The Fish Populations of the Lower Forth Estuary, including the Environmental Impact of Cooling Water Extraction.

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Declaration

The work presented in this thesis is the result of my own investigations. It has not been nor will be submitted in candidature for any degree, in this or any other university.

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Abstract

The present study investigated the fish populations of the lower Forth Estuary, east Scotland. Cooling water extraction by the 2400 MW Longannet Power Station (LPS) inevitably removes a certain quantity of fish from the estuary, all of which experience mortality. The present study employed a sampling regime of greater intensity than previous studies to investigate the extent of mortalities from January 1999 – December 2000. Collections of fish impinged on intake screens were made eight times monthly, at LW or HW of spring or neap tides during the day or by night. Marine species dominated the assemblage of fish collected, with sprat, herring, and whiting contributing > 80% of total abundance. Sprat was twice as abundant as herring in 1999, while the proportions were very similar in 2000. Total abundance of all species collected in 1999 was estimated at 1.09×10^7, while the value of 3.29×10^7 in 2000 was three times larger. These figures were the largest recorded among British estuarine and marine power stations, but were precisely the correct order based on an exponential relationship between total impingement and water abstraction rate established from data from other locations. Validation of the estimated total biomass of fish removed was given by comparison with the known total mass of all materials disposed to landfill. Statistical analysis of impingement data showed that tidal range and season were the most important environmental variables influencing the rate of removal of fish from the estuary. That light was not significant for most species is attributed to high levels of turbidity and the resulting low visibility by day and night.

Demersal and benthic fish abundances collected from 1982 – 2000 in 30 annual trawls at three sites in the mid-lower Forth Estuary were analysed. Species tended to be present in greatest abundance at the most seaward of the sites. Patterns of seasonal abundance reflected those observed in the impingement study at LPS, and catches tended to be greatest at LW. Total species richness showed no significant trend over time, whilst total annual abundance of fish captured in trawls showed a significant negative trend. This was largely due to significant declines in the two most abundant species, namely whiting and eelpout, attributable in the latter case to increasing temperatures. Changes in the ichthyofaunal composition were largely driven by whiting, eelpout, cod and plaice. Eight of ten common species showed no significant trend in abundance over the length of the time series, suggesting them to perhaps be at equilibrium densities.

Quantities of commercially fished species above minimum landing size limits that were removed by LPS were very low, and restricted to herring and occasional whiting. The quantity of juveniles that could have recruited into the fished populations was expressed as equivalent adults. The values were larger than any previously reported in the UK, primarily due to the quantities of juvenile fish impinged being greater than at any other British power station, and the importance of the Forth as a nursery area for marine species.

It was concluded that LPS is the dominant UK power station in terms of magnitude of impingement losses. It may be prudent to consider a precautionary approach to mitigate losses, and to this end options for reduction of the magnitude of impingement are discussed.

Contents

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My inspiration was **Sojourner Howe**, who was my perfect companion throughout the writing up period of the present study.

Dedication

I dedicate the present study to my mother, Jasna Birtić Greenwood, who passed away on June 11 1999, and to my father Richard Stanley Greenwood, who shares with me the same deep sense of loss.

Glossary of Abbreviations

Abbreviations in the present study are generally introduced in the text, but the following list is intended to provide a convenient reference source.

AFDS, acoustic fish deterrent system.

CA, diadromous migrant (see section 1.3.1).

CI(s), confidence interval(s).

COMAR, code of Maryland regulations.

CSS, critical swimming speed.

CV, coefficient of variation.

CVSD, coefficient of variation of standard deviation.

CW, cooling water.

CWIS, cooling water intake structure.

DD, density-dependent.

DI, density-independent.

DO, dissolved oxygen.

EIA, environmental impact assessment.

EPRI, Electric Power Research Institute.

ER, estuarine resident (see section 1.3.1).

FRPB, Forth River Purification Board.

FW, freshwater (see section 1.3.1).

GLM, generalised linear modelling (model).

GLMs, generalised linear models.

HW, high water.

ICES, International Council for the Exploration of the Sea.

KPS, Kincardine Power Station.

LPS, Longannet Power Station.

LT_{50}, lethal temperature at which 50% of tested organisms perish.

LW, low water.

MA, marine adventitious (see section 1.3.1).

MCLS, minimum commercial landing size.

MDS, multidimensional scaling.

MJ, marine juvenile (see section 1.3.1).

MS, marine seasonal migrant species (see section 1.3.1).

MW, megawatt.

MWe, megawatts of electricity.

NS, not significant.

NTU, nephelometric turbidity unit.

PRIMER, Plymouth Routines in Marine Ecological Research.

PSU, practical salinity unit(s).

SD, standard deviation.

SE, standard error.

SEPA, Scottish Environment Protection Agency.

SIMPER, similarity percentage analysis.

SL, standard length.

SSB, spawning stock biomass.

S/V, survey vessel.

TL, total length.

US EPA, United States Environmental Protection Agency.

Chapter 1. Introduction

1.1. Overview

An estuary is "a semi-enclosed coastal body of water, which has a free connection with the open sea, and within which sea water is measurably diluted with fresh water from land drainage" (Pritchard, 1967). The fact that estuaries are dynamic transition zones between freshwater and marine ecosystems is significant for the life inhabiting them, not least the ichthyofauna. Freshwater, marine, diadromous and estuarine-resident fish can be expected to form the fish assemblage of most estuaries, typically with relatively few species thriving on the abundant resources concentrated in the estuary by the action of freshwater input and tidal influence (Day *et al.*, 1989). The sheltered nature of estuaries as harbours, flat land, and of course the source of water they offer, makes this type of environment desirable for human settlement.

The fish assemblage of the Forth Estuary, East Scotland, has thus far been most fully studied by trawl studies undertaken by the Forth River Purification Board (FRPB) and its successor, the Scottish Environment Protection Agency (SEPA). Seasonal and spatial differences in the composition and abundance of the fish populations have been investigated (reviewed by Poxton, 1987). Estimates of total fish biomass and production, as well as approximate population sizes for the more abundant species, are based solely on demersal studies using either Agassiz or beam trawls (Elliott and Taylor, 1989; Elliott *et al.*, 1990). Other than one short pilot midwater trawl study in the early 1980s (FRPB, 1984), the information obtained on pelagic species is based on the comparatively small numbers of these fish that are caught in demersal trawls.

Cooling water abstracted from the Forth Estuary by the electricity generating station at Longannet has been estimated to remove substantial quantities of fish (Maitland, 1997; 1998), which are lost from the estuary. The present study intended to undertake a more intensive sampling programme of cooling water intake than previously attempted at this location, in order to both quantify the extent of power station mortality and to investigate hypotheses concerning the influence of environmental factors on rates of removal of fish species in cooling water. An assessment of the 19 year Agassiz trawl dataset was carried out to examine long term trends in abundance in the benthic and demersal fish assemblage, as well as seasonal and spatial trends in abundance. Novel pelagic trawling was initiated to address the paradoxical situation existing whereby a greater abundance of individuals of pelagic species were estimated to be removed from the estuary in cooling water than were assessed to be present by inadequate trawling on or near the bottom.

1.2. The study area: the Forth Estuary

The Forth Estuary stretches eastwards from the upstream limit of tidal intrusion in Stirling downstream along some 48 km to the seaward limit just east of the Forth Road Bridge (Maitland *et al.*, 1984; Wallis and Brockie, 1997). Thus the head of the estuary is at Stirling, the upper estuary is taken as being from Stirling to Alloa, the middle estuary (including Longannet) from Alloa to a line between Bo'ness and Crombie Point, and the lower estuary from Bo'ness/Crombie Point to the Queensferry road and rail bridges at the mouth of the estuary (Figure 1.1) (McLusky, 1987a).

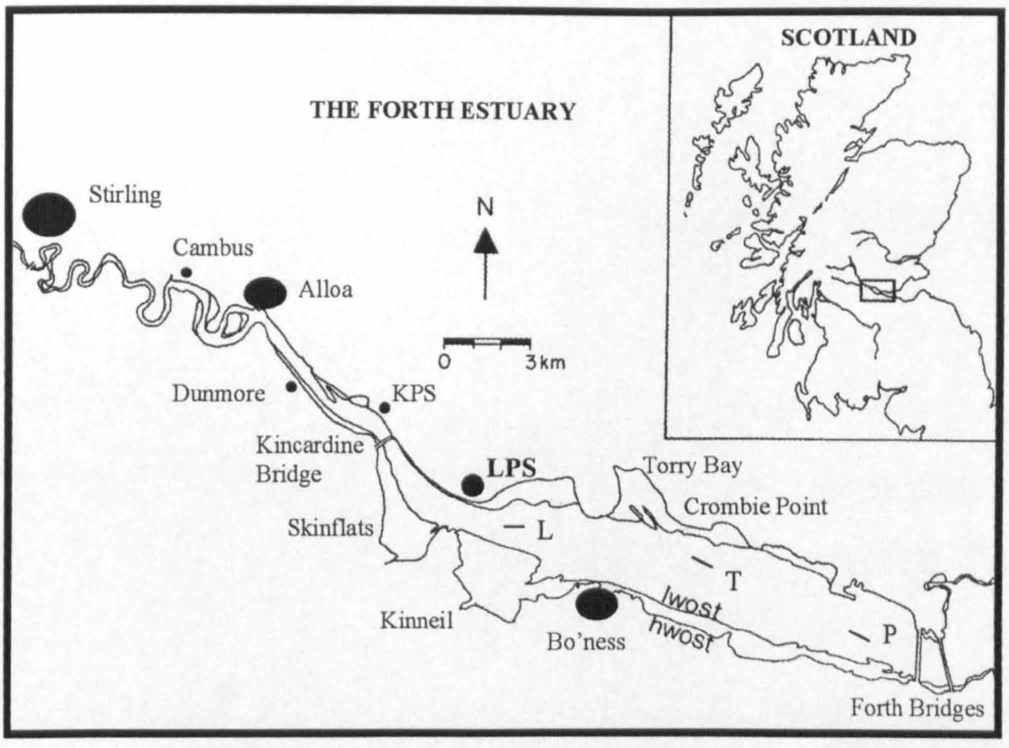


Figure 1.1. The Forth Estuary. LPS denotes position of Longannet Power Station, KPS of the now decommissioned Kincardine Power Station (see Chapter 2). Single letters indicate locations of trawls discussed in Chapter 3: P = Port Edgar, T = Tancred, L = Longannet. lwost = LW of spring tides; hwost = HW of spring tides.

Freshwater inputs to the estuary are largely from the rivers Forth and Teith and the Allan Water, with the flow rate varying with season from a mean of 24 m^3s^{-1} from June to August, increasing to 93 m^3s^{-1} between December and February (Wallis and Brockie, 1997). This reflects the increase in river flow due to rain and/or melting snow in autumn and winter. It is the ebb and flow of the tide that dominate water movements in the estuary, however. The tidal cycle is semi-diurnal, with a mean spring tidal range of 5.0 m and a mean neap range of 2.5 m (Webb and Metcalfe, 1987). From the middle estuary seawards the flood is longer in duration than the ebb, resulting in faster average ebb velocities (70-110 cms^{-1}) compared with flow (40-70 cms^{-1}); at low water a long period of slack water of about 3 hours occurs, particularly in the lower reaches such as at Rosyth (Webb and Metcalfe, 1987). Tidal excursion varies between 8-16 km, the maximal values tending to be between Cambus and Kincardine (see Figure 1.1.) (Webb and Metcalfe, 1987).

The interaction of freshwater flow and tidal fluxes of seawater influences the physico-chemical composition of the Forth Estuary. The estuary varies from being partially-mixed to well-mixed (Webb and Metcalfe, 1987; McLusky, 1989; Wallis and Brockie, 1997), with the tidal range influencing the extent to which denser, more saline water invades the upper layers of freshwater. The rate of freshwater flow affects the upstream penetration of salt. Thus a low tidal range occurring at the same time as a high freshwater flow would give salinity of 0 PSU several kilometres below Stirling, whereas a high tidal range combined with low freshwater flow would confine water with a salinity of 0 PSU to near the head of the estuary at Stirling. Surface waters of the estuary tend to ebb before the bottom waters, whereas the bottom waters flood earlier than the surface waters; in common with most northern hemisphere estuaries the flood currents are stronger on the north side of the estuary, though the cause of this has been suggested to be due to the bathymetric action of the estuary, rather than the expected effect of Coriolis' force (Webb and Metcalfe, 1987).

Along with salinity, temperature is an important physical variable that is affected by fresh and saltwater interaction. In summer, the freshwater temperature is usually greater than that of the seawater, whereas the opposite is true in the winter. Upstream, the ameliorating effect of the sea is diminished, so that the more freshwater stretches of the estuary are in phase with the surrounding land in terms of temperature change (Harrison, 1987), which may result in relatively rapid temperature variability compared to more marine areas.

The area of the estuary of primary interest to this study, between the Forth Bridges and Kincardine Bridge (Figure 1.1), is approximately 6057 ha at high water, reducing to 3850 ha at low water (Jayamanne and McLusky, 1997). The estuary's sub-tidal sedimentary regime is mostly of fine material with median diameter $< 50\ \mu m$, with coarse mixed materials in some areas, notably where the estuary is constricted (at Grangemouth docks, Kincardine Bridge and near Rosyth) and in the main channel near the north shore (Elliott and Kingston, 1987), as well as stony substratum nearer to the mouth of the estuary at Blackness and Port Edgar (Elliott and Taylor, 1989). Fine mud is found at the inter-tidal mudflats of Kinneil and Skinflats, with this mud also extending to the sub-tidal areas next to these locations. A study of 80 locations in the lower and mid-estuary by Elliott and Kingston (1987) revealed that the silt and clay content (i.e. material $< 63\ \mu m$ diameter) of these areas was generally quite high (between 40 and 80 % in many locations).

The fineness of the estuary's sediments, combined with the water movements already mentioned, makes the Forth Estuary turbid. Maximum turbidity occurs at the freshwater/brackish water interface, between 3 and 15 km downstream of Stirling; its intensity depends on freshwater flow and tidal range and varies greatly between spring and neap tides (Webb and Metcalfe, 1987; Dobson, 1997). Sediment re-suspension affects oxygen levels in the water, and the variability of freshwater flow with season is crucial in this respect: low flow combined with higher temperatures in summer causes reduced dissolved oxygen levels, whereas in winter higher freshwater flow and decreased temperature mean that oxygen levels are typically > 80 % saturation (Griffiths, 1987). Dobson (1997) suggests that a correlation between dissolved oxygen concentration and salinity in the mid and upper estuary is due to permanently trapped organic matter acting as the primary consumer of dissolved oxygen in low salinity regions, rather than being caused by suspended solids in the water column. It is important to note that oxygen depletions are rare lower down the estuary, but that any oxygen deficit in the upper estuary may act as a barrier to migratory fish needing to descend to the lower estuary and ultimately the sea. To this end, SEPA defines a minimum dissolved oxygen level environmental quality standard (EQS) of 4 $mg.l^{-1}$ (equated to a 95th percentile of 4.5 $mg.l^{-1}$) as being desirable to reduce the likelihood of stress on fish such as Atlantic salmon, *Salmo salar* (Leatherland, 1987; Elliott *et al.*, 1988).

The Forth Estuary has been described, along with the Firth of Forth, as being the "most intensively used sea area around Scotland" (McLusky, 1987a). The location of numerous major sources of industrial and municipal effluents has had a detrimental effect on water quality in the estuary. Improvements have been made in recent years with respect to the treatment of effluent before disposal into the estuary. Griffiths (1997) lists examples of these improvements such as provision of long sea outfalls and primary/secondary treatment of waste in sewage treatment works, as well as biological treatment of waste produced by companies such as Weir Paper, Quest International, Zeneca and British Petroleum. These changes have had a tangible effect, for example the return to the Forth of the anadromous smelt *Osmerus eperlanus*, a fish species sensitive to pollution (Griffiths, 1997; personal observations).

In common with most estuaries, the Forth Estuary is regarded as being biologically productive: between Kincardine Bridge and the Queensferry Bridges, it is estimated that over the whole area covered by tides there exists a total of almost 3500 tonnes wet weight of benthic biomass (McLusky, 1987b) which supports a subtidal biomass of shrimp (mostly *Crangon*) of about 110-170 tonnes (Jayamanne and McLusky, 1997).

1.3. The ichthyofauna of estuaries

General features of fish found in estuaries, particularly temperate estuaries of the northern hemisphere, are introduced as a basis to place the later information specific to the Forth in context.

1.3.1. Categories of estuarine fish

Elliott and Dewailly (1995) provide a comprehensive account of classifications of the fish in European estuaries. A fish present in an estuary generally belongs to one of the following categories:

FW - freshwater species such as perch, *Perca fluviatilis*, which penetrate the upper reaches of estuaries by voluntary migration or due to river spates;

CA - diadromous fish seasonally migrating through the estuaries, such as anadromous Atlantic salmon, *Salmo salar*, or catadromous eel, *Anguilla anguilla*, requiring to use the environment as a route linking their preferred areas for spawning and feeding;

ER - estuarine resident species, spending the whole life cycle in estuaries, for example pogge, *Agonus cataphractus*, or eelpout, *Zoarces viviparus*;

MS - marine species with seasonal migrations into estuaries as juveniles and adults, for example for overwintering purposes, like the clupeids, sprat and herring;

MJ - marine species using the estuary as a nursery for juveniles with adults present only in very low numbers, e.g. the gadoids, whiting, *Merlangius merlangus*, and cod, *Gadus morhua*, and the flatfish, plaice, *Pleuronectes platessa*, and dab, *Limanda limanda*;

MA - consists of marine adventitious species, such as ling, *Molva molva*, and lesser weever, *Trachinus vipera*, which enter the lower reaches of the estuary on an irregular basis, mostly for opportunistic feeding.

Some fish may lie out with these strict definitions. Flounder, *Platichthys flesus*, is regarded as being the classic example of an ER species (McLusky, 1989), but in fact leaves to spawn in the deeper waters beyond the estuary mouth (Elliott and Taylor, 1989). Newly settled young then ascend the estuary to use the upper turbid reaches as a nursery, and adults also re-enter. Some authors have tentatively classified flounder as CA species, e.g. Costa and Elliott (1991), others suggest either ER or CA is possible (Pomfret *et al.*, 1991). The thesis uses the abbreviations above throughout, so reference should be made to this section in order to clarify the definitions used.

1.3.2. Characteristics of estuarine fish

Abundant biological resources are concentrated in estuaries by the combined factors of freshwater input and tidal action, and this is reflected in very high levels of biomass production compared to other aquatic ecosystems. Estuaries may produce biological resources at the rate of 16 g $m^{-2}year^{-1}$, a figure that can be contrasted with the world oceanic mean of less than 1 g $m^{-2}year^{-1}$ (Day *et al.*, 1989). To exploit these resources, estuarine fish must be able to adapt to the accompanying fluctuating conditions caused by these same abiotic factors. Species distribution within estuaries is governed by many influences, but water temperature, salinity and dissolved oxygen levels are paramount. In a study of the Humber Estuary, Marshall and Elliott (1998) showed that several species of fish exhibited distributions that appeared associated significantly with temperature, these being negative correlations for whiting and sprat, whereas sole (*Solea solea*), flounder and herring displayed significant positive correlations. Salinity gave significant positive correlations with sole, plaice and pogge, but a negative correlation with three-spined stickleback (*Gasterosteus aculeatus*) distribution, the latter reflecting the fact that this is a primarily FW species, though some populations may be CA migrants. Whiting and pogge showed significant positive correlations with dissolved oxygen levels, whereas flounder and stickleback were negatively correlated to this environmental parameter. Similarly, Thiel *et al.* (1995) showed that the structure of the fish community of the Elbe Estuary, Germany, is affected by the estuary's longitudinal gradient of salinity; current velocity differences gave non-uniform distributions across the estuary; and water temperature fluctuations and low oxygen concentrations acted seasonally to alter patterns of diversity and abundance. Strategies for survival are necessarily metabolically expensive, and may consist of migration to more favourable conditions, or else the ability to tolerate change in the environment (Moyle and Cech, 1996). Relatively few species are equipped to flourish within estuaries, and typically 8-15 species make up > 90 % of the fish biomass in estuaries (Day *et al.*, 1989). The harsh nature of the estuarine environment, due to fluctuations in salinity and often strong tidal movements, combined with the fact that estuaries are geologically young environments (McLusky, 1989) and so the time to evolve to the environment has been relatively short, is reflected in the observation that there are few true ER species (Potter *et al.*, 1997).

Marine species from the MJ and MS categories may inhabit estuaries for a variety of reasons (Elliott *et al.*, 1990). Since salinities are generally lower in estuaries compared to the sea, this reduces the osmotic stress euryhaline fish experience since less water will tend to leave body fluids – this is of particular importance to juveniles, who are more vulnerable to the pressures of the environment than are adults. Avoidance of predation is an important consideration, and estuaries seem to offer an attractive environment in two respects: firstly, predators tend to be in relatively low densities in estuaries compared with neighbouring coastal waters, and secondly the predators that are present have greatly reduced hunting abilities. This latter point is due to the generally high water turbidity affecting visibility, vision being the primary sense used in predation (Guthrie and Muntz, 1993). The presence of high prey densities may be a reason for estuarine use in some species, though it is unlikely to be so for planktivorous clupeids, which are present in greatest numbers at a time of year that differs from zooplankton maxima (Elliott and Taylor, 1989). Juvenile whiting may follow the migrations of common shrimp, *Crangon crangon*, in the Severn Estuary and Bristol Channel, suggesting that this source of food is important in their estuarine distribution (Henderson and Holmes, 1989). It is generally believed that the physico-chemical characteristics of the water chiefly tend to govern distribution of estuarine fish, with biological factors being of secondary importance (Moyle & Cech, 1996).

Some authors regard MS and MJ species as 'estuarine-dependent', meaning that these species are obliged to use estuaries in at least one phase of their life cycle (e.g. Moyle and Cech, 1996). It seems that this may not be the case, however, since evidence exists for so-called estuarine-dependent species using other areas instead of estuaries, e.g. sheltered coastal waters (Blaber and Blaber, 1980; Claridge and Potter, 1984). Thus

Lenanton and Potter (1987) suggest 'estuarine-opportunistic' to be a more appropriate term for fish of the MS and MJ categories, given Hedgpeth's (1982) comment that "estuaries are transitory features in the geological sense and could not be depended on as critical environments for the survival of marine species in coastal environments".

The present study hereafter uses common names for species encountered during sampling in the Forth Estuary, with scientific nomenclature being listed in Appendix 1. Common and scientific names of species other than those sampled during the course of the present study are given in the text.

1.4. Thermal power generation and the aquatic environment

Thermal power generation involves the use of either a fossil fuel (gas, coal, oil or peat), or else the fission of radioactive elements, to produce heat. This heat is central to the Rankine cycle, a thermodynamic process whereby steam produced at high temperature and pressure is conveyed to a turbine, transmitting energy to the turbine rotor, which then drives the generator (Figure 1.2) (Langford, 1983). The steam is condensed at the end of the turbine and returned to the boiler to repeat the cycle. The significance of this to the aquatic environment is that large volumes of cooling water are required to allow the condensation process to occur: a direct-cooled 1000 MWe conventional power station uses 30 m^3s^{-1} of cooling water (Turnpenny and Coughlan, 1992). In the so-called direct-cooled (also known as 'once-through') systems, the cooling water is returned directly to the water body from whence it came (though far enough away not to affect the temperature of the water that is being withdrawn for cooling), after having taken part in the condensation process. An alternative system, utilised where water is in relatively short supply, is the re-use of the cooling water following loss of heat in cooling towers or ponds. This method, known as indirect cooling, may be two or three times more expensive than once-through cooling (Langford, 1983). There are over 40 direct-cooled power stations on the coast of Britain and the North Sea coast of continental Europe (Henderson, 2000).

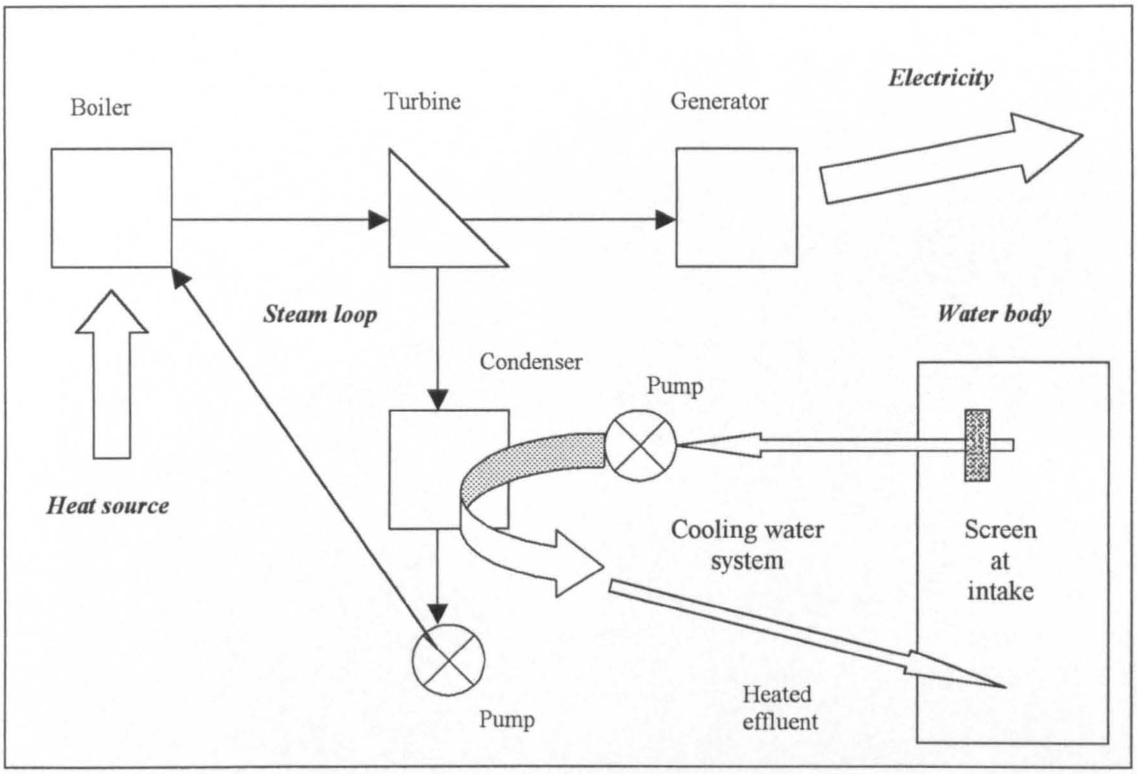


Figure 1.2. Schematic diagram of cooling water system for a direct-cooled power station (redrawn from Langford, 1983).

Concern over the effects on the aquatic environment from thermal power generation initially focussed on thermal emissions from discharge of heated cooling water. This was noted as long ago as the British Electricity Supply Act of 1919 (Langford, 1983). The elevation of water temperature due to such emissions may change the chemistry of the water in the receiving water body, thus affecting organisms in that habitat. Water chemistry may also be affected by various chemical agents used to minimise corrosion and biofouling within the power station system. Of greater importance to this study is the effect of the cooling water extraction on the aquatic environment. Water is withdrawn at an intake, which may be offshore at the end of a long tunnel, or else onshore with an intake canal leading up to it. At the point of withdrawal from the water body, coarse screens prevent large, potentially damaging materials such as logs being drawn into the cooling water system. The water, still containing a variety of biological and man-made materials, travels to fine screens of small mesh size that filter out most of the remaining items. Any organisms trapped on these fine screens are said to have been impinged (Vaughn, 1988), and are washed off the screens to either be returned to water body or else disposed of elsewhere. Very small organisms such as fish eggs and larvae may pass through the screens and enter the cooling water system proper, eventually being discharged back into the water body with the heated water, in a process termed entrainment (Vaughn, 1988). The present study concerns solely impingement of fish, though it is suggested that future studies into entrainment may be beneficial at Longannet Power Station (see Chapter 5).

The debate over whether entrainment and impingement may significantly affect fish populations in the source water body is not a new one. Studies into power station removal effects have been carried out since the 1960s. Henderson (2000) reviews 30 years' worth of fish impingement data from British and European North Sea coast power station studies, and notes that many of the power stations are sited in important nursery grounds for some commercial species of fish. For example, the 17 power stations operating in the southern North Sea remove an amount of plaice estimated at 50% of commercial plaice landings in the adjacent area.

Aside from potential adverse impacts on the aquatic environment, power stations offer a convenient means of ecological monitoring through sampling of impinged materials washed from intake screens (Turnpenny and Coughlan, 1992). The efficiency of the method is such that 119 of 122 known inshore species of fish have been collected from intake screens of power stations around the coast of England and Wales (Henderson, 1989). Long term monitoring of fish impingement in the Bristol Channel at Hinkley Point B Nuclear Power Station has yielded much valuable information on the ichthyofauna of that region, for example (Henderson and Seaby, 2001).

1.5. Aims of the present study

The present study has several key aims:

- To conduct an impingement sampling programme at Longannet Power Station in order to quantify losses of fish from the mid-lower Forth Estuary. Environmental factors and several water quality parameters are statistically modelled in order to determine their relative influences on impingement rate of the most commonly encountered species.
- To use impingement data in order to assess environmental impact in terms of removal of species deemed to be of particular importance, due to possessing status as commercially or recreationally exploited, socio-politically sensitive, or conservation listed.
- To examine the long term trends in benthic and demersal species abundance captured in Agassiz trawls since 1982, in order to assess any changes that may have occurred over the period and investigate why they may have occurred.
- To model statistically the long term Agassiz dataset in order to determine seasonal, spatial and tidal influences on abundance of benthic and demersal species in the mid-lower Forth Estuary.
- To provide estimates of abundance of common clupeids by novel midwater trawling, and to model statistically the data to give a preliminary assessment of the influence of season, location and tidal state on the abundance of the species in the mid-lower estuary.

- To evaluate the results obtained from impingement and trawl studies in order to examine the potential overall impact on the Forth Estuary ichthyofauna by Longannet Power Station.
- To assess the suitability of impingement collections as an alternative means of sampling the Forth Estuary fish populations by comparison with results obtained from trawling.

Chapter 2. Fish extraction by Longannet Power Station, 1999 & 2000: extent of impingement and influences on impingement rate.

2.1. Introduction

The abstraction of water for cooling by coastal and estuarine thermal power stations inevitably results in removal of aquatic organisms from the water body, including fish. Very small fish life stages such as eggs and larvae may pass through filter screens and through the entire cooling system. Larger fish are generally impinged on the filter screens, the minimum size of fish likely to be impinged being dependent on the size of mesh fitted (Turnpenny, 1981). Abundances of fish impinged annually may be of the order of millions of individuals (Langford, 1983). The extent of annual impingement seems to be directly proportional to the amount of water extracted for cooling water (CW) purposes: larger power stations abstract more water and so tend to remove more fish (Kelso and Milburn, 1979; Henderson and Seaby, 2000).

A variety of factors may affect both long-term and short-term impingement rate. These factors can be classified as those that increase likelihood of contact with the CW intake flow, and those that increase the likelihood of removal in the CW once in the vicinity of the intake. An example of the former is regular seasonal fluctuations in impingement rate attributable to migrations of species into areas that increase likelihood of contact with CW intakes at certain times of the year, as exemplified by large ingresses of clupeids overwintering in inshore marine and estuarine waters during the colder months of the year (*e.g.* Maes *et al.*, 1998b; Power *et al.*, 2000). Once near a CW intake, the sustainable swimming ability of fish is a major influence on probability of impingement, for this is the mode of escape from the inflow. Fish are ectothermic, meaning that swimming ability is reduced at low water temperatures, thus increasing impingement susceptibility (Turnpenny, 1983a,b). Any factor influencing swimming ability, *e.g.* level of endoparasitic infestation (Sprengel and Lüchtenberg, 1991), may affect escape probability. Escape from CW intakes is not solely dependent on swimming ability, but is dependent also on intake water flow pattern (horizontal causing greater escape than vertical) and ability of the fish to see the intake (Turnpenny, 1988a). Impingement rate has been shown to increase at night when visibility is reduced (*e.g.* van den Broek, 1979), but not in locations where turbidity is high making intakes largely undetectable by day and night (Turnpenny, 1988a; Henderson and Seaby, 1994). Tidal influence may introduce variation in impingement rate on a daily basis, with greatest rate of impingement often occurring during the period at or around LW (Sharman, 1969; Langford *et al.*, 1978). This may be because fish are more concentrated at LW than at HW, thus increasing the likelihood of contact with CW intakes (M.J. Attrill, Benthic Ecology Research Group, University of Plymouth, personal communication). In addition, intake drum screens are located in wells of tidally determined water depth, with greater turbulence and so increased likelihood of impingement occurring at LW (Langford, 1983). Within-month variations in impingement rate due to differences in tidal range may occur, and some standardised monthly sampling programmes aim to avoid such trends by collecting fish midway between spring and neap tides (*e.g.* Henderson and Holmes, 1991). An amplification of the concentration effect mentioned above presumably occurs during the extreme LW experienced during spring tides.

The Forth Estuary, east Scotland, possessed two thermal electricity generating stations, the now decommissioned Kincardine Power Station (KPS) and Longannet

Power Station (LPS) (Figure 1.1). Fish impingement studies were undertaken at the former by Sharman (1969), while impingement at LPS was investigated by Maitland (1997, 1998). The latter study assessed total impingement mortality in 1994 to have been approx. 2.16×10^7 fish in 1994 and 2.09×10^7 fish in 1996, based on fortnightly daylight samples of 30 min duration. Total impinged fish biomass in 1996 was estimated to be approx. 161.0 t. The author suggested these values to be underestimates due to the likely increase in impingement during darkness, preliminary evidence of which was found in 1997 (Maitland, 1998). Four species comprised approx. 90% of total impinged abundance in 1996, attributable to 44.3% herring, 27.7% sprat, 12.2% whiting and 5.7% sand goby. The first three of the species use the estuary seasonally, either to overwinter (herring and sprat) or as a nursery (whiting) (Elliott *et al.*, 1990), while sand goby is an estuarine resident (Elliott and Dewailly, 1995). In contrast to the field studies of Maitland (1997, 1998), Turnpenny (1997) used the impingement prediction software PISCES (v.3.0) to estimate mean LPS impingement at 7.4×10^5 individuals per annum, with mass 10.9 t. Six species were predicted to comprise almost 90% of the impinged fish abundance, namely whiting (27.1%), sand goby (19.1%), lesser pipefish (15.8%), flounder (10.1%), herring (8.5%) and sea snail (8.2%).

The present study aimed to investigate fully the nature of fish impingement at LPS. It was hypothesised that the total annual abundance of fish impinged at this location would be of the order predicted for a power station of similar water usage rate in the NE Atlantic. This was examined using a theoretical relationship linking water abstraction rate and total annual impingement proposed by Henderson and Seaby (2000). Possible links between environment and fish impingement rate were analysed using Generalised Linear Modelling. Likely influences on the rate of fish contact with the LPS intake were included in models, *e.g.* tide height, tidal range and season, as well as potential influences on probability of escape once in contact with the intake (number of CW pumps operational, freshwater flow rate, light presence/absence, temperature, salinity, turbidity, dissolved oxygen).

2.2. Materials and methods

2.2.1. Study site

LPS is a coal-fired thermal electricity generating station of the direct-cooled or once-through type. The plant commenced full operation in 1973. Total rated capacity of the facility is 2400 MW (approximately 40 % of Scottish demand), supplied by four 600 MW units. There are four CW pumps each capable of withdrawing approximately 22.75 m^3s^{-1} of cooling water through an intake structure 163 m from the shoreline. The intake consists of twelve vertical apertures, each 5.18 × 3.05 m, with coarse bars 7.62 cm wide set at 38.1 cm intervals to prevent large debris such as logs from entering the CW system. The total intake surface area is approximately 157.75 m^2. Water intake speed with four CW pumps working is thus given by the calculation

$$(4 \text{ pumps} \times 22.75 \text{ m}^3\text{s}^{-1}\text{pump}^{-1}) \div 157.75 \text{ m}^2 = 57.7 \text{ cm.s}^{-1}$$

Water entering the intake system is dosed with sodium hypochlorite to reduce biofouling and travels down two culverts. The water from each culvert then enters two screen wells horizontally, and is sucked vertically downwards through four rotating drum screens of diameter 16.46 m × 3.35 m width and 8 mm mesh size. Debris is collected on screen ledges, and is removed at the top of the screens' rotation by wash-water jets. The removed material travels down channels and into two trash-collection baskets, one each for screens 1 and 2 (the 'west' screens) and screens 3 and 4 ('east' screens). Trash baskets are emptied approximately every 8 hours, although this varies according to their degree of fullness. All material in the trash baskets is disposed of at an off-site landfill facility. Small items, including biological material such as fish eggs, pass through the screen mesh and enter the condenser system with the cooling water, after which they pass down a 1.6 km outflow canal and re-enter the estuary, downstream of the power station.

2.2.2. Sampling protocol

A sampling regime which was designed to include ranges of variables thought to influence impingement rate was implemented. Tide height and range was calculated for Grangemouth using the TIDECALC v.1.1 tidal prediction software system (Ministry of Defence Hydrographic Office, 1994). From January 1999 to December 2000, monthly sampling sessions were carried out at HW and LW of spring and neap tides (*i.e.* when tidal range was > 3.5m or < 3.5m, respectively; Lindsay *et al.*, 1996) in daylight or darkness. This resulted in eight sampling sessions per month, with some sessions being carried out on the same day at different stages of the tidal and diel cycles. 192 sampling sessions were thus scheduled over the 24 month study period, but 32 of these were unable to be undertaken due to blocking of channels by organic materials. This was most pronounced during July and August 2000, when inundation by scyphozoan medusae prevented sampling and also caused loss of generation capacity at LPS (T.Corless, ScottishPower plc, personal communication). A less pronounced influx of such medusae was also observed in 1999, and this is a common feature in British inshore areas during the summer months (Russell, 1970).

Each sampling session consisted of collecting fish that had been washed off the fine mesh drum screens following impingement, by using a 5-mm mesh handnet placed over the point of discharge into the trash basket. 10 × 3 min replicates were taken at a single active discharge. Excessive quantities of impinged debris occasionally prevented the intended sampling duration being undertaken. The actual time sampled was noted, and it was possible to standardise quantity of fish obtained per unit volume given knowledge of pump capacity (*i.e.* approx. 1.365×10^6 $l.min^{-1}$), number of pumps operational at the time of sampling, and duration of sampling. Water quality parameters (temperature, salinity, dissolved oxygen, turbidity and conductivity) were measured prior to sampling from the CW intake jetty, using an Horiba® U-10 Water Quality Checker multiprobe. Water samples at LW of spring tides occasionally had to be collected using a bucket, as the multiprobe cable did not reach the water from the jetty. Lack of availability of the multiprobe prior to its procurement or else during maintenance meant that all four physico-chemical water quality parameters were measured on 126 occasions. Collected fish were subsequently identified to species, enumerated and measured for total length (TL) and wet mass.

2.2.3. Potential errors associated with the sampling method

2.2.3.1. Number of samples taken during each sampling session

To assess whether the sampling regime was adequate in terms of reflecting differences in impingement rate during the various sampling periods each month (HW/LW of spring/neap tide in light/dark), a power analysis was undertaken using the nQuery Advisor 2.0 software package (Statistical Solutions, 1997). Three months were selected that included eight sampling sessions of equal sample volume and number of replicates, that were undertaken at the same location (*i.e.* east screens or west screens). The months chosen were November 1999, June 2000 and December 2000. For each month separately, a power analysis consisting of a one-way analysis of variance on eight groups of samples with equal numbers of replicates (n = 10 for each group in each month) was carried out. The significance level, α, used was 0.05, and the number of samples required to produce a power of 80% was calculated.

Table 2.1. Results of power analysis for fish impingement at LPS. Variance of means, V, = Σ(sample mean – group mean)/G. 'n per group' = number of replicates per group required to show significant differences between groups at α = 0.05 level.

	November 1999	June 2000	December 2000
Number of groups, G	8	8	8
Variance of means, V	1247.7	1045.03	41339.89
Common standard deviation, σ	37.193	35.419	209.853
Effect size (V/σ^2)	0.902	0.833	0.9387
Power (%)	80	80	80
n per group	3	4	3

Based on the data entered into the assessment, three replicates in each group sufficed to highlight differences for the November 1999 and December 2000 data, whereas an additional replicate (*i.e.* n = 4) was required for the June 2000 data. In all cases, performing the same analysis and entering 10 replicates as the 'n per group' gave a power of > 99%. The analysis suggested that the 10 replicates actually taken in each sampling session (group) were more than adequate to illustrate differences between samples taken each month, as shown in Table 2.1.

2.2.3.2. Sampling duration

Perhaps the major potential source of error in the present study was the relative brevity of the sampling method, with 30 min (or sometimes less) 'snapshots' of impingement by LPS being used. While being approximately four times the sampling intensity of previous works at this location (Maitland, 1997, 1998), this regime was still far less comprehensive than studies at other sites, *e.g.* the Indian Point nuclear generating station had all impinged fish collected, identified and counted from 1973-1977 (Barnthouse and Van Winkle, 1988). This was the most intensive example available but many studies have utilised a weekly 24-h sample, *e.g.* Turnpenny (1983a) at Fawley Power Station, and studies at other Hudson Estuary power plants (listed in Barnthouse and Van Winkle, 1988). The present study did not possess adequate manpower to attempt such ambitious sampling intensities, plus the large quantities of fish impinged at certain times of the year would have no doubt required subsampling in any case. To investigate the potential error in the routine sampling method, two 24-h surveys were undertaken in March and September 2000 at times intended to coincide with routine sampling sessions. The first session took place from 0500h on 30 March 2000 to 0400h the following day. One 10 min impingement sample was taken every hour on the hour. In addition, routine samples of 10 × 3 min were taken at 0550h, 1220h and 1830h on 30 March 2000 and at 0120h on 31 March 2000 (*i.e.* at LW in darkness, HW in daylight, LW in daylight and HW in darkness, respectively). The tide was a neap, having a range of approximately 2.34m. The total CW volume represented by the hourly sampling was approx. 6.552×10^8 l, while the routine technique represented a total of approx. 3.276×10^8 l CW sampled. Geometric mean impingement rates were 12.26 fish.$10^{-7}l^{-1}$ (*i.e.* fish per 10^7 l of CW sampled) (95% confidence intervals (CIs): 8.61 – 17.47) for hourly sampling and 9.51 fish.$10^{-7}l^{-1}$ (95% CIs: 7.32 – 12.37) for routine samples. Geometric means were used to account for the highly skewed nature of the data (Turnpenny and Henderson, 1993) and 95% CIs were calculated according to Fowler and Cohen (1990). The second 24-h session took place during a spring tide (tidal range 6.09m) and consisted of hourly 3 min samples taken between 0000h – 2300h on 28 September 2000. The hourly samples were of 3 min duration because of large numbers of fish and other materials being impinged on this day. Regular sampling took place at 0420h, 0850h, 1650h and 2100h (*i.e.* HW in darkness, LW in daylight, HW in daylight and LW in darkness, respectively) and consisted of 10 × 1 min samples, with the exception of the first three samples at 0420h, these being 2 min each. Routine sampling yielded fish from an approximate total of 1.1466×10^8 l CW. The hourly collections represented a total of 1.9656×10^8 l CW sampled. The hourly sampling technique gave a geometric mean impingement rate of 625.58 fish.$10^{-7}l^{-1}$ (95% CIs: 453.34 – 863.27), while the routine sampling session's mean was slightly lower, at 594.22 fish.$10^{-7}l^{-1}$ (95% CIs: 434.04 – 813.49). It can be seen that in both cases routine sampling produced an underestimate of fish impingement rate compared with hourly sampling, but that the 95% confidence intervals of the mean hourly impingement rates included the routine sampling means and so reasonable confidence in the routine method was assumed.

2.2.3.3. Residence time within system and sampling efficiency

Another likely source of error in the sampling procedure was lack of knowledge concerning residence time of fish within the cooling water system, *i.e.* time from entering the intake in the estuary to appearance in the hand net samples. It was felt that use of fluorescent dye or inert objects such as oranges and marked dead fish would not accurately represent the potential for impingement avoidance by swimming of live fish. It was also deemed unethical to release live marked fish into the system to assess residence time, however. Therefore a literature review was undertaken to obtain approximate estimates of residence time in similar systems. Maes (2000) released 246 live goldfish, *Carassius auratus*, into the inlet system of the Doel nuclear power plant, Zeeschelde estuary, Belgium. 69% of the fish were captured in impingement samples after 20 min and 80% after 1 h. The length of the inlet culvert was 540m (*c.f.* 163m at LPS). The residence time at LPS was likely to be only a few minutes, though it has been suggested that some fish, such as large cod, may be able to resist impingement for

30 min (C.J.L. Taylor, Nuclear Electric plc, personal communication). Residence time was likely to increase at HW, as there was a corresponding increase in water depth in drum screen wells; this increased the number of low turbulence areas that fish may be able to seek out and reduce likelihood of impingement (Langford, 1983; personal observations). During the March 2000 24 h session detailed above, the percentage of fish still alive in hourly impingement samples was noted and compared with tide height (Figure 2.1). This provided evidence that residence time was shorter at LW: the longer the time a fish spent in the system (as would occur at HW), the greater the stress it faced, due to various mechanical and chemical stressors (Langford, 1983). Fish sampled at HW were therefore more likely to have been in the system longer and so were more likely to experience mortality, shown by the greater percentage of dead fish with increasing tide height (Figure 2.1).

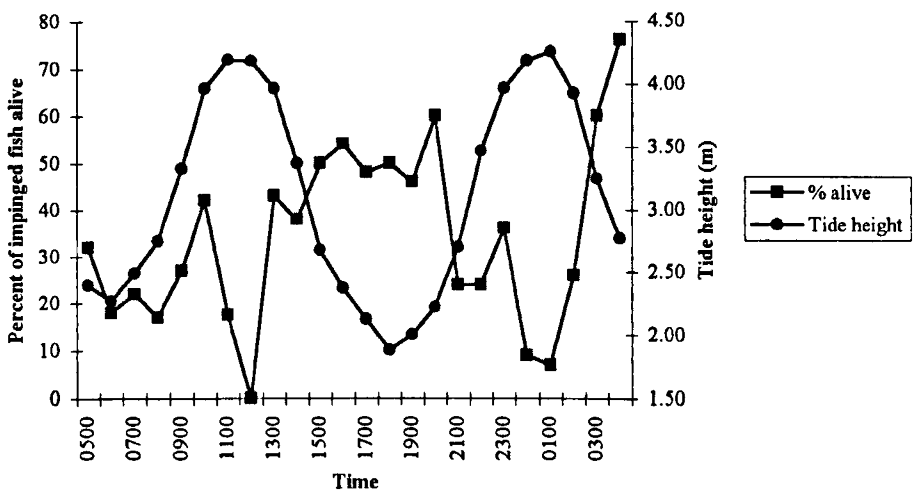


Figure 2.1. LPS: Percentage of impinged fish sampled hourly on 30-31 March 2000 that appeared to be alive, compared to tide height.

It is unlikely that all fish entering the cooling water system were impinged and displaced into trash baskets; this was illustrated by Pawson and Eaton (1999), who released 91 dye-marked juvenile sea bass into the screen well at Kingsnorth Power Station, Medway Estuary. Only 70% of the fish were recovered over the next 2 days. Since the fish were too large to penetrate the mesh of the drum screens, the authors suggest the remaining 30% that were not recorded had avoided impingement or had experienced mechanical damage that prevented identification or even that crabs living within the system had preyed upon some of the fish. In any case, this example shows that it is unlikely that all fish that are removed from the estuary by LPS will end up as impinged material on the drum screens. For the purposes of this study, it is however assumed that sampling efficiency was 100%, in terms of percentage of fish entering screen wells and subsequently being washed to trash baskets.

2.2.4. Calculation of total annual impinged fish abundance and biomass

Quantity of fish obtained in each routine sampling session was converted to number of fish per 10^7 l CW sampled. Monthly geometric mean values of fish.$10^{-7}l^{-1}$ with 95% CIs were calculated, the skew in the data having suggested that use of the arithmetic mean would have overestimated the mean rate (Turnpenny and Henderson, 1993). Where sessions had been missed, values were assigned based on interpolation of known values from that month. This was achieved by establishing mean ratios between the eight sampling sessions for months when all sessions had been undertaken. Such ratios were calculated separately for 1999 and 2000. Monthly estimates of impinged fish abundance were obtained by multiplying the mean fish impingement rate by the estimated monthly water use. Lack of detailed pump operational data meant that the average number of operational pumps observed during sampling was multiplied by mean CW abstraction rate and extrapolated to give the total monthly water use. Mass of impinged fish was estimated in a similar manner, substituting geometric mean mass per unit volume into monthly calculations. The sum of monthly fish abundance and mass in each year gave estimated annual totals of abundance and wet mass of fish impinged at LPS in 1999 and 2000. This was initially undertaken for all species together, then impinged abundance and mass was calculated separately for the ten most commonly impinged species in both years.

2.2.5. Examination of potential predictors of impingement rate

Potential predictors of fish impingement rate were assessed for statistical significance using Generalised Linear Modelling (GLM) in the software package S-PLUS (MathSoft, Inc) (Venables and Ripley, 1996). This statistical technique is increasingly popular due to flexibility in stipulation of error structure, allowing data distributions other than normal to be analysed without need for transformation of data. Overdispersion of data in the present study, caused by a few occasions when great quantities of fish were sampled, meant the data were generally suited for modelling with a negative binomial error structure. Lack of multiprobe availability to record water quality parameters during 40 sampling sessions required the separation of data into two sets. The first dataset consisted of all n=166 sampling sessions, and aimed to establish whether relationships existed between fish impingement rate and the continuous variables/factors listed as (a) in Table 2.2. Hypothesised relationships between predictors and impingement rate are included in this Table. The second dataset consisted of the n=126 samples when water quality parameters had been measured, and intended to establish the presence or absence of relationships between impingement rate and temperature, salinity, dissolved oxygen and turbidity (listed as (b) in Table 2.2).

Table 2.2. Potential predictors of fish impingement rate at LPS. (a) n=166 samples; (b) n=126 samples. * +, direct proportionality; -, inverse proportionality. † each level of the season factor represented a 3 month period, *i.e.* 1: Jan, Feb, Mar; 2: Apr, May, Jun; 3: Jul, Aug, Sep; 4: Oct, Nov, Dec. ** Freshwater flow into Forth Estuary, measured at Craigforth, Stirling (SEPA, unpublished data). ‡ 1, darkness; 2, daylight.

Predictor	Continuous variable (V) or factor (F)	Units or levels of factor	Hypothetical relationship*
(a) Season	F	1,2,3,4†	Species-dependent
Tidal range	V	m	+
Tide height	V	m	-
Freshwater flow**	V	m^3s^{-1}	Species-dependent
CW pumps operational	F	2,3,4	+
Sampling effort	V	m^3	+
Light	F	1,2‡	-
(b) Temperature	V	°C	-
Salinity	V	psu	-
Dissolved oxygen	V	$mg.l^{-1}$	-
Turbidity	V	NTU	+

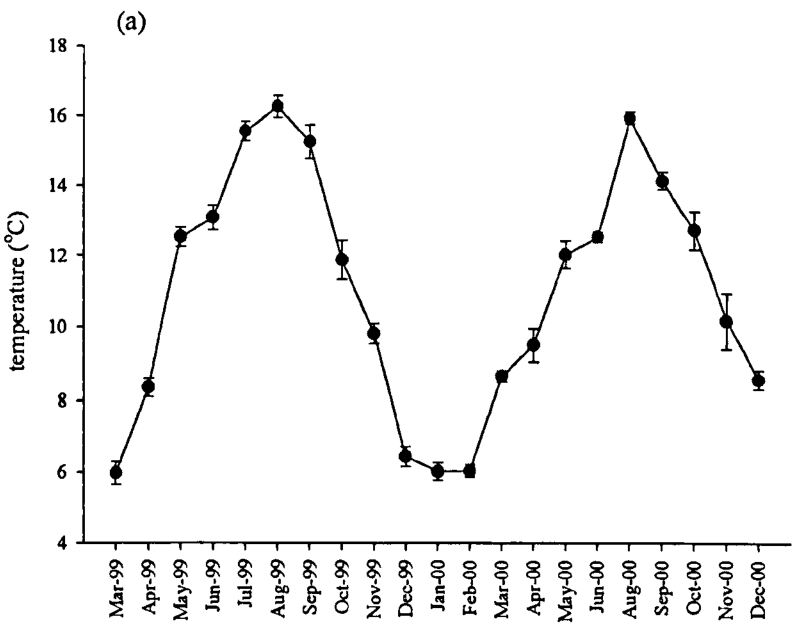
Both datasets used raw fish counts as opposed to data standardised to unit volume. Sampling effort was included as a predictor variable in GLMs to account for differences in volume of CW assessed (Table 2.2). The analysis was carried out for all species together then subsequently for each of the ten most commonly impinged species in 1999 – 2000. Predictors were assessed for significance at the 95% confidence level ($p<0.05$) and were eliminated by a stepwise deletion procedure if found to be insignificant. Data were assessed for presence of a negative binomial distribution using an S-PLUS function supplied by K. Wilson, Department of Biological Sciences, University of Stirling. GLMs thus utilised a negative binomial distribution function if appropriate, otherwise quasi-likelihood estimation with log link and variance μ was assumed. Predictors significant at the 95% confidence level were assessed for the nature of their relationship with number of fish impinged by visual inspection of plots of GLM partial residuals. Relative importance of significant predictors was established by assessment of deviance explained per degree of freedom. Interactions were not investigated, as preliminary analysis suggested them to be of little importance, and computational errors tended to occur with attempts to include excessive number of terms in GLMs.

