and tourism.

In the past, fisheries in this area were much more diverse, with crayfish (*Palinurus elephas*), lobster (*Homarus gammarus*), herring (*Clupea harengus*), cod (*Gadus morhua*) and scallops (*Pecten maximus* and *Aequipecten opercularis*) being major target species. However, by 20 years ago, the herring, whitefish and lobster stocks had collapsed, leaving *Nephrops* as the major commercial species (McHattie, 1999). *Nephrops* have been caught in Loch Torridon using baited creels for more than 30 years and due to the decline of other fisheries is now the most valuable fishery within the area. Towed underwater television (UWTV) surveys in Loch Torridon and the Inner Sound suggest that *Nephrops* stock abundance has increased in recent years and that densities are lower inside the loch than outside it (Hough, 2006).

The Torridon creel fishery is prosecuted within ICES Division VIa, which is further subdivided for stock assessment purposes into statistical rectangles. The Torridon creel fishery occupies statistical rectangles 44E4 (sub-divisions 7 & 8) and 43E4 (sub-division 1) (Figure 1.4).

The average weekly weight of *Nephrops* landed in the Torridon fishery is approximately 350 kg, with 100 to 150 t landed annually (Mason *et al.*, 2002). Approximately 50 t of herring is used annually for bait in the fishery. There are currently 13 vessels fishing within the area full time (see below). A further 17 people are employed in the packing station at Ardheslaig. A number of these people have established a co-operative (Shieldaig Export) (Figure 1.5), set up to process and ship locally-caught live *Nephrops* to European markets, where they command prices four to five times that of frozen *Nephrops*. The value of the product is further increased as small animals (35–39 mm CL) are not landed and animals with large amounts of encrusting organisms are rejected. Fishermen co-own Shieldaig Export and it is therefore in their interest to ensure the long-term sustainability of the fishery. Altogether it is estimated that 60% of the local workforce is involved in the Loch Torridon *Nephrops* creel fishery (BBC, 2003).

Although the *Nephrops* creel fishery is the major fishery within the area, the loss of the 3-mile limit in 1984 put the fishery under increased pressure, as it meant that trawlers could enter the Torridon area. Many static gear fishermen blamed trawlers for depleting stocks, owing to the effectiveness and intensity of trawling, and for causing damage to static gear and the seabed (Fulton, 1998). Following the lifting of the 3-mile limit, the Loch Torridon creel fishermen, through the Highland and Islands Fishermen's Association (HIFA), headed a campaign to establish control over the management of fishing in Loch Torridon, and in particular sought to have an area closed to mobile fishing gear.

In 2001, a zoned fisheries policy was introduced and an area including Loch Torridon and much of the Inner Sound was closed to all mobile fishing gear [The Inshore Fishing (Prohibition of Fishing and Fishing Methods) (Scotland) Amendment Order 2001]. Within the Inner Sound, there are also two testing areas for underwater weapons systems. The southernmost of these, the British Underwater Test and Evaluation Centre (BUTEC) incorporates inner and outer sea-areas. These areas were established in 1975 when all fishing was banned in the inner area and only static gear was allowed in the outer area. The inner area forms one of the largest closed areas in Europe (25 km^2). Other areas were set aside for trawling and a mixed fishery, and an area closed to trawlers seasonally (Figure 1.6). These fishing zones were designated on the understanding that the arrangements would be reviewed after 5 years and a review is currently underway.

In addition to the zoning arrangements, local fishers, under the auspices of the Torridon *Nephrops* Management Group (TNMG), have agreed a voluntary code of practice for fishing in the area closed to mobile gear. An important achievement of the TNMG was to attain accreditation by the Marine Stewardship Council (MSC) in 2003. The MSC provides independent international recognition for well-managed sustainable fisheries.

This accreditation was the first in Scotland and is currently only one of 21 worldwide, with 18 currently undergoing assessment (MSC, 2006). The main provisions of the voluntary code of practice are as follows:

1. Maximum of two sets of 800 *Nephrops* creels for vessels with two or more crew;
2. Maximum of two sets of 400 *Nephrops* creels for one-man vessels;
3. The maximum number of crab/lobster creels is two sets of 400 for two-man vessels, or two sets of 200 for one-man vessels;
4. Escape gaps/panels to be fitted to all prawn creels by end June 2003;
5. Maximum number of days fished (creels hauled) per year to be 200. Starting from 1st July 2003;
6. ALL berried prawns (ovigerous females) to be returned to the sea;
7. Only 2 sets of gear to be fished per vessel (i.e. two sets of prawn creels, or two sets of crab/lobster creels, or one set of prawn creels plus one set of crab/lobster creels);
8. Only one set of gear to be hauled each day.

Following the closure of the area to trawling in 2001, the number of creel vessels fishing full time had increased from 10 to 15. However, two vessels have recently been sold, resulting in 13 vessels fishing at present. In addition, since the zoning scheme was established, a number of vessels have begun fishing in the static gear area part time (TNMG, personal communication). Fishing effort by individual vessels has been reduced as a result of the voluntary management code, which 12 of the 13 full-time vessels have signed up to. The increase in vessels since 2001 has led to an overall increase in effort within the creel area, leading to concern for the long-term sustainability of creel fishing in the area (TNMG, personal communication).

The Loch Torridon *Nephrops* fishery has been the focus of several previous studies. The most comprehensive study to date was by Tuck & Bailey (2000), who assessed the state of the *Nephrops* stock. Three assessment methods were used: i) burrow counts from towed UWTV surveys in 1994–2000; ii) analysis of landings per unit effort data (LPUE) recorded by local fishermen since 1990; iii) analysis of length composition data from market sampling of landed creel catches (Tuck & Bailey, 2000). These three approaches each showed the stock to be in a reasonably healthy condition. Each of these analyses will be discussed further in relation to the current study in Chapter 2. Tuck & Bailey (2000) concluded that at the time of the study, the level of exploitation appeared to be sustainable and there was therefore no cause for concern about the state of the *Nephrops* stock. They did, however, advise against any increase in fishing effort.

As part of the certification assessment for MSC accreditation, another study into the Torridon *Nephrops* fishery was undertaken by Mason *et al.* (2002). As in the previous study, the security of the *Nephrops* stock was considered, but this report also focused on the fishing methods, discards and management practices. Mason *et al.* (2002) drew on a range of sources, including Tuck & Bailey (2000). In addition, they found that for every 3 kg of *Nephrops* caught, approximately 1 kg was discarded at sea. The average carapace length of *Nephrops* discards was 35.9 mm (males) and 34.9 mm (females), which is well above the minimum landing size (20 mm). Discarding in the creel fishery is very low compared with trawling and is indicative of the selective nature of creels. In relation to discarding of non-target species, Mason *et al.* (2002) found that of 115 creels examined, 69 caught no non-target species and no creel had more than two non-target species present. These findings led Mason *et al.* (2002) to conclude that the creel fishery had limited impact on species other than *Nephrops*.

As a part of the zoned management policy in the Torridon area, Fisheries Research Services (FRS) were tasked to monitor the *Nephrops* stock both within this area and in the surrounding areas using UWTV surveys and catch sampling (trawl and creel). The most recent report showed that up until 2005 there has been a general trend of increasing *Nephrops* abundance in the North Minch (trawl) and static gear zone (Tuck, 2005).

A survey of the seabed in the Inner Sound and BUTEC range was undertaken in 2005 using UWTV and acoustic methods (Atkinson *et al.*, 2006). Sediment throughout this area was found to be fairly uniform soft mud with little differentiation. Faunal surveys indicated that *Nephrops* occur at medium to low densities in the BUTEC range, but the animals present are large. The seapen *Funiculina quadrangularis*, and its associate brittlestar *Asteronyx loveni*, were also observed in high densities, with a significantly higher proportion of sea pens bearing brittlestars within the BUTEC range. The deep muds throughout the area were also characterised by the burrows of thalassinidean shrimps and several other megafaunal species (Atkinson *et al.*, 2006).

In addition to fishery studies of the Torridon fishery, several smaller-scale and more detailed studies of *Nephrops* biology and behaviour have been conducted in Loch Torridon (including, Rice & Chapman, 1971; Chapman *et al.*, 1972; Chapman *et al.*, 1975; Atkinson & Naylor, 1976; Chapman & Howard, 1979; Gaten *et al.*, 1990) and these will be considered in greater detail in the relevant sections.

Two projects have addressed the importance of the Torridon fishery to the local economy and communities (McHattie, 1999; Anon, 2000). Both studies highlighted the importance of the fishery to the communities as a whole.

From the above account it is clear that the Torridon *Nephrops* creel fishery has a number of special characteristics. Therefore, it was necessary to select some other west coast of Scotland sites in order to give an insight into differing practices within the *Nephrops* creel fishing sector.

1.4 Nephrops biology

Nephrops norvegicus belongs to the family Nephropidae, sub-family Nephropinae, and is the only species of *Nephrops* found in European waters. *Nephrops* are also known as the Dublin Bay Prawn, Prawns, Langoustine, Lagostino, Lagostin, scampi and Bokstavhummer. Owing to the importance of *Nephrops* as a commercial species, as well as its importance to soft sediment benthic communities (Chapman, 1980), there have been many studies carried out investigating a wide range of topics. These topics include morphometry, reproduction, size, age, moulting, physiology, feeding, behaviour, disease and fisheries. There have been several reviews of *Nephrops* biology and fisheries (de Figueiredo & Thomas, 1967; Farmer, 1975; Chapman, 1980; Howard, 1989; Sardà, 1995; Briggs, 1997; Bell *et al.*, 2006).

1.4.1 Morphology

Nephrops has an orange-red colour, long slender claws and prominent kidney-shaped eyes (Howard, 1989). The carapace covers the forepart of the body, while the posterior part (abdomen) is divided into six flexible segments. The sexes can be easily distinguished by examining the first pair of pleopods, which are ridged in males and feather like and flexible in females (Howard, 1989). Males show positive allometry of the chelipeds and females show positive allometry of abdomen width (Farmer, 1975).

1.4.2 Habitat

Nephrops is widely distributed on the continental shelf of Europe. Its geographical range extends from Morocco to Iceland, including the Mediterranean, in which they are found as far east as Egypt (Figueiredo & Thomas, 1967). Within this geographic range, their presence is dependent upon a seabed composed of fine cohesive mud in which they can construct burrows (Farmer, 1975; Howard, 1989). The burrows are usually quite simple, with two or more openings on the mud surface, and penetrate to a depth of about 300 mm (Rice and Chapman, 1971). They are found in depths ranging from 10 m to as deep as 800 m (Figueiredo & Thomas, 1967; Chapman & Howard, 1979). *Nephrops* spend most of their time in burrows, with burrow densities on commercial fishing grounds varying from 0.2 m^{-2} or fewer to several burrows m^{-2} (Chapman, 1980; Tuck *et al.*, 1997b). Many studies in British waters have recorded large variations in the density and size composition of *Nephrops* populations (O'Riordan, 1964; Hillis, 1972; Hillis, 1974; Chapman, 1979; Bailey & Chapman, 1983; Chapman & Howard, 1988; Briggs, 1995; Tully & Hillis, 1995; Tuck *et al.*, 1997; Sardà, 1998). One such example was highlighted by Bailey & Chapman (1983), who observed populations of small, densely aggregated *Nephrops* in the Sound of Jura. This is in contrast to observations in the northern part of the Clyde Sea Area, where they found larger individuals at lower population densities. This heterogeneous distribution of *Nephrops* populations has led ICES to classify the *Nephrops* stocks in Western Europe into 33 stocklets ('functional units'). These are not thought to be discrete stocks, but populations of individuals with similar biological characteristics, each occupying a distinct area.

The reasons for these variations seem to be unrelated to latitude, with burrow density and animal size related to sediment characteristics and hydrographic conditions (Chapman & Bailey, 1987). Around Scotland and Ireland, it has been found that areas of fine sediment are characterised by large *Nephrops* occurring in low densities; areas of coarser sediments are characterised by smaller *Nephrops* at higher densities (Howard, 1989; Tully & Hillis, 1995). This dependency on a particular sediment type has led to a discontinuous geographical distribution throughout the species range, with populations often being separated by large stretches of unsuitable sediment (Bell *et al.*, 2006). However, different relationships have been observed in other geographical regions and it appears that the major causal factor is hydrography, which affects the distribution of larvae and hence recruitment (Tuck *et al.*, 1994; Bailey *et al.*, 1995; Hill *et al.*, 1996).

1.4.3 Life History

Nephrops are dioecious and mating is promiscuous (Farmer, 1975). On average, female *Nephrops* mature at a carapace length of 21–23 mm (around 2½ years old) and males reach maturity at a carapace length of *ca* 25 mm (after 3 years), with these values varying geographically, reflecting differing growth rates. There is latitudinal variation in the timing of ovarian maturation, spawning and hatching, which appears to be related to environmental temperature (Sardà, 1995). During the summer months, the developing ovary in mature females can be seen through the carapace as a green area. Eggs are fertilised during the period from August to November, after which, eggs are spawned onto pleopods; then the ovigerous female is said to be 'berried'. In Scottish waters, the eggs are carried for around 9 months whilst they develop; this period varies with latitude, being shorter in more southerly latitudes and longer further north (Sardà, 1995). Females typically carry 2000 eggs, but possibly up to 5000, depending on their size (Sardà, 1995). There is progressive egg loss during incubation (Sardà, 1995).

As the embryos develop, there is a change in egg colour from dark green to brown prior to hatching. During this time, ovigerous females tend to spend the majority of their time within their burrows and they appear to be more catchable by creels than by trawling at this time, as they may be enticed out of their burrows by the baited creel (Watson & Bryson, 2003). Hatching in waters around Scotland and Ireland commences in late April and carries on until August (Howard, 1989). Spawning occurs either annually or biennially, with a higher prevalence of biennial spawning in higher latitudes due to the increased length of time required for the ovaries to mature (Sardà, 1995). Scottish populations contain both annual and biennial spawners (Bailey, 1984).

There are three free-swimming larval stages (zoeae), followed by the first postlarval stage, which resembles a miniature adult and explores the seabed prior to settlement in the late summer or autumn. Postlarvae / juveniles occupy burrows that are often connected to those of adults and they largely remain within their burrows for the first year of life (Chapman, 1980).

Growth in *Nephrops* is by moulting, the frequency of which varies with size, sex, season and latitude. Moulting is frequent in the first year. Sexually mature males moult twice a year until about 6 or 7 years of age, after which they moult annually. Sexually mature females generally moult only once a year throughout their life (Sardà, 1995). Growth increments at moult range between 1 and 2.5 mm CL (other values have been recorded in different populations), with evidence of a change in growth pattern at the onset of sexual maturity (Bailey & Chapman, 1983).

1.4.4 Predators

Nephrops burrow primarily to gain protection from predators, emerging to feed and engage in reproductive behaviour (Chapman, 1980). In Scottish waters, the main predators are fish, with *Nephrops* having been identified in the stomach contents of 13 fish species caught on Scottish *Nephrops* grounds (Thomas, 1965). Historically cod, *G. morhua*, were a major predator (Symonds & Elson, 1983). The curled octopus *Eledone cirrhosa* is occasionally found attacking *Nephrops* caught in creels (Howard, 1989).

1.4.5 Feeding

Nephrops are opportunistic predators and scavengers (Wieczorek *et al.*, 1999). The stomach contents of Scottish *Nephrops* were examined in detail by Thomas & Davidson (1962), who found that their diet was very varied, with polychaetes, crustaceans, molluscs and echinoderms forming the bulk of the prey, but *Nephrops* were also found to eat fish, formaniferans, coelenterates and algae. Prey can be obtained from within the burrow and adults also forage on the sediment surface, although it is thought that they do not venture far from their burrow entrance. It has also been suggested that *Nephrops* may feed on plankton filtered from water drawn through their burrows (Loo *et al.*, 1993). Feeding activity varies with season, the moult cycle and, in females, the reproductive cycle. In particular, ovigerious females appear largely to cease feeding and may go without food for up to a year, possibly supplementing their resources, by feeding opportunistically on plankton drawn into their burrows (Farmer, 1975), or from mieofana found within the burrow system (McIntyre, 1973). The diel timing of feeding is probably linked to the emergence rhythm.

1.4.6 Disease

The major disease of *Nephrops* in Scottish waters is an infection by a parasitic dinoflagellate, a species of *Hematadinium.* It was first noted in the Clyde Sea Area in the early 1980s and has since been found throughout the Scottish west coast, Fladen ground, Irish Sea and in Scandinavian waters (Stentiford *et al.*, 2001). The external characteristics of an infected animal are an orange colouration, milky white haemolymph and moribundity. *Hematadinium* infection has commonly been detected by examining removed pleopods for the presence of parasites under low power microscopy (Field & Appleton, 1995). The disease is seasonal in its occurrence, with peak prevalence during the winter and spring. The prevalence of this disease is now considered to be an important factor in not only the natural mortality of *Nephrops* populations, but also in fishing mortality, since it has been shown that infected animals spend more time outside their burrows and are less able to perform escape swimming, thereby making them more susceptible to trawling (Stentiford *et al*, 2000).

Another condition adversely affecting *Nephrops* in Scottish waters is idiopathic muscle necrosis. This is characterised by opaque lesions in the striated musculature of the abdomen. This condition, which is often compounded by bacteraemia, renders affected animals unmarketable and leads to mortality during transport (Stentiford & Neil, 2000; Ridgway *et al.*, 2006b).

1.4.7 Behaviour

Burrowing has been described as the "most characteristic feature in the behaviour of the Norway Lobster" (Dybern, 1965). Burrow construction behaviour is described in detail by Dybern (1965), Crnkovic (1968) and Rice & Chapman (1971). There is a large literature on the behaviour of *Nephrops*, much of which has focussed on the timing of emergence from burrows. Many of the early studies were prompted by the observation that trawl catches show marked diel and seasonal fluctuations (Farmer, 1974). The major method for studying the behaviour of *Nephrops* has been using trawls in conjunction with underwater television (Chapman *et al.*, 1972; Chapman *et al.*, 1975; Chapman & Howard, 1979; Newland & Chapman 1989; Newland *et al*, 1992; Burrows *et al.*, 2003). Other studies have involved direct diver observation, electronic tagging, and experimental studies in aquaria (Thomas, 1960; Dybern, 1965; Rice & Chapman, 1971; Chapman & Rice, 1971; Chapman *et al.*, 1972; Chapman *et al.*, 1975; Newland *et al.*, 1988a,b; Newland & Neil, 1990; Newland *et al.*, 1992; Richardson, 1996; Livingstone, 2001; Mouat, 2002; Aguzzi *et al.*, 2004).

Field observations show that the proportion of individuals emerging from their burrows varies markedly with time of day, season and location. A bathymetric trend in emergence behaviour has been observed, with the highest trawl catches in the shallowest waters of the continental shelf (<30 m) being observed at night; in deeper waters (~200m) peak catches were observed during the day and, at intermediate depths, around dawn and dusk (Aguzzi *et al.*, 2003; Chapman *et al*, 1972, 1975; Farmer, 1975; Atkinson & Naylor, 1976; Oakley, 1979; Mòller & Naylor, 1980). Explanations for diel changes in *Nephrops* catches have been suggested by Chapman *et al.* (1972) and Chapman *et al.* (1975), as a regulatory mechanism named the 'optimum light intensity concept'. Observations from several studies on emergence behaviour indicate that the activity of *Nephrops* is restricted to a narrow range of low light intensities. This hypothesis explains the differences in emergence patterns with depth, as the same optimal environmental illumination should be obtained at different hours of the day, at different depths.

A recent study by Aguzzi *et al.* (2003) observed that *Nephrops* in the Western Mediterranean showed different physiological adaptations to local photic conditions, implying the optimum light intensity may vary among different *Nephrops* populations. Photic responses that varied with depth were also evident in studies by Shelton *et al.* (1985) and Gaten *et al.* (1990), particularly in relation to the effect of light on the eye.

There is evidence that factors other than light may be involved in emergence, such as tides and nutritional state (Hillis, 1996; Chapman, 1980). It has been shown that endogenous factors also control emergence (Atkinson & Naylor, 1976; Hammond & Naylor, 1977; Aguzzi *et al.*, 2003). Food stimulus during the day is thought to be sufficient to override the normal inhibition by light and induce daytime activity in other species including *Jasus lalandei* (Fielder, 1965). This has also been suggested for *Nephrops* (Atkinson & Naylor, 1976) and this hypothesis was tested by Mòller & Naylor (1980) who showed that, in the laboratory, the endogenous nocturnal activity rhythm was modified by the presence of food. This has implications for the use of creels as a method of catching *Nephrops*.

There are several accounts of seasonal variations in the diel pattern of *Nephrops* emergence and sex composition of trawl catches. One reason for the variation in diel pattern may be changes in light level over the year. Seasonal changes in catches of females have been widely observed (Figueiredo & Thomas, 1967; Chapman *et al.*, 1972, 1975; Oakley, 1979). During winter months, trawls or creels in Scottish waters rarely catch females, presumably because the majority of sexually mature females remain in their burrows incubating their eggs and are largely unavailable to the fishery.

Ovigerous females may, however, emerge and be caught in creels if these are close to their burrows (see Chapter 2).

During the main emergence periods, the level of activity of individuals within a population can vary. Chapman & Howard (1979) noted that the amount of time an individual spent out of its burrow was proportional to its size. Chapman *et al.* (1975) used miniature ultrasonic transmitters to track individual *Nephrops* and found that animals do not utilise all the available time. Instead they restricted their activity to several short excursions from the burrow, with slightly longer excursions occurring around dusk and dawn. This evidence suggests that only part of the population is active at any one time.

Burrows *et al.* (2003) studied this activity using underwater video and observed that *Nephrops* movements were distinctly orientated toward or away from burrow openings, although the direction of travel away from burrows was variable. Newland *et al.* (1988a) observed that *Nephrops* orientated itself in the direction of the water current, generally walking downstream. Locomotor behaviour varied depending on current speed; animals avoid leaving their burrows when currents are strong.

1.5 Aims and Objectives

Prior to this study there was a paucity of information on the *Nephrops* creel fishery, with very little scientific analysis of the operation and sustainability of the fishery. For this reason there were two fundamental aspects requiring detailed assessment:

1. An investigation of factors determining the exploitation pattern of the *Nephrops* creel fishery.

Nephrops creels are thought to fish different components of the stock compared with trawls and, like the trawl fishery, catches are thought to vary both temporally and spatially. In order to provide a better understating of how creel fisheries exploit the stock (essential information for the management of a fishery), a broad approach to answering this fundamental question was necessary. A number of previous studies have shown that target species catchability will vary with a number of biotic and abiotic factors. This study aimed to determine which factors are the most important with regards to *Nephrops* creel catchability and, where possible, to provide a quantitative assessment of their effects on catch.

2. An evaluation of the environmental impacts of creel fishing for *Nephrops*.

Creels, when compared with towed gear, have often been considered to be environmentally benign, with ghost fishing thought to be the major source of environmental impact in creel fisheries. Reasons for this assumption include creel selectivity and assumed low physical disturbance of creels. With fisheries management moving towards a more ecosystem-based approach, the environmental impact (or lack of it) requires quantifying for all methods of fishing. To provide a better understanding of the environmental impacts associated with creel fishing the thesis aimed to determine the nature and severity of environmental impacts of the *Nephrops* creel fishery on the west coast of Scotland.

Following capture, the majority of creel-caught *Nephrops* are transported alive to European markets. In order to minimise stress, care is taken during all aspects of the transport process, with low temperatures thought to be particularly important in ensuring survival (Taylor *et al.*, 1997). In order to ensure the product goes to market in the best possible condition, a further important aspect of this work was to assess the physiological effects of capture, storage and transport of creel-caught *Nephrops*.

Furthermore, the statutory spatial management, as well as voluntary measures in the Torridon area, provided an ideal opportunity to assess the effects of management measures and their applicability to other creel fisheries.

The work reported in this thesis, provides detailed information on the *Nephrops* creel fishery, but has also enabled results to be compared with other creel fisheries, for different species. This has allowed general principles regarding creel fishing to be discussed, in addition to the first detailed assessment of some poorly understood aspects relating to creel fisheries in general.

1.6 Thesis structure

The general introduction (Chapter 1) is followed by seven chapters detailing the major studies (Chapters 2–8), and by a concluding chapter (Chapter 9). A summary of each of the seven studies can be found at the end of each chapter (Chapters 2–8).

The introduction to this thesis (Chapter 1) provides a description of *Nephrops* fisheries and *Nephrops* biology. A detailed review of relevant aspects of crustacean creel fisheries is not included in this introduction (Chapter 1); instead each aspect is reviewed separately in the relevant introductions of each chapter (Chapters 2–8). Since the same study sites are referred to in different chapters, descriptions of each site and accompanying maps have been included in this chapter (see below), with cross references from succeeding chapters where required.

The studies in chapters 2, 3 and 4 use different methods to investigate the factors determining the exploitation patterns of *Nephrops* creel fisheries. Chapter 2 focuses on the seasonal and spatial variations in *Nephrops* creel catch, using data collected from logbooks and direct fishery sampling. These data also enabled stock assessments to be carried out in the areas studied, which are also detailed in this chapter. Results are discussed in relation to how the moult and reproductive cycles of *Nephrops,* as well as differences in habitat and fishing pressure, affect the exploitation pattern. Chapter 3 describes the small-scale (microscale) distribution of fishing effort and landings from distinct fisheries, using data collected from either detailed logbooks or GPS position-recording units (PRU). This worked allowed factors effecting fishermen's behaviour and therefore distribution of effort to be investigated. Chapter 4 describes the use of underwater television to record *Nephrops* creel-related behaviour in the field and laboratory, enabling the assessment of how *Nephrops* behaviour affects catchability and therefore what components of a stock are most likely to be susceptible to creel fishing.

Chapters 5, 6 and 7 deal with environmental impacts associated with fishing for *Nephrops* with creels. Chapter 5 describes a study comparing the physical impact of *Nephrops* trawl and creel fishing, using towed underwater television surveys of the epibenthos. Chapter 6 describes an assessment of the bycatch of both target and non-target species associated with *Nephrops* creel fisheries, using direct sampling in two distinct creel fisheries. A preliminary investigation of bycatch survival is also described.

Chapter 7 focuses on the issue of ghost fishing by *Nephrops* creels, using two different methods: assessing catch and bycatch at regular intervals, by SCUBA diving in a shallow site and periodically hauling creels in a deep-water site. This study was carried out over a period of one year.

Lastly, a study of the physiological effects of capture, storage and transport of creel-caught *Nephrops* is detailed in Chapter 8. Experiments in the laboratory and field have provided valuable information with regards to best practice within the fishery.

Work undertaken in chapters 2–7 also allow for the efficacy of the management measures within the Torridon area and their applicability to other creel fisheries to be assessed. In particular, an assessment of technical measures (such as escape gaps) (Chapters 2, 4, 6, & 7) and measures to manage fishing effort (Chapters 2 & 3) was possible.

Finally, findings from all aspects of this study are drawn together to synthesise significant findings of the preceding chapters, suggest directions for future work, and consider best practice and possible fisheries management tools to ensure future sustainability of this expanding fishery (Chapter 9).

1.7 Study sites

1.7.1 Torridon area

Loch Torridon comprises three basins, Upper Loch Torridon, Loch Shieldaig and lower Loch Torrridon (the outer basin of Loch Torridon has been defined as lower Loch Torridon for the purpose of this study). lower Loch Torridon faces north-west and is exposed or moderately exposed (Figure 1.7). These loch sections are generally steep-sided with rock or boulder slopes leading to a more gently shelving bottom of sand or mud. The basins of the loch are up to 144 m deep, with the sills shallowing to depths of as little as 20 m, causing acceleration of tidal currents. Outside Loch Torridon, the Inner Sound lies between the Isle of Raasay and the Wester Ross mainland and is one of the deepest inshore sea areas in Britain, with depths of over 300 m in places. The sea bed in Loch Torridon and the Inner Sound supports dense populations of megafaunal burrowing species, including *Nephrops*, as well as a number of epifaunal species, many of which are susceptible to impacts by commercial fishing. In Loch Torridon, several areas have been identified as being of outstanding interest; sediment plains in Loch Shieldaig and Upper Loch Torridon, Upper Loch Torridon narrows, and the Ob Mheallaidh lagoon and rapids complex. The zoned fisheries policy in this area (Chapter 1; Section 1.3.5) means that only creel vessels fish the locations from which the catch and bycatch was sampled in the present study (Figure 1.8). The site chosen for a study of 'ghost fishing' was in the more exposed lower loch, at a depth of 60 m below chart datum, close to the harbour of Ardheslaig (57° 32.97'N, 5° 43.16'W) (Figure 1.9).

Loch Gairloch is west facing, has no sills and is exposed over much of its length. It is located just north of Loch Torridon (Figure 1.7). Depths of over 100 m can be found at the mouth, with sediments throughout the loch being predominantly sandy and soft mud. The deeper water contains large assemblages of sea pens and burrowing crustaceans, including *Nephrops*. Loch Gairloch is closed to mobile gear and a number of vessels fish for *Nephrops* using creels.

1.7.2 Loch Fyne

Loch Fyne is a typical steep sided fjordic loch in Argyll, Scotland, and is the longest (at approximately 70 km) and deepest (with a maximum depth of 200 m) of the Scottish sea lochs. The large volume of the loch results in a water exchange time of 13 days, the second longest of any Scottish sea loch (Davies, 1989). Sublittoral sediments in depths of less than 50 m are a mixture of mud or sand and coarser components, with soft mud in the loch basins. Davies (1989) divided Loch Fyne into three regions according to the distribution of benthic communities. The area studied in the present study included two of these regions; the upper loch and loch head (Figure 1.10). *Nephrops* trawlers occasionally fish this area, often when weather prohibits fishing in the lower loch where fishing intensity is greatest. A handful of creel vessels also fish the loch and are limited to the edges of the loch, away from the trawl tows. In addition to commercial fishing, salmon, mussels and oysters are farmed in the loch and the area is very popular with sailors, anglers and divers. The loch also supports a submarine exercise area, which includes a large area closed to fishing just south of Inveraray.

1.7.3 Loch Broom and the Summer Isles

Loch Broom is a fjordic sea loch with near-vertical sides and three distinct basins. The entrance to Loch Broom is sheltered by the Summer Isles, located a few miles to the west. Within this area there is a wide range of marine and coastal habitats, a number of which are of local and national conservation value (e.g. horse mussel (*Modiolus modiolus*) beds, maerl beds, sea-grass beds and tidal rapids). These habitats support a diverse range of flora and fauna. In the deep water, extensive areas of mud can be found, containing large assemblages of sea pens and burrowing crustaceans, including *Nephrops*. Commercial fisheries within this area are focused on shellfish, with *Nephrops* being the most important, and are fished by both creel and trawl vessels.

Other commercially exploited species include velvet, brown and green crabs caught in creels and scallops, which are either caught by divers or by dredging (Anon, 2005a). In addition to commercial fishing there are six active salmon farm sites and the loch is also used for sailing, kayaking and diving (Anon, 2005bc) (Figure 1.11).

1.7.4 Loch Sween

Loch Sween is a shallow sea loch with soft sediments, supporting dense populations of megafaunal burrowing species, including *Nephrops* (Atkinson 1987, 1989). These species lead to extensive bioturbation of the sediment, producing semi-permanent burrows and generating a surface topography of mounds and craters more varied than that found on grounds smoothed by the regular passage of trawls (see Coggan *et al.* 2001). Sediments are mostly poorly oxygenated silty-clay mud, with high organic matter content. The loch system comprises the main loch, opening to the southwest, and a number of arms also aligned on a northeast-southwest axis. The main fishing method in the upper parts of the loch is creeling. Trawlers occasionally access the more open parts of the upper loch and fish the main loch when poor weather prevents access to more open water in the Sound of Jura. The study site for ghost fishing and UWTV in the present study, is located within the sheltered upper part of the loch, in the central arm known as Sailean Mhòr, at a depth of 16 to 18 m below chart datum, northeast of the the harbour at Tayvallich. The site chosen, close to the narrow opening of Loch Craiglin, was on a creeled ground (and was thus realistic in terms of the local fishery) and its proximity to the shore reduced the risk of loss or damage to equipment caused by trawlers (56° 2'N, 5° 34.5'W) (Figure 1.12).

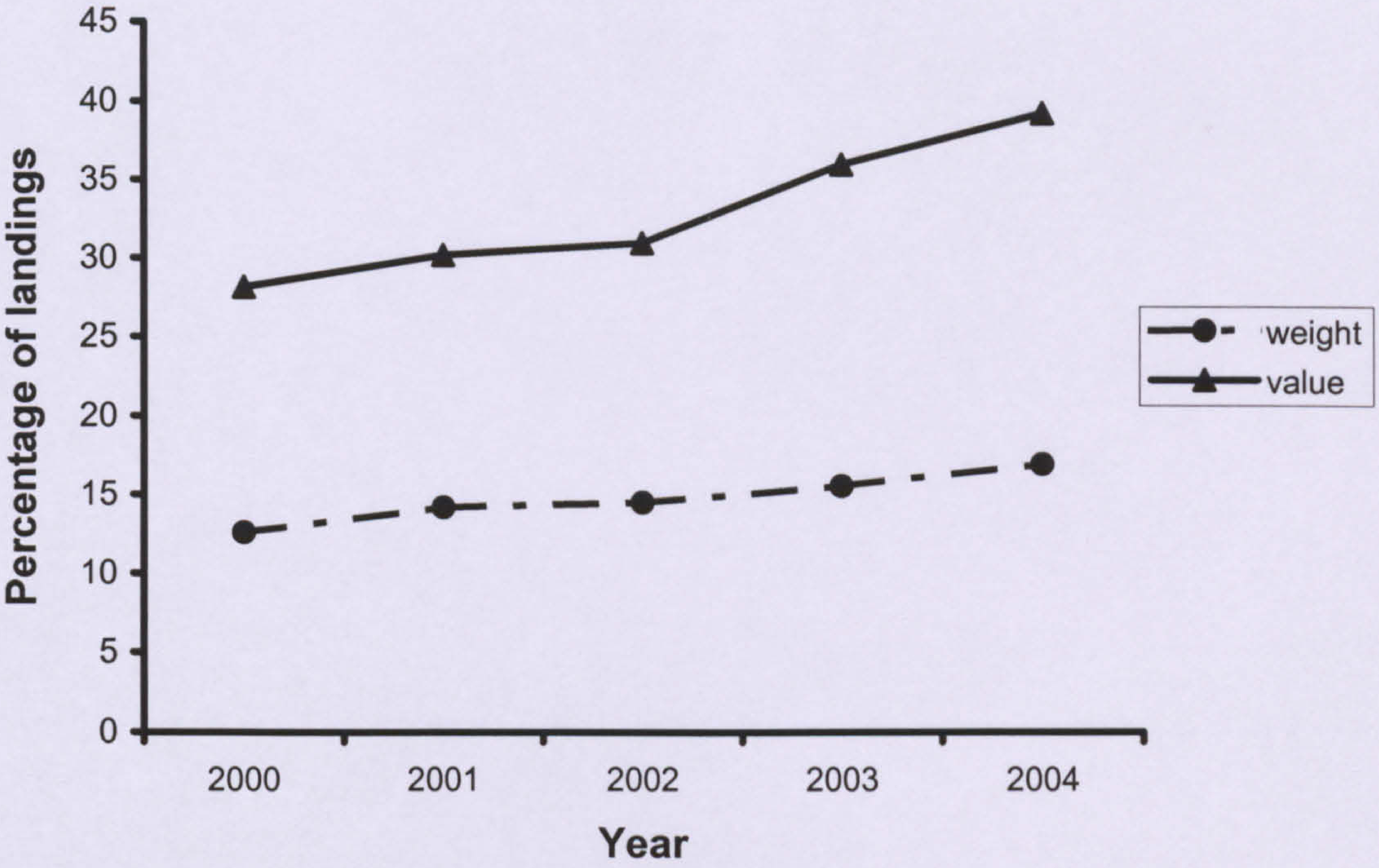

Figure 1.1 - Contribution of the creel fishery to landings of *Nephrops* on the west coast of Scotland by weight and value.

Figure 1.2 - 'Shooting' creels from a typical creel vessel in Loch Torridon (boxes of bait in the foreground).

Figure 1.3 - A *Nephrops* creel with escape gap. Note the funnel-like enterances on each side and the ridgid eye rings ('hard' eye). The ruler used as a scale is 300 mm long. (Photo: I.P. Smith)

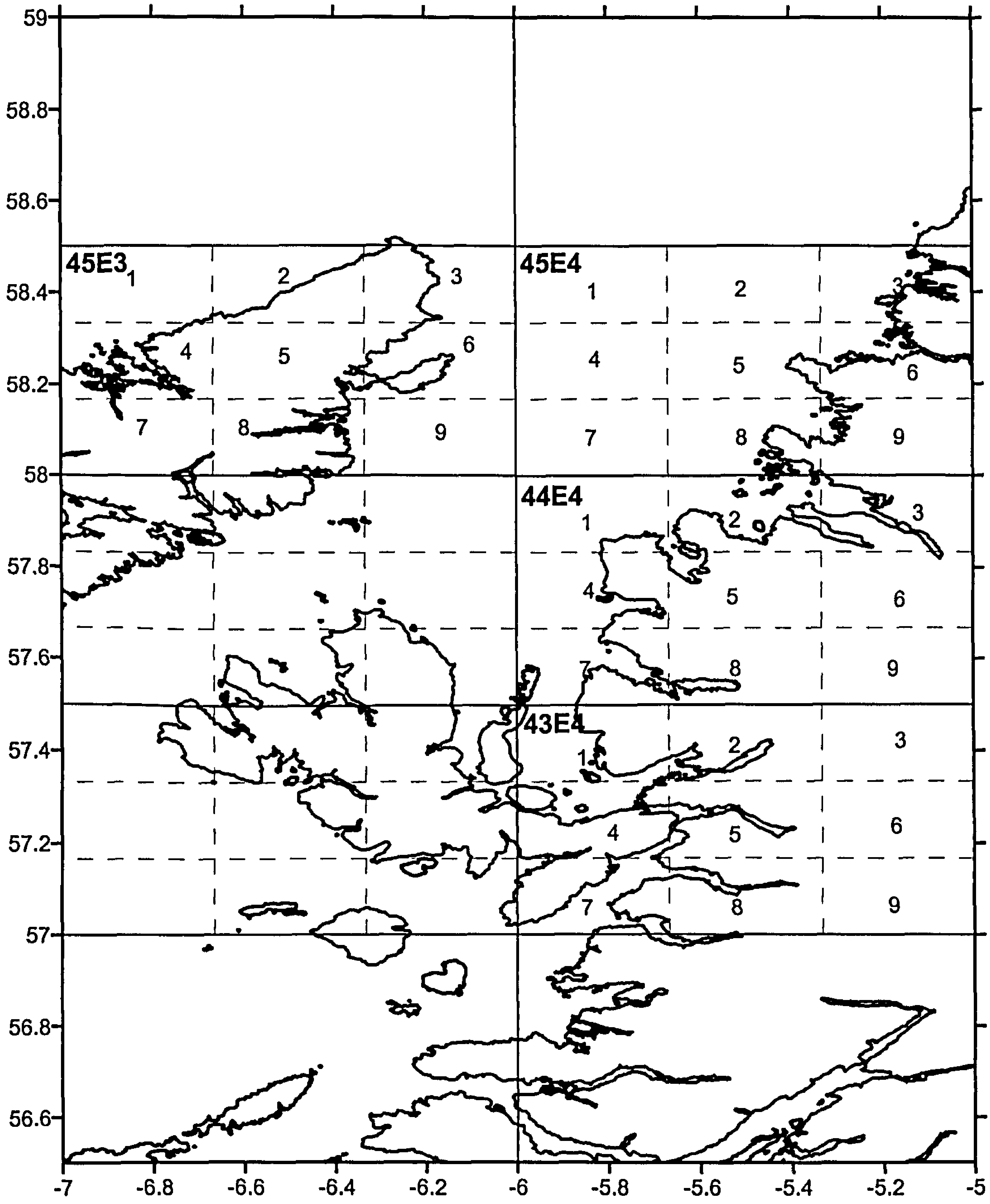

Figure 1.4 - ICES grid for the west coast of Scotland. The Torridon creel fishery occupies 44E4 (sub-divisions 7 & 8) and 43E4 (sub-division 1) (Fisheries Research Services).

Figure 1.5 - Shieldaig Export Ltd packing shed, Ardheslaig, Loch Shieldaig, with typical <10m creel vessel in the foreground.

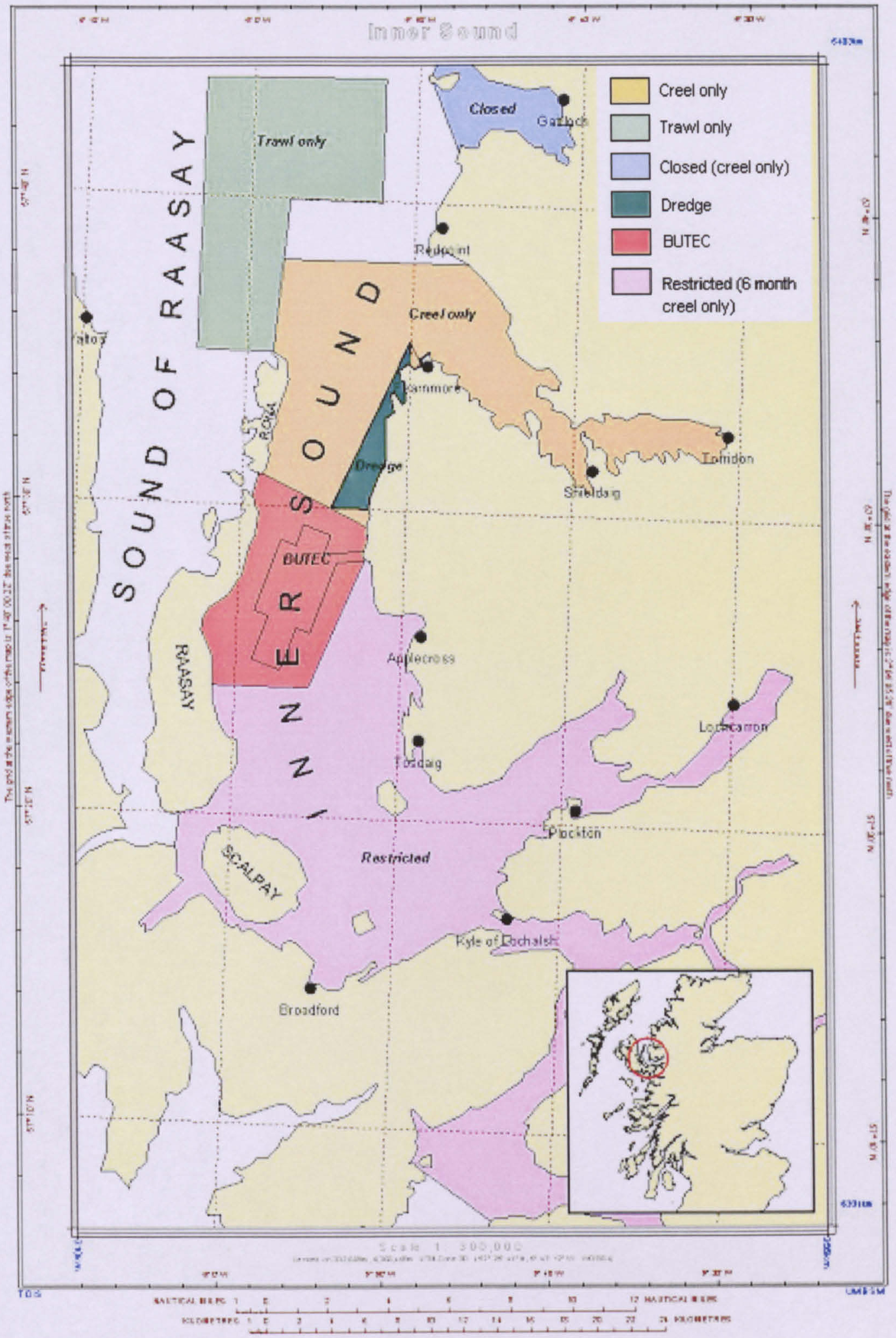

Figure 1.6 - Fishing zones in Torridon and surrounding areas (Map: T.D.I. Stevenson)

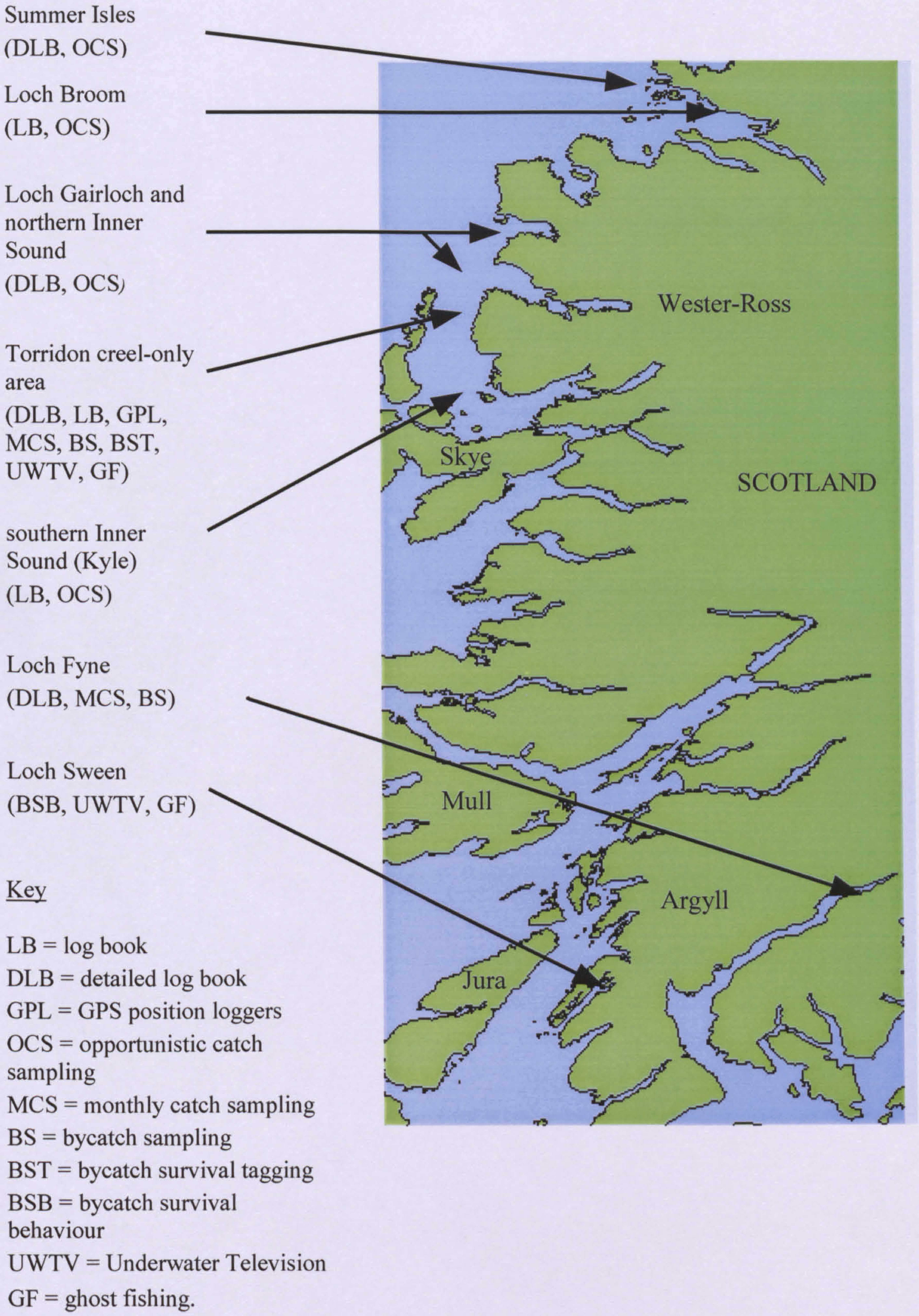

Figure 1.7 - Location of all west coast of Scotland sites studied with work undertaken at each identified

Figure 1.8 - Map of the Torridon area, showing static gear zone (dark blue shading), from which samples were taken and the BUTEC inner sea area (no fishing zone).

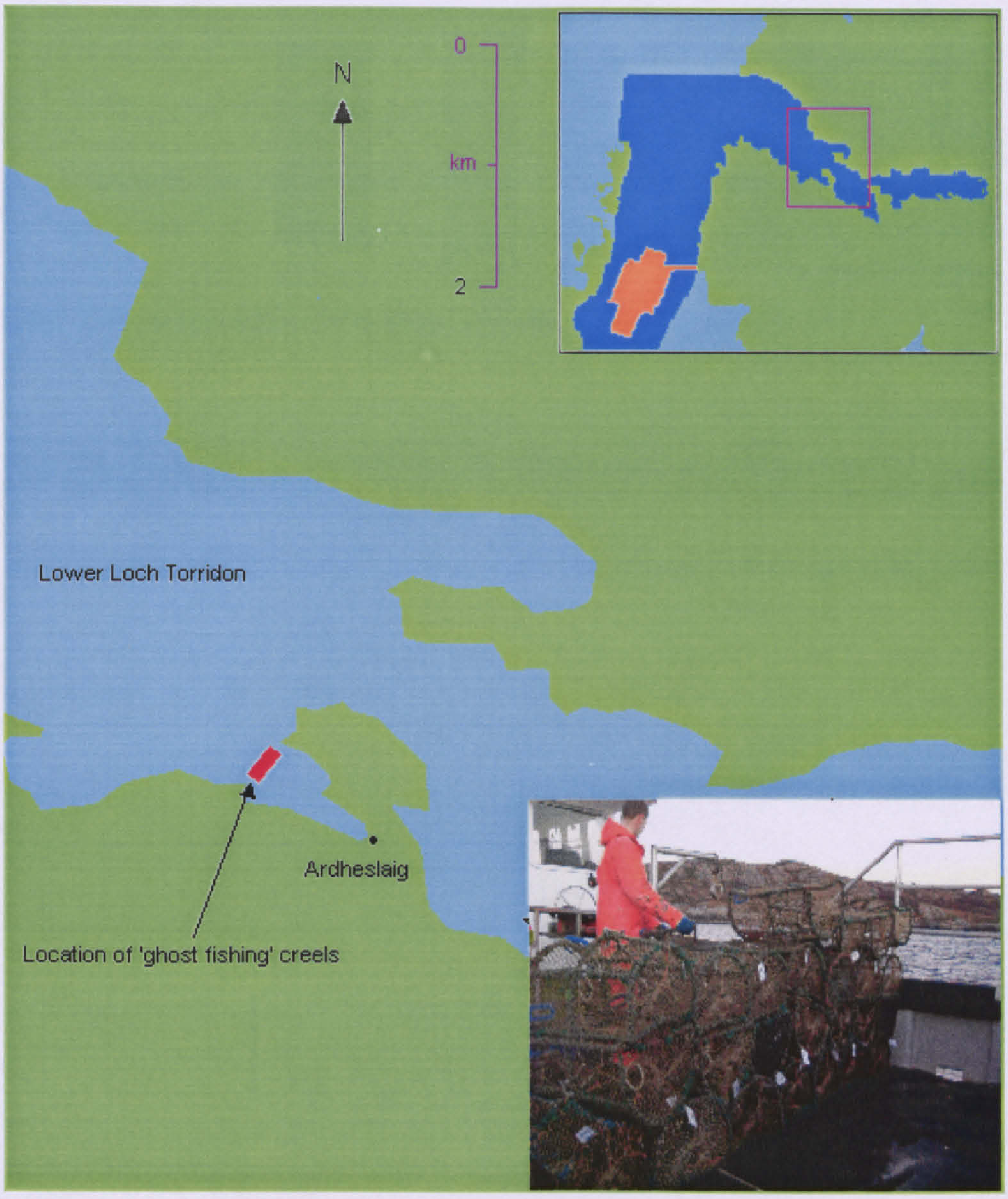

Figure 1.9 - Map of Loch Torridon, Ross-shire, Scotland, showing location of ghost fishing experiment and image of hauled ghost fishing creels.

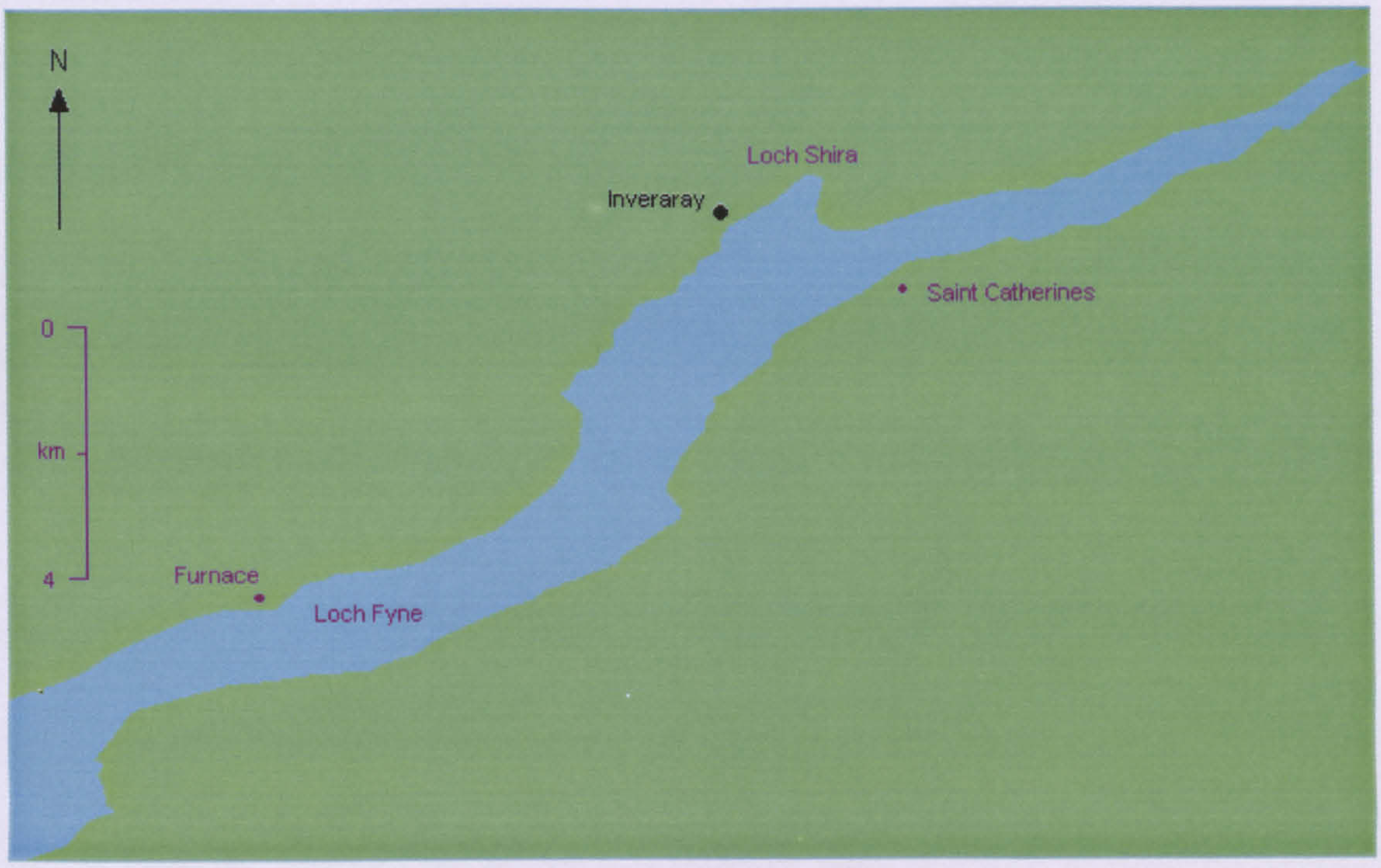

Figure 1.10 - Map of the area in upper Loch Fyne from which samples were taken.

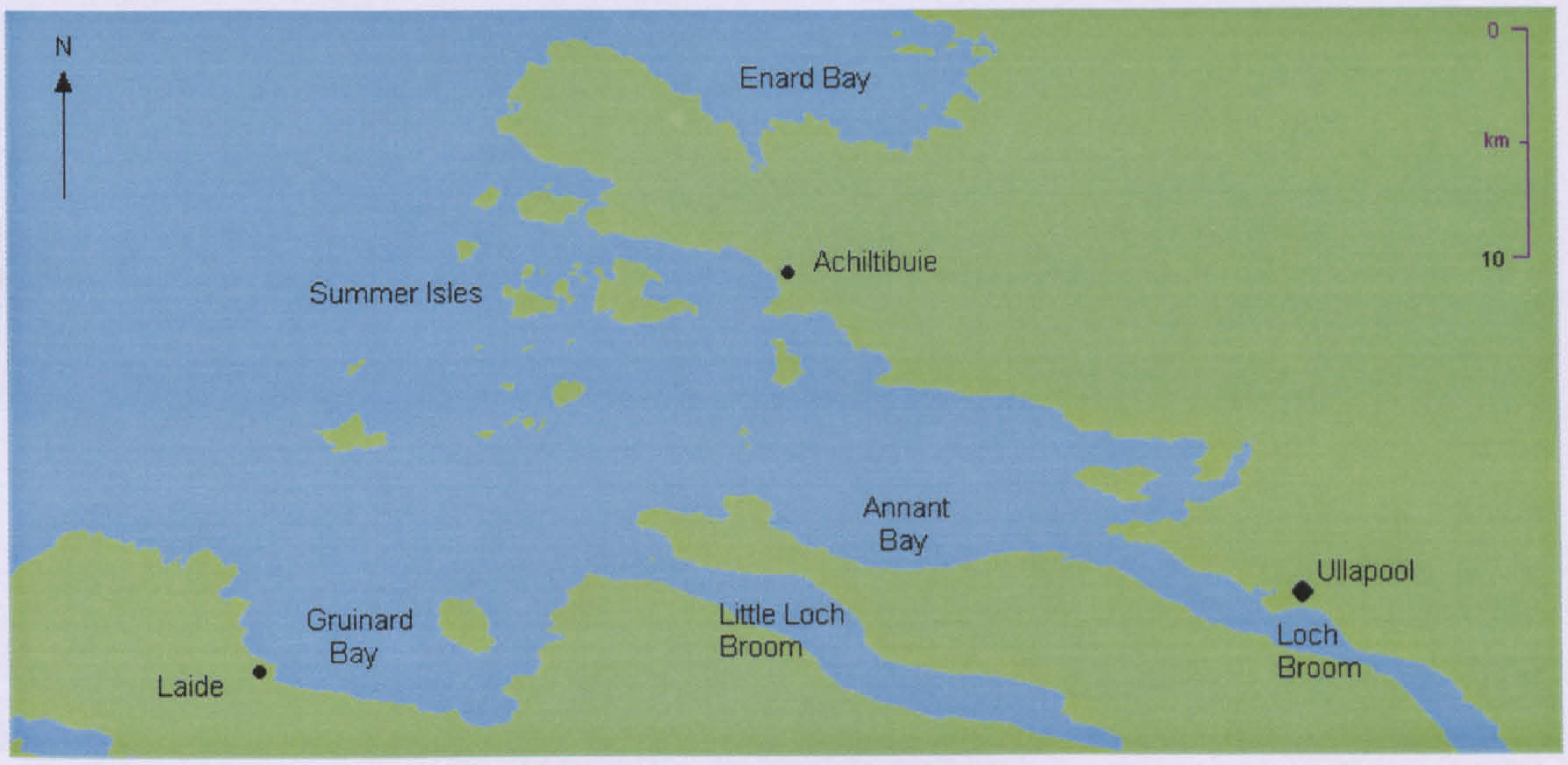

Figure 1.11 - Map of Summer Isles and Loch Broom, the area from which samples were taken.

Figure 1.12 - Map of Loch Sween, Argyll, Scotland, showing location of ghost fishing experiment, UWTV deployment and image of ghost fishing creel *in situ*

Chapter 2

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2 CHARACTERISTICS OF *NEPHROPS* CREEL FISHERIES

2.1 Introduction

An understanding of how the catch in a fishery is affected by biotic (e.g. moult and reproductive cycle, feeding rhythms and inter- or intra-specific interactions) and abiotic (e.g. environmental conditions, temperature, lunar cycle and creel design) factors is essential in order to determine how these factors may affect the catchability of the target species (Robertson, 1989). Quantification of the most important factors is essential if catch per unit effort (CPUE) is to be used as an index of abundance, a pre-requisite for the majority of stock assessments. It is also important to understand processes that may lead to differential exploitation of different components of the stock, since this may have significant effects on the dynamics and sustainability of the fishery.

Previous studies of a variety of decapod species have demonstrated that catches in baited creels are affected by the size, sex, and moult status of the animals, as well as by physical factors such as water temperature, currents and sediment type (Miller, 1990). Temporal changes in the fishery, that are not a result of changes in abundance, often occur over relatively short time scales, usually within a fishing season, and have often been shown to be in response to cyclical changes in environmental conditions (Gulland, 1964). The moult and reproductive cycles of decapods are thought to be particularly important in determining the catch composition of a number of species. Catches are often low prior to and during ecdysis and high thereafter (Miller, 1990), with the sex ratio of catches often varying in relation to differences in the timing of the male and female moult (for example, Pollock & Deb Beyers, 1979). Furthermore, in several species, females appear to be less vulnerable than males to capture in baited creels (Miller, 1990), which may reflect lower food consumption (Branford, 1979). Another factor affecting the catchability of a number of crustacean species is that of seasonal migrations, which are often undertaken in response to their moult and reproductive cycles (for example, Herrnkind, 1983; Freire & González-Gurriarán, 1998; Hunter, 1999; Bellchambers & de Lestang, 2005; Corgos *et al.*, 2006). Seasonal variations in catch in the *Nephrops* fishery were previously thought to be a result of such migrations (Thomas, 1960). This is now known not to be the case, with *Nephrops* movement shown to be restricted to relatively short excursions from their burrows (Chapman *et al.*, 1975; Chapman; 1980).

Other factors known to affect creel catches over temporal scales include seasonal and inter-annual changes in temperature, where an increase in temperature has been shown to increase activity and appetite, as well as the rate of diffusion of bait molecules (Morrissy, 1975). Monthly tidal cycles, seasonal cycles of light, turbidity, and fresh water input as well as diel changes in ambient light level and tidal levels are also known to affect creel catches (Miller, 1990). Diel cycles of environmental conditions are, however, unlikely to affect creel catches, since soak times are rarely less than one day (for a review of studies see Miller, 1990).

In contrast to the temporal variations in catch described above, spatial variations in catch are poorly understood, particularly in the context of creel fisheries. Spatial patterns in catch have been studied on a fine scale for several trawl fisheries (Rijnsdorp *et al.*, 1998; Friedlander *et al.*, 1999; Jennings *et al.*, 2000; Larcombe *et al.*, 2001; Marrs *et al.*, 2000; Marrs *et al.*, 2002; Whalley & Kemp, 2006), with effort being shown to be highly concentrated in certain areas. A few studies on creel fisheries have shown that catch and fishing effort vary spatially with changes in habitat type. These studies have, however, been conducted in discrete areas over relatively large spatial scales (Gobert & Stanisiere, 1997; Steneck & Wilson, 2001; Tremblay & Smith, 2001).

Owing to the considerable financial importance of *Nephrops* as a commercial species, as well as its importance to soft-sediment benthic communities (Chapman, 1980), there have been many studies investigating a wide range of topics. There has, in particular, been a great deal of work into the life history of *Nephrops* and its moult and reproductive cycles are now well understood (Chapter 1; Section 1.4.3). Seasonal variations in moult and reproductive cycle lead to (in Scottish waters) catch rates in *Nephrops* trawl fisheries being lower in the winter, when ovigerous females are largely confined to their burrows and (larger) males make up the majority of the catch. In the summer, the sex ratio of catches is more even, as adult females search for food following hatching of their eggs, leading to higher catch rates (Bell *et al.*, 2006). Similar patterns have been observed in the Scottish *Nephrops* creel fishery, although these data are based on a single study with relatively few sampling periods (Tuck & Bailey, 2000).

Extensive work has been undertaken investigating the spatial distribution of *Nephrops*, with results showing sediment type, among other factors, as having a major influence on the population structure of *Nephrops* (Chapter 1; Section 1.4.2). These differences in population density and structure occur over relatively small spatial scales, with resulting variations in catch from *Nephrops* trawl fisheries (Chapter 3; Section 3.1). Given the spatial heterogeneity of *Nephrops* populations it is likely that the spatial variations in catch observed in the trawl sector will occur within creel fisheries (Chapman & Bailey, 1987).

Nephrops burrowing and emergence behaviour is well understood and has a major influence on trawl catches. Emergence and therefore susceptibility to the trawl fishery is thought to be restricted to a narrow range of low light intensity (Chapter 1; Section 1.4.7). This is unlikely to be an issue for the *Nephrops* creel fishery, as creels are usually left for a minimum soak time of 2 days and such daily rhythms will therefore not affect catch.

The spatial variability of *Nephrops* stocks, as well as the seasonal availability of different components of the stock (Chapter 1; Section 1.2), complicates the assessment of *Nephrops* populations. This has led to the development of several stock assessment techniques with the aim of providing the best advice on the state of individual fisheries.

Stock assessments are carried out in order to provide fishery managers with scientific advice as to the state of the stock and sustainable levels of exploitation. In general, assessments aim to describe the past and current status of the stock and may include stock projections to make predictions about the future stock state, given various levels of exploitation. Stock assessments can be broadly categorised as either fishery-dependent or fishery-independent, depending on the data sources. Fishery-dependent assessments use data derived from fishing activities, for example, landings data, information on fishing effort and biological data obtained by sampling the catch. Fishery-independent methods are based on data collected independently from commercial fishing operations. Currently, *Nephrops* assessments use a combination of both fishery-dependent and fishery-independent methods.

The simplest method of stock assessment is to investigate trends in fishery statistics (for example, catch, effort and size). Changes in the mean size of animals caught in a *Nephrops* stock may indicate changes in recruitment and mortality (Farina & Herraiz, 2003). Catch per unit effort (CPUE) and landings per unit effort (LPUE) are often used as indicators of changes in stock abundance and to evaluate seasonal trends in catchability (Eiriksson, 1999). However, such methods can give misleading results as CPUE and LPUE can remain high when a stock is in decline, if improvements in gear allow for more efficient fishing (Bell *et al.*, 2006). In order to interpret large data sets of LPUE and CPUE, statistical methods, including generalised additive modelling (Hastie & Tibshirani, 1990) have been employed to account for sources of variation, including location, vessel type and season (for example, Tuck & Bailey, 2000; Marrs *et al.*, 2000, 2002b).

In some cases, fishery-dependent analytical assessments are used to provide management advice for *Nephrops* stocks. Length-based assessments, including length cohort analysis (LCA) (Jones, 1979), have, in the past, been the main method for assessing *Nephrops* populations. The reason for using a length-based, rather than age-based approach is that, as with all arthropods, *Nephrops* grow by shedding their exoskeleton, which means there is no hard structure with annual growth rings that can be used for ageing. Consequently, the age-based assessment procedures used for finfish cannot be readily applied. The only method for ageing wild-caught crustaceans is based on the accumulation of lipofuscin pigment in the nervous system; however, this technique is impractical to apply and difficult to calibrate (Sheehy *et al.*, 1996).

LCA uses the length composition of the catch with estimates of natural mortality and von Bertalanffy growth parameters to derive estimates of stock numbers, biomass and size-specific fishing mortality rates of the exploited component of the stock. Yield curves can be generated to predict the short and long-term effects on yield and stock biomass on a per-recruit basis, in relation to the changes within the fishery (Briggs, 2000). This approach assumes that the fishery is in equilibrium with constant recruitment, and stable exploitation patterns but given that this requirement is rarely met, the reliability of the results is uncertain. Additionally, LCA allows no conclusions to be drawn about recruitment overfishing, potentially enabling misleading conclusions to be drawn in the case of declining stocks (Bell *et al.*, 2006).

The use of LCA is increasingly being replaced by methods applicable to dynamic stocks. These assessment methods involve estimating the age structure of a population by fitting a series of normal curves to the length-frequency distribution (a process known as 'slicing'), enabling the use of age-based assessments developed for temperate fish stocks, such as virtual population analysis (VPA) and extended survivor analysis (XSA) (Anon., 1995; Shepherd, 1996).

The most commonly employed fishery-independent method for assessing *Nephrops* stocks is based on estimates of burrow density from towed underwater television (UWTV) surveys. This type of survey was pioneered in Scotland and first described by Chapman (1985). The ICES Advisory Committee for Fisheries Management (ACFM) recently recommended this technique for estimating *Nephrops* stock biomass (Lordan *et al.*, 2002). This method is described in Chapter 5, Section 5.2.1.2. Video footage is examined to estimate the density of burrows. Burrow density is converted to population density using an estimate of the proportion of occupied burrows. Population biomass is then estimated using size composition data from either LCA or trawl surveys (Bailey *et al.*, 1993). Since 2005, the approach has been to apply harvest rate options to the stocks for which there is reliable UWTV data. This harvest rate is the proportion of the population estimated by UWTV that can be removed as catch. This number is adjusted to take account of discards and then a biomass conversion provides a value for landings. At present, for most stocks including that of the N. Minch area, the harvest rate is 20% (Bailey, 2005; Keltz & Robb, 2006).

A major advantage of UWTV survey-based estimates over LCA and age-based assessments is that they are made independently from the fishery, offering greater stability in stock estimates. In addition, they are not affected by behavioural factors, such as variable emergence rates and creel or trawl selectivity. Fishery-dependent assessment approaches have a further disadvantage of relying on commercial landings data which may be unreliable due to under- or mis-reporting, as well as being based on assumptions that are more applicable to finfish stocks, such as homogeneity of the stock and equal redistribution criteria following the capture of part of the stock (Tuck *et al.*, 1997a).

There are several other methods of investigating *Nephrops* stocks, such as the back calculation of female spawning stock abundance from annual larval production (Nichols *et al.*, 1987; Milligan & Nichols, 1988; Tuck *et al.*, 1997a; Briggs *et al.*, 2002), fishery-independent trawl and creel surveys and tagging studies (Chapman, 1982).

To date there has been little previous work to investigate how spatial and temporal differences in the availability of *Nephrops* to the creel fishery affect catch. The present study investigates how the population biology of *Nephrops* affects the availability of animals to the creel fishery, as well as investigating small- and large-scale spatial variations in catch composition. This information is essential to improve understanding of how the creel fishery exploits the stock, enabling more informed management decisions to be taken and improving stock assessments. To achieve this, two major approaches were taken: *Nephrops* landings and discards were sampled on board commercial creel vessels and logbooks were provided to fishermen to record their daily catch and effort. Additionally, a more detailed assessment of the spatial variations in catch and effort will be discussed in Chapter 3 and an assessment of how *Nephrops* behaviour affects the catch is given in Chapter 4.

2.2 Methods

2.2.1 Catch sampling

Monthly or bi-monthly sampling trips were undertaken in 2004 and 2005 on commercial creel vessels in the Torridon area, and in 2005 in Loch Fyne, as well as periodic sampling in three other areas: the Kyle of Lochalsh, Loch Gairloch and Ullapool, and the Summer Isles. The months sampled and number of creels sampled per area are indicated in Table 2.1. Descriptions and the location of the fisheries are provided in Chapter 1, Section 1.7, Figures 1.7, 1.8, 1.10 & 1.11.

The volume of *Nephrops* catch in relation to the high work rate of the commercial fishermen often led to some fleets not being examined. For those fleets for which the catch was analysed, the Carapace Length (CL) of each *Nephrops* was measured in 1-mm size intervals (to the millimetre below) from the rear of the orbit to the posterior edge of the carapace at the mid-line (standard FRS protocol). In addition, the following features were recorded for each animal: sex, moult status (post-moult - soft exoskeleton or pre-moult - hard exoskeleton) and, if it was ovigerous, the gross egg stage as revealed by colour (normal – green eggs, late – brown eggs). All marketable animals were measured as they were caught and placed in tubes immediately to minimise stress. Where possible, undersized *Nephrops* were also measured as they were caught; however, the total number caught often meant there was not time to measure the undersized animals during the hauling process and they were measured subsequently.

For each creel fleet assessed, position, depth, soak time, number of creels, presence of escape gaps and sediment type were recorded. Sediment type was assessed using the local knowledge of the fishermen involved, and was defined as either 'soft mud', 'sandy mud', 'shells and mud', or 'rocks and mud'. In some cases the categories of 'sandy mud', 'shells and mud', and 'rocks and mud', were combined and described as 'hard ground' in the analysis.

Between 2004 and 2006, a total of 47,099 *Nephrops* were sampled from 26,137 creels. In all ports other than Torridon the fishery was sampled during one day of fishing (6–13 fleets of creels). The Torridon fishery is split between Loch Torridon and the Inner Sound; for this reason where weather and time permitted a second day of fishing was undertaken to provide data from both areas. In all cases data were combined from all fleets sampled during each month and the CPUE calculated ((total catch/total number of creel-hauls)×100). Summary plots have been produced from data collected to illustrate spatial and temporal patterns in catch composition.

The effects of management measures implemented within the Torridon area (Chapter 1; Section 1.3.5) on *Nephrops* catch composition, were assessed by comparing catch sampling data collected during 2004 and 2005 with data from 2001 and 2002 prior to the implementation of escape gaps (provided by J. Drewery, FRS). Catch composition of creel fisheries in the Torridon area was also compared with that in other areas not subject to such regulations.

2.2.2 Logbook landings per unit effort (LPUE)

Participating fishermen (each vessel was provided with a confidential vessel code) were asked to complete daily logbooks noting the estimated landings (kg) of three size categories of *Nephrops*: small (34–40 mm CL), medium (40–48 mm CL) and large (>48 mm CL), as well as the total number of creels hauled during the day. Where appropriate, the ICES fishing ground grid reference and weekly landings were noted (Appendix 1; Figure A.1). Data were provided from 18 fishing vessels in total, from five west coast sites. All vessels fishing in the Torridon area (signed up to the management plan; n = 12) and each vessel sampled for catch in other areas (Loch Fyne n = 1, Kyle n = 1, Gairloch n = 2 and Ullapool n =2) were provided with logbooks. From these data, daily LPUE values (kg 100-creels^{-1}) for three size categories of *Nephrops* were calculated. The months during which fishermen provided logbook data are shown in Table 2.2.

The vessels that operate in the Torridon fishery and land into Shieldaig Export do not retain 'small' animals. Any small animals that do not escape through the escape gaps in the creels are returned to the sea. Consequently, Torridon size categories differed from those used elsewhere and were defined as: small (39–45 mm CL), medium (46–48mm CL) and large (>49 mm CL). Therefore, when comparing data from the Torridon fishery with those from other areas, 'small' Torridon data were defined as 'medium', and 'medium' and 'large' values were combined into the 'large' category.

The relationship between LPUE and time of year ('month'), 'area', 'year' and 'vessel' has been examined within a generalised additive modelling (GAM) framework. GAMs are a generalisation of multiple regression that use additive, non-parametric functions of the predictor variables to improve the values of the transformed dependedt variable (Hastie & Tibshirani, 1990). A GAM was fitted to LPUE (square-root transformed) of each size category in relation to a smoothed month term and the factors 'year' (2004 and 2005) and 'area' (the geographical areas given above) using the MGCV module (Wood, 2000) of 'R' version 2.2.0 (Venables & Smith, 2005).

The seasonal plots show the partial effect of the smoothed month term on the transformed LPUE variable, taking account of the effects of year and area. The area plots show the partial effect of the smoothed area term on the transformed LPUE variable, taking account of the effects of month and year. The annual plots show the partial effect of the smoothed year term on the transformed LPUE variable, taking account of the effects of month and area. The model was run using LPUE data from all ports and also with Torridon LPUE data only.

2.2.3 Data analysis

The combined effects of *Nephrops* size (30-58mm CL) and location (Inner sound and Upper Loch Torridon) on CPUE were tested with two-way ANOVA. CPUE data were $\log_{10}$-transformed before analysis.

The combined effects of *Nephrops* size from the total catch (30-50mm CL), males (30-55mm CL) and from females (30-48mm CL) in the Torridon area and sediment type (Soft mud, Sandy mud, Shelly mud and Rocky Mud), on CPUE were tested with two-way ANOVA. CPUE data were $\log_{10}$-transformed before analysis.

2.2.4 Length-based stock assessment

In relation to the Torridon area, LCA was conducted using length composition data from monthly sampling trips in 2004 and 2005, enabling seasonal variations in the size composition of the catch and discards to be accounted for. LCA was carried out using LBA3 software (ICES, 1999). Total landings data were taken from records compiled by Shieldaig Export Ltd. (logbook landings commenced in April 2004). Using these data, the length-frequency distributions (LFD) of *Nephrops* landings were estimated for the Torridon area in 2004 and 2005. In addition, to investigate the efficacy of the management measures relating to discarding (discarding ovigerous females and larger animals than other fisheries), a further LCA was conducted assuming all discards >37 mm CL were landed.

In respect of the other ports, LCA was conducted on data obtained from periodic catch sampling (monthly in Loch Fyne) and logbook data returned by individual fishermen. In some cases, catch sampling was not conducted regularly throughout the year (Gairloch, Kyle and Ullapool). Some month's logbook data were missing (Loch Fyne, Gairloch, Kyle and Ullapool), and in these cases the average LFD of the catch the month before and after the missing month was used. As in the Torridon area, the LFD's of *Nephrops* landings was estimated. Growth parameters were as used in previous creel fishery assessments (ICES, 1991; ICES, 1999; Tuck & Bailey, 2000) and the sexes were assessed separately (Tables 2.3 & 2.4). Natural mortality was assumed to be 0.3 for males of all ages and in all years. Natural mortality was assumed to be 0.3 for immature females, and 0.2 for mature females (mature females>26mm CL). LCAs for each area were based on data from creel catches only. This was considered valid because trawling was not possible in all areas fished by creels, due to zonal management policies, unsuitable topography for mobile gear, or long-standing agreements between sectors.

2.3 Results

2.3.1 Seasonal variations in catch rate and composition

In Torridon and the other study areas, larger numbers of smaller animals were present in the catch during the spring (May), compared with the winter (February), when larger animals predominated (Figure 2.1). CPUE varied throughout the year, with a general trend of the lowest catch rates from October to March, followed by an increase until May, remaining high until August and then decreasing (Figure 2.2). In addition to the marked seasonal pattern of CPUE, there were some clear spatial and inter-annual differences.

Seasonal patterns in CPUE differed among size categories (Figures 2.3–2.6). Creel catches were dominated by small animals (<41 mm CL) and the total landings therefore reflected the quantity of this size category. Highest CPUE of small animals was recorded in the summer months in most areas (Figure 2.3). CPUE of animals of 41–48 mm CL show a similar seasonal pattern, although the peak in CPUE occurs slightly earlier and there was considerable variation among areas (Figure 2.4). CPUE of animals in the 49–56 mm CL category was more similar among areas and the seasonal pattern differed from that of smaller animals, with peak catches in December to March (Figure 2.5). CPUE of the largest size category (>56 mm CL) was comparatively low and variable throughout the year, with minimum values occurring in May to June (Figure 2.6).

Seasonal variations in CPUE appeared to be strongly correlated with numbers of males and females caught. The summer peak in catch rate observed in Figure 2.2 coincided with a peak in female CPUE (Figure 2.7). Male CPUE varied over the year, with higher values from January to March (Figure 2.8). The seasonal changes in CPUEs of different size categories were therefore largely accounted for by the differences in the sizes of the two sexes (Figure 2.9) and seasonal patterns in catch rates for each sex (Figure 2.10). The dramatic increase of percentage of females in the catch during summer followed a large increase in the number of females in the catch (Figure 2.10).

There were two peaks in the catch rate of post-moult (soft exoskeletons) males, one large peak occurred between December and February, and a further smaller peak between May and August in some areas/years (Figures 2.11 & 2.13). A large peak in the catch rate of post-moult (soft exoskeletons) females was evident between April and July (Figures 2.12 & 2.13).

There were two peaks in the CPUE of ovigerous females (Figures 2.14 & 2.15). A large peak occurred between August and December, and a second much smaller peak between February and May (Figures 2.14 & 2.15). The highest catch rates of non-ovigerious females were observed between these two peaks of ovigerous females (May–August) (Figure 2.16).

2.3.2 Spatial variations in catch composition

Catches in Loch Gairloch consisted of larger numbers of smaller animals compared with the Torridon area, where at the same time of year larger animals were present in higher numbers (Figure 2.17). Differences in size composition of the catch were evident on a smaller scale within the Torridon fishery. For example, the pattern of differences in CPUE of different size catogries of *Nephrops* varied significantly between Upper Loch Torridon which comprised higher numbers of small animals, and the Inner Sound, where the catch comprised a lower number of larger animals (2-way ANOVA, $F_{28,232}=4.97$, $P<0.001$; Figure 2.18). These were not consistent throughout the year (Figure 2.2). Sediment from which sampled catches were taken consisted of a larger proportion of soft mud in the Inner Sound compared with Loch Torridon (Figure 2.19). The catch rate of small- and medium-sized animals was lower from areas where the sediment was described as soft mud compared with 'hard' ground (Figure 2.20). A difference between sexes was also apparent, with a slightly greater CPUE of small males on 'hard' ground when compared with soft ground, where the CPUE of large males was slightly higher (Figure 2.21). The female component of the catch consisted almost entirely of small- and medium-sized animals, with a higher CPUE on 'hard' ground than on soft mud (Figure 2.22). In contrast to males, the mean size of females caught on soft mud was smaller than on harder substrata (Figures 2.21 & 2.22). These patterns of differences in CPUE of different size catogries of *Nephrops* varied significantly by substratum type in all three examples (Total catch - 2-way ANOVA,

$F_{75,4654}=3.20$, $P<0.001$; Males – $F_{75,4654}=3.01$, $P<0.001$; Females - $F_{54,3401}=2.13$, $P<0.001$).

2.3.3 Landings per unit effort (LPUE) from logbooks

The seasonal and annual variations in the Torridon data differed little from the data aggregated from all ports. Accordingly, data from all ports will be presented, with examples from Torridon displayed in Appendix B (Figures B.1–B.8).

There were two peaks in LPUE of both 'medium' and 'all sizes' categories in February and June, with the February peak being more pronounced for all size categories combined (Figure 2.23) and the June peak being more pronounced for the medium-sized animals (Figure 2.25). The LPUE of small animals increased steadily from a winter minimum to a peak in June, and then declined sharply until September, where it remained low for the rest of the year (Figure 2.24). There was a large peak in the LPUE of large animals in February, with a rapid decrease until April, remaining low thereafter (Figure 2.26).

Differences in total adjusted LPUE (after removing the effects of 'month' and 'year') were observed between all areas except the Torridon area and Kyle (southern Inner Sound), with the highest values found in the Summer Isles and Loch Fyne and the lowest values in Loch Gairloch and Loch Broom (Figure 2.27). The highest adjusted LPUE of small animals was also recorded in the Summer Isles area; however, the value for small animals was lowest in Loch Fyne (Figure 2.28), where the adjusted LPUE of large animals was highest (Figure 2.30). Adjusted LPUE from Loch Broom was low in all size categories compared with other areas (Figures 2.28-2.30). Similar adjusted

LPUE values in the Kyle and Torridon fisheries were observed with some variation among size categories (Figures 2.28-2.30).

Considerable differences were found in reported LPUE among participating vessels in the Torridon area (Figure 2.31).

Logbook data showed higher catch rates in 2004 than 2005 (Figure 2.32). There was no difference in the number of small animals caught (Figure 2.33) but there were more medium and large animals caught in 2004, implying that a size-year interaction term would have been significant, if it were included in the model (Figures 2.34 & 2.35).

2.3.4 Effects of management measures on catch composition

Figures 2.36–2.40 illustrate the differences in catch composition between two contrasting fishing areas (Torridon and Ullapool). The catch in Ullapool consisted of a larger number of smaller animals per unit effort compared with the Torridon fishery, where catch rates were lower but average size was greater. Large numbers of animals between 25 and 35 mm CL were discarded in the Ullapool fishery, whereas the majority of discards within the Torridon fishery were between 35 and 42 mm CL, or larger ovigerous females (Figures 2.36–2.41).

The CPUE of smaller (27–40 mm CL) *Nephrops* of both sexes was lower in 2004–2005 (after the introduction of escape gaps in 2003) than in 2001–2002 (Figures 2.42 & 2.43). The quarterly CPUE values of medium, large and extra large animals show little variation in catch rate between years (2001, 2002, 2004 and 2005) (Figures 2.45–2.47). A large number of small animals were caught in 2001 (194 *Nephrops* per 100 creel-hauls), when compared with 2004 and 2005 (66 and 49 *Nephrops* per 100 creel-hauls, respectively), with evidence for higher numbers of small animals in 2002 (101 *Nephrops* per 100 creel-hauls) (Figure 2.44).

2.3.5 Length-based stock assessments

Estimates of F (instantaneous rate of fishing mortality), F_{max} (the fishing mortality at which equilibrium yield per recruit would be at a maximum) and $F_{0.1}$ (the value of F at which the slope of the yield per recruit curve is 10% of the initial slope) from the LCA are given in Table 2.5.

LCA input data in the form of length frequency distributions of annual calculated *Nephrops* removals from each area are given in Appendix B (Figures B.9–B.15).

2.3.5.1 Females

Values of F were <0.05 and <0.21 for females from all areas using ICES 1999 and 1991 growth parameters, respectively. F was less than $F_{0.1}$ for both sets of growth parameters in all areas, suggesting that females were underexploited (Table 2.5). For example, in the Torridon area in 2004, a doubling of fishing effort (assuming a direct relationship between effort and fishing mortality) would give an increase in predicted long-term yield per recruit of 17.2% and 58.1%, for ICES 1991 and 1999 growth parameters, respectively (Figures 2.48 & 2.49). Further graphs of predicted female yield per recruit and spawning stock biomass changes with fishing mortality from Torridon and other areas are given in Appendix B (Figures B.16–B.27)

2.3.5.2 Males – Torridon

Values of F varied in both years using different growth parameters; however, in both cases F was below F_{max} (Table 2.5). The LCA using ICES (1991) parameters produced flat-topped long-term yield per recruit curves for both years and an F value slightly below F_{max} but above $F_{0.1}$ (Figures 2.48 & 2.50). In contrast, the LCA using ICES (1999) parameters produced F values below $F_{0.1}$ for both years, indicating stocks were underexploited, with a doubling of effort corresponding to a fishing mortality less than F_{max} (100% increase in effort would give an increase in predicted long-term yield per recruit of 23.9% and 32% for 2004 and 2005, respectively) (Figures 2.49 & 2.51).

Values of F were slightly lower in 2005 (0.37 and 0.12 for ICES 1991 and 1999 growth parameters, respectively) than in 2004 (0.42 and 0.16 for ICES 1991 and 1999 growth parameters, respectively), suggesting fishing mortality was lower in 2005 (Table 2.5).

When the model was run assuming all discards >37 mm CL from 2005 were landed, the estimate of F increased by only 0.01 for both sexes using either set of growth parameters (Table 2.5).

2.3.5.3 Males – All Ports

For all ports, values of male F estimated using the two sets of growth parameters differed considerably (Table 2.5). This was particularly evident in the southern Inner Sound (Kyle), where F was above F_{max} with the ICES (1991) growth parameters, suggesting the stock was overexploited, but with the ICES (1999) growth parameters, F was below $F_{0.1}$, indicating that the stock was underexploited (a doubling of fishing effort would give an increase in long-term yield per recruit of 17.4%) (Figures 2.52 & 2.53). In the three other ports (Ullapool, Gairloch and Loch Fyne), F was below F_{max} using both growth parameters. F was above $F_{0.1}$ in all three ports using 1991 parameters and in Ullapool and Gairloch using 1999 parameters (Table 2.5). The lowest value of F (0.12) was observed in Loch Fyne using 1999 parameters (a doubling of fishing effort would give an increase in long-term yield per recruit of 30.8%) (Figures 2.54 & 2.55), with F values lower than those observed in the Torridon area (Table 2.5). Further YPR curves and SSBPR curves are given in Appendix B (Figures B.28–B.33).

2.4 Discussion

The availability of male and female *Nephrops* to the fishery appears to vary seasonally. Variations in both numbers and catch composition are related to seasonal patterns in the moult cycle and, for females in particular, the reproductive cycle (Farmer, 1975), as they affect feeding activity and consequently the attractiveness of bait within creels.

In the present study, two peaks in the catch rate of post-moult (soft exoskeletons) males were evident, with one large peak occurring between December and February and a further smaller peak between May and August. Mature males are known to moult twice a year until about 6 or 7 years of age, after which they moult annually (Sardà, 1995). It is therefore likely that the large peak occurring between December and February represents the annual moult of larger males and the semi-annual moult of smaller individuals. The smaller peak in post-moult (soft exoskeletons) males between May and August probably reflects the other moulting period of smaller, twice-yearly moulting males. Sexually mature females are generally thought to moult only once a year throughout their life (Sardà, 1995) and there was correspondingly only a single large peak in the catch rate of post-moult (soft exoskeletons) females, between April and July, with this increase in females being responsible for the dramatic increase of percentage of females in the catch during the summer. The relationship between apparent catchability and moult stage observed in the present study is similar to that described by Miller (1990), who reviewed data on several decapod species (including; *Panulirus cygnus*, *Jasus lalandii*, *Homarus americanus* and *Cancer pagurus*) and suggested a general trend of decreasing catchabilty as ecdysis approaches, with catches near zero for several days pre- and post-moult and a period of high catches as soon as the exoskeleton of the mouthparts is hard enough for feeding.

For a number of crustacean species, females have been shown to be less vulnerable to creels than males (Miller, 1990), a result also observed in the current study throughout most of the year. In the fisheries studied, high catch rates of females were observed from June to August, in contrast to other months when catch rates were low. Adult females appear to become available to the fishery in the spring, following the hatching of their eggs between April and May (Thomas & de Figueiredo, 1965; Farmer, 1975; Chapman, 1980; Bell *et al.*, 2006). Following this, females moult and mate before the exoskeleton hardens, after which their ovaries mature during the spring and summer months. Females are seen in large numbers in the catch during this period, presumably reflecting increased feeding activity. Spawning then occurs between August and October (Thomas & de Figueiredo, 1965; Farmer, 1975; Chapman, 1980; Bell *et al.* 2006), after which ovigerous females tend to remain in their burrows, feeding little until their eggs hatch in the spring (Farmer, 1975). A reduction in feeding by ovigerous females has been observed in other crustacean species, e.g. *Homarus gammarus* by Branford (1979), *H. americanus* by Templeman & Tibbo (1945), *Carcinus maenas* by Ropes (1968), although ovigerous female *Nephrops* may feed opportunistically on plankton drawn into their burrows (Loo *et al.*, 1993), or from mieofana found within the burrow system (McIntyre, 1973).

It has previously been suggested that creel catches contain a higher proportion of ovigerous females than trawl catches, suggesting that ovigerous females may be enticed from their burrows by bait (Watson & Bryson, 2003). Ovigerous females are reported to make up 20–30% of the catch in Scottish creel fisheries and 50% of the catch in Swedish creel fisheries during the winter periods when catches of ovigerous females in the trawl fishery are low (20% of catch) (Eggert & Ulmestrand, 2000; Bell *et al.*, 2006). This difference in exploitation pattern has led to concern that an intensive creel fishery may lead to recruitment overfishing (Eggert, 1998). In the current study, the proportion of ovigerous females in the catch was lower than previously observed, although there was a degree of variation between fisheries. It therefore appears that the susceptibility of ovigerous female *Nephrops* to creels varies both temporally and spatially. Spatial variations may be a reflection of differences in stock characteristics between areas. Results from this study have shown that fewer females were caught on soft mud in the Torridon area than on 'hard' ground. It is therefore likely that the catch of ovigerous females will differ with the sediment characteristics of the geographic location. It is also probable that other factors such as differences in bait, duration of incubation period (Sardà, 1998), within burrow food supply (McIntyre, 1973; Loo *et al.*, 1993) and general nutritional status of females will affect the susceptibility of ovigerous females to capture between locations.

There appear to be two peaks in the number of ovigerous females in the catch: a large peak following spawning between August and December and a second much smaller peak, just prior to egg hatching and subsequent moulting between February and May. It is possible that these two peaks are a reflection of changes in female behaviour, with a greater tendency to leave their burrows just after spawning and prior to hatching. However, Scottish populations of *Nephrops* are known to contain both annual and biennial spawners (Bailey, 1984, Bailey *et al.*, 1989). It is therefore possible that the second smaller peak in ovigerous females reflects a smaller proportion of the female population that spawn biennially. These alternatives could be assessed further if the stage of the eggs was recorded in future studies.

Both catch sampling and logbook records from the Torridon area and all other ports indicated similar seasonal patterns of catch composition. These two commonly used methods of recording catch have previously been compared in other fisheries and have been found to indicate similar seasonal patterns of catch composition (Fox & Starr, 1996; Starr & Vignaux, 1997). The peak in LPUE between February and March in the areas studied is due to increased catch rates of large, post-moult (soft exoskeletons) males. This is followed by a similar peak from June to July, due to an increase in smaller, post-moult (soft exoskeletons) females, prior to spawning. After spawning, the number of females caught is low and is reflected by the low LPUE from September until December. A similar seasonal pattern in LPUE was evident in logbook data from creel fisheries in the North Minch collected from 1990–2000 (Tuck & Bailey, 2000).

One difference between catch sampling and logbook results is that the peak of large males in February is much more pronounced in the logbook data. This reflects the fact that catch-sampling data are expressed in numbers, whereas the logbook data are expressed in biomass. Since males are larger, on average, than females, an increase in their numbers will have a proportionally larger effect on LPUE. In addition to corroborating the findings from the catch sampling, logbook data have highlighted the importance of considering the catch in terms of biomass as well as numbers. Considering only catch numbers could potentially lead to an underestimate of the importance of males.

Data collected from logbooks also enabled the assessment of annual differences in LPUE, with a higher LPUE observed in 2004 than in 2005, from both the Torridon area and all ports. This difference was evident from the catch sampling data in the Torridon area, where the average CPUE in 2004 was 185 *Nephrops* per 100 creel-hauls compared with 164 *Nephrops* per 100 creel-hauls in 2005. Logbook data show that the reduction in landings in 2005 was most marked in the large size category (>48 mm CL). A reduction in the proportion of large individuals in the catch could be a symptom of overfishing. It is, however, very difficult to conclude anything from a change from one year to the next; a much longer term dataset would be required to separate natural variations in population size structure and differences in location of fishing from the effects of fishing pressure.

Nephrops populations are known to vary in their density and size structure among locations. Around Scotland, areas of fine sediment are generally characterised by large *Nephrops* at comparatively low population densities and areas of coarser sediment are characterised by smaller *Nephrops* at higher densities (Howard, 1989; Tully & Hillis, 1995). High-density populations may exhibit density-dependent growth (Tuck *et al.*, 1997b; Parslow-Williams *et al.*, 2002). It has also been suggested that a major causal factor in *Nephrops* distribution and population structure is hydrography, since this affects the distribution of settling post-larvae (Bailey *et al.*, 1995; Tuck *et al.*, 1994). Indeed, the sediment relationships are almost certainly a reflection of the hydrographic regime, so both aspects are interrelated.

Sampling CPUE data and LPUE data from logbooks both showed variations in the catch rates of different size categories of *Nephrops* among and within areas. These results suggest there are a number of factors influencing the catch rate of *Nephrops* in different fisheries. Sediment effects, hydrographic influences, fishing pressure and, in the Torridon area, management measures, are all possible factors leading to spatial variations in *Nephrops* catch composition (Addison & Lovewell, 1991). On a large scale, it is very difficult to resolve the effects of these factors. This is particularly true in the Torridon area, as management measures mean that the effects of habitat may be confounded with the effects of fishing practice.

Variations in catch composition were also evident on a smaller scale within the Torridon area. There are evident differences in total LPUE among the 13 vessels fishing within the Torridon area, which may reflect different fishing practices employed by individual fishermen, as well as the different areas fished. To reduce the effect of fishing practice, CPUE data collected from the same vessel were used to compare differences in catch composition within the Torridon area. Catches in Upper Loch Torridon comprised high numbers of small animals, in contrast to the Inner Sound, where the catch was lower but contained greater numbers of larger animals. Several previous studies have found that areas considered to be inhabited by the same *Nephrops* stock have considerable spatial heterogeneity in stock structure (Tully & Hillis, 1995; Maynou *et al.*, 1998; Tuck *et al.*, 1997b; Maynou & Sardà, 2001). These differences may be related to sediment type (Howard, 1989; Tully & Hillis, 1995). For example, sites in Loch Torridon have a higher proportion of mixed sediments compared with the Inner Sound, where the majority of sites sampled have soft mud sediments. Of particular interest is the lower number of small animals caught from soft sediments, which appears to be due to a lack of small females in the catch. It is therefore possible that the higher numbers of small animals caught in Upper Loch Torridon may be partly due to a greater availability of small females in this area.

It is difficult to evaluate the effects of management measures applied to fisheries in the Torridon area by comparing it with other areas without such measures, as gears used, habitat and population characteristics will vary. Discarding practices are different in Loch Torridon, with larger animals than in other ports and ovigerous females being returned. This is acceptable to fishers owing to the zoned fisheries policy (Chapter 1; Section 1.3.5), which allows deep muddy habitats normally exploited elsewhere by trawlers to be fished by creels, yielding larger animals. The high monetary value of these animals more than compensates for the rejection of small animals. In addition, because only creels are used to fish this area, fishers perceive that discarded animals will eventually be caught again at a larger, more valuable size, and in the case of ovigerous females, that they will make a direct contribution to the recruitment of juvenile *Nephrops*, because the egg-bearing proportion of the broodstock is protected.

The efficacy of discarding ovigerous females is, however, not fully understood, as the viability of eggs following discarding is unknown. Furthermore, *Nephrops* fisheries have been shown to catch a higher proportion of large males and this selectivity could result in sperm limitation. This may be particularly true for large females, which may be unable to find a mate of a suitable size (Smith & Jamieson, 1991; Mouat, 2002). It must be noted that although there may be some uncertainties as to the efficacy of returning ovigerous females, it is more than likely that this practice will be beneficial to the fishery.

The comparison of *Nephrops* catch composition from years prior to and after the introduction of escape gaps in 2003 show that this feature is associated with a marked reduction in the number of animals <41 mm CL caught. This finding suggests that the introduction of escape gaps may have reduced the numbers of the smallest component of the catch. There may, however, be other factors leading to the reduction in small animals following the introduction of escape gaps. Towed (UWTV) surveys of the North Minch showed a sharp increase in burrow density between 2001 and 2003, which coincided with an increase in CPUE of the smaller size categories within the North Minch during this period (ICES, 2005). The pre-gap data from the Torridon area also shows a large increase in the CPUE of small animals (<41 mm CL) during 2001. This result, together with findings from the North Minch, suggests there was a large increase in recruitment to this size class during 2001. It is therefore clear that although there was a reduction in small *Nephrops* following the introduction of escape gaps, this reduction should also be viewed in the context of changes in population structure.

The landing of larger, higher-value animals in the Torridon area, is perceived as being beneficial to the fishery and may therefore be used as a management measure in other *Nephrops* creel fisheries. Prior to the instigation of such measures, differences in *Nephrops* populations must be considered and accounted for. For example, the size structure of the catch from the Ullapool fishery is such that if animals below 40 mm CL were discarded, as in the Torridon area, a large proportion of the catch would be returned (63% by number), with commensurate financial loss and discard mortality (Chapter 6). If such a change were to be considered desirable, it would need to be phased in over several years. Furthermore, escape gaps have been shown to release animals above the maximum discarding size in this fishery (35 mm CL) (Livingstone, 2001; J. Adey, unpublished observations). It is therefore very important to assess each fishery individually in terms of the actual cost of implementing such measures and the direct cost of losing the smaller component of the catch. Escape gaps could be selected in relation to the specific size distribution of *Nephrops* within populations fished, though this may create difficulties of enforcement.

The present findings suggest that as a result of management measures in the Torridon area, fishermen target larger, higher-value animals, as well as to some degree reducing the number of *Nephrops* discards by modifying their gear to minimise capture of small animals. It is essential to assess each fishery in terms of the exploited stock, tailoring the management of the fishery in order to protect both the stock and also the livelihood of the fishermen. From discussions with fishermen both within and outwith the Torridon area, it is clear that to encourage compliance with these management measures, it is desirable that fishermen have a sense of ownership of the resource, providing them with an incentive to leave some components of the stock in the sea, thus avoiding the "Tragedy of the Commons" (for example, Hardin, 1968; Berkes, 1985; Berkes, 1986; Jentoft & Kristoffersen, 1989; Davidse *et al.*, 1999; Acheson & Gardner, 2004). It is worth noting that the minimum landing size for *Nephrops* on the west coast of Scotland is 20 mm CL, so creel fishermen are discarding animals well above the minimum legal landing size.

Although it appears that sediment may be an important factor in determining the population characteristics of *Nephrops* within the Torridon area and possibly also in the grounds fished by vessels from other ports, this study has provided evidence of a number of other factors (both environmental and fishery related) that are important in determining the structure of the exploited population and therefore catch composition. It is very difficult to separate such factors on the spatial scale and duration of sampling undertaken in this study. To gain an accurate picture of how the fishery and habitat affect the catch composition, it is essential to assess the fishery on a much smaller scale. A novel technique to assess the spatial distribution of catch and effort on a smaller scale has been trialled in the current study and will be discussed in Chapter 3.

Following the apparent increase in creel fishing effort since the closure of the Torridon area to trawling in 2001, and the implementation of the management measures (Chapter 1; Section 1.3.5), it is important to assess the state of the stock in the area. The assessment of stocks in other areas will also provide a good overview of the fishery as a whole, and will enable a comparison of the Torridon area with more 'typical' creel fisheries. Female LCA assessments in all areas showed similar patterns with very low estimates of F using both growth parameters, with F never exceeding $F_{0.1}$ in each area. Because female F values were much lower than natural mortality, results from these analyses must be considered with caution (Jones, 1981), and therefore values produced from the assessments will not be considered in detail. It is clear from analyses undertaken that in all areas females are lightly exploited, further illustrating that although a certain proportion of females may be enticed from their burrows within a creel fishery (Watson & Bryson, 2003), this number is likely to be low with little detrimental effect on the female population.

As with *Nephrops* trawl fisheries (Eggert & Ulmestrand, 2000; Bell *et al.*, 2006), it is apparent in the creel fishery that the male component of the stock is at the greatest risk of growth overfishing. When assessing males, there were considerable differences in estimates from LCA using different growth parameters. LCA is known to be sensitive to input parameters (Jones, 1981; Tuck & Bailey, 2000; Marrs *et al.*, 2000). Analyses show that using ICES (1991) growth parameters, stocks are either slightly under- or slightly overexploited, but in all cases F was above $F_{0.1}$. Using ICES (1999) parameters, all stocks were shown to be under-exploited, with even a 100% increase in effort resulting in F-values of less than F_{max}. This was particularly evident in the case of the fishery in the southern Inner Sound (Kyle), where either growth overfishing or underfishing was indicated, depending on which input parameters were used.

Present results indicate that stocks of males in the Torridon area and Loch Fyne are at less risk of growth overfishing than those in the southern Inner Sound (Kyle), Loch Gairloch and Ullapool. Estimated fishing mortalities in Loch Gairloch and Ullapool were greater than $F_{0.1}$ using either set of growth parameters. $F_{0.1}$ has been successfully used as a precautionary management reference point for the Icelandic *Nephrops* stocks (Dobby & Bailey, 2006); thus it may be concluded that in order to avoid overexploitation in Loch Gairloch and Ullapool, a reduction in effort may be appropriate, but more data are required to validate this suggestion given the sporadic nature of sampling at these ports. Higher values of F in the southern Inner Sound (Kyle), Loch Gairloch and Ullapool may be a result of higher fishing pressures or differences in growth rates and densities between areas (Thomas, 1965; Bailey *et al.*, 1986; Chapman & Howard, 1988; Tully & Hillis, 1995; Tuck *et al.*, 1997b; Maynou & Sardà, 2001; Tuck *et al.*, 2001). It is difficult to make any firm conclusions with regards to differences among areas, since there were differences in the sampling regime; monthly catch data were available from the Torridon and Loch Fyne fisheries, whereas catch data from other ports were collected opportunistically, often not adequately representing seasonal changes.

It appears that the *Nephrops* stock in the Torridon area is currently at significantly less risk of growth overfishing (F=0.16 in 2004; F=0.12 in 2005) than it was during the periods from 1996 to 1998 (F=0.39) (Tuck & Bailey, 2000) and 1998 to 2000 (F=0.33) (ICES, 2001) (using ICES (1999) growth parameters). It is possible that this may be a reflection of the spatial scale on which the assessments were made, with the scope of the previous assessment encompassing creel fisheries throughout the North Minch area, including fisheries such as Loch Gairloch, where in the current study values of F were similar to those of the previous assessments. Catch data used for the present assessment were much more robust (more regular catch sampling and more reliable total monthly landings values) than those of previous assessments. The conclusion that the Torridon area is currently at less risk of growth overfishing than it was during the periods of 1996 to 1998 is supported by findings from UWTV estimates. The population of *Nephrops* indicated by burrow counts was relatively stable between 1994 and 2001, followed by an increase in abundance over a wide area of the North Minch and surrounding waters after the implementation of the zoned management policy (ICES, 2005). It is likely that the lower value of F in the Torridon area, compared with previous assessments, is a reflection of higher population densities following an increase in recruitment and an increase in the average size of the catch after the introduction of escape gaps, compensating for the increase in effort in the area during this time.

When LCA was performed using data from the Torridon area in 2005, with an assumption that all animals above 37 mm CL would be landed, the estimated fishing mortality was little different from that obtained with actual fishing practice. This result may have arisen because the majority of discards above 37 mm CL consist of females. Since females appear to be lightly exploited, landing larger females would make little difference to the overall fishing mortality. If this is the case, it suggests that the policy of discarding larger and ovigerous *Nephrops* may have little effect on the susceptibility of the stock to growth overfishing. There may be a beneficial effect of releasing ovigerous females on recruitment to the area (or surrounding areas); however, only a small proportion of females will be caught (and discarded) and, as described above, the issues of egg viability and possible sperm limitation need to be resolved.

When assessing these results it is important to consider the inherent problems with LCA. A major drawback is that LCA is applicable only to stocks that have equilibrated to a constant exploitation and recruitment pattern (Hilborn & Walters, 1992; Bell *et al.*, 2006). This is not likely to be the case within the creel fisheries examined, in particular the Torridon fishery, owing to the apparent variability in *Nephrops* recruitment (ICES, 1999; ICES, 2005) combined with increases in fishing effort following the closure of this area to trawling in 2001 (Chapter 1; Section 1.3.5). As data was only collected over two years these variations in recruitment and fishing effort are likley to affect the observed outputs. Furthermore, LCA estimates are more reliable when fishing mortality is significantly higher than natural mortality (Jones, 1981), which was probably not the case in the areas studied. In addition, the assumption of redistribution following fishing (Caddy, 1975) is almost certainly invalid in *Nephrops* fisheries. Although the precise values of the LCA outputs may be uncertain, the overall conclusion that stocks in all areas studied are not currently overexploited is probably robust. To assess risk of recruitment overfishing, more information is needed about the stock-recruitment relationship in this area (ICES, 1999, 2001, 2005).

In summary, the findings from this study show catch composition in the *Nephrops* creel fishery is strongly influenced by seasonal patterns in the moult cycle of both sexes and in the female reproductive cycle. These phenomena lead to variations in both the size distribution and sex ratio of the catch throughout the year. Spatial variations were also evident, with catches among and within fisheries varying in the numbers of constituent animals and in their size distributions. There is evidence for considerable stock heterogeneity within small areas, although the causal factors are hard to assess on a large spatial scale. Voluntary management measures (use of escape gaps) in the Torridon area appear to be effective in reducing the catch of 'undersized' *Nephrops* (i.e. too small to be commercially useful). There is some evidence that the management of the fishery in the Torridon area has had a positive effect on the stock, although it is difficult to isolate this from natural variation. Analysis conducted in other areas suggests that exploitation is sustainable at current stock levels. It must also be considered that whilst the management measures imposed in the Torridon area appear to be effective, these measures are not directly transferable to other fisheries where measures must be tailored to suit their specific requirements.

2.5 Summary

- Studies were carried out using a combination of fishery sampling on board creel vessels and fishers' logbooks.

- Fishery sampling was undertaken on a monthly basis in Loch Fyne and the Torridon area and opportunistically in Ullapool, Gairloch and Kyle (southern Inner Sound). Total landings of small, medium and large *Nephrops* and number of creels hauled were noted on a daily basis from all boats.

- *Nephrops* catch is strongly influenced by seasonal patterns in the moult cycle, and in females by the reproductive cycle.

- A distinct seasonal pattern in catch composition was observed, with a peak catch in post-moult (soft exoskeletons) males at the beginning of the year, followed by a peak in female catch after the mating period in the spring. Numbers of females in the catch were reduced until spawning in the autumn, after which they were present in small numbers in the catch. Similar seasonal patterns were observed at all study sites.

- Seasonal patterns in catch composition lead to the exploitation of different components of the stock throughout the year.

- Considerable stock heterogeneity was observed both between and within fisheries, from fishery sampling and logbook data. A possible effect of sediment type was observed, with lower numbers of small animals being found in soft sediments, due to a lack of small females in the catch.

- LCA assessment suggest females in all areas were fished below $F_{0.1}$ and all male stocks were fished below F_{max}, suggesting that none of the fisheries investigated was being growth overfished.

- Escape gaps appear to be effective at reducing the unwanted bycatch of small *Nephrops.*

- There is some evidence to suggest that the management of the fishery in the Torridon area has had a positive effect on the stock.

- Management measures must be tailored to suit specific requirements of a given fishery and a sense of ownership is required for management measures to succeed.

Table 2.1 - Number of a) creel fleets and b) total creels sampled per boat for *Nephrops* catch and bycatch in different areas on monthly basis.

a)

		2004												2005												2006
Area	Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan
Torridon	Upper Loch Torridon	-	-	1	-	-	-	-	-	-	3	-	-	-	-	-	-	-	-	3	-	-	-	-	-	-
	lower Loch Torridon	-	-	1	7	-	-	-	-	3	4	3	7	3	-	-	2	-	1	4	-	-	2	5	-	-
	Inner Sound	4	3	6	5	7	7	5	-	4	3	4	-	10	6	5	3	7	6	7	7	-	4	2	5	-
Ullapool	Loch Broom	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	8	-	-	-
	Summer Isles	-	-	-	-	-	-	6	-	-	-	-	-	-	-	-	-	-	4	-	-	-	4	-	-	-
Gairloch	Loch Gairloch	-	-	-	-	-	-	9	-	-	-	-	-	10	-	-	-	-	-	-	-	-	-	-	-	-
	Loch Gairloch / IS	-	-	6	-	-	-	-	-	-	-	-	-	6	-	-	-	-	-	-	-	-	-	-	-	-
Kyle	South Inner Sound	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	6	-	8	-	-
Loch Fyne	Upper Loch Fyne	-	-	-	-	-	-	-	-	-	-	-	-	-	13	-	7	9	7	10	-	-	9	8	12	9

b)

		2004												2005												2006
Area	Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan
Torridon	Upper Loch Torridon	-	-	115	-	-	-	-	-	-	345	-	-	-	-	-	-	-	-	345	-	-	-	-	-	-
	lower Loch Torridon	-	-	115	709	-	-	-	-	345	460	345	805	345	-	-	230	-	115	460	-	-	230	575	-	-
	Inner Sound	455	329	689	575	694	779	575	-	460	345	460	-	1150	690	575	345	805	690	805	805	-	460	230	572	-
Ullapool	Loch Broom	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	480	-	-	-
	Summer Isles	-	-	-	-	-	-	360	-	-	-	-	-	-	-	-	-	-	239	-	-	-	240	-	-	-
Gairloch	Loch Gairloch	-	-	-	-	-	-	365	-	-	-	-	-	479	-	-	-	-	-	-	-	-	-	-	-	-
	Loch Gairloch / IS	-	-	379	-	-	-	-	-	-	-	-	-	396	-	-	-	-	-	-	-	-	-	-	-	-
Kyle	South Inner Sound	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	360	-	440	-	-
Loch Fyne	Upper Loch Fyne	-	-	-	-	-	-	-	-	-	-	-	-	-	635	-	389	502	394	492	-	-	425	395	648	492

Table 2.2 - Months from which logbook and GPS logger data have been collected, where S=Simple logbook, D=Detailed logbook, P=Part month of logger data and F=Full month of logger data.

Boat Number	Area		2004												2005											
			Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1	Torridon	Logbook	-	-	-	-	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S
		Data Logger	-	-	-	-	-	P	F	P	-	P	F	F	-	P	-	-	P	F	P	F	P	P	P	-
2	Torridon	Logbook	-	-	-	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S
		Data Logger	-	-	-	-	P	P	-	-	P	F	P	P	-	-	-	-	F	F	F	P	-	-	-	-
3	Torridon	Logbook	-	-	-	-	D	D	D	D	D	D	D	D	D	D	D	D	D	D	D	D	D	D	S	S
		Data Logger	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
4	Torridon	Logbook	-	-	-	S	S	S	S	S	S	S	S	S	S	S	S	-	-	-	-	-	-	-	-	-
		Data Logger	-	-	-	-	P	F	P	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
5	Loch Gairloch	Logbook	-	-	-	-	-	-	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	
		Data Logger	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
6	Torridon	Logbook	-	-	-	-	D	D	D	D	D	D	D	D	D	D	D	D	D	D	D	D	D	S	S	S
		Data Logger	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
7	Loch Fyne	Logbook	-	-	-	-	-	-	-	-	-	-	-	-	-	-	D	D	D	D	D	D	D	D	D	-
		Data Logger	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
8	Torridon	Logbook	-	-	-	-	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S
		Data Logger	-	-	-	-	P	F	F	-	-	-	-	P	P	-	-	-	-	P	F	P	F	F	P	-
9	Kyle	Logbook	-	-	-	-	-	-	-	-	-	S	S	S	S	S	S	S	S	S	S	S	S	S	S	-
		Data Logger	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
10	Torridon	Logbook	-	-	-	D	D	D	D	D	D	D	D	D	D	D	D	D	-	-	-	-	-	-	-	-
		Data Logger	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
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		Data Logger	-	-	-	-	P	F	F	P	-	-	P	F	P	P	-	-	-	-	-	-	-	-	-	-
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		Data Logger	-	-	-	-	P	F	F	P	-	-	P	F	P	P	-	-	-	-	-	-	-	-	-	-
13	Torridon	Logbook	-	-	-	-	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	-
		Data Logger	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
14	Ullapool	Logbook	-	-	-	-	-	-	D	D	D	D	D	D	D	D	D	-	-	-	-	-	-	-	-	-
		Data Logger	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
15	Torridon	Logbook	-	-	-	-	-	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S
		Data Logger	-	-	-	-	-	-	-	-	P	F	F	P	P	P	-	-	P	F	F	P	-	-	-	-

16	Torridon	Logbook	-	-	-	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S
		Data Logger	-	-	-	-	-	P	P	-	-	P	F	F	P	P	-	-	-	-	-	-	-	-	-	-
17	Loch Gairloch	Logbook	-	-	-	-	D	D	D	D	D	D	D	D	D	D	D	D	D	D	D	D	-	-	-	-
		Data Logger	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
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		Data Logger	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

Table 2.3 - Input parameters used in LCA for Torridon *Nephrops* creel fishery (ICES, 1991).

	Growth parameters		Size at		Length weight parameters	
	L_∞	k	maturity (mm)	Terminal *F*	*a*	*b*
Males	92	0.124		0.3	0.00028	3.24
Immature females	92	0.124			0.00085	2.91
Mature females	65	0.087	27	0.025	0.00085	2.91

Table 2.4 - Input parameters used in LCA for Torridon *Nephrops* creel fishery (ICES, 1999).

	Growth parameters		Size at		Length weight parameters	
	L_∞	k	maturity (mm)	Terminal *F*	*a*	*b*
Males	70	0.16		0.3	0.00028	3.24
Immature females	70	0.16			0.00085	2.91
Mature females	60	0.06	27	0.025	0.00085	2.91

Table 2.5 - Summary of values of F, F_{max} and $F_{0.1}$ from LCA on each area (* indicates F > $F_{0.1}$ and ** indicates F > F_{max}).

Location	Year	Input Parameters	Sex	F	F_{max}	$F_{0.1}$
Torridon	2004	ICES (1991)	M	0.42*	0.55	0.27
			F	0.15	0.33	0.20
		ICES (1999)	M	0.16	0.84	0.24
			F	0.04	0.22	0.14
	2005	ICES (1991)	M	0.37*	0.61	0.29
			F	0.09	0.41	0.22
		ICES (1999)	M	0.12	0.75	0.25
			F	0.03	0.21	0.18
	2005 (discards >37mm CL included)	ICES (1991)	M	0.38*	0.62	0.29
			F	0.10	0.54	0.24
		ICES (1999)	M	0.13	0.77	0.26
			F	0.04	0.24	0.20
Ullapool	2005	ICES (1991)	M	0.68*	0.69	0.40
			F	0.21	0.61	0.25
		ICES (1999)	M	0.38*	0.70	0.30
			F	0.05	0.27	0.20
Gairloch	2004	ICES (1991)	M	0.59*	0.68	0.37
			F	0.14	0.50	0.26
		ICES (1999)	M	0.33*	0.73	0.26
			F	0.04	0.27	0.19
Southern Inner Sound (Kyle)	2005	ICES (1991)	M	0.61**	0.57	0.28
			F	0.12	0.56	0.24
		ICES (1999)	M	0.25	0.86	0.31
			F	0.04	0.23	0.17
Loch Fyne	2005	ICES (1991)	M	0.33*	0.46	0.24
			F	0.08	0.51	0.25
		ICES (1999)	M	0.12	0.69	0.23
			F	0.04	0.24	0.23

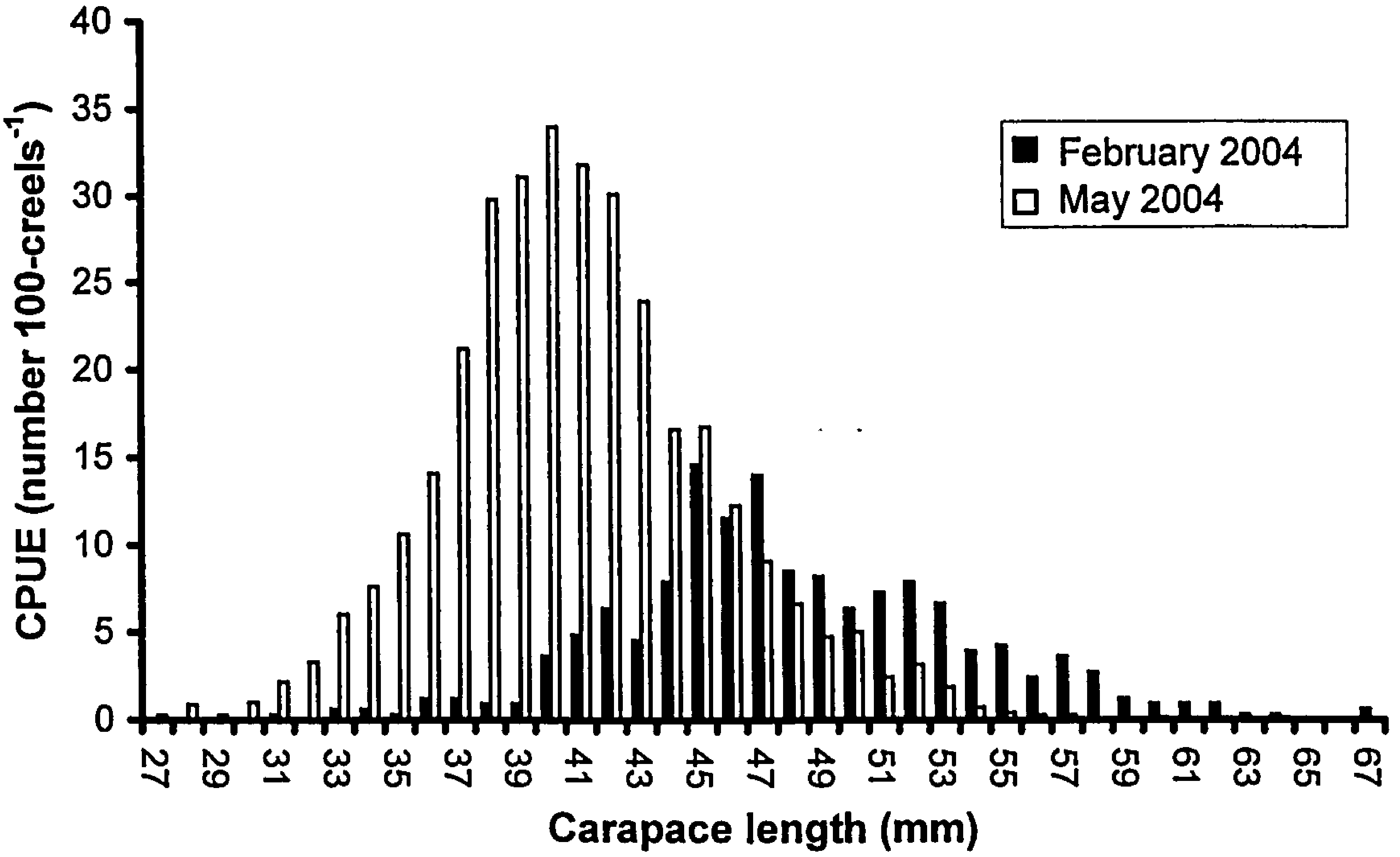


Figure 2.1 - Length-frequency distributions of *Nephrops* standardised to mean number caught per hundred creels (CPUE) in February (Peak male catch, n=329 creels) and May 2004 (Peak Female catch, n=694) in the Torridon area.

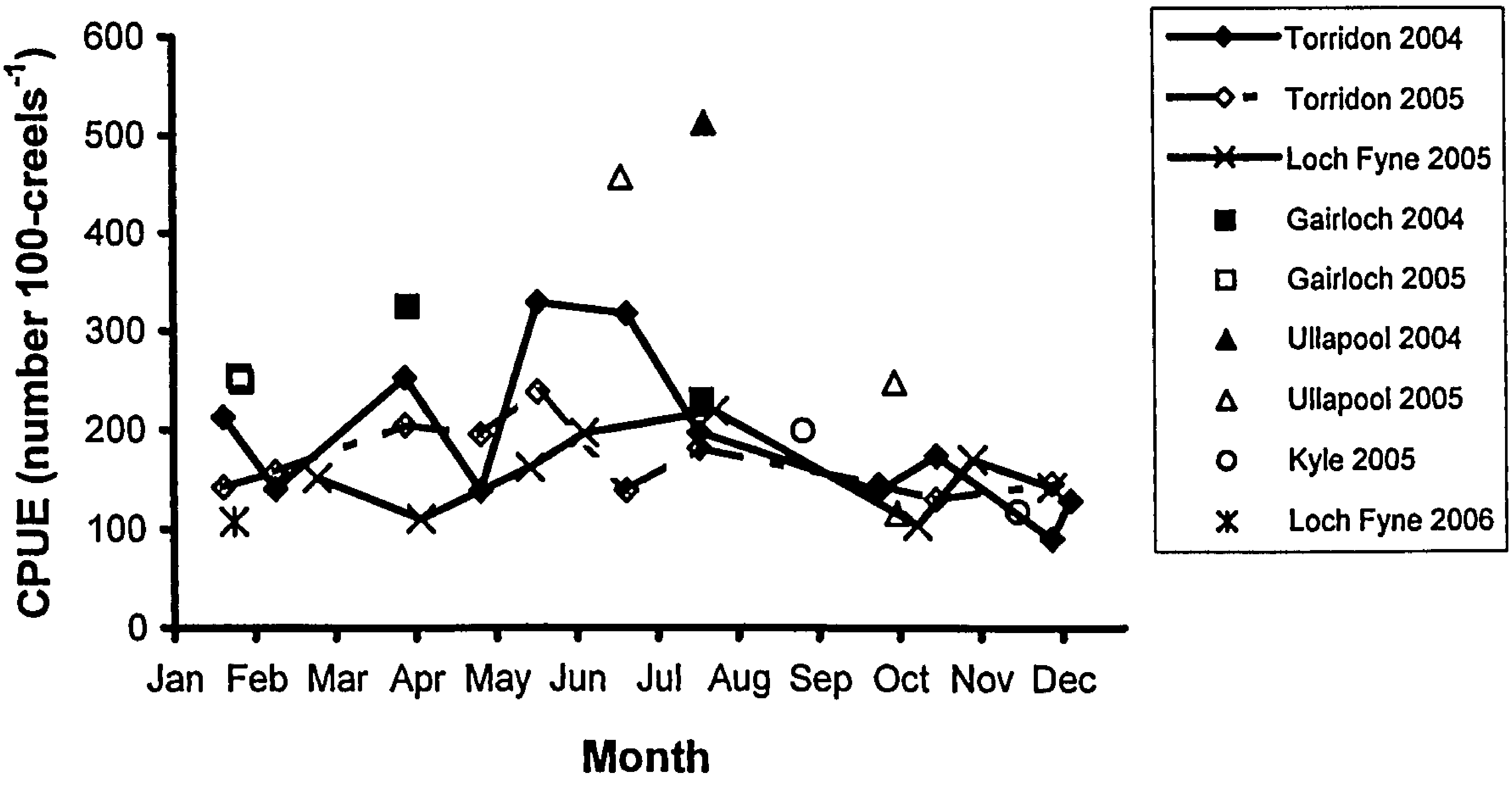


Figure 2.2 - CPUE of total *Nephrops* catch from different areas (Gairloch, n=1 619; Kyle, n=800; Loch Fyne, n=4 372; Torridon, n=17 535; Ullapool, n=1 319).

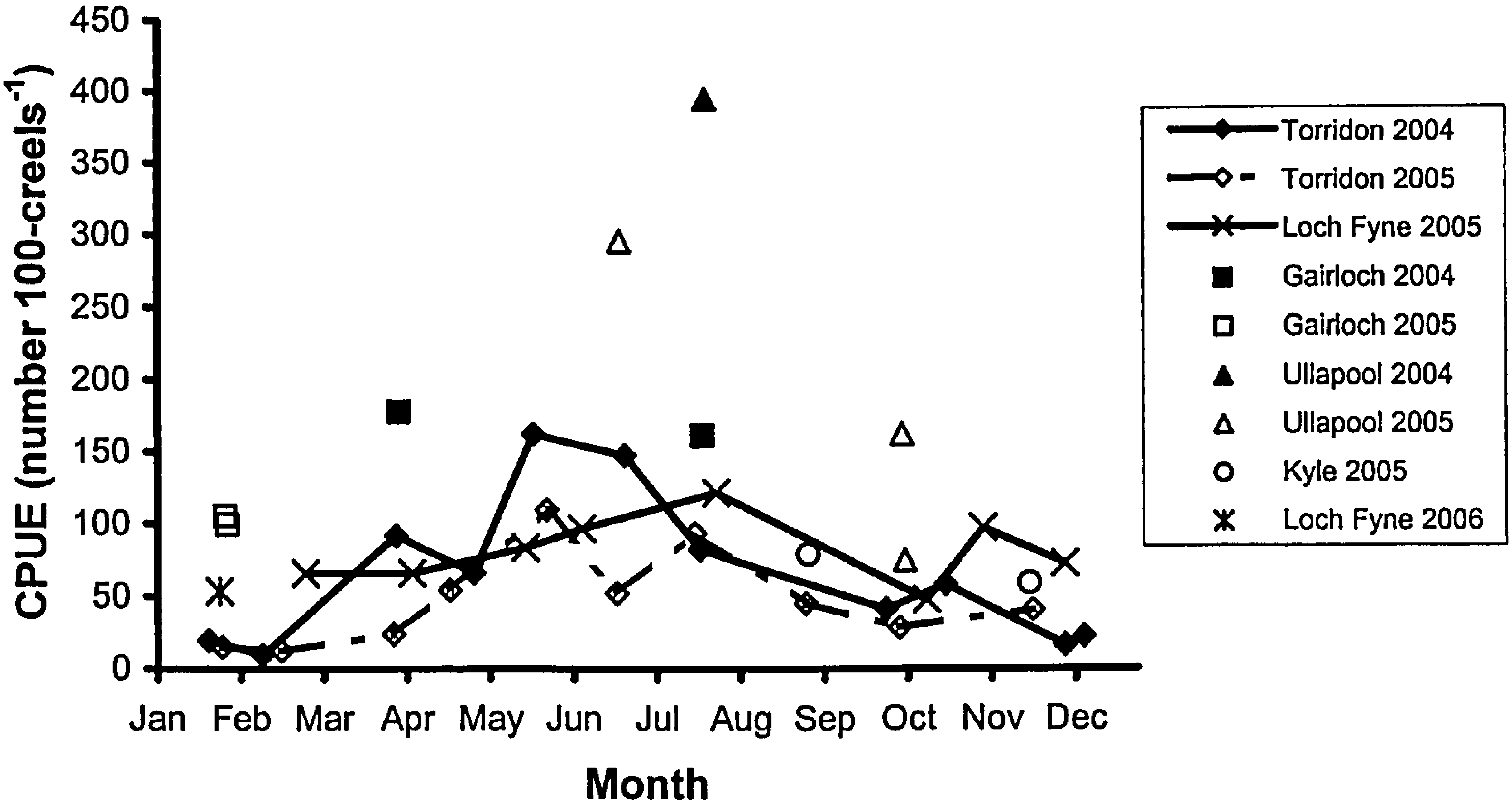


Figure 2.3 - CPUE of *Nephrops* <41 mm CL from different areas.

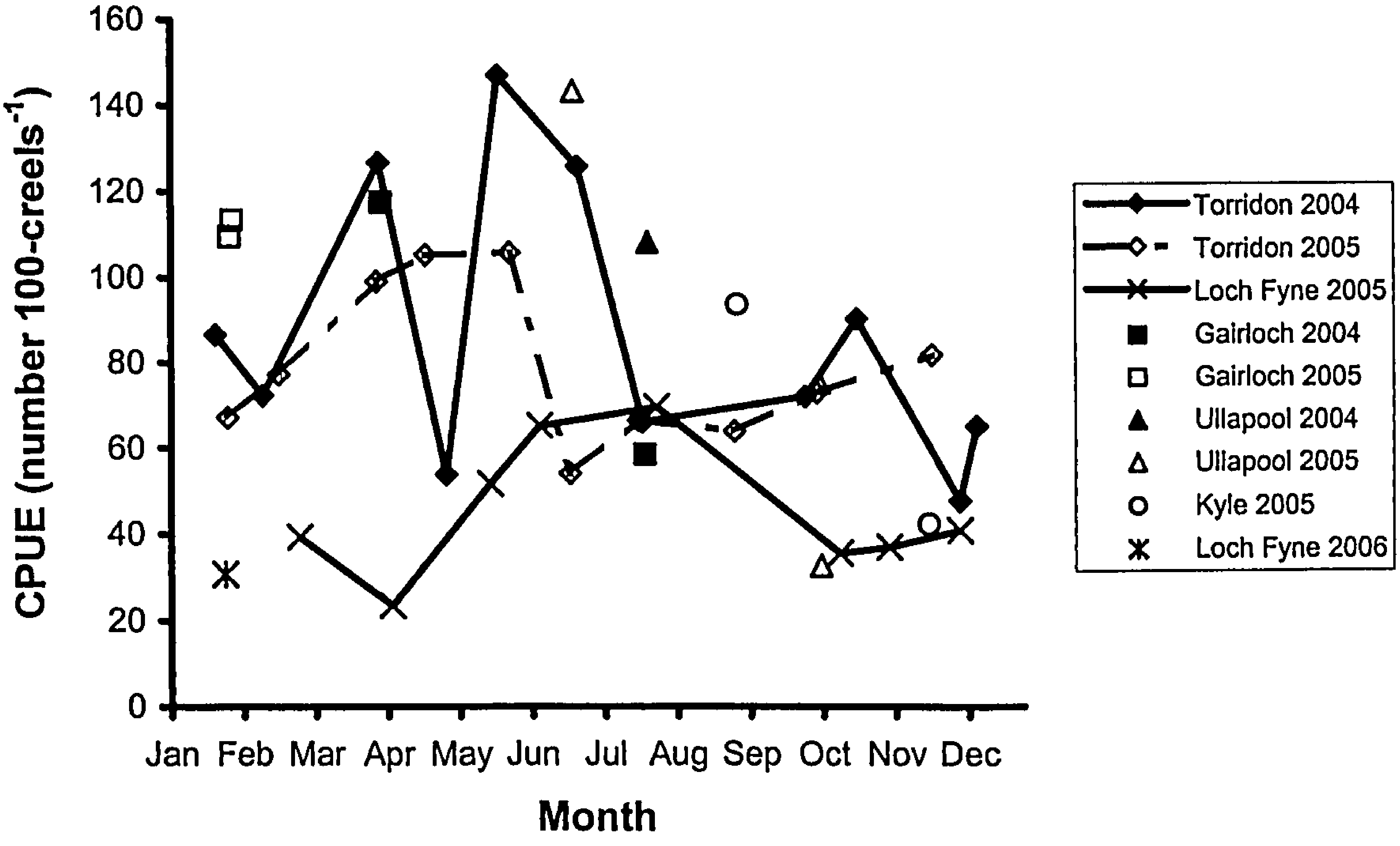


Figure 2.4 - CPUE of *Nephrops* 41–48 mm CL from different areas.

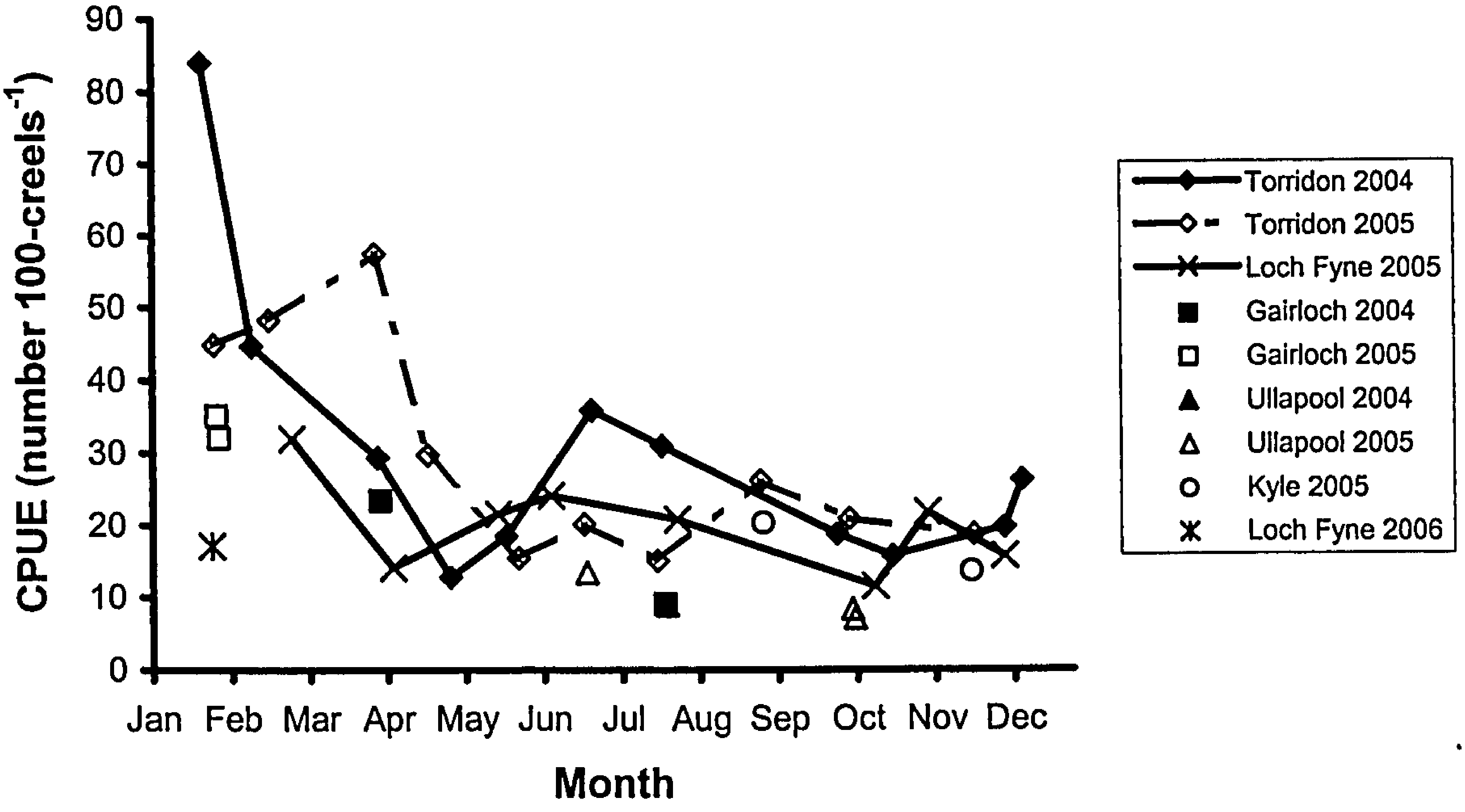


Figure 2.5 - CPUE of *Nephrops* 49–56 mm CL from different areas.

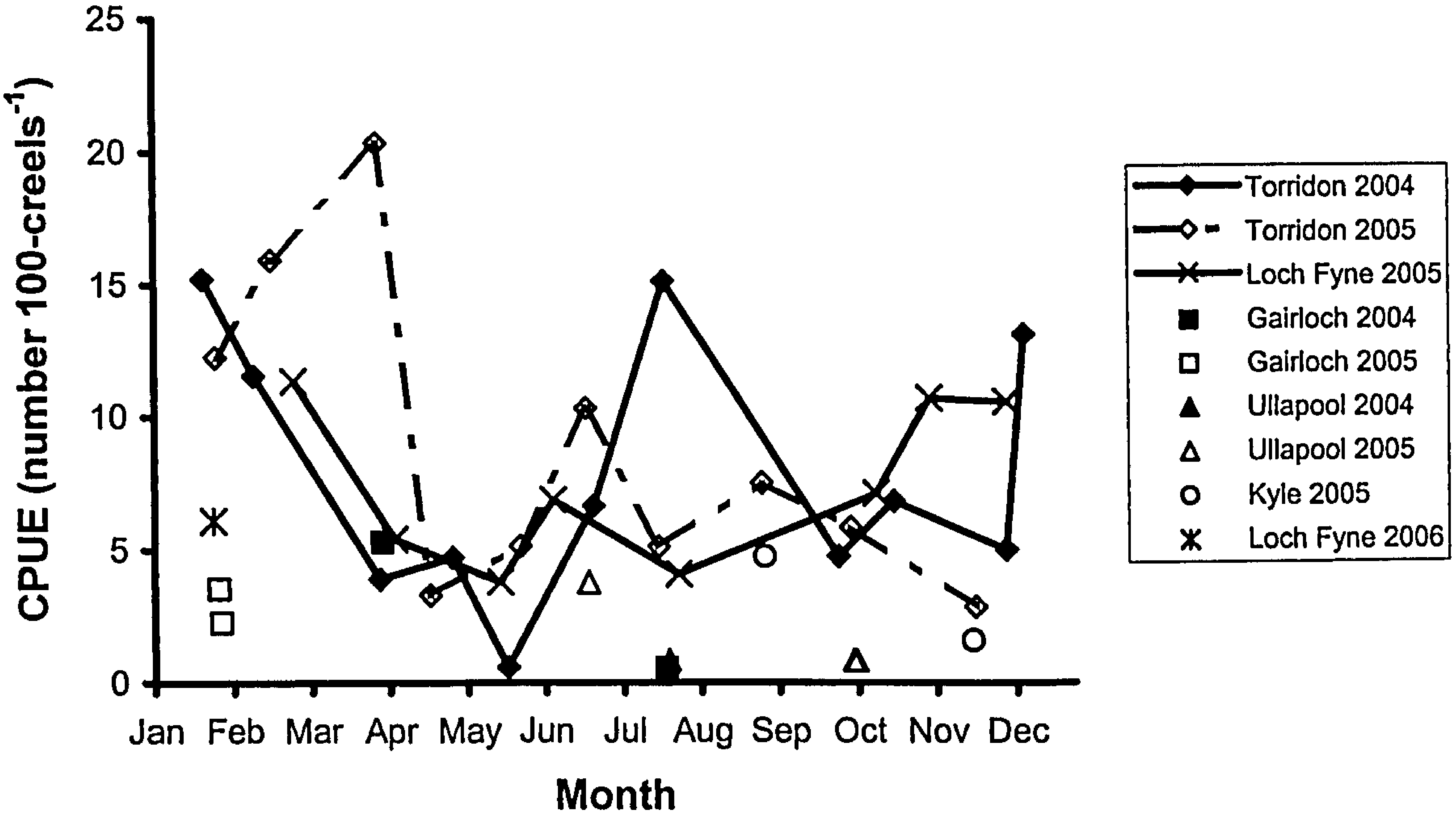


Figure 2.6 - CPUE of *Nephrops* >56 mm CL from different areas.

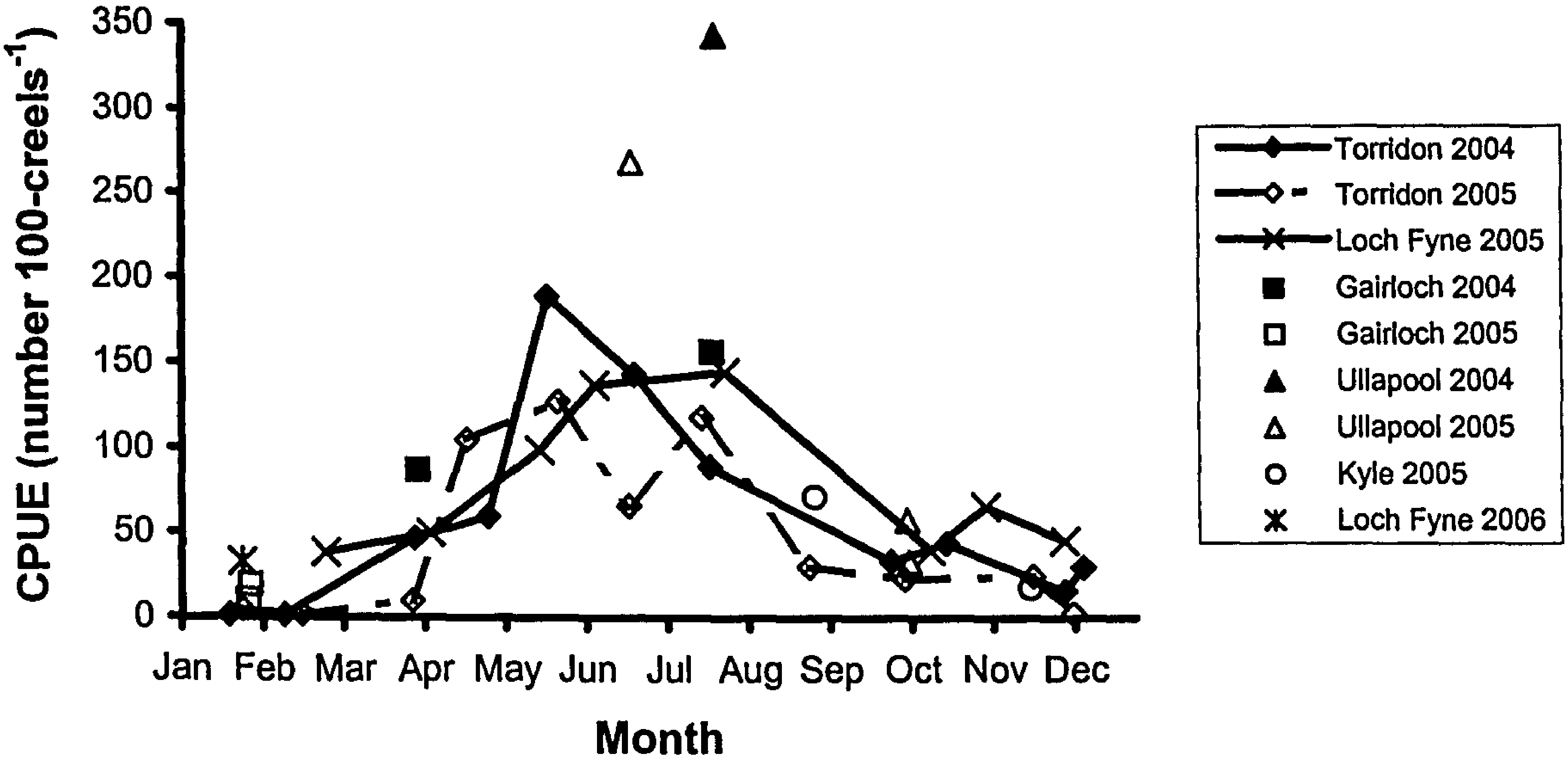


Figure 2.7 - CPUE of all sizes of female *Nephrops* from different areas.

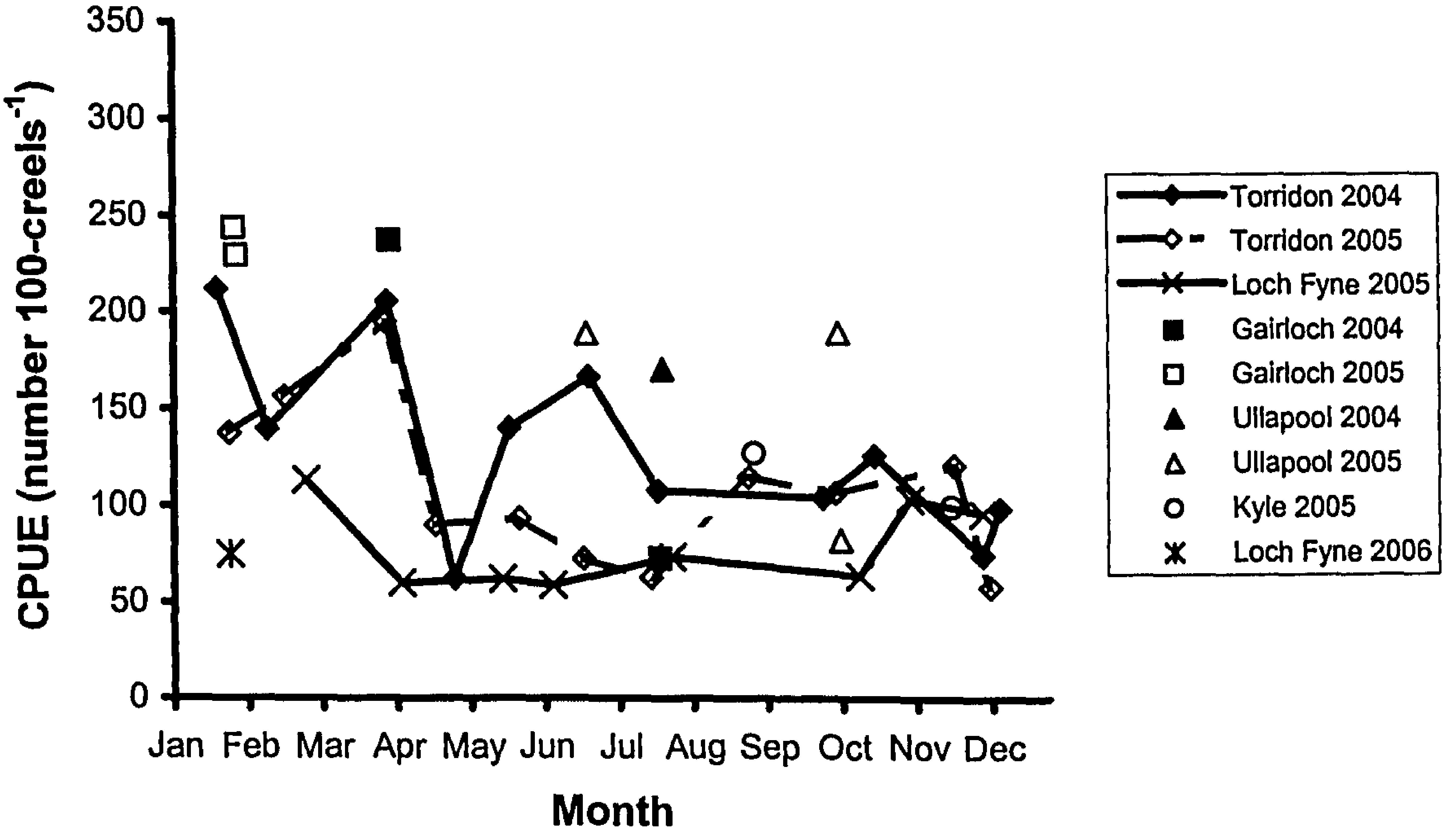


Figure 2.8 - CPUE of all sizes of male *Nephrops* from different areas.

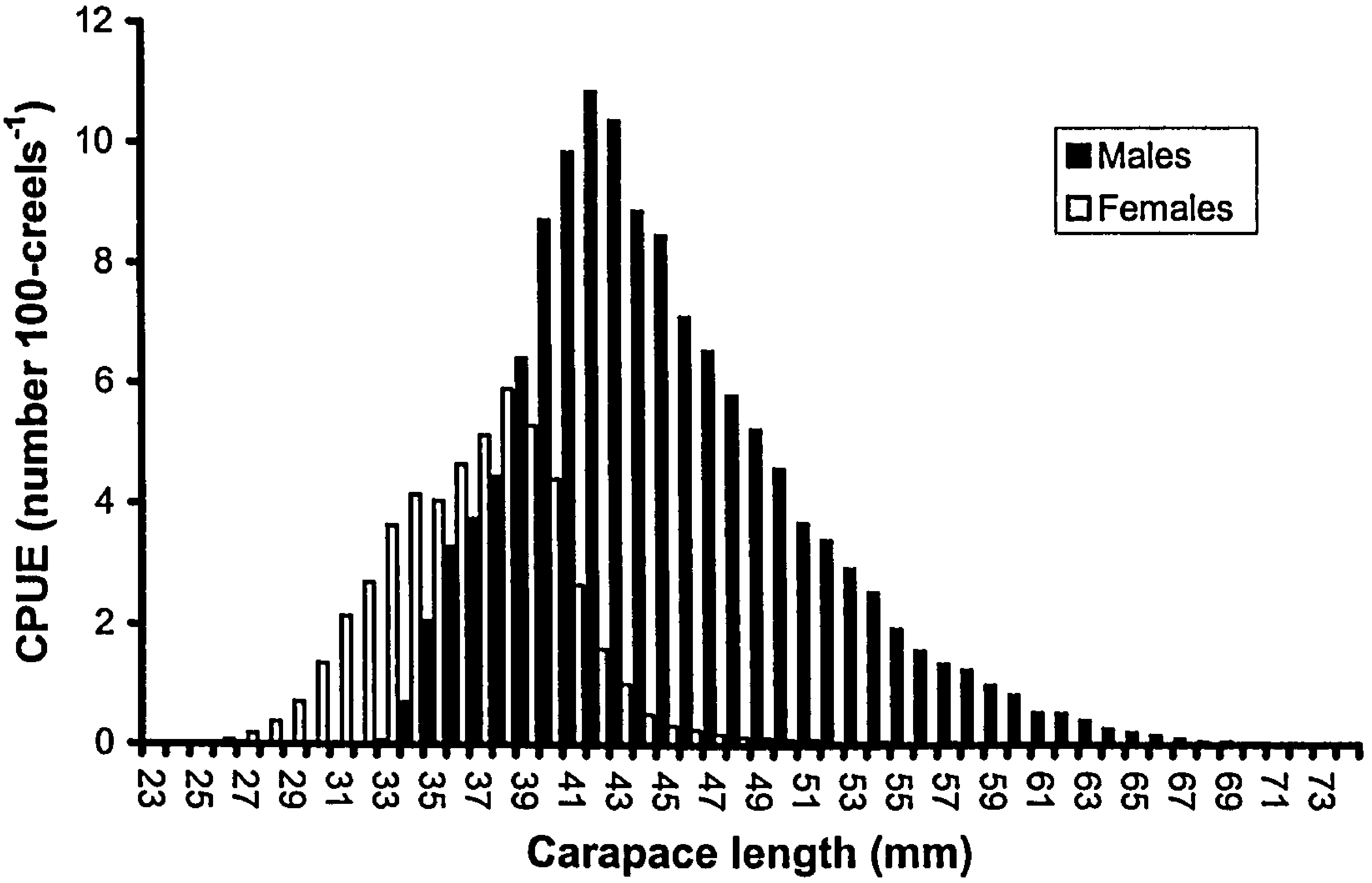


Figure 2.9 - Length-frequency distributions of male and female *Nephrops* standardised to mean number caught per hundred creels (CPUE), aggregated across all areas and months (n=25 645 creels).

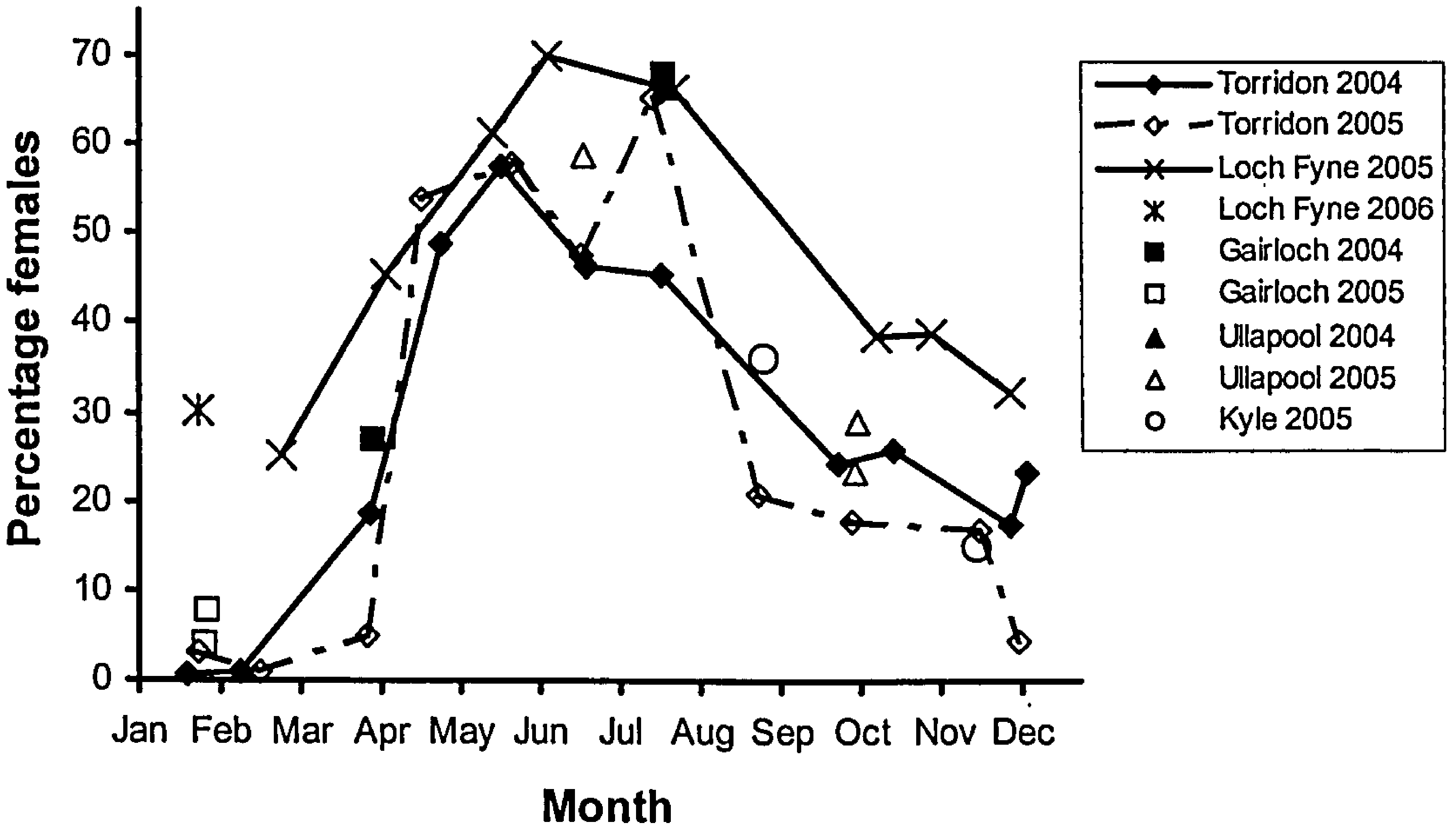


Figure 2.10 - Percentage of female *Nephrops* in the catch from different areas.

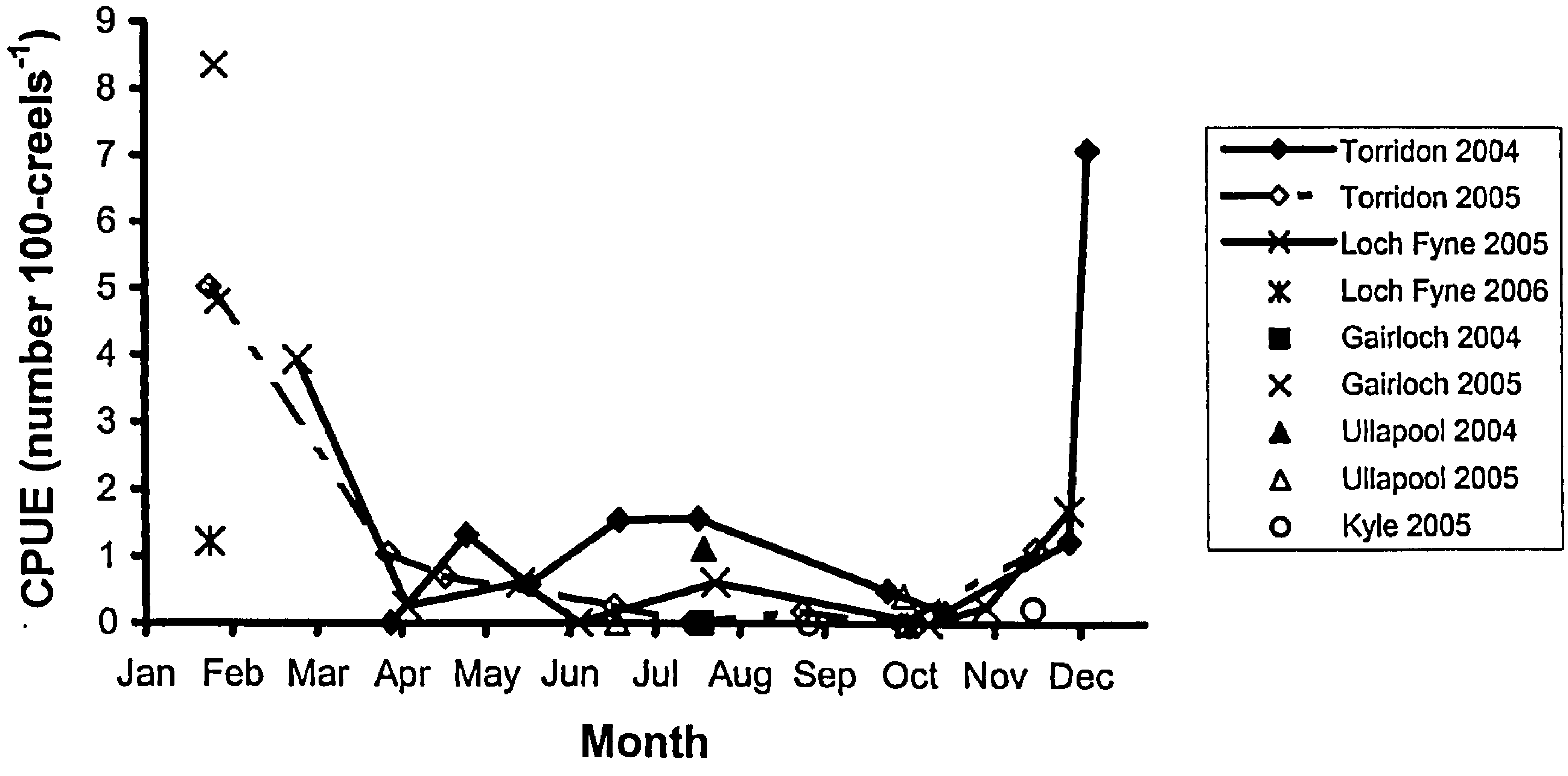


Figure 2.11 - CPUE of post-moult male *Nephrops* from different areas.

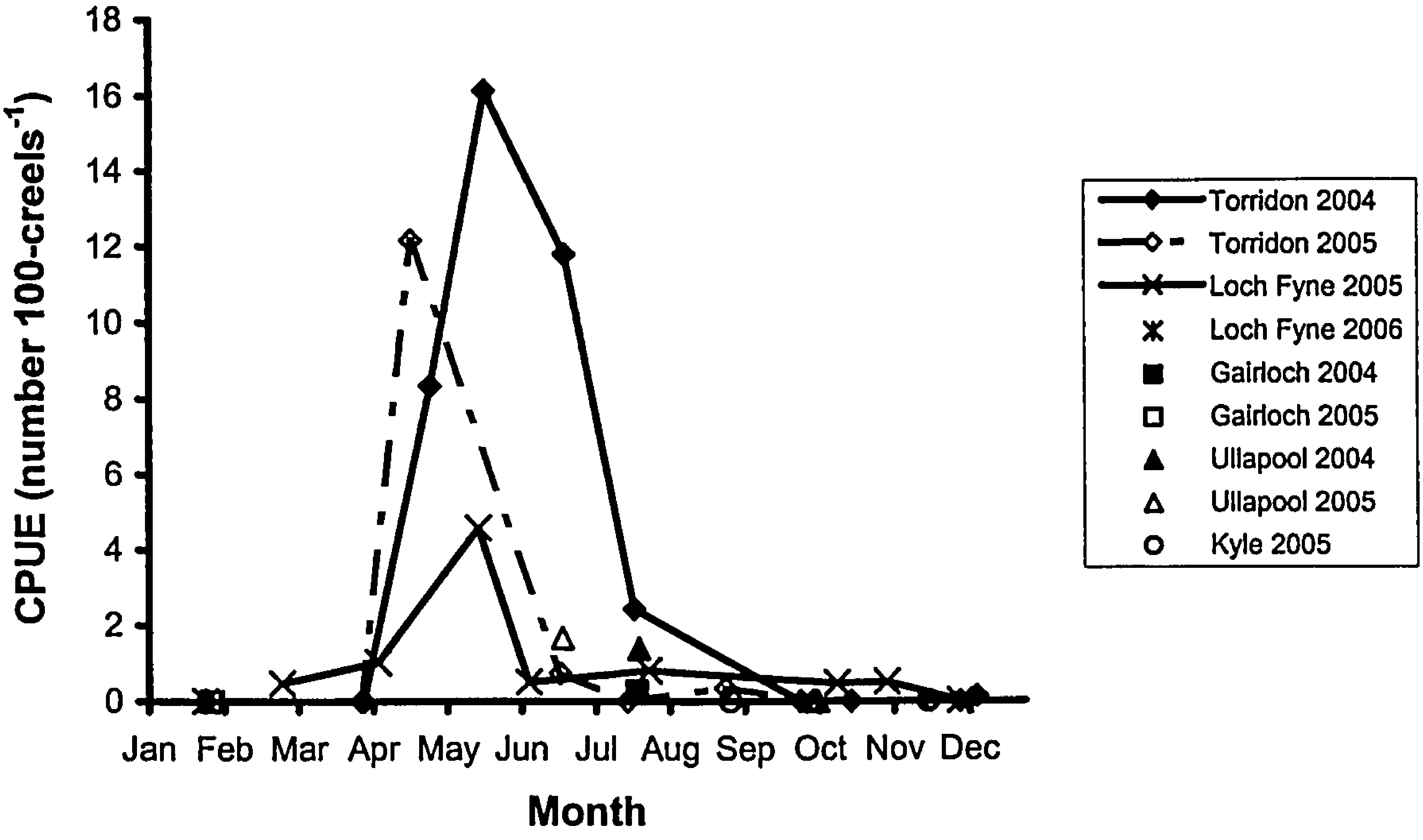


Figure 2.12 - CPUE of post-moult female *Nephrops* from different areas.

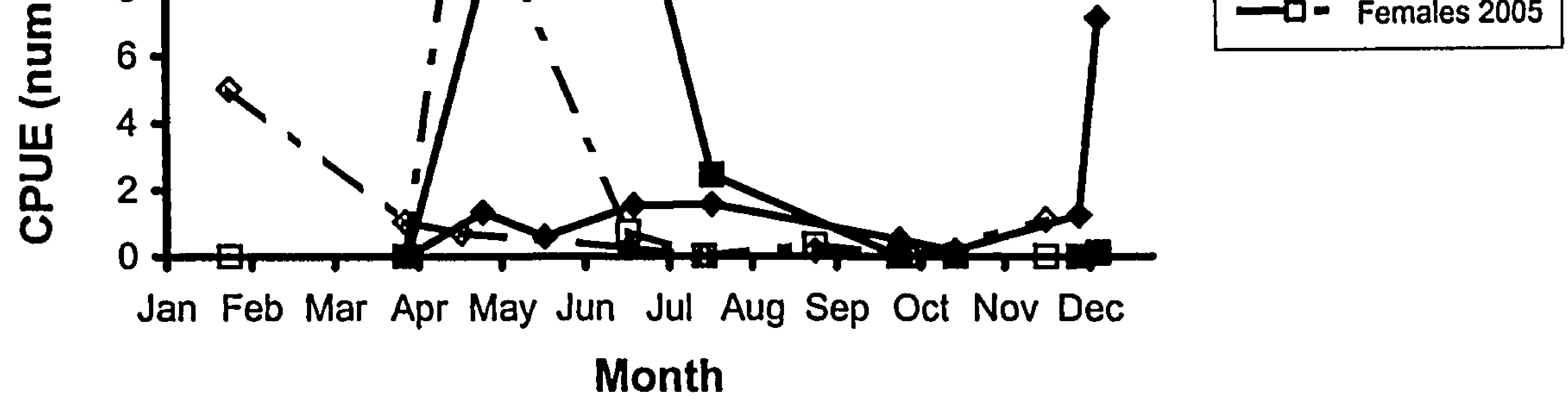


Figure 2.13 - CPUE of post-moult male and female *Nephrops* from the Torridon area (n=17 535 creels).

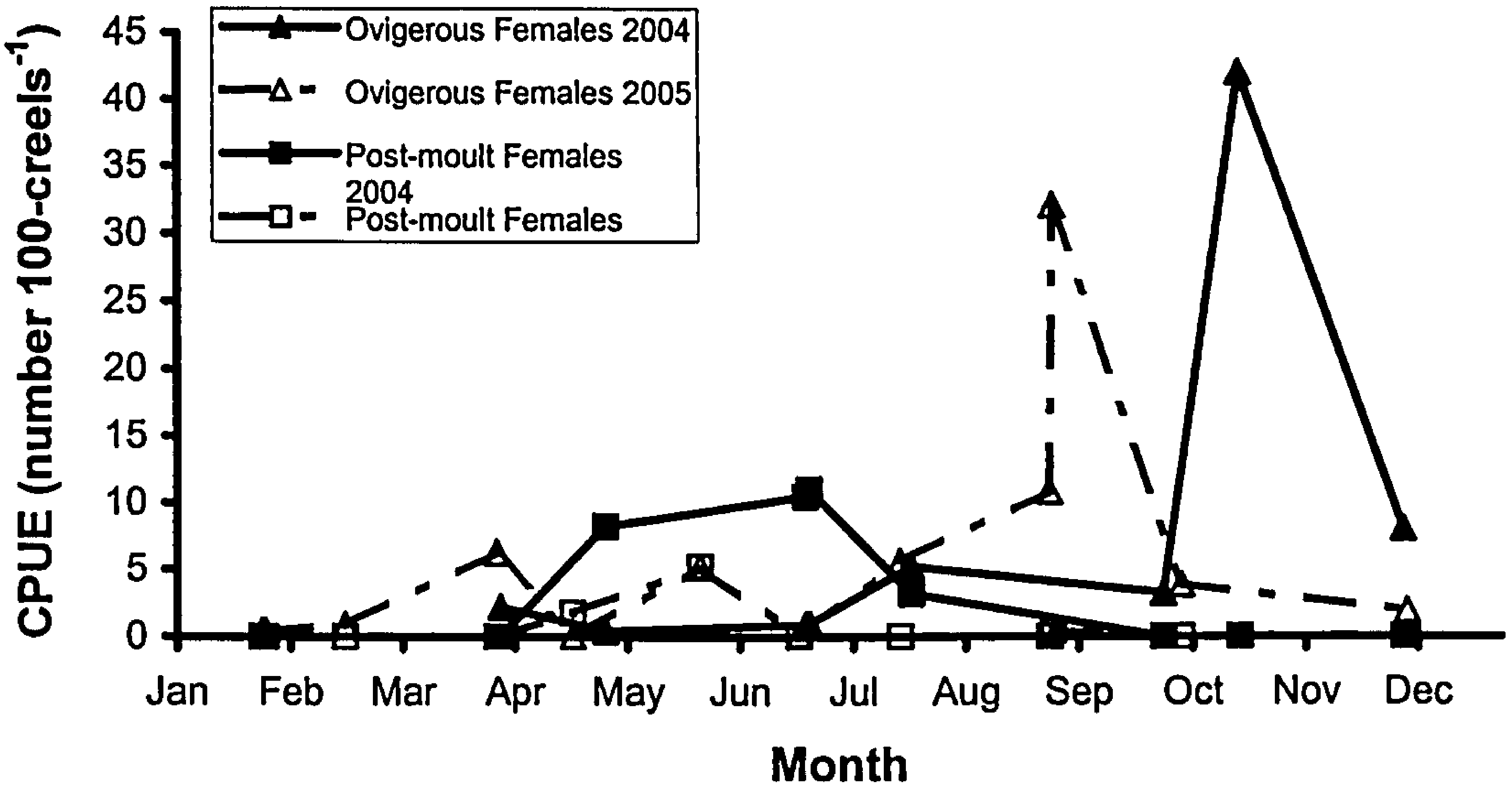


Figure 2.14 - Catch per unit effort of ovigerous and post-moult female *Nephrops* in deep sites (<150 m) with soft mud in the Torridon area in relation to time of year, by year.