2.3. Results

2.3.1 Water quality parameters near the LPS water intake

Results of the water quality parameter measurements showed that temperature averaged about 6°C in the coldest months of January and February 2000, whilst approaching 16°C in August of both years (Figure 2.2a). Salinity fluctuated rather irregularly compared with temperature, such a pattern being caused by missing samples either at LW or HW skewing monthly means high or low respectively (Figure 2.2b) and differences in freshwater flow (Figure 2.2e and see Table 2.9). In general, salinities were typical of a mid-lower estuarine site, with monthly means ranging from about to 23 to approaching full strength seawater.

Figure 2.2. Variation in water quality parameters: (a) temperature, (b) salinity, (c) dissolved oxygen, (d) turbidity, as measured off LPS intake jetty (values are means ± SE); (e) daily freshwater flow values recorded at Craigforth, Stirling (SEPA, unpublished data).



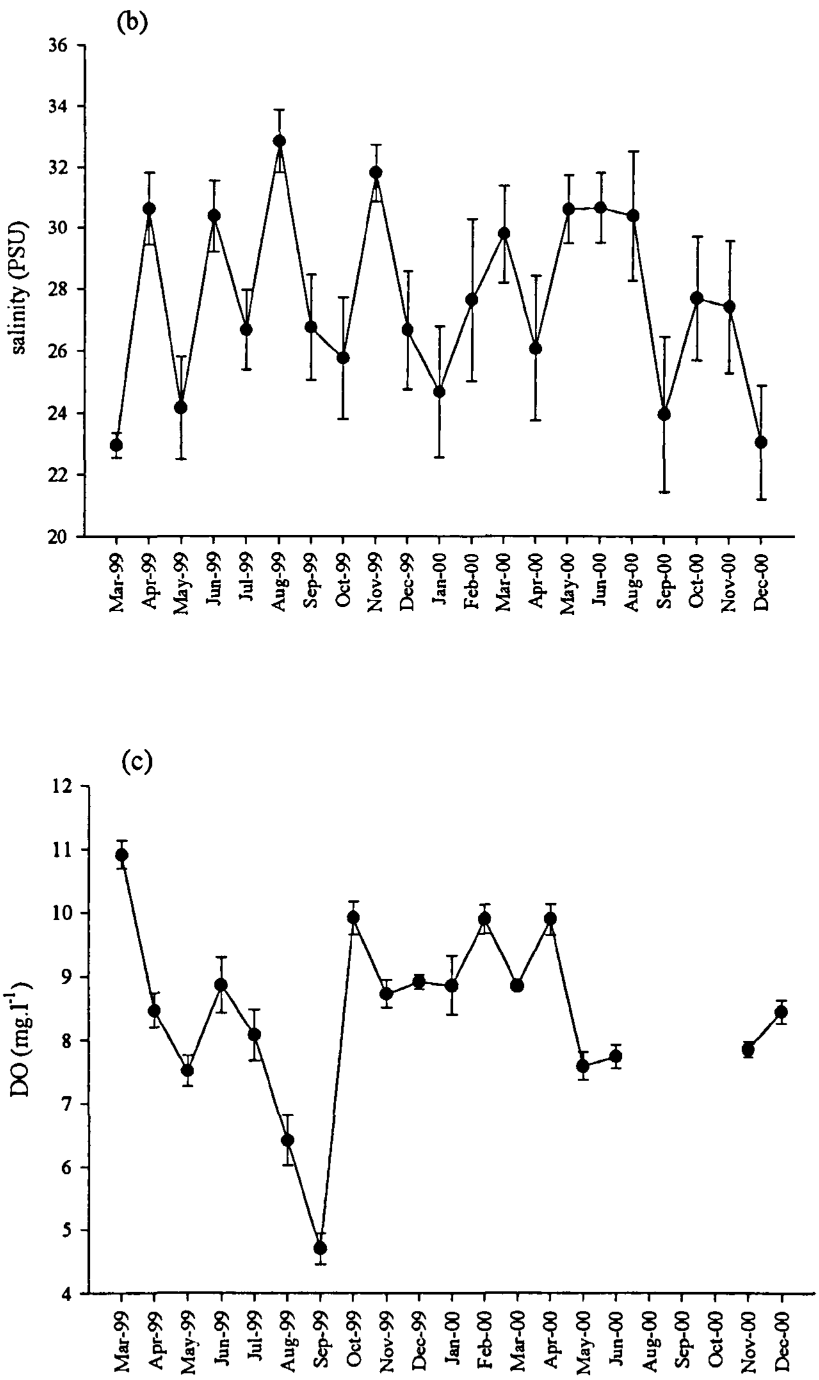
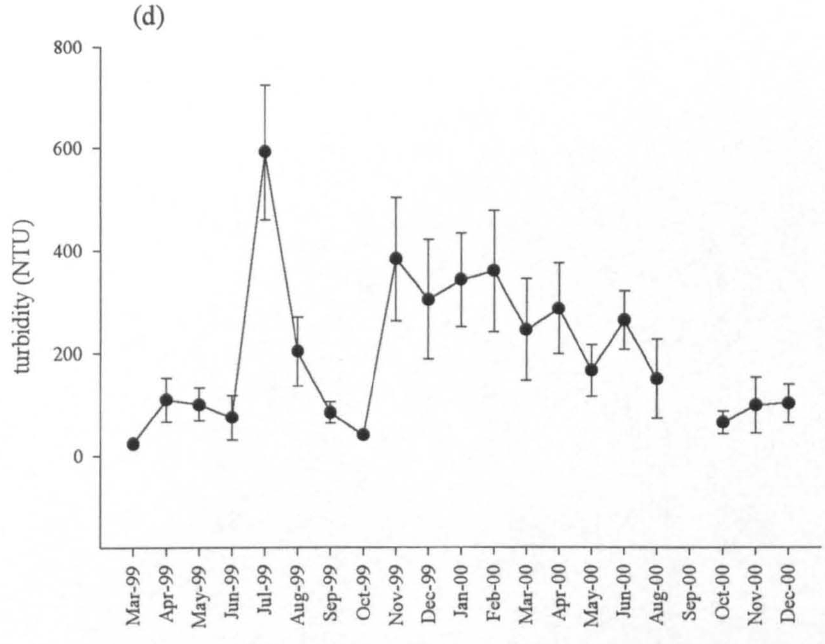


Figure 2.2 cont.



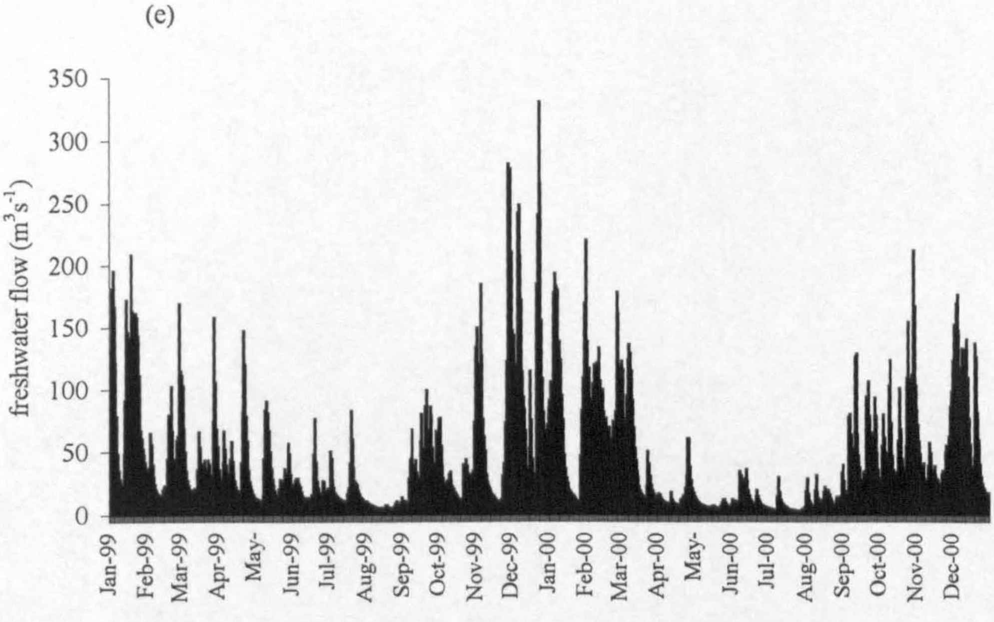


Figure 2.2. cont.

Dissolved oxygen levels were generally highest from late autumn to spring (9 – 10 mgl^{-1}), with lows in summer (5 – 8 mgl^{-1}), though the inoperation of the DO probe in summer 2000 meant data for this period were lacking (Figure 2.2c). Turbidity levels were generally very high, averaging 200 – 400 NTU for much of the study (Figure

2.2d). Freshwater flow, as measured at the estuary head (Craigforth, Stirling), was greatest between November 1999 and March 2000, with minimal levels being recorded from June – September in both years of the study (Figure 2.2e).

2.3.2 Composition of LPS impingement samples, 1999 – 2000

Detailed raw data from the LPS impingement surveys are found on the CD-ROM located in Appendix 2 of the present study. A total of 35550 fish were collected in 1999, and 101981 in 2000, representing 39 species from 25 families (see Appendix for scientific nomenclature) (Table 2.3). In each year 34 species were collected, with the only differences between years resulting from single occurrences of several species that were mostly either MA species and so using the estuary irregularly (haddock, mackerel, boreal pearlside, greater sandeel, lesser weever) or else MJ species that use estuaries routinely as nurseries but are near the northerly limits of their geographic range in the Forth and so are uncommon (*e.g.* sand smelt). All species collected have been previously recorded from the Forth, with the exception of the silvery pout, an offshore gadoid normally associated with the deeper waters of the North Sea; this species was collected once in each year of the study. Of the 39 species collected, ten each were in the ER and MJ ecological guilds, eight were MA species, five were CA migrants, four were MS species and 2 were FW (Table 2.3).

Table 2.3. Species obtained by sampling intake screens at LPS, January 1999 - December 2000. n = 83 samples in both years. * 'Gobies' reflects the taxonomic uncertainty in identification of what is likely to be sand goby, *Pomatoschistus minutus*. 'Ecological guild' follows definitions proposed by Elliott and Dewailly (1995) (see Chapter 1). Habitat: P, pelagic; D, demersal; B, benthic. Species ranked according to total collected in 2000.

	1999		2000	
Species (ecological guild, habitat)	Total collected (% of all fish)	Frequency of occurrence in samples	Total collected (% of all fish)	Frequency of occurrence in samples
Sprat (MS, P)	17304 (48.7%)	98.8%	43292 (42.5%)	100%
Herring (MS, P)	8531 (24.0%)	94.0%	39157 (38.4%)	100%
Whiting (MJ, D)	3411 (9.6%)	100%	5895 (5.8%)	96.4%
'Gobies'* (ER, B)	2580 (7.3%)	81.9%	5498 (5.4%)	83.1%
Plaice (MJ, B)	1023 (2.9%)	86.7%	3292 (3.2%)	95.2%
Smelt (CA, P)	526 (1.5%)	63.9%	1524 (1.5%)	73.5%
Lesser pipefish (ER, B)	426 (1.2%)	68.7%	841 (0.8%)	91.6%
Flounder (ER, B)	837 (2.4%)	90.4%	808 (0.8%)	92.8%
Cod (MJ, D)	289 (0.8%)	48.2%	764 (0.7%)	57.8%
Pogge (ER, B)	138 (0.4%)	50.6%	226 (0.2%)	68.7%
River lamprey (CA, B)	137 (0.4%)	56.6%	149 (0.15%)	43.4%
Dab (MJ, B)	7 (0.02%)	3.6%	100 (0.1%)	22.9%
Saithe (MJ, D)	88 (0.2%)	19.3%	76 (0.07%)	31.3%
Common sea snail (ER, B)	129 (0.4%)	33.7%	67 (0.07%)	25.3%
3-spined stickleback (FW, P)	38 (0.1%)	18.1%	66 (0.07%)	33.7%
Salmon (CA, P)	4 (0.01%)	3.6%	56 (0.05%)	6.0%
Eel (CA, P)	31 (0.09%)	21.7%	38 (0.04%)	28.9%
Sea trout (CA, P)	9 (0.03%)	9.6%	32 (0.03%)	19.3%
Lesser sandeel (ER, B)	6 (0.02%)	7.2%	31 (0.03%)	20.5%
Eelpout (ER, B)	14 (0.04%)	15.7%	17 (0.02%)	15.7%
Fatherlasher (ER, B)	6 (0.02%)	3.6%	16 (0.02%)	16.9%
Sole (MJ, B)	2 (<0.01%)	1.2%	6 (<0.01%)	6.0%
5-bearded rockling (MS, B)	1 (<0.01%)	1.2%	5 (<0.01%)	6.0%
Grey gurnard (MS, B)	1 (<0.01%)	1.2%	3 (<0.01%)	3.6%
Ling (MA, D)	1 (<0.01%)	1.2%	2 (<0.01%)	2.4%
Bib/pout (MJ, D)	0		2 (<0.01%)	2.4%
Butterfish (ER, B)	4 (0.01%)	4.8%	2 (<0.01%)	2.4%
Silvery pout (MA, D)	1 (<0.01%)	1.2%	1 (<0.01%)	1.2%
Common dragonet (MA, B)	1 (<0.01%)	1.2%	1 (<0.01%)	1.2%
Pollack (MJ, D)	1 (<0.01%)	1.2%	1 (<0.01%)	1.2%
Perch (FW, P)	2 (<0.01%)	2.4%	1 (<0.01%)	1.2%
Sand smelt (MJ, P)	0		1 (<0.01%)	1.2%
Boreal pearlside (MA, P)	0		1 (<0.01%)	1.2%
Greater sandeel (MA, B)	0		1 (<0.01%)	1.2%
Lesser weever (MA, B)	0		1 (<0.01%)	1.2%
Thick-lipped grey mullet (MS, D)	6 (0.02%)	3.6%	0	
15-spined stickleback (ER, D)	1 (<0.01%)	1.2%	0	
Haddock (MA, D)	1 (<0.01%)	1.2%	0	
Mackerel (MA, P)	1 (<0.01%)	1.2%	0	
Sea bass (MJ, D)	1 (<0.01%)	1.2%	0	

Clupeids (sprat and herring) accounted for 70 – 80% of all fish collected in both years. Sprat was more abundant than herring in both years, though there was little difference in total abundance in 2000, in contrast to the previous year when sprat were more than twice as abundant as herring. These species occurred in almost all samples taken during 1999 – 2000. Whiting and gobies ranked third and fourth in total abundance in both years of sampling but, though present in far fewer quantities than the clupeids, they occurred in a high proportion of samples. Thus whiting occurred in approximately the same proportion of samples as clupeids, while gobies were present in just over 80% of samples in both years. Plaice were the fifth most abundant taxon collected in both years, comprising about 3% of the total number of fish. The relative contribution of flounder decreased by about two-thirds between 1999 and 2000, resulting in this species moving from sixth to eighth rank, though it occurred more frequently in samples of the latter year. Smelt contributed 1.5% of total abundance in both years, but was found more frequently in samples in 2000. Another relatively abundant species that was more frequent in its occurrence in 2000 was lesser pipefish, present on over 90% of sampling occasions compared with < 70% in 1999. This species made up approximately 1% of total fish numbers in both years. The proportion of total collected abundance attributable to cod remained quite similar between years, at 0.7 – 0.8%, while frequency of occurrence increased by approximately 10% between years. Pogge and river lamprey both contributed about 0.4% of total numbers in 1999, and occurred in 50-55% of samples collected at LPS. This contrasts with the observed situation in 2000 when both species contributed half or less of the previous year's proportions; pogge were present in almost 70% of samples taken in this year, whereas river lamprey decreased to be present in just over 40% of samples. In both 1999 and 2000 the eleven most abundant species accounted for > 99% of total abundance.

2.3.3 Abundance and mass estimates of fish impingement at LPS, 1999 – 2000

It was estimated that approximately 1.09×10^7 fish were impinged in 1999 (sum of monthly minimal 95% CIs = 4.61×10^6; sum of monthly maximal 95% CIs = 2.63×10^7) with a wet mass of 75.7 t (95% CIs: 36.2 t – 160.6 t). The estimated total for 2000 was 3.29×10^7 individuals (95% CIs: $1.31 \times 10^7 - 9.03 \times 10^7$) with mass 161.3 t (66.5 t – 411.7 t). Estimates of annual abundance and mass of the ten most common species impinged are given in Table 2.4.

Table 2.4. LPS impingement, 1999-2000. Estimated annual abundances and masses of ten most commonly collected fish. Values based on mean calculations shown in bold, with ranges of sums of minimal and maximal 95% confidence intervals in parentheses.

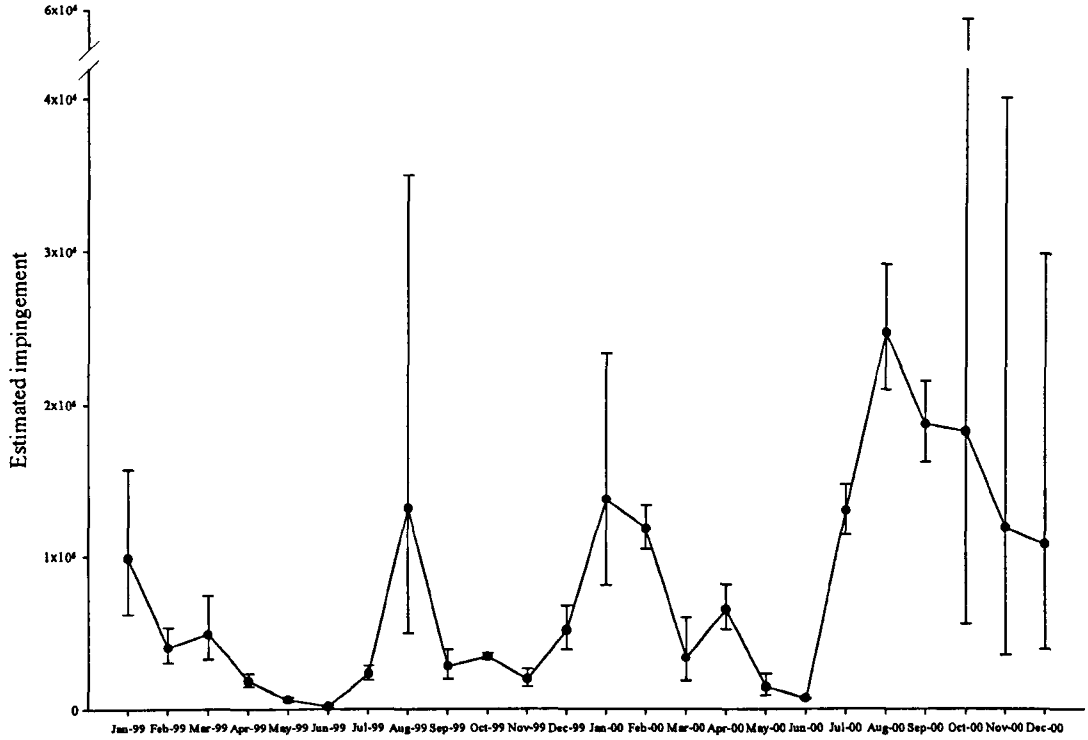
	1999		2000	
	abundance	mass (kg)	abundance	mass (kg)
sprat	**5.00×10^6** ($3.17 - 8.64 \times 10^6$)	**1.75×10^4** ($1.06 - 3.50 \times 10^4$)	**1.35×10^7** ($8.88 \times 10^6 - 2.48 \times 10^7$)	**3.92×10^4** ($2.16 - 9.15 \times 10^4$)
herring	**2.27×10^6** ($1.73 - 3.06 \times 10^6$)	**1.48×10^4** ($1.07 - 2.13 \times 10^4$)	**1.19×10^7** ($6.19 \times 10^6 - 2.43 \times 10^7$)	**4.29×10^4** ($2.34 - 9.05 \times 10^4$)
whiting	**1.21×10^6** ($7.92 \times 10^5 - 2.02 \times 10^6$)	**1.68×10^4** ($1.17 - 2.59 \times 10^4$)	**2.22×10^6** ($1.41 - 3.54 \times 10^6$)	**5.09×10^4** ($3.11 - 8.43 \times 10^4$)
gobies	**4.88×10^5** ($2.83 \times 10^5 - 1.02 \times 10^6$)	**1.29×10^3** ($5.07 \times 10^2 - 3.99 \times 10^3$)	**1.02×10^6** ($5.47 \times 10^5 - 3.07 \times 10^6$)	**2.37×10^3** ($1.01 - 9.86 \times 10^3$)
smelt	**3.24×10^5** ($1.32 \times 10^5 - 1.66 \times 10^6$)	**2.22×10^3** ($8.11 \times 10^2 - 7.78 \times 10^3$)	**3.74×10^5** ($2.45 - 6.20 \times 10^5$)	**3.12×10^3** ($1.53 - 7.34 \times 10^3$)
plaice	**3.08×10^5** ($1.93 - 5.69 \times 10^5$)	**1.92×10^3** ($1.24 - 3.36 \times 10^3$)	**1.19×10^6** ($7.04 \times 10^5 - 2.20 \times 10^6$)	**5.16×10^3** ($2.10 \times 10^3 - 1.41 \times 10^4$)
flounder	**2.73×10^5** ($1.99 - 5.29 \times 10^5$)	**5.45×10^3** ($3.30 - 1.02 \times 10^3$)	**1.74×10^6** ($1.23 - 2.56 \times 10^6$)	**9.64×10^3** ($6.31 \times 10^3 - 1.65 \times 10^4$)
lesser pipefish	**1.52×10^5** ($9.84 \times 10^4 - 2.91 \times 10^5$)	**8.84×10^1** ($4.96 \times 10^1 - 2.47 \times 10^2$)	**2.73×10^5** ($1.79 - 4.82 \times 10^5$)	**1.47×10^2** ($8.71 \times 10^1 - 3.58 \times 10^2$)
cod	**1.05×10^5** ($6.48 \times 10^4 - 2.13 \times 10^5$)	**7.69×10^2** ($4.61 \times 10^2 - 1.34 \times 10^3$)	**2.62×10^5** ($1.80 - 4.37 \times 10^5$)	**3.24×10^3** ($1.38 - 9.08 \times 10^3$)
pogge	**5.36×10^4** ($1.61 \times 10^4 - 2.45 \times 10^5$)	**2.71×10^2** ($2.33 - 3.45 \times 10^2$)	**1.02×10^5** ($4.40 \times 10^4 - 2.96 \times 10^5$)	**3.20×10^2** ($2.17 - 6.08 \times 10^2$)

Sprat, herring and whiting were estimated to be the top ranked species in terms of impinged abundance in both 1999 and 2000 (Table 2.4). Mass ranks differed, due to the relative differences in size of the species. Thus whiting was numerically subordinate to herring in 1999, but provided a greater estimated proportion of mass, while in 2000, whiting were estimated to be of greater total mass compared to both sprat and herring. Gobies were assessed as being the fourth-ranked taxon in terms of abundance impinged in 1999, though the small mean body mass meant the total mass impinged was relatively low. The most commonly encountered flatfish species, plaice and flounder, were estimated to have undergone the greatest relative differences in total annual impingement between 1999 and 2000, in the range of approx. 4-6 × greater abundance impinged in the latter year. Indeed, all of the taxa listed in Table 2.4 were estimated to have undergone an increase in total abundance impinged between 1999 and 2000, though in the case of smelt the difference was of a relatively low order. All other taxa were impinged in numbers at least twice as great as the previous year.

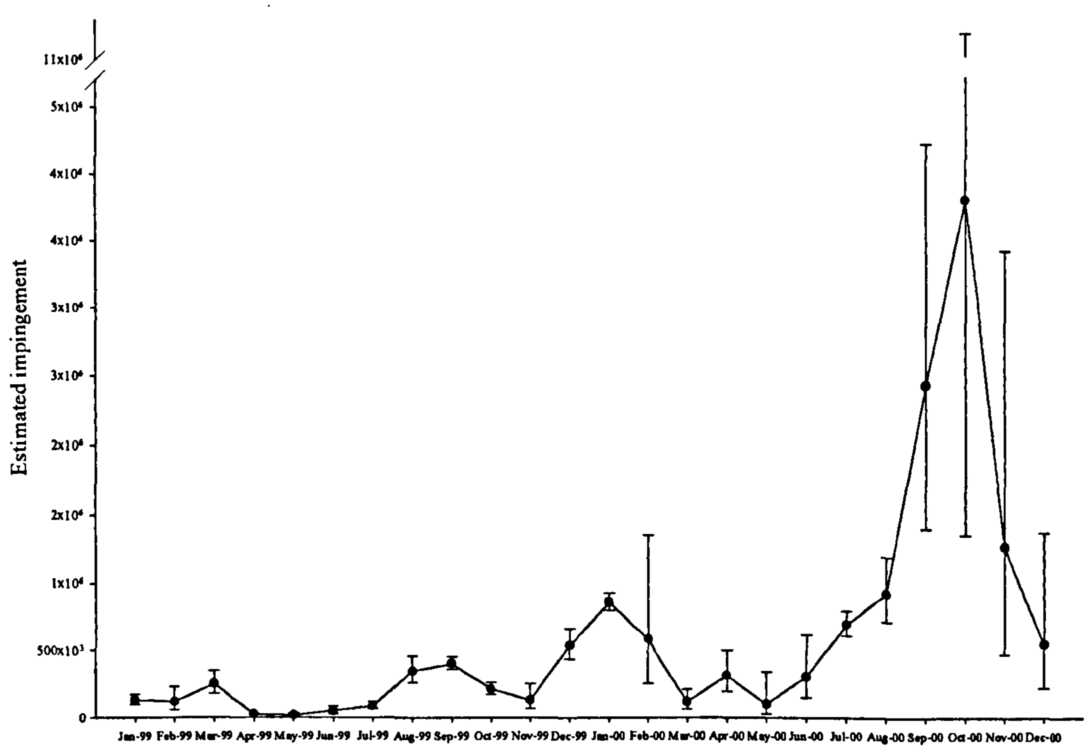
Monthly species-specific impingement varied considerably during the study period (Figure 2.3). Sprat were most abundant in August of each year, whilst generally being present in highest numbers during the cooler months of the year compared with the early summer months (fig 2.3a). Herring impingement was greatest in the final quarter of 2000, with elevated numbers also observed the previous December – February, and lowest abundances tending to be in the March – June period each year (Figure 2.3b). Whiting exhibited similar August increases as sprat, but peak estimated abundance occurred in December 2000, while the periods April – July 1999 and March – June 2000 were of greatly reduced impingement (Figure 2.3c). Estimated impingement of the 'gobies' taxon showed bimodal peaks of abundance in January and October 2000, with impingement throughout 1999 being at a much lower level than the following year, particularly in the April – July period (Figure 2.3d). Monthly impingement in smelt was estimated to have been greatest in June – July in both years, while yearly minima tended to occur from October – December (Figure 2.3e). Plaice impingement in 1999 peaked in August, and numbers from November of that year to

January 2000 were elevated compared to the first seven months of 1999 (Figure 2.3f). Peak plaice impingement was estimated to have occurred in August 2000, with numbers remaining high thereafter in relation to the early part of that year. Both years of the study showed biomodality in peaks of flounder impinged abundance: the first of these was in March – April and the second around August, with marked decreases in impingement between these periods (Figure 2.3g). Lesser pipefish impingement was greatest in March – May 1999 while the following year the same trend was observed but approximately one month later (Figure 2.3h). In both years impingement was least in July or August (Figure 2.3h). The peak in impingement abundance of cod differed somewhat between years, being July in 1999 and October in 2000, with numbers generally high over the period at, or approaching, winter (Figure 2.3i). Minimal impingement occurred in the months of May and June in each year (Figure 2.3i). Pogge impingement was highest from December 1999 – February 2000, with a secondary peak in November 2000, while yearly lows occurred in the summer – late autumn period of each year (Figure 2.3j). The influence of season on impingement rate is discussed in section 2.4.3.2.

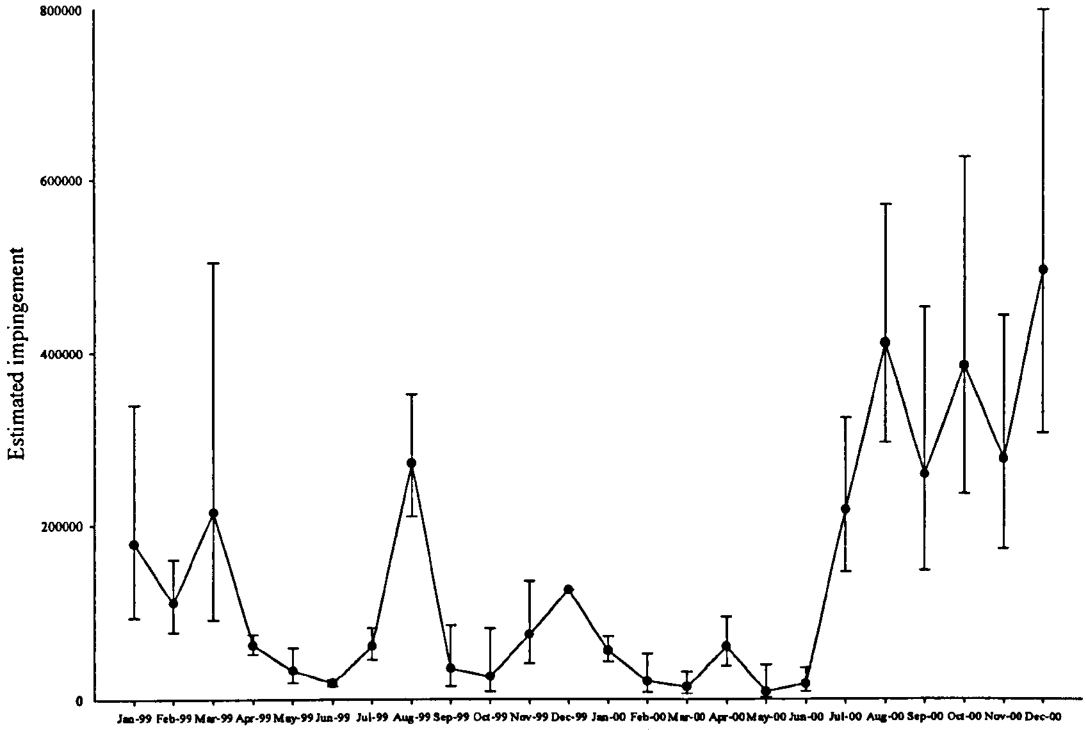
Figure 2.3. Estimated monthly impinged abundance ± 95% CIs of species impinged at LPS, 1999 – 2000.



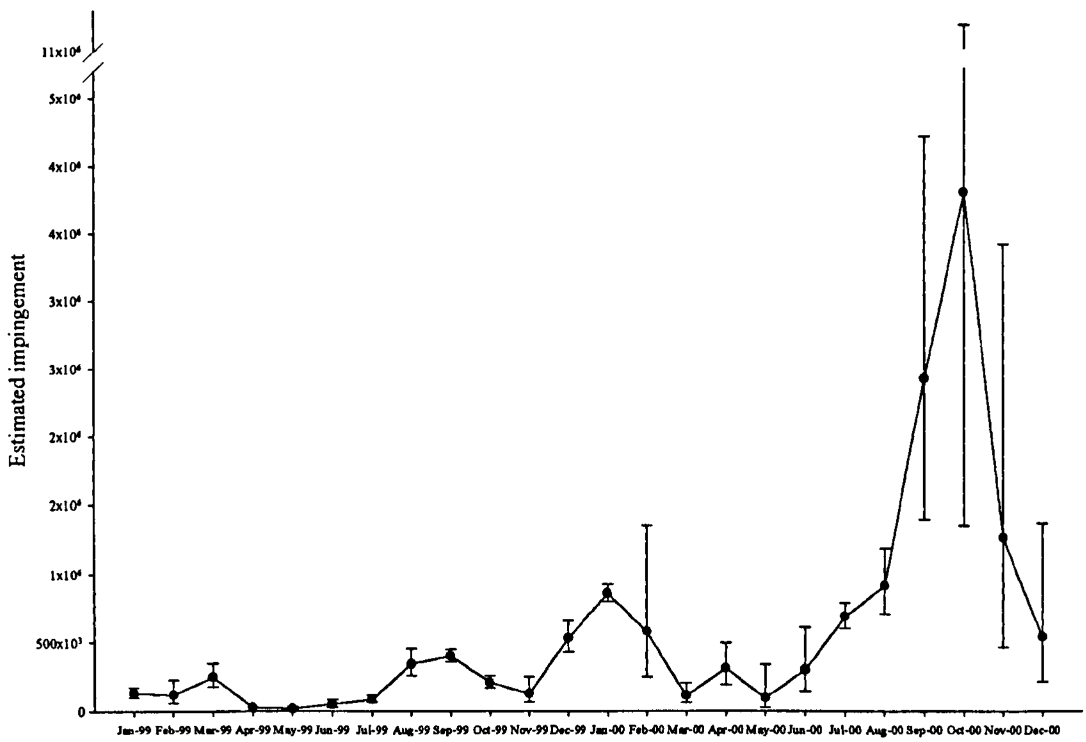
(a) sprat



(b) herring

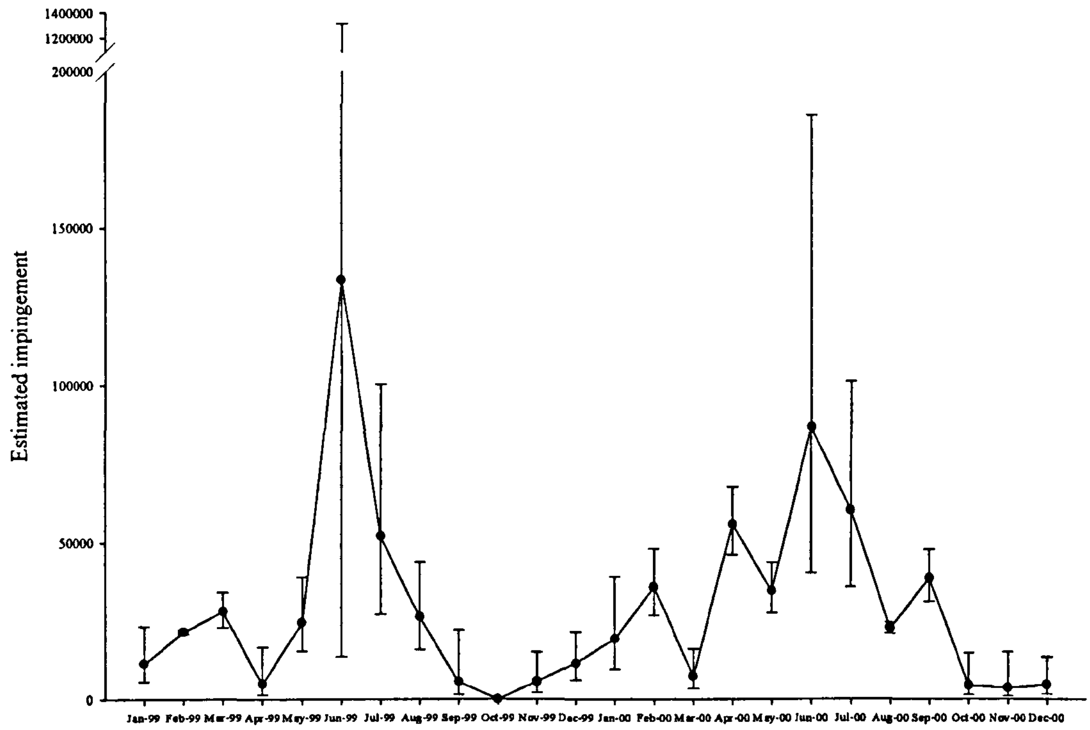


(c) whiting

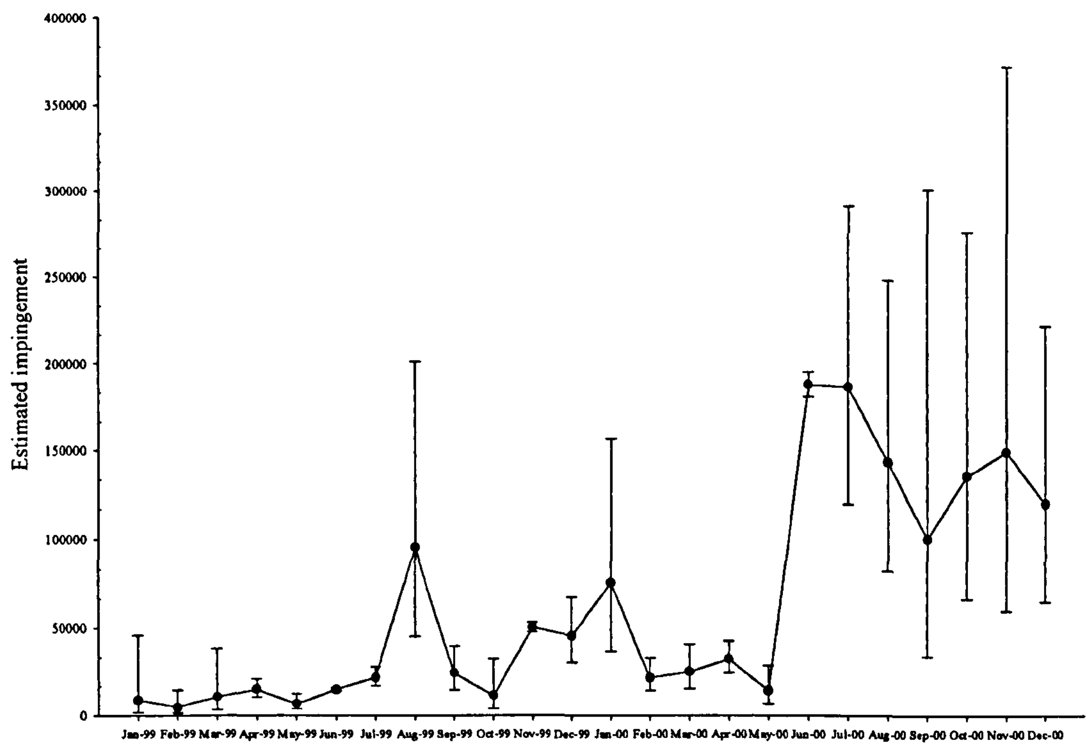


(d) gobies

Figure 2.3 cont.

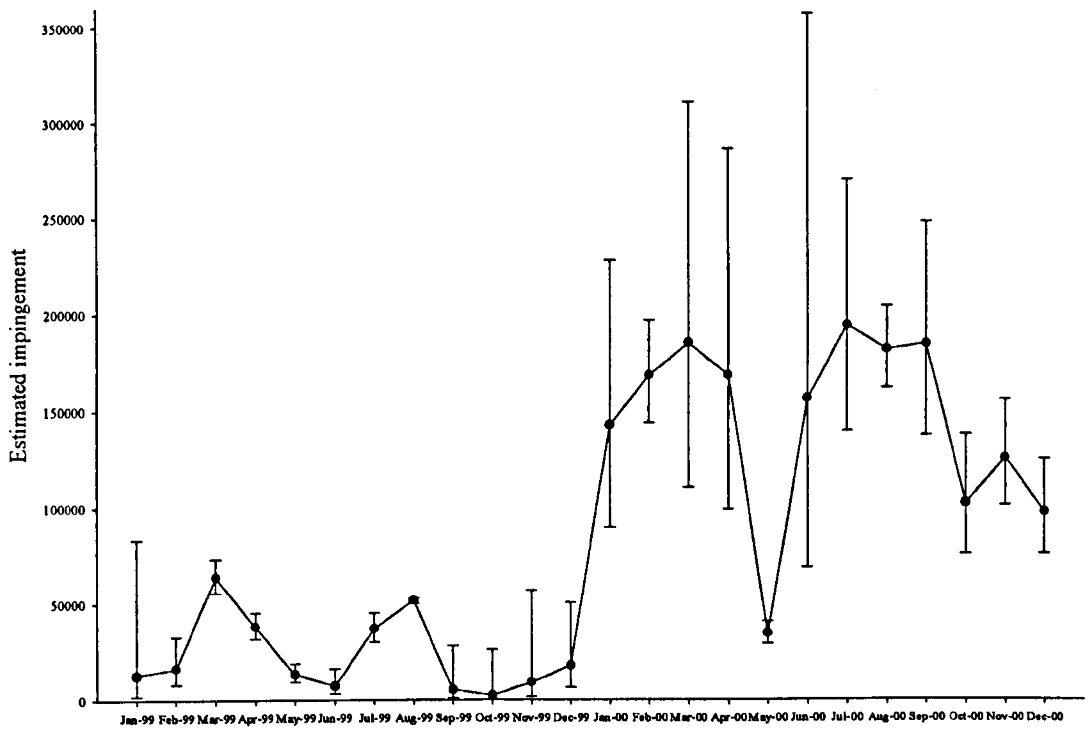


(e) smelt

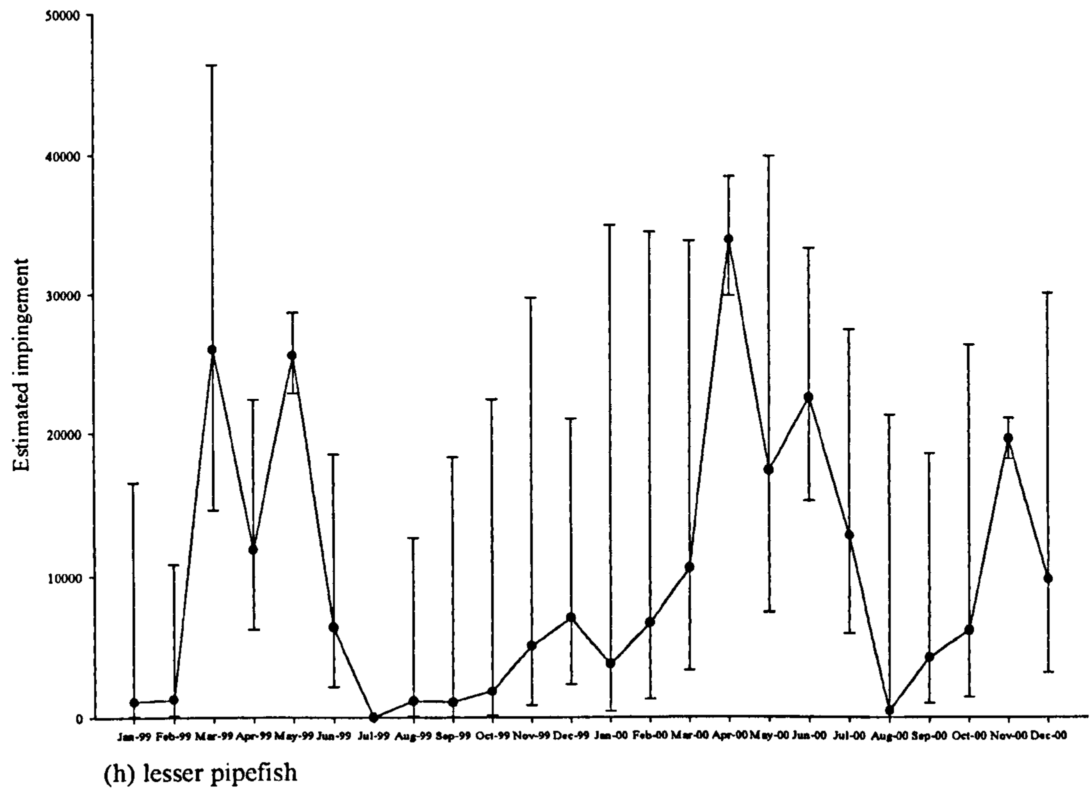


(f) plaice

Figure 2.3 cont.

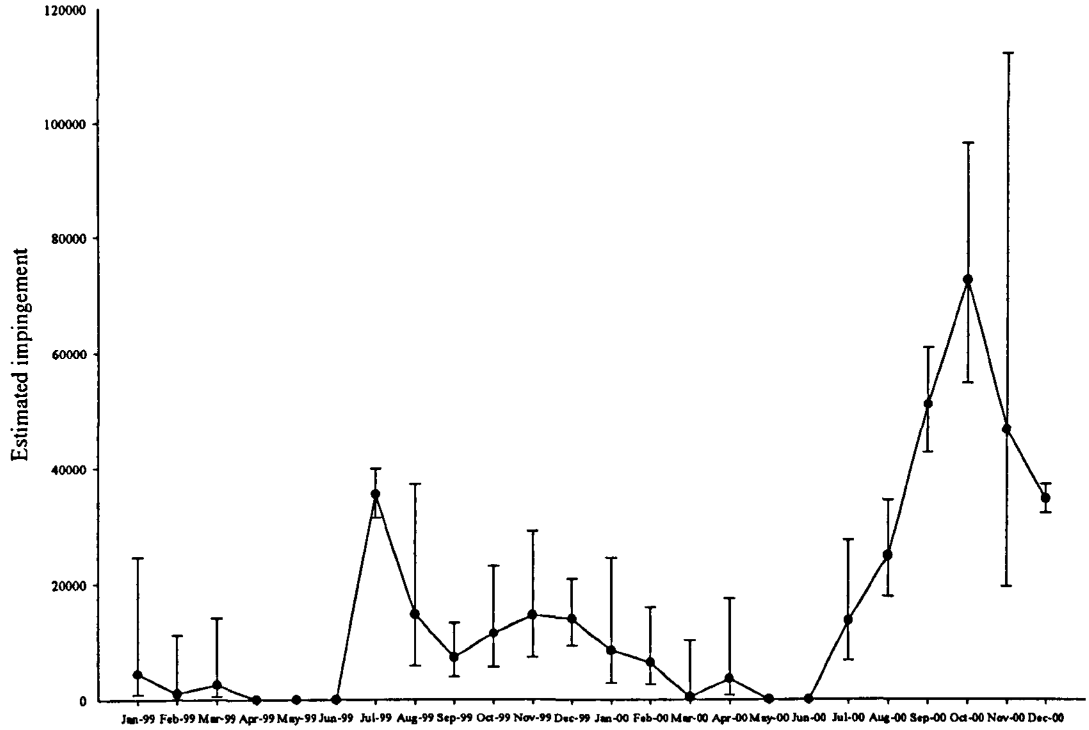


(g) flounder

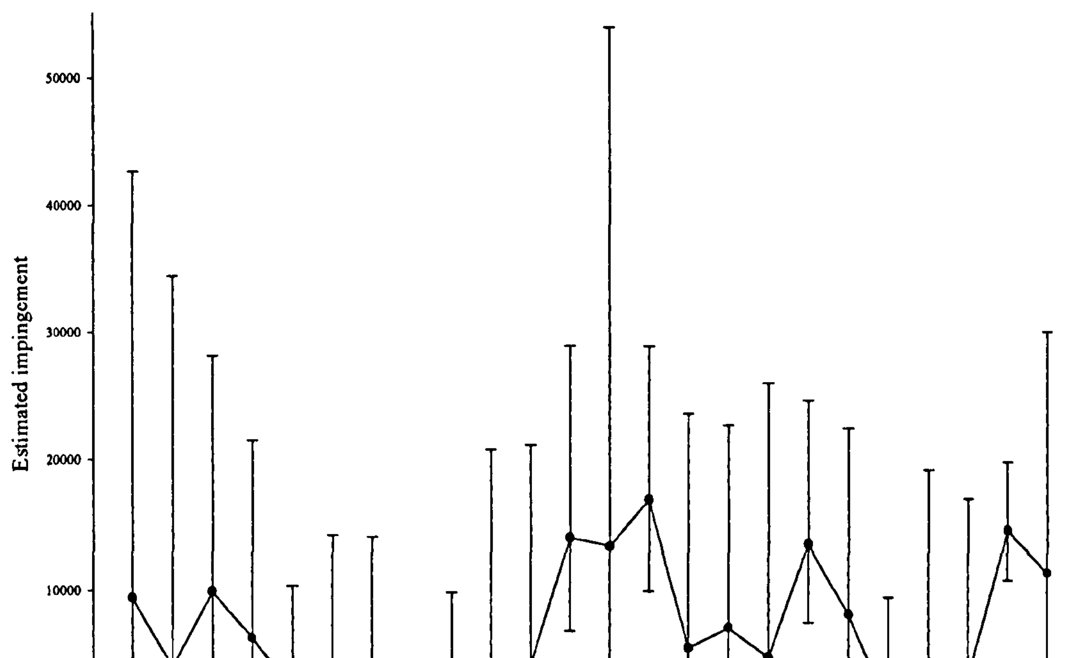


(h) lesser pipefish

Figure 2.3 cont.



(i) cod



(j) pogge

Figure 2.3 cont.

2.3.4 Predictors of fish impingement rate at LPS

In general, the predictors entered into the GLMs summarised in tables 2.5 and 2.6 explained relatively little of the null deviance. Summaries of results of GLMs of the full LPS dataset (n=166 samples) are presented in Table 2.5. When fish counts for all species were combined, tidal range and season explained the greatest quantity of deviance per degree of freedom, with the former being the most important factor overall (Table 2.5a). Tidal range was positively related to fish impingement rate. Season 3 (Jul-Sep) had a greater positive association with impingement rate than season 4 (Oct-Dec), while season 2 (Mar-Jun) exhibited a greater negative proportionality than season 1 (Jan-Mar). Freshwater flow was shown to be positively related to combined species impingement, while the number of CW pumps operational during sampling was shown to be positively related to impingement when 4 pumps were employed, but negatively related when 2 or 3 pumps were working, the former tending to be more strongly related to lower fish impingement than the latter. Tide height explained least deviance amongst significant predictors, and was inversely proportional to fish impingement rate. Sampling effort and the presence/absence of daylight were not established as significant in GLMs of all species combined.

Table 2.5. Summary of results of GLMs on n=166 dataset from LPS. Details of predictors in Table 2.2. 'd.f.' is degrees of freedom; 'deviance' is amount of deviance explained by predictor; 'significance': * = $p<0.05$, ** = $p<0.01$, *** = $p<0.001$, NS = not significant ($p>0.05$); 'relationship': + signifies direct proportionality with dependent variable, - signifies inverse proportionality, relative strengths of factor levels indicated where more than 2 levels are present. † indicates negative binomial distribution applied in GLM; ‡ indicates quasi-likelihood estimation applied in GLM. NULL signifies total deviance in null model.

(a) all species†	d.f.	deviance	significance	relationship
NULL		337.7		
season	3	65.04	***	+: 3>4 -: 2>1
tidal range	1	42.68	***	+
tide height	1	4.26	*	-
fw flow	1	11.97	***	+
pumps	2	11.65	**	+: 4 -: 2>3
effort	1		NS	
light	1		NS	

(b) sprat†	d.f.	deviance	significance	relationship
NULL		324.0		
season	3	69.91	***	+: 3>4 -: 2>1
tidal range	1	19.96	***	+
tide height	1		NS	
fw flow	1	15.76	***	+
pumps	2	15.22	***	+: 4 -: 2>3
effort	1		NS	
light	1		NS	

(c) herring†	d.f.	deviance	significance	relationship
NULL		328.3		
season	3	70.61	***	+: 3>4 -: 2>1
tidal range	1	39.34	***	+
tide height	1		NS	
fw flow	1	11.97	***	+
pumps	2	11.65	**	+: 4 -: 2>3
effort	1		NS	
light	1		NS	

Table 2.5 cont.

(d) whiting[†]	d.f.	deviance	significance	relationship
NULL		306.6		
season	3	101.14	***	+: 3>4 -: 2>1
tidal range	1		NS	
tide height	1	4.98	*	-
fw flow	1	7.80	**	+
pumps	2	11.65	NS	
effort	1		NS	
light	1		NS	

(e) plaice[†]	d.f.	deviance	significance	relationship
NULL		324.4		
season	3	29.95	***	+: 4>3 -: 2>1
tidal range	1	57.11	***	+
tide height	1	12.92	***	-
fw flow	1	7.80	*	+
pumps	2	11.65	NS	
effort	1		NS	
light	1		NS	

(f) gobies[‡]	d.f.	deviance	significance	relationship
NULL		24091.7		
season	3	5490.42	***	+: 4>1 -: 2>3
tidal range	1	6054.88	***	+
tide height	1	677.79	**	-
fw flow	1	3121.51	***	+
pumps	2		NS	
effort	1		NS	
light	1	588.74	**	-

(g) flounder[‡]	d.f.	deviance	significance	relationship
NULL		2120.4		
season	3		NS	
tidal range	1	318.63	***	+
tide height	1		NS	
fw flow	1		NS	
pumps	2		NS	
effort	1		NS	
light	1		NS	

(h) pogge[‡]	d.f.	deviance	significance	relationship
NULL		445.6		
season	3	67.43	***	+: 4>1 -: 3>2
tidal range	1	114.22	***	+
tide height	1	15.22	**	-
fw flow	1	10.76	*	+
pumps	2		NS	
effort	1	11.07	*	+
light	1		NS	

Table 2.5 cont.

(i) **smelt**‡	d.f.	deviance	significance	relationship
NULL		6386.9		
season	3	1264.04	***	+: 2 0: 1 -: 4>3
tidal range	1	1392.54	***	+
tide height	1	905.19	***	-
fw flow	1		NS	
pumps	2		NS	
effort	1	765.46	***	+
light	1		NS	

(j) **lesser pipefish**‡	d.f.	deviance	significance	relationship
NULL		2207.8		
season	3	597.80	***	+: 2>4 -: 3>1
tidal range	1		NS	
tide height	1	509.87	***	-
fw flow	1		NS	
pumps	2		NS	
effort	1	56.42	*	+
light	1		NS	

(k) **cod**‡	d.f.	deviance	significance	relationship
NULL		2434.1		
season	3	957.55	***	+: 4>3 -: 2>1
tidal range	1	288.68	***	+
tide height	1	257.47	***	-
fw flow	1	45.01	***	+
pumps	2		NS	
effort	1	33.51	**	-
light	1	31.12	**	-

Given the clupeids' numerical dominance of all species collected at LPS, it is unsurprising that the individual species GLMs for sprat and herring displayed many shared features with the GLM of all species combined (Table 2.5.a-c). Both species lacked a common predictor present in the combined model, tide height, but the remaining four predictors were the same, as was the nature of their relationships, though season explained the greatest proportion of deviance in sprat, whereas tidal range was most important in herring. Herring displayed an inversely proportional impingement rate with light intensity, though the relationship was only weakly significant. Of the more commonly occurring impinged species, only whiting lacked a significant relationship with tidal range; the significance of season was of the same nature as for the clupeids, and since around 80% of all fish impinged during 1999-2000 were sprat, herring or whiting, it is obvious why the combined species GLM was similar to these three species' individual GLMs. Whiting impingement was weakly negatively related to tide height, and increasing levels of freshwater flow tended to coincide with greater impingement of this species (Table 2.5d).

The remaining individual species GLMs on the full dataset all exhibited significant positive relationships between fish impingement abundance and tidal range, with the exception of lesser pipefish (Table 2.5e-k). Tidal range accounted for the greatest level of deviance in all of these GLMs, apart from in cod, where season was more important per degree of freedom. Where significant relationships existed between impingement rate and predictors, the nature of the relationship was as hypothesised (see Table 2.2), except for cod which possessed a weakly negative relationship between sampling effort and impingement rate. Seasonality of impingement was exhibited by most species. The impingement rates of plaice, goby, pogge and cod were shown to be most positively related to season 4 (tables 2.5e,f,h,k), the final quarter of the year, whereas lesser pipefish and smelt tended to be impinged at the greatest rate during the second quarter of the year (tables 2.5i,j). Flounder was the solitary species to possess a single significant predictor, that of tidal range (Table 2.5g).

Summarised results of GLMs of the n=126 sample dataset investigating relationship between fish impingement rate and water quality parameters are shown in Table 2.6. In all species' models significant relationships, when present, between water quality variables and fish impingement rate were always of the hypothesised nature, *i.e.*

inversely proportional for temperature, salinity and dissolved oxygen, and directly proportional for turbidity. As with the GLMs of the full dataset (see Table 2.5 above), the influence of sprat and herring on the combined species GLM is large (Table 2.6a-c). All three models show temperature and dissolved oxygen to be the only significantly related variables, with temperature explaining a greater amount of the total deviance. In the GLM of whiting impingement, dissolved oxygen was the most significantly related variable, followed by salinity and temperature, which is only weakly significant (Table 2.6d). Plaice exhibited a similar pattern, though turbidity was the weakest significant predictor rather than salinity (Table 2.6e). Temperature explained the greatest amount of deviance in gobies and pogge (Table 2.6f,h), with salinity most related to impingement rate in lesser pipefish and cod (Table 2.6j,k). Flounder impingement rate was significantly related solely to turbidity (but only very weakly) (Table 2.6g), while smelt impingement was not significantly related to any water quality parameter (Table 2.6i).

Table 2.6. Results of GLMs on n=126 dataset from LPS. Details of predictors in Table 2.2. 'd.f.' is degrees of freedom; 'deviance' is amount of deviance explained by predictor; 'significance': * = p<0.05, ** = p<0.01, *** = p<0.001, NS = not significant; 'relationship': + signifies direct proportionality with dependent variable, - signifies inverse proportionality, relative strength of factor levels indicated where more than 2 levels are present. † indicates negative binomial distribution applied in GLM; ‡ indicates quasi-likelihood estimation applied in GLM. NULL signifies total deviance in null model.

(a) all species†	**d.f.**	**deviance**	**significance**	**relationship**
NULL		192.4		
temperature	1	41.82	***	-
salinity	1		NS	
dissolved oxygen	1	31.88	***	-
turbidity	1		NS	

(b) sprat†	**d.f.**	**deviance**	**significance**	**relationship**
NULL		187.4		
temperature	1	33.57	***	-
salinity	1		NS	
dissolved oxygen	1	27.05	***	-
turbidity	1		NS	

Table 2.6 cont.

(c) herring[†]	**d.f.**	**deviance**	**significance**	**relationship**
NULL		184.8		
temperature	1	30.06	***	-
salinity	1		NS	
dissolved oxygen	1	19.03	***	-
turbidity	1		NS	

(d) whiting[†]	**d.f.**	**deviance**	**significance**	**relationship**
NULL		186.8		
temperature	1	9.65	**	-
salinity	1	11.70	***	-
dissolved oxygen	1	19.07	***	-
turbidity	1		NS	

(e) plaice[†]	**d.f.**	**deviance**	**significance**	**relationship**
NULL		169.6		
temperature	1	15.31	***	-
salinity	1		NS	
dissolved oxygen	1	20.66	***	-
turbidity	1	5.02	*	+

(f) gobies[‡]	**d.f.**	**deviance**	**significance**	**relationship**
NULL		19928.4		
temperature	1	4454.57	***	-
salinity	1	1062.44	*	-
dissolved oxygen	1	3378.25	***	-
turbidity	1		NS	

(g) flounder[†]	**d.f.**	**deviance**	**significance**	**relationship**
NULL		140.5		
temperature	1		NS	
salinity	1		NS	
dissolved oxygen	1		NS	
turbidity	1	4.47	*	+

(h) pogge[‡]	**d.f.**	**deviance**	**significance**	**relationship**
NULL		393.0		
temperature	1	121.19	***	-
salinity	1		NS	
dissolved oxygen	1	26.45	**	-
turbidity	1	20.82	**	+

(i) smelt[‡]	**d.f.**	**deviance**	**significance**	**relationship**
temperature	1		NS	
salinity	1		NS	
dissolved oxygen	1		NS	
turbidity	1		NS	

Table 2.6 cont.

(j) lesser pipefish‡	d.f.	deviance	significance	relationship
NULL		1909.1		
temperature	1		NS	
salinity	1	244.33	**	-
dissolved oxygen	1		NS	
turbidity	1		NS	

(k) cod‡	d.f.	deviance	significance	relationship
NULL		1818.1		
temperature	1	127.14	*	-
salinity	1	285.82	***	-
dissolved oxygen	1	244.44	***	-
turbidity	1		NS	

2.4 Discussion

2.4.1. Species composition of impinged LPS ichthyofauna

Species richness is inversely proportional to latitude between the poles and the tropics (Begon *et al.*, 1996). Data of impinged fish species composition at twelve marine and estuarine power stations in England and Wales analysed by Henderson (1989) suggested the relationship between species richness (S) and latitude (L) to be:

$$S = -7.85\ L + 478.8$$

Application of this equation to LPS (latitude 56° N) would predict a species richness of 39.2. The 39 species collected in 1999-2000 fulfil this prediction precisely (Table 2.3). Loss of species richness with increasing latitude is most likely to be caused by diminishing availability of ecological niches caused by the altered climatic regime (Henderson, 1989). Thus a species such as sand smelt, dependent on warm summer temperatures to build up sufficient fat reserves to overwinter (Henderson *et al.*, 1988), was present only as a single individual in LPS impingement samples (Table 2.3), but is the most commonly impinged species at Fawley Power Station on the English south coast (Henderson *et al.*, 1984). Routine sampling at LPS in 1996 by Maitland (1998) yielded 25 species, though the total annual sampling effort was approximately 25% that of the present study.

Habitat utilisation of LPS-impinged fish, in terms of favoured vertical position in the water column (pelagic, demersal or benthic), differs somewhat from predictions based on the same twelve power stations mentioned above. With 39 species present, one would predict five pelagic, 12 demersal and 22 benthic species (Henderson, 1989).

The LPS samples were of the same order for demersal species only (11 species), whereas the pelagic complement was richer almost to the same number of species as the benthic component of the impinged ichthyofauna was poorer than predicted (ten and 18 species respectively, Table 2.3). The discrepancy is not related to a large number of infrequent pelagic species occurring compared to demersal or benthic, as species contributing <0.01% of the total sampled consisted of only two pelagic species, compared to five benthic and seven demersal (Table 2.3). Mathieson *et al.* (2000) noted that pelagic species dominated total ichthyofaunal abundance in the Kincardine intertidal marsh and subtidal areas nearby, which contrasted with the numerical dominance by benthic species in the Humber, Westerschelde, Mira and Cadiz estuaries.

The major difference in the composition of impingement samples between the 1999-2000 data of the present study and those of Maitland (1997, 1998) collected during 1996, is that the most abundant species over the past two years was sprat as opposed to herring (Table 2.7).

Table 2.7. Composition of LPS impingement samples in the years 1996, 1999 and 2000. ‡Data from Maitland (1997, 1998); †data from this study.

	1996‡	**1999†**	**2000†**
Herring	44.3%	24.0%	38.4%
Sprat	27.7%	48.7%	42.5%
Whiting	12.2%	9.6%	5.8%
Sand goby	5.7%	7.3%	5.4%
Lesser pipefish	3.3%	1.2%	0.8%
Smelt	2.1%	1.5%	1.5%
Plaice	1.2%	2.9%	3.2%
Cod	1.2%	0.8%	0.7%
Flounder	1.2%	2.4%	0.8%
River lamprey	0.2%	0.4%	0.2%
	99.1%	98.8%	99.3%

Maitland's 1996 sampling coincided with the lowest North Sea sprat abundance indices since 1991, whereas values for 1999-2000 were 3 – 5× greater (ICES, 2000). Herring

North Sea spawning stock biomass has increased since 1996, and recruitment at age 0 was much greater in 1999 and 2000 than in 1996 (ICES, 2000), but it seems that greater increases in sprat abundance have been reflected in sprat becoming the dominant species in terms of impingement mortalities at LPS (Table 2.7). Sprat was the more abundant of the two common clupeids taken in the Forth between 1981 and 1989 by Elliott *et al.* (1990), whereas the impinged clupeids at Kincardine Power Station (KPS) in the 13-month period from November 1961 – November 1962 was made up of approx. 8.92×10^5 herring and 7.91×10^5 sprat (Sharman, 1969). Power *et al.* (2000) suggested that herring tend to numerically dominate in the upper reaches of estuaries in all seasons except for winter, thus at LPS, situated in the mid-lower Forth estuary, one may expect sprat to outnumber herring.

An increase in North Sea spawning stock biomass and recruitment of plaice over the past 5 years (ICES, 2001a) may have contributed to the rise in proportion of this species to approx. 3% of total impinged abundance, compared with just over 1% in 1996 (Table 2.7). Plaice ranked 5th in total abundance, a similar rank to the 6th place observed in 1961-62 at KPS (Sharman, 1969). Indeed, the major difference between the relatively similar LPS surveys of the 1990s (Table 2.7) and the KPS survey of the 1960s was that cod was assessed to be the third most commonly impinged species in the latter (Sharman, 1969). This presumably reflects the greater abundance of cod in the North Sea at that time (Hislop, 1996).

2.4.2. Extent of LPS impingement

2.4.2.1. Abundance and biomass estimates of LPS impingement, 1999 - 2000

Quantities of fish estimated impinged at LPS varied between 1999 and 2000. The 2000 estimate of approx. 3.29×10^7 fish of mass 161.3 t was 2-3 × greater than the corresponding abundance and biomass estimates for the previous year. This may be the result of a combination of factors, but those that seem most likely based on the available evidence are increased abundances of fish in the lower Forth Estuary, as shown by trawl studies (see Chapter 3) and the increased demand on LPS for electricity generation. Possible explanations for the former are explored in chapters 3 and 4. LPS generated more electricity in 2000 than in any year since full operation began in 1973, a situation that arose due to a shortfall in electrical supply caused by the inoperation of Hunterston 'B' Nuclear Power Station (L. McSporran, ScottishPower plc, personal communication). This resulted in greater annual water abstraction than in the previous year, and therefore greater removal of fish from the estuary, since water use rate is thought to positively influence the extent of impingement (section 2.4.2.2). The seemingly great difference in total quantity of fish impinged between consecutive years is not without precedent, for the Paluel Nuclear Power Station, France, was estimated to have impinged 2.04×10^9 fish in 1984, compared with 2.7×10^8 fish the following year (Henderson, 2000).

Influxes of fish at British coastal and estuarine power stations are often attributable to presence of large shoals of sprat (Turnpenny, 1983a). The Dungeness 'A' Nuclear Power Station in Kent was forced to cease generation on seven occasions between 1969 and 1980 for this very reason (Turnpenny, 1983a). Power output was diminished, though not halted, at KPS in December 1962, due to impingement of > 136 t of clupeids, mostly sprat (Sharman, 1969). Contemporary evidence for the presence of large quantities of sprat in the Forth estuary was given in February 2001, when a large shoal of sprat entered Rosyth Naval Dockyard and died through lack of sufficient oxygen (BBC Online News, 2001). The quantity of fish removed was > 8 t. Thus the 56.7 t of sprat estimated to have been impinged at LPS from Jan 1999 – Dec 2000 seemed a not inconceivable quantity. Further evidence for this order of impingement being possible is given in section 2.4.2.2. The estimated 2.73×10^5 and 1.74×10^6 flounder impinged in 1999 and 2000 respectively was comparable to the 1.13×10^6 flounder impinged at the Cordemais Power Station, Loire Estuary, France (Robin, 1991). The French data referred only to 0-group individuals, however, while LPS flounder were of a variety of ages, and the Cordemais site was in the mesohaline section of the estuary, a more preferred nursery area for juvenile flounder (Robin, 1991). This presumably explained why a power station of much lower water use (22 m^3s^{-1}) impinged comparable quantities of flounder to LPS.

The universal increase in total annual impingement exhibited by the ten most abundant taxa was likely to be a reflection of an increase in fish abundance in the Forth Estuary between 1999 and 2000, as noted above. Some evidence for similar increases being typical of the inshore areas of the British east coast is shown by comprehensive surveys carried out at various bottom trawl stations in the English inshore regions of the Humber, the Wash and the Thames (Rogers, 2000a,b,c). The results of these surveys where applicable to species commonly impinged at LPS, in each region, are summarised in Table 2.8.

Table 2.8. Summary of change in trawl catch rates per 1000 m^2 by CEFAS, September 2000, compared with September 1999 (Rogers, 2000a,b,c). Increases: +++, ≥ 100%; ++, 50-100%; +, 0-50%. Decreases: - - -, 100%; - -, 50-100%; -, 0-50%. ~, no change.

	Humber	Wash	Thames
plaice			
0-group	- -	-	-
1-group	+	+++	+
2-group	~	~	- - -
flounder	+++	+++	++
gobies	++	+	-
pogge	+++	+++	+++
whiting	+++	+++	+++

Although plaice 0- and 2-group fish declined in all three regions, the 1-group fish showed increases, and this is of significance as most plaice impinged at LPS are either 0- or 1-group individuals. Large increases in the inshore abundances of the remaining species shown in Table 2.3 suggest a common influence in the higher abundance of fish captured in 2000 compared to 1999. This is explored further in chapters 3 and 4.

2.4.2.2. Comparison of estimated LPS impingement with predictions based on water extraction rate

As water abstraction rate increases, so fish impingement also increases. Data for 89 power plants in the Great Lakes provided the relationship:

$$\log_{10}(I) = 0.414 + 1.844 \log_{10} \text{MWe capacity } (r^2 = 55\%)$$

where I = number of fish impinged per annum and MWe represents maximum electrical output (MW) (Kelso and Milburn, 1979).

Thus the relationship was not linear, but exponential. Henderson and Seaby (2000) also suggested that water abstraction rate is a good predictor of annual fish impingement rate, based on data from 13 British and European North Sea coastal and estuarine power stations. The authors noted there to be a "relative unimportance" of locality (*i.e.* coast or estuary) on the numbers impinged, and that pumping rate was of greater significance. The proposed exponential relationship was:

$$I = 9 \times 10^{-7}\, G^{3.055}\ (r^2 = 84\%)$$

where I = number of fish impinged per annum, G = maximum cooling water abstraction rate in US gall.s^{-1} (the relationship was presented in a document intended primarily for an American readership).

Data used were based on studies listed by Henderson (2000). Applying the equation to the LPS CW inflow rate of 2.41×10^4 US gall.s^{-1} (*i.e.* 91 m^3s^{-1}) gives a predicted annual impingement of approx. 2.19×10^7 fish. Impingement at LPS in 1999 and 2000 was estimated to be 1.09×10^7 and 3.29×10^7 fish respectively. The mean of these two values is 2.19×10^7, a remarkable validation of the Henderson and Seaby (2000) equation. The fit of the LPS data to the simple abstraction rate model is further illustrated in Figure 2.4.

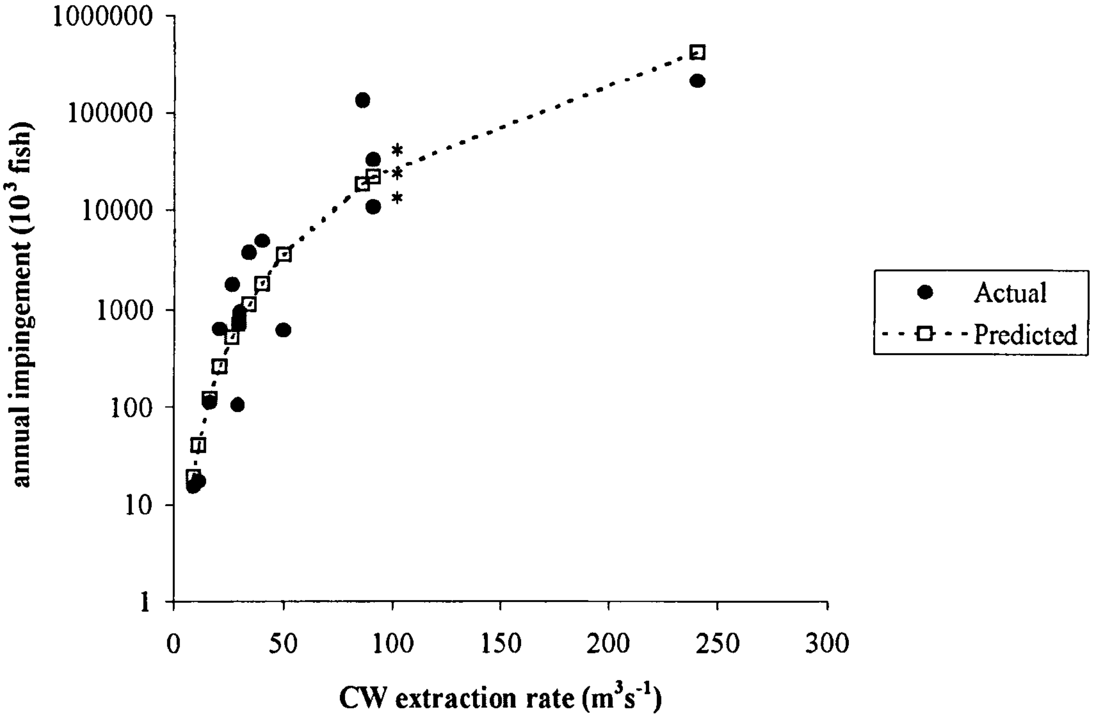


Figure 2.4. Relationship between cooling water extraction rate and annual impingement of fish for 13 British and NW European power stations, with predicted and actual values for LPS denoted by asterisks (*). Data from Henderson and Seaby (2000) and this study.

This simple analysis seems to supply good evidence for the hypothesis that LPS annual impingement of fish is of the order expected for a power station of this capacity in the NE Atlantic geographical area. It is interesting to note also that the previous estimates of impingement at LPS by Maitland (1997) were very similar, being of the order of 2.16×10^7 and 2.09×10^7 fish per annum in the years 1994 and 1996 respectively. As previously noted, these estimates were extrapolated from studies with sampling intensities 25% or less of the present study, and carried out during daylight hours only. Lack of any great difference between estimates generated from studies with and without night sampling is of significance and will be discussed further in section 2.4.3. There are some sites that are not adequately represented by the Henderson and Seaby (2000) equation, such as that of Wylfa Nuclear Power Station, Wales, where the rocky nature of the surrounding area makes the site very unusual (Henderson and Seaby, 2000). Cockenzie Power Station in the Firth of Forth impinges a minimal quantity of fish (Maitland, 1997), and this may be related to the surrounding inshore area being of notably low productivity (D.S. McLusky, University of Stirling, personal communication). The major discrepancy between the annual impinged abundance and biomass estimated from the present study and the predictions of the PISCES v.3 software, *i.e.* 7.4×10^5 individuals of mass 10.9 t (Turnpenny, 1997), is likely to be due to the software having been developed with data from power stations at lower latitudes than LPS (C.J.L. Taylor, Nuclear Electric, personal communication).

There seems to be little decline in marine productivity with latitude, as abundance of fish remains approximately equal per unit of CW abstracted over the entire annual period, as suggested by Henderson (1989). The ability of fewer species to maintain viable populations at higher latitudes, because of disappearance of ecological niches (see section 2.4.1), suggests that the remaining species may exist in greater abundance than when in competition with other species at lower latitudes.

2.4.2.3. Comparision of estimated fish biomass impinged with known values of all materials disposed to landfill

All impinged materials that are washed off the drum screens into the trash baskets at LPS are emptied into skips and ultimately buried in a landfill site, as previously mentioned. Since cost of burial is measured per tonne disposed, accurate records of wet mass of all materials buried were available for the LPS impingement study period, *i.e.* January 1999 – December 2000. A comparison of the estimated wet mass of fish impinged at LPS with total wet mass of all materials disposed to landfill is presented in Figure 2.5.

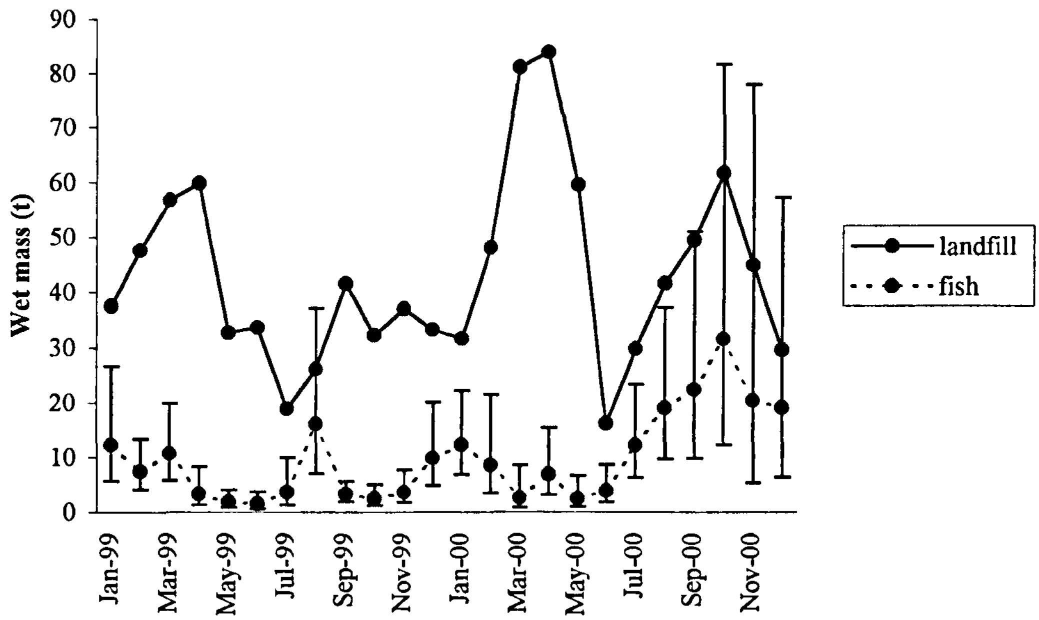


Figure 2.5. Estimated wet mass of fish impinged at LPS, January 1999 – December 2000, compared with known wet mass of all impinged materials disposed of to landfill. Fish estimates ± 95% confidence intervals.

The comparison gives confidence in the estimates of mass fish impinged at LPS that were calculated in the present study. It is apparent that in no month was the estimated impinged fish biomass greater than the total known wet mass of materials disposed of to landfill (Figure 2.5), excepting the upper 95% confidence intervals in August 1999 and September – December 2000. Proportion of total wet mass impinged that was estimated to be attributable to fish biomass tended to be least during the March – May periods in both years. This was likely to be due to the general seasonal decrease in fish impingement rate (see section 2.4.3.2) coupled with an influx of leaves that presumably were gradually washed down the Forth during the autumn and winter months (personal observations). The greater proportion of estimated fish mass in total wet mass of impinged material in the third and fourth quarters of the year was exemplified by the period July – December 2000, where the mirroring of the rise and fall of total mass impinged by fish mass is apparent. Fish were estimated to make up a relatively consistent 50% or so of the total mass during this period. The mass of all materials disposed to landfill in 1999 was 456.74 t, which increased to 578.08 t in 2000. This rise of 121.34 t was consistent given the estimated increase in fish biomass impinged and also the overall increased water extraction rate which would have been likely to increase the quantity of all materials impinged (see section 2.4.2.1).

2.4.3. Assessment of significant predictors of impingement rate at LPS

2.4.3.1. Water quality parameters as predictors of LPS impingement

Of eleven GLMs formulated for the reduced n=126 water quality dataset, temperature and dissolved oxygen were statistically significant in eight GLMs, salinity in four GLMs, and turbidity in three GLMs. The association of water quality parameters with fish impingement rate was as hypothesised, namely direct proportionality for turbidity and impingement, while temperature, dissolved oxygen and salinity were inversely related to impingement rate (Table 2.6). Caution should be applied in consideration of the results of the analyses for, as noted by Myers (1998), "It is difficult not to find environmental variables that are nominally statistically significant in an exploratory analysis". This is especially true of the following discussion and of section 2.4.3.2, where various predictors may be significantly correlated (Table 2.9).

Table 2.9. Pearson correlation coefficients for predictors of fish impingement at LPS. Associated probabilities of statistical significance: NS = not significant; * = p<0.05; ** = p<0.01; *** = p<0.001.

	turbidity	dissolved oxygen	temperature	salinity	tide height	tidal range
dissolved oxygen	0.079 NS					
temperature	-0.081 NS	-0.611 ***				
salinity	-0.111 NS	-0.131 NS	0.207 *			
tide height	-0.259 **	0.174 NS	0.018 NS	0.524 ***		
tidal range	0.368 ***	-0.318 ***	0.032 NS	0.211 *	-	
freshwater flow	0.054 NS	0.106 NS	-0.352 ***	-0.472 ***	0.007 NS	-0.232 **

Turbidity is a major influence on teleost fish behavioural ecology, as vision is well developed in most species (Guthrie and Muntz, 1993). Turbidities near the LPS intake were generally very high, ranging from a low mean monthly value of 23 NTU in March 1999 to a mean of 593 NTU in July 1999 (Figure 2.6d). A positive association between turbidity levels and quantity of fish impinged was demonstrated for three species, namely flounder, plaice and pogge. It should be noted that, in the context of generalised linear modelling, the explanatory power of these relationships was relatively weak ($p>0.001$ in all cases, Table 2.6e,g,h). The successful avoidance of removal in abstracted water requires visualisation of the CW intake to trigger the fish optomotor reflex, allowing station to be maintained against an inflow current by stabilisation of the visual field with reference to prominent components in this field (Turnpenny, 1988a). Thus greater turbidity would be likely to decrease the distance from the intake that the optomotor reflex could be triggered, exposing the fish to greater intake velocities and increasing likelihood of impingement. Intake velocity at the Doel Nuclear Power Station, Belgium, for example, increased from 6.3 cms^{-1} at 10m from the intake, to 45.8 cms^{-1} at the coarse screens (Maes *et al.*, 1998a). It can be argued that this mechanism may have been responsible for the increased impingement of these three species. However, there are additional, not necessarily exclusive, possible explanations. The visibility hypothesis outlined above is an example of a mechanism that is applicable when fish are already in the region of the intake, but turbidity may also influence the likelihood of fish being in the region of the intake. It has been suggested that one reason fish may enter estuaries is to utilise areas of high turbidity to decrease likelihood of perception by visual predators (Cyrus and Blaber, 1987; Elliott *et al.*, 1990). If the area near the LPS intake is particularly turbid compared with adjacent estuarine sites, it may be more attractive to fish. This could lead to increased likelihood of entrainment by the CW intake flow and enhance impingement. A decrease in turbidity would lead fish to seek other areas. Turbidity will tend to increase as tide height drops due to washing off of sediment from mud flats and prolonged suspension of particles in the LW channel (McLusky, 1989). This situation is augmented during the extremes of high and low water during spring tides. Thus high turbidity is linked to tide height and tidal flow. The implications of the associations of impingement rate with tide height and tidal range are discussed fully in section 2.4.3.2, and they are likely to interact with various abiotic factors, especially the water quality parameters

The salinities measured at LPS are characteristic of a mid-lower estuarine site (Figure 2.2b), approaching full strength seawater at HW, whilst generally not falling below 20 PSU at LW (McLusky, 1989). Associations between salinity and fish impingement, when statistically significant, were negative, so that impingement was enhanced at lower salinities. This was evident in whiting, goby, pipefish and cod. Most fish species present in estuaries are euryhaline (Moyle and Cech, 1996). This is especially true of the clupeids (Blaxter and Hunter, 1982). It seems unlikely that the limited range of observed salinity levels will have influenced the presence of euryhaline species in the vicinity of the LPS intake, though infrequently impinged stenohaline marine species may have been more likely to penetrate further up the estuary at higher salinities. The rarity of such species excluded them from meaningful statistical analysis. Influence of salinity on sustainable swimming speeds may have been of greater significance. Turnpenny (1983b) tested the hypothesis that salinities isotonic to fish blood, *i.e.* 17, may reduce the work that is needed to maintain fish homeostasis and so free more oxygen for sustained locomotion. This was not substantiated in investigations of critical swimming speed (CSS) of sprat and herring by the same author, as there was no effect of salinity on CSS in the range 18-33. It is tempting to interpret the results of the present study as providing evidence that marine fish are less able to escape impingement at lowered salinities, as their preferred marine conditions intuitively would allow them better physiological performance. The association of salinity with tide height, tidal range and season may better explain the significance of salinity in prediction of impingement rate, and is discussed in section 2.4.3.2.

The reaction of fish to power station CW intake flows is akin to that of the reaction to trawls: perception of the intake, followed by counter-current orientation into the flow and sustained swimming opposing the current, matching the intake velocity initially, eventually followed by fatigue and dropping back into the intake or else swimming forwards to safety (Turnpenny, 1988a). Sustainable swimming as opposed to burst swimming is used, with the latter only being employed if fish are startled by sudden movements or noise (*e.g.* throwing of objects into the water; Turnpenny, 1988a). Turnpenny (1988a) suggests that the adoption of sustained swimming appears to be due to evolved criteria for an escape response not being met in CW flows, with the result that burst swimming is not employed. Turbidity has already been mentioned in relation to intake perception, and now the effects of temperature and dissolved oxygen on sustained swimming performance are considered. Sustainable swimming is an aerobic process, and a decrease in ambient oxygen below a threshold level (Figure 2.6) has been shown to reduce sustainable swimming speed in a number of species (reviewed by Beamish, 1978).

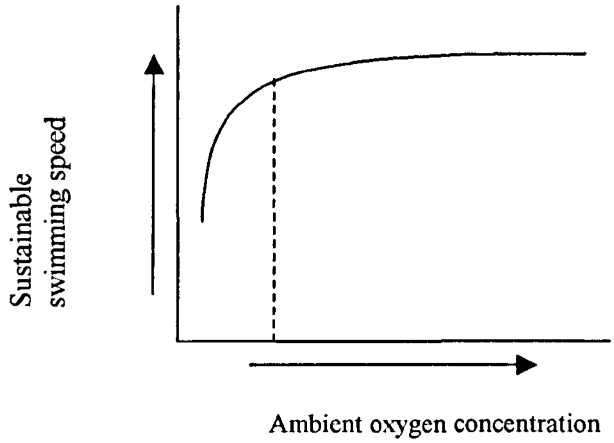


Figure 2.6. Theoretical relationship between ambient oxygen concentration and sustainable swimming speed. Dashed line indicates critical oxygen concentration below which sustained swimming speed is greatly reduced. After Beamish (1978).

It was thus hypothesised that lower concentrations of dissolved oxygen in the region of the LPS intake would increase fish impingement rate by decreasing the maximum sustainable swimming speed of fish. This was suggested to be the case in sprat, herring, gobies, whiting, cod and plaice (Table 2.6). The hypothesised reason for increased impingement at lower levels of dissolved oxygen, *i.e.* decreased sustained swimming capacity, would be valid for species that exhibit sustained swimming performance, in this case all the aforementioned species excluding gobies. The latter is a benthic taxon that moves primarily by browsing and darting, thus sustained swimming is not applicable to it (Turnpenny, 1988a). Before further discussion of the significance of dissolved oxygen to fish impingement at LPS, it is necessary to consider the importance of water temperature.

The relationship between sustainable swimming speed and temperature generally exhibits a directly proportional relationship up to a maximum, followed by a decline thereafter (Figure 2.7) (Beamish, 1978). This is due to the process being oxygen-limited, with a reliance on aerobic metabolism to produce ATP. The amount of oxygen available for activities excluding basal metabolism decreases, so that swimming performance is reduced at the upper ranges of thermal tolerance (Beamish, 1978). Turnpenny and Bamber (1983) showed that the median critical swimming speed of sand smelt increased from 2.70 – 5.73 body lengths.s^{-1} between 5.9 and 18.5°C. This would have corresponded to the ascending portion of the relationship illustrated in Figure 2.7.

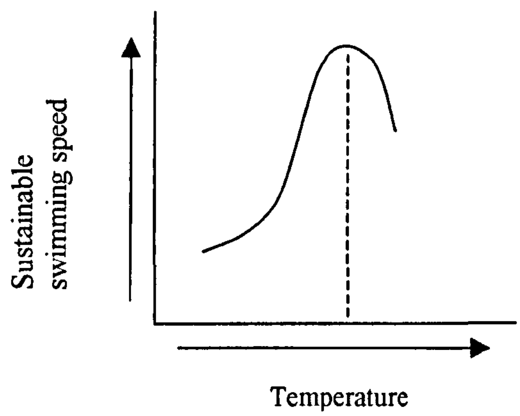


Figure 2.7. Theoretical relationship between sustainable swimming speed and water temperature (after Beamish, 1978). Temperature at which maximum sustainable swimming speed occurs is denoted by the dashed line.

The present study hypothesised that temperatures below optimum levels, being detrimental to swimming ability, would be positively related to impingement rate, and this was illustrated in GLMs of all species combined, sprat, herring, whiting, cod, goby, pogge and plaice (Table 2.6). In August 1999, temperatures averaged 16.25°C and impingement of sprat and herring was relatively great, in the order of 5.0×10^6 and 3.4×10^5 individuals respectively. If such temperatures exceeded the optimum required to achieve maximum sustainable swimming speed (Figure 2.7), then this could have explained the high level of impingement in this warm month. The LT_{50} (48h) of herring held in stock tanks of 8.7 – 11.2°C was found to be approx. 19 - 21°C by Brawn (1960), which would suggest that the herring present in the Forth at temperatures > 16°C may have exhibited reduced swimming capacity and so enhanced impingement when in the vicinity of the LPS intake. Juvenile herring (44 – 57 mm SL) tested for swimming performance at temperatures of 16.2 – 16.3°C were found to be able to sustain speeds of approx. 10 body lengths.s^{-1} (Turnpenny, 1983b). Herring are generally found at temperatures from 0 - 18°C (Brawn, 1960). The optimum temperature for swimming would therefore seem to be around 9 - 13°C, and impingement was lowest in April – June in each year (Figure 2.3b), the months when temperatures were at these levels (Figure 2.2a). This may of course also have been due to emigration from the estuary, leaving a reduced abundance present, since Power *et al.* (2000) showed that herring tend to leave the Thames Estuary at temperatures >10°C. The importance of season in influencing impingement rate is discussed in section 2.4.3.2. It could be suggested that the relatively low dissolved oxygen levels in August 1999, averaging approx. 6.42 mg.l^{-1}, may also have diminished sustained swimming ability. Dissolved oxygen levels measured in the following month were even lower, with a mean of 4.70 mg.l^{-1} (cf the suggested level of ≤ 4.5 mg.l^{-1} at which many marine fish species become stressed; Poxton and Allouse, 1982) whilst mean temperature had also actually fallen slightly. Sprat impingement was estimated at 5.9×10^5, an order of magnitude lower than the previous month. This could have been caused by the reduction in temperature enhancing swimming ability, while the reduction in dissolved oxygen did not produce a detrimental effect since it may still have been above the critical level illustrated in Figure 2.6. Herring exhibited a slight increase in estimated total impingement, to 4.0×10^5, which subsequently decreased in the following two months, whilst dissolved oxygen levels increased (Figure 2.2c). The available evidence could be interpreted as suggesting that in summer, low dissolved oxygen levels may enhance likelihood of impingement, while in winter low temperatures would cause a similar phenomenon. If summer water temperatures exceed the temperatures necessary for maximum sustainable swimming speed, then enhanced impingement may also occur in summer. As shall be seen in section 2.4.3.2, however, the complex interaction of environmental factors means that these potential influences may be difficult to isolate.

2.4.3.2. Environmental and operational predictors of LPS impingement

As with the water quality parameters of temperature, salinity, dissolved oxygen and turbidity discussed in section 2.4.3.1, the potential impingement influences of tide, season, light, freshwater flow and number of operational pumps may act to determine presence of fish in the vicinity of the LPS CW intake, or to moderate likelihood of impingement once near the intake, or both.

Of all the variables examined for significant association with impingement rate tidal range was often the most important. Tidal range was significant in nine of eleven GLMs, exceptions being for whiting and lesser pipefish (Table 2.5). It explained the greatest proportion of model deviance in seven of the nine GLMs in which it was present, including that of all species considered together. There was a direct proportionality with impingement rate. Various plausible mechanisms exist to explain this highly significant association. Ebb and flow current velocities are enhanced during spring tides, compared with neaps. In the Forth Estuary, maximum ebb and flood current velocities may be 110 and 70 cms^{-1} respectively during spring tides, whilst only 70 and 40 cms^{-1} respectively during neaps (Webb and Metcalfe, 1987). These velocities are at or above the maximum sustainable swimming speeds of most of the impinged fish sampled at LPS (Turnpenny, 1988a), and so fish would be expected to move with the tidal flow, evidence for which was given by Welsby *et al.* (1964) whilst observing shoals of clupeids using sonar from the CW intake of KPS. The greater tidal excursion associated with an increased tidal range would theoretically move more fish into the vicinity of the CW intake per unit time, thus enhancing impingement levels compared with lower tidal range and current velocities. The increased quantities of passively impinged debris during spring tides lend support to this theory. In the Severn, greatest impinged species richness occurs during spring tides, and may be attributable to increased tidal movements pushing more species into the immediate vicinity of power station intakes (P.A. Henderson, Pisces Conservation Ltd., personal communication). An alternative explanation might be that increased tidal velocities during spring tides contributed to increased CW intake flow velocities, leading to velocities exceeding those constituting the maximum sustainable swimming speed of fish. Since tidal flows are presumably largely perpendicular to the CW intake flow at LPS, this possibility is unclear and would require hydrodynamic modelling of the type discussed by Turnpenny (1988a). The fact that sampling at LPS was undertaken at slack water, with very little tidal movement, would suggest against this and the previous suggestion for effect of tidal range on impingement rate. A final possibility for positive association between tidal range and impingement rate is that extreme low waters experienced during spring tides are likely to result in a greater concentration of fish, especially in an estuary like the Forth where extensive mudflats exist, leading to increased likelihood of contact with the CW inflow. This is the mechanism believed to cause greatest impingement at Severn Estuary power stations during spring tides (P.A. Henderson, Pisces Conservation Ltd., personal communication), and was the rationale for sampling undertaken at the West Thurrock Power Station, London, as large numbers of fish were anticipated during spring tides (M.J. Attrill, University of Plymouth, personal communication). The increased numbers of leaves impinged during spring tides may be due to the extreme HW reaching farther up intertidal areas and displacing more debris than during neap tides. If there was a maximal concentrating effect of LW during spring tides, then it might be reasonable to assume that there would be a maximal diluting effect of HW springs, leading to lowest impingement at this time, something that was not observed at LPS. However, if residence time within the CW system was much greater than initially believed, then this could have resulted in many fish being impinged at HW that had actually entered the screen wells at LW or during the flood tide. The actual mechanism governing increased impingement during spring tides may be a combination of all three of the above possibilities.

Tide height itself was shown to exhibit a weakly significant inverse proportionality with impingement in all models except those of sprat, herring and flounder. The possible influence of LW in concentrating fish in the estuarine channel has been discussed in the context of differences between spring and neap tides. Trawls on the mudflats near LPS at Torry Bay, Kinneil and Skinflats by Elliott and Taylor (1989) revealed flounder to be the dominant species utilising the intertidal area, taken on 94% of occasions. Sprat and herring were taken on 35% and 53% of occasions respectively. The lack of a significant relationship between impingement rate and tide height in these species tends to undermine the argument for a concentration effect of more fish having been impinged at LW due to increased contact with the intake. Maes (2000) also observed no effect of tidal state (HW/LW/ebb/flood) on impingement rate of sprat and herring at the Doel Nuclear Power Station, upper Zeeschelde estuary, Belgium. Most benthic species were impinged in significantly greater numbers at HW and ebb tide at Doel, suggested by Maes (2000) to be due to their utilisation of intertidal mudflats close to CW intake, which required close passage to the intake to be reached.

An alternative explanation for generally enhanced LW impingement at LPS might be the actual behaviour of fish in CW filter screen wells following removal from the estuary. Water height in the wells varies with tide height, with the proximity to the screen surface increasing as tide height falls. As turbulence in the water of the well increases with the approach of LW, there is greater probability of impingement due to currents exceeding maximum sustainable swimming speeds (Langford, 1983). At HW the opposite is true, and "quiet areas" are more plentiful, sometimes allowing even dead, floating fish to apparently avoid impingement (personal observations). As duration of residence within the CW system may not differ greatly between HW and LW, this may explain the relative weakness of the statistical significance of tide height. Tide height was the most important variable associated with impingement of lesser pipefish (Table 2.5j). The preference of this species for water of < 180cm depth (Wheeler, 1969) may be evidence that a retreat from the intertidal area with ebb tides increases likelihood of contact with the LPS CW intake.

Light was hypothesised to be inversely proportional to rate of impingement, since decreased visibility at low light levels would have hindered fish perception of the CW intake structure and decreased escape potential, a situation analogous to the situation discussed with regard to turbidity in section 2.4.3.1. This phenomenon was observed at Kingsnorth Power Station by van den Broek (1979), who recorded impinged abundances three times greater by night than by day, and suggested that some benthic species are more active by night, thus are more likely to be impinged than when buried in the substrate. Light intensity influence on catch rate was investigated in the highly turbid Zeeschelde estuary for both a power station CW intake (Maes, 2000) and stow nets (Maes *et al.*, 1999a). The former study showed significantly greater impingement of sand goby and sprat at night than by day, while six other species, including herring, lacked any significant differences. Only four of 34 species exhibited greater catch abundance by night in the latter study, including dab and flounder (Maes *et al.*, 1999a). Where significant associations were demonstrated in the present study, *i.e.* in the case of herring, gobies and cod, the relationship was indeed as hypothesised, but was only weakly statistically significant (Table 2.5c,f,k). In the majority of GLMs, light was removed at an early stage in the stepwise deletion procedure. It seems that the generally high turbidity levels near the LPS intake may obscure visibility sufficiently to greatly reduce the possible effects of light on intake perception. This effect has been suggested to account for lack of day-night differences in impingement in the Severn Estuary (Turnpenny, 1988a). The behaviour of herring and cod alters at night, and may have contributed slightly to increased impingement during darkness. Herring shoals tend to break up at night, and individual fish tend to decrease swimming speed (Blaxter and Batty, 1990). The combination of dispersal over a greater area, combined with reduced activity may increase both the likelihood of contact with the CW intake and also decrease the ability of fish to resist removal in the water flow. Cod shoals also disperse at night (Wheeler, 1969) and could be prone to the former effect too. It is notable that Maitland (1998) suggested enhanced impingement occurred at LPS during the hours of darkness, based on several 24-h surveys undertaken in 1997. The present study suggests that light is not a significant factor in determining impingement rate, which would explain the agreement of estimates of total annual impingement calculated using sampling undertaken only during daylight hours (Maitland, 1997) and a combination of day and night samples (the present study), as highlighted in section 2.4.2.2.

Number of cooling water pumps operational was shown to be a significant predictor of magnitude of impingement at Fawley Power Station by Turnpenny (1983a), and the direct proportionality between CW abstraction rate and total annual impingement was investigated in section 2.4.2.2. Maitland (1997, 1998) suggested that increased electricity demand during the colder months of the year led to greater generation and water abstraction, and that this led to increased impingement more than any other influence. It was hypothesised that impingement rate at LPS would be directly proportional to number of CW pumps operating. Data from the present study suggested that operation of four CW pumps was directly proportional to impingement rate, while operation of two or three pumps was inversely proportional. This was demonstrated in GLMs of all species together, as well as for sprat and herring (Table 2.5a,b,c). A similar quantity of water was sampled in each sampling session, evidence of which is given by the fact that sampling effort was only weakly significant in models of pogge, smelt, lesser pipefish and cod impingement (Table 2.5h-k). Therefore the influence of pump rate may have been due to increased CW intake velocity with more pumps working, estimated at 57.7 cms^{-1} with four pumps operational, and falling to 43.3 cms^{-1} with three pumps. Increased intake velocities would have been more likely to exceed maximum sustainable swimming speeds, especially if in combination with high levels of turbidity. The greater need for four CW pumps operating during the winter, when fish are likely to be present in greater numbers due to inshore migrations and when temperature-dependent swimming ability is reduced, urges the relatively weak explanatory power of number of pumps operational to be treated with caution.

Season was the most frequently occurring significant environmental variable in the modelling, present in ten of eleven GLMs undertaken, and explained the greatest proportion of deviance per d.f. in analyses of sprat, whiting and cod impingement (Table 2.5b,d,k). First and foremost, fish must be in the vicinity of the intake to stand any chance of removal in the abstracted CW. Seasonality in impinged abundance is evident in a variety of species, including those present in the estuary throughout the year or else for limited periods. The latter include marine species using the estuary opportunistically (Potter *et al.*, 1997) as an overwintering ground (sprat, herring) or else as a nursery (whiting, cod, plaice) (Elliott *et al.*, 1990). The tendency of the clupeid species to enter the estuary mostly in the cooler months of the year may be an effort to reduce metabolic rate at a time when feeding is minimal (Elliott *et al.*, 1990), since the estuary is colder than the sea in winter. Abundant estuarine food resources are likely to attract juvenile gadoids, and although there is major overlap in cod and whiting's food preferences, the present data suggest the final quarter of the year to be a greater positive influence on the presence of cod, whereas the period from July to September is most positively associated with whiting impingement. This may be a reflection of a temporal partitioning of resources between these two ecologically similar species, as suggested by Elliott *et al.* (1990). The importance of estuarine turbidity in concealing juveniles from predators has already been discussed. Pogge, termed an estuarine resident by Elliott and Dewailly (1995), exhibited a positive association between impingement abundance and the seasonal factors encompassing the period between October and March, while there was a negative association during the remainder of the year (Table 2.5h). This seasonality was previously noted for pogge only in the southern part of the British range (Wheeler, 1969), so this may be evidence of offshore-inshore migrations occurring in the Forth, possibly due to a warming of the climate. The potential climate-related changes in the Forth ichthyofauna are discussed further in Chapter 3. Healey (1971) noted that sand goby was abundant in the Ythan estuary from July – February, but could not identify the reason for decrease in abundance from March – June. The present study suggested a somewhat later occurrence in the lower Forth Estuary, the seasons of greatest abundance (Table 2.5f) being similar to pogge. Of the ten most commonly impinged species, smelt and lesser pipefish were unusual in having season two (April – June) as the most positively associated with impingement (Table 2.5i,j). In smelt, this is known to be the period associated with the upstream pre-spawning migration of anadromous adults (Lyle and Maitland, 1997). No seasonality in lesser pipefish estuarine occurrence is noted in the literature. Lack of season as a significant factor influencing flounder impingement is due to the peaks and troughs of impingement straddling the boundaries between seasons (Figure 2.3g). The bimodality in abundance peaks, *i.e.* in approximately March and June – July, was not noted in Forth trawl studies by Elliott and Taylor (1989). Only in June and the following several months was abundance high compared to the rest of the year.

Migrations into the LPS vicinity thus may be responsible for much of the significance attached to the season factor. The association of season with impingement rate is likely to encompass other variables that emerged as significant in their own right. Freshwater flow was a positively-related significant variable in GLMs of all species combined, sprat, herring, whiting, plaice, gobies, pogge and cod. As these species tended to be impinged during seasons when freshwater input was highest, generally from October – March, it seems that the mostly rather weak significance of the freshwater flow association may have been an artefact of the greater importance of season on impingement rate. There is a possibility that river flow into estuaries may provide olfactory cues to trigger inshore migration of marine juvenile species (Whitfield, 1999). Elevated impingement during the colder months may be caused by a combination of three environmental and operational features. Seasonal migrations of species into the Forth estuary firstly introduce fish into areas near the LPS intake. Low water temperatures and increased CW water abstraction intake rates then diminish fish escape potential based on swimming ability alone. These three factors would all be included in the levels of the factor 'season' that were part of the modelling process.

2.5. Conclusions

The composition of the impinged icthyofauna at LPS consisted of 39 species from January 1999 – December 2000. This is in accordance with the hypothetical decline in biodiversity with increasing latitude. Habitat use differed somewhat compared to inshore areas of England and Wales, with greater proportions of pelagic than benthic species, though demersal species were of similar proportions. Species composition did not differ markedly from previous studies at this location, though sprat ranked first in overall abundance, in contrast to herring's dominant position in previous years. Annual abundances of impinged fish were exactly of the order predicted for a power station of LPS water abstraction capacity in a typical NW European coastal or estuarine location. Season and tidal range were shown to be the most important predictors of impingement, being most likely to influence presence of fish in the vicinity of the LPS CW intake. Other relatively weakly associated variables were mostly those influencing escape potential of fish once near the intake, such as temperature and dissolved oxygen. Turbidity was positively related to impingement in some species, and the generally high levels of turbidity were likely to contribute to the exclusion of light as a significant factor. The present study at LPS seems to confirm various hypotheses of predictors of impingement that have been based on previous research. The relatively low sampling intensity led to statistical models with comparatively low predictive power, and increased sampling effort would presumably have increased the clarity of some of the trends observed.

Chapter 3. Fish populations in the mid- and lower Forth Estuary: temporal, tidal and spatial variations in fish abundance as assessed by demersal and pelagic trawling.

3.1. Introduction

Variation in abundance of fish may be caused by a suite of interacting factors. Interannual variability in survival of pre-adult phases, particularly of the generally more fecund marine species, influences between-year differences in abundance (*e.g.* Phillipart *et al.*, 1996). Whiting recruitment (survival to age 1; Daan *et al.*, 1990) offers a clear example: in 1968 an estimated 9.13×10^9 fish entered the various North Sea stocks, while the following year this number was reduced to 1.08×10^9 (ICES, 2001a). Thus numbers present in inshore areas, including estuaries, may differ considerably between years. Distribution of fish in such areas may be dependent on their preferences for specific features of the environment. Substratum preferences in demersal and benthic species, for example, may influence fish in estuaries to choose one location over another, *e.g.* cod and whiting tend to be associated with soft bottoms of sand, mud and fine gravel (Elliott and Dewailly, 1995). The influence of abiotic factors in determining species distribution was introduced in section 2.4.3. Migrations into estuaries at particular times of the year are a feature of many fish species, and contribute to intra-annual monthly variations in abundance. This was clearly illustrated for the Forth Estuary (Elliott *et al.*, 1990), and is a feature common in temperate estuaries (*e.g.* Day *et al.*, 1989). Use of intertidal areas by species such as plaice and flounder at HW (*e.g.* Wolff *et al.*, 1981) may reduce abundances of these fish in samples taken in the subtidal.