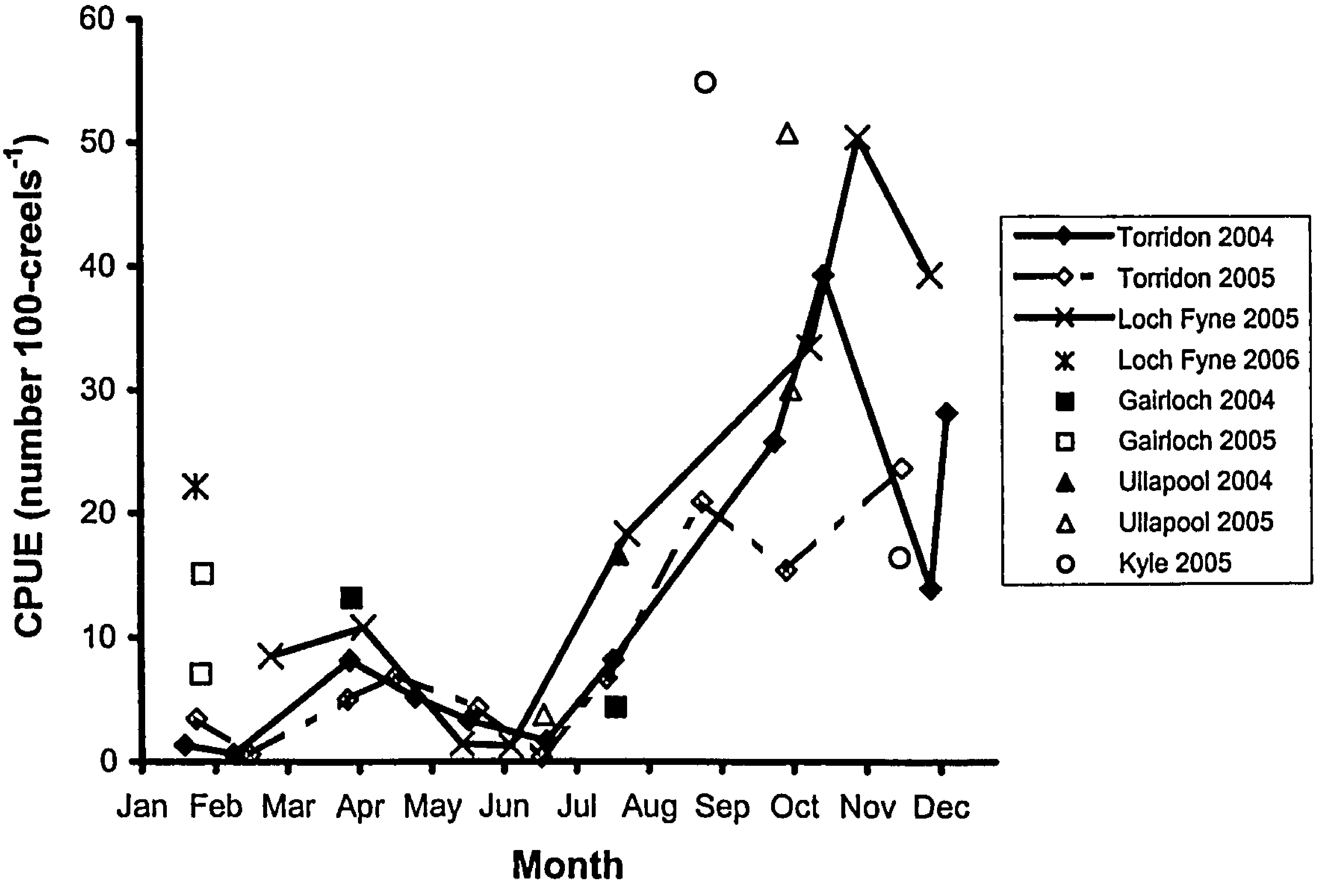


Figure 2.15 - CPUE of ovigerous female *Nephrops* from different areas and years in relation to time of year.

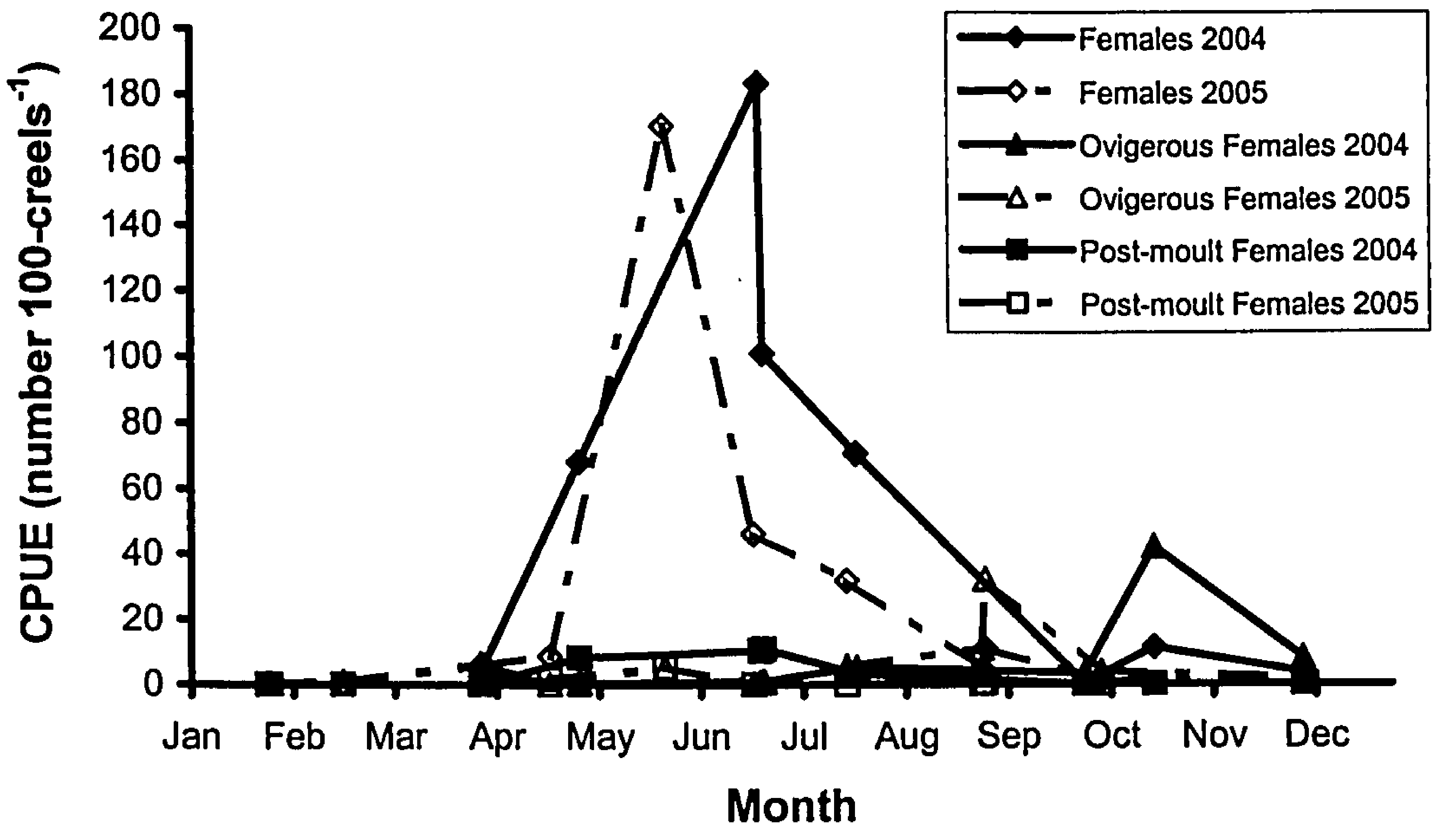


Figure 2.16 – Catch per unit effort of female *Nephrops* in deep mud sites (<150 m) in the Torridon area in relation to time of year, by reproductive and moult status and year. Values are means from on-board catch sampling from one vessel (n=7 563 creels). ("Females" include all non-ovigerous, pre moult females).

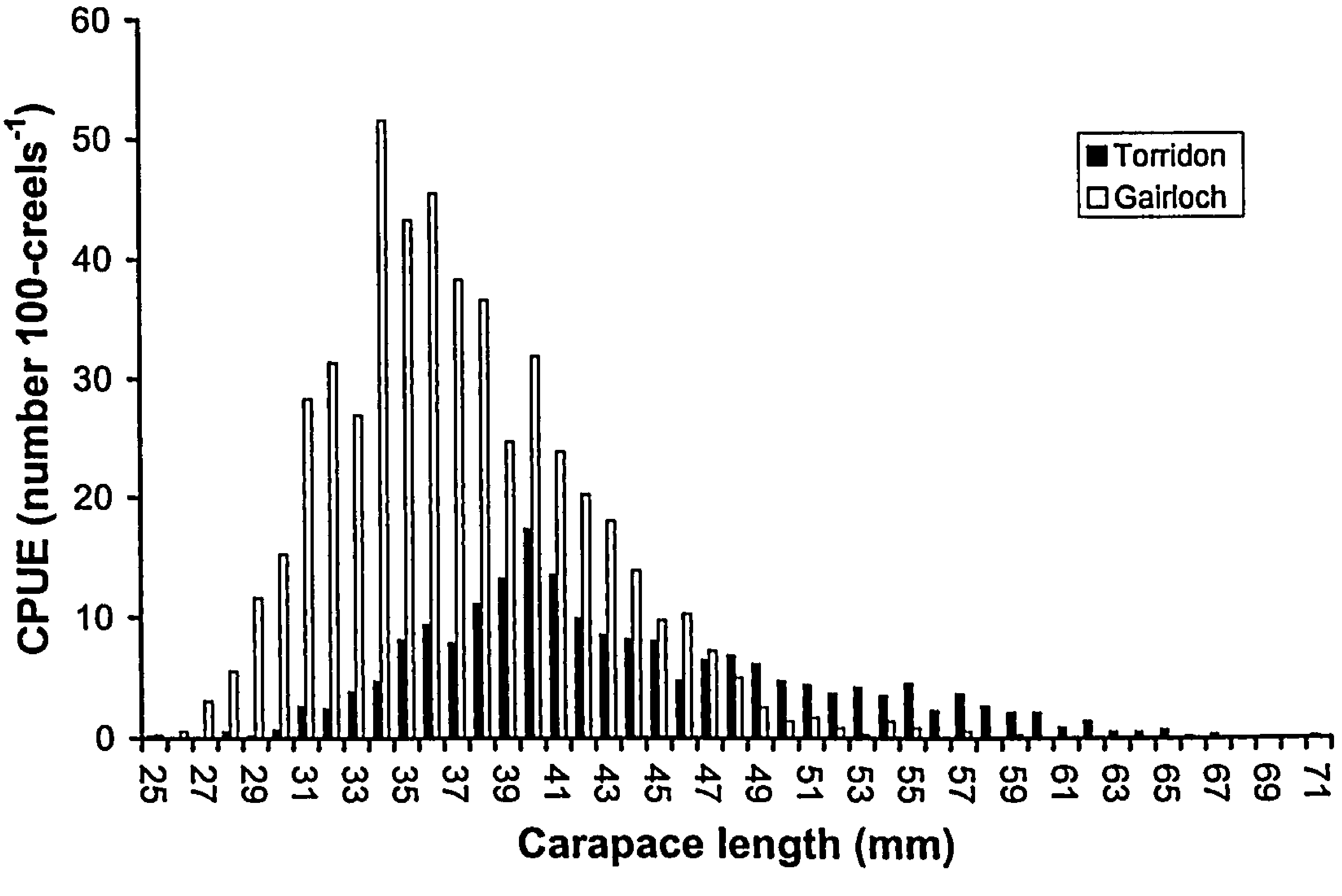


Figure 2.17 - Length-frequency distributions of *Nephrops* standardised to mean number caught per hundred creels (CPUE) from Torridon (n=75) and Gairloch (n=375) in July 2004.

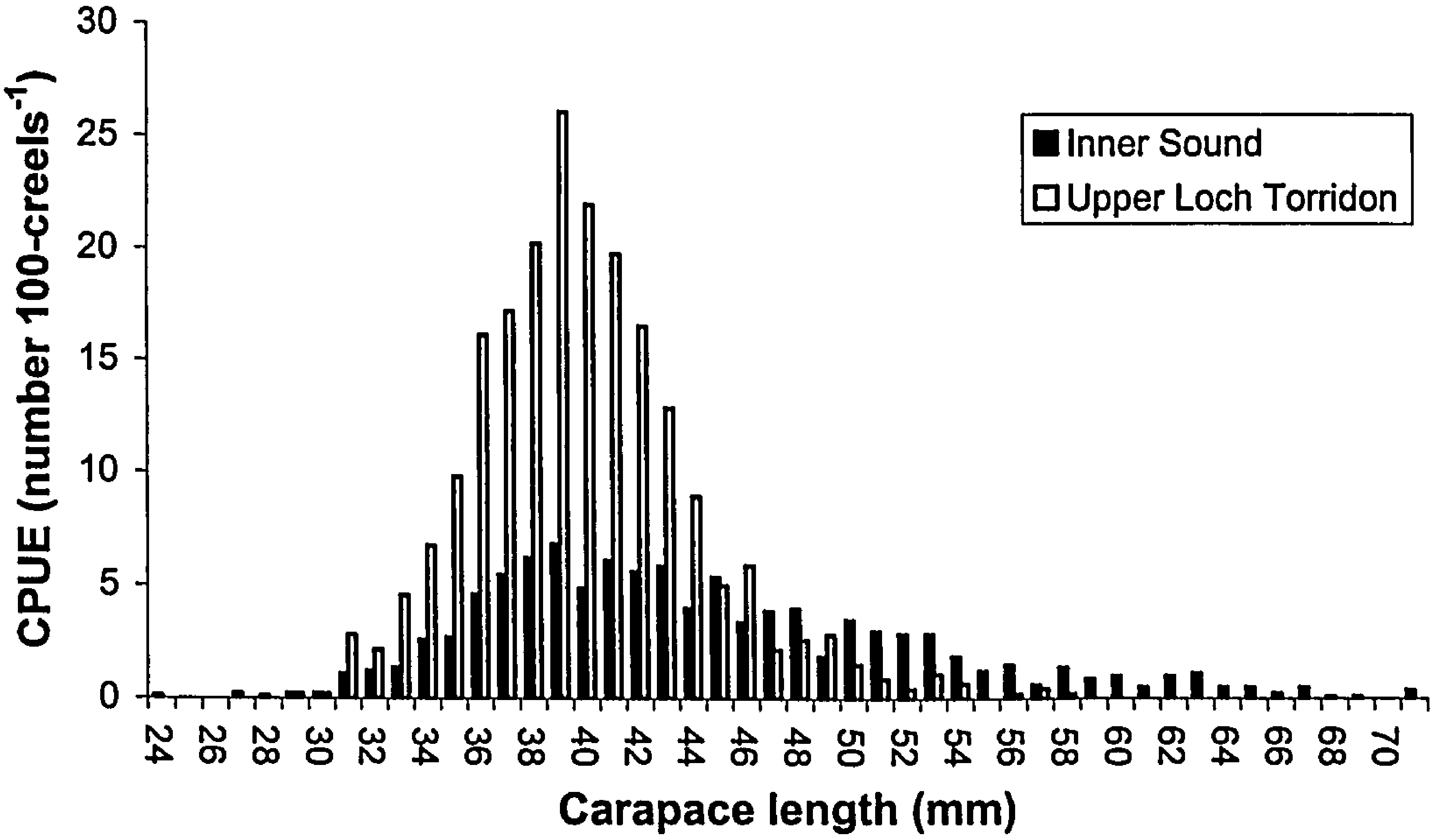


Figure 2.18 - Length-frequency distributions of *Nephrops* standardised to mean number caught per hundred creels (CPUE) from two areas within the Torridon static gear zone: Inner Sound (n=805) and Upper Loch Torridon (n=460) in July 2005.

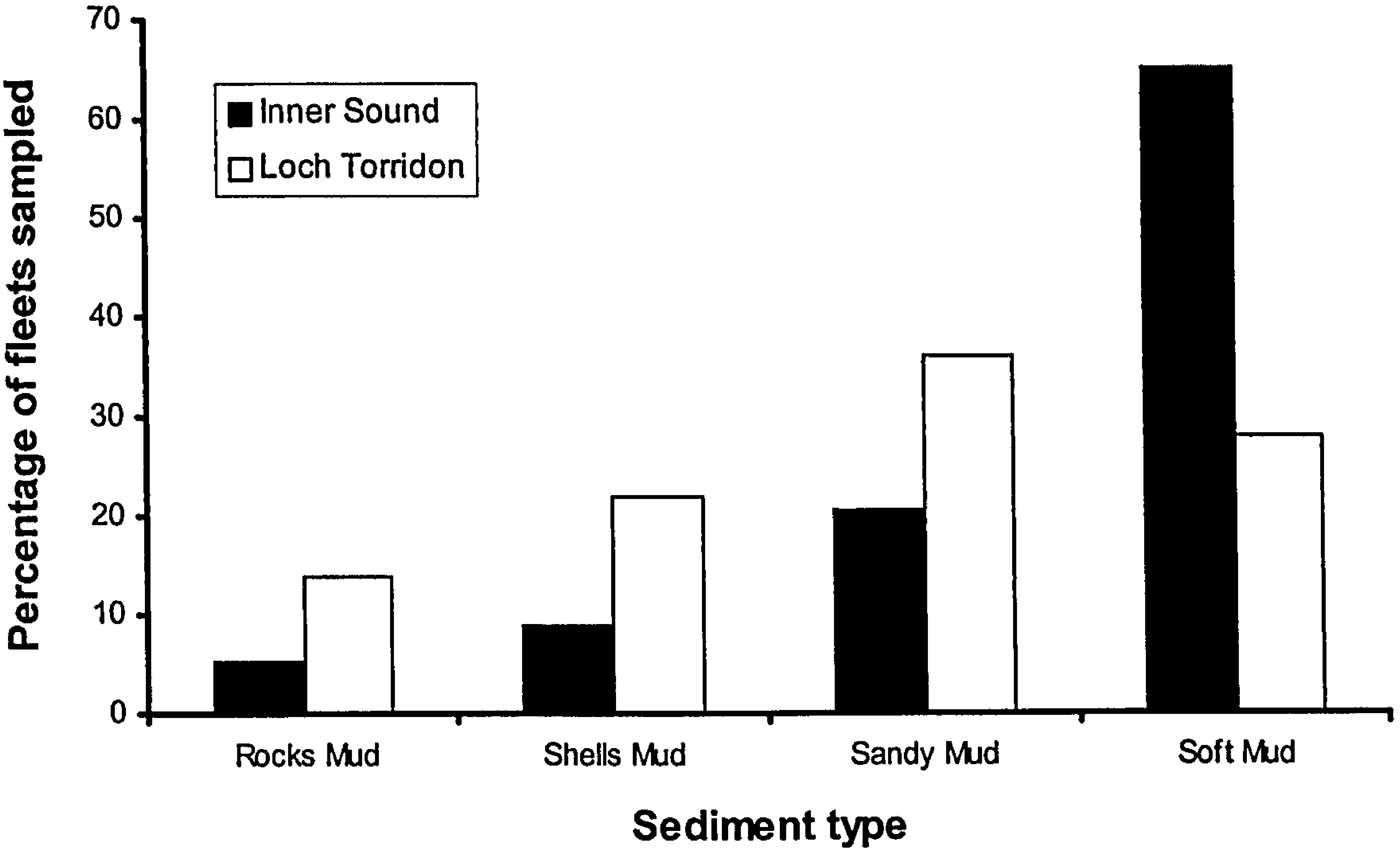


Figure 2.19 – The distribution of sampled creel fleets amoung sediment types, fished in Inner Sound (n=112) and Loch Torridon (n=50).

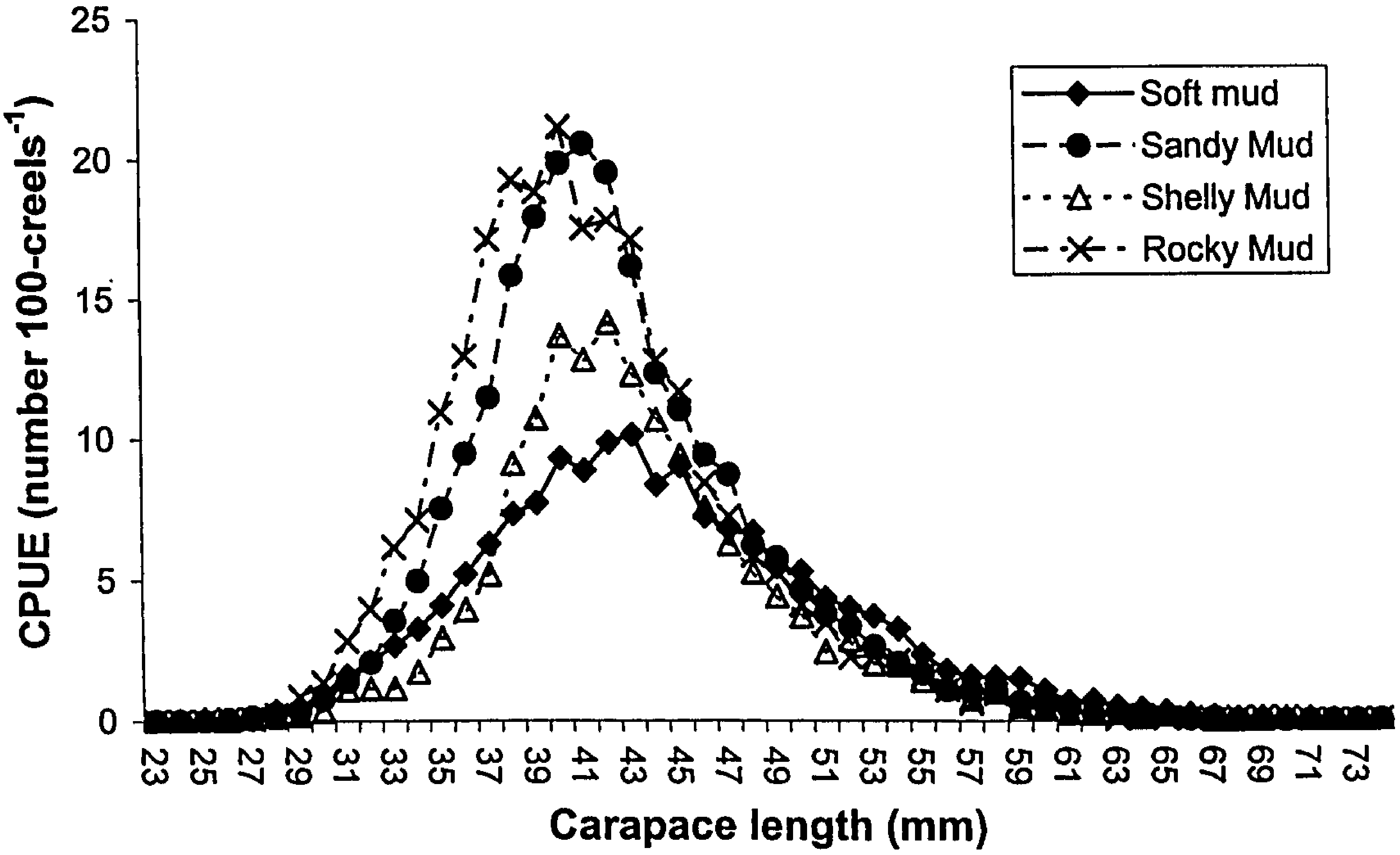


Figure 2.20 - Length-frequency distributions of *Nephrops* standardised to mean number caught per hundred creels (CPUE) from different sediment types in the Torridon static gear zone: soft mud (n=11653) sandy mud (n=4533) shelly mud (n=2399) and rocky mud (n=2173).

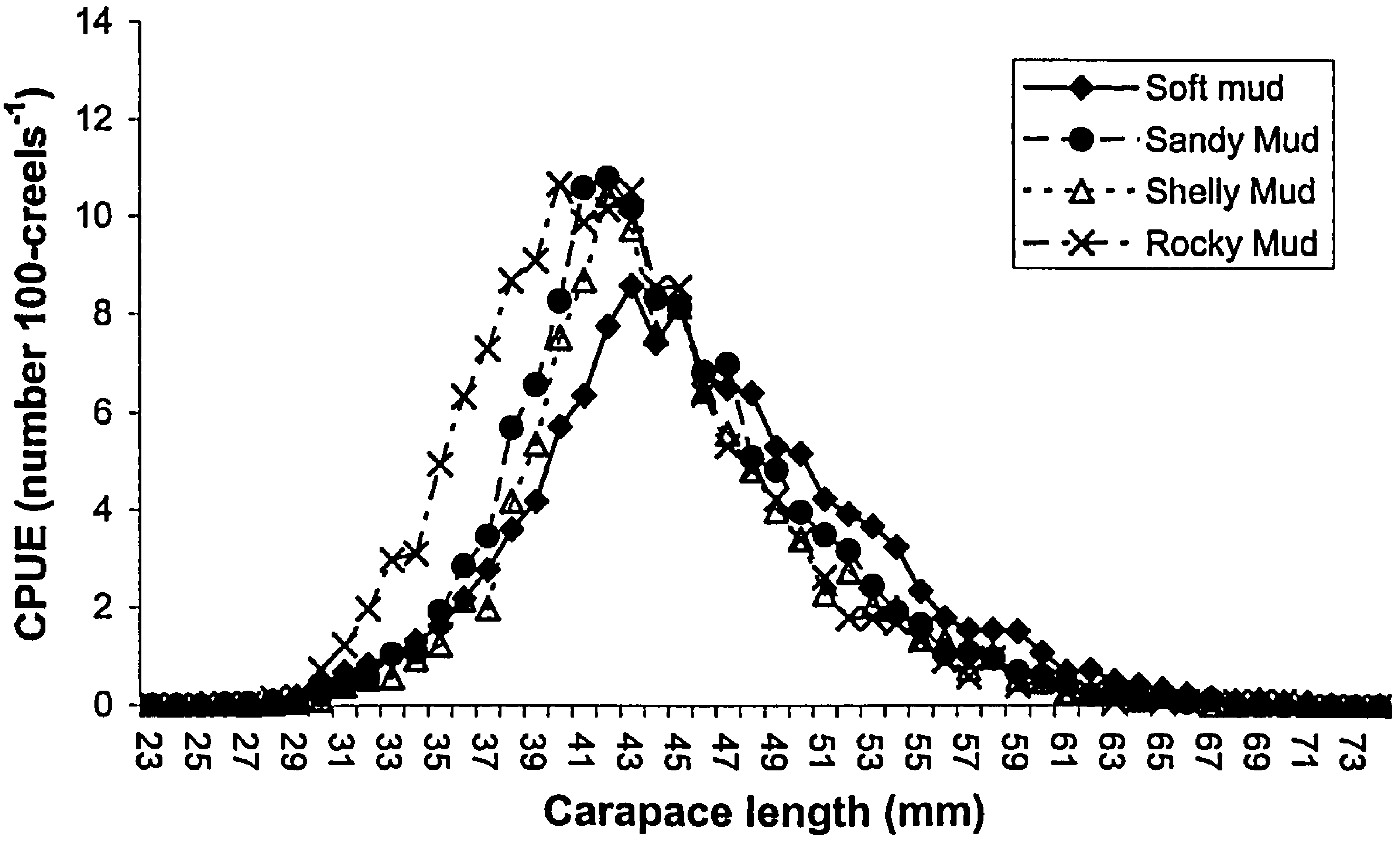


Figure 2.21 - Length-frequency distributions of male *Nephrops* standardised to mean number caught per hundred creels (CPUE) from different sediments in the Torridon static gear zone: soft mud (n=11653) sandy mud (n=4533) shelly mud (n=2399) and rocky mud (n=2173).

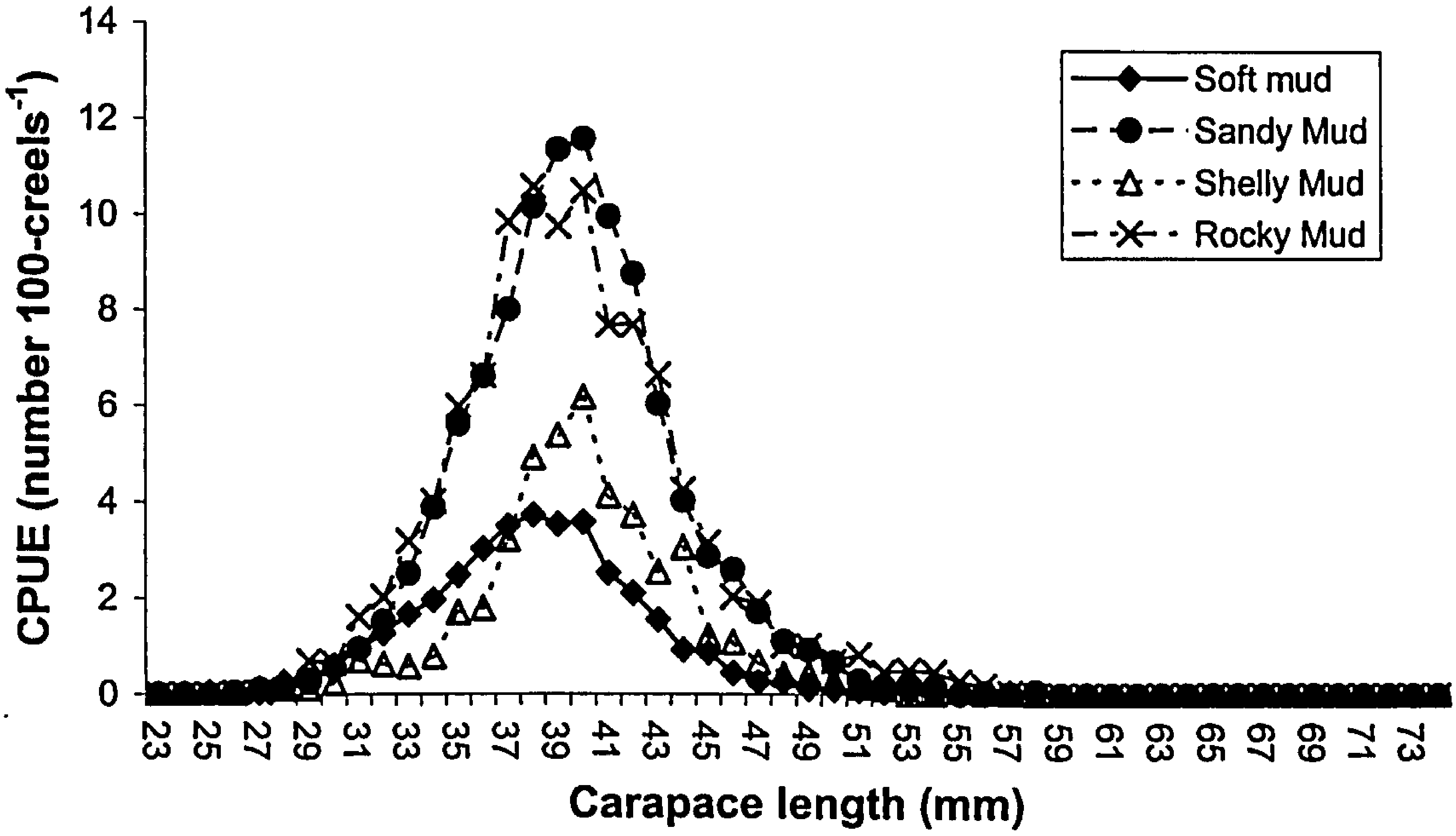


Figure 2.22 - Length-frequency distributions of female *Nephrops* standardised to mean number caught per hundred creels (CPUE) from different sediments in the Torridon static gear zone: soft mud (n=11653) sandy mud (n=4533) shelly mud (n=2399) and rocky mud (n=2173).

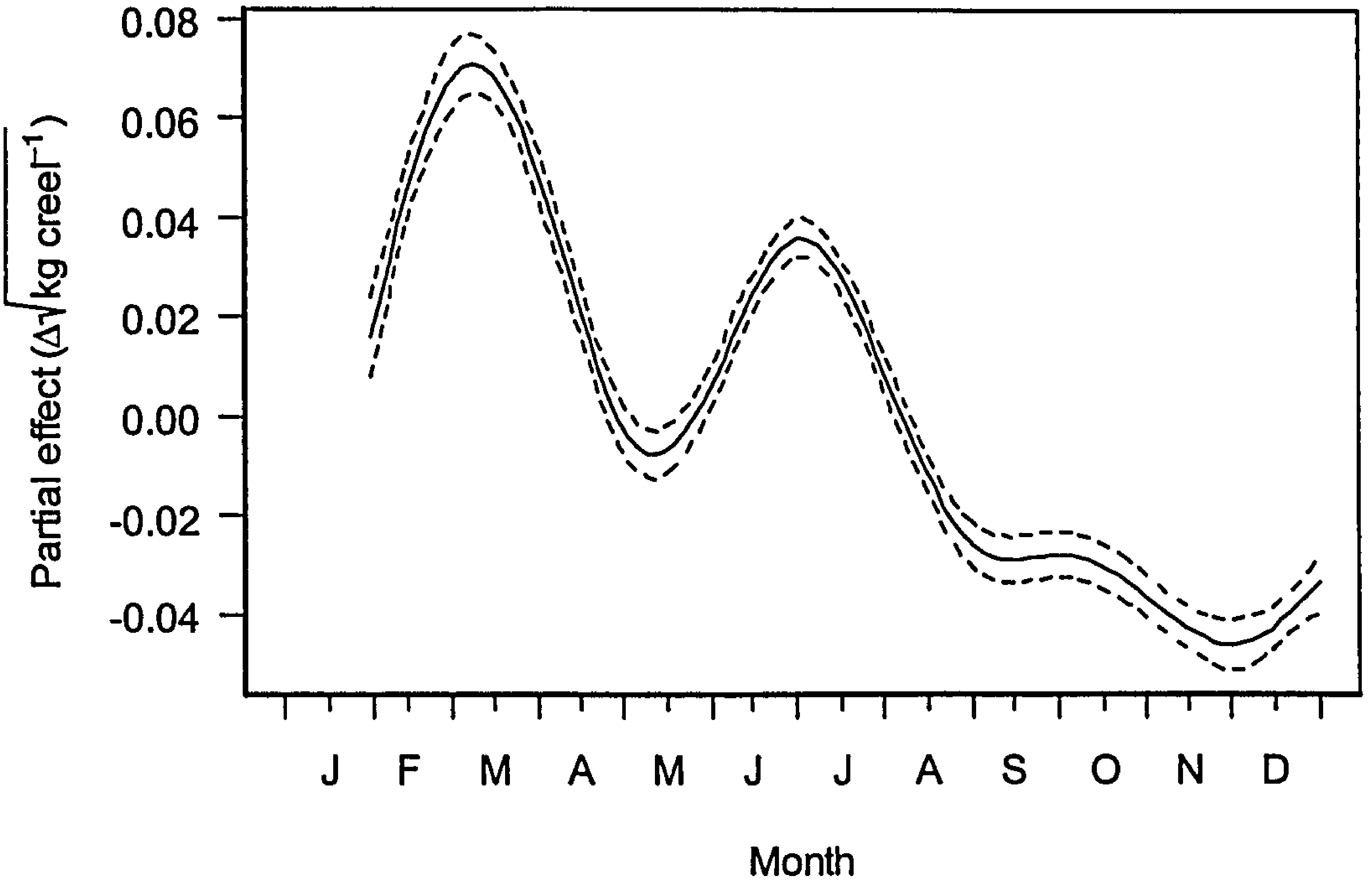


Figure 2.23 - Seasonal changes in LPUE of all categories of *Nephrops* from all ports. The solid curve is the partial effect of a smoothed month term in a generalised additive model (Positive or negative values indicate an increase or decrease in LPUE given the combined effect of the other variables, see text for details); the dashed lines indicate the 95% confidence interval of the month effect.

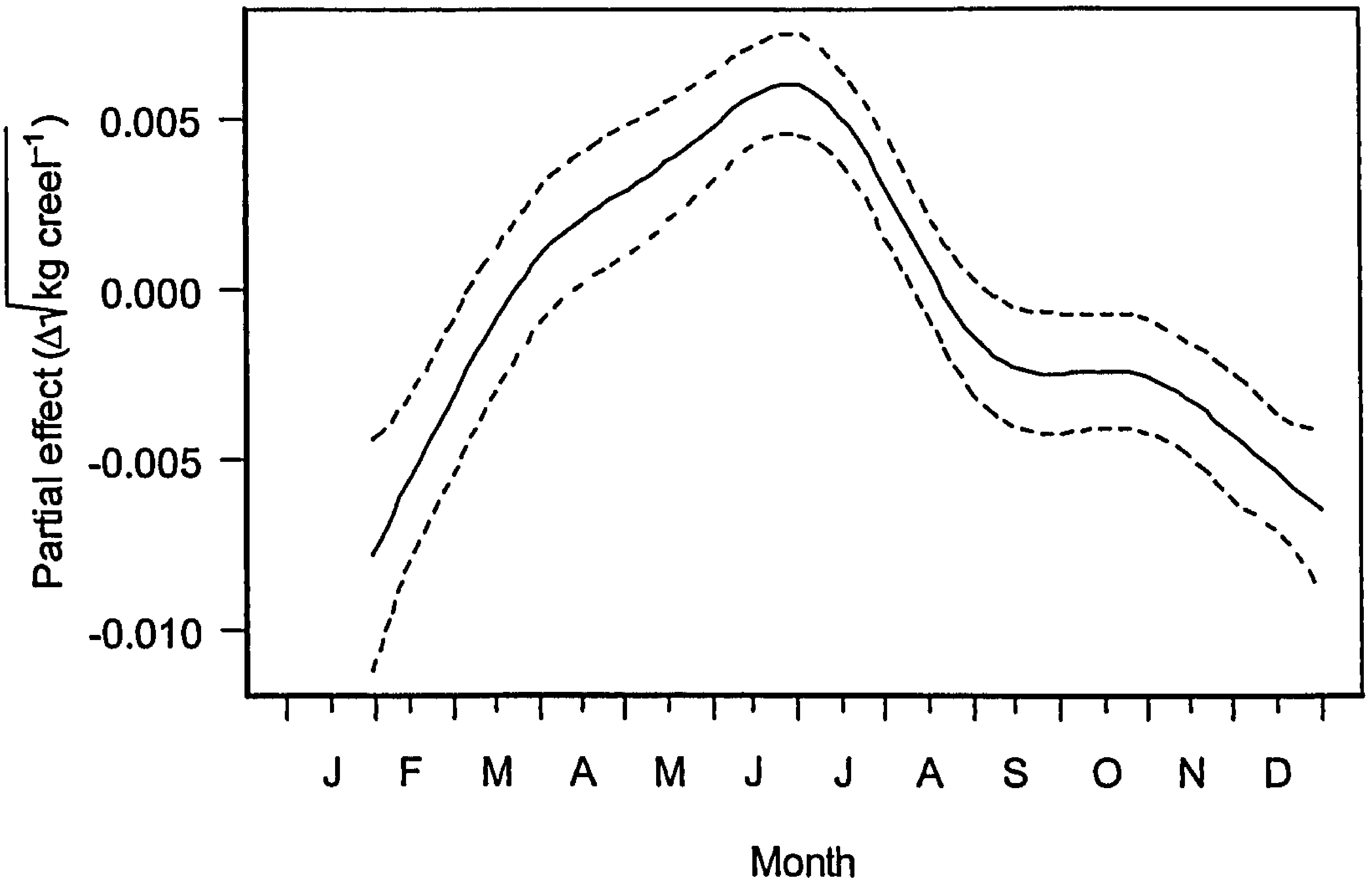


Figure 2.24 - Seasonal changes in LPUE of small *Nephrops* (34–38 mm CL) from all ports. The solid curve is the partial effect of a smoothed month term in a generalised additive model (Positive or negative values indicate an increase or decrease in LPUE given the combined effect of the other variables, see text for details); the dashed lines indicate the 95% confidence interval of the month effect.

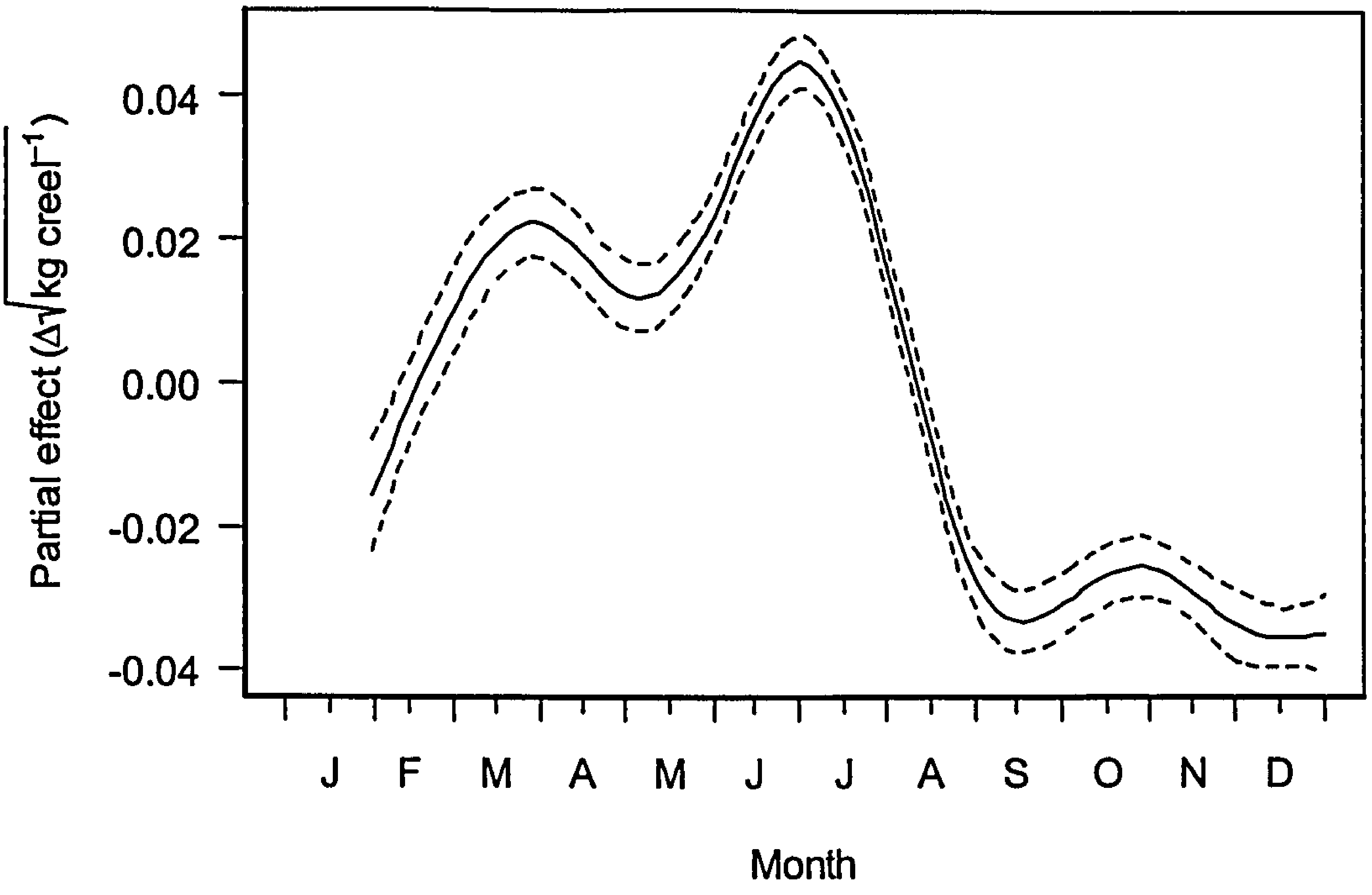


Figure 2.25 - Seasonal changes in LPUE of medium-sized *Nephrops* (39–47 mm CL) from all ports. The solid curve is the partial effect of a smoothed month term in a generalised additive model (Positive or negative values indicate an increase or decrease in LPUE given the combined effect of the other variables, see text for details); the dashed lines indicate the 95% confidence interval of the month effect.

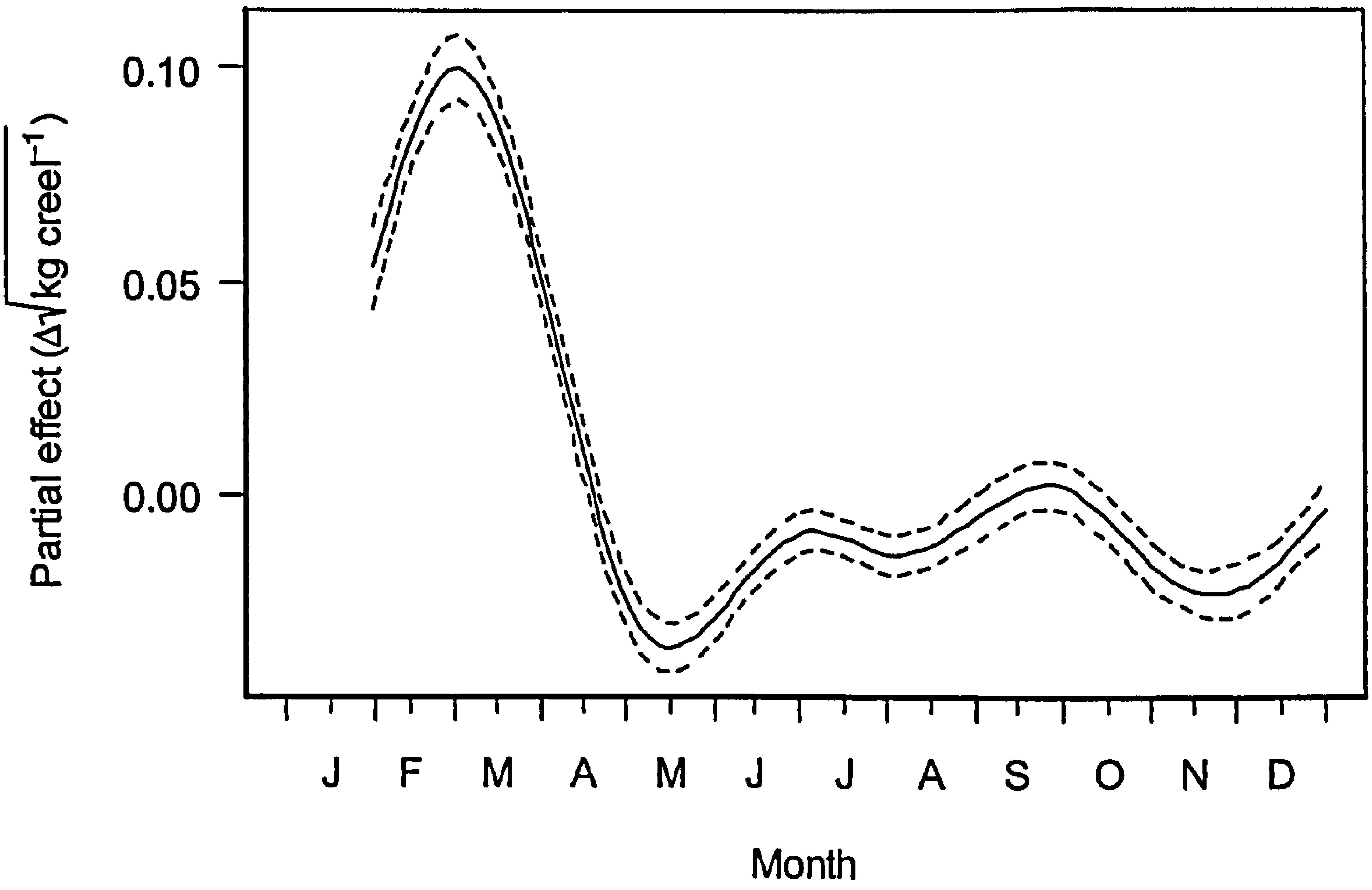


Figure 2.26 - Seasonal changes in LPUE of large *Nephrops* (>48 mm CL) from all ports. The solid curve is the partial effect of a smoothed month term in a generalised additive model (Positive or negative values indicate an increase or decrease in LPUE given the combined effect of the other variables, see text for details); the dashed lines indicate the 95% confidence interval of the month effect.

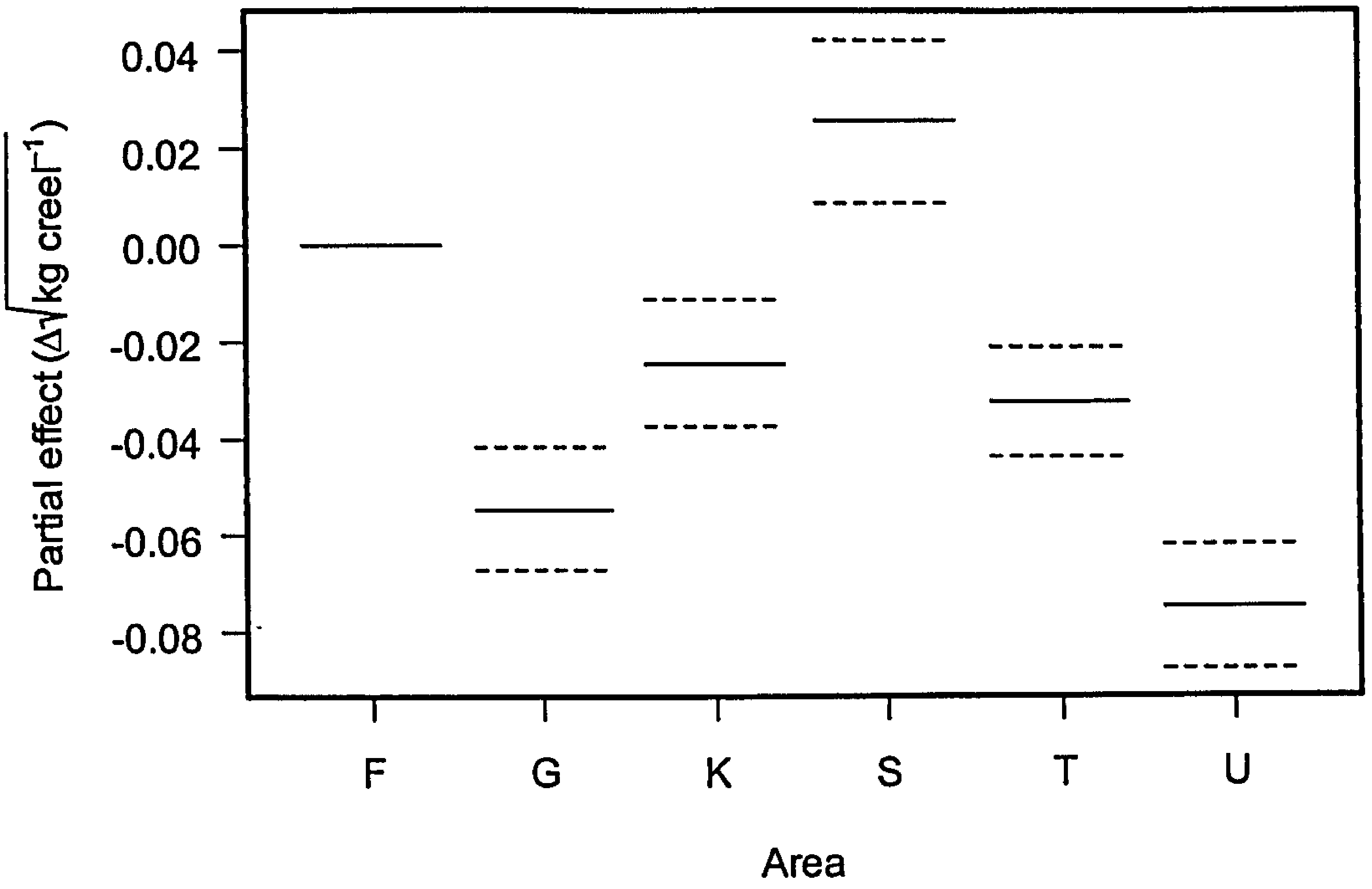


Figure 2.27 - Differences in LPUE of all categories of *Nephrops* among ports (F = Loch Fyne, G = Loch Gairloch, K = Kyle, S = Summer Isles, T = Torridon Area, U= Loch Broom). The solid lines indicate the partial effects of the 'area' term in a generalised additive model (Positive or negative values indicate an increase or decrease in LPUE given the combined effect of the other variables, see text for details); the dashed lines indicate the 95% confidence interval of the area effect.

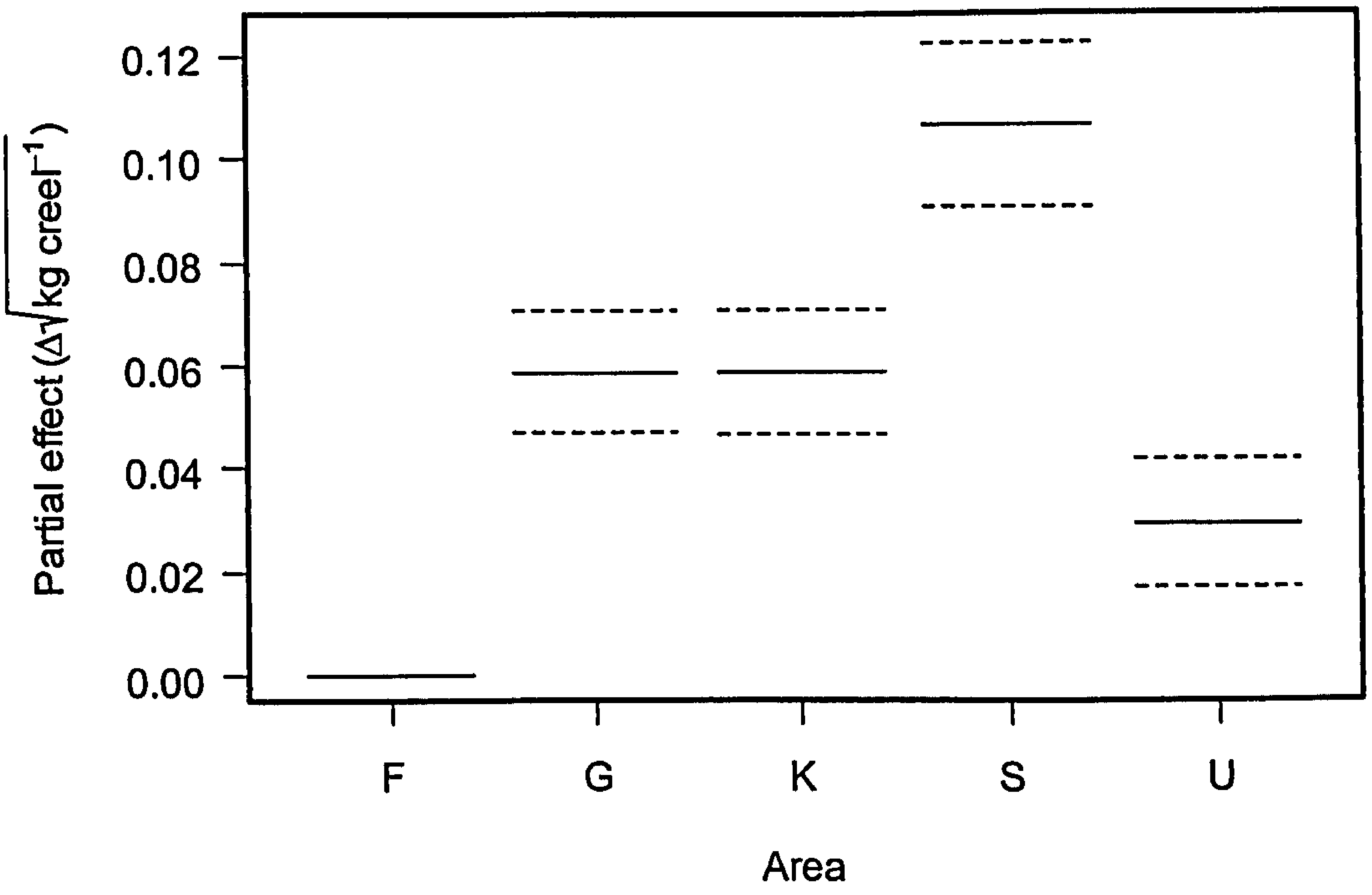


Figure 2.28 - Differences in LPUE of small *Nephrops* among ports (F = Loch Fyne, G = Loch Gairloch, K = Kyle, S = Summer Isles, T = Torridon Area, U= Loch Broom). The solid lines indicate the partial effects of the 'area' term in a generalised additive model (Positive or negative values indicate an increase or decrease in LPUE given the combined effect of the other variables, see text for details); the dashed lines indicate the 95% confidence interval of the area effect.

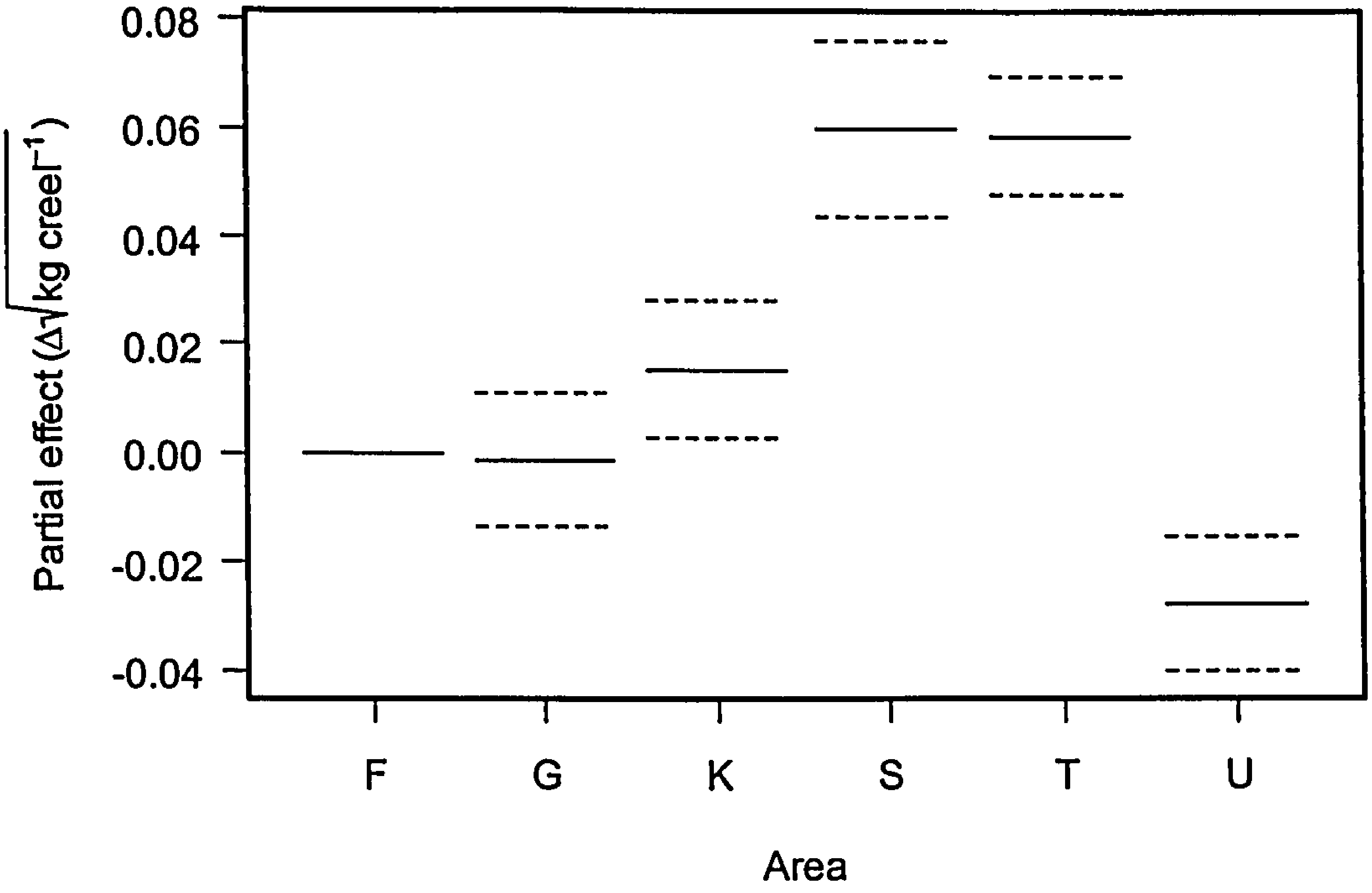


Figure 2.29 - Differences in LPUE of medium *Nephrops* among ports (F = Loch Fyne, G = Loch Gairloch, K = Kyle, S = Summer Isles, T = Torridon Area, U= Loch Broom). The solid lines indicate the partial effects of the 'area' term in a generalised additive model (Positive or negative values indicate an increase or decrease in LPUE given the combined effect of the other variables, see text for details); the dashed lines indicate the 95% confidence interval of the area effect.

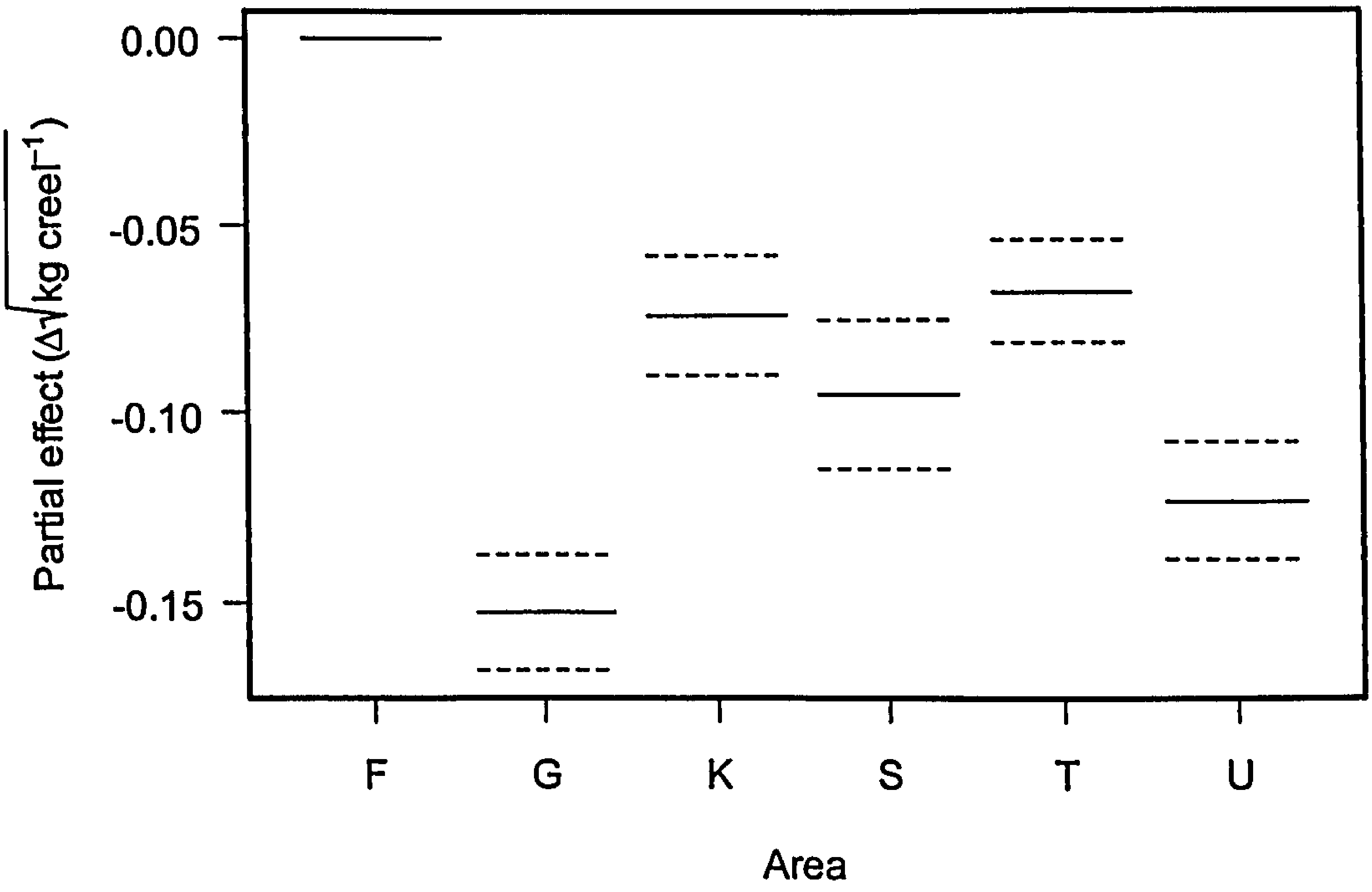


Figure 2.30 - Differences in LPUE of large *Nephrops* among ports (F = Loch Fyne, G = Loch Gairloch, K = Kyle, S = Summer Isles, T = Torridon Area, U = Loch Broom). The solid lines indicate the partial effects of the 'area' term in a generalised additive model (Positive or negative values indicate an increase or decrease in LPUE given the combined effect of the other variables, see text for details); the dashed lines indicate the 95% confidence interval of the area effect.

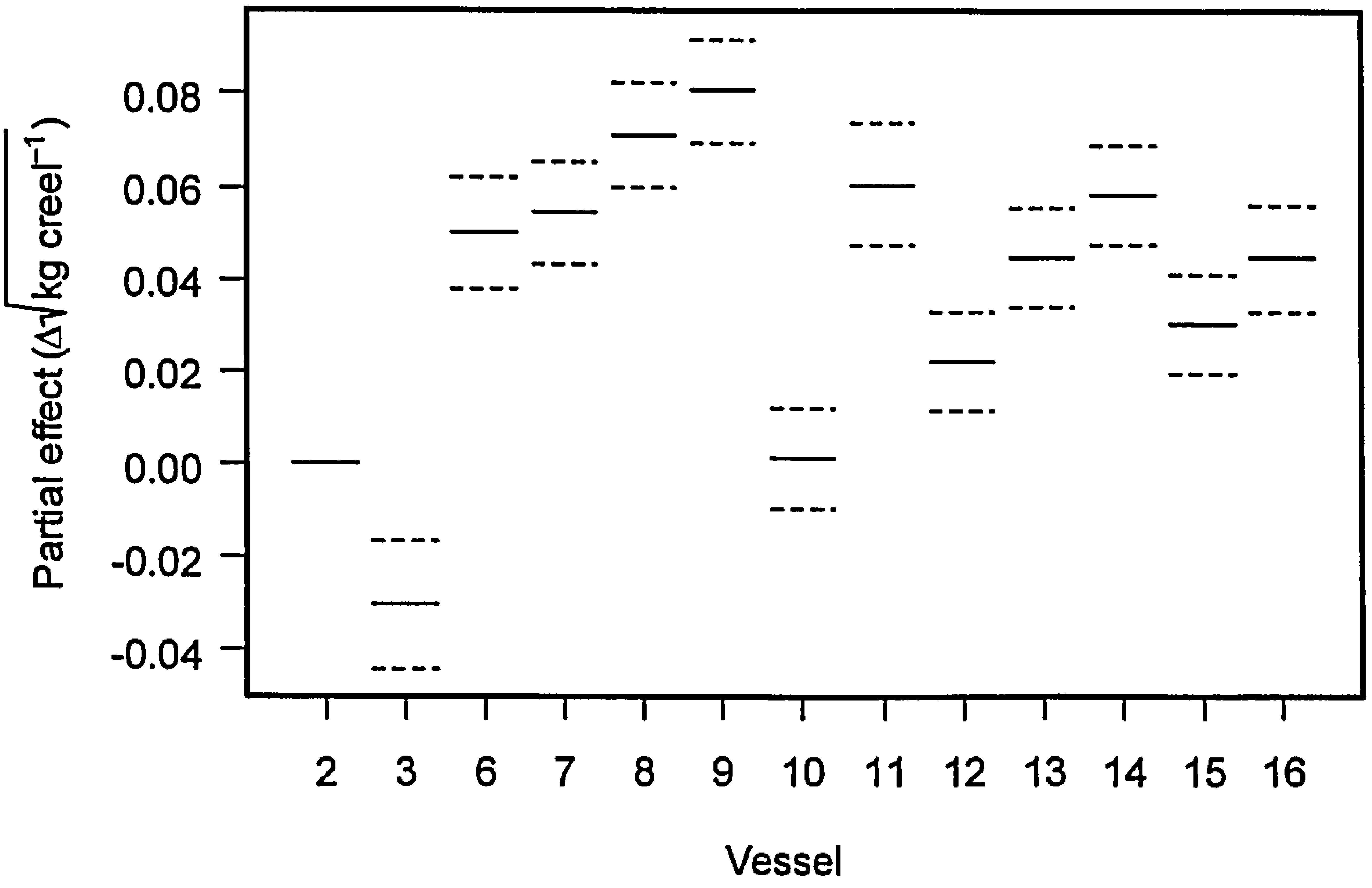


Figure 2.31 - Regional changes in LPUE of all categories of Nephrops from different boats within the Torridon area. The solid lines represent the partial effect of a 'vessel' term in a generalised additive model (Positive or negative values indicate an increase or decrease in LPUE given the combined effect of the other variables, see text for details); the dashed lines indicate the 95% confidence interval of the 'vessel' effect.

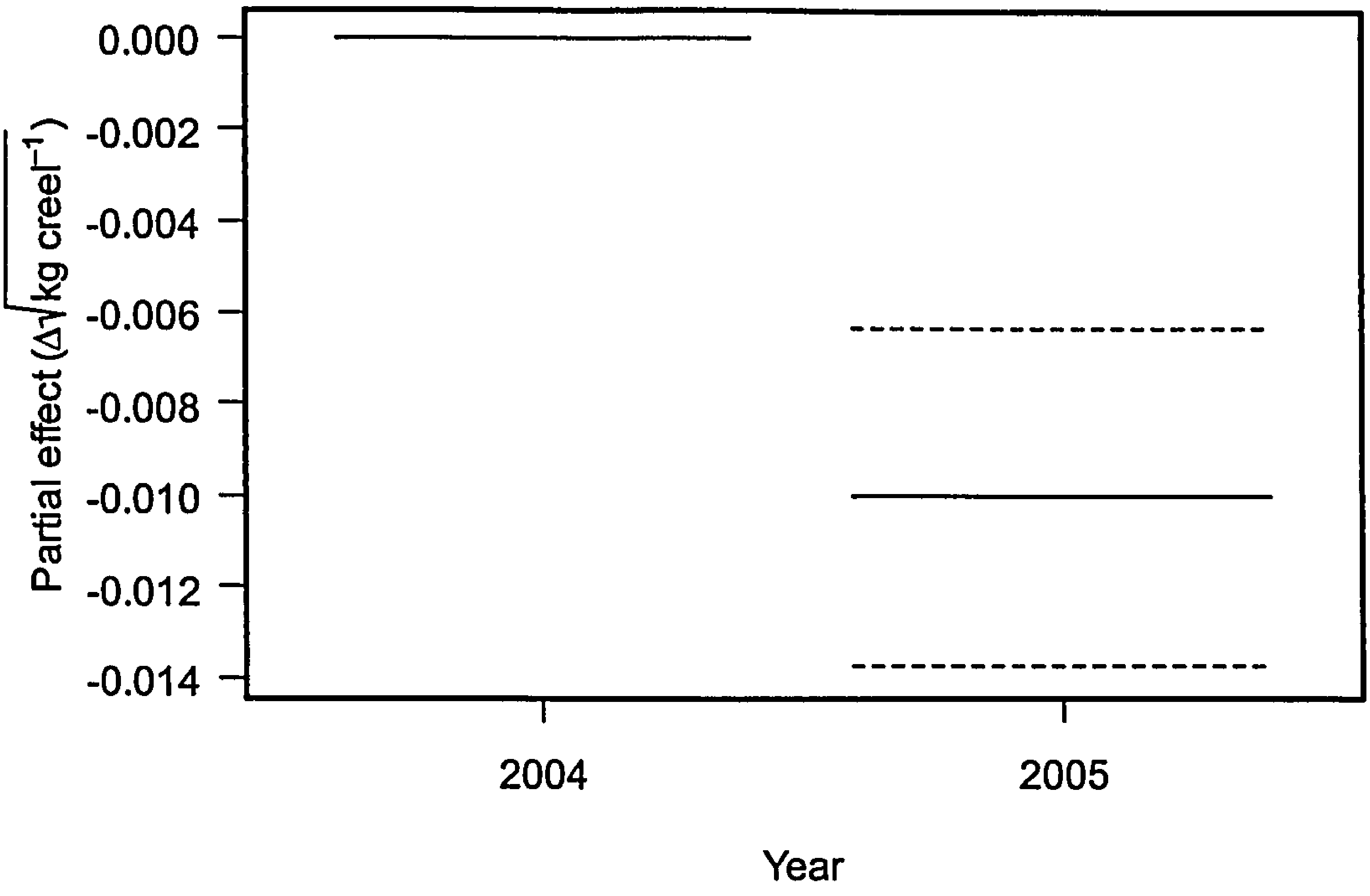


Figure 2.32 - Annual difference in LPUE of all categories of *Nephrops* from all ports. The solid lines represent the partial effect of a 'year' term in a generalised additive model (Positive or negative values indicate an increase or decrease in LPUE given the combined effect of the other variables, see text for details); the dashed lines indicate the 95% confidence interval of the year effect.

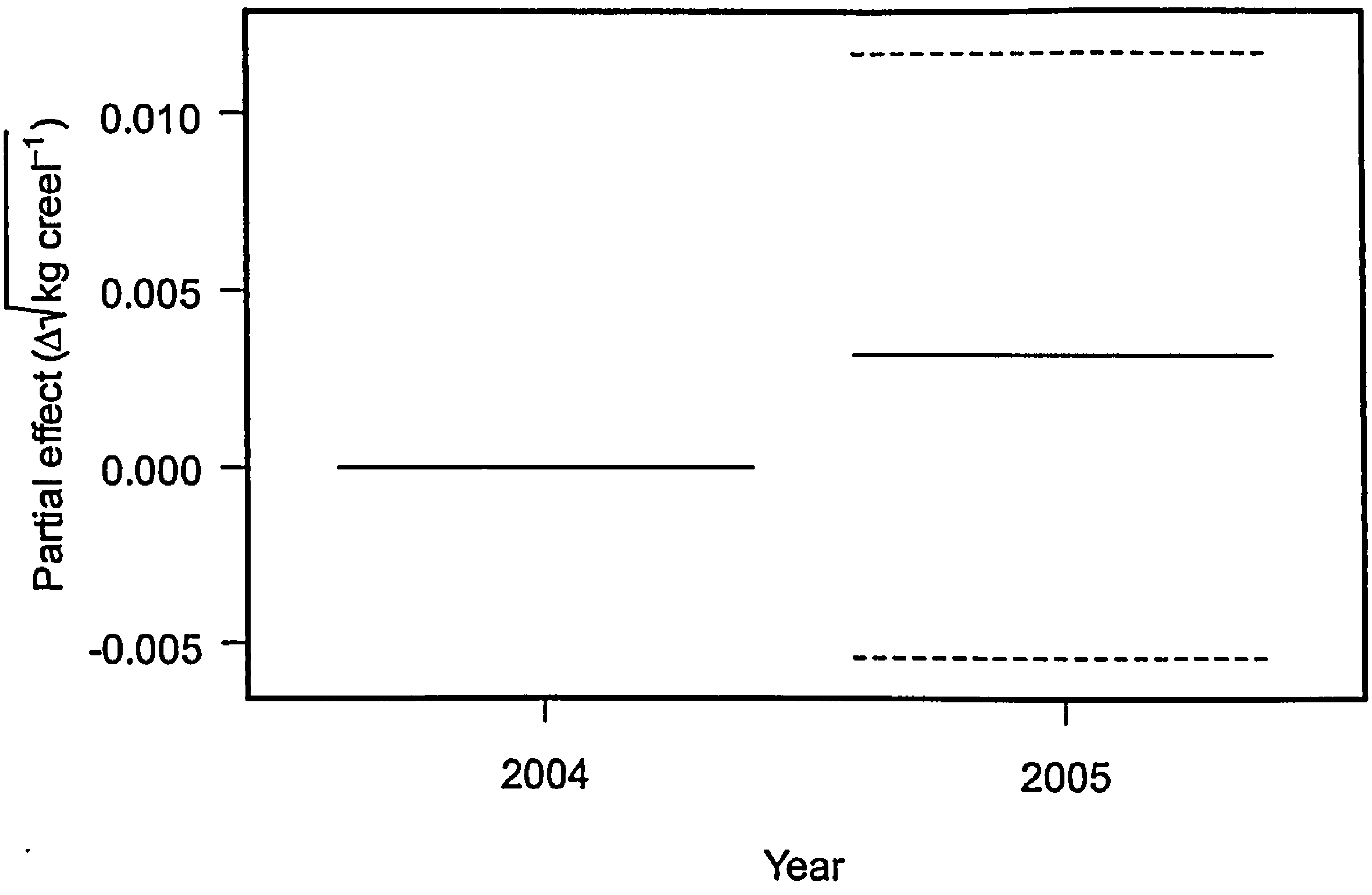


Figure 2.33 - Annual difference in LPUE of small *Nephrops* (34–38 mm CL) from all ports. The solid lines represent the partial effect of a 'year' term in a generalised additive model (Positive or negative values indicate an increase or decrease in LPUE given the combined effect of the other variables, see text for details); the dashed lines indicate the 95% confidence interval of the year effect.

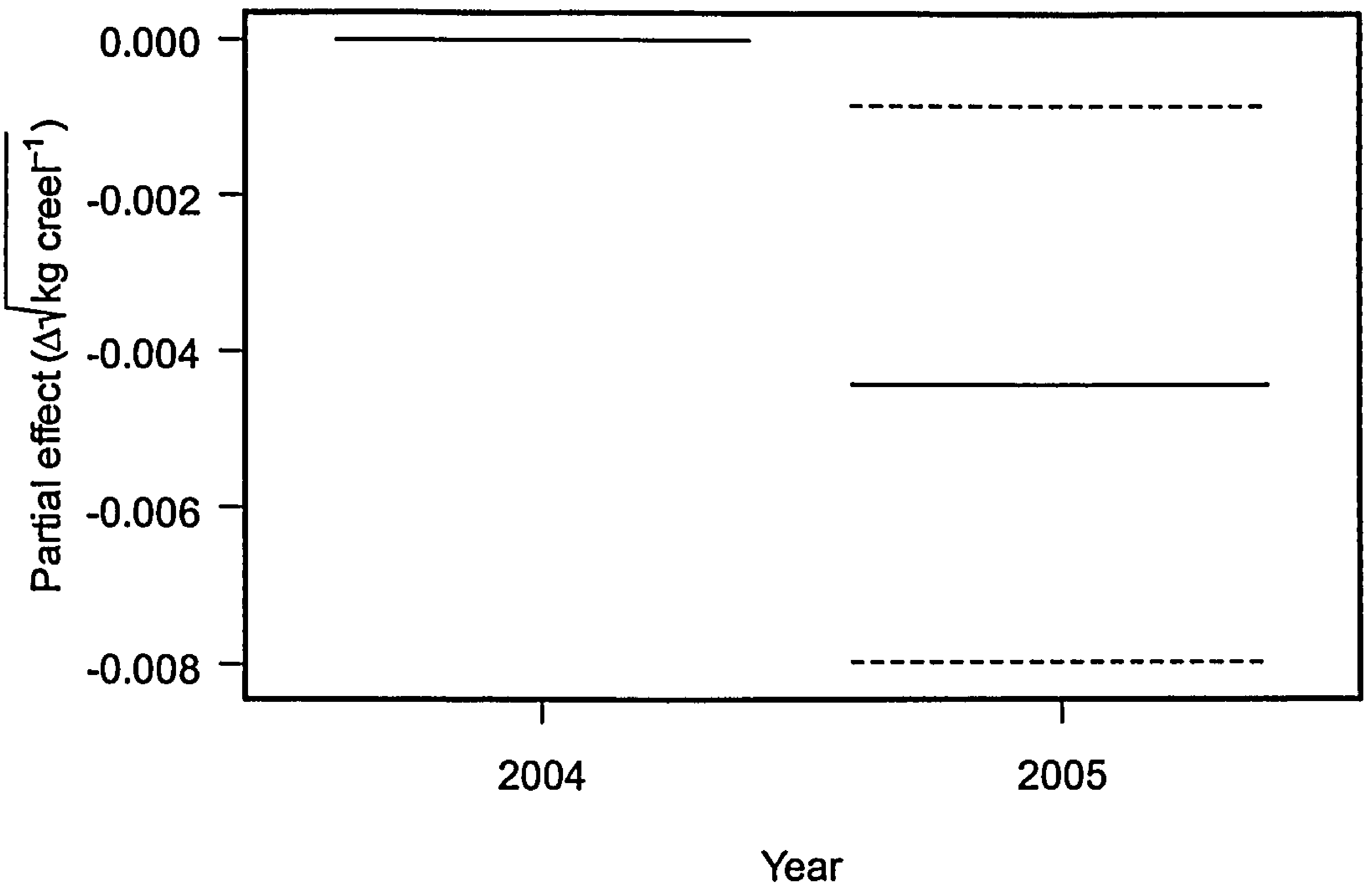


Figure 2.34 - Annual difference in LPUE of medium *Nephrops* (39–47 mm CL) from all ports. The solid lines represent the partial effect of a 'year' term in a generalised additive model (Positive or negative values indicate an increase or decrease in LPUE given the combined effect of the other variables, see text for details); the dashed lines indicate the 95% confidence interval of the year effect.

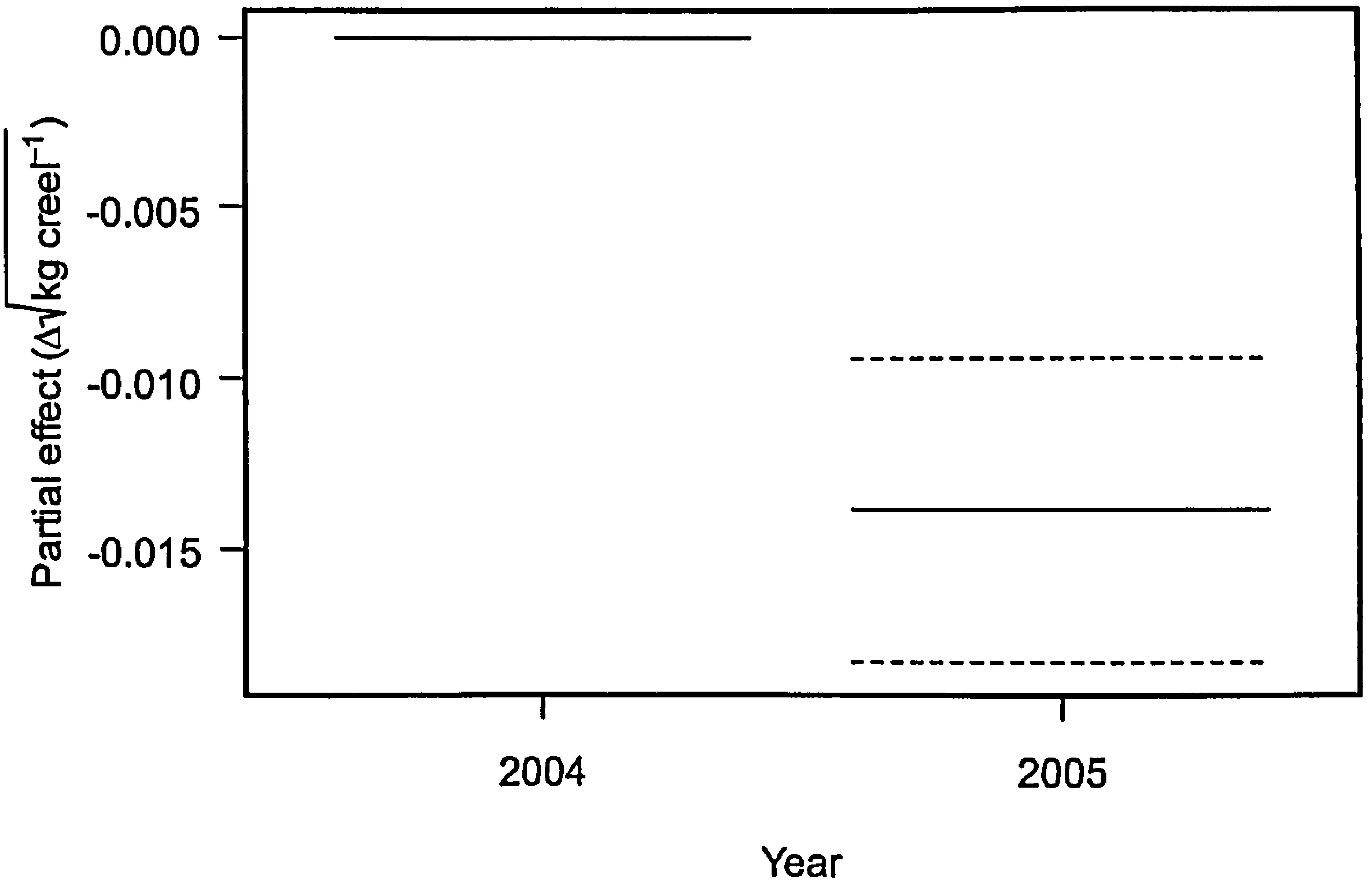


Figure 2.35 - Annual difference in LPUE of large *Nephrops* (>48 mm CL) from all ports. The solid lines represent the partial effect of a 'year' term in a generalised additive model (Positive or negative values indicate an increase or decrease in LPUE given the combined effect of the other variables, see text for details); the dashed lines indicate the 95% confidence interval of the year effect.

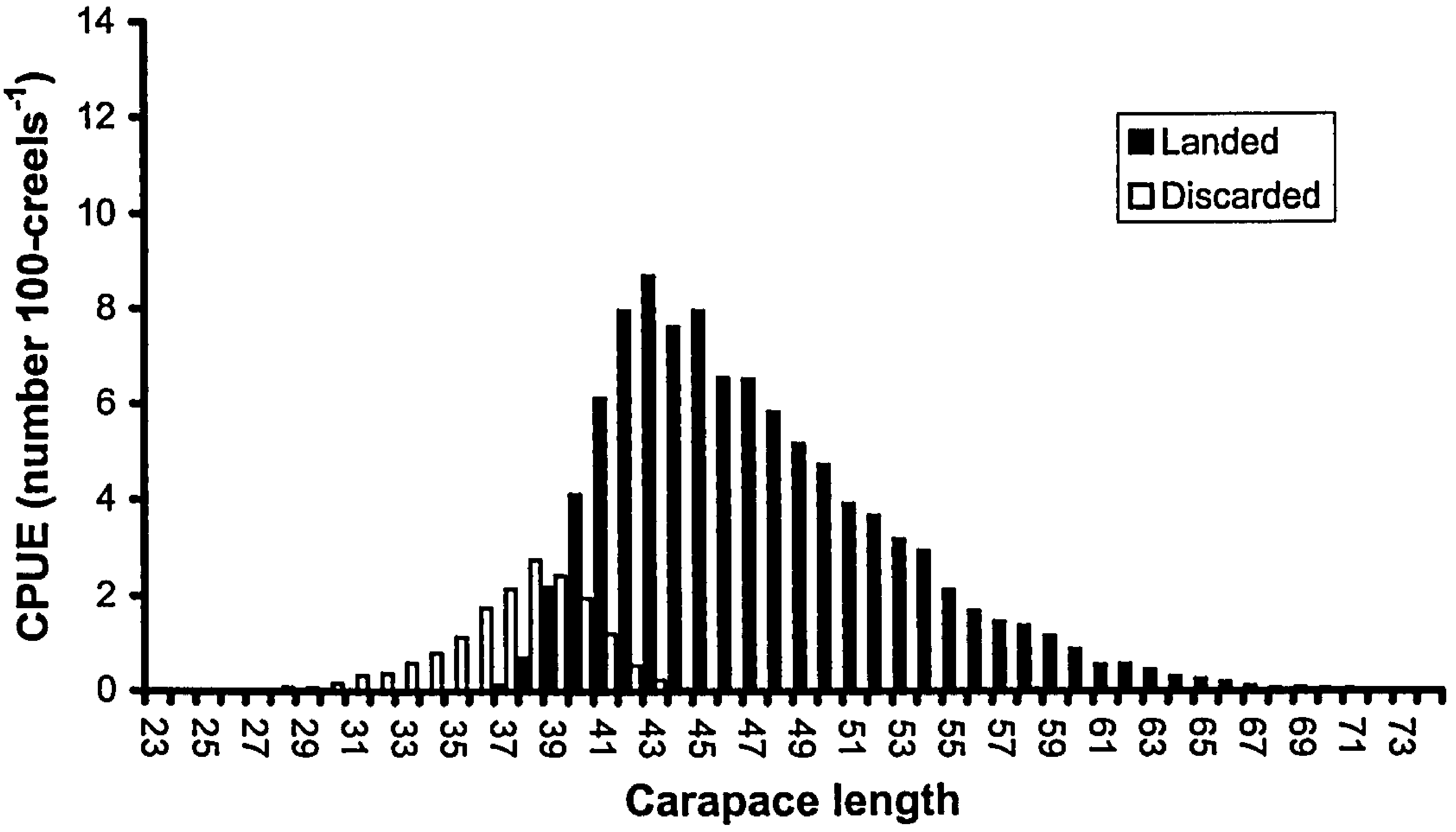


Figure 2.36 - Length-frequency distribution of CPUE data for male *Nephrops* landings and discards from Torridon 2004–2005 (Torridon n=17 535 creels).

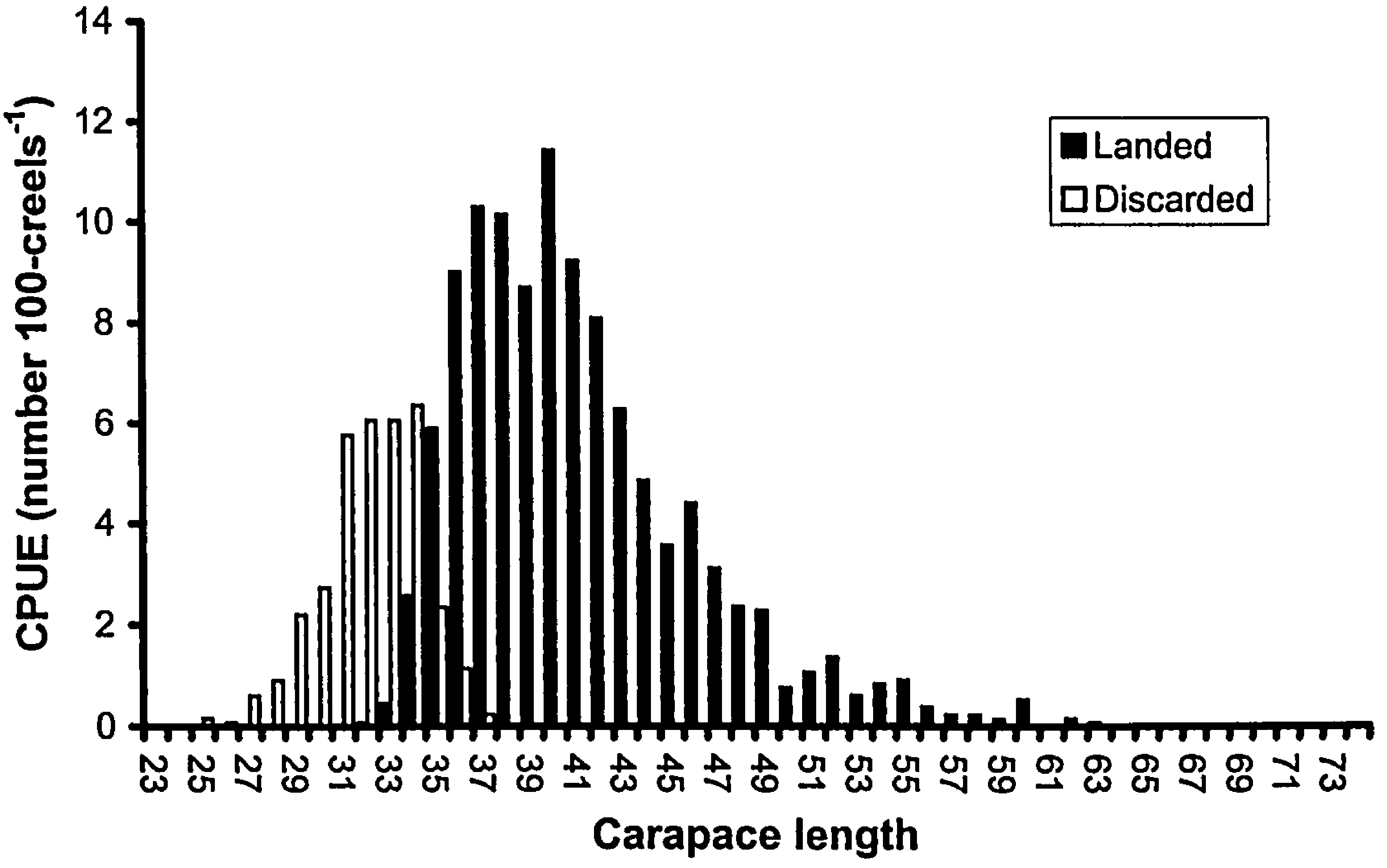


Figure 2.37 - Length-frequency distribution of CPUE data for male *Nephrops* landings and discards from Ullapool 2004–2005 (Ullapool n=1319 creels).

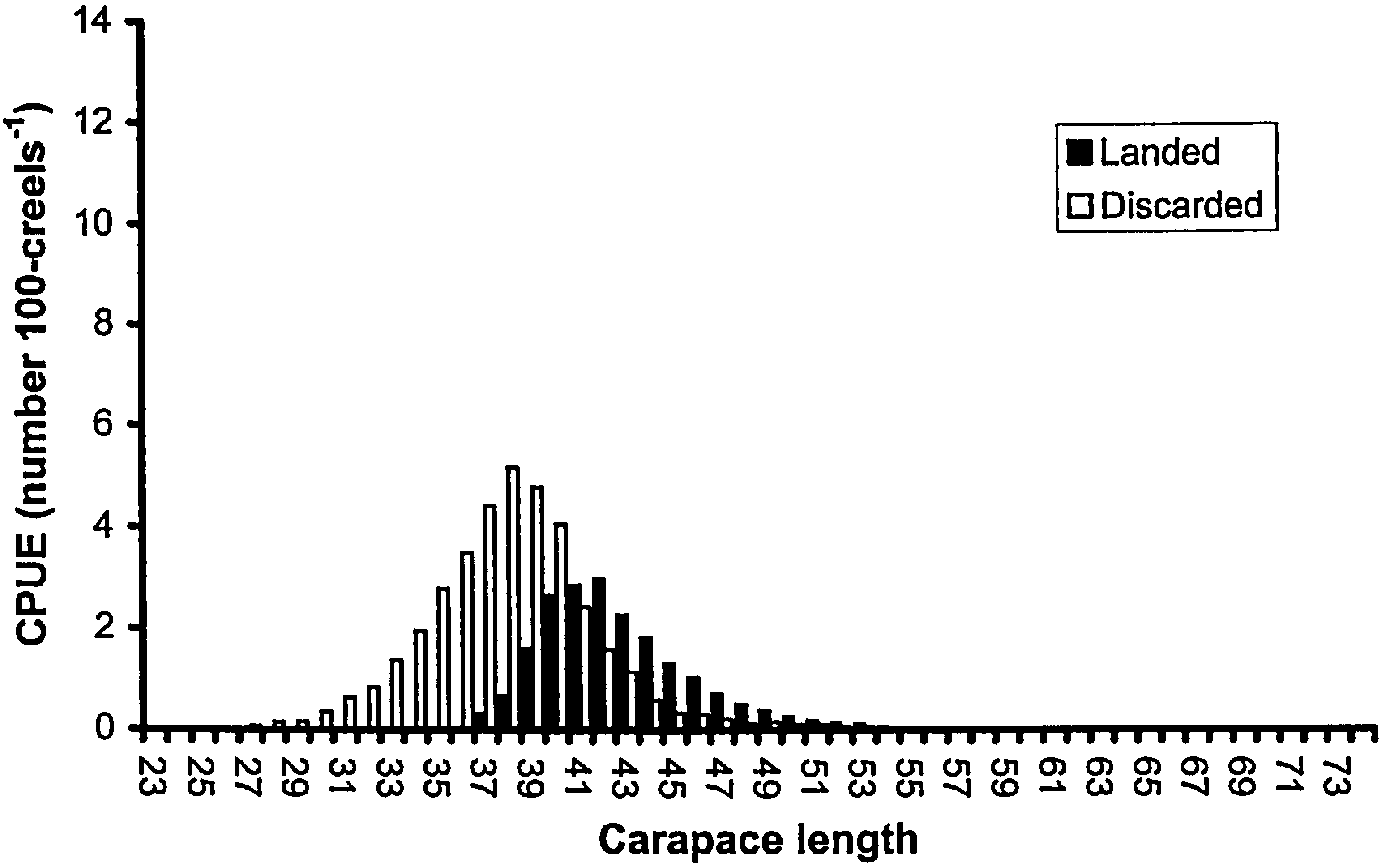


Figure 2.38 - Length-frequency distribution of CPUE data for female *Nephrops* landings and discards from Torridon 2004–2005.

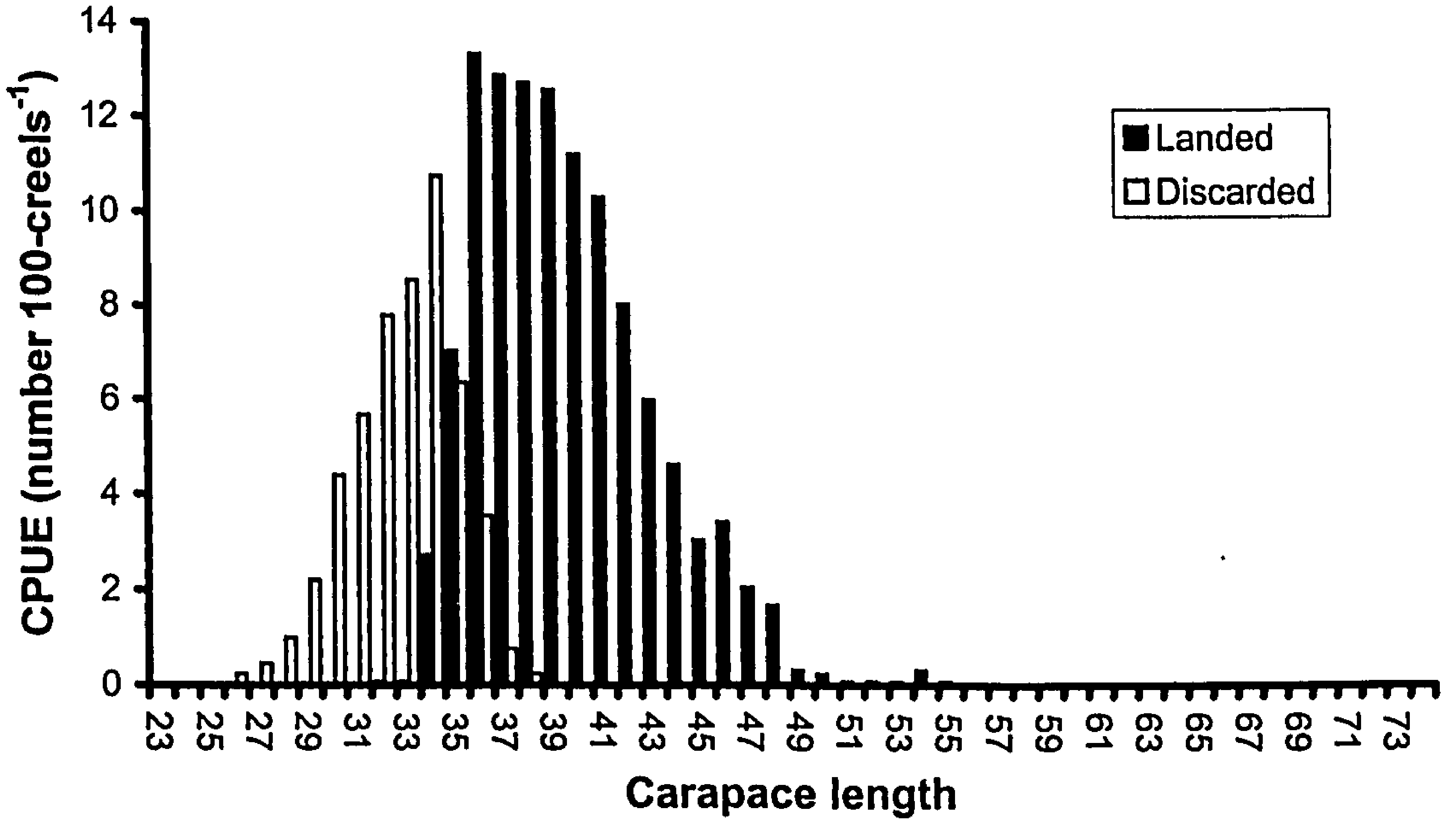


Figure 2.39 - Length-frequency distribution of CPUE data for female *Nephrops* landings and discards from Ullapool 2004–2005.

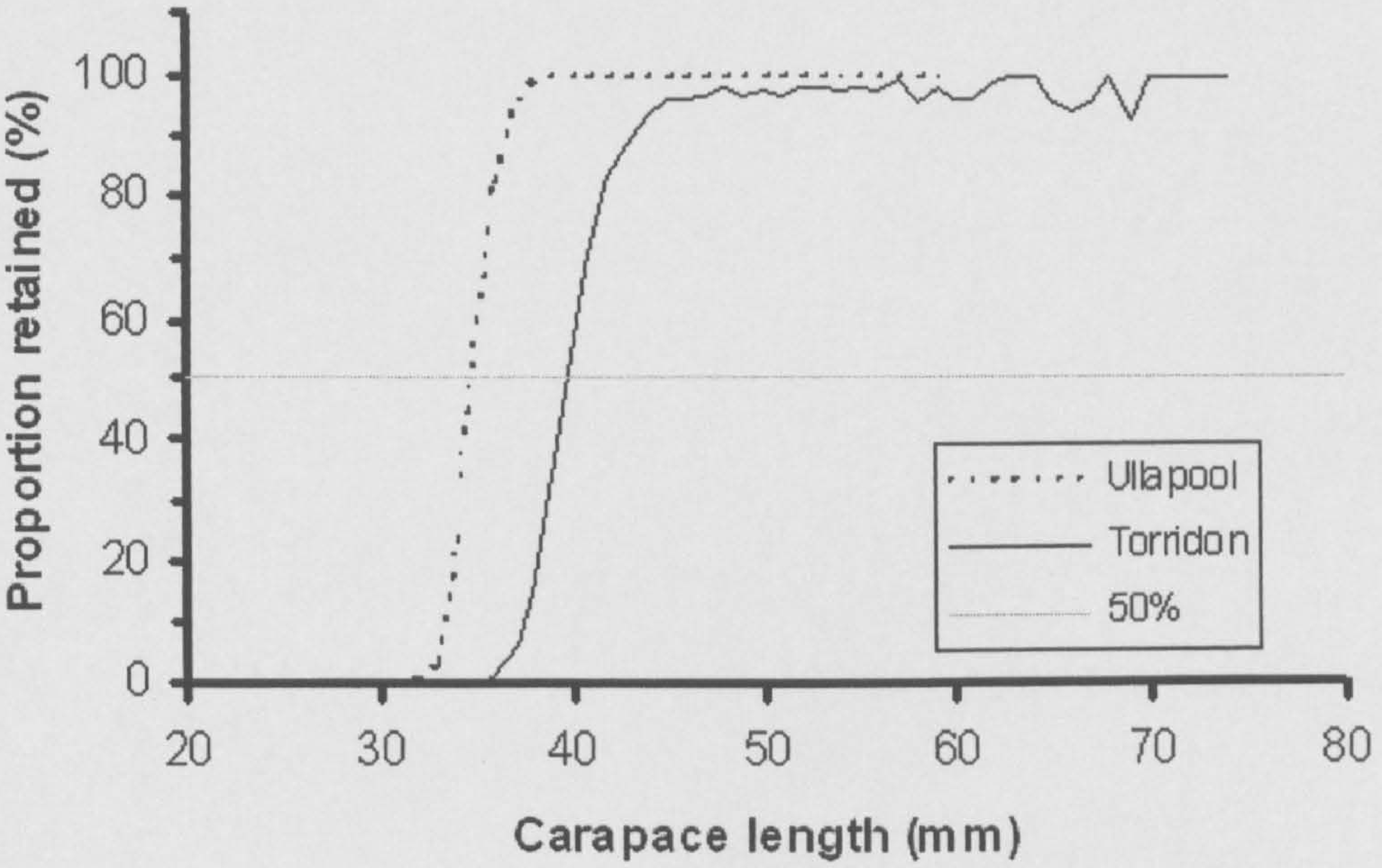


Figure 2.40 - Catch selection curve for Torridon and Ullapool fisheries 2004 – 2005.

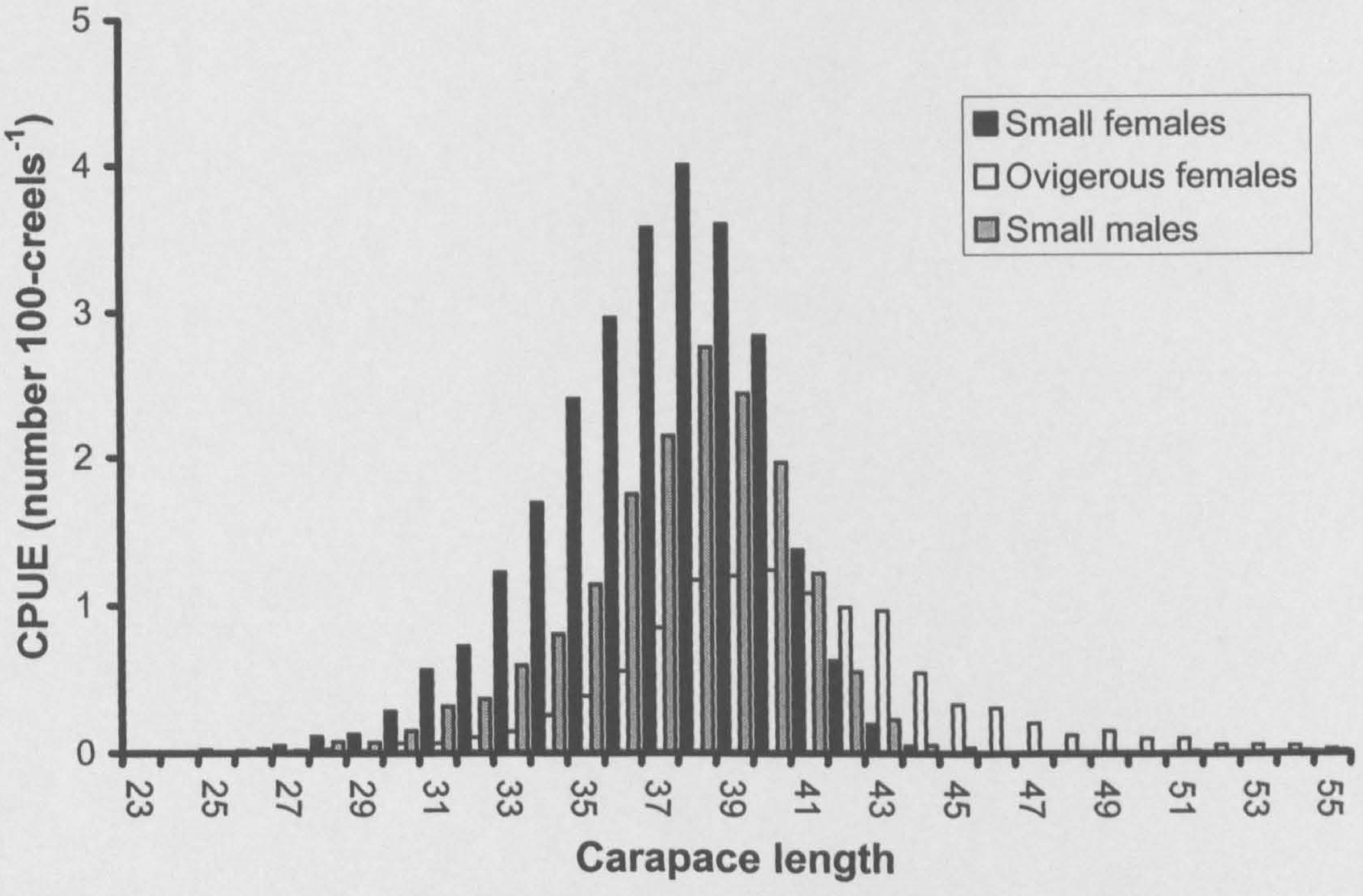


Figure 2.41 - Length-frequency distribution of CPUE data for *Nephrops* discard categories from Torridon area 2004–2005.

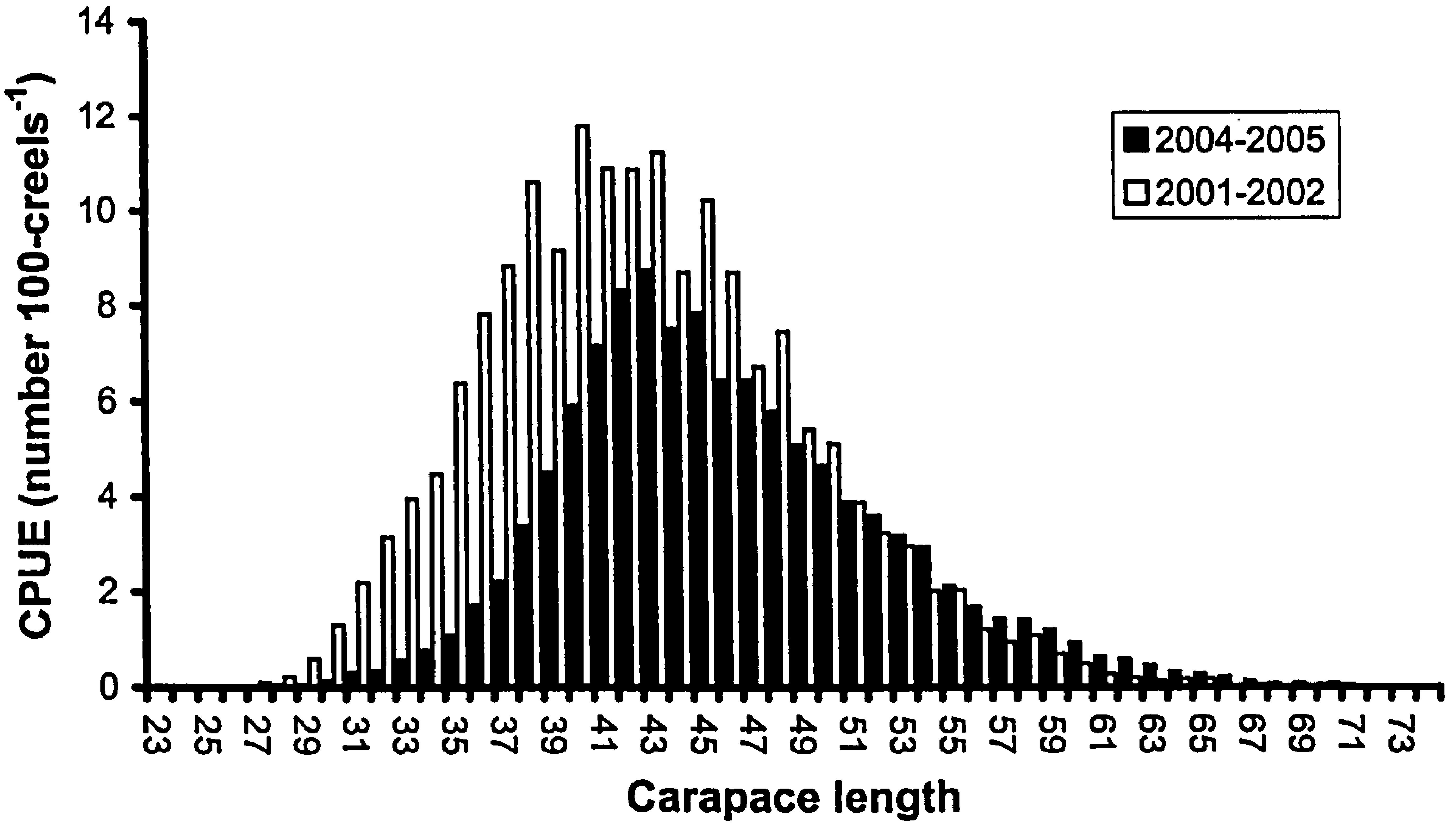


Figure 2.42 - Length-frequency distribution of CPUE data for male *Nephrops* landings and discards from Torridon 2004–2005 with escape gaps (n=18110 creels) and 2001-2002 without escape gaps (n=3678 creels).

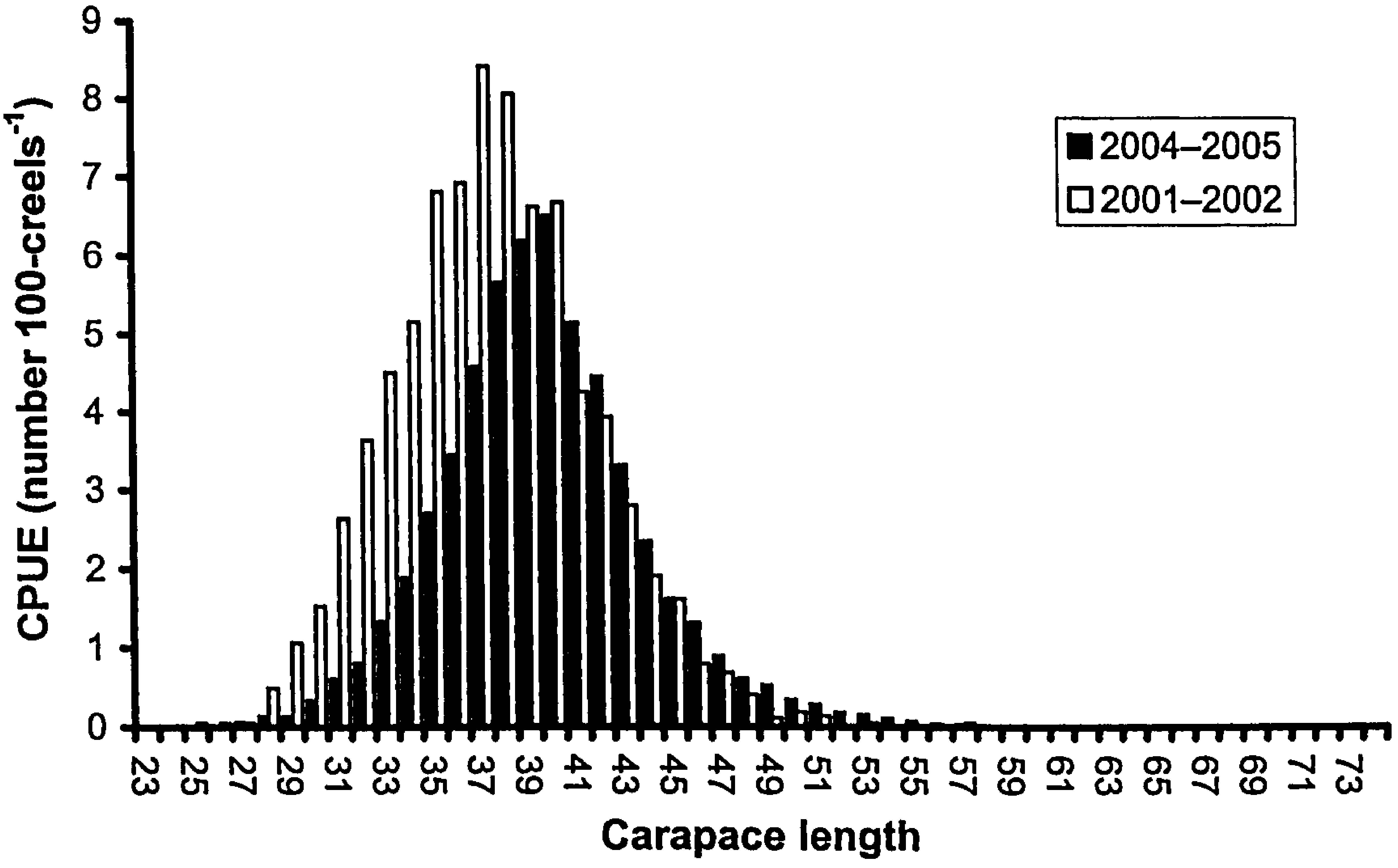


Figure 2.43 - Length-frequency distribution of CPUE data for female *Nephrops* landings and discards from Torridon 2004–2005 with escape gaps (n=18110 creels) and 2001–2002 without escape gaps (n=3678 creels).

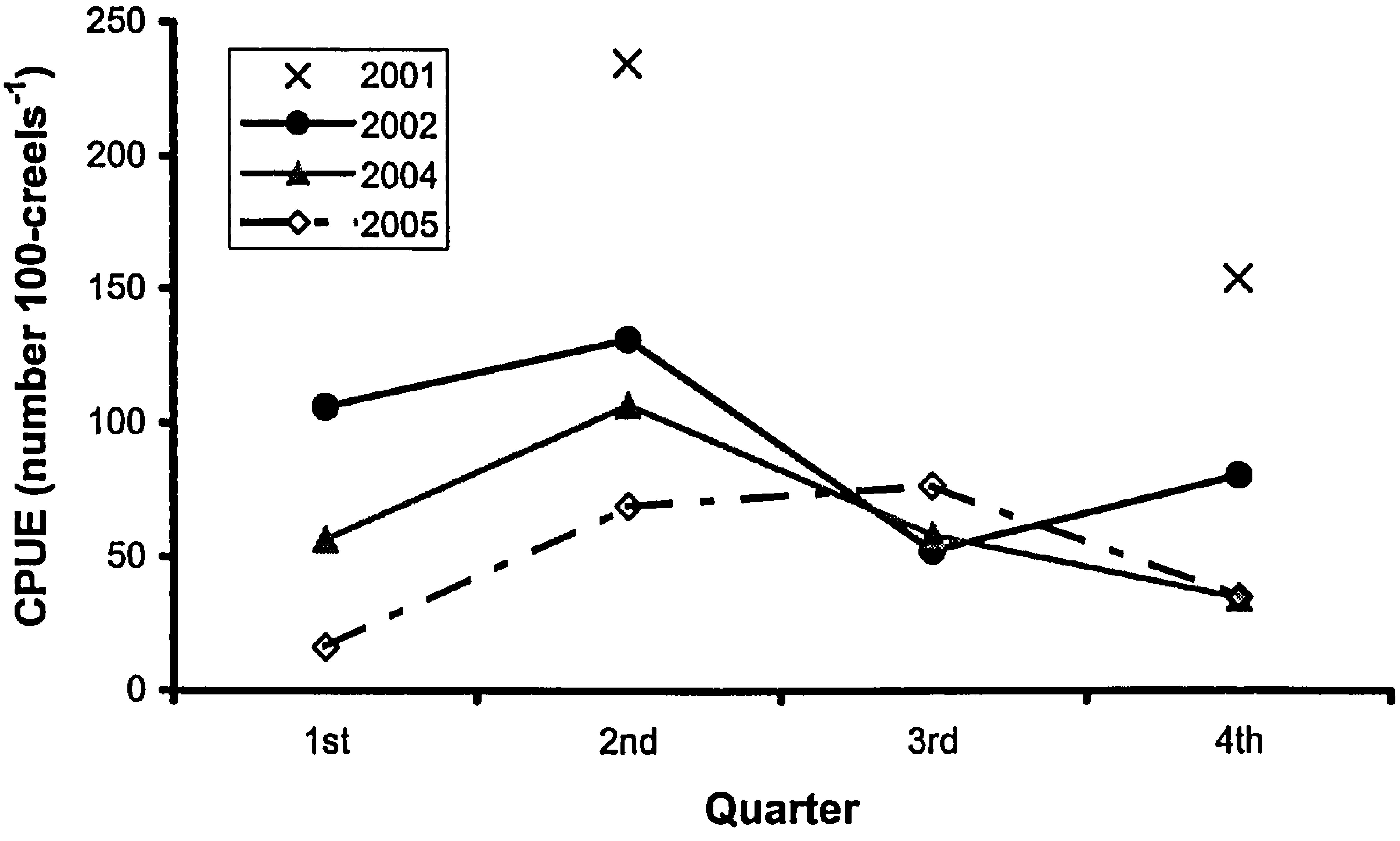


Figure 2.44 – Quarterly CPUE of *Nephrops* <41 mm CL from the Torridon area in different years (2001 n=1410 creels, 2002 n=1928, 2004 n=8680, 2005 n=8855).

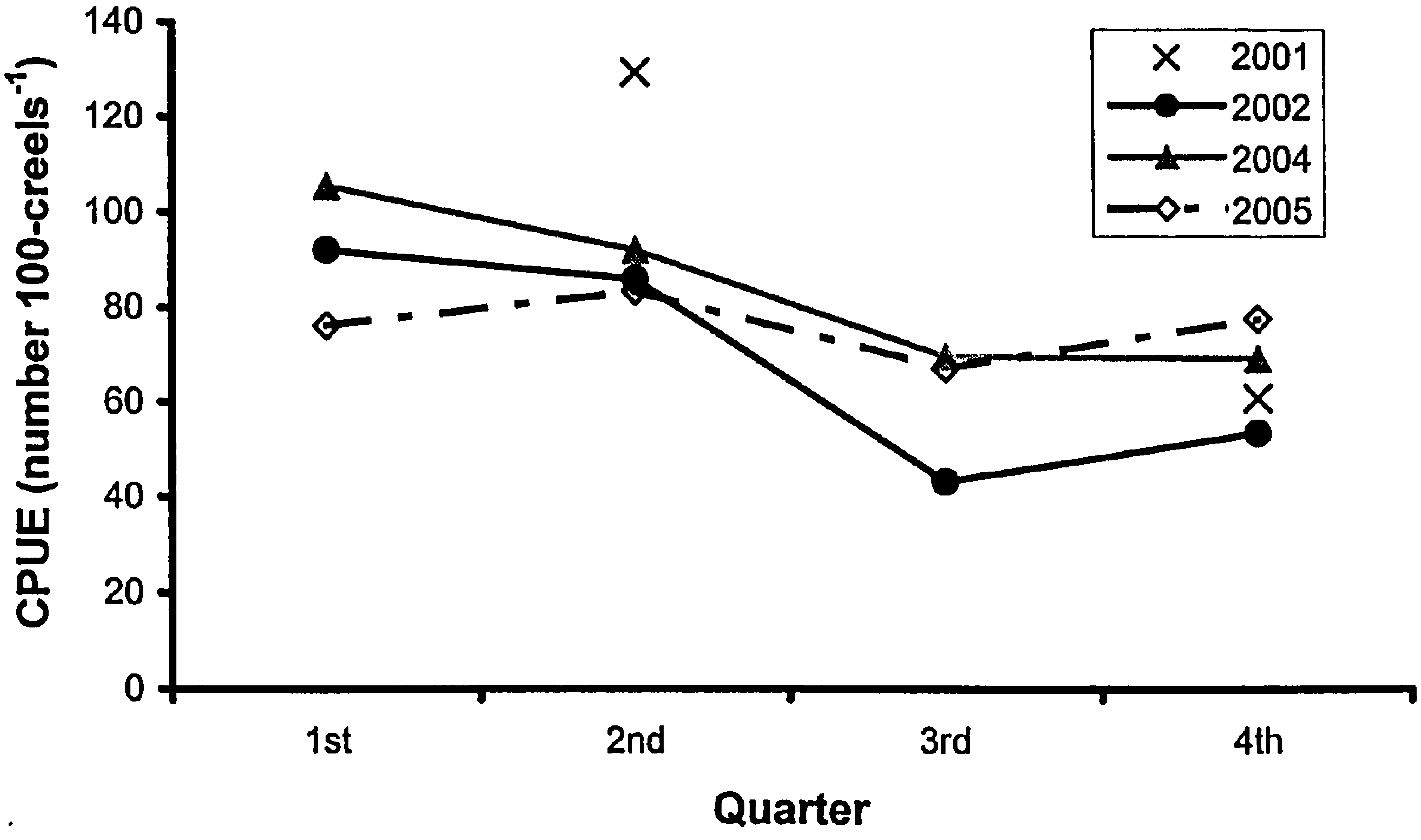


Figure 2.45 – Quarterly CPUE of *Nephrops* 41–48 mm CL from the Torridon area in different years.

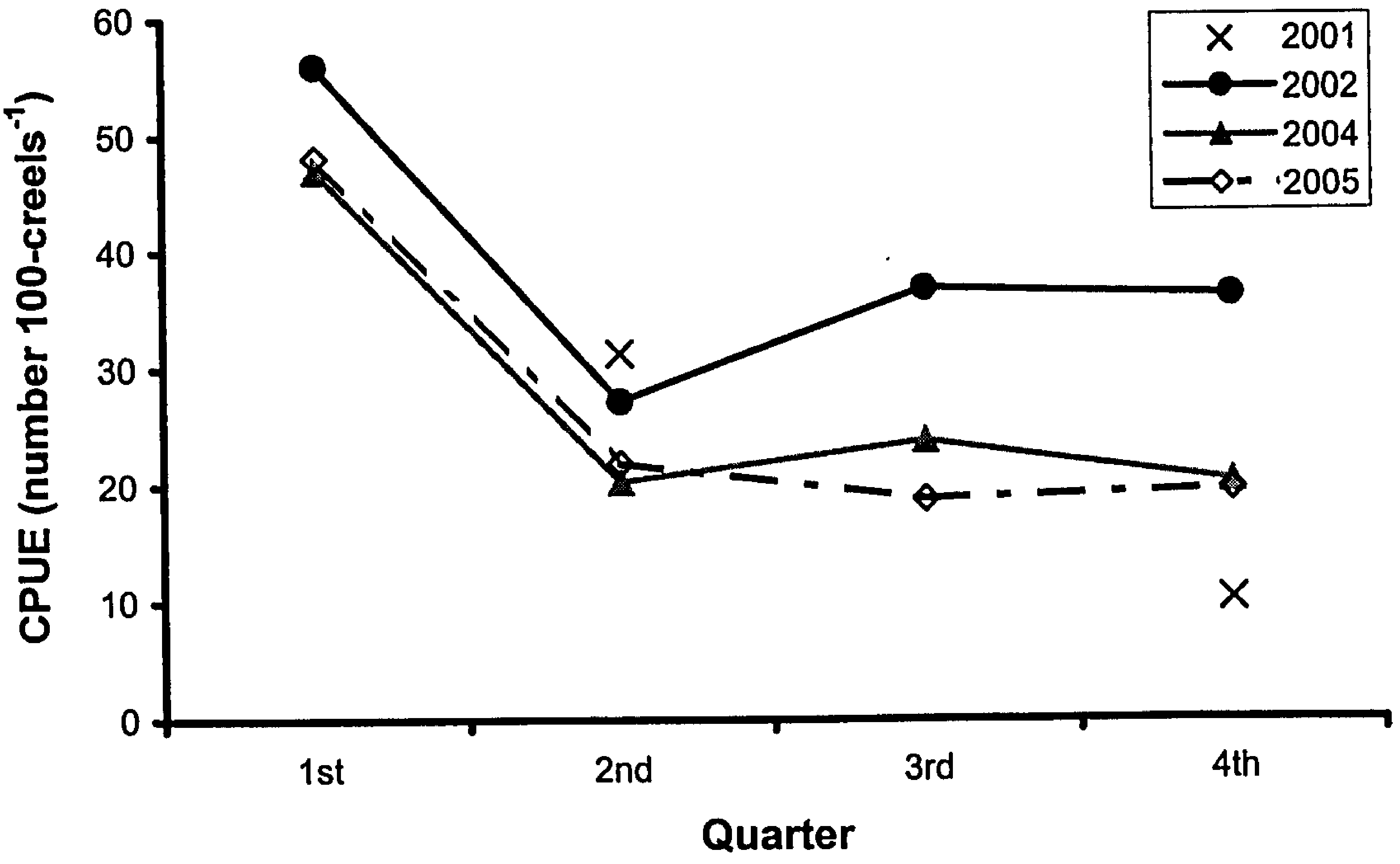


Figure 2.46 – Quarterly CPUE of *Nephrops* 49–56 mm CL from the Torridon area in different years.

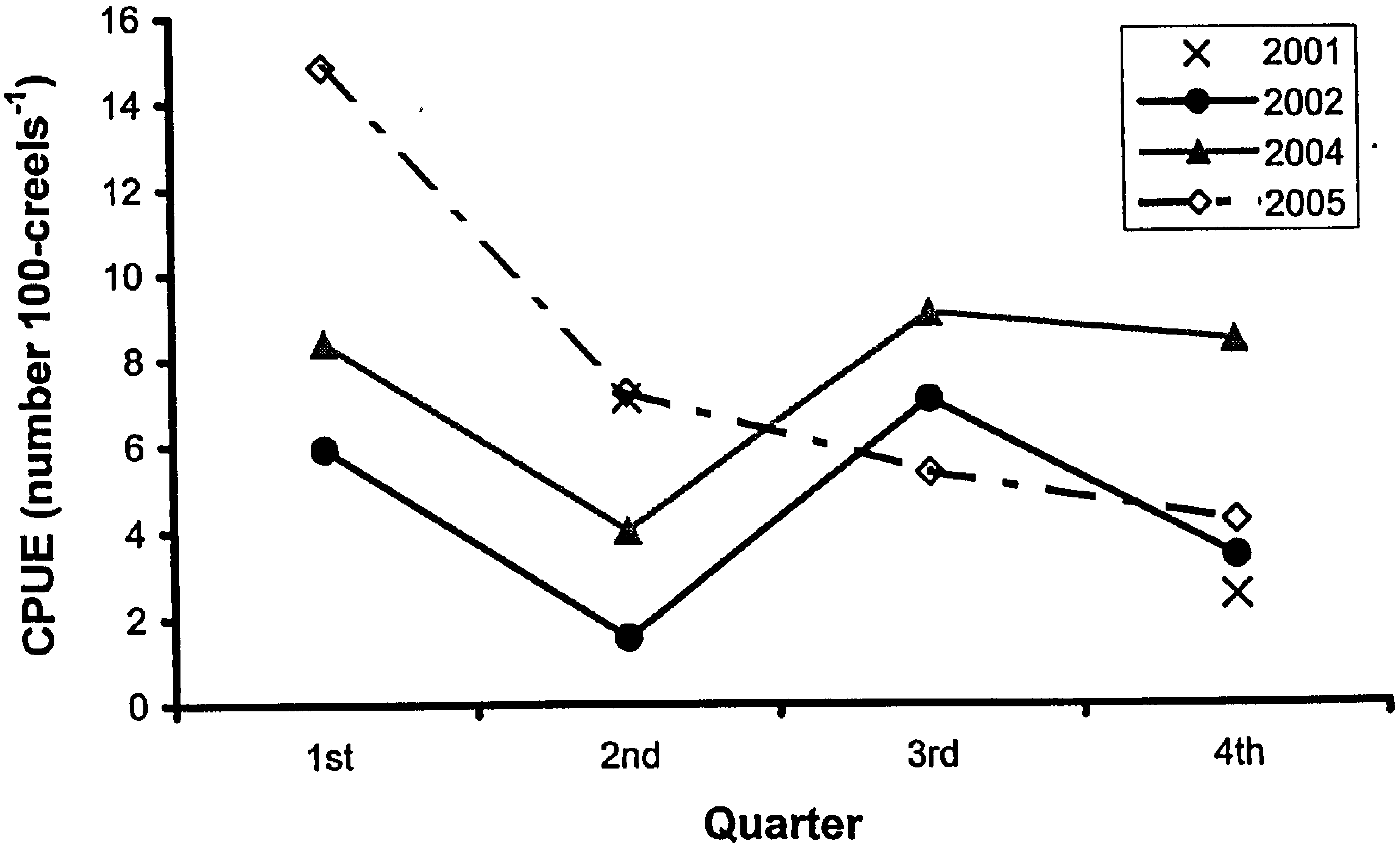


Figure 2.47 – Quarterly CPUE of *Nephrops* >57 mm CL from the Torridon area in different years.

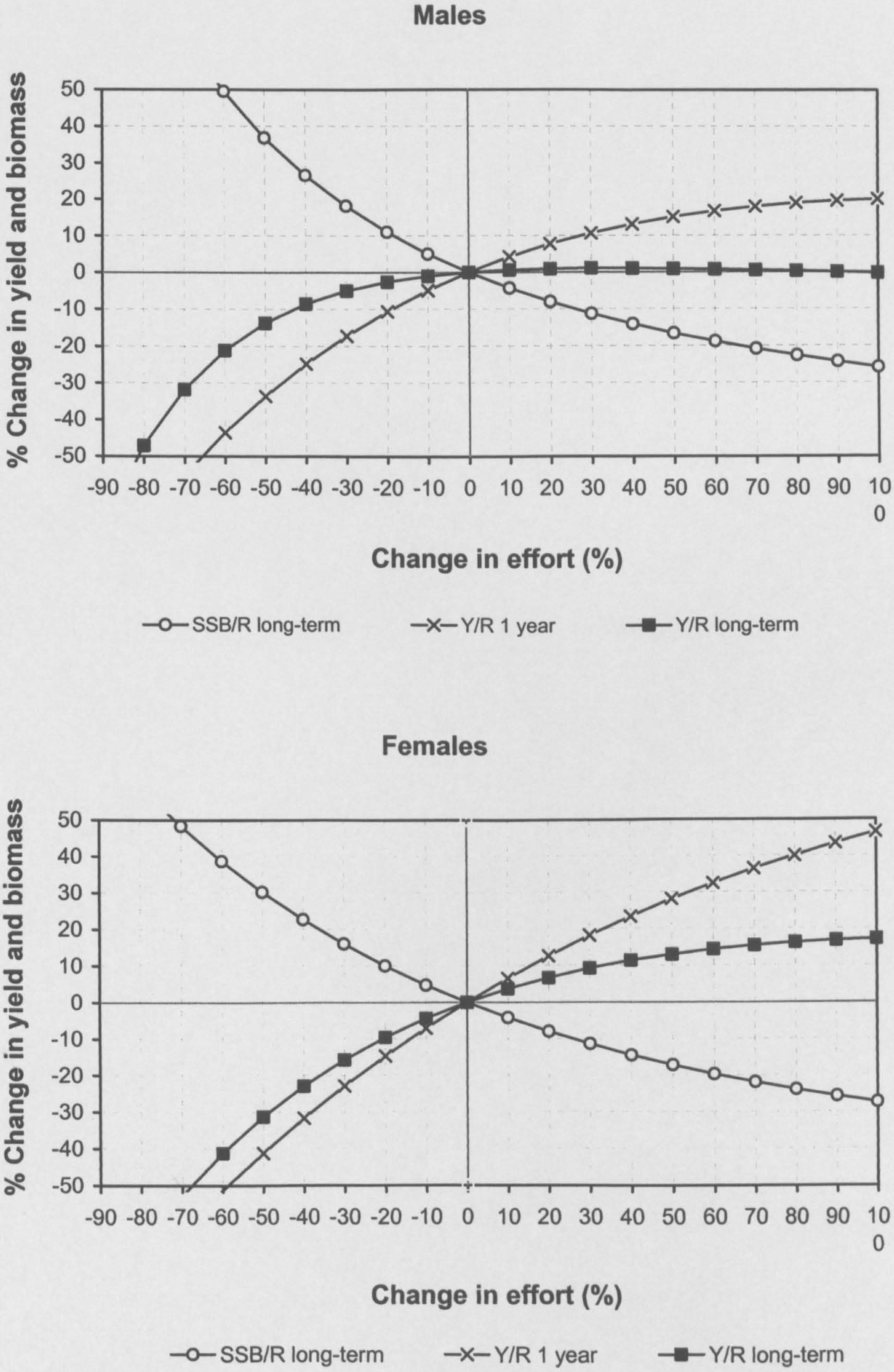
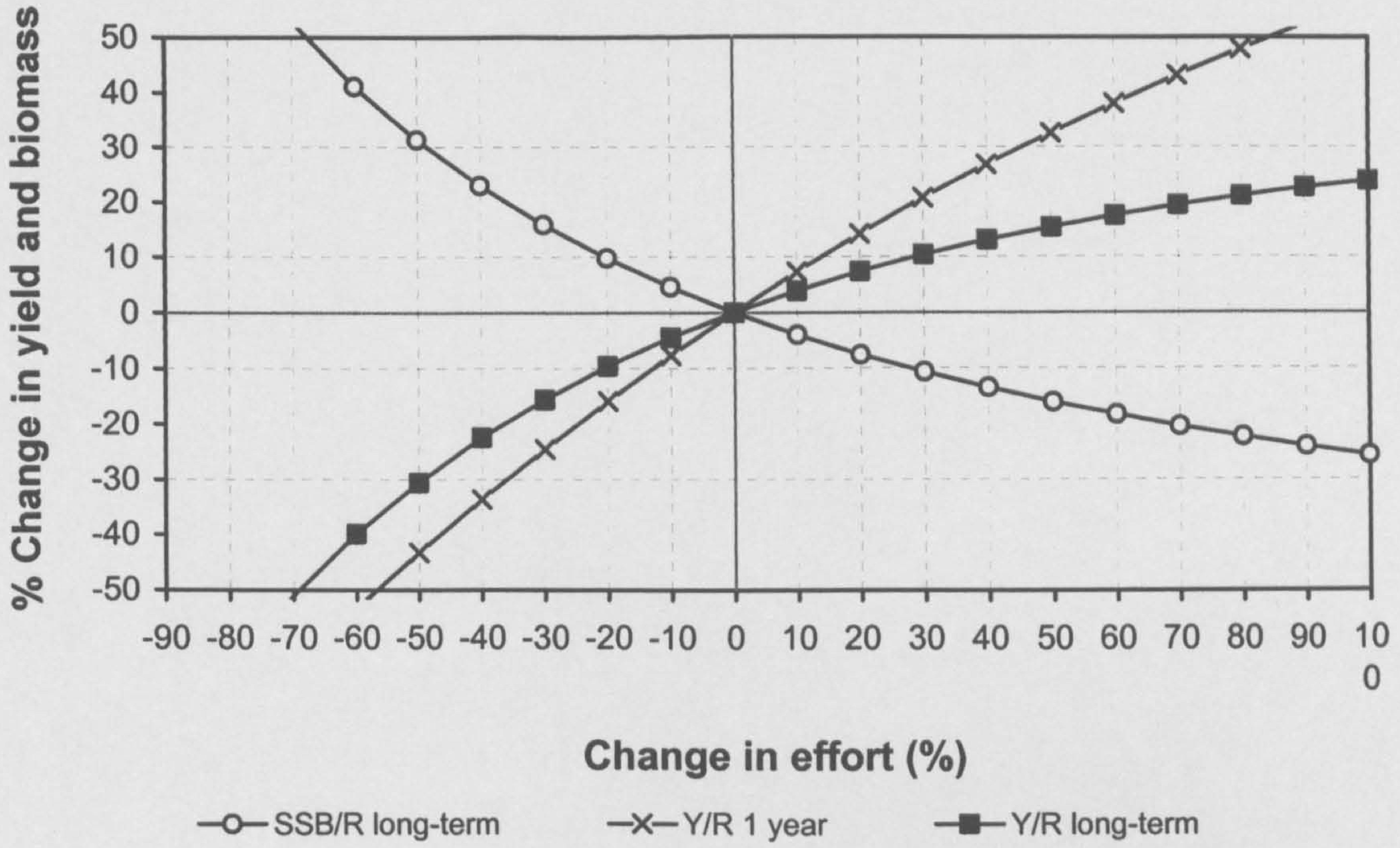


Figure 2.48 - Predicted percentage changes in male and female long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in the Torridon creel fishery 2004, using ICES (1991) growth parameters.

Males



Females

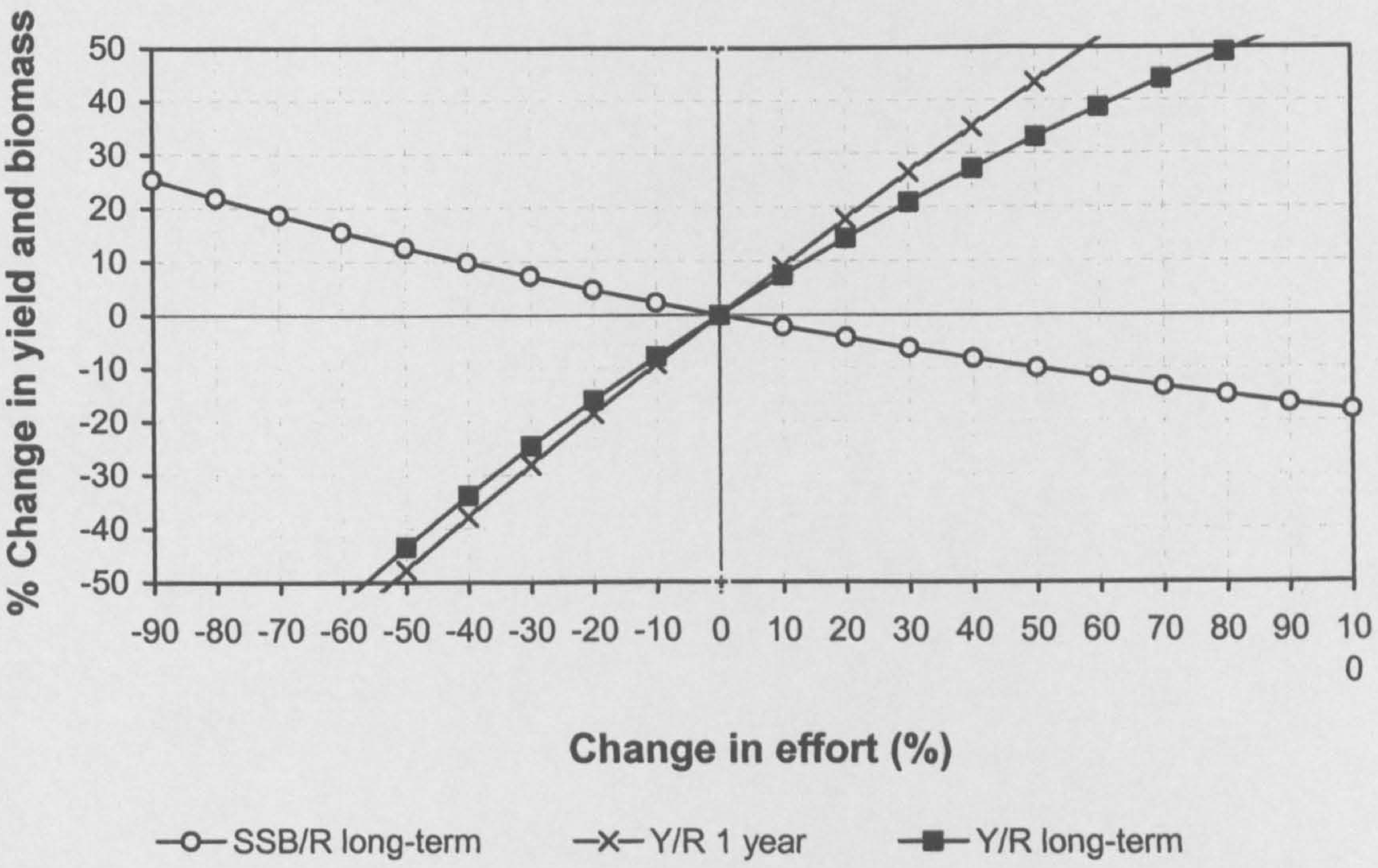


Figure 2.49 - Predicted percentage changes in male and female long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in the Torridon creel fishery 2004, using ICES (1999) growth parameters.

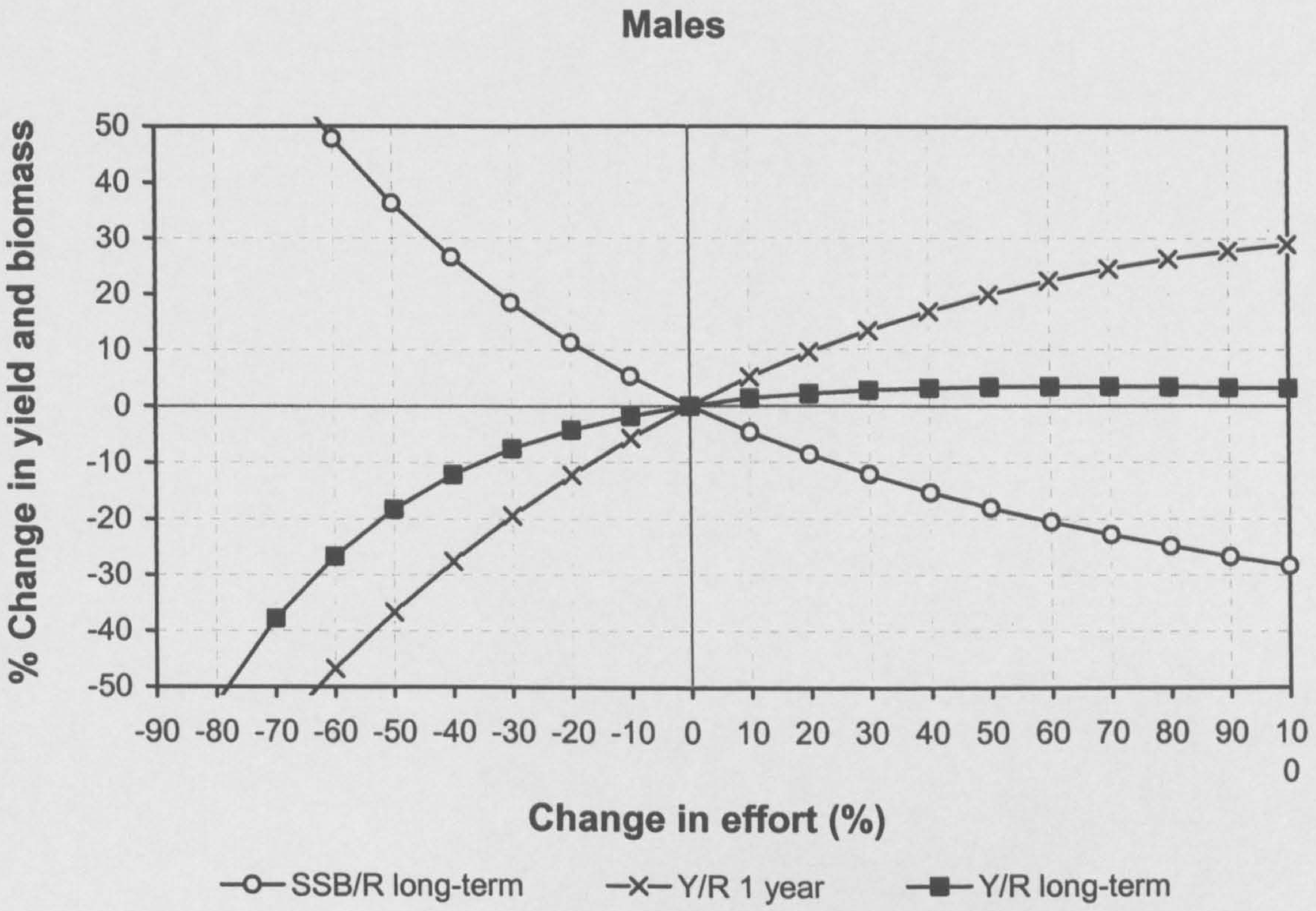


Figure 2.50 - Predicted percentage changes in male long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in the Torridon creel fishery 2005, using ICES (1991) growth parameters.

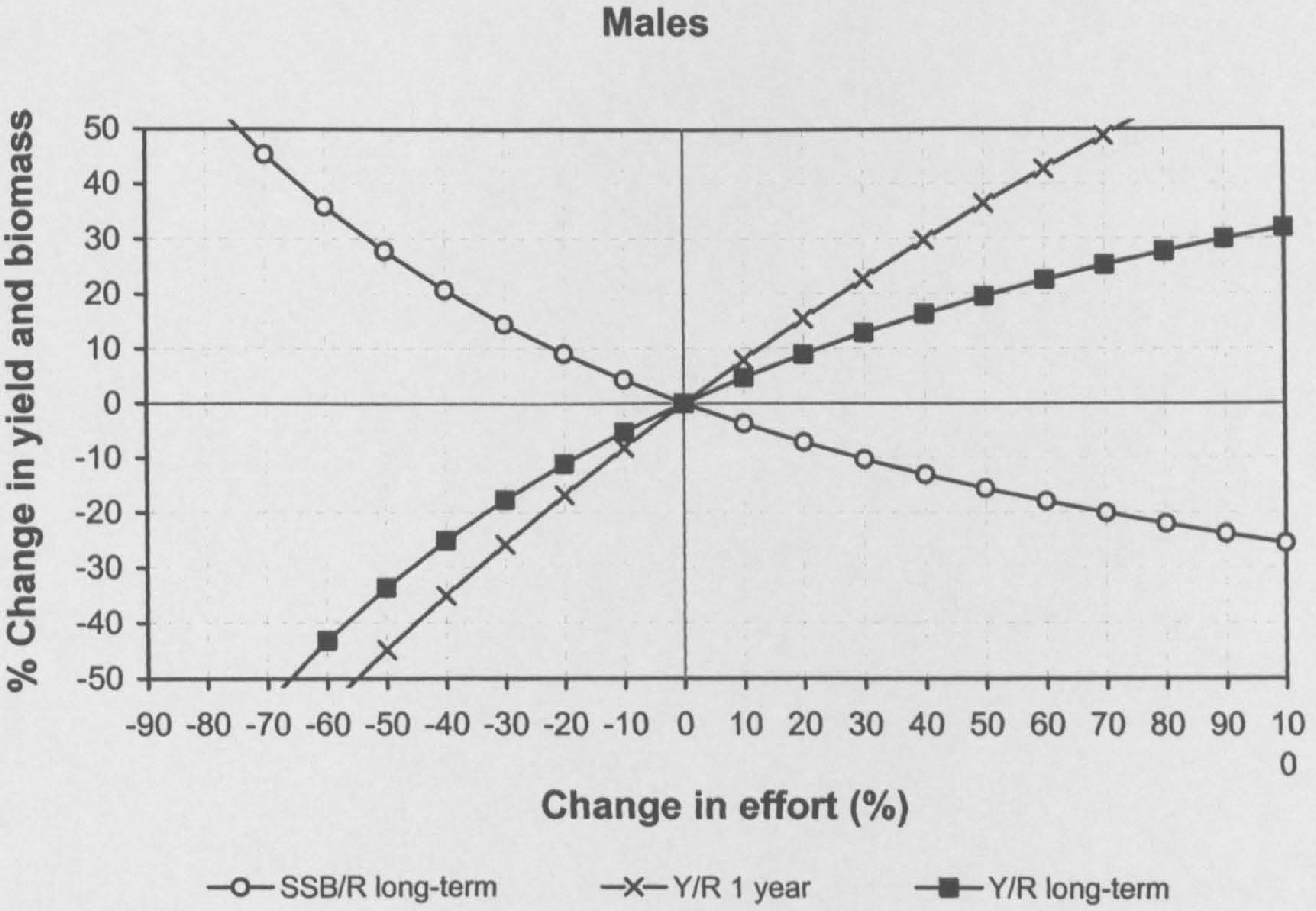


Figure 2.51 - Predicted percentage changes in male long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in the Torridon creel fishery 2005, using ICES (1999) growth parameters.

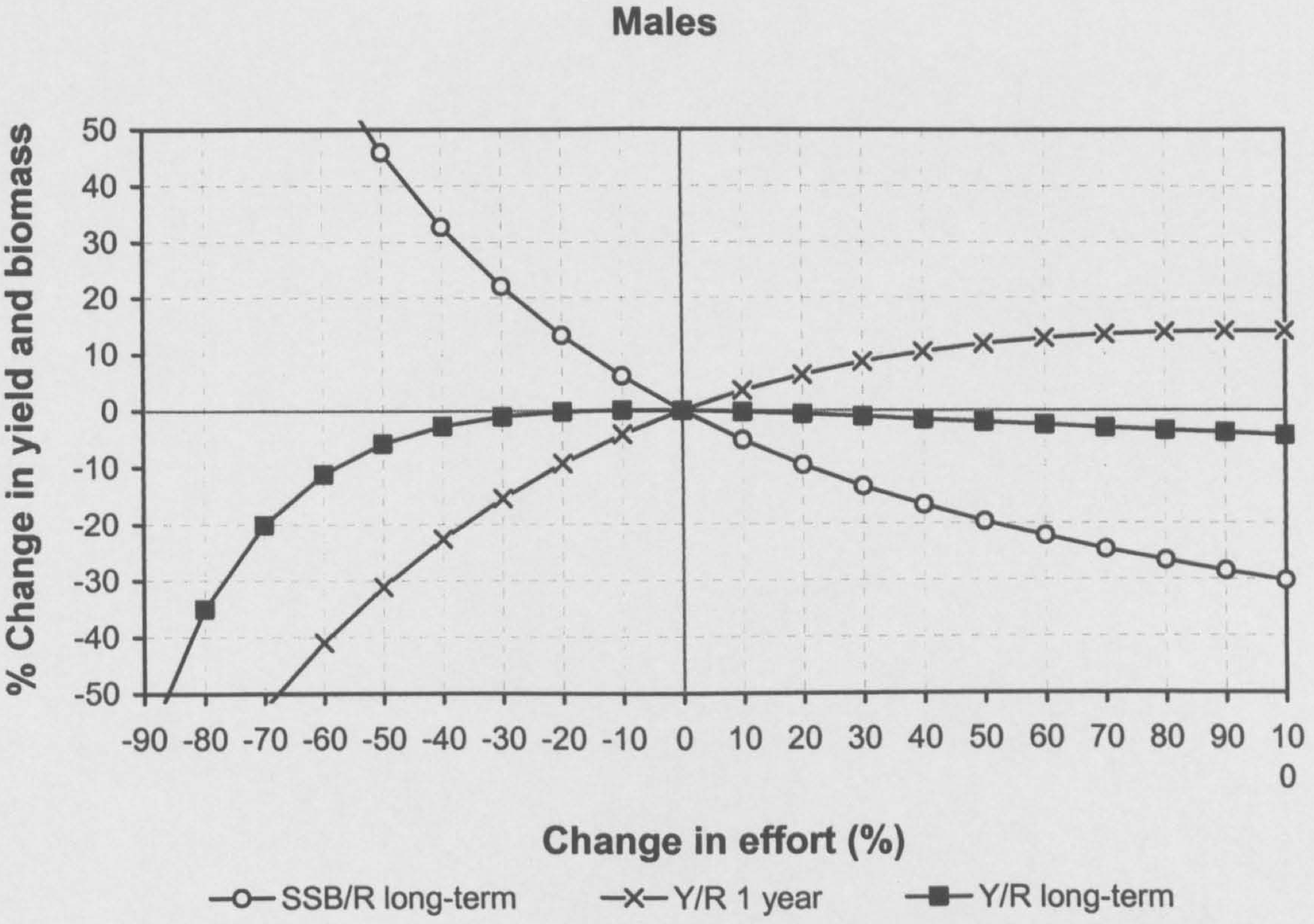


Figure 2.52 - Predicted percentage changes in male long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in the southern Inner Sound (Kyle) creel fishery in 2005, using ICES (1991) growth parameters.

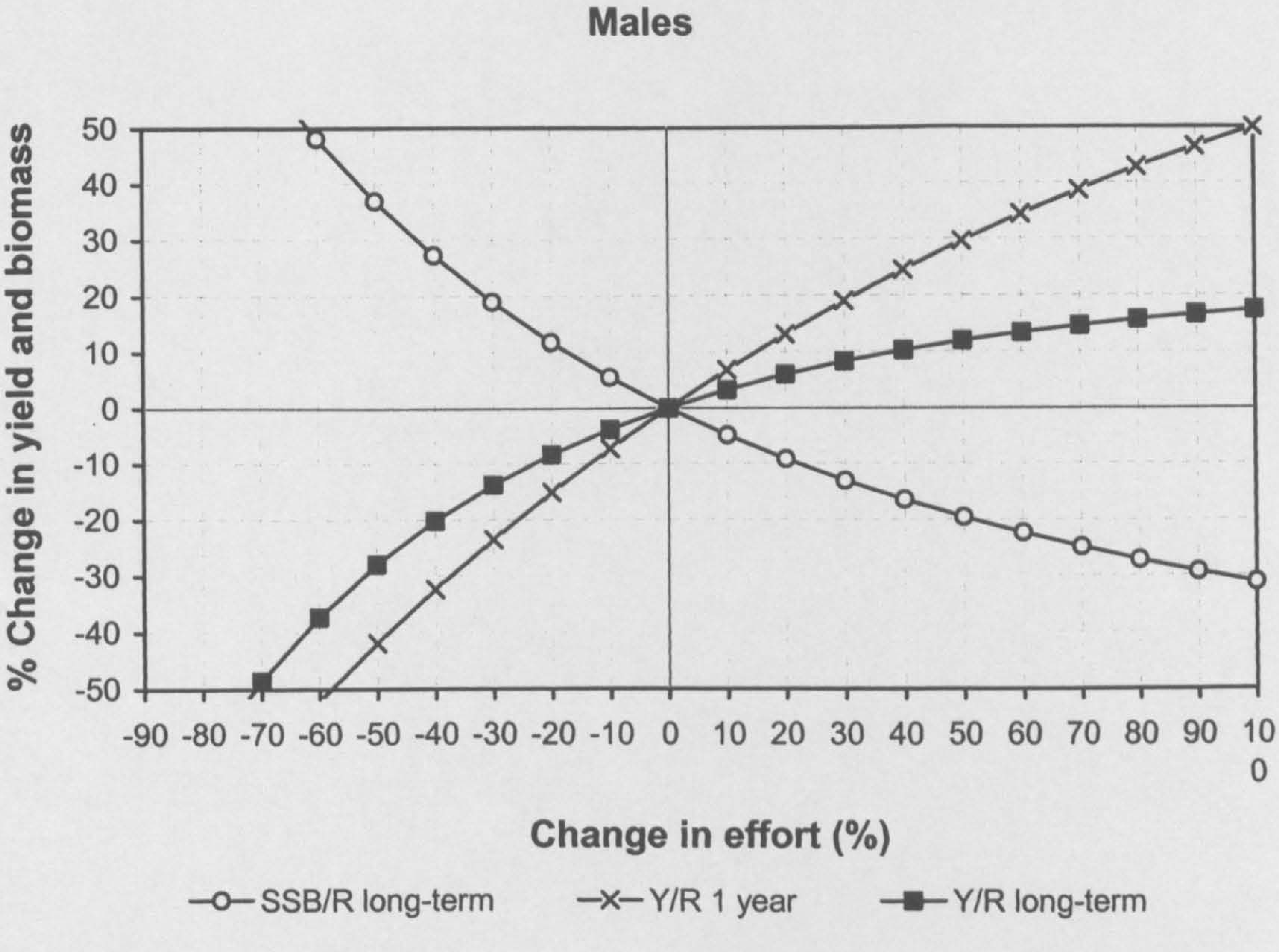


Figure 2.53 - Predicted percentage changes in male long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in the southern Inner Sound (Kyle) creel fishery in 2005, using ICES (1999) growth parameters.

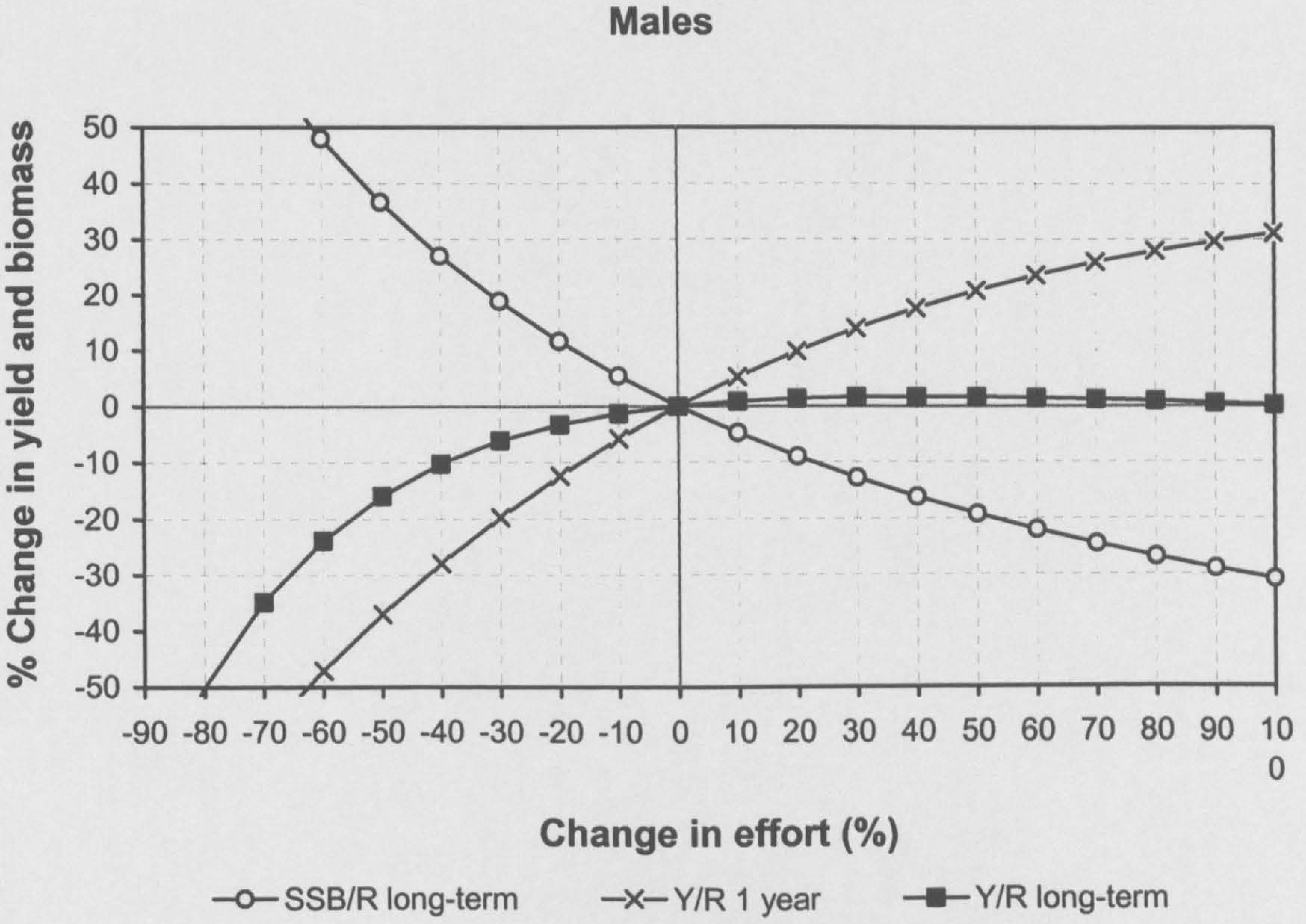


Figure 2.54 - Predicted percentage changes in male long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in the Loch Fyne creel fishery in 2005, using ICES (1991) growth parameters.

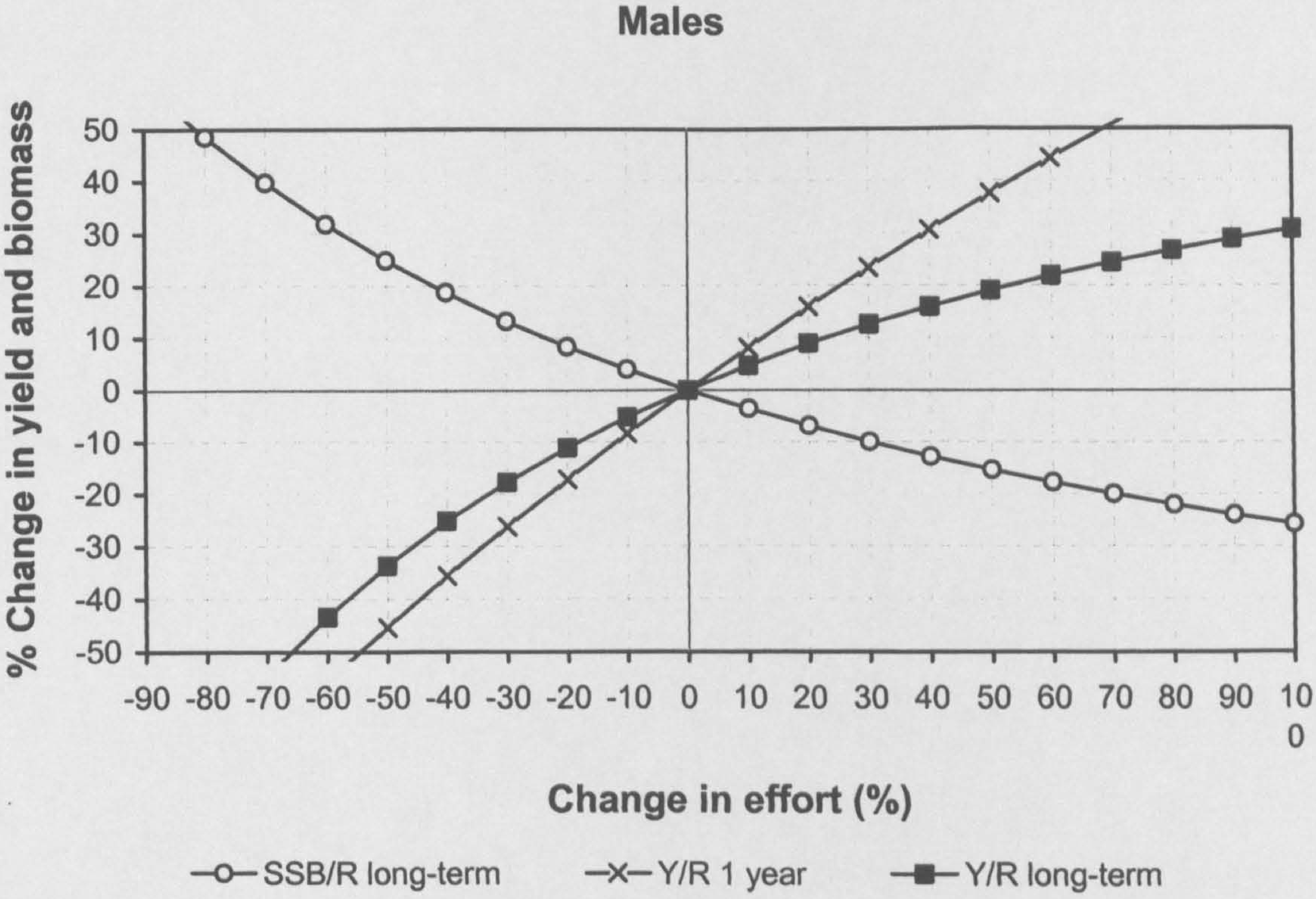


Figure 2.55 - Predicted percentage changes in male long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in the Loch Fyne creel fishery in 2005, using ICES (1999) growth parameters.

Chapter 3

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3 THE MICROSCALE MAPPING OF EFFORT AND LANDINGS IN *NEPHROPS* CREEL FISHERIES

3.1 Introduction

Effective fisheries management is essential if the future sustainability of a fishery is to be guaranteed. For effective management to be possible the quality and quantity of information available must be such that management decisions are based on the best possible science. Of particular importance is the scale at which data are collected. Accurate locational catch and effort data are important for the management of spatially imposed quotas (Meaden & Kemp, 1997), as well as to allow for the consideration of irregular temporal and spatial distributions of target species, variations in effort, variations in habitat distribution, inequalities in socio-economic aspects of the fishery and the derivation of meaningful ecological indices of fisheries disturbance (Meaden & Do Chi, 1996; Rijnsdorp *et al.*, 1998). This has led to the suggestion that a high resolution of fishing effort data is required if fisheries research is to be applicable to commercial fleets (Kaiser, 1998).