Interspecific differences in latitudinal range of distribution are hypothesised to influence interannual differences in abundance of fish, as has been demonstrated in other animal groups (see references in Henderson and Seaby, 1999). Populations near to the latitudinal centre of their geographic range would be expected to show less interannual variability in abundance than those nearer the periphery (Miller *et al.*, 1991), for the peripheral populations are close to specific tolerance thresholds and are more likely to encounter better adapted species, either in competition or through predation (Henderson and Seaby, 1999). Evidence of increased population stability with decreasing latitude, *e.g.* in sole (Rijnsdorp *et al.*, 1992) and long rough dab, (Walsh, 1994), suggests the theoretical relationship between stability and geographic range to be asymmetric (Phillipart *et al.*, 1998) (Figure 3.1).

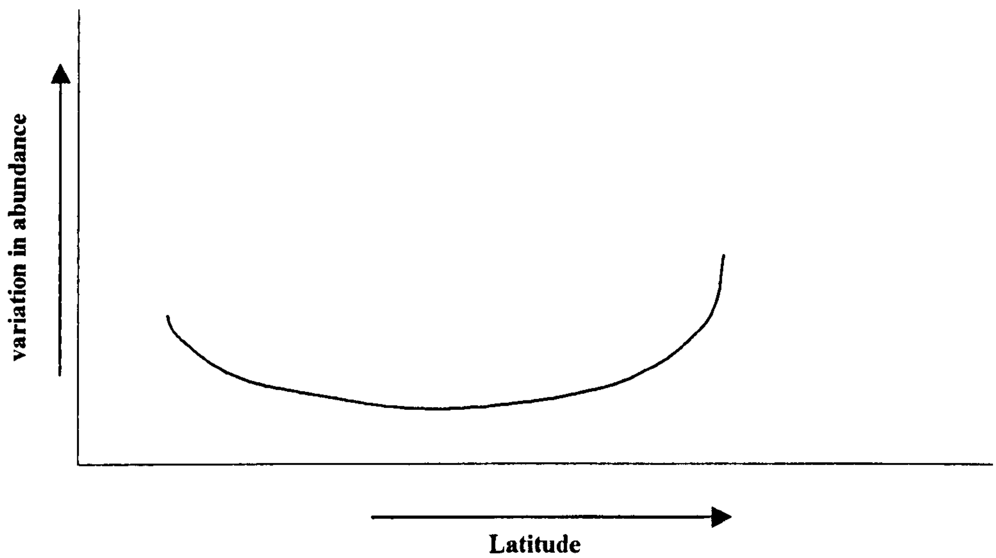


Figure 3.1. Hypothetical variation in abundance of a species in relation to latitude (after Phillipart *et al.*, 1998).

Evidence accumulated in favour of the above hypothesis thus far has been somewhat equivocal, but Henderson and Seaby (1999) suggested that comparisons between different geographical locations may be affected by lack of standardised sampling techniques.

The importance of the Forth Estuary in providing habitats for a variety of fish species was highlighted in works by Elliott and Taylor (1989) and Elliott *et al.* (1990). These studies were based on regular sampling of the Forth ichthyofauna using bottom trawls, initially monthly from October 1981 – December 1985, then with a reduced frequency of approximately every quarter until the present day. The two decades' worth of information is a valuable resource in allowing changes in the mid-lower Forth Estuary fish populations to be assessed. The present study selected monthly data from the early 1980s to conform as closely as possible to the current sampling regime. Rather than attempting to calculate absolute abundances of demersal and benthic species, as carried out by Elliott *et al.* (1990), the current study used trends in monthly abundances to examine changes in the mid-lower estuarine ichthyofauna. Simple examination of monthly plots of fish abundance illustrated trends in abundance of the frequently encountered species. Possible influences of tide height, month, and trawl station on catch abundance were modelled using the GLM technique introduced in Chapter 2. More detailed exploratory analysis of the data was then undertaken using non-parametric statistical techniques designed primarily to investigate change in marine communities. These methods, based on calculation of similarity in species abundance and percentage occurrence in the assemblage between different years, allowed the species responsible for any major changes in the ichthyofauna to be established. The hypothesis that species sampled in the Forth that were closer to the centre of their geographical ranges would exhibit less intra-annual variation than those nearer the periphery of their ranges was tested. The hypothesis was further investigated in relation to variations in population abundances of the same species sampled from the Severn

Estuary by Henderson and Seaby (1999) over a similar time period (1981 – 1998) as the present study.

Pelagic trawling was carried out in 1984 (FRPB, 1984), with reasonable success, but was discontinued. The most common pelagic species in the Forth Estuary, sprat and herring, were sampled in part by bottom trawling, and Elliott *et al.* (1990) considered the data, thus obtained, to be suitable in reflecting trends in abundance. Agassiz trawling in 1999 – 2000 captured almost no clupeids, despite LPS impingement sampling (Chapter 2) revealing that these species were present in the estuary in relative abundance. Pelagic species were therefore excluded from long-term analyses of Agassiz trawl data. Preliminary assessment of novel pelagic trawling that commenced in January 1999 was undertaken. These pelagic data were modelled in a similar fashion to the benthic and demersal data mentioned above, using GLMs, but in addition attempts were made to produce preliminary quantitative estimates of the abundance of clupeids in the mid-lower Forth Estuary. These data would be more likely to be of the true order of magnitude than those generated by Elliott *et al.* (1990) which were likely to be underestimates, based on limitations of the bottom trawls in sampling the pelagic populations.

3.2. Materials and methods

3.2.1. Agassiz trawl study

3.2.1.1. Field study

Trawling took place at three mid-lower Forth Estuary stations, Port Edgar, Tancred and Longannet (Figure 1.1), from January 1982 – December 2000, using the SEPA survey vessel (S/V), the Forth Ranger. Each sample consisted of an approx. 0.8 km haul with a 2 m Agassiz trawl with stretched mesh of 15 mm. All fish collected were identified to species, enumerated and measured for TL and wet mass. Trawling was undertaken at HW and LW in five months of the year, giving a total of 30 trawls per year, *i.e.* ten trawls at each site. Sampling in individual years took place approximately three months apart, *i.e.* in January, March/April, June/July, September/October and December, with minor exceptions. Only 24 trawls were undertaken in 1986, owing to lack of sampling in January.

3.2.1.2. Generalised Linear Modelling of Agassiz data

Abundance data for the ten most commonly captured benthic and demersal species were analysed using Generalised Linear Modelling (see section 2.2.5) to investigate various hypotheses associated with sampling conditions. In all cases n=564 trawls. LW catches were hypothesised to be greater than those at HW due to the use of mudflats by all species under investigation leading to a lower concentration of fish in the subtidal area at HW (Elliott and Taylor, 1989). Affinities of particular species with certain trawl stations were investigated without any specific hypothesis in mind.

Differences in specific abundance due to month were hypothesised to be likely to exhibit known patterns of estuarine abundance, these patterns having been assessed by previous authors (Elliott and Taylor, 1989; Elliot *et al.*, 1990; Chapter 2). Included in the models were station (levels 1,2,3 = Port Edgar, Tancred, Longannet), tide (levels 1,2 = LW, HW) and month (levels 1,2,3,4,5 = Jan, Apr, Jun, Sep, Dec approx.), as well as interactions between all possible pairs of factors (station × tide, station × month, and tide × month). Models were based on either a negative binomial distribution or quasi-likelihood estimation, as detailed in section 2.2.5. Relationships between response and predictors were assessed by visual inspection of partial residual plots. The nature of significant interactions was examined using the interaction plot function of Minitab 13 (Minitab Inc, 2000). The modelling procedure was repeated twice with random data subsets of n=282 trawls, as a means of assessing the validity of the original GLMs.

3.2.1.3. Non-parametric exploratory analysis of Agassiz trawl data

Exploratory analyses of the 19-year Agassiz trawl dataset were carried out using the PRIMER v.5 software package (PRIMER-E Ltd., 2000). These analyses aimed to elucidate any changes in the composition of the ichthyofaunal assemblage of the demersal and benthic species in the mid-lower Forth Estuary, as well as identifying the species responsible for such changes. This was achieved by treating each calendar year (January – December) as one sample and comparing similarity of the species caught in each sample in abundance and percentage terms. First, a Bray-Curtis similarity matrix of untransformed yearly abundance data was computed for all demersal and benthic species recorded during Agassiz trawling, followed by hierarchical agglomerative clustering and non-metric multidimensional scaling (MDS), the CLUSTER and MDS routines, respectively, of PRIMER (Clarke and Warwick, 1994). In this way, any changes in the ichthyofauna could initially be explored by visualisation of MDS plots, and such trends investigated further by comparing the MDS plots to groupings of years judged to be similar in terms of species composition by clustering. The MDS and cluster analyses were followed by similarity percentage analysis (SIMPER) in order to identify the main species contributing to differences between groups of years identified by clustering. This was undertaken for groups of years determined to be approx. >60% similar and >70% similar. Data for 1986 were excluded from abundance data analyses, due to the reduced sampling effort in this year. The procedure was repeated for untransformed species percentage composition data, including data for 1986.

3.2.1.3. Investigation of population size variability in relation to species latitudinal range

The approximate geographical centres of the ten most commonly caught species' ranges were determined from Wheeler (1969). These species were then ranked in order of predicted variability in population size (measured as coefficient of variation of standard deviation (CVSD) of the mean of total annual abundance), according to the hypothetical relationship between recruitment and latitude proposed by Phillipart *et al.* (1998), and this was compared to the observed values. The observed CVSDs were then compared to data for eight of these species occurring in the Severn Estuary that were sampled over a similar time period (1981 – 1998) by Henderson and Seaby (1999). The relative magnitudes of the CVSDs were assessed based on predicted differences attributable to differences in latitude of the two estuaries (56.0°N and 51.3°N for the Forth and Severn respectively) compared with species' estimated centres of geographical range.

3.2.1.4. Preliminary assessment of sampling error associated with Agassiz trawling

An assessment of the sampling error ('repeatability') of results obtained by Agassiz trawling, using data from SEPA surveys in the Tay Estuary between 13 October 1998 and 21 February 2001 obtained with the same trawl and vessel as in the Forth Estuary studies. Each haul was of 0.8 km at 2 knots. Trawling took place at two sites, Abertay and Ladyshoal. Most sampling days involved two sequential replicate hauls at each station, covering the same area, though occasionally there were three replicates, and on one occasion six (Ladyshoal, 30 April 1999). As was expected from such few replicates, coefficients of variation (CV) in abundance of individual species were mostly > 100%, indicating large variation in abundances between replicate tows. For example, abundance of one in the first trawl, followed by zero in the second tow, gave a CV of >100%, so figures for species that were not abundant were treated with caution. Similarly, CVs of 0% were obtained in the case of there being equal numbers of fish in all trawl replicates carried out, something that only occurred in cases where a single fish or a maximum of two fish were caught in each trawl. Figure 3.2 illustrates the problem of assessing sampling error of the Agassiz gear in the present study.

Low total abundance of fish will often produce a CV of 0% or 100% (DeAlteris *et al.*, 1989): a wide variety of CVs occurred at low abundances of fish caught in replicate trawls (Figure 3.2a-d). Figure 3.2b and 3.2d give some scant evidence for sampling error being reduced in whiting and cod as total abundance of fish caught in the trawls increases. This interpretation should be viewed with caution, however, as it is based on data collected on very few days, and each of these days only usually involved twin replicated tows at each site. Trawling in the Forth generally yielded more fish than trawls undertaken in the Tay. Thorough testing of sampling error of both Agassiz and pelagic trawls in the Forth Estuary is yet to be carried out. Suffice to say that all previous studies in this area (*e.g.* Elliott and Taylor, 1989; Elliott *et al.*, 1990) were subject to the same sampling error.

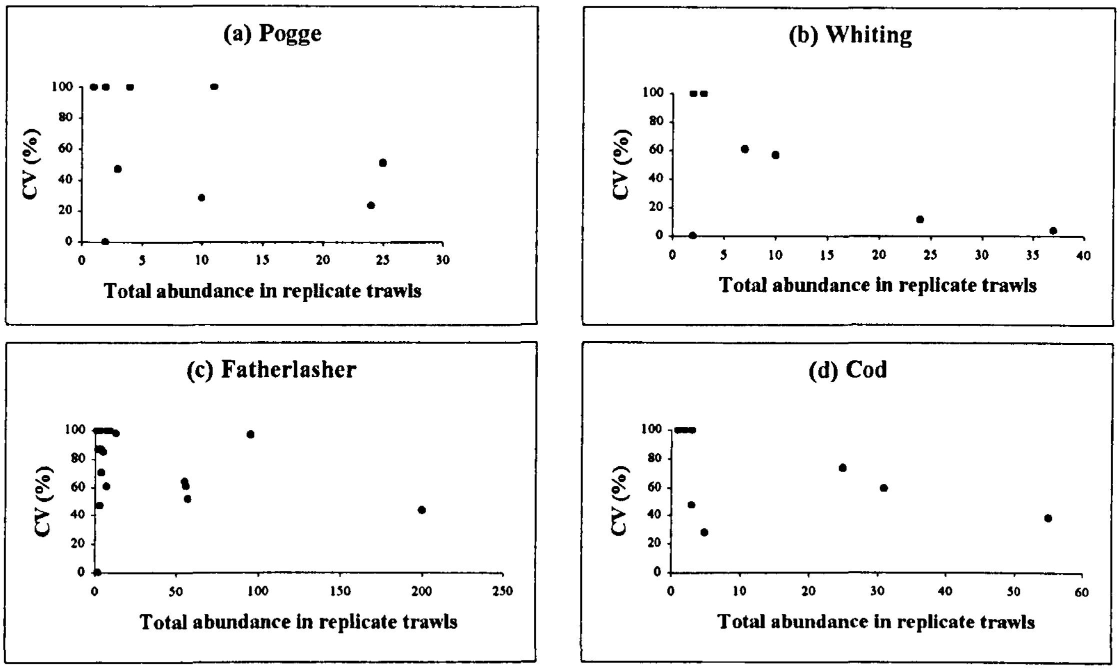


Figure 3.2. Assessment of Agassiz trawl repeatability (sampling error), data from Tay Estuary 1999-2001. CV = coefficient of variation; 'total abundance in replicate trawls' = total number of fish collected in all replicated trawls on a single day at either Abertay or Ladyshoal.

3.2.2 Pelagic trawl study

3.2.2.1. Field study

Pelagic trawling by S/V Forth Ranger commenced in January 1999, and took place on the same days, at the same stations, and for the same distance/duration as Agassiz trawling (see section 3.2.1.1.). The gear was a modified commercial sprat net, of opening 7.1m wide by 7.3m high, though depth gauge measurements showed that the average height during trawling was approx. 2.5m. Internal stretched cod-end mesh size was 1.2cm. Fish captured were analysed as detailed in section 3.2.1.1. When necessary, subsamples of 30 fish were taken for mass and length measurements, and the remaining fish were enumerated. Poor operation of the pelagic gear in January 1999 necessitated exclusion of these data from subsequent analyses; data for January 2001 were included so that a full 24-month period was obtained. LW pelagic trawling in July 2000 was curtailed by inundation of scypohozoan medusae causing tearing of the net (see section 2.2.2.). This gave a total of 57 trawls between April 1999 and January 2001.

3.2.2.2. Calculation of approximate abundances in the mid-lower Forth Estuary

Data for sprat and herring abundances obtained in each trawl were multiplied by 3.33, which assumed a gear efficiency of 33%, *i.e.* similar to that of the Agassiz gear (Kuipers, 1975). Monthly arithmetic and geometric mean abundances per trawl with 95% confidence intervals were computed. Assuming that each pelagic haul represented fish from an approximate trawled volume of 11680 m^3, the concentration of fish per m^3 was calculated. A uniform distribution of fish between the Forth Bridges and Dunmore was assumed (Figure 1.1), representing a mean volume of water of approximately 5.02×10^8 m^3 (SEPA, unpublished data). The boundaries of this area were based on the seaward limit of the Forth Estuary, as defined by McLusky (1987a), and the historical location of herring fisheries in Airth parish (The Statistical Accounts of Scotland, Vol. 3, p. 488). Extrapolation of mean fish abundance per unit volume to the volume above thus gave estimates of clupeid abundance in the Forth based on pelagic trawling. Extrapolations were based on three estimates of mean abundance: arithmetic mean with 95% CIs, geometric mean with 95% CIs, and arithmetic mean with 95% CIs calculated geometrically. The last of these calculations aimed to compensate for the possible underestimation of abundance during calculations of the geometric mean (Fowler and Cohen, 1990).

3.2.2.3. Generalised Linear Modelling of pelagic data

Data of sprat and herring abundance were treated as detailed for Agassiz-caught species in section 3.2.1.2, in order to investigate similar hypotheses. Both GLMs were based on negative binomial distributions of data.

3.3. Results

Agassiz and pelagic trawl data for the period 1999 – 2000 are presented in a CD-ROM (see Appendix 2).

3.3.1. Agassiz trawl study, 1982 – 2000.

3.3.1.1. Specific compositon of catch and abundance trends of common species

Thirty demersal and benthic species were collected in Agassiz trawls from 1982 – 2000 (Table 3.1). In addition the pelagic species of sprat, herring and smelt were also caught, but are not considered here.

Whiting was the most abundant of the species captured, followed closely by eelpout. Pogge, flounder and plaice were quite similarly abundant over the 19-year period, with approx. 2000 individuals of each species having been sampled. Gobies, cod and dab occurred in the range 950 – 1300 individuals, while fatherlasher and sea snail were similarly abundant at 400 individuals trawled. These ten most abundant species comprised 97.8% of the total demersal and benthic species quantity caught by Agassiz trawling from 1982 – 2000.

Table 3.1. Composition of Agassiz trawl catches, 1982 – 2000. Maximum: maximum yearly total; minimum: minimum yearly total; mean: arithmetic mean of all years; total: total fish caught 1982 – 2000.

	maximum	minimum	mean	total
whiting	483	36	226.9	4310
eelpout	669	17	207.7	3946
pogge	182	55	114.8	2181
flounder	349	20	103.4	1965
plaice	255	29	90.4	1718
gobies	196	3	68.1	1293
cod	293	2	62.1	1179
dab	137	4	50.2	953
fatherlasher	86	4	21.7	413
sea snail	73	5	21.1	401
lesser sandeel	62	0	5.7	108
butterfish	18	0	4.9	93
river lamprey	12	0	2.4	46
long rough dab	18	0	2.0	38
grey gurnard	19	0	1.9	36
lesser pipefish	3	0	0.9	18
saithe	8	0	0.8	15
great pipefish	6	0	0.5	10
lemon sole	4	0	0.5	9
Dover sole	3	0	0.5	9
pollack	7	0	0.4	8
common dragonet	2	0	0.3	6
ling	3	0	0.2	4
eel	2	0	0.2	4
5-bearded rockling	1	0	0.2	4
red gurnard	4	0	0.2	4
Montagu's sea snail	3	0	0.2	3
poor cod	1	0	0.1	1
15-spined stickleback	1	0	0.1	1
lesser spotted dogfish	1	0	0.1	1

The yearly total sampling effort of 30 trawls tended to catch more fish in the first decade of the time series, with peaks in 1987 – 1988 (Figure 3.3). Number of species taken per year generally declined between the mid-1980s and the mid-1990s, though there was an increase to previous highs in 1997 – 1998. A major decline in species number to the lowest ever annual total of 12 in 1999, was followed by 20 species being recorded in 2000, a 19-year high. The lowest total annual abundance of fish was also recorded in 1999, with an increase to levels similar to those of most of the 1990s the following year (Figure 3.3).

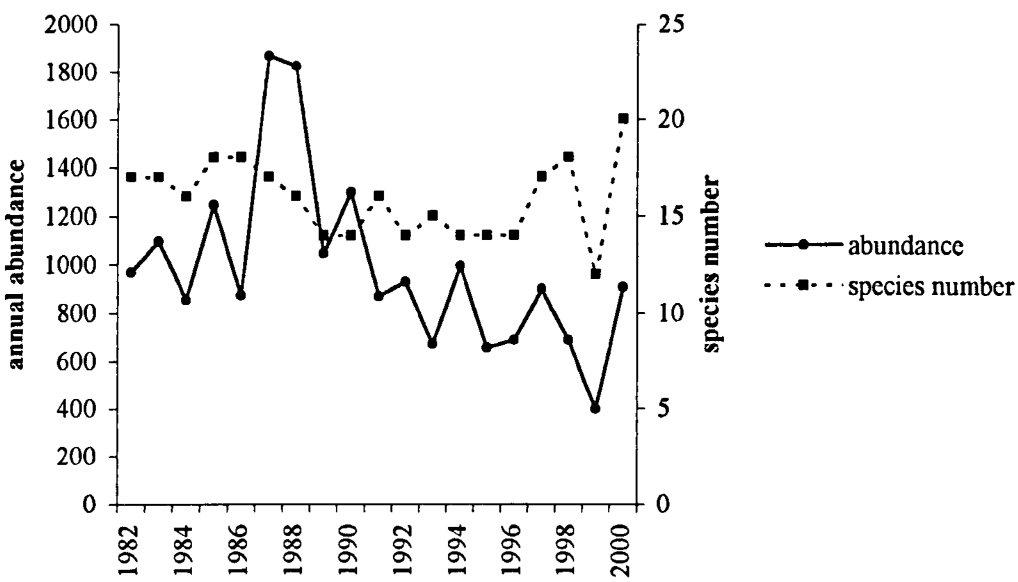


Figure 3.3. Trends in total annual abundance and number of benthic and demersal species taken by Agassiz trawling, lower Forth Estuary, 1982 – 2000. Sampling effort = 30 trawls in all years except 1986, when n = 24 trawls.

There was no statistically detectable decrease in species richness over time, but the trend of a decrease in annual abundance of fish was highly significant:

$$\ln \text{(mean annual abundance per trawl)} = 3.83 - 0.0381\ T\ (r^2 = 31.5\%, p = 0.007)$$

where T is year of study (n = 19 years in total).

Whiting, eelpout, dab and gobies exhibited 19-year lows in abundance in 1999 – 2000 (Figure 3.4a,e,i). Whiting showed peaks in abundance in 1987-88 and 1994-95, with less marked fluctuations in abundance in the years prior to these periods (Figure 3.4a). Following a smaller peak in 1997-98, the lowest recorded abundances in the 19-year series were evident in 1999-2000. The linear trend for this species was significantly negative, being described as:

$$\ln \text{(whiting abundance + 1)} = 4.3 + 0.0252\ T\ (r^2 = 6.7\%, p = 0.007)$$

where T is month of sampling (n = 94 months in total).

The other main Forth gadoid species, cod, was particularly abundant in 1990-91 and the latter months of 2000, with mean catch being 3-5 × that of the remaining years, which tended to average around two fish per trawl. There was no significant linear trend. Plaice was similar to cod in showing peak abundances in 1990-91 and 2000, and lack of any significant linear trend (Figure 3.4c). Flounder abundance peaked in 1987-88, and generally higher abundances in the 1980s compared with less abundant but more stable catches in the 1990s, contributed to a lack of any linear trend in abundance (Figure 3.4d). Dab were occasionally caught in very large numbers, but often catches were zero, and this species exhibited least evidence of any trend in abundance of all ten most common species (Figure 3.4e). In sharp contrast, eelpout showed the clearest trend of all species, this being a significant decline in abundance caused by steadily diminishing catches throughout the 1990s following an enhanced period of abundance from 1987-89 (Figure 3.4f). The regression equation obtained for this species was:

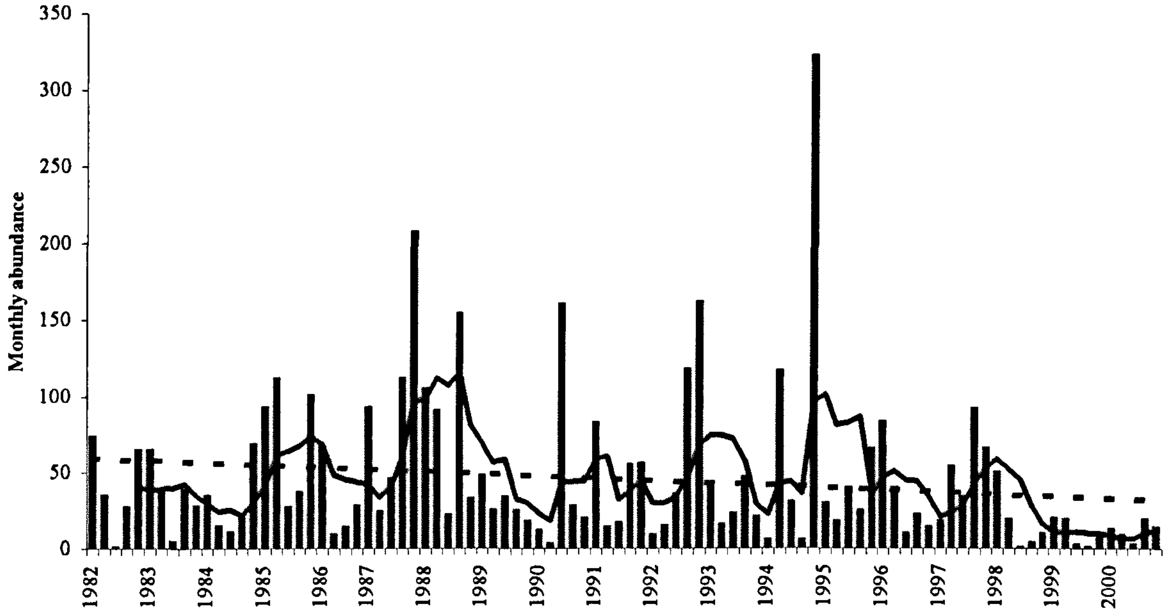
$$\ln(\text{eelpout abundance} + 1) = 3.86 - 0.0107\ T \quad (r^2 = 30.2\%,\ p < 0.001)$$

where T is month of sampling (n = 94 months in total).

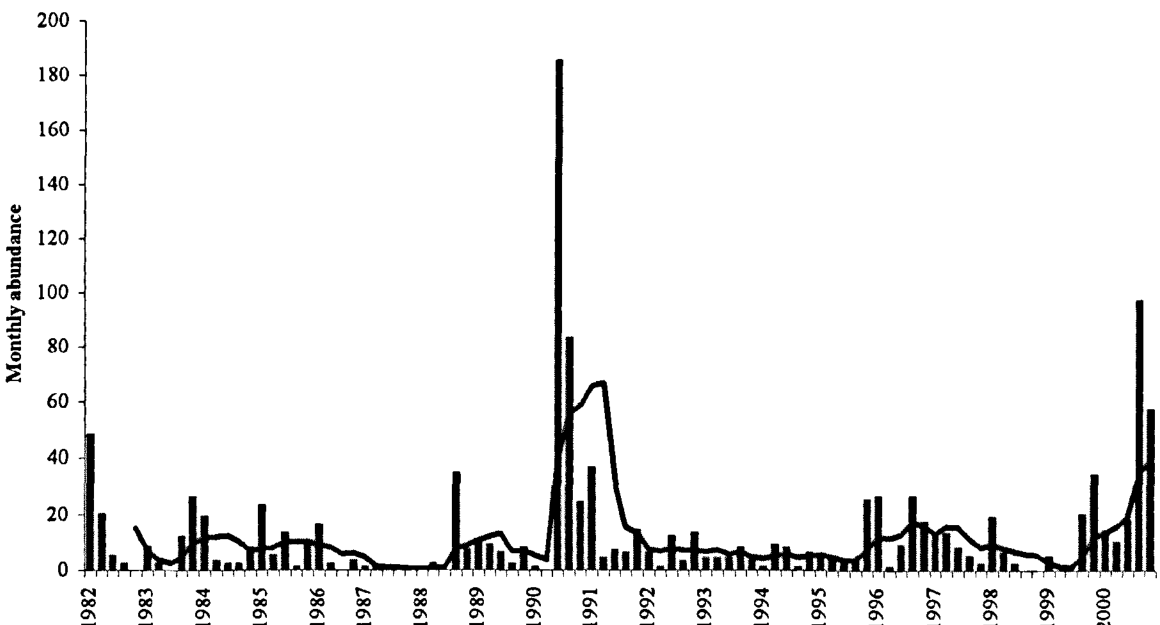
Pogge abundance reached a plateau from 1987-90, followed by something of a decline in the early 1990s, and levels of generally greater abundance in the 1990s than the previous decade (Figure 3.4g). This did not result in a significantly positive linear trend in abundance, however. The most positive trend in abundance of fish captured during Agassiz trawling was exhibited by fatherlasher, thanks to relatively high abundances in 1998-99 and 2000 (Figure 3.4h), though the linear regression did not indicate that this increase was statistically significant. High abundance in the early 1980s was followed by mostly low catches, with the exception of 1992-93, until the recent period of increased numbers. Three peaks of goby abundance occurred between 1985 and 1991, with only one period of relatively high abundance later in the time series (1997-98) combined with deep troughs of low catches in 1994-95 and 1999-2000 (Figure 3.4i). This species was similar to most others in exhibiting an absence of any statistically significant linear trend in abundance over the study period. The abundance of sea snail in trawls was second only to dab in its lack of any clear trend during the 19-year period (Figure 3.4j). A relatively high period of abundance in 1988-89 was the peak in abundance of this species, but numbers were generally low.

Figure 3.4. Monthly abundance of common fish species caught in the lower Forth Estuary by Agassiz trawling, 1982 – 2000. Each month's data represents sum of six trawls (HW and LW at three stations). Black line represents yearly moving average (period of five months of sampling), dashed lines indicate significant negative linear trends in abundance of whiting and eelpout.

(a) whiting



(b) cod



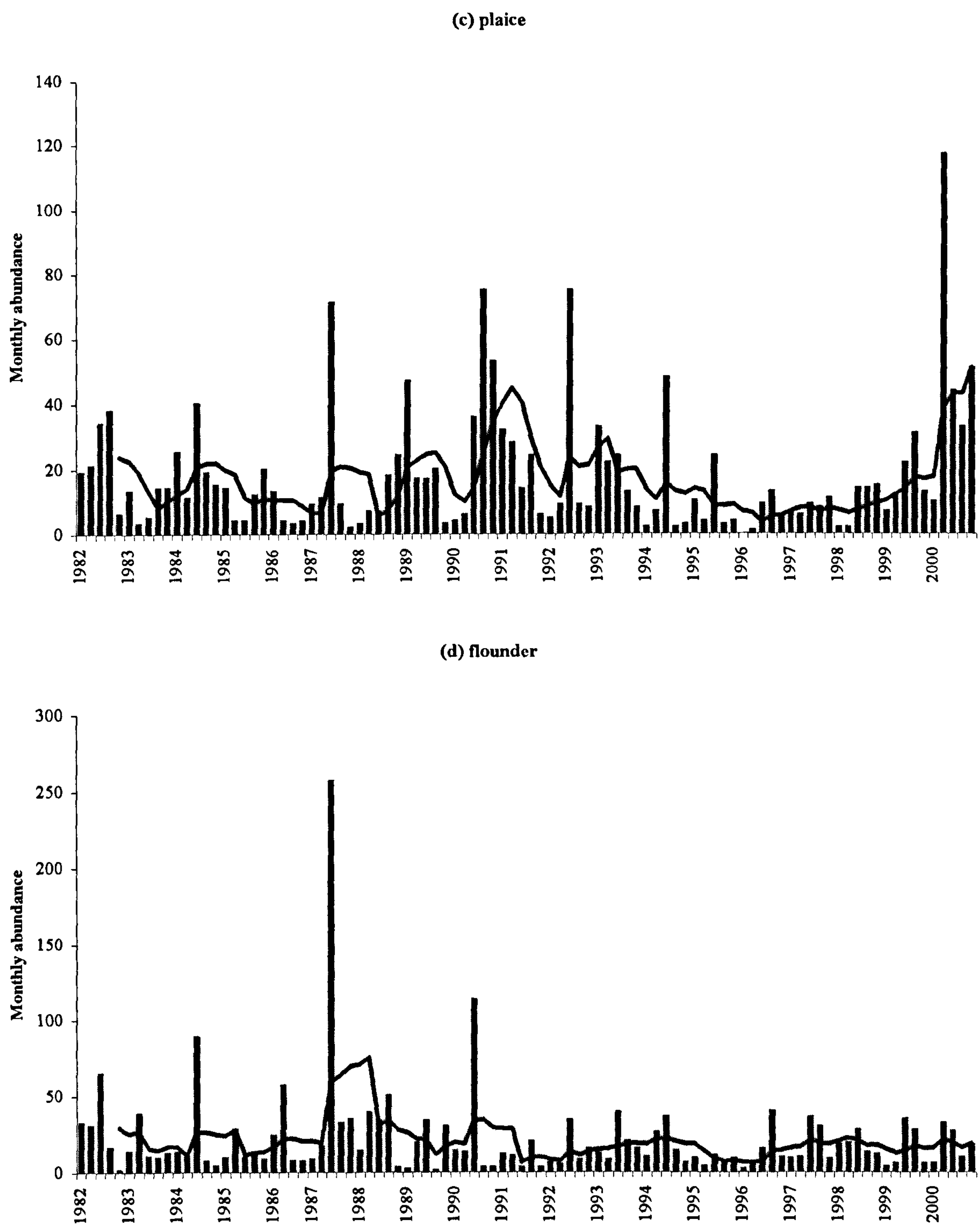
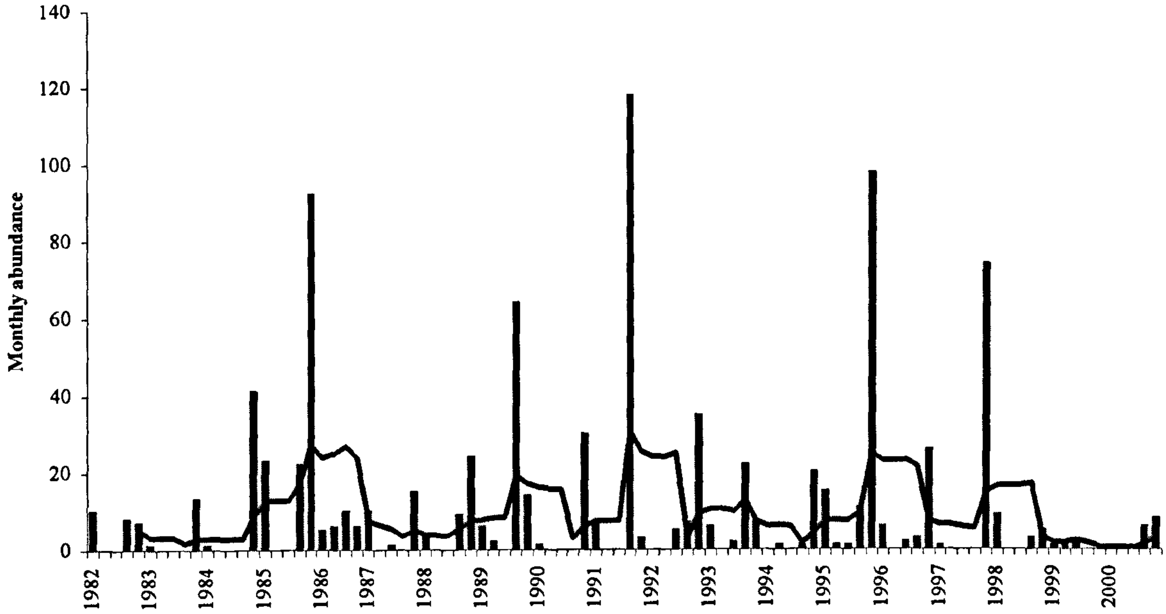


Figure 3.4 cont.

(e) dab



(f) eelpout

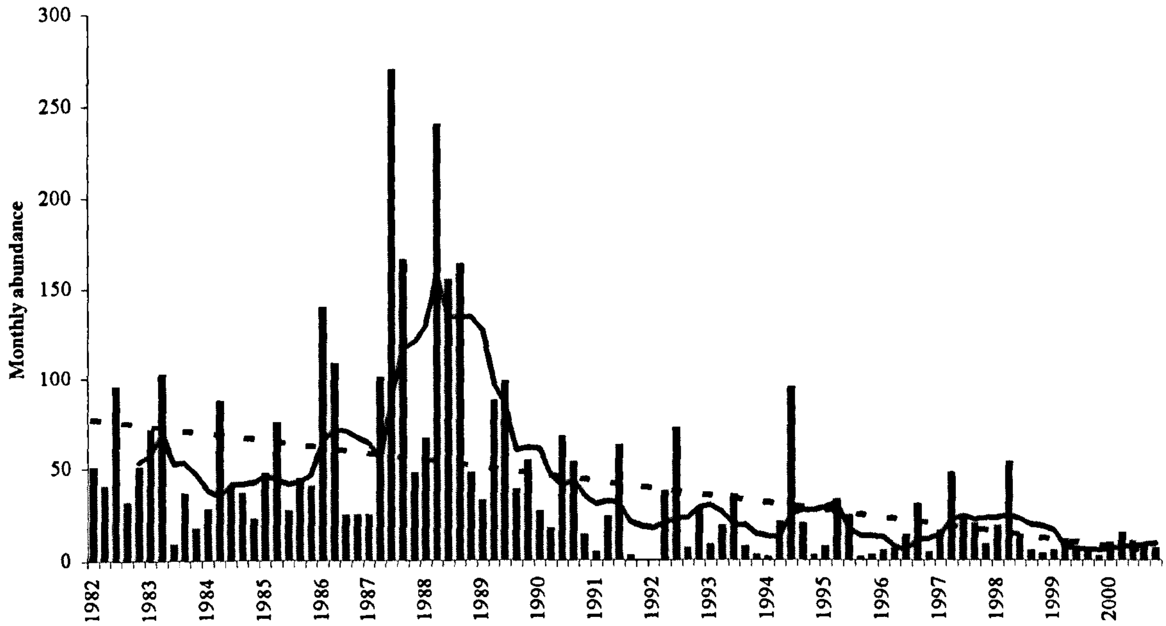
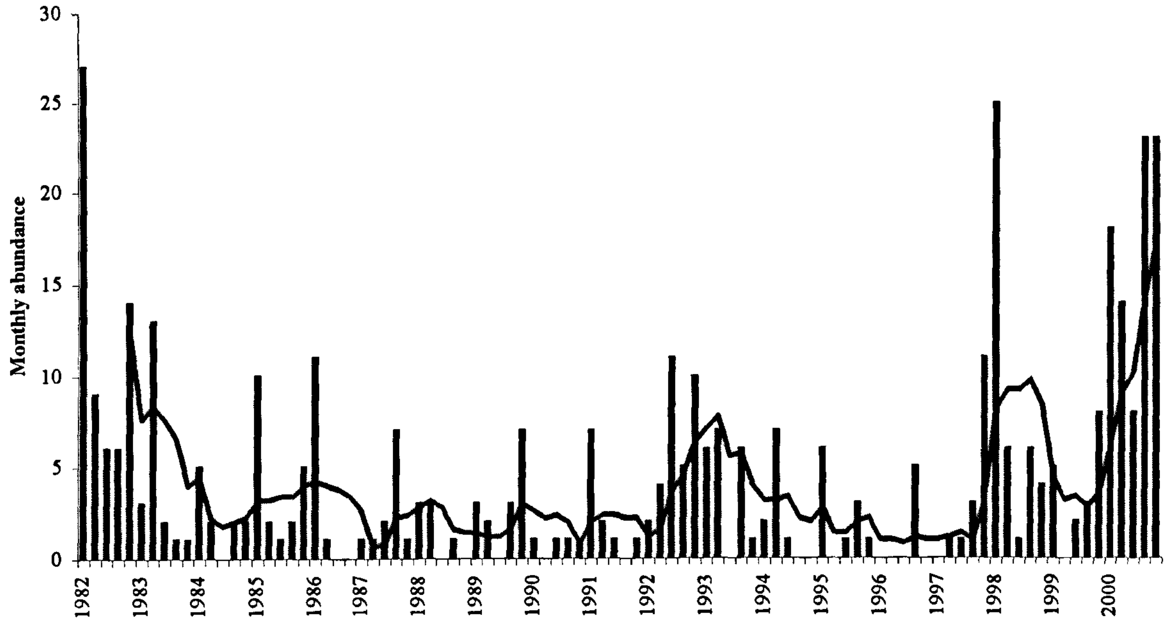


Figure 3.4 cont.

(h) fatherlasher



(i) gobies

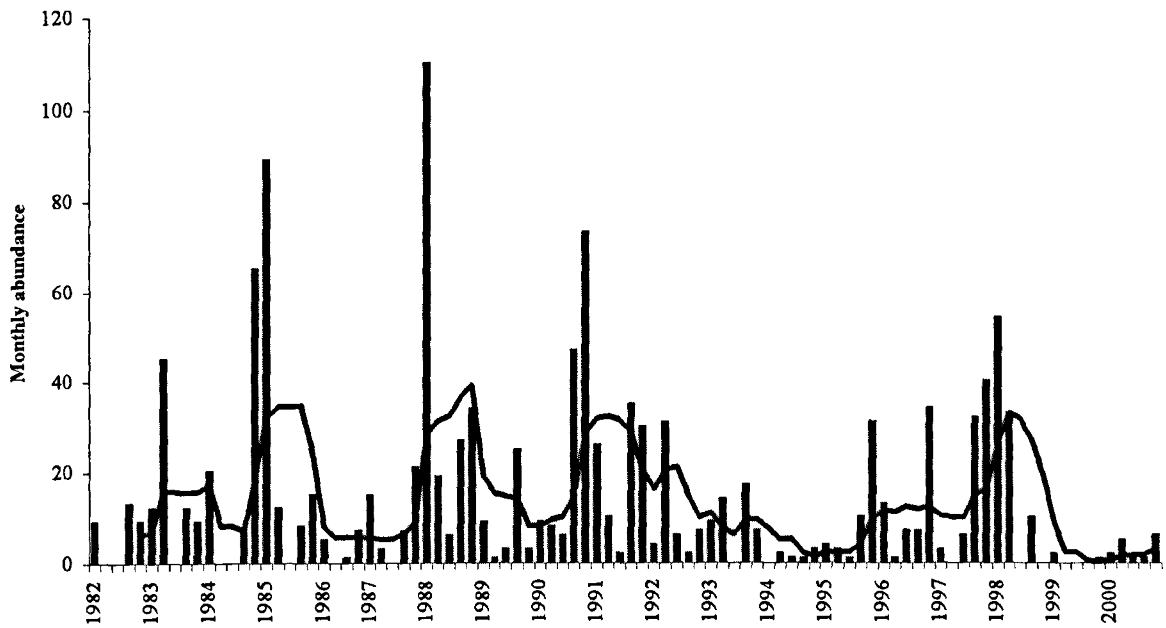


Figure 3.4 cont.

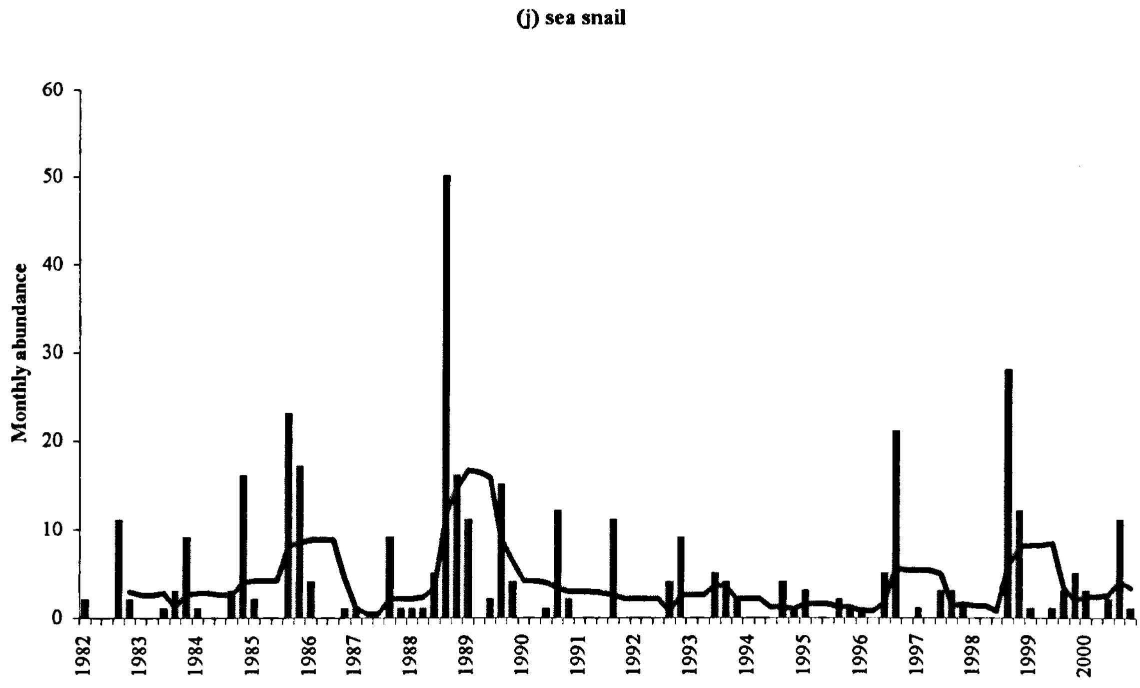


Figure 3.4 cont.

3.3.1.2. Generalised Linear Modelling of Agassiz data

GLMs of the 19-year Agassiz trawl dataset generally explained rather little of the null deviance. Despite this, both first and second retests of the models using random subsets of half the total number of data produced almost precisely the same results, indicating reasonable reliability of the GLMs of the entire datasets. As the results of these retests are almost identical to the primary GLMs, the results are not reproduced for sake of clarity. The station factor explained the greatest proportion of deviance per degree of freedom in models of plaice, flounder, cod, eelpout and fatherlasher abundance (Table 3.2c-g). Month explained most deviance in the pogge, gobies, dab and sea snail models (Table 3.2b,h-j), while tide was the most significantly related factor only in the whiting GLM (Table 3.2a). When tide was a significant factor predicting abundance of fish caught in the gear, the relationship was usually that of LW being positively related to fish abundance, and HW being negatively related. This was the case for pogge, flounder, eelpout, and fatherlasher (Table 3.2b,d,f,g respectively), and also for plaice and sea snail, where tide was only marginally statistically insignificant (Table 3.2c,j). Whiting was the only species to show a significant positive relationship between trawl abundance and HW (Table 3.2a).

Table 3.2. Summary of results of GLMs on lower Forth Estuary Agassiz trawl dataset, 1982 – 2000 (n=564 trawls). 'd.f.' is degrees of freedom; 'deviance' is amount of deviance explained by predictor; 'significance': * = p<0.05, ** = p<0.01, *** = p<0.001, NS = not significant (p>0.05); 'relationship': + signifies direct proportionality with dependent variable, - signifies inverse proportionality, relative strengths of factor levels indicated. Station: PE = Port Edgar, T = Tancred, L = Longannet. Month: 1 = January, 2 = late March/April/early May, 3 = late June/July/early August, 4 = September/October, 5 = December. † indicates negative binomial distribution applied in GLM; ‡ indicates quasi-likelihood estimation applied in GLM. NULL signifies total deviance in null model.

(a) whiting†	d.f.	deviance	significance	relationship
NULL		703.6		
tide	1	10.9	***	+: HW -: LW
station	2	16.1	***	+: PE>T -: L
month	4	30.3	***	+: 5>1>4 -: 3>2
tide × station	2	8.2	*	See Figure 3.5a
tide × month	4		NS	
station × month	8	34.2	***	See Figure 3.6a

(b) pogge†	d.f.	deviance	significance	relationship
NULL		696.5		
tide	1	15.8	***	+: LW -: HW
station	2	25.2	***	+: PE>L -: T
month	4	65.4	***	+: 4>1>5 -: 2>3
tide × station	2	14.2	***	See Figure 3.5b
tide × month	4		NS	
station × month	8	36.6	***	See Figure 3.6b

Table 3.2 cont.

(c) **plaice**†	d.f.	deviance	significance	relationship
NULL		625.3		
tide	1		NS (p=0.062)	+: LW -: HW
station	2	53.0	***	+: T>L -: PE
month	4	23.2	***	+:3>4 -: 1>2>5
tide × station	2		NS	
tide × month	4		NS	
station × month	8		**	See Figure 3.6c

(d) **flounder**†	d.f.	deviance	significance	relationship
NULL		945.9		
tide	1	13.8	***	+: LW -: HW
station	2	195.4	***	+: L -: T>PE
month	4	112.3	***	+: 3>4 0: 2 -: 1>5
tide × station	2		NS	
tide × month	4		NS	
station × month	8	55.0	***	See Figure 3.6d

(e) **cod**‡	d.f.	deviance	significance	relationship
NULL		2877.1		
tide	1		NS	
station	2	400.2	***	+: PE -: T>L
month	4		NS	
tide × station	2		NS	
tide × month	4		NS	
station × month	8		NS	

(f) **eelpout**†	d.f.	deviance	significance	relationship
NULL		828.6		
tide	1	13.9	***	+: LW -: HW
station	2	79.3	***	+: PE>L -: T
month	4	108.1	***	+: 3>2 -: 5>1>4
tide × station	2		NS	
tide × month	4		NS	
station × month	8	75.5	***	See Figure 3.6e

Table 3.2 cont.

(g) **fatherlasher**[‡]	d.f.	deviance	significance	relationship
NULL		632.9		
tide	1	12.5	**	+: LW -: HW
station	2	137.7	***	+: PE -: T>L
month	4	20.8	**	+: 1>5>2 -: 3>4
tide × station	2	7.9	*	See Figure 3.5c
tide × month	4		NS	
station × month	8		NS	

(h) gobies[†]	d.f.	deviance	significance	relationship
NULL		624.9		
tide	1		NS (p=0.057)	+: LW -: HW
station	2	8.7	*	+: PE>T -: L
month	4	100.8	***	+: 1>5>4 -: 3>2
tide × station	2		NS	
tide × month	4		NS	
station × month	8	18.9	*	See Figure 3.6f

(i) dab[‡]	d.f.	deviance	significance	relationship
NULL		2341.9		
tide	1		NS	
station	2	184.8	***	+: T>L -: PE
month	4	576.4	***	+: 5>4 -: 2>3>1
tide × station	2		NS	
tide × month	4		NS	
station × month	8		NS	

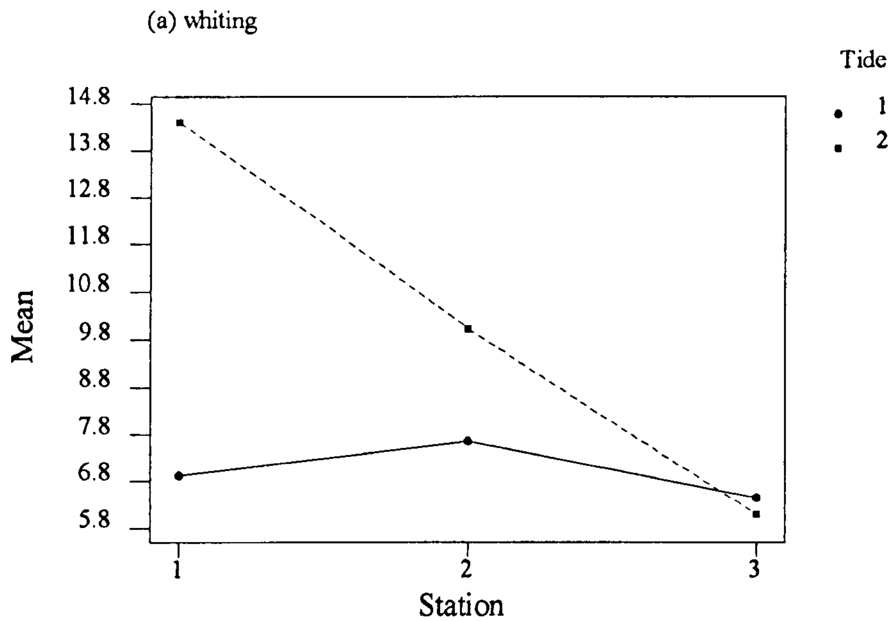
(j) sea snail[‡]	d.f.	deviance	significance	relationship
NULL		702.4		
tide	1		NS (p=0.057)	+: LW -: HW
station	2	42.3	***	+: L -: PE>T
month	4	146.5	***	+: 4>5 -: 2>3>1
tide × station	2		NS	
tide × month	4		NS	
station × month	8		NS	

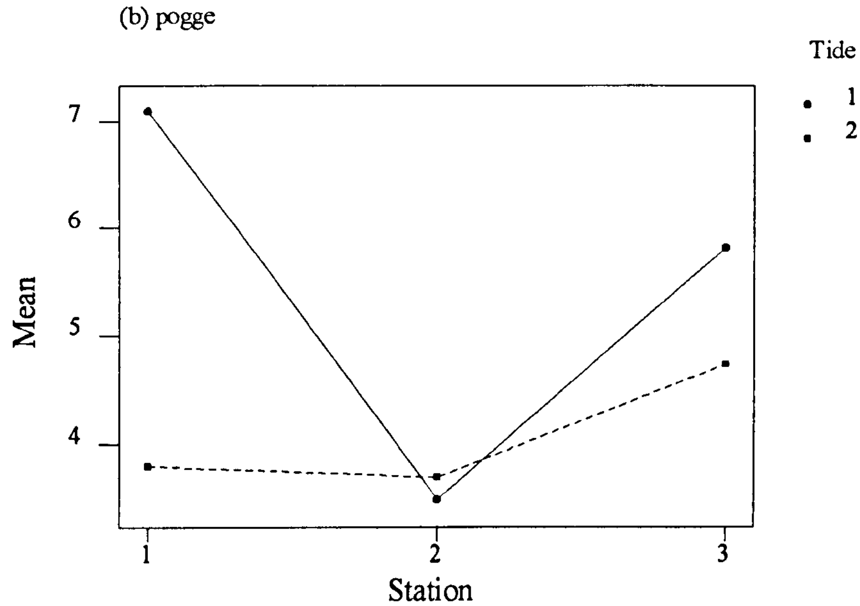
The relationships between the various species' trawl abundances and the month factor were quite diverse. Greatest positive relationships were shown between abundance and month 1 (January/February) in fatherlasher and gobies (Table 3.2g,h), month 3 (June/July) in plaice, flounder and eelpout (Table 3.2c,d,f), month 4 (September/October) in pogge and sea snail (Table 3.2b,j), and month 5 (December) in whiting and dab (Table 3.2a,i). Month 2 (April/May) did not show the greatest positive relationship with any species, but did exhibit the greatest inverse proportionality with abundance of pogge, dab and sea snail (Table 3.2b,i,j). Plaice and flounder abundance were most negatively related to the first level of the month factor (Table 3.2c,d), while the same was true of whiting, fatherlasher and gobies in month 3 (Table 3.2a,g,h), and of eelpout in month 5 (Table 3.2f).

The most pronounced positive relationships between trawl abundances and trawl station were observed with greatest regularity for Port Edgar. This was evident in whiting, pogge, cod, eelpout, fatherlasher and gobies (Table 3.2a,b,e-h). Tancred showed the greatest direct proportionality with abundances of plaice and dab (Table 3.2c,h), while flounder and sea snail exhibited this trend at the Longannet station (Table 3.2d,j). Greatest inverse proportionality with catch data was evident most often at the Tancred station, this being the case in the models of pogge, flounder, cod, eelpout and fatherlasher abundance (Table 3.2b,d-g). Plaice, dab and sea snail models suggested greatest inverse proportionality at the Port Edgar site (Table 3.2c,i,j), and a similar trend was displayed by whiting and gobies at Longannet (Table 3.2a,h).

Explanatory power of the interaction terms tested in the modelling process tended to be rather weak compared to the main effects, as evident from the relatively low amount of deviance explained per degree of freedom in most cases. The tide × month interaction was not significantly related to fish abundance in any of the models. The tide × station interaction was significant in three models (Figure 3.5). Whiting abundance at HW was much greater than at LW at Port Edgar, but declined to similar levels as the LW trawls at Longannet (Figure 3.5a). Catches of pogge at LW tended to be much higher than those undertaken at HW at Port Edgar and Longannet, but reasonably similar at Tancred (Figure 3.5b). Fatherlasher abundance tended to be considerably greater at LW than at HW at Port Edgar, while there was little difference between the two tidal states at the other stations (Figure 3.5c).

Figure 3.5. Plots of significant tide × station interaction from GLMs of Forth Estuary Agassiz trawl data, 1982 -2000. Tide: 1 = LW, 2 = HW. Station: 1 = Port Edgar, 2 = Tancred, 3 = Longannet. Mean = mean abundance per trawl.





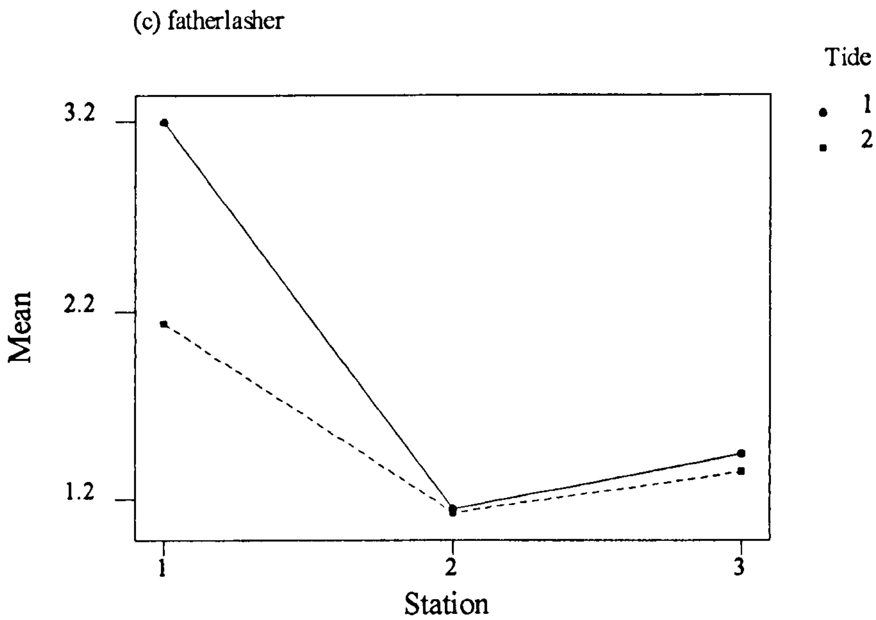
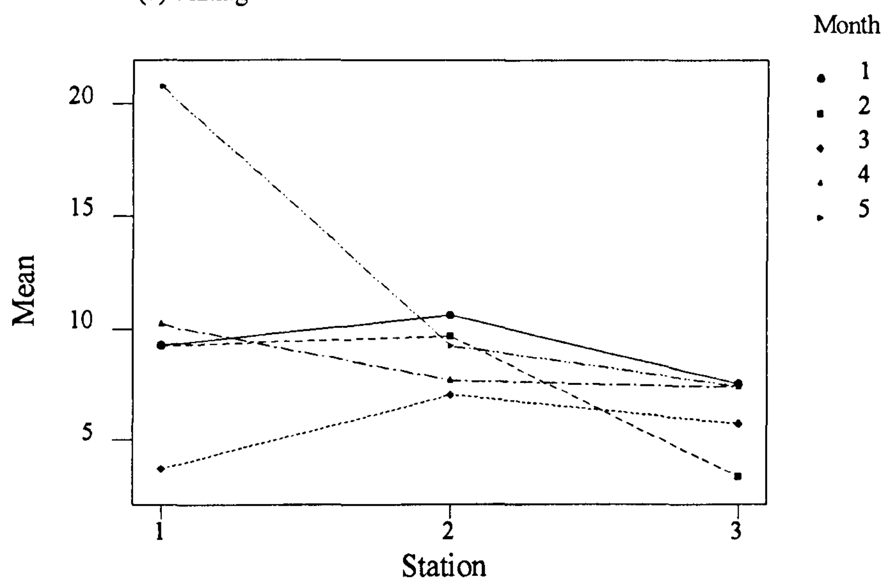


Figure 3.5 cont.

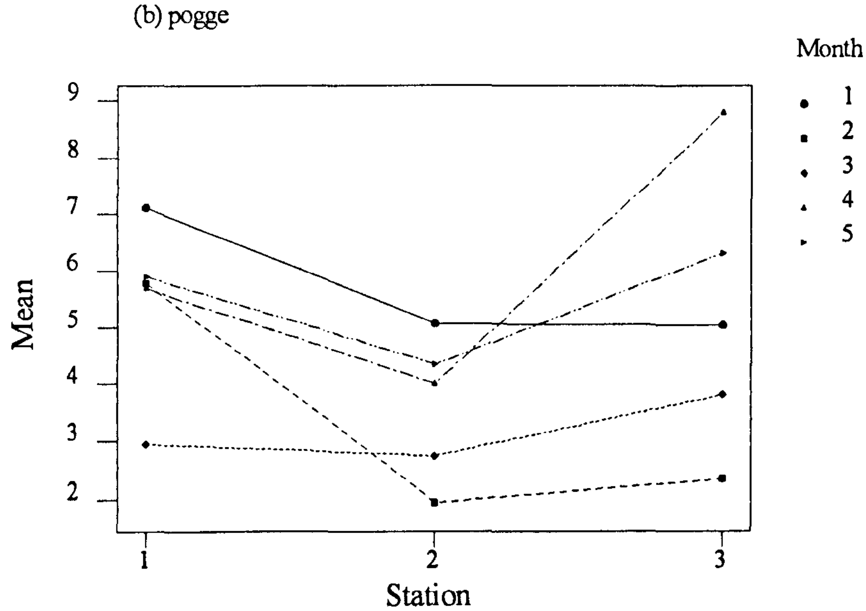
The numerous levels of factor in interaction plots of station × month made these data less easy to interpret, so the most important features were assessed. Monthly variation in catch of whiting was greatest at Port Edgar, with both greatest and least catches occurring in December and June, respectively (Figure 3.6a). Pogge exhibited similar wide variations in catch at Longannet, where the most abundant mean catches were observed in September, whilst also possessing a low in April similar to that at Tancred in the same month (Figure 3.6b). Plaice abundance in April tended to be markedly greater at Tancred than at the other two stations, while at Port Edgar catches were low throughout the year and showed little variation; maximum abundances in this species were in June at Longannet (Figure 3.6c). Flounder were caught in very similar abundance at Port Edgar and Tancred throughout the year, while values at Longannet were generally higher and in June showed a pronounced peak (Figure 3.6d). Eelpout abundance in all months, except April, tended to be greater at Port Edgar and Longannet than at Tancred, with these differences being of greater magnitude in June and September; largest catches were in April at Port Edgar and June at Longannet, with abundance at Tancred only greater than Longannet in April (Figure 3.6e). Abundance of gobies was minimal at all stations in June, while being highest at Port Edgar in January and December, and at Tancred in December (Figure 3.6f).

Figure 3.6. Plots of significant station × month interaction from GLMs of Forth Estuary Agassiz trawl data, 1982 -2000. Station: 1 = Port Edgar, 2 = Tancred, 3 = Longannet. Month: 1 = January/February, 2 = April/May, 3 = late June/July/early August, 4 = September/October, 5 = December. Mean = mean abundance per trawl.

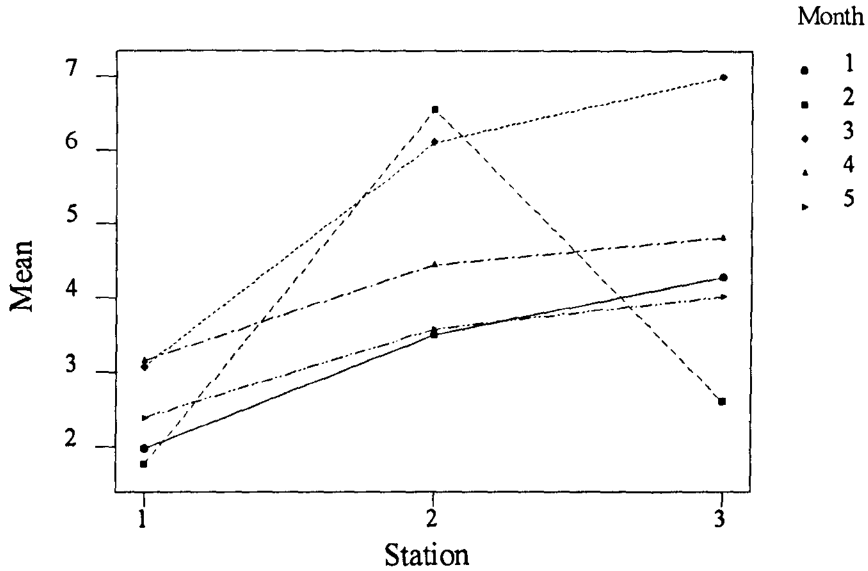
(a) whiting



(b) pogge



(c) plaice



(d) flounder

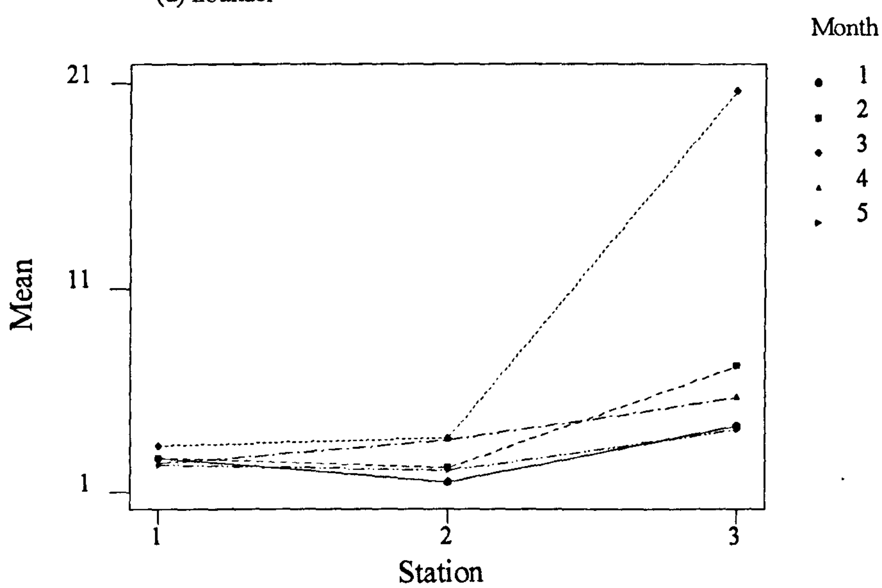
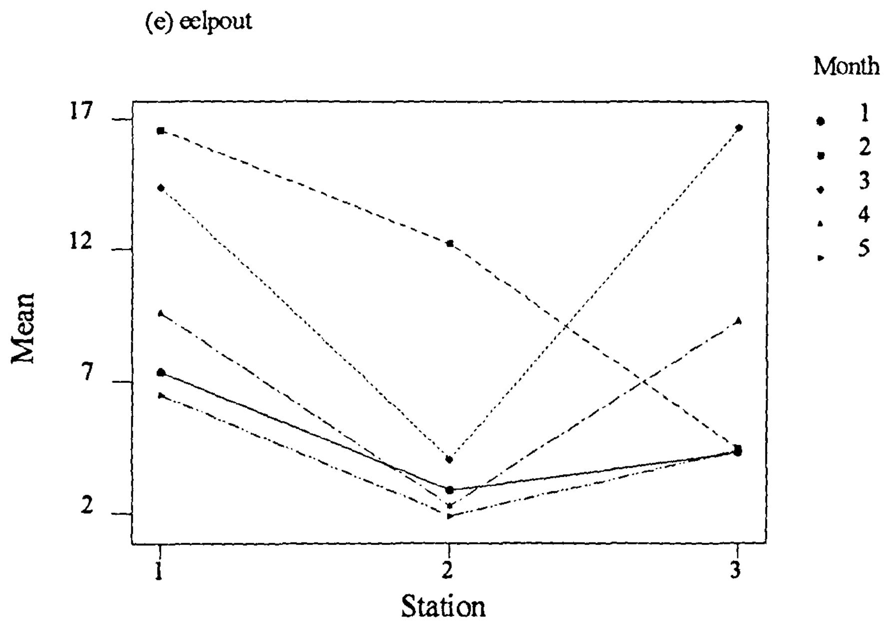


Figure 3.6 cont.



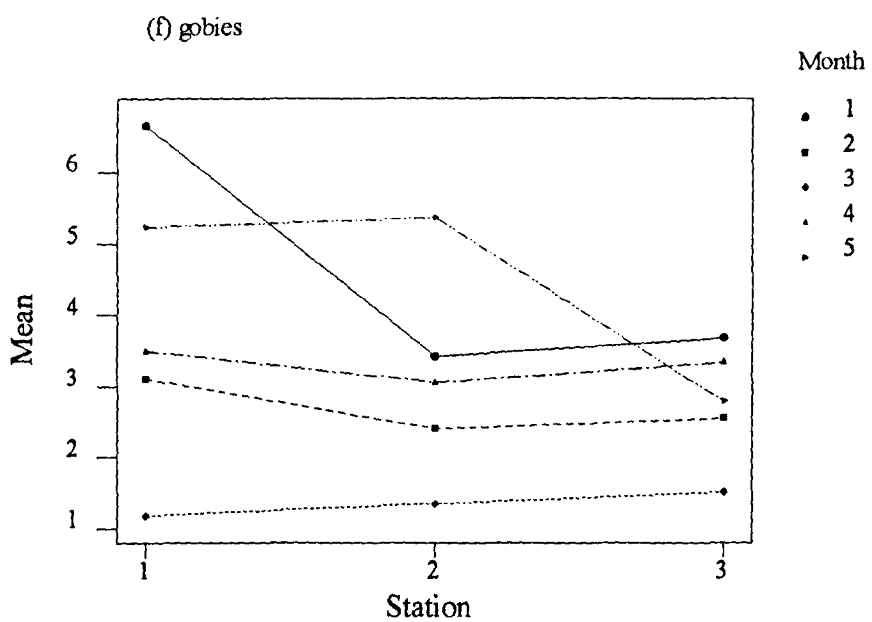


Figure 3.6 cont.

3.3.1.3. Exploratory analysis of changes in demersal and benthic ichthyofauna

Hierarchical agglomerative clustering and MDS plots of the Agassiz data highlighted similar groupings of years, whether considering raw abundance data (Figures 3.7a and 3.8a) or percentage composition of annual catches (Figures 3.7b and 3.8b). MDS plot stress values of 0.11 and 0.12 suggest that these 2-dimensional representations of multi-dimensional data were quite revealing, especially in combination with the cluster analysis. Clustering of raw abundance data gave three distinct groups at the 60% similarity level: group 1 (1999 and 2000), group 2 (1987 and 1988) and group 3 (all remaining years) (Figure 3.7a). This was in agreement with the MDS plot of the same data, with 1999 and 2000 on the left of the ordination, 1987 and 1988 on the right of the plot, and the remaining years generally nearer the centre (Figure 3.8a).

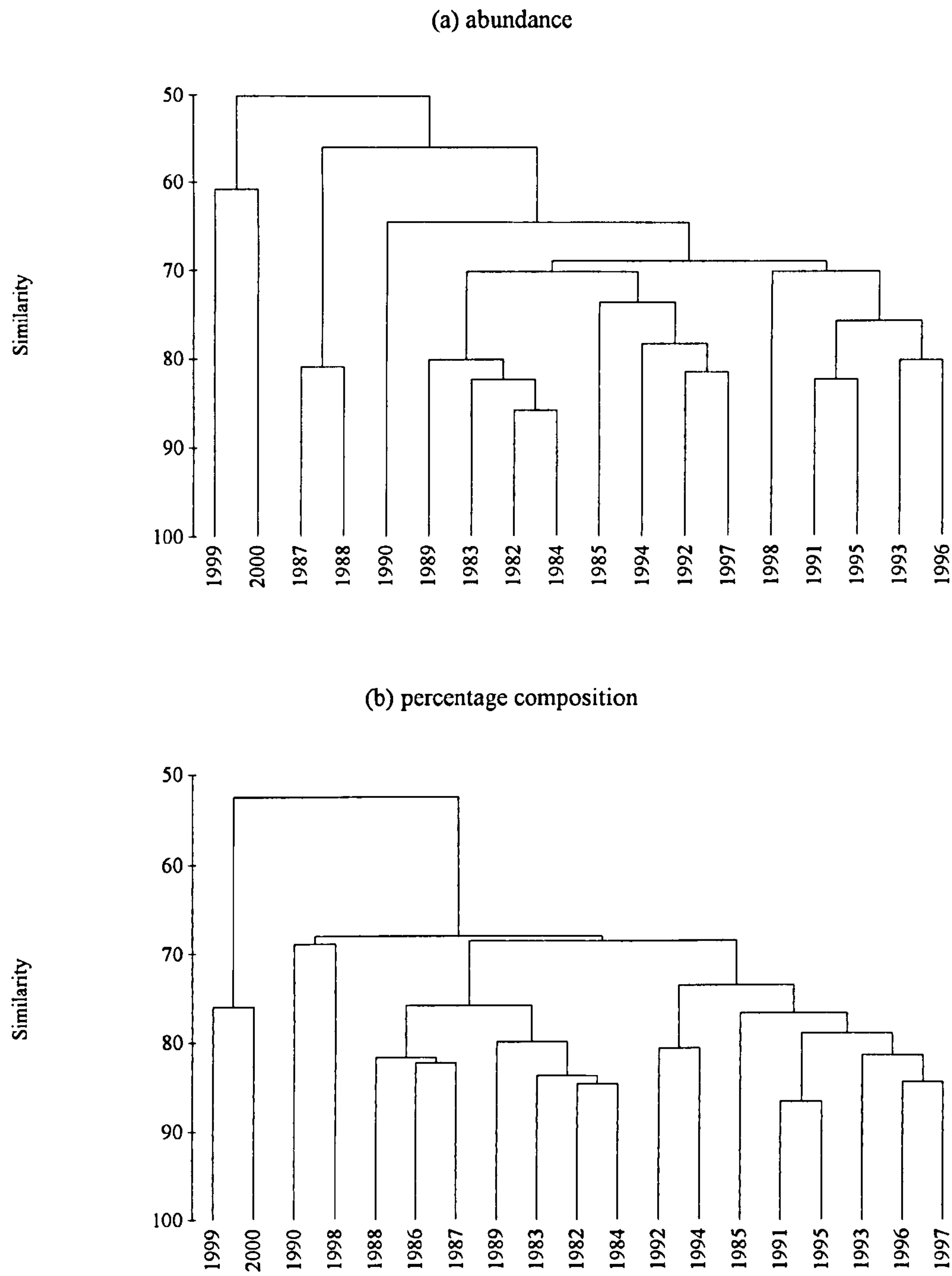


Figure 3.7. Dendrograms of hierarchical agglomerative clustering of Agassiz trawl data, lower Forth Estuary 1982 – 2000, based on (a) sums of abundances of species taken in each year, and (b) percentage contribution of each species to total annual abundance. Data for 1986 excluded from (a) due to reduced sampling effort. Values computed in relation to Bray-Curtis similarity matrices.

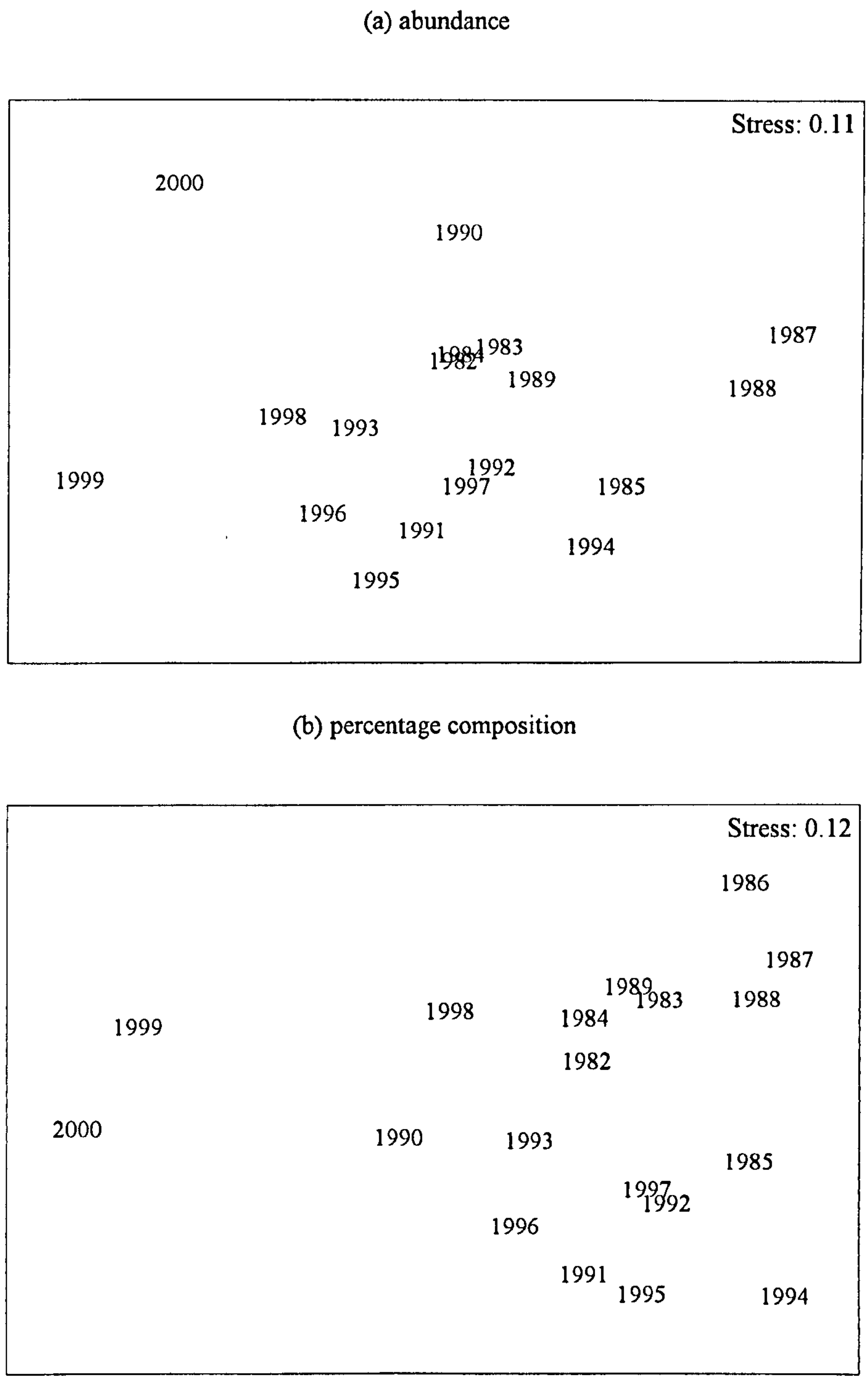


Figure 3.8. MDS plots of Agassiz trawl data, lower Forth Estuary 1982 – 2000, based on (a) sums of abundances of species taken in each year, and (b) percentage contribution of each species to total annual abundance. Data for 1986 excluded from (a) due to reduced sampling effort.

Similarity percentage analyses (SIMPER) for these groupings are reproduced in Table 3.3. The most dissimilar were groups 1 and 2, with whiting and eelpout's relative abundance in 1987-88 contrasting with their scarcity in 1999-2000, and thus providing the bulk of the dissimilarity (approx. 60%) (Table 3.3c). The relatively high ratio of average dissimilarity to standard deviation of these two species, surpassed only by that of pogge, suggested them to be consistently good indicators of changes between the two periods, (Clarke and Warwick, 1994). Similar trends were noted between groups 2 and 3 (Table 3.3a), the groups with greatest average dissimilarity. Comparison between groups 1 and 3 showed eelpout and whiting to again be the major contributors to dissimilarities (Table 3.3b).

Table 3.3. Summary output of SIMPER analysis of >60% similar raw Agassiz abundance data, 1982 – 2000 (excluding 1986). Av. Abund = average abundance, Av. Diss = average Bray-Curtis dissimilarity, Diss/SD = average dissimilarity/standard deviation, Contrib% = percentage of total dissimilarity attributable to species, Cum.% = cumulative percentage dissimilarity. See text for details of years contained in groups.

(a)
Groups 3 & 2
Average dissimilarity = 43.94

	Group 3	**Group 2**				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
eelpout	159.93	637.00	17.51	4.03	39.84	39.84
whiting	227.29	443.50	8.17	2.13	18.59	58.43
flounder	85.50	244.50	5.78	1.43	13.16	71.59
gobies	72.64	121.00	2.86	1.36	6.51	78.10
pogge	109.43	178.00	2.50	2.00	5.70	83.80
cod	60.71	23.00	1.55	0.73	3.53	87.33
dab	60.29	31.00	1.28	0.95	2.90	90.23

(b)
Groups 3 & 1
Average dissimilarity = 47.33

	Group 3	**Group 1**				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
whiting	227.29	47.00	11.66	1.75	24.63	24.63
eelpout	159.93	32.00	8.10	1.52	17.11	41.74
plaice	85.07	168.00	6.02	1.45	12.72	54.46
cod	60.71	128.50	5.45	1.40	11.52	65.98
gobies	72.64	9.00	4.21	1.81	8.90	74.88
dab	60.29	10.00	3.39	1.13	7.17	82.05
fatherlasher	19.57	52.00	2.24	1.32	4.73	86.78
pogge	109.43	100.50	1.87	1.50	3.96	90.74

(c)
Groups 2 & 1
Average dissimilarity = 67.96

	Group 2	**Group 1**				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
eelpout	637.00	32.00	24.55	6.17	36.12	36.12
whiting	443.50	47.00	16.08	5.53	23.65	59.78
flounder	244.50	80.00	6.67	1.33	9.82	69.60
gobies	121.00	9.00	4.58	1.25	6.74	76.34
cod	23.00	128.50	3.98	1.41	5.86	82.20
plaice	80.50	168.00	3.65	1.15	5.38	87.58
pogge	178.00	100.50	3.14	6.75	4.62	92.20

The 70% similarity level of raw abundance data consisted of group 1 (1999), group 2 (2000), group 3 (1987 and 1988), group 4 (1990), group 5 (1982, 1983, 1984, 1985, 1989, 1992, 1994, and 1997), and group 6 (1991, 1993, 1995, 1996, and 1998) (Figure 3.7a). SIMPER analyses of comparisons between each pair of groups are reproduced in Table 3.4. Groups 5 and 6 were the least dissimilar (Table 3.4d), while groups 1 and 3 were most dissimilar (Table 3.4h). In both cases eelpout and whiting contributed most to the average dissimilarity (approx. 50 – 65%), while eelpout was a reliable discriminating species, along with pogge, in terms of highest dissimilarity to S.D. ratios. The importance of eelpout and whiting was also apparent in all other comparisons with group 3, due to the high abundance of these species in 1987-1988 (Table 3.4a,c,e,l), as well as between group 1 and group 5 (Table 3.4g). Large abundance of cod in 1990 (group 4) and 2000 (group 2) led to this species generally accounting for approx. 20-30% of dissimilarity in comparisons of these groups with others (Table 3.4b,c,f,i,n). The relatively low cod contribution to dissimilarity between these two groups reflects the high abundances exhibited in both (Table 3.4m). The relatively large increase in abundance of cod and plaice between 1999 and 2000 contributed 60% of the dissimilarity between these two years, which comprised groups 1 and 2 respectively (Table 3.4o). The low abundances of plaice and cod in much of the 1990s gave rise to these two species again providing the bulk of dissimilarity between groups 2 and 6 (Table 3.4n). Dissimilarity between 2000 (group 2) and much of the 1980s (group 5, see above) was driven in almost equal part by the former's increased abundance of cod and plaice and decrease in whiting and eelpout, compared with the latter (Table 3.4k). These four species contributed just over 70% of dissimilarity between these groups. Decreases in whiting, gobies, eelpout and plaice accounted for a similar level of dissimilarity between group 1 and group 6 (Table 3.4j).

Table 3.4. Summary output of SIMPER analysis of >70% similar raw Agassiz abundance data, 1982 – 2000 (excluding 1986). Av. Abund = average abundance, Av. Diss = average Bray-Curtis dissimilarity, Diss/SD = average dissimilarity/standard deviation, Contrib% = percentage of total dissimilarity attributable to species, Cum.% = cumulative percentage dissimilarity. See text for details of years contained in groups. Hash marks (#) indicate inability to compute due to <3 years of data under consideration.

(a)
Groups 5 & 3
Average dissimilarity = 38.08

	Group 5	Group 3				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
eelpout	210.75	637.00	15.06	4.59	39.53	39.53
whiting	269.25	443.50	6.50	1.77	17.06	56.59
flounder	88.50	244.50	5.43	1.40	14.26	70.85
gobies	64.50	121.00	2.87	1.33	7.52	78.37
pogge	115.63	178.00	2.23	1.51	5.84	84.22
dab	58.50	31.00	1.16	1.09	3.06	87.27
sea snail	21.88	42.00	1.13	1.20	2.98	90.25

(b)
Groups 5 & 4
Average dissimilarity = 33.30

	Group 5	Group 4				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
cod	40.38	293.00	11.01	13.48	33.04	33.04
whiting	269.25	223.00	4.07	1.21	12.23	45.28
plaice	86.50	174.00	3.80	3.18	11.41	56.68
gobies	64.50	143.00	3.44	2.14	10.33	67.02
eelpout	210.75	176.00	2.95	1.67	8.87	75.89
flounder	88.50	146.00	2.46	1.94	7.39	83.28
pogge	115.63	82.00	1.77	1.27	5.30	88.58
dab	58.50	31.00	1.43	1.08	4.30	92.88

(c)
Groups 3 & 4
Average dissimilarity = 44.62

	Group 3	Group 4				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
eelpout	637.00	176.00	14.68	9.32	32.90	32.90
cod	23.00	293.00	8.59	9.92	19.25	52.16
whiting	443.50	223.00	7.01	4.20	15.71	67.87
flounder	244.50	146.00	3.31	0.75	7.41	75.28
pogge	178.00	82.00	3.06	20.12	6.85	82.12
plaice	80.50	174.00	2.98	2.99	6.68	88.80
gobies	121.00	143.00	2.38	2.46	5.34	94.14

Table 3.4 cont.

(d)

Groups 5 & 6

Average dissimilarity = 31.05

	Group 5	Group 6				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
eelpout	210.75	75.40	7.77	1.91	25.04	25.04
whiting	269.25	161.00	7.17	1.16	23.08	48.12
dab	58.50	69.00	2.83	1.36	9.11	57.23
pogge	115.63	105.00	2.26	1.58	7.27	64.50
plaice	86.50	65.00	2.16	1.27	6.96	71.46
flounder	88.50	68.60	2.05	1.45	6.60	78.06
gobies	64.50	71.60	1.99	1.45	6.41	84.47
cod	40.38	46.80	1.22	1.39	3.91	88.39
sea snail	21.88	19.00	0.94	1.18	3.02	91.41

(e)

Groups 3 & 6

Average dissimilarity = 53.16

	Group 3	Group 6				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
eelpout	637.00	75.40	21.99	11.75	41.36	41.36
whiting	443.50	161.00	11.07	4.28	20.83	62.19
flounder	244.50	68.60	6.84	1.58	12.87	75.06
gobies	121.00	71.60	2.95	1.27	5.55	80.60
pogge	178.00	105.00	2.84	3.27	5.34	85.94
dab	31.00	69.00	1.68	0.96	3.16	89.10
plaice	80.50	65.00	1.30	1.33	2.44	91.54

(f)

Groups 4 & 6

Average dissimilarity = 38.94

	Group 4	Group 6				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
cod	293.00	46.80	12.28	7.94	31.53	31.53
plaice	174.00	65.00	5.46	2.97	14.02	45.56
eelpout	176.00	75.40	5.02	5.62	12.90	58.46
flounder	146.00	68.60	3.84	3.20	9.86	68.32
gobies	143.00	71.60	3.59	2.53	9.21	77.53
whiting	223.00	161.00	3.18	1.24	8.16	85.69
dab	31.00	69.00	2.13	0.92	5.47	91.16

(g)

Groups 5 & 1

Average dissimilarity = 50.10

	Group 5	Group 1				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
whiting	269.25	36.00	16.59	2.02	33.12	33.12
eelpout	210.75	17.00	13.64	2.81	27.22	60.35
gobies	64.50	3.00	4.35	1.77	8.69	69.04
dab	58.50	4.00	3.81	1.55	7.60	76.64
pogge	115.63	99.00	2.54	1.52	5.07	81.71
flounder	88.50	67.00	2.37	1.77	4.73	86.43
plaice	86.50	81.00	1.87	3.27	3.72	90.16

Table 3.4 cont.

(h)

Groups 3 & 1

Average dissimilarity = 69.55

	Group 3	Group 1				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
eelpout	637.00	17.00	27.65	11.65	39.76	39.76
whiting	443.50	36.00	18.15	8.28	26.10	65.85
flounder	244.50	67.00	7.87	1.21	11.32	77.17
gobies	121.00	3.00	5.29	1.10	7.61	84.78
pogge	178.00	99.00	3.52	17.03	5.06	89.84
cod	23.00	61.00	1.69	1.29	2.42	92.26

(i)

Groups 4 & 1

Average dissimilarity = 56.84

	Group 4	Group 1				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
cod	293.00	61.00	13.68	#######	24.07	24.07
whiting	223.00	36.00	11.03	#######	19.40	43.46
eelpout	176.00	17.00	9.38	#######	16.49	59.96
gobies	143.00	3.00	8.25	#######	14.52	74.48
plaice	174.00	81.00	5.48	#######	9.65	84.13
flounder	146.00	67.00	4.66	#######	8.20	92.32

(j)

Groups 6 & 1

Average dissimilarity = 40.28

	Group 6	Group 1				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
whiting	161.00	36.00	11.12	2.67	27.60	27.60
gobies	71.60	3.00	6.09	2.95	15.11	42.72
dab	69.00	4.00	5.73	1.24	14.23	56.95
eelpout	75.40	17.00	5.22	4.38	12.96	69.91
plaice	65.00	81.00	2.98	2.38	7.41	77.32
cod	46.80	61.00	2.20	1.80	5.46	82.78
flounder	68.60	67.00	1.85	2.00	4.60	87.38
pogge	105.00	99.00	1.84	1.78	4.57	91.95

(k)

Groups 5 & 2

Average dissimilarity = 50.48

	Group 5	Group 2				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
whiting	269.25	58.00	11.02	1.82	21.83	21.83
plaice	86.50	255.00	8.83	6.00	17.49	39.32
eelpout	210.75	47.00	8.46	2.28	16.76	56.07
cod	40.38	196.00	8.17	9.64	16.19	72.27
fatherlasher	22.63	86.00	3.32	3.83	6.59	78.85
gobies	64.50	15.00	2.67	1.62	5.29	84.15
dab	58.50	16.00	2.20	1.19	4.36	88.50
pogge	115.63	102.00	1.86	1.65	3.69	92.20

Table 3.4 cont.

(l)

Groups 3 & 2

Average dissimilarity = 66.38

	Group 3	Group 2				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
eelpout	637.00	47.00	21.45	11.47	32.32	32.32
whiting	443.50	58.00	14.00	7.65	21.09	53.41
plaice	80.50	255.00	6.35	5.42	9.56	62.97
cod	23.00	196.00	6.28	6.20	9.46	72.44
flounder	244.50	93.00	5.48	1.03	8.25	80.69
gobies	121.00	15.00	3.87	0.99	5.83	86.52
fatherlasher	9.50	86.00	2.78	17.62	4.19	90.71

(m)

Groups 4 & 2

Average dissimilarity = 36.39

	Group 4	Group 2				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
whiting	223.00	58.00	7.49	#######	20.57	20.57
eelpout	176.00	47.00	5.85	#######	16.08	36.66
gobies	143.00	15.00	5.81	#######	15.96	52.62
cod	293.00	196.00	4.40	#######	12.09	64.71
fatherlasher	4.00	86.00	3.72	#######	10.22	74.94
plaice	174.00	255.00	3.68	#######	10.10	85.04
flounder	146.00	93.00	2.40	#######	6.61	91.65

(n)

Groups 6 & 2

Average dissimilarity = 45.22

	Group 6	Group 2				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
plaice	65.00	255.00	11.80	4.74	26.09	26.09
cod	46.80	196.00	9.25	5.12	20.46	46.55
whiting	161.00	58.00	6.29	2.11	13.91	60.46
fatherlasher	17.80	86.00	4.21	4.69	9.31	69.76
gobies	71.60	15.00	3.45	2.30	7.63	77.39
dab	69.00	16.00	3.21	1.00	7.10	84.49
eelpout	75.40	47.00	1.73	1.95	3.83	88.32
flounder	68.60	93.00	1.51	1.02	3.35	91.67

(o)

Groups 1 & 2

Average dissimilarity = 39.20

	Group 1	Group 2				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
plaice	81.00	255.00	13.32	#######	33.98	33.98
cod	61.00	196.00	10.34	#######	26.37	60.35
fatherlasher	18.00	86.00	5.21	#######	13.28	73.63
eelpout	17.00	47.00	2.30	#######	5.86	79.49
flounder	67.00	93.00	1.99	#######	5.08	84.57
whiting	36.00	58.00	1.68	#######	4.30	88.87
gobies	3.00	15.00	0.92	#######	2.34	91.21

The major differences between the final two years of the time series, 1999 and 2000, which were highlighted initially by the raw abundance data, were confirmed when the same analyses were repeated using the percentage contribution of each species to total annual abundance. At the >60% similarity level, the cluster dendrogram delineated two groups, similar to those of the MDS plot (Figure 3.8b), and consisting of group 1 (all years except 1999 and 2000) and group 2 (1999 and 2000) (Figure 3.7b). The SIMPER analysis revealed that the decline in eelpout and whiting percentage contribution to the total annual abundance, allied with the increase in cod and plaice proportions in the annual abundance, were almost in equal part to account for approx. 64% of the dissimilarity in these groups (Table 3.5).

Table 3.5. Summary output of SIMPER analysis of >60% similar Agassiz percentage composition data, 1982 – 2000. Av. Abund = average abundance, Av. Diss = average Bray-Curtis dissimilarity, Diss/SD = average dissimilarity/standard deviation, Contrib% = percentage of total dissimilarity attributable to species, Cum.% = cumulative percentage dissimilarity. See text for details of years contained in groups.

Groups 1 & 2
Average dissimilarity = 47.58

	Group 1	Group 2				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
whiting	24.07	7.70	8.19	1.84	17.21	17.21
plaice	8.27	24.21	7.97	2.91	16.75	33.96
eelpout	20.21	4.74	7.74	1.51	16.26	50.22
cod	5.47	18.43	6.73	2.87	14.14	64.36
pogge	11.93	18.03	4.10	1.43	8.62	72.98
gobies	7.46	1.20	3.16	1.70	6.64	79.62
fatherlasher	2.01	7.00	2.59	1.90	5.44	85.06
flounder	10.19	13.51	2.46	1.36	5.17	90.23

When similarity levels of approximately >70% were investigated, four groups existed: group 1 (1982 – 1989, excluding 1985), group 2 (1985 and 1991 – 1997), group 3 (1990 and 1998) and group 4 (1999 and 2000) (Figure 3.7b). Eelpout accounted for greatest proportion of dissimilarity between group 1 and groups 2, 3 and 4 (Table 3.6a,b,d) and, despite low overall contribution to dissimilarity, was by far the most consistent discriminating species between groups 3 and 4, as illustrated by the high dissimilarity/S.D. ratio (Table 3.6f). Whiting's relatively great contribution to overall fish abundance throughout most of the 1990s, as well as 1985, represented as group 2, accounted for approx. 25% of the dissimilarity between this group and groups 3 and 4, when this species' contribution was lower (Table 3.6c,). Cod made a relatively large, though inconsistent contribution to dissimilarities between group 3 and groups 1 and 2 (Table 3.6b,c), while the proportion of total dissimilarity between group 4 and groups 1 and 2 were relatively lower, but the dissimilarity/S.D. ratios were the highest of all species, suggesting consistency over all comparisons between groups (Table 3.6d,e). The relatively high proportion of total fish abundance attributable to plaice in 1999 – 2000 (group 4) meant that this species was a major contributor of dissimilarity between this group and the remaining groups, at approx. 16 – 20% of total dissimilarity (Table 3.6d,e,f).

Both raw abundance and percentage composition exploratory analyses served to highlight changes in the composition of the lower Forth Estuary ichthyofauna in similar ways. The decline of an MJ species, whiting, as well as two resident taxa, eelpout and gobies, in combination with recent increases in MJ species plaice and cod, appeared to drive much of the observed differences in the 19-year dataset.

Table 3.6. Summary output of SIMPER analysis of 70% similar Agassiz percentage composition data, 1982 – 2000. Av. Abund = average abundance, Av. Diss = average Bray-Curtis dissimilarity, Diss/SD = average dissimilarity/standard deviation, Contrib% = percentage of total dissimilarity attributable to species, Cum.% = cumulative percentage dissimilarity. See text for details of years contained in groups.

(a)
Groups 1 & 2
Average dissimilarity = 31.48

	Group 1	Group 2				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
eelpout	30.72	12.70	9.01	2.38	28.62	28.62
whiting	19.38	30.55	5.67	1.33	18.00	46.62
dab	3.64	8.93	3.01	1.18	9.57	56.19
flounder	12.03	8.09	2.55	1.30	8.09	64.28
pogge	10.02	13.40	2.35	1.59	7.46	71.74
plaice	7.91	8.12	2.15	1.45	6.83	78.57
gobies	5.90	7.55	2.11	1.58	6.70	85.27
cod	3.42	5.30	1.40	1.06	4.45	89.72
sea snail	1.95	1.97	0.77	1.26	2.46	92.18

(b)
Groups 1 & 3
Average dissimilarity = 32.28

	Group 1	Group 3				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
eelpout	30.72	13.49	8.62	2.54	26.70	26.70
cod	3.42	13.26	5.16	1.10	15.97	42.67
gobies	5.90	12.56	3.33	1.70	10.32	52.99
pogge	10.02	12.75	3.22	1.53	9.98	62.96
whiting	19.38	14.55	2.65	1.32	8.21	71.17
plaice	7.91	10.14	2.24	1.50	6.95	78.12
flounder	12.03	12.17	1.52	1.40	4.70	82.82
fatherlasher	1.88	3.20	1.45	1.28	4.49	87.32
sea snail	1.95	3.49	1.25	1.40	3.86	91.17

(c)
Groups 2 & 3
Average dissimilarity = 31.77

	Group 2	Group 3				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
whiting	30.55	14.55	8.00	1.86	25.19	25.19
cod	5.30	13.26	4.85	1.17	15.26	40.44
dab	8.93	2.43	3.28	1.20	10.32	50.76
pogge	13.40	12.75	3.22	1.91	10.14	60.90
gobies	7.55	12.56	2.55	1.47	8.03	68.93
flounder	8.09	12.17	2.27	1.41	7.14	76.07
plaice	8.12	10.14	2.21	1.37	6.95	83.02
eelpout	12.70	13.49	1.54	1.40	4.85	87.87
fatherlasher	1.82	3.20	1.45	1.71	4.56	92.44

Table 3.6 cont.

(d)

Groups 1 & 4

Average dissimilarity = 49.77

	Group 1	Group 4				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
eelpout	30.72	4.74	12.99	3.82	26.10	26.10
plaice	7.91	24.21	8.15	2.97	16.37	42.48
cod	3.42	18.43	7.50	3.97	15.08	57.56
whiting	19.38	7.70	5.84	2.97	11.74	69.30
pogge	10.02	18.03	4.40	1.31	8.83	78.13
fatherlasher	1.88	7.00	2.67	1.94	5.36	83.49
gobies	5.90	1.20	2.35	1.30	4.72	88.21
flounder	12.03	13.51	2.10	1.43	4.21	92.42

(e)

Groups 2 & 4

Average dissimilarity = 48.33

	Group 2	Group 4				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
whiting	30.55	7.70	11.43	2.77	23.64	23.64
plaice	8.12	24.21	8.05	2.86	16.65	40.29
cod	5.30	18.43	6.56	3.00	13.58	53.88
eelpout	12.70	4.74	3.98	2.09	8.24	62.11
dab	8.93	1.38	3.78	1.37	7.81	69.92
pogge	13.40	18.03	3.72	1.52	7.69	77.61
gobies	7.55	1.20	3.24	2.13	6.70	84.31
flounder	8.09	13.51	2.98	1.39	6.17	90.49

(f)

Groups 3 & 4

Average dissimilarity = 36.92

	Group 3	Group 4				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
plaice	10.14	24.21	7.04	2.39	19.06	19.06
gobies	12.56	1.20	5.68	6.18	15.38	34.44
cod	13.26	18.43	4.67	1.33	12.65	47.09
pogge	12.75	18.03	4.63	1.47	12.53	59.62
eelpout	13.49	4.74	4.37	16.47	11.85	71.47
whiting	14.55	7.70	3.42	2.01	9.27	80.74
fatherlasher	3.20	7.00	2.30	1.41	6.22	86.96
flounder	12.17	13.51	1.63	1.74	4.42	91.38

3.3.1.4. Population abundance variability in relation to latitude

The rank of species collected in the Forth based on predicted CVSDs (coefficients of variation of standard deviations) of annual abundance differed somewhat from that predicted based on the relative proximity of each species to the centre of its geographical range (Table 3.7).

Table 3.7. Ten most abundant species caught by Agassiz trawl, lower Forth Estuary, 1982-2000 (excluding 1986): coefficient of variability of standard deviation (CVSD) of mean annual abundance observed ranking compared to that hypothesised, then compared to Severn Estuary data (Henderson and Seaby, 1999). Centres of range derived from Wheeler (1969). 'Gobies' = *Pomatoschistus* spp., assumed comparable to sand goby of Severn data.

species	latitudinal range (centre), °N	Forth CVSD (%)	observed rank (*cf* predicted)	Severn CVSD (%)	observed relationship of Forth to Severn CVSDs (*cf* predicted)
pogge	48 – 73 (60.5)	32.79	1 (7)	70.95	< (<)
plaice	35 – 70 (52.5)	56.95	2 (4)	116.64	< (>)
whiting	43 – 73 (58)	57.86	3 (1)	50.55	> (<)
flounder	30 – 73 (51.5)	67.95	4 (6)	42.77	> (>)
'gobies'	37 – 72 (54.5)	69.87	5 (2)	55.49	> (<)
dab	46 – 73 (59.5)	80.73	6 (5)	60.47	> (<)
sea snail	50 – 73 (61.5)	88.10	7 (8)	70.48	> (<)
eelpout	51 – 73 (62)	91.52	8 (9)	NA	-
fatherlasher	45 – 73 (59)	95.01	9 (3)	NA	-
cod	48 – 80 (64)	109.36	10 (10)	117.80	< (<)

Pogge CVSD was the lowest, despite being predicted to be relatively high. Dab, sea snail and flounder were within one place of their predicted ranks, while cod exhibited greatest variation in annual abundance, as predicted. Plaice, whiting, gobies and flounder showed moderate agreement with predictions, while fatherlasher rank was considerably lower than predicted (Table 3.7).

Of eight species comparisons with Severn estuary data, only three were of the nature hypothesised (Table 3.7). Pogge catches in the Forth were far less variable than those in the Severn, while the opposite was true of flounder, both as predicted. The CVSDs of cod and whiting in the two estuaries were relatively similar, while respectively nominally fulfilling and rejecting the hypothesised relationships. The CVSDs of plaice, gobies, dab and sea snail were markedly different from the predicted relationships (Table 3.7).

3.3.2. Pelagic study, April 1999 – January 2001.

3.3.2.1. Species composition of catches and approximate clupeid abundances

Sprat and herring dominated pelagic trawl catches between April 1999 and January 2001, as expected. These two species accounted for 7019 individuals of a total pelagic abundance of 7842 fish, or 89.5%. The clupeid catch was dominated by sprat, with 4959 fish (63.2% of total pelagic abundance), while herring contributed 2060 individuals (29.3%). The remaining fish sampled were mostly whiting (569 individuals, 8.2% of total pelagic trawl abundance) and smelt (51 individuals). Almost 90% of whiting captured (500 individuals) were taken at LW. Other benthic and demersal species were taken, usually at LW, probably attributable to the gear dipping towards the bottom.

Mean estimated abundance of sprat in the Forth Estuary between Dunmore and the Forth Bridges from April 1999 – January 2001 was approx. 1.31×10^7 individuals (arithmetic 95% CIs: $6.00 \times 10^6 - 2.01 \times 10^7$; geometric 95% CIs: $8.10 \times 10^6 - 2.11 \times 10^7$) when calculated using the arithmetic mean of abundances per unit volume trawled, and 2.02×10^6 (95% CIs: $1.25 - 3.26 \times 10^6$) when using the geometric mean. Herring mean abundance was estimated at 5.30×10^6 individuals (arithmetic 95% CIs: $2.47 - 8.13 \times 10^6$; geometric 95% CIs: $3.40 - 8.24 \times 10^6$) by arithmetic mean calculation, and 9.94×10^5 (95% CIs: $6.39 \times 10^5 - 1.55 \times 10^6$) using the geometric mean of herring caught per unit volume trawled. Maximum and minimum estimated abundances over the study period are provided in Table 3.8. Estimated abundances in the ten months during which trawling took place are illustrated in Figures 3.9 and 3.10.