The distribution of fishing effort in the past has been studied using a variety of methods over different spatial scales. In the UK, the majority of fishing effort statistics are based on logbook data or official landings and are given at the scale of an ICES rectangle (1° longitude × 0.5° latitude) (Robinson, 2003). Finer-scale assessments have been carried out using logbooks or interviews to produce spatial data at a resolution of 10' squares (Caddy, 1975), as well as the reporting of location on a regular basis and aerial surveys of fishing activity (Robinson, 2003). A more accurate method of determining the microscale distribution of effort is the use of automated position recording units (Marrs *et al.* 2000, 2002a,b). These are global positioning system (GPS)-based position recording units (PRU), which record the latitude and longitude at regular intervals, allowing the speed of the vessel to be determined, from which fishing locations can be deduced. The term 'microscale' has been used previously (Rijnsdorp *et al.*, 1998; Marrs *et al.* 2000, 2002a,b), but has not been well defined. In this study, microscale is used to refer to areas of <1 km^2.

A number of studies investigating the microscale distribution of fishing effort have shown that trawl vessels do not fish at random: effort is usually highly concentrated in certain areas (Rijnsdorp *et al.*, 1998; Friedlander *et al.*, 1999; Jennings *et al.*, 2000; Larcombe *et al.*, 2001; Marrs *et al.*, 2000; Marrs *et al.*, 2002a,b; Whalley & Kemp, 2006). Fishers may operate in a non-random fashion for a number of reasons, including the patchy distribution of target species, distance of fishng grounds from port and the avoidance of grounds that are potentially dangerous, or exclusion zones (Rijnsdorp *et al.*, 1998). An example of an uneven distribution of effort was observed in the Clyde Sea Area *Nephrops* trawl fishery where the stock is non-uniform across the ground, relating to spatial differences in larval supply and the complex sedimentology of the Clyde (Marrs *et al.*, 2000, 2002b).

The distribution of effort in the Dutch beam trawl fishery in the southern North Sea was patchy over large spatial scales, with the distribution only becoming random (in 90% of cases) on a scale of 1 × 1 nautical miles (Rijnsdorp *et al.*, 1998). It was estimated from eight of the most intensively fished rectangles that 5% of the sea bottom was trawled less than once in 5 years, 29% less than once a year, 30% between one and two times a year and 9% more than five times a year, with the most intensive fishing occurring along the borders of closed areas and on certain offshore grounds (Rijnsdorp *et al.*, 1998). Fishing intensity in the Clyde Sea Area *Nephrops* trawl fishery (Marrs *et al.*, 2000, 2002b) was higher than in the Dutch beam trawl fishery (Rijnsdorp *et al.*, 1998), but fishing effort was similarly concentrated in certain areas. In the Clyde study, microscale mapping of landings and effort provided a more accurate assessment of the spatial variability in the fishery, allowing for small scale patterns of size distribution and growth rate of the *Nephrops* stock to be considered (Marrs *et al.*, 2000, 2002a,b).

These findings have considerable implications for the assessment of stocks at the scale of an ICES rectangle. Stock assessments applied to *Nephrops* stocks include virtual population analysis (VPA) and length cohort analysis (LCA), which rely heavily on inputs from commercial catch data and on assumptions that are often more applicable to finfish stocks, including stock homogeneity (Chapman & Howard, 1988; Tuck *et al.*, 1997a) (Chapter 2). These assessment methods are also sensitive to input parameters, which, in *Nephrops*, are known to vary with sediment type (Bailey *et al.*, 1986; Tully & Hillis, 1995; Maynou & Sardà, 2001). Microscale mapping is important in the assessment of *Nephrops* stocks, as it allows for stratified stock assessments (at a finer resolution than ICES rectangles), reducing the problems associated with assessing heterogeneous stocks (Marrs *et al.*, 2000, 2002a,b). There are, however, limitations to this approach, including the requirement of good sediment data and biological data (population parameters and length-frequency distribution of removals) for each stratum. Therefore, if a similar approach were taken in other areas, considerable groundwork (and therefore costs) would be necessary to collect the relevant biological information (Marrs *et al.*, 2002b).

High-resolution effort data are also valuable in assessing the environmental impact of fishing. Where the activity of a fishing fleet has been averaged over a large area, the estimated environmental impact may be misleading (Jennings & Kaiser, 1998). The inclusion of high-resolution effort data significantly affects estimated ecological disturbance at the scale of the ICES rectangle (Robinson, 2003). For example, Rauck (1985) estimated that every square metre of seabed in the North Sea was on average trawled 5–7 times a year on the scale of an ICES rectangle. In contrast, Rijnsdorp *et al.* (1998) observed that even in the most heavily trawled areas of the North Sea, only 9% of areas are fished more than 5 times a year. This finding not only suggests there may be lower ecological disturbance, but also indicates that even some sensitive species may be able to survive in these areas.

In the Torridon area *Nephrops* creel fishery, the current resolution used by ICES for recording fishing effort and landings for stock assessment purposes is an ICES rectangle (covering an area of 3,275 km^2 at 58°N), with the North Minch functional unit (FU11) being made up of 6 rectangles (44—46 E3—E4) (ICES, 2002). Such aggregation of data on a large scale may be a particular issue for *Nephrops* creel fisheries, in which there is considerable spatial variation in catch composition, both between and within creel fisheries (Chapter 2).

The spatial scale on which the data were analysed in Chapter 2 may have been inappropriate given the known scale of variation in habitat and stock characteristics. When compared with trawl fisheries, *Nephrops* stocks exploited by creel fisheries are likely to be more heterogeneous, as these fisheries are largely confined to areas that are inaccessible to trawlers, due to varied seabed topography. Consequently, creel fishing grounds tend to encompass a variety of biotopes, characterised by a number of environmental factors, including; sediment type, depth and current regime. This potential stock heterogeneity within typical creel fisheries is likely to lead to an unequal distribution of effort within the fishery, particularly since the gear used allows small areas to be targeted, compared with the large area covered by a single trawl tow.

In order to provide a better understanding of the spatial distribution of effort and landings from the *Nephrops* creel fishery, the microscale distribution of effort and landings from a number of areas on the west coast of Scotland was investigated. Two methods were used: initially detailed logbooks were issued to fishers to record the position and catch from fleets of creels from a number of areas. Subsequently, one fishery was selected and there, a combination of PRUs and voluntary logbooks was used.

3.2 Methods

3.2.1 Mapping of effort and landings using logbook data

The distribution of effort and catch was studied in four creel fishing areas: Loch Fyne, Torridon (Inner Sound), Gairloch and Ullapool (Summer Isles) (Chapter 1; Figure 1.7).

3.2.1.1 Data collection

Spatially resolved catch and effort data were recorded by fishermen (each vessel was provided with a confidential vessel code) using specially designed logbooks in which the location, number of creels and landed weight of small (34–38 mm CL), medium (39–47 mm CL) and large (>47 mm CL) *Nephrops* were noted for each fleet of creels fished on a daily basis (Appendix A; Figure A.2). Recording was carried out over a 1–2 year period from April 2004. The months from which data were collected are displayed in Table 2.2 (Chapter 2).

3.2.1.2 Generation of maps

Vector files of fishing locations were produced using the GIS package IDRISI (Clarke University, Worcester MA). Each vector file displayed the midpoint location of each fleet of creels fished on a quarterly basis from each fishery. For each quarter, vector files were produced mapping either the effort (number of creels fished per fleet) or the landed weight (kg per fleet) for each of the three size categories. Each of the vector files was converted into a raster image (pixel size approximately 200 × 200 m), for each quarter, from all four fisheries (IDRISI 'vector / raster' function; 'polygon to raster' option). Total quarterly landings were calculated by adding the landings per pixel of each size category (IDRISI image calculator). Quarterly effort and landings raster images were summed to produce raster images of landings (*Nephrops* categorised as, small, medium, large and total catch) and effort over the entire study period from each port. Maps of LPUE (kg 100-creels^{-1}) were produced from each area by dividing the landings (each size category) by the effort in each pixel and multiplying this value by 100 (IDRISI image calculator).

The spatial pattern of effort and landings of vessels fishing from Loch Gairloch were examined in relation to maps of sediment distribution (Section 3.2.5).

3.2.2 Mapping of effort and landings using position recording units

3.2.2.1 Position recording unit design

Position recording units (PRU) comprised a data-logger (Skyforce LOG503-2.05 GPS logger^{TM}) connected to the fishing vessel's GPS receiver. The logger received National

Marine Electronics Association (NMEA) 0183 test strings from the GPS receiver (standard communications protocaol for maritime electronic equipment). The string was processed by the data-logger, and the date, time, latitude, longitude, instantaneous speed and direction of travel were stored at 5-minute intervals on a removable 16 MB Compact Flash Memory card (Marrs *et al.*, 2002b). An internal lithium battery powered the loggers. Data was downloaded using a terminal emulator programme (HyperTerminal, Microsoft Corporation) *via* a 9-way D-type PC interface cable and stored in a plain text file in comma separated variable (CSV) format. In some cases, it was not possible to connect the logger to the vessel's own GPS receiver, so a separate GPS receiver (CMC Electronics Smart Antenna R S232, combined GPS and antenna) was incorporated into the PRU (Marrs *et al.*, 2002b).

3.2.2.2 Data collection

PRUs were installed on seven creel vessels from which data were collected from June 2004 until November 2005. Limited battery capacity necessitated the replacement of batteries on a monthly basis. The fishermen, for the most part, undertook this task. Memory cards were removed and the data were downloaded where possible (every 2–4 months).

Data collected from loggers were used to calculate the straight-line distance between consecutive location records. It was possible to determine the speed of the vessel throughout the logging period, as the time interval between records was known. Vessel speed was plotted against time of day making it possible to examine the speed profile of the vessel on a daily basis, and identify the position of each fleet of creels fished, with periods of slow speed (<1 knot) indicating creel-hauling operations (Figure 3.1).

Positional data were used in conjunction with logbook data completed by the fishermen on a daily basis, detailing the weight (kg) of *Nephrops* caught in four size categories (small, medium, large and extra large) as well as the number of creels hauled on a daily basis (Appendix A; Figure A.1).

3.2.2.3 Generation of maps

Vector files were produced using the GIS package IDRISI. Each vector file displayed the location of the fleets fished on a specific day, with a series of points spanning the length of each fleet of creels. Four vector files were produced for each vessel for each day, showing either the numbers of creel-hauls (effort) or the landings (kg) from one of the three size categories selected (large and extra large categories were combined). Vector files were converted into raster images (pixel size roughly 90 × 90 m) for each category from each day (IDRISI vector / raster function; polygon to raster option). Daily raster images were combined to produce monthly images (IDRISI image overlay function).

Each fleet of creels was represented on the raster image by a certain number of pixels, depending on the length of the creel fleet. Fishing effort and landed weight of *Nephrops* for each fleet were mapped by assigning each pixel a value corresponding to the total value for the fleet divided by the number of pixels representing the fleet (IDRISI image calculator).

Monthly raster images were further combined to produce quarterly images for effort and landings (IDRISI image overlay function). Total quarterly landings were calculated by adding the monthly landings per pixel of each size category (IDRISI image calculator). Due to the patchiness of data collection it was decided to combine the quarterly data from 2004 and 2005 (IDRISI image calculator). Quarterly effort and landings raster images were summed to produce raster images of landings (small, medium, large and total) and effort over the entire study period from each vessel. Raster images were combined from all seven vessels. LPUE (kg 100-creels^{-1}) was calculated by dividing the landings per pixel by the effort per pixel and then multiplying this value by 100.

A scaled raster image of the estimated annual distribution of effort for the whole vessel fishing fleet, was produced by applying a raising factor (total fleet effort divided by estimated total annual effort of monitored vessels) to the raster image displaying the total effort from all seven monitored vessels. Total annual fleet effort was estimated using data on the number of vessels fishing in the static gear zone and the number of creels each vessel fishes on a daily basis (supplied by Shieldaig Export Ltd.). Total annual effort was estimated by multiplying the maximum number of creels fished on a daily basis by all thirteen vessels (800 or 400 creels for two- and one-man boats, respectively (as required by MSC certification); Chapter 1; Section 1.3.5) by the maximum of number of fishing days allowed (200 days per year). Other vessels are known to fish in the area occasionally; however, the level of effort from these is unknown. Assuming that not all vessels will fish the maximum of 200 days or the maximum number of creels every day, this potential overestimate may to some degree account for these other vessels (annual effort was not estimated from logbooks as they were not completed by all boats fishing in the fleet). From these data, the number of creels fished in each pixel was estimated and illustrated in a histogram (IDRISI image histogram).

From both sediment and area maps, total and quarterly LPUE and effort patterns were extracted in relation to sediment type and location and displayed as bar charts (IDRISI attribute values extraction).

3.2.3 Test for the level of uniformity of fishing effort and landings – Moran's contiguity ratio

The spatial pattern of fishing effort was assessed with Moran's contiguity ratio (I; Cressie, 1991) calculated with the IDRISI AUTOCORR function (King's case). Moran's contiguity ratio is a coefficient of spatial autocorrelation, ranging from –1 to +1. Positive autocorrelation indicates that adjacent pixels are similar to each other (indicating clustering), while negative autocorrelation indicates that adjacent pixels are dissimilar to each other (indicating uniformity). Values tending towards 0 indicate randomness.

3.2.4 Relationship between fishing effort and landings

Regression analysis was conducted on the sampled vessels in each area to assess the relationship between the total landings or LPUE for that pixel (dependent variable) and the number of times a pixel was fished (total effort, independent variable). Analysis was undertaken using the IDRISI REGRESS function. Owing to the possibility of spatial autocorrelation, the normal significance tests for a regression cannot be relied upon, therefore the strength of these relationships was assessed on the basis of the values of the coefficient of determination (R^2).

3.2.5 Generation of sediment and area maps from the Torridon area

In order to assess differences in LPUE among different locations within the Torridon area, a vector file was produced which separated the area into Upper Loch Torridon, middle Loch Torridon (Loch Sheildaig), lower Loch Torridon, northern Inner Sound, southern Inner Sound, BUTEC range, mixed fishing area and trawl-only area (Figure 3.3). To enable catch and landings data to be extracted from each area, the vector file was converted into a raster image.

Sediment maps were produced to relate differences in effort distribution and catch rates to sediment type. Initially a map was produced using BGS sediment data supplied by Fisheries Research Services, but unfortunately data were not available at a sufficient spatial resolution, particularly within Loch Torridon (Figure 3.4). In order to provide a more detailed sediment map, several fishermen were asked to produce a map of the sediment distribution of the area covering Loch Torridon and the northern Inner Sound (no information was provided on the southern Inner Sound). Areas were classified as being either; hard, a hard edge (area of hard ground adjacent to soft sediments), sandy mud or mud (data collected by J. Drewery, FRS & P. Davidson, HIFA). There was a degree of discrepancy between maps, so an average map was generated and digitised to produce a vector file at Fisheries Research Services, which was subsequently converted into a raster image (IDRISI vector / raster function; polygon to raster option) to allow landings and effort data to be extracted with regards to sediment type (Figure 3.5).

3.3 Results

3.3.1 Mapping of effort and landings using logbook data

In total, 57 days of fishing were recorded from the Summer Isles, 480 days from Loch Gairloch (3 vessels), 148 days from the Torridon area (Inner Sound) and 71 days from Loch Fyne. From these data, considerable spatial variability in effort was evident in all areas (Figures 3.6, 3.8, 3.9, 3.11, 3.13). A similar variability in LPUE was also observed (Figures 3.7, 3.10, 3.12, 3.14), with the highest LPUE recorded from the Summer Isles area ($13.82 \text{ kg } 100\text{-creels}^{-1}$) and the lowest LPUE from the Inner Sound ($9.20 \text{ kg } 100\text{-creels}^{-1}$).

Fishing by a single monitored vessel in the Summer Isles area was restricted to a narrow band of open sea, where the LPUE and effort was high (Figure 3.6 & 3.7). There was evidence that fishing effort was significant spatially autocorrelated indicating effort was clustered ($Z=4.51$, $I=0.23$).

Three vessels from Loch Gairloch operated in Loch Gairloch itself, but most effort was concentrated in the northern Inner Sound, where the LPUE was highest (Figures 3.8 & 3.10). There was evidence that fishing effort was significant spatially autocorrelated indicating effort was clustered ($Z=11.61$, $I=0.25$), with the highest effort occurring at the northern edge of the static gear zone (Figure 3.8). Fishing also occurred in the mixed fishing area, but was confined to areas on hard edges and close to rocky ground (Figures 3.8 & 3.9).

Creel fishing is spread over a wide area in the Inner Sound, occurring inside and outside the static gear zone (Figure 3.11). Effort and LPUE did not appear to be evenly distributed throughout the area (Figures 3.11 & 3.12). There was evidence that fishing effort was significant spatially autocorrelated indicating effort was clustered ($Z=5.29$, $I=0.17$). Increased effort around the southern boundary of the BUTEC range is apparent, with corresponding areas of high LPUE (Figures 3.11 & 3.12).

Fishing in Loch Fyne occurs over the majority of the upper loch, but is largely confined to the sides of the loch, with results showing some points to occur on land (a reflection of reduced accuracy of relying on logbook-derived positional data) (Figure 3.13). There was no evidence that fishing effort was spatially autocorrelated indicating effort was randomly distributed (Z=-1.13, I=-0.08).

Regression analysis of the data from the Summer Isles area, Loch Gairloch, Inner Sound and Loch Fyne showed there was no relationship between LPUE and fishing effort (R^2= 0.47%, 0.34%, 0.01% and 0.40%, respectively). There was a strong positive relationship between total landings and fishing effort (R^2= 88.25%, 94.39%, 70.98% and 83.28%, respectively).

3.3.2 Mapping of effort and landings using GPS position recording units

Owing to the limited battery life of PRUs, it was not possible to obtain a continuous record of effort and landings over the 2-year study period. The months from which data are available for the seven vessels fitted with PRUs are illustrated in Table 2.2 (Chapter 2). Few data were collected in the 1st quarter of 2004 and 2005, with only 23 days of fishing recorded from four vessels. For the 2nd, 3rd and 4th quarters, 201, 188 and 160 days of data were recorded, respectively.

The estimated total annual effort in the Torridon area was 1,760,000 creel hauls. Using this figure it was possible to estimate the number of creels fished in each pixel on an annual basis. In the pixels where there was some fishing, the minimum effort in a pixel (90 × 90 m) was 25 creel-hauls, the maximum was 954 creel-hauls and the mean number of creel-hauls per pixel was 123.3, with 25% of pixels being fished by 40 or fewer creel-hauls, 50% by 89 or fewer creel-hauls and 75% by 169 or fewer creel-hauls. The modal fishing intensity (19.7% of pixels) was 30–40 creel-hauls per year (Figure 3.16).

There was evidence that fishing effort was significant spatially autocorrelated indicating effort was clustered (Z=127.45, I=0.60). This clustering of effort in the Torridon area is illustrated in Figure 3.15. Only a limited amount of fishing within the boundaries of the BUTEC range and trawl-only zones (where static gear is banned) was observed. Small numbers of creels were deployed in the mixed fishery area (Figure 3.17). The greatest numbers of creels were deployed in the Inner Sound and lower Loch Torridon, which constitute most of the fishing grounds in the static gear zone. A large area of low effort occurs in the middle of the northern Inner Sound, coinciding with the Rona 'Noise Range' (a facility for measuring the radiated acoustic signature of surface ships, submarines and underwater vehicles), which is periodically closed to fishing (Figure 3.15). Although fewer creels were fished in middle Loch Torridon (Loch Shieldaig) and Upper Loch Torridon, effort was still high in these areas (Figures 3.15 & 3.17). Other areas within the static gear zone, where no fishing has been observed, occur in areas of hard sediment (Figures 3.5 & 3.15 & BGS, Admiralty chart). Some fishing did occur on ground defined by fishermen as 'hard' or as 'hard edges'; however, the majority occurred on ground defined as 'muddy' or 'sandy to firm mud', with roughly equal effort (per pixel) on each ground type (Figure 3.18).

There was a degree of spatial variation in the total LPUE (Figure 3.19), with the highest LPUE from the few creels fished in the trawl only area. High LPUE values were also observed in the mixed and Inner Sound areas, with the lowest values being recorded in middle Loch Torridon (Loch Shieldaig) and lower Loch Torridon (Figure 3.20). Total LPUE differed little among different sediment types (Figure 3.21).

Regression analysis showed there to be no significant relationship between fishing effort and LPUE (R^2= 0.19%), but there was a strong positive relationship between fishing effort and total landings (R^2= 91.14%).

The LPUE of small (39–45 mm CL) animals appeared to be highest in Upper Loch Torridon and near the northern limit of the northern Inner Sound (Figure 3.22). LPUE values in these areas were almost twice those of the other areas, with particularly low LPUE of small animals in the mobile gear zone, BUTEC range and southern Inner Sound (Figure 3.23). The two areas of highest small-animal LPUE appear to occur where there is a mixture of hard and soft sediments, with the highest LPUE on sandy to firm mud and hard edges (Figure 3.5 & 3.24). This difference was more evident when catch was assessed from one vessel, for which the LPUE of small animals was 30% higher in areas sandy to firm mud than in muddy areas (Appendix C; Figures C.1–C.3).

Landings of medium (46–49 mm CL)-sized animals differed spatially, with some evidence of high LPUE values close to land or hard areas (Figure 3.25). On a larger scale there was little difference in LPUE between areas (Figure 3.26) or sediment type (Figure 3.27).

There is an evident spatial pattern in the LPUE of large (>49 mm CL) animals, with the highest landings in the Inner Sound (Figure 3.28). In contrast to the finding for small animals, the highest LPUE values were recorded in the southern Inner Sound, BUTEC range and trawl-only area; LPUE was lower in Loch Torridon, particularly Upper Loch Torridon (Figure 3.29). Unlike small animals, the highest LPUE of large animals occurred in muddy areas, although the actual difference between other sediment types is low (Figure 3.30). A clearer pattern was observed from one vessel; in this case the LPUE of large animals in muddy areas was around 21% higher than from sandy to firm mud areas (Appendix C; Figures C4–C6).

Combined quarterly landings and effort data (2004 and 2005) were analysed to assess temporal changes in the spatial distribution of effort and landings. There was an apparent increase in effort in lower Loch Torridon during the year, coinciding with a decrease in effort in the southern and northern Inner Sound (Figures 3.31–3.33). Effort also varied with sediment type: there were high levels of effort in muddy areas during the 1st quarter of 2004 and 2005 and higher effort on the hard and sandy grounds during the 2nd quarter of 2004 and 2005 (Figure 3.34).

The LPUE for all size categories in all areas was higher in the 2nd quarters compared with the 4th quarters (Figures 3.35 & 3.36). Total LPUE was progressively lower in all areas in the 2nd, 3rd and 4th quarters, with high LPUE values in the mobile gear zone and Inner Sound area, compared with lower Loch Torridon and Upper Loch Torridon in the 1st quarters (Figure 3.37). A similar progressive reduction in total LPUE was observed on each sediment type (Figure 3.38).

The LPUE of small animals was higher in all areas in the 2nd quarters when compared with the 4th quarters, with a particular reduction in LPUE in Loch Torridon and in the north of the northern Inner Sound (Figures 3.39 & 3.40). LPUE of small animals was low in quarters 1 and 4 and highest during the 2nd quarters. This pattern was found in all areas and sediment types (Figures 3.41 & 3.42).

The LPUE of medium-sized *Nephrops* appeared to be higher in all areas in the 2nd quarters compared with the 4th quarters (Figures 3.43 & 3.44). Total LPUE was progressively lower in all areas in the 1st, 2nd, 3rd and 4th quarters, except for the 1st quarter of these years in Loch Torridon, where values were low (Figure 3.45). A similar progressive seasonal reduction in the LPUE of medium-sized *Nephrops* was observed for each sediment type (Figure 3.46).

For large animals a different temporal pattern in the LPUE was recorded, with very little difference between the 2nd and 4th quarters (Figures 3.47 & 3.48). A higher LPUE of large animals was evident during the 1st quarter when compared with the 2nd, 3rd, and 4th quarters. This seasonal pattern was found when data were categorised by either area (Figure 3.49) or sediment type (Figure 3.50).

3.4 Discussion

The use of detailed logbooks and GPS-linked PRUs to investigate small-scale spatial variations of *Nephrops* LPUE has illustrated that there may be considerable stock heterogeneity within relatively small areas. In addition to mapping small-scale variations in LPUE, these methods provide details about fishing practices that relate to how the fishery exploits the stock. Such fishing practices are influenced by a number of factors other than *Nephrops* population characteristics. These include competition amongst fishermen for the best fishing grounds, time of year and benthic topography, all of which will affect the behaviour of individual fishers and therefore stock exploitation. The influence of such factors will be discussed here in detail in relation to microscale distributions of effort and landings and are also considered on a larger spatial scale using Generalised Additive Modelling in Chapter 2. If fisheries research is to be applicable to the activities of commercial fleets, it is essential to consider spatial variations in effort and landings, both on a large scale among areas and on a much smaller scale within fisheries (Kaiser, 1998).

The use of logbooks did not provide the spatial resolution achieved with PRUs, with only point locations being available, which are not accurate representations of a fleet of creels. A degree of human error is also likely where fishermen noted fleet positions. The use of logbooks did, however, highlight some clear differences in the exploitation patterns between the four fisheries investigated, with considerable variability in effort and LPUE observed in all areas, reflecting the complex nature of different fishing grounds.

Interactions with other fisheries were apparent in Loch Fyne, where fishing by the monitored vessel occurs over the majority of the upper loch, but is largely confined to the sides of the loch, owing to displacement of creeling by trawling along the central axis of the loch. Interestingly, effort was randomly distributed over the area fished, a possible reflection of the mixed topography along the edges of the loch and restricted fishing area, leading to small-scale variations in *Nephrops* population structure. Such variations will lead to small-scale differences in catch, and a more random distribution of effort. Interactions between sectors (trawl and creel) were also evident in the Summer Isles, where fishing was restricted to a narrow band of open sea, where seabed topography is less suitable for fishing with towed gear. The concentration of effort in particular locations in this area may be a reflection of having to fish certain areas due to competition for space from other creel vessels. In contrast, in Loch Fyne there is very little competition with other creel vessels, allowing effort to be spread out over a wider area (along the edges of the loch).

The zonal management (Chapter 1; Section 1.3.5) in the Inner Sound enables creel fishing over a wide area, except in the BUTEC no-fishing area. This zonal policy also affects the fishing practice of the three monitored vessels fishing from Loch Gairloch. The majority of their fishing was divided between two areas in which mobile gear is prohibited (Loch Gairloch and the northern Inner Sound), with the majority of this effort focused in the Inner Sound. Although no relationship between LPUE and effort was observed, there does appear to be a higher LPUE in the Inner Sound area, which may be responsible for the higher effort in the area. A number of other creel vessels fish within Loch Gairloch, suggesting that the Inner Sound provides more available space. Some creel fishing was also undertaken in the mixed fishery area, with effort focused around hard ground where the use of mobile gear is restricted by the benthic topography.

These results suggest that fishing effort may to some degree be targeted in areas where catch rate is high (Hilborn & Ledbetter, 1979; Gillis *et al.*, 1993; Hutton *et al.*, 2004). It is clear that effort was also directed in specific areas due to a combination of competition for space with other creel and trawl vessels (Hilborn, 1985; Gillis & Peterman, 1998; Rijnsdorp *et al.*, 2000a) and management restrictions (Pasco *et al.*, 2001; Marchal *et al.*, 2002).

A previous study investigating the microscale distribution of effort from the *Nephrops* trawl fleet in the Sound of Jura showed that trawl vessels were partly excluded from areas of mixed topography, enabling creel vessels to fish in these areas; however, trawling was also restricted in the centre of the Sound by the presence of a creel fishery

(Marrs *et al.*, 2002b). These results suggest that in some cases in mixed fisheries, creel fishers may fish in areas other than those that trawl vessels cannot reach.

The spatial data recorded in logbooks showed large variations in the spatial pattern of effort and landings between fisheries, with competition for suitable fishing ground being a major factor determining the distribution of effort. Although this work has provided a good overview of the difference in the spatial distribution of effort and landings between the different fisheries examined, no firm conclusions can be drawn with regards to the reasons for the observed differences. This is due to the sampling intensity within each fishery: only a small number of vessels were used to assess the exploitation patterns in each fishery, which often comprises several vessels. This small sample may therefore not be representative of the fishery as a whole. Also, as previously identified, the point data did not take into account the length of the creel fleet and there may be a degree of error involved in the position recording, with these factors necessitating a high resolution for analysis of the spatial information.

The use of PRUs in combination with logbooks in the Torridon area enabled a detailed assessment of the microscale distribution of landings and effort. PRUs allowed for the exact position of the entire fleet of creels to be determined, providing a much finer spatial resolution than could be achieved from logbooks alone. The amount of data produced (number of fleets of creels) using this method was much greater than from the detailed logbook study described above. Importantly, data were collected from 7 of the 15 vessels fishing in the Torridon area over a two-year period, providing representative spatial coverage of the fishing grounds.

One area that might be marginally underrepresented is the southern Inner Sound, as a PRU fitted on a vessel known to fish in this area malfunctioned. A further problem with the analysis was that catch was recorded on a daily basis, leading to incidences where catches were aggregated over different areas. Even with these minor problems, this technique allowed a much more realistic map of the spatial distribution of effort and landings to be constructed than could be achieved from logbooks alone, providing information on the fine-scale behaviour of creel fishermen and how this may respond to a variety of factors.

The management measures within the Torridon area mean that this fishery is not typical of the majority of creel fisheries. There are, however, some advantages of carrying out this kind of study in this area. It contains a number of *Nephrops* habitats that are found in other fisheries, including loch environments and deep mud grounds, and because no mobile gear is allowed in the area, it allows conclusions to be drawn with regards to the creel fishery alone.

Fishing effort is almost entirely limited to the static gear zone in which there are clear locations where the ground type is unsuitable for creel fishing. Effort varied considerably within the areas fished. This non-random distribution of effort within fisheries has been observed in a number of previous studies (Rijnsdorp *et al.*, 1998; Friedlander *et al.*, 1999; Jennings *et al.*, 2000; Marrs *et al.*, 2000, 2002a,b). The spatial pattern of effort in the Torridon area is likely to be due to a number of factors. For example, the lower effort from a large area in the northern Inner Sound corresponds with the location of the Rona 'Noise Range', which is periodically closed to fishing. Additionally, the nature of creel fishing means that once hauled, to save time and fuel, most fleets of creels are returned close to where they were hauled. This is particularly true in heavily fished areas, such as Torridon, where space is limited. Such practices inevitably lead to a non-random pattern of fishing effort (clustering), where heavily fished pixels occur alongside other heavily fished pixels. Competition for space will inevitably lead to vessels often being unable to target the most productive grounds, a finding also observed in the Clyde Sea Area trawl fishery (Marrs *et al.*, 2000, 2002a,b).

A fisherman is likely to continue to fish in an area where catches are high (Hilborn & Ledbetter, 1979; Gillis *et al.*, 1993; Hutton *et al.*, 2004), although in the present study, when the relationship between total effort and LPUE was assessed over the whole area, this was not found to be the case. There was a possible correlation between LPUE and effort was in the Inner Sound (particularly around the BUTEC range), but no such pattern was observed in lower Loch Torridon, where areas of high effort do not necessarily relate to areas of high catch rate. This pattern may arise in part because of the proximity of Loch Torridon to port, compared with the Inner Sound and the effect of this on fuel costs. Furthermore, the loch is more sheltered, allowing vessels to continue fishing during bad weather. This suggests that a greater degree of spatial targeting of the most productive grounds may occur when the cost of reaching these areas is greater, which in turn is dependent on the fishermen's ability to target these more productive grounds (Abrahams & Healey, 1980; Hilborn, 1985; Sampson, 1991; Gillis 1999; Salthaug & Aanes, 2003).

Differences in the spatial pattern of LPUE of different size categories of *Nephrops* were recorded throughout the static gear zone. Variations in the catch rates of small and large *Nephrops* were particularly apparent. This variation was mirrored in data collected from catch sampling, which showed that realitivly more larger *Nephrops* were caught in the Inner Sound than in Loch Torridon (Chapter 2). In contrast, the LPUE of both the total and medium-sized *Nephrops*, was in places higher, but overall was observed to be fairly evenly distributed over most of the study area. This is possibly a reflection of the factors influencing the distribution of large and small *Nephrops* having an equal effect on landings of medium-sized and total *Nephrops*. Only in lower and middle Loch Torridon did the catch rate appear to be slightly reduced in comparison with the other areas, possibly an indication of the intensity of fishing in this area.

Microscale mapping allowed analysis of the precise areas where catch rate was highest. The two distinct areas of high catch rate of small *Nephrops* (Upper Loch Torridon and the north of the Inner Sound) contain a mixture of sediments, with the highest LPUE observed around patches of sandy to firm mud. This relationship of higher catches of small animals on harder sediments was particularly evident when the spatial distribution of the LPUE of small animals was assessed from a single vessel, where the potential effect of different fishing practices between vessels was removed. The higher catch rate of larger animals in the Inner Sound appears to reflect the greater proportion of seabed consisting of muddy sediment. As with the small animals, when assessing the spatial LPUE of large animals from a single vessel, the difference in LPUE among sediment types was even more evident. These differences in the spatial distribution of LPUE show a clear relationship between the ground type and *Nephrops* catch composition.

Within the Inner Sound, high levels of fishing intensity were observed around the boundaries of the BUTEC no fishing area. This pattern of high effort was associated with relatively high numbers of large animals in the catch, suggesting that there may be a 'spill over' of large animals from the closed area (Chapman *et al.*, 1975; Chapman; 1980), leading to an increase in fishing intensity (Russ & Alcala, 1996; Sale *et al.*, 2005). A study of the microscale distribution of the beam trawl fishery in the North Sea showed similar areas of high fishing intensity along the borders of closed areas (Rijnsdorp *et al.*, 1998).

It has been well documented that around Scotland, areas of fine sediment are generally characterised by large *Nephrops* at comparatively low population densities, and areas of coarser sediments are characterised by smaller *Nephrops* at higher densities (Chapman & Bailey, 1987; Howard, 1989; Tully & Hillis, 1995). This pattern is evident in the catch composition within the Torridon area. When the spatial distribution of total catch is assessed, the combination of higher catches of small animals in Loch Torridon and higher catches of larger animals in the Inner Sound leads to an overall LPUE that, although patchy, does not vary a great deal throughout the area. The considerable stock heterogeneity recorded in the Torridon area has also been observed in other *Nephrops* fisheries, including the Clyde Sea Area where *Nephrops* have been shown to exhibit considerable variation at the scale of a few kilometres (Thomas, 1965; Tuck *et al.*, 1997b; Tuck *et al.*, 2000; Marrs *et al.*, 2000, 2002a,b).

It is likely that factors other than sediment will determine the size and density of *Nephrops* populations within the Torridon area. This will be particularly true of the differences observed between Loch Torridon and Inner Sound, as the hydrographical characteristics of these sea areas are likely to be very different. This will affect the larval supply to areas as well as food availability, both of which directly affect the *Nephrops* population structure. Future attention should be given to considering larval supply in the area, taking into account residual current flows, water mass residence time and larval supply routes (Bailey *et al.*, 1995).

Of particular interest is the observation that the highest catches of large animals were from a fleet of creels fished in the trawl-only area. It is possible that little trawling occurs close to the boundary of the no-trawling zone, as there may be a risk of accidentally towing away creels. If this is the case, it is likely that very little fishing by either method will occur in this area, allowing *Nephrops* to grow to larger sizes. Unfortunately little is known about trawl effort in this area. However, in other fisheries, trawlers are known to operate very close to areas fished by creels. For this reason it is likely that this ground does experience a normal level of fishing effort from the trawl fishery, implying, if this is the case, that large *Nephrops* may be present in high numbers on this trawl ground. It has been suggested that trawl fisheries catch a higher proportion of smaller animals than those in the creel fishery (Bjordal, 1983; Tuck and Bailey, 2000). If this were the case in this area, it would suggest that the two sectors exploit different components of the population. This aspect would merit further study.

There was little difference in the distribution of effort over a quarterly basis, except for a possible transfer of effort from the Inner Sound into lower Loch Torridon during the winter, when weather conditions would prohibit fishing in the more exposed Inner Sound area. Seasonal patterns of LPUE mirrored those observed in Chapter 2, with catch being related to patterns in the moult cycle and, for females in particular, the reproductive cycle (Farmer 1975). For a detailed discussion of the seasonal variations in catch in the Torridon area see Chapter 2, Section 2.4. Although behavioural factors are likely to have the largest effect on catch, effort was high during the beginning of the year in areas where catch rates of large animals were high. It is therefore possible that the greater effort in these areas led to a significant proportion of larger animals being removed during the beginning of the year and thus not being available to the fishery until a new cohort has grown to this size (the next winter). In general, however, as would be expected, even during the months when a particular component of the stock was caught in low numbers, catches were still higher in particular areas, suggesting that catch is density-dependent throughout the year.

There was some evidence of variation in the seasonal distribution of effort on different ground types. A greater proportion of effort was focused on muddy grounds during the 1st quarter of the year, when large *Nephrops,* typical of this type of ground, were found in their highest numbers. In contrast, during the 2nd quarter of the year, a greater proportion of effort was focused on sandy to firm mud, rocky ground and hard edges where small *Nephrops* were found in their highest numbers (Howard, 1989; Tully & Hillis, 1995). This suggests a certain degree of targeting of effort may occur as different components of the *Nephrops* stock become available to the fishery on different grounds. Such targeting of effort was observed in a study in the Clyde and Sound of Jura trawl fisheries, where fishing occurred disproportionately on sandy or coarse mud grounds (Marrs *et al.*, 2002b).

Results from this study suggest that current assessment methods employed for creel fisheries fail to take into account the spatial variability of landings and effort and may therefore be prone to error. This is particularly relevant for analytical assessments such as LCA and VPA, for which one of the main assumptions is homogeneity of the stock (Chapman & Howard, 1988; Tuck *et al.*, 1997a). A possible solution may be to split landings and effort according to sediment type rather than traditional ICES rectangles. This stratified approach has been shown to provide a potentially more accurate assessment of the Clyde Sea Area *Nephrops* stock, which, like the creel fisheries examined, may vary in terms of population parameters over small spatial scales (Marrs *et al.*, 2000). In order to carry out such a stratified approach for creel fisheries, a considerable amount of further study is essential, both in terms of carrying out similar studies in other areas, as well as providing more fundamental background biological information (population parameters and length frequency distribution of removals) on *Nephrops* populations and the variety of habitats found within creel fisheries. Furthermore, such assessments are very labour intensive and would require considerably more funding if this approach were to be used on a routine basis (Marrs *et al.* 2000).

The information provided from the assessment of the annual distribution of effort, enabled an estimate of the annual *Nephrops* removals within the Torridon static gear zone. Using the average number of creels fished per pixel (123 per 8100 m^2), the average removals per creel of 1.50 *Nephrops* (average number landed and assumed 10% discard mortality) and the burrow density from the area (fluctuated around 0.3 m^{-2} over the past 6 years, Tuck (2005)), the average annual exploitation rate was calculated to be just 7.6%. This information appears to corroborate the conclusion from LCA (Chapter 2; Section 2.5) that the stock within the Torridon area is not currently being overexploited (The recommended harvest ratio for the North Minch is 20%; Bailey, 2005). A similar result was observed from the Sound of Jura and Fladen Ground trawl fisheries, where the annual removals in 2001 were calculated to be 7.8% and 6% of the population, respectively. Both of these values are low in comparison with other

European stocks (Marrs *et al.*, 2002b). Although these results suggest that removals from these two fisheries are very similar, the value in the current study was calculated from only the fished pixels, whereas the previous study used the entire fishery area. Annual removals from the Torridon area are likely to be significantly lower if calculated for the whole area.

Work undertaken during the present study has shown that the environmental impact of creeling is low, but there are some issues relating to bycatch and physical disturbance (Chapters 5 & 6). The heterogeneous environments that make up the majority of creel fisheries will lead to a number of different habitats being found over relatively small spatial scales. Such spatial variation was observed to affect bycatch and the impact on important structural species such as seapens, in the Torridon area (Chapters 5 & 6). The heterogeneity of environments combined with the observed patchy distribution of effort in the Torridon area suggests that the impact of creel fishing will not be evenly spread. At present, there is insufficient information on the biological characteristics of the Torridon area and the potential impact of creeling to allow any conclusion to be drawn with regard to potential microscale impacts of creel fishing within the area. Previous work has suggested that uneven distribution of effort within a fishery may be beneficial, as areas of low effort may provide refuges for some sensitive species, which act as potential sources of recruits to recolonise surrounding heavily impacted areas (Pitcher *et al.*, 2000).

The present study has highlighted the potential of PRUs to provide fishery information with a high spatial and temporal resolution, enabling the assessment of factors determining the exploitation pattern of the fishery. The *Nephrops* stocks studied were found to be very spatially heterogeneous, with sediment type correlating with catches of small and large animals. There was some qualitative evidence that effort was targeted on areas of high catch rates. However, other factors, including; distance from port, sea conditions, management restrictions and competition for resources from both creel vessels and trawl vessels, all determine the spatial distribution of effort. This detailed understanding of the behaviour of fishing vessels is essential to the management of a fishery, in particular an expanding fishery such as the one studied.

3.5 *Summary*

- Microscale mapping of fishing effort and landings was initially undertaken using detailed logbooks, completed by fishermen, which provided the location, number of creels, and weight of small, medium and large *Nephrops* landed for each fleet of creels fished by the sample vessels. Areas investigated were around the Summer Isles, Gairloch, the Inner Sound and Loch Fyne.

- A non-random distribution of effort and landings was observed in all areas except Loch Fyne.

- Differences in the spatial pattern of effort and landings were observed between fisheries, with a major factor determining the distribution of effort being competition for suitable fishing ground from other creel and trawl vessels. Other factors included management measures and bottom topography, with little apparent effect of catch rate.

- A second method used a combination of logbook and PRUs, on seven boats, representative of the fishing fleet, to provide a more accurate assessment of the microscale distribution of effort and landings in the Torridon area.

- Fishing effort was almost entirely confined to the static gear zone and was non-randomly distributed throughout the area with distinct areas of high effort.

- Results scaled to the entire fishing fleet showed that on average a pixel (90 × 90 m) was fished by 123 creels.

- The patchy distribution of effort will lead to an uneven distribution of environmental disturbance.

- A patchy distribution of LPUE was observed throughout the area, illustrating considerable stock heterogeneity throughout, with high catch rates of small animals being correlated with fishing on sandy to firm mud and high catch rates of large animals on softer mud. This illustrates that the distribution of sediment has an evident effect on catch rate.

- Some qualitative evidence for targeting of effort in areas of high LPUE was observed in the Inner Sound. In the majority of areas this was not the case, with other factors such as distance from port and competition for ground from other creel vessels being found to be more plausible reasons for the observed distribution of effort.

- In the Torridon area the highest catch rates of large animals were observed in the trawl-only area.

- Seasonal variations in catch rate mirrored those observed in Chapter 2, with some evidence for a seasonal increase in effort on grounds where catches of different components of the stock become more readily available to the fishery.

- The estimated annual removals from the Torridon area were calculated to be 7.6% of the total population, suggesting that exploitation was relativity low.

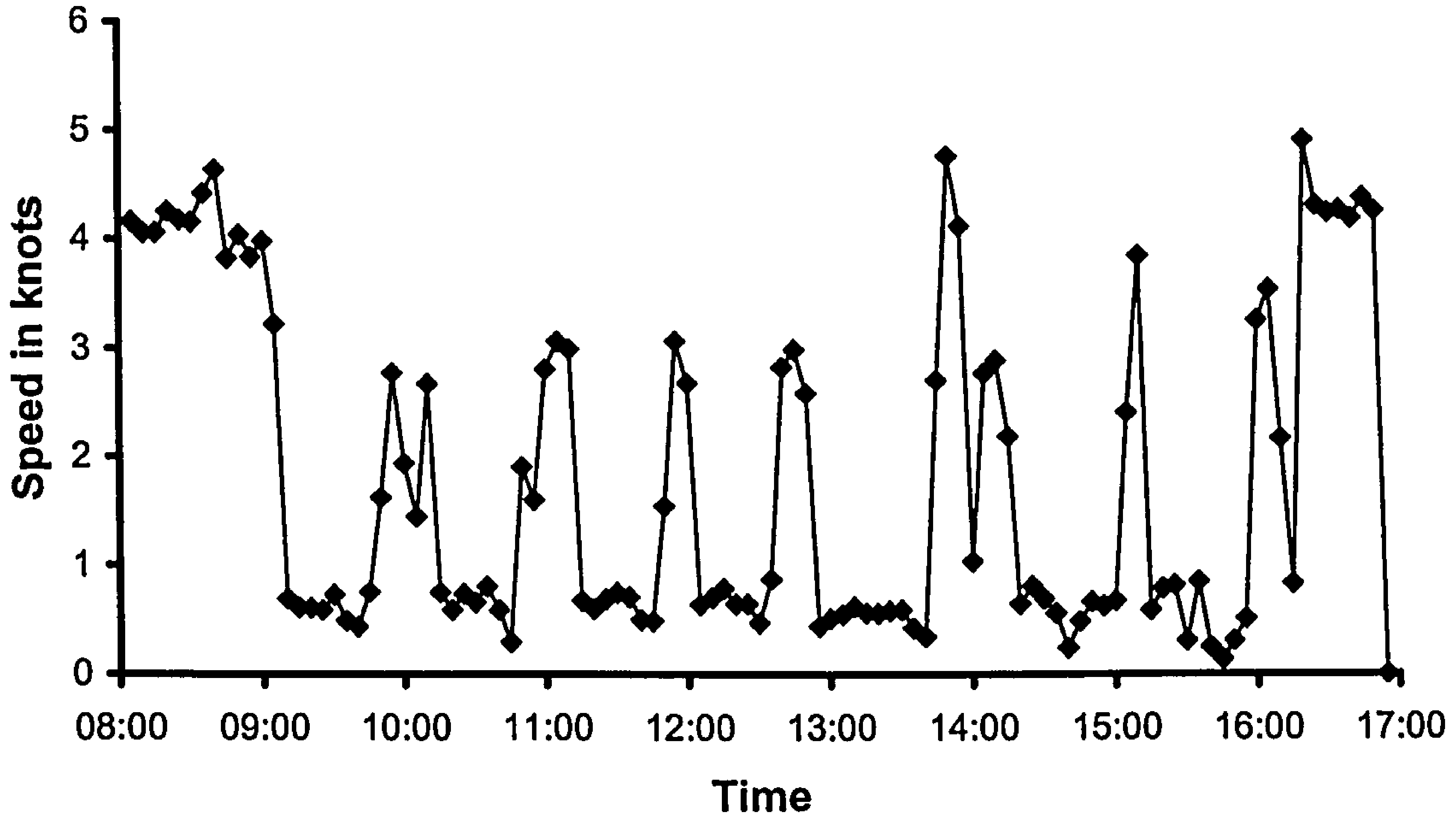

Figure 3.1 – Variations in speed (determined from GPS-records) of a creel fishing vessel for a typical day in the Inner Sound area in March 2004.

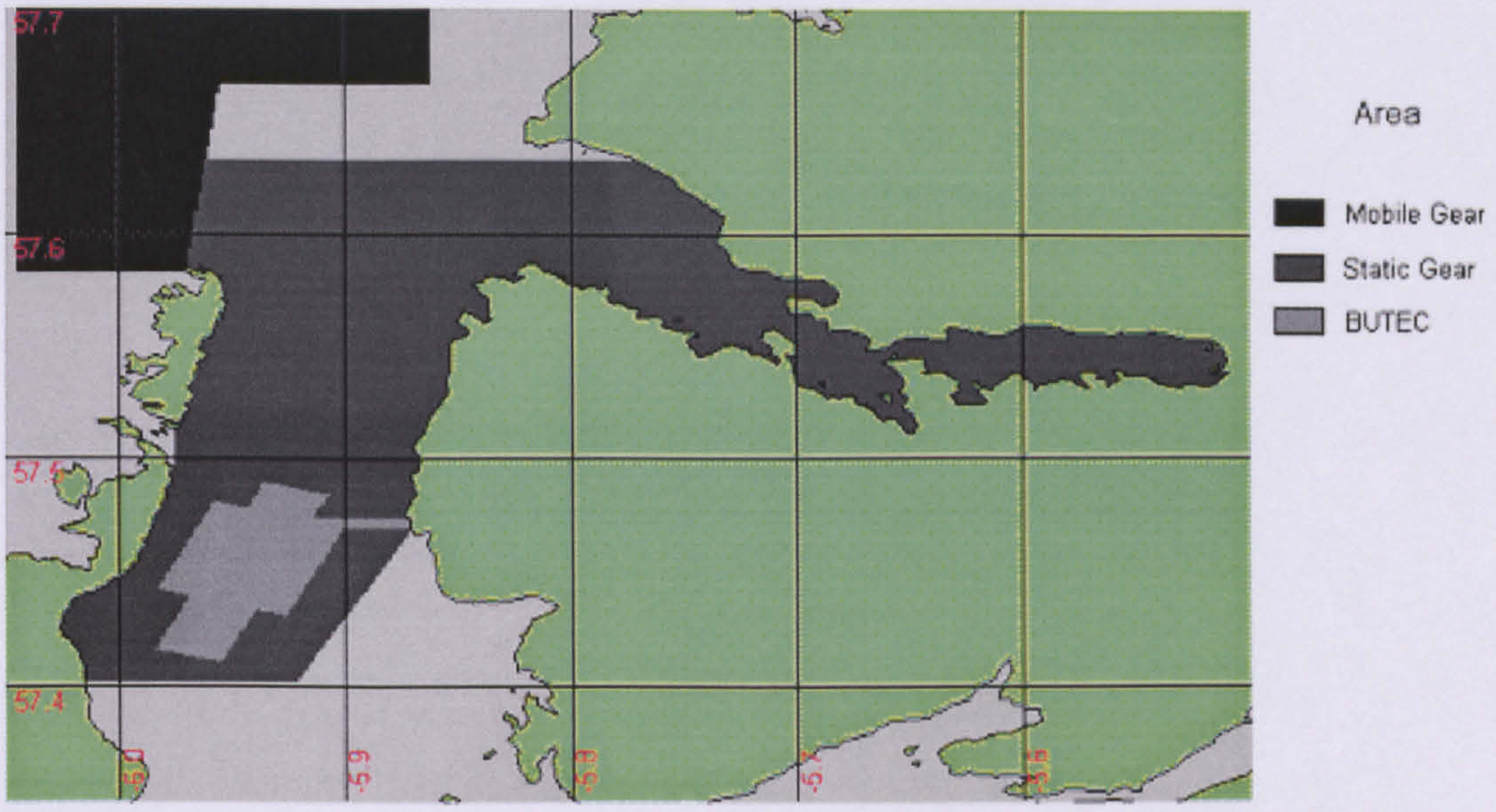

Figure 3.2 – Fishing zones in the Torridon area. Grid lines are labelled with decimal latitudes and longitudes.

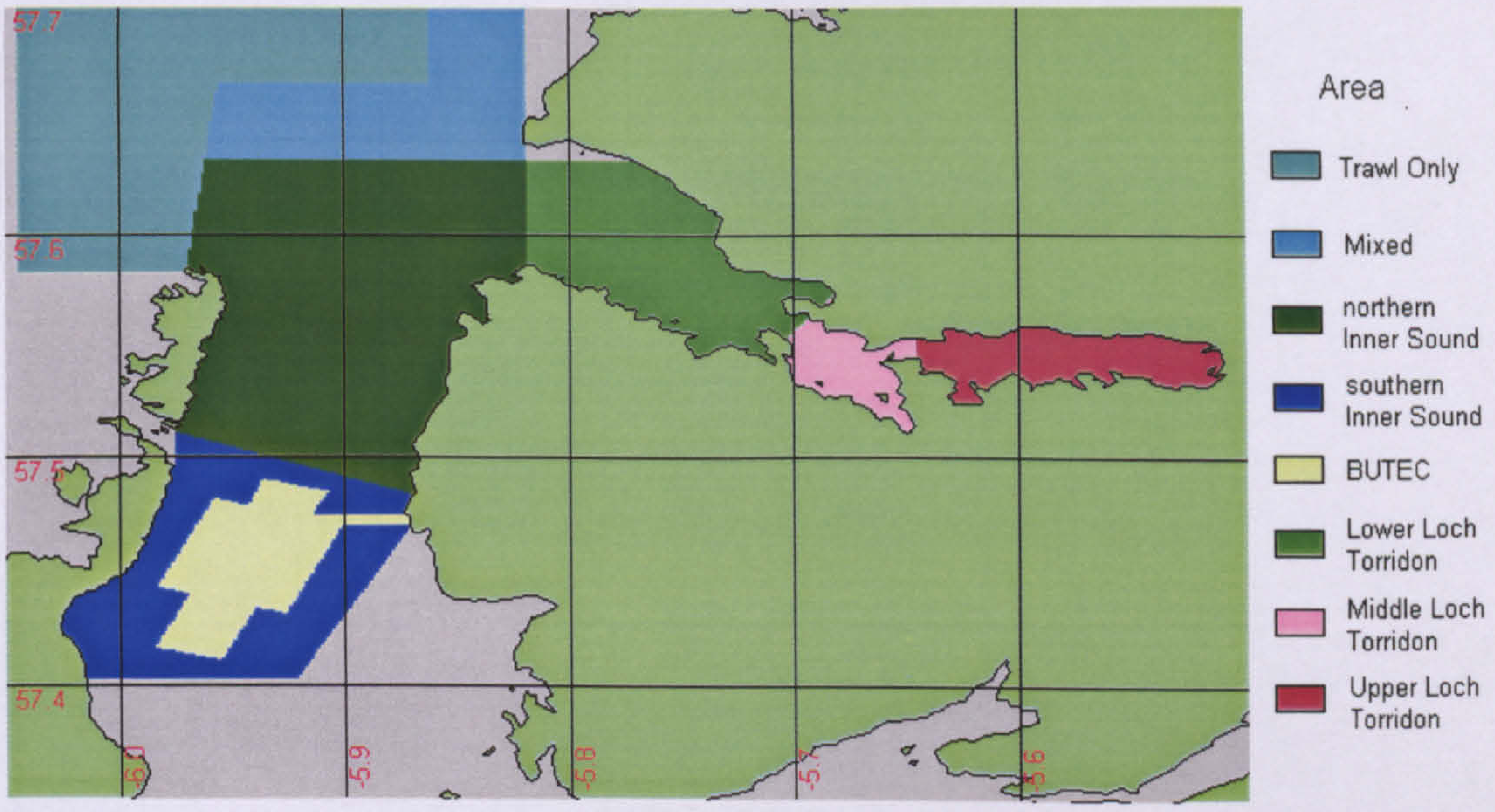

Figure 3.3 – Division of fishing zones in the Torridon area used to describe spatial variations in effort and catch.

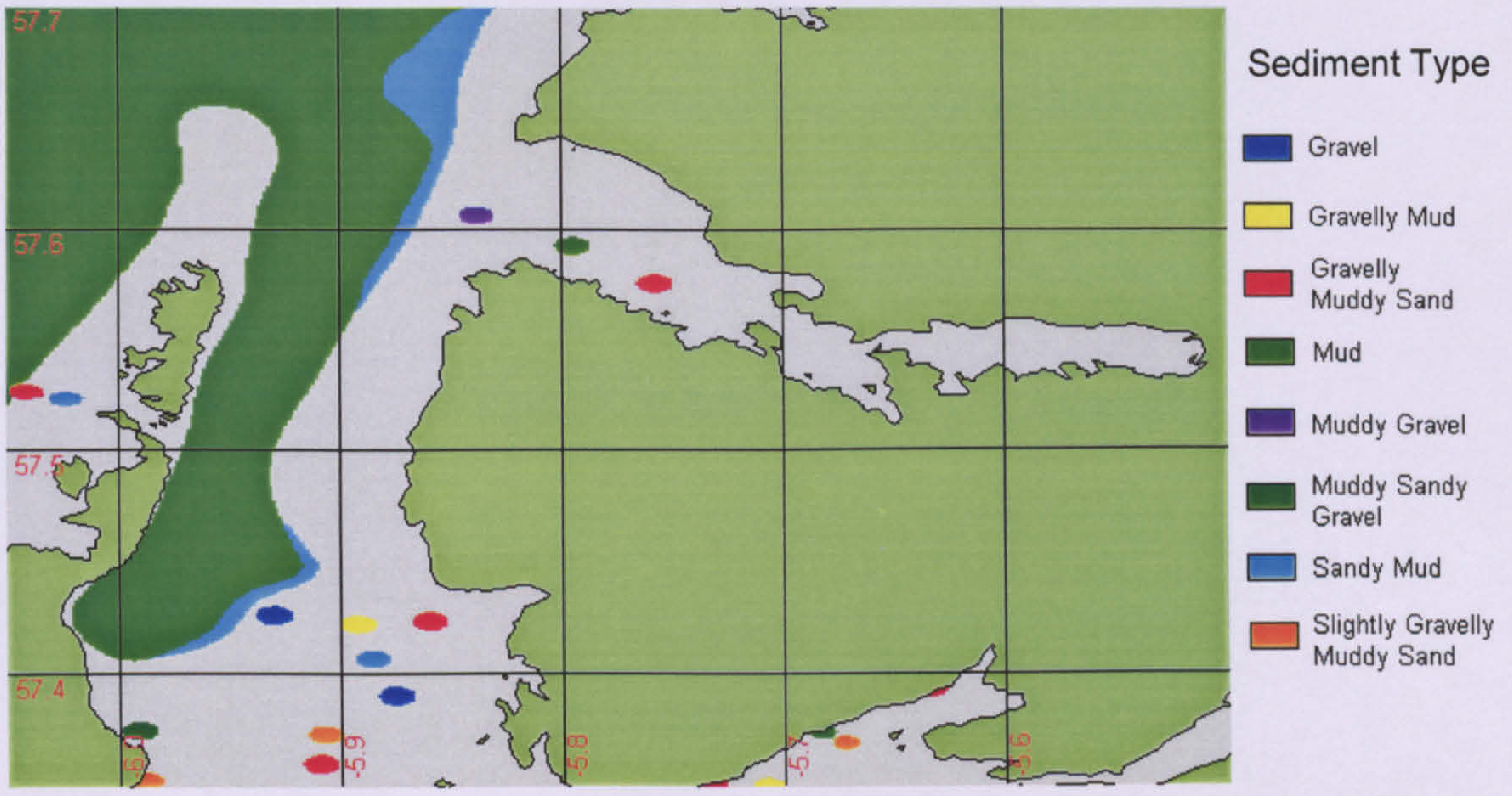

Figure 3.4 – Sediment distribution in the Torridon area as indicated by British Geological Survey data (grey area represents no data).

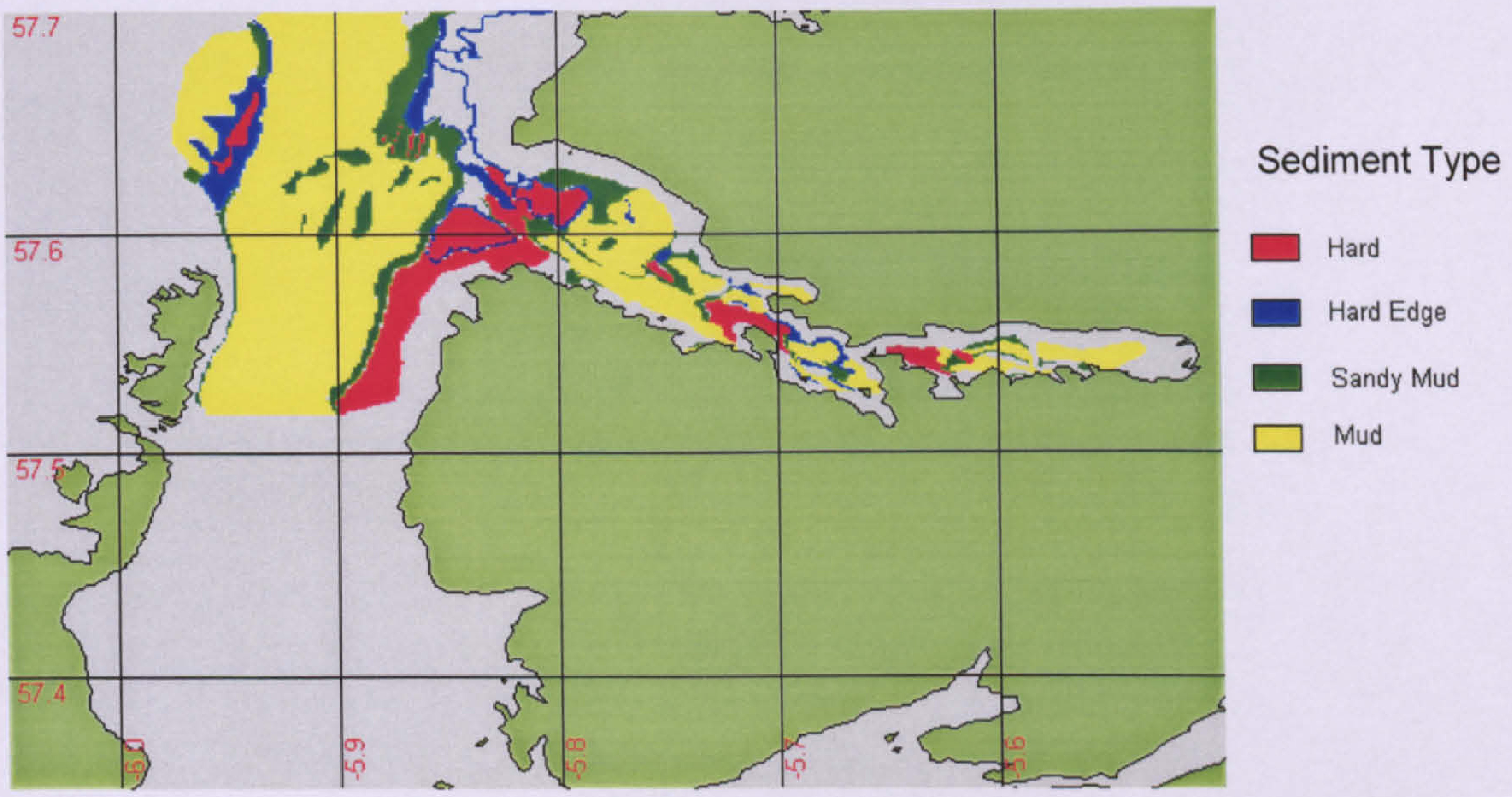

Figure 3.5 – Sediment distribution in the Torridon area based on local fishermen's descriptions (grey area represents no data).

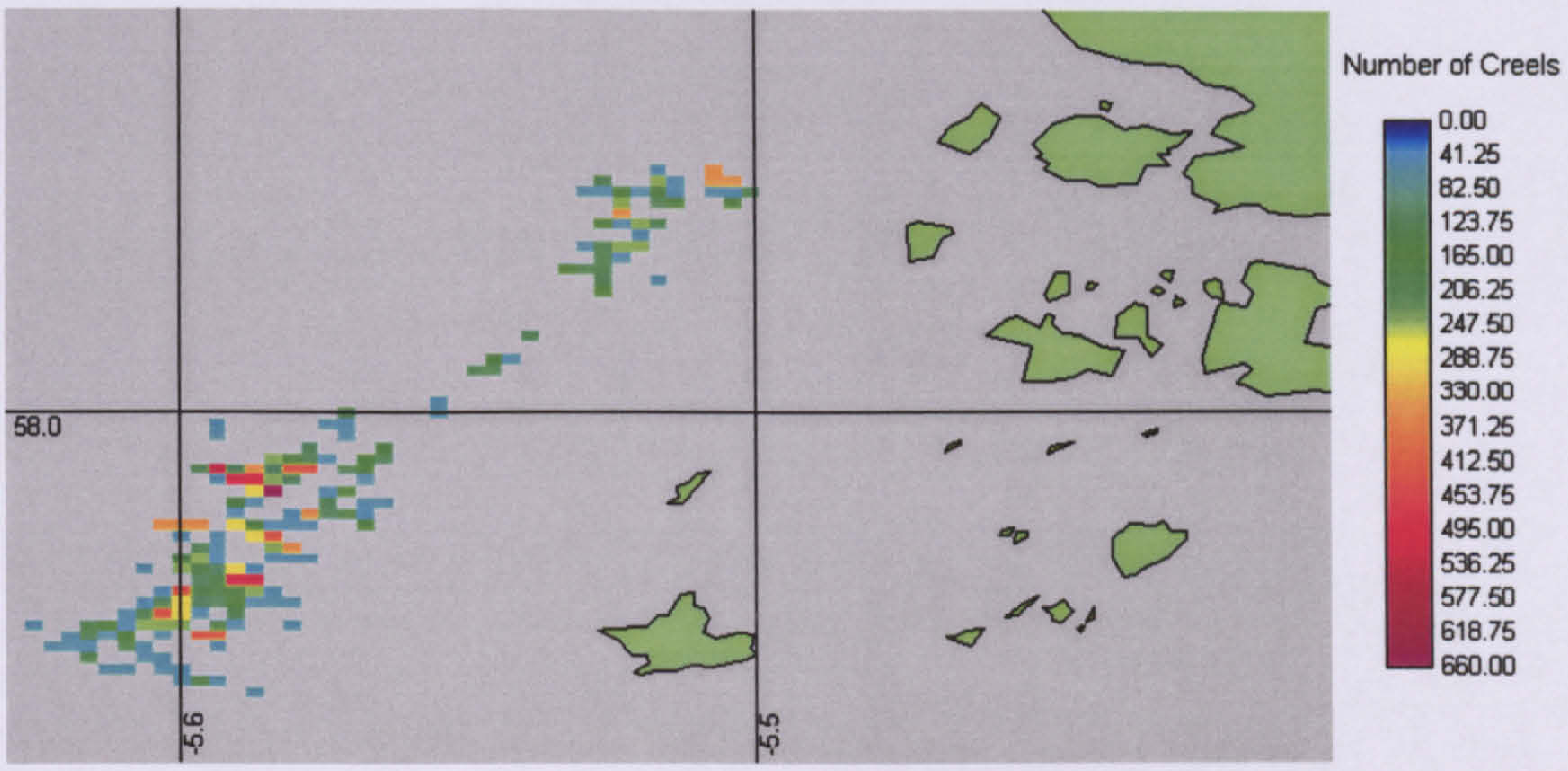

Figure 3.6 - Spatial pattern of total effort based on logbook data from one vessel in the Ullapool (Summer Isles) area, September 2004–March 2005 (57 days fishing).

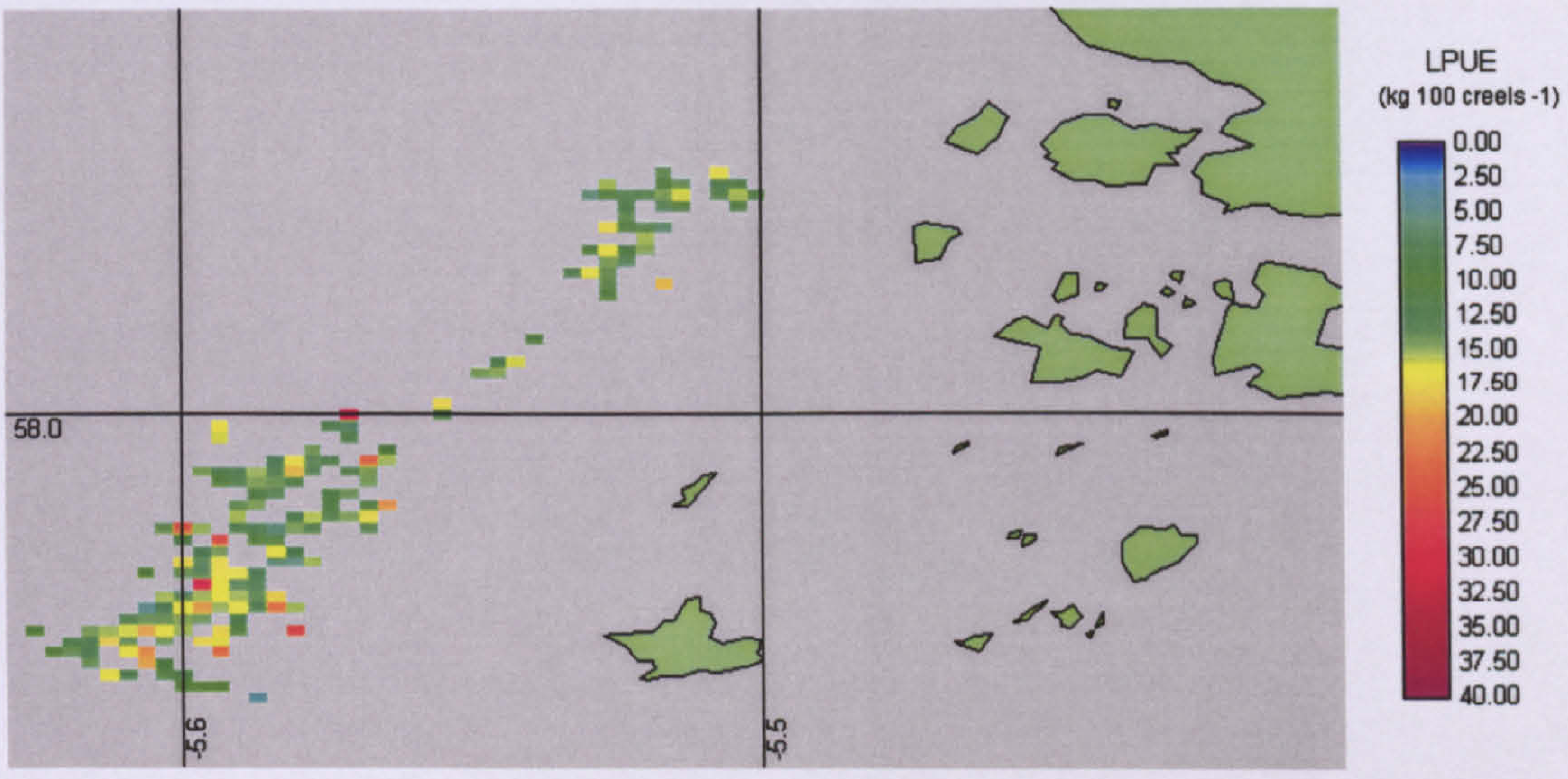

Figure 3.7 - Spatial pattern of total LPUE (kg 100-creels^{-1}) of *Nephrops* based on logbook data from one vessel in the Ullapool (Summer Isles) area, September 2004–March 2005 (57 days fishing).

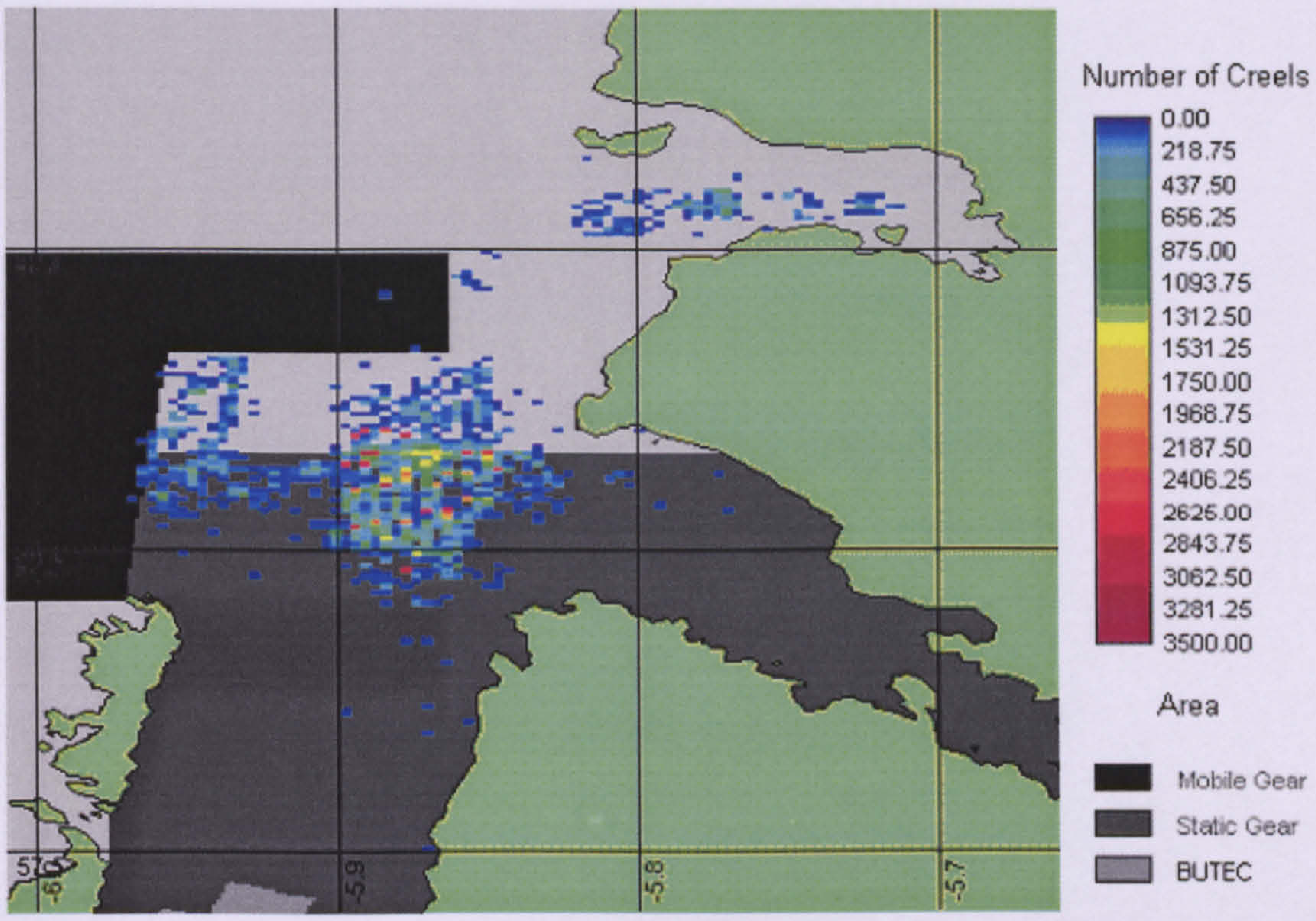

Figure 3.8 - Spatial pattern of total effort based on logbook data from three vessels in the Gairloch and Torridon areas, March 2004–September 2005 (480 days fishing).

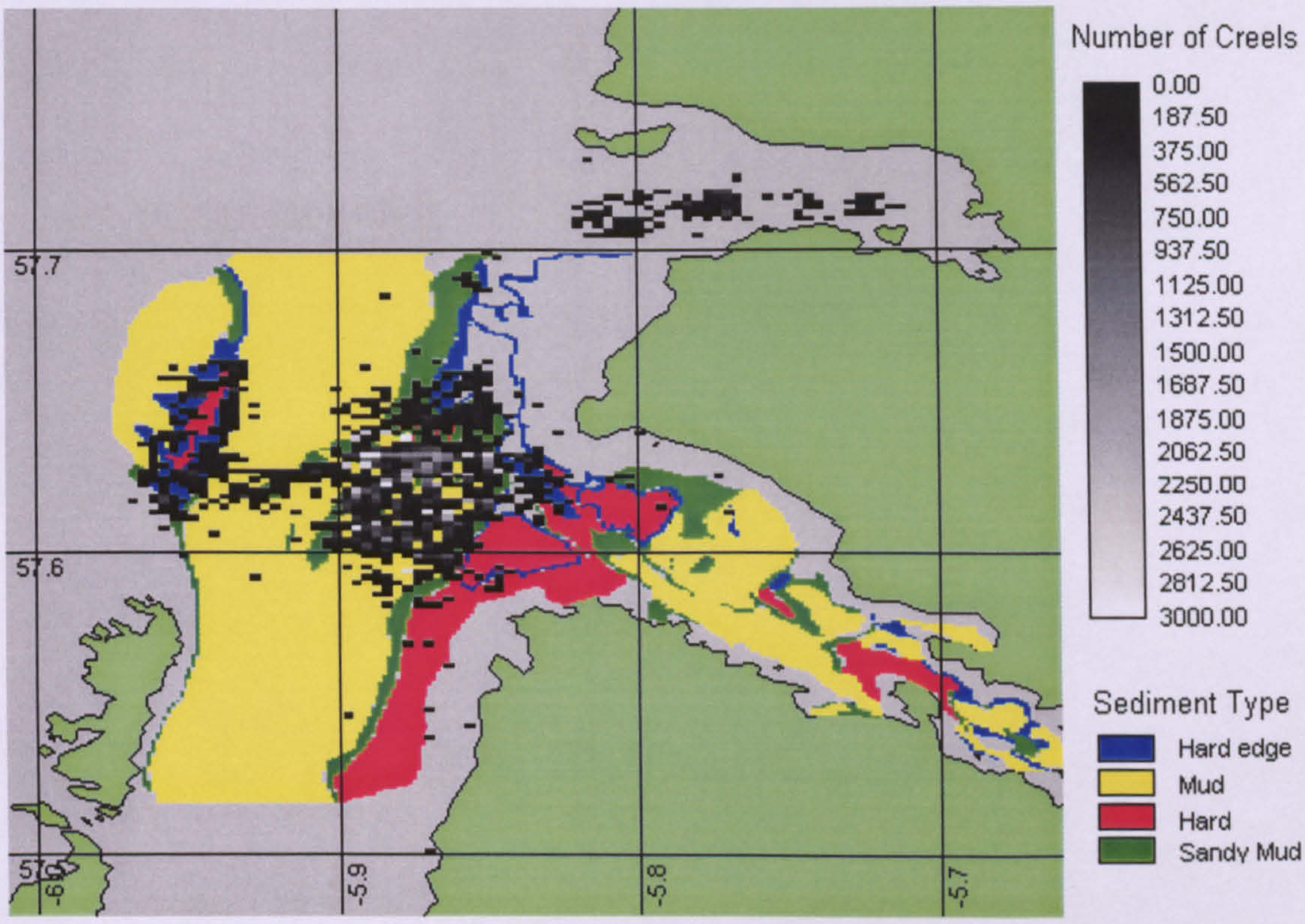

Figure 3.9 - Spatial pattern of total effort based on logbook data from three vessels in the Gairloch and Torridon areas, March 2004–September 2005 (480 days fishing), showing sediment distribution (derived from fishermen).

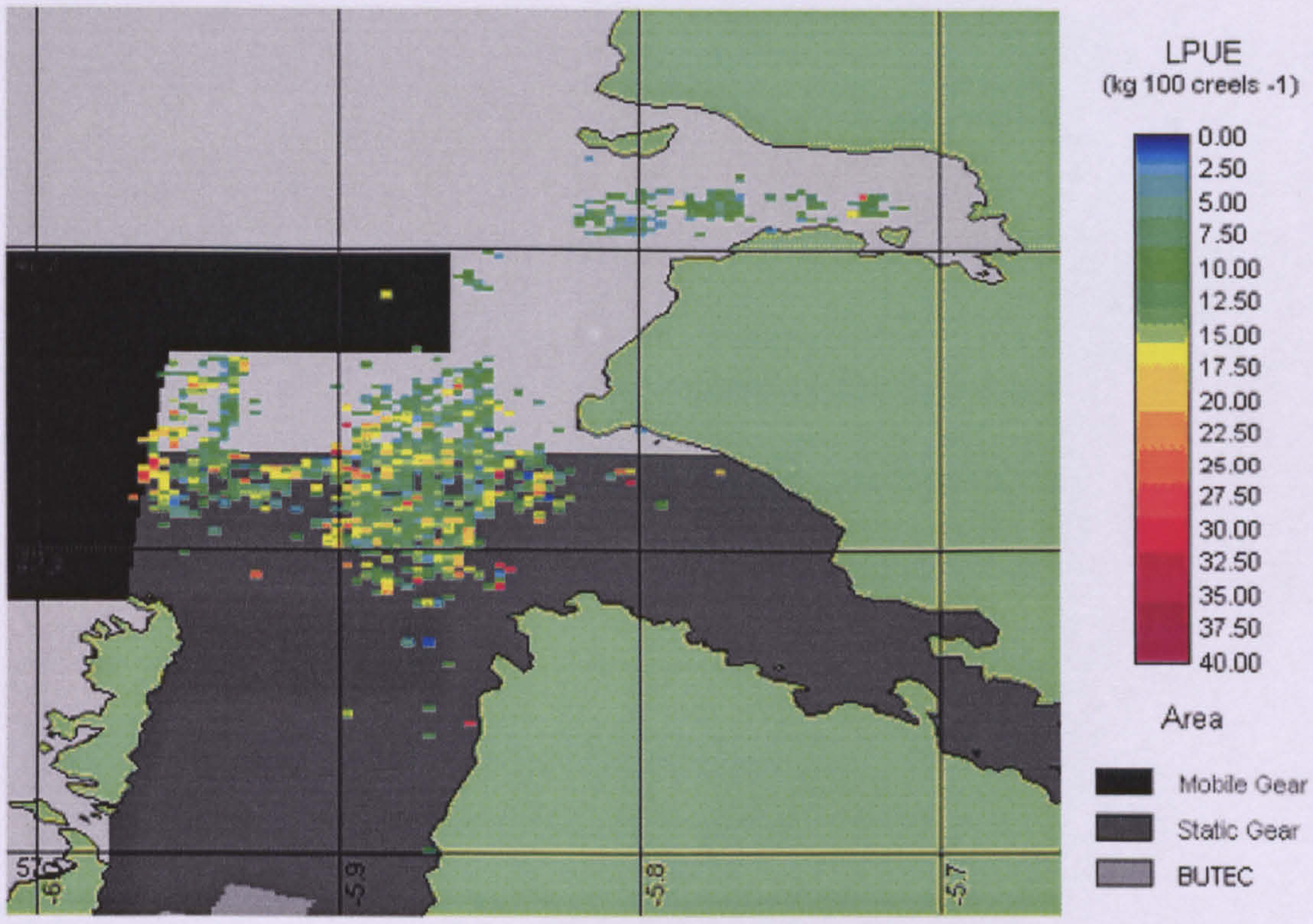

Figure 3.10 - Spatial pattern of total LPUE (kg 100-creels^{-1}) of *Nephrops* based on logbook data from three vessels in the Gairloch and Torridon areas, March 2004–September 2005 (480 days fishing).

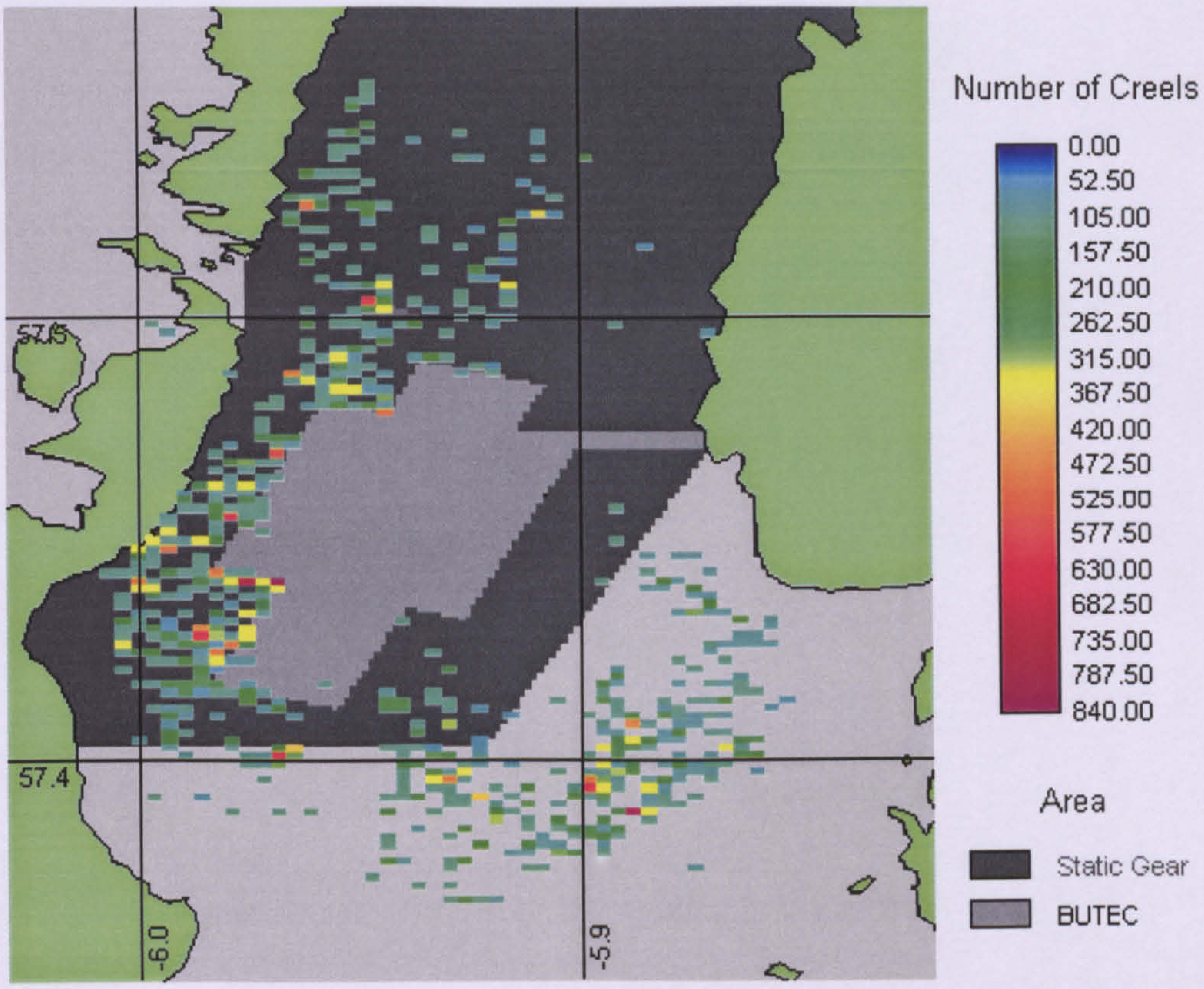

Figure 3.11 - Spatial pattern of total effort based on logbook data from one vessel in the Torridon area, March 2004–September 2005 (148 days fishing).

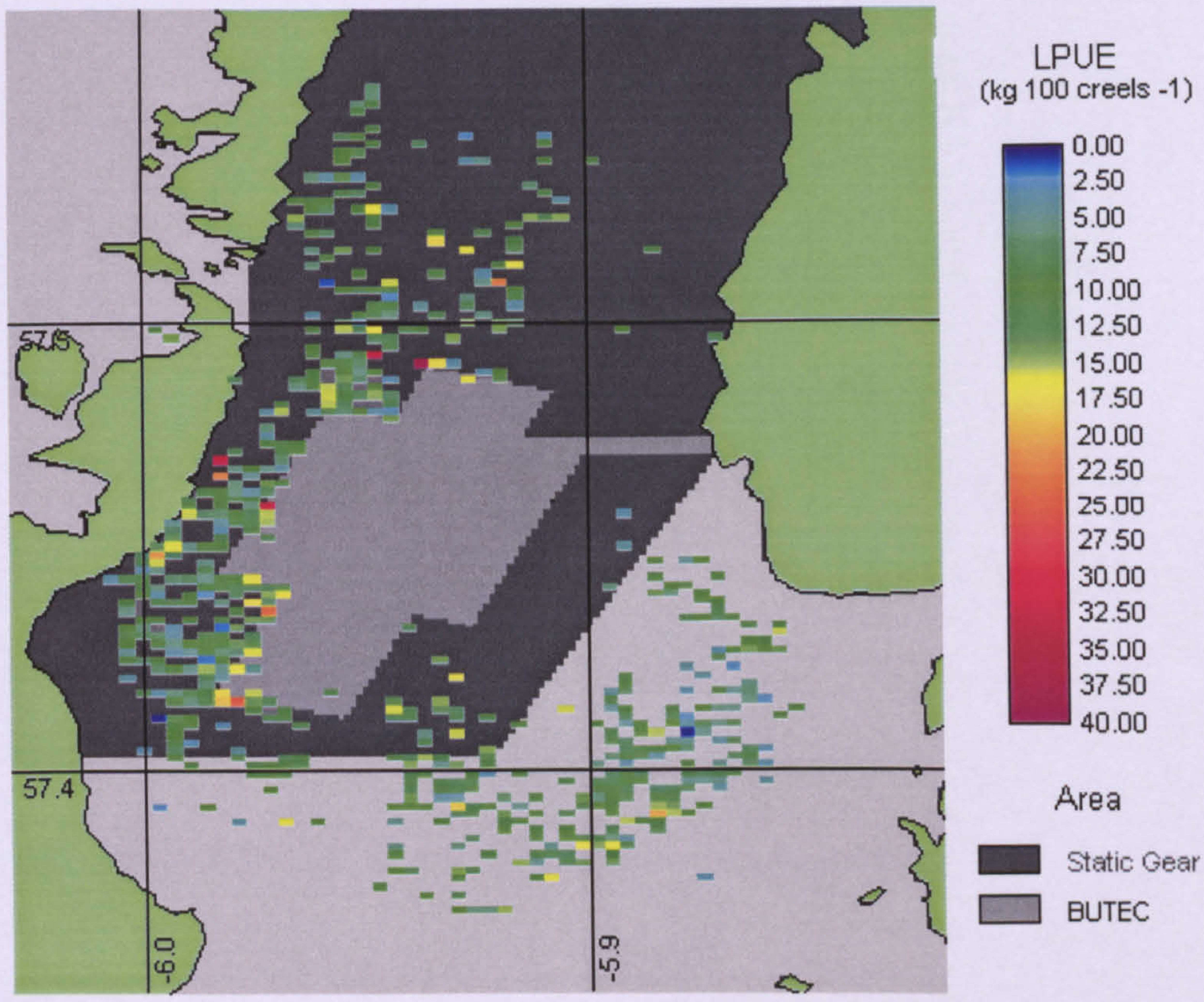

Figure 3.12 - Spatial pattern of total LPUE (kg 100-creels^{-1}) of *Nephrops* based on logbook data from one vessel in the Torridon area, March 2004–September 2005 (148 days fishing).

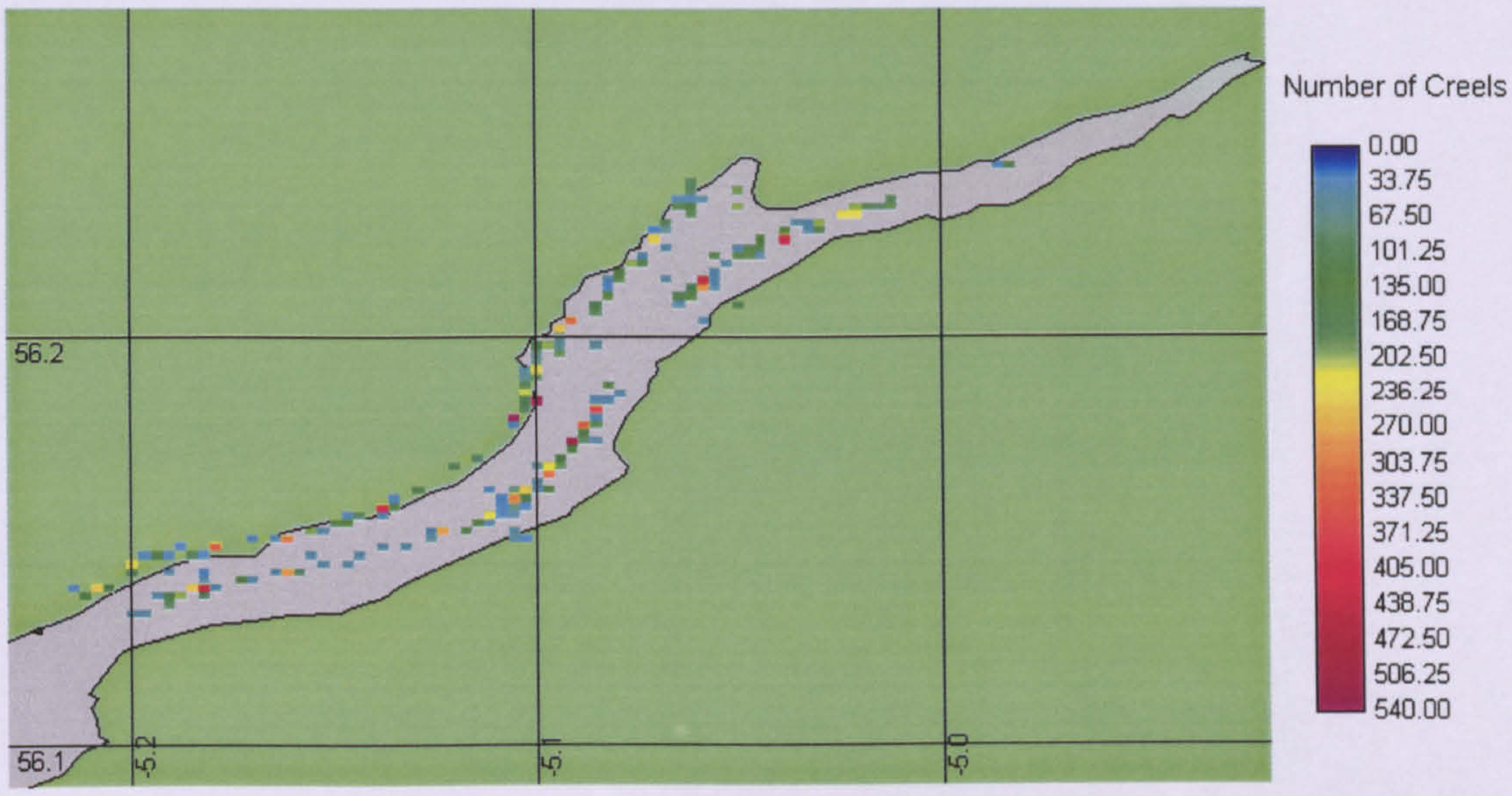

Figure 3.13 - Spatial pattern of total effort based on logbook data from one vessel in Loch Fyne, January–July 2005 (71 days fishing) (some data on land due to positional recording error in logbook, but trend is clear).

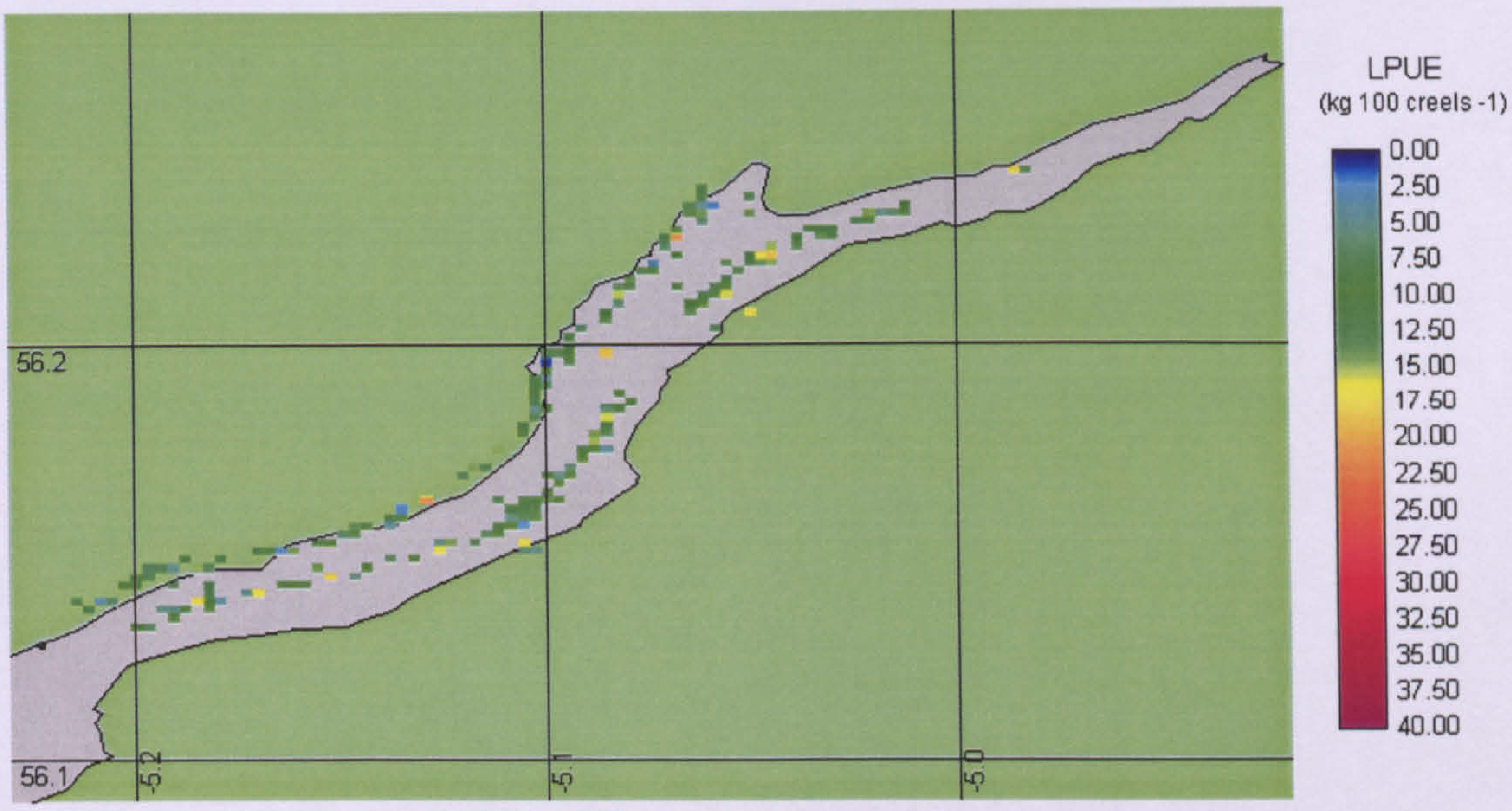

Figure 3.14 - Spatial pattern of total LPUE (kg 100-creels^{-1}) of *Nephrops* based on logbook data from one vessel in Loch Fyne, January–July 2005 (71 days fishing).

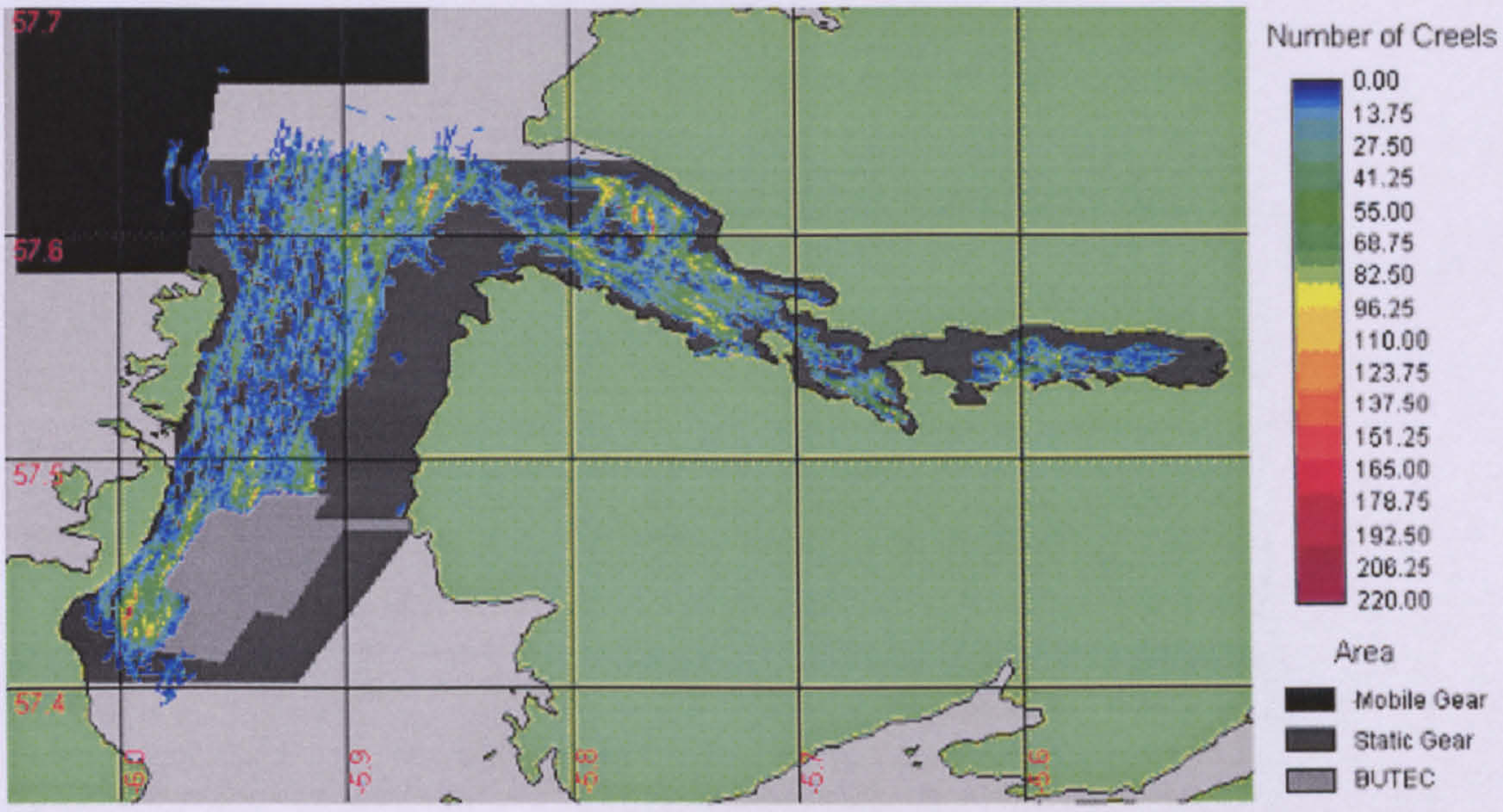

Figure 3.15 - Spatial pattern of scaled effort from the Torridon (Total annual fleet effort) *Nephrops* fishery, using GPS data logger and logbook data (original data collected from seven vessels in 2004 and 2005 (572 days fishing, 389,883 creels)).

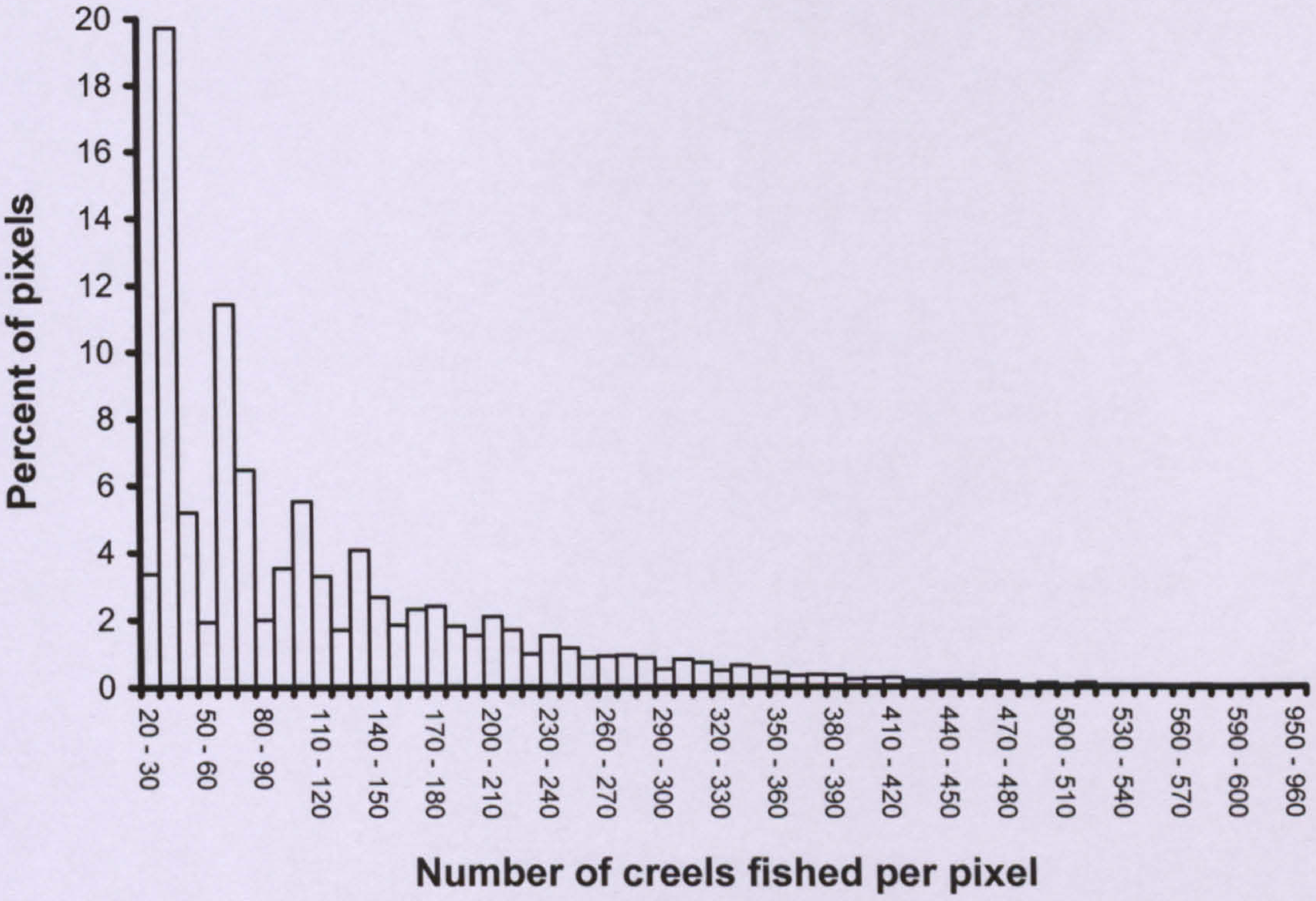

Figure 3.16 – Frequency distribution of fishing effort per pixel (90 × 90 m) scaled per year for the whole fleet of vessels and those pixels in which non-zero effort was recorded.

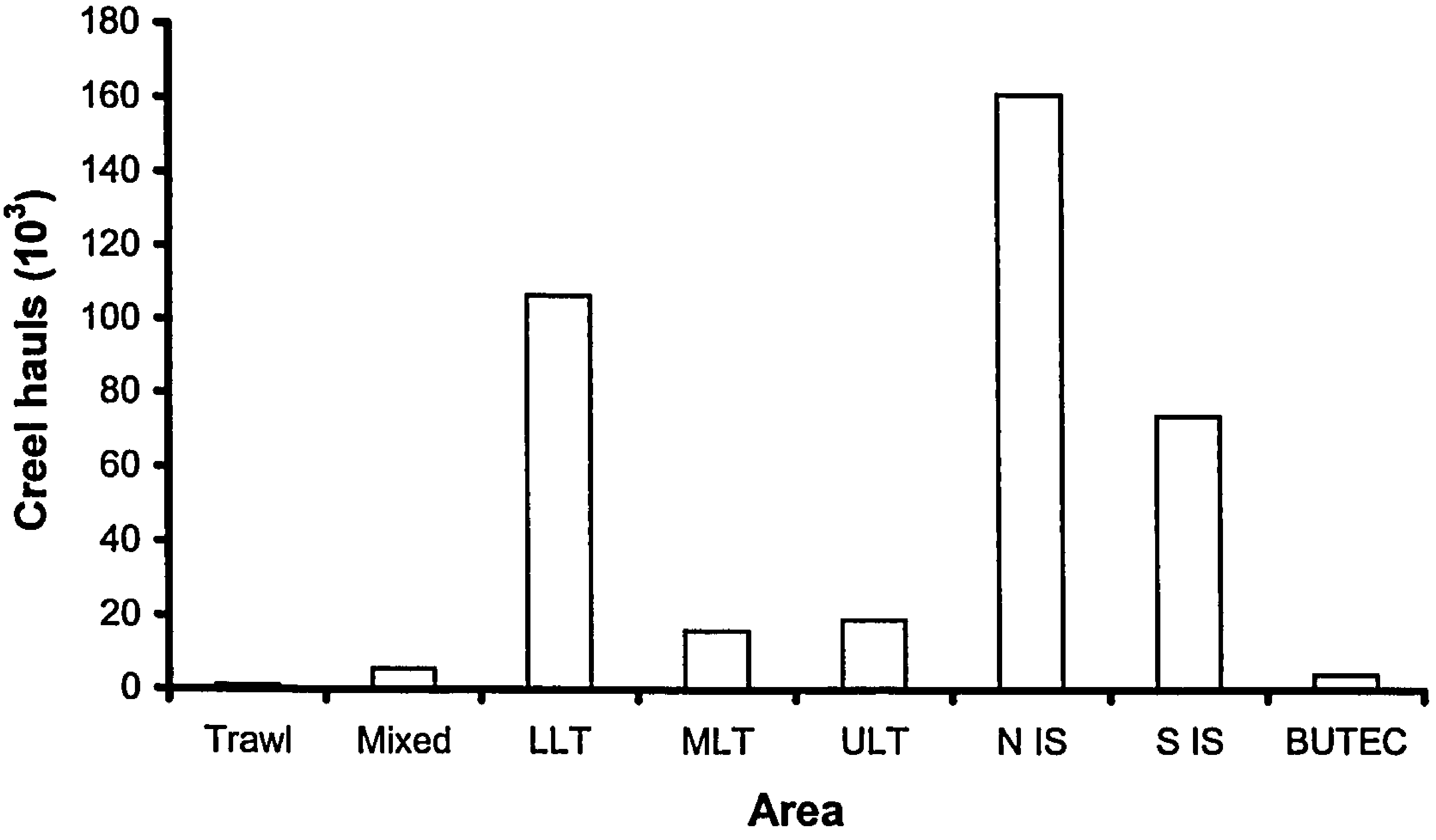

Figure 3.17 - Distribution of total effort in different fishing areas (data scaled to one year for whole fleet), where Trawl=trawl only zone, Mixed=Mixed fishery, LLT=lower Loch Torridon, MLT=middle Loch Torridon, ULT=Upper Loch Torridon, N IS=north Inner Sound, S IS=south Inner Sound and BUTEC=BUTEC no fishing zone.

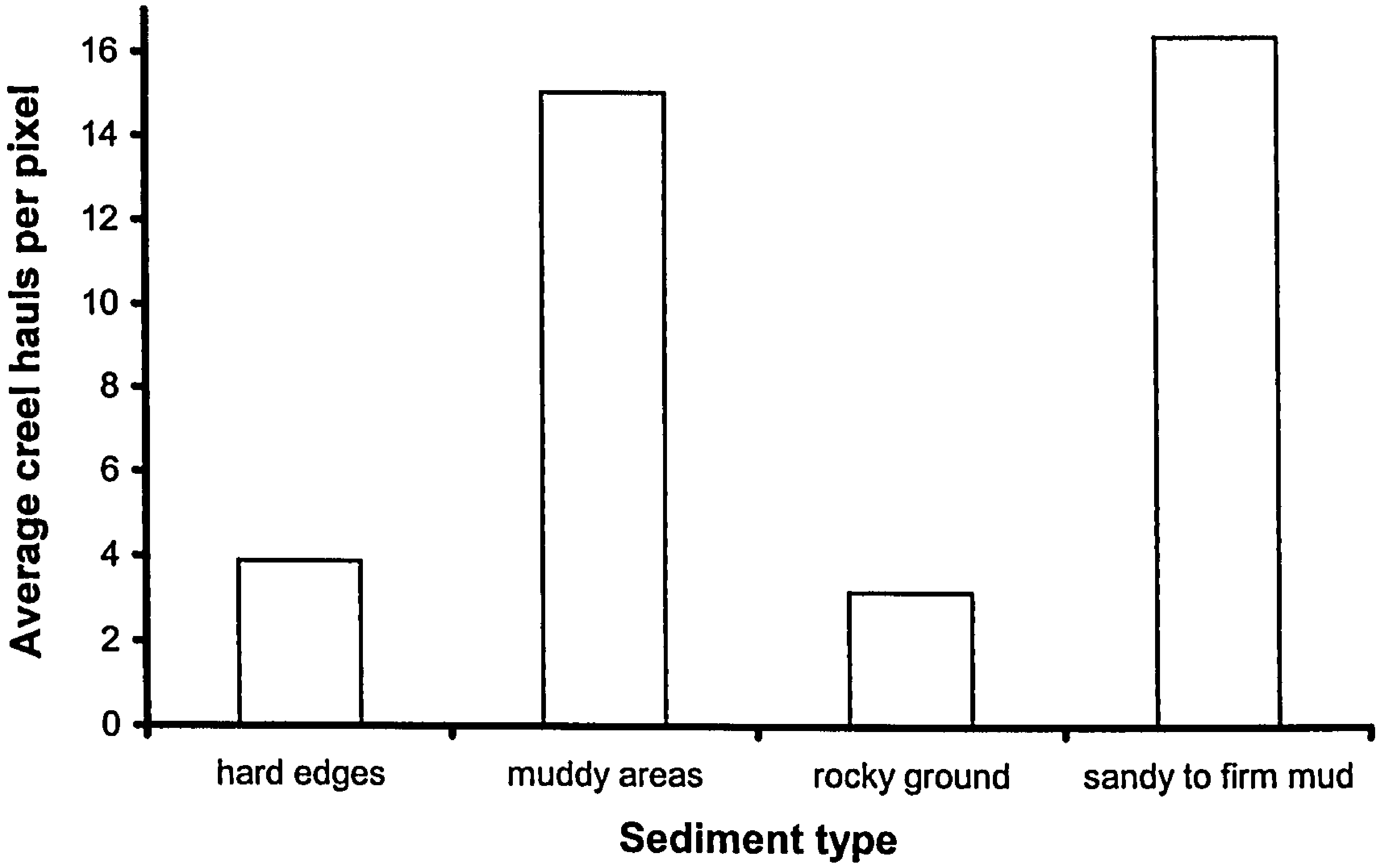

Figure 3.18 - Distribution of average effort in the Torridon area per pixel (90 × 90m) from different sediment types, seven vessels in 2004 and 2005.

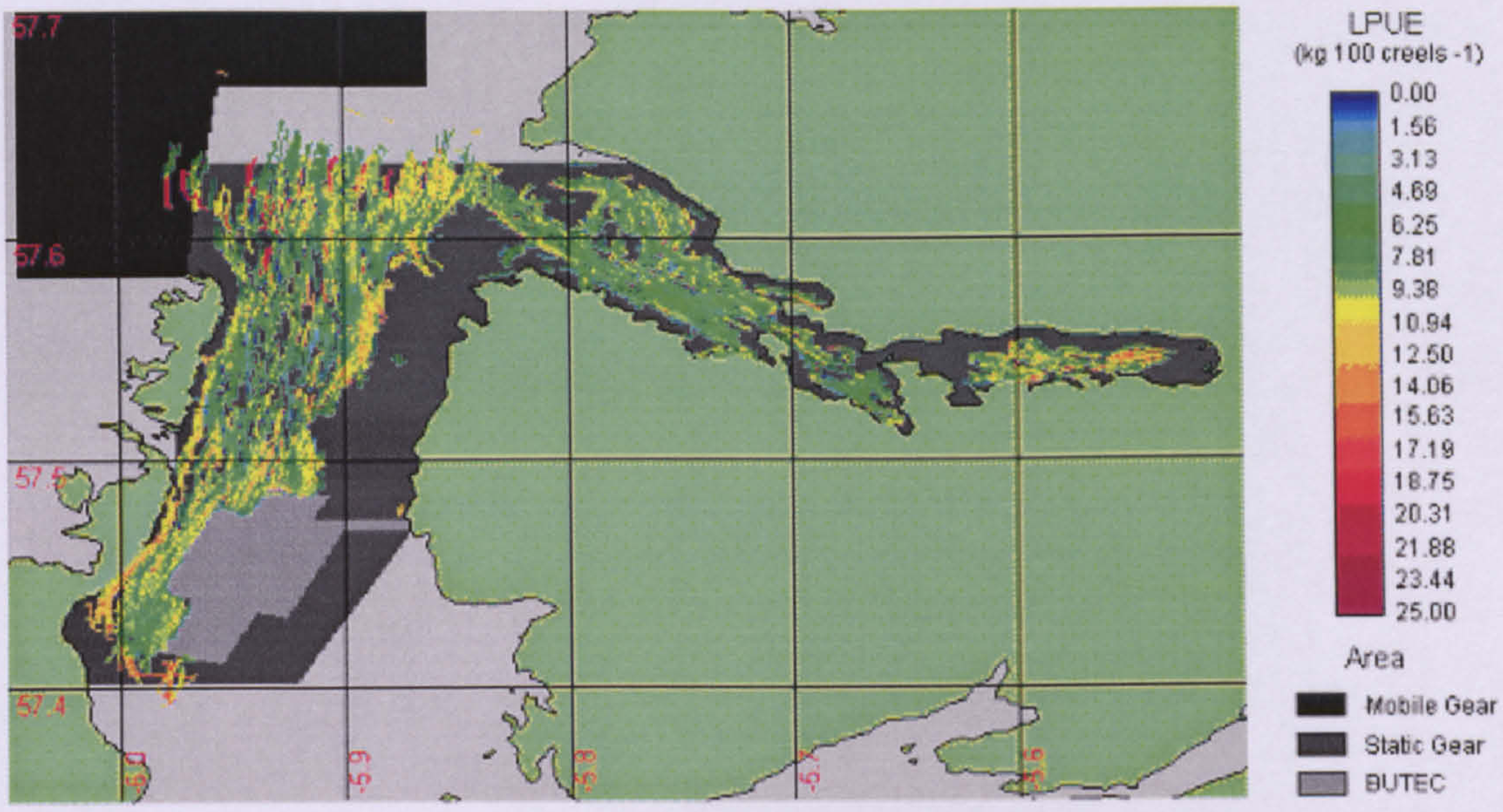

Figure 3.19 - Spatial pattern of LPUE (kg 100-creels^{-1}) of *Nephrops* (all categories) from the Torridon fishery, using GPS data logger and logbook data, seven vessels in 2004 and 2005 (572 days fishing).

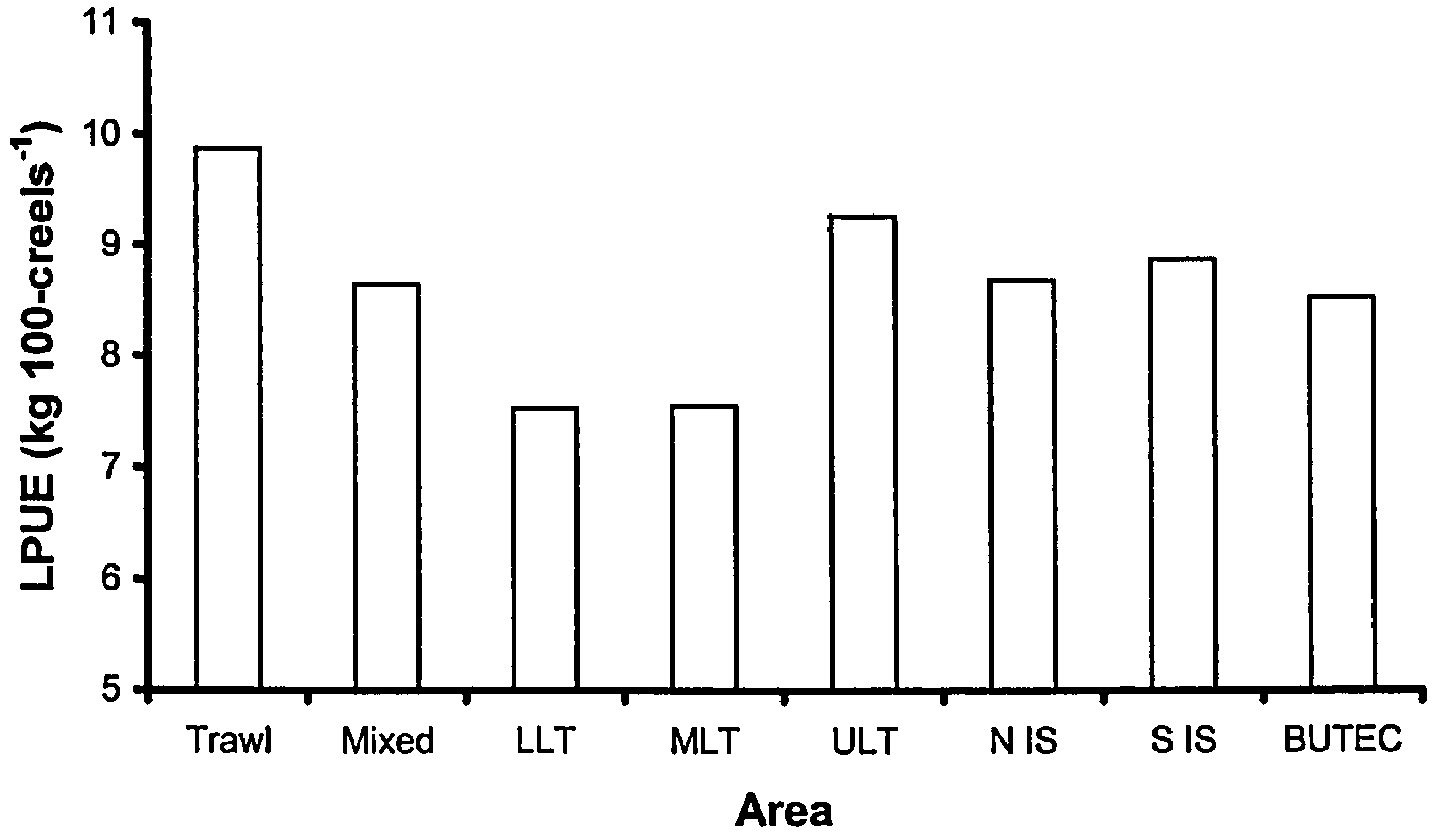

Figure 3.20 – Average LPUE of *Nephrops* (all categories) averaged over pixels in different fishing areas in the Torridon creel fishery, seven vessels in 2004 and 2005, where Trawl=trawl only zone, Mixed=Mixed fishery, LLT=lower Loch Torridon, MLT=middle Loch Torridon, ULT=Upper Loch Torridon, N IS=north Inner Sound, S IS=south Inner Sound and BUTEC=BUTEC no fishing zone.

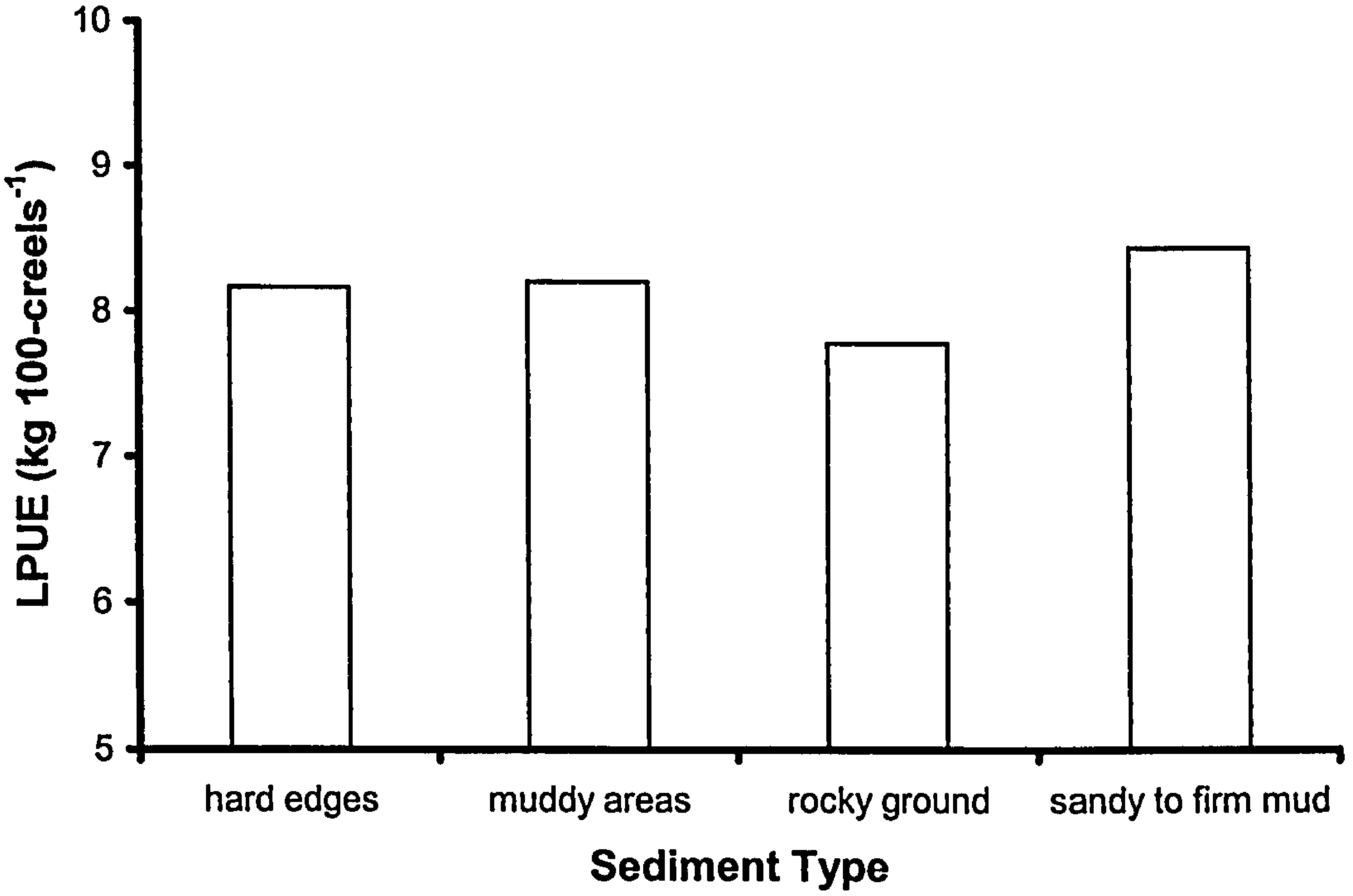

Figure 3.21 – Average LPUE of *Nephrops* (all categories) averaged over pixels in areas of different sediment types, seven vessels in 2004 and 2005.

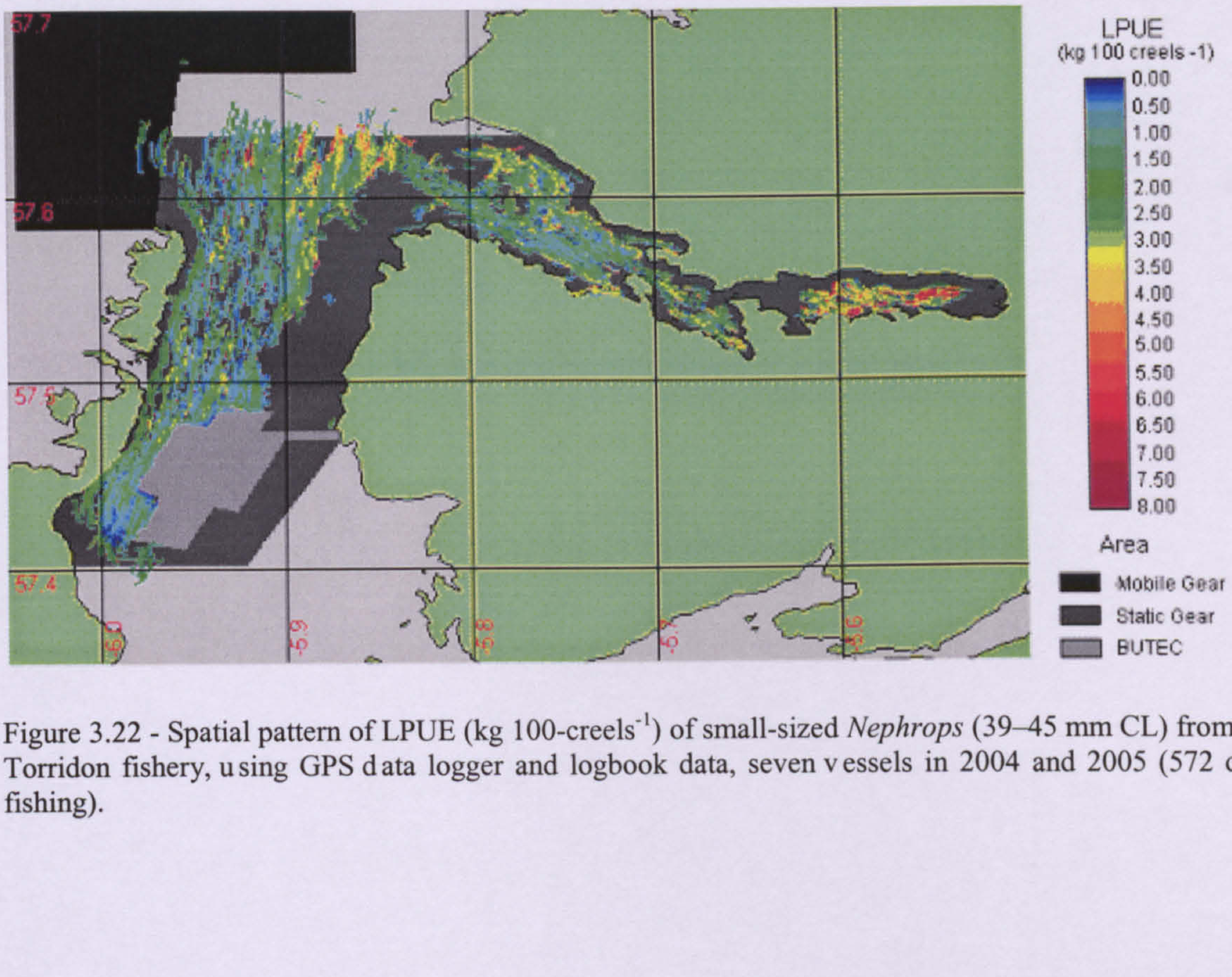

Figure 3.22 - Spatial pattern of LPUE (kg 100-creels^{-1}) of small-sized *Nephrops* (39–45 mm CL) from the Torridon fishery, using GPS data logger and logbook data, seven vessels in 2004 and 2005 (572 days fishing).

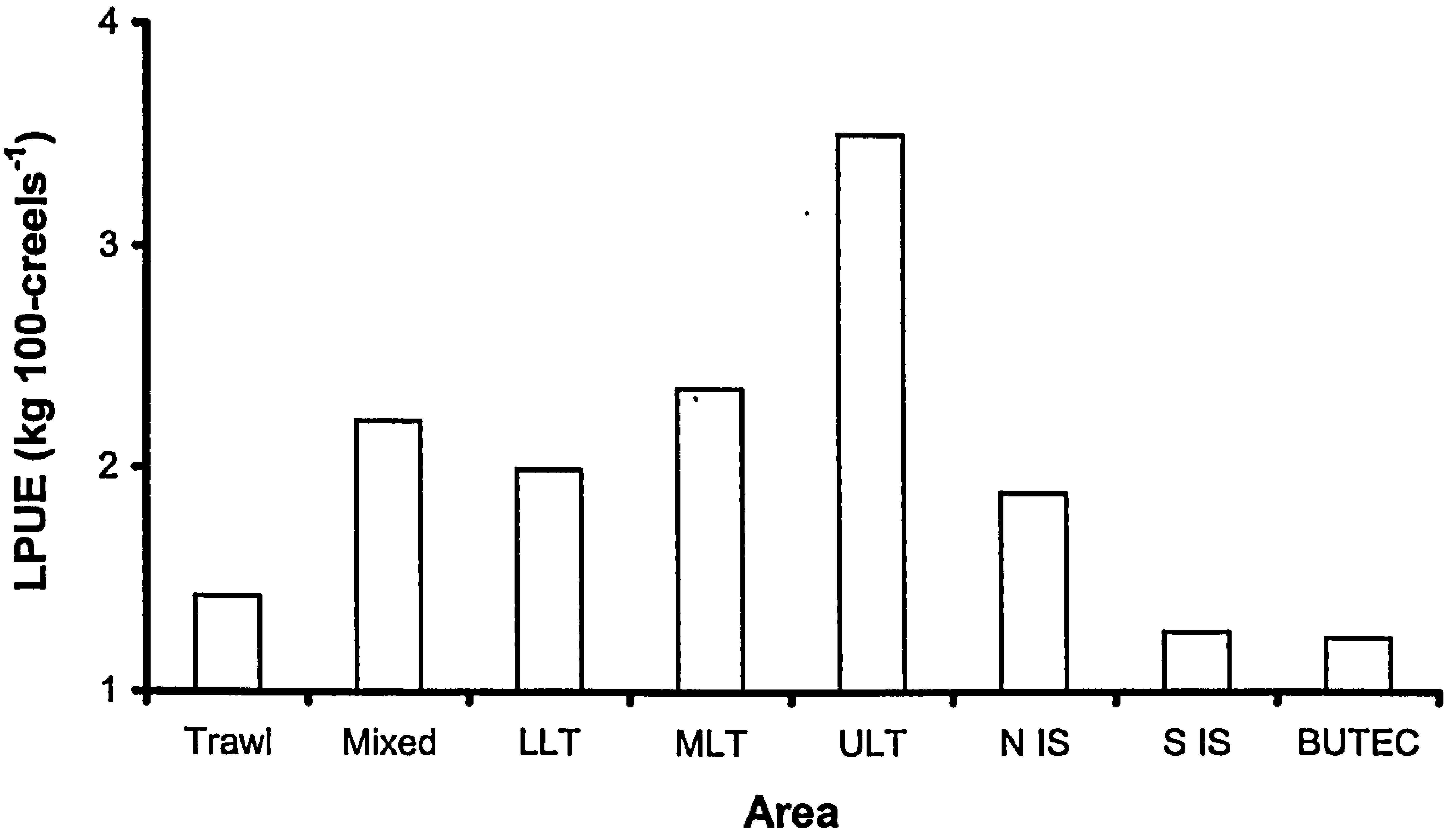

Figure 3.23 – Average small-sized *Nephrops* LPUE per pixel in different fishing areas, seven vessels in 2004 and 2005 where Trawl=trawl only zone, Mixed=Mixed fishery, LLT=lower Loch Torridon, MLT=middle Loch Torridon, ULT=Upper Loch Torridon, N IS=north Inner Sound, S IS=south Inner Sound and BUTEC=BUTEC no fishing zone.

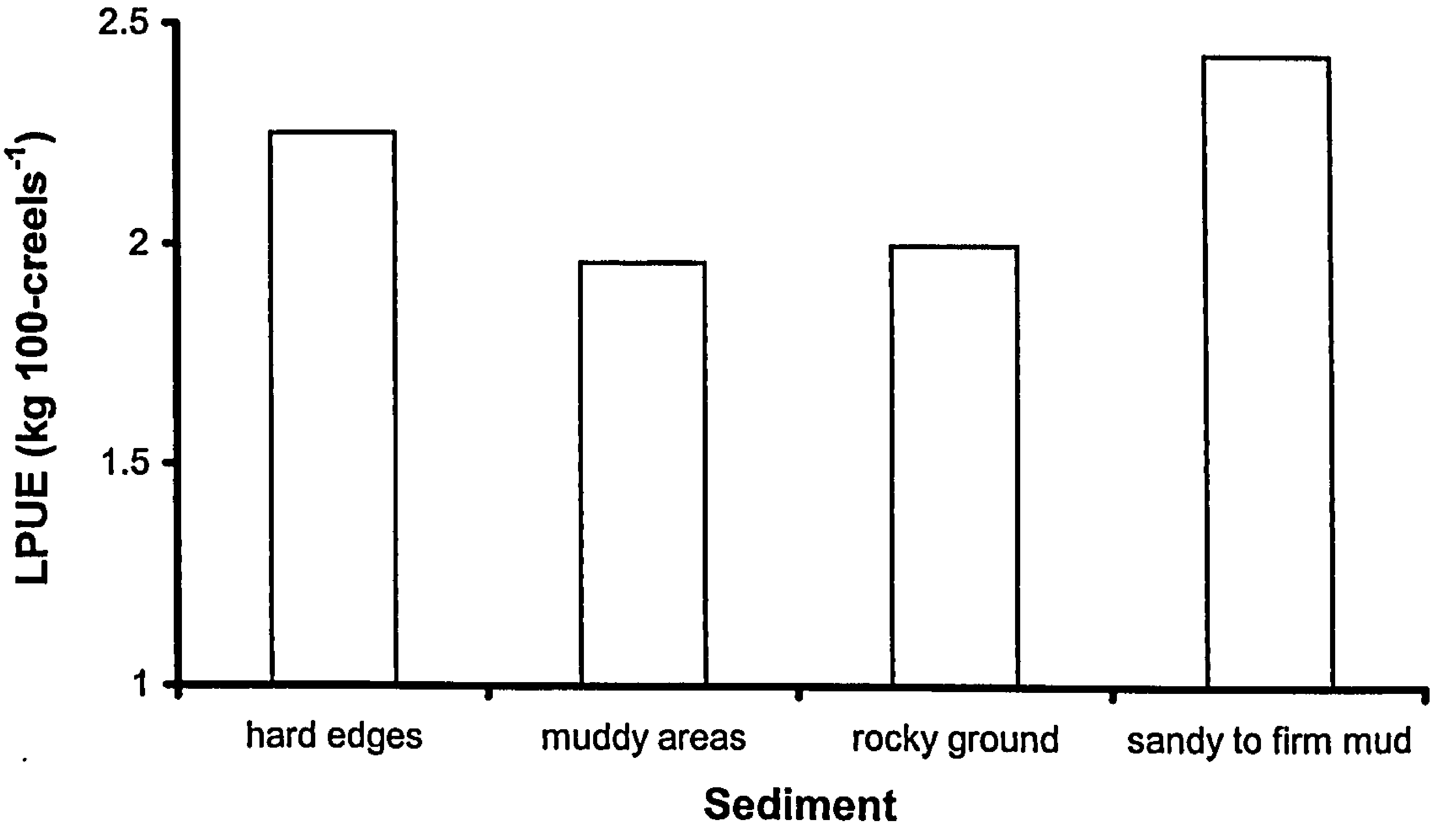

Figure 3.24 – Average LPUE of *Nephrops* (small) averaged over pixels in areas of different sediment types, seven vessels in 2004 and 2005.

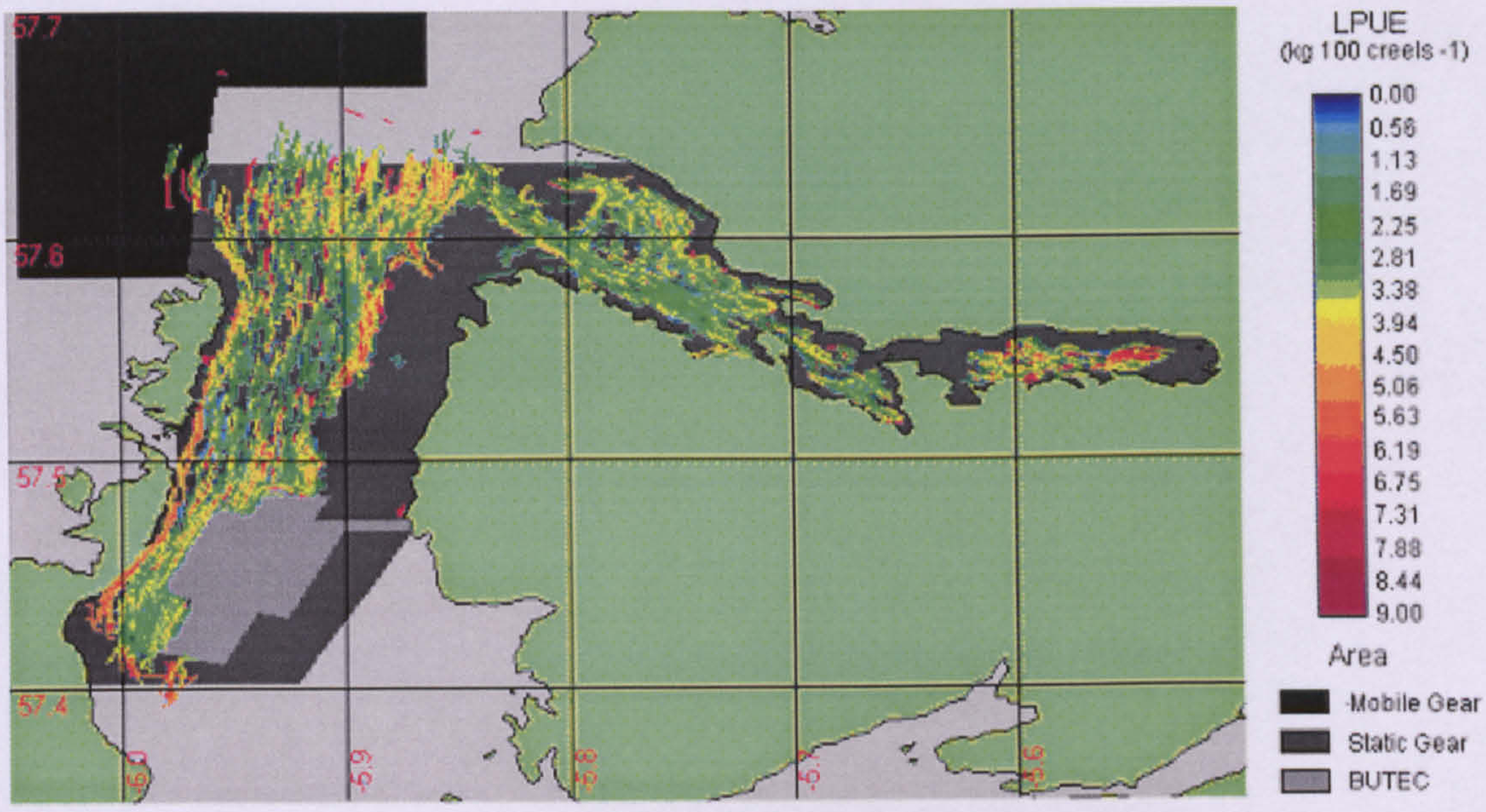

Figure 3.25 - Spatial pattern of LPUE (kg 100-creels^{-1}) of medium-sized *Nephrops* (46–49 mm CL) from the Torridon fishery, using GPS data logger and logbook data, seven vessels in 2004 and 2005 (572 days fishing).

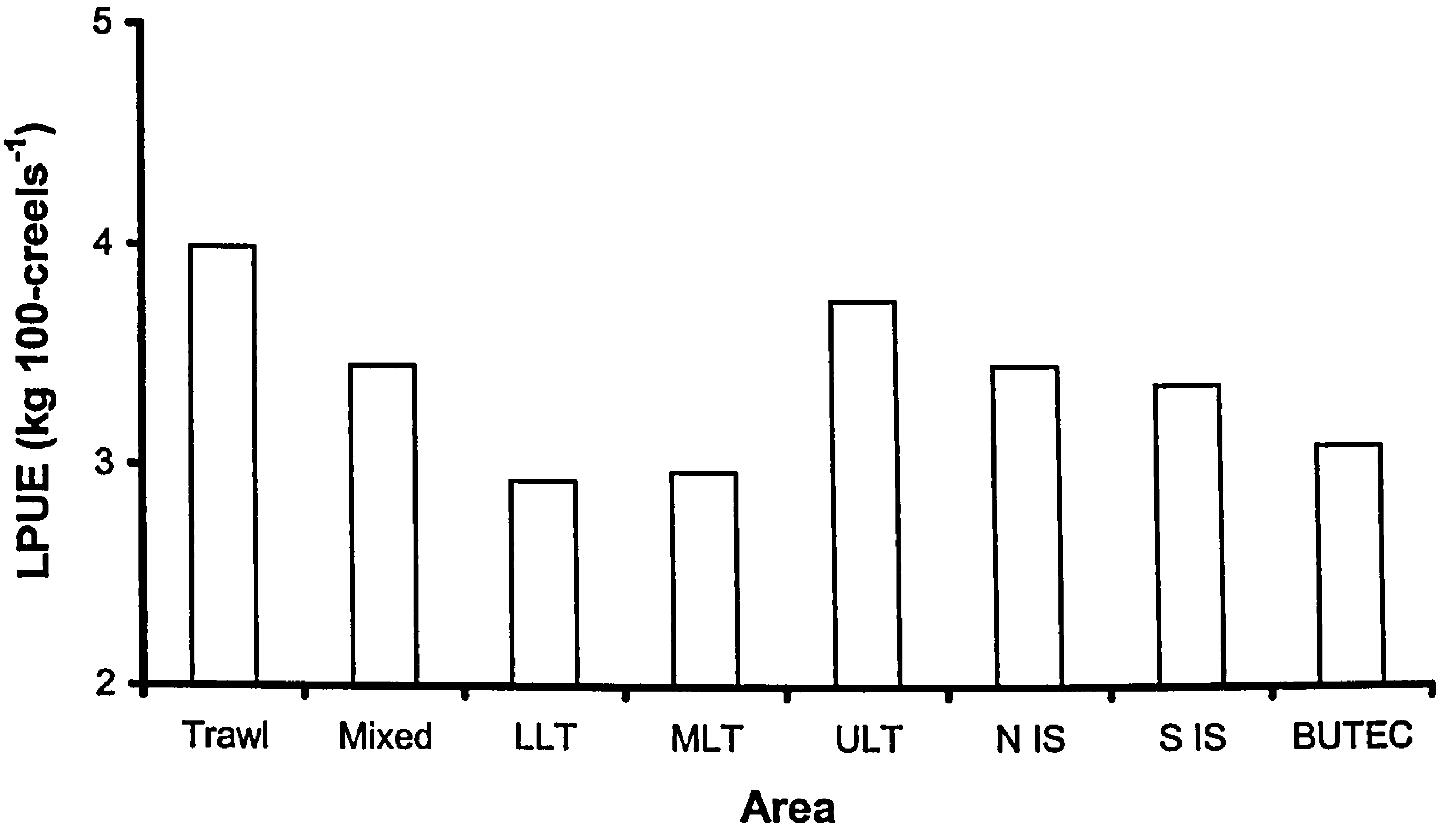

Figure 3.26 – Mean LPUE of medium-sized *Nephrops* averaged over pixels in different fishing areas, seven vessels in 2004 and 2005, where Trawl=trawl only zone, Mixed=Mixed fishery, LLT=lower Loch Torridon, MLT=middle Loch Torridon, ULT=Upper Loch Torridon, N IS=north Inner Sound, S IS=south Inner Sound and BUTEC=BUTEC no fishing zone.

Figure 3.27 – Average LPUE of *Nephrops* (medium) averaged over pixels in areas of different sediment types, one vessel in 2004 and 2005.

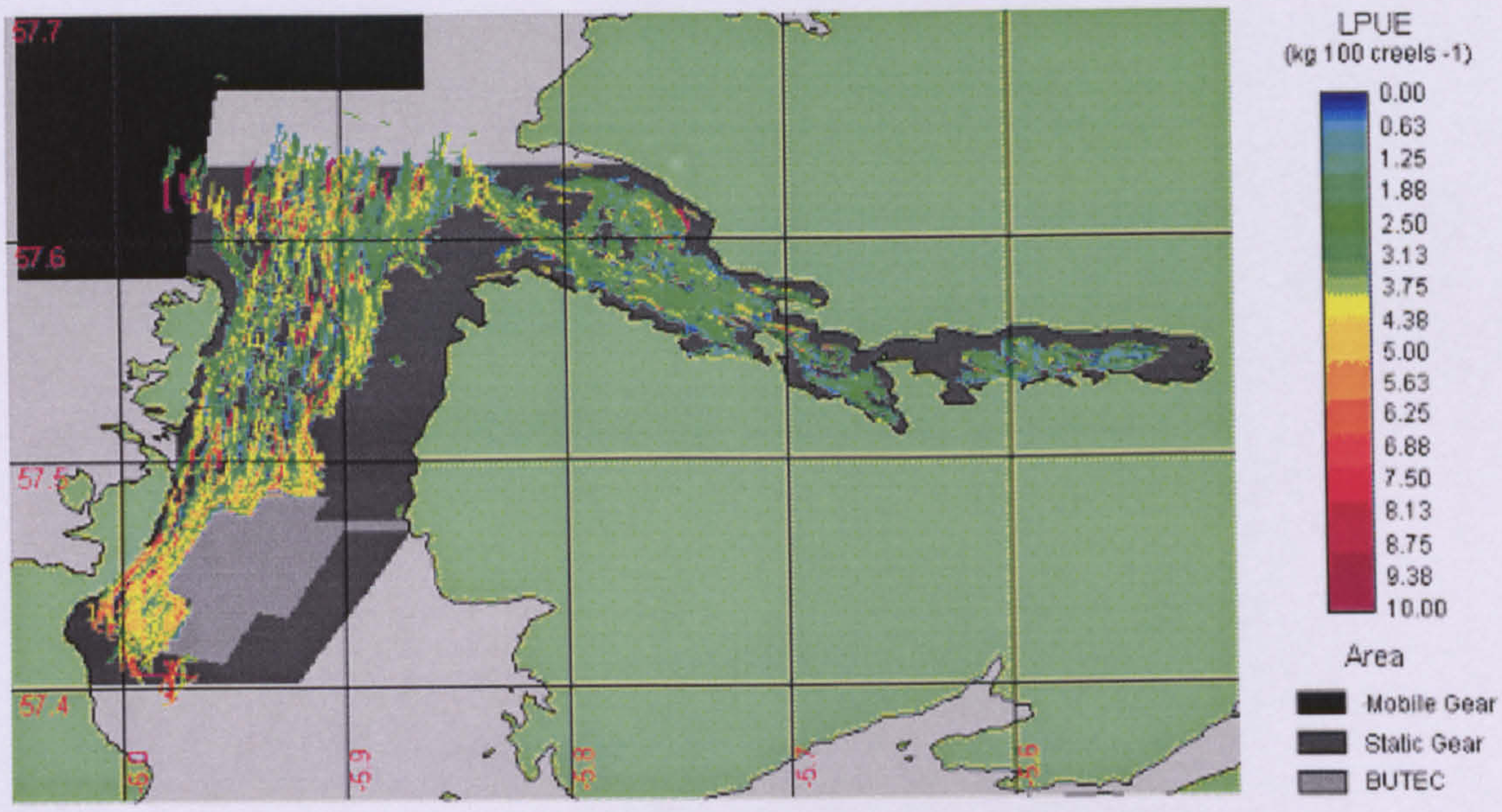

Figure 3.28 - Spatial pattern of LPUE (kg 100-creels^{-1}) of large-sized *Nephrops* (>49 mm CL) from the Torridon fishery, using GPS data logger and logbook data, seven vessels in 2004 and 2005 (572 days fishing).

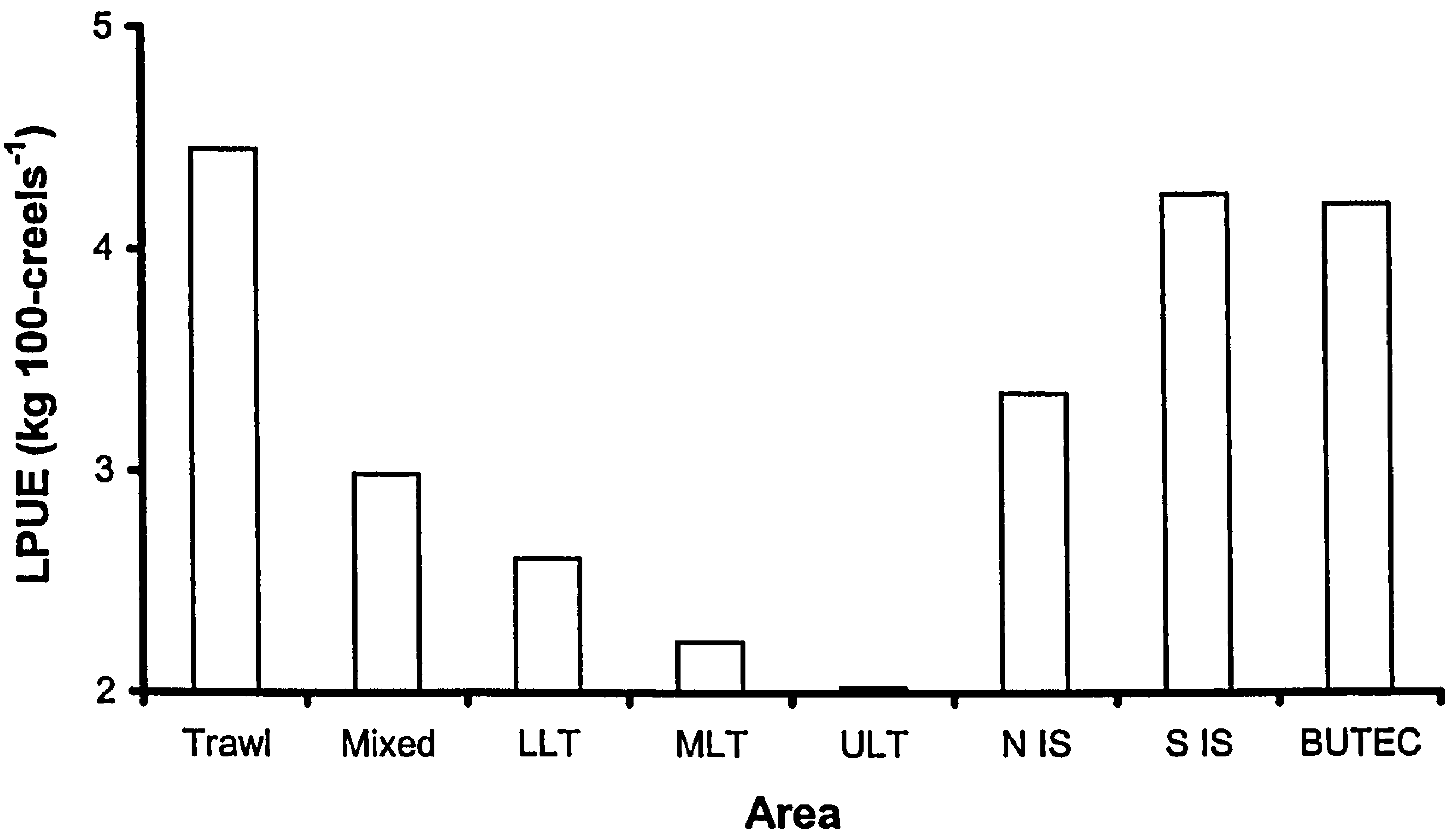

Figure 3.29 – Average large-sized *Nephrops* LPUE per pixel in different fishing areas, seven vessels in 2004 and 2005, where Trawl=trawl only zone, Mixed=Mixed fishery, LLT=lower Loch Torridon, MLT=middle Loch Torridon, ULT=Upper Loch Torridon, N IS=north Inner Sound, S IS=south Inner Sound and BUTEC=BUTEC no fishing zone.

Figure 3.30 – Average LPUE of *Nephrops* (large) averaged over pixels in areas of different sediment types, one vessel in 2004 and 2005.

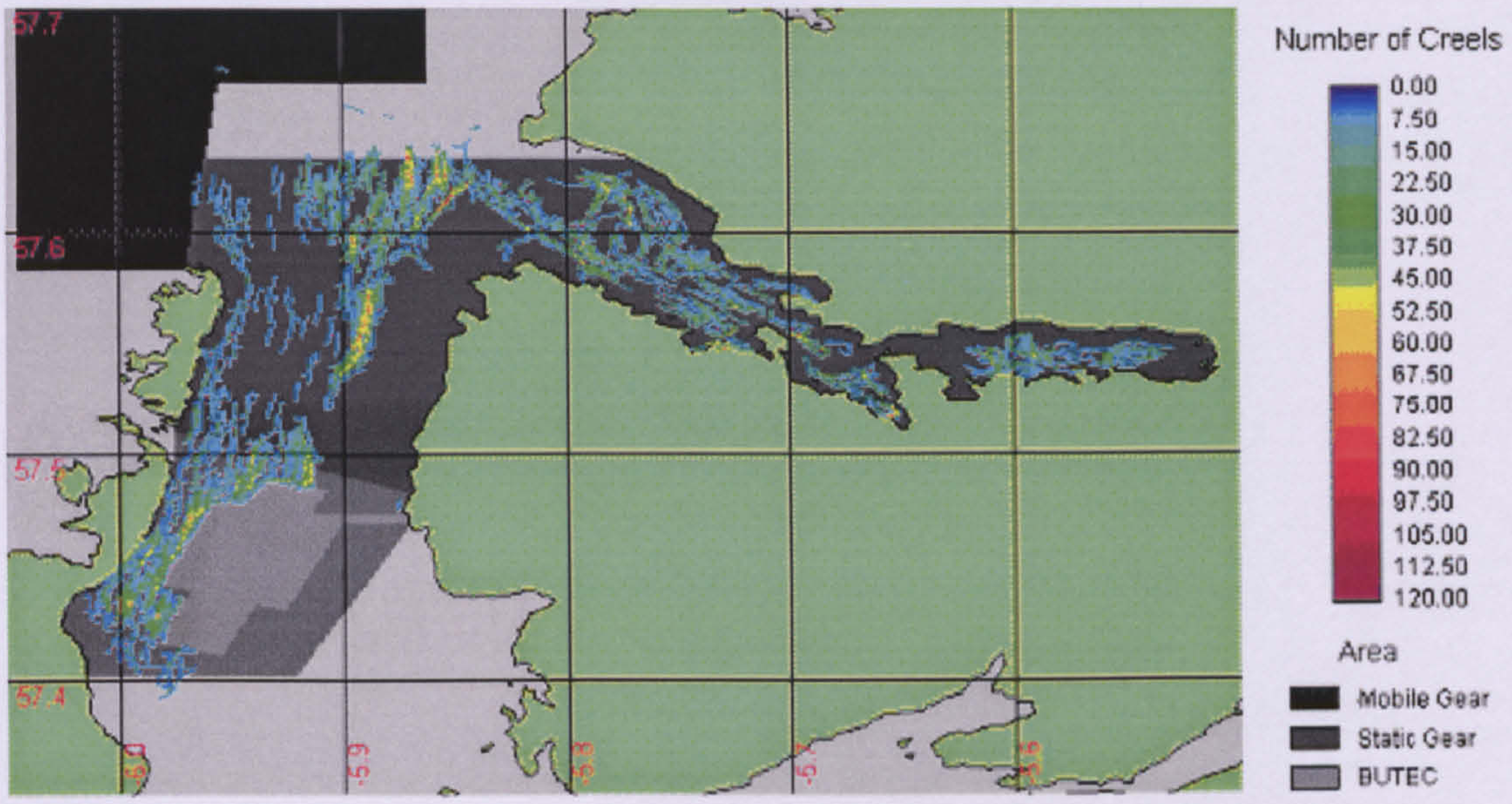

Figure 3.31 - Spatial pattern of effort from the Torridon *Nephrops* creel fishery, using GPS data logger and logbook data, seven vessels, 2nd quarter in 2004 and 2005 (201 days fishing).

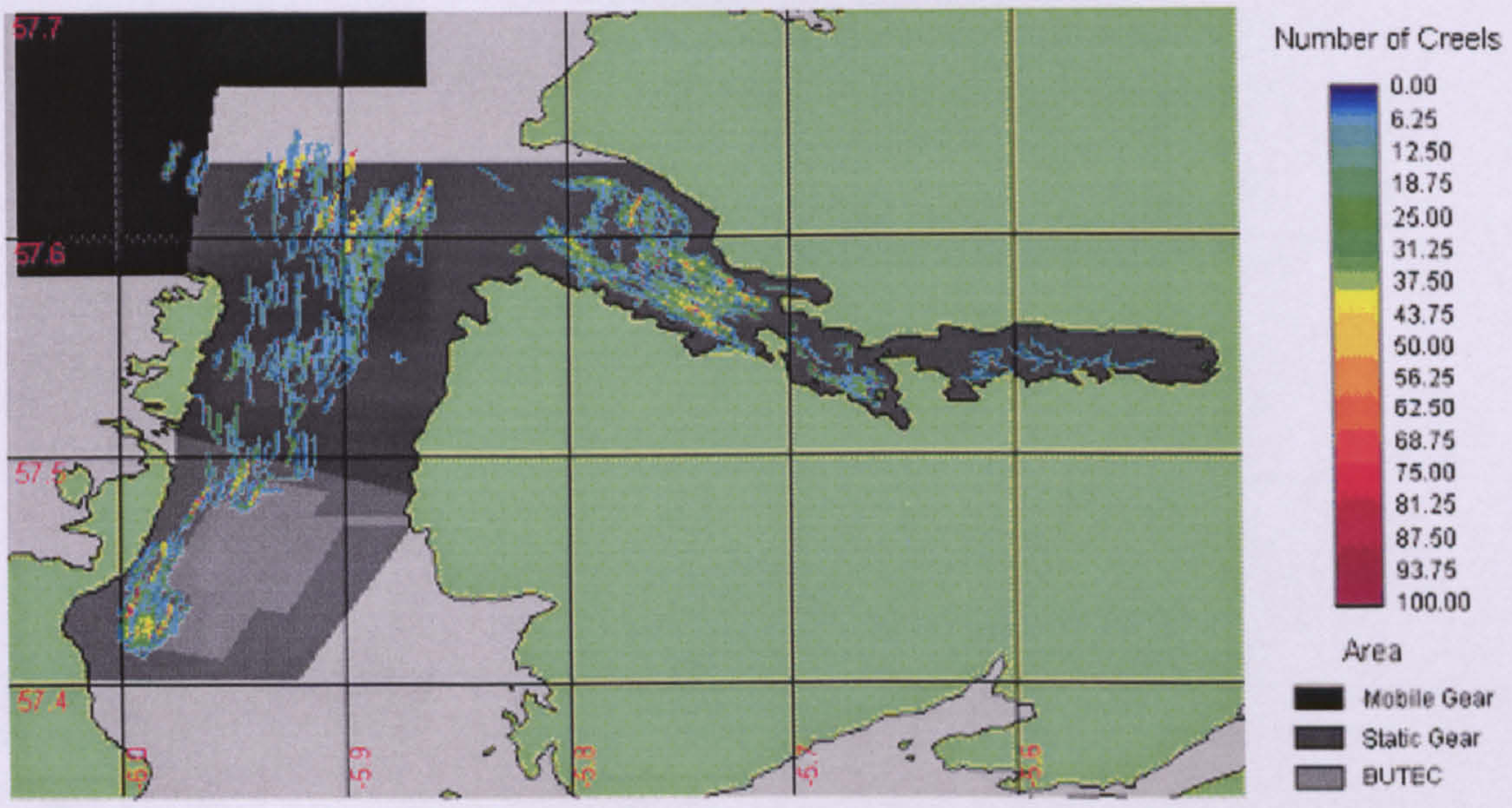

Figure 3.32 - Spatial pattern of effort from the Torridon *Nephrops* fishery, using GPS data logger and logbook data, seven vessels, 4th quarter in 2004 and 2005 (160 days fishing).

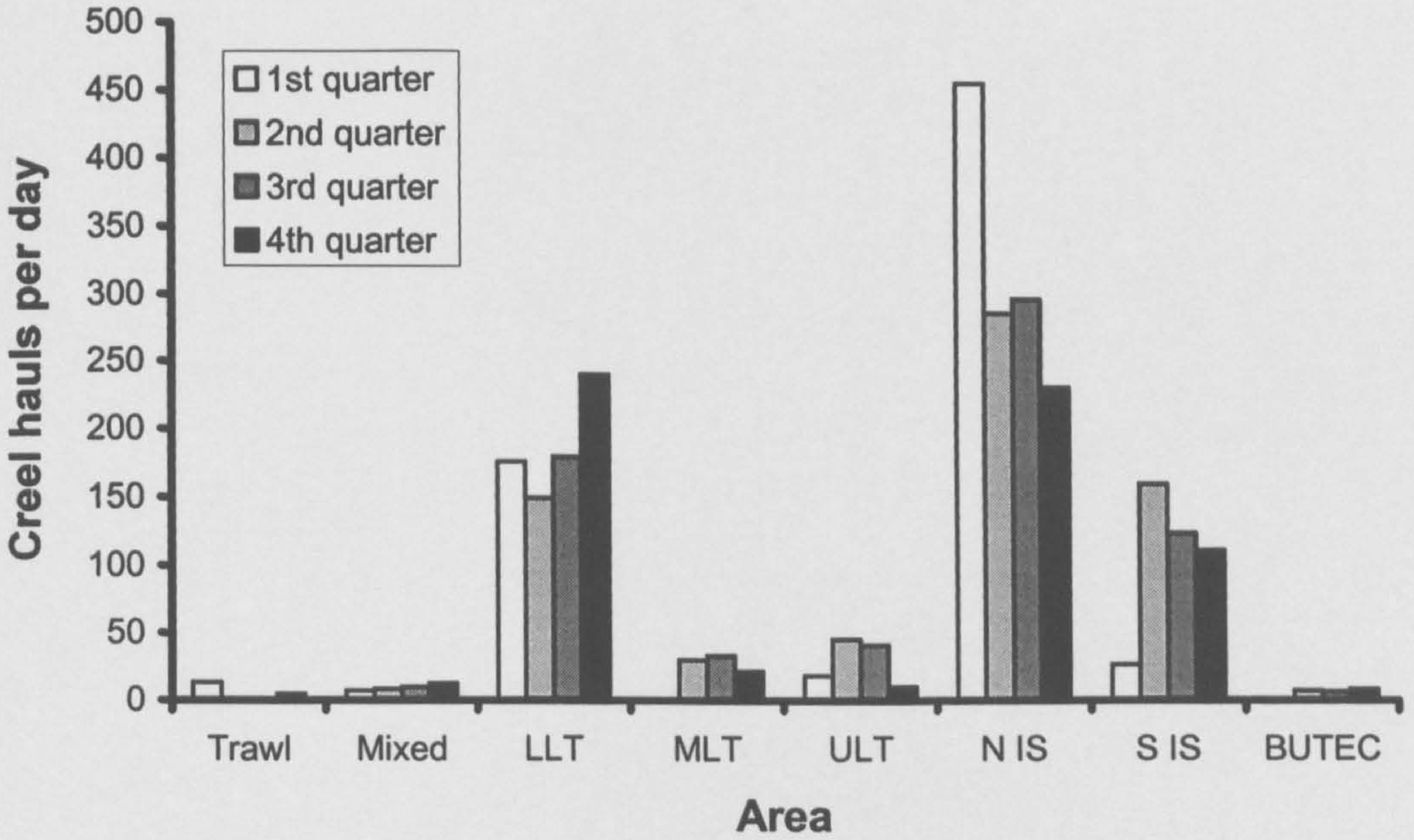

Figure 3.33 – Mean daily fishing effort in different areas of the Torridon creel fishery by year quarter. Based on data from position recording units on seven vessels in 2004 and 2005, where Trawl=trawl only zone, Mixed=Mixed fishery, LLT=lower Loch Torridon, MLT=middle Loch Torridon, ULT=Upper Loch Torridon, N IS=north Inner Sound, S IS=south Inner Sound and BUTEC=BUTEC no fishing zone.