Table 3.8. Months with estimated maximum and minimum clupeid population sizes, mid-lower Forth Estuary, April 1999 – January 2001. Data estimated from pelagic trawling.

	maximum abundances		**minimum abundances**	
	arithmetic mean (geometric 95% CIs)	geometric mean (95% CIs)	arithmetic mean (geometric 95% CIs)	geometric mean (95% CIs)
sprat	April 2000: $\mathbf{3.45 \times 10^7}$ ($1.27 \times 10^6 - 9.39 \times 10^8$)	January 2000: $\mathbf{1.18 \times 10^7}$ ($5.95 \times 10^6 - 2.35 \times 10^7$)	September 2000: $\mathbf{1.93 \times 10^6}$ ($1.57 - 2.37 \times 10^6$)	December 2000: $\mathbf{7.33 \times 10^5}$ ($6.22 \times 10^4 - 8.65 \times 10^6$)
herring	January 2001: $\mathbf{1.41 \times 10^7}$ ($1.81 \times 10^6 - 1.10 \times 10^8$)	January 2001: $\mathbf{2.05 \times 10^6}$ ($2.62 \times 10^5 - 1.60 \times 10^7$)	December 2000: $\mathbf{1.14 \times 10^6}$ ($2.32 \times 10^5 - 5.64 \times 10^6$)	December 2000: $\mathbf{3.55 \times 10^5}$ ($7.21 \times 10^4 - 1.75 \times 10^6$)

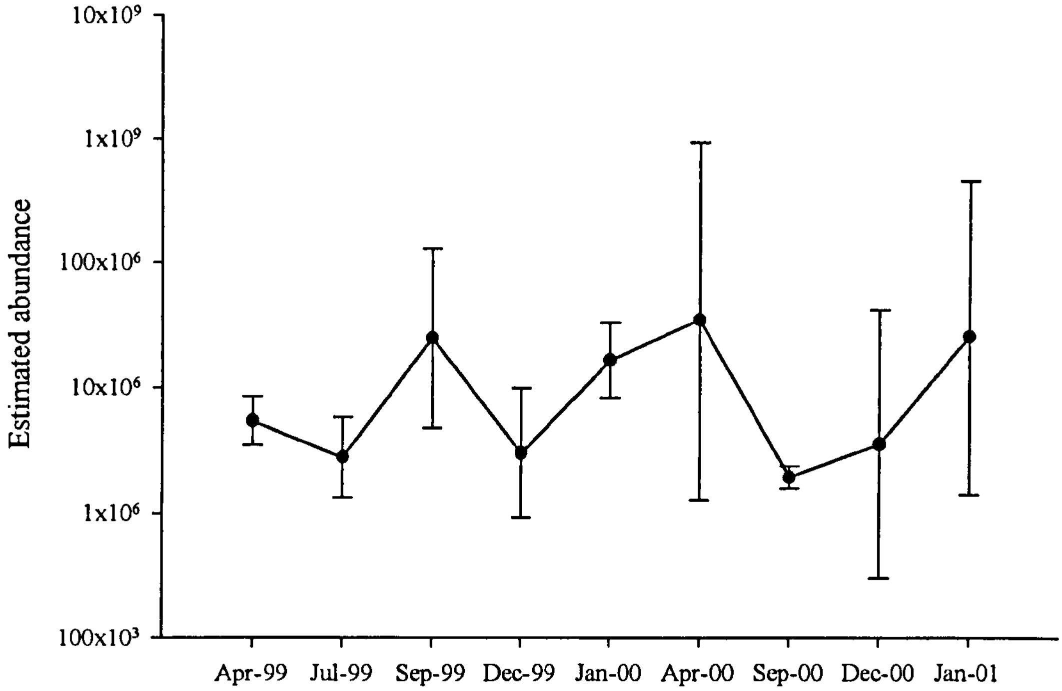


Figure 3.9. Estimated abundances of sprat in the Forth Estuary (Dunmore – Forth Bridges), April 1999 – January 2001, based on pelagic trawl catches Data for July 2000 excluded due to gear damage (see text). Values are extrapolations of arithmetic means of 6 trawls per month, ± 95% geometric CI.

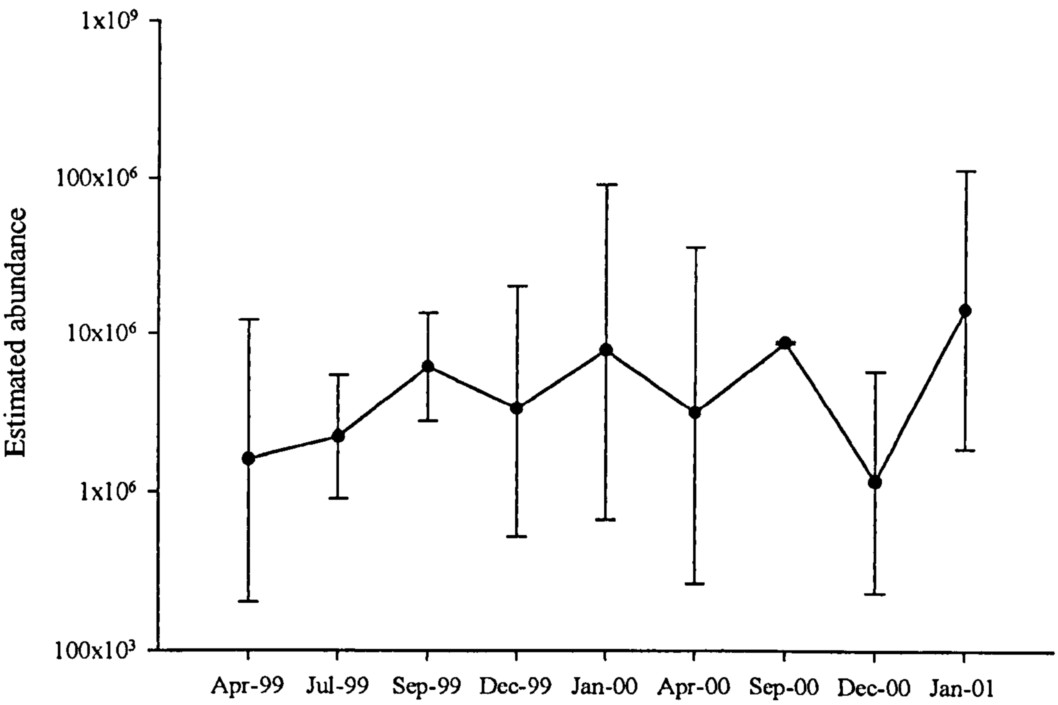


Figure 3.10. Estimated abundances of herring in the Forth Estuary (Dunmore – Forth Bridges), April 1999 – January 2001, based on pelagic trawl catches. Data for July 2000 excluded due to gear damage (see text). Values are extrapolations of arithmetic means of 6 trawls per month, ± 95% geometric CI.

3.3.2.2. Generalised Linear Modelling of sprat and herring abundance

GLMs of sprat and herring abundances in relation to the predictor factors of tide, month and trawl station explained relatively little of the null deviance, as was the case with the Agassiz and power station models. Sprat abundance was significantly related to tidal state, LW having a directly proportional relationship with numbers captured, while HW was inversely proportional (Table 3.9a). Month was the only other factor that was significant, and the greatest positive relationship with abundance was shown by month 2 (April/May), followed by months 1 (January) and 4 (September). Inverse proportionality was demonstrated with the months of July and December (3 and 5 respectively), with the former showing the stronger relationship (Table 3.9a).

Herring abundance was significantly related to all of the factors and interactions modelled, with the exception of the station × month interaction (Table 3.9b). Tide was only included in the model due to the significance of both of its interaction terms, while it was actually marginally non-significant ($p = 0.07$). The nature of the relationship was as exhibited by sprat, LW positively related to abundance (Table 3.9b). Station was weakly significant, with direct proportionality between herring abundance and trawls carried out at Longannet and Port Edgar (the former being more positively related than the latter), and a negative relationship was suggested with data collected at Tancred (Table 3.9b). Sampling in months 4 (September) and 1 (January) was most positively related to herring abundance, while April, June and December samples were inversely proportional to abundance (Table 3.9b).

Table 3.9. Summary of results of GLMs on lower Forth Estuary pelagic trawl dataset, April 1999 – January 2001 (n=57 trawls). 'd.f.' is degrees of freedom; 'deviance' is amount of deviance explained by predictor; 'significance': * = $p<0.05$, ** = $p<0.01$, *** = $p<0.001$, NS = not significant ($p>0.05$); 'relationship': + signifies direct proportionality with dependent variable, - signifies inverse proportionality, relative strengths of factor levels indicated. Station: PE = Port Edgar, T = Tancred, L = Longannet. Month: 1 = January, 2 = late March/April/early May, 3 = late June/July/early August, 4 = September/October, 5 = December. Both models utilised negative binomial distributions. NULL signifies total deviance in null model.

(a) sprat	d.f.	deviance	significance	relationship
NULL		97.4		
tide	1	13.6	***	+: LW -: HW
station	2		NS	
month	4	28.5	***	+: 2>1>4 -: 3>5
tide × station	2		NS	
tide × month	4		NS	
station × month	8		NS	

(b) herring	d.f.	deviance	significance	relationship
NULL		131.8		
tide	1	3.3	p = 0.07	+: LW -: HW
station	2	7.3	*	+: L>PE -: T
month	4	20.7	***	+: 4>1 -: 2>3>5
tide × station	2	6.1	*	See Figure 3.11a
tide × month	4	22.4	***	See Figure 3.11b
station × month	8		NS	

The interaction of tide × station explained least proportion of null deviance per degree freedom of the significant factors, and suggested HW abundance to be slightly greater than LW only at Port Edgar, while the abundance at Longannet at LW was much greater than at HW (Figure 3.11a). The other significant interaction, tide × month, explained the greatest proportion of null deviance, and showed LW catches to be much greater than those at HW in January, and slightly greater in April and September, while in the other two months HW yielded slightly more herring (Figure 3.11b).

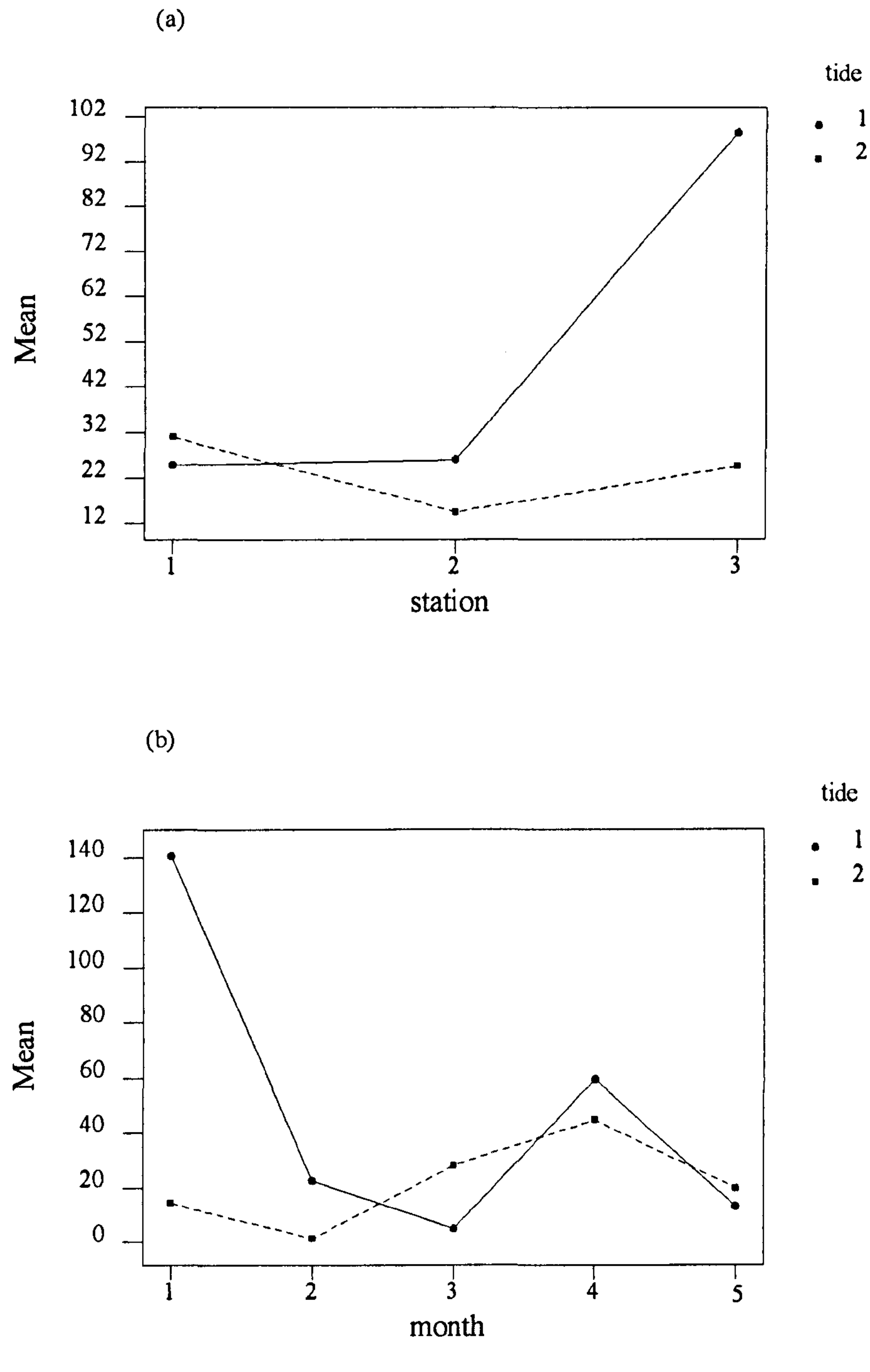


Figure 3.11. Plots of significant interaction terms from GLMs of herring abundance, lower Forth Estuary: (a) tide × station, (b) tide × month. Tide: 1 = LW, 2= HW. Station: 1 = Port Edgar, 2 = Tancred, 3 = Longannet. Month: 1 = January, 2 = April, 3 = July, 4 = September, 5 = December. Mean = mean abundance per trawl.

3.4. Discussion

3.4.1. Assumptions made during trawling

Various assumptions were made in the Agassiz and pelagic trawl studies. Assessment of relative abundance, such as that between each trawl in the present study, depended on the assumption that the relationship:

$$\frac{\Delta C}{\Delta f} = \frac{qN}{A}$$

holds true, where C = catch, f = trawling effort, A is area or volume occupied by the stock, N = total stock abundance, and q is the catchability coefficient (Gee, 1983). Intrinsic and extrinsic factors affect the catchability coefficient of trawls: it is based upon two elements, q_1 (gear efficiency) and q_2 (availability and vulnerability of fish to the trawl) (Gee, 1983). Gear efficiency (q_1) is assumed to be constant throughout a tow, but in practice may vary, for example due to a change in substrate. Availability of fish to trawling (q_2) is a measure of the proportion of fish in a study area that are likely to be in the path of the trawl at the time of sampling. Estimation of availability may assume an even spatial distribution throughout the study area, but aggregations of fish on a small scale (*e.g.* in shoals; Misund, 1994) or larger scales (*e.g.* migration to intertidal areas for feeding purposes; Wolff *et al.*, 1981) may significantly affect such an assumption. The known diel vertical migrations of herring, *i.e.* towards the sea surface at dusk and towards the sea bed at dawn (Blaxter and Batty, 1990), may have affected estimates of abundance from midwater trawls in the present study, assuming herring shoaling behaviour in the Forth Estuary to be similar to that in the ocean. Michalsen *et al.* (1996) provided evidence that q_2 is related to tidal rhythm for Barents Sea cod and haddock, since the vertical distribution of the species varies with tidal current; fish descended as tidal current decreased in speed, and so were more available to bottom trawl capture at this phase of the tide. This effect seemed to be advanced and expanded if co-occurring with sunrise, illustrating that the interaction of these two environmental influences is important.

Many studies simply state efficiency of the gear to indicate catchability coefficient. Studies of the Forth Estuary fish using beam and Agassiz trawls assumed an efficiency of 33% (*i.e.* catchability coefficient of 0.33), based on Kuipers' (1975) study that assessed the efficiency of a 2 m beam trawl in catching juvenile plaice. Efficiency of the same gear is likely to differ between species (see above), as shown by Kjelson and Johnson (1978) for a 6.1 m otter trawl used in sampling marked fish of a known density in an enclosed North Carolina tidal embayment: juvenile pinfish, *Lagodon rhomboides*, were sampled at a rate of 48%, whereas juvenile spot, *Leiostomus xanthurus*, experienced a sampling efficiency of 32% (*i.e.* catchability coefficients of 0.48 and 0.32, respectively). Precision of measurement of efficiency in the study was variable, with the standard error ranging from 9 – 58% of the mean.

Fish behaviour in the process of capture by trawls is of great commercial and scientific interest. Vulnerability of fish available for sampling (*i.e.* those in the path of an oncoming trawl) varies inter- and intraspecifically. Initially, the sound of the vessel may cause an avoidance response; demersal fish in the pelagic zone typically swim downwards (possibly increasing the proportion caught by demersal trawls), whereas pelagic fish tend to swim off at right angles to the vessel's motion (Godø, 1994). Vision is important to most fish (Guthrie and Muntz, 1993), and perception of trawl components often depends on fishes' visual abilities and the colour of the trawl used. Wardle (1993) suggests pelagic nets to be most effective in catching fish if fitted with a black top, white base and black and white striped sides. High turbidity in the Forth (see section 2.3.1) may have caused the lack of this optimum colour of netting to not be as significant as during fishing in clearer waters.

When fish are already in the mouth of the trawl, it is still not a foregone conclusion that they will be captured. This was mentioned in section 2.4.3.1 as analogous to avoidance of removal in power station CW intake flows (Turnpenny, 1988a). Smaller fish are better at escaping beneath demersal trawls, an effect reduced by fitting tickler chains to the bottom of the trawl's mouth (Godø, 1994). No tickler chains were fitted to the Agassiz trawls in the present study. Haddock were shown to escape upwards over the head of the trawl more consistently than cod (Godø, 1994). Scaling effects are of great importance when fish are in the mouth of the trawl, for larger fish are more likely to escape capture by enduring pursuit from the trawl by swimming in the same direction as the trawl; small fish need to swim more energetically to attain the same speed, and so tire more easily and are caught (Wardle, 1993). Smaller fish may pass through the mesh of the net's cod end, although this is less likely with the small mesh sizes used for scientific purposes, as in the present study. Godø *et al.* (1990) hypothesised that catch rates of large fish would decrease with increasing trawl duration, due to the greater swimming capacity of large fish compared to smaller individuals. Testing this with varying trawl durations did not, however, produce significant change in mean length of fish caught, for the species tested (cod, haddock and long rough dab).

In general, trawl surveys are limited in their accuracy by the following assumptions (Godø, 1994):

i) variability in gear selection for size or species of fish is constant, so that comparisons can be made between surveys subject to certain constraints;

ii) the size of the sampling unit (*i.e.* trawling effort) is constant under all conditions;

iii) the trawl performs equally well in differing conditions such as depth, weather and bottom substrate type.

The Agassiz and pelagic trawling carried out in the present study assumed these conditions, though testing of gear efficiency and related features would be needed to quantify errors associated with the techniques. Resource constraints have prevented this from being carried out adequately thus far.

3.4.2. Trends in demersal and benthic species' abundances and changes in composition of the lower Forth Estuary ichthyofauna

Total number of benthic and demersal species collected over the 19-year period of Agassiz trawling was thirty, which, when the occasional pelagic species captured were also included, is close to the quantity of species expected for a temperate estuary of this latitude (Henderson, 1989; see Chapter 2). Considerable changes in the composition of the ichthyofaunal assemblage were observed to have taken place over the time period, culminating in a fish community structured quite differently in 1999 – 2000 from that observed in previous years. While these two years were very distinct from all others, they were also quite dissimilar to each other. The change stemmed from 1999 possessing the least total quantity of species and total abundance of fish over the entire period: it was a poor year for many species, with abundances of whiting, gobies, eelpout and dab being at minimum values for the 19-year period. In addition, cod and plaice abundances in 2000 were at second highest and highest levels observed, respectively. The exploratory non-parametric multidimensional analyses employed in the present study suggested that most differences in the composition of the Forth

Estuary ichthyofauna were due to the differing abundances of whiting, eelpout, cod and plaice. Analysis using percentage of the total abundance attributable to each species showed that, apart from the very different years of 1999 and 2000, there was generally a split between most years sampled in the 1980s and those from 1990 onwards. Whilst no trend in benthic or demersal species richness over the study period was apparent, there was a statistically significant decrease in abundance of fish captured during trawling. This was attributable largely to the significant declines in abundance of whiting and eelpout, which were the most abundant species taken over the entire period, but with greatly reduced numbers towards the end of the time series. There were no other species exhibiting statistically detectable linear trends in abundance, though more species (flounder, gobies, dab, sea snail) showed non-significant negative trends in abundance than positive (plaice, cod), and this would also have contributed to depressed total fish abundances as the time series progressed. It is important to consider whether the decline in fish abundance in the mid-lower Forth Estuary may have been a local or regional phenomenon.

Eelpout was the second most abundant species sampled in 19 years of Agassiz trawling, and showed the most significant negative trend in abundance of the ten most common species sampled. From 1981 – 1989, eelpout was estimated to be the most abundant fish species in the Forth Estuary, with a long-term average of 3.26×10^6 individuals (Elliott *et al.*, 1990). The peak in abundance of the Forth Estuary eelpout in 1988 was followed by a steady decline into the 1990s with abundance being half or less than the previous decade, and a downward trend clearly in evidence (Figure 3.4f). This was very similar to the trend in eelpout catch per unit effort (CPUE) observed in inshore areas near the Ringhals Nuclear Power Station, Kattegat, Sweden (Figure 3.12) (S.

Thörnqvist, Swedish National Fisheries Board, personal communication), located on the eastern edge of the North Sea at approximately the same latitude as the Forth.

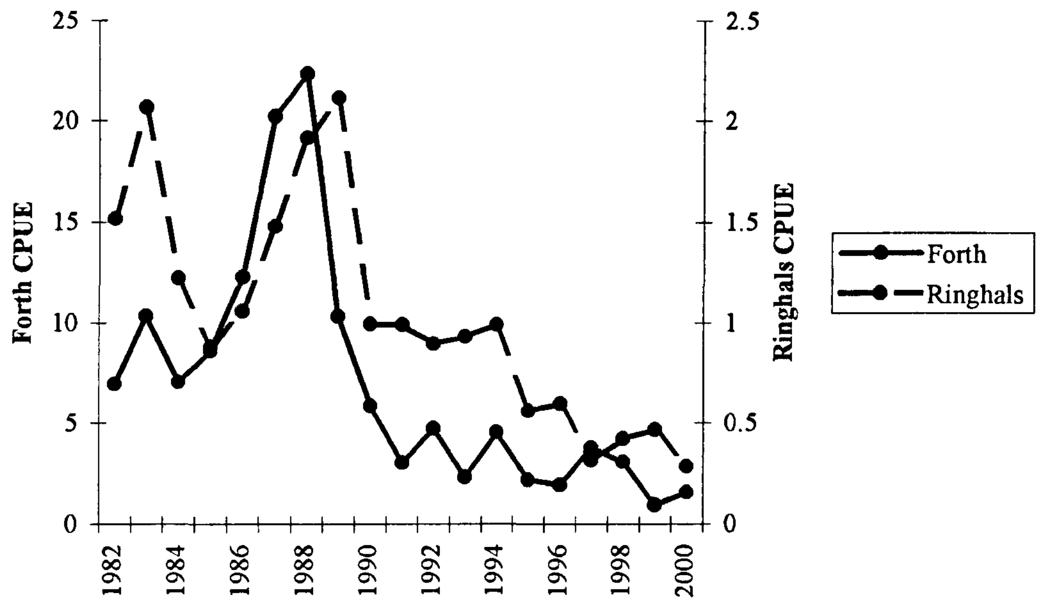


Figure 3.12. Similarity in CPUE of eelpout, lower Forth Estuary and Swedish inshore waters off Ringhals Nuclear Power Station. Data are mean number of fish per trawl (Forth) or per fyke net night (Ringhals). Data from this study and Thörnqvist (personal communication).

Peaks of eelpout abundance in 1987 were recorded during trawl sampling in inshore regions of the east English coast between Flamborough Head and North Foreland, followed by catches in the 1990s that were very small compared with the previous decade (Rogers *et al.*, 1998). The most likely reason for the decline of eelpout at these locations, which are relatively near to the southern edge of its range (Wheeler, 1969), is a general upward trend in water temperatures over the past two decades (Henderson and Seaby, 2001), associated with warming of the global atmosphere. The greatest ever mean temperature anomalies of both land and ocean were observed in 1998, as illustrated for most of the northern hemisphere in Figure 3.13.

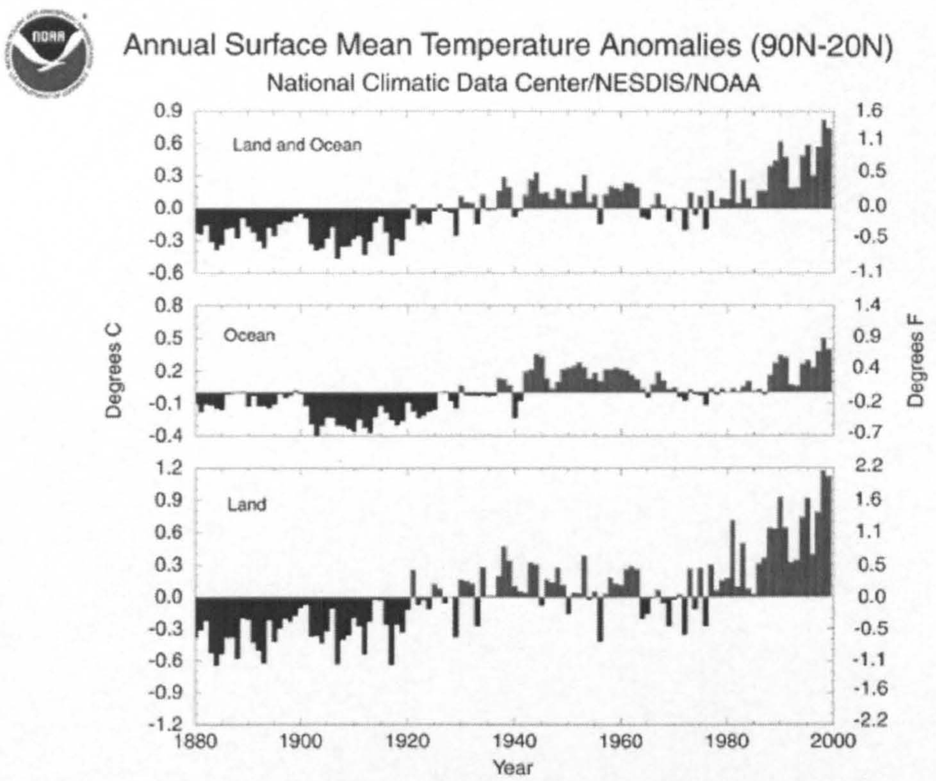


Figure 3.13. Annual Surface Mean Temperature Anomalies, 1880 – 1999, for latitude 20 - 90°N. Reproduced from http://www.ncdc.noaa.gov/ol/climate/research/1999/ann/ann99.html (National Oceanic and Atmospheric Administration National Climatic Data Center).

The thermal tolerance of eelpout is closely correlated with the southern limit of its range, and an increase in temperature would be expected to cause a northward shift in distribution (van Dijk *et al.*, 1999). The importance of eelpout abundance in indicating possible climate change was the subject of the EU project "Effects of climate induced temperature change on marine coastal fishes", results of which are in preparation. Adequate year-round water temperature data for the Forth Estuary are lacking, but significant negative correlations between the natural logarithm of annual mean of monthly abundance of eelpout in the Forth and January water temperature one year earlier of two offshore North Sea stations at similar latitudes were observed (Table 3.10).

Table 3.10. Significant negative correlations between natural logarithm of eelpout CPUE and sea bottom temperature in the previous year, based on Agassiz trawl data from 1981 – 2000 (n = 19 years). Values are Pearson correlation coefficients with associated probabilities in parentheses. Temperature data from ICES Oceanographic Database and Services, http://www.ices.dk/ocean/

	Bottom sea temperature in January of previous year	
	Approx. 55°N	Approx. 57°N
ln (annual mean of eelpout monthly abundance)	-0.686 (p = 0.001)	-0.621 (p = 0.005)

The major peaks in eelpout abundance in both Forth and Kattegat data (Figure 3.12) would seem to agree reasonably well with the low temperature anomalies of the mid-1980s, while the period of eelpout decline between 1990 and 2000 coincides with the most positive temperature anomalies in the 120-year time series (Figure 3.13). Given that the period from December – February is when eelpout are born (Miller and Loates, 1997), the temperature at this time may be of great importance in determining survival of young fish. Eelpout reach 140mm and 200mm TL by the end of the first and second years of life respectively (Wheeler, 1969), and the length-frequency distribution of eelpout caught in the lower Forth Estuary suggests that most fish were of these ages (Figure 3.14). This may partially explain the lag of one year in the negative correlation of eelpout abundance and January water temperature.

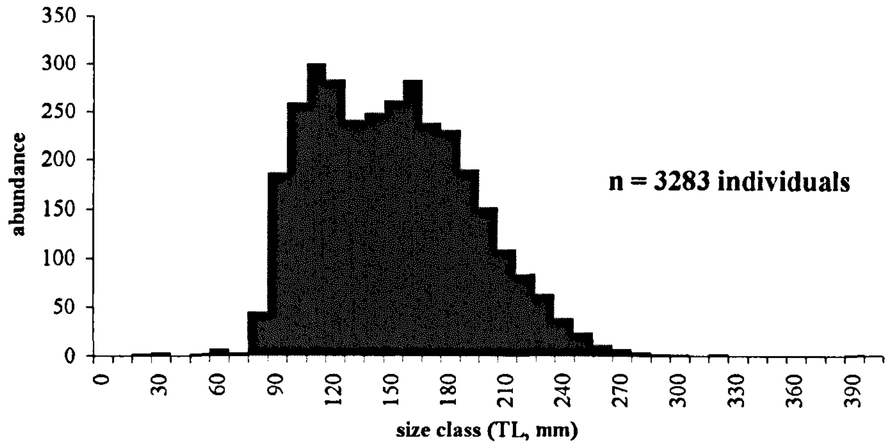


Figure 3.14. Length-frequency distribution of eelpout sampled from the lower Forth Estuary, 1987 - 2000.

Were eelpout abundances negatively correlated to temperature in the same year, rather than the previous year, an alternative explanation to decreasing abundances could be movement of adults to cooler waters offshore, beyond the range of the sampling gear, as was illustrated by sea snail in the Severn Estuary (Henderson and Seaby, 1999). This species was generally low in abundance during the relatively warm winter of 1996 – 1997, with the exception of the month of January, when temperatures were unusually cool (4.5°C) and sea snail abundances were 1 – 2 orders of magnitude greater than in the preceding or following months (Figure 3.15). Sea snail reproduce in spring, so that this pulse of increased abundance could only have been explained by inshore movement (Henderson and Seaby, 1999). With increasing temperatures in the Severn, as elsewhere in the northern hemisphere and indeed the world, annual sea snail catches in this location steadily declined from 1981 - 2001, probably due to increased use of deeper waters away from the sampling point (Henderson and Seaby, 2001). The same was not true of the Forth, with abundances captured in 1998 being the second greatest recorded, after 1988, and lack of any significant positive or negative trends in abundance over the same time period (Figure 3.4j). As previously noted, 1998 was the warmest year so far (Figure 3.13), and natural logarithm of total annual sea snail abundance in Agassiz trawls was significantly *positively* correlated with January bottom water temperature in the same year at the 55°N North Sea station mentioned above (Pearson correlation coefficient = 0.502, p = 0.029). This unexpected result may have been a spurious correlation, but can largely be explained by high sea snail abundances in a relatively few years when temperatures were unusually high.

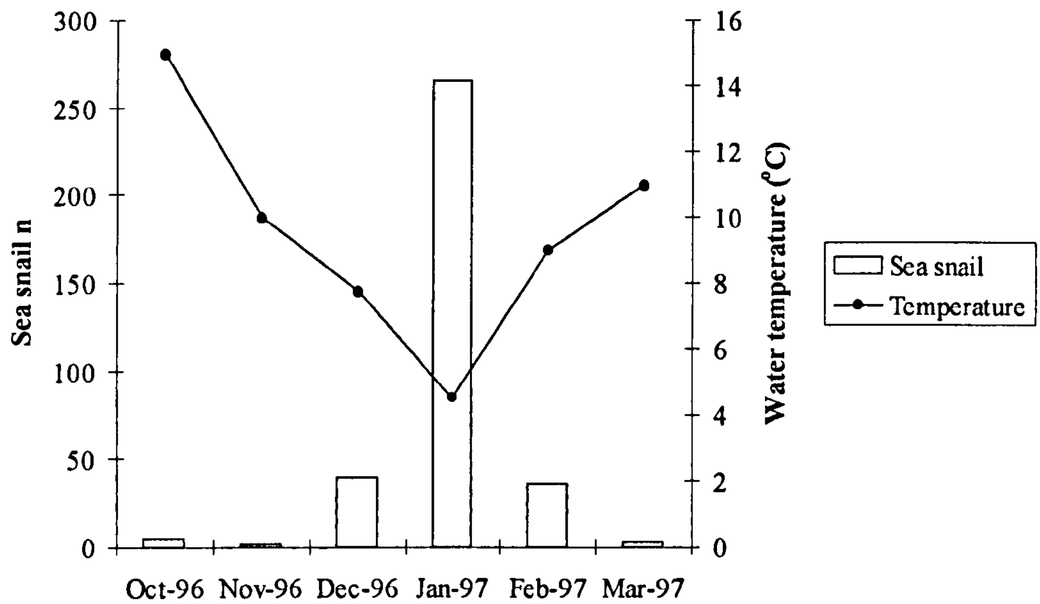


Figure 3.15. Evidence for possible inshore-offshore migration controlling abundance of sea snail impinged at Hinkley Point B power station, Severn Estuary. Data from Henderson and Seaby (1999).

While not as clear, a statistically significant decline in Agassiz trawl catches over the study period was exhibited in whiting as for eelpout, and the mean catches in the final three years of the dataset were the lowest recorded. No such significant decrease was observed in either or cod or plaice. These three species enter the Forth from the North Sea primarily as juveniles (Elliott *et al.*, 1990), so one might expect reasonable agreement between stock assessment data from the North Sea and adjacent areas and Agassiz data from the Forth. While spawning stock biomass (SSB) of all three species declined significantly from 1981 – 2000 (ICES 2001a), only plaice and cod recruitment of age 1 fish showed a significant linear fall over the same time period (Figure 3.16). It seems counter-intuitive that whiting was the only species of the three not to exhibit a significant decline in North Sea abundance of recruits, while the juveniles of the same species were unique in showing a decline in numbers in the Forth Estuary. Cod and plaice recruitment significantly declined in the North Sea, but showed no such trend in the Forth, exactly the opposite effect to whiting. Availability of data more specific to the east coast of Scotland may correct these unusual observations, as the data in Figure 3.16 integrate results from many countries and regions of the North Sea. Processes governing abundance of these 0+ MJ species are most likely to be of importance beyond inshore areas, since spawning and early life history prior to metamorphosis occurs in offshore regions of the North Sea, *e.g.* at depths of about 37m in the southern North Sea in the case of plaice (Wheeler, 1969). There is evidence that recruitment to age 1 of some commercially important species is governed by climatic factors. This was suggested by Fox *et al.* (2000) in an assessment of Northeast Atlantic plaice, that egg mortality is increased at higher temperatures and therefore that lower temperatures would favour increased recruitment. This could explain the rather depressed levels of plaice in the Forth during the 1990s (Figure 3.4c), but not the high values of 1990 and 2000 when rather high positive temperature anomalies were observed at the time of spawning (Figure 3.13). Planque and Frédou (1999) used a meta-analysis of Atlantic cod stocks to show that recruitment was positively related to temperature in northern stocks, negatively related in southern stocks, and that there was no relationship between recruitment and temperature in stocks near the middle of the temperature range. The exploitation of important commercial species may complicate the trends in recruitment of the MJ species, something that would not be the case for species that are not the subject of capture fisheries, such as eelpout.

It was noteworthy that in 1999, the year following the warmest on record (1998; Figure 3.13), species richness and total abundance of demersal and benthic fish captured by Agassiz trawling were both at lows for the 19-year time series (Figure 3.3). Negative impacts of warm winter temperatures causing reduced numbers of eelpout have already been suggested and it may be possible that this phenomenon produced low recruitment in a number of other species, such as goby and dab. Warm temperatures in 1999 may have discouraged estuarine use by MJ and MA species and influenced increased residence in cooler waters outside the estuary, for 1999 was only marginally less warm than 1998 (Figure 3.13). The same arguments could be applied to increased abundances of fish in the year 2000, being due to cooler waters in either 1999 or 2000 affecting recruitment or inshore-offshore movements respectively, or a combination of the two.

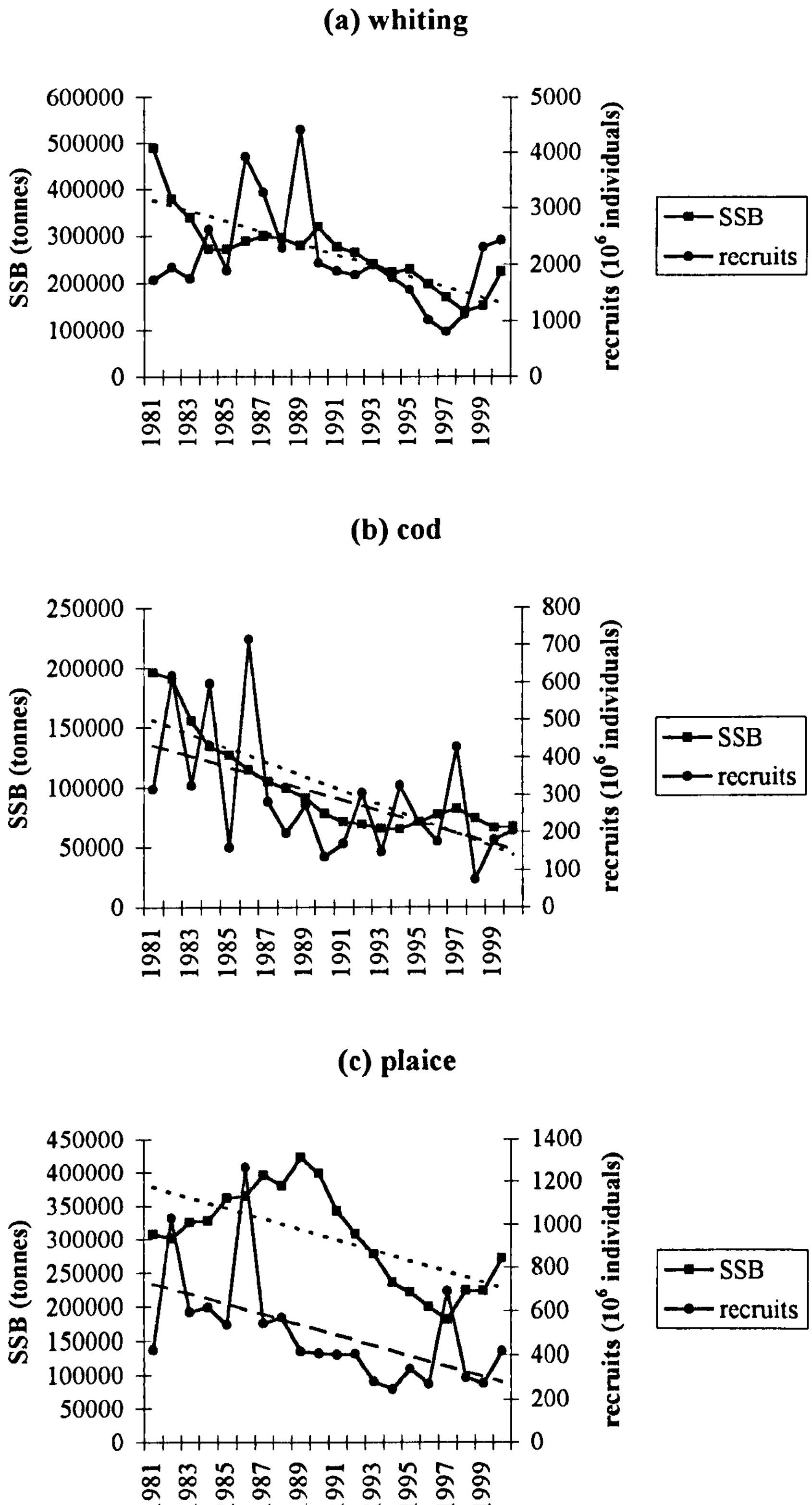


Figure 3.16. Spawning stock biomass (SSB) and recruits at age 1 of (a) whiting in ICES areas IV and VII D, (b) cod in ICES subarea IV and divisions VII D and III A , and (c) plaice in ICES area IV (North Sea). Significant negative linear trends for SSB (- - -) and recruitment (— — —) are indicated. Data from ICES (2001a).

How do changes in the Forth's ichthyofaunal assemblage compare with those observed in the Severn region over a similar time period? The significant decline in total abundances of benthic and demersal species in the Forth contrasted strongly with the increase observed in the outer Severn Estuary, where total abundance of fish taken during standardised monthly sampling at Hinkley Point A in 2000 – 01 had risen significantly to three times its level in 1981 (Henderson and Seaby, 2001). This agreed with the approx. fourfold increase in total abundance of all fish sampled from the intake screens of Oldbury Power Station in the mid-Severn estuary when comparing the periods July 1972 – June 1977 and July 1996 – June 1998 (Potter *et al.*, 2001). Although these Severn power station impingement data included pelagic species, which showed positive trends in abundance, increases in two major gadoid species, whiting and cod, were also evident in the Hinkley data (Henderson and Seaby, 2001). Mean annual abundance of whiting on Oldbury intake screens was more than twice as great in 1996 – 1998 than from 1972 – 1977. Some species that did not exhibit any long-term trends in the Forth, such as flounder, also seemed to be increasing in the outer Severn, though Henderson and Seaby (2001) state that further years of data are needed to confirm this statistically. There was a "modest" decrease in abundance of flounder in the mid-Severn Estuary (Potter *et al.*, 2001), paralleling a decrease in the Thames Estuary that may be attributable to increased predation of 0+ individuals by ctenophores (Thomas, 1998). This decline was not true of sand goby, which exhibited a 4.2 × increase in mean annual abundance between the 1970s and 1990s (Potter *et al.*, 2001). The low numbers of dab in the final three years of the Hinkley Point study are similar to those observed in the Forth and reductions in abundances with increasing water temperature were suggested to be likely (Henderson and Seaby, 2001). Species richness in the outer Severn also increased, due to increased occasional captures of warmer water species. The abundances of whiting, cod and plaice have increased between 1981 – 2000, with the enhanced numbers of cod being suggested as "curious" by Henderson and Seaby (2001), given the decline in North Sea stocks noted above, and the increasing temperatures within the Severn. Abundance of 0-group cod sampled at Hinkley Point in 2000 – 01 was the highest on record. It is suggested that conditions in both the outer and inner Severn Estuary have generally become more favourable for fish (Potter *et al.*, 2001; Henderson and Seaby, 2001), and this is reflected in the increases in total abundance over the last 20 – 30 years. Potter *et al.* (2001) believe reductions in emissions of industrial effluents such as cadmium over the past 20 – 30 years may have had more of an influence on improving water quality for fish in the Severn than reductions in organic waste disposal. This was due to the Severn generally being well-mixed because of its great tidal range and so not being as likely to suffer oxygen depletion problems characteristic of other previously polluted estuaries such as the Thames (Thomas, 1998). Further discussion of this subject is given in Chapter 5.

3.4.3. Temporal, spatial and tidal influences on abundance of demersal and benthic species

The fact that random selections of data used to rerun GLM analyses provided almost exactly the same results as the original models utilising the full dataset gave confidence that the results were of biological significance and not merely spurious relationships caused by sampling errors. Statistical analysis of potentially important factors influencing abundance of fish caught in Agassiz trawls confirmed that the temporal trends in abundance of fish captured in the mid-lower Forth Estuary in this study compared favourably to previous studies in this estuary and similar temperate locations. Examination of all GLMs formulated allowed several observations to be made. As noted by Elliott *et al.* (1990), there was a marked seasonality of fish abundance in the Forth. The abundances of fish caught in September – October and December were found to be higher than at other times of the year, this being attributable to the influxes of 0+ MJ migrants over autumn and winter. Fish abundance tended to be least in April and June/July, due primarily to emigration of the same MJ species. These trends were in good agreement with peaks and troughs in total estimated fish biomass impinged at LPS (Figure 2.5). Possible reasons for increased abundance of fish in the estuarine habitat during these months were discussed in section 2.4.3.2. Temporal trends in abundance confirmed the hypothesis that they would reflect trends seen by other workers in the Forth (Elliott and Taylor, 1989; Elliott *et al.*, 1990). This was unsurprising given that the data used in the present study were merely an extension of the same dataset. Abundances of fish over the 19-year period also exhibited comparable trends with the 1999 – 2000 dataset from LPS (Chapter 2).

While the ten species analysed consisted nominally of four MJ (whiting, cod, plaice, dab) and six ER species (flounder, pogge, eelpout, fatherlasher, gobies, sea snail) (Elliott and Dewailly, 1995), it is important to note that flounder may be the only species that can be regarded as a true estuarine resident (Wheeler, 1969). The other ER species also possess significant inshore coastal populations, and are essentially marine and estuarine fish, prompting Mathieson *et al.* (2000) to suggest a necessity for subclassification of these species to recognise the difference. The majority of species were to be found in greatest abundance at the most marine site, Port Edgar, suggesting a possible preference for the near perpetual full strength seawater at this location. The extent to which populations from the estuary mix with those from the Firth and beyond may also influence the numbers of fish at the boundary of the truly marine environment. Heterogeneity of Forth Estuary and Firth of Forth populations of eelpout was suggested by differences in mean vertebral counts of the two areas (Poxton, 1987), possibly indicating the estuarine population to have a reasonably restricted geographic range and therefore a true preference for more saline waters. Salinities remain high at Tancred for much of the time (A.S. Hill, Scottish Environment Protection Agency, personal communication), and plaice and dab were most positively associated with this trawl site. Only flounder and sea snail were most common at Longannet, and in the case of the former species this observation was not unsurprising given the preference for waters of reduced salinity (Wheeler, 1969).

Were salinity the only factor governing distribution of fish in the mid-lower estuarine environment of the Forth, one might have expected species with highest abundance at Port Edgar to show progressively less abundance at the Tancred and Longannet sites. This was true of whiting and gobies, while pogge, cod, eelpout and fatherlasher tended to be more abundant at Longannet than Tancred. This may be explained by the increased likelihood of capture of fish at Longannet, due to the relatively narrow nature of the estuary at this station compared with Tancred producing a greater concentration of fish at LW. Port Edgar was also in a narrower section of estuary than Tancred, and may explain the greater abundance of flounder at the former station, when one might intuitively expect the less saline Tancred site to be preferable for this species. Differences in abundance between the sites were relatively small in this species, however. The brief migrations of flounder to the Firth for spawning (Elliott *et al.*, 1990) may complicate this interpretation, however. Whether or not there was a genuine biological explanation for greatest abundances of plaice and dab being at Tancred would require further studies into possible local differences in substrate type or prey availability between the sites.

The importance of tide in determining abundance of fish caught in trawls was evident in most of the ten species modelled. Pogge, flounder, eelpout, fatherlasher, plaice, gobies and sea snail exhibited negative relationships between abundance in trawls and tide height, as originally hypothesised; in the case of the latter three species the relationship was marginally statistically insignificant. The best explanation for this observation seems to be the concentrating effect of LW meaning more fish are present in the estuarine channel, as discussed in section 2.4.3.2. At HW the dilution effect of increased water volume and fish moving onto mudflats to feed would have been expected to produce this result, as observed for all of these species except fatherlasher by Elliott and Taylor (1989). LW at Port Edgar gave the greatest abundances of pogge and fatherlasher, possibly due to the combination of the concentration effect and also reduced salinity influencing a preference for the most marine site (see above). Whiting were the only species for which HW was positively related to abundance in trawls, contrary to expectations since Elliott and Taylor (1989) noted use of intertidal mudflats by this species, and this was most evident at Port Edgar, then Tancred, while at Longannet there was little to choose between abundance at HW or LW. This may be evidence for this MJ species being essentially estuarine-opportunistic (Potter *et al.*, 1997), rather than estuarine-dependent: if whiting used the inshore areas of the Firth of Forth as a nursery as well, then the influx of water into the estuary at HW may facilitate immigration of many more individuals, which then leave again with the ebb of the tide. Given that the whiting population in the Forth from 1981 – 1987 was estimated by Elliott *et al.* (1990) to be approx. 2.65×10^6 individuals with contemporary abundances likely to be less than this (section 3.4.2 above), and the present study suggested that LPS impingement of whiting in 1999 – 2000 was at values approaching this estimate (see section 2.3.3), then a flux of individuals between the estuary and offshore areas seems not unreasonable. The lack of differences between HW and LW catches of whiting at Longannet contrasts with the significantly increased impingement of the species observed at LW at LPS (section 2.4.3.2) and would lend support to the idea that the height of the water in the screen wells at LPS is of greater significance than that of the concentration effect of fish in the estuary.

Interactions between trawl station and month of sampling were the commonest of the significant interactions, occurring in whiting, plaice, pogge, eelpout and gobies. There was very little difference in abundance of whiting between months at the Tancred and Longannet sites, with abundances at the former being higher (see above). Most pronounced differences occurred at Port Edgar, where the greatest abundances, as at the other sites, at all times occurred in December, while minimum abundance was at the same site in June/July. This was consistent with a large influx of whiting during the winter, and low abundance in summer. Little difference in abundance at the two other sites suggested reasonable numbers of whiting remaining in the estuary throughout the year, as is now the case for the Severn Estuary and Bristol Channel, where before there was a more marked seasonality (Henderson and Seaby, 2001). Flounder and eelpout were the major exceptions to the general pattern of high winter and low summer abundance in the estuary. Elliott and Taylor (1983, 1989) showed that flounder were notably more abundant in the subtidal and intertidal areas of the lower estuary during the summer months of 1981 and 1982; this agreed with the interaction of month 3 (June/July) and the Longannet station (*i.e.* the station closest to the large areas of mudflat at Kinneil and Skinflats; Figure 1.1) producing the strongest positive relationship with abundance of flounder captured in trawls. This is explained by the tendency of flounder to migrate to intertidal areas to feed at HW (Wolff *et al.*, 1981; Wirjoatmodjo and Pitcher, 1984), and to virtually cease feeding during the colder months of winter (Wheeler, 1969). Thus flounder were most abundantly captured at LW in the Longannet area during the summer months. Eelpout's presence in the estuary was most pronounced during the spring (month 2) at Port Edgar and summer (month 3) at Longannet, corresponding to the return of adults from deeper waters in the Firth following winter spawning and the recruitment of young-of-the-year juveniles respectively (Elliott *et al.*, 1990). It seems that the rapid growth of the species in its first year (Wheeler, 1969) may be facilitated by consumption of abundant resources on the intertidal flats, since the species are known to use the mudflats (Elliott and Taylor, 1989). Intertidal use may have become less common with warming of the estuary and the general increase in temperature of the estuarine waters in summer, presumably proportionally more than the surrounding sea, may explain the fewer eelpout in the estuary in the 1990s (see section 3.4.2). The seasonality of pogge in exhibiting inshore-offshore migrations was introduced in section 2.4.3.2, and the GLM of the 19-year Agassiz dataset seemed to provide some evidence of this, for abundance at Longannet was greatest in autumn and winter, while being at a minimum in late spring and summer, in complete agreement with the impingement trends at LPS. The inshore migration was for spawning purposes, with adults migrating to deeper waters offshore (Wheeler, 1969). Plaice are not as tolerant of reduced salinities as flounder (Wheeler, 1969), so the tendency for greater abundance at Tancred than Longannet may be largely explained by this factor. High abundance of the species at Longannet, especially of young-of-the-year, in the summer time may again be caused by movements in relation to the intertidal flats, as suggested for flounder and other species. Plaice were the third most common species encountered during intertidal trawling at Kinneil, Skinflats and Torry Bay in 1981 – 1982 by Elliott and Taylor (1989), being present on 41% of occasions. Goby was relatively uniformly distributed among the three stations throughout the period from April to September, but was of relatively low magnitude. In contrast, the Port Edgar and Tancred sites yielded far more fish than at Longannet in winter. This agrees reasonably well with the pattern of impingement at LPS (Chapter 2) and of abundance in the Ythan Estuary (Healey, 1971). The species does not penetrate far up estuaries (Wheeler, 1969), so greater abundance at Port Edgar and Tancred, as noted above, was not surprising. Wheeler (1969) also noted that gobies, meaning specifically sand goby, are annual, and this explained the observed pattern of abundance well: abundances were at a minimum in June/July, at the time before hatching of young of the year had truly commenced; by September, abundances had risen, but were still quite modest, in contrast to the very large increases over winter at the more marine stations, which would have been almost completely of the young of the year, now of a reasonable size to be captured more consistently by Agassiz trawling; in late spring (April), following some overwintering mortality, the population had declined and spawning was commencing. The death of adults following spawning meant that the June/July population was the minimum observed in the estuary, completing the cycle. Offshore migrations at the end of autumn, as noted by Wheeler (1969) may also have enhanced numbers at Port Edgar during this period of the year.