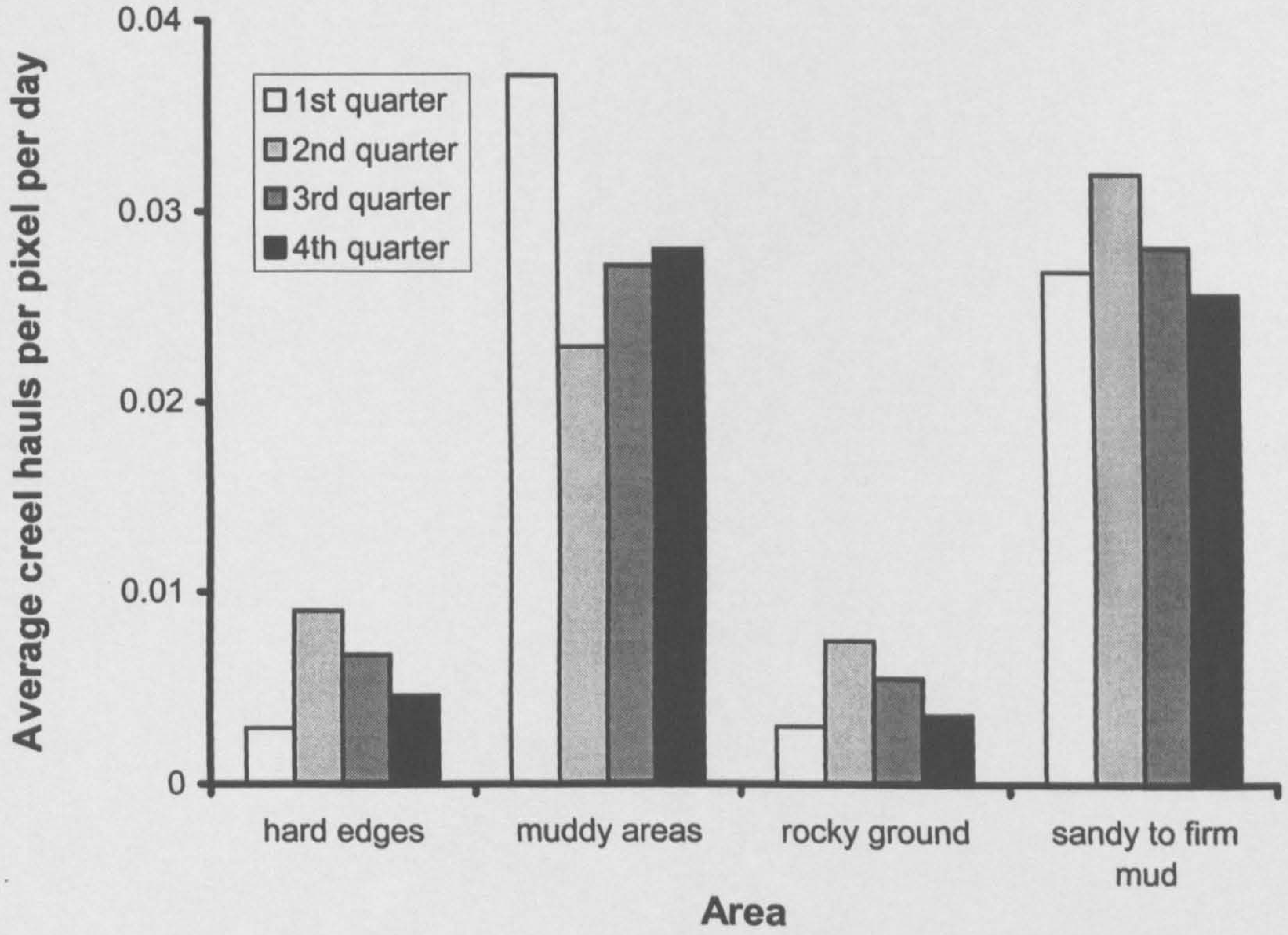

Figure 3.34 – Mean daily fishing effort on different sediment types within the Torridon creel fishery by year quarter. Based on data from position recording units on seven vessels in 2004 and 2005.

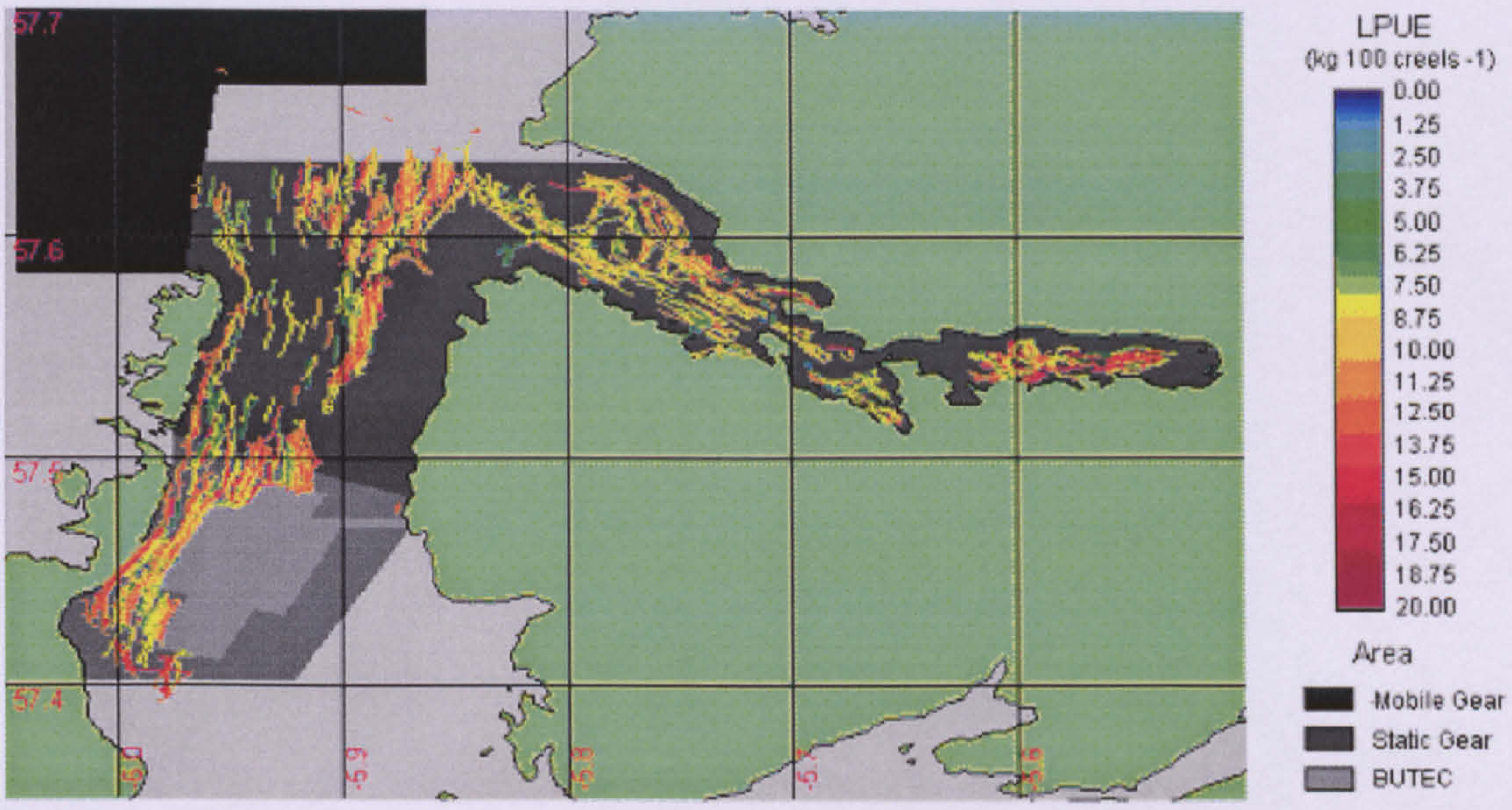

Figure 3.35 - Spatial pattern of LPUE (kg 100-creels^{-1}) of total *Nephrops* catch from the Torridon *Nephrops* fishery, using GPS data logger and logbook data, seven vessels, 2nd quarter, 2004–2005 (201 days fishing).

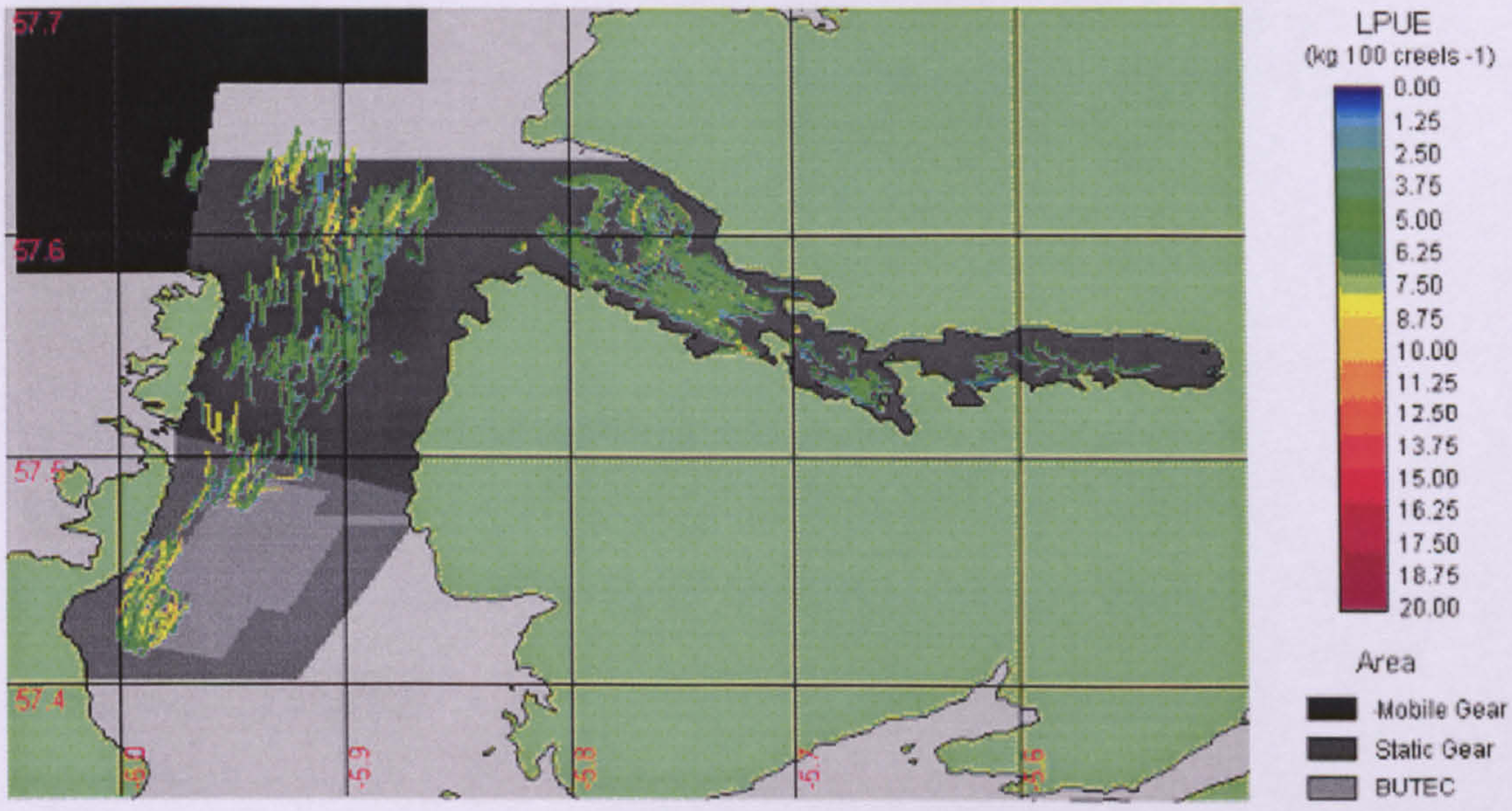

Figure 3.36 - Spatial pattern of LPUE (kg 100-creels^{-1}) of total *Nephrops* catch from the Torridon *Nephrops* fishery, using GPS data logger and logbook data, seven vessels, 4th quarter in 2004 and 2005 (160 days fishing).

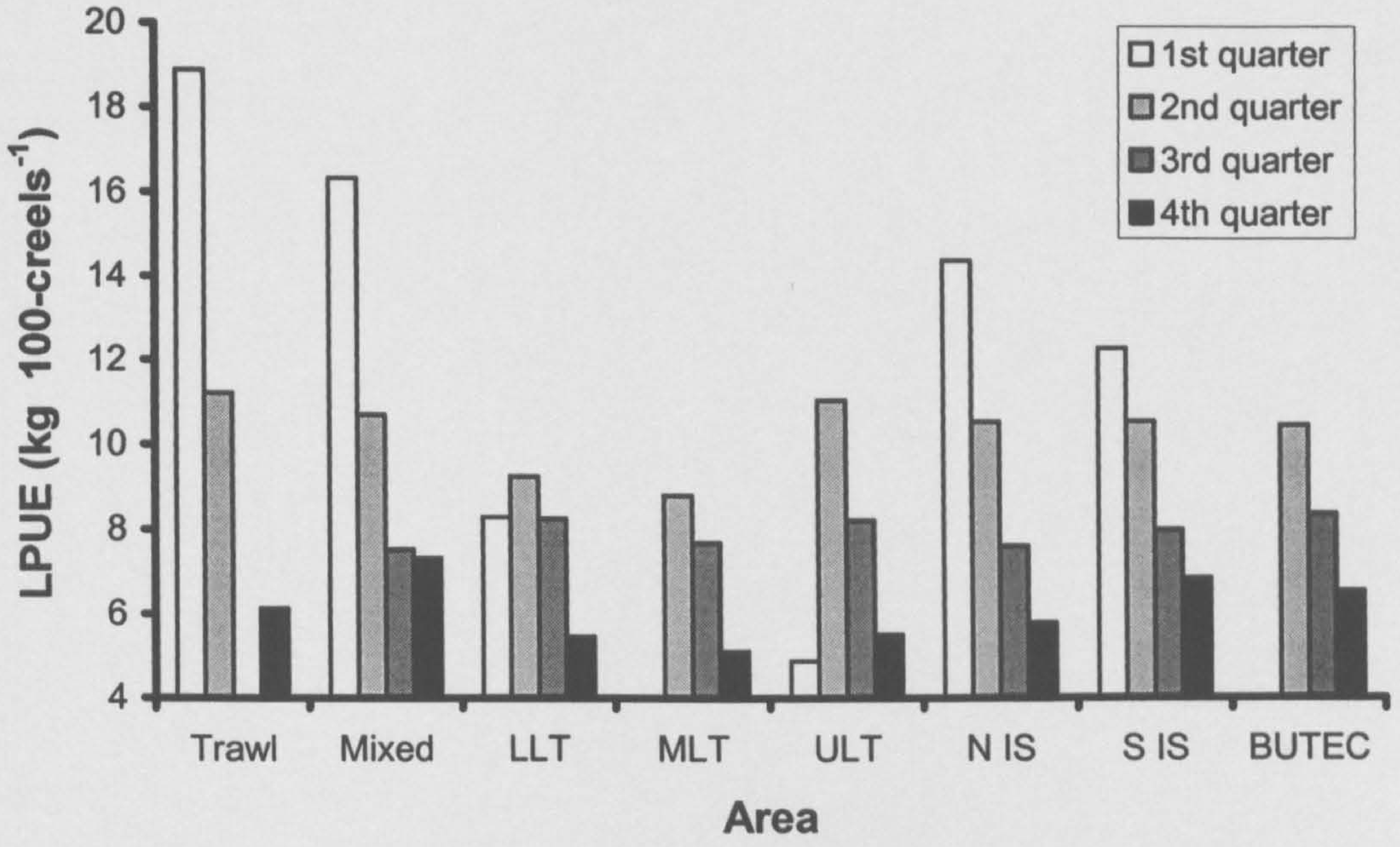

Figure 3.37 – Mean daily fishing LPUE (all categories) in different fishing areas within the Torridon creel fishery by year quarter. Based on data from position recording units on seven vessels in 2004 and 2005, where Trawl=trawl only zone, Mixed=Mixed fishery, LLT=lower Loch Torridon, MLT=middle Loch Torridon, ULT=Upper Loch Torridon, N IS=north Inner Sound, S IS=south Inner Sound and BUTEC=BUTEC no fishing zone.

Figure 3.38 – Mean daily fishing LPUE (all categories) on different sediment types within the Torridon creel fishery by year quarter. Based on data from position recording units on seven vessels in 2004 and 2005.

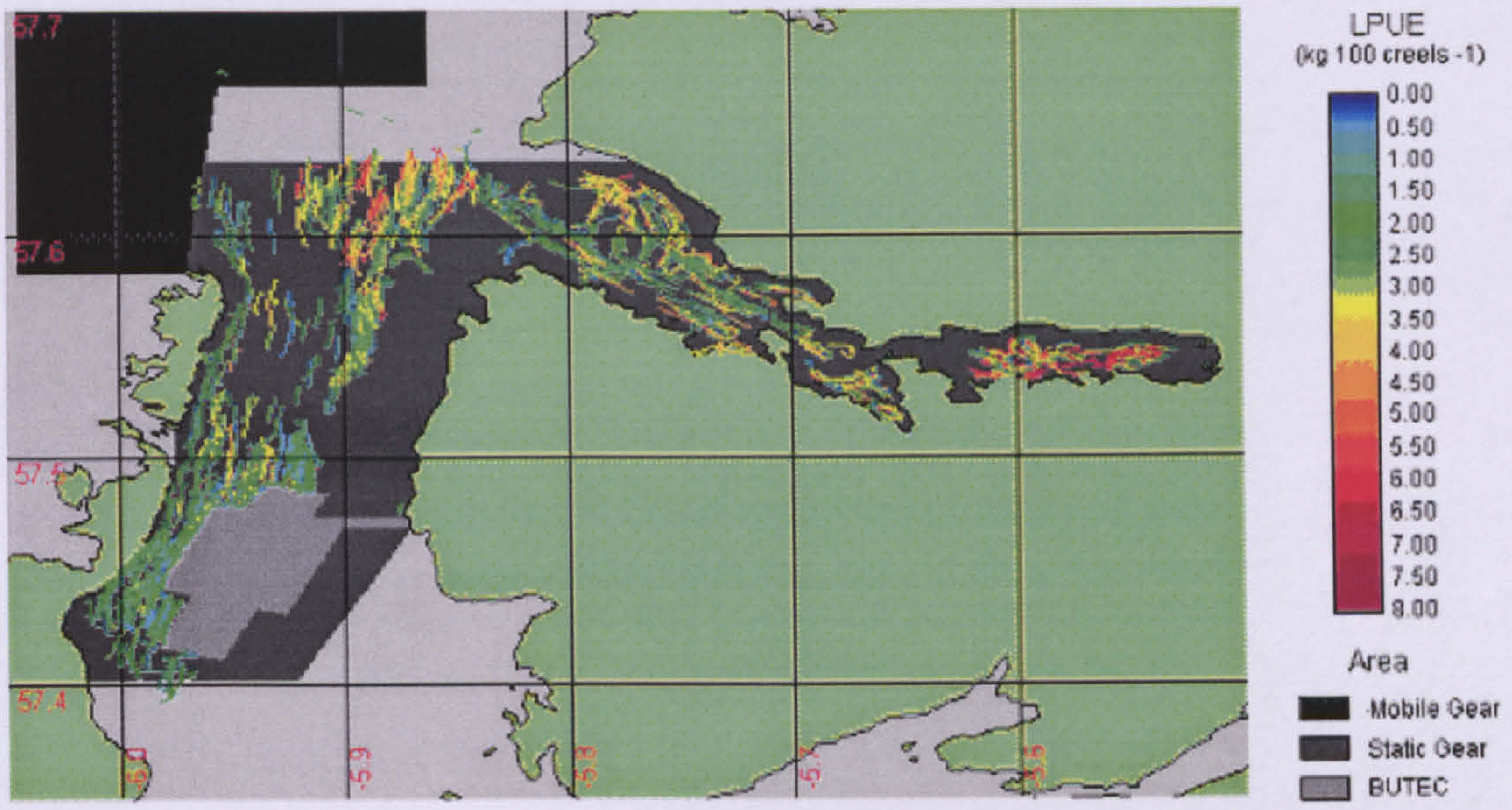

Figure 3.39 - Spatial pattern of LPUE (kg 100-creels^{-1}) of small-sized *Nephrops* (39–45 mm CL) from the Torridon *Nephrops* fishery, using GPS data logger and logbook data, seven vessels, 2nd quarter in 2004 and 2005 (201 days fishing).

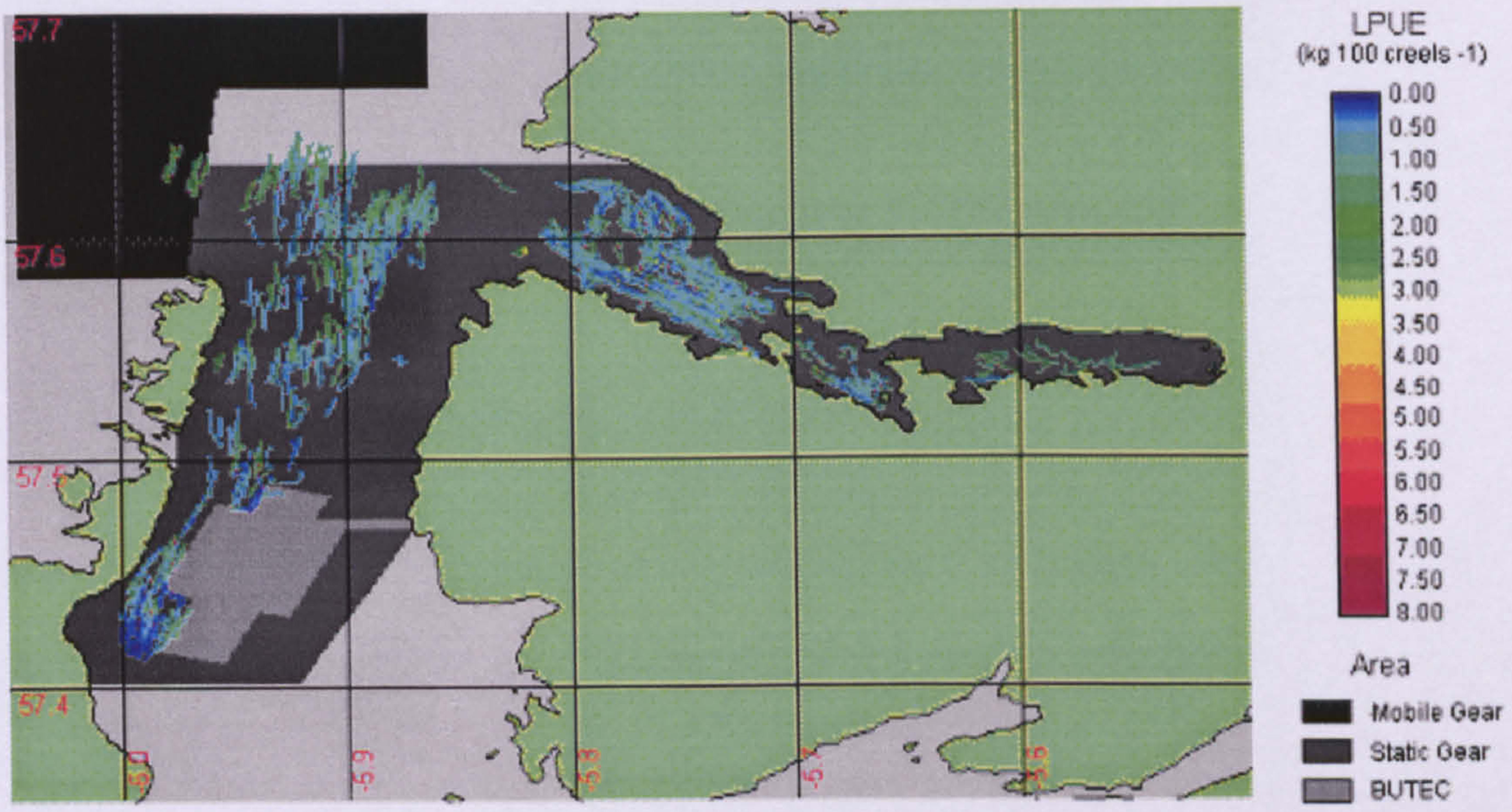

Figure 3.40 - Spatial pattern of LPUE (kg 100-creels^{-1}) of small-sized *Nephrops* (39–45 mm CL) from the Torridon *Nephrops* fishery, using GPS data logger and logbook data, seven vessels, 4th quarter in 2004 and 2005 (160 days fishing).

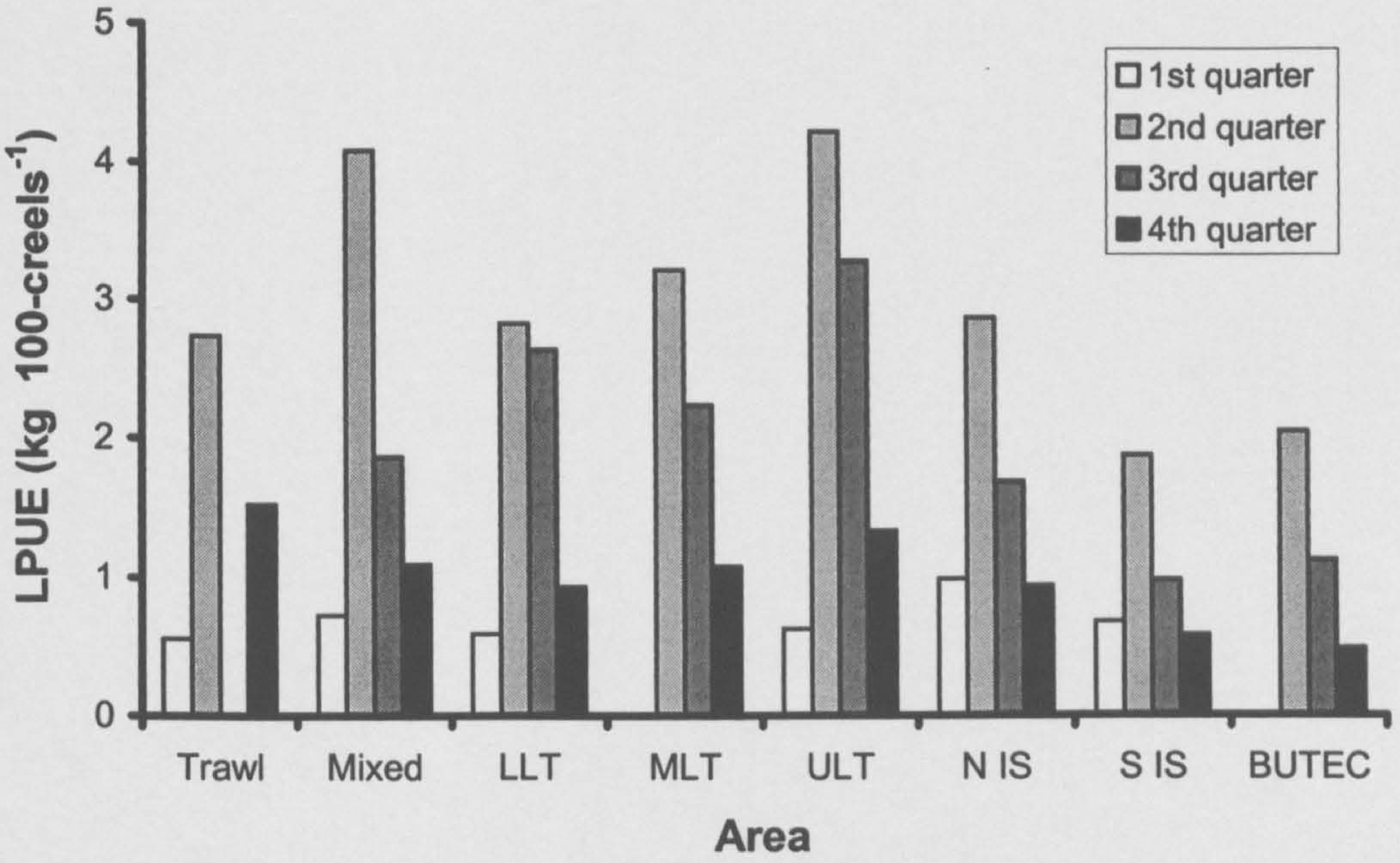

Figure 3.41 – Mean daily fishing LPUE (small) in different fishing areas within the Torridon creel fishery by year quarter. Based on data from position recording units on seven vessels in 2004 and 2005,, where Trawl=trawl only zone, Mixed=Mixed fishery, LLT=lower Loch Torridon, MLT=middle Loch Torridon, ULT=Upper Loch Torridon, N IS=north Inner Sound, S IS=south Inner Sound and BUTEC=BUTEC no fishing zone.

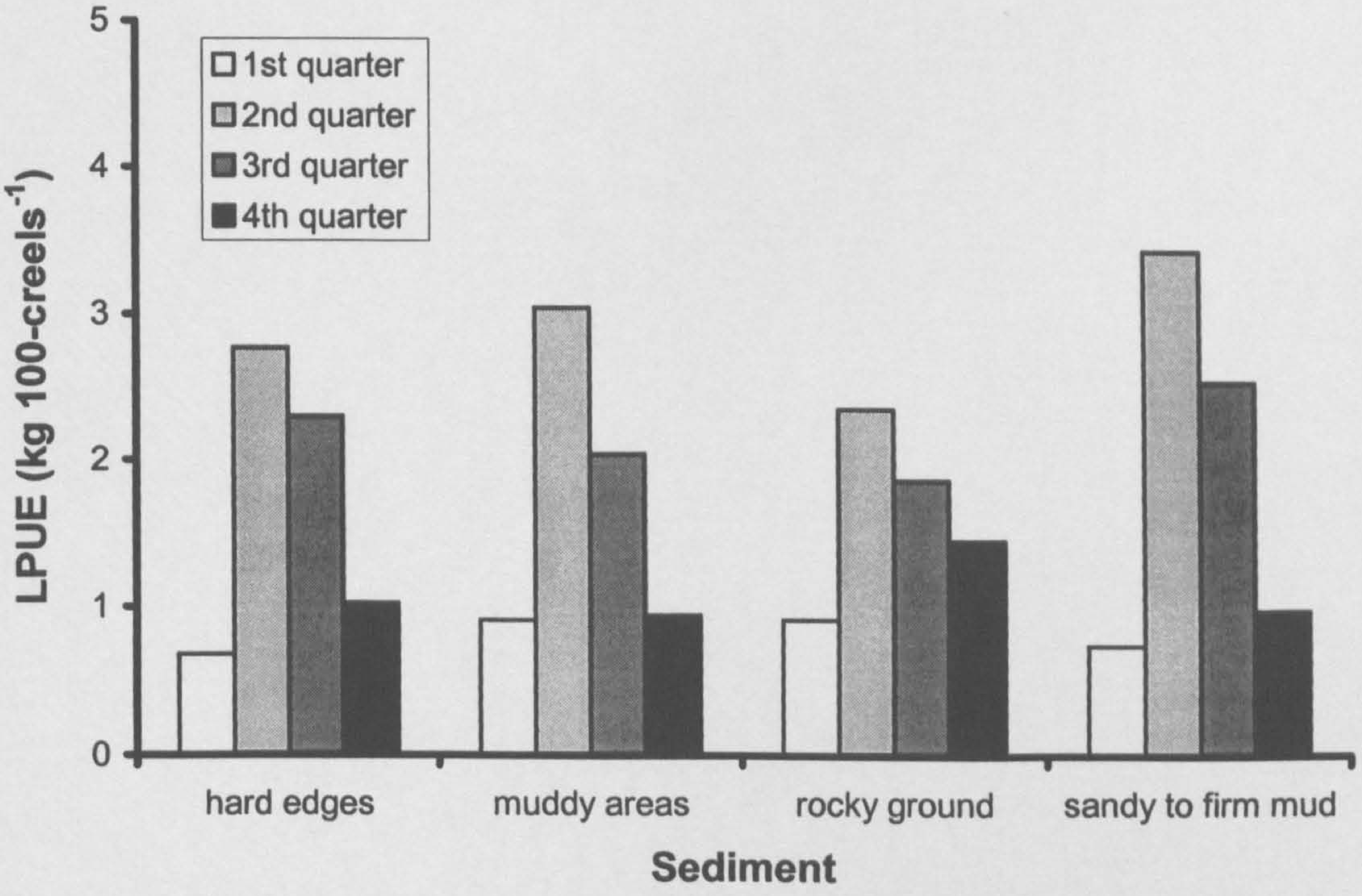

Figure 3.42 – Mean daily fishing LPUE (small) on different sediment types within the Torridon creel fishery by year quarter. Based on data from position recording units on seven vessels in 2004 and 2005.

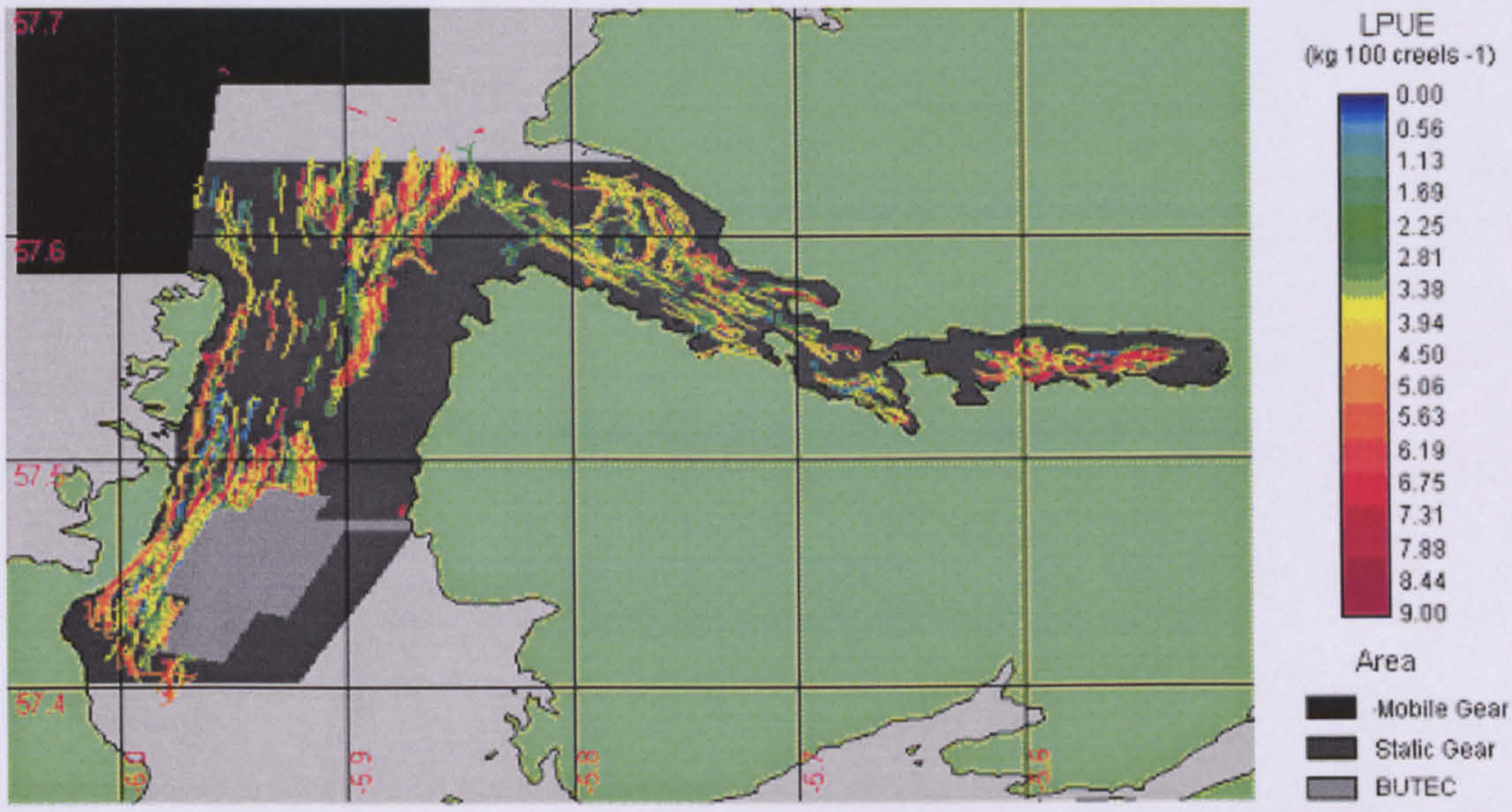

Figure 3.43 - Spatial pattern of LPUE (kg 100-creels^{-1}) of medium-sized *Nephrops* (46–49 mm CL) from the Torridon *Nephrops* fishery, using GPS data logger and logbook data, seven vessels, 2nd quarter in 2004 and 2005 (201 days fishing).

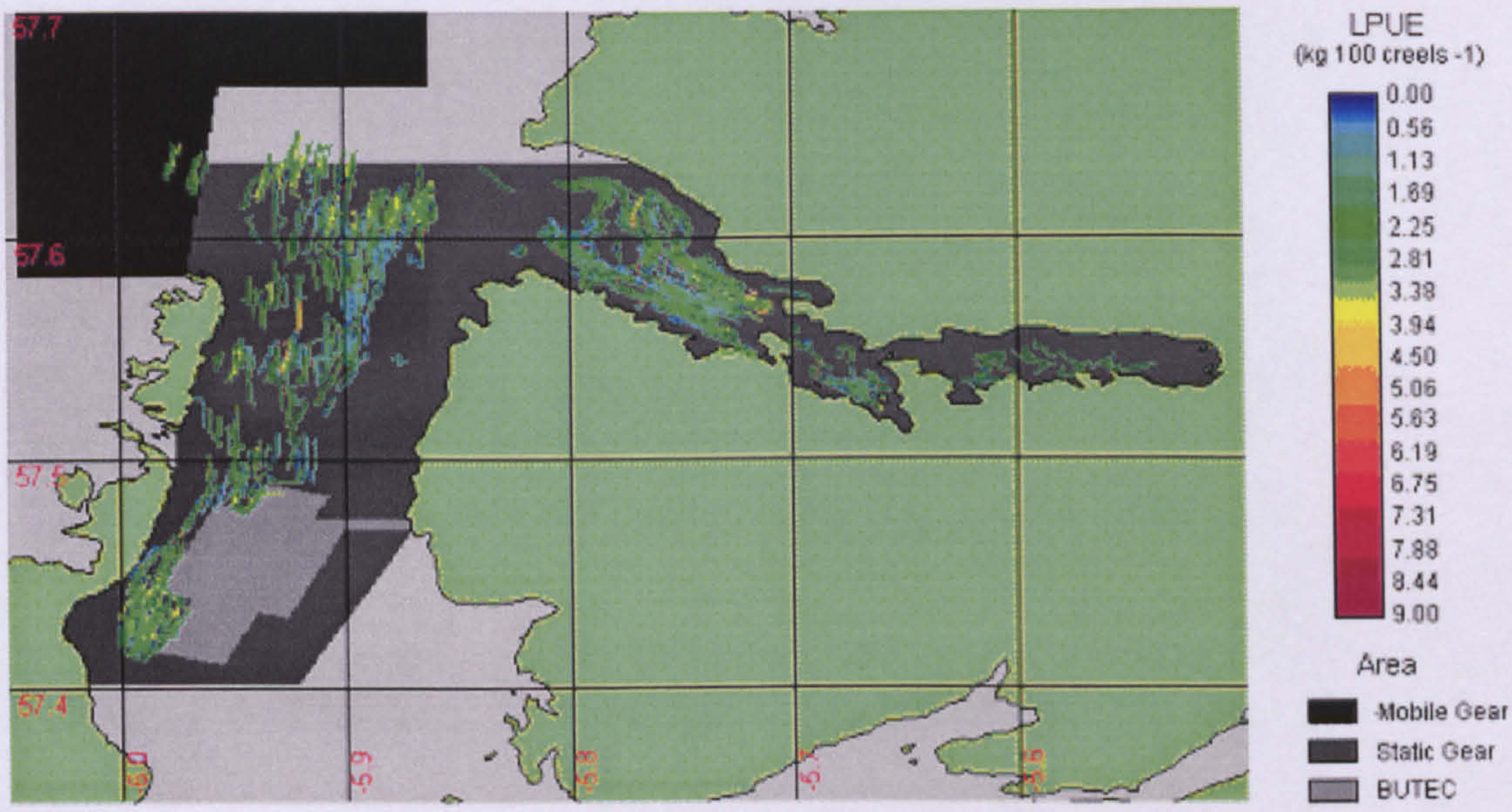

Figure 3.44 - Spatial pattern of LPUE (kg 100-creels^{-1}) of medium-sized *Nephrops* (46–49 mm CL) from the Torridon *Nephrops* fishery, using GPS data logger and logbook data, seven vessels, 4th quarter in 2004 and 2005 (160 days fishing).

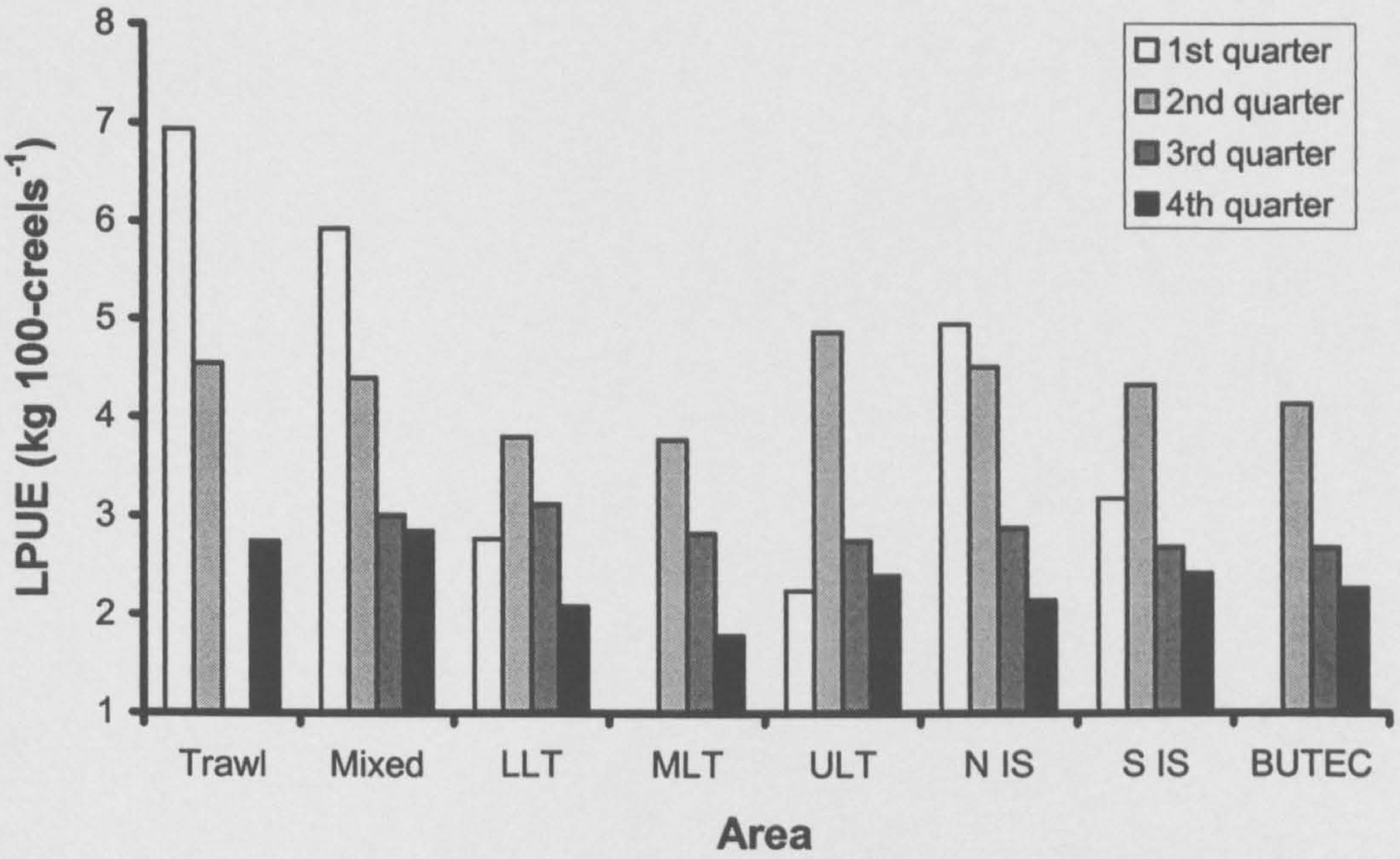

Figure 3.45 – Mean daily fishing LPUE (medium) in different fishing areas within the Torridon creel fishery by year quarter. Based on data from position recording units on seven vessels in 2004 and 2005, where Trawl=trawl only zone, Mixed=Mixed fishery, LLT=lower Loch Torridon, MLT=middle Loch Torridon, ULT=Upper Loch Torridon, N IS=north Inner Sound, S IS=south Inner Sound and BUTEC=BUTEC no fishing zone.

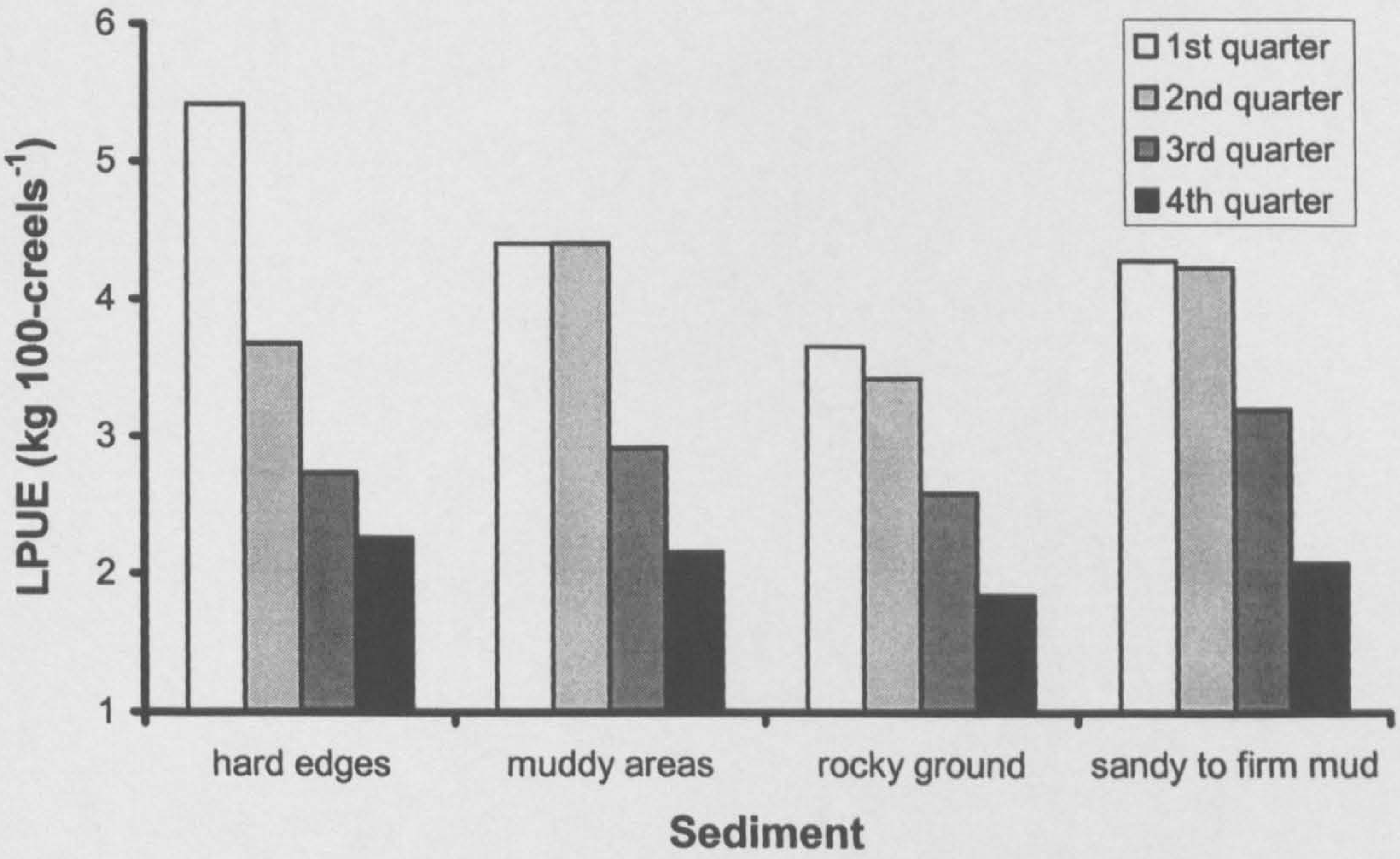

Figure 3.46 – Mean daily fishing LPUE (medium) on different sediment types within the Torridon creel fishery by year quarter. Based on data from position recording units on seven vessels in 2004 and 2005.

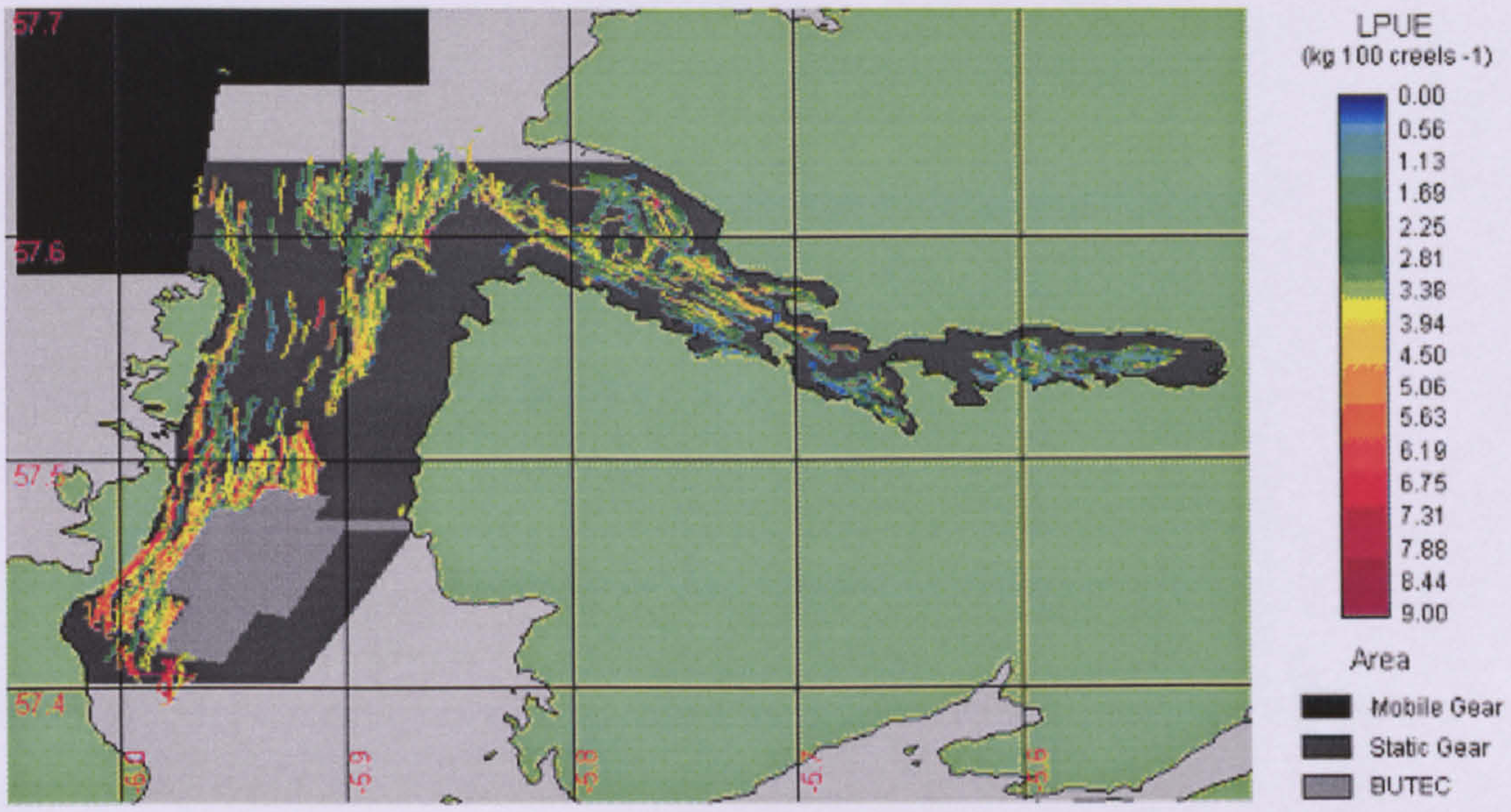

Figure 3.47 - Spatial pattern of LPUE (kg 100-creels^{-1}) of large-sized *Nephrops* (>49 mm CL) from the Torridon *Nephrops* fishery, using GPS data logger and logbook data, seven vessels, 2nd quarter in 2004 and 2005 (201 days fishing).

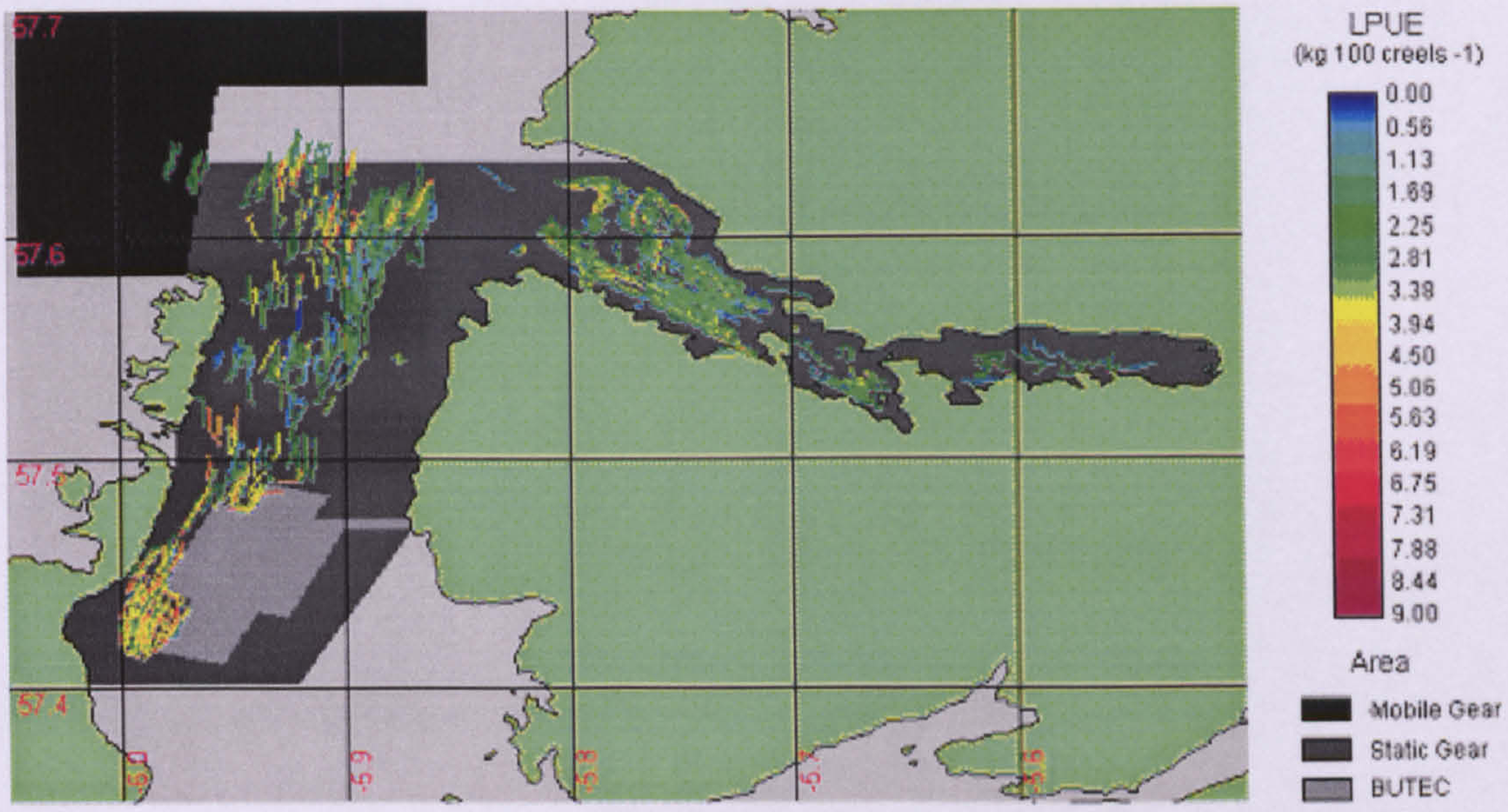

Figure 3.48 - Spatial pattern of LPUE (kg 100-creels^{-1}) of large-sized *Nephrops* (>49 mm CL) from the Torridon *Nephrops* fishery, using GPS data logger and logbook data, seven vessels, 4th quarter in 2004 and 2005 (160 days fishing).

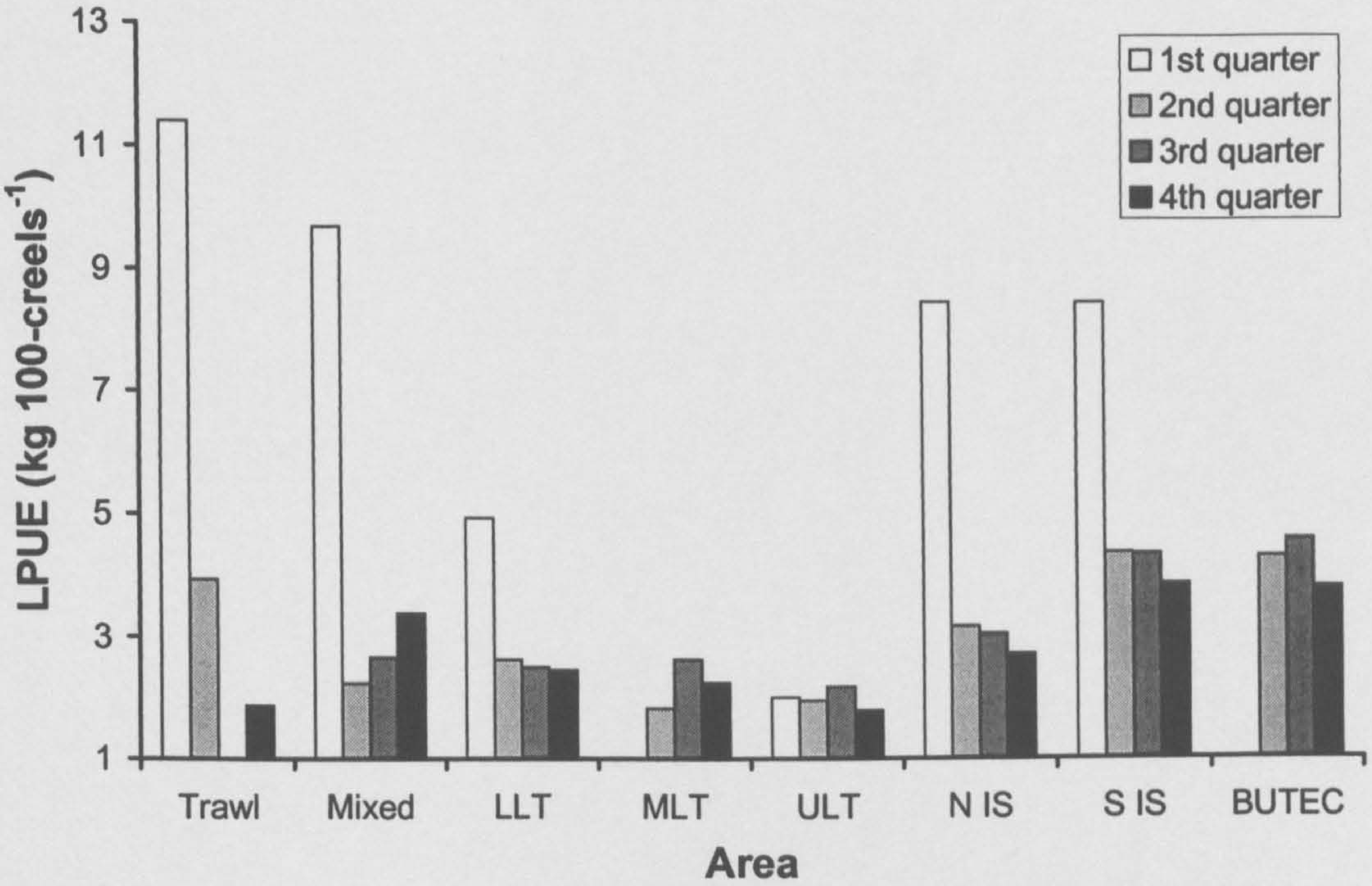

Figure 3.49 – Mean daily fishing LPUE (large) in different fishing areas within the Torridon creel fishery by year quarter. Based on data from position recording units on seven vessels in 2004 and 2005, where Trawl=trawl only zone, Mixed=Mixed fishery, LLT=lower Loch Torridon, MLT=middle Loch Torridon, ULT=Upper Loch Torridon, N IS=north Inner Sound, S IS=south Inner Sound and BUTEC=BUTEC no fishing zone.

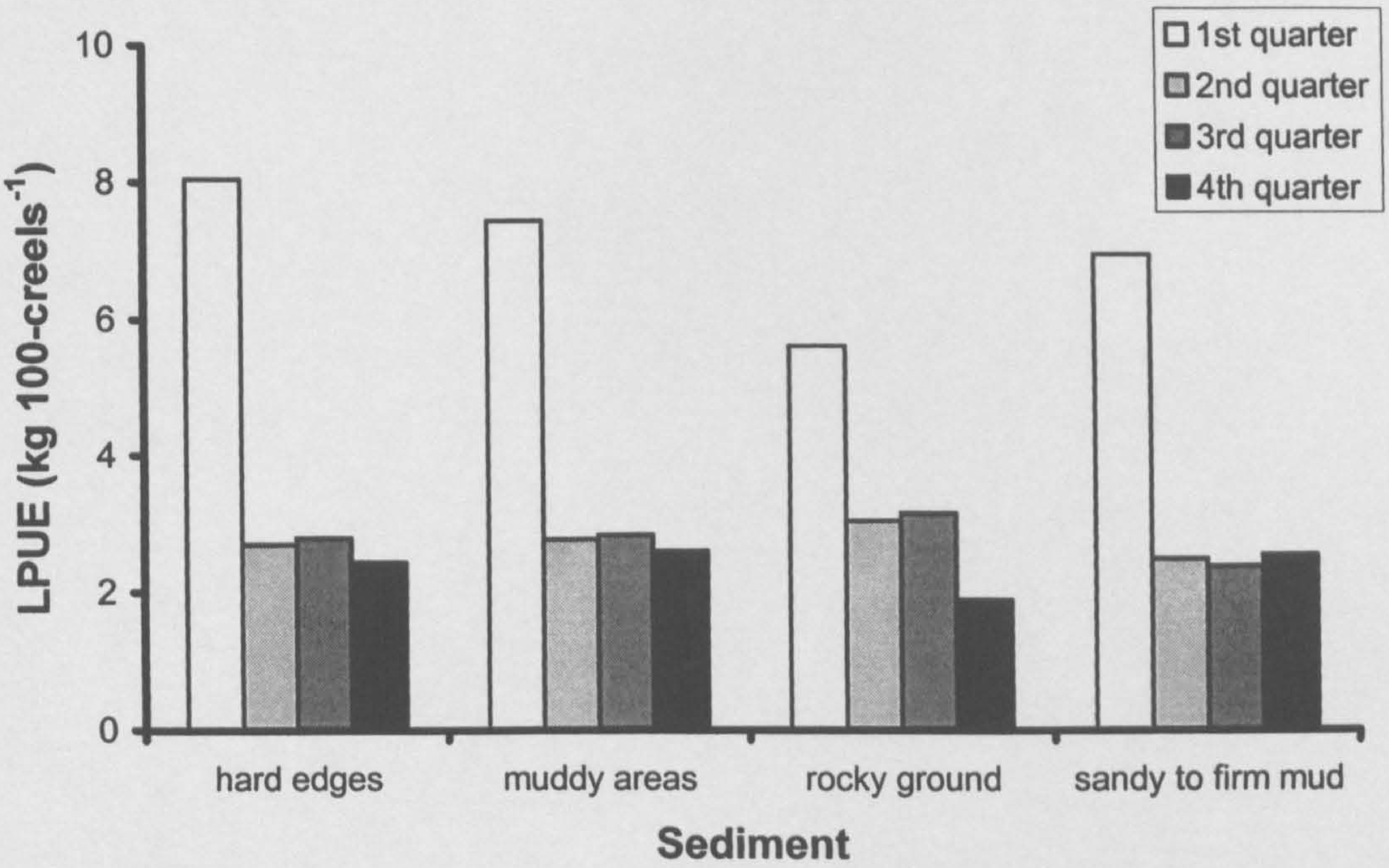

Figure 3.50 – Mean daily fishing LPUE (large) on different sediment types within the Torridon creel fishery by year quarter. Based on data from position recording units on seven vessels in 2004 and 2005.

Chapter 4

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4 CREEL-RELATED BEHAVIOUR

4.1 Introduction

There have been few previous studies of *Nephrops* behaviour in relation to fishing with baited creels. More is known about trawling through direct observations, measurements of catch performance and mesh selectivity, and through studies of activity rhythms in relation to vulnerability to trawls. Main & Sangster (1985) and Thorsteinsson (1986) found that most *Nephrops* outside their burrows, in the path of the net, were captured, while those in the openings of their burrows escaped capture by withdrawing into them. When outside the path of the net mouth, *Nephrops* have also been seen to escape by swimming or using repeated tail flick motions to avoid the approaching trawl (Main & Sangster, 1985).

The catchability of *Nephrops* in creels is strongly influenced by the behaviour of the animal, by factors affecting entry to the creels, such as feeding patterns and agonistic interactions, although very little work has been undertaken to quantify this. The behavioural responses of other decapod crustacean species towards baited creels have received more attention. These studies have found that there are a number of variables affecting catch rates and the sex ratios and size composition of catches (for example, Miller, 1978; Shelton, 1981; Karnofsky & Price, 1989; Miller, 1990; Miller & Addison, 1995; Zhou & Shirley, 1997; Jury *et al.*, 2001; Vazquez Archdale *et al.*, 2003).

Most of these studies have involved direct observations of the behavioural responses of crustaceans to creels. Of these studies, the majority have been conducted in aquaria (Miller, 1978; Miller & Addison, 1995; Zhou & Shirley, 1997), but some are field studies (Bjordal, 1986; Jury *et al.*, 2001; Vazquez Archdale *et al.*, 2003). Each approach has advantages and disadvantages. For example, in aquaria the conditions are unnatural, but allow greater experimental control; whereas in the field, conditions are natural, but the identity and history of individuals sampled are unknown and there is limited scope for experimental manipulation. To attempt to alleviate some of the aforementioned problems, Karnofsky & Price (1989) set up a semi-natural environment for lobsters, *Homarus americanus*, in a large aquarium. This system differed from previous aquarium studies, as the tank covered an area of 180 m^2, allowing detailed observation of the behaviour of a mixed population of known individuals in relation to a creel.

These studies have shown that all crustacean creels selectively sample their target population. Some of this selectivity is intentional (e.g. escape gaps and mesh size), but much is not, with other factors influencing creel selectivity (Jury *et al.*, 2001). For example, it has been observed in *Homarus* spp. that behavioural interactions disproportionately reduce the tendency of smaller individuals to enter creels (Miller & Addison, 1995). Larger *Nephrops* have been shown to spend more time per day foraging (Chapman & Howard, 1979) and therefore cover larger areas, leading to a greater chance of capture (Chapman *et al.*, 1975). Additionally, larger *Nephrops* win a greater proportion of agonistic encounters (Bjordal, 1986; Richardson, 1996; Mouat, 2002), deterring smaller animals from entering creels. This has led to the widespread view that, in general, the catch from crustacean creels is not representative of the stock in the sea. Furthermore, since feeding and locomotor activity are related to water temperature and the moult and reproductive (in females, at least) cycles (Chapman, 1980), the selectivity of baited creels is expected to vary seasonally.

Direct observations of crustaceans around creels have shown a general trend of low capture rates regardless of the experimental approach. Karnofsky & Price (1989) found that only 2% of *H. americanus* that approached the creels were captured. Similarly, when Jury *et al.* (2001) studied *H. americanus* in the field, they found that less than 1% of lobsters that approached creels were caught. In both studies, the overwhelming majority of lobsters that entered the creel eventually escaped (92% and 94%, respectively). Several factors affect the catch in crustacean creels, including animal size, sex and environmental conditions. In particular, interactions between individuals inside and outside the creel may be responsible for low and variable catch rates (Richards *et al.*, 1983; Miller, 1990).

Aquarium observations of *Nephrops* behaviour in creels with escape gaps, showed that animals too big to walk through escape gaps could roll through 90° to pass through the gaps on their sides (Livingstone, 2001). Such behaviour has been observed in other species (Brown, 1982; Lovewell and Addison, 1989). Livingstone (2001) observed that large animals tended to be dominant over others in the aquarium and were therefore the most likely to enter the creels. Richardson (1996) found that blinding by high light intensities (Chapter 6; Section 6.1) had no apparent effect on *Nephrops* responses to a baited creel and that catch rates were very low for blind and visually intact animals.

The most comprehensive study to date on *Nephrops* behaviour toward creels was carried out by Bjordal (1986), who found low catch rates (6.1% of observed animals) and selection for large individuals. Suggested reasons for these findings were: low motivation for food search, difficulties in locating creel entrances, aggression between individuals around the creel and the creel being an unnatural object. Bjordal (1986) concluded that there would be an initial selection against smaller *Nephrops* because they are likely to have shorter feeding excursions and are therefore less likely to encounter a creel. Furthermore, once the creel was encountered, smaller individuals were more likely to withdraw (Bjordal, 1986).

The current study aimed to improve the knowledge of *Nephrops* behaviour in relation to creel fishing. Much more needs to be understood about creel selectivity and the field behaviour of the target organism. The selectivity of *Nephrops* creels has been investigated with regards to how animal size determines both creel avoidance and outcome of inter- or intra-specific interactions. To achieve this, creel-related behaviour has been observed in large tanks, enabling more detailed observations on a known population. The introduction of crab species to creels enabled the assessment of the effect of their presence on *Nephrops* behaviour. However, findings from aquarium studies should be interpreted with caution, since the conditions do not mimic those of their natural environment. To complement observations under controlled conditions, the behaviour of *Nephrops* was observed in the field using underwater video at two distinct sites: one shallow-water site and one deep-water site. Results from field studies not only provide essential information of *Nephrops* behaviour in the natural environment, but also enable confirmation of results from aquaria (Miller & Addison, 1995).

4.2 Materials and Methods

4.2.1 Field observations

4.2.1.1 Loch Sween

A shallow-water site in Loch Sween was selected to set up underwater closed-circuit television (CCTV) equipment. The site is located in upper Loch Sween (Sailean Mhòr), Argyll, and was chosen as it was shallow enough for diving, close to a shore-based power supply and had a reasonable number of *Nephrops* burrows close–by, although burrow density (<1 m^{-2}) was low when compared to deep water *Nephrops* grounds (>1 m^{-2}). The location of the site is shown in Chapter 1, Figure 1.12.

The CCTV equipment consisted of a frame (0.8 × 0.8 × 1.4 m, anchored 200 mm into sediment) supporting an underwater video camera (Kongsberg Simrad OE1358, mounted *ca* 850 mm above sediment surface) and lamps (Versabeam, Remote Ocean Systems × 2, mounted *ca* 550 mm above sediment surface) with infrared filters (Infrared, Kodak), below which a baited creel was placed (Figure 4.1). The creel design was a standard *Nephrops* creel as used in the fishery, fitted with an escape gap and baited with salted herring (Chapter 1; Figure 1.3). The site had an underwater cliff close to the shore, which dropped to a muddy seabed with *Nephrops* burrows at approximately 18 m depth. The camera frame and creel were positioned on the seabed approximately 30 m from the cliff by divers, who were in direct audio communication with the surface party. Divers (including the author) rebaited the creel each day during recording periods. This scene was recorded with a time-lapse video recorder (Panasonic AG6730 time lapse, connected to the shore by a cable produced by MacArtney, Aberdeen) for successive periods of 17–77 h (24 h on each 3 h tape), on three occasions: in February 2004, May 2004 and November 2004, yielding 362 h of time-lapse video recording. During the deployment in February a side and end view of the creel were recorded; in May and November a view from directly above the creel was recorded (Figure 4.2).

4.2.1.2 Loch Torridon

To overcome problems associated with being close to the shore in Loch Sween, and to observe *Nephrops* behaviour on the deeper grounds of the Torridon area, a self-contained CCTV system was developed by University Marine Biological Station Millport (UMBSM) and Fisheries Research Services (FRS) Aberdeen. This system comprised a ROS 20/20 Navigator TV Camera video camera, two Bowtech in-house infrared LED lamps, a Sony GV-D1000E Digital Mini Video Recorder in underwater housing and one underwater battery unit (housing 12 V 12 A h lead acid batteries), all mounted on a robust steel frame (1500mm × 1500mm × 1500mm) that held the creel in position beneath the camera (Figure 4.1). This system could be used down to a depth of 200 m and placed anywhere that the research vessel could operate. A major disadvantage compared with the shore-based system is that it was limited to 3 h recording time. This meant the equipment had to be retrieved, serviced and re-deployed several times during daylight hours. Deployments of the self-contained CCTV system were undertaken during December 2004 from the FRV *Clupea*.

4.2.2 Aquarium-based observations of behaviour

Experiments were carried out at the Fish Behaviour Unit at the FRS Marine Laboratory, Aberdeen. *Nephrops* were collected at night under red light from a commercial creel vessel, placed into tubes, and stored in salt water in a light-proof container. Transport to aquaria was undertaken at night in salt water to avoid stress and prevent blinding

(Chapter 6; Section 6.1). On arrival in Aberdeen, specimens were stored separately in cages (allowing adequate water exchange) in large tanks. Animals were kept in the holding tanks for an acclimatisation period of two weeks prior to behavioural observations.

Observations were made in two 7 m × 3 m glass-reinforced plastic tanks filled with sea water to a depth of 1 m. Artificial burrows (inverted sections of plastic guttering 300 mm in length) were placed in the tanks in random positions and a single *Nephrops* was placed into each 'burrow'. Animals were acclimatised for 24 h in the observation tank before a baited creel with an escape gap was placed in the centre of the tank and video recording commenced. The creel was a standard *Nephrops* creel as used in the fishery.

The video recording system consisted of two Panasonic colour video cameras viewing the entire tank, with another focusing on the creel (Mini Bowcam with integrated lighting, Bowtec), allowing a good overall view and a detailed image of the creel (Figure 4.3). The images from the two cameras were combined on one monitor using a multiplexer (B/W Multiplexer, model B4D MUX). Lighting was provided by existing fluorescent luminaries fitted with red plastic sleeves (Encapsulite, Leighton Buzzard, Bedfordshire), as well as red spotlights. This produced a wavelength of light mainly outside the visible range of *Nephrops* (Gaten *et al.*, 1990). This scene was recorded with a time-lapse video recorder (Panasonic AG6730 time lapse). An additional lamp with a green filter (Encapsulite) that produced dim light within the range of *Nephrops* spectral sensitivity (as determined by electroretinography; Johnson *et al.*, 2002) was operated on a time switch to simulate the day-night cycle (12L:12D).

Nephrops behaviour was recorded for 48 h, following which the creel was removed and the number, sex and carapace length of captured animals was noted. Individual *Nephrops* were used in only one observation session each.

The effect of the presence of crabs was assessed using a stocking density of 10 *Nephrops* with a size distribution of 36–50 mm CL. Experiments were carried out with a baited creel containing five *Liocarcinus depurator*, five *Carcinus maenas* and two *Cancer pagurus*, compared with a baited creel containing no crabs (Figure 4.3). Four replicates of each experimental condition were conducted with representative sizes of *Nephrops*. Two trials were run simultaneously, with the sequence of treatments randomised in each.

4.2.3 Video tape analysis

Video recordings were first reviewed to classify the types of behaviour exhibited by *Nephrops*. Using this information, behaviour was recorded in terms of:- 1. approaches to the creel, 2. creel-related behaviour (entry to the eye) and 3. agonistic interactions. The method of recording enabled data to be analysed in relation to how interactions affected catchability.

1. Approaches

An approach was defined as any individual *Nephrops* entering the field of view of the camera and pausing in the vicinity of the creel. A separate approach was recorded if more than 2 minutes elapsed after an individual left the field of view, or if the next individual to enter the field of view was obviously different (e.g. in size). Individuals could not be easily distinguished, so many *Nephrops* will have been recorded approaching multiple times. The presence of individuals and their size (small, medium or large) were recorded every minute, enabling the residence time at the creel and any contact with the creel or interactions with other animals to be recorded. Since *Nephrops* were not individually recognisable, it was not possible to apply a known size to individuals; however, size was estimated on each occasion by comparing the size of the animal to that of the creel. Size categories were kept constant throughout the video analysis.

2. Creel-related behaviour

The time at which a *Nephrops* entered the creel eye was noted. The type of eye entry was also noted for each minute the animal was in the eye. *Nephrops* always entered the creel eye by moving forwards and usually by inserting their antennae and chelipeds in the eye first. Eye entries were classified by the furthest extent to which an individual *Nephrops* entered as: 'cheliped entry', where part of a cheliped entered the creel eye; 'carapace entry', where part of the carapace entered the creel eye; 'telson entry', where the telson entered the creel eye (this does not imply telson-first entry); and 'capture', where the animal passed completely through the creel eye and entered the creel. This method provided information on the time that individuals spent in the creel eye, as well as the degree of entry (Figures 4.4 & 4.5). If an individual that had entered the creel subsequently escaped, the escape route was noted (i.e. through the creel eye or escape gap) (Figure 4.6).

3. Agonistic interactions

An agonistic encounter was scored if an animal acted aggressively towards another (Huntingford & Turner, 1987) and if the opponent apparently responded to this action (Richardson, 1996) (Figure 4.7). If an agonistic encounter occurred between two animals, the time it occurred was noted to the nearest minute, along with the approximate size of the animals involved (small, medium or large). Because the time at the creel and the approximate size of animals involved in the bout were known, it was possible to determine whether the winner / loser of the bout was smaller, larger or of equal size and which (if any) animals left the vicinity of the creel following the bout. The loser of the bout was defined as the animal that retreated following an interaction and the winner was the animal that forced the loser to retreat (Richardson, 1996). If both of the animals retreated simultaneously, the outcome was recorded as a draw. Using this information it was possible to investigate how residency at the creel affected the outcome of the bout.

Other qualitative observations, such as the presence of crabs or seals in the field experiments, were noted (Figure 4.8).

In the aquarium experiments, a view of the whole tank was examined on an hourly basis to assess the number of animals out of their artificial burrows.

4.2.4 Data analysis

The relationship between the number of *Nephrops* entering the creel eye and their residence time at the creel was investigated using a rank correlation analysis for both field and aquaria studies.

The relationship between the number of *Nephrops* outside their burrows, approaching the creel, entering the creel eye and engaging in agonistic encounters, and the duration of the experiment, was tested using least squares regression for both treatments where crabs were present or absent from the creel.

The effect of the presence or absence of crabs in the creel on the number and average duration of approaches, the number of animals caught and the number of agonistic encounters, was tested using a Kruskal-Wallis test.

The occurrence of different extents of entry to the creel eye was analysed in relation to the presence or absence of crabs in the creel with a Scheirer-Ray-Hare test (Sokal & Rohlf, 1995), and in relation to *Nephrops* size category with two-way ANOVA of arcsine-transformed percentage of each entry type by relative size category data.

The percentage of larger *Nephrops* winning an agonistic encounter, in relation to size of animals in bout, was tested using a Kruskal-Wallis test.

To test for an effect of resident *Nephrops* size on success in agonistic interactions, the mean percentage of interactions won by the larger, smaller or equal-sized resident animal was tested using a Kruskal-Wallis test. Additionally, the average duration of residency at the creel for larger, smaller or equal sized *Nephrops* was analysed in relation to the outcome of the agonistic interaction (win or lose) with two-way ANOVA.

The percentage of agonistic interactions that were followed by the winner, loser, both or neither contestants leaving the vicinity of the creel, was analysed in relation to the relative size of the winning contestants with two-way ANOVA of arcsine-transformed percentage of each outcome category by relative size category. Pairwise comparisons were made with Tukey's test, where appropriate.

All analysis was undertaken using MINITAB version 13 (Minitab Inc., State College, Pennsylvania).

4.3 Results

4.3.1 Loch Sween

From analysis of 343 h of video recordings from the deployments in Loch Sween during February, May and November 2004, there were 154 approaches by *Nephrops* to the creel (individual animals may approach a number of times). The number of approaches varied considerably between deployments, with an average of 1.2, 0.6 and 0.1 approaches per hour during deployments in February, May and November, respectively.

During May and November 2004 (300 h of observation), there were 79 approaches by *Nephrops* to the creel, of which 42 (53%) resulted in an entry to the creel eye, but only two of these (5%) resulted in entry to the creel (February data not included because the creel eyes were not clearly visible, owing to the positioning of the creel). Following entry, both captured animals immediately commenced feeding on the bait and neither escaped from the creel (Figure 4.9). The two captured animals were both caught during the May deployment. Only three agonistic interactions were observed, of which one was followed by both *Nephrops* leaving the vicinity of the creel shortly afterwards. On most (97%) of the occasions when an individual *Nephrops* left the field of view, it did so without any apparent prior interspecific interaction.

The majority of approaches to the creel occurred during the night, with 100%, 83% and 96% of approaches occurring during darkness (darkness defined as when illumation by infrared lamps was required for camera operation; from 45 min to 1.5 h before sunset and after sunrise) in February, May and November, respectively (Figures 4.10–4.12). The largest number of approaches during the daytime occurred in May, when the period of darkness was shortest (8 h) (Figure 4.11). There appeared to be a reduction in activity around the creels just after midnight (GMT) in February and just before midnight in November. The creel eyes were obscured in the February deployment, but in May and November the observed number of eye entries followed a similar temporal pattern to that of approaches (Figures 4.11 & 4.12).

To investigate the effect of *Nephrops* size on creel eye entry, results from the three observation periods were combined, as eye entry numbers were low. In May and November, 47%, 60% and 45% of approaches led to an eye entry for large, medium and small *Nephrops*, respectively. *Nephrops* size appeared to affect the degree of entry into the creel eye. There was some evidence that a higher proportion of eye entries by 'large' animals progressed to carapace or telson entry compared with eye entries by 'medium' or 'small' animals, and both of the captured individuals were large (Figure 4.13). There was also a significant positive relationship between the number of eye entries and time in the field of view (r_s=0.7, d.f.=42, *P*<0.001) (also significant with outlier removed) (Figure 4.14).

Liocarcinus depurator and *C. maenas* generally appeared within the field of view in less than 10 min (longer for *C. pagurus*). Both *L. depurator* and *C. maenas* entered and exited through the creel eye and escape gap, whereas *C. pagurus* were too large to pass through the creel eye, but spent long periods on top of the creel, or in the funnels leading to the creel eyes. Interactions between crabs and *Nephrops* involving cheliped displays, were common and were often followed by the *Nephrops* leaving the field of view. This was particularly so with encounters with *C. pagurus*. A greater number of crabs were observed in and around the creel during the November deployment, compared with the February and May deployments. The greater abundance of crabs in November was associated with a reduced number of approaches by *Nephrops*. A young seal (*Halichoerus grypus*) was observed in the vicinity of the creel during the November deployment, displacing it on several occasions. No *Nephrops* were observed in the vicinity of the creel 30 minutes prior to or after a seal was seen at the creel (Figure 4.8).

4.3.2 Loch Torridon

Unfortunately, no useable video was recorded, as the LED lamps were not powerful enough to illuminate the creel. Only a few deployments were possible, owing to adverse weather conditions. For these reasons no results from the CCTV study in Loch Torridon are available.

4.3.3 Aquarium study

In eight 48-h observation periods (four periods with crabs and four without) there were totals of 1674 and 1827 approaches by *Nephrops* to the creel containing crabs and no crabs, respectively.

Of the 1674 approaches to the creel containing crabs, *Nephrops* entered the creel eye on 188 occasions (11%) and of those only four individuals entered the creel (2%). Following creel entry, one animal escaped through the creel eye (Figure 4.15). These results imply an average of 42 approaches per individual before capture (although since individuals could not be identified, this figure should be treated with caution). A similar pattern in eye entries was observed when the creel contained no crabs: 173 (9.5%) of the 1827 approaches led to the animal entering the eye, from which, 12 (7%) entered the creel. of which two escaped (17%) one *via* the escape gap and one *via* the creel eye (Figure 4.16). There was an average of 15 approaches per captured individual, a much lower value than when crabs were present. Departures from the creel were preceded by agonistic encounters in 278 cases (17% of departures) when crabs were present and in 355 cases (20%) when crabs were absent (Figures 4.15 & 4.16).

A similar pattern in approach behaviour was observed in the two treatments (crabs and no crabs), with no difference in the number (Figure 4.17; H_1=0.08, P=0.773) or duration (Figure 4.18; H_1=0.08, P=0.773) of approaches. There was no significant interaction between the effects of treatment type (crabs and no crabs) and eye entry type (Scheirer-Ray-Hare interaction, $F_{2,23}$=0.02, P=0.978), indicating that relative occurrence of eye entry types was not dependent on the presence or absence of crabs in the creel. The presence or absence of crabs in the creel had no effect on the extent of eye entry

(Scheirer-Ray-Hare treatment, $F_{1,23}$=0.08, P=0.787), but the frequency of the different types of eye entry differed significantly (Scheirer-Ray-Hare eye entry type, $F_{2,23}$=7.14, P=0.005): there were significantly more cheliped eye entries than both carapace and telson entries (Tukey, P<0.05, for both comparisons) (Figure 4.19). There were significantly more *Nephrops* caught in the creel without crabs than were caught in the creel with crabs (H_1=4.4, P<0.05) (Figure 4.20).

There was a difference between the two treatments in the relationship between the percentage of *Nephrops* outside their burrows and elapsed time during the observation period. When crabs were present in the creel, the percentage of animals outside their burrows did not vary significantly over time (R^2=0.08; $F_{1,46}$=3.77, P=0.058). In contrast, when there were no crabs in the creel, there was a significant decrease in the percentage of animals outside their burrows with time (R^2=0.55; $F_{1,46}$=56.49, P<0.001) (Figure 4.21). There was a slight but significant decrease in the number of approaches over time when no crabs were present (R^2=0.43; $F_{1,46}$=35.35, P<0.001), however, where crabs were present there was no change over time (R^2=0.00; $F_{1,46}$=0.00, P=0.969) (Figure 4.22).

There were apparent differences in residence time at the creel with *Nephrops* size; however, there was a large amount of variability both within and between treatments. There was a significant interaction between *Nephrops* size and experiment type (field or aquaria; crabs and no crabs) (Scheirer-Ray-Hare interaction, $F_{4,3550}$=16.77, P<0.001), indicating that the effect of *Nephrops* size differed among experiment types (Figure 4.23).

There was no significant interaction between the effects of size and eye entry type (2-way ANOVA interaction, $F_{6,91}=0.47$, $P=0.634$), indicating that the relative occurrence of eye entry types was not dependent on the size of *Nephrops*. The size of *Nephrops* had no effect on the type of eye entry (2-way ANOVA size, $F_{2,91}=0.27$, $P=0.767$) but, as previously observed, the type of eye entry varied significantly (2-way ANOVA eye entry type, $F_{3,91}=51.98$, $P<0.001$). There were significantly more cheliped entries than carapace, telson and full entry (capture) types (Tukey, $P<0.001$ for each comparison), and significantly more carapace entries than full entries ($P<0.001$) (Figure 4.24).

There was a significant increase in the number of *Nephrops* eye entries with residence time at the creel ($r_s=0.448$, d.f.=407, $P<0.001$) (Figure 4.25). For both treatments, a decrease in the number of eye entries with time was evident (crabs: $R^2=0.12$, $F_{1,46}=6.16$, $P<0.05$; no crabs: $R^2=0.53$, $F_{1,46}=22.85$, $P<0.001$). There was a difference in the pattern of decrease. Where no crabs were present, particularly high numbers of eye entries occurred initially, with the pattern following a logarithmic decrease (Figure 4.26). A similar pattern was observed for creel entries although the numbers of entries was too small to draw any firm conclusions.

There was no significant difference in the mean number of agonistic interactions when the creel contained crabs and when it did not ($H_1=2.08$, $P=0.149$) (Figure 4.27). There was a difference between treatments in the relationship between number of agonistic encounters per unit time and elapsed time during the experiment. When crabs were present in the creel, there was no significant relationship between the number of agonistic encounters and elapsed time ($R^2=0.01$; $F_{1,46}=0.58$, $P=0.449$). In contrast, there was a logarithmic decrease in the number of agonistic encounters with time when there were no crabs in the creel ($R^2=0.49$; $F_{1,46}=12.43$, $P=0.001$) (Figure 4.28).

There was a significant effect of relative size of contestants on the outcome of the bout ($F_{2,62}=357.42$, $P<0.001$). Significantly more agonistic interactions were won by larger contestants (89.9±3.6%) than by smaller animals (7.6±3.3%) (Tukey, $P<0.05$). In turn, significantly more interactions were won by smaller animals than contests that were drawn (i.e. no evident winner) (2.5±1.4%). There was no significant difference in the percentage of larger animals winning an encounter when different-sized pairings were investigated (Figure 4.29; $H_1=4.96$, $P=0.084$).

Most interactions occurred between a 'resident' individual (which had been in the vicinity of the creel for at least 1 min) and an 'intruder' (which had been in the vicinity of the creel for less time than the resident animal). The percentage of the resident *Nephrops* winning an encounter varied significantly between resident animals that were larger, smaller or of equal size to the intruder ($H_2=16.94$, $P<0.001$). Of particular interest is the observation that a higher percentage of smaller animals won an encounter when resident at the creel, than when the smaller animal was an intruder (Figure 4.30). There was no significant interaction between the relative size of the resident animal and the result (win or lose) of an agonistic encounter with duration of residency (2-way ANOVA interaction, $F_{2,631}=2.10$, $P=0.123$), indicating that the outcome of an agonistic interaction following residency at a creel is not dependent on the size of *Nephrops*. The size of resident *Nephrops* had no effect on the duration of residency (2-way ANOVA size, $F_{2,631}=1.08$, $P=0.34$) but the duration of residence differed significantly among outcomes (2-way ANOVA outcome, $F_{1,631}=256.58$, $P<0.001$), with the average duration at the creel being significantly greater for those resident animals winning a bout (Tukey, $P<0.001$) (Figure 4.31).

After an agonistic interaction, the loser, winner, both or neither contestants could leave the vicinity of the creel (i.e. the camera's field of view). The relative frequencies of these events appeared to be unaffected by the winning animals being larger, smaller or of equal size to the loser (2-way ANOVA interaction, $F_{6,84}=1.78$, $P=0.586$). These different subsequences were not equally common (2-way ANOVA departure category, $F_{3,84}=61.10$, $P<0.001$): the most common subsequence was that only the loser departed and the least likely was that only the winner departed. The percentage of interactions in which the winner left was the least, and not significantly different from the percentage of interactions when both individuals left. The percentage of interactions in which the looser left was significantly higher (Figure 4.32, Tukey, $P<0.001$).

A few of the *Nephrops* used in aquarium experiments were missing a cheliped. Of the six size-matched agonistic interactions, in which one animal was missing a cheliped, all were won by the intact animal. Of the 49 interactions in which the intact animal was smaller than an opponent missing a cheliped, 45 (92%) were won by the intact (smaller) animal. No agonistic interactions were observed in which the smaller contestant was missing a cheliped. None of the *Nephrops* missing a cheliped was observed to enter the creel eye.

4.4 Discussion

The low catch rate of *Nephrops* creels observed in the present field and aquarium studies concur with the findings from all three previous comparable studies (Bjordal, 1986; Richardson *et al.*, 1996; Livingstone, 2001), as well as observations in other species targeted by creels (Miller, 1978; Miller 1980; Karnofsky & Price, 1989; Miller, 1990; Zhou & Shirley, 1997; Jury *et al.*, 2001). In contrast to findings from other species, analysis of *Nephrops* behaviour in the field and aquarium, as well as from ghost fishing experiments (Chapter 7), indicates that following capture, the majority of animals do not escape. This suggests that the low catch rate of *Nephrops* in creels is almost solely due to their low tendency to enter creels, whereas in other species, low catch rates appear to result from a combination of low entry rates and subsequent escape (Karnofsky & Price, 1989; Jury *et al.*, 2001). Bjordal (1986) suggested that the low catch rate of *Nephrops* in creels was due to low motivation for food search, difficulty in locating the entrances, aggressive behaviour between *Nephrops* and a degree of gear avoidance. The present study clearly illustrates the potential importance of both intra- and inter-specific behavioural interactions by *Nephrops*, as well as responses to the creel itself, in determining the number and type of *Nephrops* that enter the creel.

Although catch rates were low, there were many approaches to the creel by *Nephrops* in the present field and aquarium studies. In the field study, it was not possible to determine the number of individual *Nephrops* that approached the creel. Under controlled conditions in an aquarium, an average of 42 and 15 approaches were made per capture, depending respectively on whether there were crabs present in the creel or not. From the three field deployments, a total of 77 approaches to the creel were made per capture. If the relationship between the number of approaches and creel entry in the field is similar to that observed in the aquarium, this result suggests that a large proportion of individual *Nephrops* approaching the creel may, following a number of approaches, eventually enter. A similar finding has been made for *H. americanus*, where results from field and aquarium studies indicated that the number of individuals that approached was similar to the number eventually captured (Karnofsky & Price, 1989; Jury *et al.*, 2001). Although this finding implies that the catchability of *Nephrops* in creels may be greater than previously thought (Bjordal, 1983), it must be treated with a degree of caution, as values from aquarium studies have been obtained under artificial conditions, with a number of factors making comparisons of field and aquarium studies difficult. Furthermore, only two animals were captured during the field studies, both in May, with none of the approaches during the other two deployments leading to the capture of any *Nephrops*.

There are several reasons why observations in aquaria are not directly comparable with field studies. One striking difference observed in aquaria was that, on average, around 50% of animals were outside their artificial burrows at any one time, whereas in the field, *Nephrops* spend most of their time within burrows or at their openings (Chapman & Howard, 1979; Chapman, 1980). American lobsters, *H. americanus*, were similarly more prone to remaining outside shelter in an aquarium/mesocosm study and it was suggested that the lack of predators in the tank made animals less wary of being exposed (Karnofsky & Price, 1989). It is likely that the high percentage of *Nephrops* outside their burrows in aquaria is a combination of a lack of predators together with the shelters being poor representations of natural burrows (a suggestion also made by Mouat (2002)). Experimental animals were starved prior to observation sessions and stocking densities were high, which, together with the high degree of burrow emergence, may explain the high levels of activity around the creel (Chapman & Howard, 1979). Approaches to the creel were less frequent in the field and correspondingly fewer agonistic encounters were observed. In contrast, a large number of agonistic encounters were observed around the creel in the aquarium, which may explain the lower catch rate. Several previous studies indicate that aggressive behaviour reduces the catch rate with baited traps (*H. americanus* – Richards *et al.*, 1983; Karnofsky & Price, 1989; Jury *et al.*, 2001, *N. norvegicus* - Bjordal, 1983). Furthermore, the bait odour in aquaria is confined to the space of the tank and is likely to fill the whole body of water, resulting in the chemical attractants and stimulus gradient being different from that of the field.

The pattern of *Nephrops* activity observed in this study may reflect the nocturnal habits of *Nephrops* in shallow water. The majority of approaches to the creel and subsequent entries to the creel eye occurred during periods of darkness (scotophase). The observed activity pattern is similar to patterns found in previous studies in shallow waters (<40 m), similar to those investigated in this study, in which peak emergence (and therefore trawl catch) occurred throughout the period of darkness (Chapman & Rice, 1971; Hillis, 1971; Chapman *et al.*, 1972; Chapman & Howard, 1979). This pattern varies with depth, with peaks of activity observed around dawn and dusk at depths of 40–100 m, and at depths of >100 m, activity is greatest during the day (Höglund & Dybern, 1965; Simpson, 1965; Hillis, 1971; Chapman *et al.*, 1972; Farmer, 1974; Chapman & Howard, 1979; Redant & De Clerck, 1984). In conjunction with the present findings, these previous results suggest that the diel pattern of approaches towards, and entry into creels, varies with depth.

Significant crepuscular activity around creels was only observed during May, when the duration of darkness was shortest (8 h). This may reflect a greater motivation for foraging during lighter periods when the period of darkness, and therefore feeding, is reduced. Additionally, catches in the fisheries studied in Chapter 2 were highest during the spring, suggesting that during this period *Nephrops* are more likely to leave their burrows in search of food (with this increase in activity related to the moult and reproductive cycle in *Nephrops* – Chapter 2).

Diel variations in activity observed in the field were not observed in aquaria, perhaps because the lighting used to simulate the day night cycle may have been inappropriate (only one small light was used) – a problem also identified by Mouat (2002). There were temporal changes in activity in the aquarium, with the percentage of *Nephrops* outside their burrows, approaches and eye and creel entries decreasing over the 48-h trial when no crabs were present in the creel. This decrease in the number of eye and creel entries was particularly evident during the first few hours. The decrease in activity with time suggests that in this trial the attraction of *Nephrops* to the creel decreased with time. Catchability in creels has been shown to reduce with soak time, as the fishing power becomes reduced with increasing catch (gear saturation) and the bait odour diminishes over time (Miller, 1978; Miller, 1980; Smith & Jamieson, 1989; Miller, 1990). When crabs were present in the creel there was no significant change in the percentage of animals outside their burrows, approaching the creel, and entering the creel over time. In addition, the decrease in the number of *Nephrops* entering the creel eye was much less marked when crabs were present in the creel, than when they were absent. This observed difference suggests that following the introduction of the creel the presence of crabs led to the cessation of the initial high level of activity observed when crabs were absent in the creel, suggesting that crabs may reduce catchability of *Nephrops* before they reach the creel. It is also possible that the presence of crabs in the creel feeding on the bait would stop the bait odour diminishing over time, and therefore activity may remain constant over the trial period. There may, however, be other viable explanations for the observed difference. Comparing trials with bait protected or unprotected from crabs could test this hypothesis. As catches were significantly greater in trials with no crabs, it is likely that where no crabs are present, it is gear saturation that is affecting the activity in and around the creel, although diminishing bait odour is still likely to have an effect.

There was an increase in eye entries with time spent at the creel by individual animals in both field and aquarium studies, although animals were stationary for much of the time spent in the vicinity of the creel. This provides evidence that although *Nephrops* may actively seek the entrance, they have difficulties finding the creel entrances, even following extended search periods (Bjordal, 1983).

It was also observed in field studies that the number of approaches to the creel per hour reduced during the three study periods, with very little creel-related activity during the November deployment. Catch sampling from the creel fishery (Chapter 2) has shown that the lowest catches occur during the autumn and winter, providing a possible explanation for the low number of approaches during this period. Other, local factors may also have been responsible, such as changes in the *Nephrops* density due to commercial fishing, or the influence of high densities of crabs and the presence of a seal (see below).

Field observations indicated that the presence of crabs (*C. pagurus, L. depurator* and *C. maenas*) in and around the creel limited the number of *Nephrops* approaching the creel and entering the creel eye, thus reducing the number of *Nephrops* entering the creel. There is evidence of a similar phenomenon during ghost fishing (Chapter 7). When the effect of crabs was tested under controlled conditions in the aquarium, there was no difference in *Nephrops* activity around the creel when crabs were present or absent. There were, however, significantly more *Nephrops* caught in creels without crabs, suggesting that crabs affected the behaviour of *Nephrops* at the point of entry to the creel only. This apparent discrepancy between the field and aquarium may be a result of differences in the number and size of crabs present in the two studies. At times, higher densities of crabs were observed in the field and, when *C. pagurus* was present, they were much larger than those used in the aquarium. Furthermore, rather than being inside the creel, they were often observed on top of the creel or in the funnels leading to the creel eyes. *Nephrops* in the field study therefore usually encountered crabs before passing through the eye and often left the field of view following such an encounter.

There have been few previous studies on interspecific interactions between target and non-target species in relation to creel catchability. Where predators of the target species are present in and around a creel, reduced catches have been observed in several species (Ritchie, 1972; Addison, 1995; Miller & Addison, 1995; Brock, 2004). In a case where the target species (*H. americanus*) was the predator, Richards *et al.* (1983) found that the presence of crabs had no significant effect on catch of *H. americanus*. *Homarus americanus* has been observed to compete aggressively with certain crab species to enter the creel (Jury *et al.*, 2001). Interactions between *Nephrops* and the crab species observed in this study are different from those observed previously from other species, with the presence of crabs apparently deterring *Nephrops* from entering the creel. Although no firm conclusions can be drawn regarding the presence of a seal during the November deployment, seals are commonly considered by fishers as damaging to both gear and catch (Moore, 2003). It is likely that in part the seal's presence was responsible for the low number of animals observed during November, as no *Nephrops* were observed during periods when the seal was present.

The two animals caught during the present field study were large (49 and 52 mm CL). In the aquarium the majority of animals caught were medium-sized (43-48 mm CL). These results suggest that creels are more selective of larger animals, although to confirm this in the field further study would be necessary. A positive relationship between catchability and animal size in crustacean creel fisheries has been observed for several species (for a review of studies see Miller (1990)). A general trend of size selectivity in creels shows a catch rate near zero where animals are small enough to pass freely through the creel mesh, increasing over intermediate sizes, followed by the catch rate levelling off and in some cases decreasing for large animals (Miller, 1990). In the case of *Nephrops,* the differences in the size of animals caught by trawling and creeling is well recognised (Chapman & Howard, 1979; Bjordal, 1983; Bell *et al.*, 2006). Both sectors exploit animals over the same size range, but trawl catches generally contain a higher proportion of small animals and the size distribution of creel catches is either symmetrical or skewed towards large animals (Bjordal, 1983). It is possible that differences observed in the sectors could be due to differences in the grounds that are fished for example creels can be located precisely near to reefs, wrecks and other ground unsuitable for trawling, often leading to the two sectors exploiting different patches in the same general area. It is, however, more likely that creels select larger animals in the population than are caught by trawls (Chapman & Howard, 1979; Bjordal, 1983; Tuck and Bailey, 2000; Eggert & Ulmestrand, 2000; Bell *et al.*, 2006).

There may be several reasons for the selection of larger animals by creels. It is likely that a greater proportion of larger animals would approach the creel, as large animals are thought to have greater foraging ranges because of higher food requirements and faster walking rates (Morrissy, 1973). Large *Nephrops,* in particular, have been observed to spend more time per day foraging than small *Nephrops* (Chapman & Howard, 1979). This may be particularly relevant as it has been suggested that bait odour does not induce *Nephrops* to leave its burrow, and that attraction only occurs when the animals are stimulated during feeding excursions (Chapman & Howard, 1979), although the validity of this suggestion is as yet unclear. The occurrence of ovigerous females within creels would, for these animals at least, argue against this. Mòller & Naylor (1980) did, however, show that, in the laboratory, the endogenous nocturnal activity rhythm was modified by the presence of food. It is likely that smaller animals will have a reduced chance of reaching the creel than larger animals with greater feeding ranges. Unfortunately it was not possible to test this hypothesis in the present study, as individual animals in both the field and aquaria were not readily identifiable.

Once at the creel, a further size selection of *Nephrops* has been observed, with larger animals found to have a higher threshold for gear avoidance and the presence of conspecifics, or other species (Bjordal, 1983; Livingstone, 2001). In a previous aquarium study, the largest *Nephrops* consistently displaced smaller individuals and were the only ones to enter the creel (Livingstone, 2001). In the current study, larger *Nephrops* did not spend more time in the vicinity of the creel, suggesting that duration of residence at the creel may not be a good predictor of the probability of creel entry. Results from the field study did indicate that with increasing size, *Nephrops* were more likely to enter further into the creel eye and of those entering only large animals were trapped. This was not observed in the aquarium, perhaps because of the unnatural conditions. There is scope for further investigation of the effect of *Nephrops* size on gear avoidance and capture.

No conclusions can be drawn regarding the effects of agonistic encounters on selectivity under natural conditions, owing to the small number of encounters observed. It has, however, been suggested that such encounters are common around *Nephrops* creels (Chapman & Rice, 1971) and may have a large influence on catch (Miller, 1990). In the present aquarium study, larger *Nephrops* won the majority of agonistic encounters regardless of the extent of size difference, a finding also reported by Richardson (1996) and Mouat (2002). In most cases, following an agonistic encounter, the losing animal (in the majority of cases the smaller animal) leaves the vicinity of the creel. When one animal was resident at the creel, if the animals were size matched, the resident animal had a greater probability of winning an agonistic encounter. Even when the resident animal was smaller than its opponent it still won, on average, more encounters. A further observation also suggests that the duration of residence is important in determining the outcome of an agonistic encounter. On average, *Nephrops* winning a bout were resident at the creel for significantly longer than those that lost, regardless of the size of the resident animal. This 'prior residence effect' has been demonstrated in a number of decapod crustaceans (Peeke *et al.,* 1995), as well as in relation to burrow occupancy for *Nephrops* (Chapman & Rice, 1971; Farmer, 1974).

These findings suggest that there may be a selection of larger animals at the creel following agonistic interactions, with larger animals being more likely to win the encounter, thus staying in the vicinity of the creel, where their chances of entering will increase with residence time. It would be useful to know the extent to which animals inside the creel affect the entry of *Nephrops* from outside. Unfortunately, in the field study only two animals were caught and no such interactions were observed. Conversely, in the aquarium study, the large amount of activity meant it was not possible to distinguish such interactions.

Once caught, a further size selection was observed, with one of the two small *Nephrops* caught in aquarium trials leaving the creel through the escape gap. Escape gaps have been shown to reduce numbers of undersized catch in a number of species (for example, Templeman, 1958; Bowen, 1963; Bain, 1967; Krouse, 1978; Nulk, 1978; Maynard *et al*, 1987; Lovewell & Addison, 1989; Miller, 1990; Shanks *et al.*, 1997; Livingstone, 2001; Stewart & Ferrell, 2002). In the present study, the captured animal escaped by rolling through 90° to pass through the gap on its side, a manoeuvre which has been observed previously in *Nephrops* that are too big to walk through the escape gap (Livingstone, 2001). It appears that escape gaps are not 100% effective at allowing small animals to escape, but are likely to enable a significant proportion of undersized *Nephrops* to escape (Livingstone, 2001).

Nephrops with only one cheliped were not caught. Loss of a cheliped appeared to greatly reduce the chance of winning an agonistic encounter, even against smaller opponents. It is therefore likely that where activity around the creel is high, as observed in the aquarium, animals with only one cheliped are unlikely to enter the creel, as they will be more readily displaced by conspecifics. Conflicting results were obtained by Mouat (2002), who suggested that the loss of a cheliped resulted in a compensatory increase in aggressiveness. Results from the current study, agree with a number of studies of other species (for a review of studies see Juanes & Smith (1995)). Furthermore, none of the *Nephrops* that were missing an appendage were observed to enter the creel eye, suggesting that gear avoidance in these animals was greater than in intact *Nephrops.* Although further work is needed to clarify our understanding of the effect of limb damage on catchability, the current study indicates that damage may limit *Nephrops* catch through intraspecific interactions and gear avoidance. Cheliped loss has been observed in a significant number of discarded animals (J. Adey, personal observation). The loss of a cheliped is likely to reduce an animal's competitiveness, which may be particularly important in conflicts between individuals for occupancy of burrows and when competing for food (Richardson, 1996). The loss of a cheliped may therefore result in increased mortality of discarded animals (Chapman, 1981). It is therefore desirable that care be taken during the discarding process to avoid unnecessary injuries. Since intact animals have greater market value, the possible avoidance of creels by animals lacking a cheliped is interesting and merits further investigation.

This study has indicated that the operation of *Nephrops* creels is characterised by a high ratio of approaches to entries. The present results provide evidence that low catchability in creels is particularly related to the following: creel avoidance, difficulty in finding creel entrances and, once in the creel eye, a reluctance to pass through the eye. This suggests there may be a tendency to avoid unnatural objects, which may be more influential than the attractiveness of the odour plume. The presence of crabs or other *Nephrops* appears to further increase creel avoidance. These observations concur with those of Bjordal, (1983). There is evidence of an effect of body size at certain stages of the capture process. Larger animals were less likely to be displaced from the vicinity of the creel following agonistic encounters, entered further into the creel eye and were ultimately more likely to be captured. This, combined with likely greater foraging range and longer activity periods of larger *Nephrops* (Chapman & Howard, 1979), indicates that creels are selective for larger animals. Assessing the size selection of creels in more detail in the field would require knowledge of the fished population of *Nephrops.*

The likely selectivity of creels for larger animals, both prior to (creel attraction and behaviour at the creel) and following capture (use of escape gaps, where fitted) has several advantages to the fishery. It allows fishermen to exploit larger, higher-value animals, of which the majority will have had an opportunity to reproduce before being caught. Furthermore, creel selectivity reduces the number of small *Nephrops* being brought to the surface and discarded, a process that increases mortality (Chapter 6).

4.5 Summary

- Behavioural studies were conducted using CCTV in the field and in aquaria.

- In the field, activity around the creel was greatest during periods of darkness.

- A reduction of creel-related activity with time in aquaria provides some evidence that gear saturation may occur within the creel fishery.

- The catchability of *Nephrops* in creels was low: a large number of approaches to the creel by *Nephrops* were observed when compared with the number of entries.

- The presence of crabs in the creel reduced the number of *Nephrops* entering. Where crabs were present in large numbers, in particular where large *C. pagurus* were found on the outside of the creel, the number *Nephrops* approaching the creel appeared to be reduced.

- Results provide further evidence to support the theory that low entry rates to *Nephrops* creels occur due to a low motivation for food search, creel avoidance (i.e. inhibitory effects), difficulty in finding creel entrances and intra- and inter-specific interactions (Bjordal, 1983).

- *Nephrops* creels appeared to capture large animals selectively in both field and aquarium studies. Results suggested larger animals have a higher threshold for disturbances, such as the presence of unfamiliar objects such as gear and the presence of conspecifics or other species.

- Some evidence was provided to suggest larger *Nephrops* were more likely to enter further into the creel eye, implying reduced gear avoidance in larger animals.

- Behavioural interactions between *Nephrops* may play a particularly important role in determining catch composition.

- A selection of larger animals at the creel following agonistic interactions was observed, with larger animals being more likely to win such encounters, thus staying in the vicinity of the creel, where their chances of entering will increase.

- The loss of a cheliped markedly reduced an animal's ability to win an agonistic encounter and appeared to increase gear repellency, suggesting that the loss of an appendage dramatically reduces catchability.

- Evidence of a residency effect was observed, with duration of residency important in determining the outcome of an agonistic encounter.

- Escape gaps were shown to allow a proportion of small animals to escape.

- The observed selectivity of *Nephrops* creels is important in ensuring the sustainability of this fishery.

Figure 4.1 - Underwater CCTV system used in Loch Sween (left), self-contained CCTV video system aboard FRV *Clupea* (right) and view of creel in place in Loch Sween (bottom). For dimensions see text (Photo: A. Weetman, top right; I.P. Smith, bottom).

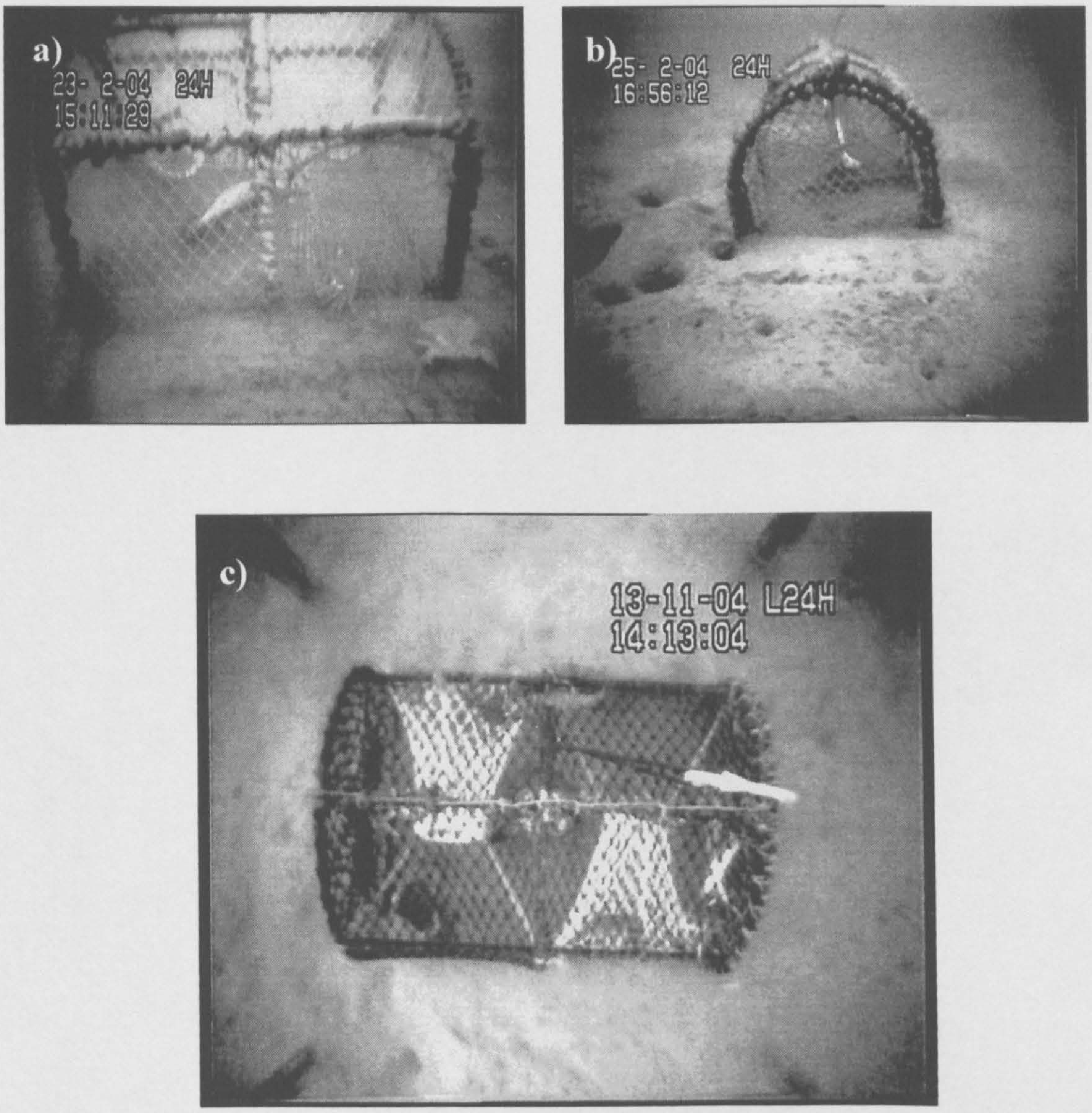

Figure 4.2 – Images of the three angles from which data were recorded during field trials a) Feburary 04, b) Feburary 04, c) May and November 04.

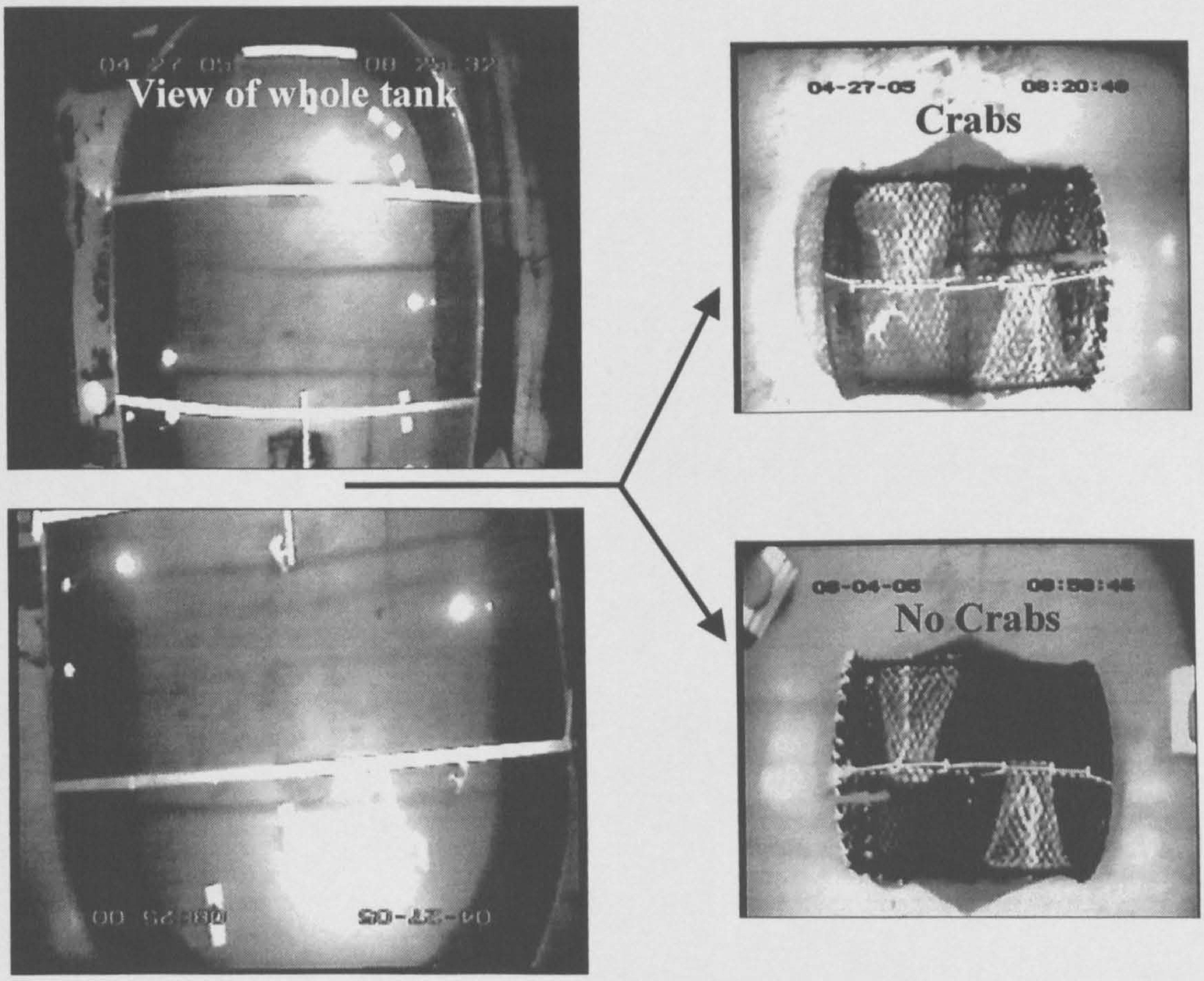

Figure 4.3 – Images from video cameras in aquarium study of *Nephrops* behaviour around baited creels: left, whole tank view; right, close-up view of creel in centre of tank.

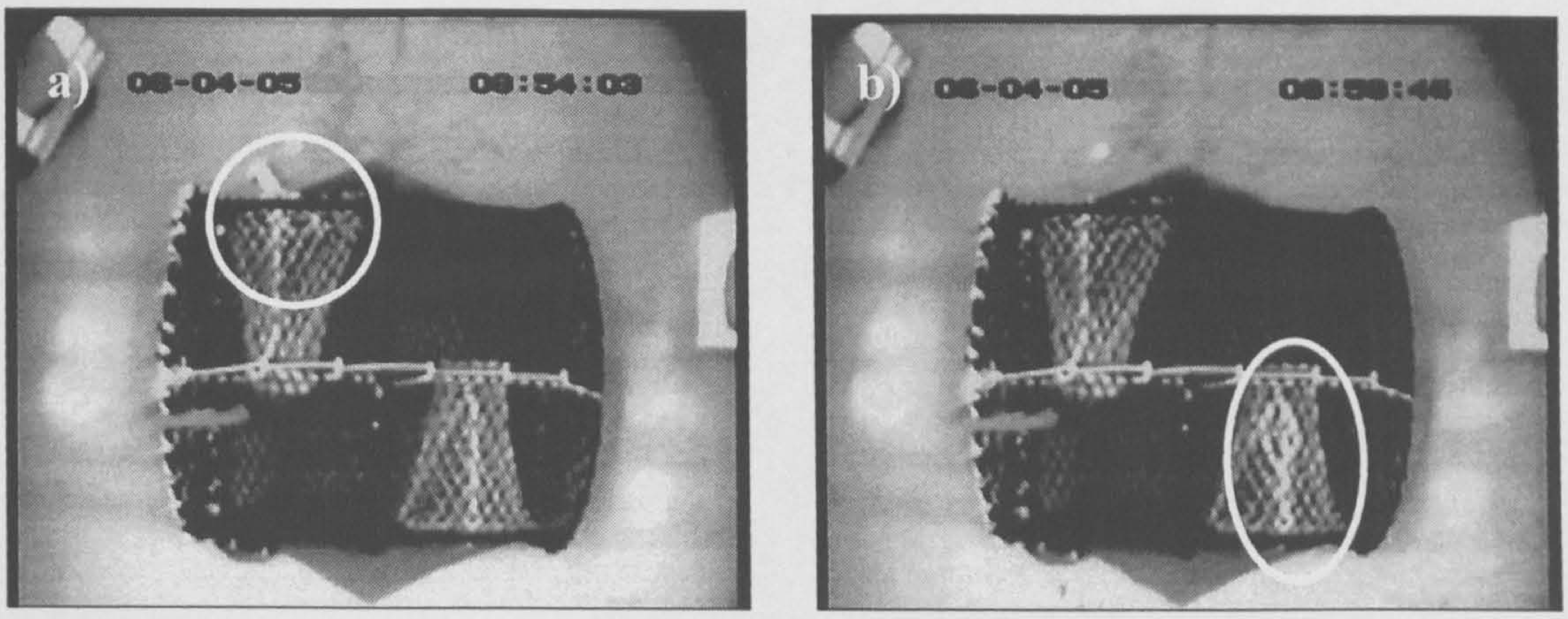

Figure 4.4 – Images of *Nephrops* entering the creel eye in aquaria a) cheliped entry, b) telson entry

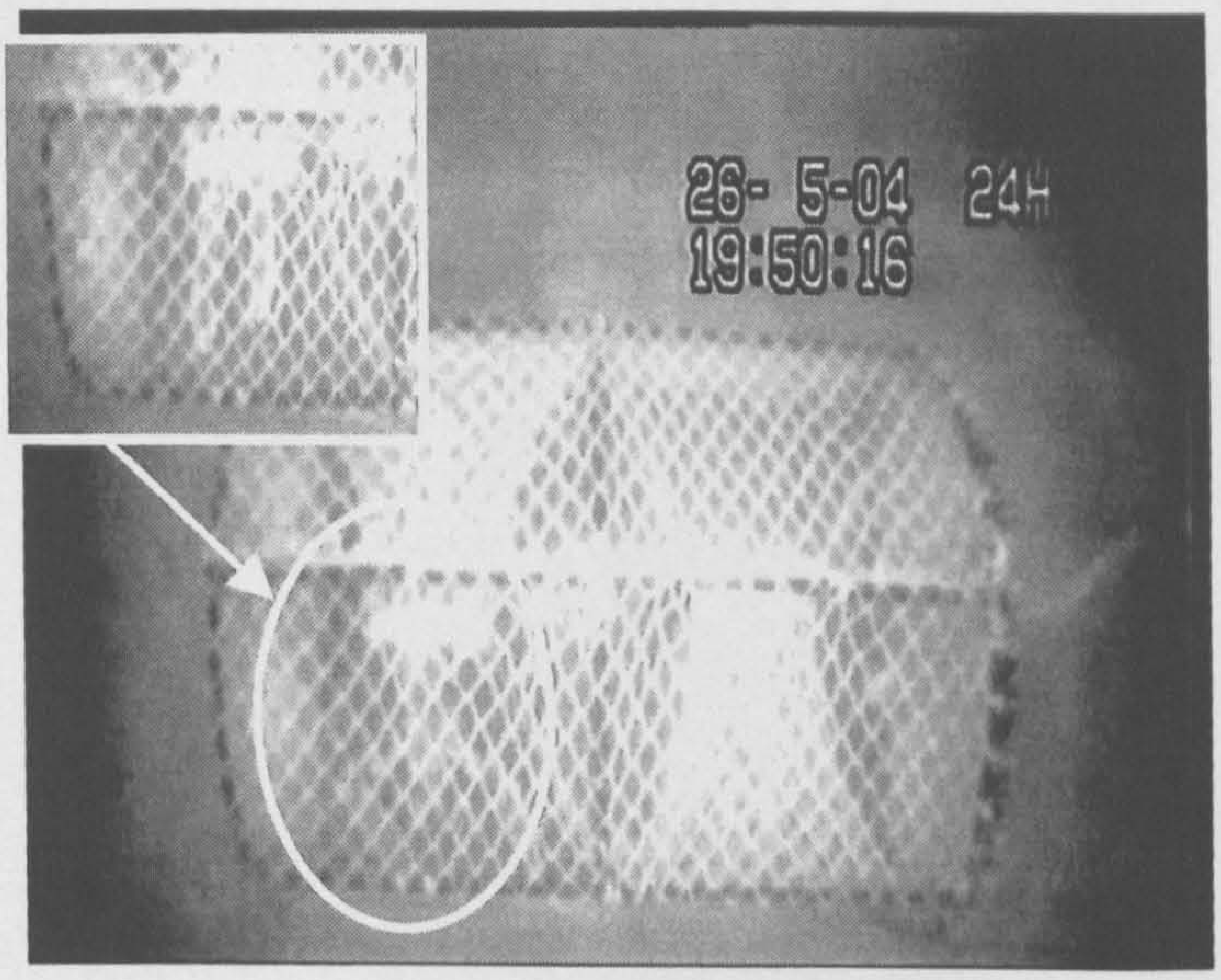

Figure 4.5 – Typical field behaviour of *Nephrops*, creel entry.

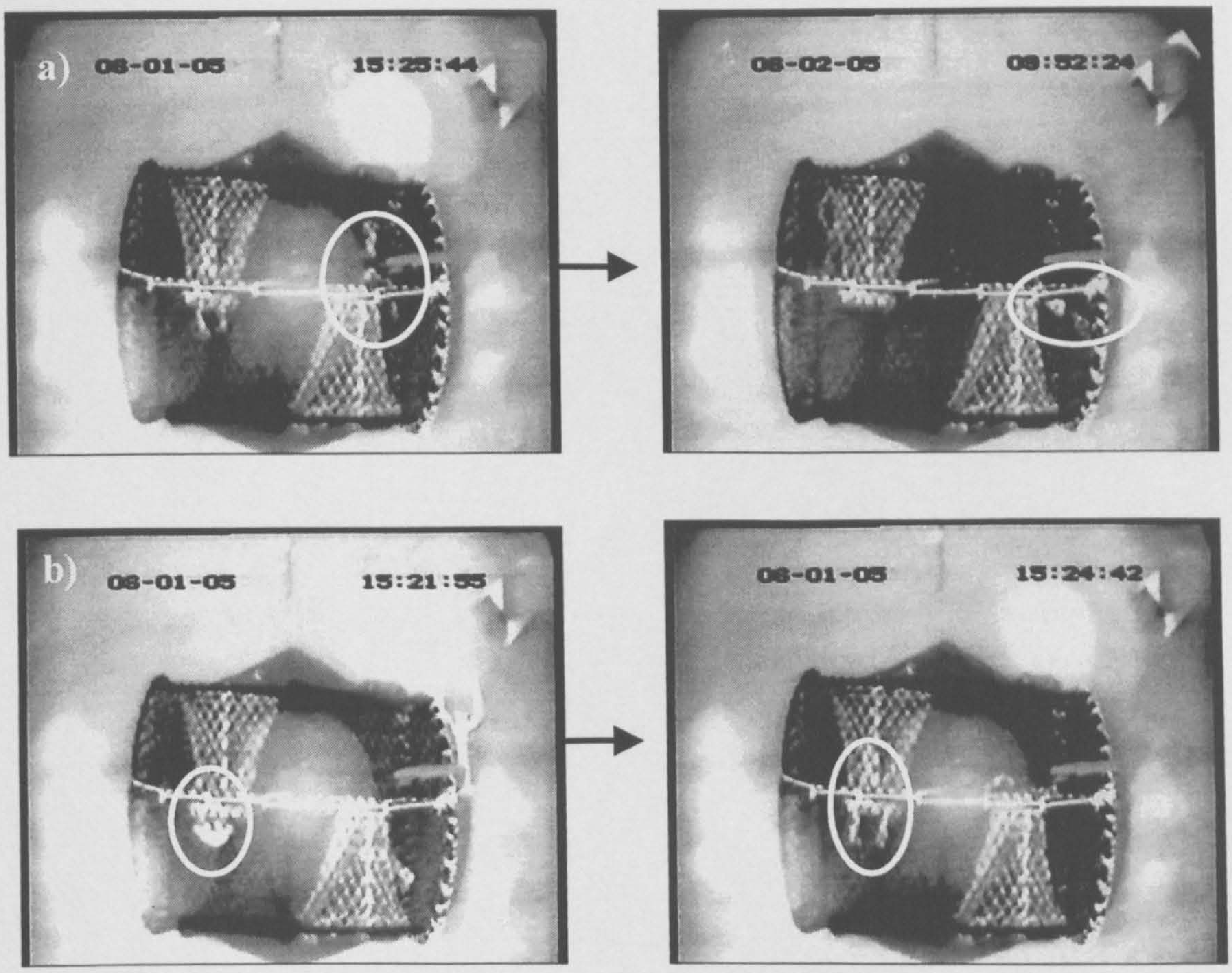

Figure 4.6 – Images of *Nephrops* escaping from the creel in aquaria a) through the escape gap, b) through creel eye.

Figure 4.7 – Typical field behaviour of *Nephrops*, agonistic encounter.

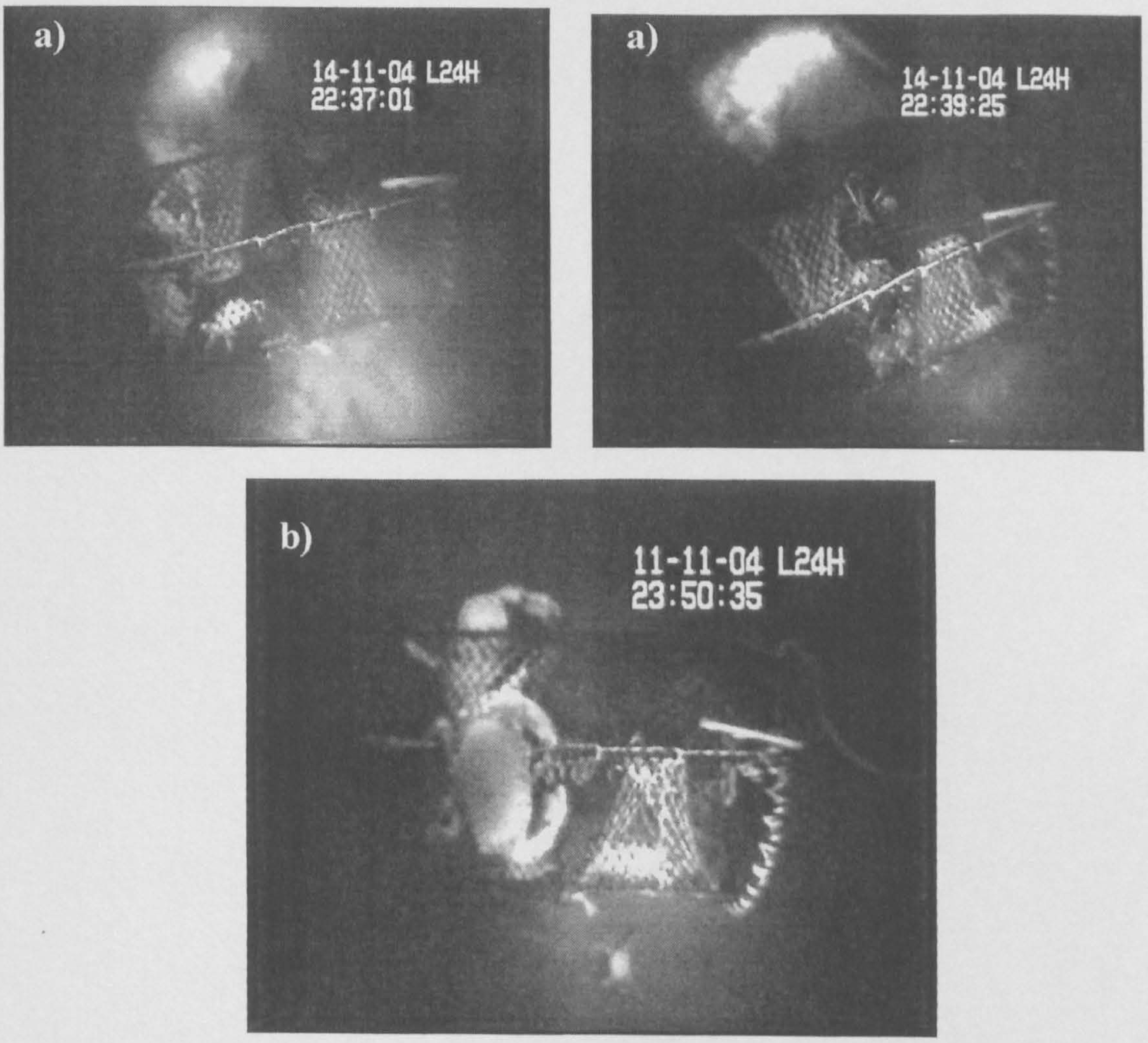

Figure 4.8 – Images of the influence of other species to creel catchability in field studies by a) seal, b) crabs

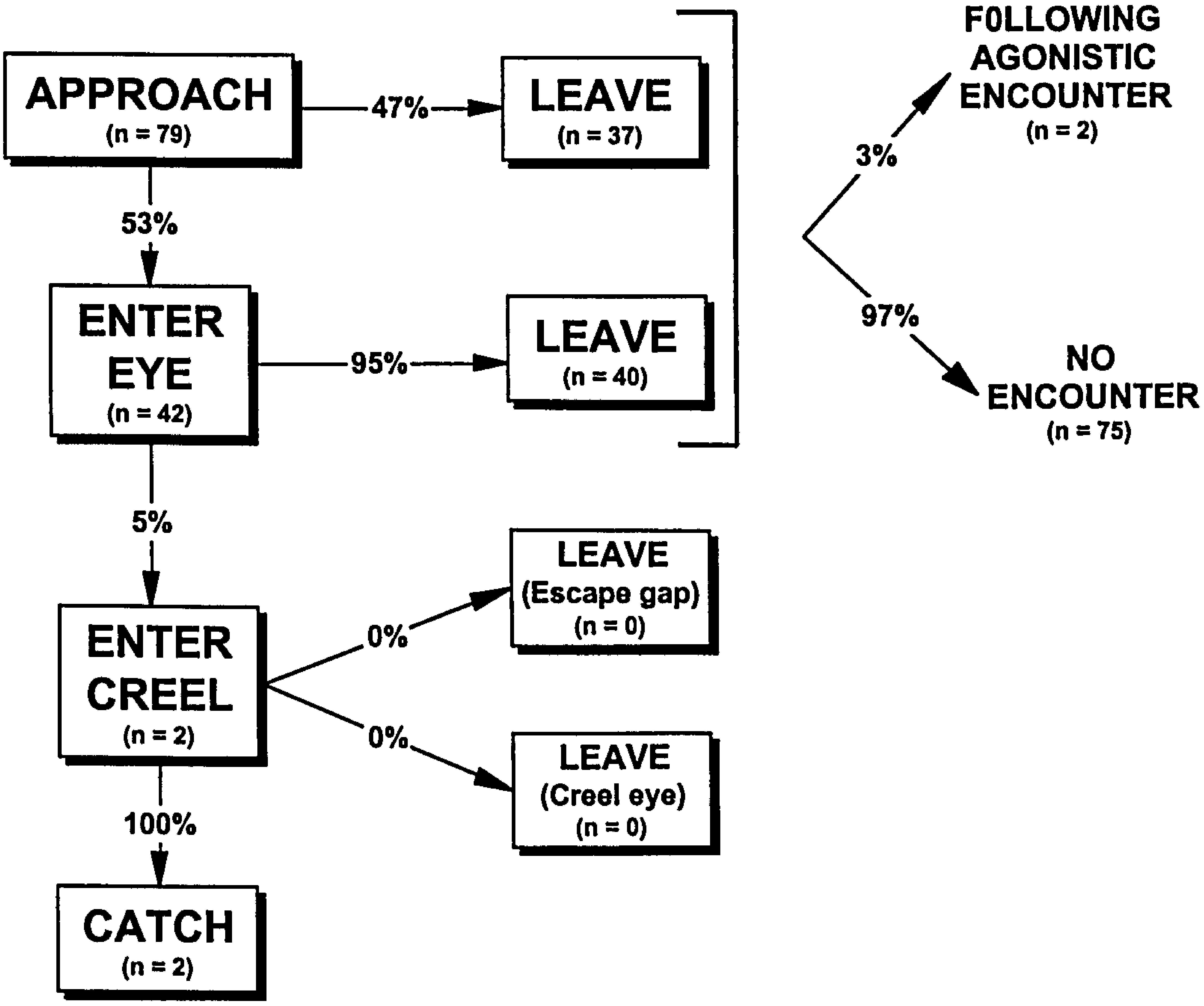

Figure 4.9 - Summary of behavioural responses of *Nephrops* to a baited creel, observed in field deployments in May and November 2004 (February deployment omitted). From 300 h of video.

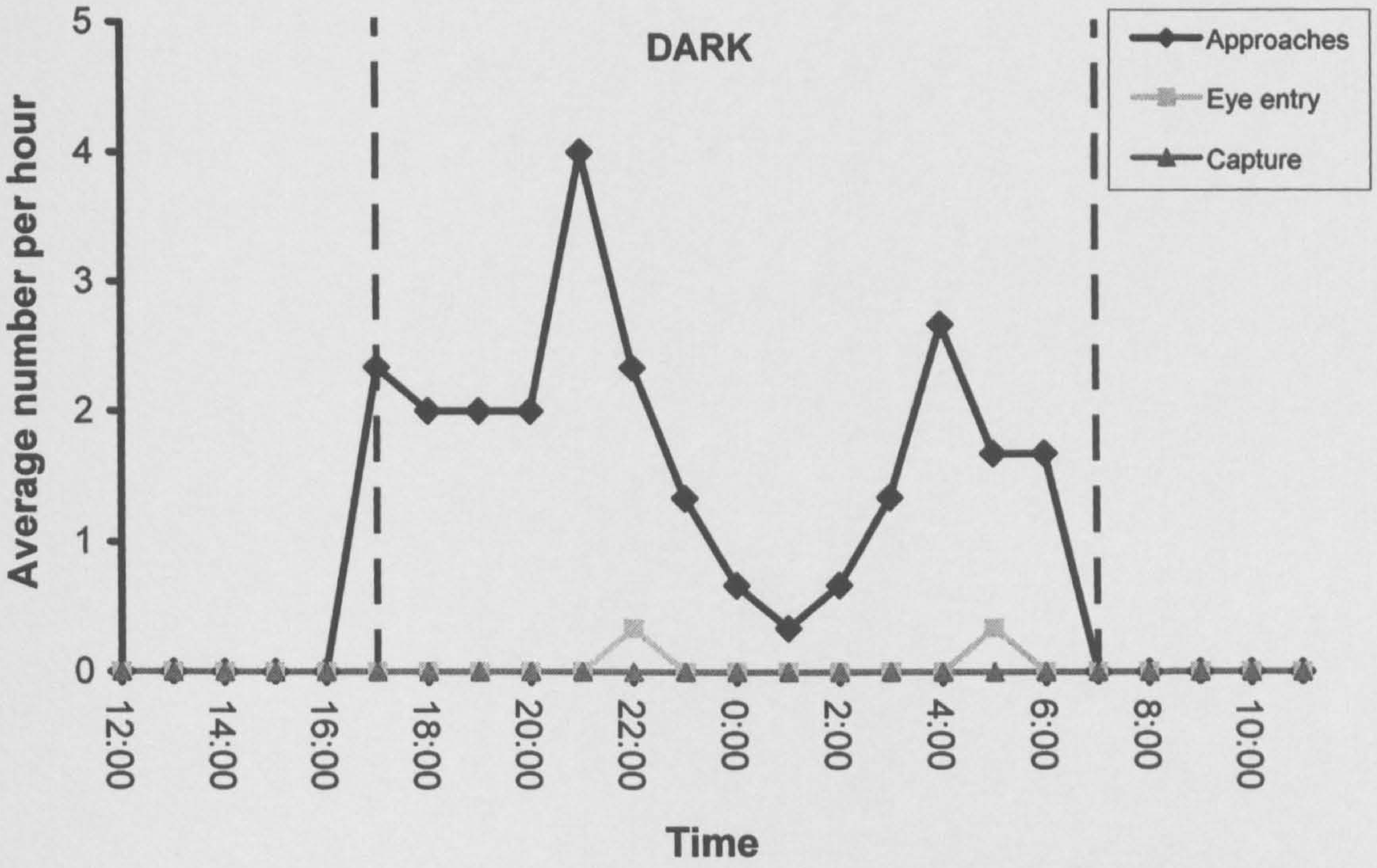

Figure 4.10 - Average number of approaches, creel eye entries and caught animals during three, 24-h periods in February 2004 (Eyes obscured, leading to eye entries being underrepresented).

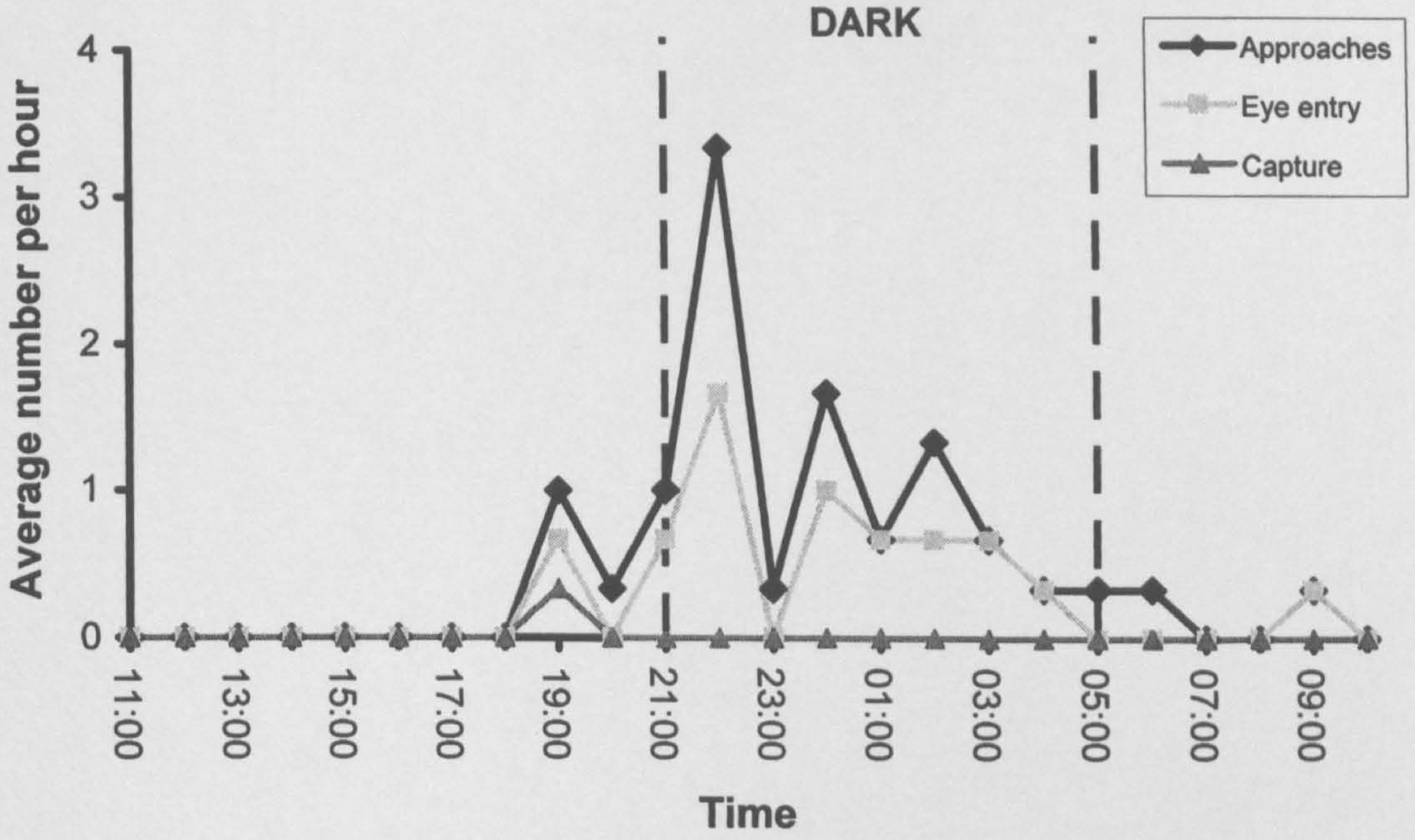

Figure 4.11 - Average number of approaches, creel eye entries and caught animals during three, 24-h periods in May 2004.

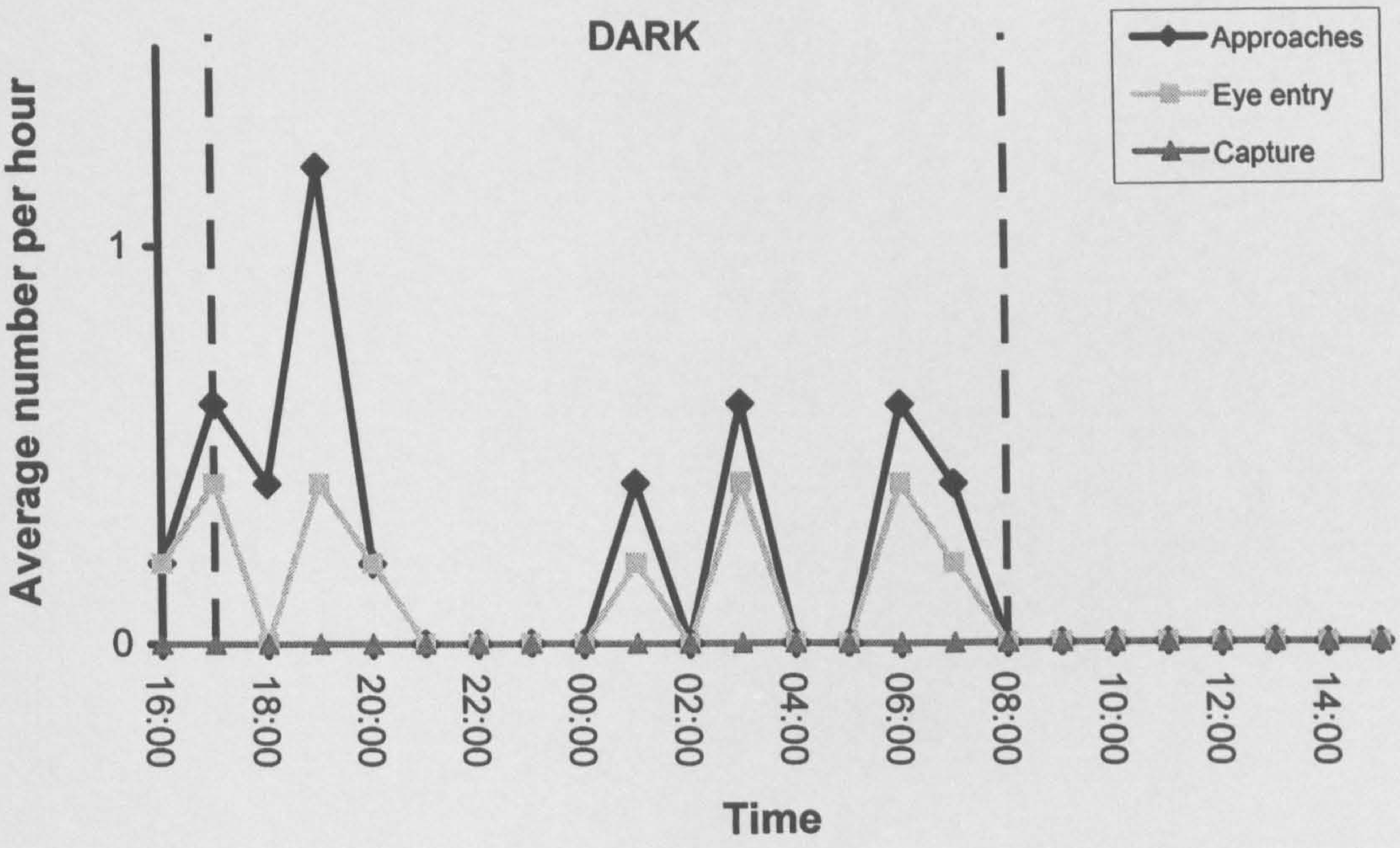

Figure 4.12 - Average number of approaches, creel eye entries and caught animals during five, 24-h periods in November 2004.

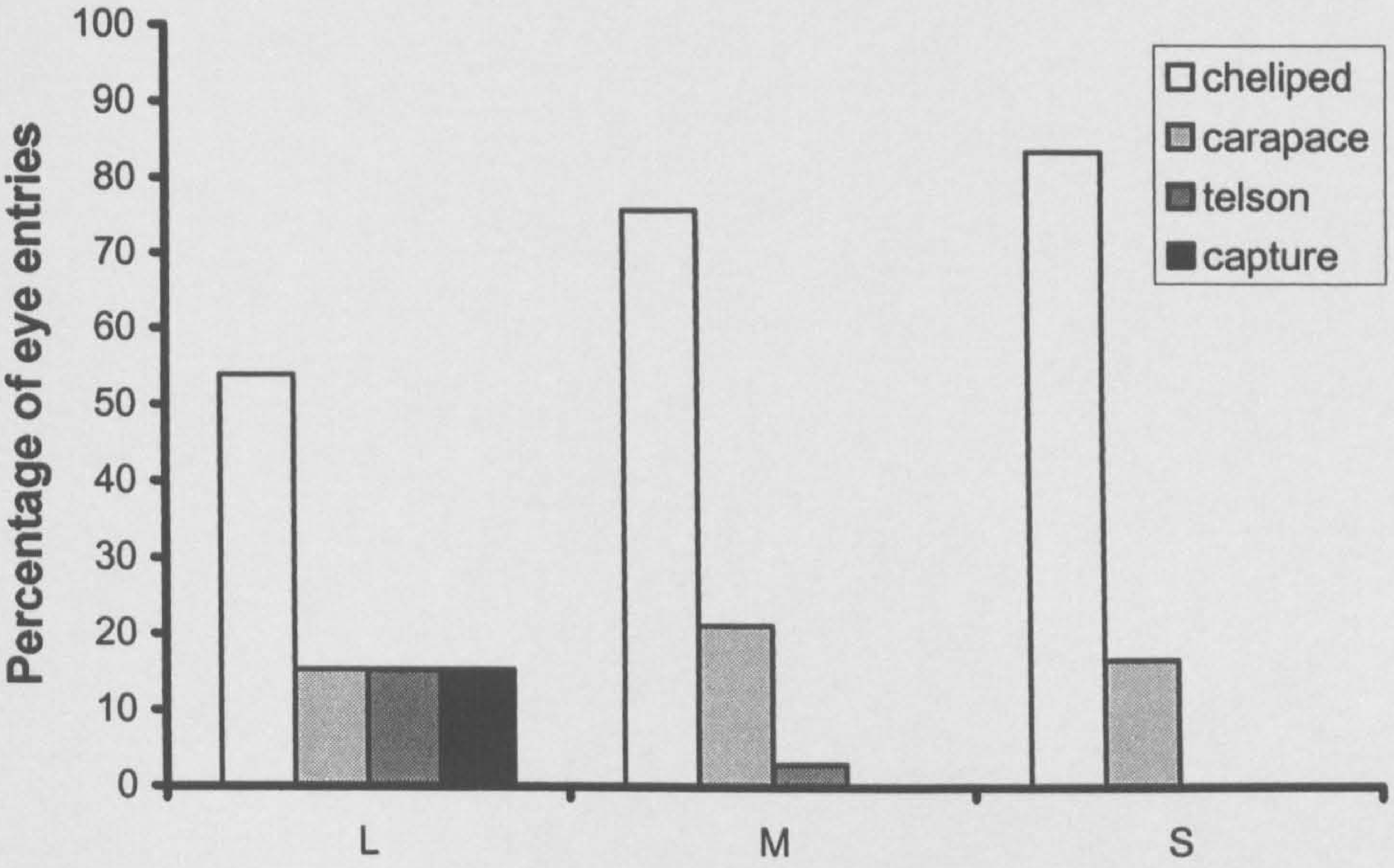

Figure 4.13 – Percentage of eye entries and degree of *Nephrops* entry into creel eye for large (L), medium (M) and small (S) *Nephrops* during 11 trials in Loch Sween (values combined).

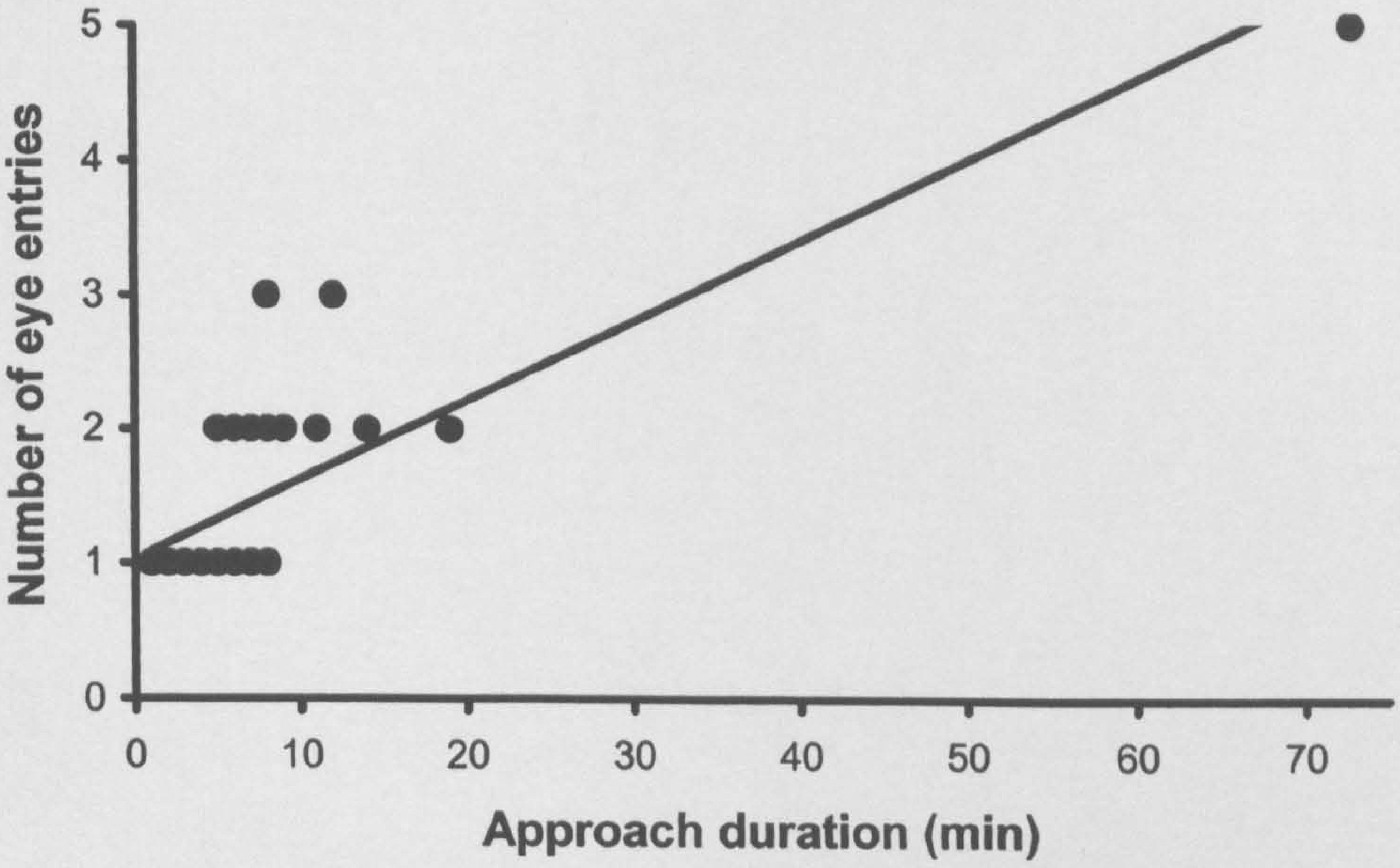

Figure 4.14 – The relationship between time at creel and the number of creel eye entries by individual *Nephrops* in Loch Sween, linear trend line displayed (February deployment omitted).

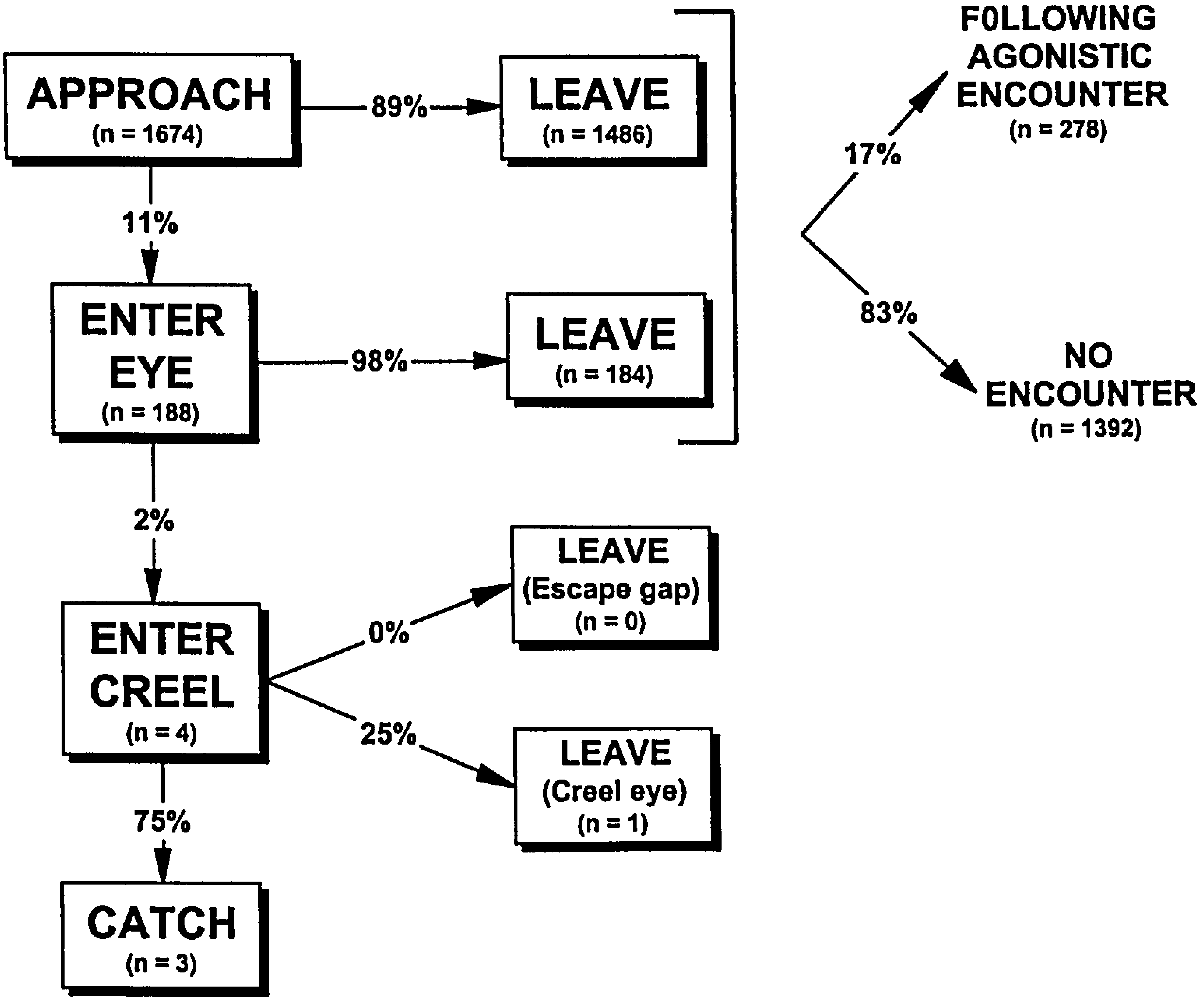

Figure 4.15 - Summary of behavioural responses of *Nephrops* to a baited creel containing crabs, observed in aquarium experiments with a stocking density of 10 animals in a 7 × 3 m tank. From 4, 48-h experiments.

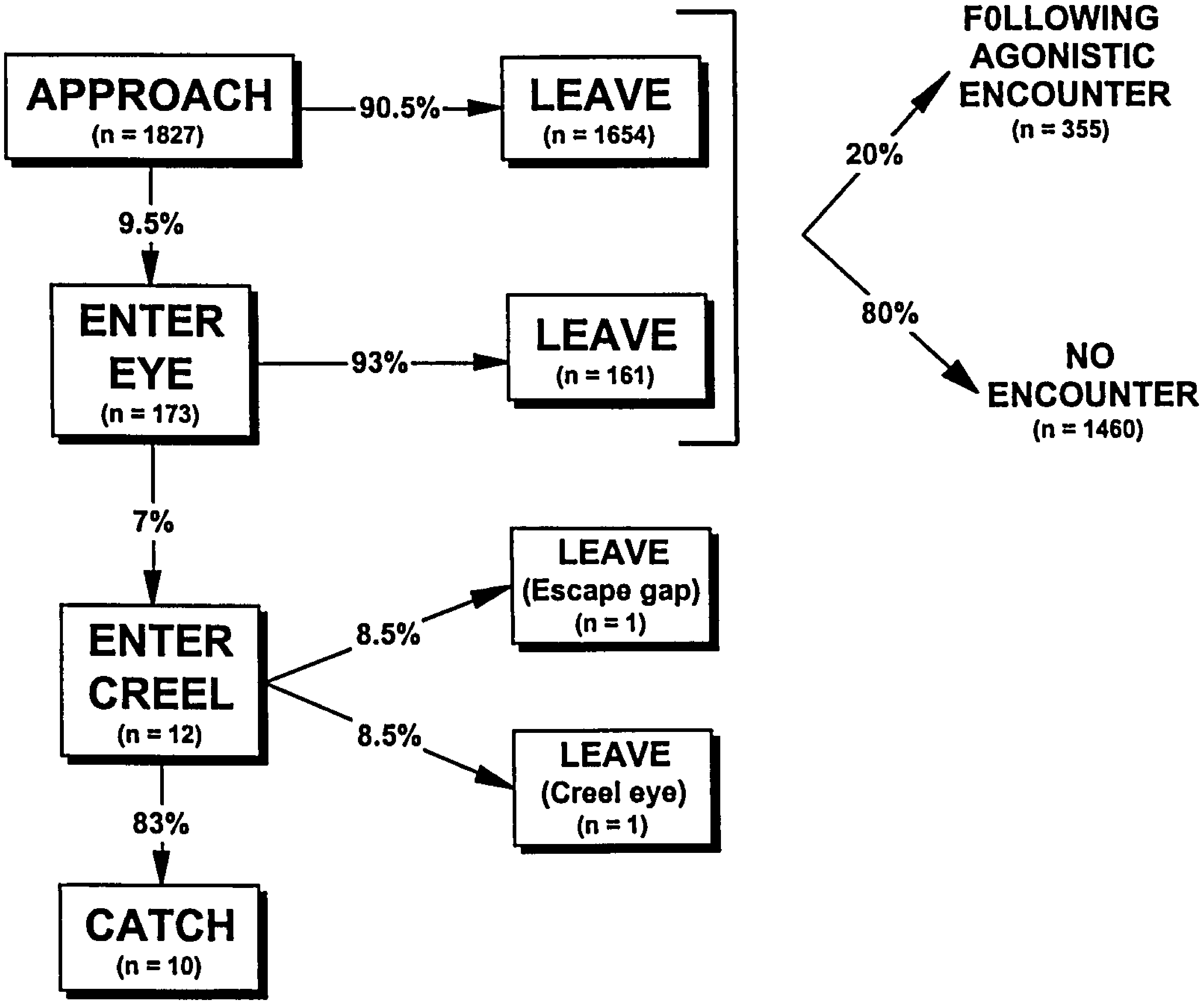

Figure 4.16 - Summary of behavioural responses of *Nephrops* to a baited creel, observed in aquarium experiments with a stocking density of 10 animals in a 7 × 3 m tank. From 4, 48-h experiments.

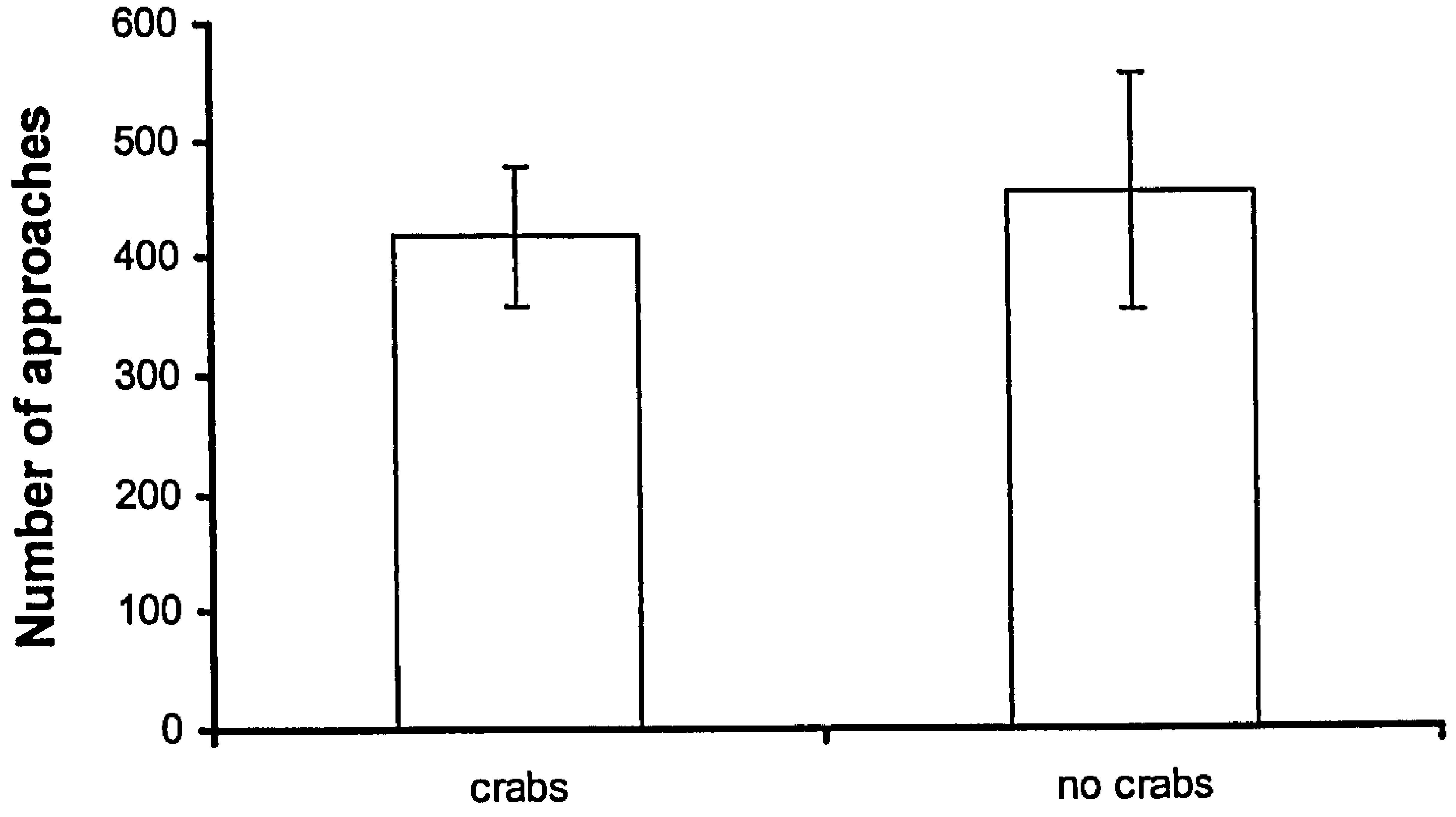

Figure 4.17 – Number of approaches to creel for both treatments during 48-h period in aquaria (4 replicates from each treatment, mean ± Standard Deviation (SD)).

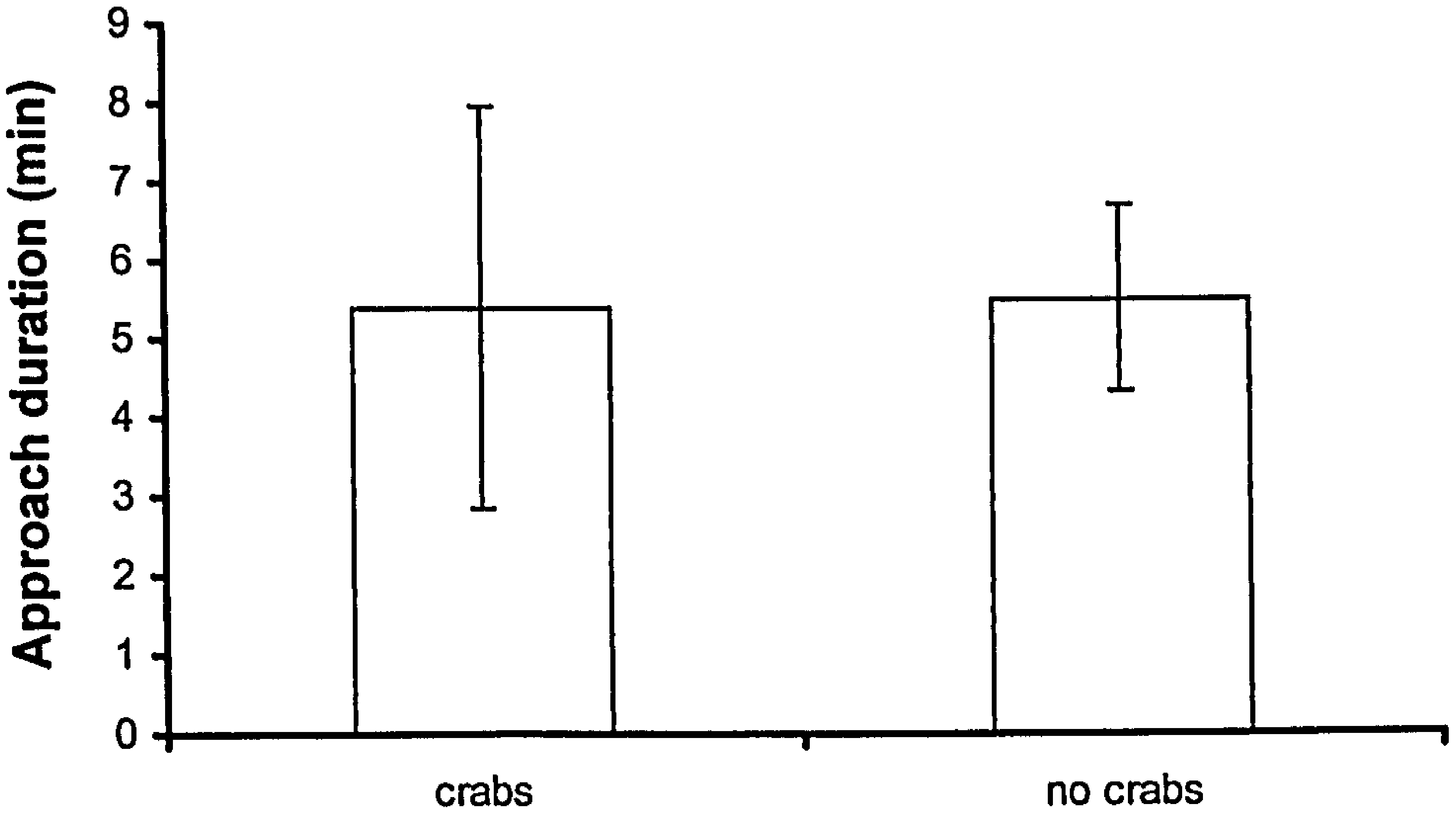

Figure 4.18 – Average approach duration for both treatments during 48-h period in aquaria (4 replicates from each treatment, mean ± SD).

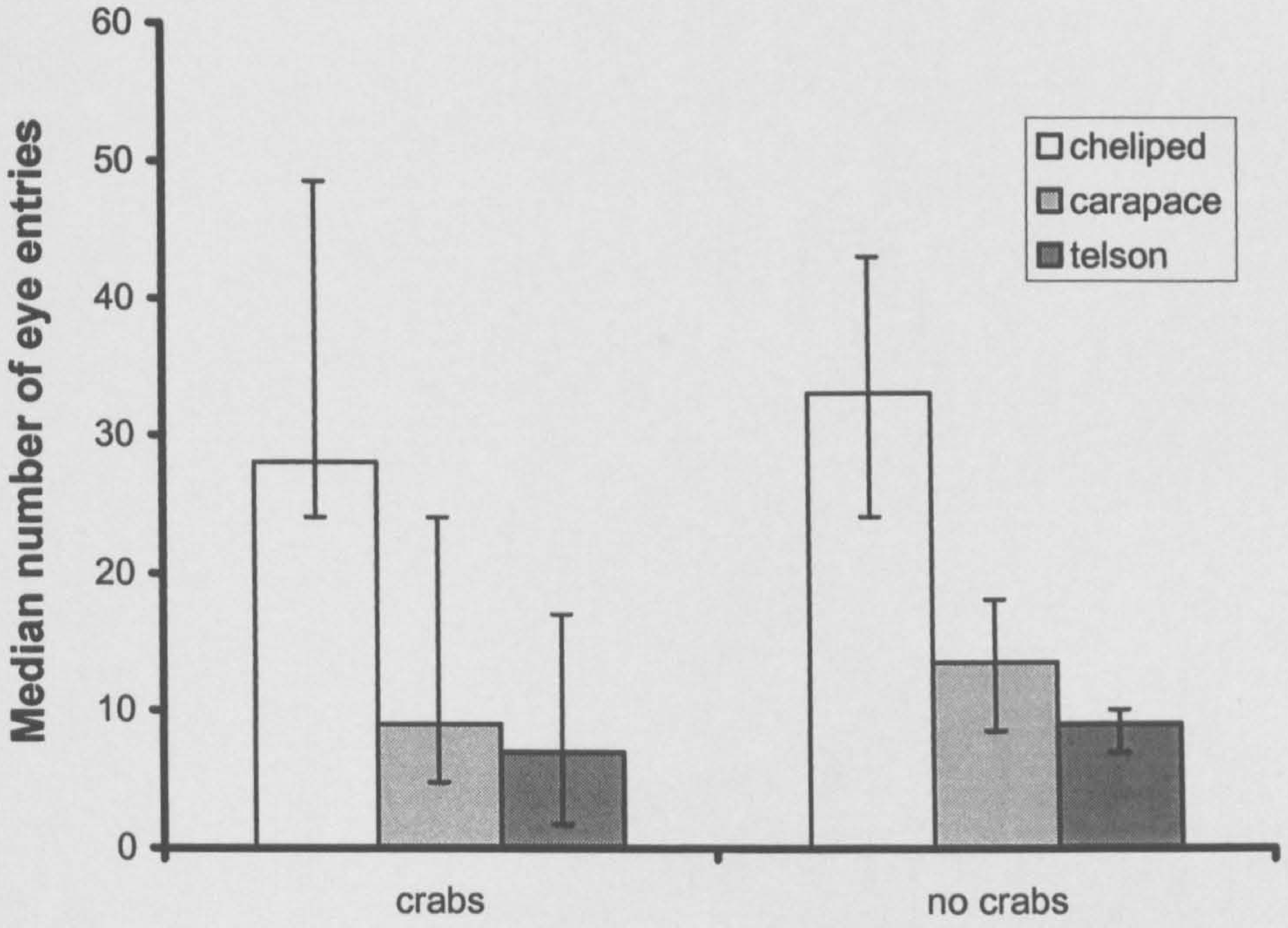

Figure 4.19 – Number and degree of *Nephrops* entry into creel eye for both treatments during 48-h period in aquaria (4 replicates from each treatment, median ± Inter Quartile Range (IQR)).

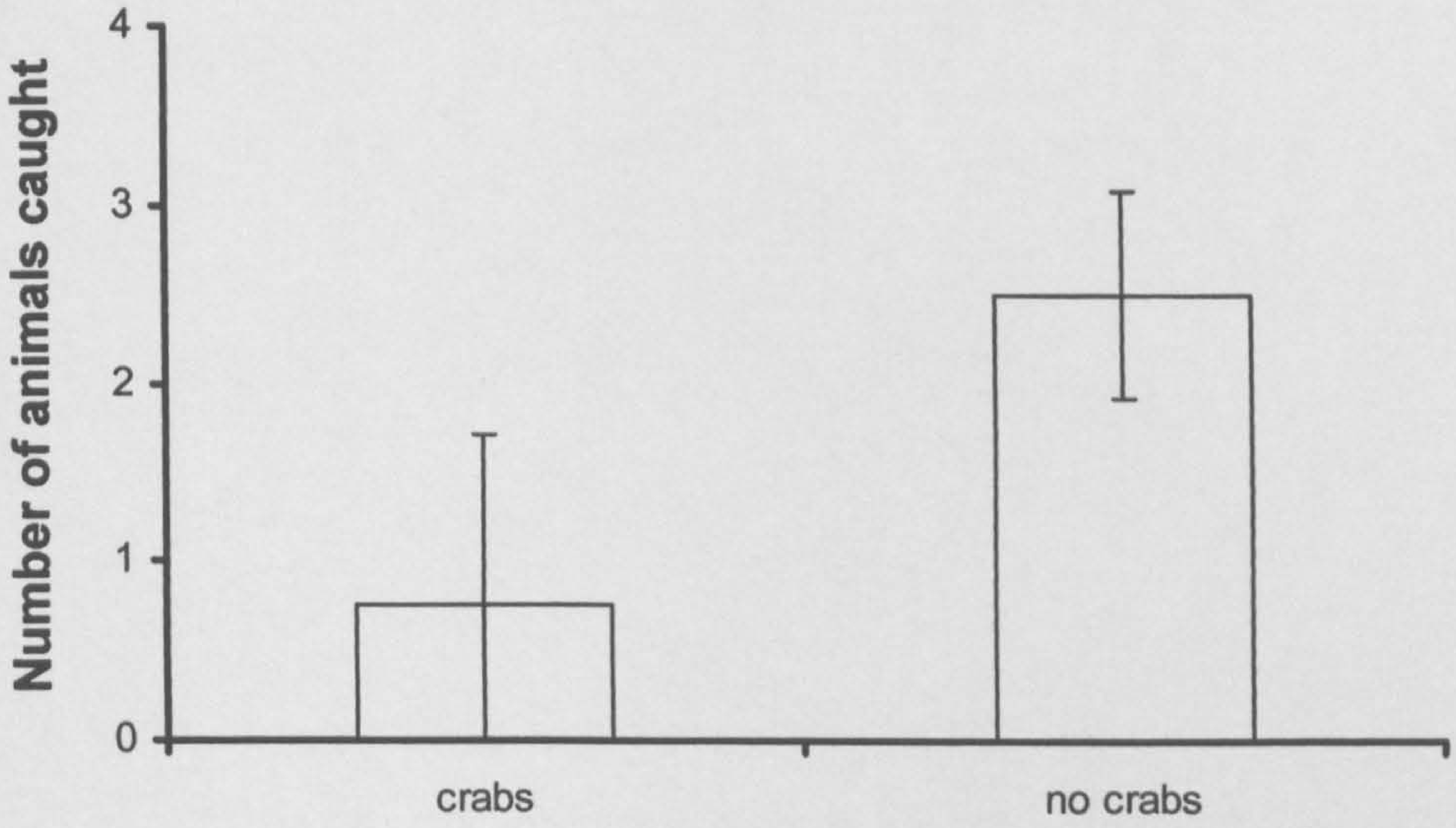

Figure 4.20 – Number of *Nephrops* caught in both treatments during 48-h period in aquaria (4 replicates from each treatment, mean ± SD).

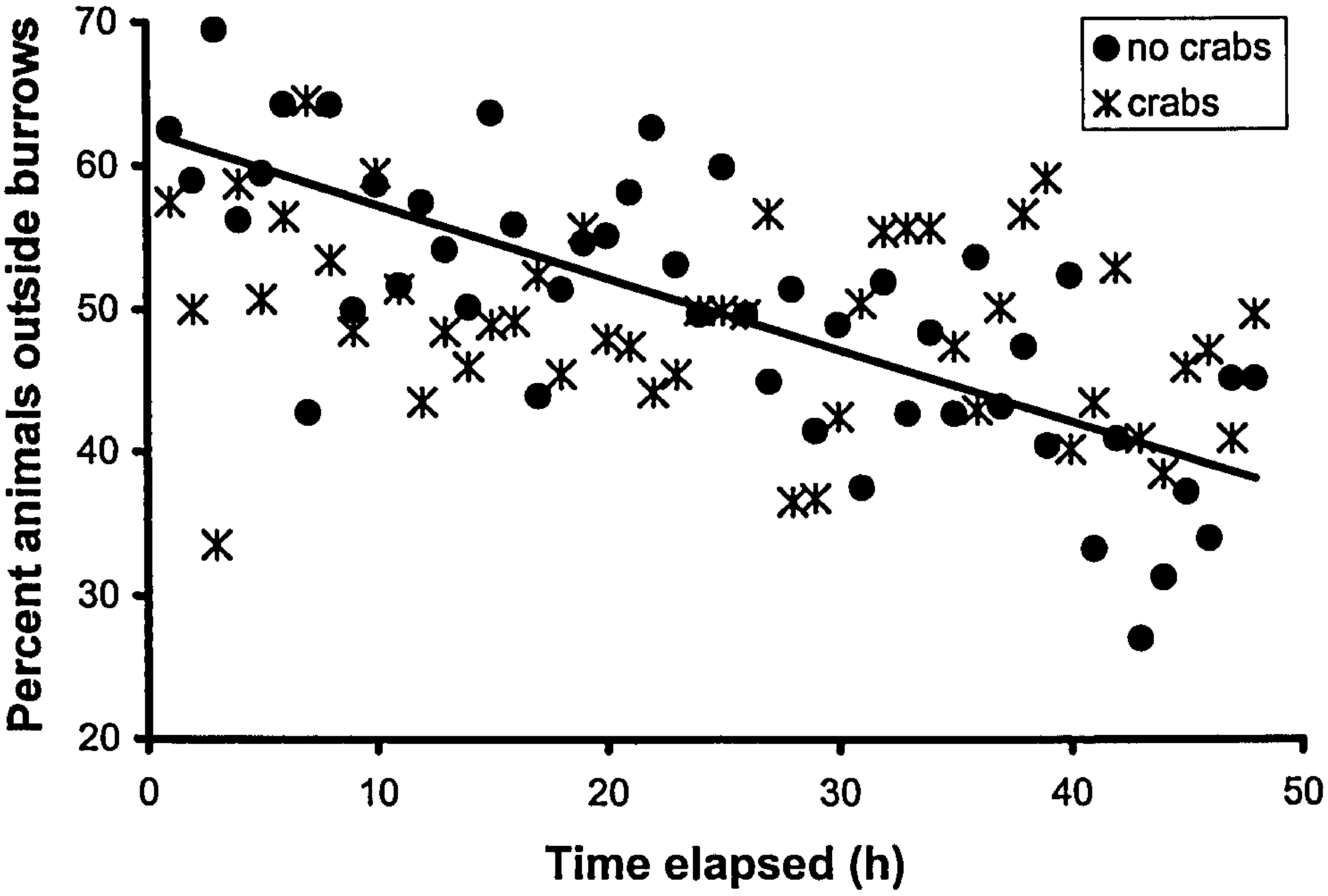

Figure 4.21 – Percentage of *Nephrops* outside burrows in the treatments with the creel containing crabs and where no crabs were present during a 48-h period in aquaria. Regression line displayed (no crabs, unbroken line).

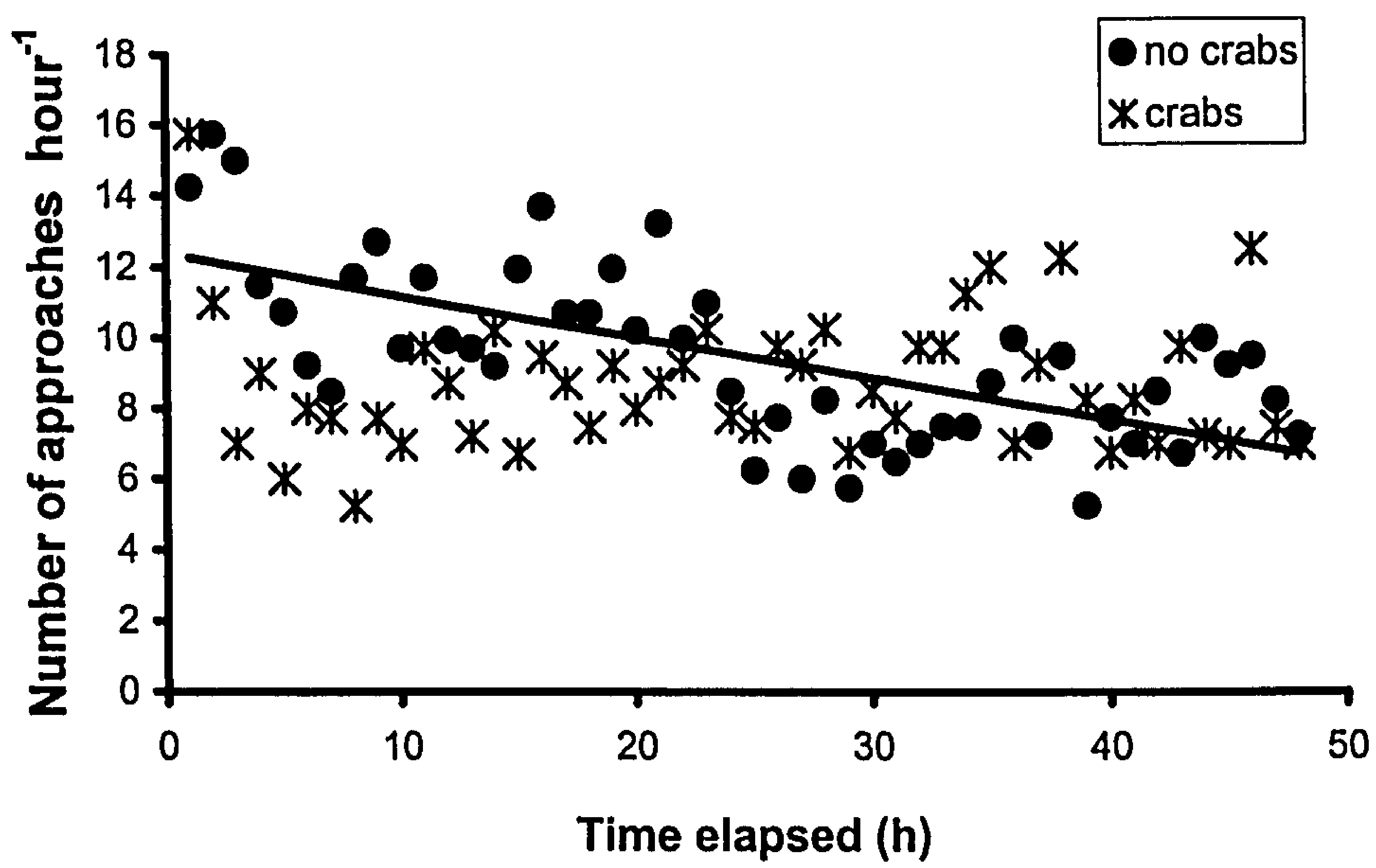

Figure 4.22 – Number of *Nephrops* approaches to a creel in the treatments with the creel containing crabs and where no crabs were present during a 48-h period in aquaria. Regression line displayed (no crabs, unbroken line).

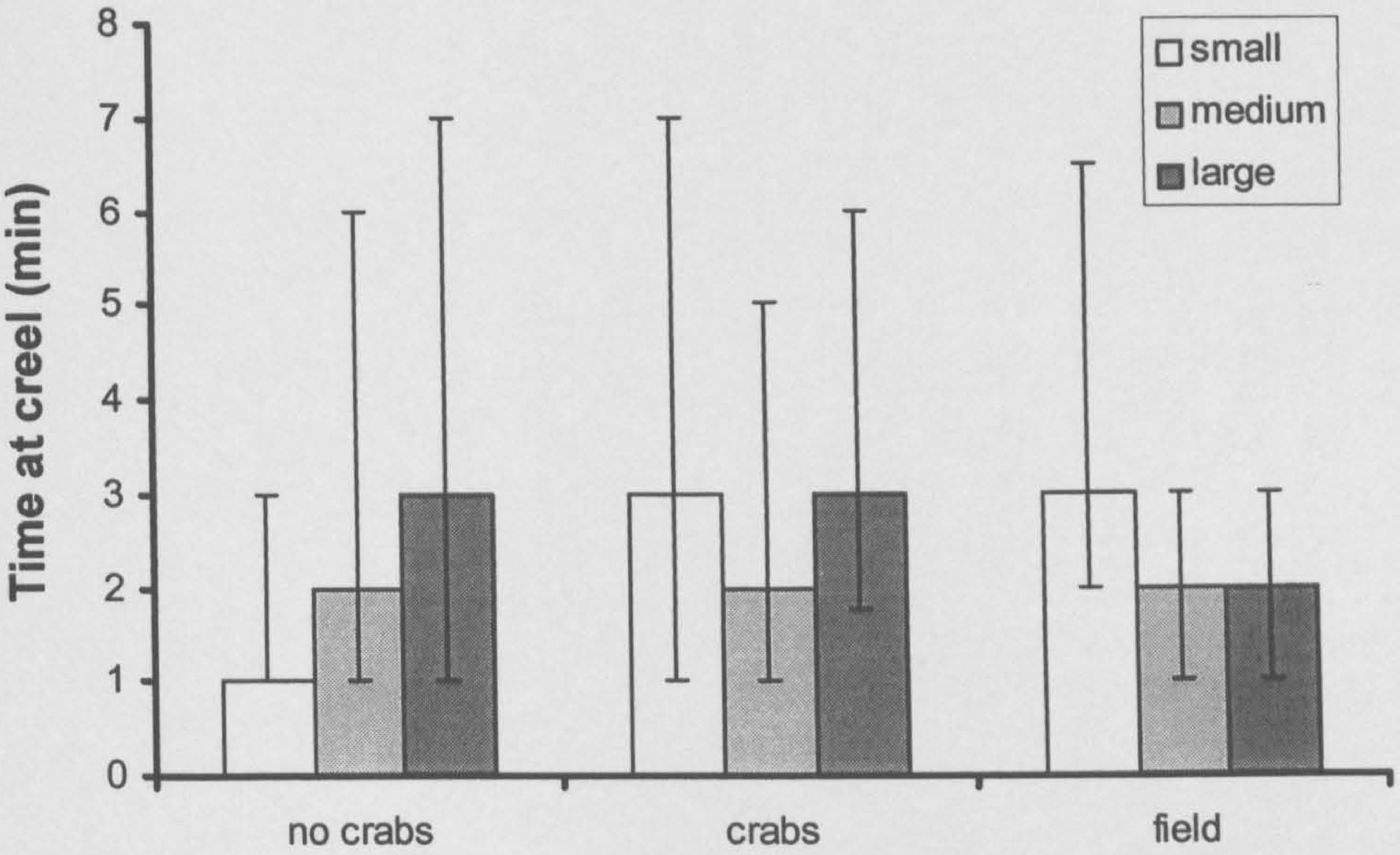

Figure 4.23 – Median duration at creel for different sized *Nephrops* during a 48-h period in the field and in aquaria where crabs were present or absent (median ± IQR).

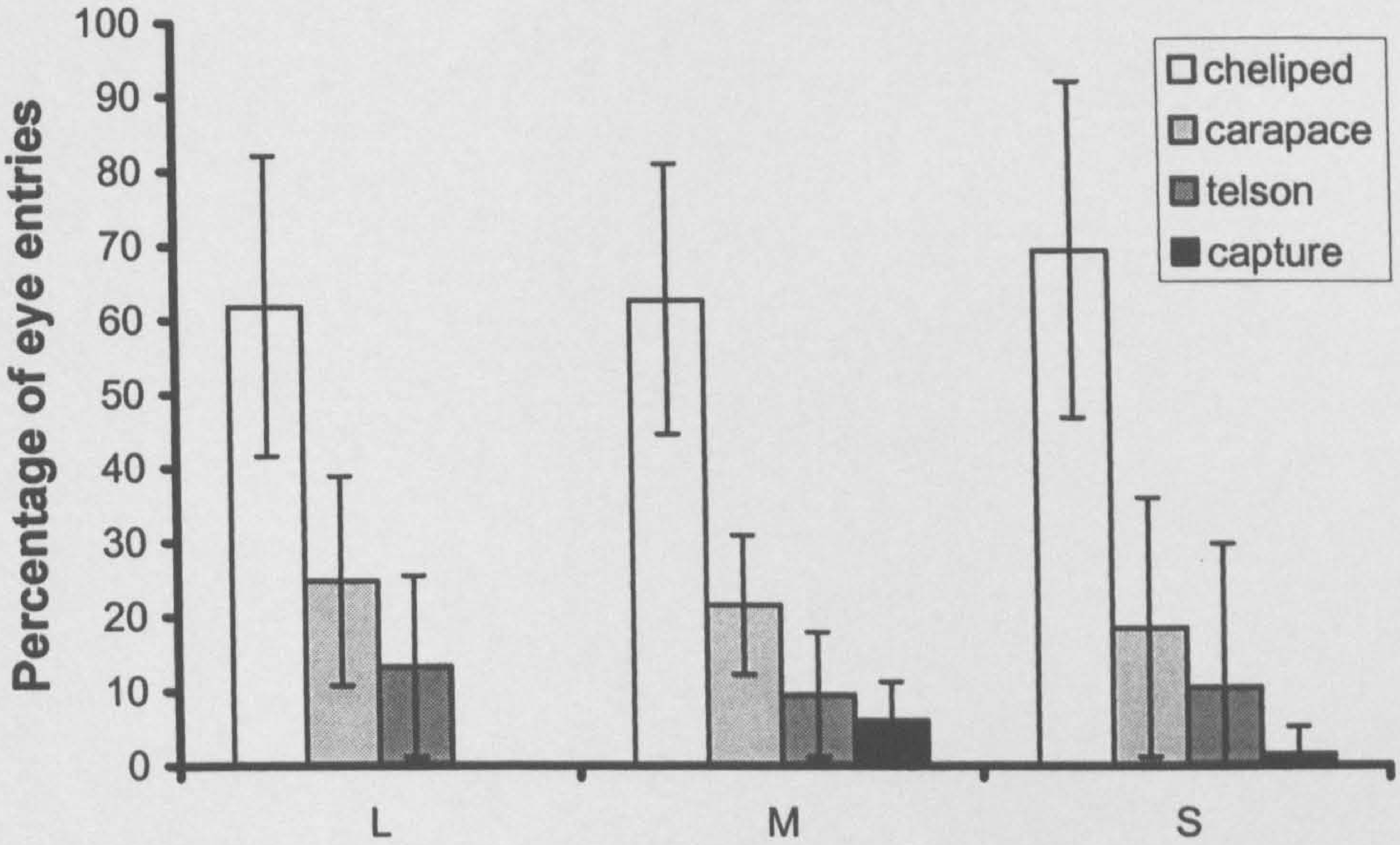

Figure 4.24 – Percentage of eye entries and degree of *Nephrops* entry into creel eye for large medium and small Nephrops during 8 trials in aquaria (mean ± 95% Confidence Limits).

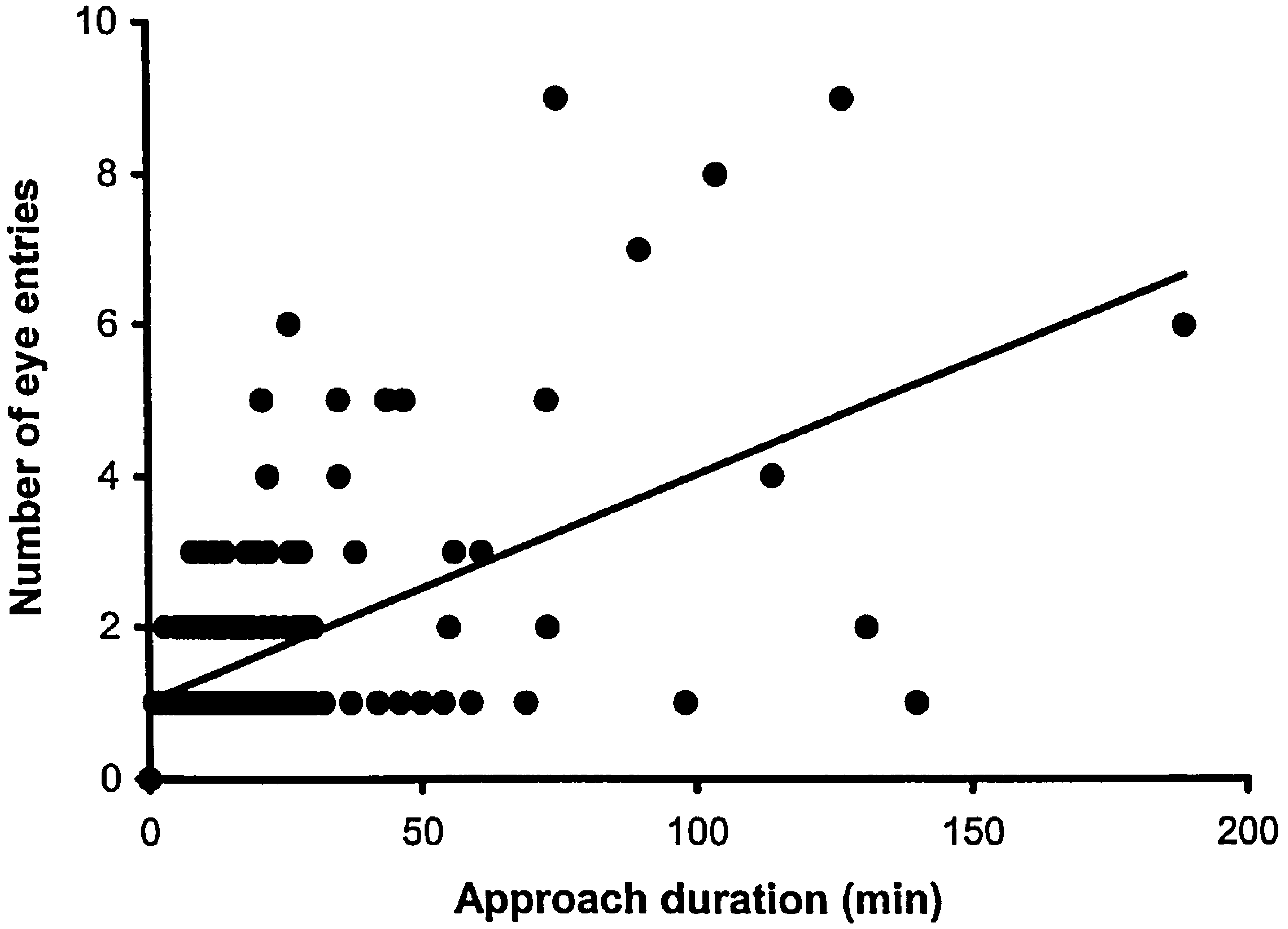

Figure 4.25 – The relationship between time at creel and the number of creel eye entries by individual *Nephrops* in aquaria. Regression line displayed.

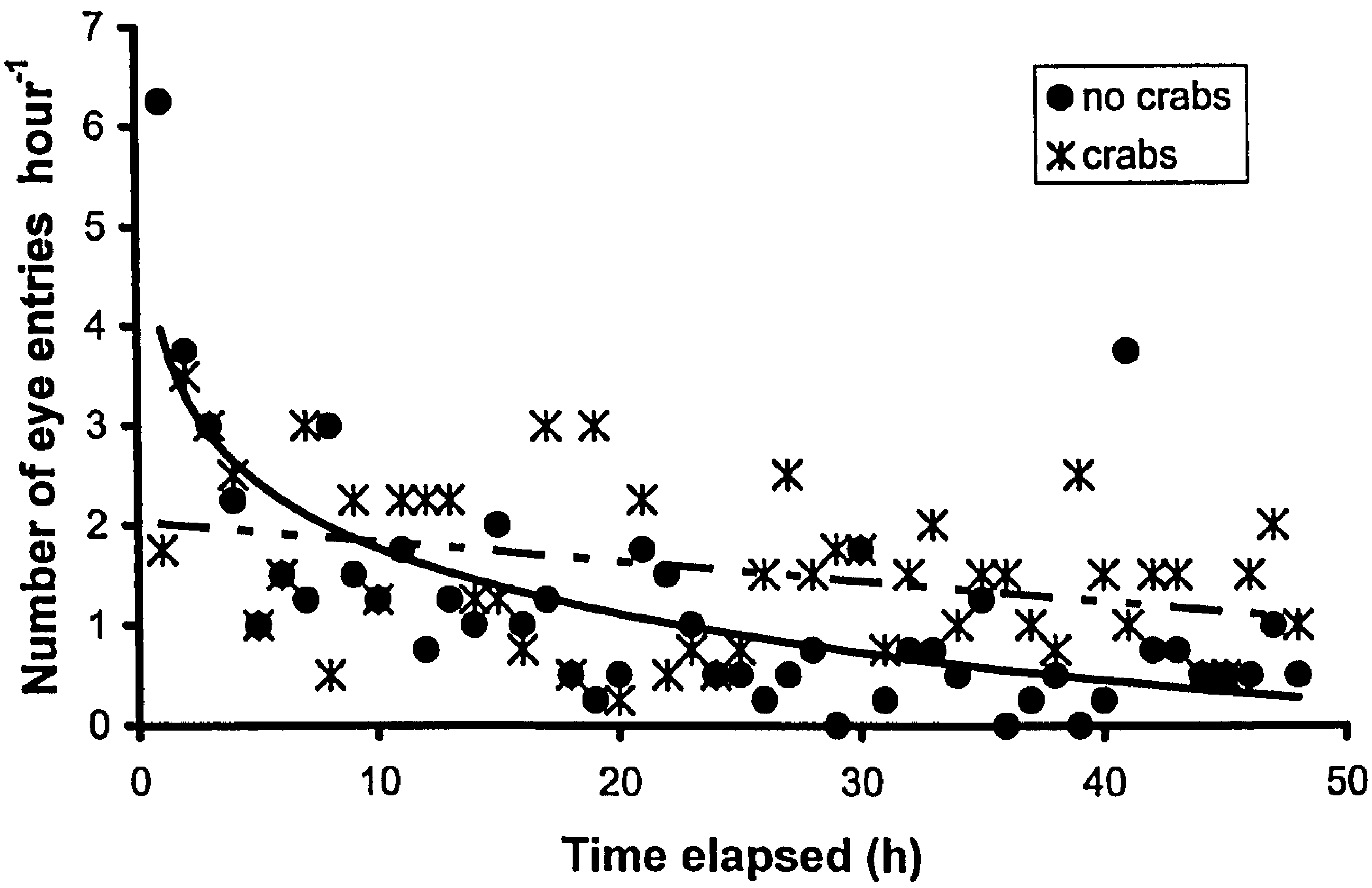

Figure 4.26 – Number of *Nephrops* entering the creel eye, in the treatments with the creel containing crabs and where no crabs were present during a 48-h period in aquaria. Regression lines displayed (no crabs, unbroken line; crabs, broken line).

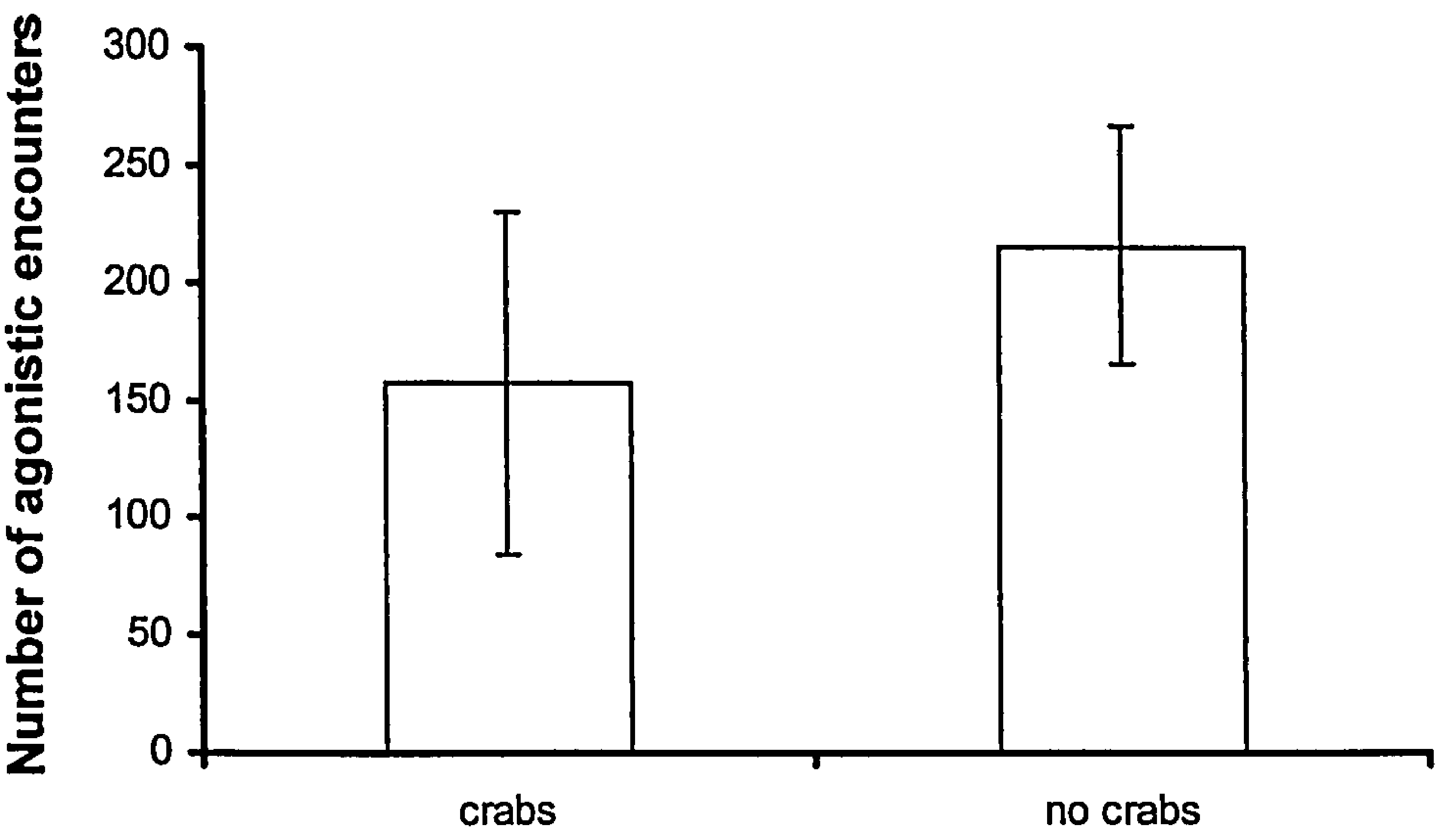

Figure 4.27 – Number of agonistic encounters between *Nephrops* in both treatments during 48-h period in aquaria (4 replicates from each treatment, mean ± SD).

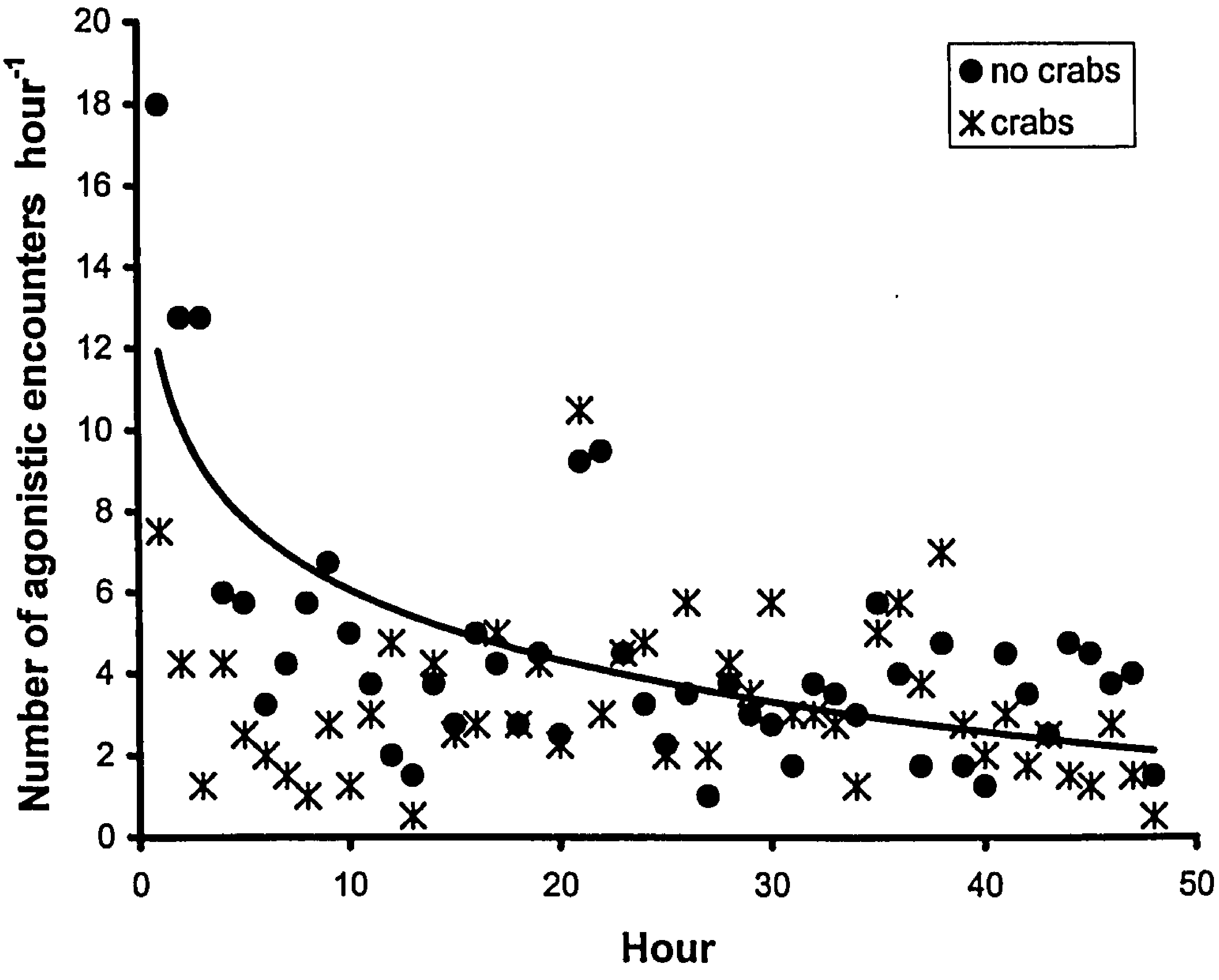

Figure 4.28 – Number of agonistic interactions between *Nephrops* in the treatments with the creel containing crabs and where no crabs were present during a 48-h period in aquaria. Regression line displayed (no crabs, unbroken line).

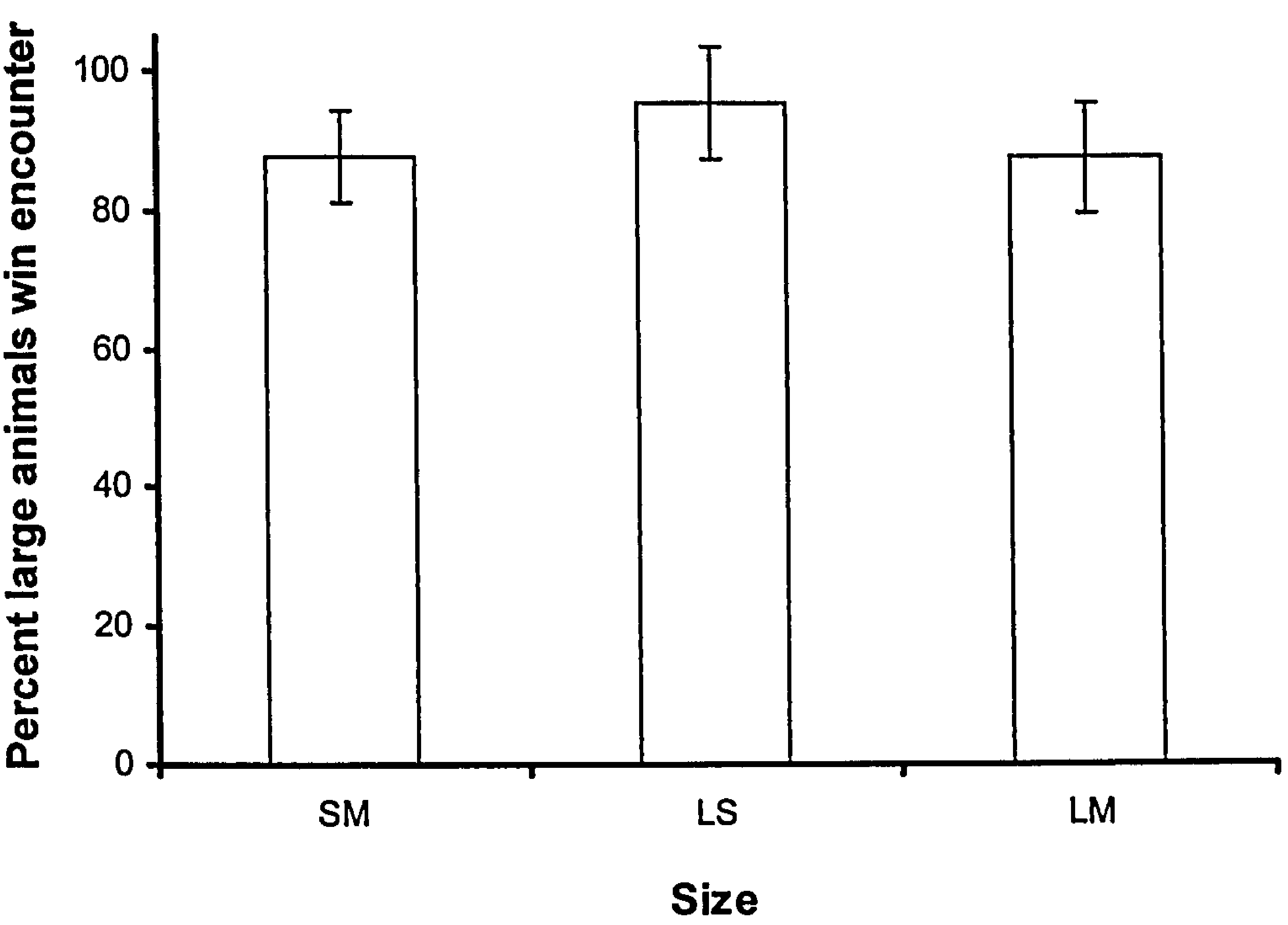

Figure 4.29 – Percentage of larger *Nephrops* winning encounter in relation to size of animals in bout in aquaria, where: SM = small *Nephrops* vs. medium *Nephrops*; LS = large *Nephrops* vs. small *Nephrops*; LM = large *Nephrops* vs. medium *Nephrops* (both treatments (crabs/no crabs) combined, mean ± SD)

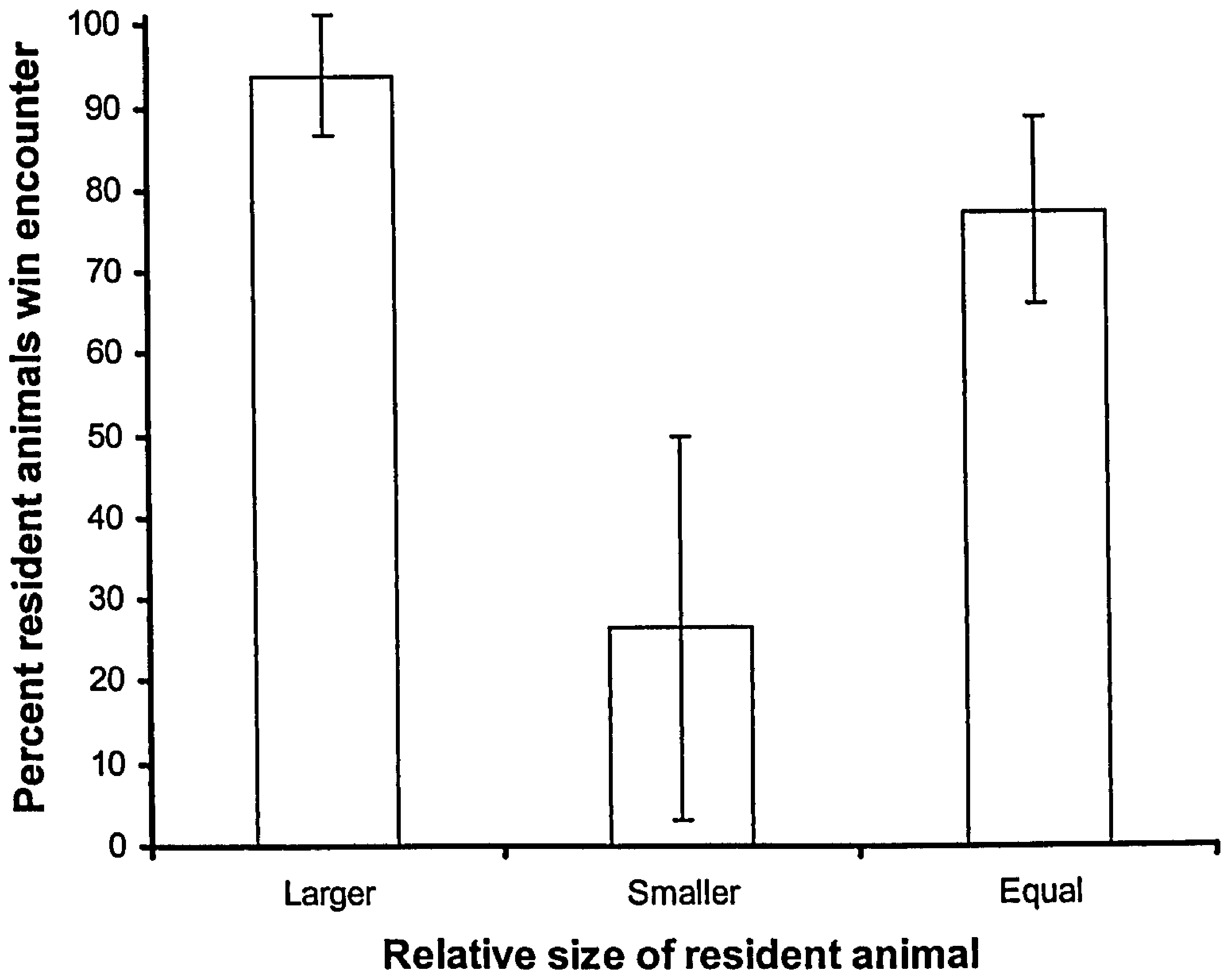

Figure 4.30 – Percentage of larger, smaller or equal sized *Nephrops* winning encounter when resident at the creel, from studies in aquaria (both treatments (crabs/no crabs) combined, mean ± SD).

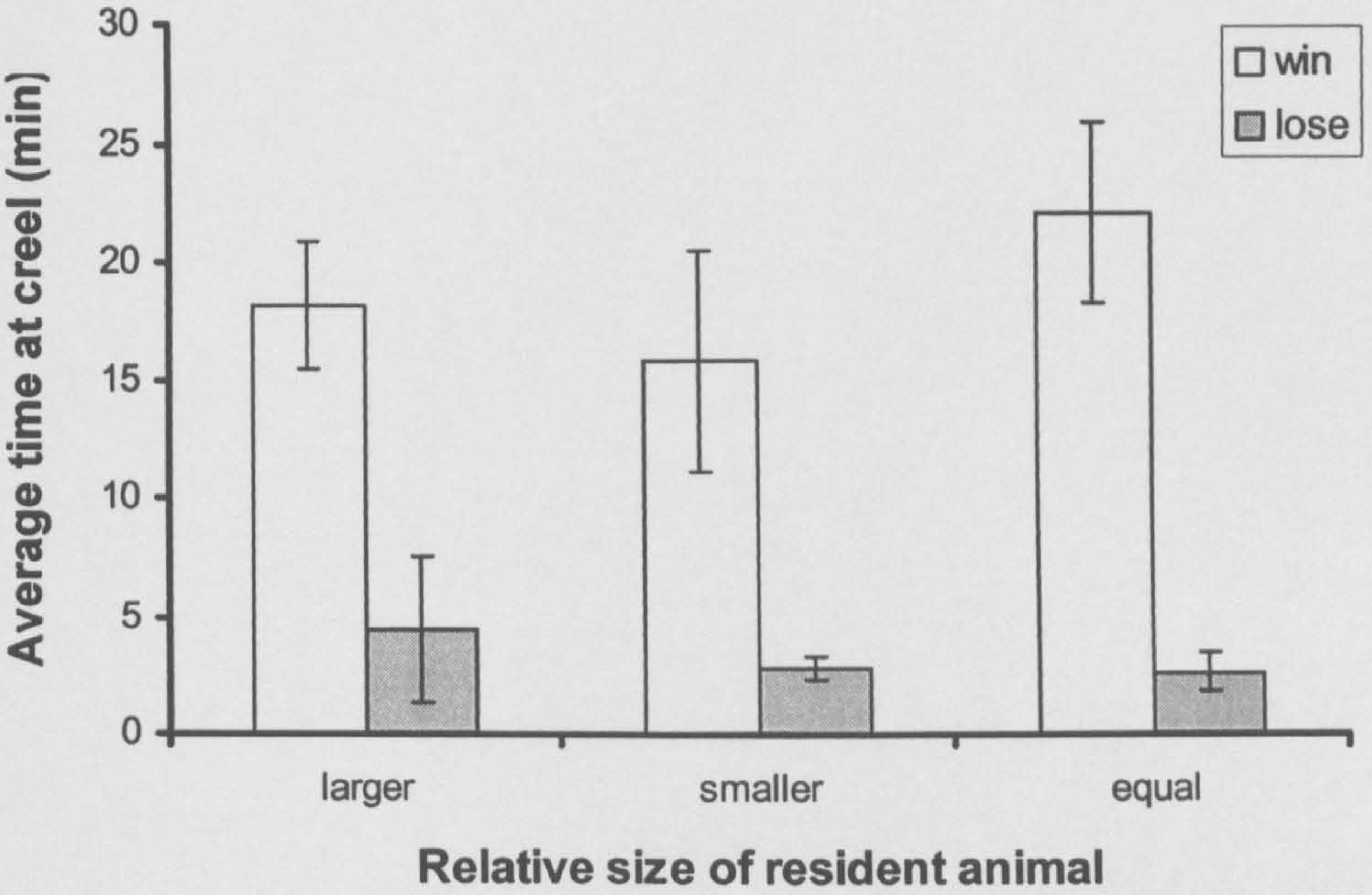

Figure 4.31 – Average duration of large, smaller or equal sized resident *Nephrops* at a creel for animals winning (n=215, 51 and 155, respectively) or losing (n=23, 159 and 34, respectively) an encounter, from studies in aquaria (both treatments (crabs/no crabs) combined, mean ± 95% Confidence Limits).

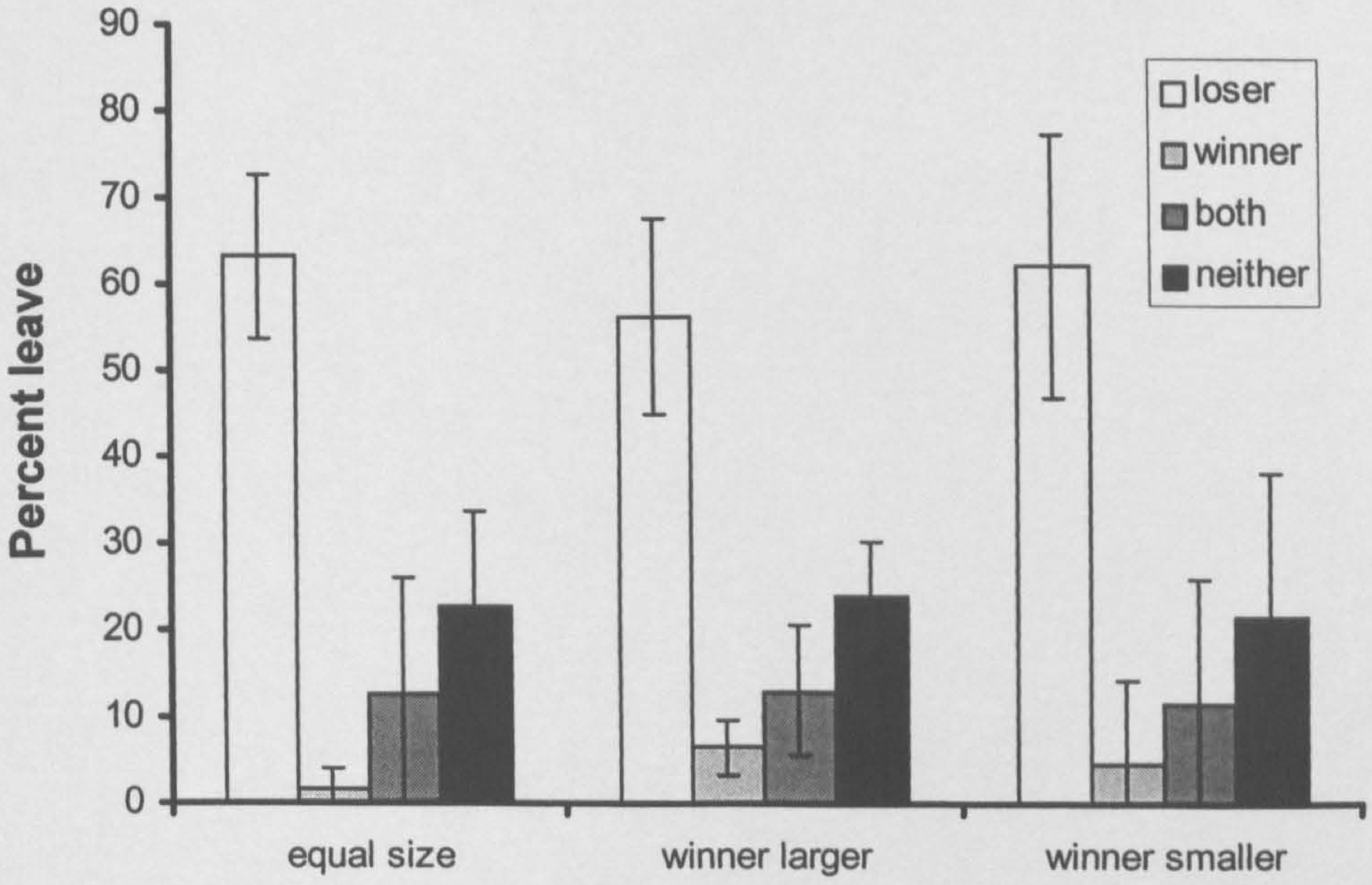

Figure 4.32 – Percentage of *Nephrops* leaving the area surrounding the creel in relation to the outcome of the bout, with relative size of winning *Nephrops* examined in aquaria (both treatments (crabs/no crabs) combined, mean ± 95% Confidence Limits).

Chapter 5

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5 PHYSICAL IMPACT OF *NEPHROPS* FISHING METHODS

5.1 Introduction

There is increasing awareness and understanding of the effect of fishing gear on the structure of benthic communities (De Groot, 1984; Jones, 1992; Kaiser & Spencer, 1996; Tuck *et al.*, 1998; Collie *et al.*, 2000; Ball *et al.*, 2000; Maynou & Sardà, 2001; Queirós *et al.*, 2006; Tillin *et al.*, 2006). Fishing with bottom-towed gear has been linked with disturbances to benthic communities leading to changes in species diversity, community structure, trophic structure and productivity (Thrush & Dayton, 2002). Tuck *et al.* (1998) observed benthic disturbances 18 months after experimental trawling of a previously unfished site and suggested that such disturbances can cause long-term changes in community structure. Further studies have shown significant decreases in numbers of species, biomass and species richness following experimental trawling (Kaiser & Spencer, 1996; Ball *et al.*, 2000; Cryer *et al.*, 2002). Such changes are apparent in the southern North Sea, where eighty years of increasingly intensive beam trawling appear to have completely re-structured the benthic ecosystem (Coggan *et al.* 2001). In comparison to fishing using bottom-towed gears, creels might be considered to be environmentally benign, since they are highly selective for the target species and have a low physical disturbance to the seabed (Jennings & Kaiser, 1998). This difference is highlighted in a study by Ziegler (2006) who compared of the impacts of the two methods of fishing for *Nephrops* (trawl and creel), calculating that an area of seabed covering 3900m^2 is kept in a permanently disturbed condition, per kg *Nephrops* landed in the trawl fishery compared to an area of only 62m^2 in the creel fishery.

European states are coming under increasing pressure through the EU Habitats Directive, as well as a number of non-statutory sources, to assess the ecological effects of all fishing activities. This pressure is likely to increase as fisheries management becomes more focused on an ecosystem-based approach, rather than managing single species. It is therefore essential to quantify the impacts of different fishing methods on the benthic environment, to underpin fishery management decisions.

Of particular interest are the effects of fishing gear on 'structural' species that extend above the seafloor, providing refuge and nursery habitats for a wide variety of fish and invertebrate species (Troffe *et al.*, 2005). A number of structural species have been shown to be susceptible to fishing gear (Hughes, 1998; Eno *et al.*, 2001; Wassenburg *et al.*, 2002; Roberts & Hirshfield, 2004; Troffe *et al.*, 2005). A recent comparison of the effects of shrimp beam trawls and prawn creels in British Columbia, Canada, indicated that creels caused more damage to colonies of the sea whip *Halipteris willemoesi* than trawling, by uprooting colonies (Troffe *et al.*, 2005). This study highlights the importance of monitoring the effects of both trawl and creel fisheries and shows that creel fisheries are not necessarily environmentally benign.

Deep mud habitats in sea lochs on the west coast of Scotland and in the northeast Irish Sea support the three British species of sea pen that occur in British waters: *Funiculina quadrangularis* (and its associated brittle star *Asteronyx loveni), Virgularia mirabilis* and *Pennatula phosphorea* (two other species may be rare occurrences, but not in NW Scotland). The biotopes in which these three species are found are characterized by soft mud in areas sheltered from wave action and strong currents (Hughes, 1998). Seven biotope types (Connor *et al.*, 2004) within the circalittoral mud habitat type support one or more of these species, principally 'SS.SMU.CFiMu.SpnMeg' (sea pens and burrowing megafauna in circalittoral fine mud) and 'SS.SMU.CFiMu.SpnMeg.Fun' (sea pens, including *F. quadrangularis* and burrowing megafauna in undisturbed circalittoral fine mud) (Connor *et al.*, 2004, Greathead, *et al.*, in press.).

Funiculina quadrangularis is considered to be reduced in numbers in these biotopes and has been classified as a nationally rare species (Jones *et al.*, 2000), with its associated brittle star *A. loveni* being of national conservation importance (Hughes, 1998). These biotopes also support large populations of *Nephrops*, for which there are important creel and trawl fisheries. The potential damage to *F. quadrangularis* caused by both fisheries is of concern, with the perceived impacts being greatest in the trawl fishery. This has led to proposed management measures, such as the exclusion of mobile fishing gear from defined areas, to protect examples of these biotopes (Hughes, 1998).

There have been several investigations into the fisheries effects on these sea pen species (Hoare & Wilson, 1976; Atkinson, 1989; Howson & Davies, 1991; Kinnear *et al.*, 1996; Tuck *et al.*, 1998; Hughes, 1998; Eno *et al.*, 2001; Greathead *et al.*, in press.). Kinnear *et al.* (1996) used observations by divers and underwater television to describe the effects of the creel fishery on the three sea pen species. This led to two main findings: firstly, sea pens bend as a result of the pressure wave generated ahead of the sinking creel, protecting the sea pen from the impact. Secondly, after being smothered or uprooted, sea pens are able to reinsert and right themselves. Kinnear *et al.* (1996) concluded that *Nephrops* creels had no lasting effect on the muddy substratum or its associated species, including sea pens.

Both *V. mirabilis* and *P. phosphorea* have the ability to withdraw into the sediment in response to disturbance (Hughes, 1998). This reaction may afford a degree of protection from damage by trawl gear, but there is conflicting evidence about the impacts of fishing on these species. Some studies have indicated little effect on *V. mirabilis* (Howson & Davies, 1991; Tuck *et al.*, 1998), while others have found lower population densities of either species in more-intensively trawled areas (Hoare & Wilson, 1976; Atkinson, 1989; Kinnear *et al.*, 1996). In contrast to *V. mirabilis* and *P. phosphorea*, *F. quadrangularis* does not have the ability to withdraw into the sediment when disturbed (Hughes, 1998). A comparison of the densities of these three species between areas of high and low intensity of trawling, found significantly higher densities of *F. quadrangularis* in the area of lower trawling intensity and no differences in the densities of the other two species between areas (Greathead *et al.*, in press.). A recent study recorded densities of *F. quadrangularis* and its associated brittle star *A. loveni* in areas where no fishing occurs and at sites within a creel fishing area. The number of *F. quadrangularis* associated with *A. loveni* was significantly greater within the no-fishing area than in the adjacent creel-fished area, suggesting a possible impact of creeling activities (Atkinson *et al.*, 2006).

The studies discussed above illustrate the lack of clear understanding of the impacts of both the *Nephrops* creel and trawl fishery on the benthic environment, with considerable uncertainty about the effects on the sea pen species *F. quadrangularis, P. phosphorea* and *V. mirabilis.* The uncertainty may arise because much of the information has been derived from studies not specifically addressing the question of fishery impact. There is therefore a need for a dedicated study of the effects of the different methods of fishing for *Nephrops* on these sea pen species and community structure as a whole.

The closure of the Inner Sound and Loch Torridon to mobile fishing gear in 2001 (The Inshore Fishing (Prohibition of Fishing and Fishing Methods) (Scotland) Amendment Order 2001), as well as the presence of a large fisheries exclusion zone (BUTEC range) (Chapter 1; Section 1.3.5) within this area, provides a unique opportunity to study the effects of fishing on species abundance, in particular those species considered to be more vulnerable to fishing activities, such as sea pens.

To this end the present study aimed to:

1. Compare the population density of the three sea pen species and the brittle star *A. loveni* between zones of trawl and creel fishing and among areas of different environmental conditions.
2. Compare epibenthic community composition in zones of trawl and creel fishing and in areas of different environmental conditions.
3. Compare the incidence of damaged sea pens between the creel-only and trawled zones.
4. Assess the relationship between observed population density of sea pens and the brittle star *A. loveni* and the quantity of these species caught as bycatch on commercial creel vessels.

5.2 Methods

5.2.1 Population density estimates

5.2.1.1 Study sites

The study area was located in Loch Torridon, the Inner Sound and North Minch areas in Wester Ross, Scotland. The zoned fisheries policy in this area (Chapter 1; Section 1.3.5) allowed the comparison of the epibenthos in relation to fishing methods. In the northern Inner Sound, Loch Torridon and Loch Gairloch, *Nephrops* is fished using creels. The North Minch is fished almost exclusively by trawlers and the southern Inner Sound and Loch Kishorn are closed to trawling for six months of the year. Within the northern Inner Sound there are two areas for testing naval warfare systems: the British Underwater Test and Evaluation Centre (BUTEC) and the Rona Noise Range, a facility for measuring the acoustic signature of military vessels. The BUTEC range consists of an inner area, which is closed to all fishing, and an outer area within which fishing by static gear is allowed (The British Underwater Test and Evaluation Centre Byelaws 1984). The outer area is now incorporated within a larger mobile gear exclusion zone defined by the 2001 fishing order. Fishing with mobile gear is also prohibited within the Rona range. A number of biotopes can be found within this area, but the majority of sites surveyed were characterised by soft mud (SS.SMU.CfiMu.SpnMeg and SS.SMU.CfiMu.SpnMeg.Fun) supporting dense populations of megafaunal species such as those summarised by Connor *et al.* (2004).

5.2.1.2 Survey procedure

Towed underwater television (UWTV) surveys have been conducted by Fisheries Research Services (FRS), Aberdeen, from FRV *Clupea* in the North Minch since 1994 and in the Torridon area since 2000 to estimate the density of *Nephrops* burrows as part of a statutory stock assessment. A further UWTV survey focussing on sea pens was conducted by University Marine Biological Station Millport from RV *Aora* in the Inner Sound in February 2005 (Atkinson *et al.* 2006).

A colour television camera and lights mounted on a sledge (Shand & Priestley, 1999) were towed slowly (<1 knot) for 10 minutes astern of the research vessel (Figure 5.2). An umbilical cable attached to the towing warp carried power and the video signal, except that in more recent surveys, electrical cables were incorporated within the towing cable. The camera viewed a strip of seabed of known width and the sledge was fitted with an odometer with a 1-m circumference wheel, allowing for the distance travelled and therefore the area surveyed to be measured. The video signal was passed to a time and date generator, video recorder and display screen on board the vessel.

5.2.1.3 Laboratory analysis

In the present study, recordings from stations occupied during FRS surveys in 2000, 2002 and 2003 were examined. Analysis was undertaken using DVD player software, allowing for control of the speed of playback, facilitating accurate species counts. All distinguishable animals were identified to the lowest possible taxonomic level and the number of each species noted for the whole recording at each station was examined (Figure 5.3). Numbers of individuals were totalled and divided by the area of seabed viewed to obtain absolute densities at each site. Differences in quality of recordings meant it was necessary to identify some animals to higher taxonomic groups. The results from additional sites in the northern Inner Sound and BUTEC range surveyed by UMBSM (Atkinson *et al.*, 2006), were included in the analysis of the densities of *F. quadrangularis* and *A. loveni.*

Owing to the variability in environmental conditions among sites, it was necessary to select sites with similar environmental conditions from each fishing area (creel and trawl) (Figure 5.1). Sites with similar depths and hydrographic conditions (open sea areas) were compared, and sites where the bottom was visibly rocky were rejected (Figure 5.1).

A significant proportion of the data used in this section were extracted from the FRS UWTV records by an undergraduate student, Lauren Kirkland. However, all data were checked, the dataset was extended and edited, positional resolution was improved, and all analysis was redone in the present work.

5.2.2 Bycatch species analysis

Catches in the commercial creel fishery in the Loch Torridon area were sampled to estimate the incidence of the sea pen species *F. quadrangularis, P. phosphorea* and *V. mirabilis* and the brittle star *A. loveni* in the bycatch. The sampling methodology and sample sites are described in Chapter 6, Sections 6.2.1 & 6.2.2.

5.2.3 Statistical analysis

5.2.3.1 Sea pen densities

Sea pen densities were analysed in relation to area (Figure 5.1: Upper Loch Torridon, lower Loch Torridon, northern Inner Sound, southern Inner Sound, North Minch, Loch Gairloch), depth ranges (<50 m, 50–100 m, 100–150 m, >150 m) and fishing method (creeling or trawling). When comparing fishing methods, sites from the southern Inner Sound were excluded, since both fishing methods are used in this area. Areas open to all fishing methods were considered to be, in effect, trawl zones. Mean, standard deviation and number of samples of *F. quadrangularis, P. phosphorea, V. mirabilis* and *A. loveni* were calculated. The patchy distribution of these species meant the majority of data were not normally distributed, so differences between groups were tested with non-parametric (Kruskal-Wallis) tests. Where more than two levels of a factor were analysed (location and depth), pair-wise comparisons among groups were made with Dunn's test (Dunn, 1964). ANOVA was used in one instance to compare $\log_{10}$-transformed *F. quadrangularis* density between creeled and trawled areas from all stations. All univariate analysis was undertaken using MINITAB version 13 (Minitab Inc., State College, Pennsylvania).

5.2.3.2 Damaged sea pens

Proportions of different sea pen species damaged or dead at each site were calculated by dividing the number of damaged/dead sea pens by the total number of sea pens observed at each site (considering only sites where sea pens were present). Sea pens were considered damaged if they were no longer in an upright position or visibly broken. Differences in prevalence of damaged *F. quadrangularis* and *V. mirabilis* colonies between creeled and trawled areas, were tested with one-way ANOVA. Both data sets were arcsine-transformed and the *V. mirabilis* data were further $\log_{10}$-transformed.

5.2.3.3 Species diversity

Simpson's index of species diversity, D, was calculated for each station:

$$D = 1 - \sum(p_i)^2$$

Where p_i = the proportion of individuals of species i

Differences in species diversity in relation to fishing method, location and depth were tested with a one-way ANOVA on cube root–transformed diversity indices.

5.2.3.4 Community composition analysis

Two-Way Indicator Species Analysis (TWINSPAN: Hill, 1979) was used to classify the stations in terms of the species composition. Five pseudo-species cuts (species density) of: 0.00, 0.01, 0.02, 0.05 and 0.10 m^{-2} were used.

Multivariate differences in species composition in relation to area, depth range and fishing zone were analysed with analysis of similarities (ANOSIM), using Bray-Curtis similarity indices calculated from fourth-root-transformed data and the maximum permutations of 999 (Clarke & Warwick, 1994). Multiple pair-wise comparisons following ANOSIM were made with a Bonferonni adjustment, dividing the normal significance level (0.05) by the total number of pair-wise comparisons, to protect against Type I error. Multidimensional Scaling (MDS) ordination was carried out to give a visual representation of the similarities among stations. The contribution of individual species to the Bray-Curtis similarity was analysed with SIMPER (Clarke & Warwick, 1994).

5.2.3.5 Bycatch species

The mean and standard deviation of the number of *F. quadrangularis, P. phosphorea, V. mirabilis* and *A. loveni* from each fleet of creels was calculated. Catch rate of these species is expressed as catch per unit effort (CPUE, number 100-creels^{-1}). The distribution of CPUE of these species was not normally distributed, so differences in relation to location, sediment type and depth were tested with Kruskal-Wallis tests, followed by multiple comparisons with Dunn's test.

5.3 Results

From the 142 FRS stations analysed, 29 species or taxonomic groups were identified. It is likely that the number of species observed on the tapes was in reality greater; however, the image quality only allowed for identification to a higher taxonomic group in several species. The most common species observed included all three species of sea pen *F. quadrangularis, P. phosphorea* and *V. mirabilis.* The common crustaceans were *Nephrops, Munida rugosa* and hermit crabs (F. Paguridae), mostly *Pagurus bernhardus.* Large numbers of feather stars (F. Antedonidae, not possible to differentiate between species at resolution of camera) and small fish from the family Gadidae were also observed (Table 5.1).

5.3.1 Sea pen densities

5.3.1.1 All sites

Differences in mean density of all three sea pen species and the brittle star *A. loveni* among areas are illustrated in Figures 5.4 and 5.5. The mean density of each species within areas had large standard deviations, probably reflecting patchy distributions (Table 5.2).

The density of *F. quadrangularis* differed significantly among locations (Table 5.2). This species was observed in all areas except Loch Gairloch. Highest densities of *F. quadrangularis* were observed in the northern Inner Sound, followed by lower Loch Torridon and Upper Loch Torridon (Figure 5.4). These three areas account for the majority of sites analysed where only creel fishing occurred and therefore significantly more *F. quadrangularis* were found within creeled areas than trawled areas (Figure 5.5). This species was also observed in the North Minch and the southern Inner Sound, but at lower densities (Figure 5.4). Pair-wise comparisons showed significant differences between Loch Gairloch to both the northern Inner Sound and the North Minch (Table 5.3). *Funiculina quadrangularis* occurred in all depth categories (Figure 5.5), but at significantly different densities, although no pair-wise comparisons were significant (Table 5.3).

The greatest density of *V mirabilis* was in Upper Loch Torridon, with small numbers observed in all other areas (Figure 5.4). The density of *V. mirabilis* varied significantly among areas, but no pair-wise comparisons were significant (Table 5.3). *Virgularia mirabilis* occurred at greater density at sites where only creel fishing occurred and no significant differences among depth ranges were detected (Table 5.3).

Pennatula phosphorea was found in highest density in Loch Gairloch, followed by Upper Loch Torridon, and occurred in very small numbers or was absent in other areas (Figure 5.4). All sites in Loch Gairloch and Upper Loch Torridon are in zones where only creel fishing occurs and the majority of these sites are shallow (0–100 m). Accordingly, significantly higher densities of *P. phosphorea* were observed in creel zones and in the shallowest depth categories (Table 5.3).

Asteronyx loveni was observed in small numbers at all sites except Loch Gairloch, where none was observed. The highest densities were observed in the northern Inner Sound, followed by the southern Inner Sound (Figure 5.4). Significant differences were found between the northern Inner Sound, lower Loch Torridon and Loch Gairloch (Table 5.3). Significantly higher densities were observed in the creel zone, a reflection of the majority of animals being found in the northern Inner Sound. No significant variation was noted with depth (Table 5.3).

5.3.1.2 Selected sites — fishing method

Differences in mean density of the three sea pen species and the brittle star *A. loveni* are illustrated in Figure 5.6. Low mean densities of *P. phosphorea* and similar mean densities of *V. mirabilis* were observed in both types of fishing areas. Larger mean densities of both *A. loveni* and *F. quadrangularis* were observed in the creel zone. The large standard deviation in Table 5.4 reflects the patchy distribution of these species. The numbers of both *A. loveni* and *F. quadrangularis* were significantly greater in the creel zone than the trawl zone (Table 5.5). No significant differences were found among fishing zones for *P. phosphorea* and *V. mirabilis.*

5.3.1.3 Selected sites including no fishing area — *F. quadrangularis* and *A. loveni* only

Data from the BUTEC range were analysed separately, as only densities for *F. quadrangularis* and *A. loveni* were available. The density of *F. quadrangularis* and *A. loveni* differed significantly among zones (Figure 5.7 & Table 5.7). The density of *F. quadrangularis* was highest in the creel zone and this value was significantly greater than the density in the trawl zone, but not significantly different from the density in the no-fishing zone (Table 5.7). In contrast, the density of *A. loveni* was highest in the no-fishing zone, this value being significantly different from the trawl zone, but not significantly different from that in the creel zone.

5.3.2 Damaged sea pens

Only 0.01% of *P. phosphorea* in the creel zone and none in the trawl zone were observed to be damaged. For both *F. quadrangularis* and *V. mirabilis*, the prevalence of damage (Figure 5.8) was significantly higher in the creel zone (10.7% and 18.6%, respectively) than in the trawl zone (5.5% and 5.4%, respectively) (Table 5.9).

5.3.3 Species diversity

The values of Simpson's diversity index ranged between 0.1041 and 0.8811 and there were no significant differences in species diversity among areas ($F_{5,136}$=1.45, P=0.212), depth ranges ($F_{3,138}$=0.97, P=0.410) or fishing zones ($F_{1,140}$=0.10, P=0.749).

5.3.4 Community composition analysis

5.3.4.1 All sites

The first division of the TWINSPAN analysis (eigenvalue = 0.347, good division; Table 5.10) separated the sites into a large group (group 0) containing sites from the North Minch, Northern Inner Sound and Loch Torridon, and a smaller group (group 1) with sites in lower Loch Torridon, Upper Loch Torridon, Loch Gairloch and the northern Inner Sound (Table 5.11)

The indictor species identified by TWINSPAN (most important species in determining the groups) were *F. quadrangularis* in group 0 and *Porania pulvillus, Asterias rubens, Cancer pagurus* and *M. rugosa* in group 1 (Table 5.12).

The second split of group 0, into groups 00 and 01 (eigenvalue = 0.149, poor division; Table 5.10) produced a large group comprising almost exclusively North Minch sites, which are deep trawled sites (group 00). The other smaller group (group 01) contains North Minch sites, sites from Loch Torridon and one site from the northern Inner Sound, representing a mixture of fishing methods and depths (Table 5.11).

Carcinus maenas and two groups of fish species (Family Gadidae and Order Pleuronectiformes) were TWINSPAN indicator species for group 00 and three sea pen species (*F. quadrangularis*, *P. phosphorea* and *V. mirabilis*) and *P. pulvillus* were indicator species for group 01 (Table 5.12).

The second split of group 1 into groups 10 and 11 (eigenvalue = 0.243, good division; Table 5.10) produced two small groups with little difference in fishing methods, area or depth range (Table 5.11). This is likely to be due to group 1 being small to begin with, making any further division unnecessary.

MDS plots of species density are shown in Figure 5.9 with all three factors being displayed and the boundaries for TWINSPAN groups 00, 01 and 1 overlaid. Stations with similar epibenthic communities occur closer together on the plot. Separations between the trawl and creel zones are apparent in Figure 5.9a, with TWINSPAN group 00 containing the densely clustered trawled sites. All trawl sites were located in the North Minch, which is mostly a deep-water habitat. The creel sites, the majority of which are in TWINSPAN groups 01 and 1, are more scattered than the trawl sites (Figure 5.9a). This scatter is likely to be due to the different depths and areas of the creel sites supporting a wider variety of species assemblages (Figure 5.9b,c). The difference in species composition is reflected by the large distance of these sites from the deeper, trawled North Minch sites (Figure 5.9b).

These subjective assessments of the MDS plots were confirmed by ANOSIM, which indicated significant differences in the community composition between the North Minch and all three loch environments (Upper Loch Torridon, lower Loch Torridon and

Loch Gairloch), as well as significant differences between lower Loch Torridon and Loch Gairloch (Table 5.13). In addition, there was a significant difference in the community composition between zones fished by trawling to those fished by creeling only. Significant differences were observed among all depth ranges except >150 m and 101–150 m (Table 5.13).

The importance of indicator species in producing the TWINSPAN groups is illustrated in Figure 5.10a–g). The relative density of indicator species identified by TWINSPAN analysis has been plotted on MDS plots, showing the location of each site, with simplified TWINSPAN groups illustrated. The largest numbers of Pleuronectiformes and Gadidae occurred in group 00, characterising the trawled zone in the North Minch (Figure 5.10a,b). *Funiculina quadrangularis*, *V. mirabilis* and *P. phosphorea* occur in the largest numbers in group 01 (Figure 5.10c,d,e). These sites comprise largely creeled sites from Loch Torridon and the northern Inner Sound. *Munida rugosa* and *P. pulvillus*, occur in the largest numbers in group 1 (Figure 5.10f,g). This group contains stations almost exclusively from loch environments, many of which are shallow sites.

The average dissimilarity was high between all areas, depth ranges and fishing zones, with the highest dissimilarity between Loch Gairloch and the North Minch and <50 and >150 m (Table 5.14). The importance of the three sea pen species in the dissimilarities between groups are similar to those described by the univariate analyses of species abundances, with the high densities of *F. quadrangularis* in Upper and lower Loch Torridon and northern Inner Sound accounting for a large percentage of dissimilarity to the other sites and therefore between creel and trawled areas (Table 5.14). High densities of *P. phosphorea* in Loch Gairloch account for a large percentage of the dissimilarity of this area to all other areas (Table 5.14). In addition to the sea pen species, the presence of high densities of gadoids in the North Minch and *M. rugosa* in the North Minch and southern Inner Sound are important in distinguishing these areas (Table 5.14).

The species identified by SIMPER as being the most important in determining the dissimilarity between sites (Table 5.14) are similar to those identified as indictor species by TWINSPAN (Table 5.12). The relative importance of these species in defining differences among areas, depth ranges and fishing zones is also similar.

5.3.4.2 Selected sites – fishing method

Analysis of all sites showed significant differences between the epibenthic species composition of creeled and trawled areas (Table 5.13). However, there were also differences among areas and depth ranges, suggesting that any effects of fishing method may be confounded with effects of environmental conditions. It was therefore important to compare species composition in areas where different fishing methods are used but that have similar environments. The same sites chosen for the univariate sea pen density analyses were selected.

The MDS indicates that, with respect to species composition, five of the six trawl sites are more similar to each other than to the creel-only sites, two creel sites differ from trawl sites, while one creel site and one trawl site are more similar to each other than they are to any of the other sites in either fishing zone (Figure 5.11).

Despite the apparent differences illustrated by the MDS, species composition did not differ significantly between creel and trawl sites (ANOSIM, R=0.29, *P*=0.11). Clearly, a larger sample size is needed to assess adequately differences in species composition between areas fished by different methods.

From these selected sites, the average dissimilarity between creel and trawled areas was high, with the higher densities of *F. quadrangularis* in the creel area accounting for 44% of the dissimilarity (Table 5.15).

5.3.5 Bycatch species analysis

The numbers of all three sea pen species and *A. loveni* taken in bycatch were very variable, as reflected in high standard deviations of CPUE (Table 5.16). Numbers of *F. quadrangularis* caught as bycatch were very low (Figure 5.12) compared with the density observed in each area (Figure 5.4). The CPUE of *F. quadrangularis* differed significantly among depth ranges, with pair-wise comparisons showing that CPUE was significantly higher at depths >150 m than at 50–100 m. CPUE was higher in areas where the bottom type was described as "soft" by fishermen (Table 5.17). The CPUE of *F. quadrangularis* did not vary significantly with location (Table 5.17).

Asteronyx loveni was caught in the highest numbers in the Inner Sound (Figure 5.12). Of particular interest is the proportion of *A. loveni* caught in comparison with its host *F. quadrangularis* (Figure 5.12). Higher densities of *F. quadrangularis* were observed than *A. loveni* in all areas (Figure 5.4). However, *A. loveni* was caught in similar numbers to *F. quadrangularis* in the bycatch, suggesting that *A. loveni* is more susceptible to capture by the creeling than its host, *F. quadrangularis*.

CPUE of both *V. mirabilis* and *P. phosphorea* was significantly higher in the two shallower loch environments than in the Inner Sound (Figure 5.12). This difference reflects the greater abundance of these species in the shallower loch environments (Figure 5.4). The mean CPUE of these species did not differ significantly between soft and hard ground (Table 5.17).

5.4 Discussion

From the towed UWTV surveys, the densities of all three sea pen species varied with permitted fishing zone, area and depth. It was important in this study to take account of environmental conditions while assessing the effects of fishing, since most of the 'trawl' and 'creel' sites were located in areas with different hydrographic and physical characteristics. The trawl sites examined were located in the North Minch, which is an area of open sea where much of the seabed is deep (average depth of trawl sites 140 m) with soft mud (Admiralty chart). In contrast, the majority of creel sites were in Loch Torridon and Loch Gairloch, which have variable depth (average depth of creel sites 90 m), sediment type (Admiralty chart) and hydrographic conditions. These differences in habitat must be considered when interpreting the results.

Significantly higher densities of all three sea pen species and *A. loveni* were observed in the creel areas than the trawl areas, suggesting that trawling has a more negative effect on these species. However, Hughes (1998) noted that both *V. mirabilis* and *P. phosphorea* were very common in Scottish sea lochs, which may explain their higher abundance in the creel areas. Comparing the density of these species in different depth ranges and areas showed little variation in the densities of *V. mirabilis,* reflecting this species' tolerance of a wide range of environmental conditions (Jones *et al.*, 2000). In contrast to a study by Greathead *et al.* (in press.), *P. phosphorea* occurred at highest densities in shallow sea lochs, but at very low densities in the northern Inner Sound (a deeper, creel-only zone), suggesting that low densities of this species in trawled zones may have been due to habitat rather than fishing method. The observed density of *V. mirabilis* and *P. phosphorea* may vary considerably according to how many are emergent at the time of counting, so that estimates of density may not be reliable. This is particularly true for *V. mirabilis*, which retracts much more readily if disturbed (Kinnear *et al.*, 1996). This source of error would affect the conclusions of this study only if the proportion of individuals emerged at the time of the survey differed among fishing zones, in relation to depth, time of day or prior disturbance, for example.

Funiculina quadrangularis was found in a wide range of depths, but highest densities were recorded in the component parts of Loch Torridon and in the northern Inner Sound. *Funiculina quadrangularis* was absent from Loch Gairloch, possibly due to unsuitable substratum or other undetermined environmental conditions in this area. This species was also observed in the North Minch and southern Inner Sound, but at much lower densities than in the northern Inner Sound and Loch Torridon. *Asteronyx loveni* was found in greatest numbers in the deep, open-sea environments of the northern Inner Sound, and at much lower densities in the loch environments despite the presence of their host. Significantly higher densities of *A. loveni* were observed in the northern Inner Sound compared with the North Minch. The higher densities of both *F. quadrangularis* and *A. loveni* in the northern Inner Sound compared with the North Minch area (areas of similar environmental characteristics) suggest an effect of trawling on the density of these species.

The present findings highlight the necessity of selecting sites with similar environmental conditions, as it is not possible to distinguish fishing effects clearly by comparing areas with different habitat characteristics. Unfortunately, only a small number of sites from different fishing zones were sufficiently similar in habitat to allow direct comparison. The sites selected for comparison were in the northern Inner Sound (creel-only zone) and North Minch (open to all fishing methods, but in effect a trawl zone) and were deep-water, open-sea environments with a soft mud substratum.

Comparing the selected sites, no significant difference was observed in the densities of *V. mirabilis* and *P. phosphorea*, indicating that they were not differentially affected by creeling and trawling, perhaps owing to their ability to withdraw into the sediment when disturbed (Howson & Davies, 1991; Tuck *et al.*, 1998; Hughes 1998) and the ability of these species to re-insert themselves if removed (Eno *et al.*, 2001). As noted above, *P. phosphorea* was found only in very small numbers at deep, muddy, open-water sites, so it is not a particularly useful indicator of the effects of fishing in this type of habitat.

The density of *F. quadrangularis* was approximately 14 times higher at selected sites in the creel-only zone than at similar sites in the trawl-only zone, with this difference being statistically significant despite the small sample size (number of stations). *Asteronyx loveni* was present in the creel-only zone, but absent from the sites selected in the trawled zone, this difference also being statistically significant. These findings provide stronger circumstantial evidence than the overall comparison described above that trawling reduces the density of these species compared with creeling. Nevertheless, the possibility that these species occur at lower density in the trawled zone for reasons other than fishing cannot be conclusively excluded. Hughes (1998) suggested that trawling has a negative impact on the tall sea pen (*F. quadrangularis*), as might be expected from its morphology and inability to retract into the sediment, and another study reported lower abundances in heavily trawled areas compared with a lightly trawled area (Greathead *et al.*, in press.).

Analysis of the sites from the BUTEC no-fishing area enabled a further assessment of the relative effects of trawl and creel fishing on *F. quadrangularis* and *A. loveni*. Although there is no fishing within the BUTEC inner sea-area, it should be noted that there may be disturbance to the sea bed from operations involved in testing underwater military equipment (Atkinson *et al.*, 2006). The densities of both *F. quadrangularis* and *A. loveni* were lowest in the trawled zone and highest in the creel-only zone (*F. quadrangularis*) or the no-fishing zone (*A. loveni*). For both species, densities in the creel-only zone and the no-fishing zone were not significantly different, although the small sample size (number of stations) limited the statistical power of the comparisons. Had a greater number of comparable stations been surveyed, it would have been possible to draw more-certain conclusions about whether the differences in density of both species between the creel zone and no-fishing zone, and the difference in density of *F. quadrangularis* between the trawl zone and no-fishing zone, were real or the result of chance sampling variation.

Atkinson *et al.* (2006) compared the prevalence of *A. loveni* on *F. quadrangularis* at two sites in the BUTEC inner area and two sites in the creel zone. The prevalence (expressed as the percentage of sea pens visibly bearing brittle stars) at sites in the no-fishing zone (83%) was significantly higher than that at sites in the creel zone (33%) (Atkinson *et al.*, 2006). In the current study although there was a difference in the relative densities of *A. loveni* and *F. quadrangularis* between the no-fishing zone and the creel zone this difference was not found to be significant.

Taken together, these findings suggest that trawling reduces the density of the tall sea pen, *F. quadrangularis*, and its associated brittle star, *A. loveni*. There is no evidence that creeling adversely affects the density of *F. quadrangularis*, despite a proportion of them being displaced by creels and creel lines. The study by Eno *et al.* (2001) showed that displaced *F. quadrangularis* colonies are able to re-anchor themselves. However, there is evidence, albeit from a small number of sites, that creeling may reduce the density of the brittle star, *A. loveni*, relative to the density of its sea pen host.

Quantification of the proportion of damaged sea pens was difficult, because identification was subjective, which must be considered when assessing the results. There is, however, evidence of damaged *V. mirabilis* and *F. quadrangularis* from both creel and trawl zones, with the highest prevalence for both species in the creel zone. This result is in contrast with previous findings suggesting that the creel fishery would cause little mortality of either species, in particular *V. mirabilis*, as its ability to withdraw rapidly is thought to protect it from damage by creels (Kinnear *et al.*, 1996, Hughes, 1998). The larger proportion of damaged colonies may be a reflection of the higher densities observed in the creel zone, or is perhaps a result of greater sediment disturbance within the trawl fishery causing dead colonies to be buried quicker. Another consideration is that damaged sea pens brought to the surface in trawls may be discarded at a distance from their site of capture, possibly outside the fishing zone or the study area. On the face of it, these findings suggest that, in contrast to the evidence from population densities, creeling causes higher mortality of *F. quadrangularis* and *V. mirabilis* than trawling. However, it is difficult to interpret the quantity of dead colonies in terms of mortality without knowing the longevity of visible colony remains.

The diversity of epibenthic species, as expressed by Simpson's diversity index, did not differ among areas. Jennings & Reynolds (2000) and Cryer *et al.* (2002) suggested that simple indices of biodiversity such as Simpson's index are unlikely to be sensitive or useful indicators of community change and that multivariate techniques are more appropriate.

Differences in the composition of epibenthic species between creel and trawl zones, and among areas and depth ranges, were apparent from multivariate analysis. Both TWINSPAN and the MDS divided the majority of sites into groups that corresponded to trawled and creeled zones and there were statistically significant differences in the community composition between these zones. However, as with the univariate analyses of sea pen density, effects of fishing were confounded with habitat differences, with evidence that species composition was related to both depth and geographical area.

Considering those sites selected to have similar habitat characteristics, species composition did not differ significantly between the creel zone and the trawl zone. To assess an effect of fishing method on community composition, a much larger sample size is desirable. In the present study, the distribution of sites enabled only three sites from the creel zone to be compared with six trawled sites. The patchy nature of many of the species found in these soft mud environments makes fishing effects difficult to distinguish from natural variability, particularly when such small sample sizes are used. In addition, the inherent difficulties of using underwater television techniques make identification to species level difficult, with small or cryptic species being missed

(Holme & Barrett, 1977). This loss of taxonomic resolution is likely to reduce the sensitivity of multivariate comparisons (Clarke & Warwick, 1994).

Several species were identified as contributing significantly to dissimilarity of species composition among sites. The relative abundance of the three sea pen species was important at many of the sites, with *P. phosphorea* being an important distinguishing species in shallower sites, mostly in Upper Loch Torridon and Loch Gairloch. *Funiculina quadrangularis* was typical of the species assemblages in Loch Torridon and the northern Inner Sound. When sites from similar habitats were compared, higher densities of *F. quadrangularis* in creeled areas accounted for nearly 50% of the dissimilarity between sites from creel and trawled zones.

Numbers of all three sea pen species and *A. loveni* caught as bycatch in the Torridon area were found to vary in accordance with density estimates derived from towed UWTV surveys. *Virgularia mirabilis* and *P. phosphorea* were caught in the largest numbers in shallower sites of Upper and lower Loch Torridon and *A. loveni* was caught in the largest numbers in deeper waters of the Inner Sound. As with the density estimates, the numbers of *F. quadrangularis* caught as bycatch appeared to be fairly evenly distributed between Loch Torridon and the Inner Sound. No comparable bycatch data were available for the trawl fishery.

Numbers of *V. mirabilis* and *P. phosphorea* caught as bycatch, particularly in Upper Loch Torridon, appear to be quite high. Eno *et al.* (2001) observed that creels may uproot some *P. phosphorea*. It is possible that following their initial retraction, they may emerge through the netting of the creel, leading to them being removed from the sediment when the creels are hauled. The high densities of *V. mirabilis* and *P. phosphorea* observed in the loch environments, despite intense creel fishing effort there, lead to the conclusion that survival of these species discarded from creel vessels is high, although there may be some discard mortality.

In contrast, numbers of *F. quadrangularis* caught as bycatch were found to be very low, supporting the conclusion that creel fishing has little impact on *F. quadrangularis* populations. From the UWTV surveys of the creel zone of the Inner Sound, *A. loveni* occurred at lower density than *F. quadrangularis*, but the abundance of *A. loveni* relative to *F. quadrangularis* was higher in the bycatch, suggesting that *A. loveni* is more susceptible to creel fishing than its host sea pen. It is possible that the majority will go back undamaged and individual *A. loveni* have been observed separate from their host, *F. quadrangularis* (Atkinson *et al.*, 2006), so they may be able to find a new host. However, there is likely to be some discard mortality, which would tend to reduce the abundance of *A. loveni* in creel fishing areas. Work by Bergmann & Moore (2001a) has suggested that discard mortality of trawled starfish has been underestimated in the past. For the brittle star, *Ophiura ophiura,* they showed that the combination of trawling damage and air exposure resulted in 100% mortality after two weeks. Although the main damage to *O. ophiura* was likely to have been caused by the trawling process, this was accelerated by air exposure. Similarly, if *A. loveni* have been damaged by the creeling process, their demise may be sealed by air exposure.

In summary, although the slender sea pen, *V. mirabilis*, and the phosphorescent sea pen, *P. phophorea*, can be caught in moderate quantities in the creel fishery, high densities of these species persist in areas of intense creel fishing. There are no no-fishing areas in the typical habitat of these species for comparison, but it appears that creel fishing does not have a major adverse effect on them. The density of *V. mirabilis* and *P. phosphorea* is low in the trawled zone, but this may be principally due to the habitat characteristics of those areas (deep, soft mud) being less suitable for these species, particularly *P. phosphorea*. However, there is evidence that the densities of the tall sea pen, *F. quadrangularis*, and that of its symbiotic brittle star, *A. loveni*, are reduced by trawling. The comparatively low bycatch of *F. quadrangularis* in the creel fishery and the high densities of this species in the creel zone suggest that creeling does not adversely impact this species. Evidence was contradictory from the prevalence of damaged and dead colonies, requiring further investigation. There are indications from the present study and that of Atkinson *et al.* (2006) that creeling may be more damaging to *A. loveni* than to its host sea pen, *F. quadrangularis*. The evidence for this is based on rather few samples, however, so further work is desirable to confirm or refute this conclusion.

As with the univariate analyses of sea pen densities, the multivariate analyses to investigate possible fishing effects on epibenthic species composition were confounded by differences in habitat characteristics in the different fishing zones. There were significant differences in species composition among areas that were probably largely due to differences in environmental conditions. However, the sea pen species were important determinants of dissimilarity of species composition among sites. Since *F. quadrangularis* density appears to be affected by trawling, some of the differences between the trawled zone and elsewhere may have been due to the fishing practices there. A direct comparison of species composition between the creel and trawl zones based on sites with similar habitat characteristics did not reveal a significant difference, but again this was perforce based on few samples. The data on species composition and sea pen density analysed in the present study were derived from surveys designed for another purpose (*Nephrops* stock assessment). Future studies to investigate the effects of fishing practices on the epibenthos in the Torridon area should endeavour to survey sufficient stations of similar habitat characteristics in zones fished by creeling or trawling.

5.5 Summary

- Towed UWTV surveys from 2000–2005 were assessed for the numbers of all observed species from sites in areas fished by creels and by trawls.

- There were significant differences in species composition among areas, but it is thought that the differences were largely due to environmental conditions.

- The sea pen species *V. mirabilis, P. phosphorea* and *F. quadrangularis* (and associated brittle star *A. loveni*) were all found in lower densities in the trawled areas when compared to areas fished solely by *Nephrops* creels.

- Reduced densities of both *V. mirabilis* and *P. phosphorea* in the trawled area may be principally due to the habitat characteristics of those areas being less suitable for these species, particularly *P. phosphorea.*

- There is evidence that the densities of the tall sea pen, *F. quadrangularis*, and that of its symbiotic brittle star, *A. loveni*, are reduced by trawling.

- Towed UWTV from the BUTEC fishery exclusion zone provided evidence that creeling may be more damaging to *A. loveni* than to its host sea pen, *F. quadrangularis.*

- *V. mirabilis* and *P. phophorea*, were caught in moderate quantities in the creel fishery, with a comparatively low bycatch of *F. quadrangularis.*

- High densities of all three sea pen species were observed in areas where bycatch was greatest, suggesting creeling does not adversely impact these species.

- Damaged and dead colonies of the sea pens *V. mirabilis* and *F. quadrangularis* were more prevalent in creeled areas, when compared with areas fished by trawls. This requires further investigation.

Table 5.1 - Population density (100 m^2)$^{-1}$ of species/taxonomic groups (F. = family; C. = class; O. = order) estimated from UWTV surveys by FRS from all FRS sites (n = number of individual animals observed).

Species	Mean Density (± SD) (100 m^2)$^{-1}$	n	Comments
Cnidaria			
Funiculina quadrangularis	6.21 ± 11.91	2171	
Pennatula phosphorea	6.08 ± 36.08	2626	
Virgularia mirabilis	3.86 ± 9.79	1181	
Cerianthus lloydii	0.04 ± 0.17	16	
Pachycerianthus multiplicatus	0.14 ± 0.31	48	
Urticina eques	0.03 ± 0.4	7	
Annelida			
Sabella pavonina	0.03 ± 0.2	19	
Crustacea			
Nephrops norvegicus	2.68 ± 2.84	874	
F. Paguraidae	1.09 ± 2.79	410	Mostly *Pagurus bernhardus*
Munida rugosa	1.28 ± 4.1	563	Includes *Munida sarsi*
Cancer pagurus	0.37 ± 1.12	174	
Carcinus maenas	0.61 ± 0.89	203	Includes *Liocarcinus depurator*
Mollusca			
Buccinum undatum	0.05 ± 0.18	22	Includes *Neptunea antiqua*
Pecten maximus	0.01 ± 0.07	2	
Aequipecten opercularis	1.09 ± 7.61	270	
Echinodermata			
F. Antedonidae	1.48 ± 8.08	881	
Luidia ciliaris	0.05 ± 0.25	28	
Porania pulvillus	0.32 ± 0.82	162	
F. Solasteridae	0.05 ± 0.36	23	
Asterias rubens	0.38 ± 1.58	148	
C. Ophiuroidea	0.08 ± 0.41	28	Does not include *Asteronyx loveni*
Asteronyx loveni	0.13 ± 0.52	43	
Echinus esculentus	0.37 ± 2.22	105	
F. Spatangidae	0.01 ± 0.09	2	
C. Holothurioidea	0 ± 0.03	2	
Chondrichthyes			
F. Rajidae	0.01 ± 0.07	5	
Scyliorhinus canicula	0.07 ± 0.31	21	Includes *Galeus melastomus*
Osteichthyes			
F. Gadidae	5.28 ± 5.51	1811	Mostly *Trisopterus minutus*
O. Pleuronectiformes	0.4 ± 0.63	115	Mostly *Pleuronectes platessa*

Table 5.2 - Descriptive statistics of densities (m^{-2}) of *F. quadrangularis*, *V. mirabilis*, *P. phosphorea* and *A. loveni* in different areas, depth ranges and fishing zones (* Inner Sound (S) data removed due to area being fished by both methods).

		Species			
Location		*F. quadrangularis*	*V. mirabilis*	*P. phosphorea*	*A. loveni*
Inner Sound (N)	mean	0.311	0.028	0	0.014
	s.d.	0.478	0.029	0.001	0.015
	n	5	4	4	5
Upper Loch Torridon	mean	0.078	0.101	0.222	0.001
	s.d.	0.122	0.213	0.695	0.003
	n	26	26	26	26
lower Loch Torridon	mean	0.088	0.010	0.010	0.001
	s.d.	0.103	0.015	0.027	0.002
	n	15	15	15	15
Loch Gairloch	mean	0	0.002	0.608	0
	s.d.	0	0.002	1.160	0
	n	4	4	4	4
North Minch	mean	0.045	0.029	0.003	0.001
	s.d.	0.048	0.031	0.007	0.004
	n	89	89	89	89
Inner Sound (S)	mean	0.033	0.012	0	0.007
	s.d.	0.042	0.012	0	0.015
	n	5	4	4	5
Depth					
<50 m	mean	0.074	0.171	1.020	0
	s.d.	0.138	0.318	1.366	0
	n	7	7	7	7
50-100 m	mean	0.052	0.048	0.037	0
	s.d.	0.095	0.123	0.101	0.001
	n	35	35	35	35
101–150 m	mean	0.077	0.026	0.002	0.002
	s.d.	0.159	0.033	0.007	0.005
	n	60	60	60	60
>150 m	mean	0.050	0.026	0.001	0.003
	s.d.	0.047	0.023	0.002	0.007
	n	42	40	40	42
Fishing Zones*					
Trawl	mean	0.045	0.029	0.003	0.001
	s.d.	0.048	0.031	0.007	0.004
	n	89	89	89	89
Creel	mean	0.098	0.059	0.171	0.002
	s.d.	0.187	0.160	0.603	0.006
	n	50	49	49	50

Table 5.3 - Summary of Kruskal-Wallis test results (test statistic, *H*, degrees of freedom, d.f., and significance level, *P*) comparing densities of *F. quadrangularis, V. mirabilis, P. phosphorea* and *A. loveni* from different areas, depths and fishing zones. Dunn's test for multiple comparisons also shown for significant comparisons. (* Inner Sound (S) data removed due to area being fished by both methods, **ANOVA test on log(x+1)-transformed *F. quadrangularis* densities).

Species	*H*	d.f.	*P*	Multiple comparisons	*P*
	Location				
F. quadrangularis	13.22	5	< 0.05	IS (N) vs. LG	< 0.0001
				NM vs. LG	< 0.005
V. mirabilis	15.19	5	0.010	None	
P. phosphorea	25.43	5	< 0.0001	NM vs. ULT	< 0.0001
A. loveni	22.10	5	0.001	NM vs. IS (N)	0.0001
				IS (N) vs. ULT	< 0.001
				IS (N) vs. LG	< 0.005
	Depth				
F. quadrangularis	7.90	3	< 0.05	None	
V. mirabilis	4.20	3	0.240		
P. phosphorea	44.30	3	< 0.0001	> 150 vs. 50 - 100	< 0.0001
				> 150 vs. < 50	< 0.0001
				101 - 150 vs. 50 - 100	< 0.0001
				101 - 150 vs. < 50	< 0.0001
A. loveni	4.72	3	0.193		
	Fishing Zones*				
*F. quadrangularis***	F=6.39	1 137	< 0.05		
V. mirabilis	6.98	1	< 0.01		
P. phosphorea	17.52	1	< 0.0001		
A. loveni	5.15	1	< 0.05		

Table 5.4 - Descriptive statistics of densities (m^{-2}) of *F. quadrangularis, V. mirabilis, P. phosphorea* and *A. loveni*, from sites selected for their similar environmental conditions, in different fishing zones.

		Species			
Fishing Zones		*F. quadrangularis*	*V. mirabilis*	*P. phosphorea*	*A. loveni*
Trawl	mean	0.0278	0.0312	0.00149	0
	s.d.	0.0294	0.0406	0.00365	0
	n	6	6	6	6
Creel	mean	0.388	0.0377	0.0006	0.01729
	s.d.	0.514	0.0264	0.00103	0.01419
	n	4	3	3	4

Table 5.5 - Summary of Kruskal-Wallis test results (test statistic, *H*, degrees of freedom, d.f., and significance level, *P*) comparing densities of *F. quadrangularis, V. mirabilis, P. phosphorea* and *A. loveni* from different fishing zones.

	Fishing Zones		
Species	*H*	d.f.	*P*
F. quadrangularis	5.50	1	< 0.05
V. mirabilis	1.07	1	0.302
P. phosphorea	0.13	1	0.724
A. loveni	5.57	1	< 0.05

Table 5.6 - Descriptive statistics of densities (number m^{-2}) of *F. quadrangularis* and *A. loveni*, from sites selected for their similar environmental conditions, from different fishing zones.

		Species	
Fishing Zones		*F. quadrangularis*	*A. loveni*
Trawl	mean	0.0278	0
	s.d.	0.0294	0
	n	6	6
Creel	mean	0.388	0.01729
	s.d.	0.514	0.01419
	n	4	4
No Fishing	mean	0.0957	0.07564
	s.d.	0.0263	0.00415
	n	2	2

Table 5.7 - Summary of Kruskal-Wallis test results (test statistic, *H*, degrees of freedom, d.f., and significance level, *P*) comparing densities of *F. quadrangularis* and *A. loveni* from different fishing zones. Dunn's test for multiple comparisons also shown for significant comparisons.

	Fishing Zones				
Species	*H*	d.f.	*P*	Multiple comparisons	*P*
F. quadrangularis	6.88	2	<0.05	Creel vs. Trawl	<0.05
A. loveni	8.97	2	<0.05	Trawl vs. No Fishing	<0.005

Table 5.8 - Descriptive statistics of proportion of damaged *F. quadrangularis* and *V. mirabilis*, from different fishing zones. (* Inner Sound (S) data removed due are being fished by both methods).

		Species	
Fishing Zone*		*F. quadrangularis*	*V. mirabilis*
Trawl	mean	0.0548	0.054
	s.d.	0.1134	0.1998
	n	86	79
Creel	mean	0.1069	0.1858
	s.d.	0.1781	0.3351
	n	34	33

Table 5.9 - Summary of ANOVA test results (test statistic, *F*, degrees of freedom, d.f., and significance level, *P*) comparing proportion of damaged *F. quadrangularis* and *V.miribilis* from different fishing zones (* Inner Sound (S) data removed due are being fished by both methods).

	Fishing Zones*		
Species	*F*	d.f.	*P*
F. quadrangularis	8.12	1 118	0.005
V. mirabilis	15.35	1 110	<0.0001

Table 5.10 - TWINSPAN group properties

Group	Division	Eigenvalue	Samples
0	1	0.347	116
00	2	0.149	82
01			34
1	1	0.347	24
10	2	0.243	8
11			16

Table 5.11 - Percentage distribution of sites from different areas, depth ranges and fishing zones across divisions produced from TWINSPAN analysis (ULT = Upper Loch Torridon; LLT = lower Loch Torridon; LG = Loch Gairloch; IS = Inner Sound; NM = North Minch).

		Group					
		0	00	01	1	10	11
Fishing Type	Creel	23.3	9.8	61.8	91.7	87.5	93.8
	Trawl	76.7	90.2	38.2	8.3	12.5	6.3
Depth (m)	< 50	0.9	0.0	2.9	20.8	37.5	12.5
	50–100	20.7	13.4	38.2	45.8	50.0	43.8
	101–150	46.6	43.9	52.9	29.2	12.5	37.5
	>150	31.9	42.7	5.9	4.2	0.0	6.3
Location	ULT	12.9	6.1	29.4	50.0	62.5	43.8
	LLT	8.6	2.4	23.5	20.8	12.5	25.0
	LG	0.0	0.0	0.0	16.7	12.5	18.8
	IS	1.7	1.2	2.9	8.3	0.0	12.5
	NM	76.7	90.2	44.1	4.2	12.5	0.0

Table 5.12 - Indicator species and taxonomic groups (F. = family; O. = order) from each TWINSPAN group.

Group	Indicator Species
0	*Funiculina quadrangularis*
00	*Carcinus maenas*, F. Gadidae, O. Pleuronectiformes
01	*Funiculina quadrangularis, Pennatula phosphorea, Virgularia mirabilis, Porania pulvillus*
1	*P. pulvillus, Asterias rubens, Cancer pagurus, Munida rugosa*
10	*P. phosphorea, Buccinum undatum*
11	*P. pulvillus, B. undatum, M. rugosa*

Table 5.13 - Summary of ANOSIM of community composition at sites from different areas, depth ranges and fishing (only significant comparisons shown). Probabilities adjusted using Bonferonni correction to protect against type I errors (* Inner Sound (S) data removed due are being fished by both methods).

	R	*P*	Multiple comparisons	R	*P*
Area	0.548	0.001	NM vs. ULT	0.560	0.015
			NM vs. LLT	0.536	0.015
			NM vs. LG	0.969	0.015
			LLT vs. LG	0.397	0.030
Depth range	0.191	0.001	>150 vs. 50–100	0.270	0.006
			>150 vs. <50	0.916	0.006
			101–150 vs. 50–100	0.182	0.006
			101–150 vs. <50	0.665	0.006
			50–100 vs. <50	0.289	0.048
Fishing Zones*	0.439	0.001			

Table 5.14 - Analysis of dissimilarities (SIMPER) based on species abundance and taxonomic group (F. = family), from different areas, depth ranges and fishing zones.

Factor groups		Average Dissimilarity	Species	Mean Abundance		Cumulative percentage
Factor 1	Factor 2			Factor 1	Factor 2	
Location						
North Minch	Inner Sound (N)	70.6	*F. quadrangularis*	0.05	0.34	36.89
			F. Antedonidae	0	0.12	52.65
			M. rugosa	0	0.07	66.92
North Minch	Upper Loch Torridon	75.5	F. Gadidae	0.06	0.04	15.55
			F. quadrangularis	0.05	0.08	30.85
			V. mirabilis	0.03	0.1	43.54
			P. phosphorea	0	0.22	56.23
			N. norvegicus	0.03	0.03	65.02
North Minch	lower Loch Torridon	69.68	*F. quadrangularis*	0.05	0.09	26.42
			F. Gadidae	0.06	0.04	42.24
			M. rugosa	0	0.04	53.3
			F. Antedonidae	0	0.06	63.4
North Minch	Inner Sound (S)	67.83	*M. rugosa*	0	0.09	25.2
			F. quadrangularis	0.05	0.03	40.98
			F. Gadidae	0.06	0.02	56.44
			C. pagurus	0	0.03	65.07
North Minch	Loch Gairloch	93.23	*P. phosphorea*	0	0.61	28.53
			F. Gadidae	0.06	0.01	41.29
			F. quadrangularis	0.05	0	50.4
			F. Paguraidae	0.01	0.03	57.55
			N. norvegicus	0.03	0	64.35
Inner Sound (N)	Upper Loch Torridon	81.35	*F. quadrangularis*	0.34	0.08	28.54
			F. Antedonidae	0.12	0.02	41.43
			F. Gadidae	0.09	0.04	53.1
			M. rugosa	0.07	0.01	63.46
Inner Sound (N)	lower Loch Torridon	71.7	*F. quadrangularis*	0.34	0.09	36.67
			F. Antedonidae	0.12	0.06	55.64
			M. rugosa	0.07	0.04	69.49
Inner Sound (N)	Inner Sound (S)	71.65	*F. quadrangularis*	0.34	0.03	35.3
			M. rugosa	0.07	0.09	51.28
			F. Antedonidae	0.12	0	66.17
Inner Sound (N)	Loch Gairloch	91.54	*F. quadrangularis*	0.34	0	24.37
			P. phosphorea	0	0.61	47.55
			F. Gadidae	0.09	0.01	59.33
			F. Antedonidae	0.12	0	69.58
Upper Loch Torridon	lower Loch Torridon	79.22	*F. quadrangularis*	0.08	0.09	20.96
			P. phosphorea	0.22	0.01	33.46
			F. Antedonidae	0.02	0.06	43.53
			V. mirabilis	0.1	0.01	52.93
			F. Gadidae	0.04	0.04	62.03

Upper Loch Torridon	Inner Sound (S)	79.68	*M. rugosa*	0.01	0.09	17.85
			F. quadrangularis	0.08	0.03	30.71
			P. phosphorea	0.22	0	42.18
			V. mirabilis	0.1	0.01	51.88
			F. Gadidae	0.04	0.02	60.01
Upper Loch Torridon	Loch Gairloch	86.9	*P. phosphorea*	0.22	0.61	32.72
			F. quadrangularis	0.08	0	40.29
			A. opercularis	0.06	0.01	47.78
			V. mirabilis	0.1	0	54.99
			F. Paguraidae	0.02	0.03	61.22
lower Loch Torridon	Inner Sound (S)	70.8	*M. rugosa*	0.04	0.09	24.71
			F. quadrangularis	0.09	0.03	48.43
			F. Antedonidae	0.06	0	57.66
			F. Gadidae	0.04	0.02	66.34
lower Loch Torridon	Loch Gairloch	88.98	*P. phosphorea*	0.01	0.61	29.31
			F. quadrangularis	0.09	0	46.03
			M. rugosa	0.04	0.02	53.49
			F. Gadidae	0.04	0.01	60.47
Inner Sound (S)	Loch Gairloch	85.88	*P. phosphorea*	0	0.61	30.08
			M. rugosa	0.09	0.02	46.25
			F. quadrangularis	0.03	0	53.45
			F. Paguraidae	0	0.03	60.43
Depth						
>150 m	<50 m	91.44	*P. phosphorea*	0	1.02	37.99
			M. rugosa	0	0.04	46.26
			F. Gadidae	0.06	0.01	54.52
			V. mirabilis	0.03	0.17	62.65
>150 m	50–100 m	68.17	*F. quadrangularis*	0.05	0.05	17.73
			F. Gadidae	0.06	0.04	35.22
			V. mirabilis	0.03	0.05	47.36
			N. norvegicus	0.03	0.03	57.37
			P. phosphorea	0	0.04	64.52
>150 m	101–150 m	59.06	*F. quadrangularis*	0.05	0.08	24.4
			F. Gadidae	0.06	0.06	46.61
			N. norvegicus	0.03	0.03	58.22
			V. mirabilis	0.03	0.03	69.4
101–150 m	50–100 m	71.49	*F. quadrangularis*	0.08	0.05	19.62
			F. Gadidae	0.06	0.04	35.91
			V. mirabilis	0.03	0.05	47.04
			N. norvegicus	0.03	0.03	56.12
			M. rugosa	0.01	0.03	63
101–150 m	<50 m	90.74	*P. phosphorea*	0	1.02	37.27
			F. quadrangularis	0.08	0.07	46.83
			F. Gadidae	0.06	0.01	54.95
			M. rugosa	0.01	0.04	62.94

50–100 m	<50 m	87.37	*P. phosphorea*	0.04	1.02	38.3
			M. rugosa	0.03	0.04	47.43
			V. mirabilis	0.05	0.17	56.25
			F. quadrangularis	0.05	0.07	63
Fishing Zones						
Trawl	Creel	74.5	*F. quadrangularis*	0.04	0.1	19.4
			F. Gadidae	0.06	0.04	34.18
			P. phosphorea	0	0.17	44.97
			V. mirabilis	0.03	0.06	54.98
			N. norvegicus	0.03	0.02	63.12

Table 5.15 - Analysis of dissimilarities (SIMPER) based on species abundance and taxonomic group (F. = family), from sites of similar environmental characteristics where fishing methods differ.

Fishing Zones		Average Dissimilarity	Species	Mean Abundance		Cumulative percentage
				Trawl	Creel	
Trawl	Creel	70.9	*F. quadrangularis*	0.03	0.46	43.84
			F. Antedonidae	0.04	0.16	65.17
			F. Gadidae	0.07	0.09	74.44
			M. rugosa	0.01	0.06	81.40
			V. mirabilis	0.03	0.04	85.72
			N. norvegicus	0.03	0.04	88.19
			A. loveni	0	0.02	90.26

Table 5.16 - Descriptive statistics of bycatch CPUE (number 100-creels^{-1}) of *F. quadrangularis*, *V. mirabilis*, *P. phosphorea* and *A. loveni* from different locations, depths and sediment types.

Factor		Species			
Location		*F. quadrangularis*	*V. mirabilis*	*P. phosphorea*	*A. loveni*
Inner Sound	mean	1.494	0	0	2.146
	s.d.	2.063	0	0	2.980
	n	76	76	76	76
lower Loch Torridon	mean	0.696	3.740	7.980	0.304
	s.d.	1.463	14.84	19.91	1.015
	n	40	40	40	40
Upper Loch Torridon	mean	1.565	56.5	44.3	1.043
	s.d.	1.429	63.1	74	1.556
	n	5	5	5	5
Depth					
<50 m	mean	0.745	45.300	47.300	0
	s.d.	1.057	55.8	63.3	0
	n	7	7	7	7
50–100 m	mean	0.537	3.380	6.040	0.205
	s.d.	1.483	15.13	16.29	0.644
	n	34	34	34	34
101–150 m	mean	1.141	0	0.272	1.250
	s.d.	1.337	0	0.882	1.904
	n	16	16	16	16
>150 m	mean	1.679	0	0	2.399
	s.d.	2.150	0	0	3.117
	n	64	64	64	64
Sediment Type					
Soft	mean	1.595	4.88	3.73	2.175
	s.d.	2.105	21.61	15.53	3.029
	n	69	69	69	69
Hard	mean	0.753	1.84	5.45	0.585
	s.d.	1.441	12.1	24.6	1.448
	n	52	52	52	52

Table 5.17 - Summary of Kruskal-Wallis test results of bycatch CPUE (number 100-creels^{-1}) of *F. quadrangularis*, *V. mirabilis*, *P. phosphorea* and *A. loveni* from different areas, depth ranges and sediment (soft and hard) types. Dunn's test for multiple comparisons also shown, for significant comparisons.

Species	*H*	d.f.	*P*	Multiple comparisons	*P*
	Location				
F. quadrangularis	4.81	2	0.057		
V. mirabilis	31.95	2	< 0.0001	IS vs. ULT	< 0.0001
				LLT vs. ULT	< 0.0001
				LLT vs. IS	< 0.05
P. phosphorea	35.94	2	< 0.0001	LLT vs. IS	< 0.0001
				IS vs. ULT	0.0001
A. loveni	18.84	2	< 0.0001	LLT vs. IS	0.0001
	Depth				
F. quadrangularis	11.83	3	< 0.01	50-100 vs. >150	< 0.001
V. mirabilis	37.07	3	< 0.0001	<50 vs. >150	< 0.0001
				<50 vs. 100-150	< 0.0001
				<50 vs. 50-100	< 0.0001
P. phosphorea	46.51	3	< 0.0001	<50 vs. >150	< 0.0001
				<50 vs. 100-150	< 0.0001
				<50 vs. 50-100	< 0.0001
				50-100 vs. >150	< 0.0005
A. loveni	23.20	3	< 0.0001	50-100 vs. >150	< 0.0001
				<50 vs. >150	< 0.005
	Sediment Type				
F. quadrangularis	6.39	1	< 0.05		
V. mirabilis	1.13	1	0.287		
P. phosphorea	1.80	1	0.179		
A. loveni	12.81	1	< 0.0001		

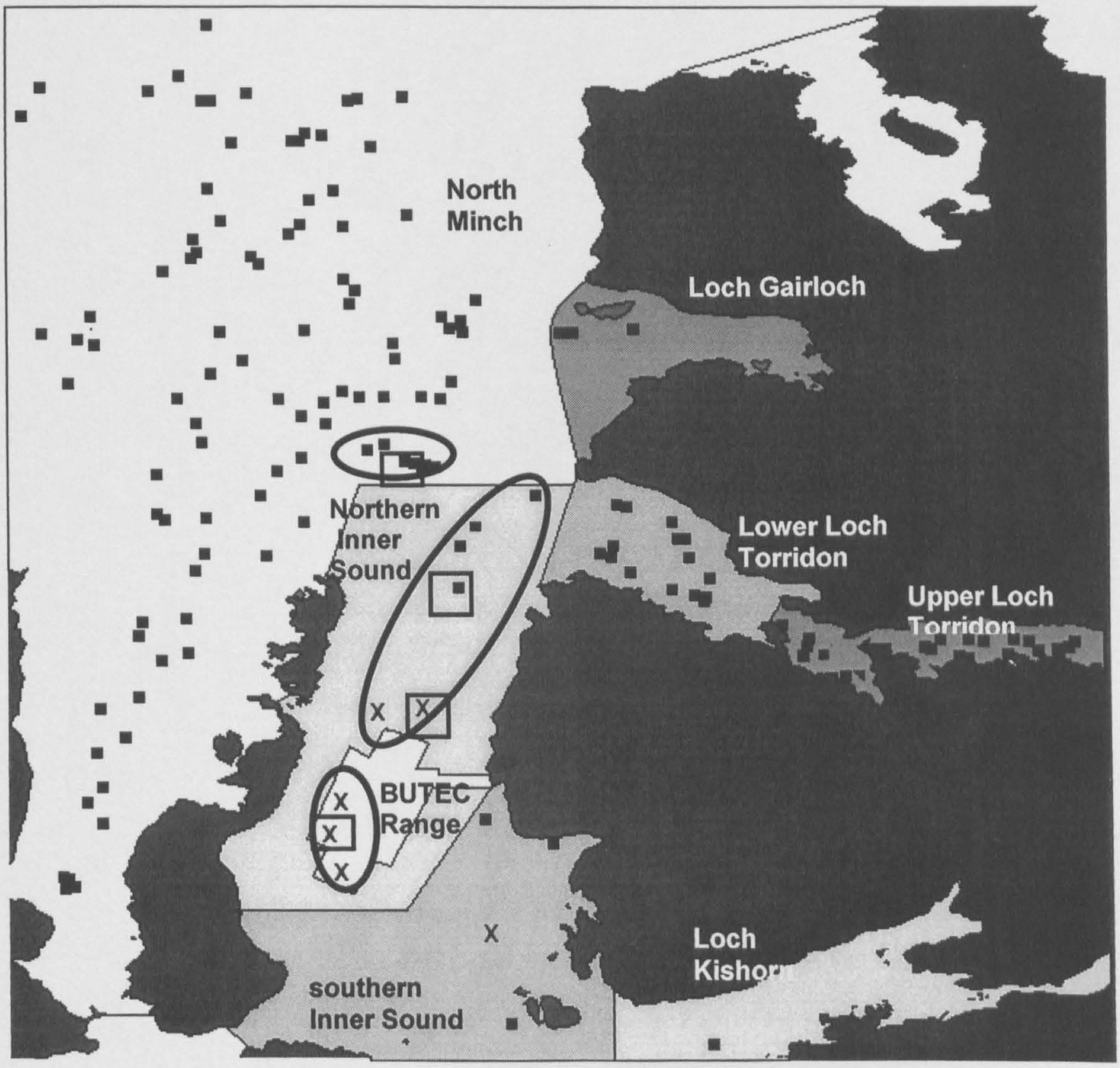


Figure 5.1 - Location of stations from FRS (FRV *Clupea*) towed-TV surveys (squares) and RV *Aora* towed-TV survey (crosses), analysed for megafaunal abundances. Sites within circles are selected sites for fishing method comparison (trawl only, n = 6; creel only, n = 4; no fishing, n = 2), sites within squares discounted samples due to a rocky substratum (n = 4).

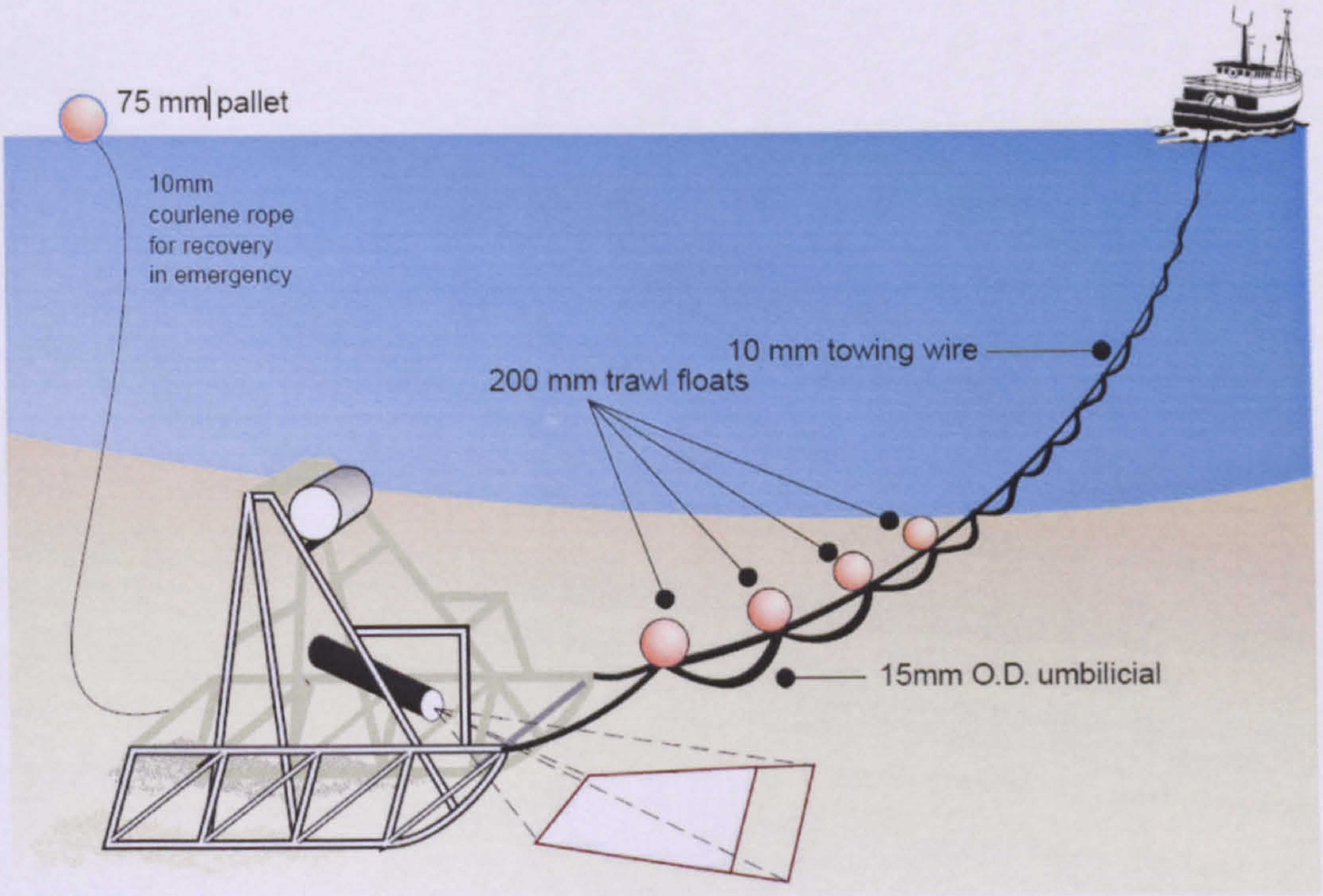


Figure 5.2 - Towed sledge with composite towing wire and electric cable for underwater television surveys of the sea bed (from Shand & Priestley, 1999).

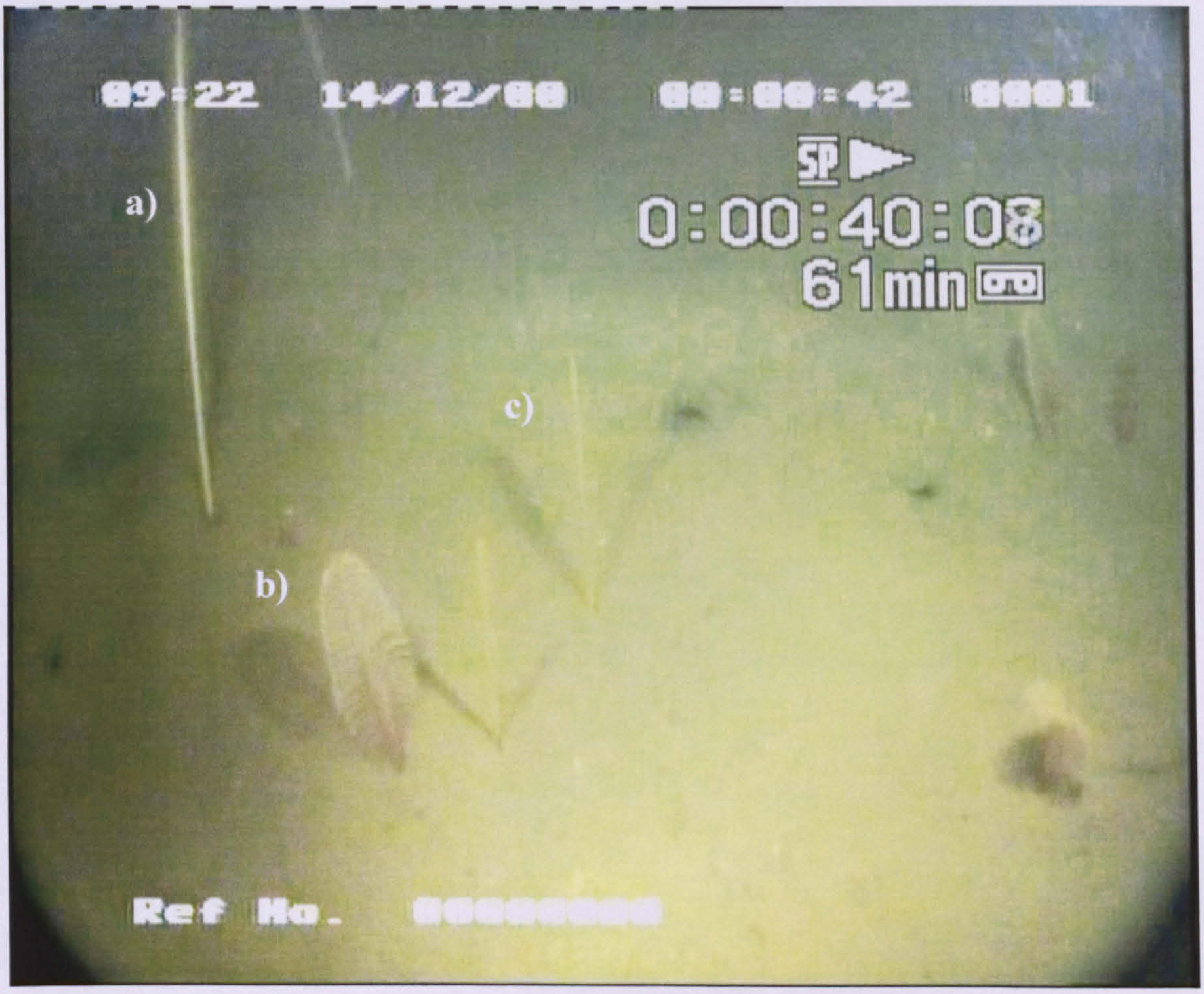


Figure 5.3 - Typical view from towed TV showing a) *F. quadrangularis,* b) *P. phosphorea* and c) *V. mirabilis* (Upper Loch Torridon).

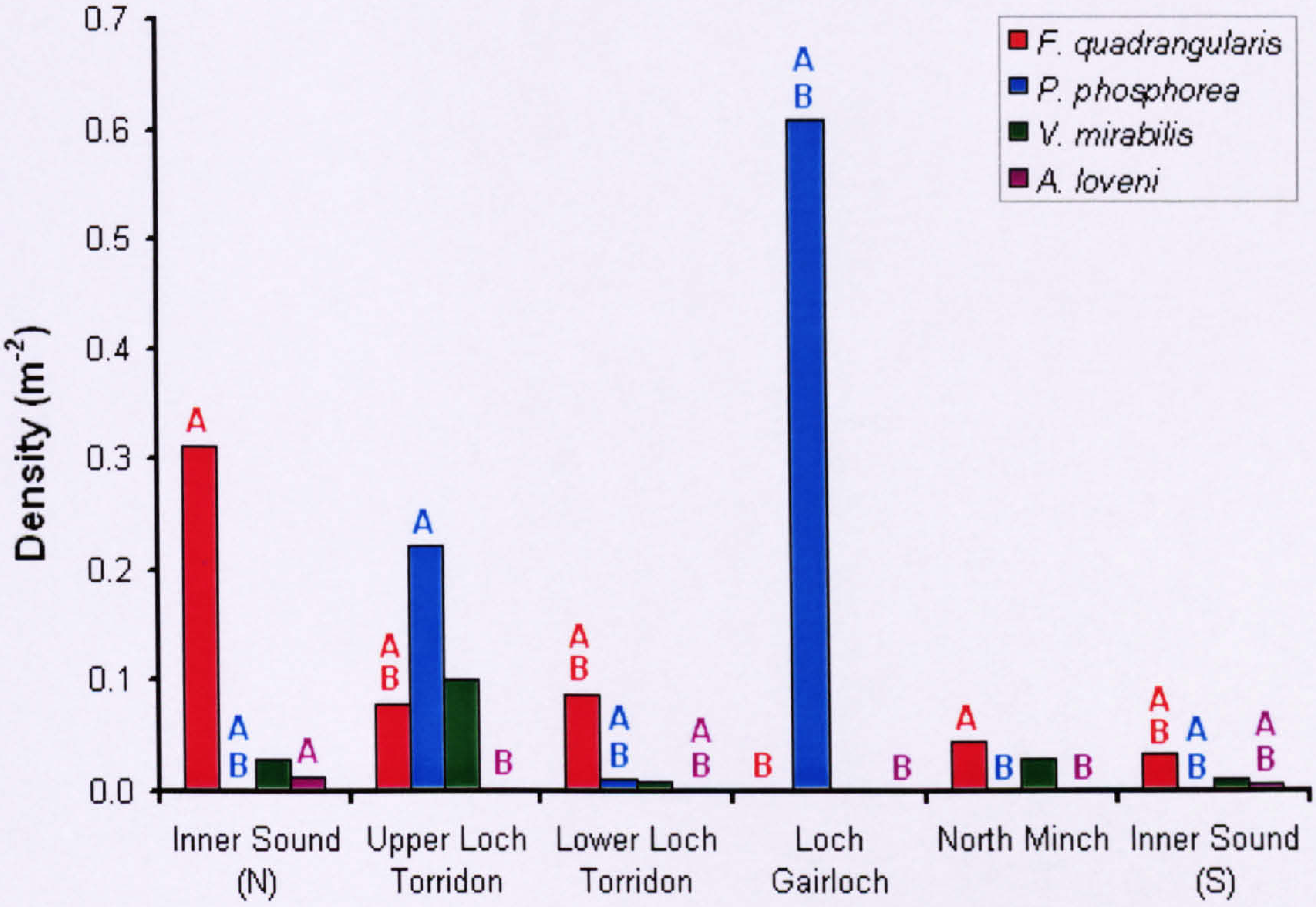


Figure 5.4 - Mean densities of *F. quadrangularis, V. mirabilis, P. phosphorea* and *A. loveni* in different areas. Within each species different letters indicate significant differences ($P<0.05$).

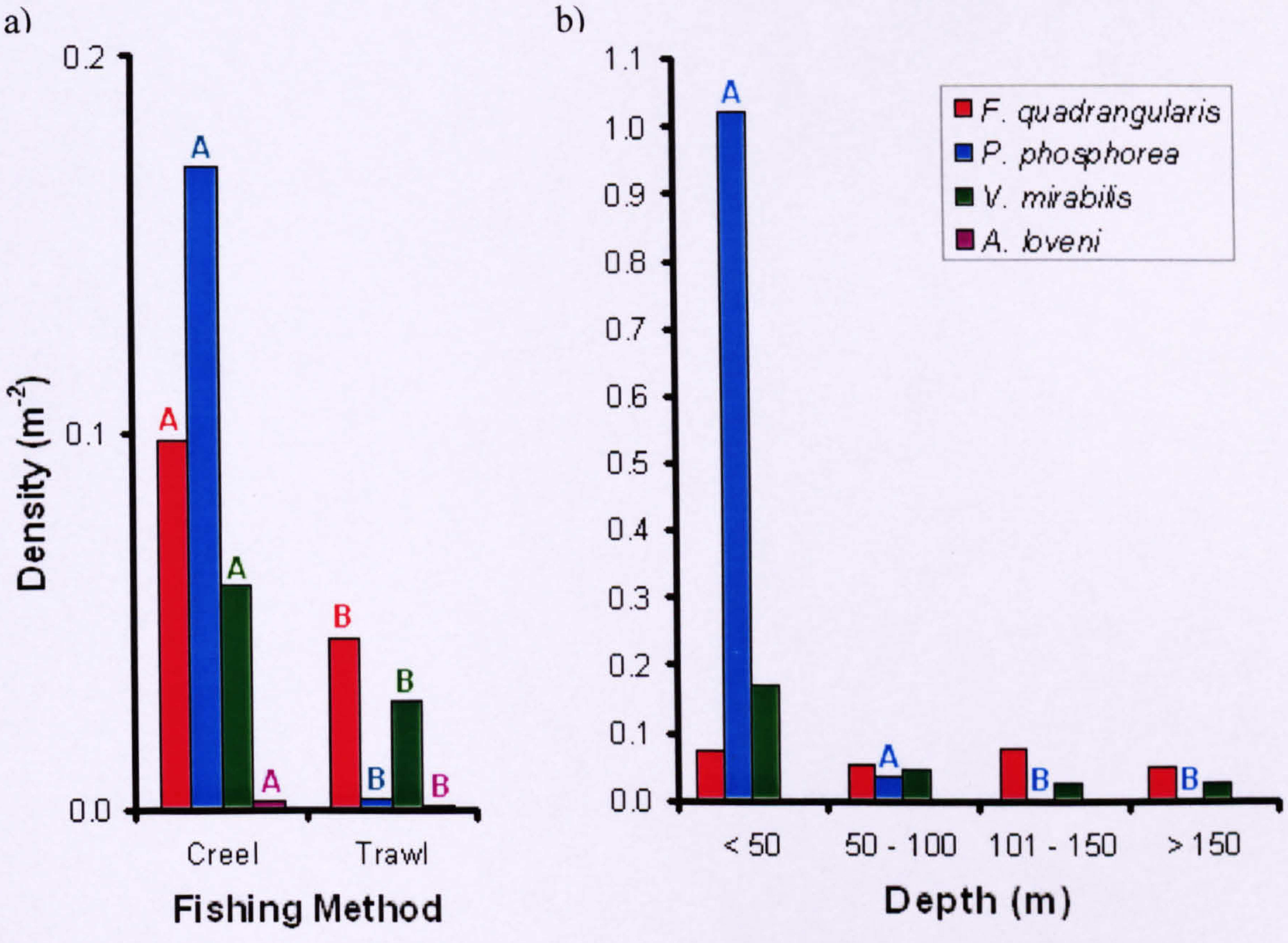


Figure 5.5 - Mean densities of *F. quadrangularis, V. mirabilis, P. phosphorea* and *A. loveni* in (a) zones where fishing methods differ and (b) different depth ranges. Within each species different letters indicate significant differences ($P<0.05$).

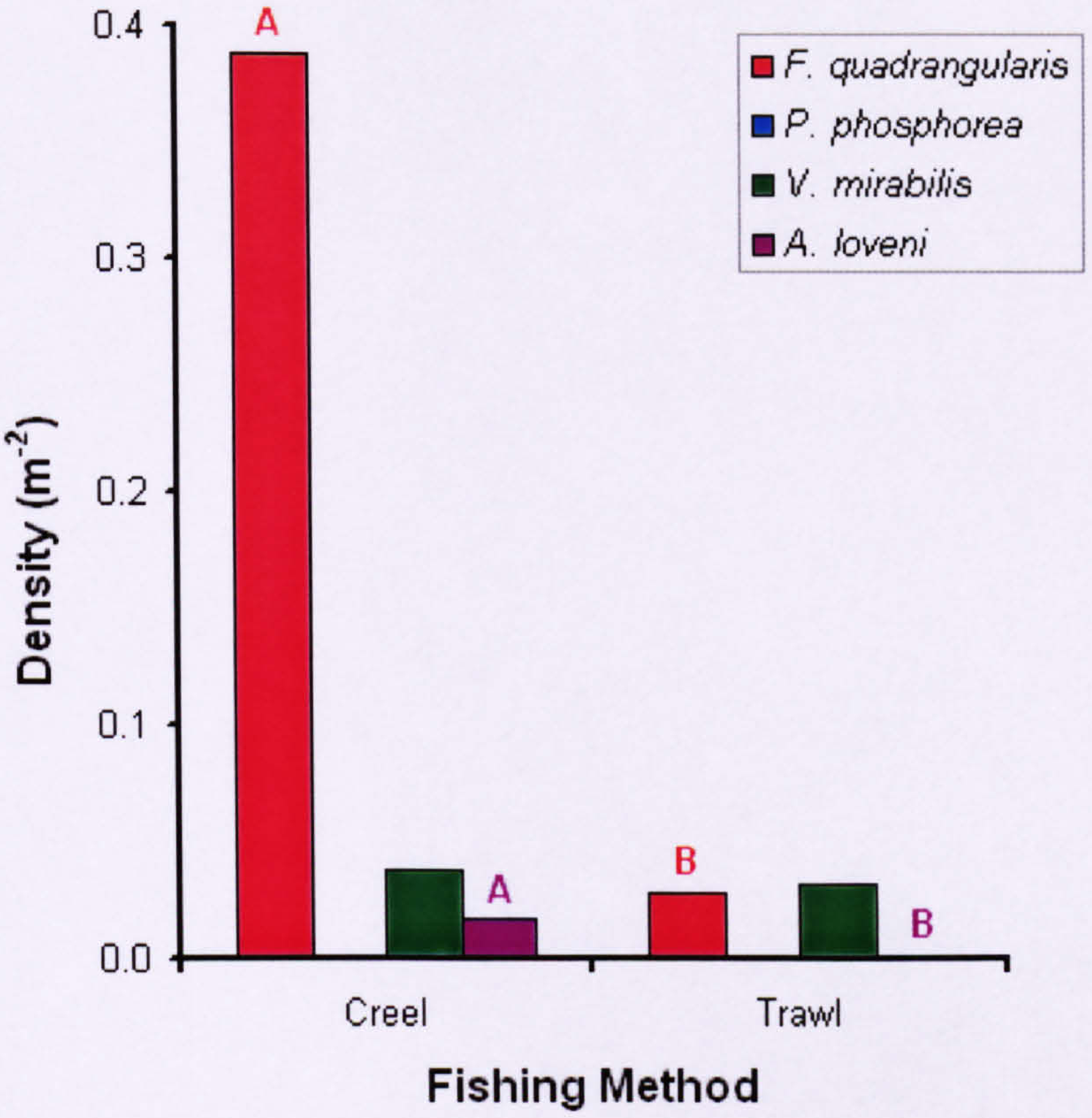


Figure 5.6 - Mean densities of *F. quadrangularis*, *V. mirabilis*, *P. phosphorea* and *A. loveni* from sites selected for their similar environmental conditions, in different fishing zones. Within each species different letters indicate significant differences (*P*<0.05).

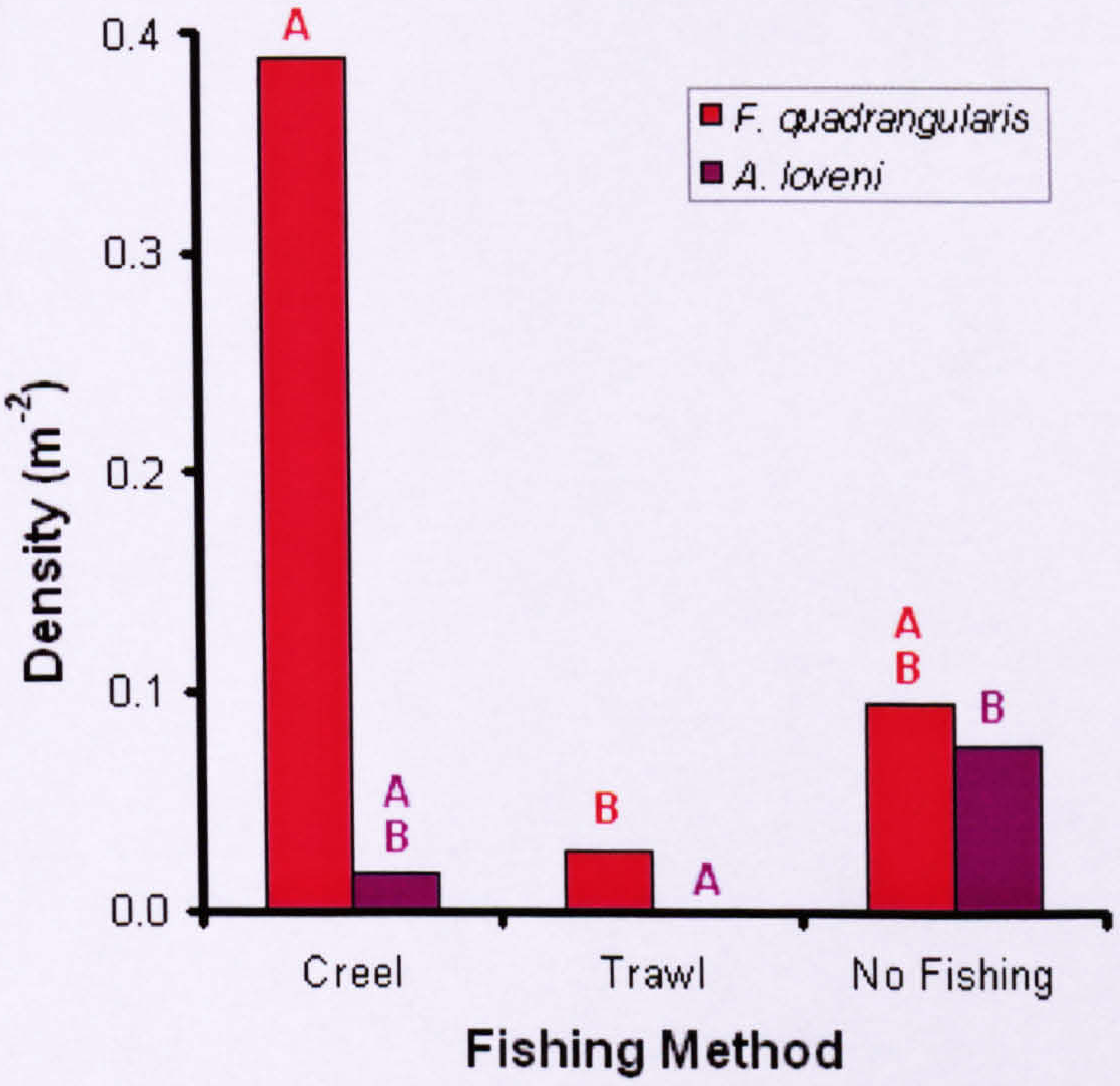


Figure 5.7 - Mean densities of *F. quadrangularis* and *A. loveni* from sites selected for their similar environmental conditions, in different fishing zones. Within each species different letters indicate significant differences (*P*<0.05).

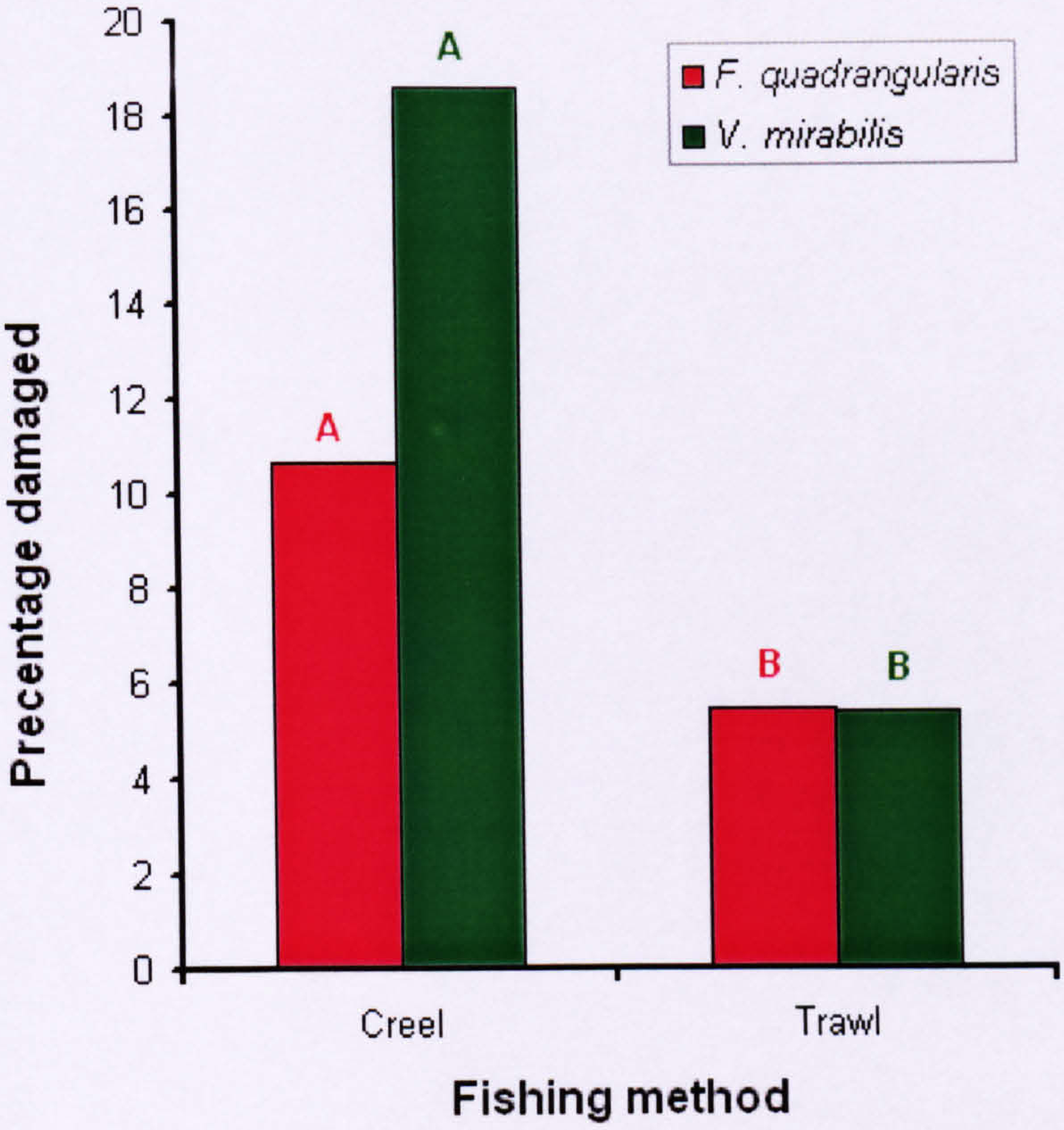


Figure 5.8 - Mean percentage of damaged *F. quadrangularis* and *V. mirabilis*, from different fishing zones. Within each species, different letters indicate significant differences ($P<0.05$).

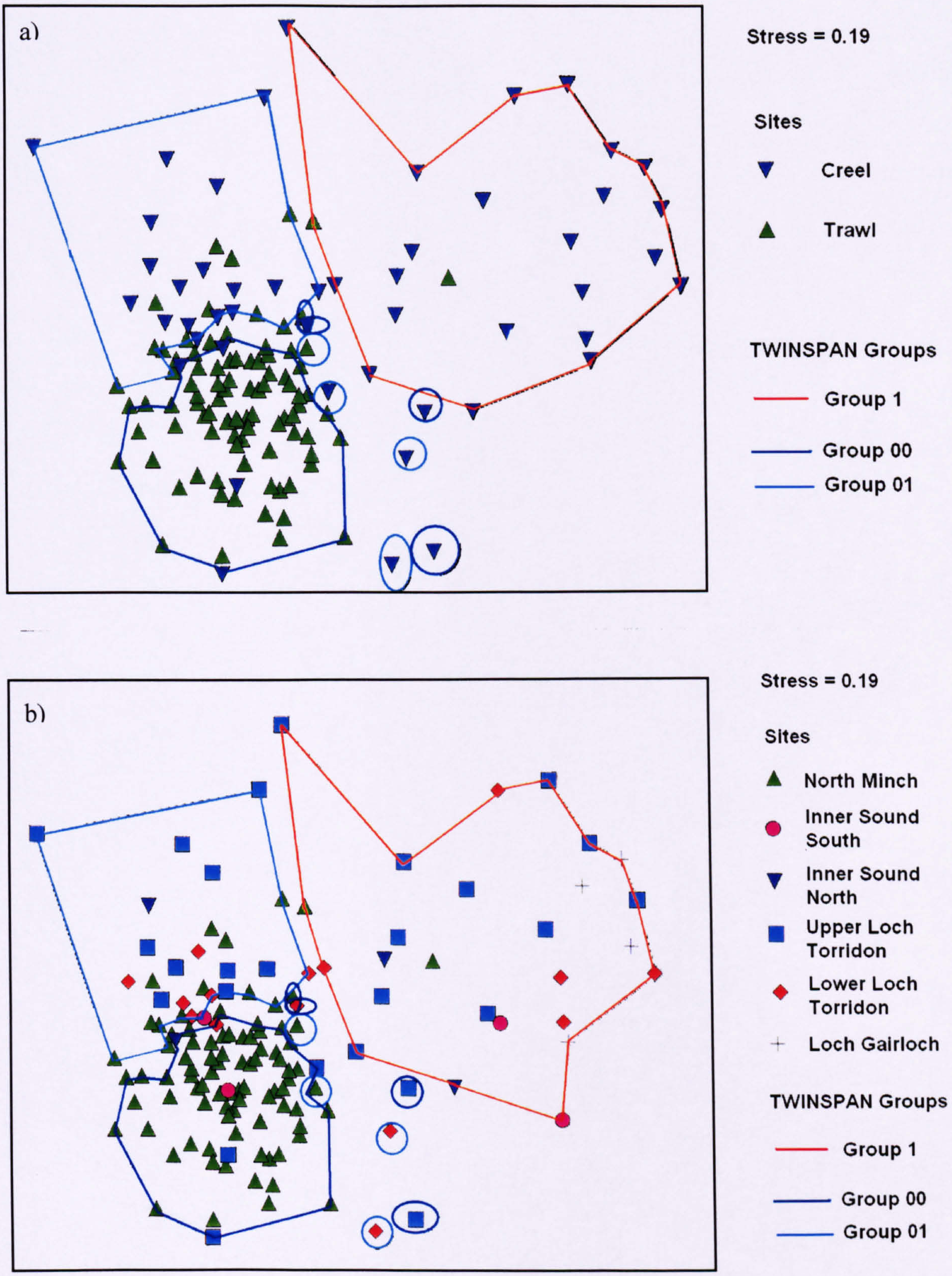


Figure 5.9 - MDS plots of stations arranged by their similarity of species composition, with overlay of TWINSPAN group boundaries at level 1, 00 and 01. Plots shown for (a) fishing method, (b) area and (c) depth range (when is stress<20 the ordination is useful).

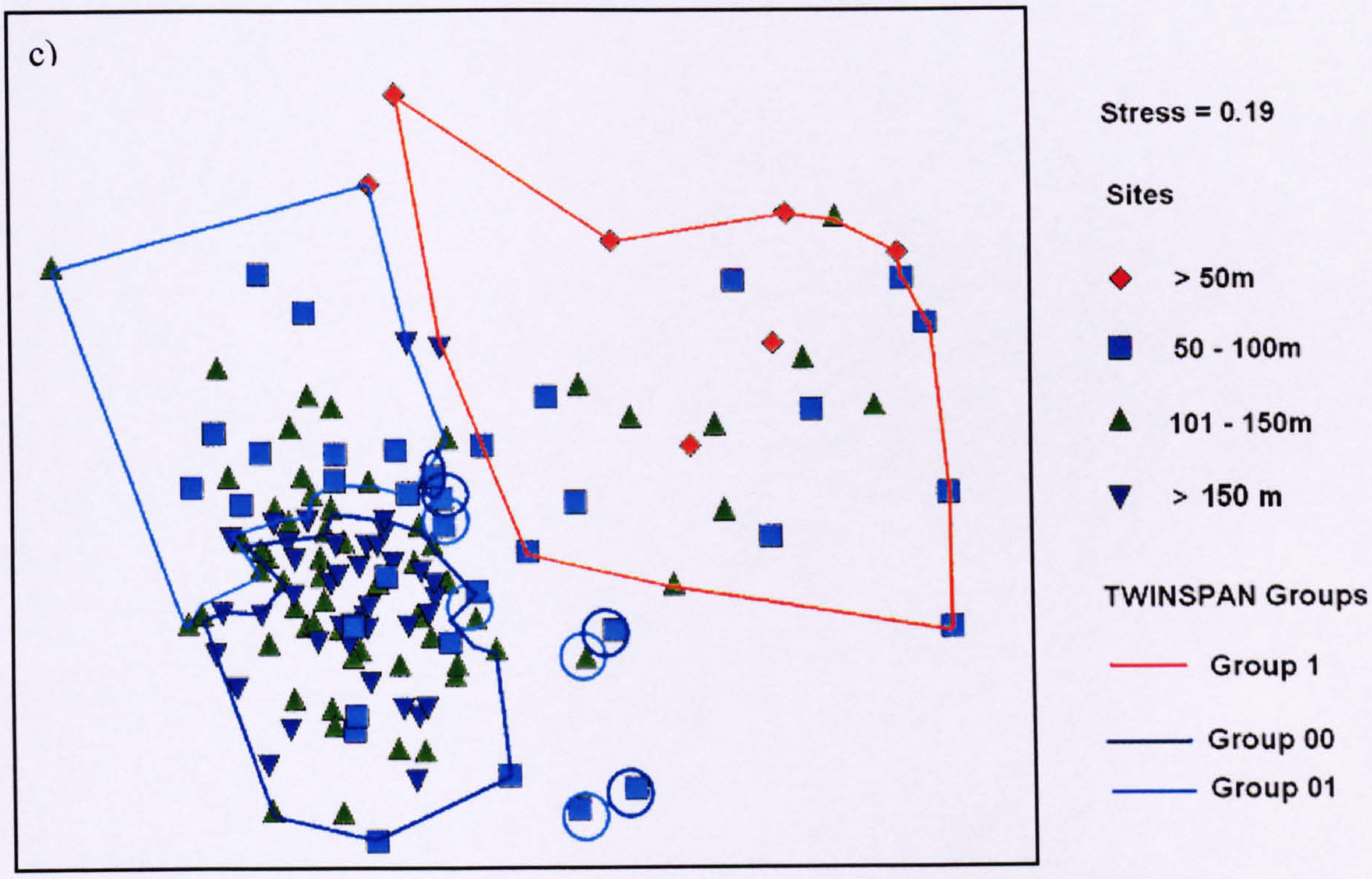


Figure 5.9 – (continued).

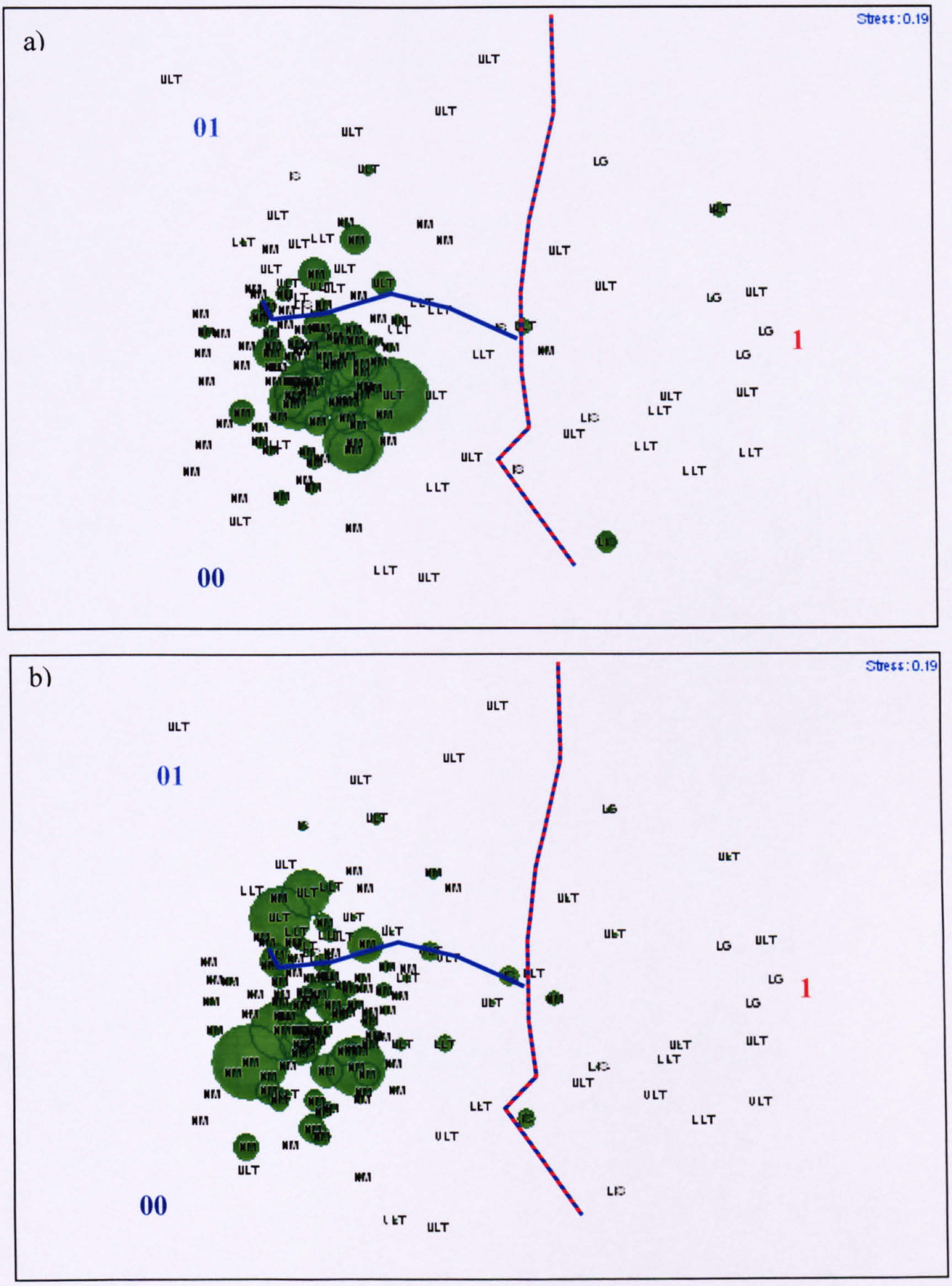


Figure 5.10 - MDS plots of stations arranged by their similarity of species composition. Bubble size represents relative density of representative indicator species, chosen from each TWINSPAN group. Simplified TWINSPAN groups (00, 01, 1) are illustrated. Plots shown for TWINSPAN group 00; a) O. Pleuronectiformes and b) F. Gadidae, group 01; c) *F. quadrangularis*, d) *V. mirabilis* and e) *P. phosphorea*, group 1; f) *M. rugosa* and g) *P. pulvillus*. (IS - Inner Sound (N), LIS – Inner Sound (S), NM – North Minch, LLT – lower Loch Torridon, ULT – Upper Loch Torridon and LG – Loch Gairloch) (when is stress<20 the ordination is useful).

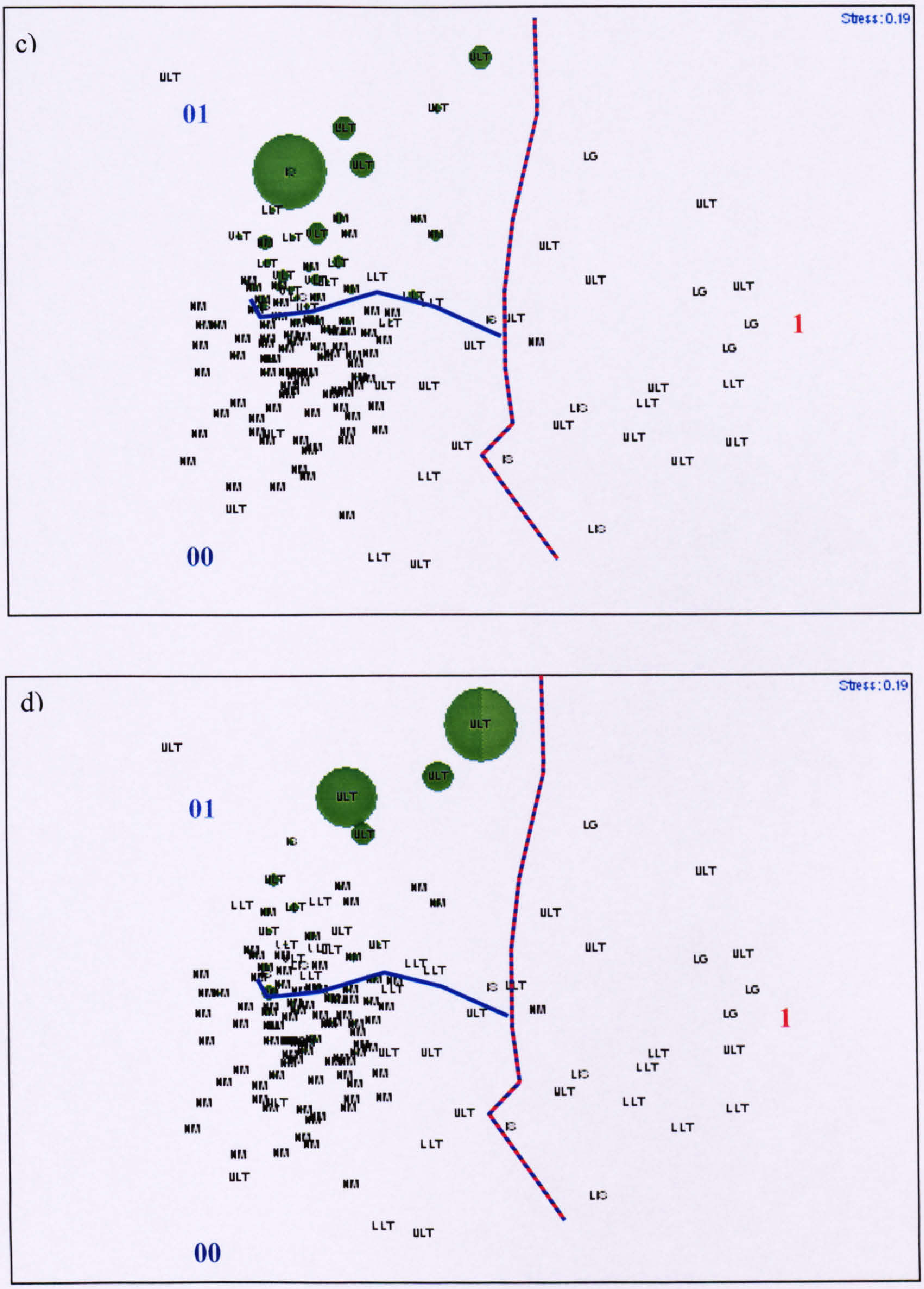
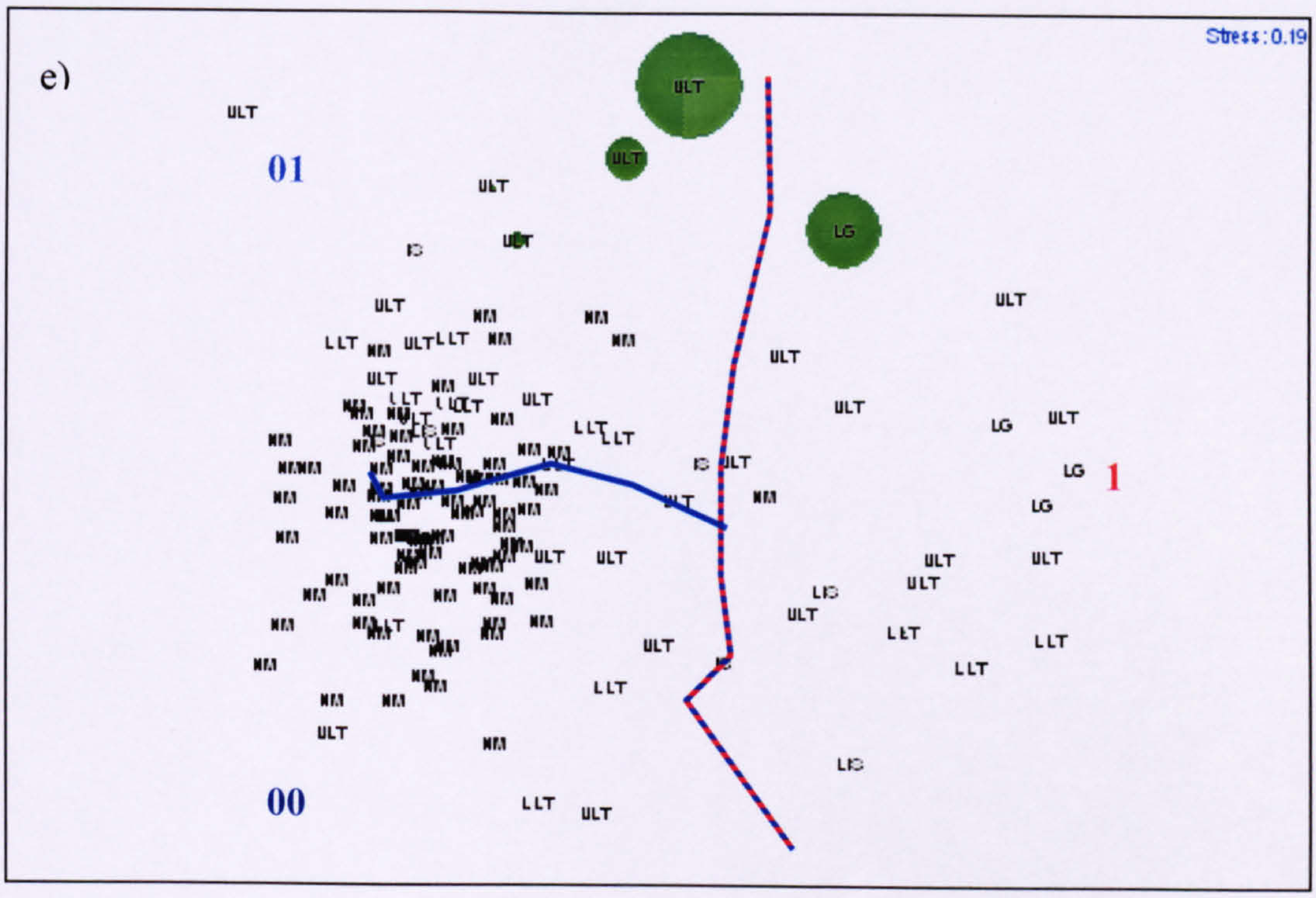


Figure 5.10 - (continued).