3.4.4. Interannual variation in demersal and benthic species abundances

The data of the present study contributed little in the way of supportive evidence for the hypothesis that between-year variation in fish population size increases with distance from the latitudinal centre of the species' range (Miller *et al.*, 1991). Comparison of the same species between the Forth and the Severn gave three of eight correct predictions, not dissimilar to that of random guessing. The ordinal predictions of species variability within the Forth were not as poor as the inter-estuarine comparison, but still of dubious quality. The observation that ranges may be shifting northwards due to an upward trend in seawater temperatures may have complicated any existing relationships (Phillipart *et al.*, 1998). Of species commonly sampled from intake screens at Hinkley Point B Power Station in the Severn Estuary, sole were theoretically the furthest north from the centre of their geographical range, and would therefore be hypothesised to exhibit most interannual variation in numbers impinged. In fact, the species was the third most stable of 26 species investigated, after conger, *Conger conger*, and flounder (Henderson and Seaby, 1999). Pogge and fatherlasher were the species showing unexpectedly low and high variability in population abundance in the Forth. Subjective determination of the centre of species' geographical ranges from reference sources means that the true centre of the range was unlikely to be estimated completely accurately, and this major potential error may have generated false predictions based on the hypothesis. Differences in sampling technique may have introduced lack of compatibility in the Forth and Severn datasets (Henderson and Seaby, 1999). The case for power stations being an excellent sampling tool for fish has been emphasised on many occasions (*e.g.* Henderson, 1989; Maes *et al.*, 1998a), so the species concerned may not have been caught with equal efficiency. Comparisons of commercially exploited species with unfished species may be complicated by fishing mortality altering natural patterns of variation. As noted by Henderson and Seaby (1999), the attractiveness of the latitude-population variability hypothesis of Miller *et al.* (1991) was its simplicity, but lack of evidence is mounting against it. Henderson and Seaby (1999) offer an alternative hypothesis, that population variability shows no trend with either latitude or longitude and that variability within the permanent range is determined by local factors such as disease, giving a chaotic pattern of recruitment variability. The sea snail in the Severn, at the southern edge of the species' range, are annual and yet have greater stability of numbers than those nearer the centre of the range in the Forth, which live for several years (Henderson and Seaby, 1999). This stability of the annual Severn population is remarkable, and contrasts with the huge variations in abundance of other annual species such as sand goby (Rogers *et al.*, 1998). To avoid extinction, the southern sea snail of the Severn possesses great stability in year-to-year recruitment. This in itself is compelling evidence against the hypothesis of Miller *et al.* (1991).

3.4.5. Preliminary assessment of the pelagic fish populations

Sprat and herring constituted the great majority of fish taken during pelagic trawling between April 1999 and January 2001, whilst whiting were the next most abundant species and most of the remaining species were benthic or demersal species caught incidentally when the gear dipped towards the substrate. The difficulties of sampling with a net of opening 2.5 – 7m high in water depths as shallow as 6m are apparent, so it was no surprise that occasionally non-pelagic species were taken. Herring constituted 29.3% of total clupeid catch, which is somewhat less than the approx. 44% taken during impingement sampling at LPS (Chapter 2), but very similar to the 26.4% derived in the initial pelagic study of January – April 1984 (FRPB, 1984).

Estimated sprat and herring population sizes were greater than the long term averages provided by Elliott *et al.* (1990). The arithmetic means of abundance were an order of magnitude larger, while the geometric means were approximately double (Table 3.11). The estimates all suggest sprat populations to be between two and three times more abundant than herring at this location. Maximum estimated abundances, which occurred in the period from January – April, were very similar to those noted by Elliott *et al.* (1990) (January – March).

Table 3.11. Estimated sizes of clupeid populations in the lower Forth Estuary, based on data from the present study and Elliott *et al.* (1990).

	This study (1999 – 2000)		**Elliott *et al.* (1990) (1981 – 1989)**
	arithmetic mean	geometric mean	
sprat	1.37×10^7	2.02×10^6	1.18×10^6
herring	5.30×10^6	9.94×10^5	5.9×10^5

These data suggest the pelagic gear offers more accurate estimates of clupeid population size than Agassiz trawling, as expected. While sprat and herring abundance in the North Sea was rather low in the early 1980s, there were much greater values later in the same decade (ICES 2000), so that the estimates of Elliott *et al.* (1990) were calculated at times of both relative abundance and scarcity of the two species compared with the present day. This may be taken as evidence that the increased population estimates derived for 1999 - 2000 are not merely a result of increased North Sea abundances of the two species. It is apparent that the method of estimation of abundances was rather crude. Gear efficiency was not known, so a value similar to that of the Agassiz trawl was adopted, where a lower value would have resulted in proportionally greater estimates of abundance. The shoaling nature of clupeids may have suggested greater abundance on some occasions while underestimating numbers on others. The need for greater frequency of trawling to reduce this type of error is obvious, but often was not possible because of limited sampling resources. As was clear in monthly estimates of abundance (Figures 3.9 and 3.10), confidence limits were generally exceptionally wide, symptomatic of the shoaling behaviour of the species. The arbitrary selection of the region between Dunmore and the Forth Bridges as the volume of water that contained the clupeids was necessary to aid calculations of absolute abundance, but the likely movement of species into and out of this area due to tidal flows and active migrations introduces a degree of inaccuracy into calculations of this type. More accurate measurements of abundance may be possible in the future by utilisation of acoustic techniques, though the development of adequate methods for generally shallow estuaries is a relatively new science, facing different challenges to the more established sonar used in open sea assessments (Trevorrow *et al.*, 2000). Since movement of clupeids in the estuary tends to be directed by tidal currents (Welsby *et al.*, 1964), use of fixed nets of known opening area combined with flow meters may allow abundances of fish to be more effectively estimated. Use of stow nets by local fishermen utilises high spring tidal currents to catch fish in the lower estuary (personal observations), and a comparison with results from trawl studies would be of interest.

Generalised linear modelling of relatively few pelagic data obtained between April 1999 and January 2001 must be treated with caution, in particular in light of the relatively large dataset available for Agassiz trawling (section 3.4.3). Despite this, some preliminary features were of note. Both sprat and herring abundance was significantly enhanced at LW compared to HW, possibly due to increased concentration of fish available to trawling in the subtidal region at LW. Elliott and Taylor (1989) noted that herring and sprat used the mudflats at Kinneil, Skinflats and Torry Bay relatively frequently at HW, so this would tend to increase numbers present in the subtidal at LW. The significant interaction of tide × station in herring showed that this effect was particularly pronounced at the Longannet station, which is immediately adjacent to the mudflats and situated in relatively shallow water in a relatively narrow channel. Observations regarding effects of the month factor on abundances probably require many more additional years of data to be reliable.

3.4.6. An overview of data from the upper Forth Estuary

Data collected during HW beam trawling at six sites in the upper estuary between Kincardine and Stirling (see Elliott and Taylor, 1989, for map) in the summer months were not considered in detail in the present study, due to lack of sampling in 1986-87 and 1994-95 as well as wide variation in number of months sampled, but are of note as a comparison to data from the lower estuary discussed in detail above. Abundances of fish captured per trawl were similar to those in the mid-lower estuary, despite hauls being of only 0.6 km length (*cf* 0.8 km in lower estuary, see section 3.2.1.1). In general an increase in total number of fish captured was observed between 1983 and 1993, with a levelling-off from 1996 – 1999 (Table 3.12). This trend was largely attributable to increases in abundance of smelt, whiting, flounder and gobies. The abundance of gobies continued to rise until 1996, then stabilised at around 15 individuals per trawl from 1997 – 1999. Flounder declined in abundance at the end of the period to levels similar to those recorded early in the time series, while whiting declined to the lowest recorded abundances from 1996 – 1999.

Table 3.12. Abundance of fish sampled in upper Forth Estuary, 1983 - 1999. Data represent mean number of fish collected per trawl per year. Data from A.S. Hill (Scottish Environment Protection Agency, personal communication).

	total	smelt	sprat	herring	flounder	goby	whiting	plaice	others
1983	5.7	0.0	0.0	0.9	2.3	0.0	2.4	0.0	0.0
1984	2.6	0.0	0.1	0.1	2.0	0.3	0.1	0.3	0.0
1985	10.3	0.0	2.7	1.3	5.0	0.0	1.2	0.2	0.0
1986					*No sampling undertaken*				
1987					*No sampling undertaken*				
1988	17.0	0.0	4.3	0.5	4.1	5.9	1.7	0.0	0.4
1989	15.2	0.1	0.8	2.7	9.1	1.8	0.5	0.0	0.1
1990	43.7	0.2	8.3	5.9	1.9	6.8	20.0	0.1	0.4
1991	37.1	9.5	2.5	0.1	13.9	3.1	0.8	5.3	1.8
1992	31.1	6.4	1.2	0.1	18.2	0.7	3.9	0.0	0.5
1993	58.3	16.1	6.0	2.3	17.4	2.6	8.8	4.9	0.3
1994					*No sampling undertaken*				
1995					*No sampling undertaken*				
1996	48.7	2.3	0.2	0.1	15.8	25.2	1.8	0.7	2.7
1997	25.1	5.1	2.0	0.1	3.6	13.9	0.0	0.1	0.3
1998	24.2	4.5	0.5	0.0	1.5	17.0	0.0	0.0	0.7
1999	30.1	7.7	1.8	1.4	4.5	13.8	0.2	0.6	0.2

The re-establishment of the pollution-sensitive smelt in the Forth, following disappearance in the late 1960s due to overfishing, siltation and low dissolved oxygen levels (Blaber *et al.*, 2000), was "eagerly anticipated" by Poxton (1987). The upper estuarine data, together with high levels of impingement at LPS (Maitland 1997, 1998; Chapter 2), indicate that the species has indeed made a good recovery from the late 1980s to the present day, and this was suggested to be due to improved water quality in the upper estuary by Griffiths (1997). The reduction in emissions of effluents into the upper estuary is likely to be the cause of increased fish abundances in this area, but the overall rises in fish numbers are of relatively smaller magnitude than the decreases in abundance of fish in the mid-lower estuary, since the latter region comprises over 90% of the volume of the Forth Estuary (FRPB, 1978).

3.5. Conclusions

Total abundances of demersal and benthic fish in the mid-lower Forth Estuary significantly declined between 1982 and 2000, largely due to significant decreases in abundance of whiting and eelpout. While the abundance of fish in the upper estuary increased over the same period, due to improvements in water quality, these two species did not increase in abundance in the upper estuary, suggesting that they were reduced in numbers from the estuary as a whole. Large differences in water volume of the upper and lower estuaries suggest reductions in fish abundance in the latter area to be of greater significance than increases in the former. For eelpout the change in total abundance may be related to increasing water temperatures affecting recruitment. Trends in abundance of juvenile whiting, cod and plaice in the Forth did not reflect trends for the same species in the North Sea and adjacent areas. Populations of all other species in the lower estuary showed no significant positive or negative trends in abundance. Decline in abundance of Forth fish populations may be due to decreased recruitment or else emigration from the estuary in response to unfavourable conditions such as increasing water temperatures. Measurement of water quality parameters such as temperature, dissolved oxygen, salinity and turbidity were not undertaken, so the influence of these factors on the ichthyofauna could not be modelled. The addition of these data to regular trawling undertaken by SEPA would be of use in elucidating further the distribution patterns of fish in the estuary. The species in the lower Forth Estuary are primarily of marine origin and reflected this by tending to be found in greatest abundance at the most seaward site of Port Edgar. Abundance at Longannet was often also higher than at the Tancred station, because of the proximity of extensive mudflats used for feeding purposes by many species. This was also reflected in increased catches at LW, when most fish were concentrated into a smaller area of estuarine channel. The exception was whiting, and it may be that HW brings many individuals of this species from the Firth of Forth into the lower estuary. The pelagic fish populations in 1999 – 2000 were dominated by sprat and herring, with the former more abundant than the latter, and preliminary estimates of abundance were an order of magnitude greater than those obtained by extrapolation of bottom trawl data. Continuation of the pelagic trawling programme may allow trends in abundance within the lower estuary to be modelled more accurately over the coming years.

Chapter 4. Environmental impact of Longannet Power Station on the Forth Estuary ichthyofauna: impingement of commercial, recreational and threatened fish species.

4.1. Introduction

Assessment of environmental impact on fish populations due to losses caused by impingement or entrainment during CW abstraction is a 30-year old subject that to this day causes much debate amongst those involved in power station research. The United States Clean Water Act § 316(b), for example, states that "the location, design, construction, and capacity of cooling water intake structures reflect the best technology available for minimizing adverse environmental impact", but without specific definitions of terms such as "adverse" and "impact", the legislation is open to a variety of interpretations. The US Environmental Protection Agency (US EPA) is in the process of re-addressing this issue and proposing federal laws on CW abstraction (US Federal Register, 2000), though with not inconsiderable difficulties (see EPRI, 2000; Henderson and Seaby, 2000). Several different definitions of adverse environmental impact are being considered (US Federal Register, 2000), including:

1. Adverse aquatic environmental impacts occur whenever there will be entrainment or impingement damage as a result of the operation of a specific cooling water intake structure. The critical question is the magnitude of any adverse impact. The exact point at which adverse aquatic impact occurs at any given plant site or water body segment is highly speculative and can only be estimated on a case-by-case basis by

considering the species involved, magnitude of the losses, years of intake operation remaining, ability to reduce losses, etc. (US EPA, 1977).

2. Impingement or entrainment of 1% or more of the aquatic organisms in the near-field area [defined as the area around the intake structure from which organisms are drawn onto the screens or into the cooling system] as determined in a one-year study.

3. Any impingement or entrainment of aquatic organisms.

An alternative definition was proposed by UWAG (2000): "Adverse environmental impact is a reduction in one or more representative indicator species that (1) creates an unacceptable risk to the population's ability to sustain itself, to support reasonably anticipated commercial or recreational harvests, or to perform its normal ecological function and (2) is attributable to the operation of the cooling water intake structure".

British legislation covering operation of power stations is represented by the 1989 Electricity Act. Schedule 9 § 3(3) requires electricity generators to "avoid, so far as possible, causing injury to fisheries or to the stock of fish in any waters". Thus a similar problem regarding definitions of terms such as "injury", as was noted above for US legislation, clearly exists.

Individual power stations may impinge numerous organisms from a variety of species. The US EPA (1977) suggested that "critical aquatic organisms" be selected for study of environmental impacts of power stations, such organisms tending to be removed in CW flows and being one or more of the following:

1. representative, in terms of their biological requirements, of a balanced, indigenous community of fish, shellfish, and wildlife;
2. commercially or recreationally valuable (*e.g.*, among the top ten species landed – by dollar value);
3. threatened or endangered;
4. critical to the structure and function of the ecological system (*e.g.*, habitat formers)
5. potentially capable of becoming localised nuisance species;
6. necessary, in the food chain, for the well-being of species determined in 1 – 4;
7. one of 1 – 6 and have high potential susceptibility to entrapment-impingement and/or entrainment.

This was very similar to the process of identifying appropriate 'important' marine species in order to conduct appropriate environmental impact assessments (EIA) by the UK power industry. Species were classified as 'important' if rare, locally representative, commercially significant, of socio-political concern, or else of importance as food resources for other creatures (Bamber, 1990).

The American approach to EIA of CW impingement and entrainment mortalities was to formulate mathematical models of fish population dynamics and investigate whether compensatory mechanisms would offset losses due to CW abstraction (*e.g.* papers in Barnthouse *et al.*, 1988). These density-dependent models were deemed inappropriate to study environmental impacts of British power stations by the Central Electricity Research Laboratories (CERL) (P.A. Henderson, Pisces Conservation Ltd., personal communication), and so a procedure was developed to estimate the quantity of mature adult fish that would have accrued to commercial fisheries had they not experienced mortality as juveniles due to impingement or entrainment. This technique, known as the equivalent adult value (EAV) method (Turnpenny, 1989), suggested tonnages of equivalent adult fish lost at Heysham 1, Hinkley B, Fawley and Kingsnorth power stations to be trivial compared to catches from adjacent commercial fishing grounds (Turnpenny, 1988b). Thus although catches of fish seemed numerically large, the preponderance of juveniles meant that the large quantities involved only represented a relatively small total number of fish that would have survived to adulthood. The EAV method applied to commercial species impinged at LPS was undertaken by Turnpenny (1997) using the PISCES (v.3) software prediction system (see section 2.4.2.2), and estimated that the abundance of juvenile herring, whiting, cod, dab, plaice and sole impinged annually at LPS would represent approximately 9.7 t of fish that would have existed as adults.

The present study aimed to use the EAV technique employing actual field data collected during the impingement study from January 1999 – December 2000 (Chapter 2), to assess the potential environmental impact of loss of equivalent adult whiting, cod, plaice, and herring through loss of juveniles by LPS CW abstraction. In addition, the direct loss of these species above minimum commercial landing sizes, as well as the losses of sprat, a commercial species with no minimum size limit to catch, was estimated. The extent of impingement of Atlantic salmon and sea trout smolts, as well as eel, was investigated due to their recreational and socio-political importance. In addition, the impingement of the potentially threatened river lamprey, a species listed by the International Union for Conservation of Nature and Natural Resources, was also determined.

4.2. Materials and methods

4.2.1. Impingement of fish species above minimum commercial landing sizes

Impinged whiting, cod, plaice, herring and sprat were collected and analysed from January 1999 – December 2000, as detailed in section 2.2.2. Abundances and wet mass of the first four of these species above minimum commercial landing size (MCLS) were estimated, according to the technique described in section 2.2.4. The same procedure was undertaken for all sprat estimated to have been impinged during the same time period, as no MCLS exists for this species. Abundances of each species were then converted into approximate monetary values by use of available data for value and total live weights of fish landed into Scottish ports from ICES area IVb by UK vessels in 1999.

4.2.2. Potential future losses of commercial species through impingement of juveniles

4.2.2.1. Basis of the Equivalent Adult Value (EAV) Method

The EAV method (Turnpenny, 1989) was used to estimate the approximate number of adult whiting, cod, plaice and herring that would have existed had juveniles not experienced mortality on the LPS intake screens. The EAV of a juvenile fish is defined as the average lifetime fecundity of an adult that has just reached maturity which is required to replace that juvenile (Turnpenny, 1988b; 1989). The EAV method assumes that the population is in equilibrium; therefore births balance deaths and the population growth rate, λ, equals one. Each EAV value is calculated as:

$$EAV = \frac{1}{S_t F_a}$$

where S_t = fractional probability of survival from spawning to time t, and F_a is the average lifetime fecundity of an adult. F_a is defined as:

$$F_a = \sum_{j=a}^{m} P_j S_j E_j R_j$$

where a is the age at which > 50 % of fish mature; m is the number of age classes in the population, P_j is the fraction of females that are mature in age class j, S_j is the survival probability from the age at which > 50% mature to age class j, E_j is the average fecundity of mature females of age class j, and R_j is the proportion of females in age class j.

Therefore as an immature fish grows older, the probability of its survival to maturity increases, and this is reflected in an increase in the EAV factor used to calculate the number of just-mature (*i.e.* the age when > 50% of the year class is mature) adults that would have resulted from that individual. Since the EAV method was intended to calculate the number of just-mature adults, incorporation of fish that are beyond this stage of the life cycle yields EAV factors > 1, *i.e.* impingement of one such fish represents more than one just-mature individual. This phenomenon occurs rarely in power station studies since most impinged fish are juveniles and therefore possess EAV factors < 1.

4.2.2.2. Potential losses of equivalent adult whiting, cod and plaice

Potential losses of equivalent adult whiting, cod and plaice through impingement of juveniles were assessed using the EAV method as originally presented by Turnpenny (1989). Size class measurements for each species were summed by quarter (January – March, April – June etc.) to conform to the procedure of Turnpenny (1989), which had originally used data collected monthly but analysed quarterly. Each fish was assumed to have been caught in the middle of the quarter, so that ages relative to the assumed date of capture were calculated based on known periods of peak spawning. Proportions of different age groups of each species were estimated by examination of length-frequency histograms and using lengths-at-age suggested by Wheeler (1969). These proportions were then multiplied by the estimated total number of fish impinged during the appropriate quarter using previously calculated abundance data (section 2.4.2.1), to give estimated abundances of each age group impinged at LPS. The abundances of each age group were multiplied by the appropriate EAV factor (Table 4.1) to give abundances of adults of age when > 50% maturity occurs. Some error was inevitably introduced by use of EAV factors over a decade old, since parameters such as survival probabilities are liable to change over time, but the values employed were expected to give results of the correct order of magnitude. The abundances of equivalent adults thus obtained were multiplied by mass-at-age values obtained from the literature to give estimates of total mass of equivalent adults potentially lost through impingement of juveniles. Total adult equivalent masses were converted into monetary terms using available data for value and total live weights of fish landed into Scottish ports from ICES area IVb by UK vessels in 1999.

Table 4.1. EAV factors employed in calculations of potential equivalent adult losses of whiting, cod, and plaice at LPS, 1999 – 2000. 0, I, II refer to age group of fish, *i.e.* those born in same year as impingement, those born in previous year, and those born two years earlier, respectively. Data from Turnpenny (1989).

	whiting			cod			plaice		
	0	**I**	**II**	**0**	**I**	**II**	**0**	**I**	**II**
January - March	-	0.506	0.932	1.05×10^{-5}	7.4×10^{-2}	0.149	7.62×10^{-6}	0.122	0.23
April - June	2.48×10^{-3}	0.595	1.10	3.98×10^{-2}	8.4×10^{-2}	0.167	1.37×10^{-3}	0.145	0.258
July – September	0.35	0.70	1.27	4.7×10^{-2}	0.10	0.187	1.62×10^{-2}	0.163	0.289
October – December	0.42	0.82	1.68	6.0×10^{-2}	0.124	0.215	6.86×10^{-2}	0.193	0.306

4.2.2.3. Potential losses of equivalent adult herring

Assessment of the potential loss of equivalent adult herring at LPS, through loss of juveniles, employed the EAV method of Turnpenny (1989), as detailed above, but with certain alterations. The majority of herring entering the Forth Estuary are likely to originate from spawning grounds off Buchan Ness, and therefore were born in August – September (Harden-Jones, 1968). Employing the EAV factors given by Turnpenny (1989) would have assumed that the herring had been spawned in November, since the original factors were calculated for herring stocks originating from different spawning grounds. This would have introduced potentially important errors into the calculations. Instead, a birthday of 1 September was assumed for most herring collected in LPS impingement samples. EAV factors employed were estimated from values given for both spring and autumn-spawning herring by Turnpenny (1989), based on the assumption that the fish were collected in the middle of each month (Table 4.2).

Table 4.2. EAV factors employed in calculations of potential loss of equivalent adults due to impingement of juveniles at LPS, 1999 – 2000. † indicates months for which the EAV factors apply in the case of herring assessed to have originated from Buchan Bank (born on 1 September), while ‡ indicates months for spring-spawned fish (born on 1 March). Approx. ages in days are shown, assuming that fish were sampled in the middle of each month. – indicates months for which EAV factors were not estimated due to fish of this age having not been impinged.

	First year of life		Second year of life		Third + year of life	
Month of impingement	**Age**	**EAV factor**	**Age**	**EAV factor**	**Age**	**EAV factor**
September†/March‡	15	-	380	0.757	745	0.998
October†/April‡	45	-	410	0.775	775	1.07
November†/May‡	75	-	440	0.7885	805	1.21
December†/June‡	105	-	470	0.802	835	1.21
January†/July‡	135	-	500	0.802	865	1.35
February†/August‡	165	0.6875	530	0.8345	895	1.35
March†/September‡	195	0.682	560	0.869	925	1.52
April†/October‡	225	0.6985	590	0.869	955	1.52
May†/November‡	255	0.715	620	0.869	985	1.69
June†/December‡	285	0.715	650	0.869	1015	1.69
July†/January‡	315	0.715	680	0.9335	1045	1.69
August†/February‡	345	0.736	710	0.9335	1075	1.69

The possibility that herring other than those from the Buchan stock also enter the Forth Estuary was suggested by the occasional skew observed in length-frequency histograms of impinged herring from LPS (Figure 4.1). A possible explanation was that impingement of spring-spawned herring caused the unusual additional modes on histograms. A commercial fishery on herring spawning in the Firth of Forth from January – April existed until a stock collapse in 1946 (Thomas and Saville, 1972). Catches of adult spring-spawning herring in the Firth of Forth averaged approx. 5000 t from 1896 – 1927, then increased to a peak of over 18000 t in 1938, followed by a decline and the aforementioned collapse immediately following World War II (Thomas and Saville, 1972). The same authors suggested that since fishing effort was markedly reduced due to war, the decline in the stock was more probably due to emigration of herring from the area or a recruitment failure due to some change in the environment rather than overfishing. It may be that the spring-spawning Firth stock remains in greatly reduced numbers, or else that fish spawned in spring from other areas find their way into the Firth, and that they are occasionally impinged at LPS. As this theory was postulated retrospectively, no morphometric confirmatory analyses of herring assumed to be spring spawners were undertaken. It was therefore assumed that these herring were born on 1 March, and EAV factors based on the estimated age of the fish in each month were employed (Table 4.2).

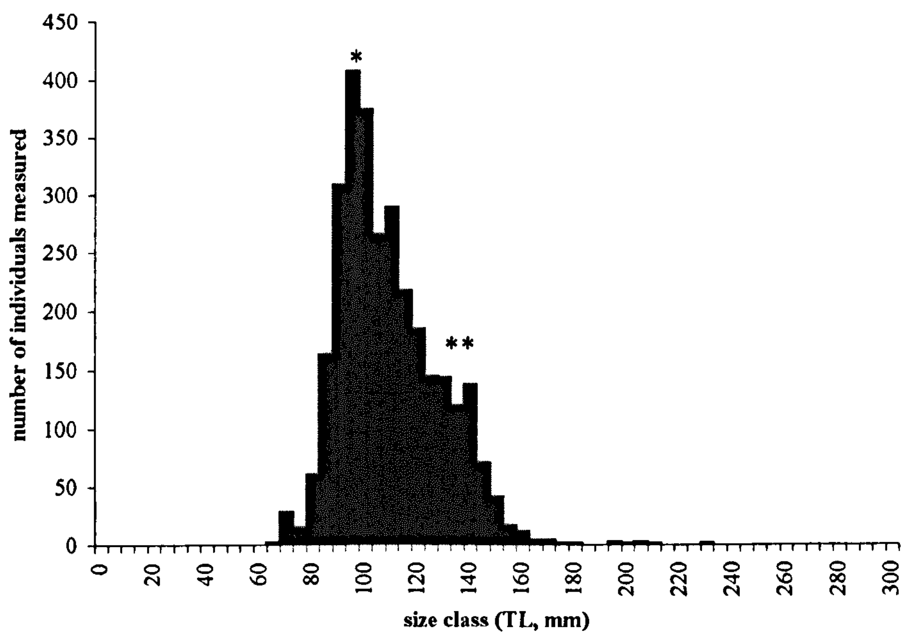


Figure 4.1. Example of possible mixing of Buchan-spawned and spring-spawned herring in impingement samples from LPS in January 2000. Mode of Buchan herring spawned in August – September 1998 indicated by *, that of herring potentially spawned in February – March 1998 by **.

4.2.3. Extent of impingement of salmonid smolts, spring/summer 2000

Identification of salmonid smolts (Atlantic salmon and sea trout) in 1999 was uncertain due to lack of adequate identification keys. The extent of LPS impingement of salmonid smolts was assessed during the 2000 downstream migration season, for which suitable identification sources became available (K. Lockhart, University of Stirling, personal communication). Based on first and last occurrences of smolts in impingement samples, salmon were assumed to have been migrating past the LPS CW intake from 16 April – 31 May 2000, while sea trout were assumed to have undertaken downstream migrations between 16 April and 16 June. Number of fish impinged per unit volume in each of the sampling sessions during the assumed duration of the downstream migration was followed by determination of the geometric mean number of fish per unit volume over the same period. The mean values for each species were then multiplied by the volumes of CW abstracted during the two periods of downstream migration, these volumes having been estimated according to the method in section 2.2.4, to give the estimated total abundances of smolts impinged at LPS in 2000.

4.2.4. Extent of impingement of river lamprey and eel

River lamprey and eels were collected and measured from the intake screens at LPS from January 1999 – December 2000 using techniques described in section 2.2.2. Abundances of the species were estimated as detailed in section 2.2.4.

4.3. Results

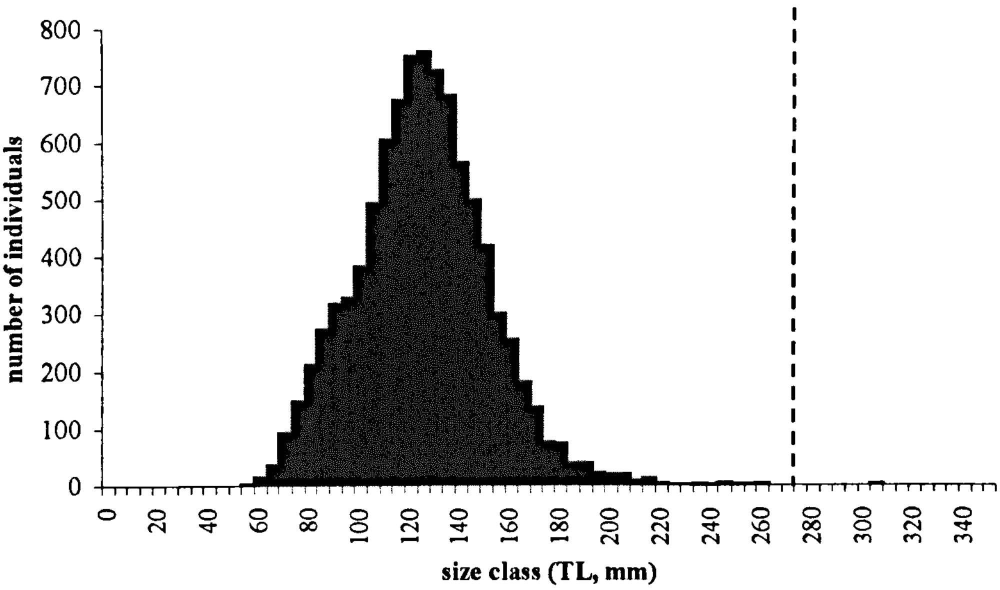
4.3.1. Impingement of commercial species ≥ MCLS

Very few individuals of the larger-bodied commercially important species (whiting, cod, plaice, herring) collected in impingement samples from LPS were ≥ minimum commercial landing size (MCLS) (Figure 4.2a-d). Most of these fish were 0+ or 1+ juveniles. Sprat do not have an MCLS (J.N. McCallum, Scottish Fisheries Protection Agency, personal communication), and the catch consisted mostly of 0+ and 1+ fish (*i.e.* those approx ≤ 130mm; Wheeler, 1969), though with a very small proportion of individuals approaching the maximum TL (160mm; Miller and Loates, 1997) (Figure 4.2e).

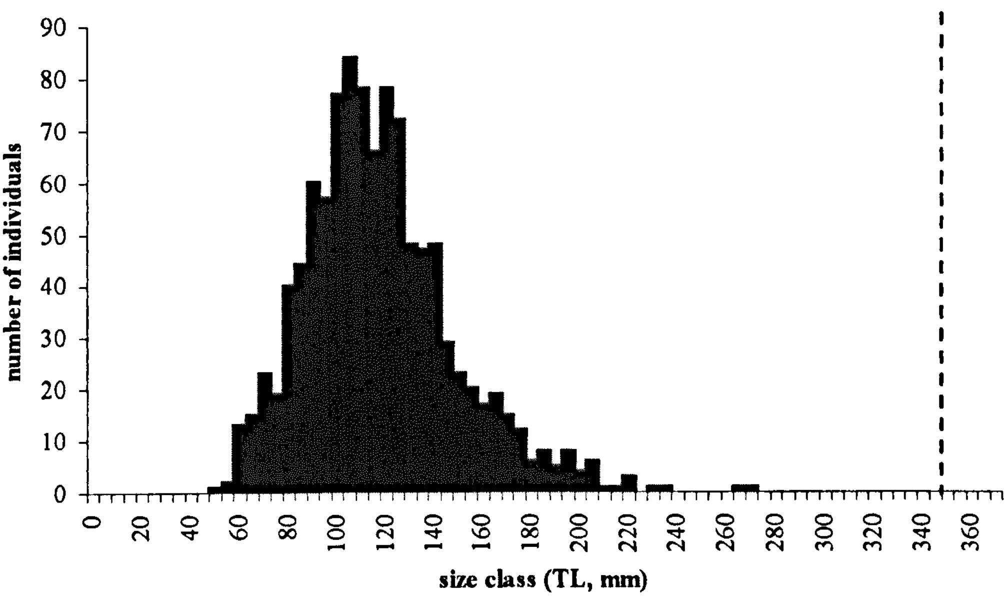
No cod or plaice ≥ MCLS were taken during the course of the study, and the single whiting of 302 mm taken in April 2000 extrapolated to approximately 400 individuals of this size over the whole study period, with a very low monetary value (Table 4.3). Herring ≥ 200mm TL were rare in relation to other size classes of this species (Figure 4.2d), but were the most commonly impinged of the species with commercial landing size limitations. The estimated 2.15×10^4 herring impinged ≥ MCLS would have been worth just over £360 at 1999 values (Table 4.3). Due to the fact that sprat of all sizes are marketable in one form or another, and there is a lack of MCLS for this species, it was unsurprising that sprat contributed the dominant portion of commercial species directly lost to impingement at LPS in 1999 – 2000. The estimated 1.85×10^7 individuals of this species of nearly 57 t wet mass were worth marginally below £3000 (Table 4.3).

Figure 4.2. Length frequency distributions of commercial species measured from impingement at LPS, January 1999 – December 2000. Dashed lines (- - -) indicate minimum commercial landing sizes (MCLS) for whiting, cod, plaice and herring (no MCLS exists for sprat).

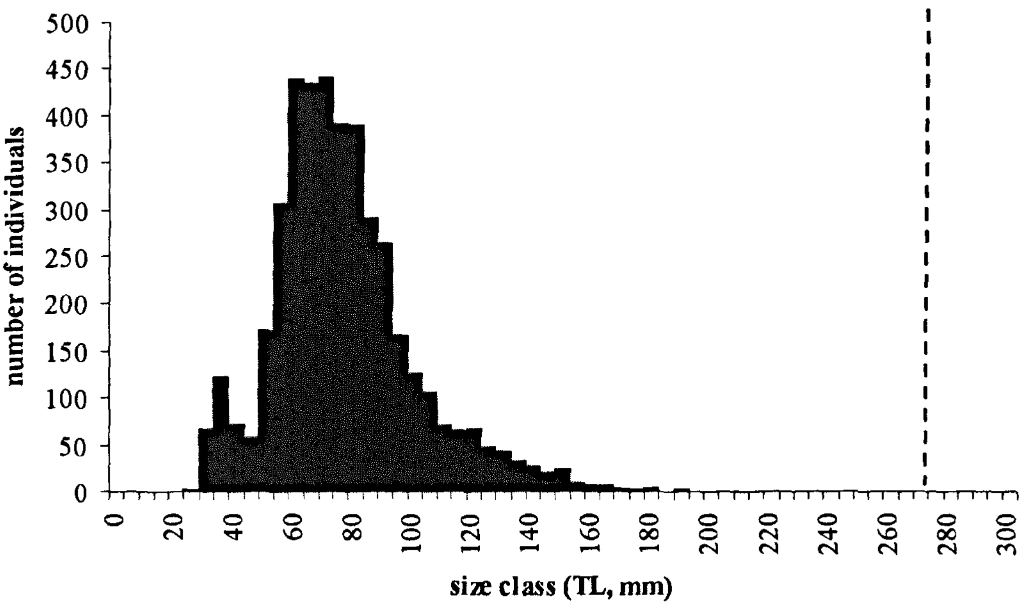
(a) whiting



(b) cod



(c) plaice



(d) herring

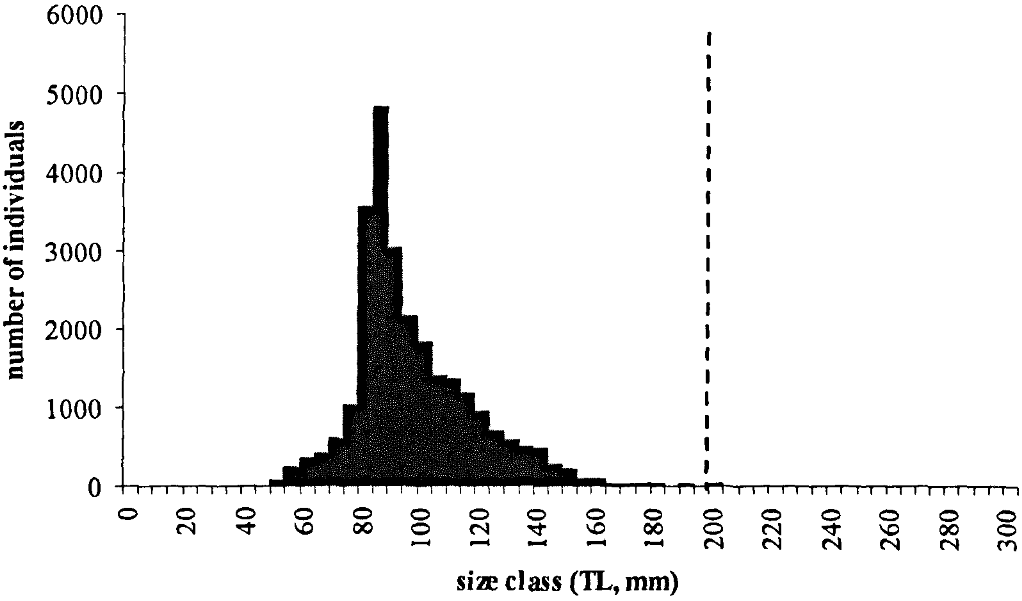


Figure 4.2 cont.

(e) sprat

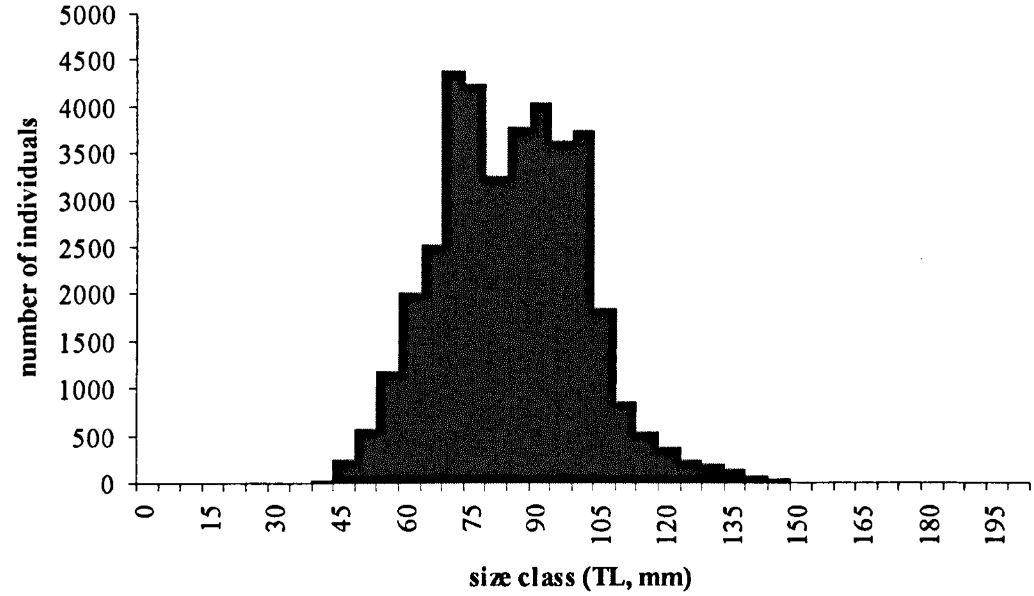


Figure 4.2 cont.

Table 4.3. Estimated abundance, wet mass, and value of commercially important species ≥ minimum commercial landing size (MCLS) impinged at LPS, January 1999 – December 2000. Ranges of 95% confidence intervals indicated in parentheses. MCLS values obtained from ICES (2000, 2001a). Monetary values based on data of fish landed into Scottish ports from ICES area IVb by UK vessels in 1999 (Scottish Executive Publications, 2000). No MCLS exists for sprat.

species	MCLS	Estimated abundance of impinged fish ≥ MCLS	Estimated wet mass of impinged fish ≥ MCLS (kg)	Estimated value of impinged fish ≥ MCLS
whiting	27cm	**400** (254 – 629)	**58.8** (37.3 – 92.5)	**£33** (£21 – £52)
cod	35cm	0	0	0
plaice	27cm	0	0	0
herring	20cm	**2.15×10^4** ($1.53 - 3.11 \times 10^4$)	**2.62×10^3** ($1.86 - 3.79 \times 10^3$)	**£361** (£274 – £523)
sprat	-	**1.85×10^7** ($1.20 - 3.35 \times 10^7$)	**5.67×10^4** ($3.22 \times 10^4 - 1.26 \times 10^5$)	**£2894** (£1643 – £6450)

4.3.2. Potential loss of adults through impingement of juveniles of commercial species

All four commercially important species exhibited substantial increases from 1999 to 2000 in estimates of abundance and mass of equivalent adults that were lost through impingement of juvenile fish (Tables 4.4 – 4.8). This was a reflection of the increased impingement of fish in the latter year of the study, as discussed in Chapter 2. The period from October – December in both years gave maximum estimated losses of equivalent adults in all species, with the exception of whiting in 1999 when the phenomenon was observed during the January – March interval (Table 4.4a). In contrast, the first quarter of 2000 gave lowest estimated abundance of equivalent adult whiting lost to impingement at LPS (Table 4.4b), with this period exhibiting a similar characteristic for cod and plaice in 1999 (Tables 4.5a and 4.6a, respectively). Loss of equivalent adults tended to be at minimal levels during the second quarter of the year, April – June; this was true for plaice and herring in both years (Tables 4.6 – 4.8,), and for whiting in 1999 (Table 4.4a) and cod in 2000 (Table 4.5b). In all four species the bulk of potential equivalent adult losses were provided by impingement of juvenile fish less than a year old, *i.e.* those entering inshore waters for the first time following birth. The herring assessed to have originated from Buchan Ness spawning grounds outnumbered those believed to have come from spring-spawning stock by an order of magnitude in 1999, both in terms of actual abundance impinged and potential losses of equivalent adults (Tables 4.7a and 4.8a); in 2000, the Buchan herring were an additional order of magnitude more abundant than their spring-spawned cousins (Tables 4.7b and 4.8b).

Table 4.4. Estimated impingement of 0 – 2-group whiting at LPS with estimated abundance and biomass of equivalent adults that would have existed had the impinged fish not experienced mortality. EAV (equivalent adult value) factors obtained from Turnpenny (1989). Mass values calculated from mean biomass of age 2 whiting of 0.174kg (ICES, 2001a). Values in parentheses are ranges of 95% confidence intervals.

(a) 1999		Estimated impinged abundance	Abundance of equivalent adults	Mass of equivalent adults (kg)
January – March	0-group	0	0	0
	1-group	**4.78×10^5** ($2.49 - 9.50 \times 10^5$)	**2.42×10^5** ($1.26 - 4.81 \times 10^5$)	**4.21×10^4** ($2.19 - 8.36 \times 10^4$)
	2-group	**2.67×10^4** ($1.39 - 5.31 \times 10^4$)	**2.49×10^4** ($1.30 - 4.95 \times 10^4$)	**4.33×10^3** ($2.25 - 8.61 \times 10^3$)
April – June	0-group	0	0	0
	1-group	**9.83×10^4** ($7.40 \times 10^4 - 1.35 \times 10^5$)	**5.85×10^4** ($4.41 - 8.02 \times 10^4$)	**1.02×10^4** ($7.66 \times 10^3 - 1.39 \times 10^4$)
	2-group	**1.48×10^4** ($1.12 - 2.03 \times 10^4$)	**1.63×10^4** ($1.23 - 2.24 \times 10^4$)	**2.84×10^3** ($2.14 - 3.89 \times 10^3$)
July – September	0-group	**3.50×10^5** ($2.57 - 4.92 \times 10^5$)	**1.22×10^5** ($8.98 \times 10^4 - 1.72 \times 10^5$)	**2.13×10^4** ($1.56 - 3.00 \times 10^4$)
	1-group	**1.56×10^4** ($1.15 - 2.20 \times 10^4$)	**1.09×10^4** ($8.02 \times 10^3 - 1.54 \times 10^4$)	**1.90×10^3** ($1.40 - 2.68 \times 10^3$)
	2-group	**2.28×10^3** ($1.67 - 3.21 \times 10^3$)	**2.89×10^3** ($2.12 - 4.07 \times 10^3$)	**5.03×10^2** ($3.69 - 7.09 \times 10^2$)
October – December	0-group	**2.24×10^5** ($1.73 - 3.41 \times 10^5$)	**9.43×10^4** ($7.27 \times 10^4 - 1.43 \times 10^5$)	**1.64×10^4** ($1.27 - 2.49 \times 10^4$)
	1-group	**1.47×10^3** ($1.13 - 2.23 \times 10^3$)	**1.20×10^3** ($9.28 \times 10^2 - 1.83 \times 10^3$)	**2.09×10^2** ($1.62 - 3.18 \times 10^2$)
	2-group	**2.67×10^2** ($2.06 - 4.05 \times 10^2$)	**4.48×10^2** ($3.46 - 6.81 \times 10^2$)	**78** ($60 - 1.19 \times 10^2$)
total		**1.21×10^6** ($7.92 \times 10^5 - 2.02 \times 10^6$)	**5.73×10^5** ($3.69 - 9.70 \times 10^5$)	**9.98×10^4** ($6.42 \times 10^4 - 1.69 \times 10^5$)

(b) 2000		Estimated impinged abundance	Abundance of equivalent adults	Mass of equivalent adults (kg)
January – March	0-group	0	0	0
	1-group	**7.65×10^4** ($4.85 \times 10^4 - 1.33 \times 10^5$)	**3.87×10^4** ($2.45 - 6.72 \times 10^4$)	**6.73×10^3** ($4.27 \times 10^3 - 1.17 \times 10^4$)
	2-group	**1.30×10^4** ($8.21 \times 10^3 - 2.25 \times 10^4$)	**1.21×10^4** ($7.66 \times 10^3 - 2.10 \times 10^4$)	**2.10×10^3** ($1.33 - 3.65 \times 10^3$)
April – June	0-group	0	0	0
	1-group	**4.75×10^4** ($2.67 - 9.47 \times 10^4$)	**2.87×10^4** ($1.59 - 5.63 \times 10^4$)	**4.92×10^3** ($2.77 - 9.80 \times 10^3$)
	2-group	**3.79×10^4** ($2.13 - 7.54 \times 10^4$)	**4.16×10^4** ($2.34 - 8.30 \times 10^4$)	**7.25×10^3** ($4.08 \times 10^3 - 1.44 \times 10^4$)
July – September	0-group	**8.83×10^5** ($5.87 \times 10^5 - 1.34 \times 10^6$)	**3.09×10^5** ($2.05 - 4.70 \times 10^5$)	**5.38×10^4** ($3.58 - 8.17 \times 10^4$)
	1-group	**3.82×10^3** ($2.54 - 5.80 \times 10^3$)	2.67×10^3 ($1.78 - 4.06 \times 10^3$)	**4.65×10^2** ($3.09 - 7.06 \times 10^2$)
	2-group	0	0	0
October – December	0-group	**1.15×10^6** ($7.14 \times 10^5 - 1.87 \times 10^6$)	**4.85×10^5** ($3.00 - 7.84 \times 10^5$)	**8.44×10^4** ($5.22 \times 10^4 - 1.36 \times 10^5$)
	1-group	**1.65×10^3** ($1.02 - 2.67 \times 10^3$)	**1.35×10^3** ($8.39 \times 10^2 - 2.19 \times 10^3$)	**2.36×10^2** ($1.46 - 3.81 \times 10^2$)
	2-group	0	0	0
total		**2.22×10^6** ($1.41 - 3.54 \times 10^6$)	**9.20×10^5** ($5.80 \times 10^5 - 1.49 \times 10^6$)	**1.60×10^5** ($1.01 - 2.59 \times 10^5$)

Table 4.5. Estimated impingement of 0 – 2-group cod at LPS with estimated abundance and biomass of equivalent adults that would have existed had the impinged fish not experienced mortality. EAV (equivalent adult value) factors obtained from Turnpenny (1989). Mass values calculated from mean biomass of age 4 cod of 3.305kg (ICES, 2001a). Values in parentheses are ranges of 95% confidence intervals.

(a) 1999		**Estimated impinged abundance**	**Abundance of equivalent adults**	**Mass of equivalent adults (kg)**
January – March	0-group	**0**	**0**	**0**
	1-group	**7.45×10^3** ($1.25 \times 10^3 - 4.65 \times 10^4$)	**5.51×10^2** ($93 - 3.44 \times 10^3$)	**1.82×10^3** ($3.07 \times 10^2 - 1.14 \times 10^4$)
	2-group	**5.32×10^2** ($90 - 3.32 \times 10^3$)	79 ($13 - 4.94 \times 10^2$)	**2.62×10^2** ($44 - 1.63 \times 10^3$)
April – June	0-group	**0**	**0**	**0**
	1-group	**0**	**0**	**0**
	2-group	**0**	**0**	**0**
July – September	0-group	**5.75×10^4** ($4.13 - 9.05 \times 10^4$)	**2.70×10^3** ($1.94 - 4.26 \times 10^3$)	**8.93×10^3** ($6.42 \times 10^3 - 1.41 \times 10^4$)
	1-group	**0**	**0**	**0**
	2-group	**0**	**0**	**0**
October – December	0-group	**3.98×10^4** ($2.21 - 7.30 \times 10^4$)	**2.39×10^3** ($1.33 - 4.38 \times 10^3$)	**7.89×10^3** ($4.38 \times 10^3 - 1.45 \times 10^4$)
	1-group	**0**	**0**	**0**
	2-group	**0**	**0**	**0**
total		**1.05×10^5** ($6.48 \times 10^4 - 2.13 \times 10^5$)	**5.72×10^3** ($3.37 \times 10^3 - 1.26 \times 10^4$)	**1.89×10^4** ($1.12 - 4.15 \times 10^4$)

(b) 2000		**Estimated impinged abundance**	**Abundance of equivalent adults**	**Mass of equivalent adults (kg)**
January – March	0-group	**0**	**0**	**0**
	1-group	**1.51×10^4** ($5.45 \times 10^3 - 5.05 \times 10^4$)	**1.12×10^3** ($4.04 \times 10^2 - 3.73 \times 10^3$)	**3.69×10^3** ($1.33 \times 10^3 - 1.23 \times 10^4$)
	2-group	**0**	**0**	**0**
April – June	0-group	**0**	**0**	**0**
	1-group	**3.62×10^3** ($7.50 \times 10^2 - 1.75 \times 10^4$)	**3.05×10^2** ($63 - 1.47 \times 10^3$)	**1.01×10^3** ($2.08 \times 10^2 - 4.87 \times 10^3$)
	2-group	**0**	**0**	**0**
July – September	0-group	**8.74×10^4** ($6.58 \times 10^4 - 1.20 \times 10^5$)	**4.11×10^3** ($3.09 - 5.65 \times 10^3$)	**1.36×10^4** ($1.02 - 1.87 \times 10^4$)
	1-group	**2.07×10^3** ($1.56 - 2.85 \times 10^3$)	**2.07×10^2** ($1.56 - 2.85 \times 10^2$)	**6.84×10^2** ($5.15 - 9.41 \times 10^2$)
	2-group	**0**	**0**	**0**
October – December	0-group	**1.52×10^5** ($1.05 - 2.43 \times 10^5$)	**9.14×10^3** ($6.32 \times 10^3 - 1.46 \times 10^4$)	**3.02×10^4** ($2.09 - 4.82 \times 10^4$)
	1-group	**1.85×10^3** ($1.28 - 2.95 \times 10^3$)	**2.29×10^2** ($1.58 - 3.66 \times 10^2$)	**7.57×10^2** ($5.23 \times 10^2 - 1.21 \times 10^3$)
	2-group	**0**	**0**	**0**
total		**2.62×10^5** ($1.80 - 4.37 \times 10^5$)	**1.51×10^4** ($1.02 - 2.61 \times 10^4$)	**4.99×10^4** ($3.37 - 8.63 \times 10^4$)

Table 4.6. Estimated impingement of 0 – 2-group plaice at LPS with estimated abundance and biomass of equivalent adults that would have existed had the impinged fish not experienced mortality. EAV (equivalent adult value) factors obtained from Turnpenny (1989). Mass values calculated from mean biomass of age 4 plaice of 0.349kg (ICES, 2001a). Values in parentheses are ranges of 95% confidence intervals.