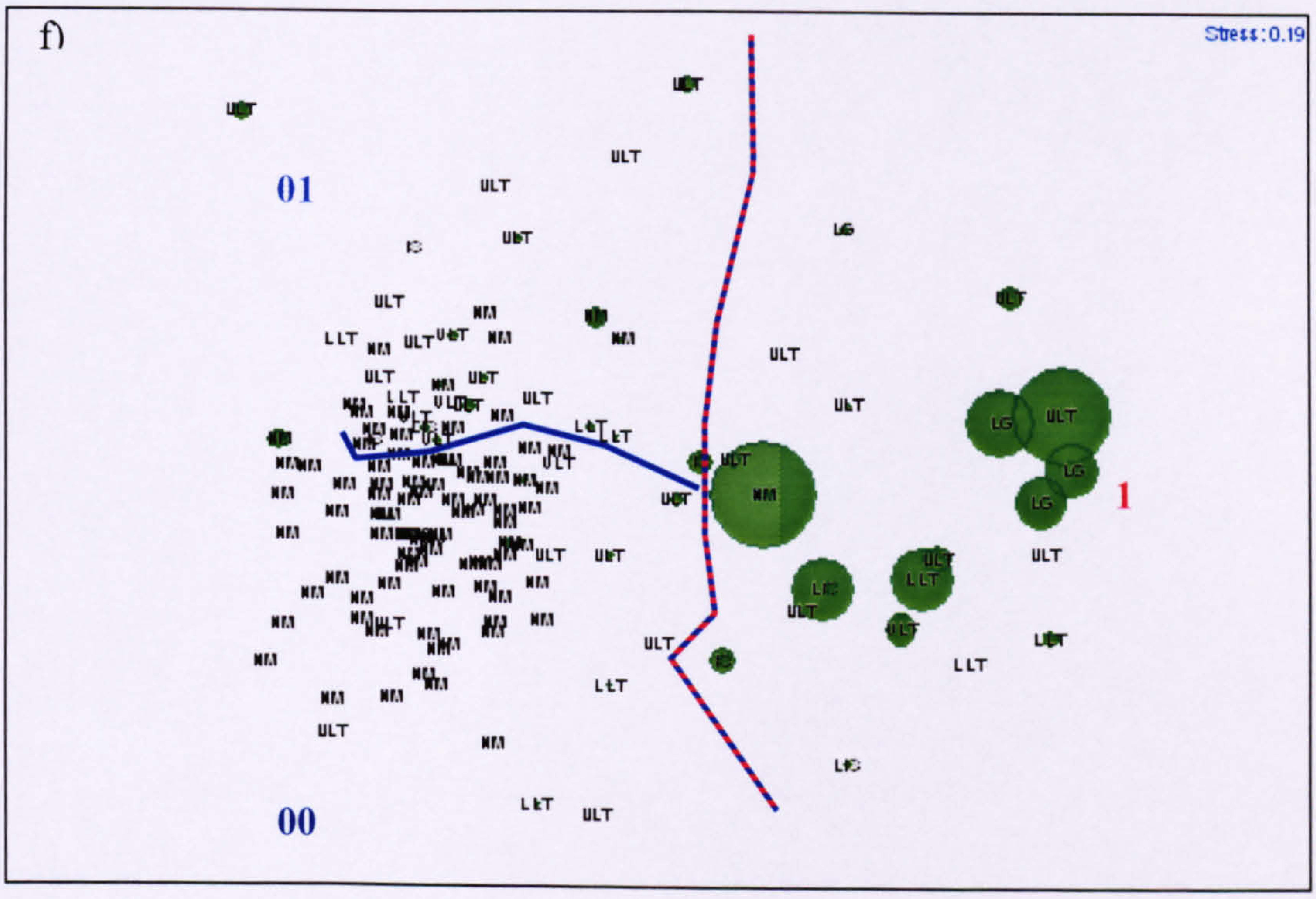


Figure 5.10 - (continued).

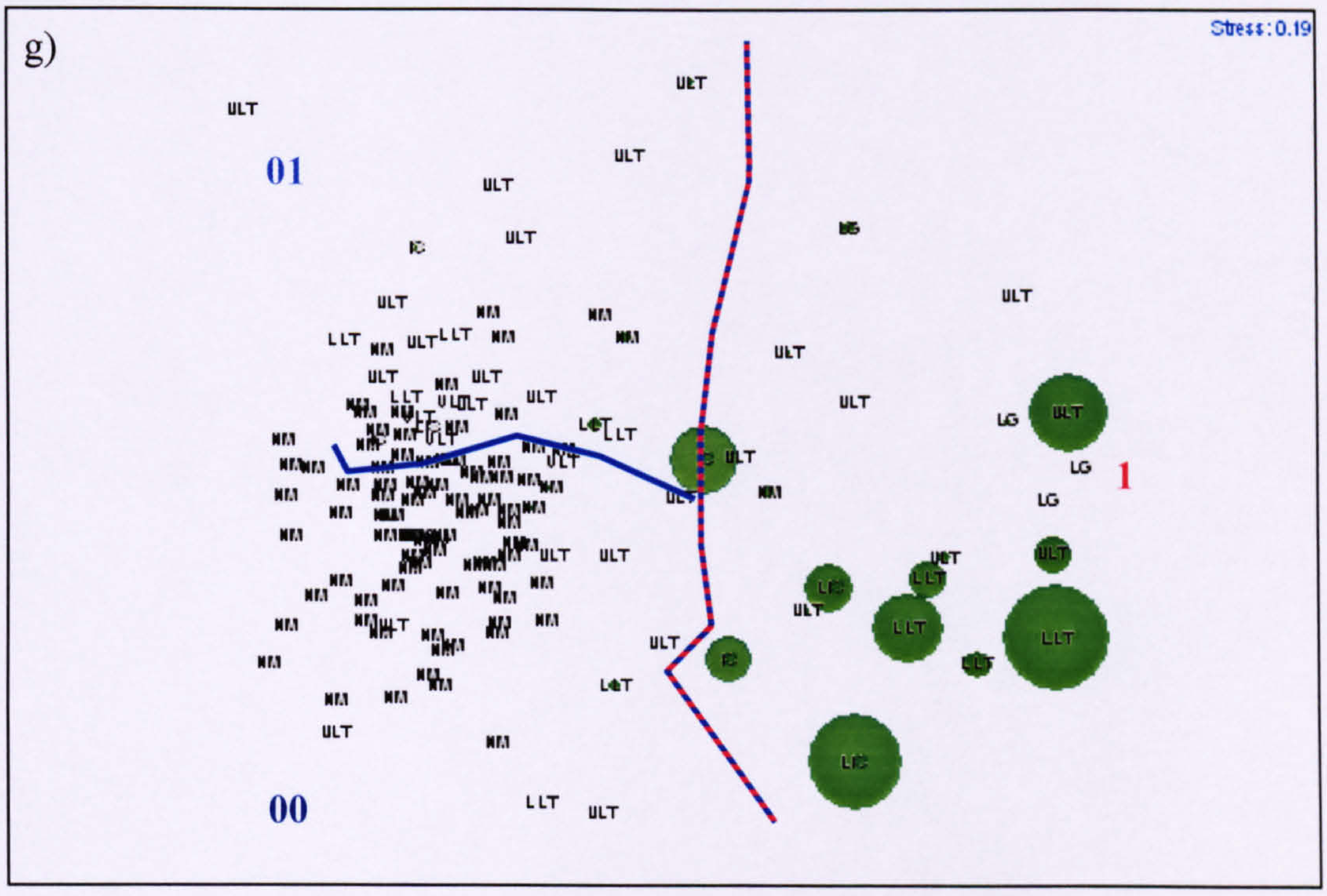


Figure 5.10 - (continued).

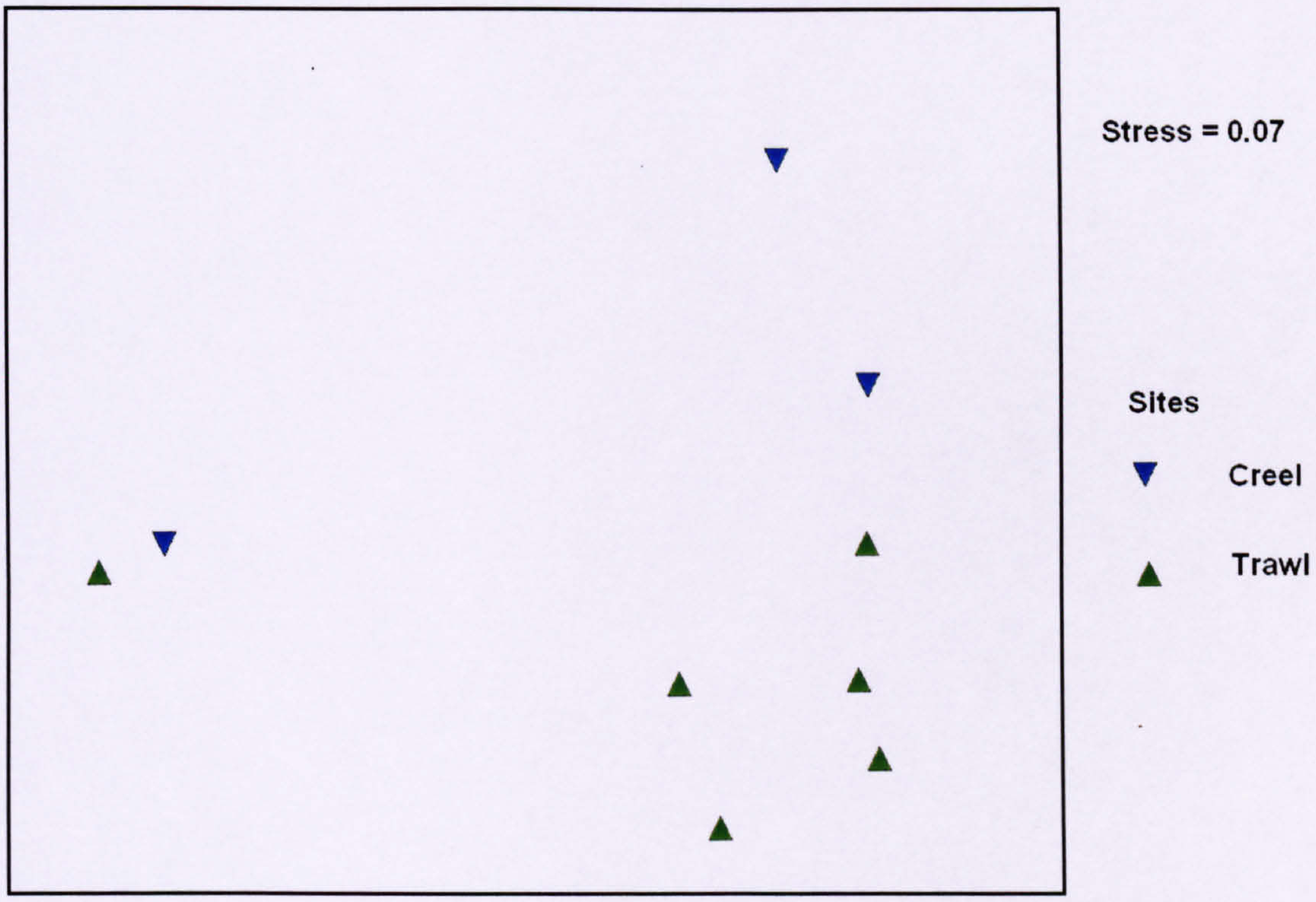


Figure 5.11 - MDS plots of species composition from areas where permitted fishing methods differ but environmental conditions were similar (when is stress<10 the ordination is good).

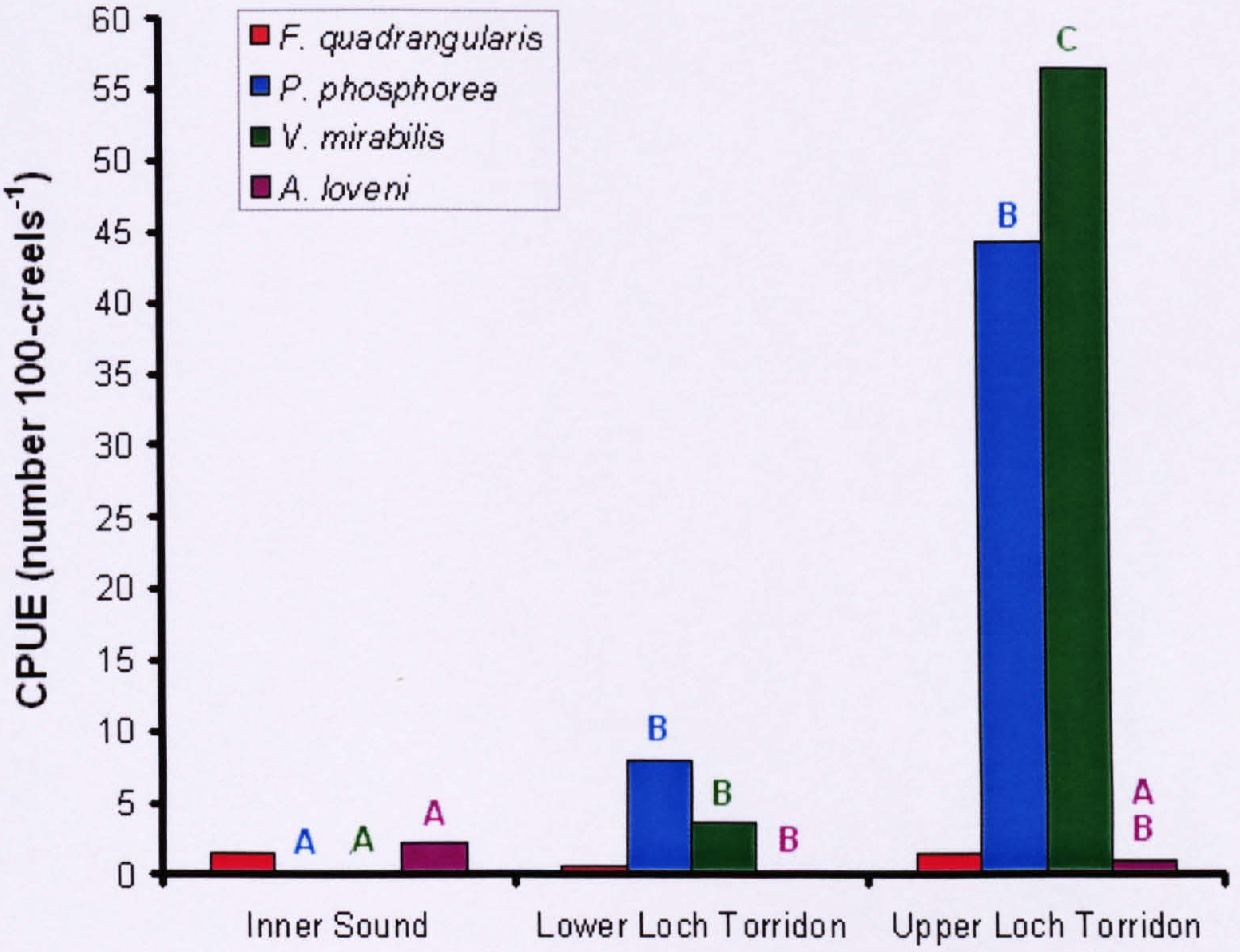


Figure 5.12 - Mean CPUE (number 100-creels^{-1}) of *F. quadrangularis, V. mirabilis, P. phosphorea* and *A. loveni* from different areas. Within each species different letters indicate significant differences ($P<0.05$).

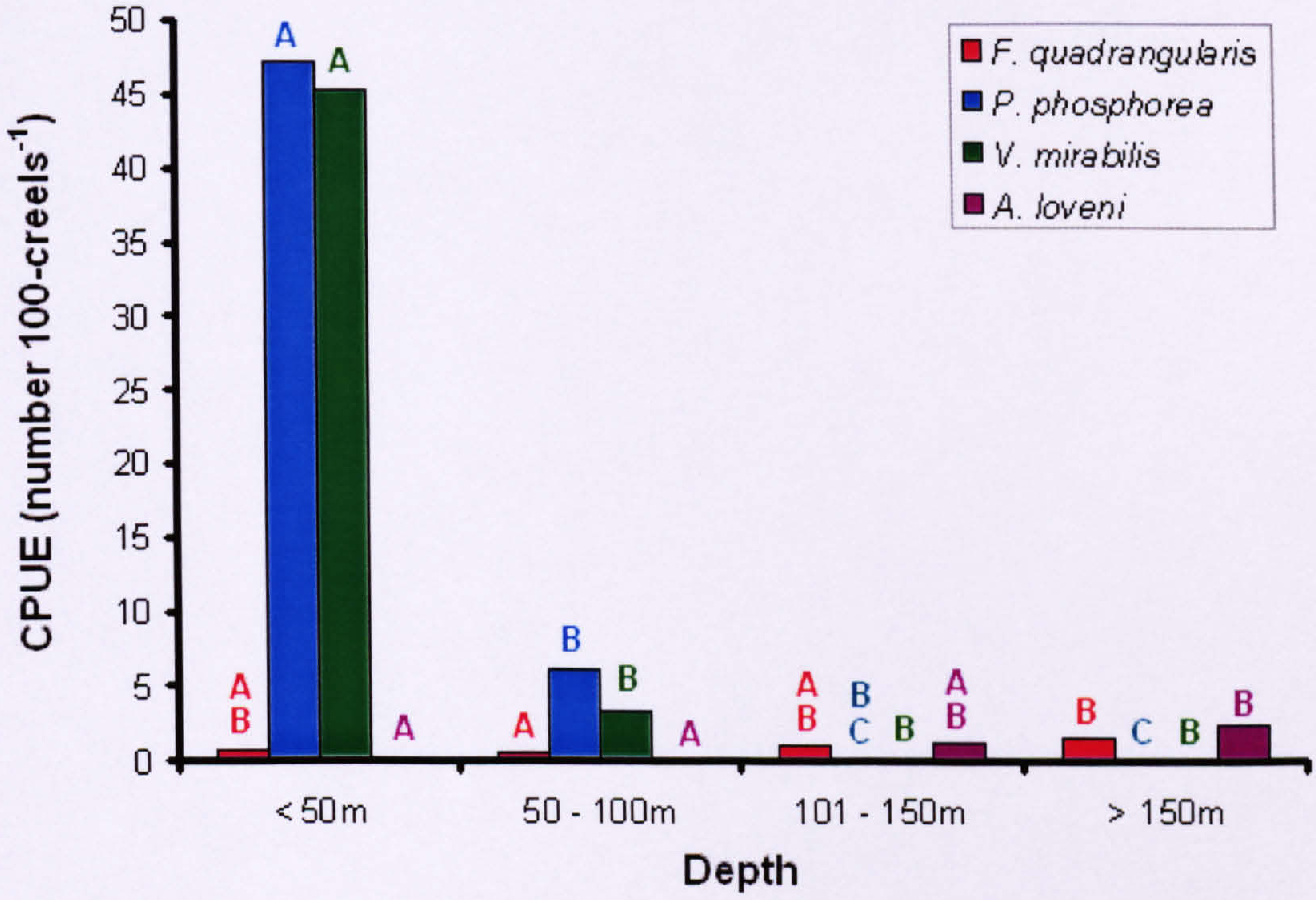


Figure 5.13 - Mean CPUE (number 100-creels^{-1}) of *F. quadrangularis, V. mirabilis, P. phosphorea* and *A. loveni* from different depths. Within each species different letters indicate significant differences ($P<0.05$).

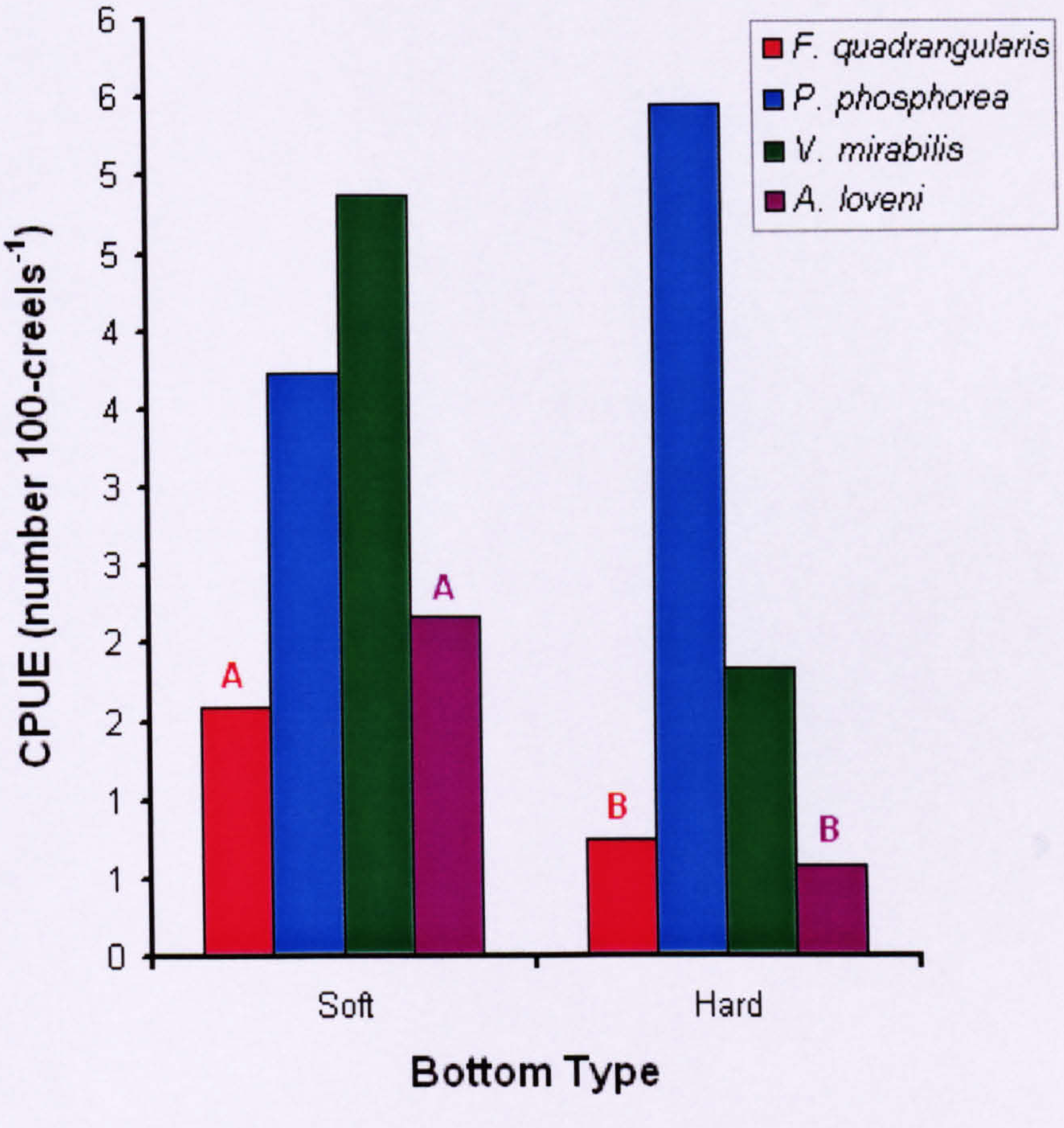


Figure 5.14 - Mean CPUE (number 100-creels^{-1}) of *F. quadrangularis, V. mirabilis, P. phosphorea* and *A. loveni* from different sediment types. Within each species different letters indicate significant differences (P<0.05).

Chapter 6

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6 BYCATCH COMPOSITION AND SURVIVAL

6.1 Introduction

Discarding bycatch is important in world fisheries, accounting for around 25% of the total world catch (Alverson *et al.*, 1994). Discards from a commercial fishing vessel can include animals of the target species that are undersized or above quota, or other species of no commercial value. Bycatch wastes valuable resources leading to endangered or rare species being threatened and large-scale ecosystem changes. Discard patterns can be highly variable due to differences in sociological, environmental and biological factors (Catchpole *et al.*, 2005). This waste of potential food resources has led to ecological, social and economic arguments to decrease bycatch, highlighting the issue to both the industry and public (Harrington *et al.*, 2005). Several international agreements promote the reduction of discards by operational and technological methods (FAO, 1995; UN, 1995).

There is particular concern regarding shrimp fisheries, in which the bycatch may be five to ten times greater than the catch of the target species. *Nephrops* trawl fisheries are ranked fifth in the world with respect to discard ratio (number of bycatch to number of target species) (Alverson *et al.*, 1997). Levels of discards from trawl fisheries vary with differing bathymetries and hydrographic conditions. In the Clyde Sea Area, the *Nephrops* trawl fishery discards an average of 9 kg of unwanted catch per 1 kg of *Nephrops* landed (Bergmann *et al.*, 2002a). Bycatch is often made up of commercially important species; for example, whiting accounted for an estimated 72% of discards by weight in the northeast English coast *Nephrops* fishery (Catchpole *et al.*, 2005).

Unlike trawling, crustacean creels are thought to be highly size- and species-selective (Miller, 1990), implying that the amount of bycatch of both target and non-target species is significantly lower than for trawl fisheries. For example, discards in the USA shrimp trawl and bottom trawl fishery account for 46.9% and 25.1% of catch (by weight), compared to the creel fisheries where only 4.5% of the catch is discarded (Harrington *et al.*, 2005). There is very little quantitative information on the levels of discard from *Nephrops* creel fisheries. This is particularly true for non-target species, for which only two small studies appear to have been undertaken; one reported in Mason *et al.* (2002) as part of the MSC assessment of the Loch Torridon *Nephrops* fishery and another by Kinnear *et al.* (1996) in Badentarbet Bay, Ullapool.

Bycatch can be reduced by a number of means, with a reduction in fishing effort being the simplest. Gear modifications can also reduce bycatch, for example, by an increase in mesh size and change of mesh shape (Campos *et al.*, 2003). Technical measures, such as the insertion of a panel of square mesh in a trawl net, have been shown to reduce the catch of juvenile fish by up to 80% without a reduction in prawn catch (Briggs, 1992). A change to more selective fishing gears is also a useful tool in reducing bycatch, examples of this include conversion of trawl fishers to creels, and switching from dragged gear to groundfish longlining (NMFS, 2003).

Closing defined areas to fishing is a method of protecting a proportion of the populations of target and non-target species from harmful effects of fishing. However, although seasonal and permanent closures are important in protecting certain habitats and, in some cases, breeding stocks of target species, closures may result in increased effort intensity outside the closed areas, with concomitant environmental damage (Childress, 1997). Managing discards can also be a method of reducing bycatch. One example of this is banning the discard of commercial fish species, which has been shown to decrease discards and act as an incentive to adopt more selective fishing techniques, so only the larger more valuable fish are caught (Catchpole *et al.*, 2005).

Survival of *Nephrops* discards from trawlers has been well studied, using aquaria or cages on the seabed (Gueguen & Charuau, 1975; Chapman, 1981; ICES, 1997; Ulmestrand, 1997; Ulmestrand, 1998; Castro *et al.*, 2003) or tagging studies (Chapman *et al.*, 2000) to monitor mortality levels. Chapman (1981) listed several factors affecting discard survival:

- Damage during fishing and landing,
- Effects of changes in temperature, salinity, pressure and light intensity during ascent and decent,
- Exposure in air on deck (limited during creeling), and
- Predation by seabirds, fish and other animals during decent and on the seabed.

Previous studies on *Nephrops* discard survival have focused on trawl-caught animals, showing mortalities ranging from 23–70% (Gueguen & Charuau, 1975; Chapman, 1981; ICES, 1997; Ulmestrand, 1998; Castro *et al.*, 2003). Lower mortality levels have been observed in animals that were able to escape from trawl nets. For example, Ulmestrand *et al.* (1998) observed 19% mortality of escapees, compared with 69%

mortality of discards. Survival of non-target species discarded in *Nephrops* trawl fisheries varies markedly among taxa. Long-term mortality of *Asterias rubens, Ophiura ophiura* and *Liocarcinus depurator* was 22–96%, 100% and 26%, respectively (Bergmann & Moore 2001a,b).

Studies on the survival of creel-caught animals have been largely limited to using them as control animals when looking at the survival of trawl-caught animals (Harris *et al.*, 1997; Ulmestrand *et al.*, 1997). A reason for this is that it is generally assumed that the discard mortality of creel-caught animals is very low. Chapman (1981) observed that 97% of *Nephrops* survived for 9 days after being caught and Ulmestrand *et al.* (1998) showed discard survival from creels to be similar to that of escapees from the trawl fishery (*ca* 81%). There have been no studies of the mortality of the other species caught as bycatch in *Nephrops* creel fisheries. It is known that creeling causes little physical damage to non-target benthic species (Eno *et al.*, 2001). However, fish with swim bladders may die as a result of changes in hydrostatic pressure.

A major concern about the survival of both trawl- and creel-caught animals after they have been returned is the effect of light-induced retinal damage. Brief exposure (<1 min) of the *Nephrops* eye to light can damage a large proportion of the retina and dioptric apparatus (Gaten *et al.*, 1990). Blind Norway lobsters can survive, grow and reproduce as successfully as fully sighted individuals (Chapman *et al.*, 2000), with loss of sight having little effect on their behaviour (Richardson, 1996).

Another important aspect of discard (target and non-target) survival is predation by seabirds, fish and other animals during decent and on reaching the seabed (Chapman, 1981). Evans *et al.* (1994) studied aspects of this in relation to the *Nephrops* trawl fishery, concluding that previous studies had overestimated *Nephrops* survival from trawling, with the true figure being close to 0%. Mortality of discards from trawlers due to seabird predation has been estimated from 14% to 70% (Blaber & Wassenberg, 1989; Berghard & Rosner, 1992; Wieczoerek *et al.*, 1999; Bertellotti & Yorio, 2000; Hill & Wassenberg, 2000; Martinez-Abrain *et al.*, 2002). Walter & Becker (1997) found that seabirds following a brown shrimp (*Crangon crangon*) trawler took 41% of flatfish, 79% of roundfish, 23% of invertebrate species and 10% of undersized shrimps. There are large differences among these studies in the estimated mortality of discards due to seabirds. This is not surprising, considering the variations in discard practices between commercial fishing fleets, as well as season, quantities of discards and methods used to compile data (Garthe & Huppop, 1998). It has been observed that seabirds took *Nephrops* discarded from trawlers when discarded individually, but when discarded in conjunction with other species such as roundfish they were taken to a much lesser extent. The least-preferred discard species were the fast-sinking *Echinus esculentus* and *Buccinum undatum* (Wieczorek *et al.*, 2000).

It is possible that there is significant mortality of discards due to pelagic or demersal predators (Wassenberg & Hill, 1990; Hill & Wassenberg, 1990). This may be particularly true if the discard species have been injured or are returned to grounds where conditions are unsuitable (Evans *et al.*, 1994).

The physiological damage caused to both discarded and landed animals can have adverse effects on their long-term survival. Harris *et al.* (1997) found that discard species caught by trawling had high levels of physiological disruption and stress (dehydration, reduced blood volume, increased blood protein concentrations, respiratory and metabolic acidosis, high blood lactate and glucose concentrations) compared with those caught by creeling. Such debilitation is of particular concern in *Nephrops* creel fisheries, where the marketable animals are usually exported live.

The perception of high survival in creel-caught discards, owing to less physical and physiological damage than in trawl-caught animals (Chapman, 1981; Harris *et al.*, 1997), has meant that there has been little research in this area. Discard mortality in creel fisheries needs to be assessed in relation to predation by seabirds, and predation by fish and other animals during descent and on the seabed. Predation of creel discards may be significant, since discards are thrown back individually as they are removed from the creels, thereby allowing predators to 'pick off' individuals on the surface, in the water column and on the bottom.

To make informed management decisions regarding the *Nephrops* fishery there is a need to assess bycatch levels associated with the *Nephrops* creel sector. The present study aims to give an accurate description of the species caught as bycatch within parts of the *Nephrops* creel fishery and how these species vary geographically and seasonally, both in type, number and biomass. Survival of both target and non-target species is of particular importance with regards to the management plan applied in that component of the Torridon fishery that is subject to MSC certification (Chapter 1; Section 1.3.5). Within this component of the fishery, ovigerous females are discarded, and the smaller animals within the catch are discarded at a larger size than in other similar fisheries, leading to a higher proportion of discards. Herein, bycatch survival has been investigated only at a pilot study level, providing some general conclusions and highlighting possible areas of future study.

6.2 Methods

6.2.1 Study sites

Descriptions and the location of the Torridon and Loch Fyne fisheries are provided in Chapter 1, Sections 1.7.1 & 1.7.2, Figures 1.7, 1.8 & 1.10.

6.2.2 Sampling procedure

Bycatch data collection was undertaken at the same time as *Nephrops* catch sampling, therefore the sampling procedure was as described in Chapter 2, Section 2.2.1.

6.2.2.1 Non-target species

An attempt was made to record the number of each species in every fleet of creels fished in a day; however, this was not always possible due to the volume of bycatch in relation to the high work rate of the commercial fishermen. This led to some fleets not being examined. Bycatch assessments were carried out in conjunction with *Nephrops* catch analysis (Chapter 2). For this reason it was necessary for the fishermen to keep all the bycatch in a container during the hauling process, following which species were identified and counted. The sampling procedure did not enable the quantification of pandalid shrimps in the Torridon area (they could easily fall through the meshes of the creel and were small and easily overlooked). In Loch Fyne some fleets of creels contained large numbers of *Munida rugosa*, *Pagurus bernhardus* and *Buccinum undatum*, requiring, in some cases, an estimate of their number to be made rather than a precise count. The creels used at each location were of the same design, except that those in the Torridon area had escape gaps and those in Loch Fyne did not.

6.2.2.2 Target species

Sampling of target species as described in Chapter 2, Section 2.2.1.

6.2.2.3 Biomass estimation

To express results in terms of biomass as well as number, it was necessary to obtain average weight values for all non-target discard species. This was achieved by weighing a representative sample of each of the most common species from the Torridon and Loch Fyne areas. For less common species, which comprised a small proportion of bycatch weight, it was necessary to collect biomass data from material caught using a research vessel (RV *Aplysia*, University Marine Biological Station Millport), fishing in the vicinity of the Isle of Cumbrae, Clyde Sea Area. In the case of a few species rarely caught, and for which no weights were available, biomass was estimated for each from that of similar species (Table 6.2).

To provide biomass estimations for *Nephrops* landings and discards it was necessary to convert the length-frequency catch data of males and females using the formulae below:

Males

$$W = 2.8 \times 10^{-4} CL^{3.24}$$

Females

$$W = 8.5 \times 10^{-5} CL^{2.91}$$

where W is weight (g) and CL is carapace length (mm) (ICES, 1999).

6.2.3 Pilot survival studies

6.2.3.1 Predation by seabirds

Mortality of target and non-target species from seabirds has been quantified by observing numbers of each species eaten by seabirds and the number that were able to escape during normal commercial fishing, enabling the percentage mortality to be calculated. This was possible in the creel fishery because the discarding practice allowed the fate of each animal to be assessed. In addition, the wind speed estimated on the Beaufort scale by the fisherman was recorded.

6.2.3.2 Post-discard behaviour

To assess post-discard behaviour of *Nephrops* (an indication of their susceptibility to predation) individuals caught in creels were returned to the sea bed and their behaviour was observed for a period of 10 min. *Nephrops* were caught in Loch Sween using commercial creels during November 2004. Once on the surface, they were immediately placed into a keep creel with all the exits sealed and returned to the bottom by divers. On the sea bed, animals were carefully removed from the keep creel and were followed for 10 min, during which time a qualitative assessment of their behaviour was made. The large size of animals caught at the site made it necessary to use animals larger than those that would normally be discarded by the commercial fishery.

6.2.3.3 Recapture of discards

Assessment of the recapture rate of discarded *Nephrops* provides an indication of their long-term survival. To achieve this, a tag and recapture study was undertaken. Initially it was necessary to find an adequate method for tagging the animals. Previously Chapman *et al.* (1989) used streamer tags successfully for tagging *Nephrops*. These tags had the advantage of being clearly visible and were not lost during the moult; however, aquarium trials indicated that animals tagged using this method (37-53 mm CL; 20% female) suffered 100% mortality, perhaps because the tags had been inserted too deeply (Figure 6.1). Another two types of tagging method were tested; tail notching and cable ties, both of which were found to have no adverse effect on the animals, when compared with control animals (Figures 6.1 & 6.2). Tail notching had the advantage of the mark being left after the moult, whereas cable ties would be lost. Tail notches were considered to be insufficiently conspicuous for fishermen to recognise reliably during the discarding process, so cable ties were chosen for the present study.

The trial was undertaken from a commercial creel vessel in the Inner Sound. Animals that would normally have been discarded were immediately tagged by placing a cable tie around the proximal part of each cheliped. The ends of each tie were removed so as not to impair the movement of the discarded animal. The tagged animals were then placed into a 'tube' so as not to be damaged by other animals, and released after the fleet of creels had been hauled. In total, 542 animals were tagged and released into different areas in the Inner Sound. The trial was undertaken at the beginning of September 2005 and continued until the end of November 2005 to coincide with a time when very few creel-caught animals of either sex are in a post-moult condition (Chapter 2; Section 2.3.1) and therefore tag loss would be minimised. All boats fishing within the Inner Sound were supplied with a logbook in which the location, date and number of any recaptured animals were recorded.

6.2.4 Statistical analysis

6.2.4.1 Bycatch composition

The mean and standard deviation of the biomass and abundance of non-target discards from each area were calculated. Catch rate of these species is expressed as catch per unit effort (CPUE, as number 100-creels^{-1}).

6.2.4.2 Community composition analysis

Two-Way Indicator Species Analysis (TWINSPAN: Hill, 1979) was used to classify the stations in terms of the species composition. Five pseudo-species cuts (CPUE (number 100-creels^{-1})) of: 0, 3, 5, 10 and 50 were used.

Multivariate differences in species composition in relation to area, depth range and sediment type were analysed with analysis of similarities (ANOSIM), using Bray-Curtis similarity indices calculated from fourth-root-transformed data and the maximum permutations of 999 (Clarke & Warwick, 1994). Multiple pairwise comparisons following ANOSIM were made with a Bonferonni adjustment, dividing the normal significance level (0.05) by the total number of pairwise comparisons, to protect against Type I error. Non-metric Multidimensional Scaling (MDS) ordination was carried out to give a visual representation of the similarities among stations. The contribution of individual species to the Bray-Curtis similarity was analysed with SIMPER (Clarke & Warwick, 1994).

6.2.4.3 Pilot survival studies

Differences in *Nephrops* discard mortality by seabirds at different wind speeds was tested by a chi-squared test of numbers eaten or not at different wind speeds. The relationship between proportion eaten and wind speed was also tested with Spearman rank correlation. The relationship between *Nephrops* mortality and the number of seabirds present was tested by Spearman rank correlation.

6.3 Results

6.3.1 Bycatch composition

From the 17,809 creels examined for bycatch, 52 species or taxonomic groups were identified. In addition to the species described, some *Loligo forbesi* and *Scyliorhinus canicula* eggs were observed but not recorded.

The most common species observed included five crustacean species – *P. bernhardus, Cancer pagurus, L. depurator, Carcinus maenas* and *M. rugosa* as well as the whelk, *B. undatum* and the gadoid, *Trisopterus minutus*. The large standard deviations reflect the different bycatch compositions observed both among and within areas (Table 6.1).

A large proportion of the catch in *Nephrops* creels comprised non-target species, with an average of 1.1 kg of non-target discards per 1 kg of landed *Nephrops*. *Nephrops* discards accounted for a much smaller proportion of the catch, with an average of 0.2 kg kg^{-1} (Table 6.3). A comparison between samples from Loch Fyne and the Torridon area show considerable differences in the biomass of both target and non-target species discarded (Figure 6.4). The CPUE of non-target species discarded in Loch Fyne (15.2 kg 100-$creels^{-1}$) was over twice that of the Torridon area (6.5 kg 100-$creels^{-1}$) (Table 6.4). The opposite was true of *Nephrops* discards, with a CPUE of 2 kg 100-$creels^{-1}$ of *Nephrops* destined for discard in Torridon, compared with 0.7 kg 100-$creels^{-1}$ in Loch Fyne (Figure 6.4).

The biomass of non-target discards was found to vary among sites in the Torridon area, with the greatest biomass observed in lower Loch Torridon (Table 6.3). *Nephrops* discards also varied among locations in Torridon, with a similar average CPUE of discard category animals in the Inner Sound (1.8 kg 100-creels^{-1}) and lower Loch Torridon (1.6 kg 100-creels^{-1}), compared with Upper Loch Torridon where the biomass of *Nephrops* discards was more than twice as high (3.8 kg 100-creels^{-1}) (Figure 6.5).

Differences in the biomass of non-target discards among areas reflect the species composition of the bycatch. Crustaceans were the most common group in all of the areas except the Inner Sound, where Chondrichthyes was the most important in terms of biomass (Figure 6.5). A large biomass of molluscs was observed in Loch Fyne, but were found in much smaller quantities in the Torridon area. In addition, both fish species groups (Chondrichthyes, Osteichthyes) were important in terms of biomass within the Loch Fyne bycatch. The proportion of bycatch consisting of Echinodermata and Cnidaria was low in all areas (Figure 6.5).

Several species occurred in large quantities in terms of biomass, with catch varying among habitats. Large biomasses of *P. bernhardus, C. maenas* and *B. undatum* were observed in the two inner loch environments, Loch Fyne and Upper Loch Torridon. In Loch Fyne there was also a high biomass of *L. depurator* and *M. rugosa* in the bycatch. A large biomass of *C. pagurus* was observed at the deeper sites, lower Loch Torridon and the Inner Sound. A high biomass of *Scyliorhinus canicula* was recorded in the Inner Sound and Loch Fyne and *T. minutus* was fairly ubiquitous in the catch throughout the areas (Figure 6.6).

Bycatch composition was assessed in terms of abundance, as well as biomass, since the importance of smaller species may be under-represented when considered in terms of biomass alone. This is evident when comparing the species composition of bycatch in two different habitats: Upper Loch Torridon and the Inner Sound (Chapter 1; Section 1.7.1). A comparison of the abundance and biomass of non-target discards in these areas shows similar proportions of non-target species in terms of biomass (Table 6.7). When expressed in numbers, an average CPUE of 39 individuals per hundred creels (100-creels^{-1}) was found in the Inner Sound compared with 335 in Upper Loch Torridon, an almost 10-fold difference (Table 6.7; Figures 6.7 & 6.8). The reason for this difference is that in the Inner Sound, *S. canicula* accounted for 57.4% of non-target bycatch by biomass, but just 11.1% by abundance. Conversely, the biomass of Cnidaria accounted for only 4.7% of the biomass of non-target bycatch in Upper Loch Torridon, compared with 35.9% by abundance (Figures 6.6 & 6.9). This difference is also reflected in other smaller animals, such as *M. rugosa* and *P. bernhardus*, which contributed to a higher proportion of the catch in terms of number, compared with larger animals, including *C. pagurus*, which contributed to higher proportions of the catch in terms of biomass (Figures 6.6 & 6.9).

6.3.2 Non-target species community composition analysis

The first division of the TWINSPAN analysis (eigenvalue = 0.543, good division; Table 6.9) separated the sites into two distinct groups, with group 0 comprising almost exclusively shallower loch sites (Loch Fyne and Upper Loch Torridon), and group 1 containing deeper loch sites (lower Loch Torridon and the deep Inner Sound sites). Both groups contain slightly higher numbers of sites with soft ground than coarser ground (Table 6.10).

Indicator species identified by TWINSPAN (most important species in determining the groups) were *B. undatum, C. maenas, P. bernhardus, L. depurator* and *M. rugosa* in group 0 and *P. pulvillus* and *C. pagurus* in group 1 (Table 6.11), reflecting the different environments in each group.

The second split of group 0, into groups 00 and 01 (eigenvalue = 0.206, acceptable division; Table 6.9) produces a small group dominated by shallow sites from Upper Loch Torridon (group 00). The other larger group contains almost exclusively sites from Loch Fyne with the majority being shallow water sites (group 01). As with the first split, both groups contain larger numbers of sites with soft ground (Table 6.10).

Virgularia mirabilis was the only indicator species for group 00 and no indicator species were identified for group 01, highlighting the importance of this species in separating these two similar environments (Table 6.11).

The second split of group 1 into groups 10 and 11 (eigenvalue = 0.36, good division; Table 6.9) produces a group dominated by sites from lower Loch Torridon with a mixture of depths (group 10) and a group containing deep sites almost exclusively from the Inner Sound (group 11). The proportion of sediment types vary between groups, with group 10 containing a larger number of sites with hard ground and group 11 containing a larger number of sites with soft ground (Table 6.10).

The indicator species in group 10 were *C. pagurus, P. bernhardus* , *L. depurator* and *M. rugosa* , typical of a mixed sediment loch environment and in group 11 were *Asteronyx* *loveni* and *S. canicula*, typical of the deep muddy habitat of the Inner Sound (Table 6.11).

Non-metric multidimensional scaling (MDS) plots of species CPUE are shown in Figure 6.10, with all three factors (location, depth and sediment type) being displayed. In addition to the MDS plots, the boundaries for TWINSPAN groups 00, 01, 10 and 11 have been overlaid. Stations with similar epibenthic communities appear closer together on the plot.

Clear separations between the four areas are apparent, with the largest distance (and therefore difference in non-target bycatch composition) occurring between the shallow loch environments (Loch Fyne and Upper Loch Torridon) and the deep open sea area of the Inner Sound (Figure 6.10a,b).

The sites within TWINSPAN group 00 are densely clustered sites from Loch Fyne, with Upper Loch Torridon sites grouped around the Loch Fyne sites, showing the similarity in terms of non-target bycatch composition between the areas (Figure 6.10a). The dense clustering of Loch Fyne sites indicates the similarity in benthic assemblages found at similar depths (Figure 6.10b), but different ground types (Figure 6.10c).

The lower Loch Torridon and Inner Sound sites show a clear separation with two distinct clusters. Only a few sites from the Inner Sound are found within the lower Loch Torridon group (10) and only one lower Loch Torridon site is found within the Inner Sound group (00) (Figure 6.10a). This separation of these two locations is probably a reflection of different environmental conditions, with the Inner Sound sites located in deeper waters with softer sediments than those from lower Loch Torridon (Figure 6.10b,c). These sites are less densely clustered than the Loch Fyne site, indicating a wider variety of non-target bycatch.

Subjective assessments of the MDS plots were confirmed by analysis of similarities (ANOSIM), which showed significant differences in community composition among all areas. Significant differences were observed between all depth ranges except the two shallowest ones (<50 m and 50–100 m). No significant difference in community composition with ground type was observed (Table 6.12).

The importance of indicator species in producing the TWINSPAN groups is illustrated in Figure 6.11. The relative density of indicator species identified by TWINSPAN analysis has been plotted on MDS plots, showing the location of each site, with simplified TWINSPAN groups illustrated. The results show the importance of these species in defining the TWINSPAN groups. The largest numbers of *V. mirabilis* occurred in group 00, distinguishing Upper Loch Torridon from Loch Fyne (Figure 6.11a). The crab *C. pagurus* occurred in largest numbers in group 10 (Figure 6.11b). The brittle star *A. loveni* has been shown to be typical of deep-water muddy environments (Chapter 5), which is reflected in its abundance at the Inner Sound sites in group 11 (Figure 6.11c).

A SIMPER test was used to identify which species contributed most to the dissimilarity between the two sites. The average dissimilarity was high between all locations, depths and between sediment types, with the highest dissimilarity between the two shallow loch environments (Loch Fyne and Upper Loch Torridon) and the deep Inner Sound

(Table 6.13). This difference is represented by the distance of sites from these groups in Figure 6.10a. The bycatch in both of the two shallow loch environments (Loch Fyne and Upper Loch Torridon) contained large numbers of *P. bernhardus*, but the sites are distinguished by high numbers of *M. rugosa* in Loch Fyne and high numbers of *Virgularia mirabilis* and *Pennatula phosphorea* caught in Upper Loch Torridon (Table 6.13). *Munida rugosa* and *C. pagurus* were both important in distinguishing lower Loch Torridon, and the Inner Sound was separated by low numbers of most species except *T. minutus* (category includes some other gadoids), which occurred in higher numbers at this location (Table 6.13).

Large numbers of the species *M. rugosa, P. bernhardus, B. undatum* and *C. pagurus* were found within the shallower sites, with smaller numbers being observed in the deeper sites (Table 6.13). These species were also important in distinguishing sites with different sediment types, being found in larger numbers on hard ground, but this distinction is not as clear as that with depth and location (Table 6.13).

Considering all areas, no significant difference was found in community composition among sediment types (Table 6.12). There was an indication that there may be a difference in the Torridon sites (Figure 6.10c). This is likely to result from the method for defining sediment (Chapter 2; Section 2.2.1). An experienced fisherman is likely to have a good idea of the sediment type in his local area. However, the definition of 'hard' and 'soft' ground may vary between areas. For example, the sediment in the Torridon area is softer than that found in Loch Fyne and therefore what a fishermen defines as 'soft' in Loch Fyne may actually be considered 'hard' in the Torridon area.

For this reason it is necessary to compare differences in community composition with sediment type in each area separately.

MDS plots of community composition on different sediment types are illustrated in the (Figure 6.12 a) Torridon area and (Figure 6.12 b) Loch Fyne. A separation between hard and soft ground in the Torridon area is evident (Figure 6.12 a) (ANOSIM, R=0.223, *P*=0.001). In Loch Fyne, there was no clear separation between hard and soft ground (R=-0.047, *P*=0.842) (Figure 6.12 b), suggesting that defining this group by sediment type is not appropriate. It should be stressed that the terms soft and hard are relative: 'soft' is soft mud, 'hard' is muddy sediment with admixtures of coarser material (sand, shell, etc).

In the Torridon area, the average dissimilarity between hard and soft areas was high, with the higher numbers of all species identified caught on the hard ground. The abundances of *M. rugosa* and *C. pagurus* were particularly important contributors to the dissimilarity (Table 6.14).

6.3.3 Pilot survival studies

6.3.3.1 Predation by seabirds

The majority of seabirds following *Nephrops* creel boats in the Torridon area were herring gulls (*Larus argentatus*), with other species observed in much smaller numbers (Figure 6.15). Overall, the percentage of *Nephrops* discards taken by seabirds was 8.6%, with a significant increase in the mean percentage of animals eaten with increasing wind speed (Figure 6.13, χ^2=76.6, $P<0.001$; r_s=1, $P<0.05$). Pairwise comparisons showed mortality levels at Beaufort scale of 4 to be higher than those at 0, 1 and 2. In addition, mortality was higher at a Beaufort scale of 3 compared to 0 (Figure 6.13).

There was an indication that the proportion of *Nephrops* discards consumed increased with the number of sea birds present (Spearman rank correlation, r_s=0.691, P<0.05) (Figure 6.14). Individuals discarded further from the boat appeared to be more susceptible to predation, although this was not quantified. Numbers of *Nephrops* discards taken by seabirds within other fisheries appeared to differ. In Loch Fyne, for example, there appeared to be little mortality from seabirds throughout the year. This is a reflection of the low number of seabirds that follow the boat. In this area, there is a trawl fishery and this seems to be the focus of bird activity.

The mortality of other species attributable to seabird predation has also been investigated, with preliminary mortality estimates from two fleets of creels illustrated in Table 6.15.

6.3.3.2 Post-discard behaviour

Preliminary observation of *Nephrops* behaviour following discarding of five individuals, revealed a variety of behaviours, from apparent inactivity for 10 minutes, to rapid walking (appearing to the human observer as searching), followed by locating a *Nephrops* burrow opening and entering the burrow (n =2).

6.3.3.3 Recapture of discards

Of the 542 *Nephrops* that were tagged and released, during the three months of the trial, 7 were recaptured.

6.4 Discussion

For the two study areas (Torridon and Loch Fyne), for every 1 kg of landed *Nephrops* a further 1.1 kg of non-target species and 0.2 kg of *Nephrops* were discarded, with *Nephrops* landings accounting for 43.6% of the catch by weight. The discard level from the *Nephrops* creel fishery is much lower than *Nephrops* trawl fisheries, with *Nephrops* constituting, by weight, only 14–23% of the trawl catch in the Clyde Sea Area, 12% in the Farne Deeps (North Sea) and 13% in the Irish Sea (Evans *et al.*, 1994; Craeymeersh, 1994; Bergmann *et al.*, 2002a).

Bycatch composition, in terms of species composition and quantity (weight or number), was very variable among fleets of creels, in both the Torridon area and Loch Fyne. This is in contrast with findings from the trawl fishery, in which bycatch composition was similar within areas, but varied between areas (Bergmann *et al.*, 2002a). A possible reason for this difference is that the area of sea bed fished by the creel fishery is likely to be more varied, with creeling often undertaken near rocky ground or within the heterogeneous habitats of sea lochs, leading to different biotopes being fished on a relatively small scale. Furthermore, a trawl tow often covers a large area, and a fleet of creels samples only a relatively small area, which, in combination with the patchy distribution of epibenthic species observed in these habitats (Chapter 5), results in variability in bycatch from *Nephrops* creels.

Only a small proportion of the catch consisted of *Nephrops* discards, a reflection of *Nephrops* creels being highly selective for the larger animals in the population (Chapter 4). The proportion of *Nephrops* discards varied among areas, with over twice the weight of *Nephrops* discards from the Torridon fishery, when compared with Loch Fyne.

Superficially, this result seems to negate the assertion that escape gaps in creels in the Torridon fishery make these creels even more selective for larger animals, thus reducing the number of smaller animals in the creels (Livingstone, 2001). However, there is no information on how many small animals may have escaped at the sea bed in relation to those that have not. The underlying reason for the above difference is the different discarding policies between the two areas, with animals below 40 mm CL being discarded in the Torridon fishery compared with animals below 35 mm CL discarded in the Loch Fyne fishery. Furthermore, ovigerous females are also discarded in the Torridon fishery, but only during part of the study period in Loch Fyne. Sediment type is also likely to affect *Nephrops* discard composition, with areas of coarser sediments typically being characterised by smaller *Nephrops* at higher densities. A higher proportion of the catch will therefore tend to be discarded in areas of coarse sediment (Tully & Hillis, 1995). This consideration may account for variation in the proportion of discards within the Torridon area, with the highest proportions observed in Upper Loch Torridon, where the sediment is much more mixed than the lower loch or Inner Sound (Chapter 3).

The non-target discard composition varied both in terms of biomass and species composition between the Loch Fyne and Torridon fisheries, as well as among areas within the Torridon fishery. A similar result was also observed in the Clyde Sea Area trawl fisheries (Wieczorek *et al.*, 2000; Bergmann *et al.*, 2002a). In Loch Fyne, the biomass of non-target discards was over twice that in the Torridon area. The non-target discard composition is related to factors such as population density of the species in the area, their attraction to the bait and, following capture, their subsequent ability to escape. The sea loch environments contain a variety of sediment types, leading to a higher diversity of epibenthic species. In these areas the most common bycatch species were epibenthic carnivores and scavengers, with crustaceans being the most common group. This contrasts with the Inner Sound, which consists of large areas of soft mud, inhabited by fewer epibenthic species. The large proportion of the bycatch comprising fish species in the Inner Sound, with *S. canicula* being the dominant species by weight, may be a reflection of a lower abundance of mobile epibenthic invertebrates there.

Multivariate analysis of the species composition highlighted this difference, with non-target bycatch composition varying significantly in relation to location, depth and, in the Torridon area, sediment type. Similar habitat relationships were evident from towed TV surveys of the epibenthic populations in the Torridon and North Minch areas (Chapter 5). The most similar areas in terms of non-target bycatch composition were the two shallower loch sites (Loch Fyne and Upper Loch Torridon), but the high densities of the two seapen species *P. phosphorea* and *V. mirabilis* in Upper Loch Torridon distinguished these two areas. The next most similar area was lower Loch Torridon and the most different area was the Inner Sound.

The findings of the present study indicate that fish, in particular commercially exploited species, make up a small proportion of the bycatch in *Nephrops* creel fisheries. This observation is in marked contrast with a number of *Nephrops* trawl fisheries, in which the bycatch consists of a large proportion of fish, many of which are juveniles of commercially exploited species (Bergmann *et al.*, 2002a). For example, 61% of the bycatch in the Farn Deeps fishery by weight consisted of fish (Evans *et al.*, 1994), 72% of discards on the northeast English coast consisted of whiting (Catchpole *et al.*, 2005), undersized commercial fish accounted for up to 39% of the total catch in the Clyde Sea

Area (Bergmann *et al.*, 2002a) and in the Skagerrak and Kattegat fishery, less than 0.15 kg of undersized fish were caught per kg of landed *Nephrops* in the creel fishery, while 1.9 kg of undersized *Nephrops* and fish, were caught in the trawl fishery (Ziegler, 2006). The greater numbers of fish caught as bycatch within the trawl fishery, is likely to reflect trawlers focusing their effort in areas of open ground, similar to the Inner Sound, where lower numbers of epibenthic invertebrates would be present. Additionally, it is likely that fish are more susceptible to, and less likely to escape from, trawl gear.

Although habitat appears to be the major factor influencing non-target bycatch composition, differences between the Torridon area and Loch Fyne may also be due to differences in fishing gear and fishing practice. The average soak time of creels was 2.6 days in Loch Fyne, compared with 6.5 days in the Torridon area. The longer average soak time in the Torridon area is a likely result of the management restrictions (Chapter 1; Section 1.3.5) and more exposed fishing grounds resulting in vessels fishing for fewer days than in Loch Fyne. This longer soak time is likely to result in more of the non-target animals escaping from the creels (Mason *et al.*, 2002). In addition, the creels in the Torridon fishery deployed by fishermen abiding by the management agreement are fitted with escape gaps. The crabs *L. depurator* and *C. maenas* have both been observed to exit the creel through escape gaps (Chapter 4), with numbers of *L. depurator* being significantly reduced in a creel with escape gaps when tested in the field (Chapter 7). A greater proportion of *L. depurator* was observed in the bycatch in Loch Fyne than in any of the Torridon areas, suggesting that their reduced numbers in Torridon may be due partly to the presence of escape gaps.

A further possible reason for the higher proportion of non-target discards in Loch Fyne is the close proximity of this fishery to the trawl fishery (Chapter 3; Section 3.4). It is possible that the large bycatch of epibenthic scavengers in the Loch Fyne fishery may be a result of the large volume of animals discarded by the trawl fishery in the area, of which the majority sinks and becomes available to benthic scavengers (Wieczorek *et al.*, 1999). There has been a great deal of debate regarding the possible energy subsidy to benthic scavengers from trawl discards and the effect that this has on benthic scavenger populations. Groenewold & Fonds (2000) calculated that up to 13% of total annual secondary production of macrobenthos in the southern North Sea might result from trawl discards. Related studies include those of Kaiser & Spencer (1994), Evans *et al.* (1996), Castro *et al.* (1998), Wieczoreck *et al.* (1999) and Bergmann *et al.* (2002b). These studies have provided evidence that trawl discards temporarily augment the diet of scavengers, but the overall degree to which they contribute to their diet is still not fully understood. Therefore, while the larger proportion of non-target discards in Loch Fyne may be primarily an effect of habitat, the effect of trawl discards on the benthic scavenger population must not be ruled out.

The majority of previous fishery studies have assessed bycatch in terms of biomass; however, this may lead to the underestimation of the importance of smaller species in the bycatch. This was observed in the northern Clyde Sea Area trawl fishery where invertebrates were underestimated when analysed in terms of biomass alone (Bergmann, 2002a). The underestimation of certain components of the catch was particularly evident in the Torridon area with Upper Loch Torridon and the Inner Sound showing similar proportions of non-target species in terms of biomass. However, when expressed in terms of abundance, almost 10 times as many non-target animals were caught in Upper

Loch Torridon than in the Inner Sound. This result is a reflection of the different species caught in each area, with a large catch of small sea pen and crustacean species in Upper Loch Torridon and the majority of the biomass of non-target discards in the Inner Sound deriving from a few large dogfish (*S. canicula*). This observation highlights the importance of analysing the data in terms of numbers as well as biomass in order to illustrate the importance of species such as sea pens in the bycatch in certain areas (Chapter 5; Section 5.3.5).

There are a number of factors that can lead to the mortality of both target and non-target species discarded within the *Nephrops* creel fishery. Results from the pilot studies suggest that predation from seabirds is the major cause of mortality to *Nephrops* discards in the creel fishery. Although *Nephrops* discards appeared not to be the preferred food item for scavenging seabirds, they were still taken in large numbers, with an average of 8.6% by number taken in the Torridon area. There was also a strong correlation of increasing *Nephrops* mortality with wind strength, which is likely to be due to seabirds being able to gain more lift in stronger winds and therefore manoeuvre close to the moving vessel to take discards more easily. No predation by seabirds was observed in Loch Fyne, where very few seabirds were found in the vicinity of the vessel. This illustrates the need to assess seabird predation in a variety of fisheries. Two other small crustacean species *L. depurator* and *M. rugosa* were seen to be taken by seabirds, whereas seabirds appeared unable to consume large, fast sinking species. A result of the creels being pulled up from depth was that all *T. minutus* (and other small fish with swim bladders) were discarded dead and consumed by seabirds. Seabirds have been estimated to take a much larger proportion of the discarded catch in the trawl fishery than has been observed in the creel fishery (Evans *et al.*, 1994; Wieczorek *et al.*, 1999). The overall tonnage taken by the creel fishery will be low by comparison with the trawl fishery: a quantitative assessment awaits input from a wider number of sites than those studied to date.

Nephrops behaviour appeared to be unaffected following capture and subsequent return to the seabed, although there were few observations and further work on this aspect is warranted. Similarly, Richardson, (1996) observed no effect of blinding on the time *Nephrops* took to gain shelter or differences in behaviour when investigating or entering a shelter. Also, in contrast to the trawl fishery, where animals are often returned damaged and to an area distant from the fishing grounds (Evans *et al.*, 1994), creel-caught animals are discarded in the same area where they were caught, with damage usually being minimal. 1.3% of tagged animals were recaptured during the tagging trial. Chapman *et al.* (1989) observed a recovery rate of 17% from 1,215 animals over a 4–5 year period. If recoveries were made at a constant rate, this would equate to 1% recaptured during the same time period as the current trial (3 months), although it is likely that recapture rates decline with time. It is possible that the recapture rate observed in both studies could be due to a high mortality of *Nephrops* discards. It is, however, much more likely to be a reflection of the low entry rate of *Nephrops* to creels, (Chapter 4) and healthy state of the stock in this area (Chapter 2). Tag loss and under-reporting of recaptures may also have been factors, but the aquarium trial suggested that tag loss would be minimal and the study was well publicised with local fishermen and packers who were supportive of the work.

Although only preliminary observations have been carried out on the survival of discards, there is little evidence for high levels of mortality of both *Nephrops* and non-target discards from the creel fishery. It can therefore be assumed that the majority of discarded animals will return to the sea bed unharmed. This is in contrast with the trawl fishery, in which mortality of both target and non-target species is considered to be high (Gueguen & Charuau, 1975; Chapman, 1981; Evans *et al.*, 1994; ICES, 1997; Ulmestrand, 1998; Castro *et al.*, 2003). In the creel fishery, seabirds appear to be the major source of mortality to both target and some non-target species; however, further work must be undertaken in order to obtain a more robust estimate. Seabirds also receive a major energy subsidy from the trawl fishery (Furness, 1982; Monaghan & Zonfrillo, 1986; Blaber & Wassenberg, 1989; Camphuysen *et al.*, 1995; Wieczorek *et al.*, 1999), but in this case, many of the discards are injured or dead. In the creel fishery, one possible way of mitigating discard mortality from seabirds would be to alter discarding practice or to use a kind of 'chute', transferring animals directly into the water column. Such management measures could prove particularly useful during times of the year when the number of small *Nephrops* in the catch is high and therefore numbers of *Nephrops* discards are greatest (Chapter 2). Furthermore, the ability of *Nephrops* discards to re-establish following discarding needs to be investigated in much more detail. The efficacy of returning ovigerous females also requires investigation in relation to the viability of the eggs following aerial exposure. However, larval hatching work in the laboratory using both trawl and creel caught animals (for example, Salerno, 2000) suggests that the eggs of healthy females remain viable.

6.5 Summary

- Bycatch data collection was undertaken at the same time as *Nephrops* catch sampling, on a monthly basis during 2004 and 2005 in the Torridon area and during 2005 in Loch Fyne.

- All target and non-target bycatch species caught from each fleet of creels were identified and counted before being returned. A representative sample of each species was weighed in order to calculate bycatch in terms of biomass as well as numbers.

- For the two study areas, *Nephrops* landings accounted for 43.6% of the catch by weight. This value is much higher than that of the trawl fishery.

- Only a small proportion of the catch consisted of *Nephrops* discards, a reflection of *Nephrops* creels being highly selective for the larger animals in the population.

- Differences were apparent between areas, with over twice the weight of *Nephrops* discards from the Torridon fishery, when compared with Loch Fyne. This is due to larger animals and ovigerous females being discarded in the Torridon fishery.

- In contrast to the majority of trawl fisheries, commercially exploited fish made up a small proportion of the bycatch in the *Nephrops* creel fisheries studied.

- The non-target discard composition varied both in terms of biomass and species composition between the Loch Fyne and Torridon fisheries, as well as among areas within the Torridon fishery. In Loch Fyne, the biomass of non-target discards was over twice that in the Torridon area.

- Differences in bycatch between locations reflect the differences in the environments fished, with sea loch environments comprising a variety of sediment types, leading to a higher diversity and density of epibenthic species and therefore increased bycatch. This contrasts with the Inner Sound, which consists of large areas of soft mud, inhabited by fewer epibenthic species reflected in the lower bycatch.

- Upper Loch Torridon and the Inner Sound show similar proportions of non-target species in terms of biomass. However, when expressed in terms of abundance, almost 10 times as many non-target animals were caught in Upper Loch Torridon than in the Inner Sound. These results highlight the importance of analysing the data in terms of numbers, as well as biomass, in order to illustrate the importance of smaller species such as sea pens in the bycatch in certain areas.

- In comparison with the *Nephrops* trawl fisheries, the bycatch associated with the creel fishery is not only much lower, but the survival of discarded species may be higher, with predation from seabirds identified as the major source of mortality in the creel fishery.

Table 6.1 - CPUE of species and other taxa (F. = family) in bycatch from creeling in Loch Fyne and the Torridon area. Additional species found in small numbers only.

Species	Mean CPUE ± SD (number 100-creels)$^{-1}$	Comments
Cnidaria		
Funiculina quadrangularis	0.80 ± 1.66	
Pennatula phosphorea	3.24 ± 16.36	
Virgularia mirabilis	2.23 ± 14.41	
Annelida		
Aphrodita aculeata	0.01 ± 0.09	
Crustacea		
Pandalus montagui	1.05 ± 6.57	Includes other pandalids
Pagurus bernhardus	37.45 ± 73.33	Includes *Pagurus prideaux* with *Adamsia carciniopados*
Lithodes maja	0.34 ± .28	
Munida rugosa	54.84 ± 99.05	
Munida sarsi	0.74 ± 3.77	
Galathea strigosa	0.15 ± 1.48	
Hyas araneus	2.58 ± 9.34	
Macropodia tenuirostris	0.23 ± 1.66	
Atelecyclus rotundatus	0.02 ± 0.21	
Cancer pagurus	6.31 ± 12.5	
Necora puber	0.69 ± 2.33	
Liocarcinus depurator	14.42 ± 32.34	Includes *Liocarcinus holsatus*
Carcinus maenas	12.56 ± 31.38	
Monodaeus couchi	0.08 ± 1.00	
Goneplax rhomboides	0.10 ± 0.48	
Mollusca		
Neptunea antiqua	1.59 ± 5.7	
Buccinum undatum	16.79 ± 40.28	Includes *Buccinum humphreysianum*
F Sepiidae	0.01 ± 0.09	
F Loliginidae	0.00 ± 0.06	
F Octopodidae	0.00 ± 0.06	
Echinodermata		
Luidia ciliaris	0.22 ± 0.65	
Porania pulvillus	2.56 ± 4.38	
Crossaster papposus	0.03 ± 0.33	
Asterias rubens	0.79 ± 2.01	
Marthasterias glacialis	0.24 ± 1.82	
Ophiura ophiura	0.03 ± 0.26	

Asteronyx loveni	0.93 ± 2.17	
Echinus esculentus	0.15 ± 1.69	
Echinocardium cordatum	0.00 ± 0.06	Mode of capture unknown
Chondrichthyes		
Scyliorhinus canicula	2.45 ± 6.86	
Galeus melastomus	0.00 ± 0.06	
Osteichthyes		
Conger conger	0.05 ± 0.32	
Gadus morhua	0.36 ± 1.17	
Enchelyopus cimbrius	0.06 ± 0.25	May include other rocklings
Molva molva	0.12 ± 0.40	
Raniceps raninus	0.03 ± 0.15	
Trisopterus minutus	7.90 ± 10.54	Includes some *Merlangius merlangus, Trisopterus luscus* & *T. esmarki*
F Syngnathidae	0.00 ± 0.06	
Myoxocephalus scorpius	0.82 ± 3.51	
Mullus surmuletus	0.00 ± 0.06	
F Labridae	0.03 ± 0.24	
Echiichthys vipera	0.33 ± 1.54	
Pholis gunnellus	0.01 ± 0.14	
Callionymus lyra	0.01 ± 0.12	
F Gobiidae	0.34 ± 2.61	
Scomber scombrus	0.00 ± 0.06	
Pleuronectes platessa	0.43 ± 1.46	Includes *Limanda limanda* and *Zeugoptererus punctatus*

Table 6.2 - Mean weight of non-target bycatch species and other taxa (F. = family).

Species	Sample Location	Number sampled	Mean weight (g)
Cnidaria			
Funiculina quadrangularis	Torridon / Fyne	3	28.37
Pennatula phosphorea	Clyde	3	4.04
Virgularia mirabilis	Clyde	16	1.17
Annelida			
Aphrodita aculeata	Clyde	12	24.33
Crustacea			
Pandalus montagui	Clyde	13	4.31
Pagurus bernhardus	Clyde	12	20.25
Lithodes maja	Torridon / Fyne	45	115.56
Munida rugosa	Torridon / Fyne	877	25.88
Munida sarsi	Torridon / Fyne	6	23.05
Galathea strigosa	Estimate	-	25.00
Hyas araneus	Clyde	12	89.13
Macropodia tenuirostris	Clyde	4	1.49
Atelecyclus rotundatus	Clyde	2	19.73
Cancer pagurus	Torridon / Fyne	223	178.70
Necora puber	Torridon / Fyne	19	105.26
Liocarcinus depurator	Torridon / Fyne	72	33.81
Carcinus maenas	Torridon / Fyne	89	44.00
Monodaeus couchi	Estimate	-	60.00
Goneplax rhomboides	Estimate	-	15.00
Mollusca			
Neptunea antiqua	Torridon / Fyne	12	131.58
Buccinum undatum	Torridon / Fyne	71	85.92
F Sepiidae	Clyde	3	43.59
F Loliginidae	Estimate	-	70.00
F Octopodidae	Estimate	-	100.00
Echinodermata			
Luidia ciliaris	Torridon / Fyne	3	300.00
Porania pulvillus	Torridon / Fyne	33	87.07
Crossaster papposus	Torridon / Fyne	5	80.00
Asterias rubens	Clyde	12	18.28
Marthasterias glacialis	Clyde	12	139.33
Ophiura ophiura	Clyde	11	5.49
Asteronyx loveni	Torridon / Fyne	4	15.35
Echinus esculentus	Clyde	9	122.42
Echinocardium cordatum	Clyde	5	48.99

Chondrichthyes			
Scyliorhinus canicula	Torridon / Fyne	6	858.33
Galeus melastomus	Estimate	-	1000.00
Osteichthyes			
Conger conger	Estimate	-	3000.00
Gadus morhua	Torridon / Fyne	6	333.33
Enchelyopus cimbrius	Torridon / Fyne	1	50.50
Molva molva	Torridon / Fyne	5	1068.00
Raniceps raninus	Estimate		60.00
Trisopterus minutus	Torridon / Fyne	52	38.02
F Syngnathidae	Clyde	4	22.25
Myoxocephalus scorpius	Clyde	16	62.00
Mullus surmuletus	Estimate	-	400.00
F Labridae	Estimate	-	350.00
Echiichthys vipera	Estimate	-	60.00
Pholis gunnellus	Clyde	6	6.04
Callionymus lyra	Clyde	11	41.09
F Gobiidae	Clyde	1	18.66
Scomber scombrus	Estimate	-	500.00
Pleuronectes platessa	Clyde	12	43.21

Table 6.3 - Mean weight of discards per unit weight of *Nephrops* landed (kg kg^{-1}), where n is the number of fleets of creels from which the samples were taken.

	n	Discard ratio (kg-bycatch kg-*Nephrops*$^{-1}$)	
		Non-target species	*Nephrops* discards
Loch Fyne	71	2.02	0.09
Torridon Area	123	0.71	0.22
Inner Sound	78	0.65	0.18
lower Loch Torridon	40	0.90	0.23
Upper Loch Torridon	5	0.71	0.22
All Areas	194	1.10	0.20

Table 6.4 - Mean CPUE weight (kg 100-creels^{-1}) of non-target discards (± SD)

Mean CPUE ± SD (kg 100-creels^{-1})		
All areas	Loch Fyne	Torridon Area
9.70 ± 9.66	15.19 ± 11.04	6.53 ± 7.07

Table 6.5 - Mean CPUE weight (kg 100-creels^{-1}) of non-target discards grouped by Phylum or Class (± SD).

Phyla	Mean CPUE ± SD (kg 100-creels^{-1})			
	Loch Fyne	Inner Sound	lower Loch Torridon	Upper Loch Torridon
Cnidaria	0.000 ± 0.000	0.044 ± 0.058	0.054 ± 0.099	0.370 ± 0.282
Annelida	0.000 ± 0.000	0.000 ± 0.002	0.000 ± 0.000	0.004 ± 0.009
Crustacea	8.311 ± 6.531	1.281 ± 2.148	4.881 ± 3.553	6.092 ± 4.208
Mollusca	4.089 ± 5.197	0.191 ± 0.698	0.256 ± 0.667	1.055 ± 1.177
Echinodermata	0.215 ± 0.586	0.454 ± 0.444	0.517 ± 0.512	0.161 ± 0.093
Chondrichthyes	1.562 ± 4.145	3.705 ± 8.124	0.205 ± 0.697	0.000 ± 0.000
Osteichthyes	1.014 ± 1.532	0.766 ± 0.917	0.603 ± 1.352	0.269 ± 0.265

Table 6.6 - Mean CPUE by weight of individual non-target discard species and other taxa (F. = family) (± SD) ("-" indicates no species caught).

Species	Loch Fyne	Inner Sound	lower Loch Torridon	Upper Loch Torridon
	Mean CPUE ± SD (kg 100-creels^{-1})			
Cnidaria				
Funiculina quadrangularis	- -	0.044 ± 0.058	0.018 ± 0.040	0.054 ± 0.086
Pennatula phosphorea	- -	- -	0.032 ± 0.081	0.250 ± 0.261
Virgularia mirabilis	- -	- -	0.004 ± 0.017	0.066 ± 0.074
Annelida				
Aphrodita aculeata	- -	0.000 ± 0.002	- -	0.004 ± 0.009
Crustacea				
Pandalus montagui	0.012 ± 0.046	- -	- -	- -
Pagurus bernhardus	1.851 ± 1.913	0.006 ± 0.017	0.089 ± 0.194	2.331 ± 1.458
Lithodes maja	0.025 ± 0.093	0.072 ± 0.592	0.005 ± 0.022	- -
Munida rugosa	3.183 ± 3.243	0.106 ± 0.583	1.006 ± 1.866	0.167 ± 0.244
Munida sarsi	0.012 ± 0.098	0.032 ± 0.098	- -	- -
Galathea strigosa	0.010 ± 0.061	- -	- -	- -
Hyas araneus	0.601 ± 1.295	0.012 ± 0.050	0.021 ± 0.135	0.016 ± 0.035
Macropodia tenuirostris	- -	- -	0.001 ± 0.005	0.001 ± 0.001
Atelecyclus rotundatus	- -	0.001 ± 0.006	0.000 ± 0.003	- -
Cancer pagurus	0.016 ± 0.110	1.023 ± 1.931	3.374 ± 3.130	0.590 ± 0.554
Necora puber	0.141 ± 0.342	0.011 ± 0.053	0.043 ± 0.117	0.311 ± 0.646
Liocarcinus depurator	1.142 ± 1.584	0.005 ± 0.015	0.305 ± 0.347	0.182 ± 0.109
Carcinus maenas	1.317 ± 1.802	0.001 ± 0.007	0.029 ± 0.106	2.487 ± 3.360
Monodaeus couchi	- -	0.011 ± 0.095	0.000 ± 0.000	- -
Goneplax rhomboides	- -	- -	0.007 ± 0.014	0.008 ± 0.017
Mollusca				
Neptunea antiqua	0.407 ± 1.086	0.072 ± 0.292	0.146 ± 0.616	0.069 ± 0.102
Buccinum undatum	3.681 ± 4.941	0.118 ± 0.616	0.108 ± 0.236	0.986 ± 1.211
F Sepiidae	0.001 ± 0.006	- -	- -	- -
F Loliginidae	- -	- -	0.002 ± 0.010	- -
F Octopodidae	- -	0.001 ± 0.010	- -	- -

Echinodermata				
Luidia ciliaris	0.046 ± 0.233	0.047 ± 0.124	0.150 ± 0.220	- -
Porania pulvillus	- -	0.373 ± 0.445	0.345 ± 0.419	0.061 ± 0.083
Crossaster papposus	0.002 ± 0.015	- -	0.009 ± 0.055	- -
Asterias rubens	0.030 ± 0.052	0.001 ± 0.011	0.004 ± 0.009	0.076 ± 0.044
Marthasterias glacialis	0.091 ± 0.414	- -	- -	- -
Ophiura ophiura	- -	- -	0.001 ± 0.003	- -
Asteronyx loveni	- -	0.033 ± 0.045	0.005 ± 0.017	0.003 ± 0.006
Echinus esculentus	0.046 ± 0.341	- -	0.003 ± 0.017	0.021 ± 0.048
Echinocardium cordatum	- -	0.001 ± 0.005	- -	- -
Chondrichthyes				
Scyliorhinus canicula	1.562 ± 4.145	3.694 ± 8.127	0.205 ± 0.697	- -
Galeus melastomus	- -	0.011 ± 0.098	- -	- -
Osteichthyes				
Conger conger	0.181 ± 1.161	0.100 ± 0.505	0.196 ± 1.237	- -
Gadus morhua	0.277 ± 0.603	0.030 ± 0.089	0.036 ± 0.117	- -
Enchelyopus cimbrius	- -	0.006 ± 0.016	0.001 ± 0.007	0.018 ± 0.039
Molva molva	0.125 ± 0.553	0.202 ± 0.414	- -	- -
Raniceps raninus	- -	0.002 ± 0.010	0.004 ± 0.014	- -
Trisopterus minutes	0.149 ± 0.203	0.421 ± 0.498	0.340 ± 0.388	0.251 ± 0.266
F Syngnathidae	- -	- -	0.000 ± 0.003	- -
Myoxocephalus scorpius	0.137 ± 0.344	- -	0.005 ± 0.016	- -
Mullus surmuletus	0.000 ± 0.000	0.004 ± 0.039	- -	- -
F Labridae	0.022 ± 0.132	- -	0.008 ± 0.048	- -
Echiichthys vipera	0.055 ± 0.147	- -	- -	- -
Pholis gunnellus	- -	0.000 ± 0.001	- -	- -
Callionymus lyra	0.001 ± 0.008	- -	- -	- -
F Gobiidae	0.017 ± 0.080	- -	- -	- -
Scomber scombrus	0.000 ± 0.000	- -	0.011 ± 0.069	- -
Pleuronectes platessa	0.049 ± 0.097	- -	0.000 ± 0.010	- -

Table 6.7 - Mean CPUE by weight and number of non-target discards from two distinct areas within the Torridon fishery (± SD).

	Mean CPUE ± SD	
	Inner Sound	Upper Loch Torridon
Number (100-creels^{-1})	38.90 ± 35.3	335.30 ± 150.55
Biomass (kg 100-creels^{-1})	6.44 ± 8.24	7.95 ± 5.02

Table 6.8 - Mean CPUE by number of individual non-target discard species and other taxa (F. = family) (± SD) ('- indicates no species caught).

Species	Loch Fyne	Inner Sound	lower Loch Torridon	Upper Loch Torridon
	Mean CPUE ± SD (100-creels^{-1})			
Cnidaria				
Funiculina quadrangularis	- -	1.56 ± 2.05	0.63 ± 1.41	1.91 ± 3.04
Pennatula phosphorea	- -	0.01 ± 0.10	7.96 ± 19.92	61.74 ± 64.47
Virgularia mirabilis	- -	- -	3.74 ± 14.84	56.52 ± 63.05
Annelida				
Aphrodita aculeata	- -	0.01 ± 0.10	- -	0.17 ± 0.39
Crustacea				
Pandalus montagui	2.88 ± 10.67	- -	- -	- -
Pagurus bernhardus	91.41 ± 94.51	0.29 ± 0.86	4.41 ± 9.60	115.13 ± 72.01
Lithodes maja	0.21 ± 0.80	0.62 ± 5.12	0.04 ± 0.19	- -
Munida rugosa	122.99 ± 125.31	4.11 ± 22.52	38.87 ± 72.10	6.43 ± 9.44
Munida sarsi	0.51 ± 4.27	1.39 ± 4.27	- -	- -
Galathea strigosa	0.40 ± 2.44	- -	- -	- -
Hyas araneus	6.74 ± 14.53	0.13 ± 0.56	0.24 ± 1.51	0.17 ± 0.39
Macropodia tenuirostris	- -	0.04 ± 0.31	1.00 ± 3.56	0.35 ± 0.78
Atelecyclus rotundatus	- -	0.04 ± 0.31	0.02 ± 0.14	- -
Cancer pagurus	0.09 ± 0.62	5.73 ± 10.81	18.88 ± 17.51	3.30 ± 3.10
Necora puber	1.34 ± 3.25	0.10 ± 0.50	0.41 ± 1.11	2.96 ± 6.14
Liocarcinus depurator	33.79 ± 46.84	0.14 ± 0.43	9.02 ± 10.27	5.39 ± 3.22
Carcinus maenas	29.94 ± 40.96	0.03 ± 0.17	0.65 ± 2.40	56.52 ± 76.36
Monodaeus couchi	- -	0.19 ± 1.58	- -	- -
Goneplax rhomboides	- -	- -	0.43 ± 0.90	0.52 ± 1.17
Mollusca				
Neptunea antiqua	3.09 ± 8.26	0.55 ± 2.22	1.11 ± 4.68	0.52 ± 0.78
Buccinum undatum	42.85 ± 57.51	1.37 ± 7.17	1.26 ± 2.74	11.48 ± 14.09
F Sepiidae	0.02 ± 0.15	- -	- -	- -
F Loliginidae	- -	- -	0.02 ± 0.14	- -
F Octopodidae	- -	0.01 ± 0.10	- -	- -
Echinodermata				
Luidia ciliaris	0.15 ± 0.78	0.16 ± 0.41	0.50 ± 0.73	- -
Porania pulvillus	- -	4.28 ± 5.11	3.97 ± 4.82	0.70 ± 0.95
Crossaster papposus	0.02 ± 0.19	- -	0.11 ± 0.69	- -
Asterias rubens	1.64 ± 2.82	0.08 ± 0.60	0.22 ± 0.51	4.17 ± 2.41
Marthasterias glacialis	0.65 ± 2.97	- -	- -	- -
Ophiura ophiura	- -	- -	0.13 ± 0.58	- -

Asteronyx loveni	- -	2.14 ± 2.95	0.35 ± 1.07	0.17 ± 0.39
Echinus esculentus	0.38 ± 2.78	- -	0.02 ± 0.14	0.17 ± 0.39
Echinocardium cordatum	- -	0.01 ± 0.10	- -	- -
Chondrichthyes				
Scyliorhinus canicula	1.82 ± 4.83	4.30 ± 9.47	0.24 ± 0.81	- -
Galeus melastomus	- -	0.01 0.10	- -	- -
Osteichthyes				
Conger conger	0.06 ± 0.39	0.03 ± 0.17	0.07 ± 0.41	- -
Gadus morhua	0.83 ± 1.81	0.09 ± 0.27	0.11 ± 0.35	- -
Enchelyopus cimbrius	- -	0.11 ± 0.32	0.02 ± 0.14	0.35 ± 0.78
Molva molva	0.12 ± 0.52	0.19 ± 0.39	0.00 ± 0.00	- -
Raniceps raninus	- -	0.03 ± 0.17	0.07 ± 0.23	- -
Trisopterus minutus	3.93 ± 5.35	11.08 ± 13.10	8.94 ± 10.20	6.61 ± 7.00
F Syngnathidae	- -	- -	0.02 ± 0.14	- -
Myoxocephalus scorpius	2.21 ± 5.55	- -	0.09 ± 0.26	- -
Mullus surmuletus	- -	0.01 ± 0.10	- -	- -
F Labridae	0.06 ± 0.38	- -	0.02 ± 0.14	- -
Echiichthys vipera	0.91 ± 2.45	- -	- -	- -
Pholis gunnellus	- -	0.03 ± 0.22	- -	- -
Callionymus lyra	0.03 ± 0.20	- -	- -	- -
F Gobiidae	0.93 ± 4.27	- -	- -	- -
Scomber scombrus	- -	- -	0.02 ± 0.14	- -
Pleuronectes platessa	1.14 ± 2.24	- -	0.04 ± 0.19	- -

Table 6.9 - TWINSPAN group properties.

Group	Division	Eigenvalue	Samples
0	1	0.543	80
00	2	0.206	4
01			76
1	1	0.543	114
10	2	0.360	41
11			73

Table 6.10 - Percentage distribution of sites from different sediment types, depths and locations (IS - Inner Sound, LLT – lower Loch Torridon, ULT – Upper Loch Torridon and LF – Loch Fyne) across divisions produced from TWINSPAN analysis.

Factor		Group					
		0	00	01	1	10	11
Sediment Type	Hard	42.3	25	37.5	36.8	74.4	22.1
	Soft	57.7	75	62.5	63.2	25.6	77.9
Depth	< 50	44.7	100	41.7	2.8	6.8	0
	50-100	51.3	0	54.2	28.7	59.1	7.8
	101-150	1.3	0	1.4	13.9	25	6.3
	>150	2.6	0	2.8	54.6	9.1	85.9
Location	LF	88.8	0	93.4	0	0	0
	ULT	6.3	75	2.6	0	0	0
	LLT	3.8	25	2.6	32.5	87.8	1.4
	IS	1.3	0	1.3	67.5	12.2	98.6

Table 6.11 - Indicator species from each TWINSPAN group.

Group	Indicator Species
0	*Buccinum undatum, Carcinus maenas, Pagurus bernhardus, Liocarcinus depurator, Munida rugosa*
00	*Virgularia mirabilis*
01	
1	*Cancer pagurus, Porania pulvillus*
10	*Cancer pagurus, Pagurus bernhardus , Liocarcinus depurator, Munida rugosa*
11	*Asteronyx loveni, Scyliorhinus canicula*

Table 6.12 - Summary of ANOSIM of community composition at sites from different areas, depths and sediment types (only significant comparisons shown). Probabilities adjusted using Bonferonni correction to protect against type I errors.

Factor	R	*P*	Multiple comparisons	R	*P*
Location (P<0.0083)	0.776	0.001	LF vs. LLT	0.893	0.001
			LF vs. ULT	0.812	0.001
			LF vs. IS	0.922	0.001
			LLT vs. ULT	0.643	0.001
			LLT vs. IS	0.48	0.001
			ULT vs. IS	0.836	0.001
Depth (P<0.0083)	0.487	0.001	> 150 vs. 50 - 100	0.636	0.001
			> 150 vs. 101 - 150	0.28	0.001
			> 150 vs. < 50	0.873	0.001
			101 - 150 vs. 50 - 100	0.279	0.001
			101 - 150 vs. < 50	0.662	0.001
Sediment Type	0.006	0.26			

Table 6.13 - Analysis of dissimilarities (SIMPER) based on discard species abundance, from different locations, depths and sediment types.

Factor groups		Average Dissimilarity	Species	Mean Abundance		Cumulative percentage
Factor 1	Factor 2			Factor 1	Factor 2	
Location						
Loch Fyne	Inner Sound	95.86	*Munida rugosa*	122.99	4.11	30.53
			Pagurus bernhardus	91.41	0.29	53.97
			Buccinum undatum	42.85	1.37	65.65
Loch Fyne	Upper Loch Torridon	71.08	*Munida rugosa*	122.99	6.43	22.18
			Pagurus bernhardus	91.41	115.13	41.01
			Virgularia mirabilis	0	56.52	54.86
			Pennatula phosphorea	0	61.74	67.2
Loch Fyne	lower Loch Torridon	79.96	*Munida rugosa*	122.99	38.87	27.29
			Pagurus bernhardus	91.41	4.41	50.07
			Buccinum undatum	42.85	1.26	61.56
Inner Sound	Upper Loch Torridon	96.08	*Pagurus bernhardus*	0.29	115.13	31.39
			Virgularia mirabilis	0	56.52	50.02
			Carcinus maenas	0.03	56.52	65.5
Inner Sound	lower Loch Torridon	80.4	*Munida rugosa*	4.11	38.87	27.99
			Cancer pagurus	5.73	18.88	44.78
			Trisopterus minutus	11.08	8.94	55.97
			Liocarcinus depurator	0.14	9.02	64.27
Upper Loch Torridon	lower Loch Torridon	87.07	*Pagurus bernhardus*	115.13	4.41	28.65
			Virgularia mirabilis	56.52	3.74	45.7
			Carcinus maenas	56.52	0.65	60.26
Depth						
> 150	< 50	94.78	*Munida rugosa*	1.97	99.64	25.26
			Pagurus bernhardus	4.46	90.66	46.4
			Buccinum undatum	1.07	30.19	55.14
			Liocarcinus depurator	0.27	36.85	63.36
> 150	50 - 100	88.44	*Munida rugosa*	1.97	85.98	29.05
			Pagurus bernhardus	4.46	45.25	43.99
			Cancer pagurus	5.54	9.58	53.15
			Buccinum undatum	1.07	25.98	61.03
> 150	101 - 150	74.51	*Munida rugosa*	1.97	25.73	19.81
			Trisopterus minutus	12.06	9.51	36.83
			Cancer pagurus	5.54	6.01	47.87
			Pagurus bernhardus	4.46	10.91	57.09

			Scyliorhinus canicula	4.96	1.02	65.11
101 - 150	50 -100	77.68	*Munida rugosa*	25.73	85.98	31.15
			Pagurus bernhardus	10.91	45.25	47.97
			Buccinum undatum	8.18	25.98	57.5
			Cancer pagurus	6.01	9.58	66.12
101 - 150	< 50	83.81	*Munida rugosa*	25.73	99.64	26.13
			Pagurus bernhardus	10.91	90.66	47.97
			Buccinum undatum	8.18	30.19	57.41
			Carcinus maenas	2.87	34.53	65.94
50 -100	< 50	69.58	*Munida rugosa*	85.98	99.64	27.1
			Pagurus bernhardus	45.25	90.66	48.78
			Buccinum undatum	25.98	30.19	58.22
			Liocarcinus depurator	18.22	36.85	66.34
Sediment Type						
Soft	Hard	79.69	*Munida rugosa*	43.15	51.81	25.02
			Pagurus bernhardus	31.97	40.27	41.01
			Cancer pagurus	3.91	10.89	50.02
			Buccinum undatum	12.41	22.69	58.27
			Trisopterus minutus	7.51	9.19	65.95

Table 6.14 - Analysis of dissimilarities (SIMPER) based on discard species abundance, from different sediment types in the Torridon area.

		Mean Abundance		Cumulative
Average Dissimilarity	Species	Soft	Hard	percentage
70.9	*Munida rugosa*	2.27	33.6	23.14
	Cancer pagurus	5.98	15.26	38.87
	Trisopterus minutus	9.61	11.01	51.87
	Porania pulvillus	3.66	4.54	58.07
	Pagurus bernhardus	4.59	8.63	63.8

Table 6.15 - Percentage of non-target discard species taken by seabirds.

Species	n	Percentage mortality
Funiculina quadrangularis	3	0
Pagurus bernhardus	37	0
Munida rugosa	48	6.3
Cancer pagurus	62	0
Liocarcinus depurator	33	24.2
Buccinum undatum	54	0
Porania pulvillus	20	10
Trisopterus minutus	14	100

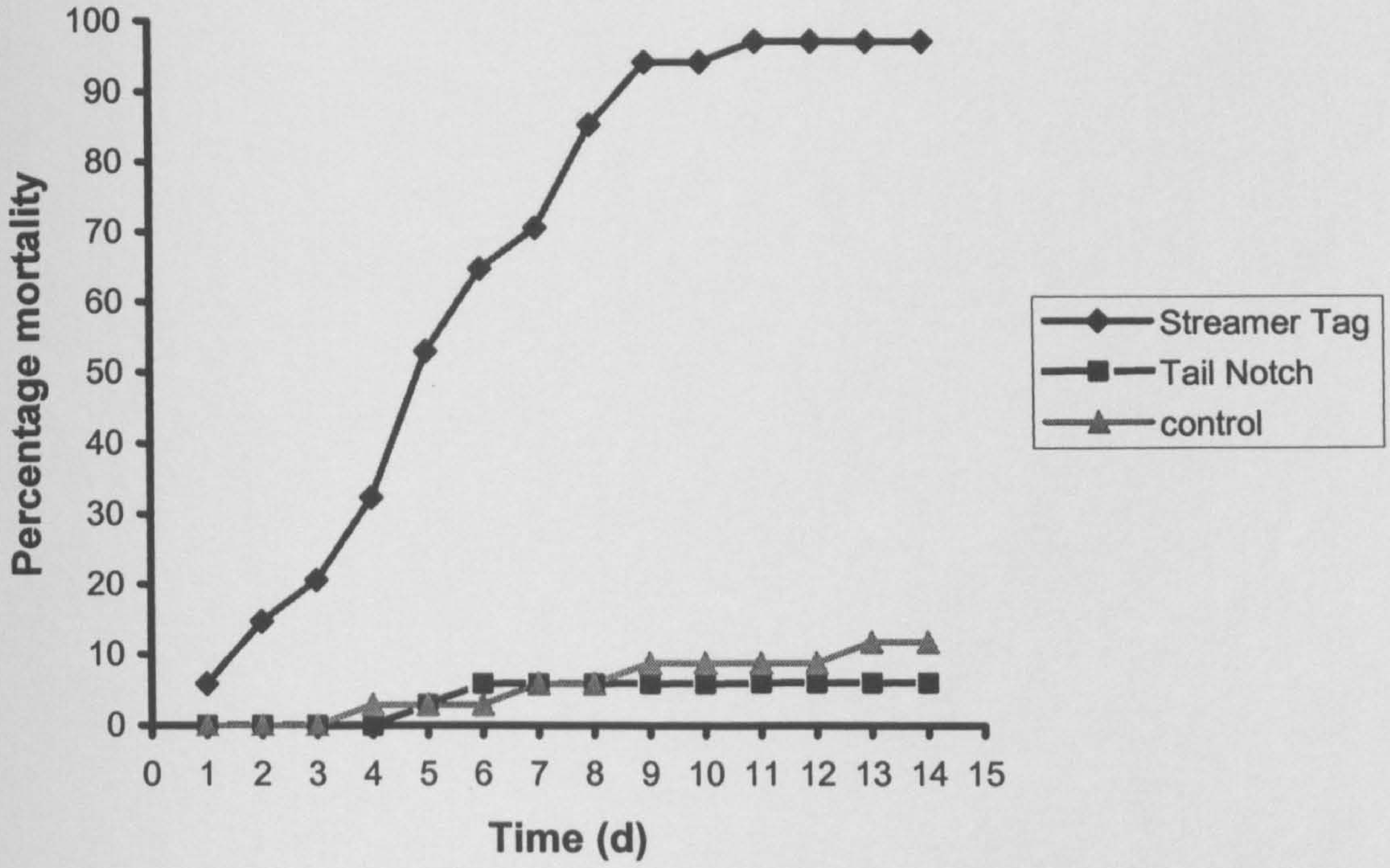


Figure 6.1 - Percentage mortality of *Nephrops* tagged with streamer tag and tail notch.

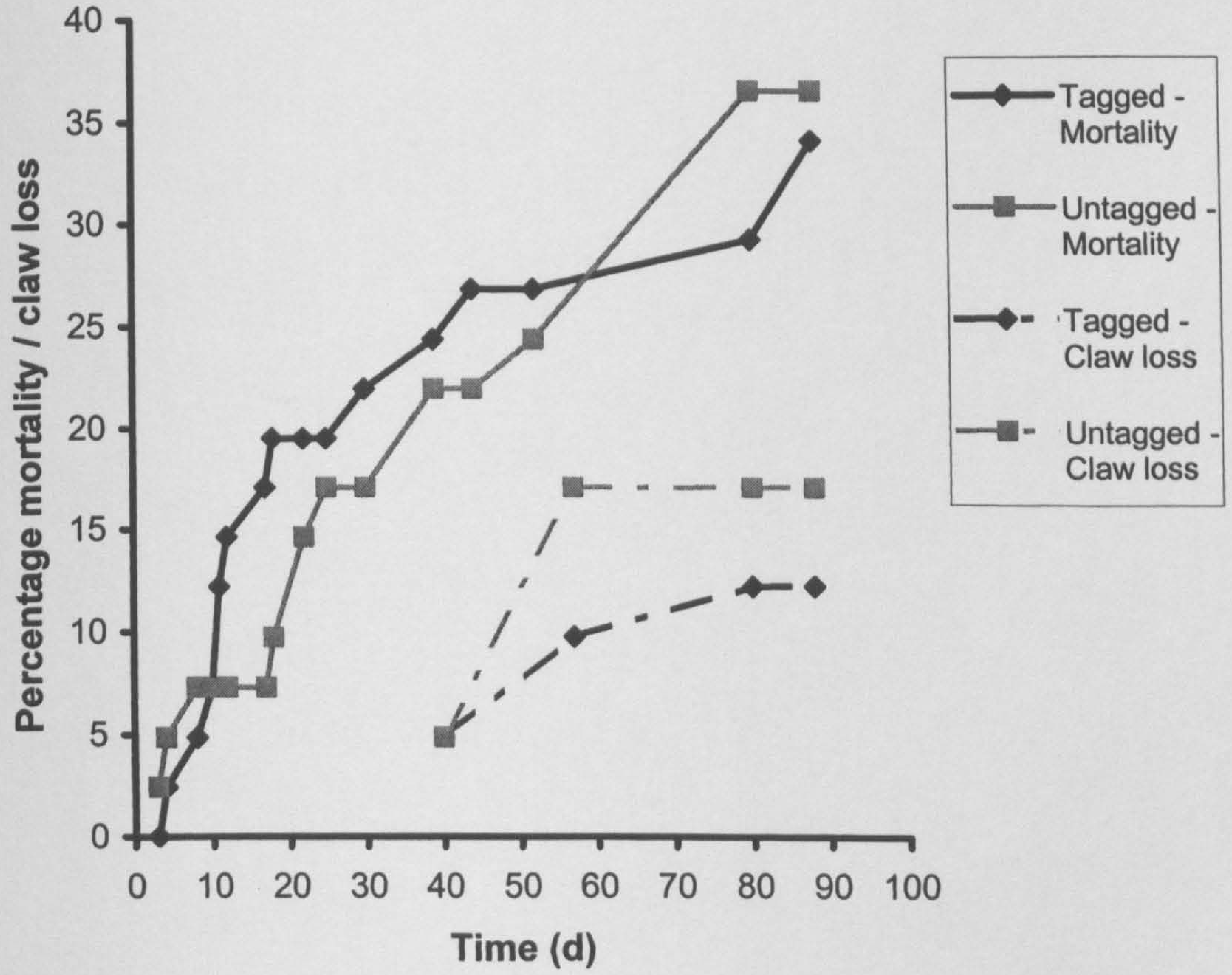


Figure 6.2 - Percentage mortality and limb loss of *Nephrops* tagged with cable ties.

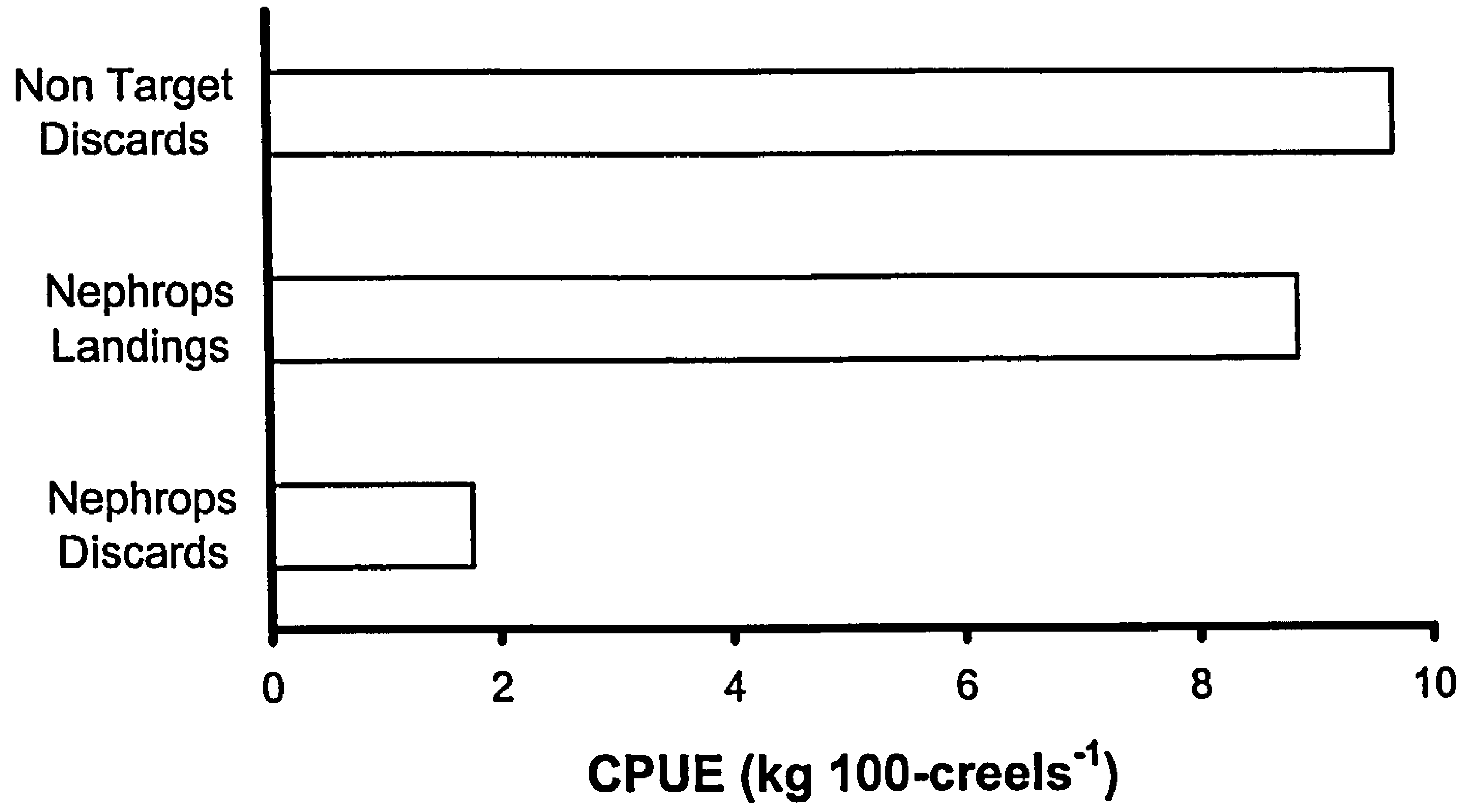


Figure 6.3 - Mean catch composition (CPUE by weight) from the Torridon and Loch Fyne *Nephrops* creel fisheries combined.

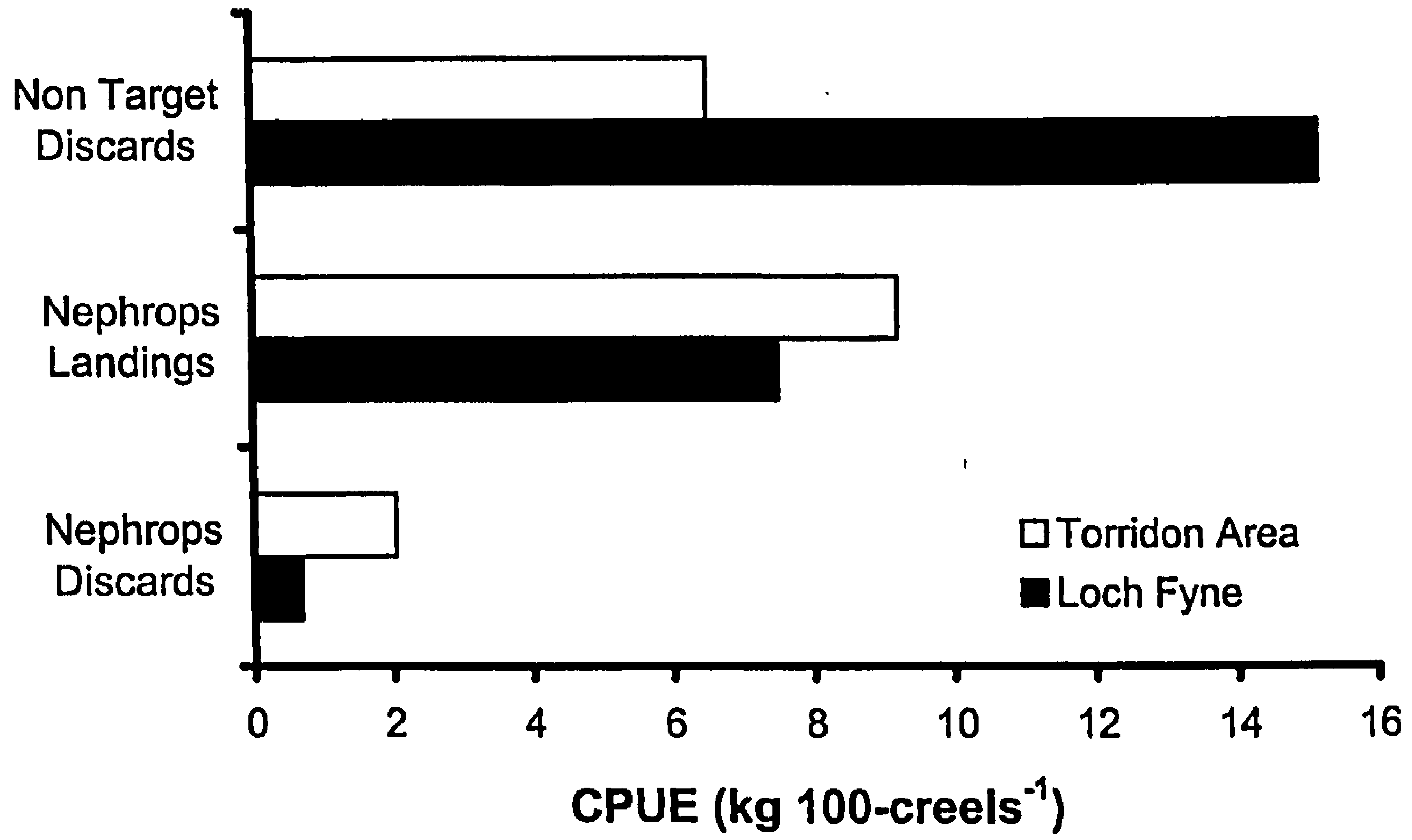


Figure 6.4 - Mean catch composition (CPUE by weight) from the Torridon and Loch Fyne *Nephrops* creel fishery.

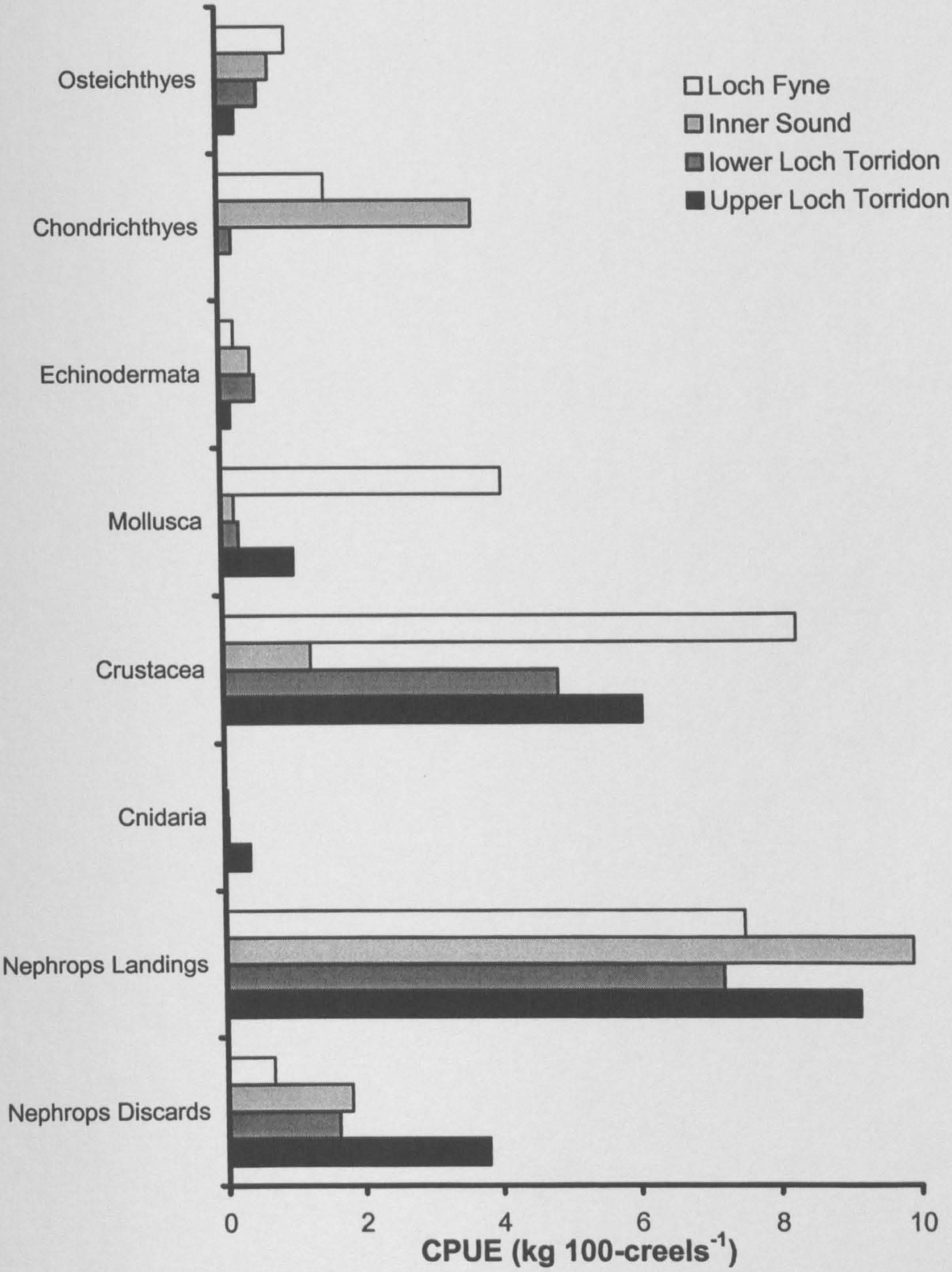


Figure 6.5 - Mean CPUE by weight of the target species and non-target species groups in the Torridon and Loch Fyne *Nephrops* creel fishery. Non-target species separated by phyla.

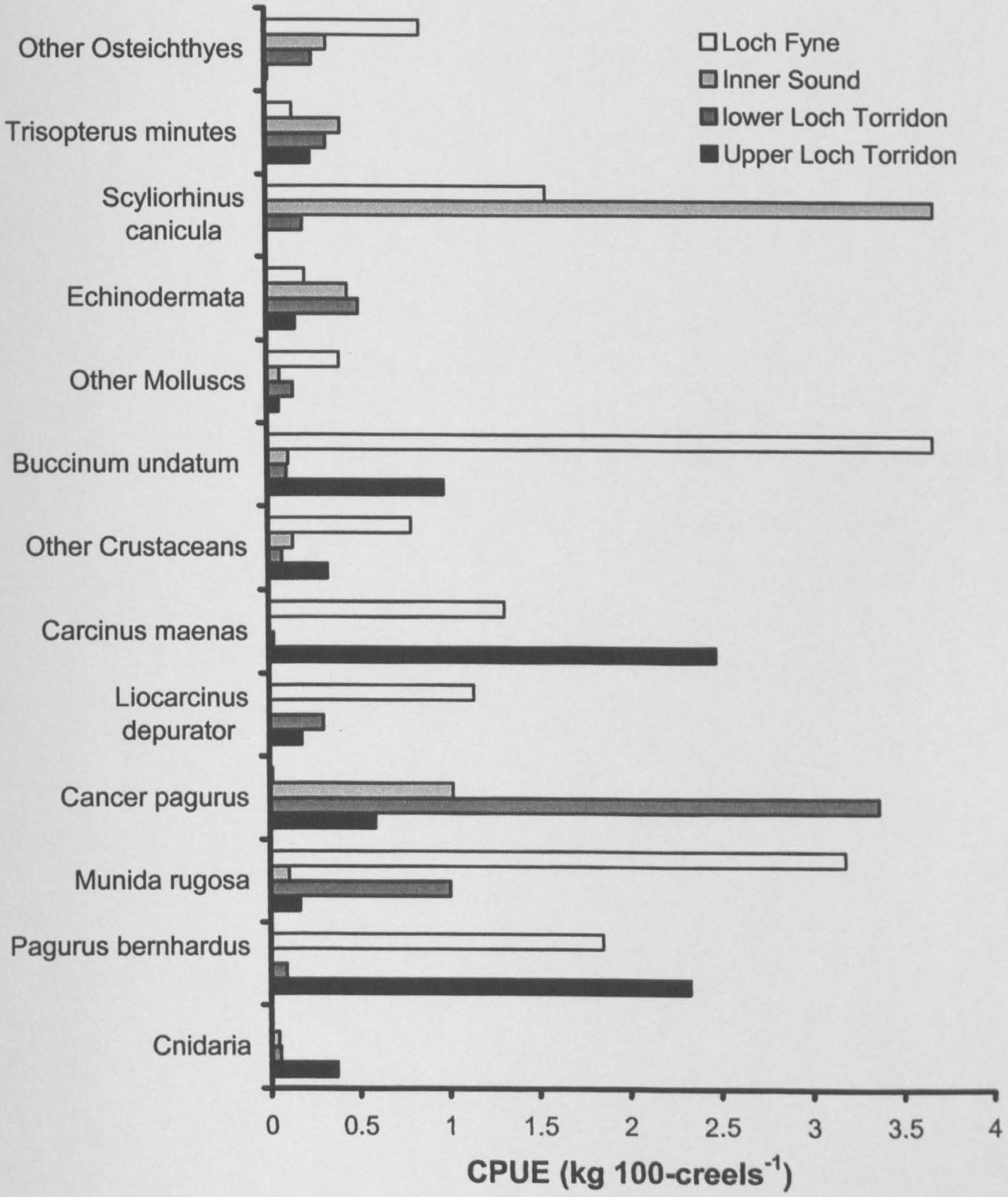


Figure 6.6 - Mean catch composition (CPUE by weight) from the Torridon and Loch Fyne *Nephrops* creel fishery. Most abundant discard species and taxonomic group shown.

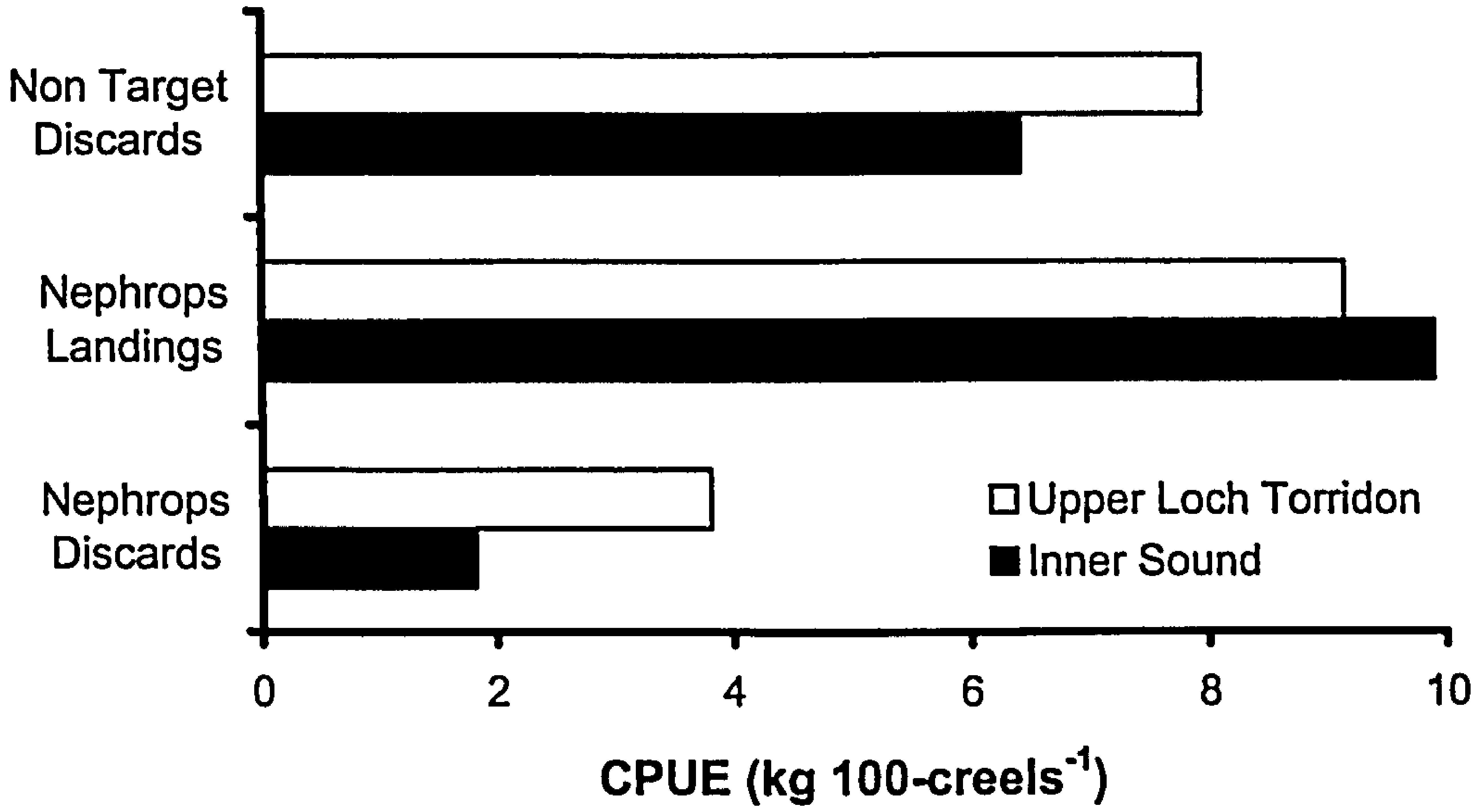


Figure 6.7 - Mean catch composition (CPUE by weight) from the Torridon *Nephrops* creel fishery.

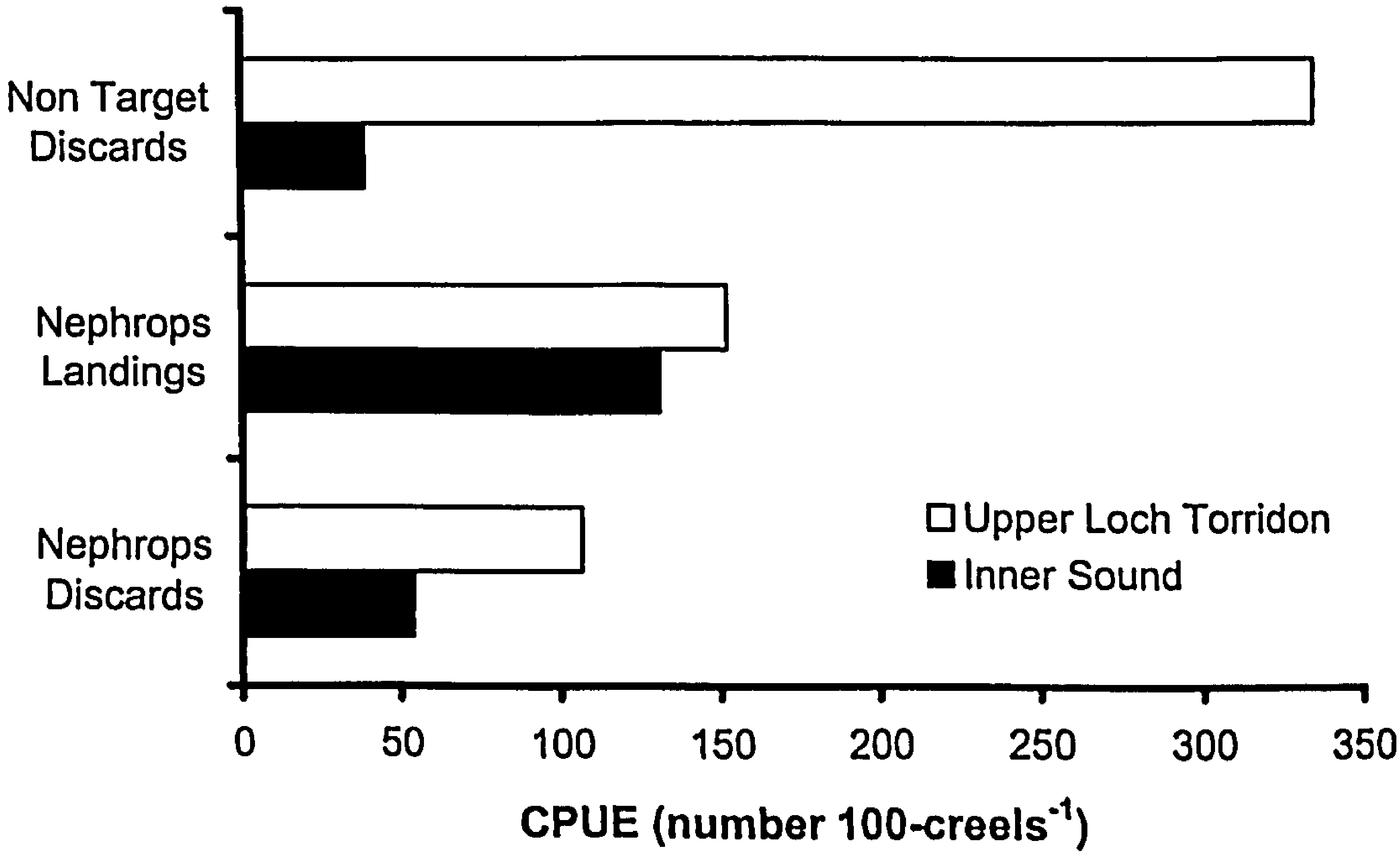


Figure 6.8 - Mean catch composition (CPUE by number) from the Torridon *Nephrops* creel fishery.

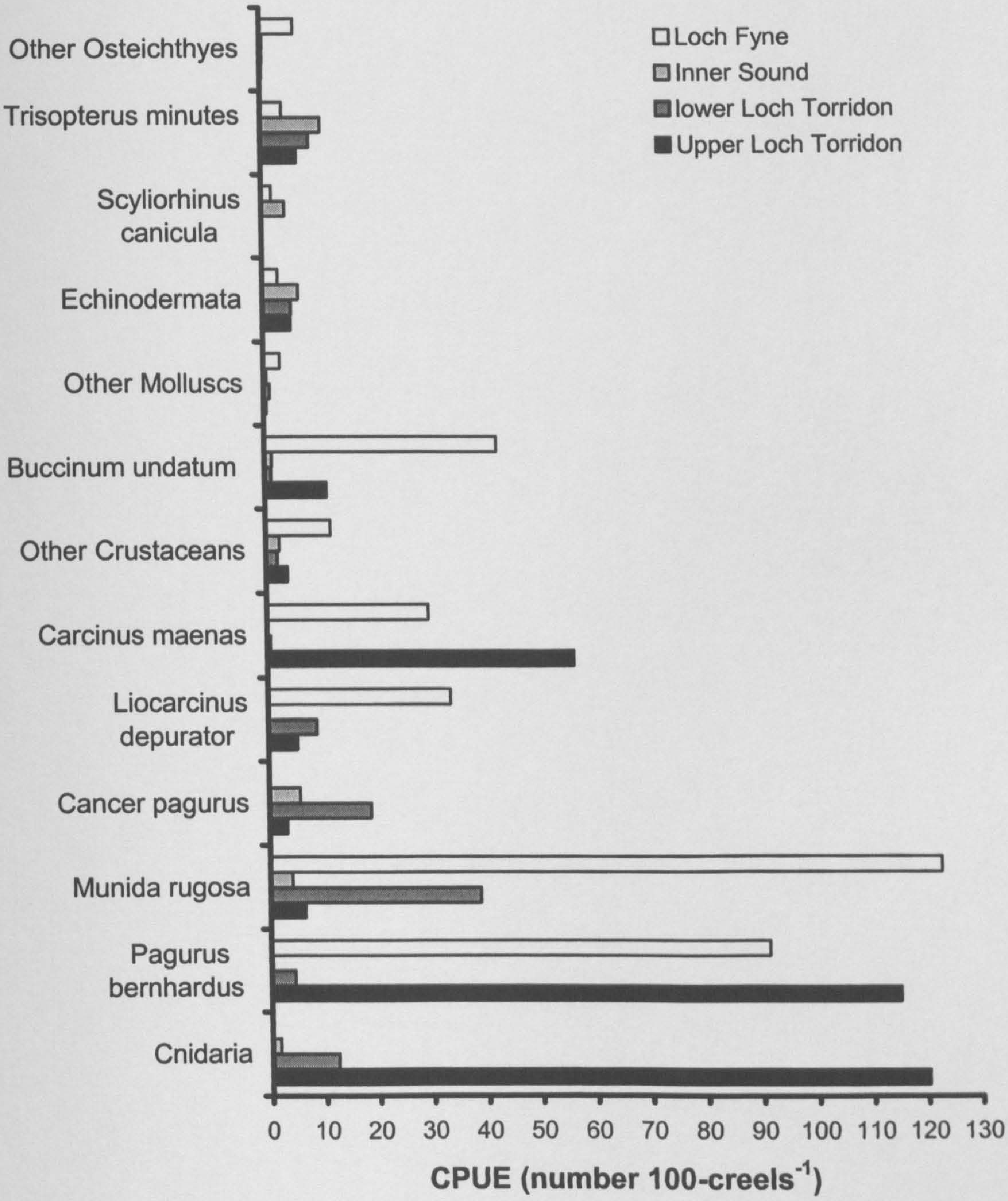


Figure 6.9 - Mean catch composition (number 100-creels^{-1}) from the Torridon and Loch Fyne *Nephrops* creel fishery. Most abundant discard species and taxonomic group shown (by weight).

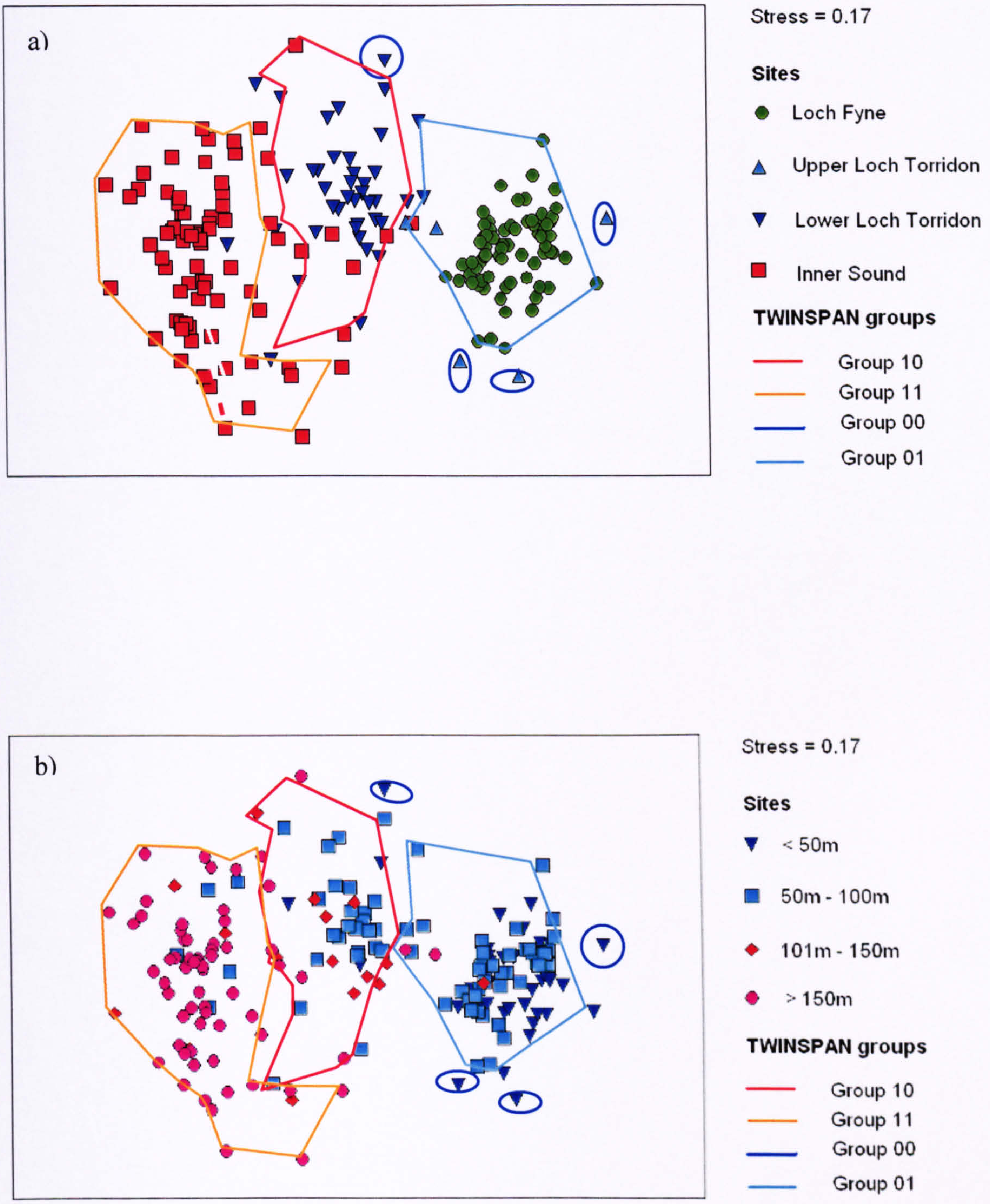


Figure 6.10 - MDS plots of stations by similarity of discard species composition by abundance, with overlay of TWINSPAN group boundaries at level 1, 00 and 01. Plots shown for (a) area, (b) depth and (c) sediment type (when is stress<20 the ordination is useful).

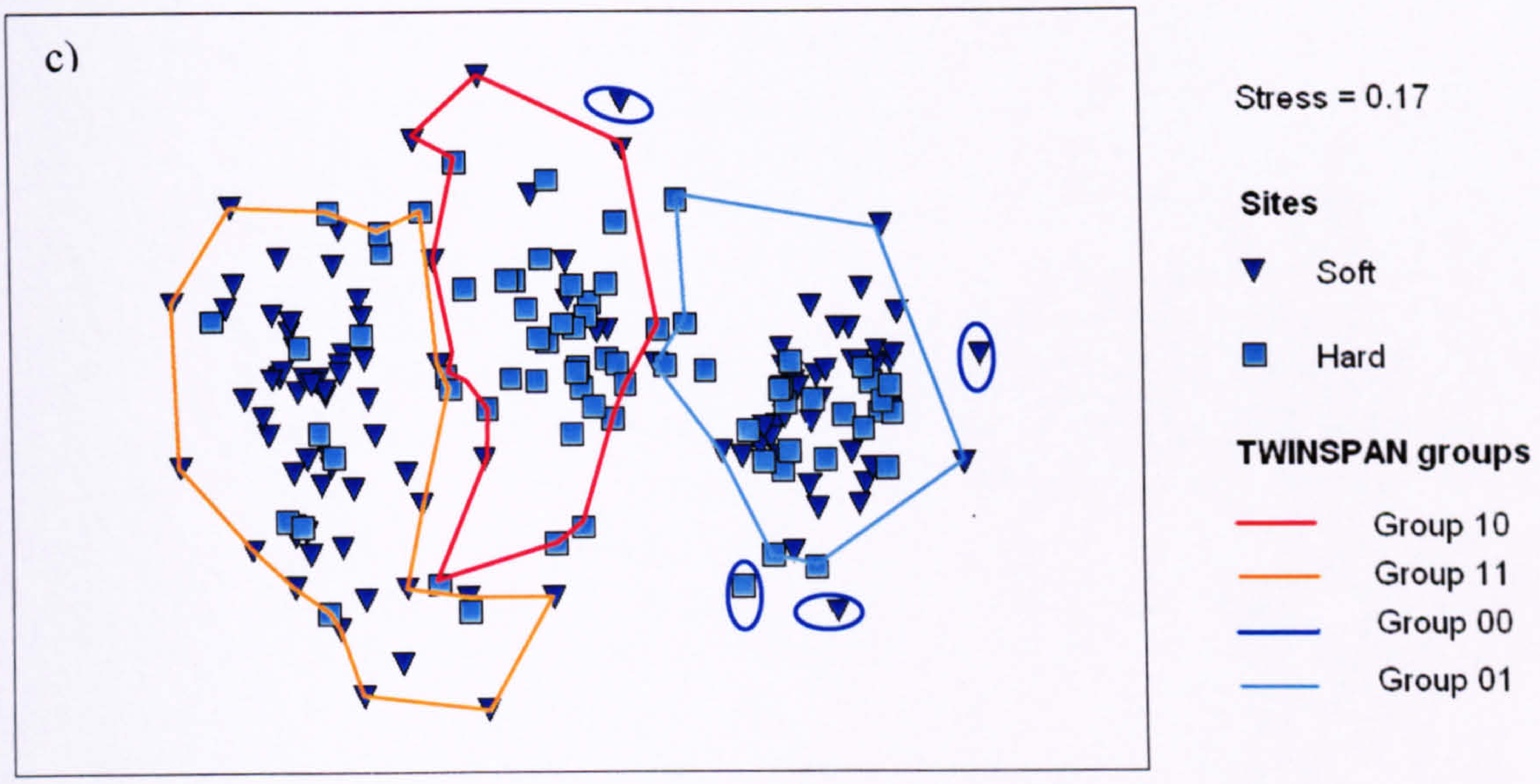
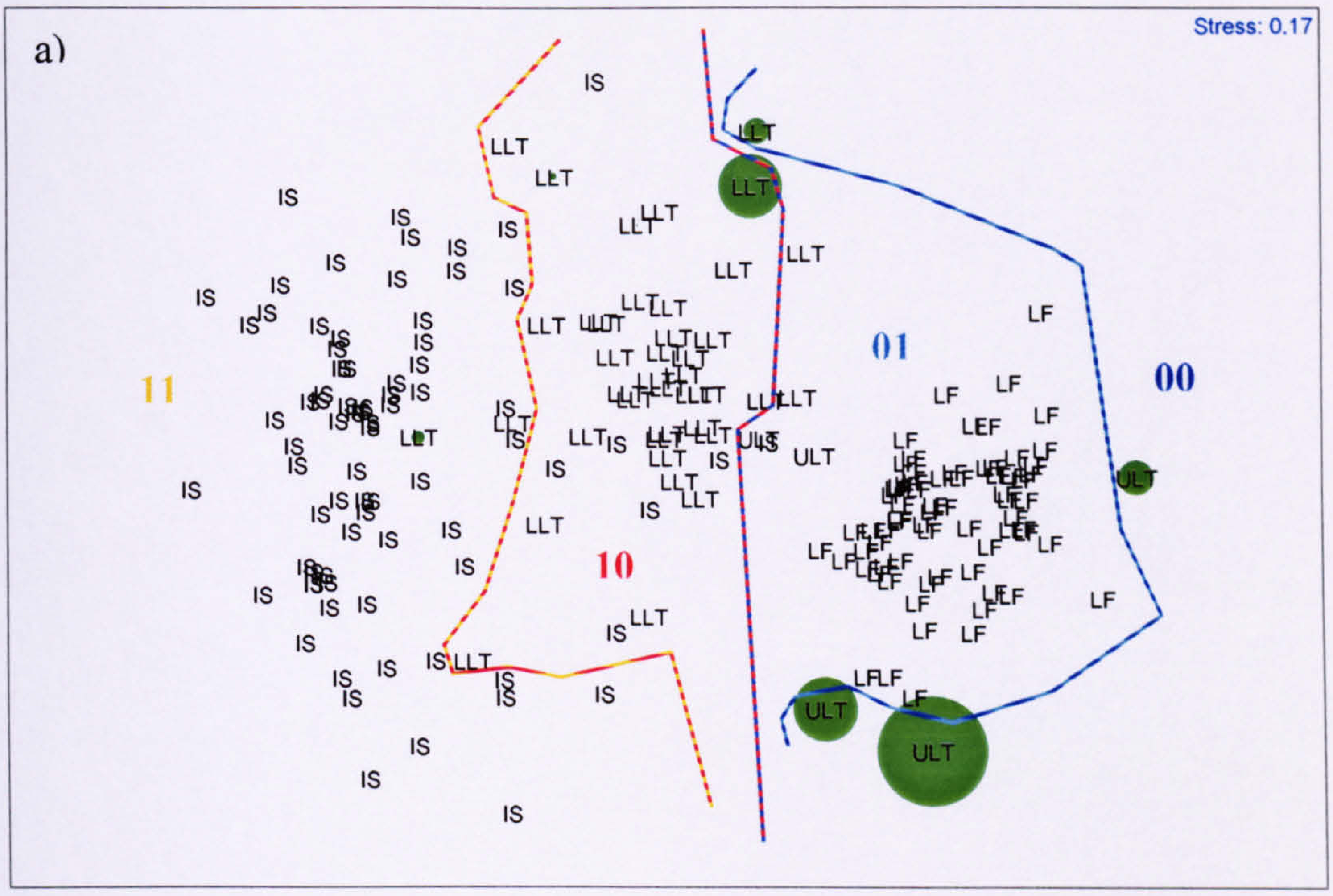


Figure 6.10 – (continued).



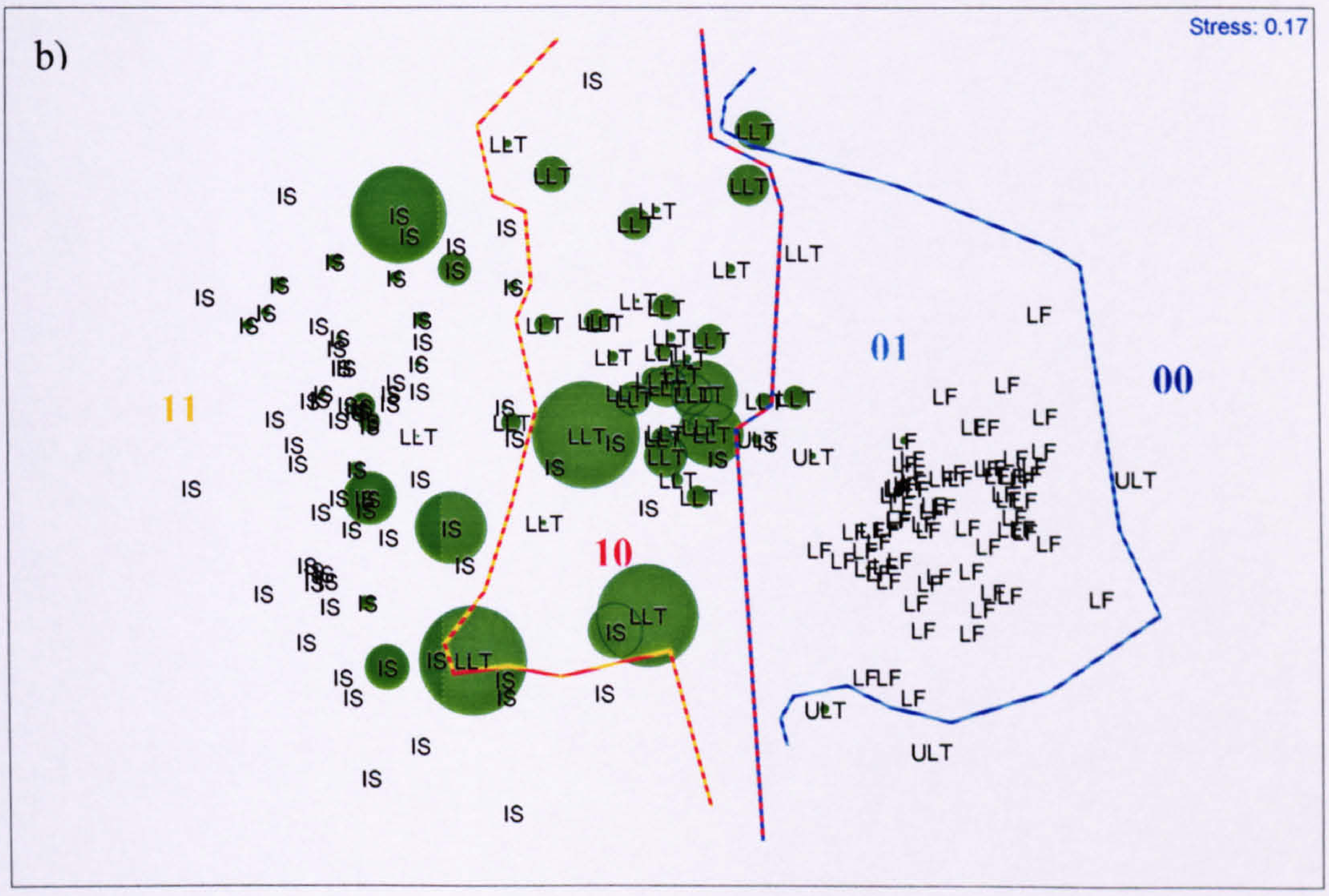


Figure 6.11 - MDS plots of stations by similarity of discard species composition by abundance. Bubble size refers to relative density of representative indicator species, chosen from each TWINSPAN group. Simplified TWINSPAN groups illustrated. Plots shown for TWINSPAN group 00; a) *V. mirabilis*, group 10; b) *C. pagurus*, group 11; c) *A. loveni*. (IS - Inner Sound, LLT – lower Loch Torridon, ULT – Upper Loch Torridon and LF – Loch Fyne) (when is stress<20 the ordination is useful).

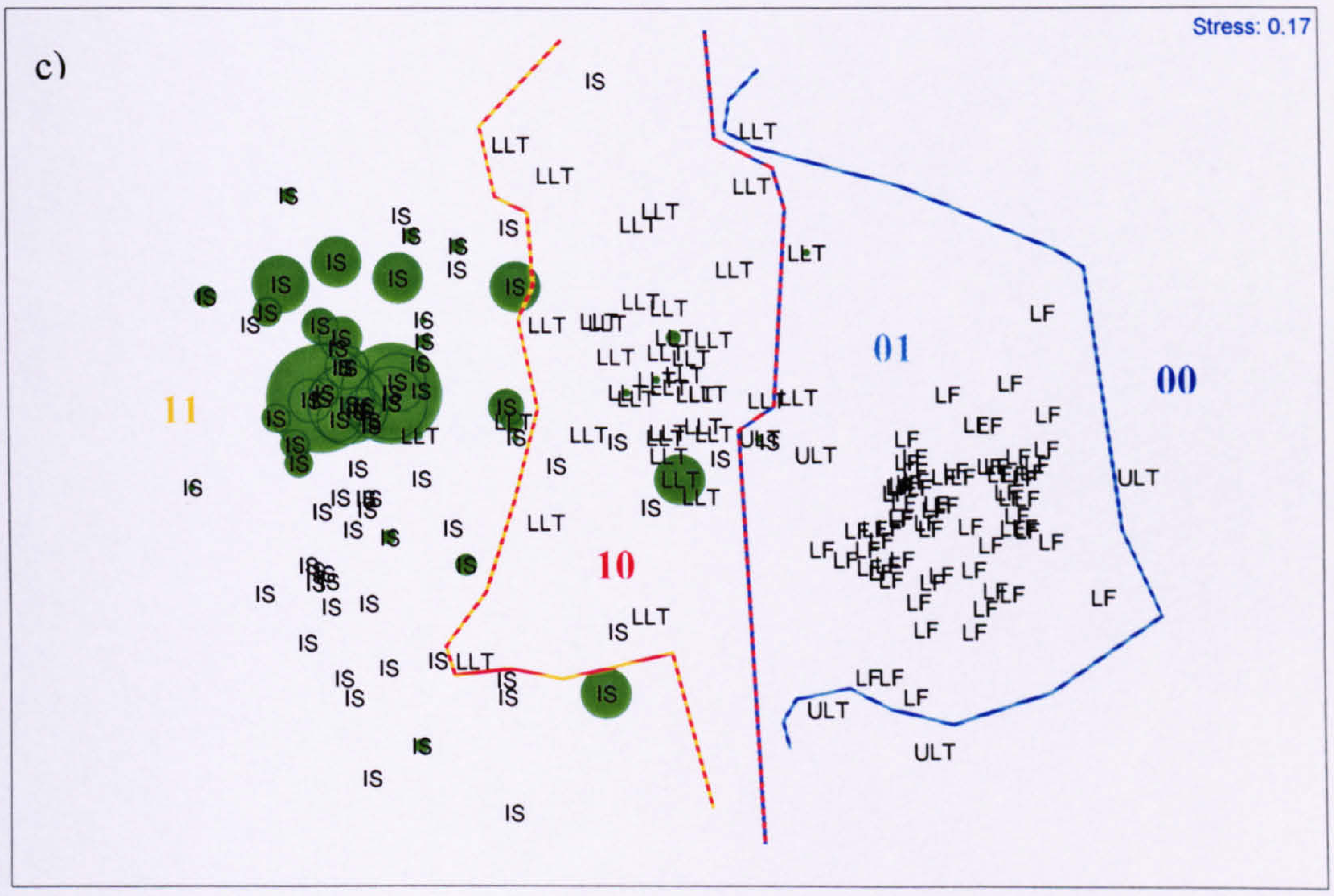
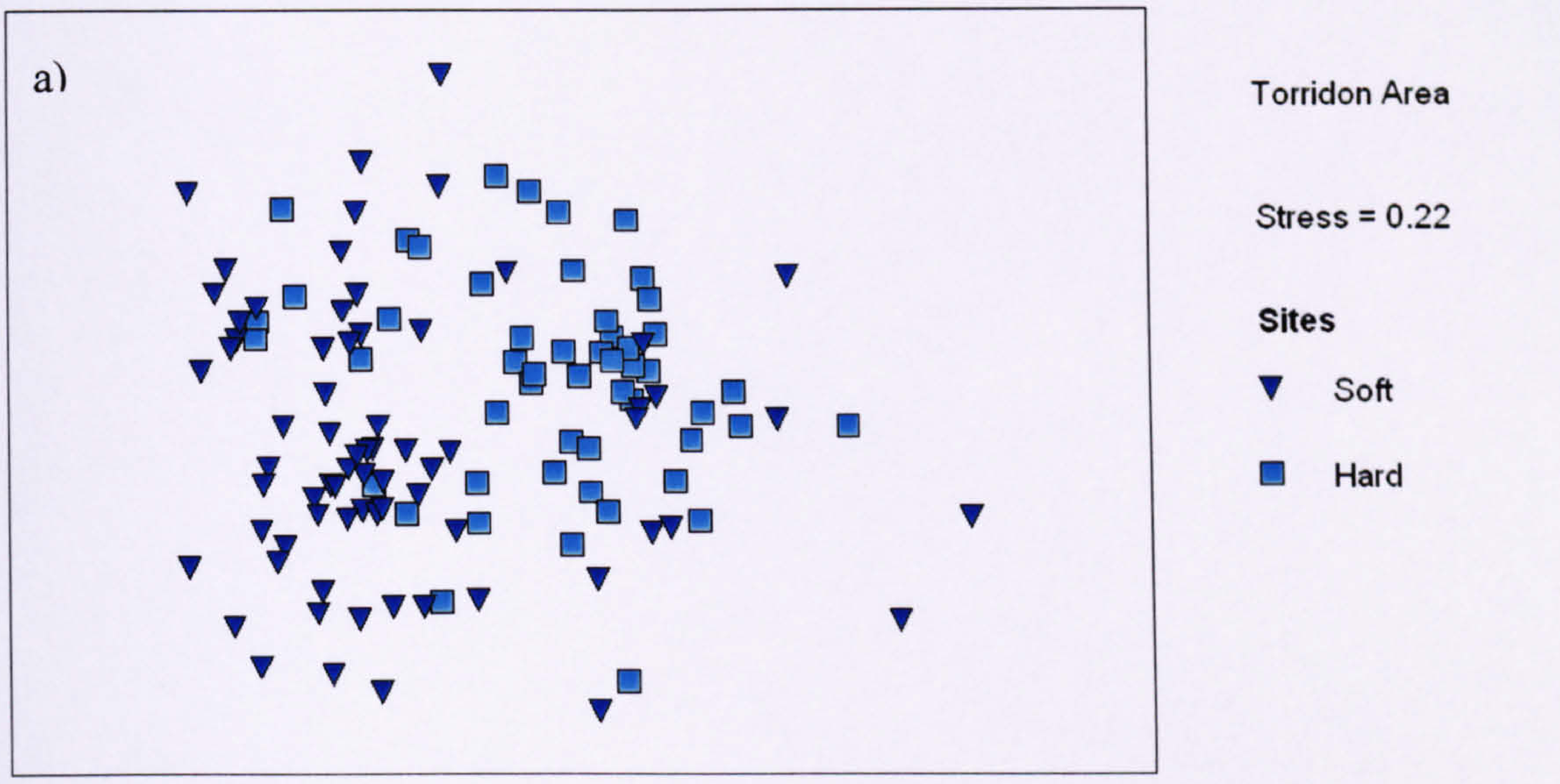


Figure 6.11 – (continued).



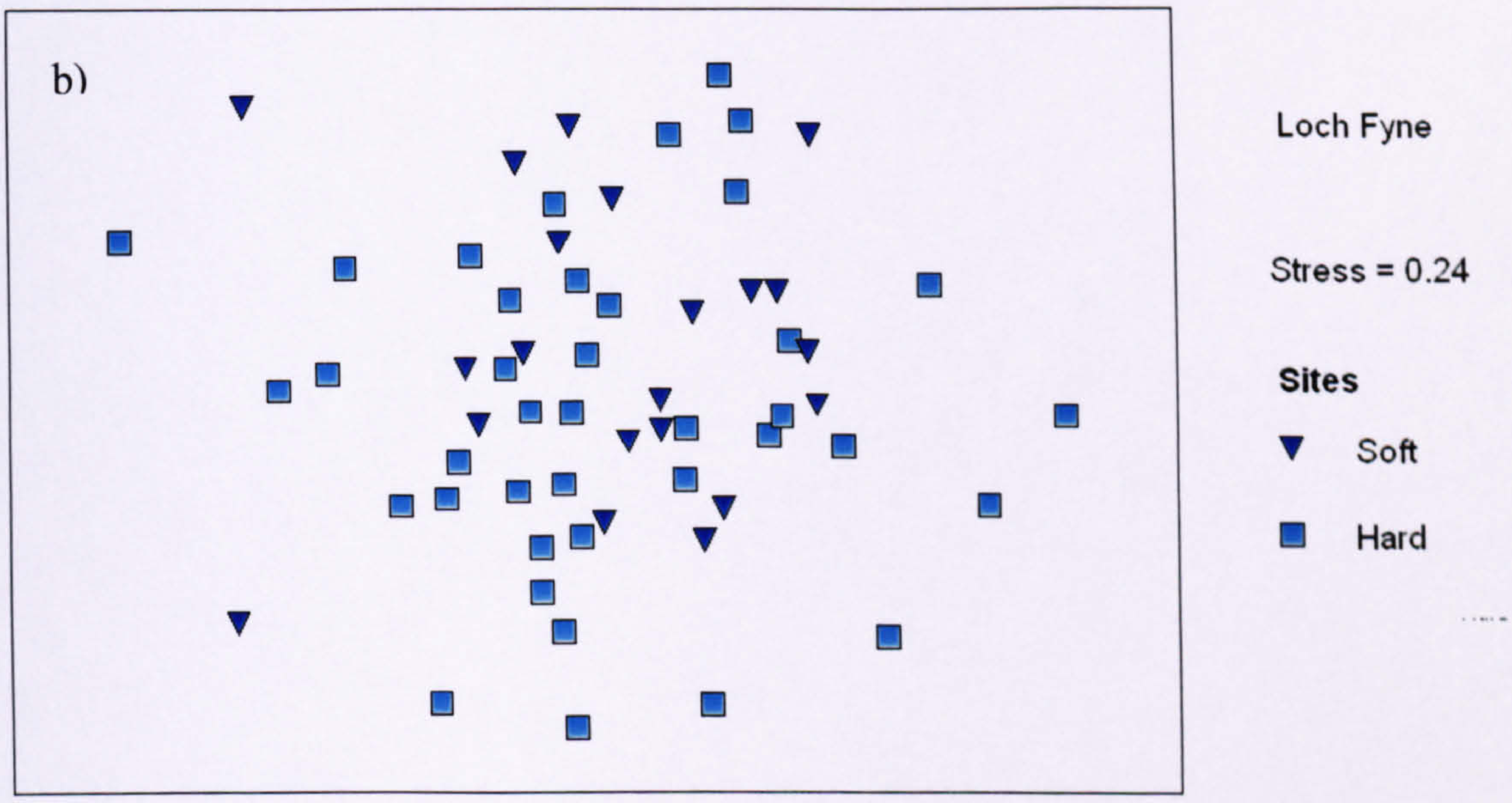


Figure 6.12 - MDS plots of stations by similarity of discard species composition by abundance from different sediment types in; (a) Torridon area and (b) Loch Fyne (when is stress>20 the ordination must be treated with caution).

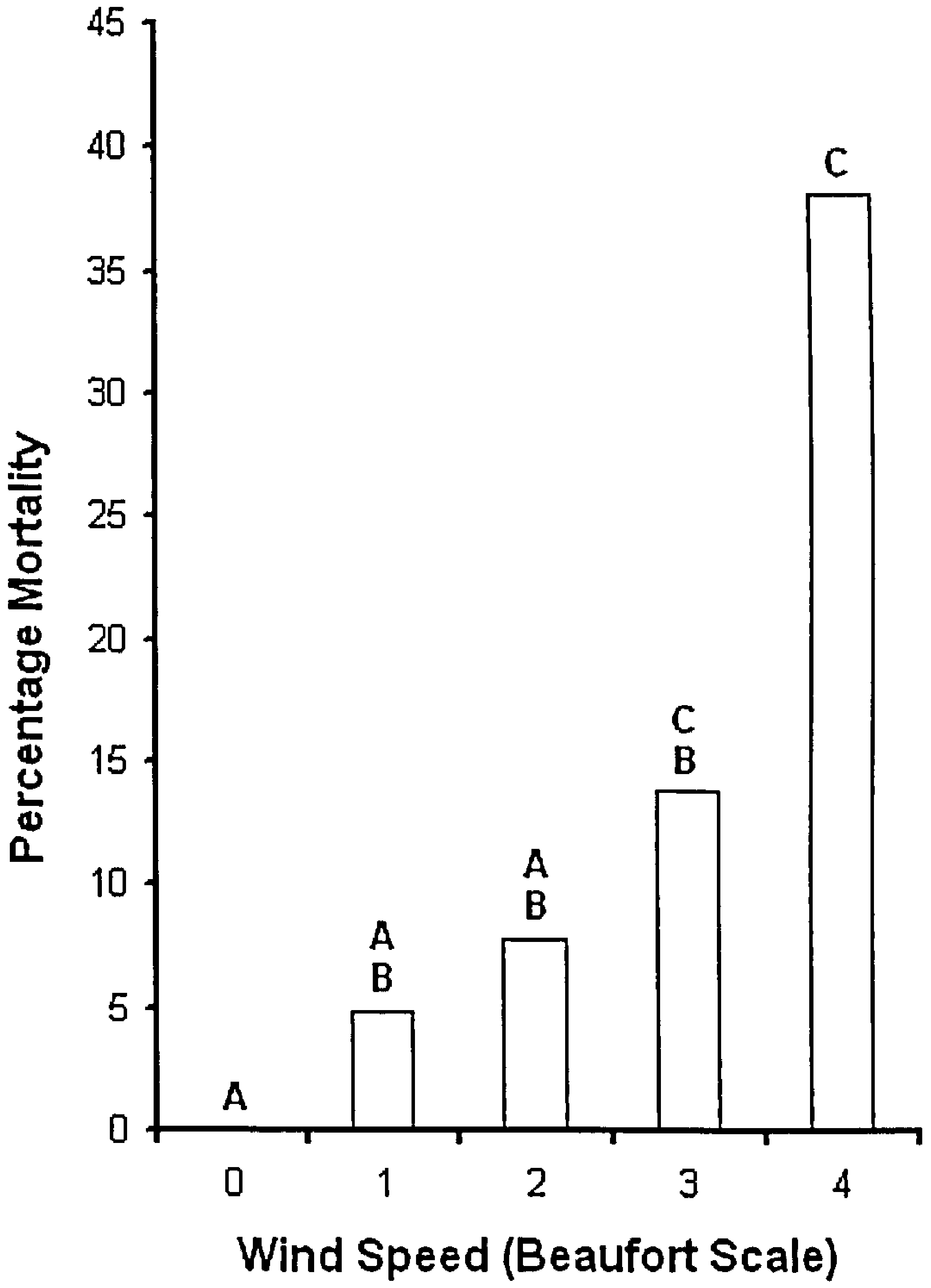


Figure 6.13 - Percentage of *Nephrops* discards from a fleet of creels consumed by seabirds at different wind speeds (11 fleets of creels sampled; 0 n =1, 1 n=4, 2 n=3, 3 n=2, 4 n=1). Within each wind speed different letters indicate significant differences (P<0.05).

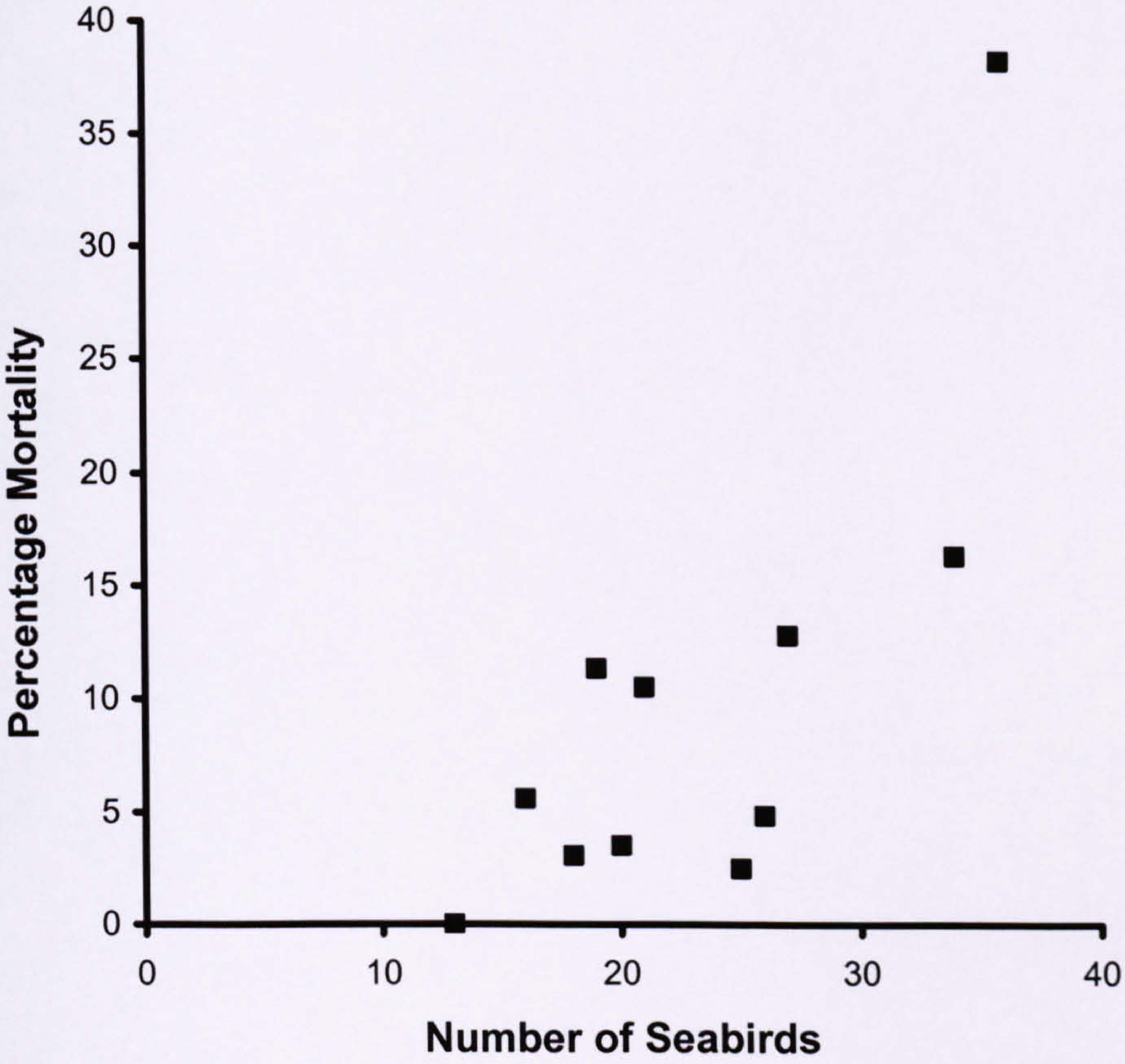


Figure 6.14 - Percentage of *Nephrops* discards consumed by different numbers of seabirds (each data point represents one fleet of creels).

Figure 6.15 - Seabirds feeding on discards from the *Nephrops* creel fishery.

Chapter 7

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7 ASSESSMENT OF 'GHOST FISHING' BY LOST GEAR

7.1 Introduction

Creels can be lost as a result of bad weather, by becoming snagged on the seabed or when they are towed away by mobile fishing gear. Lost gear has the potential to continue to fish for several years, a phenomenon known as 'ghost fishing' (Pawson, 2003). Ghost fishing has been described as one of the most serious negative impacts in the present capture fishery industry (FAO, 1995). If it were not for this potential problem, fishing with creels might be considered to be environmentally benign, since it is highly selective for the target species and has a low physical disturbance to the seabed, when compared with fishing using towed bottom gears (Jennings & Kaiser, 1998).

Little is known about the frequency of static gear loss or for how long gears continue to fish. In part, this is because fishermen are reluctant to report such incidents and also because ghost fishing studies are necessarily long term (Pawson, 2003). Estimates for the loss of pots vary greatly among studies, which is to be expected when comparing different fisheries. A fishery on a rocky substratum is likely to have greater loss than one on a soft substratum, owing to the increased likelihood of snagging and storm damage. The problem of estimating loss is highlighted by two separate studies on the North American Bristol Bay king crab (*Paralithodes camtschaticus*) fishery, which estimated the annual loss of pots to be 31,600 and 7,000, respectively (Kruse & Kimker, 1993; Paul *et al.*, 1994; Stevens, 1996, cited by Bullimore *et al.*, 2001).

Few studies have quantified the length of time lost creels or pots continue to fish. Attempts to quantify the age of lost nets from the degree of biofouling have been successful, but rates of biofouling are very dependent on local environmental conditions (Saldanha *et al.*, 2003). The duration of ghost fishing by lost creels is, however, likely to be considerable, on account of the thick, non-biodegradable netting and sturdy steel frame. Bullimore *et al.* (2001) found that parlour pots designed for crab (*Cancer pagurus*) and lobster (*Homarus gammarus*) continued to attract crustaceans and fish for more than one year.

Lost creels can cause mortality for a number of reasons. Animals may starve to death if they cannot escape, or predators or conspecifics that have also entered the creel may kill them. Predators and cannibals may subsequently escape or become trapped. Once the bait has been exhausted, a decrease in catch is often observed, but this does not mean that the creel ceases to fish, since the animals that die in the creels may effectively rebait them, leading to a continuous cycle of capture, decay and attraction for as long as the gear remains intact (Bullimore *et al.*, 2001).

There have been a number of studies to attempt to quantify the effects of ghost fishing on different species, most of which have looked at the effect of gill and trammel nets (Erzini *et al.*, 1997; Humborstad *et al.*, 2003; Revill & Dunlin, 2003; Saldanha *et al.*, 2003; Sancho *et al.*, 2003; Santos *et al.*, 2003a,b; Tschernij & Larsson, 2003). There have been several studies investigating the effects of ghost fishing by crustacean creels (High & Worlund, 1979; Breen, 1987; Parrish & Kazama, 1992; Kimker; 1994; Stevens *et al.*, 2000; Bullimore *et al.*, 2001; Herbert *et al.*, 2001; Godoy *et al.*, 2003).

A study by Breen (1987) clearly demonstrated the huge potential for creels to continue fishing and cause significant mortality. In that study, 10 baited Dungeness crab (*Cancer magister*) creels were observed by diving over a period of 12 months. Crabs comprised 55% of the ghost fishing catch, estimated to be about 7% of annual landings. Such high levels of mortality from ghost fishing have also been shown in other studies (Kimker; 1994; Bullimore *et al.*, 2001; Herbert *et al.*, 2001). In contrast, two tagging studies of king crabs (*P. camtschatica*) showed that they were able to escape from creels and ghost fishing mortality was low (High & Worlund, 1979; Godoy, 2003). Jury *et al.* (2001) found that 94% of *H. americanus* that entered creels were able to escape, further suggesting that the impact of ghost fishing will vary between species.

Most studies of ghost fishing have focused on the effect on the target populations. Bullimore *et al.* (2001) recorded catches of all species in a fleet of ghost fishing pots off the coast of Wales. They found that there was mortality not only of the target species, but also of other crustacean species and fish.

To overcome the problem of ghost fishing, several crustacean fisheries have introduced regulations requiring escape mechanisms to be fitted to creels. These escape mechanisms include openings to allow undersized animals to escape, biodegradable sections, and corrodable or biodegradable fasteners attached to opening panels (Miller, 1990). For example, the Alaska Board of Fisheries adopted a regulation in 1977, requiring escape panels made from biodegradable twine to be fitted to all crab and groundfish pots (Kimker, 1994). There is no such regulation in force in the UK.

There have been no previous attempts to quantify the effects of ghost fishing on *Nephrops* populations, yet it has been highlighted as one of the key issues in the Wester Ross Biodiversity Action Plan (Anon, 2004). Swarbrick & Arkley (2002) found that trawlers and scallop dredgers towing through fleets of *Nephrops* creels caused the majority of creel losses. There was no mention of losses due to snagging on the sea bed, which is unlikely to be a problem on the muddy sea beds typical of *Nephrops* grounds. Swarbrick & Arkley (2002) observed that it was common practice for creel fishermen to use only half of their gear and to store the rest in deep water, unbaited until required again. This is a cause for concern in relation to ghost fishing. It was reported that some fishermen left the creel doors open to prevent ghost fishing, but this practice was not universal.

There is clearly a need to quantify the economic and conservational cost of ghost fishing in the *Nephrops* creel fishery. The objectives of the current study were as follows: 1. to assess the capture and retention of both target and non-target species; 2. to quantify the resulting mortality; 3. to quantify the effect of escape panels; and 4. to observe how these factors change over time.

7.2 Methods

7.2.1 Study sites

Study sites were chosen to reflect two of the major habitats in which *Nephrops* creel fisheries are found:

1. Loch Sween, a shallow sheltered sea loch allowing for *in situ* observation by SCUBA divers.

2. Loch Torridon, a deep sea loch for hauling studies using a commercial fishing vessel.

A detailed description of each site and the location of the study sites are illustrated in Chapter 1, Sections 1.7.1 & 1.7.4, Figures 1.9 & 1.12.

7.2.2 Gear deployment and data recording

7.2.2.1 Loch Sween

Four fleets of five standard *Nephrops* creels each baited with salted herring or mackerel were placed in the study site on the 11th of November 2004. Creels fitted with escape gaps to facilitate the release of *Nephrops* of CL <40 mm, were interspersed with those without gaps. Each creel was clearly labeled to ensure that the contents of individual creels could be recorded on each sampling occasion. Creels were examined daily by divers for seven out of the first nine days, after two weeks and monthly thereafter for 346 days. Numbers and species of captured animals were recorded. The presence of dead animals was recorded, ensuring body parts were distinguished from moulted exoskeletons. Size and sex of captured *Nephrops* were noted, making it possible to distinguish individual animals between consecutive sampling occasions. Size was estimated by eye, because removing the animal for measurement would have disrupted the experiment. On the final sampling day, in addition to examination *in situ*, the creels were lifted, and the same parameters were recorded at the surface.

7.2.2.2 Loch Torridon

A fleet of 27 *Nephrops* creels, 16 with escape gaps and 11 without (all baited with salted herring), were deployed at a depth of 60 m by a commercial fishing vessel on the 11th of May 2005. Each creel was clearly labelled to ensure that the contents of individual creels could be recorded on each sampling occasion. Recording the catch of the experimental creel fleet, using an agreed protocol, was conducted by fisherman and/ or scientist when time and weather conditions permitted, resulting in the time interval between sampling being less controlled than that of the Loch Sween trial. On each sampling occasion, the creels were hauled to the surface for inspection on board the vessel and the number of each species captured in each creel was recorded. For reasons of practicality (needing to work at commercial speed, not always the same recorder), *Nephrops* size was noted as small (36–44 mm CL), medium (45–48 mm) or large (>48 mm) and sex was noted. Because the creels were hauled from depth, any fish with a swim bladder was unlikely to survive, due to the rapid change in pressure. Such fish were removed from the creels so they would not bias the results (i.e. by acting as bait). It can be assumed that this removal procedure will have had little effect on the findings, since the Loch Sween experiment had already shown that these species were able to escape from the creels. Once the catch was recorded, the creels were returned to the same location from which they were hauled.

7.2.3 Statistical analysis

Since it was necessary to vary the sampling period, the *Nephrops* catch data have been expressed as catch-per-unit-effort (CPUE) using the following formula (Bullimore *et al.*, 2001):

$$CPUE = \frac{N_{curr}}{E\left(t_{curr} - t_{prev}\right)},$$

where N_{curr} is the number of newly caught animals in the current sample, E is number of creels fishing and $t_{curr} - t_{prev}$ is the time interval since the previous sample.

Differences in numbers of selected species caught in the ghost fishing creels with escape gaps, and those without, were compared using one-way ANOVA tests. Additionally, in the Loch Sween experiment the two main phases, i.e. when bait was present (days 1–23) and when the bait was exhausted (days 24–346), were investigated. In this trial it was necessary to square-root transform the *L. depurator* data, as it was not normally distributed. The data for all other species were found to follow a normal distribution and therefore no transformations were necessary.

The length of time that individual *Nephrops* remained captured (residence time) was estimated for the Loch Sween creel deployment. Differences in residence time among *Nephrops* size categories were tested with a one-way ANOVA. The relationship between the date on which a *Nephrops* entered the creel and residence time was also investigated. These analyses were not possible with the Loch Torridon deployment, as the method of recording the size of the animals was insufficiently precise to enable the unambiguous recognition of individual animals on subsequent occasions.

7.3 Results

7.3.1 Loch Sween

Bait in the creels was consumed rapidly with only seven of the 20 creels having remnants of bait by day 9 (Table 7.1; Figure 7.1). Throughout the experiment most of the creels were left undisturbed until day 299, when five were found to have been moved from their original position. These creels were not discounted, as from discussion with commercial fishermen it is known that under normal conditions other fishing gear may often relocate ‘ghost fishing’ creels.

In total, 10 different species were observed, with only *Nephrops, Liocarcinus depurator* and *Carcinus maenas* being recorded in large numbers. *Liocarcinus depurator* appeared in large numbers in the creels on the first day, followed by a rapid decline for the following nine days, then declining more slowly until the bait was exhausted, after which numbers fluctuated at fewer than 10 animals. *Carcinus maenas* followed a more gradual increase in numbers, with the maximum being observed on day 6, after which there was a slow decline. Beyond day 23, the numbers of *C. maenas* fluctuated without obvious trend. *Nephrops* were third in abundance. The total number of *Nephrops* increased until day 7, remained relatively stable until day 60, after which there was a gradual decline (Table 7.1; Figure 7.1). Fish, including *Gadus morhua*, *Trisopterus minutus* and *Scyliorhinus canicula*, were caught in small numbers and their pattern of occurrence suggested that they were able to enter and leave creels. After three weeks when the bait was exhausted, few specimens of non-target bycatch were present in the creels (Table 7.1; Figure 7.1).

There were two observed mortalities of *Nephrops* during the trial, on days 64 and 212. Deaths of some crab species were observed between days 97 and 154 (*C. maenas* n = 12, *L. depurator* n = 5, *C. pagurus* n = 1), leading to a short-term increase in small numbers of scavengers, including *Asterias rubens* and *C. pagurus*. *Nephrops* were not found to enter the creels during the period when crab deaths were observed.

Slight increases in a number of species were observed at the end of the experiment (Figure 7.1), coinciding with a large settlement of tubicolous polychaetes (*Sabella pavonina*) on the creels. Inspection of the creels at the surface at the end of the experiment revealed that 28% of *C. maenas* and 16% of *L. depurator* had not been counted in the final *in situ* inspection. This was due to the dense growths of *S. pavonina* obscuring the interior of the creels. It must therefore be assumed that on the last two sampling occasions, the number of animals observed within the creels was slightly underestimated, but prior to this visibility was good.

Observations of *Nephrops* activity showed that the majority of entrances occurred during the first few days when bait was still present. Following this the number of entrances remained very low, with none observed between days 182 and 299 (Figure 7.2). The higher catches when the bait was present are clearly illustrated when the catch is expressed as CPUE (Figure 7.3). A low escape rate from creels was observed, with the greatest number of animals found to escape when bait was still present, fluctuating thereafter (Figure 7.2).

The large numbers of *Nephrops* caught initially, followed by low escape rate and low mortality levels, imply that the average residence time for a single *Nephrops* was 111.67 days. There was no significant effect of size ($F_{2,22}$=2.16, P=0.14) or presence of escape gaps on residence time ($F_{1,23}$=0.82, P=0.375), nor was there a significant relationship between residence time and date of entry ($F_{1,23}$=0.00, P=0.968).

The apparent mortality of *Nephrops* was very low (7%), with the majority of captured animals being able to escape after often long durations of capture (78%), with 15% of animals remaining captured. *Nephrops* observed in the creels were predominantly large and medium-sized animals (52% and 44%, respectively), with few small animals (4%).

There were significantly more *L. depurator* ($F_{1,32}$=6.99, P<0.05) in the creels without gaps and significantly more *Nephrops* ($F_{1,32}$=6.04, P<0.05) in the creels with gaps. During the time when no bait remained in the creels (days 24–346) there were significantly more *L. depurator* ($F_{1,16}$=8.11, P<0.05), *C. maenas* ($F_{1,16}$=4.9, P<0.05) (Figure 7.4) and *A. rubens* ($F_{1,16}$=4.5, P=0.05) in the creels without gaps.

7.3.2 Loch Torridon

For practical reasons, it was not possible to sample the ghost fishing fleet for 22 days after its initial deployment, following which creels were sampled over variable, but generally progressively longer time periods. Remnants of bait in the creels were present on day 22, but no traces were found by day 37.

Throughout the experiment, 14 different species were caught, with *Nephrops* being the most abundant. The majority of animals in the creels on day 22 were *Nephrops,* suggesting that during the first few weeks, large numbers of *Nephrops* were caught. This was followed by a marked decline in numbers until day 123, after which time numbers remained low (Table 7.2; Figure 7.5). During the decline in *Nephrops* numbers, *Porania pulvillus, L. depurator, Myoxocephalus scorpius* and *T. minutus* were observed to increase in numbers. The increase of *P. pulvillus, M. scorpius* and *T. minutus* was followed by a subsequent decrease in numbers. By day 168 all three species were observed either in very low numbers or were not present in the creels (Table 7.2; Figure 7.5). Following the increase of *L. depurator*, their numbers remained high for the rest of the experiment. All other species were observed in small numbers at different times during the deployment (Table 7.2; Figure 7.5), during which no mortalities of any species were observed.

Nephrops activity was more difficult to define in this experiment because of the generalised size recording of individual animals and the longer time period between sampling. However, the data suggest that the majority of creel entries occurred when the bait was still present. Following this, the number of creel entrances remained low (Figure 7.6). The larger catches when the bait was present is clearly illustrated by looking at the catch in terms of CPUE (Figure 7.7). Similarly, the decrease in entrances of *Nephrops* to creels was accompanied by a decrease in the number of escapes, following which escapes remained low (Figure 7.6). No dead *Nephrops* (either whole or fragments) were observed suggesting that the majority were able to escape (94%). The relative abundance of the three size categories of observed *Nephrops* was: small 68%, medium 31%, large 1%.

There were no significant differences in the numbers of animals of any species between creels with escape gaps and those without. There was a suggestion that there were more *L. depurator* and *Pleuronectes platessa* in creels without escape gaps; however, possibly due to the small number of animals observed, no significant differences were found ($F_{1,12}$=2.51, P=0.139 and $F_{1,12}$=2.4, P=0.147, respectivly).

7.4 Discussion

A number of different species were observed in the ghost-fishing creels in both trials, with variations in the species composition between sites reflecting different environments in which the creels were located. All species were shown to be able to exit from the creels following capture, although there was an apparent difference in a species ability to escape the creels, reflected in the time periods spent within the creel. The ability of animals to escape is likely to be due to the simple design of the single-compartment creels, as well as the 'hard' eyes providing a more accessible exit than is found in creels with more complicated design and those with 'soft' eyes (no eye ring, Chapter 1; Figure 1.3).

Nephrops was the only species observed to stay in the creels for long periods of time. This indicates that *Nephrops* creels are highly selective for their target species. *Nephrops* in the Loch Torridon deployment appeared to be able to escape more easily than those in the Loch Sween experiment. This may reflect the size of animals caught in each location, as it seems likely that smaller animals, such as those observed in the Loch Torridon deployment, would be able to escape more easily (Livingstone, 2001; Chapter 2). The difference in *Nephrops* size between trials may be a reflection of the time of year the trials commenced and environmental conditions (Chapter 2).

Results from both experiments suggest that mortality of *Nephrops* trapped in lost creels is very low, with only two observed mortalities in the Loch Sween experiment and none in the Loch Torridon study. It appears that *Nephrops* can survive for long periods within creels and the majority are eventually able to escape. The fact that there were no observed mortalities in Loch Torridon may be a reflection of their smaller size allowing them to escape more readily. The two mortalities of *Nephrops* in the Loch Sween experiment coincided with entrances of additional *Nephrops*. These entrances were not observed in the creels in which the dead animals were found, proving no direct evidence that these mortalities attracted the new *Nephrops* into the creels. It has been suggested that some dead or damaged crustaceans may release a chemical substance eliciting an avoidance reaction in others of the same species (Hancock, 1974). This avoidance of creels containing dead conspecifics was observed in *C. pagurus* (Chapman & Smith,

1974) and *Nephrops* (Chapman, 1981), suggesting that this may be the case in the *Nephrops* creel fishery.

Several uncertainties must be considered when estimating mortality rates. In particular, dead *Nephrops* may be consumed by predators or, once dead, be eaten by other animals within the creel. This may have occurred in the case of Loch Torridon, as there were often long time periods between observations and body parts may have been lost from the creels during hauling to the surface. In the Loch Sween experiment, this uncertainty is much lower, as the creels were observed *in situ*, enabling body parts to be used to indicate the number of dead animals. Furthermore, individual animals were observed to inhabit the creels for long time periods, confirming that mortality due to starvation or predation was very low.

Mortality can be overestimated if the number of animals that have entered and left the creels between observations is high, since mortality is estimated from the number of live and dead animals in the creel when the observations take place (Godoy *et al.*, 2003). The Loch Sween trial showed that individual animals were resident in the creels for long time periods, suggesting that this error is likely to be low.

Some mortalities of the crab species *C. maenas*, *L. depurator* and *C. pagurus* were observed in the Loch Sween experiment, with the majority found between February and April. It is likely that mortalities occurred due to crabs moulting in the creel, making them vulnerable to other animals. Discussions with a number of fishermen have suggested that some crabs, particularly *L. depurator*, are good bait for *Nephrops*. However, during this time of high crab mortailty, no new *Nephrops* were observed in the creels suggesting that these crabs did not act as bait for *Nephrops* during the experiment.

Changes in the numbers of species observed throughout each experiment showed similar patterns in animal abundance. The initial two phases of ghost fishing described by Miller (1990), were clearly shown in the Loch Sween deployment: a short phase of high catch rates when the bait is an attractant followed by a long phase of low catch rates as the effectiveness of the bait diminishes. The first phase was not apparent in the Loch Torridon deployment, as the creels were first sampled after 22 days. Both deployments showed that following this initial period low catch rates occurred.

Of particular concern for the phenomenon of ghost fishing is when animals die within the creel, increasing the catching effectiveness of the creel (Bullimore *et al.*, 2001; Matsuoka *et al.*, 2005). Although there was some mortality of animals observed in the Loch Sween experiment, this was followed by only small increases in numbers of a few non-target species. There is therefore little evidence from the present work to suggest that once lost, *Nephrops* creels continue to fish by self re-baiting.

Another phase was observed in the Loch Sween experiment. Towards the end of the creel deployment, an increase in the number of *C. maenas, L. depurator, A. rubens* and *Crossaster papposus* was observed, coinciding with dense aggregations of a sabellid polychaete (*S. pavonina*) on the creels. Matsuoka *et al.* (2005) noted that the 'ghost fishing' effectiveness of creels reduces with the accumulation of fouling organisms. It can therefore be suggested that these animals observed within the creels, at the end of the experiment, were not actually trapped. It is possible that they were using the creels as a shelter providing protection from predation. An analogous increase was not observed in the Loch Torridon experiment; however, on day 217, when the creels were finally lifted, several of the creels had a number of tunicates attached to them. It is possible that if these creels were left for much longer a similar result may have been found to that observed in Loch Sween.

The Loch Sween study indicated that escape gaps were effective at reducing the numbers of *L. depurator* captured in the creels and, over the long term, escape gaps also reduced the numbers of *C. maenas* and *A. rubens* retained. A probable reason for the escape gaps being so effective for *L. depurator* is that its morphology and mode of locomotion would enable it to pass through the escape gap relatively easily. For other non-target species, their size meant that the vast majority of individuals were unable to pass through the escape gap and therefore there were no differences in the numbers of animals observed between the two creel types. In contrast to *L. depurator, C. maenas* and *A. rubens*, there were significantly more *Nephrops* observed in creels without gaps. Behavioural studies indicate that *Nephrops* are deterred from entering creels containing *L. depurator* (Chapter 4). It is therefore possible that, by reducing the number of *L. depurator* present, escape gaps tend to increase the number of *Nephrops* captured in ghost fishing creels.

The experiment in Loch Torridon showed no significant differences in numbers of any of the species observed between creels with, and without, escape gaps. For non-target species, this may have been because there were too few animals for adequate analysis. The small number of non-target species and their possible deterrent effect may also account for there being no significant difference in the number of *Nephrops* between creels with and without escape gaps.

Both experiments have shown that in contrast to the conclusions of the majority of previous studies, ghost fishing is unlikely to be a problem for the *Nephrops* creel fishery, as long as the creels remain on *Nephrops* ground. From the present work, it can be assumed that after the initial capture period, when bait is still present, the majority of non-target species will be able to escape in a relatively short space of time. Those *Nephrops* that have been caught may stay in the creel for long periods, but are likely to escape eventually, with only a small percentage dying as a result of being caught. In addition to this, mortalities of target or non-target species seem to be relatively ineffective in attracting *Nephrops* into the creels. The main reasons for the low ghost-fishing effectiveness of *Nephrops* creels are; the design of the creel itself allowing non-target species to escape relatively easily ('hard' eyes), the behaviour of *Nephrops* restricting the number of animals entering the creels once the bait has been consumed and the ability of *Nephrops* to survive for long periods of time once caught (the majority of which are eventually able to escape).

It is also interesting to note that the fishermen report that, in the Torridon area, recovered lost creels contain little catch of any species. Creel loss in the area is now infrequent because of the exclusion of trawlers.

7.5 Summary

- The effects of 'ghost fishing' by *Nephrops* creels were investigated using two different methods: assessing catch and bycatch at regular intervals by SCUBA diving in a shallow site and periodically hauling creels in a deep-water site. These studies were carried out over a period of one year allowing for any evidence of ghost fishing to be identified.

- Creels were found to be very selective for their target species, with both target and non-target species observed to escape.

- Of those *Nephrops* that were caught, several remained in the creel for long periods, but the majority of these animals eventually escaped.

- There was no evidence of re-baiting from the small number of target or non-target species that died.

- Fewer crabs were retained in creels with escape gaps, which resulted in more *Nephrops* being caught in these creels.

- Lost creels do not constitute a serious issue in this fishery.

- Reasons suggested for the observed results are: the design of the creel allowing non-target species to escape relatively easily, the behaviour of *Nephrops* restricting the number of animals entering the creels and the ability of *Nephrops* to survive for long periods of time once caught, with the majority being able to eventually escape.

Table 7.1 - Numbers of individuals observed in a fleet of 20 creels deployed in Loch Sween in relation to time from deployment. The mean catch (animals / 20 creels) over the duration of the experiment, presence of bait and *Nephrops* fate also shown.

		Days from deployment																	
		1	2	5	6	7	8	9	23	64	97	128	154	182	212	261	299	346	
		Number of creels with bait remaining																	
		20	20	14	13	10	8	7	0	0	0	0	0	0	0	0	0	0	
Species	Common Name	Number of animals observed																	Mean Catch
Nephrops norvegicus	Norway lobster	7	7	16	16	17	17	17	18	16	13	11	7	6	7	6	4	4	11.12
Carcinus maenas	shore crab	19	36	34	44	33	37	30	34	18	31	25	18	13	13	21	20	24	26.47
Cancer pagurus	brown crab	0	1	0	1	2	3	3	1	3	4	5	3	0	0	1	2	1	1.76
Liocarcinus depurator	harbour crab	72	71	46	36	33	26	23	11	11	7	6	2	3	4	3	6	11	21.82
Asterias rubens	common starfish	0	0	1	1	0	0	0	1	2	2	1	2	0	0	0	3	5	1.06
Crossaster papposus	sunstar	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	2	0.24
Aequipecten opercularis	queen scallop	0	0	0	0	0	1	1	1	1	0	0	1	1	1	0	0	0	0.41
Trisopterus minutus	poor cod	1	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	0.24
Scyliorhinus canicula	dogfish	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.12
Gadus morhua	cod	1	0	2	2	1	0	1	0	0	0	0	0	0	0	0	0	0	0.41
Number of Species		5	6	6	6	5	5	6	6	7	5	4	6	4	4	5	6	8	
		Nephrops fate																	Total Fate
	Entrances	7	1	11	0	1	0	0	1	1	0	0	0	0	2	1	0	1	26
	Escapes	0	1	2	0	0	0	0	0	2	3	2	4	1	0	2	2	1	20
	Dead	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	0	2

Table 7.2 - Animals observed in entire fleet of 27 creels deployed in Loch Torridon at time intervals from first deployment of gear. Mean catch over 217 days, presence of bait and *Nephrops* fate also shown.

		Days from deployment							
		22	37	62	88	123	168	217	
		Number of creels with bait							
		14	0	0	0	0	0	0	
Species	Common Name	Number of animals observed							Mean Catch
Nephrops norvegicus	Norway lobster	35	21	11	8	2	4	3	12.00
Cancer pagurus	brown crab	0	0	0	0	2	5	5	1.71
Liocarcinus depurator	harbour crab	0	0	4	12	11	12	7	6.57
Munida rugosa	squat lobster	0	0	0	0	0	0	1	0.14
Necora puber	velvet swimming crab	0	0	0	0	0	0	1	0.14
Porania pulvillus	cushion star	1	1	9	0	3	2	3	2.71
Luidia ciliaris	spiny starfish	0	0	0	0	0	2	2	0.57
Buccinum undatum	common whelk	3	2	3	2	3	3	3	2.71
Trisopterus minutus	poor cod	2	0	2	5	1	1	1	1.71
Myoxocephalus scorpius	Father lasher	2	0	2	10	4	0	0	2.57
Zeus faber	John Dory	1	0	0	0	0	0	0	0.14
Zeugopterus punctatus	topknot	0	0	0	1	0	1	1	0.43
Pleuronectes platessa	plaice	0	3	1	0	0	1	0	0.71
Labrus mixtus	cuckoo wrasse	0	0	0	1	0	0	0	0.14
Number of Species		6	4	7	7	7	9	10	
		Nephrops fate							Total Activity
	Entrances	35	6	3	2	1	3	3	53
	Escapes		22	13	5	6	0	4	50
	Dead		0	0	0	0	0	0	0

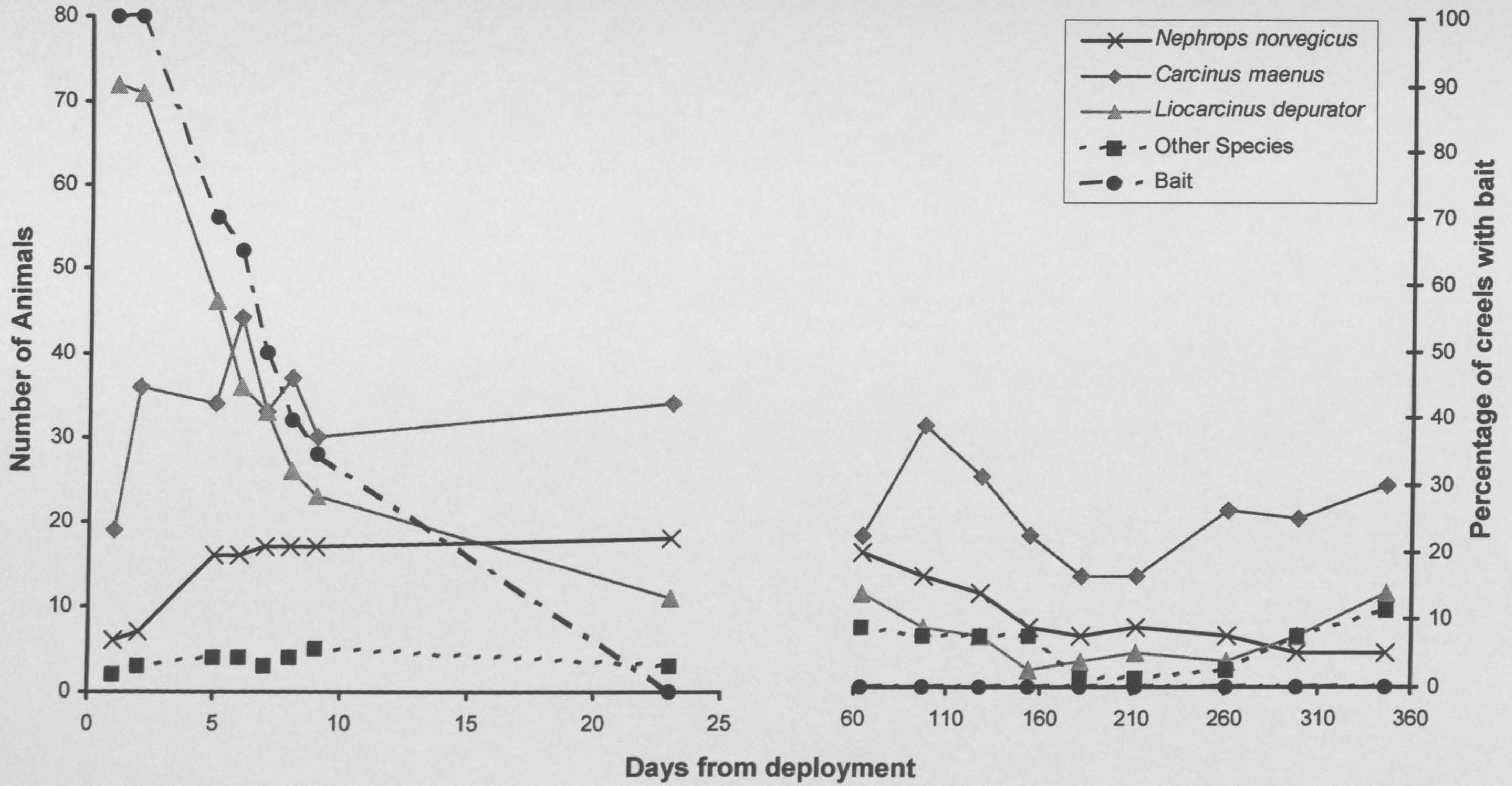

Figure 7.1 - Numbers of each species observed in a fleet of 20 creels deployed in Loch Sween in relation to time since deployment. The percentage of creels with bait remaining is also indicated (other species - total catch of *Cancer pagurus, Asterias rubens, Crossaster papposus, Aequipecten opercularis, Trisopterus minutus, Scyliorhinus canicula* and *Gadus morhua)*.

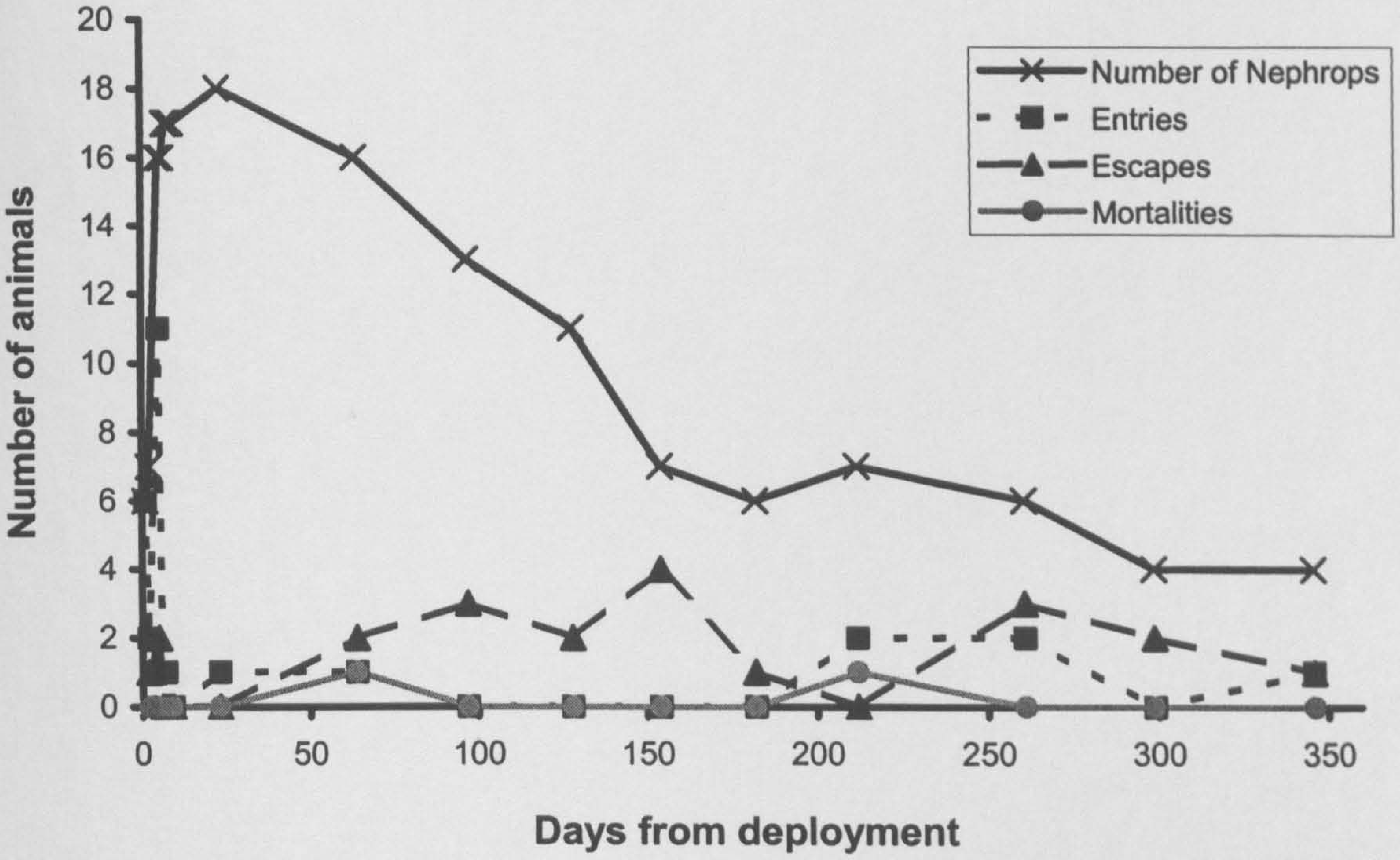

Figure 7.2 - Entries, exits and deaths of *Nephrops* in a fleet of 20 creels deployed in Loch Sween in relation to time since deployment.

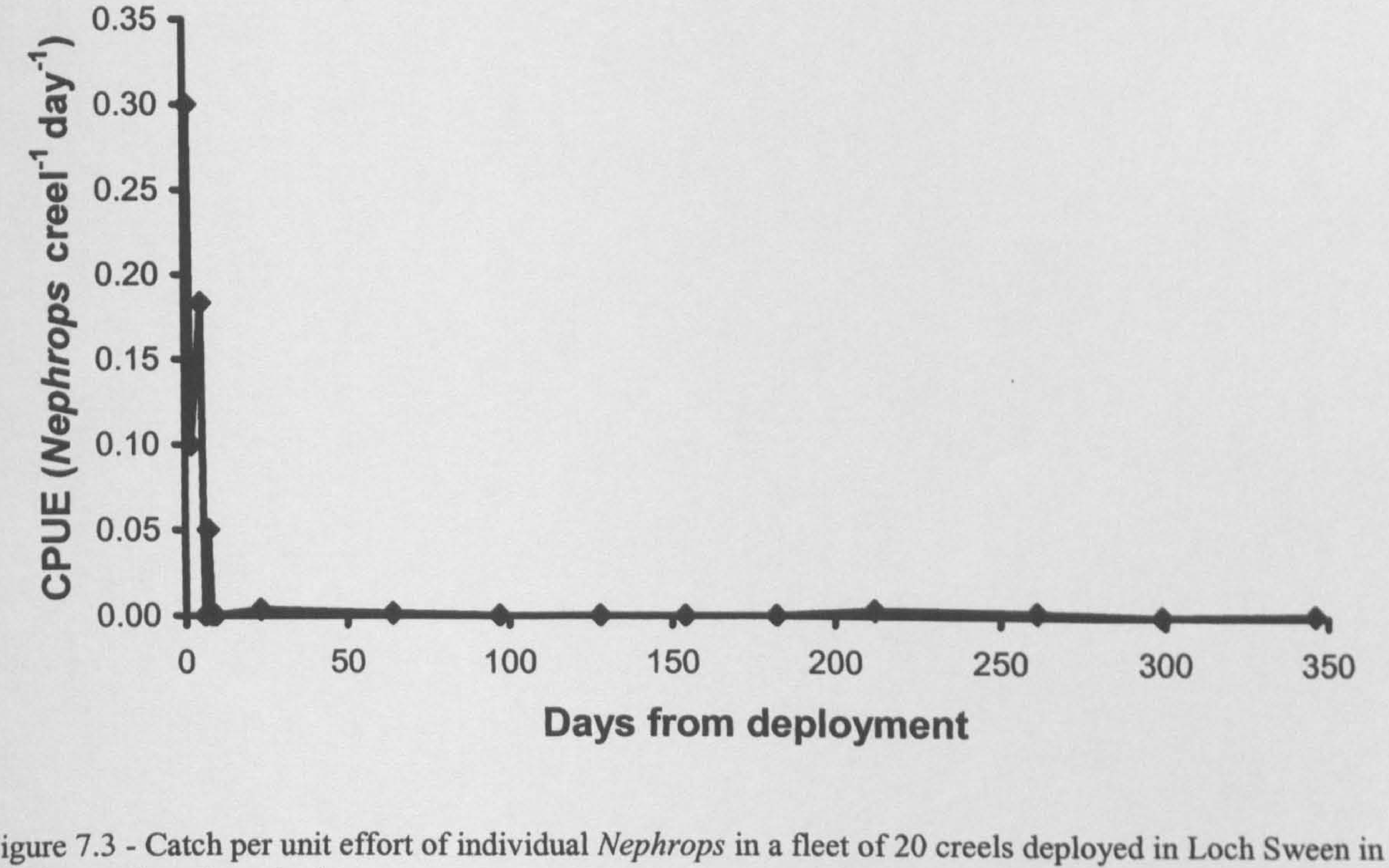

Figure 7.3 - Catch per unit effort of individual *Nephrops* in a fleet of 20 creels deployed in Loch Sween in relation to time since deployment.

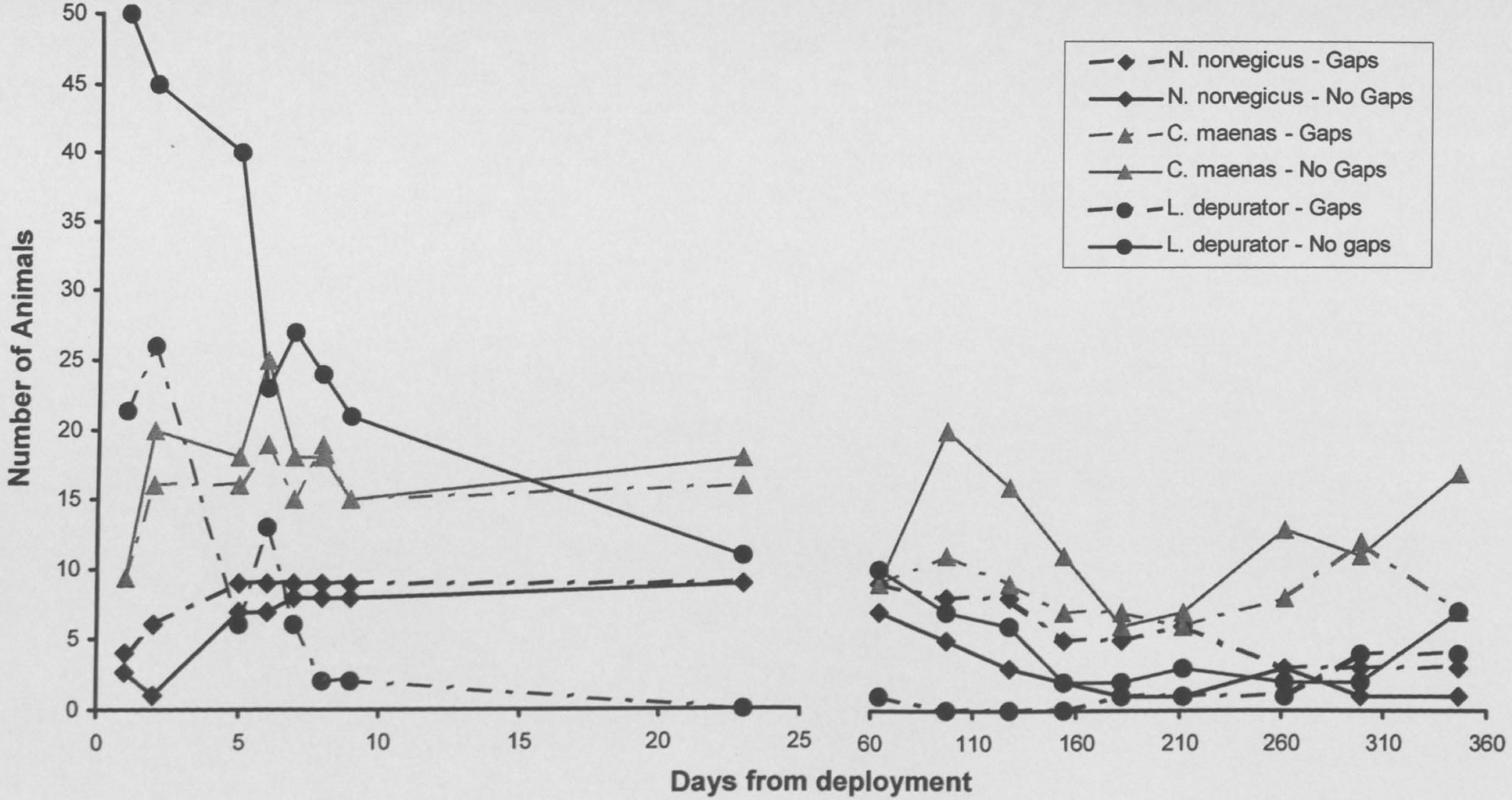

Figure 7.4 - Number of animals of in creels deployed in Loch Sween with and without gaps in relation to time since deployment.

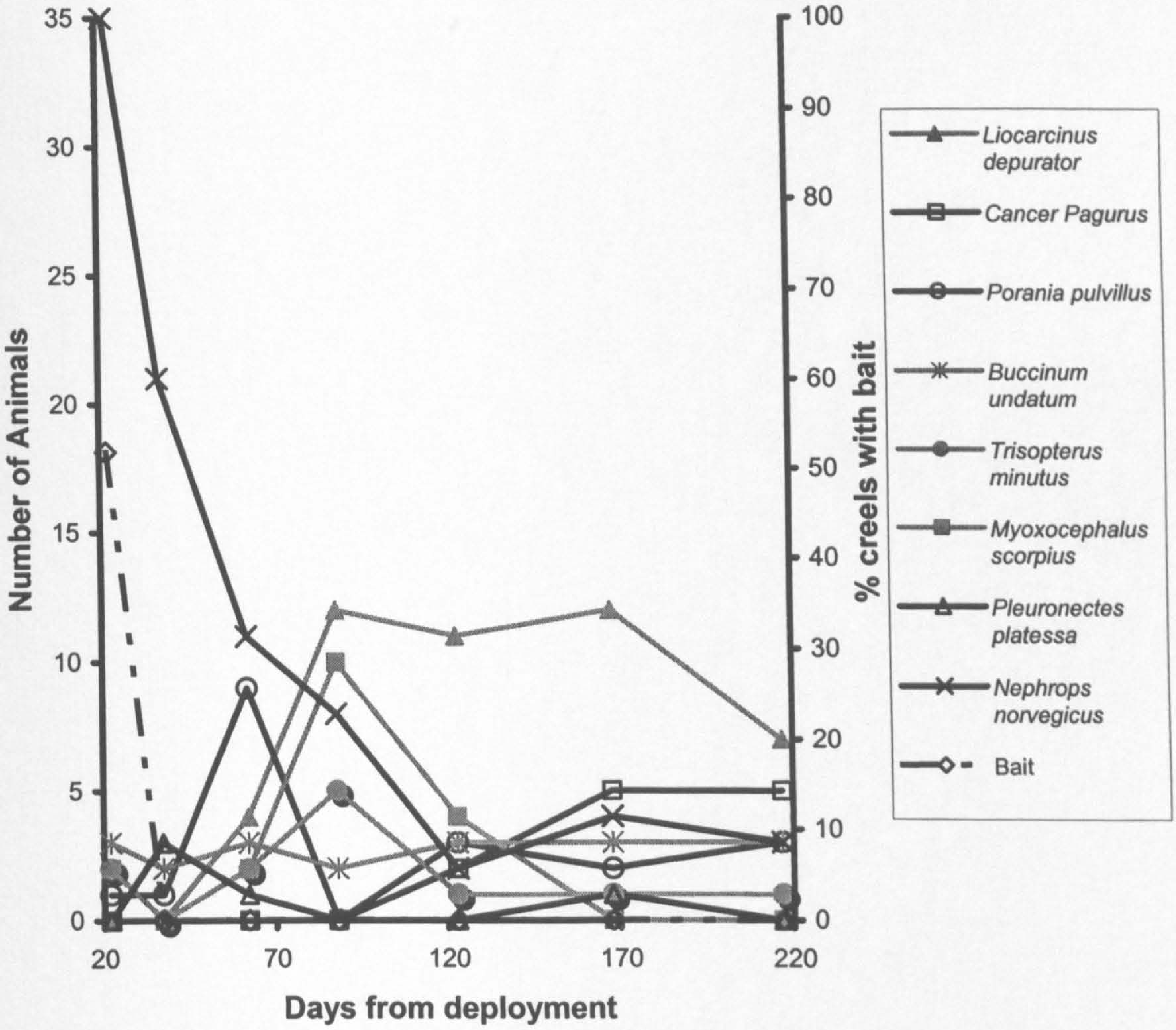

Figure 7.5 - Numbers of each species observed in a fleet of 27 creels deployed in Loch Torridon in relation to time since deployment. The percentage of creels with bait remaining is also indicated.

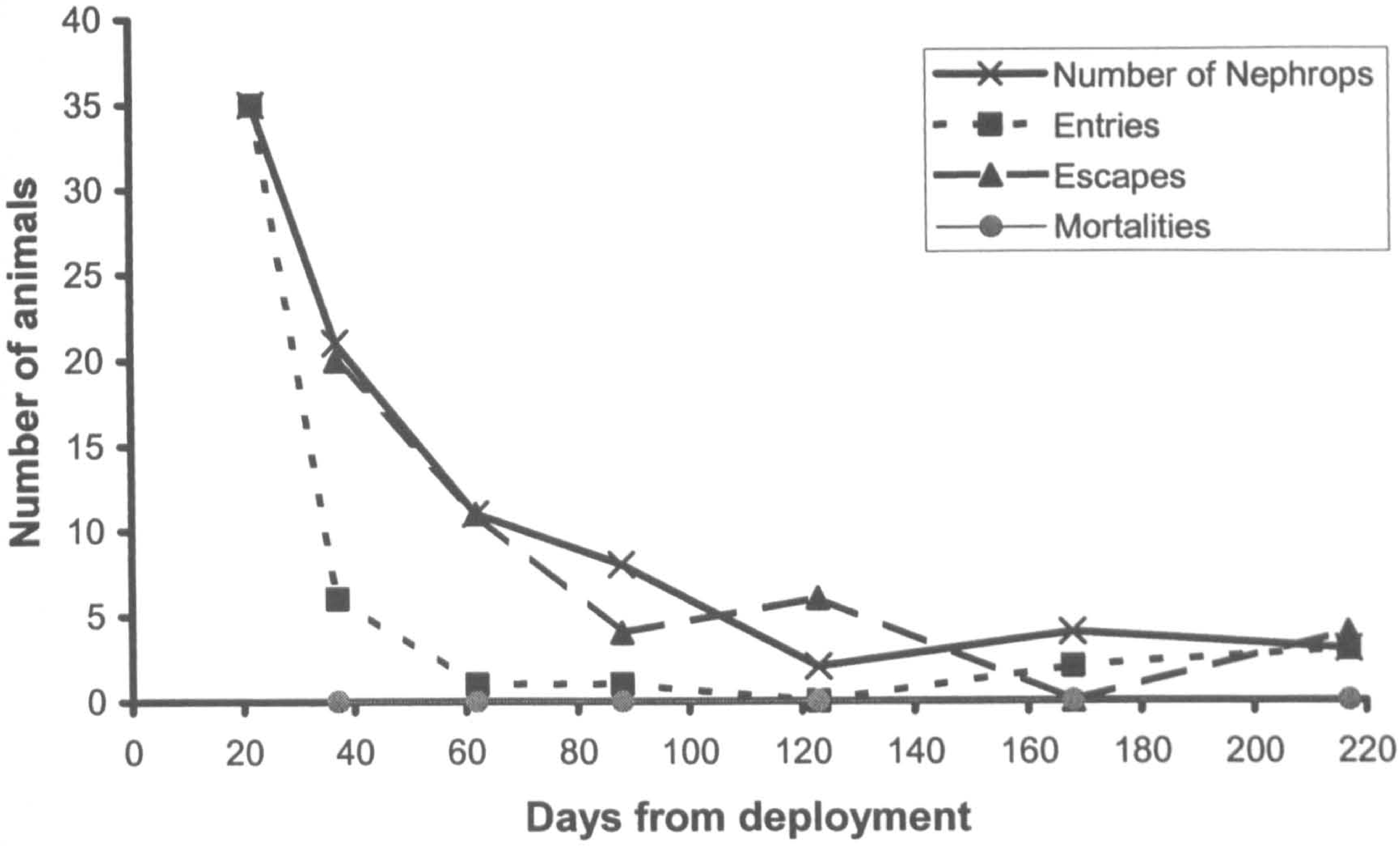

Figure 7.6 - Entries, exits and deaths of *Nephrops* in a fleet of 27 creels deployed in Loch Torridon in relation to time since deployment.

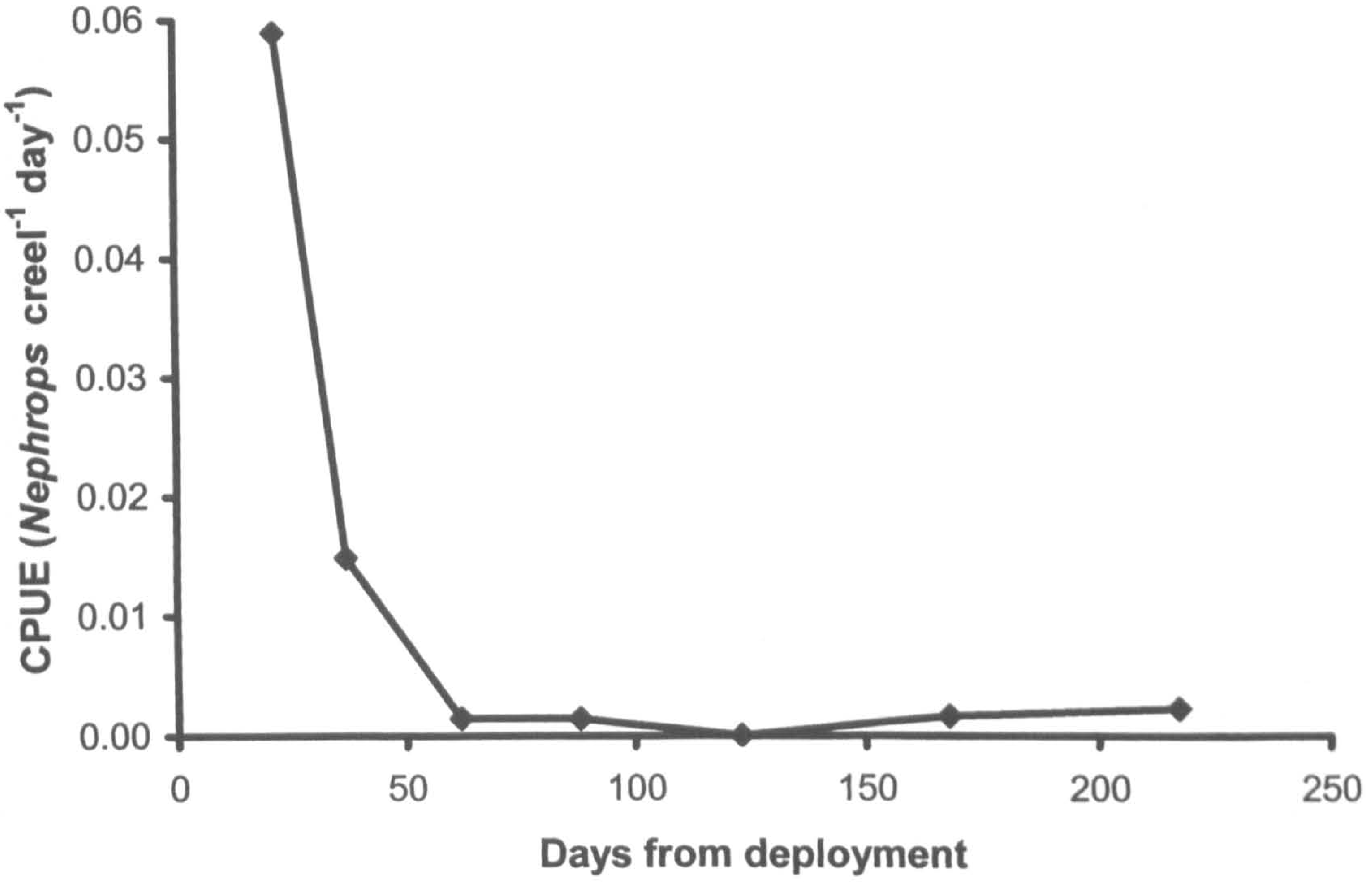

Figure 7.7 - Catch per unit effort of individual *Nephrops* in a fleet of ghost fishing creels deployed in Loch Torridon.

Chapter 8

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8 PHYSIOLOGICAL EFFECTS OF CAPTURE, STORAGE AND LIVE TRANSPORT OF *NEPHROPS*

8.1 Introduction

The capture and transport of crustaceans is a global business, leading to a number of species being subjected to long shipments. This exposes these animals to conditions outside those experienced in the wild, ultimately leading to them experiencing physiological stress. Stress responses occur when a physiological system is challenged by environmental conditions outside the range normally experienced, as one or more physical factors or 'stressors' act upon it (Barton & Iwama, 1991). These stressors can include: capture and landing, aerial exposure, physical damage, interactions with other animals, escape responses and low water quality in holding tanks (Taylor *et al.*, 1997). Stress can affect the long term survival of animals, and may lead to a decrease in product quality and value (Whyman *et al.*, 1985; Jussilla *et al.*, 1990; Beard & McGregor, 1991; Jacklin, 1996; Spanoghe & Bourne, 1997; Ruiz-Capillas, 2004; Ridgway *et al.*, 2006a).

Creel-caught *Nephrops* are exposed to a number of stressors, including physical damage due to capture and handling, aerial exposure during transport, as well as exposure to high concentrations of ammonia and low salinities when stored in tanks both at sea and on land (Section 1.3.2). *Nephrops* can be exposed to air for up to 12 h at sea depending on the method employed by the fishermen (many fishermen have seawater tanks to avoid aerial exposure at sea). As an example of handling and transport practices (at the facilities of Shieldaig Export, the company who have been collaborating with the work reported in this thesis), *Nephrops* are either stored at sea or at 3°C in cold-water tanks until they are re-packaged and transported to mainland Europe. They are packed in ice and transported in refrigerated containers for part of the journey, which may last for up to 12 h. Temperature loggers placed in a *Nephrops* crate showed they may experience temperatures ranging between 3°C and 4°C throughout the transport process, although this is likely to vary with season (Shieldaig Export, personal communication).

Previous studies have suggested that capture and handling stress in creel-caught *Nephrops* will be lower than observed in trawl fisheries, in which animals were observed to have high levels of physiological disruption and stress (dehydration, reduced blood volume, increased blood protein concentrations, respiratory and metabolic acidosis, high blood *L*-lactate and *D*-glucose concentrations) (Spicer *et al.*, 1990; Harris *et al.*, 1997; Wileman *et al.*, 1999; Ridgway *et al.*, 2006a,b). In the *Nephrops* creel fishery, the long periods of emersion during transport in particular and possible water quality issues during storage are thought to be the main causes of stress (Section 1.3.2).

The physiological effects of aerial exposure vary considerably among crustacean species (for example, McLeese, 1956; Taylor & Butler, 1978; DeFur & McMahon, 1984 a,b; Winkler, 1987; DeFur *et al.*, 1988; Spicer *et al.*, 1990; Whiteley & Taylor, 1992; Goodrick *et al.*, 1993; Ridgway *et al.*, 2006a). This variation is particularly evident between sublittoral and intertidal species, where sublittoral species such as *Nephrops* do not have the morphological and physiological adaptations of intertidal species that enable them to survive long periods of aerial exposure (Spicer *et al.*, 1990).

In sublittoral species, emersion generally leads to a reduction in oxygen uptake and subsequent internal hypoxia, as diffusion across the gills is seriously impaired by the collapse of the gill filaments in air. Animals switch to anaerobic pathways, leading to the elevation of the concentrations of *L*-lactate and *D*-glucose in the haemolymph and in the tissues (Vermeer, 1987; Taylor & Whiteley, 1989; Schmitt & Uglow, 1997a,b; Paterson & Spanoghe, 1997; Wileman *et al.*, 1999). Glucose is the main energy substrate for metabolic processes, increasing in concentration in response to stress (change from aerobic to anaerobic respiration) (Telford, 1973; Taylor *et al.*, 1997; Paterson & Spanoghe, 1997). The mobilisation of glucose from intracellular glycogen reserves is stimulated by crustacean hyperglycaemic hormone (CHH), which is in turn stimulated by low glucose levels (also modulated by *L*-lactate levels) (Kallen *et al.*, 1990; Santos & Keller, 1993). *L*-lactate is the major end product of anaerobic metabolism (converted from glucose via glycolysis) in many species, including decapod crustaceans (Vermeer, 1987; Hill, 1989; Taylor & Whiteley, 1989; Schmitt & Uglow, 1997a; Paterson & Spanoghe, 1997; Wileman *et al.*, 1999), and can accumulate in sufficient quantities to overwhelm protein and carbonic acid buffering in the haemolymph, resulting in a metabolic acidosis. This change in pH can have important effects on cells by disrupting enzymic reactions and affecting osmotic regulation and cell membrane stability (Vermeer, 1987).

Additionally, emersion has been shown to have an adverse effect on the immune system of decapod crustaceans (Le Moullac *et al.*, 1998; Lorenzon *et al.*, 2001; Fotedar *et al.*, 2001; Cheng *et al.*, 2002; Ridgway *et al.*, 2006a). It has been shown that when *Nephrops* were exposed to high temperatures in air, significant reductions in total haemocyte counts, prophenoloxidase levels and elevated haemolymph bacteraemia levels were observed, indicating that the animals were under significant immunological stress (Ridgway *et al.*, 2006a).

Temperature is known to be an important factor affecting the survival of crustaceans during transport. This may result partly from the increase in metabolic rate associated with an increase in temperature during aerial exposure, which may cause respiratory stress if the delivery of oxygen to the tissues is compromised (Taylor & Whiteley, 1989; de Fur *et al.*, 1988; Spicer *et al.*, 1990; Zainal *et al.*, 1992; Goodrick *et al.*, 1993; Taylor *et al.*, 1997; Paterson & Spanoghe, 1997; Spanoghe & Bourne, 1997; Ridgway, 2005; Ocampo *et al.*, 2005). Although the switch to anaerobic metabolism that may occur during exposure often results in a metabolic acidosis, there is some evidence that lobsters may be able to compensate for this acidosis provided they are undisturbed. However, the extent of this compensation varies with species and temperature. At 15°C, *Homarus gammarus* is able to prevent major pH shifts for 14 h, whereas *Jasus edwardsii* at 17°C showed similar compensation for just 4–8 h, and *Panulirus argus* for less than 1 h at 25°C (Taylor *et al.*, 1997).

To reduce the adverse effects of emersion, the temperature at which the animals are stored is often reduced prior to, and during transport, depressing activity and reducing metabolism and stress (Whiteley and Taylor, 1990; Goodrick *et al.*, 1993; Paterson, 1993; Spanoghe & Bourne, 1997; Morris & Oliver, 1999a,b). Conversely, rapid chilling may cause a temporary increase in metabolic rate and has also been observed in some cases to shift metabolism towards anaerobic pathways (Taylor *et al.*, 1997), leading to little overall change in the accumulation of acidic metabolites when compared with animals emersed at higher temperatures (Spicer *et al.*, 1990; Ocampo *et al.*, 2005).

Physiological stress in *Nephrops* associated with exposure to naturally occurring hypoxic conditions has been demonstrated in a number of studies (Baden *et al.*, 1990; Baden *et al.*, 1994; Baden *et al.*, 1995; Spicer & Baden 2001) and during emersion (Spicer *et al.*, 1990; Jacklin, 1996; Schmitt & Uglow, 1997a,b; Ridgway *et al*, 2006a). These studies have illustrated that under hypoxic conditions, *Nephrops* are able to compensate to a limited extent for the reduction in oxygen supply to the tissues. However, prolonged periods of emersion, particularly at high temperatures, produce a large stress response and in some cases may cause mortality, leading to industry advice that animals must be kept moist and cool following capture (Myers & Combes, 2004). The majority of these studies, however, have been over relatively short time periods and conducted at higher temperature ranges than those experienced by *Nephrops* transported to European markets.

Nephrops may also be exposed to low salinities during the capture and storage process. Low salinity exposure may initially occur if they are brought up through a reduced salinity layer on the surface that may have developed following high levels of freshwater runoff. In certain areas of *Nephrops* fishing, such as the Kattegat and Skagerrak, a halocline occurs throughout the year (Harris & Ulmestrand, 2004). During heavy rainfall, *Nephrops* may also be exposed to freshwater on deck as well as to low salinities in holding tanks, if water has been taken from near the sea surface. This has led to advice from the industry to take great care in storing *Nephrops* below the surface away from freshwater run-off (Myers & Combes, 2004).

The survival of lobster species in reduced salinities requires either extracellular osmoregulation (illustrated in *Homarus americanus* – Jury *et al.*, 1994), minimising the difference in haemolymph osmolarity, and/or intracellular isosmotic regulation (illustrated in *Panulirus longipes* – Dall, 1974), preventing movements of water across cell membranes by equilibrating intra- and extracellular osmolarities (Charmantier *et al.*, 2001). Most lobsters species are, however, not noted for their osmoregulatory ability (Mantel & Farmer, 1983), with exposure to low salinities often resulting in swelling of the abdomen, major behavioural disturbances (Dall, 1974; Shaner *et al.*, 1985), mass gains (Cornell, 1980; Harris & Ulmestrand, 2004), as well as a net water gain and solute dilution (Kirschner, 1991). *Nephrops* is a stenohaline species and is not found in low salinity environments (Mantel & Farmer, 1983; Schoffeniels & Dandrifosse, 1994); the gill cuticle is 'leaky' to a number of ions (Lignon & Gender, 1988). *Nephrops* have been shown to tolerate salinities of 28 (the convention of not quoting units is followed throughout this chapter, expressing salinity in accordance with the Practical Salinity Scale 1978), but exposure to lower salinities resulted in high mortalities (Harris & Ulmestrand, 2004).

Stress responses may be evaluated subjectively or expressed quantitatively by measured changes in physiological variables (Taylor *et al.*, 1997). Stress indicators include immunological changes, such as total haemocyte count, differential haemocyte count and phenoloxidase activity, or physiological responses such as changes in the rate of oxygen consumption, ventilation and heartbeat. The most commonly used methods are the assessment of metabolic changes (Baden *et al.*, 1990; Paterson & Spanoghe, 1997; Taylor *et al.*, 1997; Fotedar *et al.*, 2001; Ridgway *et al.*, 2006a), such as variations in *L*-lactate and *D*-glucose concentration and pH. These stress indicators have the advantage of being easy to quantify and samples can be frozen for later analysis (Taylor *et al.*, 1997). Others have suggested that the use of the more-sophisticated CHH radioimmunoassay may be a more appropriate indicator of stress (Lorenzon *et al.*, 1997).

Levels of these indicators enable the assessment of physiological stress by illustrating the degree to which an animal is relying on anaerobic metabolism under certain conditions. These indicators do not, however, show if an animal is in poor condition, just if it is temporarilly challenged (Taylor *et al.*, 1997). It is therefore important to monitor these physiological changes during simulated capture, storage and transport, to assess the point at which stress leads to mortality.

The aim of the present study was to assess the physiological stress experienced by creel-caught *Nephrops* during capture, transport and storage. The effects of emersion at a range of temperatures likely to be experienced during capture and transport have been studied in detail, as have the effects of low salinity. Understanding the physiological stresses endured by *Nephrops* during capture and transport will allow 'best practice' to be defined.

8.2 Methods

8.2.1 *Nephrops* collection

Nephrops were collected using an otter trawl (70-mm mesh size) from the Clyde Sea Area in May 2004. Animals were placed in 'tubes' (elongate, square cross section cells created by placing interlocking vertical partitions within storage boxes), kept damp using a wet cloth and transported to the University of Glasgow. They were kept in 40 l tanks with available shelter at a temperature of 10°C and salinity of 32 and allowed to recover for a period of 2 weeks before the experiments were carried out. During storage, animals were subjected to a 12 h: 12 h light:dark cycle and fed twice weekly on frozen mussels (*Mytilus edulis*). Only *Nephrops* in the intermoult stage and those with a carapace length of 28 mm or more were selected. For each group of animals tested, an equal number of each size category (small <40-mm CL, medium 40–46-mm CL, large >46-mm CL) and sex was used. *Nephrops* were not fed during the experiment.

8.2.2 Effects of aerial exposure and reduced salinity on mortality

Mortality resulting from aerial exposure and from exposure to reduced salinity was assessed in separate sets of experiments. No movement of the telson or pleopods defined mortality.

The effect of aerial exposure at temperatures of 2, 5, 10 and 15°C on the mortality of *Nephrops* was assessed in groups of 10 animals at intervals over a period of 24 h.

Mortality of groups of 10 *Nephrops* immersed in water at salinities of 15, 20, 25, 30 and 32 at 10°C was assessed every half hour up to 8 h and once more following 1 week of exposure.

8.2.3 Effects of aerial exposure on physiological parameters

An investigation into the physiological effects of aerial exposure was carried out by exposing 20 groups of 10 animals to a range of temperatures for up to 24 h. *Nephrops* were removed from tanks, placed in 'tubes' and stored in temperature-controlled rooms at 2, 5, 10 and 15°C. The humidity in the tubes measured using a hygrometer varied between 70% and 90% during the experiment. At each temperature, the concentrations of *L*-lactate and *D*-glucose in the haemolymph, as well as haemolymph pH and osmotic concentration, were assessed in 10 animals following emersion for 1, 3, 8, 16 and 24 h. Haemolymph samples from 20 'control' animals kept in aerated water at each temperature were also taken: an initial set of samples was taken from 10 animals after 1 h and from a further 10 animals after 24 h.

8.2.4 Effects of salinity on physiological parameters

The effects of exposure to low salinity were assessed using 16 groups of 10 *Nephrops* immersed in tanks of aerated seawater containing water of differing salinity (salinities of 15, 20, 25 and 30). Each tank was maintained at a temperature of 10°C. Haemolymph osmotic concentration and *L*-lactate concentration were determined in 10 animals following 1, 3, 8 and 16 h of immersion in each salinity. In addition, haemolymph osmotic concentration and *L*-lactate concentration were determined in 10 lobsters maintained in normal strength sea water (salinity = 32) after 1 h of immersion and in a further 10 animals after 24 h of immersion.

8.2.5 Collection and preparation of haemolymph samples

Haemolymph was rapidly removed from the sinus behind the 5th pereiopod using a 1 ml syringe and 25-gauge needle. A sample of haemolymph (100 µl) was mixed with an equal volume of ice-cooled 0.6 M perchloric acid (PCA) to deproteinize the sample. The samples were mixed thoroughly using a vortex mixer and centrifuged for 5 min at 13 000 *g* to precipitate the protein, following which the supernatant was removed and frozen at -20°C for later analysis.

Half of the remaining fresh haemolymph sample was immediately tested for pH (Section 8.2.6.3). The other half was immediately frozen prior to the determination of the osmotic concentration.

8.2.6 Determination of stress indicators

8.2.6.1 L-lactate

L-lactate was determined enzymatically using the method of Gutmann & Wahlefeld (1974) as modified by Hill (1989). Aliquots (50 µl) of the deproteinized haemolymph were added to 50 µl of nicotinamide adenine dinucleotide (NAD), 1000 µl of glycine-hydrazine buffer (pH 9.0) and 5 µl of *L*-lactate dehydrogenase (LDH). Ethylenediaminetetraacetic acid (EDTA) was added to the mixture to prevent copper ions associated with the haemocyanin from interfering with the assay (Engel & Jones, 1978). After thorough mixing, the samples were incubated for 2 h at 37°C and the absorbance of the sample measured at 340 nm on a UV-Vis spectrophotometer (Shimadzu, UV Mini 1240). Absorbance values were converted to *L*-lactate concentrations using a calibration curve constructed from standards of known *L*-lactate concentration (0, 0.25, 0.5, 1, 2, 4, 8, 12 and 16 mmol l^{-1}).

8.2.6.2 D-glucose

D-glucose was determined enzymatically using the method of Bergmeyer & Bernt (1974). The deproteinized haemolymph samples were neutralised by the drop-wise addition of potassium bicarbonate (2 M). Aliquots (75 µl) of each sample were diluted by the addition of the same volume of distilled water. Haemolymph *D*-glucose concentration was measured using a Sigma glucose assay kit (GAGO-20). Glucose oxidase/peroxidase reagent (300 µl) was added to the 150-µl samples (reagents were added at 30-s intervals). The samples were thoroughly mixed and incubated for exactly

30 min at 37°C. The reaction was stopped by adding 300 µl of 12 N sulphuric acid at corresponding 30-second intervals, ensuring that the samples were mixed. The absorbance of the sample was measured at 540 nm on a UV-Vis spectrophotometer (Shimadzu, UV Mini 1240). Absorbance values were converted to *D*-glucose concentrations using a calibration curve constructed from standards of known *D*-glucose concentration (0, 0.01, 0.03, 0.06, 0.08, 0.11, 0.22 and 0.44 mmol l^{-1}).

8.2.6.3 pH

The pH of haemolymph samples at the experimental temperature (10°C) was measured immediately after collection using a precalibrated microelectrode (Thermo Russell) and pH meter (Corning model 255 ion Analyser).

8.2.6.4 Osmotic concentration

Haemolymph osmotic concentration was determined by freezing point depression, using a freezing point osmometer (Osmomat 030). The osmometer was calibrated with 50 µl of deionized water and a calibration solution placed in a clean, dry measuring vessel, ensuring that no air bubbles were present. Sample measurements were made under the same conditions as for the calibration solution using 50 µl of frozen whole haemolymph samples. The thermistor probe was cleaned with deionized water between samples.

8.2.7 Field experiments

Nephrops were collected from Upper Loch Torridon using commercial creels. Animals were stored in 'tubes' and either used immediately, or stored in salt-water tanks prior to the experiment. Only *Nephrops* in the intermoult stage and with a carapace length of 28–40 mm were selected.

8.2.7.1 *Nephrops* storage

To investigate the effect of storage methods, the concentrations of *L*-lactate in the haemolymph of 40 freshly caught *Nephrops* were compared with those obtained from 20 *Nephrops* stored in 'tubes' in commercial tanks (3°C for 12 h) and 20 *Nephrops* stored in 'tubes' (Chapter 1; Section 1.3.2) in the sea (12°C for 12 h).

8.2.7.2 Exposure on deck

The effect of storage at sea was investigated by determining the concentrations of *L*-lactate and *D*-glucose in the haemolymph from 60 *Nephrops* placed in tubes on the deck of a commercial creel vessel. Haemolymph samples were taken from 20 animals immediately after capture and from further batches of 20 animals at 1, 2 and 4 h. Samples were collected during May 2004 and the average temperature and humidity in the tube throughout the experiment was measured using a combined temperature and humidity probe, with mean readings of 14.7°C and 82%, respectively.

8.2.8 Data analysis

The combined effects of duration of aerial exposure and ambient temperature on haemolymph *L*-lactate, *D*-glucose concentrations, haemolymph pH and osmolarity were tested with two-way ANOVA. *L*-lactate and *D*-glucose were $\log_{10}$-transformed before analysis. The same variables were analysed for control animals in seawater at 1 and 24 h. Owing to 100% mortality of *Nephrops* following 24 h of emersion at 15°C, it was necessary to carry out two separate tests, either excluding the data obtained at 24 h or those at 15°C.

The combined effects of duration of immersion and salinity on haemolymph *L*-lactate concentration and osmolarity ($\log_{10}$-transformed) were tested with two-way ANOVA.

Owing to 100% mortality of *Nephrops* following 3 h of immersion at a salinity of 15 and at 8 h following immersion at salinities of 15 and 20, it was necessary to carry out two separate tests, excluding the data for 8 and 16 h, or for salinities of 15 and 20.

One-way ANOVA was used to compare $\log_{10}$-transformed haemolymph *L*-lactate and *D*-glucose concentrations following 0, 1, 2 and 4 h of aerial exposure on a creel boat.

One-way ANOVA was used to compare $\log_{10}$-transformed haemolymph *L*-lactate concentrations of freshly caught *Nephrops*, of *Nephrops* stored at sea and of *Nephrops* stored in chilled tanks (see above).

Pairwise comparisons were made with Tukey's test, where appropriate. All univariate analyses were undertaken using MINITAB version 13 (Minitab Inc., State College, Pennsylvania).

8.3 Results

8.3.1 Mortality

No mortalities were recorded during the 24-h period among *Nephrops* maintained at 2 and 5°C (Figure 8.1). At 10 and 15°C mortality increased with the duration of aerial exposure, with the increase being more pronounced at 15°C. At 10°C, mortalities were observed following 16 h of exposure and at 15°C following 8 h exposure. After 24 h exposure in air, 100% of *Nephrops* at 15°C and 50% of *Nephrops* at 10°C had died (Figure 8.1).

At salinities of 25 and 30, no mortalities were observed during the experiment (Figure 8.2). When exposed to salinities of 15 and 20, mortality increased with duration of aerial exposure, with the increase being similar between salinities. Mortalities were observed after 2 h of exposure to a salinity of 15 and after 3 h for *Nephrops* maintained at a salinity of 20. Exposure to salinities of 15 and 20 for 8 h resulted in 100% mortalities (Figure 8.2).

8.3.2 Aerial exposure

During this experiment, a number of mortalities were observed at 10 and 15°C, with 60% and 30% mortality being reached after 16 h at 10 and 15°C, respectively and 100% mortality after 24 h at 15°C. There were no mortalities at either 2 or 5°C.

8.3.2.1 *L*-lactate

At test temperatures of 5, 10 and 15°C, haemolymph *L*-lactate concentration increased with the duration of aerial exposure; the increase was greater at higher temperatures (Figure 8.3). There was no significant increase in haemolymph *L*-lactate concentration with time in *Nephrops* exposed to air at 2°C (Tukey Simultaneous Tests, $P \geq 0.50$). The mean *L*-lactate concentration was higher during aerial exposure than the mean resting value of 0.72 ± 0.17 mmol l^{-1} (mean ± 95% Confidence Limits). There was a significant interaction between the effects of temperature and exposure time (2-way ANOVA interaction, excluding 24 h: $F_{9,128}=4.20$, $P<0.001$; excluding 15°C: $F_{8,125}=5.19$, $P<0.001$), indicating that the time course of changes in *L*-lactate concentration was temperature-dependent.

Following immersion in seawater (control experiment to assess the effect of temperature alone) at test temperatures of 2, 5, 10 and 15°C, no increase in haemolymph *L*-lactate concentrations was observed with time (2-way ANOVA time, $F_{1,69}$=0.86, P=0.358) (Figure 8.4), with no interaction between the effects of temperature and immersion time (2-way ANOVA interaction, $F_{3,69}$=1.26, P=0.294).

8.3.2.2 *D*-glucose

Haemolymph *D*-glucose concentrations increased with the duration of aerial exposure at test temperatures of 2, 5 and 10°C. This increase was greatest during the first 3 h, with no significant difference in *D*-glucose concentrations among temperatures of 2, 5 and 10°C (Tukey Simultaneous Tests, $P \geq 0.214$) (Figure 8.5). At 15°C, haemolymph *D*-glucose concentrations remained high for the duration of the experiment, until all test animals had died after 24 h (Tukey Simultaneous Tests, $P \geq 0.965$). The mean *D*-glucose concentration was higher than the mean resting value of 0.45 ± 0.11 mmol l^{-1} (mean ± 95% Confidence Limits) at 5, 10 and 15°C throughout the experiment and following 3 h exposure at 2°C. There was a significant interaction between the effects of temperature and exposure time (2-way ANOVA interaction, excluding 24 h: $F_{9,132}$=3.79, P<0.001; excluding 15°C: $F_{8,127}$=2.68, P<0.01), indicating that the time course of changes in *D*-glucose concentration differed according to temperature.

The results of a control experiment in which *Nephrops* were immersed in seawater at different temperatures (2, 5, 10 and 15°C) showed that higher haemolymph *D*-glucose concentrations occurred in animals maintained at 15°C (Figure 8.6) (2-way ANOVA temperature, $F_{3,68}$=8.82, P<0.001). There was no interaction between the effects of temperature and immersion time (2-way ANOVA interaction, $F_{3,69}$=1.26, P=0.294).

8.3.2.3 pH

At test temperatures of 2, 5 and 10°C, there was no relationship between haemolymph pH and the duration of aerial exposure (Tukey Simultaneous Tests, $P \geq 0.0749$), although differences between temperatures were apparent (Figure 8.7). In all cases, the pH was lower during aerial exposure than the mean resting value of 7.96 ± 0.03 (mean ± 95% Confidence Limits) for control animals in water at 10°C. At 15°C, haemolymph pH decreased during exposure (up to 16 h, after which all animals had died) (Figure 8.7). There was a significant interaction between the effects of temperature and exposure time excluding 24 h (2-way ANOVA, $F_{9,131}=8.54$, $P<0.001$), but no significant interaction excluding 15°C (2-way ANOVA, $F_{8,127}=1.97$, $P=0.55$), indicating that the time course of changes in pH was different only at 15°C.

The results of a control experiment in which *Nephrops* were immersed in seawater at different temperatures (2, 5, 10 and 15°C) showed that at 10 and 15°C, haemolymph pH values were significantly lower than those at 2 and 5°C after immersion for 1 h (Tukey Simultaneous Tests, $P<0.001$). (Figure 8.8). There was a significant interaction between the effects of temperature and immersion time (2-way ANOVA interaction, $F_{3,71}=5.65$, $P<0.005$).

8.3.2.4 Osmotic concentration

At test temperatures of 2, 5, 10 and 15°C, there appeared to be no relationship between haemolymph osmotic concentrations and duration of exposure to air (Figure 8.9). There was no significant interaction between the effects of temperature and exposure time (2-way ANOVA interaction, excluding 24 h: $F_{9,130}=0.72$, $P=0.686$; excluding 15°C: $F_{8,125}=0.42$, $P=0.904$). A significant difference was recorded, however, with time (2-way ANOVA time, excluding 24 h: $F_{3,130}=3.62$, $P<0.05$; excluding 15°C: $F_{4,125}=2.83$,

$P<0.05$). Haemolymph osmotic concentration, although variable, differed little from the mean resting value of 1090 ± 0.07 mOsm kg^{-1} (mean ± 95% Confidence Limits), indicating that the time course of changes in osmotic concentration was not temperature dependent.

The results of a control experiment in which *Nephrops* were immersed in seawater at different temperatures (2, 5, 10 and 15°C) showed no increase in haemolymph osmotic concentrations with time at temperatures of 2, 5 10 and 15°C (Figure 8.10), with no interaction between the effects of temperature and immersion time (2-way ANOVA interaction, $F_{3,69}=1.62$, $P=0.193$).

8.3.3 Salinity

During this experiment, a number of mortalities occurred at salinities of 15 and 20, with 100% mortality after 8 h at a salinity of 15 and after 16 h at a salinity of 20. There were no mortalities at salinities of 25 and 30 for the duration of the experiment (up to 16 h).

8.3.3.1 Osmotic concentration

At salinities of 15, 20 and 25 there was no significant difference (Tukey Simultaneous Tests, $P\geq0.6178$) in haemolymph osmotic concentrations during the first 3 h of immersion. After 8 h, the haemolymph osmotic concentrations of the remaining *Nephrops* immersed at a salinity of 20 and *Nephrops* immersed at a salinity of 25 had become significantly reduced (Tukey Simultaneous Tests, $P<0.05$) (Figure 8.11). At a salinity of 30, haemolymph osmotic concentration varied little throughout the 16-h immersion period (Tukey Simultaneous Tests, $P\geq0.5906$). Haemolymph osmotic concentration was, for all salinities and times throughout the experiment, lower during immersion than the mean resting value of 1070 ± 0.05 mOsm kg^{-1} (mean ± 95%

Confidence Limits) (control from undisturbed animals taken from storage tanks at a salinity of 32). There was a significant interaction between the effects of salinity and immersion time excluding salinities of 15 and 20 (2-way ANOVA, $F_{3,68}$=3.43, P<0.05), but no significant interaction excluding 8 and 16 h (including salinities of 15 and 20) (2-way ANOVA, $F_{3,69}$=0.33, P=0.801), indicating that the time course of changes in haemolymph osmotic concentration was different following 3 h of immersion.

Haemolymph osmotic concentration was higher than that of the medium throughout the experiment. During the first 3 h only, a small reduction in haemolymph osmotic concentration was observed at all salinities. After 3 h, the haemolymph osmotic concentration in surviving animals remained higher than that of the medium, but was greatly reduced when compared with initial values (Figure 8.12).

8.3.3.2 L-lactate

All haemolymph *L*-lactate concentrations were low when compared to those when *Nephrops* were emersed out of the water. At salinities of 15 and 20, haemolymph *L*-lactate concentrations increased with the duration of immersion, with the increase being greater at higher temperatures (up to 3 h, beyond which there was low survival at low salinities) (Figure 8.13). There was no change in haemolymph *L*-lactate concentration with time at salinities of 25 and 30. *L*-lactate concentration was higher during immersion at all salinities than the mean resting value of 0.72 ± 0.17 mmol l^{-1} (mean ± 95% Confidence Limits) (control from undisturbed animals taken from storage tanks), with this difference being greater at low salinities. There was no significant interaction between the effects of salinity and immersion time, excluding salinities of 15 and 20 (2-way ANOVA, $F_{3,64}$=1.76, P=0.165), but a significant interaction excluding 8 and 16 h (including salinities of 15 and 20) was observed (2-way ANOVA, $F_{3,69}$=4.63,

P=0.005), indicating that the time course of changes in haemolymph osmotic concentration was different only at salinities of 15 and 20.

8.3.4 Field experiments

8.3.4.1 *Nephrops* storage

There was a significant difference in the *L*-lactate concentration of the haemolymph of freshly caught *Nephrops*, those stored in chilled water (3°C) and those stored at sea ($F_{2,76}$=29.95, P<0.001). The mean haemolymph *L*-lactate concentration of *Nephrops* stored in the sea was significantly higher than in animals stored in chilled water, which in turn was significantly higher than in freshly caught specimens (Tukey, P<0.05; Figure 8.14).

8.3.4.2 Exposure on deck

There was 40% mortality of *Nephrops* following 4 h aerial exposure on the deck of a commercial fishing vessel. During exposure, significant increases in the mean concentration of haemolymph *L*-lactate and *D*-glucose were observed ($F_{3,67}$=55.69, P<0.001; $F_{3,68}$=19.12, P<0.001, respectively) (Figures 8.15 & 8.16). Pairwise comparisons showed that the concentrations of haemolymph *L*-lactate were significantly higher after exposure for 1, 2 and 4 h when compared with concentrations determined just after capture. *D*-glucose concentrations were also significantly higher after exposure for 1, 2 and 4 h when compared with those from animals just after capture, but there were no significant differences in the haemolymph glucose concentrations of animals exposed for 1, 2 and 4 h. Thus concentrations of *L*-lactate continued to increase during aerial exposure, whereas glucose increased over the first hour of exposure only (Figures 8.15 & 8.16).

8.4 Discussion

There have been some preliminary studies into the physiological effects of fishing and transport on *Nephrops*. The majority of these studies have focused, however, on conditions associated with trawling. The limited information that exists for creel-caught animals has been obtained primarily as control data in studies investigating the effects of trawling (Harris *et al.*, 1997; Wileman *et al.*, 1999; Harris & Andrews, 2005a,b)

8.4.1 Aerial exposure

During experiments carried out to examine the physiological responses of *Nephrops* to aerial exposure, no mortaility was observed at the low temperatures (2 and 5°C), but at higher temperatures (10 and 15°C) there was a pronounced increase in mortality with duration of aerial exposure. In addition, haemolymph *L*-lactate concentrations increased at temperatures of 5, 10 and 15°C in the laboratory and at 15°C under commercial conditions. Previous studies have shown that *L*-lactate is an indicator of stress associated with exposure to air. This may be because *L*-lactate is the major end product of anaerobic metabolism that may occur during periods of stress (Spicer *et al.*, 1990). Analogous increases in the concentration of *L*-lactate in the haemolymph of *Nephrops* were observed in two previous studies that investigated the effect of long-term aerial exposure (Spicer *et al.*, 1990; Ridgway *et al.*, 2006a). The data obtained during the current study confirmed the observations made by Ridgway *et al.* (2006a) that the greatest accumulation of *L*-lactate occurred at the highest temperatures, with this accumulation (and subsequent death of the animals) being greater at a temperature of 25°C. In contrast, results from the current study showed that at 2°C no build-up of haemolymph *L*-lactate occurred. This result suggests that, at this temperature, metabolism was reduced to a level at which it could be largely supported by aerobic metabolism fuelled by oxygen obtained from air.

Although no significant change in haemolymph *L*-lactate concentration was recorded at 2°C, *L*-lactate levels were lower at this temperature than at other test temperatures following prolonged exposure. This finding is in contrast to a study by Spicer *et al.* (1990), in which no difference in the concentration of haemolymph *L*-lactate was observed between *Nephrops* kept on ice and animals stored in sea water at 10°C. The difference between these studies may be a reflection of the way in which the animals were stored. It is possible that animals stored on ice were adversely affected by exposure to freshwater from melted ice and by contact of the cuticle with ice-cold surfaces (Taylor *et al.*, 1997).

The time course of changes in haemolymph pH following aerial exposure in this study was similar to that observed by Ridgway *et al.* (2006a). Two distinct phases were evident: an initial decrease in pH during the first hour of exposure at all temperatures (thought to reflect an immediate change in the physio-chemical equilibrium) followed by a later decrease in pH occurring only at the highest temperatures (Whitely *et al.*, 1995). The more pronounced reduction in pH that occurs during prolonged exposure is probably due to the accumulation of *L*-lactate as a result of anaerobic metabolism. At high temperatures this accumulation occurs at a rate that overwhelms the protein and bicarbonate buffering systems, leading to a reduction in pH, a process described as metabolic acidosis (Taylor & Whiteley, 1989; Bergmann *et al.*, 2001; Ridgway *et al.*, 2006a). The resulting acidosis may disrupt enzymatic reactions, osmotic regulation and cell membrane stability, potentially causing permanent physiological change (Vermeer, 1987).

The pH of the haemolymph did not appear to be closely correlated with the concentrations of *L*-lactate in the haemolymph, suggesting that *Nephrops* may be able to compensate for the build-up of *L*-lactate. Such compensation has been observed in *Nephrops* and in other species and is thought to involve the dissociation of bicarbonate from the calcium carbonate in the carapace as well as the alkalization of branchial water (DeFur & McMahon, 1984; Burnett, 1988). It is likely that at lower temperatures *Nephrops* is able to compensate for the slower build up of *L*-lactate but, when accumulation is rapid at high temperatures, this process is overwhelmed, leading to subsequent acidosis, permanent physiological damage and death. This may provide further evidence that it is the acidosis and not the accumulation of organic acids that causes mortality in *Nephrops* (Ridgway, 2005).

Glucose has been identified as an indicator of stress in crustaceans (Paterson & Spanoghe, 1997), although its concentration in the haemolymph can vary as a result of a number of factors including nutritional state, moult stage and time of day (Kallen *et al.*, 1990; Chang, 1995; Bergmann *et al.*, 2001). The present laboratory and field experiments both found an initial increase in haemolymph *D*-glucose concentration following aerial exposure, as observed in previous studies (Spicer *et al.*, 1990; Harris *et al.*, 1997; Harris & Andrews, 2005a). An increase in glucose concentrations or hyperglycaemia is thought to be caused by the mobilisation of carbohydrate stores during glycolysis (Harris *et al.*, 1997). Increased rates of glycolysis can occur during anaerobic metabolism, as glucose is mobilised from intracellular glycogen reserves in order to fuel anaerobic metabolism. This process has previously been identified during aerial exposure (Hagerman *et al.*, 1990; Morris & Oliver, 1999a,b; Harris & Andrews, 2005a). Ridgway *et al.* (2006a) reported a decrease in haemolymph *D*-glucose concentration in *Nephrops* following emersion at 25°C, suggesting that at temperatures higher than those investigated in the current study, the resulting increase in metabolism may rapidly deplete carbohydrate stores and therefore have a different effect on haemolymph *D*-glucose concentration. In the current study, there was evidence for a slower increase in the concentration of *D*-glucose in the haemolymph at 2°C. This may be due to a reduction in metabolic rate, leading to a slower use of carbohydrate stores. A similar effect was observed by Spicer *et al.* (1990) in *Nephrops* kept on ice.

There was little evidence of an increase in haemolymph osmolarity with the duration of emersion. Previous work has provided some evidence of an increase in osmotic concentration of the haemolymph following aerial exposure for 90 min on the deck of a research vessel. This increase was thought to be a result of dehydration (Harris & Ulmestrand, 2004). It is possible that the high humidity levels recorded in the 'tubes' in which *Nephrops* were stored during the present study resulted in low levels of water loss and therefore little change in haemolymph osmolarity.

In the control animals immersed in seawater at each test temperature, there was little evidence of physiological stress. It is therefore clear that it is the combination of temperature and aerial exposure that has the greatest effect on the survival of *Nephrops*.

During aerial exposure, the degree of stress experienced by animals increases with temperature, with particularly high levels of both physiological stress and rapid mortality occurring at 15°C. This may be due to *Nephrops* being unable to compensate for the rapid build-up of haemolymph metabolites and subsequent acidosis during anaerobic metabolism. Below 10°C, stress appears to be greatly reduced with no mortalities observed during the 24 h period. This was particularly evident at 2°C, where no increase in *L*-lactate was observed. It is evident that in order to achieve high survival during the live transport of *Nephrops* to foreign markets, low temperatures must be utilised.

8.4.2 Salinity

Nephrops haemolymph was found to be isosmotic with seawater (1058 mOsm kg^{-1}), which confirmed the results from several other studies (Robertson, 1961; Harris *et al.*, 1997; Harris & Ulmestrand, 2004; Harris & Andrews, 2005b). When exposed to reduced salinities, the osmolarity of *Nephrops* haemolymph decreased with time, but always remained slightly hyperosmotic, suggesting that *Nephrops* may have only limited powers of osmoregulation. This limited intracellular isosmotic regulation has also been observed in the lobster *Homarus americanus* (Dall, 1970; Jury *et al.*, 1994). It is also likely that this decrease with time is indicative of the rate of loss of inorganic ions from the haemolymph.

In the present study, *Nephrops* survived in seawater with a salinity as low as 25. A similar salinity tolerance was observed by Harris & Ulmestrand (2004), suggesting that this is their tolerance limit. They also observed that *Nephrops* haemolymph was isosmotic with seawater at reduced salinites, which was not the case in the current study. The reason for this difference requires further investigation.

Low concentrations of haemolymph *L*-lactate were observed in *Nephrops* immersed in water with a salinity of 25 and 30, suggesting that exposure at these salinities caused little stress and if some form of osmoregulation occurred, the energy demands of this process were largely met by aerobic respiration. At salinities of 15 and 20, in addition to rapid mortality after 3 h of immersion, a rapid accumulation of *L*-lactate in the haemolymph was observed. This would appear to indicate that there was a pronounced increase in metabolic rate, probably due to the energetic demands of the animals attempting to regulate the osmotic concentration of the haemolymph. A similar result was reported in *H. americanus,* in which limited osmoregulation caused an almost twofold increase in metabolic rate when lobsters were transferred into progressively lower salinities (Jury *et al.*, 1994). It is likely that the limited osmoregulatory ability of *Nephrops*, coupled with the energy demands of osmoregulation, are responsible for the poor survival of animals exposed to salinities below 25.

Harris & Ulmestrand (2004) showed that *Nephrops* suffered similar haemolymph solute loss whether exposed to a prolonged period of emersion in low salinity water, or a short period of emersion in low salinity water followed by a prolonged period of aerial exposure. It was suggested that low salinity water trapped extracorporeally would continue to exchange with the haemolymph, resulting in a net water gain and solute dilution. If animals are exposed to low salinities in holding tanks prior to transport, or brought up through a surface layer of low salinity, this potentially brief exposure to low salinity will affect the animal throughout the transport process, compounding the effects of aerial exposure.

8.4.3 *Nephrops* handling and storage

The low concentrations of *L*-lactate in the haemolymph following capture suggest low levels of stress and locomotor activity during the capture process, a result also observed in *Nephrops* by Harris *et al.* (1997) and Harris & Andrews (2005a). Haemolymph *L*-lactate concentrations (0.73 ±0.03 mmol l^{-1}) were, however, higher than values reported for undisturbed *Nephrops* by Schmitt & Uglow (1997a) (0.16 ± 0.05 mmol l^{-1}), and previous findings for creel-caught *Nephrops* by Spicer *et al.* (1990) and Harris & Andrews (2005a) (0.57 ± 0.21 mmol l^{-1} and 0.27 ± 0.127 mmol l^{-1}, respectively). This may reflect the deep-water environment from which these animals were sampled, increasing the duration of hauling and therefore stress. Haemolymph *L*-lactate concentrations of creel-caught *Nephrops* were much lower than has been observed in most studies of *Nephrops* caught by trawling (Field, 1992; Wileman *et al.*, 1999; Harris & Andrews, 2005a; Ridgway *et al.*, 2006b). This indicates that stress associated with capture in the creel fishery is lower than that of the trawl fishery.

Low haemolymph *L*-lactate concentrations were observed following storage, a result also observed by Spicer *et al.* (1990). Although lower *L*-lactate concentrations were observed in *Nephrops* stored in chilled tanks compared to those stored in keep-boxes in the sea, the actual difference in concentration was small. It is likely that chilling *Nephrops* will be beneficial during the handling and transport process, by reducing stress through depressing activity and lowering metabolism (Whiteley and Taylor, 1990; Goodrick *et al.*, 1993; Paterson, 1993; Spanoghe & Bourne, 1997; Morris & Oliver, 1999a,b).

To summarise, the present study has shown that there is little stress associated with the capture of creel-caught *Nephrops*. Furthermore, as long as animals are not exposed to reduced salinity and oxygenation is maintained, there is also very little stress associated with the storage of *Nephrops*. The greatest stressors appear to be aerial exposure and exposure to low salinities, with temperature and salinity being very important in determining stress levels and ultimately survival. It appears that stress is greatly reduced below 5 °C, with this reduction in stress being particularly evident at 2 °C. A similar clear threshold was observed with salinity, where high levels of stress and rapid mortality occurred at salinities of 20 and 15. These findings indicate that following capture, storage and transport methods will have a large effect on both the survival and quality of the product arriving at European markets. It is clear that exposure to salinities below 25 and aerial exposure at temperatures above 10°C both lead to rapid mortality. This demonstrates the need to store animals in undiluted seawater on board fishing vessels and that the chilling of *Nephrops* prior to and during transport are effective ways of enhancing survival.

8.5 Summary

- The physiological effects of capture, storage and transport of *Nephrops* were investigated in the field and under controlled conditions in the laboratory.

- Mortality and haemolymph *L*-lactate, *D*-glucose, pH and osmolarity were used as indicators of stress.

- There was little apparent stress associated with the capture of *Nephrops* by creeling (as indicated by the low haemolymph *L*-lactate and *D*-glucose concentrations following capture).

- Low stress (low haemolymph *L*-lactate concentrations) was found with storage in both in chilled seawater tanks (3°C) on land and in the sea.

- Aerial exposure causes high levels of stress, with large increases in haemolymph metabolite concentrations (particularly *L*-lactate) and rapid mortality at temperatures of 10 and 15°C.

- At low temperatures (2 and 5°C) stress during aerial exposure appears greatly reduced (particularly at 2°C), with no mortalities of *Nephrops* observed following 24 h exposure at these temperatures.

- Immersion in low salinity water also resulted in high levels of stress (rapid increases in haemolymph *L*-lactate concentration) and mortality at salinities of 15 and 20.

- When immersed at higher salinities (25 and 30) there was no evidence of stress or mortality in any of the animals studied.

- *L*-lactate was the most effective measure of sub-lethal stress.

- Following capture, the storage and transport methods employed will have a large effect on both the survival and quality of the product arriving at European markets.

- During storage, *Nephrops* must not be exposed to salinities below 25 for even a short period of time and when transported out of water temperatures above 5°C must be avoided.

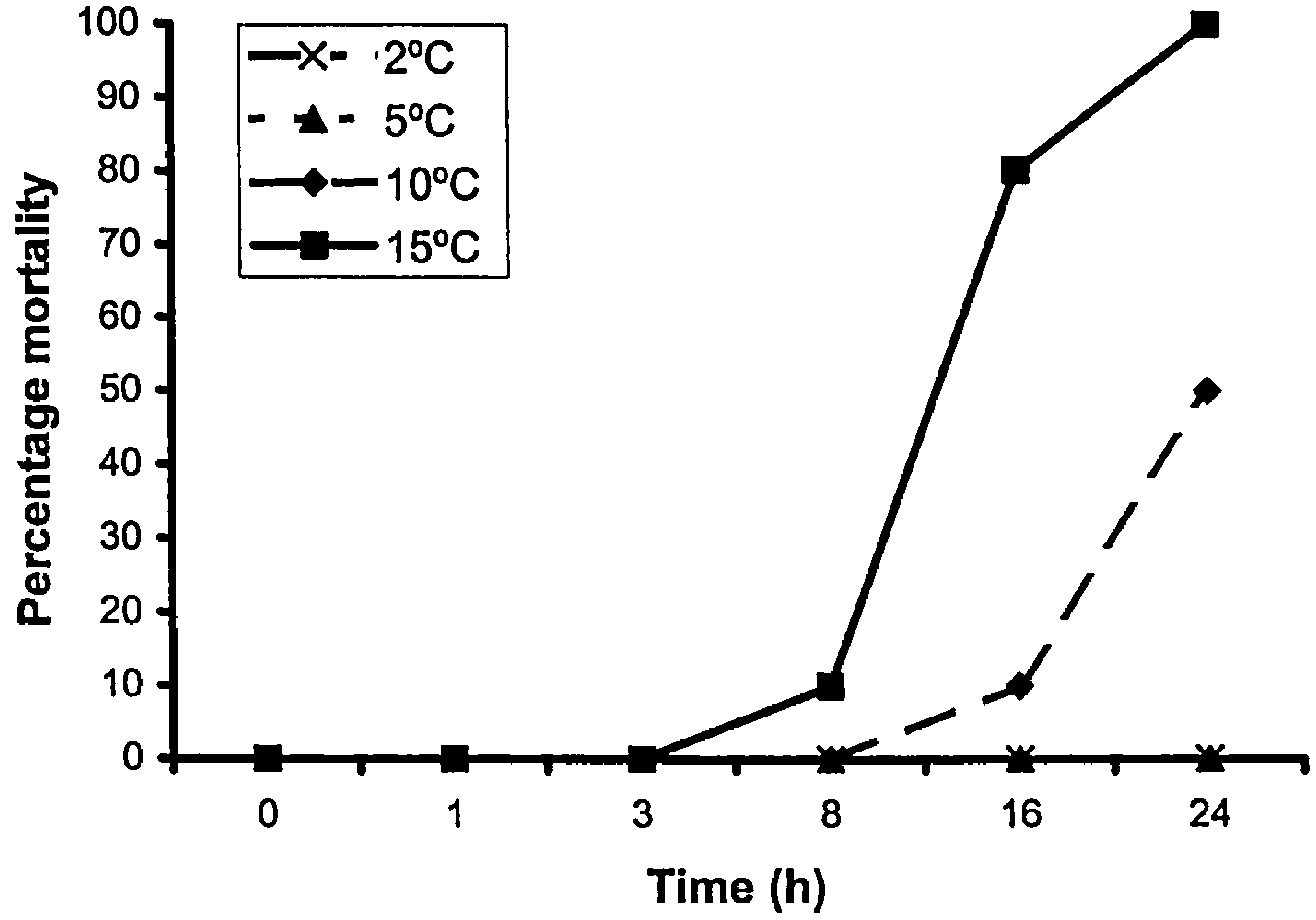

Figure 8.1 - The effects of temperature and duration of aerial exposure on the percentage *Nephrops* mortality.

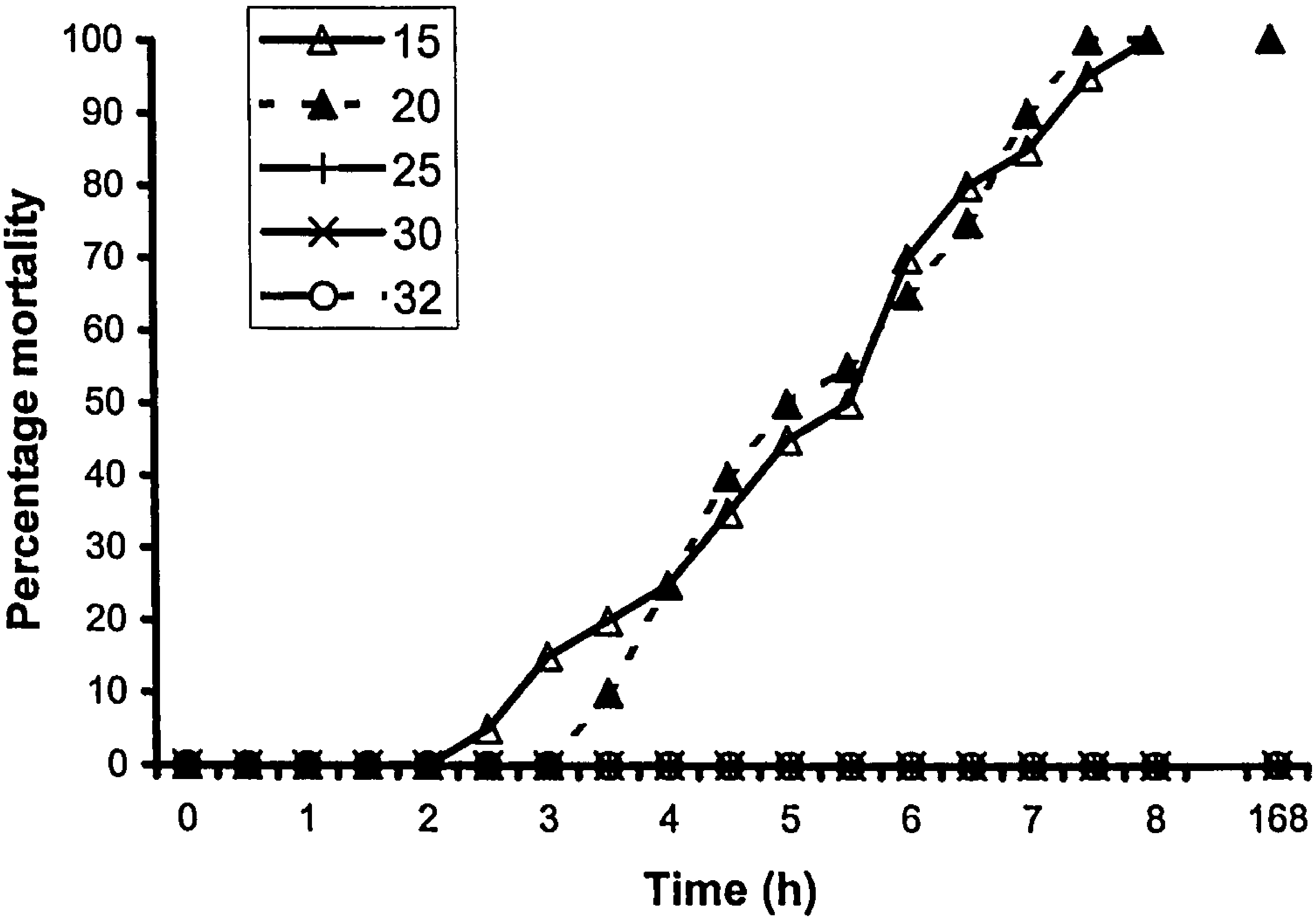

Figure 8.2 - The effects of salinity and duration of immersion on the percentage *Nephrops* mortality.

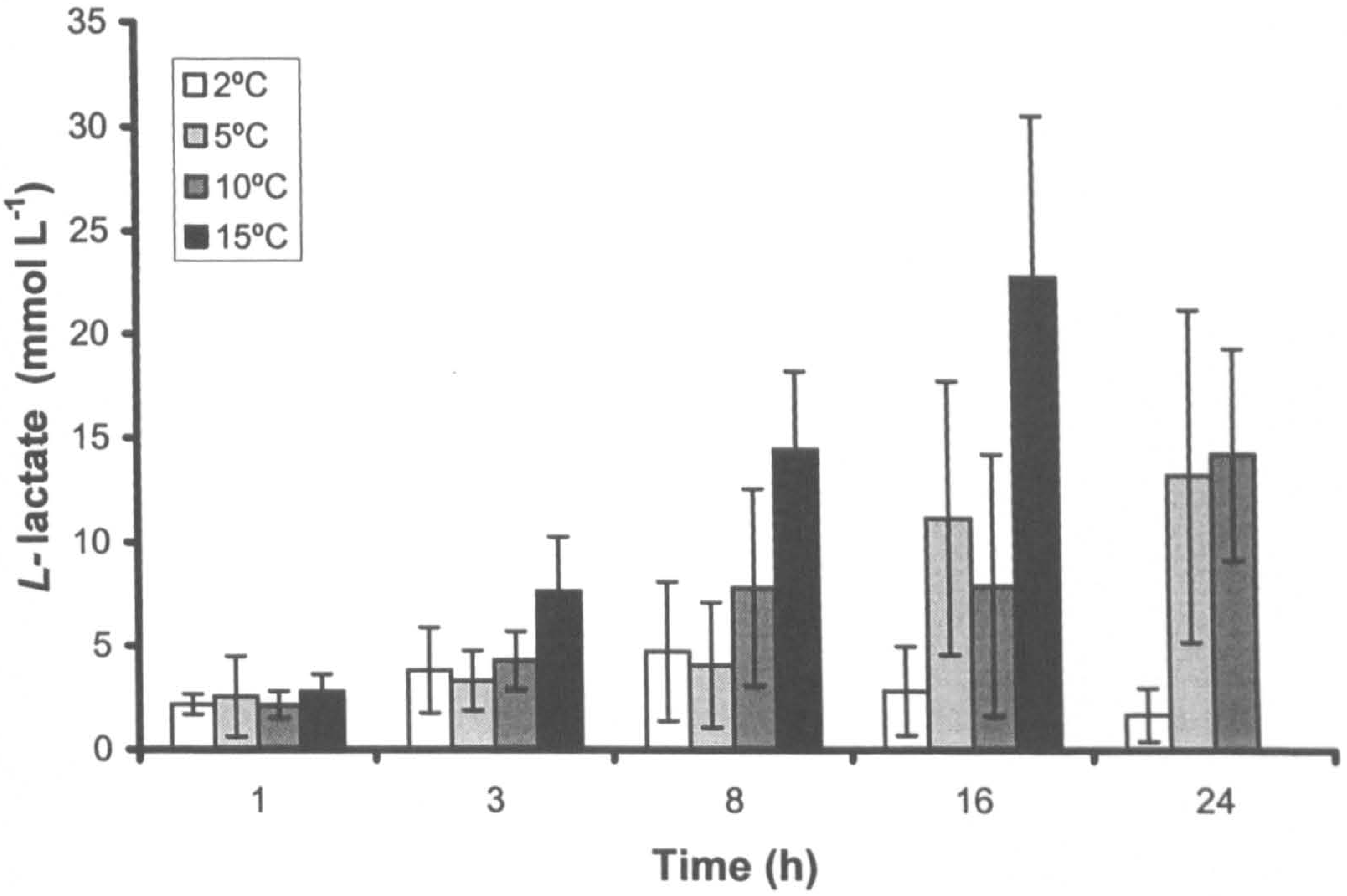

Figure 8.3 - Changes in *Nephrops* haemolymph *L*-lactate concentration with duration of air exposure at different temperatures, 100% mortality at 15°C following 24h exposure (mean resting value, 0.72 ± 0.17 mmol l^{-1}). Values are means ± 95% Confidence Limits.

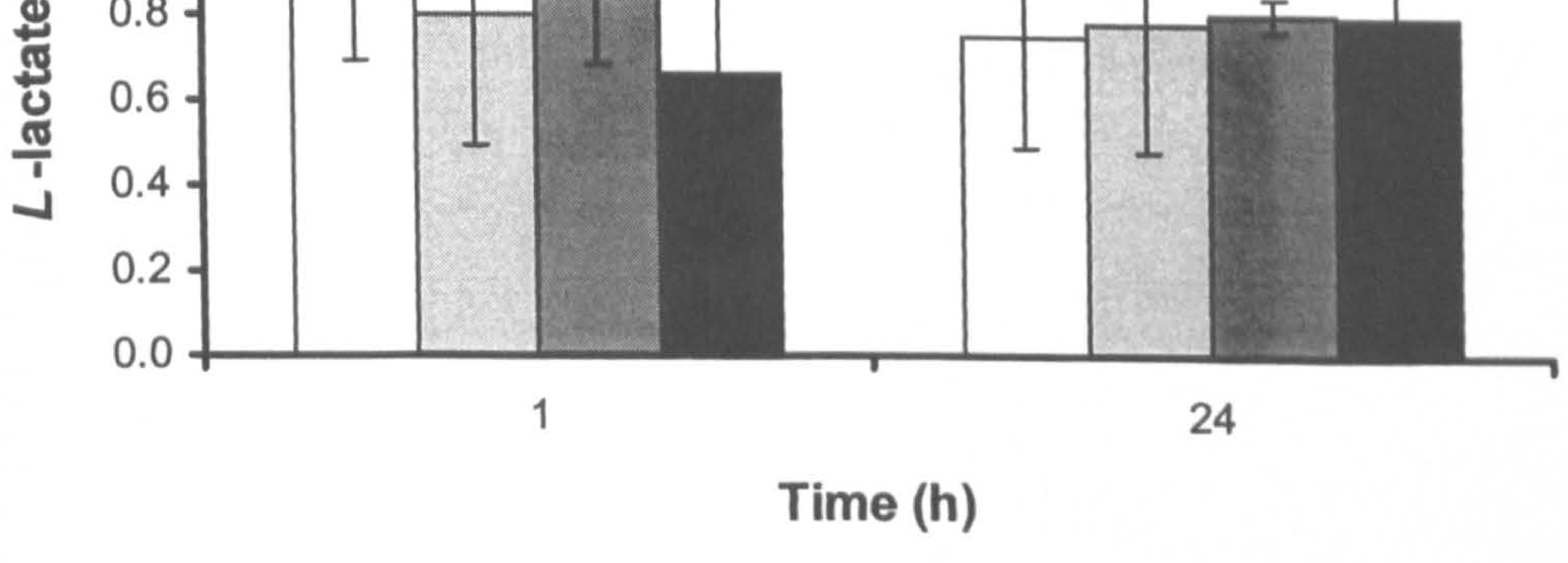

Figure 8.4 - Changes in *Nephrops* haemolymph *L*-lactate concentration with duration of immersion in sea water at different temperatures (control) (mean resting value, 0.72 ± 0.17 mmol l^{-1}). Values are means ± 95% CI.

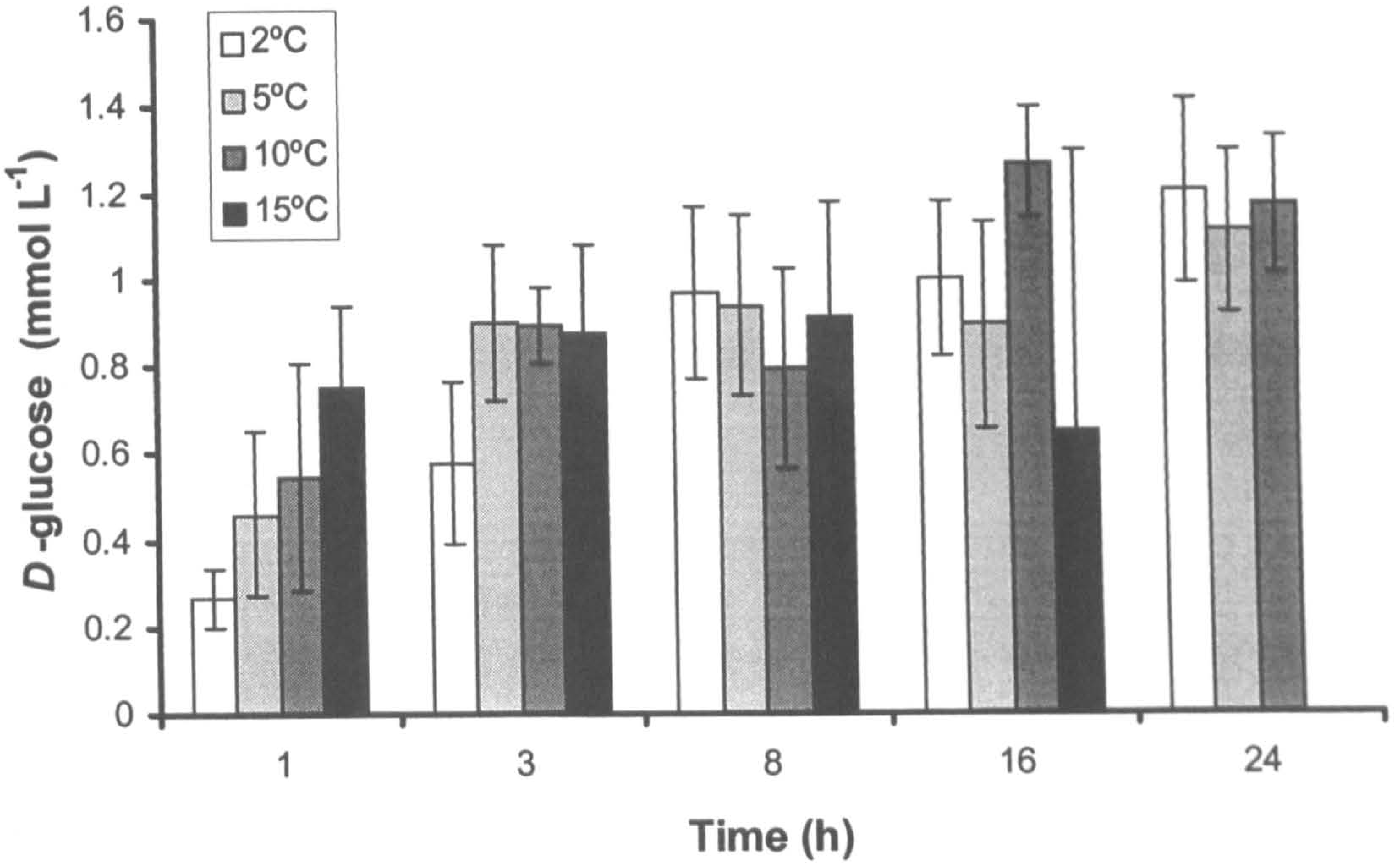

Figure 8.5 - Changes *Nephrops* haemolymph *D*-glucose concentration with duration of air exposure at different temperatures, 100% mortality at 15°C following 24h exposure (mean resting value, 0.45 ± 0.11 mmol l^{-1}). Values are means ± 95% Confidence Limits.

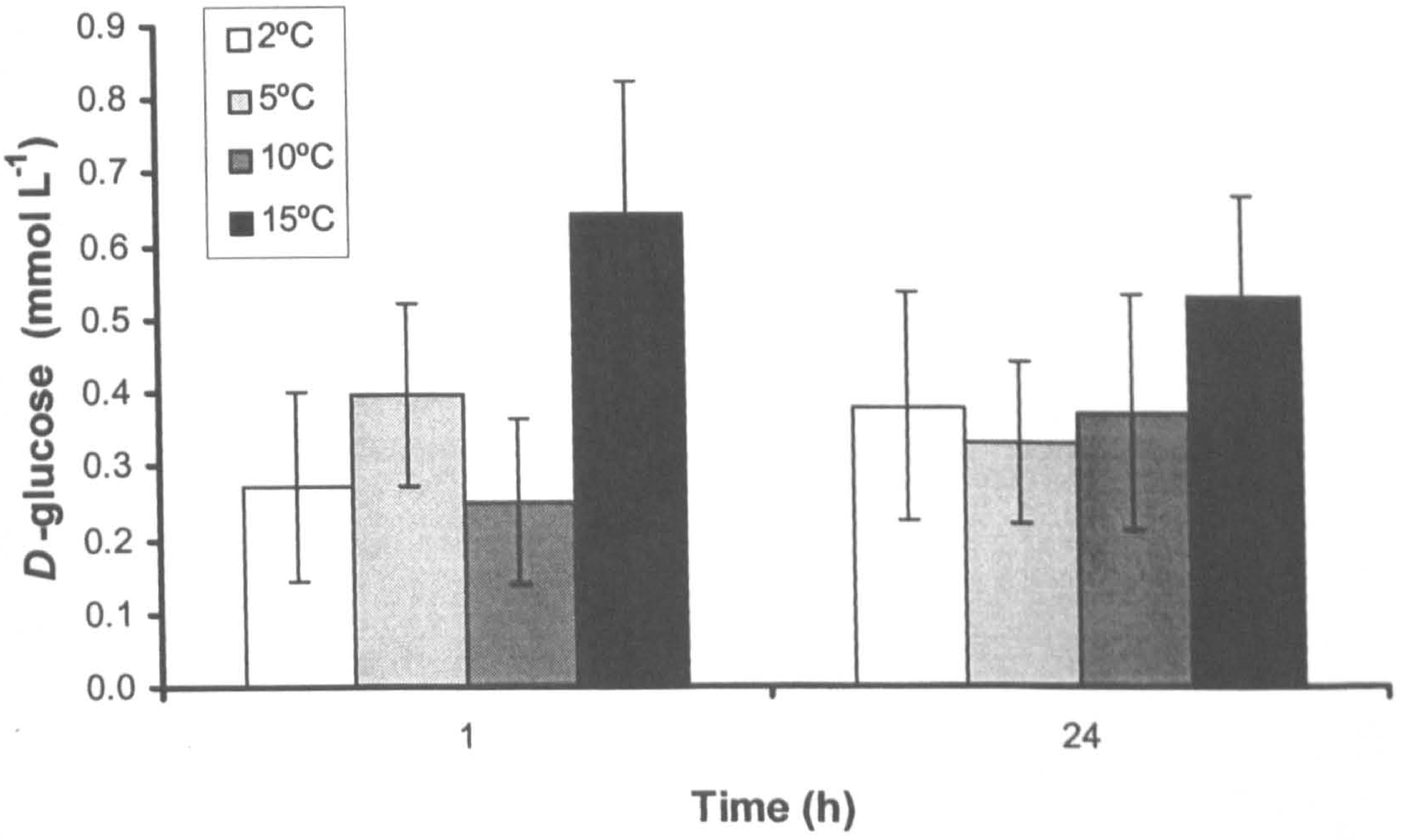

Figure 8.6 - Changes in *Nephrops* haemolymph *D*-glucose concentration with duration of immersion in sea water at different temperatures (control) (mean resting value, 0.45 ± 0.11 mmol l^{-1}). Values are means ± 95% Confidence Limits.

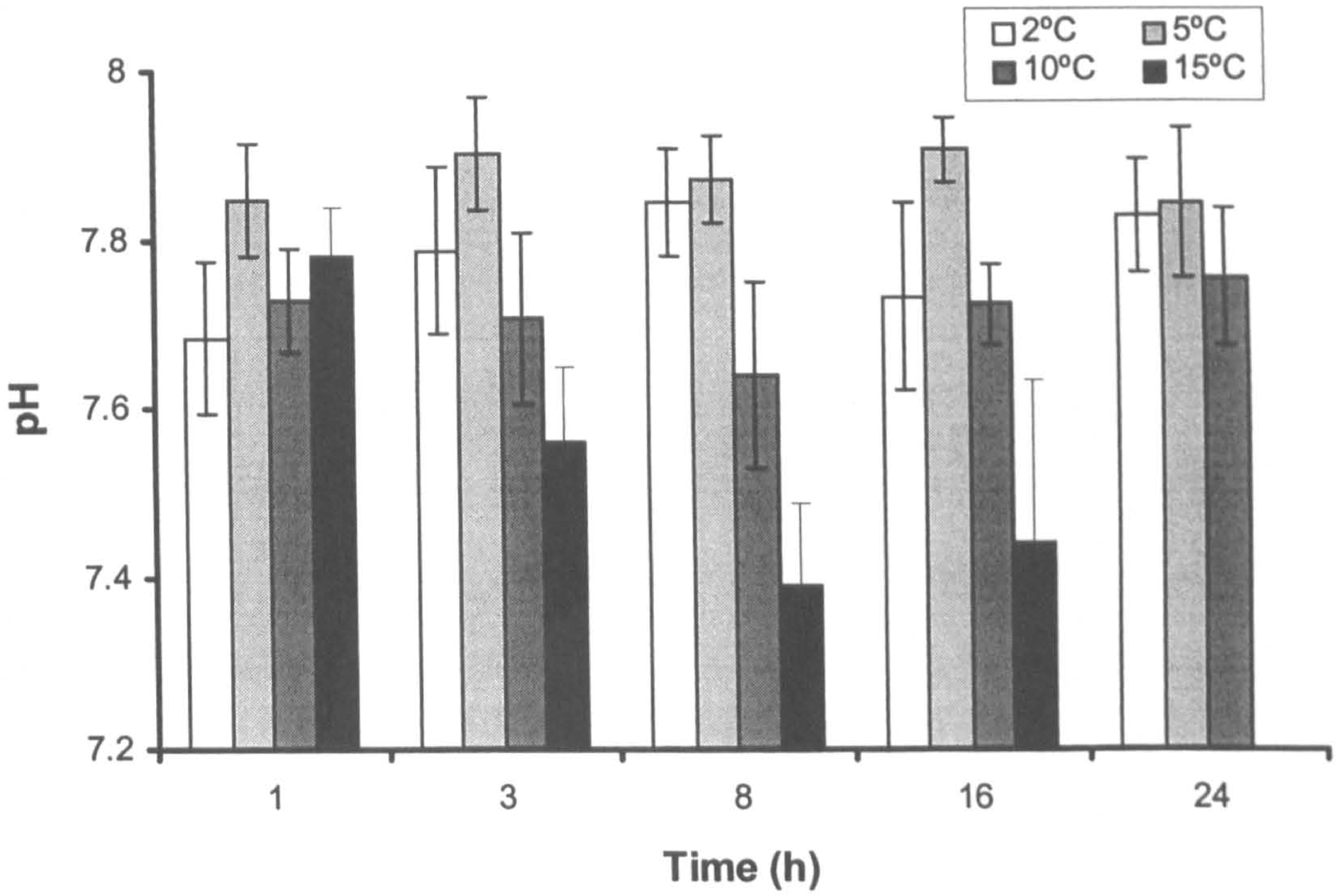

Figure 8.7 - Changes in *Nephrops* haemolymph pH with duration of air exposure at different temperatures, 100% mortality at 15°C following 24h exposure (mean resting value, 7.96 ± 0.03). Values are means ± 95% Confidence Limits.

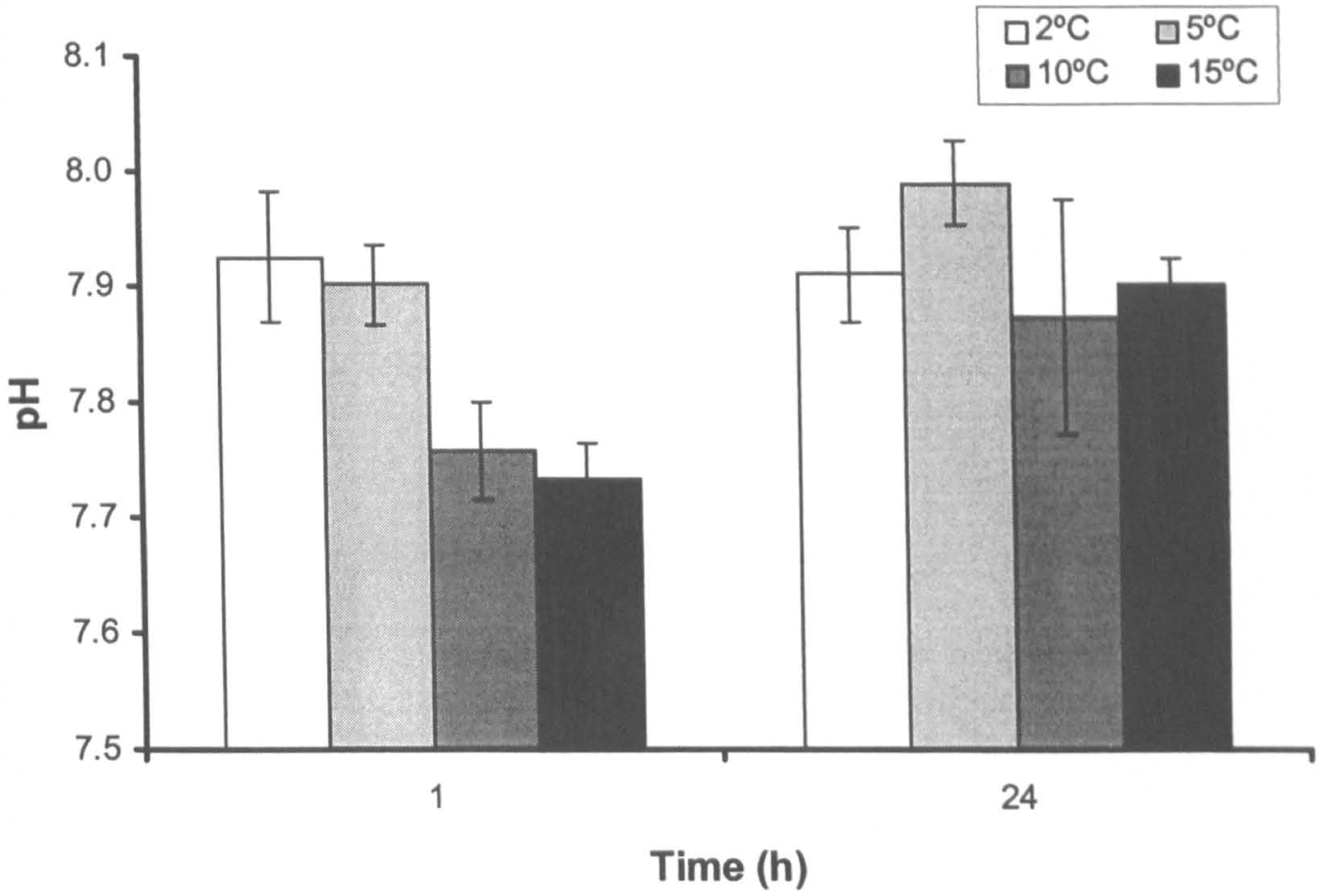

Figure 8.8 - Changes in *Nephrops* haemolymph pH with duration of immersion in sea water at different temperatures (control) (mean resting value, 7.96 ± 0.03). Values are means ± 95% Confidence Limits.

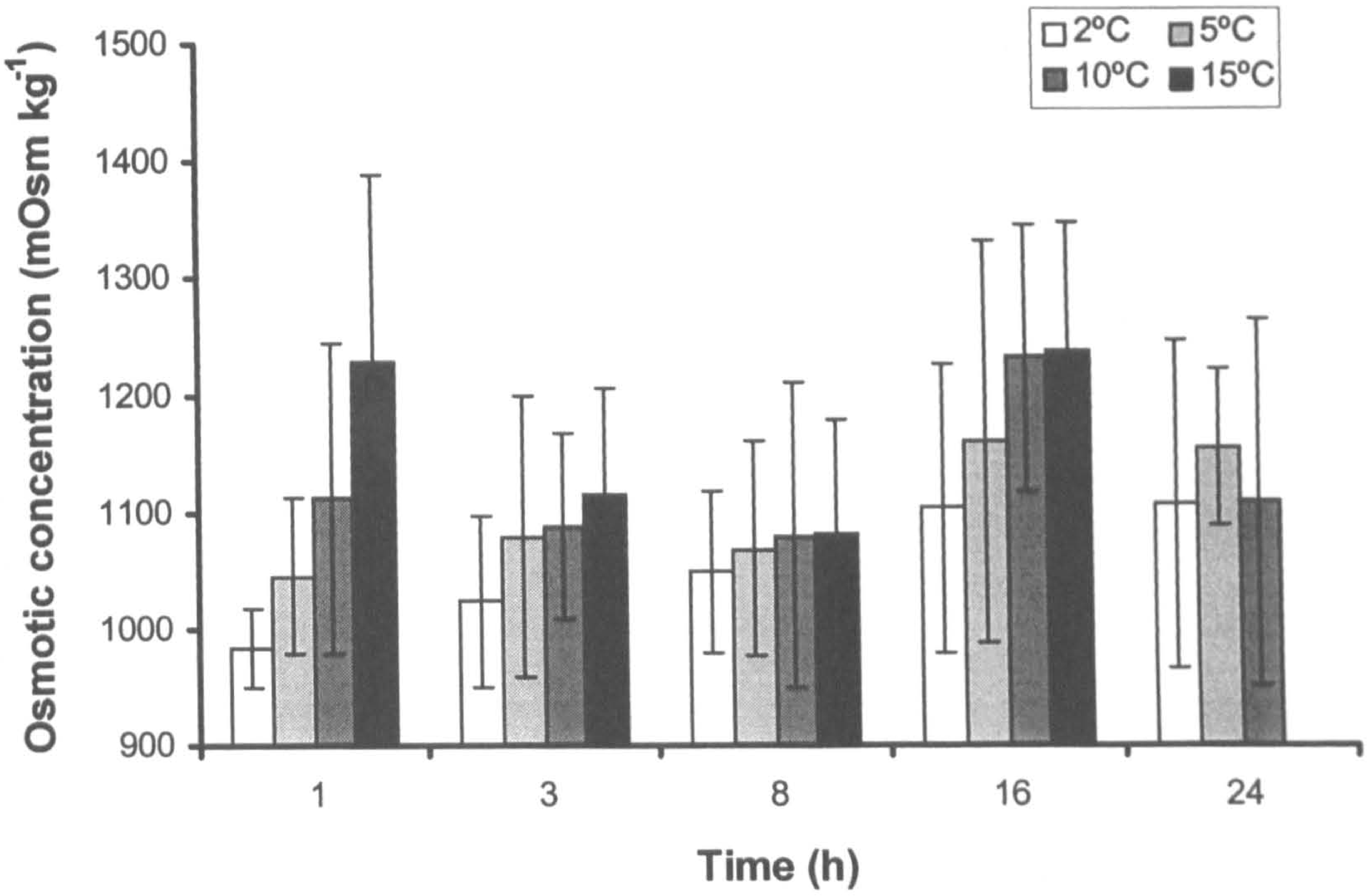

Figure 8.9 - Changes in *Nephrops* haemolymph osmotic concentration with duration of air exposure at different temperatures, 100% mortality at 15°C following 24h exposure (mean resting value, 1090 ± 0.07 mOsm kg^{-1}). Values are means ± 95% Confidence Limits.

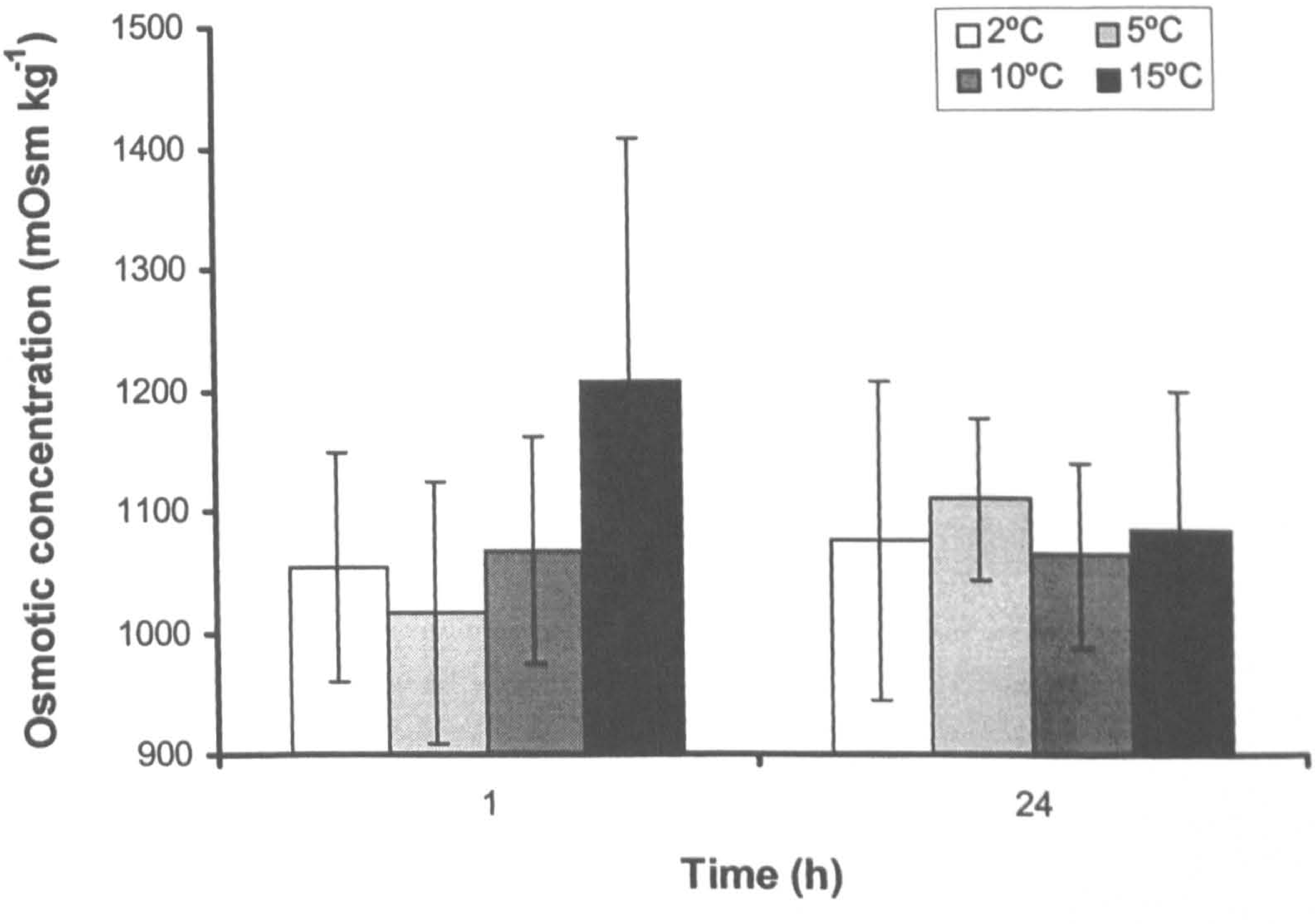

Figure 8.10 - Changes in *Nephrops* haemolymph osmotic concentration with duration of immersion at different temperatures (control) (mean resting value, 1090 ± 0.07 mOsm kg^{-1}). Values are means ± 95% Confidence Limits.

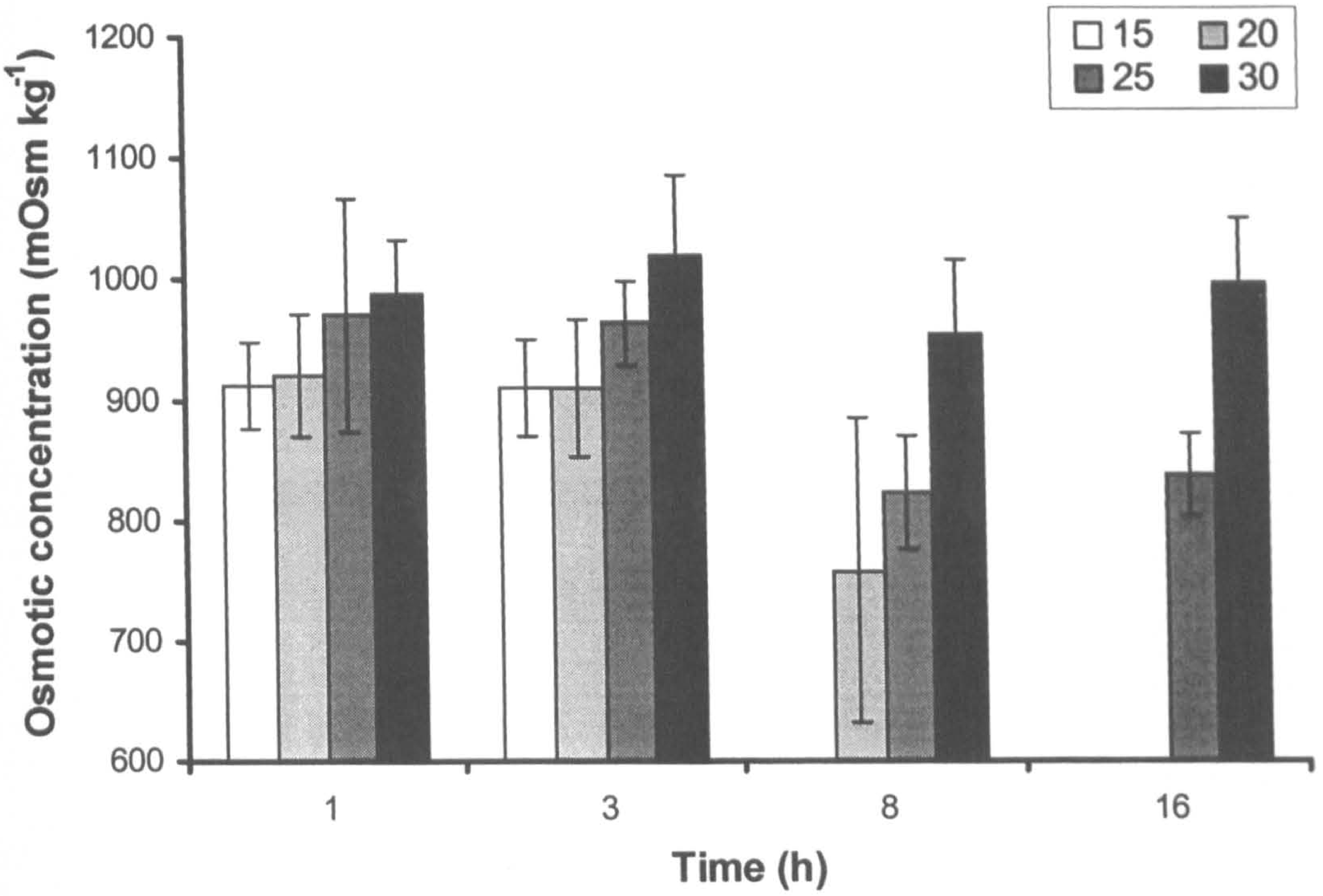

Figure 8.11 - Changes in *Nephrops* haemolymph osmotic concentration with duration of immersion at different salinities, 100% mortality at salinity 15 following 8h and 16h immersion and at salinity 20 following 16h immersion (mean resting value, 1090 ± 0.07 mOsm kg^{-1}). Values are means ± 95% Confidence Limits.

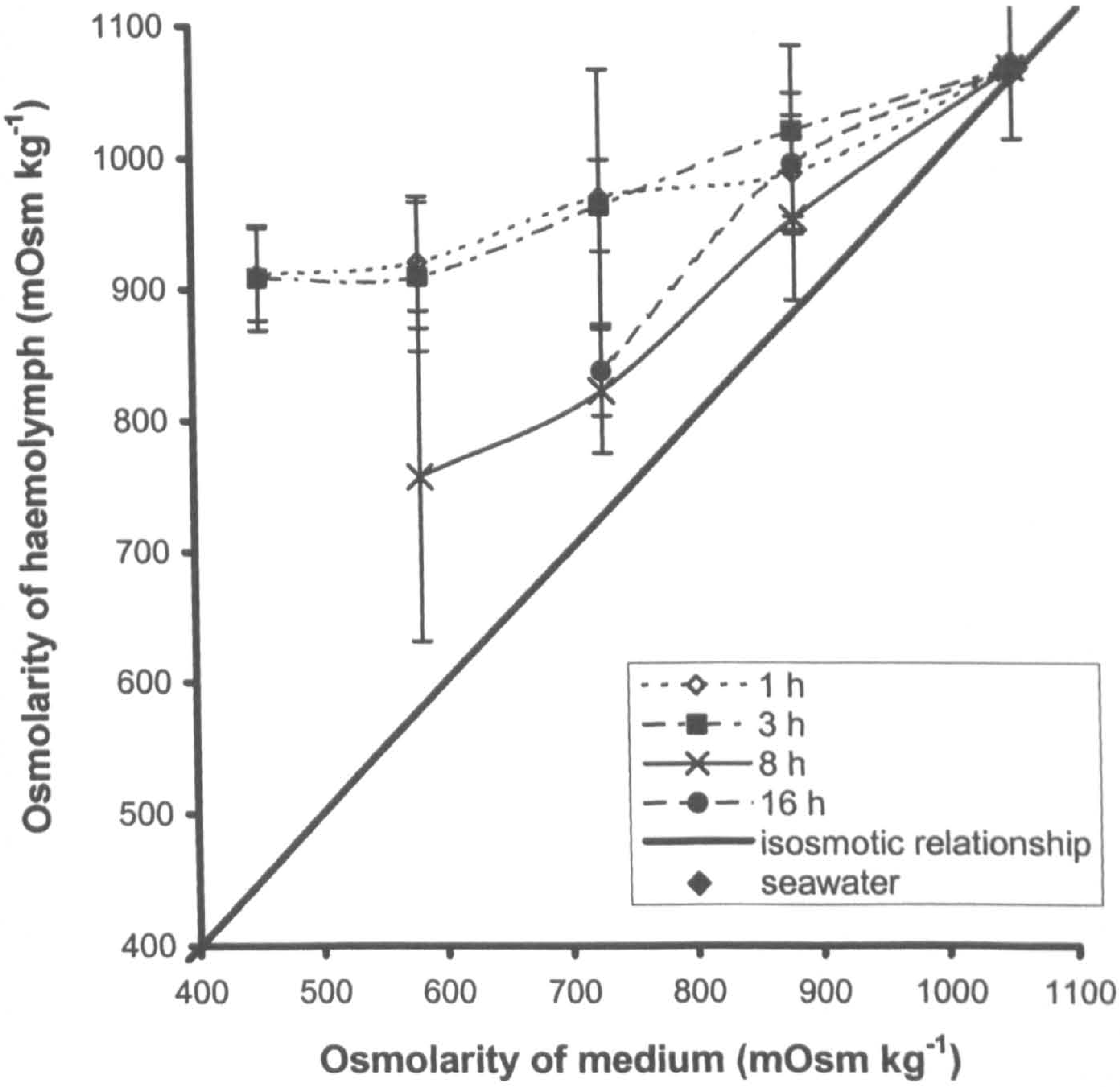

Figure 8.12 - Haemolymph osmotic concentration as a function of external concentrations following immersion for 1, 3, 8, 6 hours and in full strength seawater (black diamond) (mean ± 95% Confidence Limits). Solid diagonal line indicates an isosmotic relationship between haemolymph and medium.

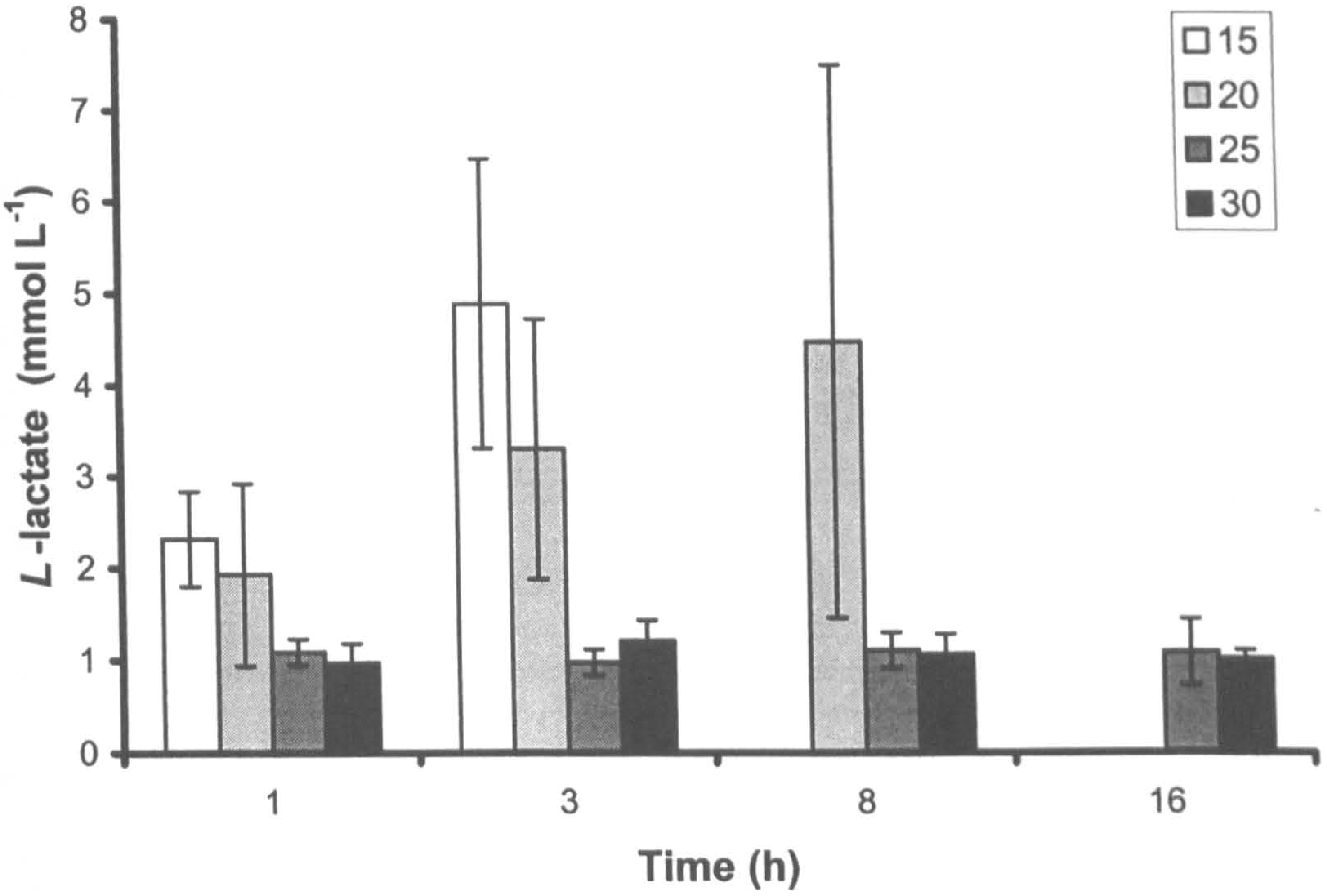

Figure 8.13 - Changes in *Nephrops* haemolymph *L*-lactate concentration with duration of immersion at different salinities, 100% mortality at salinity 15 following 8h and 16h immersion and at salinity 20 following 16h immersion (mean resting value, 0.72 ± 0.17 mmol l^{-1}). Values are means ± 95% Confidence Limits.

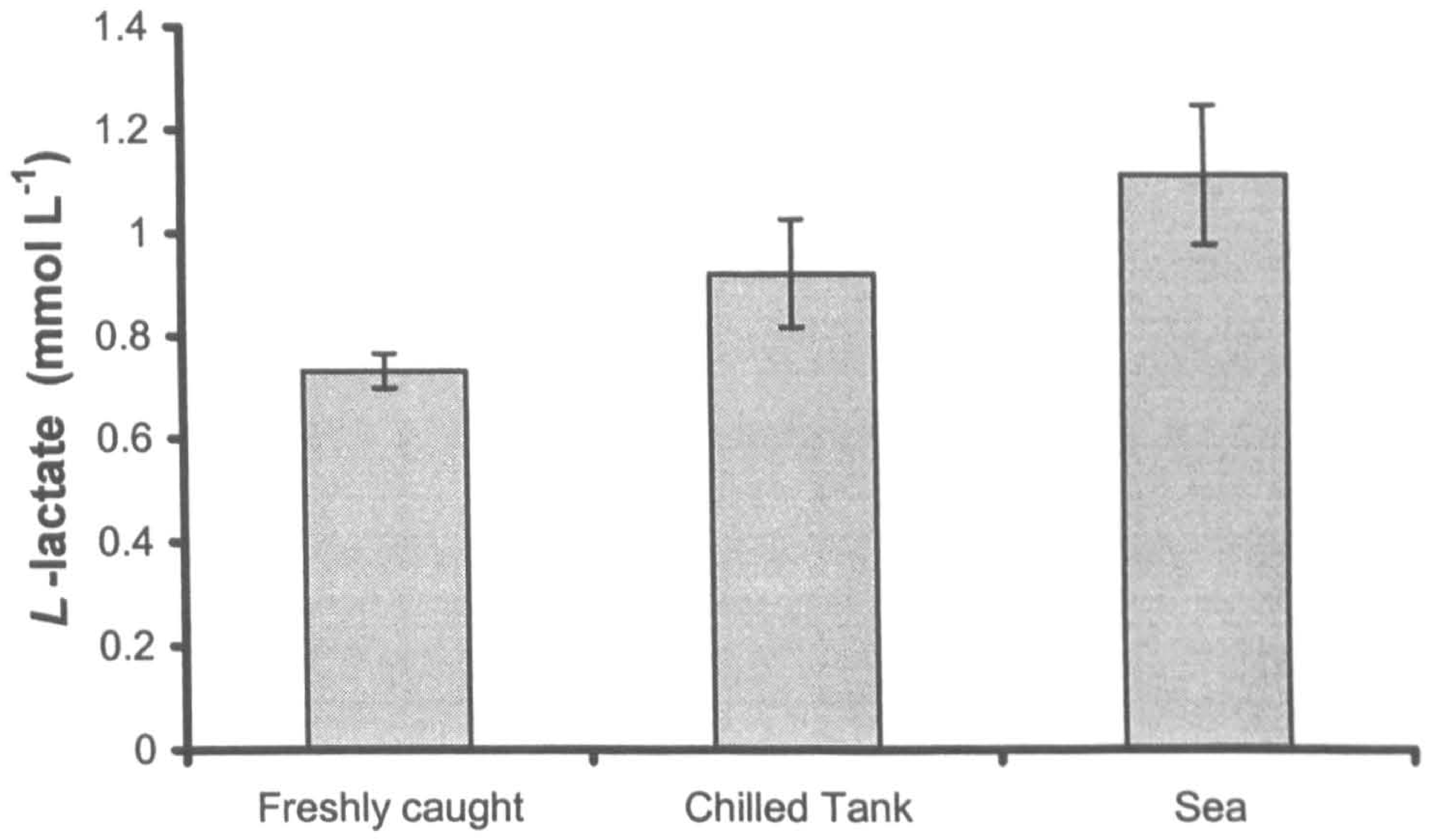

Figure 8.14 - Concentrations of *Nephrops* haemolymph *L*-lactate immediately after capture and following 12 h stored in a chilled tank and in the sea (mean resting value, 0.72 ± 0.17 mmol l^{-1}). Values are means ± 95% Confidence Limits.

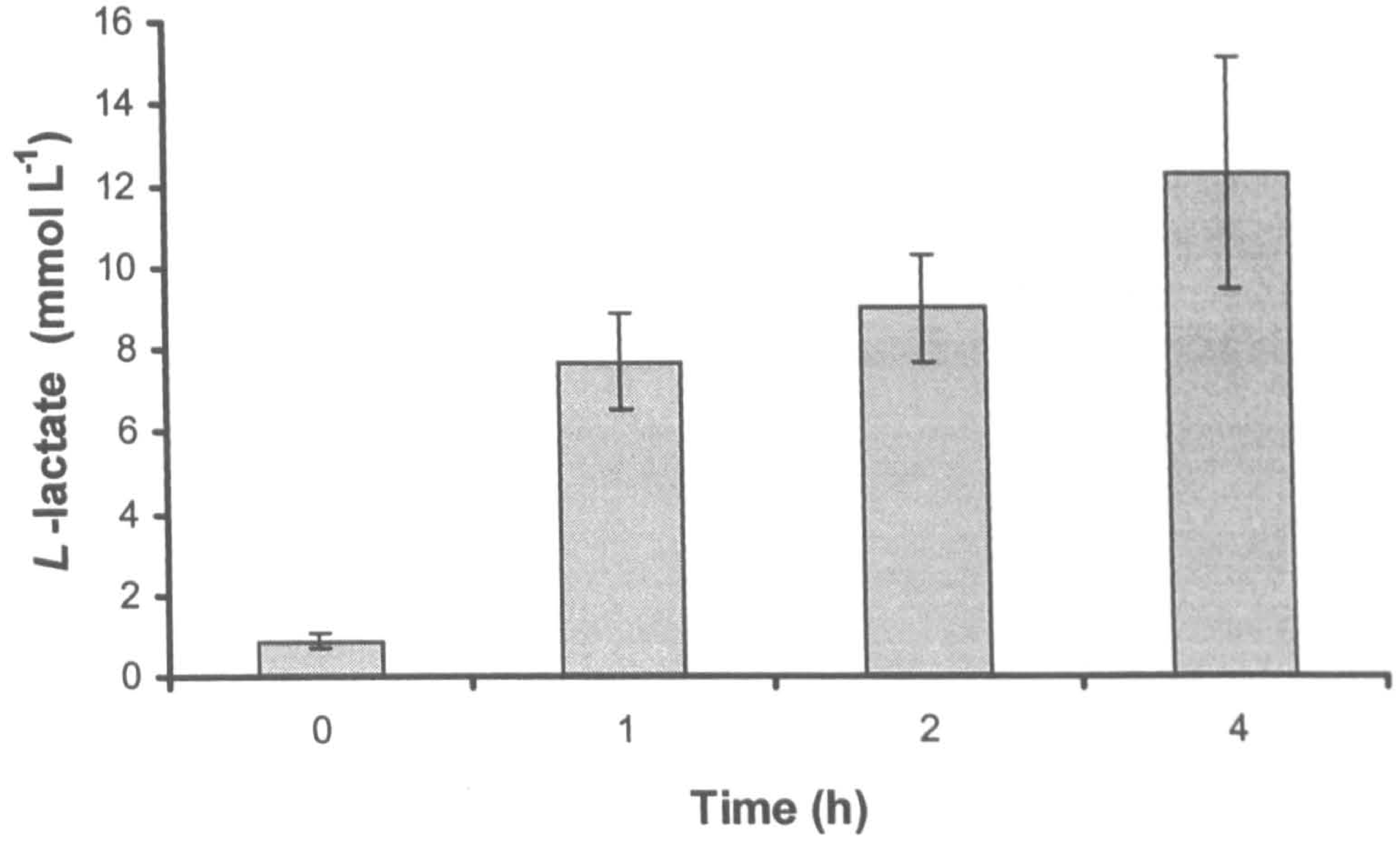

Figure 8.15 - Changes in *Nephrops* haemolymph *L*-lactate concentration with duration of air exposure on the deck of a commercial creel vessel (mean resting value, 0.72 ± 0.17 mmol l^{-1}). Values are means ± 95% Confidence Limits.

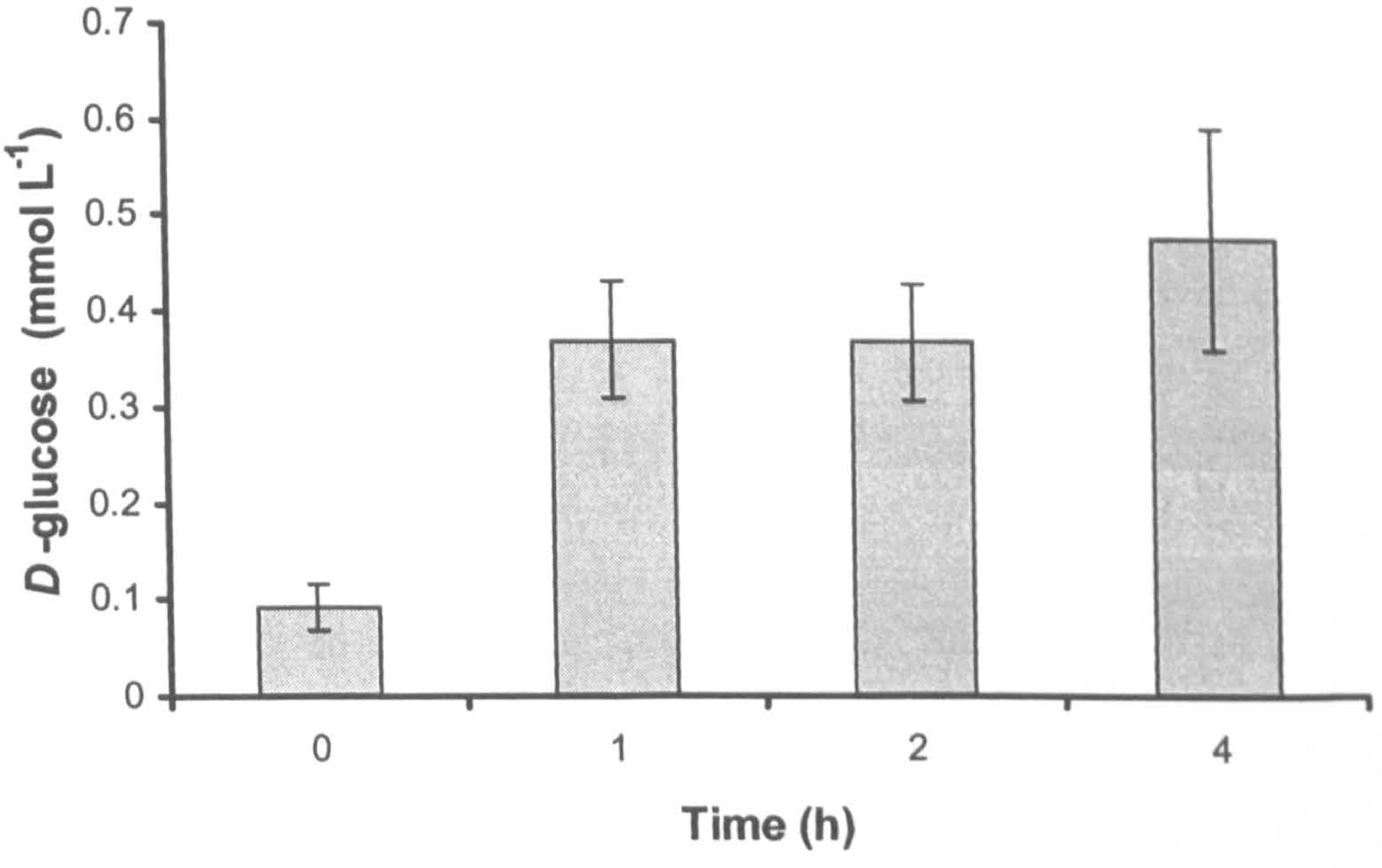

Figure 8.16 - Changes in *Nephrops* haemolymph *D*-glucose concentration with duration of air exposure on the deck of a commercial creel vessel (mean resting value, 0.45 ± 0.11 mmol l^{-1}). Values are means ± 95% Confidence Limits.

Chapter 9

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9 CONCLUSIONS

9.1 Overall conclusions and suggestions for further work

The aim of this investigation was to provide a detailed scientific assessment of a number of aspects of the *Nephrops* creel fishery. A broad approach was considered appropriate, as there was a paucity of information on this fishery. Although studies were necessarily broad ranging, they were undertaken in sufficient detail to enable a number of firm conclusions to be drawn on most aspects of the work. Investigations have provided a detailed assessment of how the biology of *Nephrops* affects its catchability, as well as of the exploitation pattern of the creel fishery. Additionally, work carried out during this study has resulted in a better understanding of the environmental impacts of the *Nephrops* creel fishery and has built on existing knowledge of the physiological effects of the capture and transport of live creel-caught *Nephrops*.

The catch of *Nephrops* in the creel fishery is strongly influenced by seasonal patterns in the moult cycle and, in females, by the reproductive cycle (Farmer 1975). These influences lead to the observed variations in the size distribution and proportions of males and females caught throughout the year. A large peak in female catch follows their moult and mating in the early spring, with catch rates falling to very low numbers after spawning. A peak in male catch rate was observed around February, following moulting. Of those animals caught, males were larger than females, providing a partial explanation for seasonal variations in catch size distribution.

Spatial variations in catches were evident both between and within fisheries, varying in the numbers of constituent animals and their size distributions. It is likely that these variations are due to considerable stock heterogeneity within small areas, an effect of sediment type and fishing pressure (Marrs *et al.*, 2000, 2002a,b). In particular, lower numbers of small animals were observed in soft sediments, caused by a lack of females in the catch. Microscale mapping of fishing effort and landings also revealed spatial variation in both the size distribution and numbers of animals caught per unit of fishing effort. There was some evidence to suggest that effort was targeted in areas where the catch rate of marketable *Nephrops* was greatest. Results also showed a seasonal increase in effort on grounds where different components of the stock become more readily available to the fishery. However, sea-bed topography, competition for resources among fishery sectors, and management measures, are all suggested as reasons affecting the behaviour of fishermen and therefore exploitation patterns.

Seasonal variations in *Nephrops* creel catch composition have been well characterised in this study, although the extent of variations between fisheries and the underlying factors are still not fully understood. Marked differences in catch composition between the fisheries sampled were observed. Unfortunately, for logistic reasons, sampling in several areas was undertaken on an opportunistic basis and did not enable a detailed comparison to be made. Further work incorporating a regular sampling regime in a number of distinct creel fisheries is essential to increase understanding of how catch composition varies between fisheries.

Microscale mapping using PRUs (Position Recording Units) was very successful in this study. There were some problems associated with the methods used, including catch being recorded on a daily basis, which led to instances where catch was aggregated over different areas. Batteries had a limited life leading to long periods when no data were collected. Furthermore, management measures in place in the Torridon area, where this trial was conducted, meant that this fishery was not representative of the majority of creel fisheries. A combination of regular catch sampling and the implementation of an improved PRU / logbook system in a number of distinct creel fisheries would provide a better understanding of spatial variations in catch composition within *Nephrops* creel fisheries.

The introduction of escape gaps in the Torridon creel fishery led to a reduction in the catch of the smallest animals. Additionally, some evidence suggests that this size selection has had a positive effect on the stock within this area. LCA undertaken during this study, of the Torridon fishery, suggests that the stock is at less risk of growth overfishing than when it was assessed during the periods of 1996 to 1998 (Tuck & Bailey, 2000). Furthermore, none of the fisheries studied was at risk of growth overfishing ($F<F_{max}$), with females being particularly lightly exploited ($F<F_{0.1}$).

It is likely that both the spatial and temporal variations in *Nephrops* catch observed in all fisheries, together with the assumptions made by LCA, mean that this approach may not be appropriate in assessing stocks exploited by creel fisheries, particularly where catch composition and landings data are sparse. To provide the best management advice on creel fisheries, more information about the micro- and large-scale variations in catch composition and fishing effort, followed by appropriate sampling based on this knowledge are required, allowing for a more stratified approach to stock assessment in these areas (Marrs *et al.* 2000, 2002b).

Behavioral studies, both in the field and in aquaria, indicate that the catch of *Nephrops* is low compared with the number of individuals attracted to the creel, with only 0.2–7% of approaches to a creel leading to capture (individual animals are likely to approach several times before capture). The present results provide further evidence that low entry rates to *Nephrops* creels occur due to: a low motivation for food search, creel repellancy (i.e. factors associated with creel avoidance), difficulty in finding creel entrances and intra- and inter-specific interactions (Bjordal, 1983). This suggests that *Nephrops* creels are not very efficient and will capture only a small proportion of the resident population. Creels were also shown to select larger *Nephrops*, with evidence that larger animals are more likely to enter the creel and less likely to leave the vicinity of the creel following an encounter with a smaller conspecific. In addition to these observations, it appeared that interspecific (crabs) interactions deterred *Nephrops* from entering a creel.

Although the present study provided valuable information on *Nephrops* creel-related behaviour, there were some discrepancies between studies in the field and aquarium. This is probably due to the unnatural conditions in the aquarium, as well as the field study site being located on the periphery of a *Nephrops* ground, where *Nephrops* population densities were low. Further fieldwork using improved technology is necessary in deep-water sites (>60 m) that are more representative of *Nephrops* creel fisheries. In such areas, it is likely that *Nephrops* densities will be higher and therefore, activity around creels will be greater, enabling a more valid comparison with aquarium observations. FRS have now overcome the problem of insufficient light output from infrared LEDs (C. Shand, pers. comm.), so progress in the study of *Nephrops* responses to creels at deep water sites should be possible in the future. Further work in aquaria is essential to enable the manipulation of factors such as stocking density, size distribution and sex ratio. This information will be invaluable in interpreting seasonal and geographic variations in catch composition in the fishery, as well as providing a better understanding of why catch composition in trawl and creel fisheries differs.

Fishery investigations, as well as behavioural observations, have shown that a number of factors influence catchability of *Nephrops*, leading to large spatial and temporal variations in creel catch composition. It must therefore be considered that whilst management measures imposed in the Torridon area (partly statutory – the zoned fishery areas, and partly non-statutory – the conditions necessary for Marine Stewardship Council Certification of a sector of the creel fishery) appear to be effective, these measures are not directly transferable to other fisheries and measures must be tailored to suit their specific requirements. Furthermore, the zoned fisheries policy in the Torridon area has highlighted the need to provide fishermen with a sense of ownership of their resource in order for such management measures to succeed (Hardin, 1968).

The physical damage from creeling is low. Sea pen species *Virgularia mirabilis* and *Pennatula phosphorea* do not appear to be adversely affected by either trawling or creeling, with *V. mirabilis* being fairly ubiquitous in all locations and depths and the densest aggregations of *P. phosphorea* being found in shallow loch systems. *Funiculina quàdrangularis* was found in a number of locations but occurred in the highest densities in the Inner Sound. Its associated brittle star *Asteronyx loveni* was found almost exclusively in deep-water environments of the Inner Sound. Both species appear to be reduced in numbers in areas where trawl fishing occurs, with densities of *A. loveni* being particularly low in these areas. Numbers of *F. quadrangularis* seem to be unaffected by creel fishing. However, there was some evidence to suggest lower numbers of *A. loveni* in these areas. There was apparently contradictory evidence from the prevalence of damaged and dead colonies, with greater numbers of damaged or dead colonies in the creeled area when compared with the trawled area. Further research is required on this issue. The differences in species composition of benthic megafauna between fishing zones are likely to be largely due to differences in habitat characteristics between zones. A direct comparison of species composition between the creel and trawl zones, with similar habitat characteristics, did not reveal a significant difference, but this was perforce based on few samples.

Data on species composition and sea pen density analysed in this study were derived from surveys designed for another purpose (*Nephrops* stock assessment). This meant that a direct comparison of the epibenthos between areas was only possible at a few sites. Future studies to investigate the effects of fishing practices on the epibenthos in the Torridon area should endeavour to survey sufficient stations of similar habitat characteristics in zones fished by creeling or trawling. The incorporation of sediment data and effort data (creel and trawl) would enable a more accurate determination of physical impacts of each fishing method.

When compared with the trawl fishery, discard levels in the *Nephrops* creel fishery are much lower (Evans *et al.*, 1994; Craeymeersh, 1994; Bergmann *et al.*, 2002a). This was found to be particularly true for commercially exploited fish, which comprised only a very small proportion of the catch in the creel fishery, compared with the proportion of commercial fish reported to be caught as bycatch in the trawl fishery (Bergmann *et al.*, 2002a). For the creel fishery, both numbers and biomass of discards varied among different areas, with the major factor being differences in community composition between areas. However, there is also some evidence of a gear effect, whereby the presence of escape gaps may reduce the catch of some crab species. The mortality of both target and non-target discards is assumed to be low, with preliminary studies indicating the only significant source of mortality may be from predation by seabirds. It is clear that in comparison with *Nephrops* trawl fisheries, the bycatch associated with the creel fishery is not only much lower, but the survival of discarded species may be higher (Evans *et al.*, 1994).

Bycatch associated with creel fisheries in Loch Fyne and in the Torridon area is now well understood. These two areas are not, however, representative of the whole fishery and similar work in other areas would be beneficial. The issue of bycatch survival was not thoroughly assessed during this study. A further assessment of predation by seabirds and the factors that influence it should be carried out in a variety of fisheries. Behavioural results suggested that once returned to the seabed, *Nephrops* were able to resume normal behaviour. In contrast, tagging studies showed very low re-capture rates, suggesting a high mortality of *Nephrops* discards (although this result is a likely reflection of the low entry rate of *Nephrops* to creels and high stock densities in this area). These preliminary trials show that the survival of *Nephrops* once in the water column and on the bottom requires further investigation. Furthermore, the efficacy of returning ovigerous females also requires investigation in relation to the viability of the eggs following aerial exposure.

In the ghost fishing trials, the majority of captured species were able to escape from the creels, with only *Nephrops* being trapped for long periods of time. This suggests that creels are very selective for their target species (despite having low catch efficiency). Dead target or non-target species within creels seem to be ineffective at attracting *Nephrops* into the creels. It can therefore be assumed that following the initial attraction to creels, once all the bait has been consumed, lost creels will cease to fish. The main reasons for the low ghost-fishing performance of *Nephrops* creels appear to be: the design of the creel itself, which allows non-target species to escape relatively easily ('hard' eyes), the behaviour of *Nephrops* restricting the number of animals entering the creels once the bait has been consumed, and the ability of *Nephrops* to survive for long periods of time once caught. In addition to these findings, there was evidence that escape gaps reduce the number of swimming crabs (*Liocarcinus depurator*) resident in the creels. Since crabs appear to deter entry of *Nephrops* to creels, the use of escape gaps may indirectly increase the *Nephrops* catch.

Further ghost fishing trials could be carried out under different environmental conditions, however, results produced from this study appear to be conclusive, showing that mortalities of both target and non-target species due to lost *Nephrops* creels are very low.

These findings suggest that, although not without impact, *Nephrops* creels have little detrimental effect on the benthic environment, with impacts appearing considerably lower than for the trawl fishery. It may therefore be considered that spatial management of *Nephrops* fisheries is a very valuable means to protect key biotopes from excessive disturbance.

There is little stress associated with the capture and storage of creel-caught *Nephrops*. The greatest stressors appear to be aerial exposure and exposure to low salinities, with temperature and salinity being very important in determining stress levels and ultimately survival. It is clear that exposure to salinities below 25 and aerial exposure at temperatures above 10°C, both lead to rapid mortality. This demonstrates the need to store animals in undiluted seawater on board fishing vessels and that chilling of *Nephrops* prior to and during transport are effective ways of ensuring survival.

The present physiological study not only complements previous studies (in particular, Ridgway *et al.*, 2006a,b), but also assessed stressors in the context of the creel fishery. Although sound conclusions can be drawn from this work, the majority of investigations were carried out in the laboratory. Each stressor was treated individually (capture, storage and transport), which does not reflect commercial conditions. Stress caused at each stage of the process is likely to be compounded. An investigation of stress under commercial conditions would provide a better understanding of the physiological effects of the process as a whole.

Throughout this study, qualitative comparisons with the trawl fishery have been made, by comparing results from this study with previous work investigating the trawl sector. This report has highlighted differences in catch composition between sectors, both seasonally and spatially, suggesting that catch in the trawl fishery is related to light-modulated emergence patterns (Chapman *et al.*, 1972, 1975), whereas creel related behaviour also influences creel catch (Bjordal, 1983). These differences may, however, simply be due to the different sectors exploiting different *Nephrops* populations (although this is unlikely to be the sole reason for the observed differences). Environmental impacts were suggested as being greater in the trawl fishery; in particular the bycatch was much higher in previous trawl studies, although bycatch has been observed to vary considerably over relatively small spatial scales.

Only qualitative comparisons were possible during the current study, since creeling and trawling take place in different areas with different *Nephrops* populations. It is not possible at present to compare the exploitation pattern of the two fishing methods directly, or to evaluate their potential effects on stocks. In order to compare the two sectors, it is essential to evaluate variation in *Nephrops* catch in relation to gear in a controlled, quantitative manner, by experimentally fishing with each gear type on the same ground. This would enable a direct comparison of catch composition of each fishing method when applied over a common area of known *Nephrops* stock density, as well as allowing a direct comparison of non-target bycatch.

9.2 The present work within the context of other crustacean creel fisheries

The current study is not only the first of its kind to investigate the *Nephrops* creel fishery, it also addresses several key issues, which are poorly understood for creel fisheries in general. This is particularly true for the environmental impacts of fishing with creels: there was little prior information about the physical impacts and bycatch associated with this method of fishing. There have also been few previous spatially explicit assessments of catch and effort from creel fisheries. These subjects have been assessed in detail in this study, and have provided data to inform several broader issues in fisheries science. *Nephrops* habitat and life history is very different from the majority of other commercially exploited crustacean species (for example, *Nephrops* lives in a soft mud habitat, occupies burrows and is non-migratory), enabling comparisons between these different creel fisheries to be made.

Spatial and temporal variations in catch of the type observed in this study occur in a number of other fisheries, using both static and mobile gear. Catch composition in the majority of crustacean fisheries is known to vary throughout the year (Miller, 1990). As with *Nephrops*, females of other species are less vulnerable to creel fishing (Miller, 1990). Results from the present study provide further evidence to support the large body of previous work, which demonstrates that catches in crustacean creel fisheries are affected by the moult and reproductive cycles of the target species (for a review see, Miller, 1990). These largely annual cycles influence feeding activity (Miller, 1990) and seasonal migrations (for example, Herrnkind, 1983; Freire & González-Gurriarán, 1998; Hunter, 1999; Bellchambers & de Lestang, 2005; Corgos *et al.*, 2006) and thus catchability, for a number of species. *Nephrops* differs from these species, in that the variations in catch observed are almost solely due to differences in feeding activity, as they move only short distances (Chapman *et al.*, 1975; Chapman; 1980). It is important to take account of the type of temporal patterns in catchability shown by *Nephrops* when designing experiments using creels or when surveying abundance by trap sampling (Greenwood, 1996). If the exploitation patterns of different fisheries are to be compared, differences in behaviour must be considered.

Spatial variations in catch are poorly understood, particularly in the context of creel fisheries. Only a handful of studies of creel fisheries have investigated this and these have been conducted in discrete areas over relatively large spatial scales (Gobert & Stanisiere, 1997; Steneck & Wilson, 2001; Tremblay & Smith, 2001). The results from the current study has for the first time in a creel fishery demonstrated that fishing effort is highly aggregated in distinct areas, with catch rate as well as a number of other factors such as competition with other boats, fishing boundaries and distance from port affecting the behaviour of creel fishers and thus the distribution of effort. The use of high-resolution spatial data is essential if fisheries research is to be applicable to the activities of commercial fleets (Kaiser, 1998). This is particularly important in the context of creel fisheries, which often fish spatially heterogeneous environments and are able to target discrete areas.

The behavioural responses of other decapod crustacean species towards baited creels have received more attention than in the case of *Nephrops*. These studies have found that there are a number of variables affecting catch rates, sex ratios and size composition of catches (for example, Miller, 1978; Shelton, 1981; Karnofsky & Price, 1989; Miller, 1990; Miller & Addison, 1995; Zhou & Shirley, 1997; Jury *et al.*, 2001; Vazquez Archdale *et al.*, 2003). The current study has shown that *Nephrops* creel related behaviour affects not only the number of animals entering the creel, but also the size of animals caught. This selectivity resulted from different levels of gear avoidance and responses to intra- and inter-specific interactions according to the size of the animal, as found in several other species (Miller, 1978; Shelton, 1981; Karnofsky & Price, 1989; Miller, 1990; Miller & Addison, 1995; Zhou & Shirley, 1997; Jury *et al.*, 2001; Vazquez Archdale *et al.*, 2003). In contrast to previous studies, where low catch rates appear to result from a combination of low creel-entry rates and subsequent escape (Karnofsky & Price, 1989; Jury *et al.*, 2001), low catch rates of *Nephrops* in creels appear to be almost solely due to the low tendency of *Nephrops* to enter creels.

In comparison to fishing using towed bottom gears, creels might be considered to be environmentally benign, since they are highly selective for their target species and have a low physical disturbance to the seabed (Jennings & Kaiser, 1998). For this reason static gear has received very little attention. Results from this study highlight the importance of monitoring environmental effects of creel fisheries, with results showing that although when compared to trawl fisheries impacts appeared significantly lower, creel fisheries should not be considered as being environmentally benign.

Physical disturbance of the seabed is particularly poorly understood, with only two previous studies providing some evidence to suggest that structural species may be damaged by static gear (Eno *et al.*, 2001; Troffe *et al.*, 2005). In the current study, creel fishing, unlike the trawl fishery, was shown to have little detrimental impact on structural species (*F. quadrangularis*). There was, however, an observed reduction in the symbiotic brittle star *A. loveni* on creel ground when compared with an area where fishing is restricted, suggesting that damage by creels may still occur to some fragile species.

Discarding bycatch is important in world fisheries, accounting for around 25% of the total world catch (Alverson *et al.*, 1994). *Nephrops* trawl fisheries are ranked fifth in the world with respect to discard ratio (number of bycatch to number of target species) (Alverson *et al.*, 1997). Unlike trawling, crustacean creels are thought to be highly size- and species-selective (Miller, 1990), implying that the amount of bycatch of both target and non-target species is significantly lower than for trawl fisheries. Considering the importance of this issue it is surprising that there has been very little previous work to assess the bycatch associated with creel fisheries. This study has been the first of its kind to provide a detailed assessment of the composition of both target and non-target species discarded in a creel fishery. Results concurred with previous suggestions that creel bycatch was significantly lower than that of the trawl fishery (Harrington *et al.*, 2005). In particular, results showed commercially exploited fish make up a small proportion of the bycatch in *Nephrops* creel fisheries. These results provide further evidence to suggest a change to more selective fishing gears could be an important tool in reducing bycatch and protecting nursery areas for commercially exploited fish stocks (NMFS, 2003).

Ghost fishing is the best-studied aspect of environmental impacts in crustacean creel fisheries. This is because ghost fishing is thought to be the major source of environmental impact in the present creel fishery industry (FAO, 1995; Jennings & Kaiser, 1998). High levels of mortality from ghost fishing have been shown in a number of species (for example, Breen, 1987; Kimker; 1994; Bullimore, 2001; Herbert *et al.*, 2001). In contrast, studies of king crabs (*Paralithodes camtschatica*) showed that they were able to escape from creels and ghost fishing mortality was low (High & Worlund, 1979; Godoy, 2003). Only one previous study has recorded catches of all species in a fleet of ghost fishing pots and mortality, not only of the target species, but also of other crustacean species and fish (Bullimore *et al.*, 2001). Both experiments in the current study showed that in contrast to the conclusions of the majority of previous studies, ghost fishing is unlikely to be a serious impact of the *Nephrops* creel fishery. It was observed that the design of the creel allowed all non-target species and the majority of *Nephrops* to escape. Creel design is essential in determining the degree to which a creel will continue to fish once lost. *Nephrops* creels are of a particularly good design in this respect, as they are very selective for their target organism, retaining them for a sufficient duration to enable capture, but if creels are lost, animals may eventually escape.

Several previous studies into the stress associated with the capture, storage and live transport of crustaceans, have provided similar results to those observed in the current study. Similar responses to aerial exposure and low salinities have been observed in a number of subtidal crustacean species not adapted to such variations in environmental conditions (Mantel & Farmer, 1983; Whyman *et al.*, 1985; Jussilla *et al.*, 1990; Beard & McGregor, 1991; Jacklin, 1996; Spanoghe & Bourne, 1997; Harris & Ulmestrand, 2004; Ruiz-Capillas, 2004; Ridgway *et al.*, 2006a). Although exposure to low salinities during the storage phase was found to cause significant stress, temperature was found to be particularly important during transport out of water, with lower temperatures observed to reduce the adverse effects of emersion. This has led to the temperature at which a number of commercially exploited crustaceans are stored being reduced prior to, and during transport (Whiteley & Taylor, 1990; Goodrick *et al.*, 1993; Paterson, 1993; Spanoghe & Bourne, 1997; Morris & Oliver, 1999a,b).

It is clear that *Nephrops* creel fisheries have many similarities to other creel fisheries exploiting different crustacean species. In general, as observed in the majority of other fisheries, the *Nephrops* creel fishery showed temporal and spatial variations in catch, low environmental impact and catch being affected by creel-related behaviour. When comparing such studies it is essential to consider differences in the biology and habitat of the species investigated, as well as creel design. *Nephrops* are fished in a different habitat to that of the majority of commercially exploited crustaceans. The impacts of fishing are therefore concentrated on dissimilar benthic assemblages, with different resiliencies to disturbance. The environment itself may affect *Nephrops* behaviour as the lack of shelter and presence of a creel in a largely featureless environment may make them less likely to enter the creel or to search for long periods, compared with fisheries in a more diverse rocky environment. Differences in gear design will also lead to variations between fisheries, with clear evidence from this study showing that the design of *Nephrops* creels significantly reduces the environmental impact of the fishery. The lightweight design of the creel reduces the physical impact and 'hard' eyes enable the majority of animals to escape if the creel is lost. Differences between fisheries may lead to successful management measures in one fishery having little or even negative effects in another.

9.3 Assessment of management measures and recommendations

In the Torridon creel fishery, escape gaps placed in creels have been shown to be an effective and relatively inexpensive technical measure, allowing smaller *Nephrops* and certain other bycatch species to escape. The size distribution of *Nephrops* in the Torridon area makes it commercially worthwhile for fishermen to leave smaller animals in the sea and harvest them at larger sizes when they are more valuable. The escape gaps used in the Torridon area, may allow animals judged to be of commercial size elsewhere, to escape. In some areas, these smaller animals may account for a large proportion of the catch. If more widely applied, it would therefore be important to adjust the size of the escape gap in relation to the stock characteristics. A possible alternative to escape gaps could be to increase the mesh size of creels enabling smaller animals to escape through the meshes.

Management of effort within the Torridon area has led to a decrease in the number of creels fished and a restriction in fishing days by individual fishermen who comply with the voluntary management policy. This reduction in effort by individual fishermen has been followed by an increase in the number of boats fishing within the area (Chapter 1; Section 1.3.5). If effort is to be managed in a creel fishery, it is important to limit not only the number of creels, but also the number of boats exploiting the resource. Furthermore, limiting effort by number of creels and days at sea may not be suitable for different fisheries. For example, reducing the number of creels fished in a day would have more impact in a fishery that exploits a stock in a very exposed sea area, where they are limited in the days they can fish, when compared with a fishery in a relatively sheltered area. Conversely, limiting the days at sea would have a more adverse effect on the sheltered fishery. These are issues would need to be addressed at both local and national levels.

From the present study, it is not possible to draw any conclusions on the effect of discarding ovigerous female *Nephrops*. At certain times of the year ovigerous females make up a substantial part of the catch (although catches in this study were not as high as previously reported) and discarding them could be interpreted as leading to significant loss of profits during these times. The benefits of returning ovigerous females to maintain the brood stock appear obvious, but require investigation. Research into the viability of the eggs of discarded females, as well as the distribution of larvae within defined areas, to establish the location of the parent spawning stock would help inform this debate.

Managing effort and the use of escape gaps could, in the short-term, lead to a reduction in profits. Such measures, even if made compulsory, are very hard to police. It is therefore important to provide fishermen with an incentive to abide by such measures.

This was achieved in the Torridon area by the zoned fisheries policy creating a large sea area where *Nephrops* could only be fished by creels, providing the fishermen with a sense of ownership of their resource (Hardin, 1968). It was assumed that by reducing the numbers of the smaller animals caught (by use of escape gaps), they would be caught again subsequently at a larger, more valuable size. The zoned fisheries policy was also successful in reducing conflict between the two sectors in the area. Furthermore, restricting trawling to defined areas may protect sensitive areas from physical damage as well as reduce the bycatch of juvenile fish in nursery grounds. Creel fishing in these areas will still have some impact; however, this impact is likely to be much lower than if the area was trawled. This is a more practical solution than completely closing an area to fishing, as it will have a less detrimental effect on the local economy. In addition to this fishery management tool, debate still surrounds the effectiveness of total closure of areas to all fishing methods, a situation that exists in part of the Torridon area because of military activity. The effect of this particular closed area on the local fishery is unknown, but a reservoir of unfished *Nephrops* is perceived as being beneficial.

Results from this study have suggested that if individual creel fisheries are to be managed using similar tools to those applied in the Torridon area, measures must be tailored to suit specific circumstances. This is important in order to protect the local stock in a way that will not adversely affect the livelihood of fishermen. In principle, establishing local inshore fisheries management groups provides a platform for this to be achieved (Scottish Executive, 2005). There are still large gaps in the current knowledge of how *Nephrops* creel fisheries exploit the stock and how this varies in different fishing areas. There is therefore a need for continued research into the

Nephrops creel fishery, to provide information essential for future management of this expanding fishery.

This need for continued research in relation to the *Nephrops* creel fishery has never been greater and is largely due to the recent increases in effort in a number of fisheries, leading to fishermen having to abandon their traditional practices of annually rotating their gear in different areas, in favour of fishing the same grounds throughout the year (Scottish west coast fishermen, personal communication). Recent reports from fishermen suggest that this change in fishing practice is beginning to have a detrimental effect on their catches (TNMG, personal communication; Keltz & Robb, 2006). If this trend continues, there may be a real danger that this fishery, which has been found to be both ecologically and economically sustainable, may encounter serious problems in the future.

Fishermen in the Torridon area have successfully limited the effort of individual boats in their fishery. Unfortunately, the number of vessels has increased in this area (Chapter 1, Section 1.3.5), as the Torridon fishermen are not able to restrict access to their area. The responsible approach taken by fishermen from Torridon, as well as other similarly well managed fisheries (for example, Acheson, 1998; MSC, 2006) should be an example to other fishers the world over; however, in the case of the Torridon fishery voluntary management measures have not been complemented by much-needed legislation from government and at present there is a real danger of the benefits of the voluntary restrictions being nullified by the activities of other fishermen. This need for adequate regulation in open access *Nephrops* creel fisheries was highlighted by Eggert & Ulmestrand (2000).

This study has shown that *Nephrops* creels exploit the fished population in a very different way, and on distinct grounds, compared with the larger areas covered by trawls. Therefore, the current stock assessment practice, which has been largely developed for the trawl fishery, may be inappropriate for the *Nephrops* creel fishery (for example, the relative inefficiency of *Nephrops* creels may well lead to a fishery becoming economically unviable before the stock is biologically 'overfished'). I believe a very different approach to the future management of the *Nephrops* creel fishery needs to be taken, in which effort is set not purely by biological criteria but instead by bio-economic ones (Richardson & Gates, 1986; Annala & Sullivan, 1997; Eggert & Ulmestrand, 2000; Allen, 2004). Future management decisions must also be considered in the context of the creel fisheries' low environmental impact. This is particularly relevant with the current move towards an ecosystem-based management approach in Scottish fisheries (Scottish Executive, 2005). Creel fisheries could, therefore be used as a management tool to allow exploitation of *Nephrops* stocks without damaging sensitive habitats and critical fish habitat. Of utmost importance is the need for governments around the world to take a more proactive approach to fisheries management, instigating management measures on the basis of the precautionary principle rather than waiting until it is too late.

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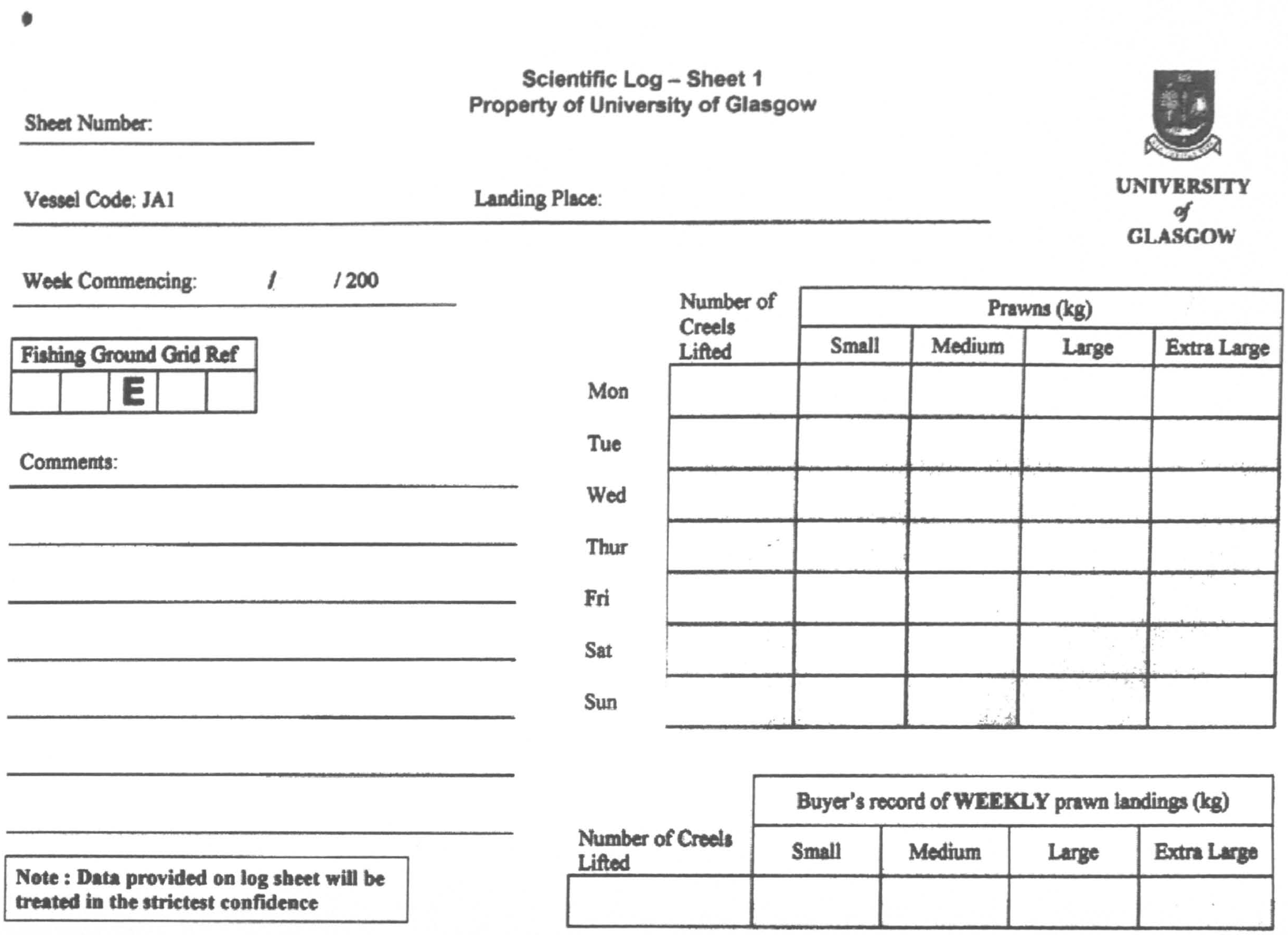
Appendices



Scientific Log – Sheet 1
Property of University of Glasgow

UNIVERSITY
of
GLASGOW

Sheet Number:

Vessel Code: JA1 **Landing Place:**

Week Commencing: / / 200

Fishing Ground Grid Ref				
		E		

Comments:

	Number of Creels Lifted	Prawns (kg)			
		Small	Medium	Large	Extra Large
Mon					
Tue					
Wed					
Thur					
Fri					
Sat					
Sun					

Number of Creels Lifted	Buyer's record of **WEEKLY** prawn landings (kg)			
	Small	Medium	Large	Extra Large

Note : Data provided on log sheet will be treated in the strictest confidence

A.1 – Weekly log sheet

Scientific Log – Sheet 2 Proportion of catch from each fleet
Property of the University of Glasgow

UNIVERSITY
of
GLASGOW

Date: / / 200

Vessel Code: JA1

Fleet Number	Location	Number of Creels	Prawns (kg)			
			Small	Medium	Large	Extra Large
1	N W					
2	N W					
3	N W					
4	N W					
5	N W					
6	N W					
7	N W					
8	N W					
9	N W					

A.2 – Daily log sheet

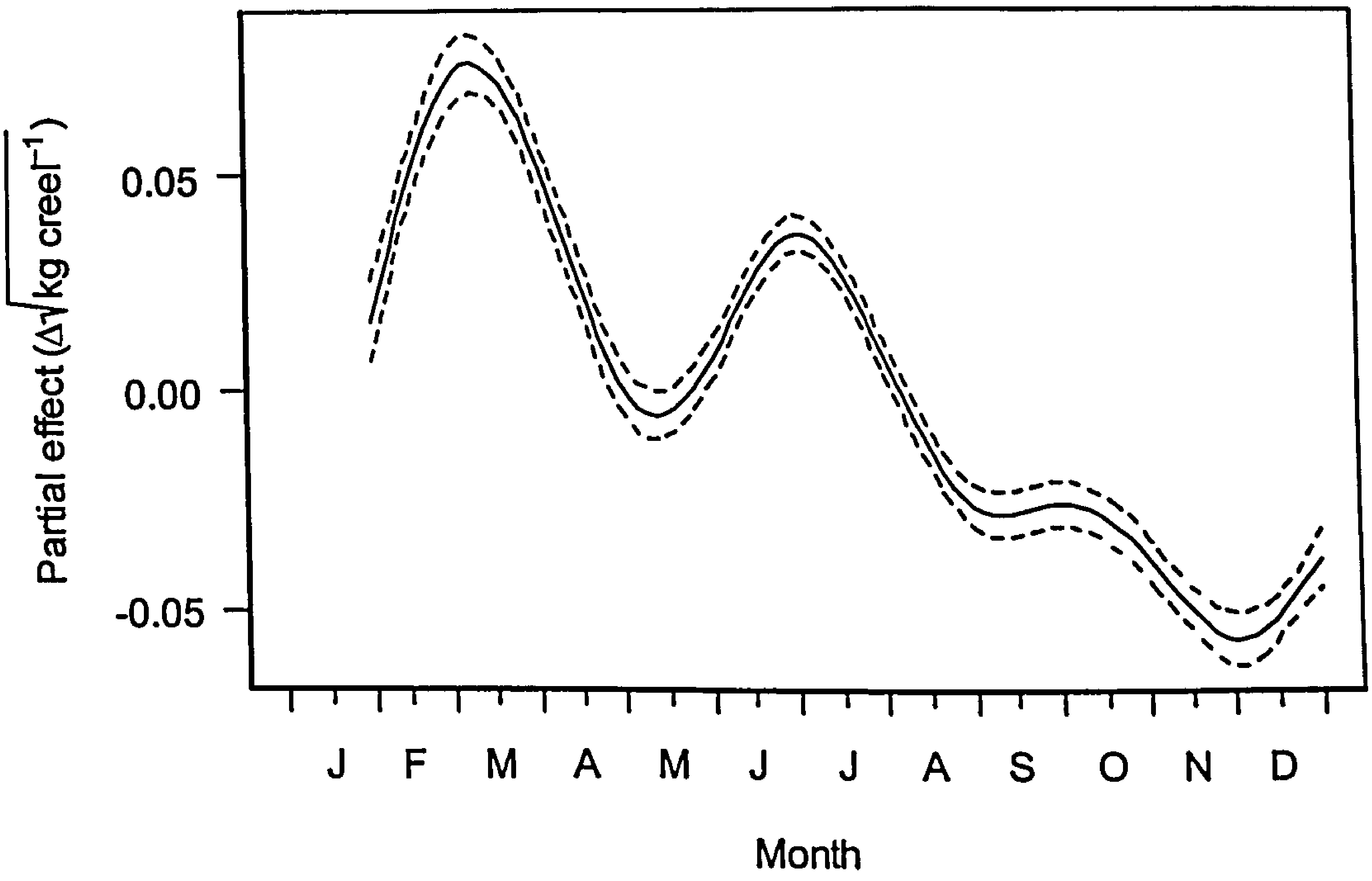


Figure B.1 - Seasonal changes in LPUE of all categories of *Nephrops* from the Torridon area. The solid curve is the partial effect of a smoothed month term in a generalized additive model (Positive or negative values indicate an increase or decrease in LPUE given the combined effect of the other variables, see text for details); the dashed lines indicate the 95% confidence interval of the month effect

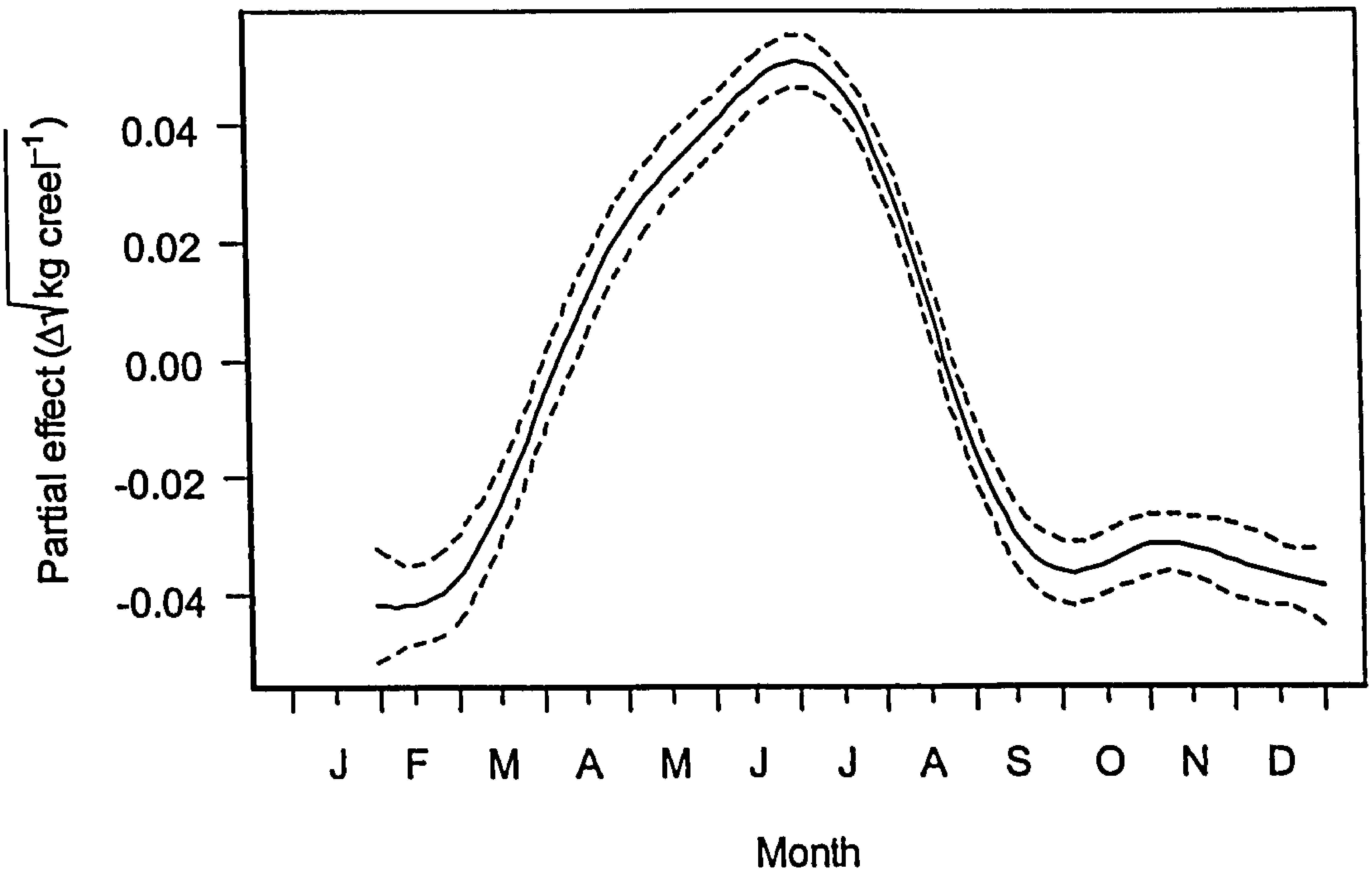


Figure B.2 - Seasonal changes in LPUE of small *Nephrops* (38–42 mm CL) from the Torridon area. The solid curve is the partial effect of a smoothed month term in a generalized additive model (Positive or negative values indicate an increase or decrease in LPUE given the combined effect of the other variables, see text for details); the dashed lines indicate the 95% confidence interval of the month effect.

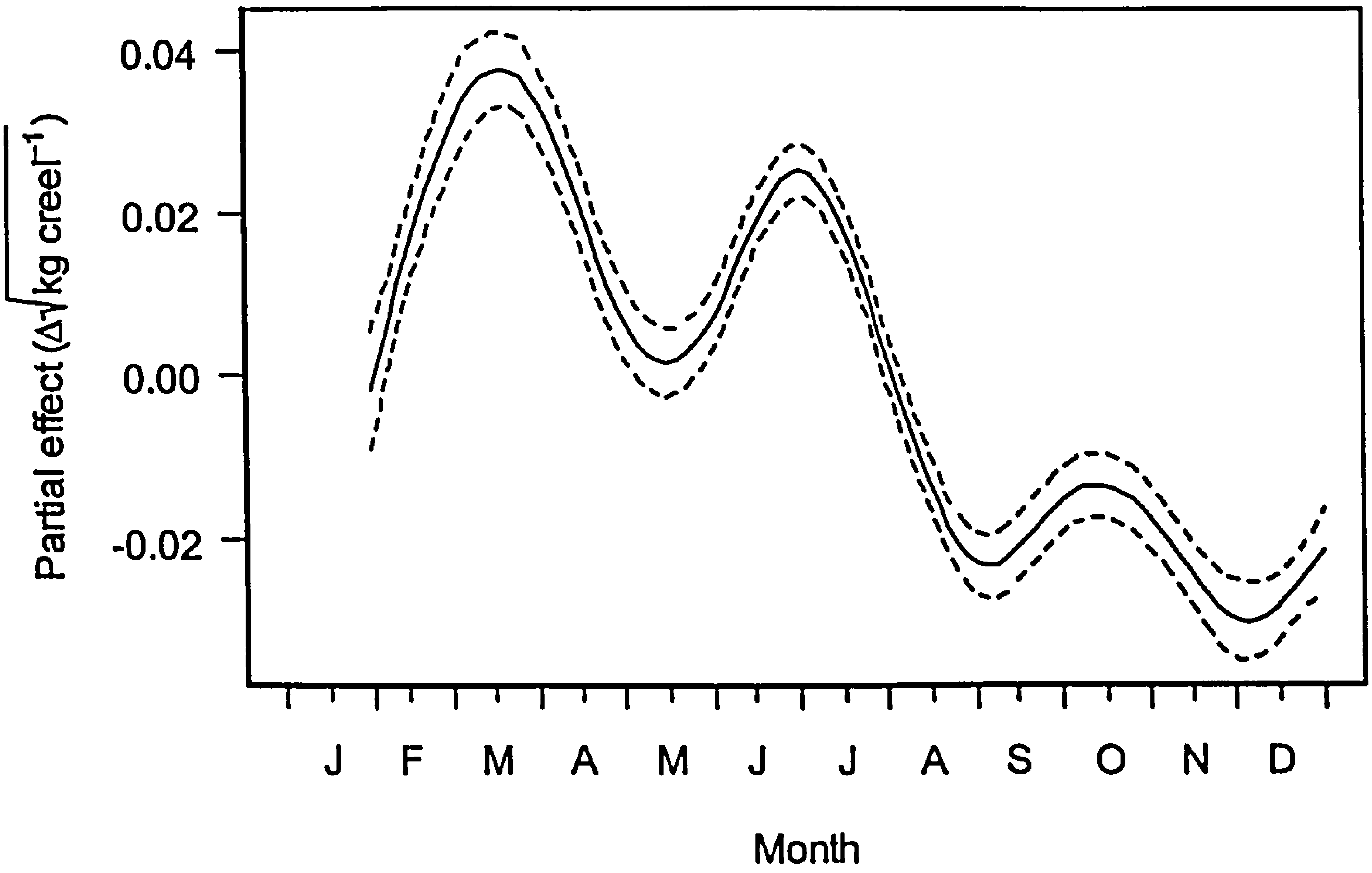


Figure B.3 - Seasonal changes in LPUE of medium-sized *Nephrops* (43–48 mm CL) from the Torridon area. The solid curve is the partial effect of a smoothed month term in a generalized additive model (Positive or negative values indicate an increase or decrease in LPUE given the combined effect of the other variables, see text for details); the dashed lines indicate the 95% confidence interval of the month effect.

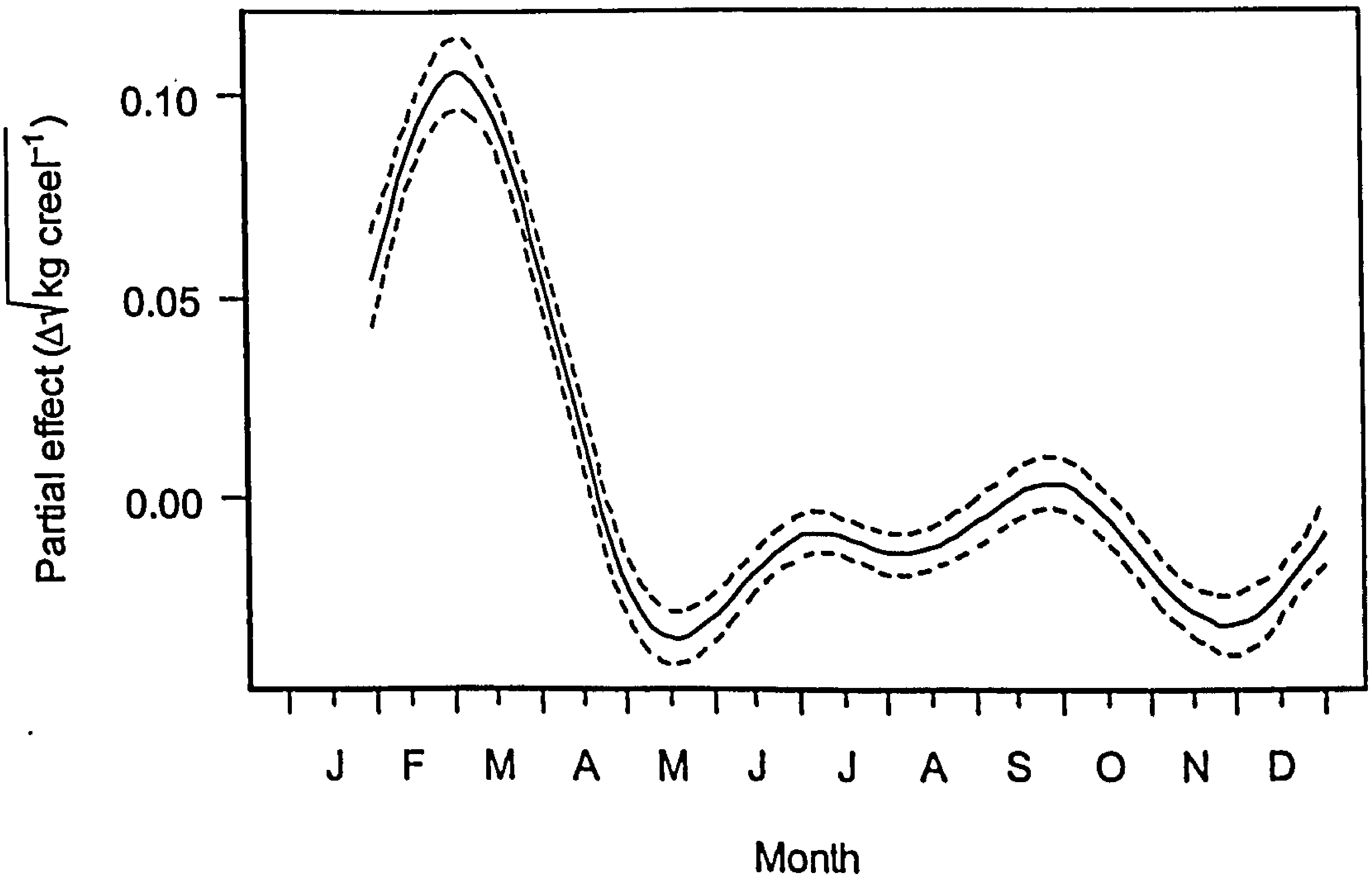


Figure B.4 - Seasonal changes in LPUE of large *Nephrops* (>48 mm CL) from the Torridon area. The solid curve is the partial effect of a smoothed month term in a generalized additive model (Positive or negative values indicate an increase or decrease in LPUE given the combined effect of the other variables, see text for details); the dashed lines indicate the 95% confidence interval of the month effect.

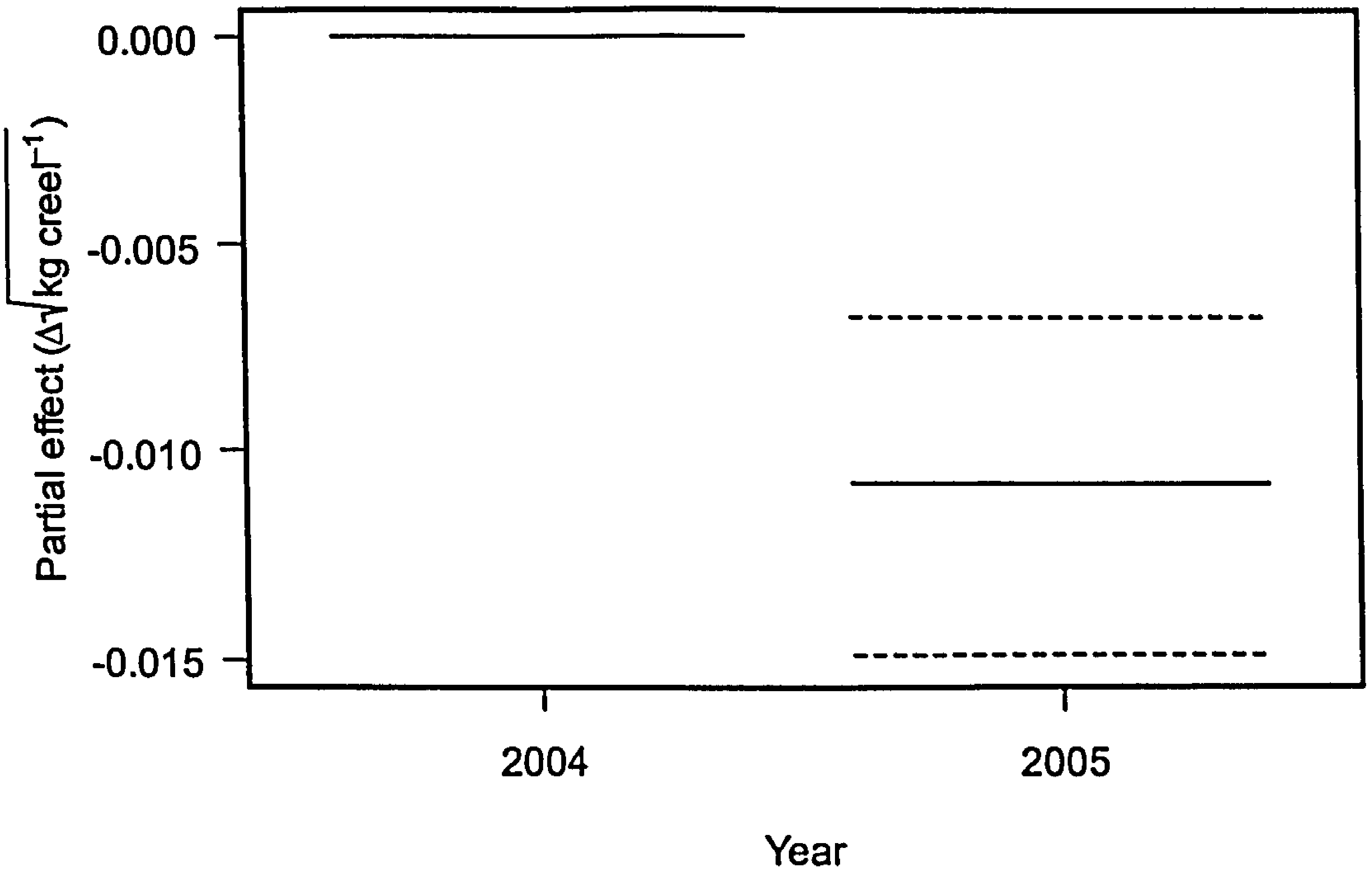


Figure B.5 - Annual change in LPUE of all categories of *Nephrops* from the Torridon area. The solid line is the partial effect of a smoothed month term in a generalized additive model (Positive or negative values indicate an increase or decrease in LPUE given the combined effect of the other variables, see text for details); the dashed lines indicate the 95% confidence interval of the month effect.

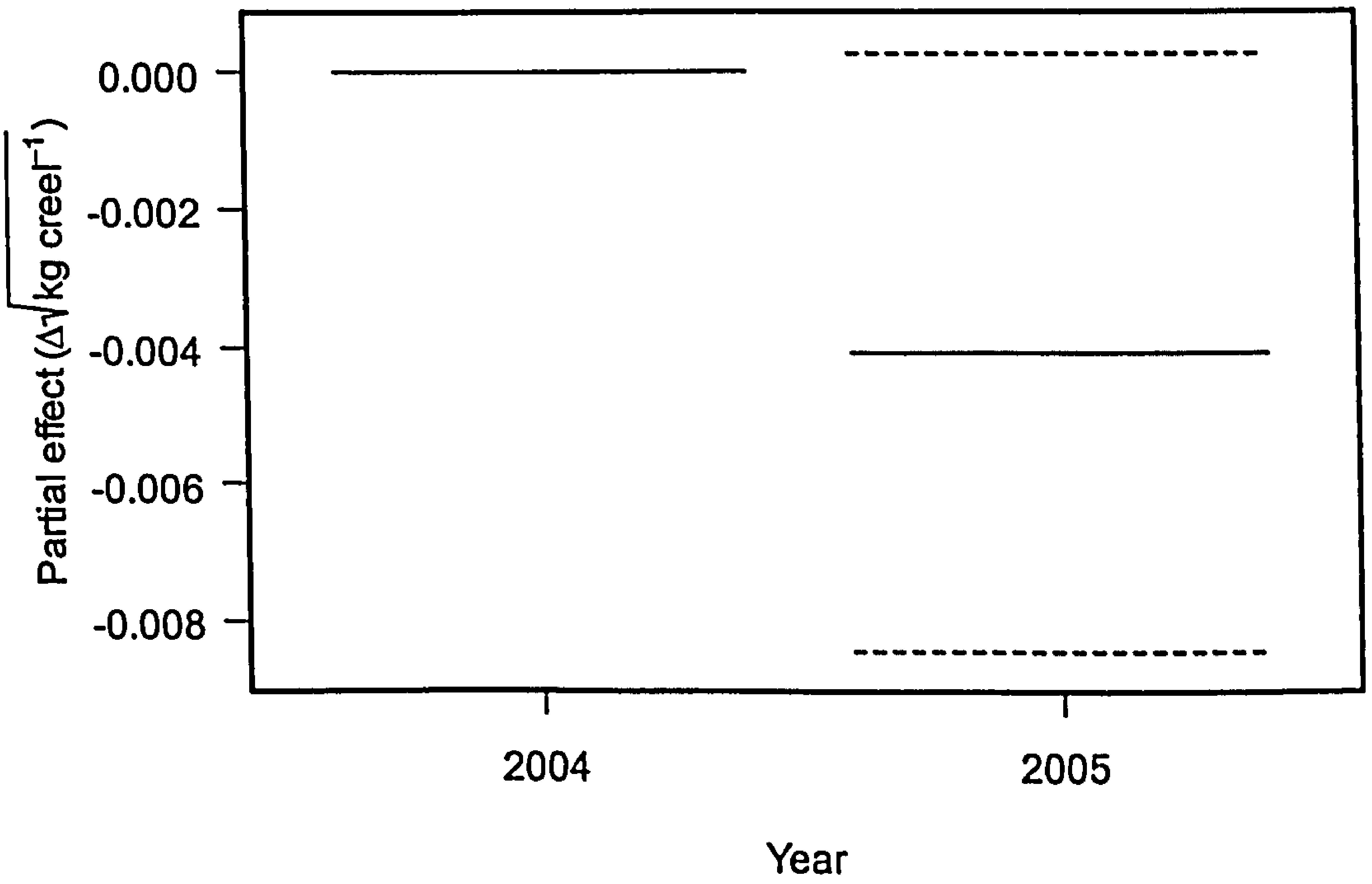


Figure B.6 - Annual change in LPUE of small *Nephrops* (38–42 mm CL) from the Torridon area. The solid line is the partial effect of a smoothed month term in a generalized additive model (Positive or negative values indicate an increase or decrease in LPUE given the combined effect of the other variables, see text for details); the dashed lines indicate the 95% confidence interval of the month effect.

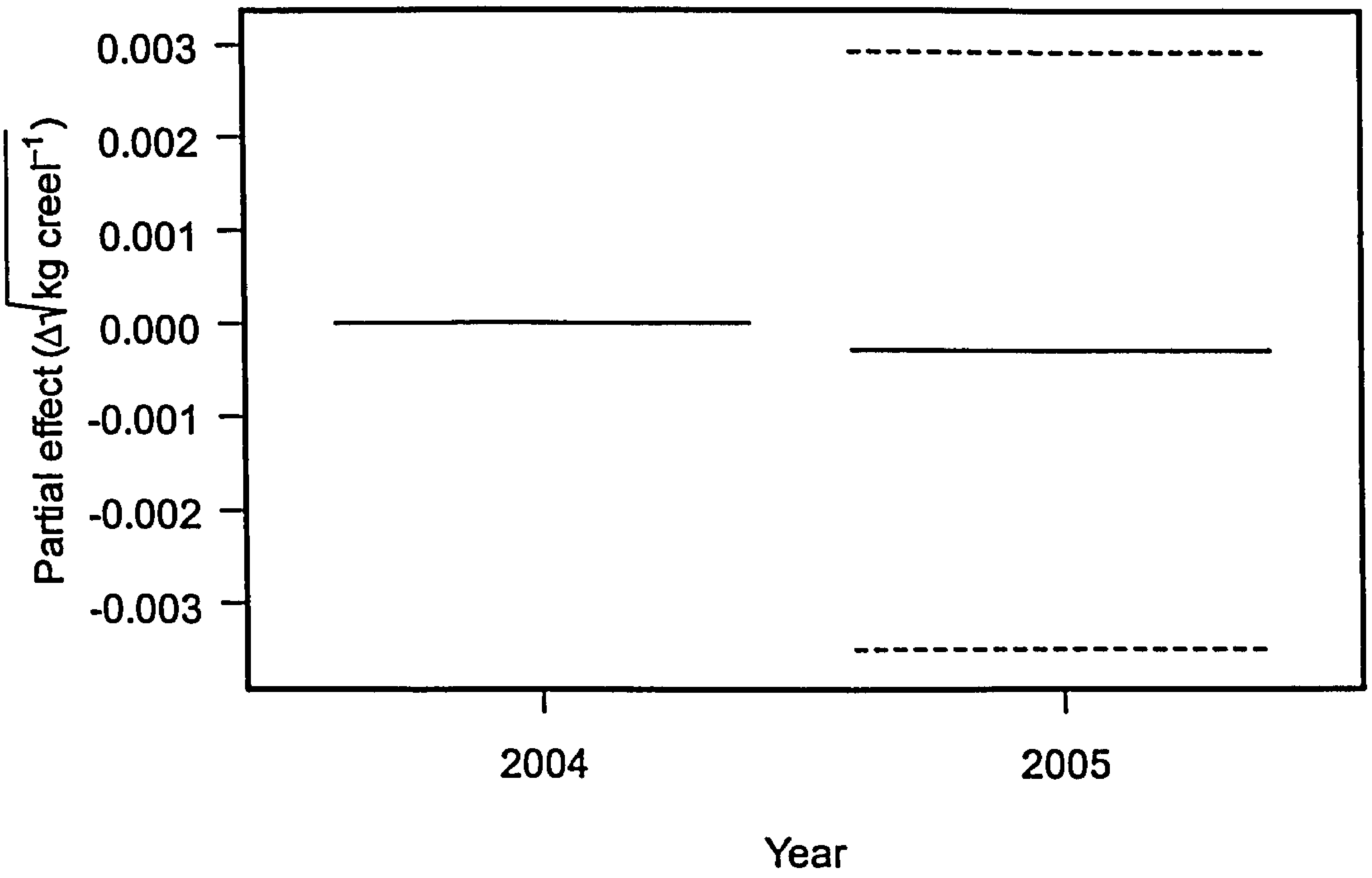


Figure B.7 - Annual change in LPUE of medium *Nephrops* (43–48 mm CL).from the Torridon area. The solid line is the partial effect of a smoothed month term in a generalized additive model (Positive or negative values indicate an increase or decrease in LPUE given the combined effect of the other variables, see text for details); the dashed lines indicate the 95% confidence interval of the month effect.

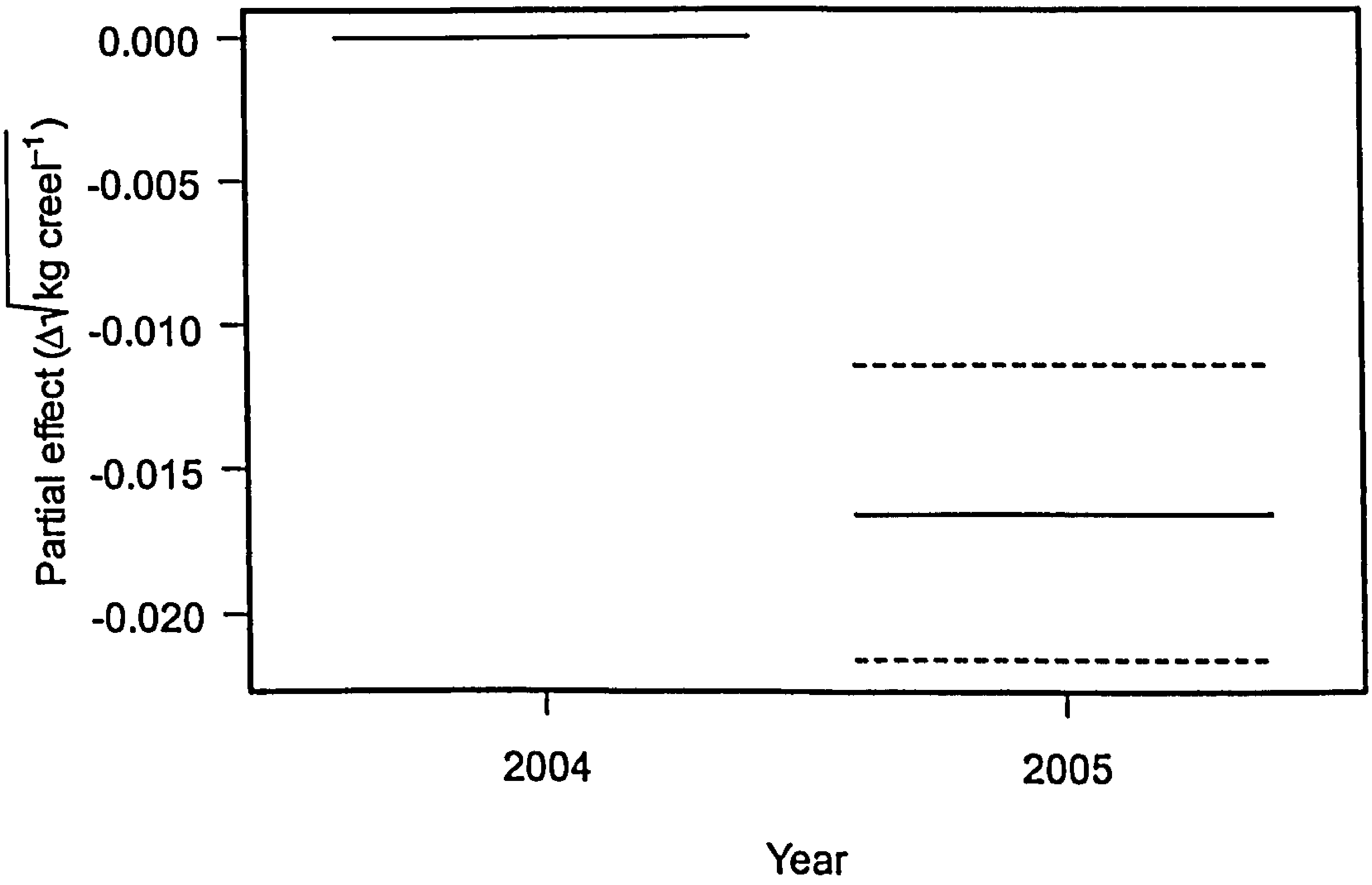


Figure B.8 - Annual change in LPUE of large *Nephrops* (>48 mm CL) from the Torridon area. The solid line is the partial effect of a smoothed month term in a generalized additive model (Positive or negative values indicate an increase or decrease in LPUE given the combined effect of the other variables, see text for details); the dashed lines indicate the 95% confidence interval of the month effect.

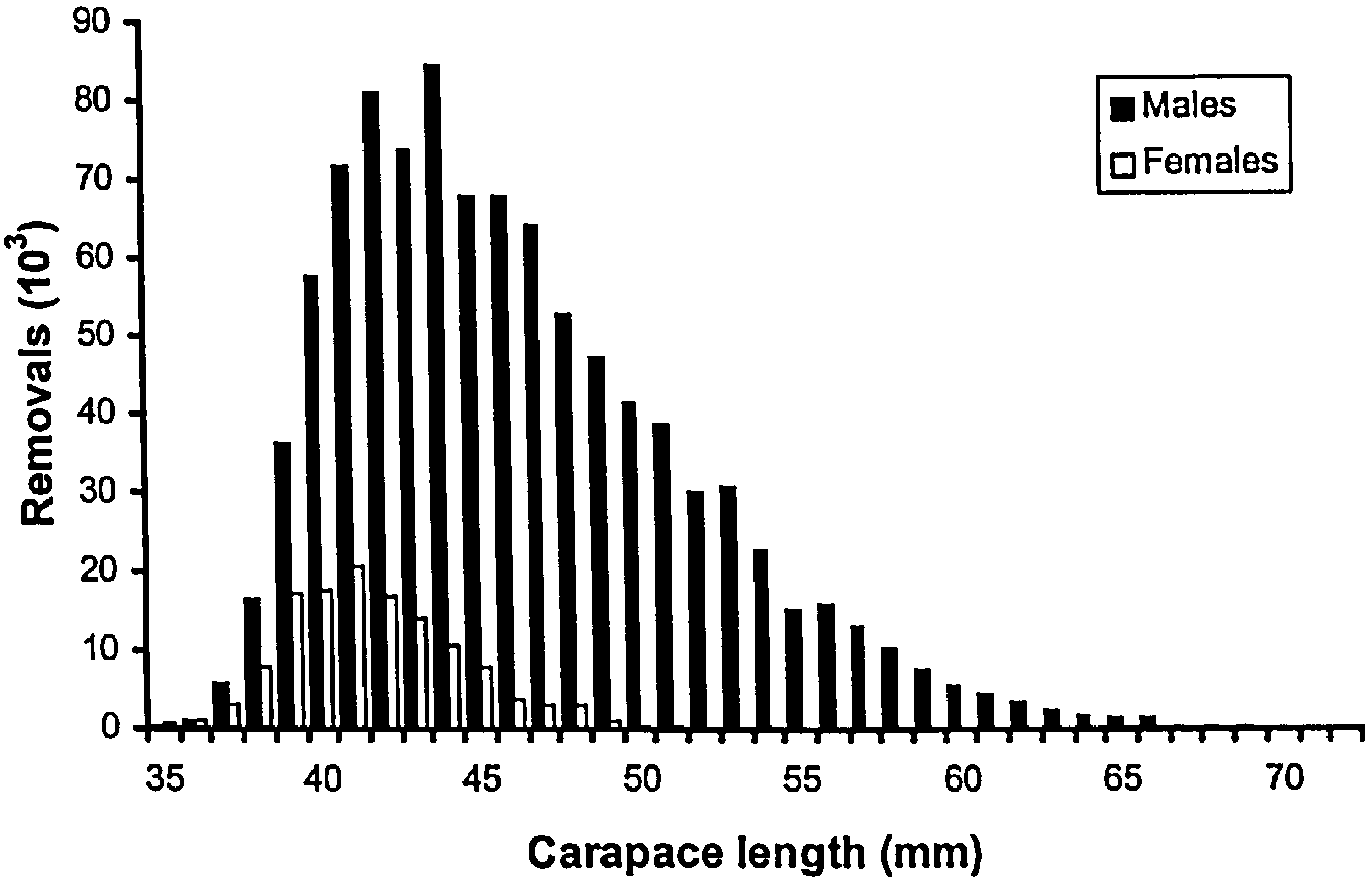


Figure B.9 - Length-frequency distribution of *Nephrops* removals (landings and moribund discards) from Torridon static gear zone in 2004 (12 boats), estimated from the annual average length frequency distribution derived from monthly catch sampling raised to the scale of total landings.

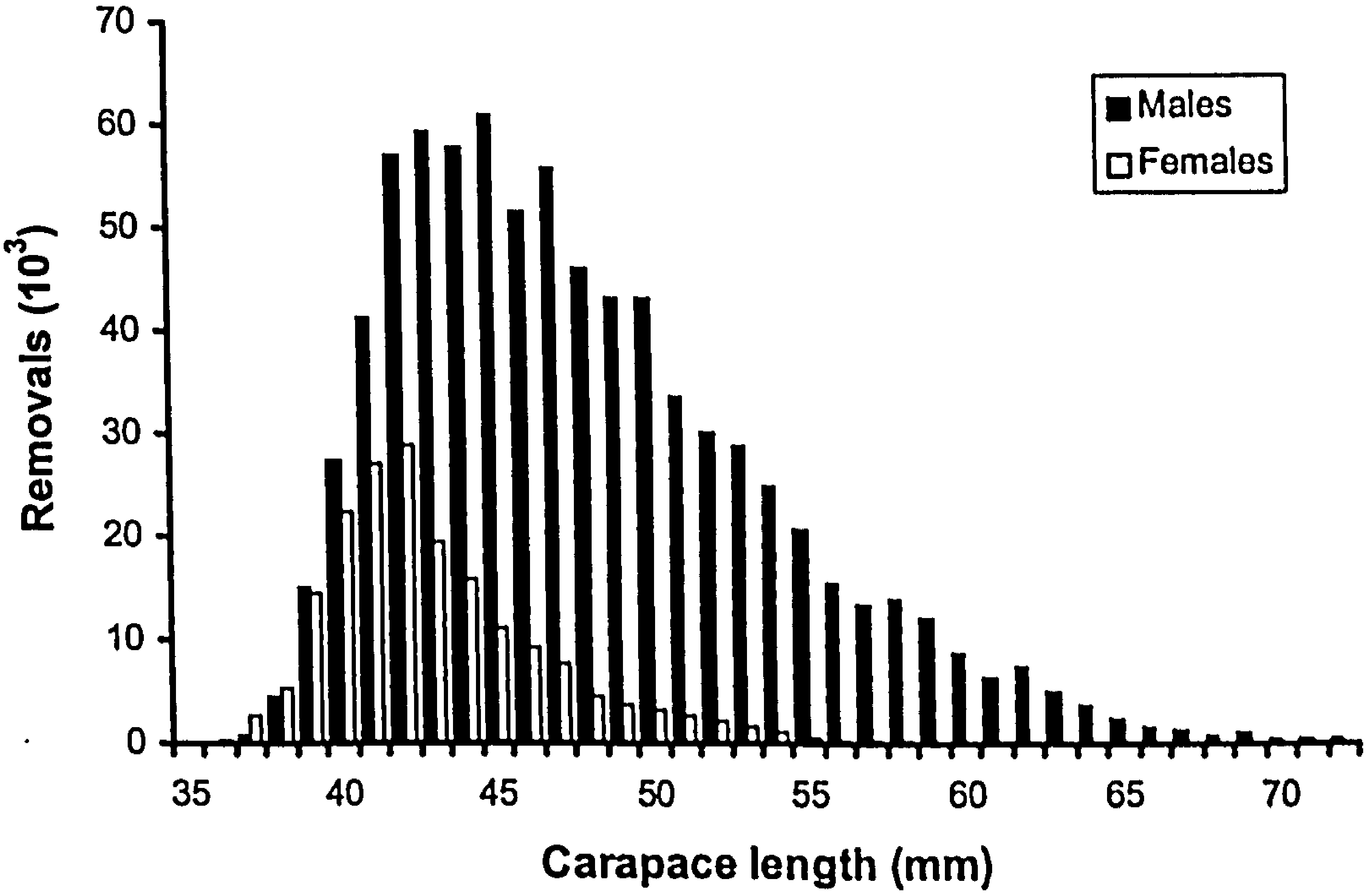


Figure B.10 - Length-frequency distribution of *Nephrops* removals (landings and moribund discards) from Torridon static gear zone in 2005 (12 boats), estimated from the annual average length frequency distribution derived from monthly catch sampling raised to the scale of total landings.

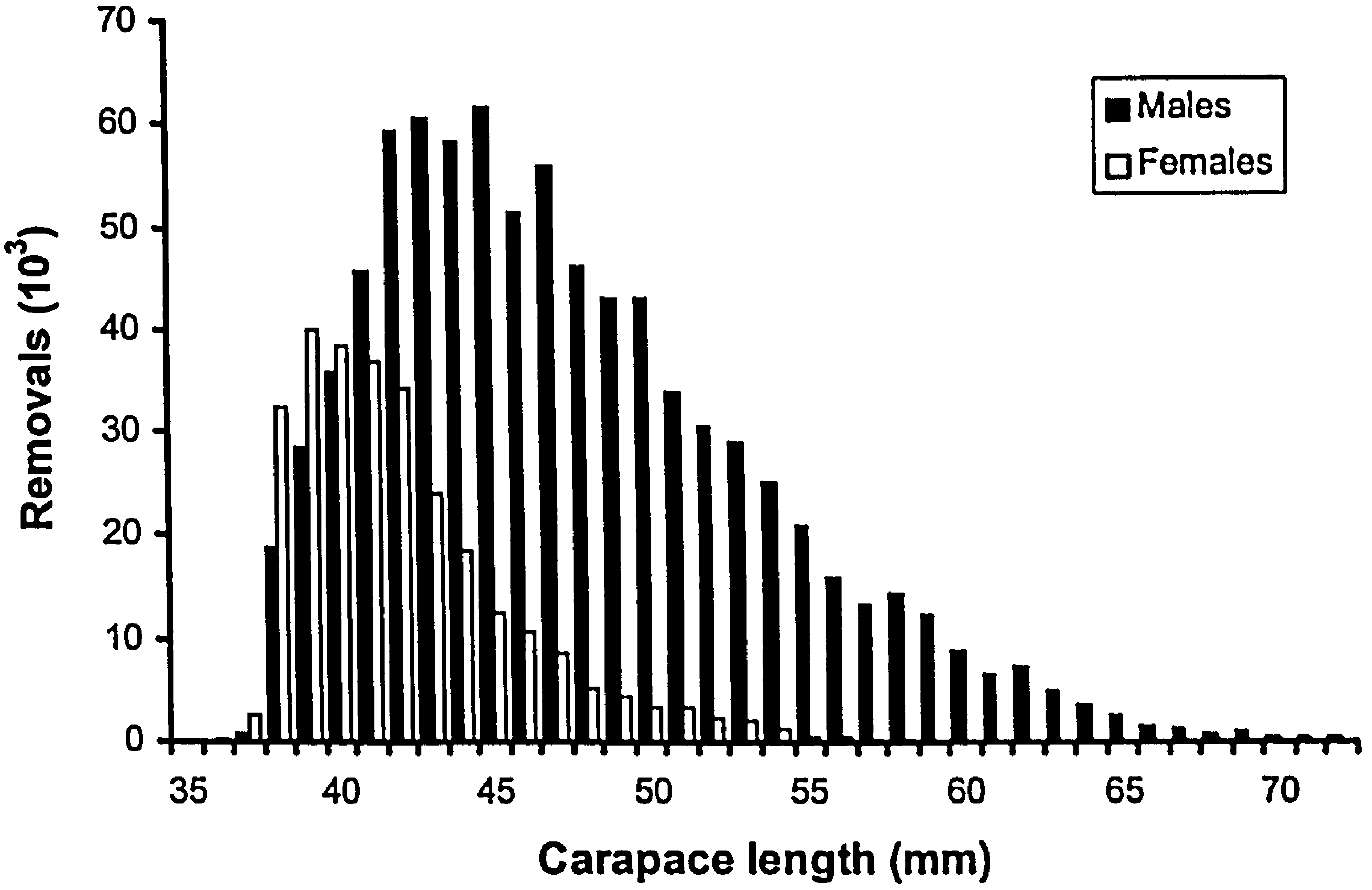


Figure B.11 – Length-frequency distribution of *Nephrops* removals (landings and moribund discards) from Torridon static gear zone in 2005 (12 boats), estimated from the annual average length frequency distribution derived from monthly catch sampling raised to the scale of total landings (discards >37 mm CL included in landings).

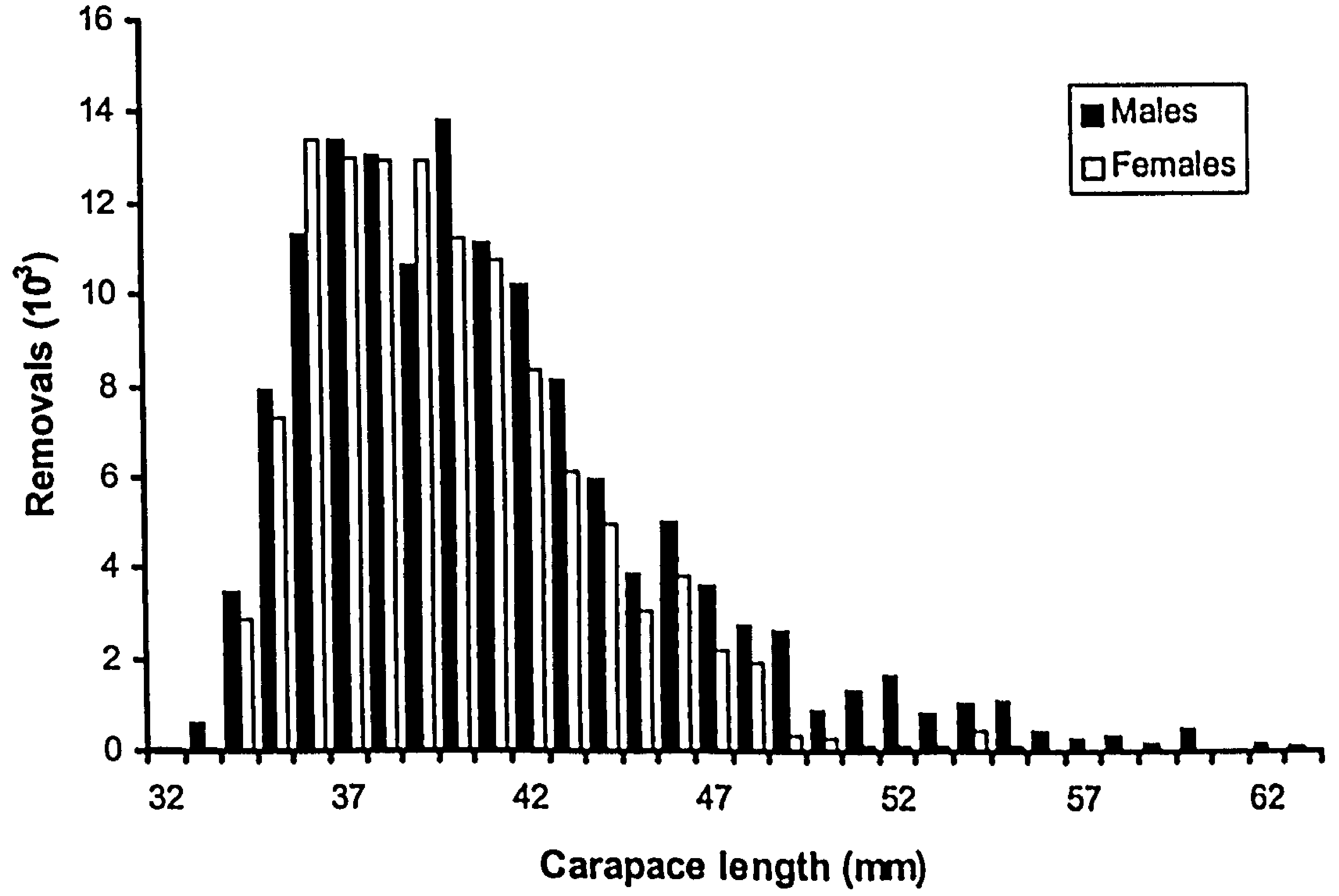


Figure B.12 - Length-frequency distribution of *Nephrops* removals (landings and moribund discards) from Loch Broom and the Summer Isles in 2005 (2 boats), estimated from the annual average length frequency distribution derived from monthly catch sampling raised to the scale of total landings.

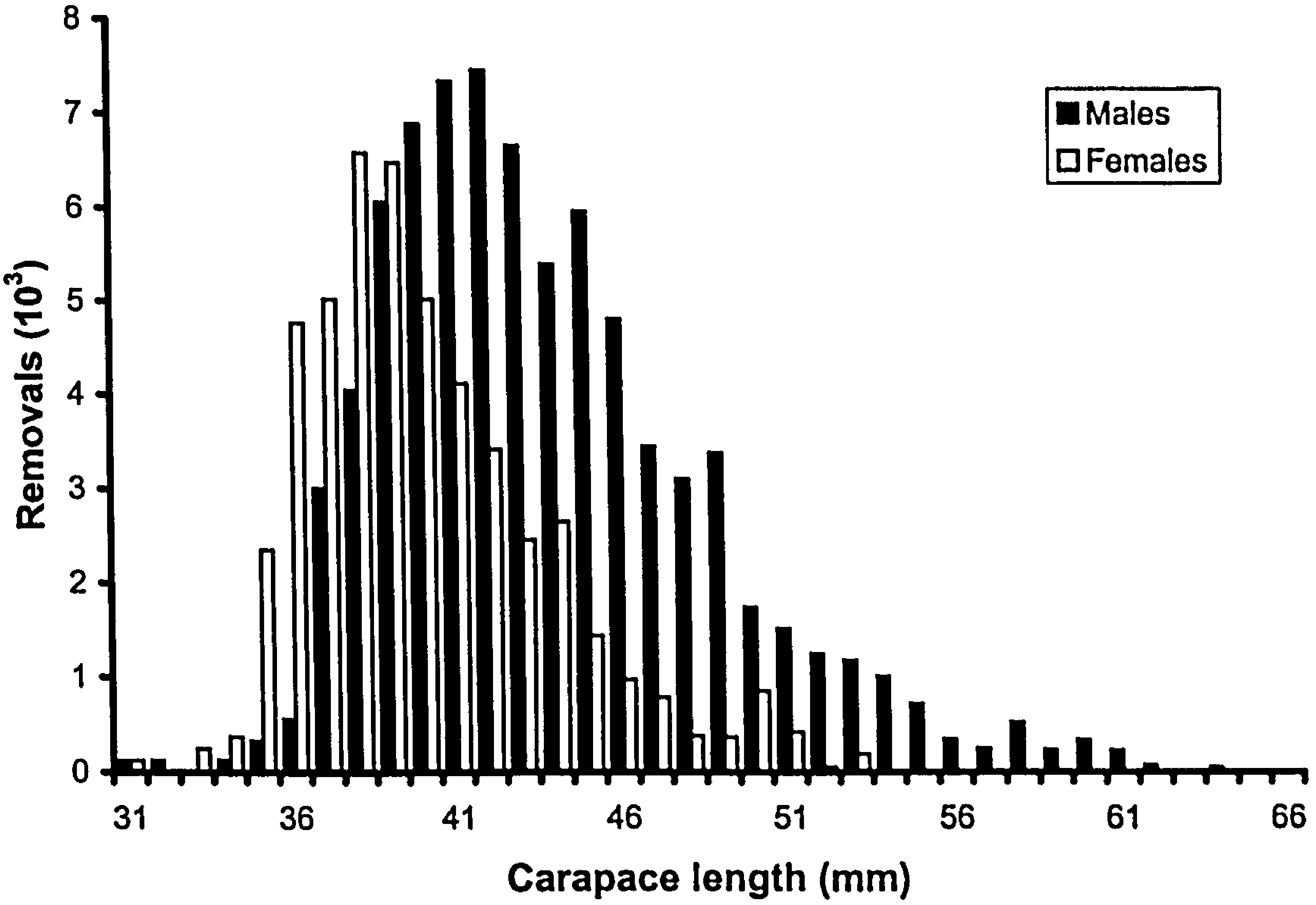


Figure B.13 - Length-frequency distribution of *Nephrops* removals (landings and moribund discards) from Loch Gairloch in 2004 (2 boats), estimated from the annual average length frequency distribution derived from monthly catch sampling raised to the scale of total landings.

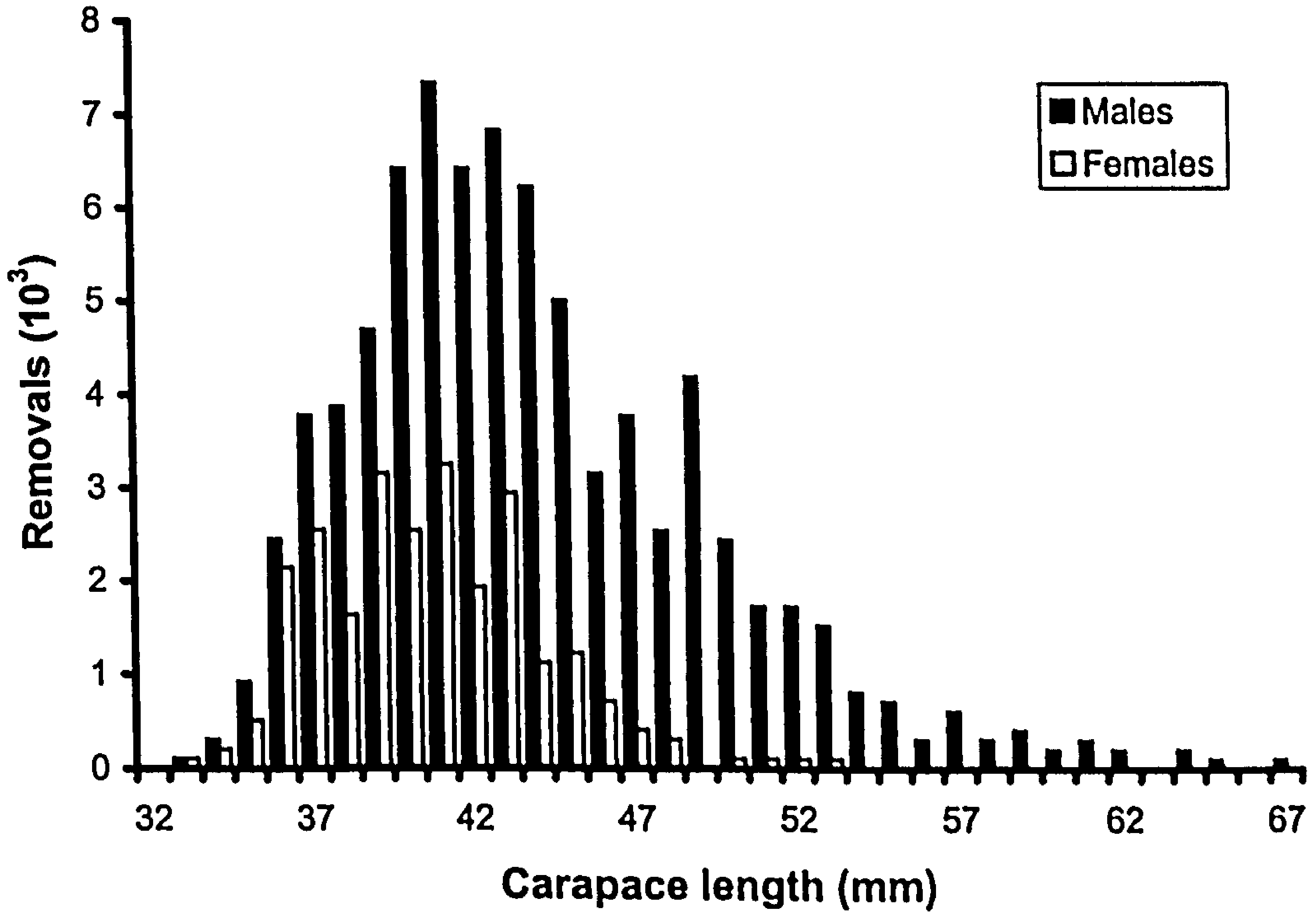


Figure B.14 - Length-frequency distribution of *Nephrops* removals (landings and moribund discards) from the Southern Inner Sound (Kyle) in 2005 (1 boat), estimated from the annual average length frequency distribution derived from monthly catch sampling raised to the scale of total landings.

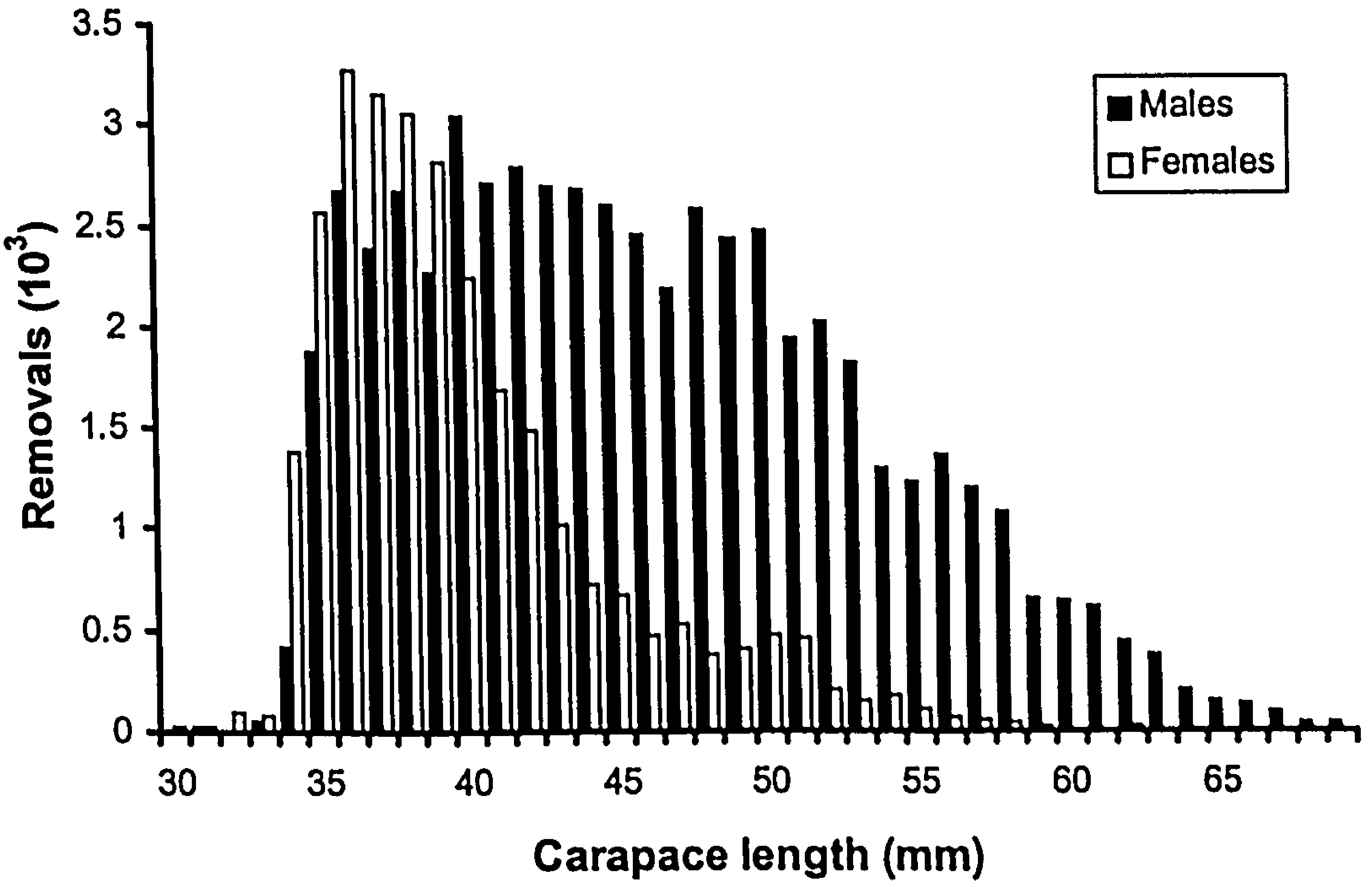


Figure B.15 - Length-frequency distribution of *Nephrops* removals (landings and moribund discards) from Loch Fyne in 2005 (1 boat), estimated from the annual average length frequency distribution derived from monthly catch sampling raised to the scale of total landings.

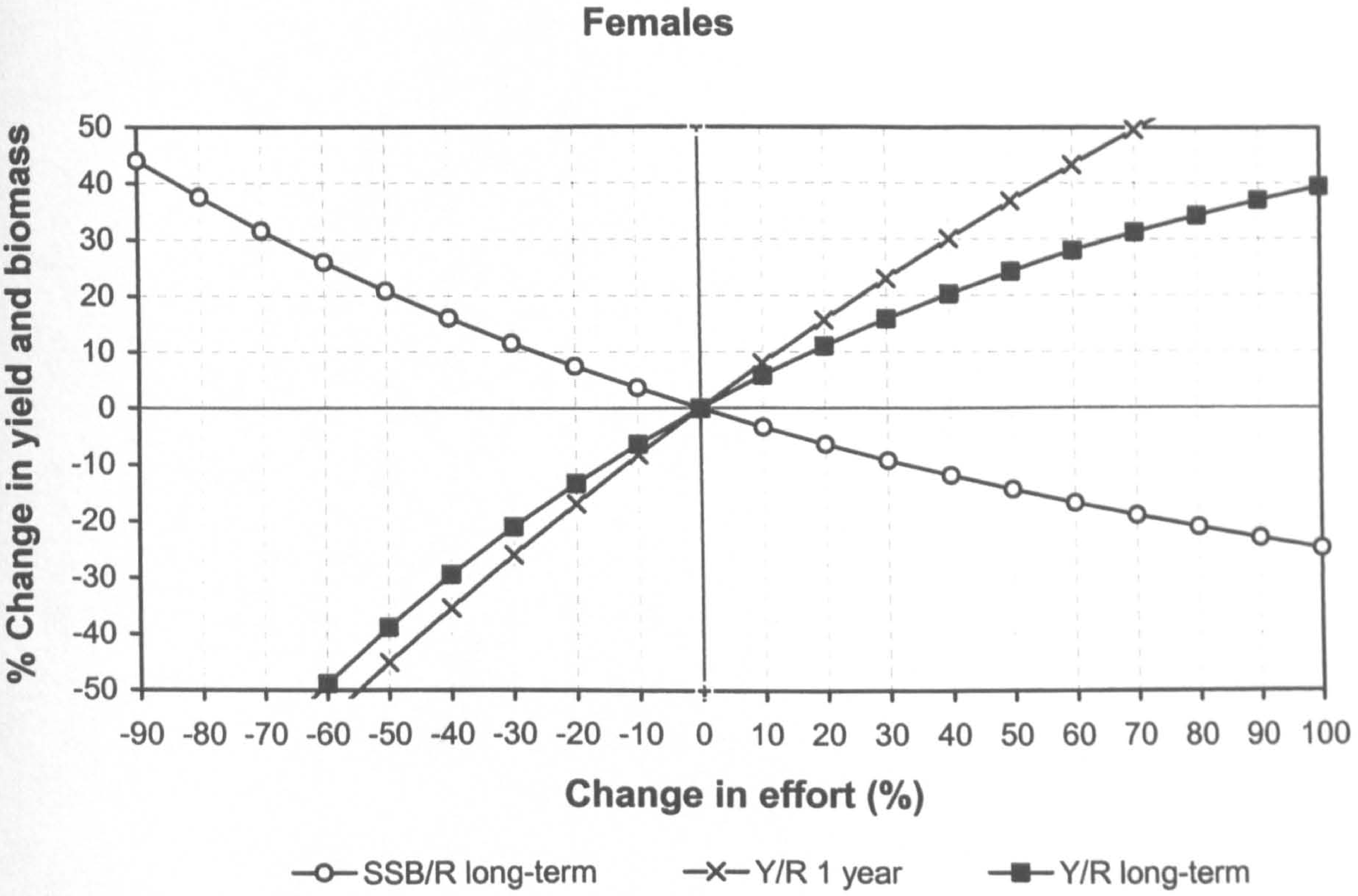


Figure B.16 - Predicted percentage changes in female long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in the Torridon creel fishery 2005, using ICES (1991) growth parameters.

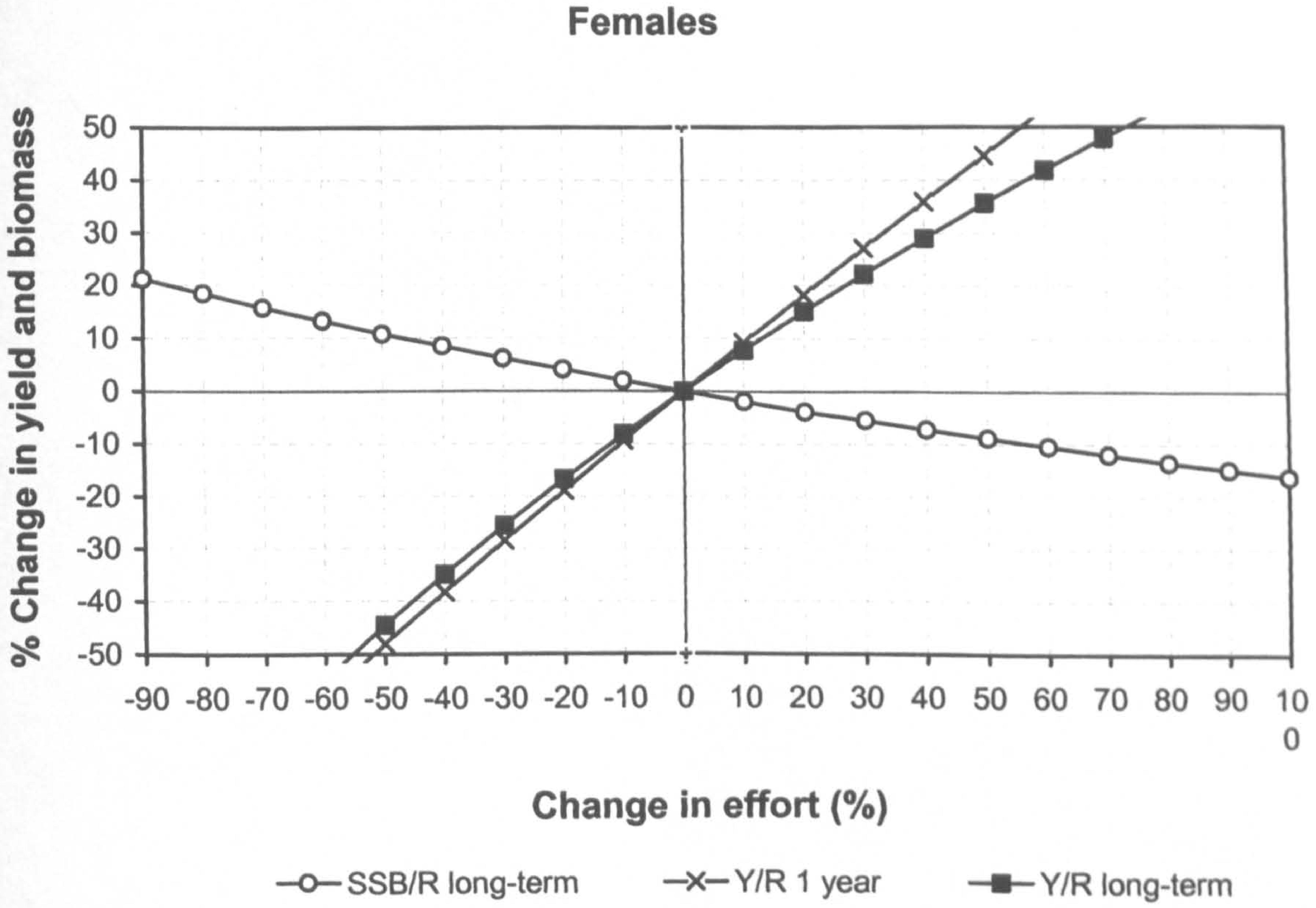


Figure B.17 - Predicted percentage changes in female long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in the Torridon creel fishery 2005, using ICES (1999) growth parameters.

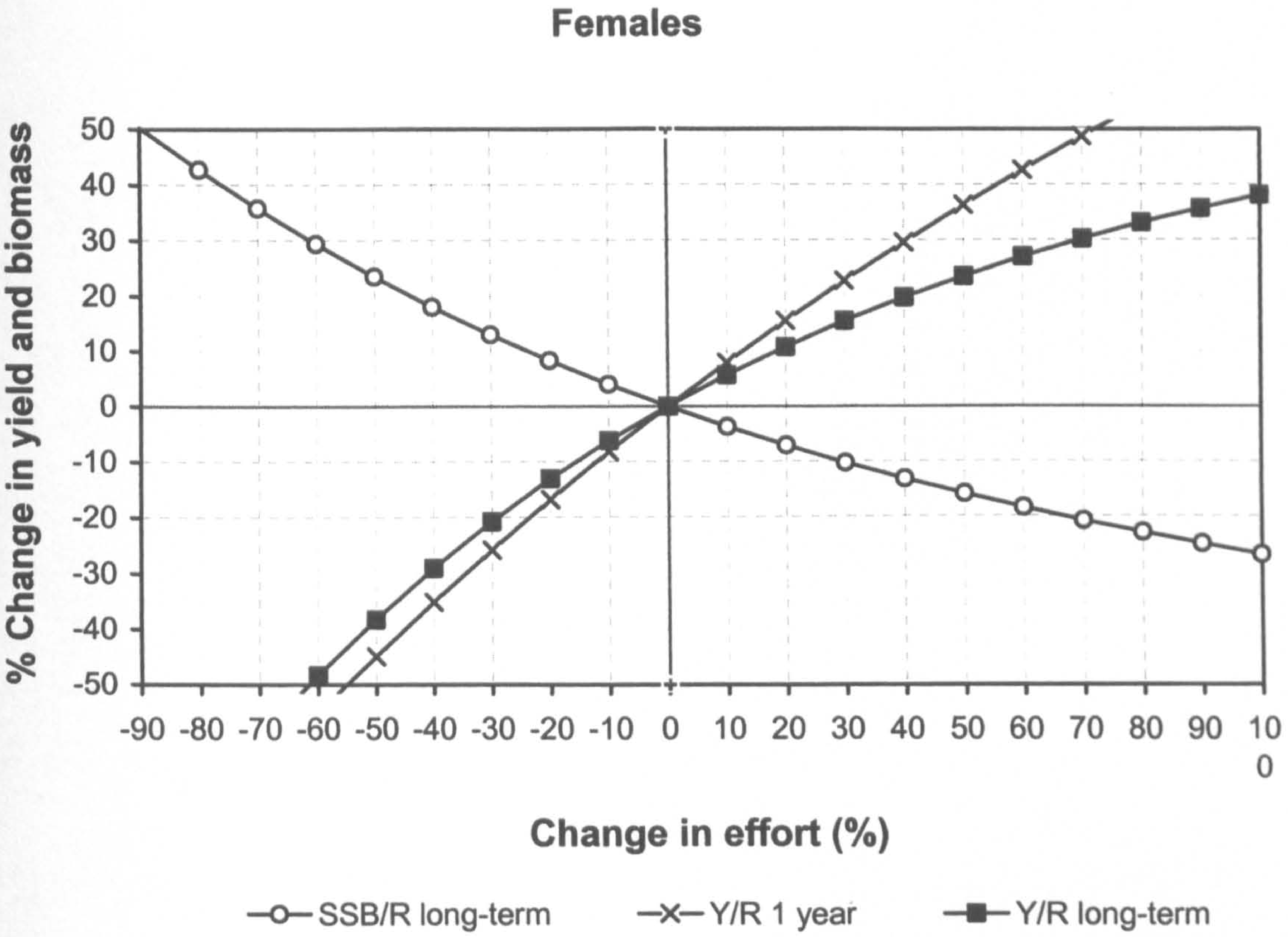


Figure B.18 - Predicted percentage changes in female long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in the Torridon creel fishery 2005, using ICES (1991) growth parameters (discards >37 mm CL included in landings).

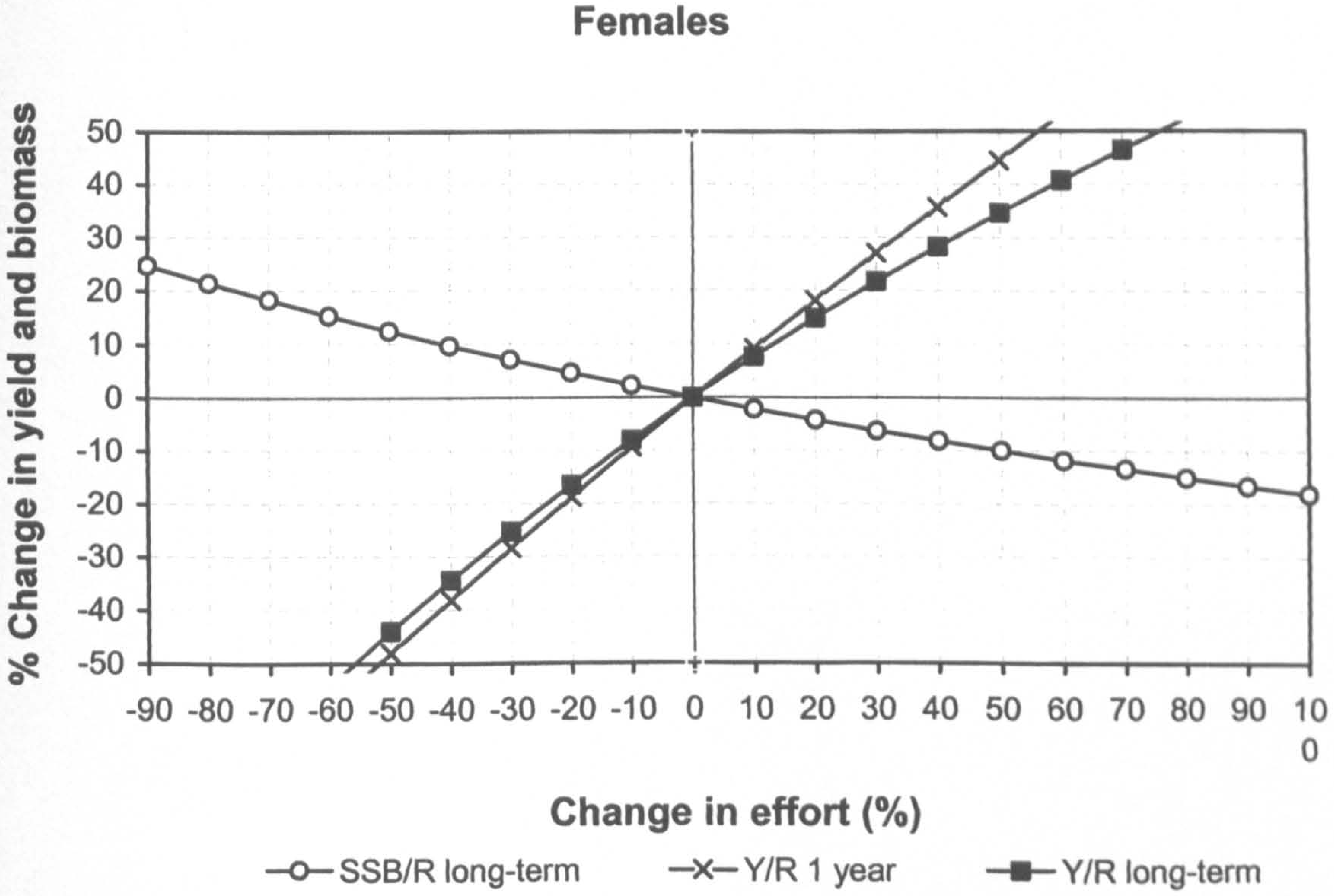


Figure B.19 - Predicted percentage changes in female long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in the Torridon creel fishery 2005, using ICES (1999) growth parameters (discards >37 mm CL included in landings).

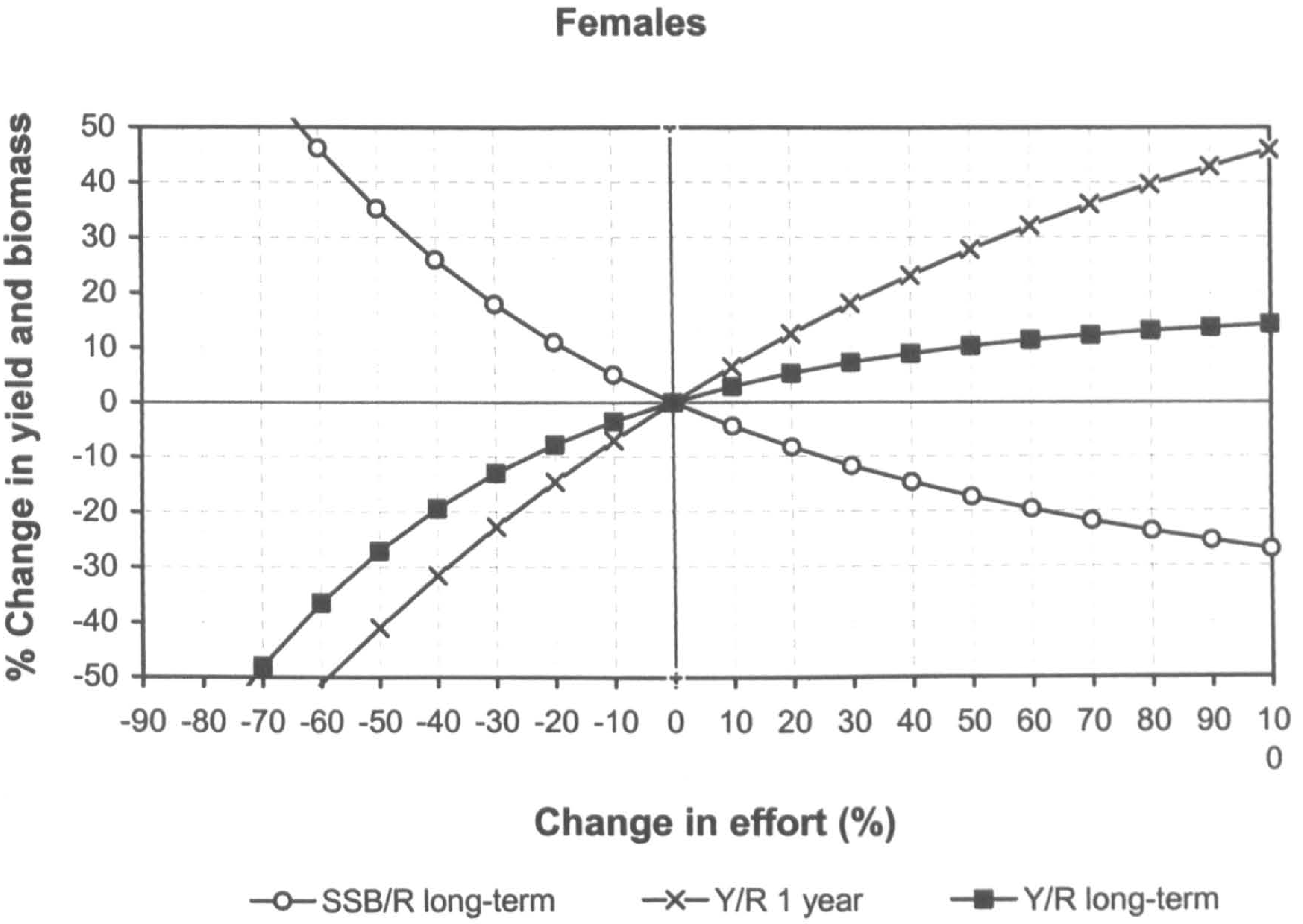


Figure B.20 - Predicted percentage changes in female long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in Loch Broom and the Summer Isles creel fishery in 2005, using ICES (1991) growth parameters.

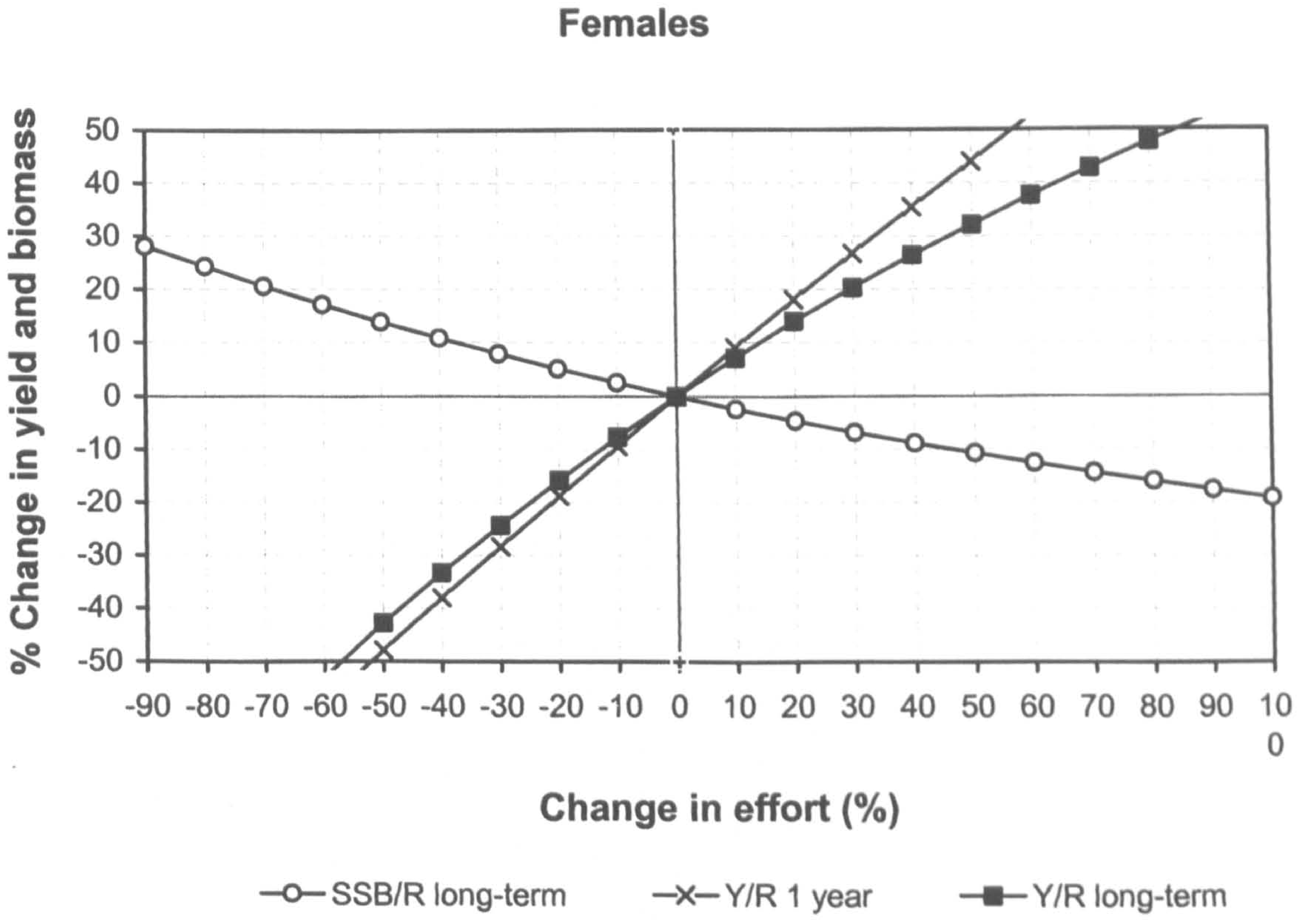


Figure B.21 - Predicted percentage changes in female long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in Loch Broom and the Summer Isles creel fishery in 2005, using ICES (1999) growth parameters.

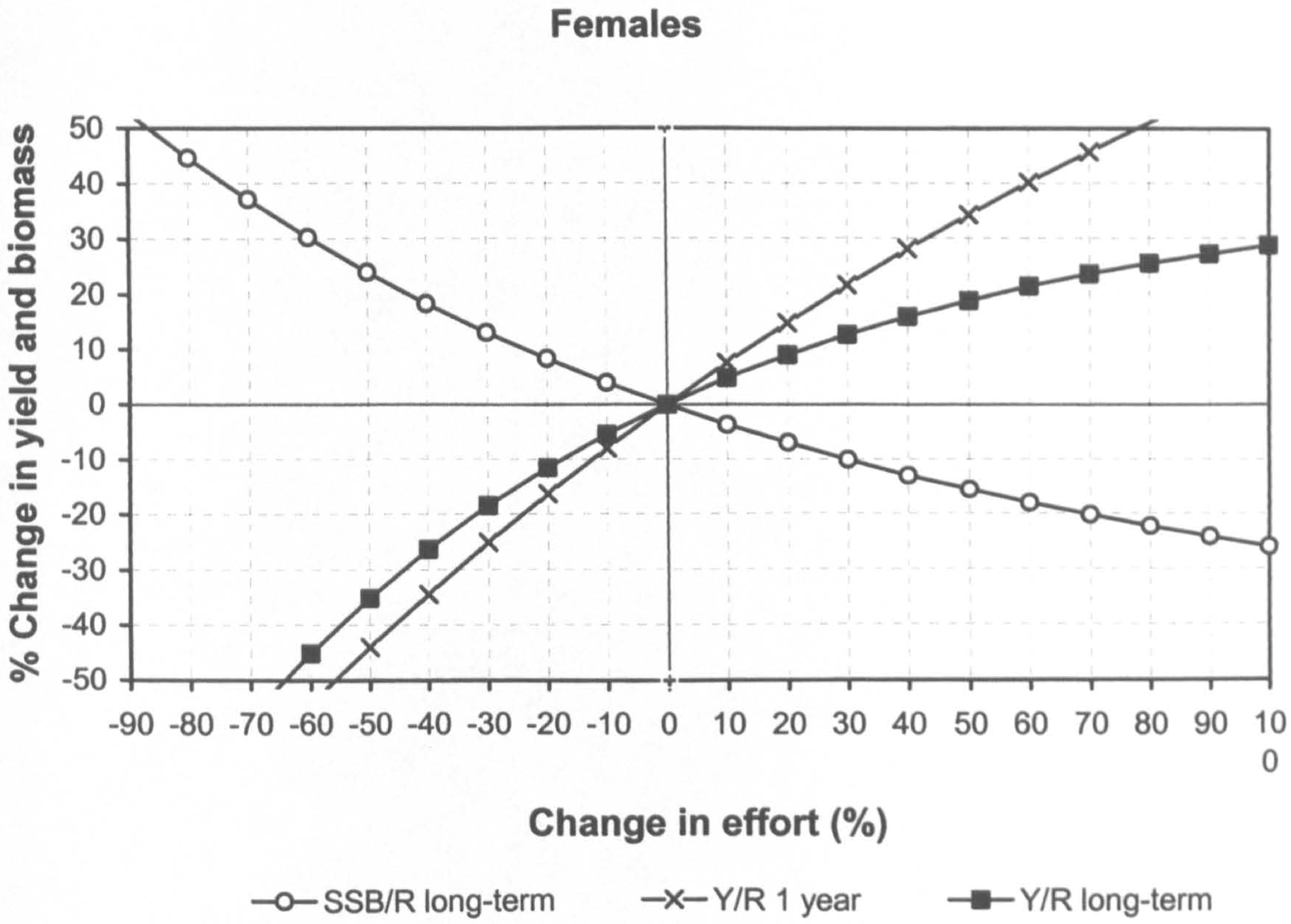


Figure B.22 - Predicted percentage changes in female long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in the Loch Gairloch creel fishery in 2004, using ICES (1991)

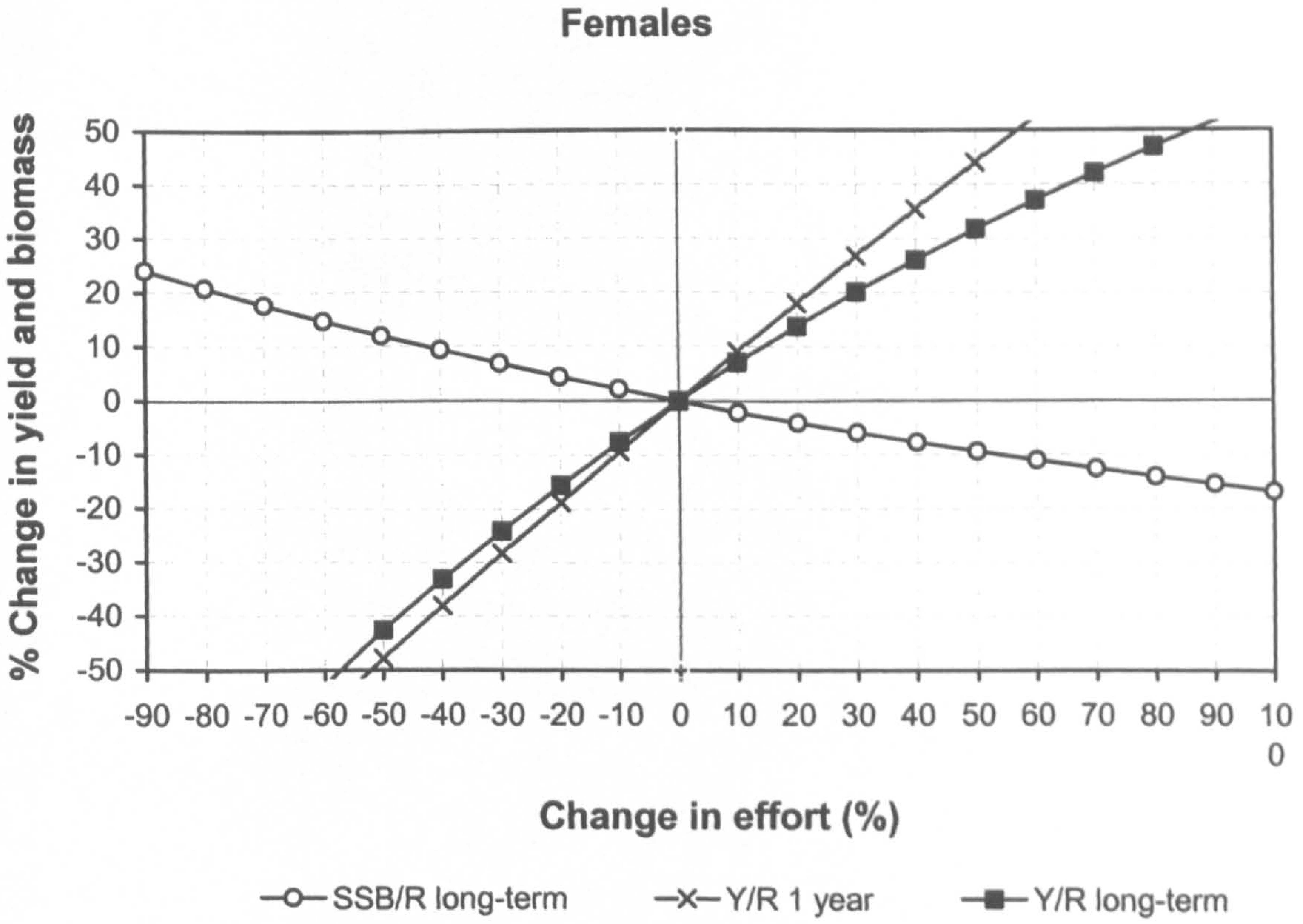


Figure B.23 - Predicted percentage changes in female long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in the Loch Gairloch creel fishery in 2004, using ICES (1999) growth parameters.

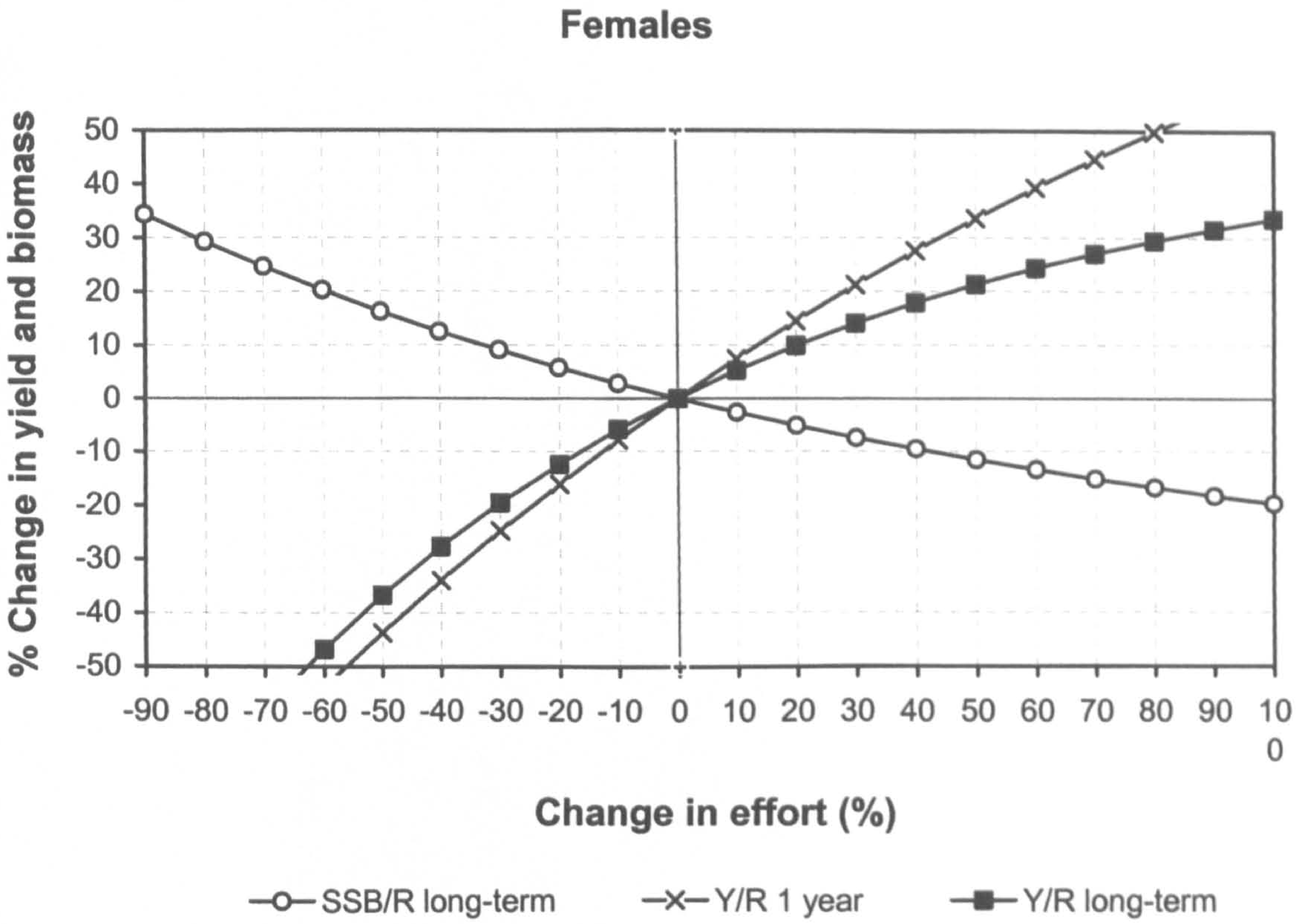


Figure B.24 - Predicted percentage changes in female long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in the Southern Inner Sound (Kyle) creel fishery in 2005, using ICES (1991) growth parameters.

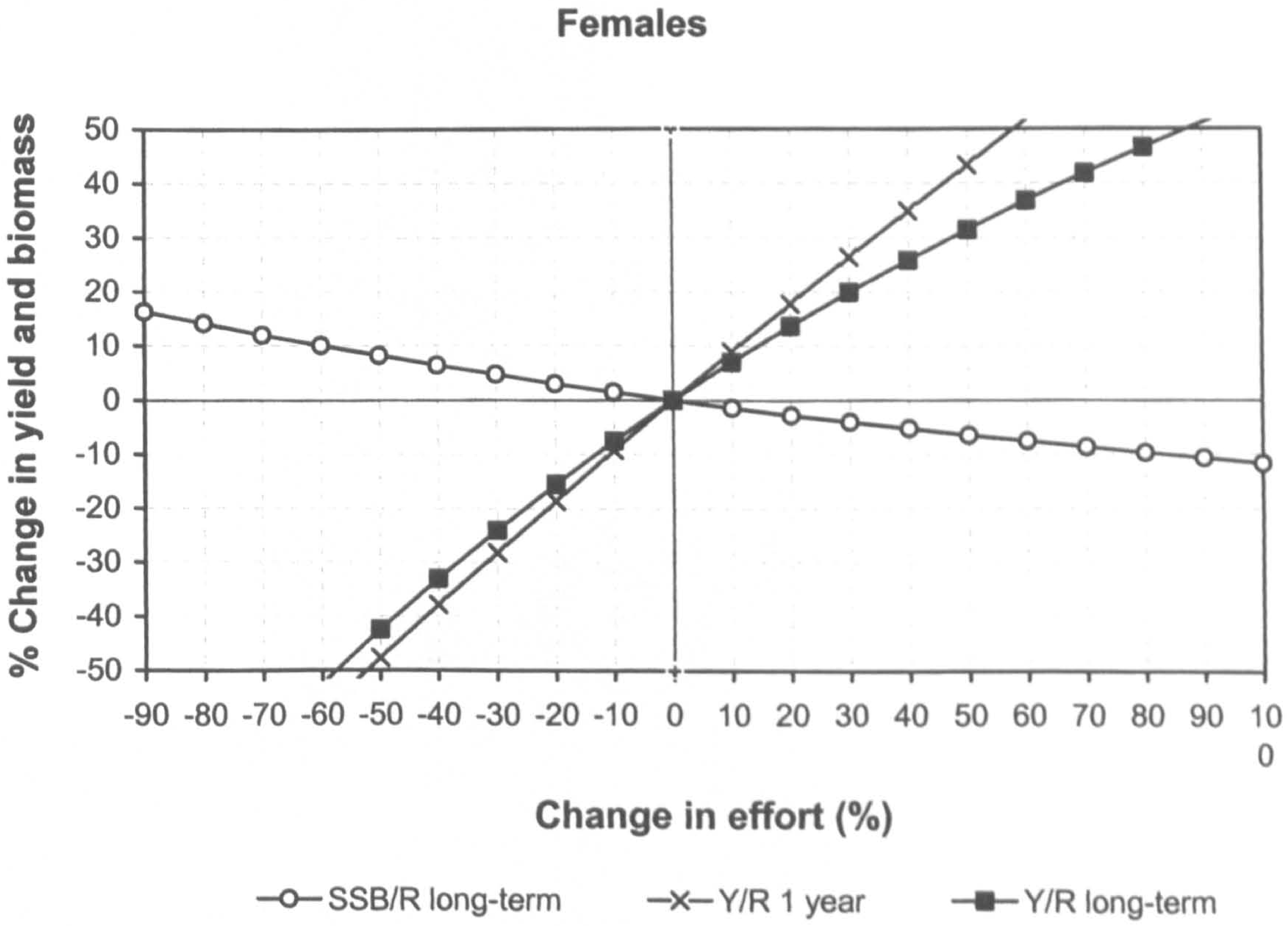


Figure B.25 - Predicted percentage changes in female long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in the Southern Inner Sound (Kyle) creel fishery in 2005, using ICES (1999) growth parameters

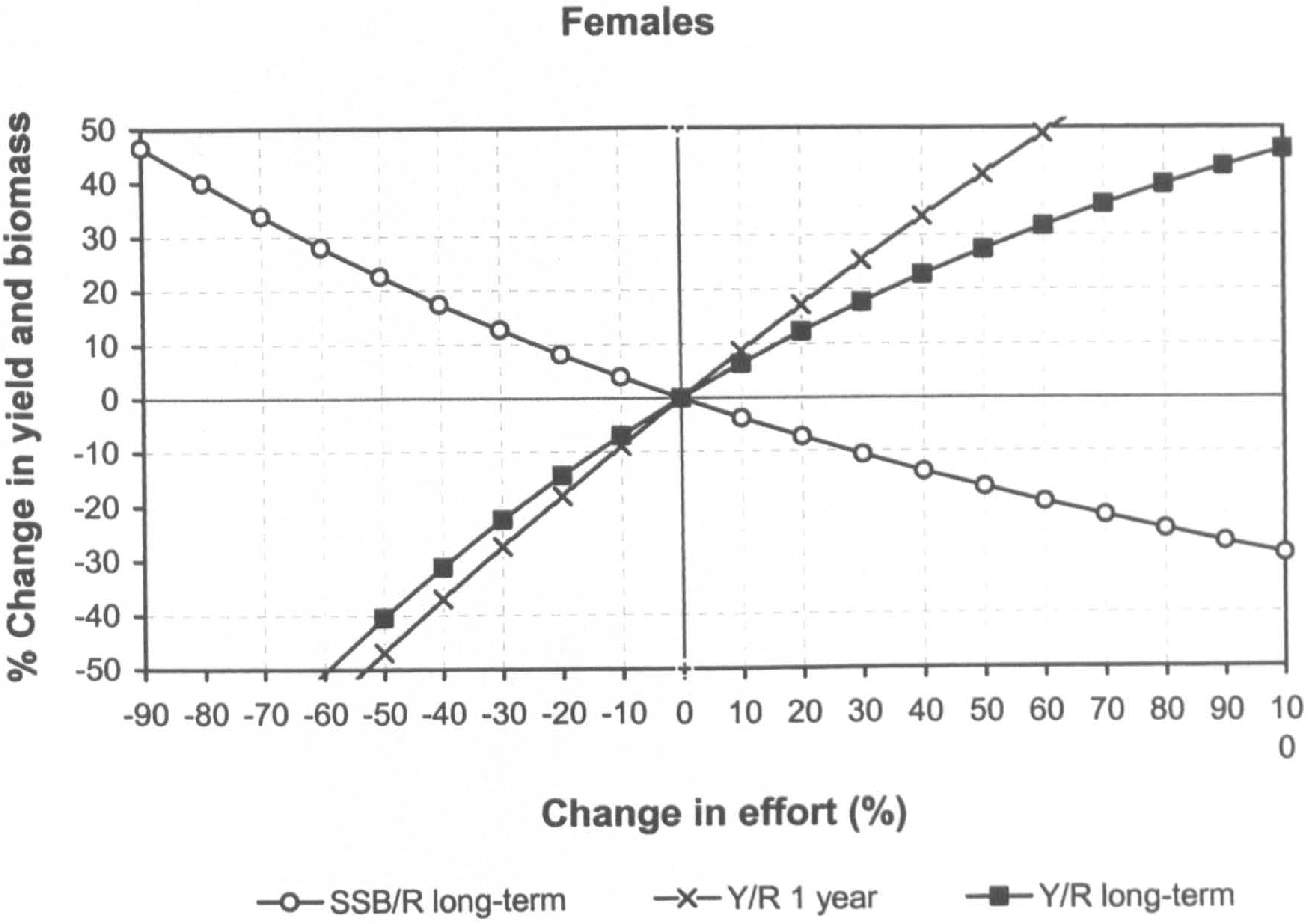


Figure B.26 - Predicted percentage changes in female long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in the Loch Fyne creel fishery in 2005, using ICES (1991) growth parameters.

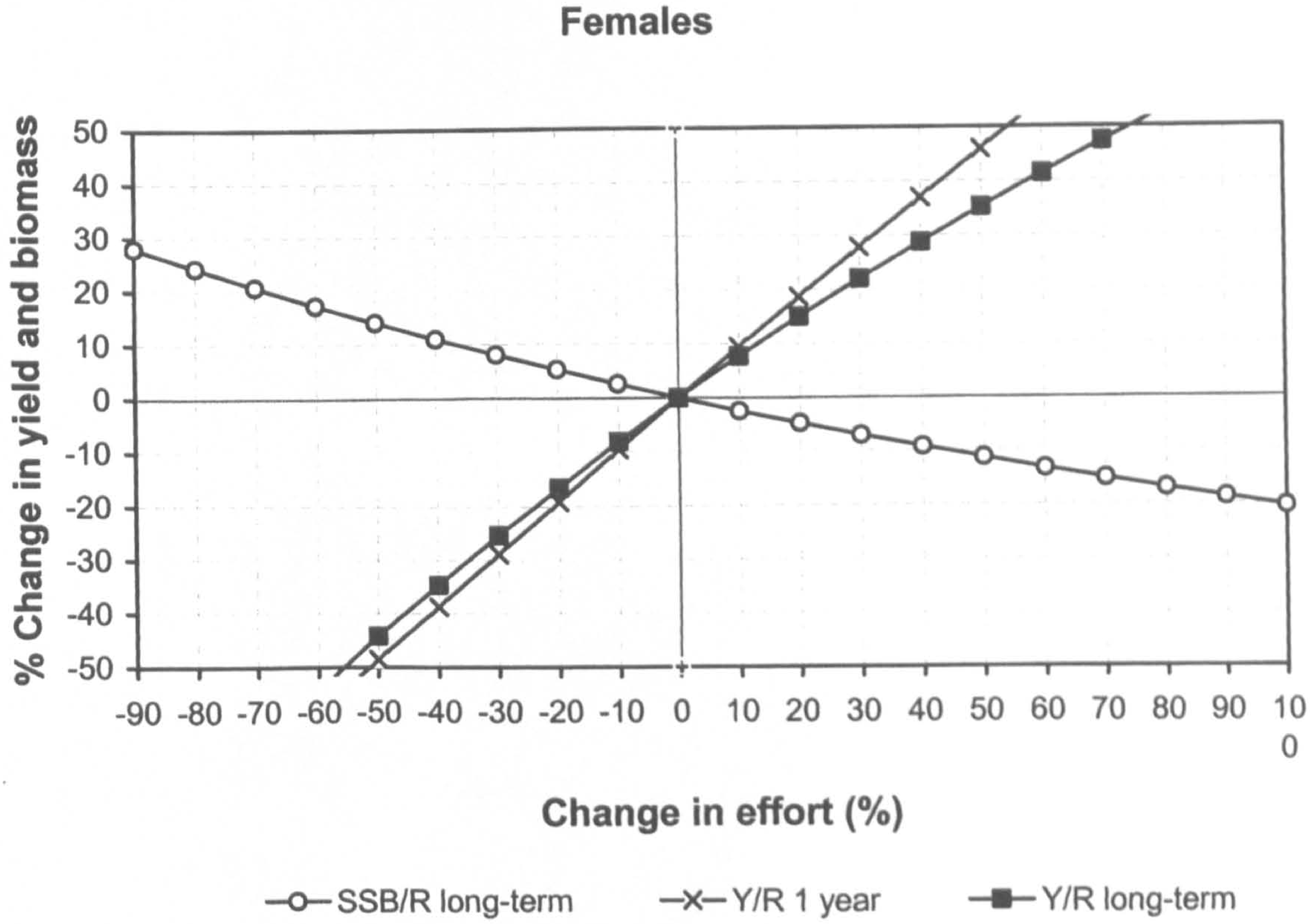


Figure B.27 - Predicted percentage changes in female long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in the Loch Fyne creel fishery in 2005, using ICES (1999) growth parameters.

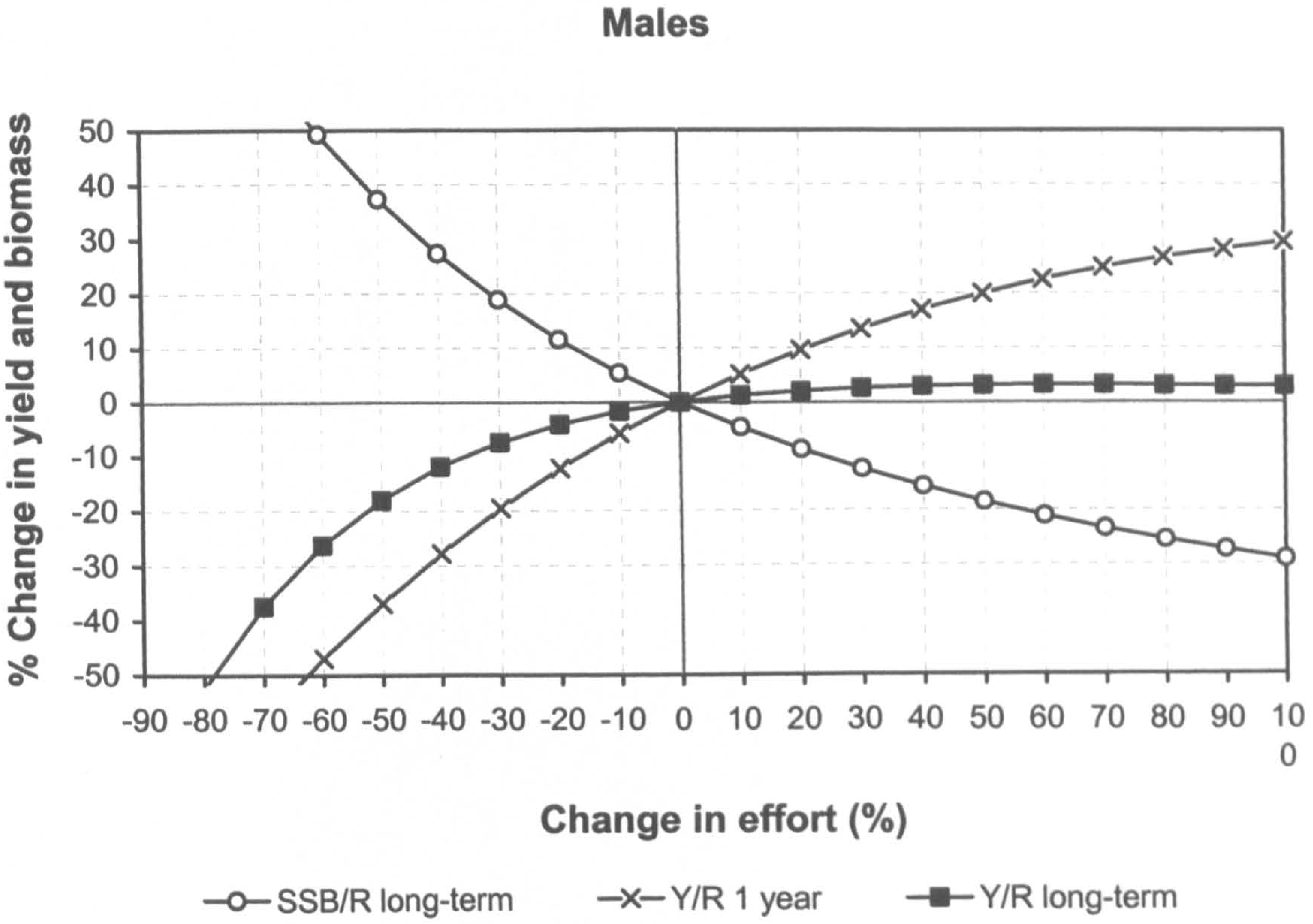


Figure B.28 - Predicted percentage changes in male long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in the Torridon creel fishery 2005, using ICES (1991) growth parameters (discards >37 mm CL included in landings).

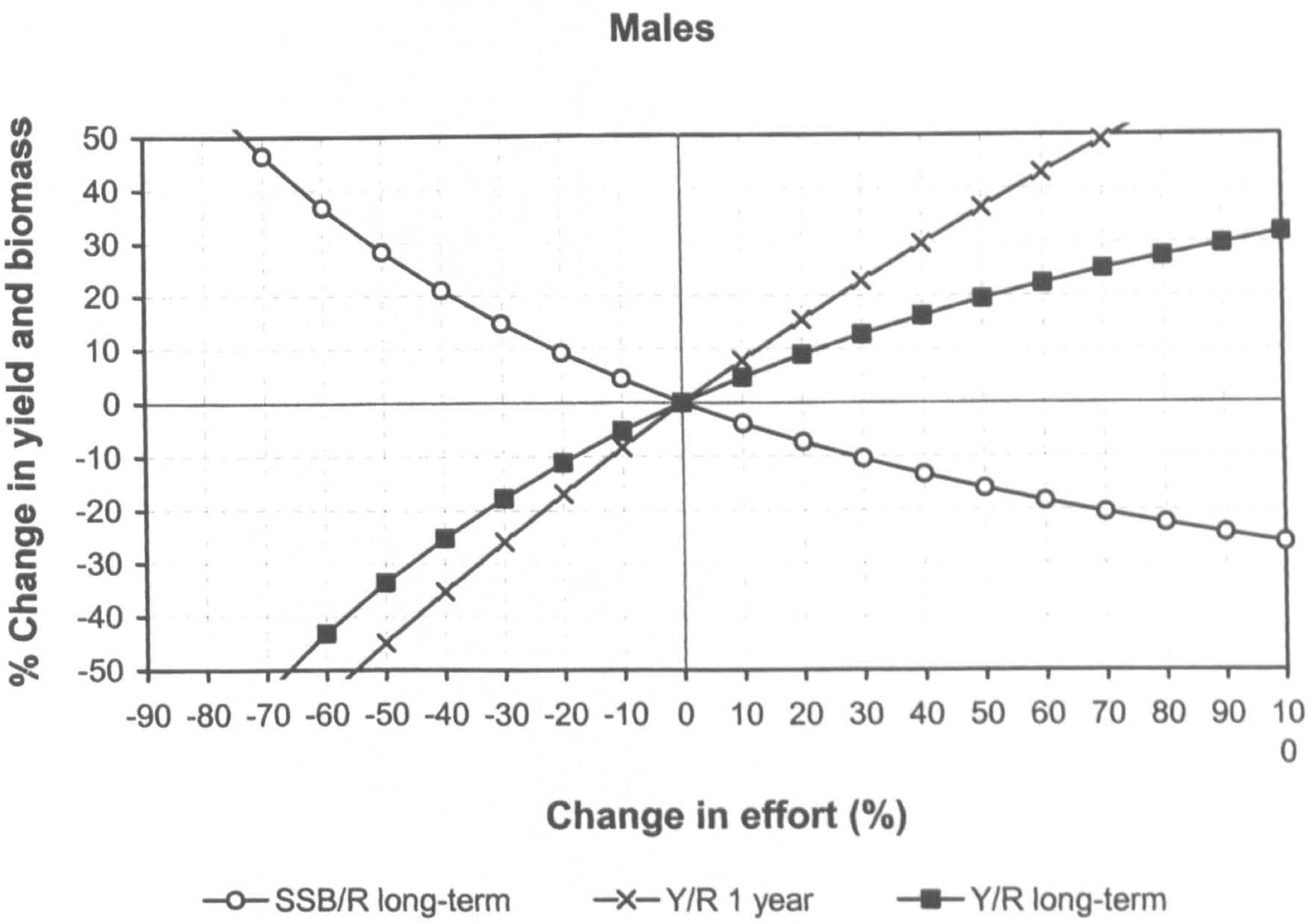


Figure B.29 - Predicted percentage changes in male long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in the Torridon creel fishery 2005, using ICES (1999) growth parameters (discards >37 mm CL included in landings)

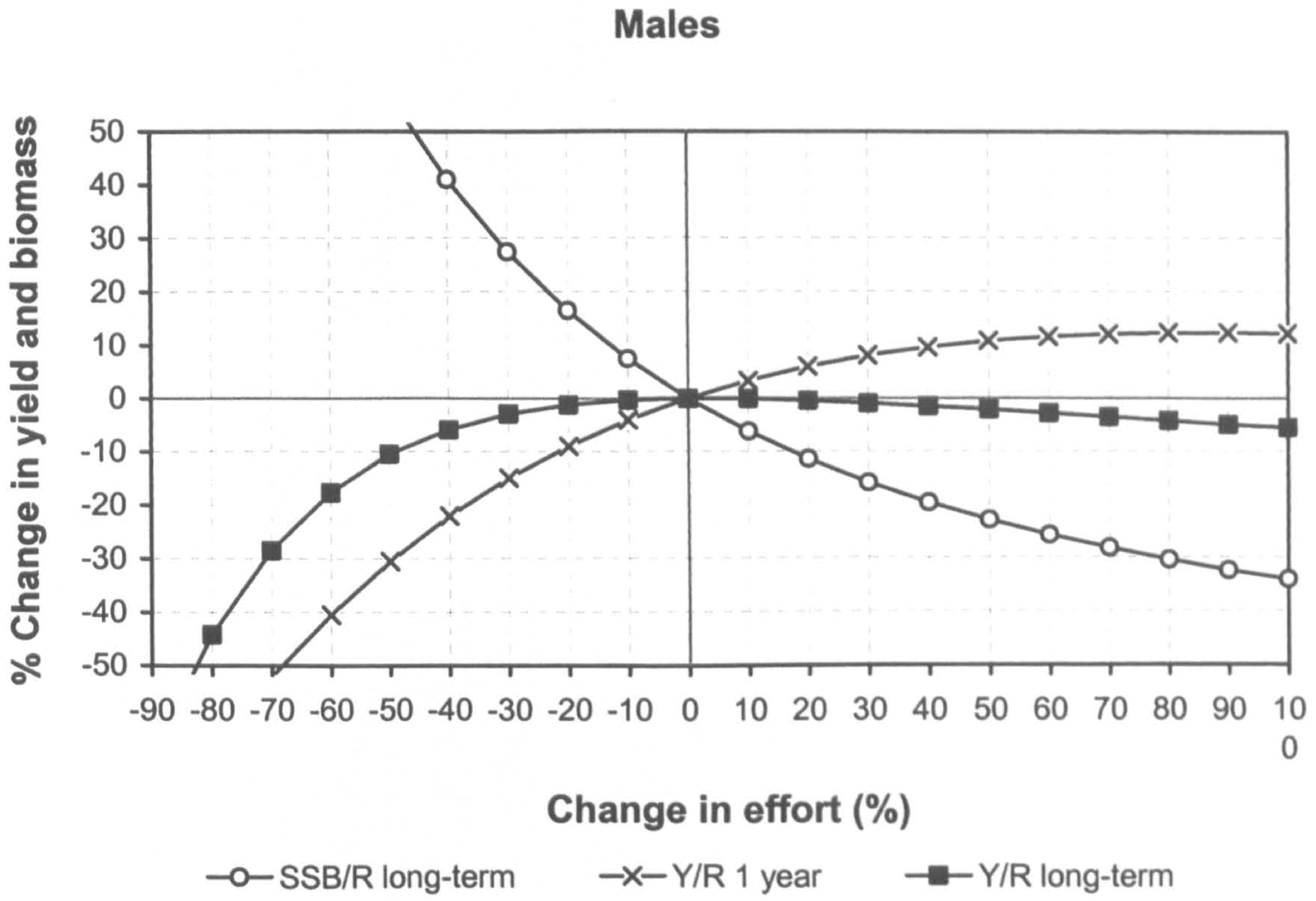


Figure B.30 - Predicted percentage changes in male long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in Loch Broom and the Summer Isles creel fishery in 2005, using ICES (1991) growth parameters.

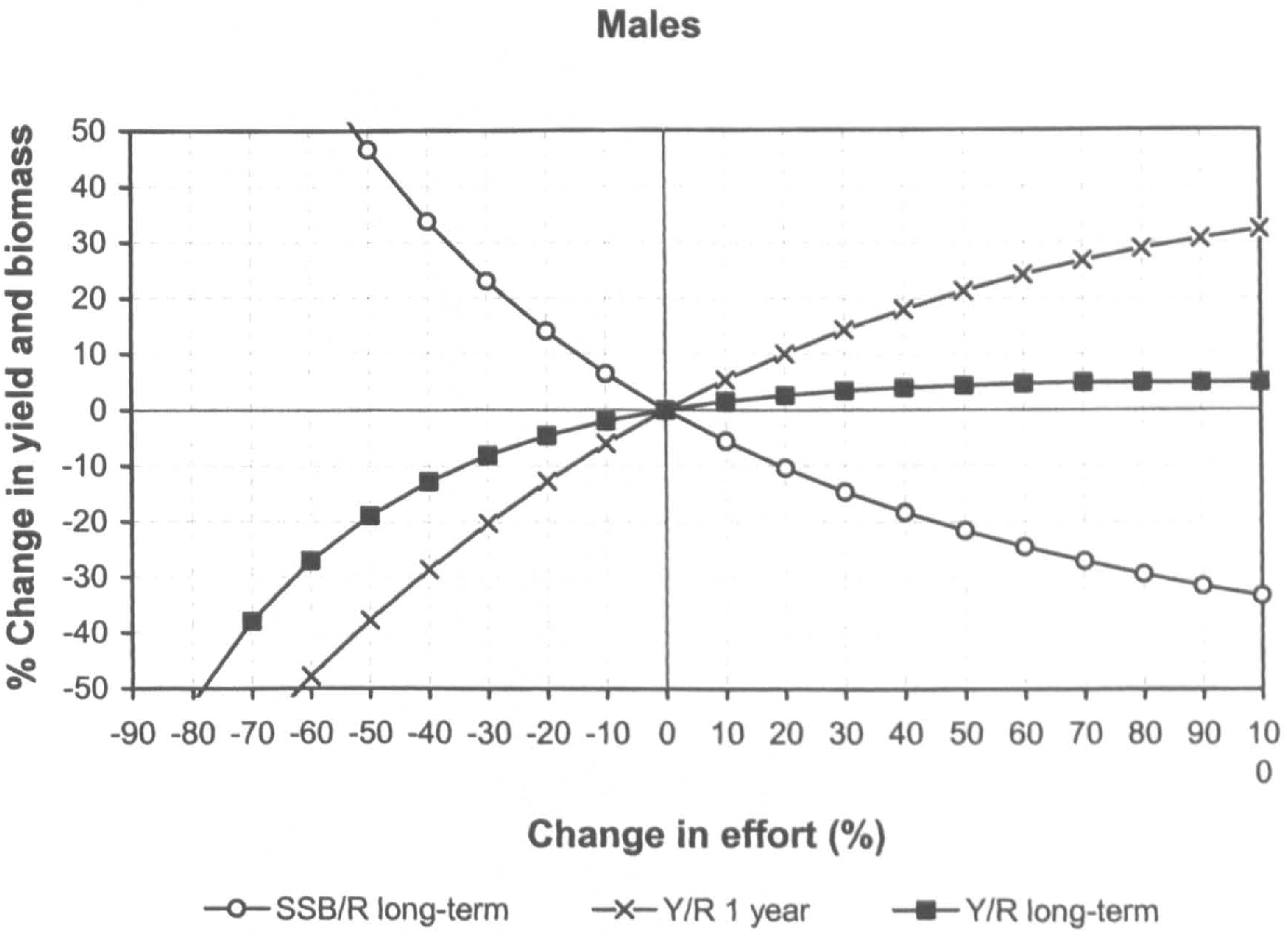


Figure B.31 - Predicted percentage changes in male long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in Loch Broom and the Summer Isles creel fishery in 2005, using ICES (1999) growth parameters.

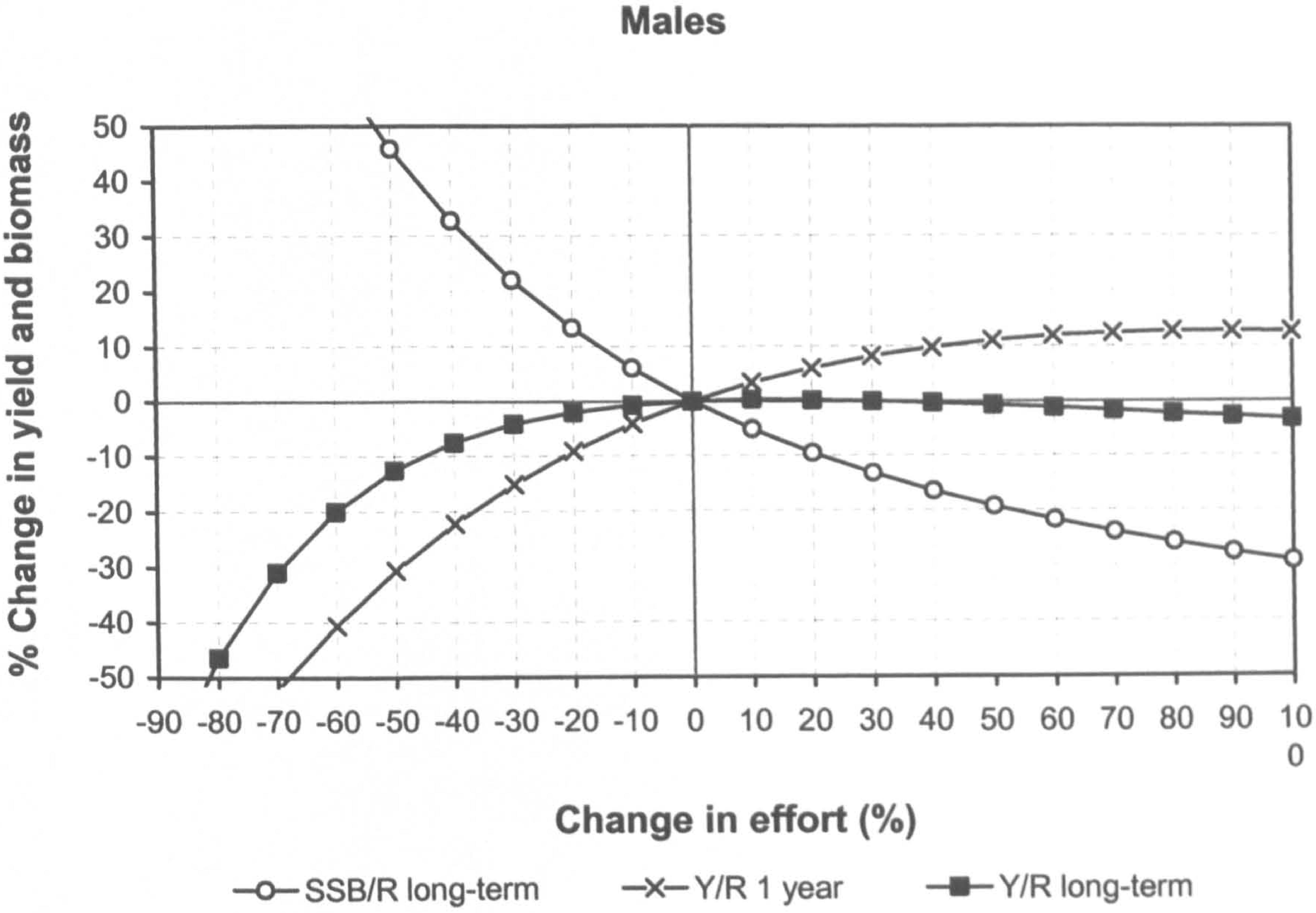


Figure B.32 - Predicted percentage changes in male long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in the Loch Gairloch creel fishery in 2004, using ICES (1991) growth parameters.

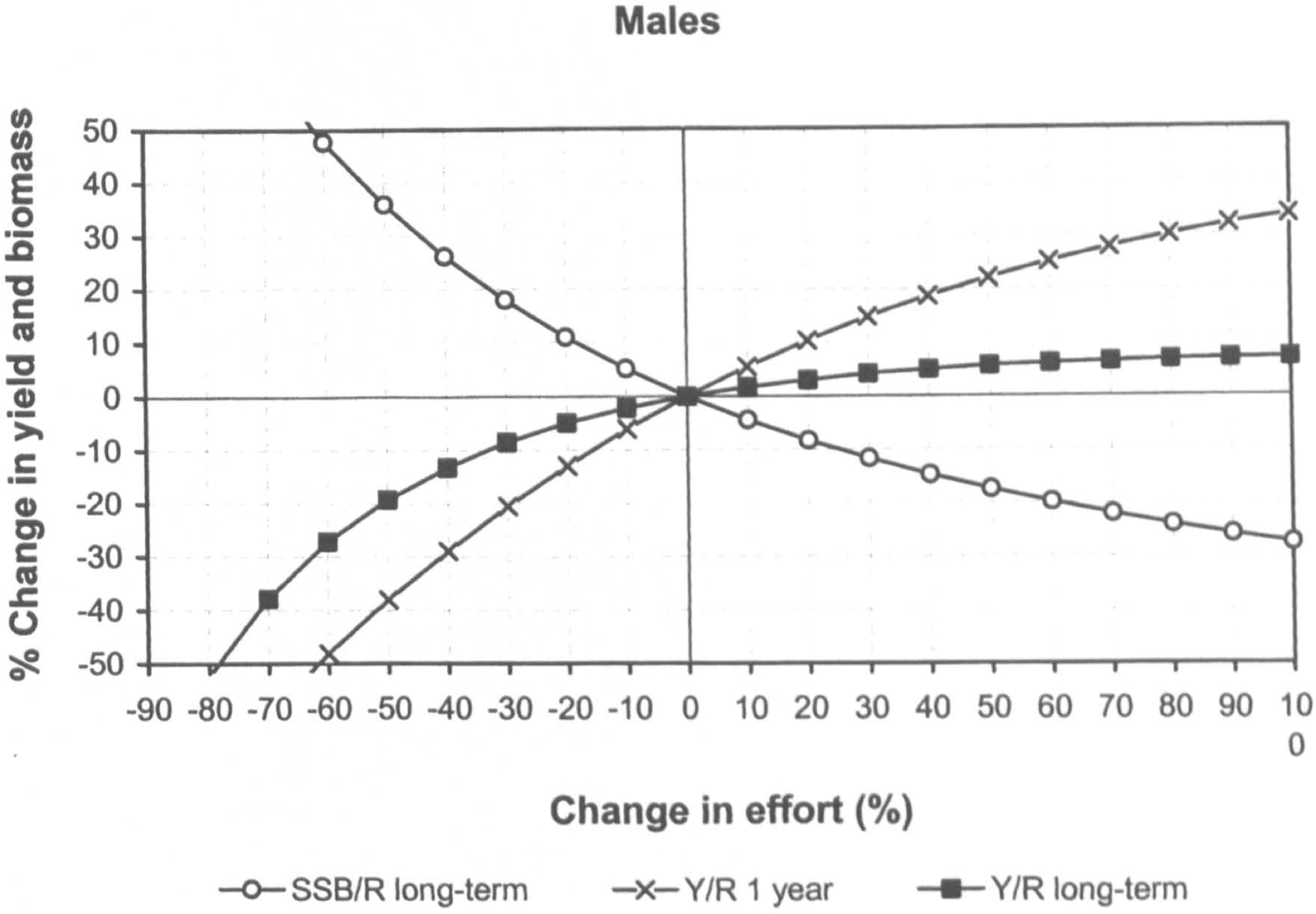


Figure B.33 - Predicted percentage changes in male long-term yield per recruit, spawning stock biomass per recruit and short-term yield per recruit following changes in effort in the Loch Gairloch creel fishery in 2004, using ICES (1999) growth parameters.

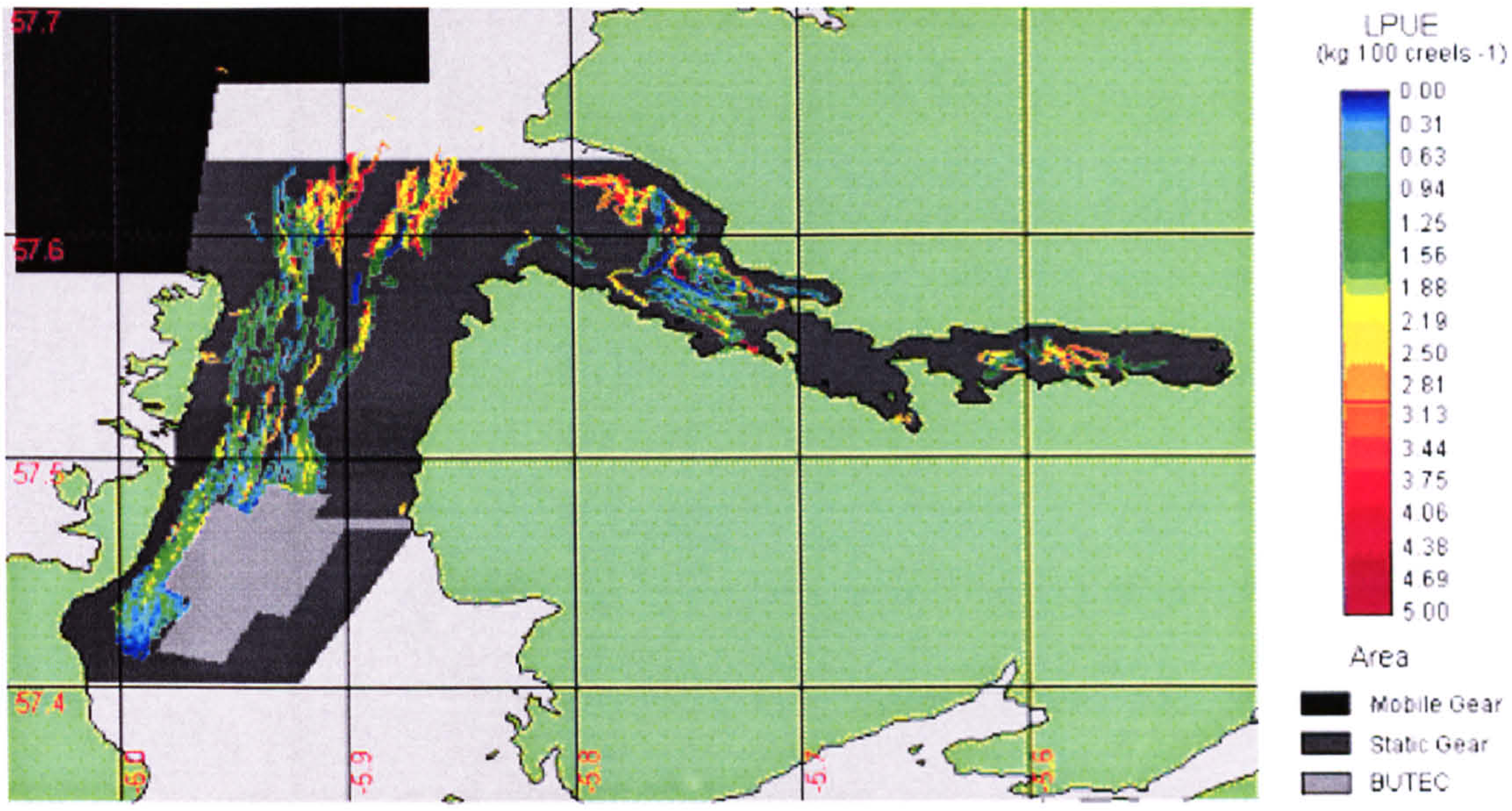


Figure C.1 - Spatial pattern of LPUE (kg 100-creels^{-1}) of small-sized *Nephrops* (39–45 mm CL) from the Torridon fishery, using GPS data logger and logbook data, one vessel in 2004 and 2005 (136 days fishing).

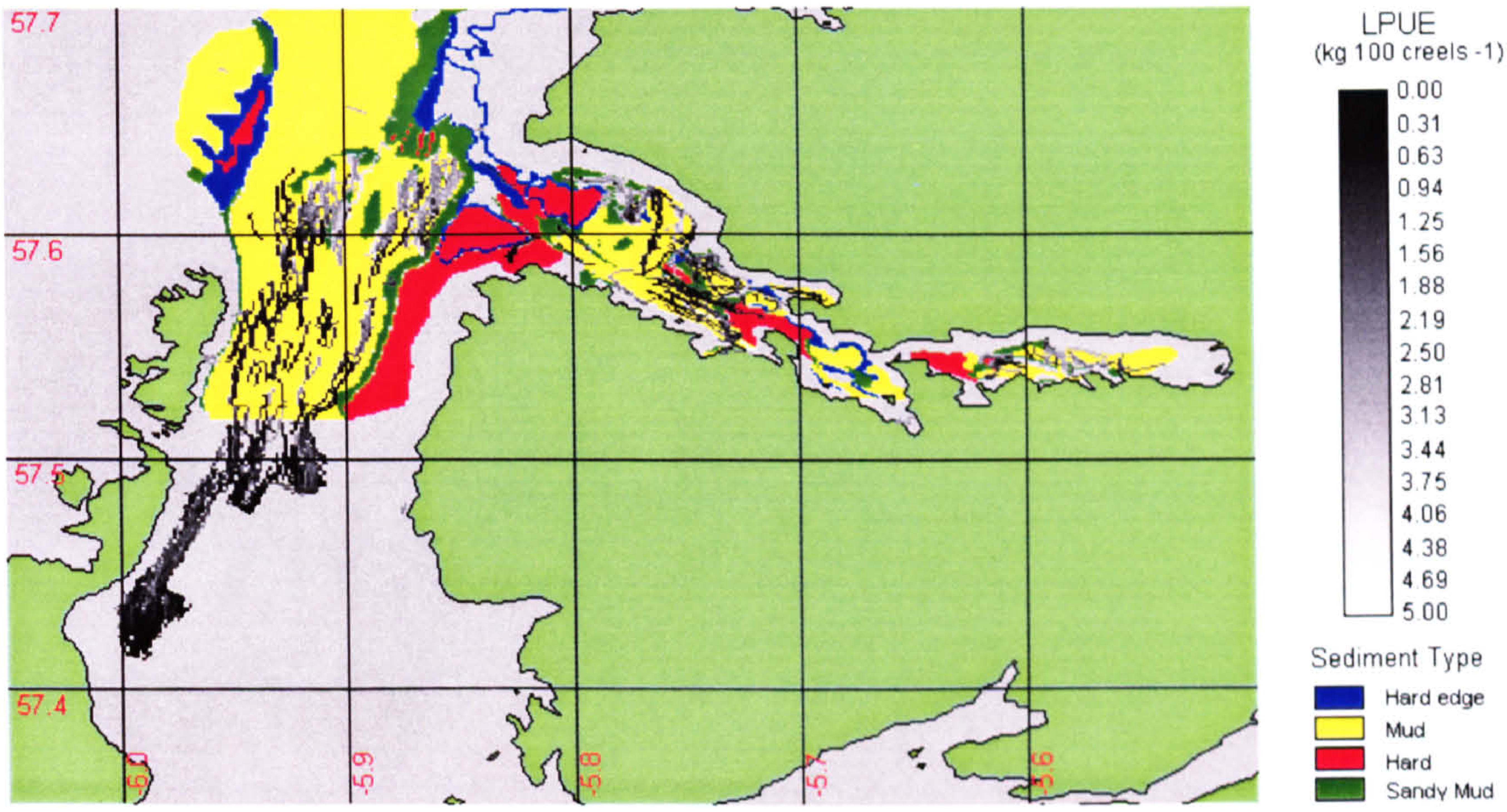


Figure C.2 - Spatial pattern of LPUE (kg 100-creels^{-1}) of *Nephrops* (39–45 mm CL) from the Torridon fishery, using GPS data logger and logbook data, one vessel in 2004 and 2005 (136 days fishing).

Figure C.3 – Average LPUE of *Nephrops* (small) averaged over pixels in areas of different sediment types, one vessel in 2004 and 2005.

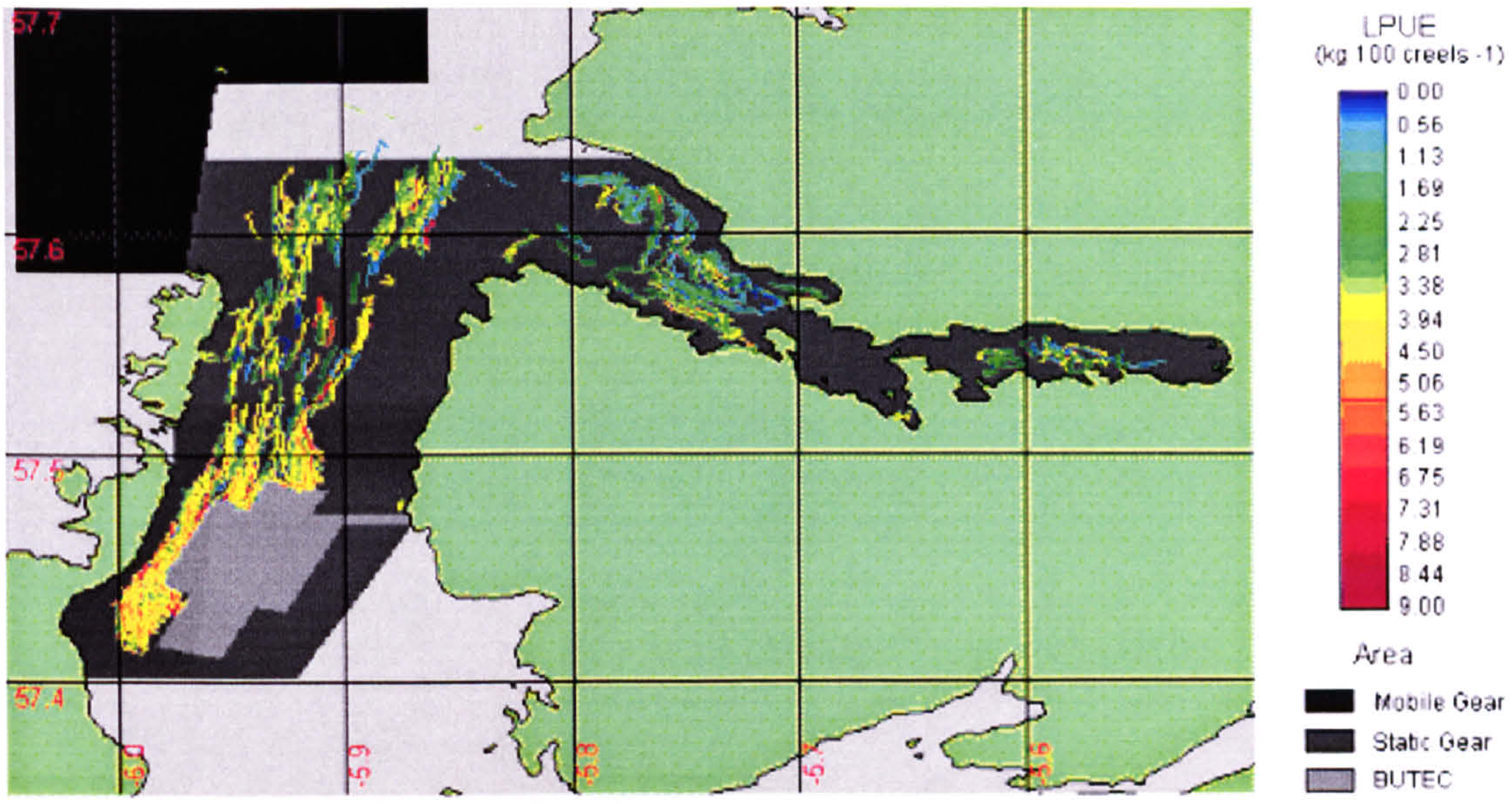


Figure C.4 - Spatial pattern of LPUE (kg 100-creels^{-1}) of large-sized *Nephrops* (>49 mm CL) from the Torridon fishery, using GPS data logger and logbook data, one vessel in 2004 and 2005 (136 days fishing).

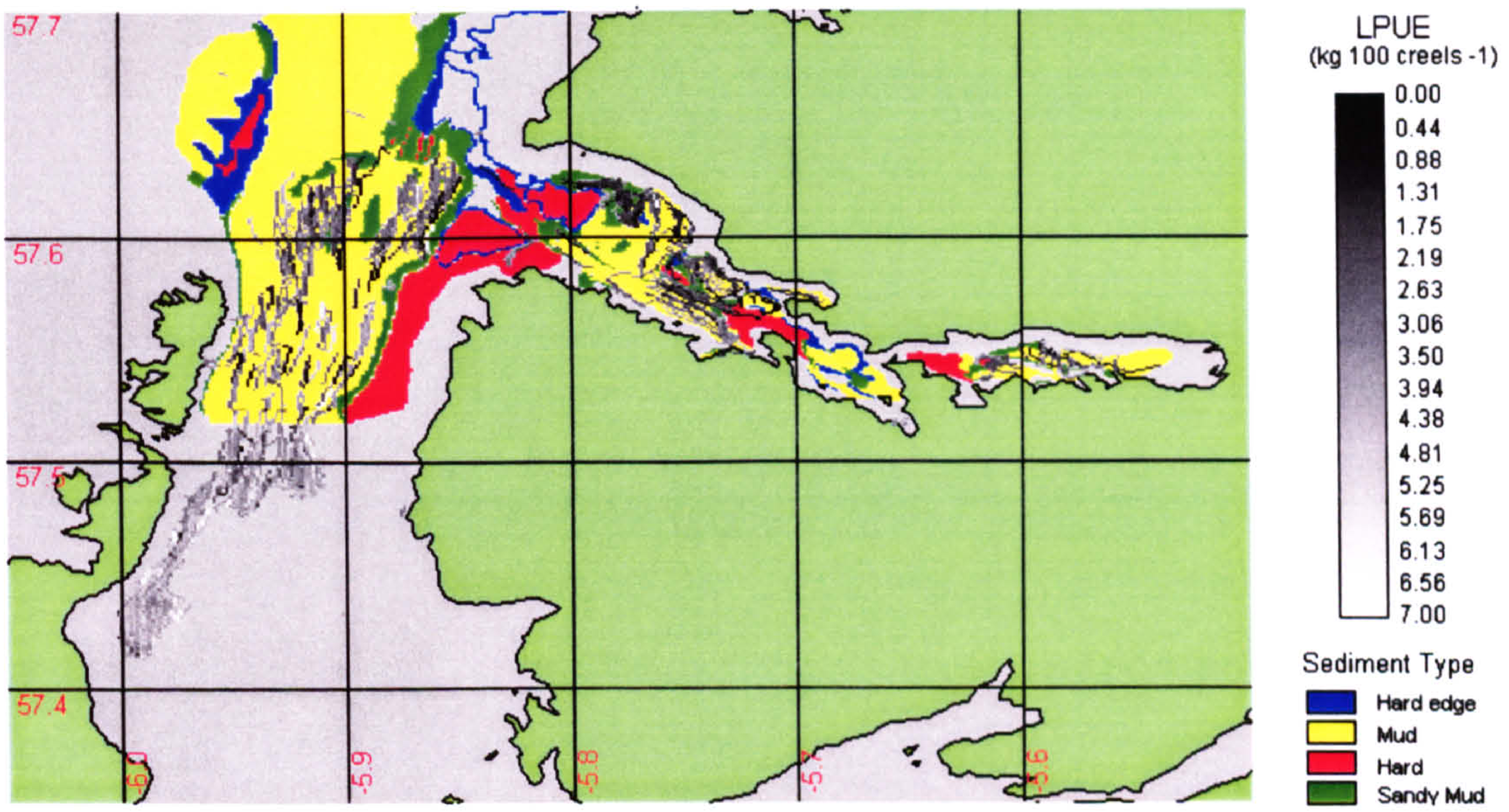


Figure C.5 - Spatial pattern of LPUE (kg 100-creels^{-1}) of large-sized *Nephrops* (>49 mm CL) from the Torridon fishery, using GPS data logger and logbook data, one vessel in 2004 and 2005 (136 days fishing).

Figure C.6 – Average LPUE of *Nephrops* (large) averaged over pixels in areas of different sediment types, one vessel in 2004 and 2005.