(a) 1999		Estimated impinged abundance	Abundance of equivalent adults	Mass of equivalent adults (kg)
January – March	0-group	**0**	**0**	**0**
	1-group	**2.18×10^{4}** ($5.46 \times 10^{3} - 9.09 \times 10^{5}$)	**2.66×10^{3}** ($6.66 \times 10^{2} - 1.11 \times 10^{4}$)	**9.28×10^{2}** ($2.32 \times 10^{2} - 3.87 \times 10^{3}$)
	2-group	**1.88×10^{3}** ($4.71 \times 10^{2} - 7.84 \times 10^{3}$)	**4.32×10^{2}** ($1.08 \times 10^{2} - 1.80 \times 10^{3}$)	**1.51×10^{2}** ($38 - 6.29 \times 10^{2}$)
April – June	0-group	**5.52×10^{3}** ($4.26 - 7.46 \times 10^{3}$)	**8** (6 – 10)	**3** (2 – 4)
	1-group	**2.64×10^{4}** ($2.04 - 3.56 \times 10^{4}$)	**3.82×10^{3}** ($2.95 - 5.17 \times 10^{3}$)	**1.33×10^{3}** ($1.03 - 1.80 \times 10^{3}$)
	2-group	**4.29×10^{3}** ($3.32 - 5.80 \times 10^{3}$)	**1.11×10^{3}** ($8.56 \times 10^{2} - 1.50 \times 10^{3}$)	**3.87×10^{2}** ($2.99 - 5.22 \times 10^{2}$)
July – September	0-group	**1.20×10^{5}** ($6.49 \times 10^{4} - 2.28 \times 10^{5}$)	**1.94×10^{3}** ($1.05 - 3.69 \times 10^{3}$)	**6.76×10^{2}** ($3.67 \times 10^{2} - 1.29 \times 10^{3}$)
	1-group	**1.40×10^{4}** ($7.59 \times 10^{3} - 2.66 \times 10^{4}$)	**2.28×10^{3}** ($1.24 - 4.34 \times 10^{3}$)	**7.96×10^{2}** ($4.32 \times 10^{2} - 1.51 \times 10^{3}$)
	2-group	**7.32×10^{3}** ($3.98 \times 10^{3} - 1.39 \times 10^{4}$)	**2.12×10^{3}** ($1.15 - 4.03 \times 10^{3}$)	**7.39×10^{2}** ($4.01 \times 10^{2} - 1.41 \times 10^{3}$)
October – December	0-group	**9.70×10^{4}** ($7.44 \times 10^{4} - 1.39 \times 10^{5}$)	**6.66×10^{3}** ($5.11 - 9.50 \times 10^{3}$)	**2.32×10^{3}** ($1.78 - 3.32 \times 10^{3}$)
	1-group	**8.32×10^{3}** ($6.38 \times 10^{3} - 1.19 \times 10^{4}$)	**1.61×10^{3}** ($1.23 - 2.29 \times 10^{3}$)	**5.60×10^{2}** ($4.30 - 8.00 \times 10^{2}$)
	2-group	**1.76×10^{3}** ($1.35 - 2.52 \times 10^{3}$)	**5.40×10^{2}** ($4.14 - 7.71 \times 10^{2}$)	**1.88×10^{2}** ($1.44 - 2.69 \times 10^{2}$)
total		**3.08×10^{5}** ($1.93 - 5.68 \times 10^{5}$)	**2.32×10^{4}** ($1.48 - 4.42 \times 10^{4}$)	**8.08×10^{3}** ($5.16 \times 10^{3} - 1.54 \times 10^{4}$)

(b) 2000		Estimated impinged abundance	Abundance of equivalent adults	Mass of equivalent adults (kg)
January – March	0-group	**0**	**0**	**0**
	1-group	**1.20×10^{5}** ($6.49 \times 10^{4} - 2.26 \times 10^{5}$)	**1.47×10^{4}** ($7.92 \times 10^{3} - 2.76 \times 10^{4}$)	**5.12×10^{3}** ($2.76 - 9.63 \times 10^{3}$)
	2-group	**2.00×10^{3}** ($1.08 - 3.77 \times 10^{3}$)	**4.61×10^{2}** ($2.49 - 8.67 \times 10^{2}$)	**1.61×10^{2}** ($87 - 3.03 \times 10^{2}$)
April – June	0-group	**9.79×10^{4}** ($8.87 \times 10^{4} - 1.11 \times 10^{5}$)	**1.34×10^{2}** ($1.22 - 1.52 \times 10^{3}$)	**47** (42 – 53)
	1-group	**1.35×10^{5}** ($1.23 - 1.54 \times 10^{5}$)	**1.96×10^{4}** ($1.78 - 2.23 \times 10^{4}$)	**6.85×10^{3}** ($6.21 - 7.79 \times 10^{3}$)
	2-group	**3.71×10^{2}** ($3.36 - 4.22 \times 10^{2}$)	**96** ($87 - 1.09 \times 10^{2}$)	**33** (30 – 38)
July – September	0-group	**4.10×10^{5}** ($2.25 - 8.00 \times 10^{5}$)	**6.64×10^{3}** ($3.64 \times 10^{3} - 1.30 \times 10^{4}$)	**2.32×10^{3}** ($1.27 - 4.52 \times 10^{3}$)
	1-group	**1.77×10^{4}** ($9.73 \times 10^{3} - 3.47 \times 10^{4}$)	**2.89×10^{3}** ($1.59 - 5.65 \times 10^{3}$)	**1.01×10^{3}** ($5.53 \times 10^{2} - 1.97 \times 10^{3}$)
	2-group	**1.11×10^{3}** ($6.08 \times 10^{2} - 2.17 \times 10^{3}$)	**3.21×10^{2}** ($1.76 - 6.26$)	**1.12×10^{2}** ($61 - 2.18 \times 10^{2}$)
October – December	0-group	**3.92×10^{5}** ($1.86 - 8.41 \times 10^{5}$)	**2.69×10^{4}** ($1.27 - 5.77 \times 10^{4}$)	**9.39×10^{3}** ($4.44 \times 10^{3} - 2.01 \times 10^{5}$)
	1-group	**1.15×10^{4}** ($5.46 \times 10^{3} - 2.48 \times 10^{4}$)	**2.23×10^{3}** ($1.05 - 4.78 \times 10^{3}$)	**7.77×10^{2}** ($3.68 \times 10^{2} - 1.67 \times 10^{3}$)
	2-group	**0**	**0**	**0**
total		**1.19×10^{6}** ($7.04 \times 10^{5} - 2.20 \times 10^{6}$)	**7.39×10^{4}** ($4.53 \times 10^{4} - 1.33 \times 10^{5}$)	**2.58×10^{4}** ($1.58 - 4.63 \times 10^{4}$)

Table 4.7a. Estimated 1999 impingement of herring originating from Buchan Ness (assumed born on 1 September each year) at LPS, with estimated abundance and biomass of equivalent adults that would have existed had the impinged fish not experienced mortality. EAV (equivalent adult value) factors obtained from Turnpenny (1989). Mass values calculated from mean biomass of age 2 herring of 0.122kg (ICES, 2000). Values in parentheses are ranges of 95% confidence intervals.

(a) 1999	Born September 1998		Born September 1997		Born September 1996	
	Abundance	**Equivalent adults**	**Abundance**	**Equivalent adults**	**Abundance**	**Equivalent adults**
January	0	0	**1.12×10^5** (8.38×10^4 – 1.50×10^5)	**9.00×10^4** (6.72×10^4 – 1.21×10^5)	**6.15×10^3** (4.59 – 8.24 $\times 10^3$)	**8.30×10^3** (6.20×10^3 – 1.11×10^4)
February	0	0	**1.11×10^5** (5.63×10^4 – 2.19×10^5)	**9.27×10^4** (4.70×10^4 – 1.83×10^5)	**2.17×10^3** (1.10 – 4.29 $\times 10^3$)	**2.93×10^3** (1.48 – 5.79 $\times 10^3$)
March	0	0	**2.25×10^5** (1.61 – 3.15 $\times 10^5$)	**1.96×10^5** (1.40 – 2.74 $\times 10^5$)	**1.11×10^4** (7.90×10^3 – 1.55×10^4)	**1.68×10^4** (1.20 – 2.36 $\times 10^4$)
April	0	0	**2.50×10^4** (2.38 – 2.63 $\times 10^4$)	**2.17×10^4** (2.06 – 2.28 $\times 10^4$)	**5.43×10^2** (5.17 – 5.71 $\times 10^2$)	**8.26×10^2** (7.85 – 8.69 $\times 10^2$)
May	**2.96×10^2** (1.51 – 5.80 $\times 10^2$)	**2.12×10^2** (1.08 – 4.15 $\times 10^2$)	**1.42×10^4** (7.24×10^3 – 2.79×10^4)	**1.23×10^4** (6.29×10^3 – 2.42×10^4)	**2.96×10^2** (1.51 – 5.80 $\times 10^2$)	**5.00×10^2** (2.55 – 9.81 $\times 10^2$)
June	**4.33×10^4** (2.53 – 7.40 $\times 10^4$)	**3.10×10^4** (1.81 – 5.29 $\times 10^4$)	**3.33×10^3** (1.95 – 5.69 $\times 10^3$)	**2.90×10^3** (1.69 – 4.95 $\times 10^3$)	**3.33×10^2** (1.95 – 5.69 $\times 10^2$)	**5.63×10^2** (3.30 – 9.62 $\times 10^2$)
July	**8.30×10^4** (6.10×10^4 – 1.13×10^5)	**5.93×10^4** (4.36 – 8.08 $\times 10^4$)	**1.10×10^3** (8.06×10^2 – 1.49×10^3)	**1.02×10^3** (7.52×10^2 – 1.39×10^3)	0	0
August	**3.31×10^5** (2.48 – 4.42 $\times 10^5$)	**2.44×10^5** (1.83 – 3.25 $\times 10^5$)	**6.39×10^3** (4.79 – 8.53 $\times 10^3$)	**5.97×10^3** (4.47 – 7.96 $\times 10^3$)	**3.06×10^3** (2.29 – 4.08 $\times 10^3$)	**5.17×10^3** (3.87 – 6.90 $\times 10^3$)
September	**3.98×10^5** (3.54 – 4.46 $\times 10^5$)	**3.01×10^5** (2.68 – 3.38 $\times 10^5$)	**1.81×10^3** (1.62 – 2.04 $\times 10^3$)	**1.81×10^3** (1.61 – 2.03 $\times 10^3$)	**2.27×10^2** (2.02 – 2.55 $\times 10^2$)	**3.83×10^2** (3.41 – 4.30 $\times 10^2$)
October	**2.06×10^5** (1.68 – 2.53 $\times 10^5$)	**1.60×10^5** (1.30 – 1.96 $\times 10^5$)	**7.25×10^2** (5.91 – 8.91 $\times 10^2$)	**7.76×10^2** (6.32 – 9.53 $\times 10^2$)	**7.25×10^2** (5.91 – 8.91 $\times 10^2$)	**1.23×10^3** (9.98×10^2 – 1.51×10^3)
November	**1.19×10^5** (6.12×10^4 – 2.31×10^5)	**9.38×10^4** (4.82×10^4 – 1.82×10^5)	**2.84×10^2** (1.46 – 5.52 $\times 10^2$)	**3.44×10^2** (1.77 – 6.68 $\times 10^2$)	**2.84×10^2** (1.46 – 5.52 $\times 10^2$)	**4.80×10^2** (2.47 – 9.33 $\times 10^2$)
December	**4.06×10^5** (3.28 – 5.03 $\times 10^5$)	**3.26×10^5** (2.63 – 4.03 $\times 10^5$)	**5.08×10^3** (4.11 – 6.30 $\times 10^3$)	**6.15×10^3** (4.97 – 7.62 $\times 10^3$)	**2.33×10^3** (1.88 – 2.89 $\times 10^3$)	**3.94×10^3** (3.18 – 4.88 $\times 10^3$)

Total abundance = 2.12×10^6 (1.61 – 2.87 $\times 10^6$)

Total abundance of equivalent adults = 1.69×10^6 (1.28 – 2.29 $\times 10^6$)

Total mass of equivalent adults = 2.06×10^5 kg (1.56 – 2.79 $\times 10^5$ kg)

Table 4.7b. Estimated 2000 impingement of herring originating from Buchan Ness (assumed born on 1 September each year) at LPS, with estimated abundance and biomass of equivalent adults that would have existed had the impinged fish not experienced mortality. EAV (equivalent adult value) factors obtained from Turnpenny (1989). Mass values calculated from mean biomass of age 2 herring of 0.122kg (ICES, 2000). Values in parentheses are ranges of 95% confidence intervals.

(b) 2000	Born September 1999		Born September 1998		Born September 1997		Born September 1996	
	Abundance	**Equivalent adults**	**Abundance**	**Equivalent adults**	**Abundance**	**Equivalent adults**	**Abundance**	*Equivalent adults*
January	0	0	**7.26×10^5** ($6.73 - 7.84 \times 10^5$)	**5.83×10^5** ($5.40 - 6.29 \times 10^5$)	**2.05×10^4** ($1.90 - 2.21 \times 10^4$)	**2.76×10^4** ($2.56 - 2.98 \times 10^4$)	**3.70×10^3** ($3.42 - 3.99 \times 10^3$)	**6.25×10^3** ($5.79 - 6.75 \times 10^3$)
February	0	0	**5.77×10^5** ($2.48 \times 10^5 - 1.34 \times 10^6$)	**4.81×10^5** ($2.07 \times 10^5 - 1.12 \times 10^6$)	**1.56×10^3** ($6.69 \times 10^2 - 3.63 \times 10^3$)	**2.11×10^3** ($9.04 \times 10^2 - 4.90 \times 10^3$)	0	0
March	0	0	**1.10×10^6** ($5.84 - 2.05 \times 10^6$)	**9.52×10^4** ($5.08 \times 10^4 - 1.78 \times 10^5$)	7.39×10^2 ($3.94 \times 10^2 - 1.38 \times 10^3$)	**1.12×10^3** ($5.99 \times 10^2 - 2.10 \times 10^3$)	0	0
April	0	0	**3.07×10^5** ($1.90 - 4.96 \times 10^5$)	**2.67×10^5** ($1.65 - 4.31 \times 10^5$)	**3.08×10^2** ($1.91 - 4.98 \times 10^2$)	**4.68×10^2** ($2.90 - 7.56 \times 10^2$)	**3.08×10^2** ($1.91 - 4.98 \times 10^2$)	**5.21×10^2** ($3.22 - 8.41 \times 10^2$)
May	**5.77×10^4** ($1.64 \times 10^4 - 2.04 \times 10^5$)	**4.13×10^4** ($1.17 \times 10^4 - 1.46 \times 10^5$)	**3.83×10^4** ($1.08 \times 10^4 - 1.35 \times 10^5$)	**3.33×10^4** ($9.43 \times 10^3 - 1.17 \times 10^5$)	0	0	0	0
June	**2.45×10^5** ($1.19 - 5.03 \times 10^5$)	**1.75×10^5** ($8.54 \times 10^4 - 3.60 \times 10^5$)	**5.24×10^4** ($2.55 \times 10^4 - 1.08 \times 10^5$)	**4.55×10^4** ($2.22 - 9.35 \times 10^4$)	0	0	0	0
July	**6.31×10^5** ($5.52 - 7.22 \times 10^5$)	**4.52×10^5** ($3.95 - 5.16 \times 10^5$)	**5.52×10^4** ($4.83 - 6.31 \times 10^4$)	**5.15×10^4** ($4.50 - 5.89 \times 10^4$)	0	0	0	0
August	**8.97×10^5** ($6.93 \times 10^5 - 1.16 \times 10^6$)	**6.60×10^5** ($5.10 - 8.54 \times 10^5$)	**1.59×10^4** ($1.23 - 2.05 \times 10^4$)	**1.48×10^4** ($1.14 - 1.92 \times 10^4$)	0	0	0	0
September	**2.41×10^6** ($1.38 - 4.19 \times 10^6$)	**1.82×10^6** ($1.05 - 3.17 \times 10^6$)	**9.11×10^3** ($5.24 \times 10^3 - 1.58 \times 10^4$)	**9.09×10^3** ($5.23 \times 10^3 - 1.58 \times 10^4$)	0	0	0	0
October	**3.78×10^6** ($1.34 \times 10^6 - 1.06 \times 10^7$)	**2.93×10^6** ($1.04 - 8.23 \times 10^6$)	**1.19×10^4** ($4.21 \times 10^3 - 3.33 \times 10^4$)	**1.27×10^4** ($4.51 \times 10^3 - 3.57 \times 10^4$)	0	0	0	0
November	**1.24×10^6** ($4.59 \times 10^5 - 3.35 \times 10^6$)	**9.78×10^5** ($3.62 \times 10^5 - 2.64 \times 10^6$)	**8.00×10^3** ($2.96 \times 10^3 - 2.16 \times 10^4$)	**9.67×10^3** ($3.58 \times 10^3 - 2.61 \times 10^4$)	0	0	0	0
December	**4.22×10^5** ($1.68 \times 10^5 - 1.06 \times 10^6$)	**3.39×10^5** ($1.34 - 8.53 \times 10^5$)	**8.97×10^3** ($3.56 \times 10^3 - 2.26 \times 10^4$)	**1.08×10^4** ($4.31 \times 10^3 - 2.73 \times 10^4$)	**8.54×10^2** ($3.39 \times 10^2 - 2.15 \times 10^3$)	**1.44×10^3** ($5.73 \times 10^2 - 3.64 \times 10^3$)	0	0
	Total abundance = 1.16×10^7 ($6.04 \times 10^6 - 2.51 \times 10^7$)							
	Total abundance of equivalent adults = 9.05×10^6 ($4.69 \times 10^6 - 1.97 \times 10^7$)							
	Total mass of equivalent adults = 1.10×10^6 kg ($5.72 \times 10^5 - 2.39 \times 10^6$ kg)							

Table 4.8a. Estimated 1999 impingement of spring-spawned herring (assumed born on 1 March each year) at LPS, with estimated abundance and biomass of equivalent adults that would have existed had the impinged fish not experienced mortality. EAV (equivalent adult value) factors obtained from Turnpenny (1989). Mass values calculated from mean biomass of age 2 herring of 0.122kg (ICES, 2000). Values in parentheses are ranges of 95% confidence intervals.

(a) 1999	Born March 1999		Born March 1998		Born March 1997	
	Abundance	**Equivalent adults**	**Abundance**	**Equivalent adults**	**Abundance**	**Equivalent adults**
January	**0**	**0**	**4.43×10^3** ($3.31 - 5.93 \times 10^3$)	**3.17×10^3** ($2.36 - 4.24 \times 10^3$)	**5.90×10^3** ($4.41 - 7.91 \times 10^3$)	**5.51×10^3** ($4.12 - 7.38 \times 10^3$)
February	**0**	**0**	**0**	**0**	**1.95×10^3** ($9.89 \times 10^2 - 3.86 \times 10^3$)	**1.82×10^3** ($9.23 \times 10^2 - 3.60 \times 10^3$)
March	**0**	**0**	**4.61×10^3** ($3.29 - 6.46 \times 10^3$)	**3.49×10^3** ($2.49 - 4.89 \times 10^3$)	**6.46×10^3** ($4.61 - 9.05 \times 10^3$)	**6.44×10^3** ($4.60 - 9.03 \times 10^3$)
April	**0**	**0**	**0**	**0**	**0**	**0**
May	**0**	**0**	**0**	**0**	**0**	**0**
June	**0**	**0**	**3.33×10^2** ($1.95 - 5.69 \times 10^2$)		**0**	**0**
July	**0**	**0**	**0**	**0**	**0**	**0**
August	**8.34×10^2** ($6.25 \times 10^2 - 1.11 \times 10^3$)	**5.66×10^2** ($4.24 - 7.55 \times 10^2$)	**0**	**0**	**0**	**0**
September	**0**	**0**	**4.54×10^2** ($4.04 - 5.09 \times 10^2$)	**3.94×10^2** ($3.51 - 4.42 \times 10^2$)	**0**	**0**
October	**0**	**0**	**0**	**0**	**0**	**0**
November	**0**	**0**	**6.82×10^3** ($3.50 \times 10^3 - 1.33 \times 10^4$)	**5.92×10^3** ($3.05 \times 10^3 - 1.15 \times 10^4$)	**0**	**0**
December	**1.53×10^4** ($1.23 - 1.89 \times 10^4$)	**1.09×10^4** ($8.81 \times 10^3 - 1.35 \times 10^4$)	**1.03×10^5** ($8.28 \times 10^4 - 1.27 \times 10^5$)	**8.91×10^4** ($7.20 \times 10^4 - 1.10 \times 10^5$)	**0**	**0**

Total abundance = 1.50×10^5 ($1.16 - 1.94 \times 10^5$)

Total abundance of equivalent adults = 1.28×10^5 ($9.92 \times 10^4 - 1.66 \times 10^5$)

Total mass of equivalent adults = 1.56×10^4 kg ($1.21 - 2.03 \times 10^4$ kg)

Table 4.8b. Estimated 2000 impingement of spring-spawned herring (assumed born on 1 March each year) at LPS, with estimated abundance and biomass of equivalent adults that would have existed had the impinged fish not experienced mortality. EAV (equivalent adult value) factors obtained from Turnpenny (1989). Mass values calculated from mean biomass of age 2 herring of 0.122kg (ICES, 2000). Values in parentheses are ranges of 95% confidence intervals.

(b) 2000	Born March 2000		Born March 1999		Born March 1998	
	Abundance	**Equivalent adults**	**Abundance**	**Equivalent adults**	**Abundance**	**Equivalent adults**
January	**1.28×10^4** ($1.19 - 1.38 \times 10^4$)		**1.28×10^4** ($1.19 - 1.38 \times 10^4$)	**9.15×10^3** ($8.48 - 9.88 \times 10^3$)	**9.27×10^4** ($8.59 \times 10^4 - 1.00 \times 10^5$)	**8.65×10^4** ($8.02 - 9.34 \times 10^4$)
February	**0**	**0**	**0**	**0**	**4.46×10^2** ($1.91 \times 10^2 - 1.04 \times 10^3$)	**4.16×10^2** ($1.79 - 9.69 \times 10^2$)
March	**0**	**0**	**0**	**0**	**2.46×10^2** ($1.31 - 4.61 \times 10^2$)	**2.46×10^2** ($1.31 - 4.61 \times 10^2$)
April	**0**	**0**	**0**	**0**	**0**	**0**
May	**0**	**0**	**0**	**0**	**0**	**0**
June	**0**	**0**	**0**	**0**	**0**	**0**
July	**0**	**0**	**0**	**0**	**0**	**0**
August	**7.55×10^2** ($5.84 - 9.77 \times 10^2$)	**5.19×10^2** ($4.01 - 6.72 \times 10^2$)	**0**	**0**	**0**	**0**
September	**8.28×10^2** ($4.76 \times 10^2 - 1.44 \times 10^3$)	**5.65×10^2** ($3.25 - 9.82 \times 10^2$)	**0**	**0**	**0**	**0**
October	**0**	**0**	**0**	**0**	**0**	**0**
November	**0**	**0**	**1.24×10^4** ($4.57 \times 10^3 - 3.34 \times 10^4$)	**1.07×10^4** ($3.97 \times 10^3 - 2.90 \times 10^4$)	**0**	**0**
December	**0**	**0**	**1.10×10^5** ($4.37 \times 10^4 - 2.77 \times 10^5$)	**9.57×10^4** ($3.80 \times 10^4 - 2.41 \times 10^5$)	**0**	**0**

Total abundance = 2.30×10^5 ($1.47 - 4.29 \times 10^5$)

Total abundance of equivalent adults = 2.04×10^5 ($1.32 - 3.77 \times 10^5$)

Total mass of equivalent adults = 2.49×10^4 kg ($1.61 - 4.59 \times 10^4$ kg)

Herring dominated estimated total wet biomass of equivalent adults lost by impingement of juvenile fish at LPS during 1999 – 2000, with values an order of magnitude greater than whiting, and two orders of magnitude greater than both cod and plaice (Table 4.9). Monetary value of the relatively large total mass of herring was approx. 1.5 × that of whiting, reflecting the lower value per unit mass of herring. The values per unit mass of cod and plaice were similar, thus the total value of the estimated wet masses lost as equivalent adults were approx. 2:1 in favour of the former species, explained by the wet mass of equivalent adults being in a similar ratio. The relatively high monetary value per unit mass of cod meant that the estimated total monetary value of this species was approx. 56% that of whiting and 36% that of herring, whilst the total wet mass was of the order of 26% and 4% respectively (Table 4.9).

Table 4.9. Estimated total wet mass and monetary value of equivalent adult whiting, cod, plaice and herring potentially lost through impingement of juveniles at LPS, January 1999 – December 2000. Values in parentheses are ranges of 95% confidence intervals. Monetary values based on data of fish landed into Scottish ports from ICES area IVb by UK vessels in 1999 (Scottish Executive Publications, 2000).

species	**mass of equivalent adults**	**value**
whiting	**259.7 t** (165.1 – 427.6 t)	**£145,612** (£92,293 - £239,031)
cod	**68.8 t** (44.8 – 127.8 t)	**£82,166** (£53,513 - £152,560)
plaice	**33.9 t** (21.0 – 61.8 t)	**£40,080** (£24,814 - £73,027)
herring	**1637 t** (756 – 2734 t)	**£225,858** (£104,288 - £377,062)

4.3.3. Extent of impingement of salmonid smolts during the 2000 downstream migration

A total of 56 Atlantic salmon and 19 sea trout smolts were collected during routine sampling sessions between 27 April and 5 June 2000. The greatest rate of impingement of both species was on 3 May, at HW of a spring tide during darkness (Table 4.10). Salmon smolts ranged from 101mm (8 g wet mass) to 145mm TL (22.1 g), while sea trout were in the range 111 mm (12.9 g) to 191 mm (63.1g).

An estimated 3.62×10^{11} l and 4.80×10^{11} l of CW was abstracted from 16 April – 31 May and 16 April – 16 June, respectively. The former period was assumed to be the downstream migratory period of salmon smolts, while the latter was that of sea trout (see section 4.2.3). During the first of these periods it was estimated that the total abundance of salmon smolts impinged at LPS was 5.18×10^3 (95% CIs: $3.99 - 6.73 \times 10^3$), while the total impinged abundance of sea trout smolts during the latter period was approximately 255 (105 – 618).

Table 4.10. Details of impingement of salmonid smolts at LPS during the 2000 downstream migration season. Abbreviations of sampling sessions: HW/LW = high/low water; S/N = spring/neap tide; D/L = darkness/daylight. Thus, for example, HWSD = high water of spring tide in darkness. - indicates sessions assumed undertaken after cessation of the salmon smolt downstream migration season (see text for details).

	fish per 10^8 l CW sampled	
sampling session	**salmon**	**sea trout**
27 April, 0240, LWND	0	0
27 April, 0900, HWNL	0	0
27 April, 1530, LWNL	1.22	2.44
27 April, 2210, HWND	0	0
3 May, 0330, HWSD	47.62	9.77
5 May, 0910, LWSL	13.43	3.66
5 May, 1720, HWSL	3.66	1.22
19 May, 2150, LWSD	4.88	0
26 May, 0200, LWND	0	1.22
26 May, 0830, HWNL	0	0
26 May, 1450, LWNL	0	0
30 May, 0050, HWND	0	0
1 June, 0250, HWSD	-	0
5 June, 1040, LWSL	-	4.88
5 June, 1840, HWSL	-	0
5 June, 2300, LWSD	-	0
9 June, 0220, LWND	-	0
13 June, 0110, HWND	-	0
13 June, 0640, LWNL	-	0
13 June, 1310, HWNL	-	0

4.3.4. Extent of impingement of river lamprey and eel

4.3.4.1. Extent of impingement of river lamprey

In 1999 an estimated 4.31×10^4 (range of sums of minimal and maximal 95% confidence intervals: $1.16 \times 10^4 - 2.45 \times 10^5$) river lamprey were impinged at LPS. A somewhat greater quantity of the species was impinged in the year 2000, this being approximately 7.23×10^4 (95% CIs: $2.39 \times 10^4 - 2.52 \times 10^5$) lamprey. Impingement was greatest during August and December 2000 and December 1999 – January 2000, while exhibiting minimal levels during the period from March – May in both years (Figure 4.3).

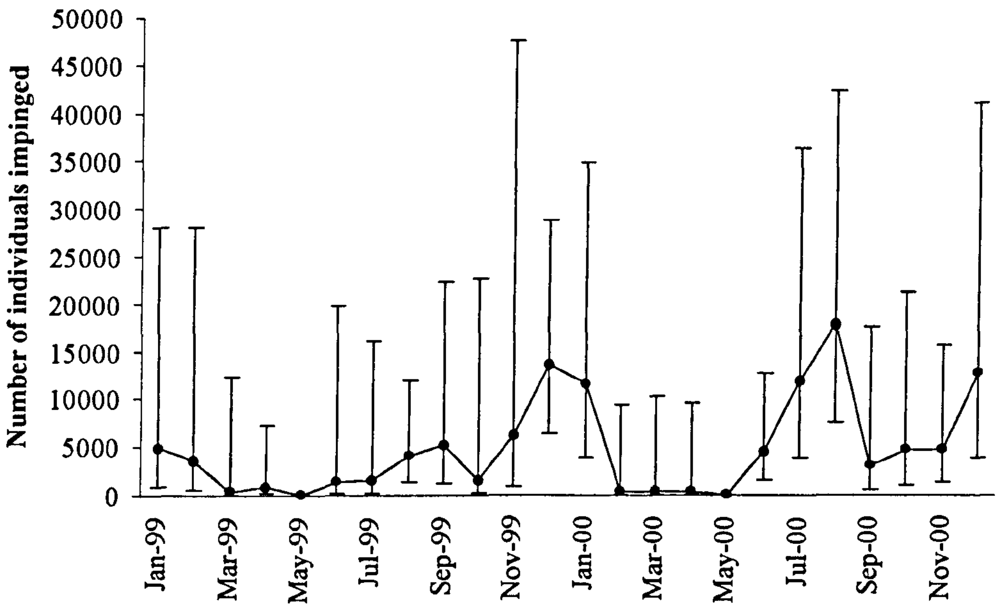


Figure 4.3. Estimated abundance of river lamprey impinged at LPS, 1999 – 2000. Error bars indicate 95% confidence intervals.

The length-frequency distribution of lamprey collected in impingement samples at LPS showed two main groups of lamprey with modes of about 100 – 120mm and 230 – 260mm respectively (Figure 4.4). The former was likely to constitute juveniles undergoing downstream migration primarily in spring and early summer, while the latter would have been a mixture of older juveniles inhabiting the estuary for feeding purposes and adults undertaking upstream migration to spawn in freshwater (Maitland *et al.*, 1984).

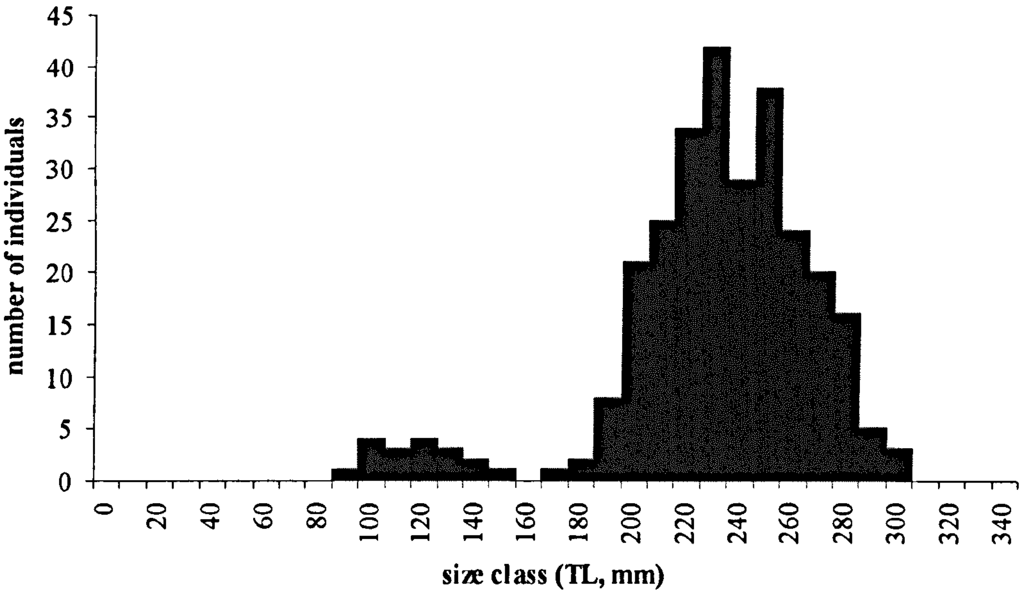


Figure 4.4. Length frequency distribution of river lamprey collected in impingement samples at LPS, January 1999 – December 2000.

4.3.4.2. Extent of impingement of eel

As with lamprey, eel impingement was greater in the latter year of the study: in 1999 an estimated 1113 individuals were impinged at LPS (range of sums of minimal and maximal 95% confidence intervals: 23 – 90742) while in 2000 the abundance was estimated at more than double the previous year's, 2441 (range of 95% CIs: 50 – 154110) fish. The wide confidence intervals were due to the very infrequent occurrence of eels in impingement samples. Peaks of impingement at LPS occurred in August 1999 and the period from May – August 2000, while numbers were lowest during the January – March period each year (Figure 4.5).

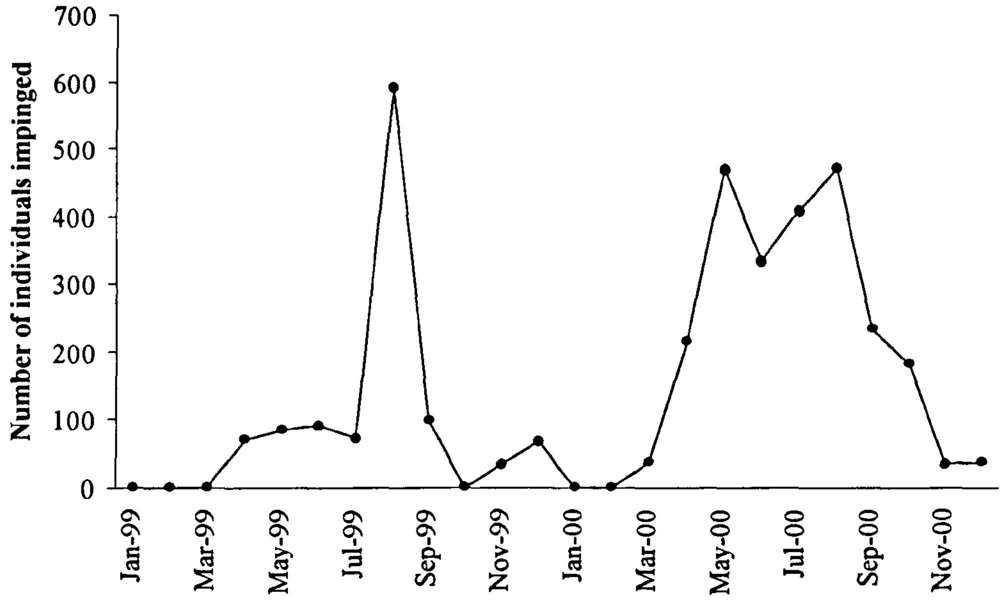


Figure 4.5. Estimated impingement of eel at LPS, January 1999 – December 2000. 95% CIs omitted for clarity.

All individuals taken were > 200 mm TL, and the majority were yellow eels of 210 – 350 mm TL, with only a few specimens being silver eels (Figure 4.6).

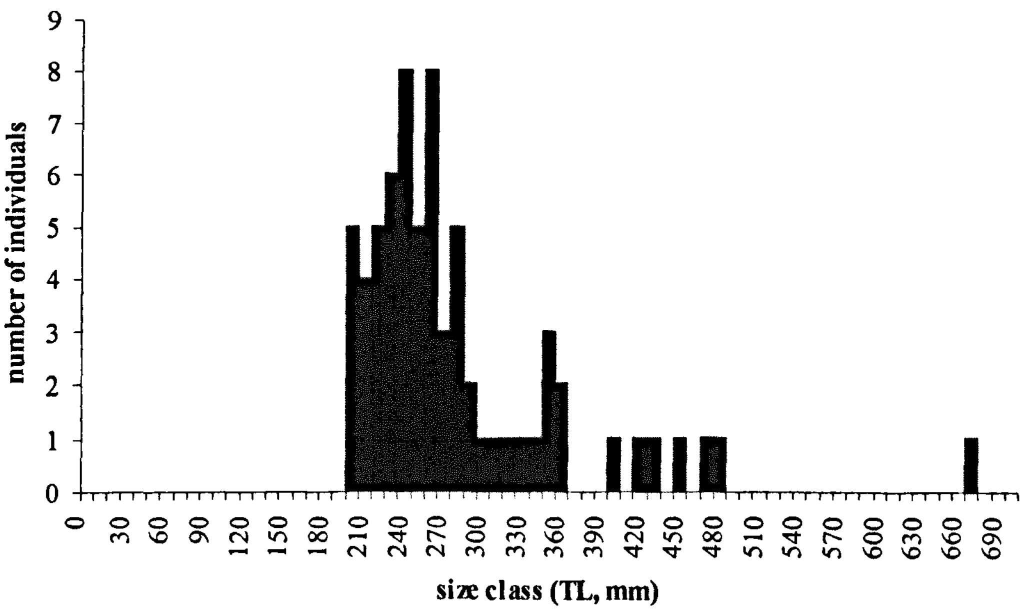


Figure 4.6. Length frequency distribution of eel collected in impingement samples at LPS, January 1999 – December 2000.

4.4. Discussion

4.4.1. Impingement of commercially important species at LPS, 1999 – 2000

Impingement of sprat, herring, plaice, whiting and cod above MCLS in 1999 - 2000 produced a quantity of fish that would have been worth approx £3300 at 1999 values. This was largely attributable to the inclusion of all sprats estimated impinged during this time period in the calculation. In reality, sprats that are > 12 cm TL (approx 15 g wet mass) are the most valuable to fishermen (J.N. McCallum, Scottish Fisheries Protection Agency, personal communication), and since relatively few sprats impinged were of this size (modal length was in the range 7 – 9 cm TL; Figure 4.2e), estimated monetary values for sprat were likely to be overestimates. Few fish above MCLS of the remaining four assessed species were taken. This was probably attributable to a) the relatively small proportions of older and larger fish in most populations; b) the function of the estuary for marine species being primarily as a nursery, with adults being less frequent visitors (Elliott *et al.*, 1990); c) the greater likelihood of adults being above critical escape lengths required to avoid removal in the cooling water inflow (Turnpenny, 1988a; see section 5.2.2). The importance of the second of these points is apparent when comparing losses of commercial species above MCLS at LPS with values suggested for the Sizewell power stations of the Suffolk coast (Table 4.11). Only values for impingement of adult herring were comparable to the other sites, reflecting the fact that adult plaice, cod and whiting are MJ species, and so were not as abundant in the Forth Estuary as coastal waters. Herring, in contrast, are MS species, with adults as well as juveniles overwintering in estuaries and inshore areas.

Table 4.11. Estimated annual mean mass of fish ≥ MCLS impinged at LPS (1999 – 2000) compared with data for existing Sizewell A and B power stations, as well as the proposed Sizewell C station (Turnpenny and Henderson, 1993).

	Annual mass of impinged fish ≥ MCLS (kg)			
	LPS	**Sizewell A**	**Sizewell B**	**Sizewell C**
whiting	29	2796	6051	12080
cod	0	708	1533	3060
plaice	0	18	38	76
herring	1310	570	1233	2460
total	1339	4092	8855	17676

Of greater potential significance in monetary estimation of value of impinged fish were the estimated equivalent adults that may have existed had juveniles not experienced mortality. The EAV method employed was not expected to give precise estimates, but order-of-magnitude approximations (Turnpenny, 1989). The crude estimates generated in the present study suggested that, from January 1999 – December 2000, the juvenile plaice, cod, whiting and herring removed by LPS may have represented future losses of adults in the region of 1.27×10^7 fish with mass 1999 t and worth approximately £494,000 at 1999 values. Since whiting, plaice, and herring are spawned in the North Sea and are not limited in their distribution solely to the Forth Estuary, it is relevant to place potential losses at LPS into the context of catches by North Sea commercial fisheries. Landings of cod and plaice from ICES fishing area IV (the North Sea) from 1995 – 1999 averaged 105851 t and 83055 t respectively (ICES, 2001a). The mean annual LPS equivalent adult losses of cod and plaice in 1999 – 2000 amounted to approximately 0.032% and 0.020% of these totals. No data on discards were available for these two species. Total herring catch from ICES areas IVa (northern North Sea) and IVb (central North Sea) in 1999 was 203839 t, of which 873 t was discarded (ICES, 2000). The annual mean based on the 2 years of data from the present study suggested that herring equivalent adults removed by LPS would have constituted 0.54 % of the total catch, or almost 1.3 × the total discarded mass. Discards often contain many undersized fish, however, so comparing the mass of equivalent adults to total mass of discards was not entirely appropriate. Length-frequency compositions of discarded fish are necessary to allow more accurate comparisons. Such data from northeast English fisheries suggest that the proportion of total whiting discards attributable to 2 year-old fish (*i.e.* those considered adults in the present study) was approximately 22.4% by mass, based on the variety of discard tendencies displayed by seine, otter, Nephrops and beam trawlers (ICES, 2001b). The mean annual total discards of whiting from ICES subarea IV from 1995 – 1999 was 22931 t (ICES, 2001a); assuming similar proportions of 2 year old fish were discarded by the North Sea whiting fleet as suggested by the NE English data above, LPS would have removed equivalent adult whiting amounting to approximately 2.6 % of the annual total of 2-year old discarded fish.

Were the estimates of equivalent adults that may have ultimately been lost due to impingement at LPS of reasonable magnitude? LPS was estimated to have impinged a total mean annual abundance of fish species of all sizes in the region of 2.2×10^7 fish (Chapter 2). This was comparable to the total quantity of fish impinged at eight power stations on the SE coast of England (Table 4.12; Henderson, 2000). Estimates of the total mass of equivalent adults of the four commercial species considered in the present study were somewhat greater at LPS than at the SE English power stations combined (Turnpenny and Henderson, 1993), though of the same order of magnitude, and the composition of the total mass varied markedly for all species excepting plaice (Table

4.12). Estimated total mass of equivalent adult whiting lost at LPS was half that of the SE English data, while cod equivalent adult mass was 4 – 5 × greater. These differences may be accounted for by differential distribution of the juveniles of these gadoid species, *i.e.* whiting tending to be more abundant in the inshore waters of SE England, while cod are more common in the Forth Estuary than the English coast. Inshore trawl data from SE England suggested that cod were scarce in the region of the power stations represented in table 4.12, with mean catches from zero to well below one fish per 1000 m^2 sampled from 1981 – 1997 (Rogers *et al.*, 1998). This contrasted with a mean of 1.2 cod 1000 m^{-2} in the Forth Estuary from 1982 – 1997 (based on data from Chapter 3). Whiting abundance in the Forth over the same time period averaged 5.5 fish 1000 m^{-2}, while the inshore regions of SE England possessed many areas with whiting at abundances of 10 – 50 fish 1000 m^{-2} (Rogers *et al.*, 1998).

Table 4.12. Comparison of estimated equivalent adult mass loss of commercially important species at LPS (mean annual values based on period from January 1999 – December 2000) and values estimated for SE English power stations (Turnpenny and Henderson, 1993). * power stations included were Sizewell A, Sizewell B, Bradwell, Littlebrook C, West Thurrock, Tilbury C, Isle of Grain and Kingsnorth. ‡ data for total impingement of all species at the SE English power stations from Henderson (2000).

		LPS, annual mean 1999 – 2000	**SE English power stations***
annual impinged abundance, all species (number of individuals)		2.19×10^7	2.84×10^7‡
equivalent adult mass of commercial species (t)	whiting	129.8	345
	cod	34.4	7.7
	plaice	16.9	11.1
	herring	818.7	435
	total	999.8	798.8

Equivalent adult estimates of herring impingement at LPS were approx. twice the total mass estimated impinged at SE English power stations (Table 4.12). Such a difference may have arisen for several reasons. Firstly, the present study employed an assumed mass at age 2 of 0.122 kg for equivalent adults, whereas the SE English estimates were undertaken at a time when a value of 0.113 kg was appropriate (Turnpenny and

Henderson, 1993). Each equivalent adult fish was therefore approx. 10% lighter. The principle period of impingement of herring in their first year of life occurs at the same time of the year (*i.e.* late summer – autumn) at LPS and Sizewell power stations, but the fish are not of the same age: as previously noted, herring impinged at LPS are generally from Buchan stock and were spawned in August – September of the year prior to impingement. The younger SE English herring, originating from the Downs spawning population, are spawned in November – January, thus have lower EAV factors for the same period of the year as Forth herring. The decreased probability of survival to maturity thus contributed to each juvenile herring from SE English locations accounting for a reduced proportion of an equivalent adult compared with juvenile herring impinged at the same time at LPS. Given these various pieces of evidence, the estimates of equivalent adult biomass of all species impinged at LPS seemed not unreasonable.

Estimates of equivalent adult tonnage losses due to impingement of juveniles at LPS in 1999 – 2001 were two orders of magnitude greater than the predicted losses from the PISCES (v.3) simulation undertaken by Turnpenny (1997) (Table 4.13). This was clearly attributable to the fact that the PISCES simulation produced an estimate of total impingement of all species that was two orders of magnitude less than that of the present study (Table 4.13).

Table 4.13. Comparison of estimated equivalent adult mass loss of commercially important species at LPS (mean annual values based on period from January 1999 – December 2000) and values predicted for LPS by the PISCES (v.3) software system (Turnpenny, 1997).

		LPS, annual mean 1999 – 2000	**LPS, PISCES simulation**
annual impinged abundance, all species (number of individuals)		2.19×10^7	7.41×10^5
equivalent adult mass of commercial species (t)	whiting	129.8	4.7
	cod	34.4	0.9
	plaice	16.9	0.05
	herring	818.7	3.9
	total	999.8	9.6

The PISCES estimate of annual EAV tonnage lost at LPS was described as being "well below the expected annual landings from a small inshore trawler" (Turnpenny, 1997). Comparisons of the present study's estimates are hard to make without precise data on the catches of individual trawlers. There were 40 pelagic vessels based in Scotland in 1999, which landed a total of 84455 t of herring in the UK and abroad (Scottish Executive Publications, 2000), suggesting a mean of approximately 2111 t per vessel. Impingement of juvenile herring at LPS in 1999 would have constituted just over 10% of the equivalent adult annual catch of one such 'average' vessel. 555 demersal vessels based in Scotland landed 20659 t, 26567 t, and 8029 t of whiting, cod and plaice respectively (Scottish Executive Publications, 2000). Not all demersal vessels fish for these species in equal proportions, due to factors such as geographical location or quota restrictions, but if it is assumed for simplicity that this was the case, then equivalent adult impingement at LPS in 1999 would have amounted to the tonnage of whiting landed by 2 – 3 theoretical average vessels, and 40% and 56% of the mass of cod and plaice respectively landed by a single such average vessel. Data for 2000 landings were unavailable for a similar comparison to be undertaken, though it would be likely that LPS impingement figures would be proportionally greater, due to total allowable catch

(TAC) of the demersal species being reduced (ICES, 2001a) and the TAC of herring remaining at approximately the same level as 1999 (ICES, 2000).

Equivalent adult tonnages of herring regarded as spring-spawned averaged just over 92 t per annum. This was well below historical levels of fishing catch of Firth of Forth spring-spawning herring (see section 4.2.2.3). Confirmation of the status of the herring assumed to be spring-spawned was not undertaken in the present study, but may be of use in the future in order to clarify this issue. Small spring-spawning stocks exist on the SE coast of Britain, *e.g.* the Wash and the Thames/Blackwater estuaries (MSC, 2000), but whether herring from these areas would be likely to enter the Forth and experience impingement is uncertain. Wood (1959) describes spring-spawned herring being taken inshore on the NE coast of England (Blyth, Sunderland and Hartlepool) that may have originated from spawning grounds in the vicinity of the Berwick Bank. It may be possible that these herring enter the Forth, along with Firth of Forth spring-spawned herring, and produce the occasional anomalies in length-frequency distributions of what are predominantly late summer/autumn-spawned herring originating from Buchan Ness.

4.4.2. Impingement of salmonid smolts during the 2000 downstream migration

It must be borne in mind that the estimates of salmonid smolt impingement were subject to various potential sources of error, not least of which was the fact that they were generated from relatively few routine sampling sessions. Better results would be obtained from more intensive sampling during the expected period of downstream migration of smolts. Although discrimination of Atlantic salmon and sea trout smolts during 1999 was uncertain, it was undoubtedly clear that far more smolts were collected during the 2000 downstream migration season than the previous year. Six smolts of indeterminate species were collected during April – May 1999, compared with 75 the following year. Volumetric extrapolation of the 1999 data yielded an approximate total of 650 smolts impinged at LPS, though with extremely wide 95% confidence intervals (6.6 – 64099 smolts). This figure was an order of magnitude less than the estimated total abundance of salmon and sea trout smolts impinged during the 2000 downstream migration season. How much this was due to routine sampling in 2000 coinciding with one particular occasion when smolts were being impinged at a great rate (*i.e.* 3 May, 0330h) was uncertain. Smolt production has been shown to vary greatly between consecutive years, *e.g.* in the North Esk, where abundance was estimated at 98000 in 1967 compared with 227000 in 1968, and a decline from 173000 in 1975 to 93000 in 1976 (Shearer, 1992). Another contributory factor to increased catches in 2000 could have been that water use by LPS during the smolt downstream migratory season was greater in 2000 than 1999. In April – June 1999 an average of 2.8 cooling water pumps were observed to be working during sampling sessions; the same period in 2000 saw an average of 3.7 pumps working. The greater water use would have resulted in greater impingement of fish, since water extraction rate is the best predictor of fish impingement rate (see sections 4.4.1. and 2.4.2.2; Henderson and Seaby, 2000).

Impinged salmon smolt abundance at LPS in 2000 constituted approximately 0.016% of the total quantity of fish estimated to have been impinged in that year. This proportion was an order of magnitude lower than estimates of 0.16 – 0.22% for smolt impingement at Severn Estuary power stations (Table 4.14), as well as the PISCES v.3 software prediction of 2481 salmon removed from a total of 741109 fish of all species (0.34%;

Turnpenny, 1997). Thus estimated salmon smolt impingement at LPS was proportionally less than that observed at other power stations, but was an still an absolute order of magnitude larger due to total quantity of fish impinged at LPS being two orders of magnitude greater (Table 4.14). Comparison of contemporary Forth data with data from the Severn that is 10 – 30 years old is not ideal, for many salmon stocks have exhibited considerable declines in recent years (WWF, 2001). The 184 salmon captured by rod and line in the Severn in 1999 (Environment Agency of England and Wales, 2000) is an order of magnitude less than 1196 salmon fished from the Forth District in the same year (Anon., 2000), and may provide albeit scant evidence of salmon being more abundant in the Forth catchment than the Severn, assuming fishing effort was comparable. If a greater abundance of salmon and smolts were present in the Forth than the Severn, this could also have contributed to greater impingement of smolts estimated to have occurred at LPS.

Table 4.14. Comparison of LPS salmon smolt impingement estimate with data from Severn Estuary and Bristol Channel power stations. * data for abstraction rates and total annual impingement of all species from Henderson (2000).

power station	**water abstraction rate (m^3s^{-1})***	**period of study**	**total annual fish impingement (all species)***	**total salmon smolt impingement**	**reference**
Oldbury-on-Severn	26.5	July 1972 – June 1977	2.5×10^5	405 (annual mean)	Claridge and Potter (1994)
Berkeley	26.5	September 1974 – July 1977	2.5×10^5	490 (annual mean)	Claridge and Potter (1994)
Uskmouth	30.3	8 April – 12 May, 1989	2.9×10^5	642	Aprahamian and Jones (1997)
LPS	91.0	April 16 – June 16, 2000	3.3×10^7	5178	this study

How many of the smolts impinged at LPS in 2000 would have been likely to survive to return to the Forth area? Estimates of survival in Atlantic salmon between smolt stage and return to the homewaters as grilse (*i.e.* salmon undergoing an upstream migration after one winter at sea) or older fish (2 SW or 3 SW, meaning upstream migration after 2 or 3 winters at sea) are not available for the salmon of the Forth. The closest studied Scottish population was in the North Esk, where total survival from smolt-to-grilse, -2 SW and -3 SW ranged between 14 – 53% for each cohort of smolts from 1964 – 1985 (Shearer, 1992). If similar values were applicable to the Forth, and assuming the above extrapolations of total smolt abundance impinged were reasonable, then perhaps approximately 560 – 3570 fish that were likely to return within 3 winters at sea were lost from the Forth catchment due to impingement. Sea trout survival from the smolt stage to return as mature adults was estimated to be 30 – 40 % in the Bresle catchment area, Upper Normandy/Picardy, France (Euzenat *et al.*, 1999). If similar rates of survival were applicable to the Forth, the potential future loss of sea trout adults from the Forth catchment due to power station impingement at LPS would be in the region of 30 – 250 individuals. Such simple calculations take no account of the possible benefits to individuals avoiding impingement that removal of conspecifics in CW flows may confer. If, for example, less competition for food results from removal of a certain number of individuals, then remaining fish may experience enhanced prospects of survival. This point is elaborated in Chapter 5.

Atlantic salmon smolts have been shown to use a nocturnal selective ebb tide transport pattern of migration (Moore *et al.*, 1995). Use of the faster flowing upper sections of the water column, in the middle of a river or estuary, seems to be an energetically efficient way of reaching the sea. Thus smolts would tend to be least susceptible to impingement at a power station intake during a nocturnal ebb tide (Aprahamian and Jones, 1997), assuming that the intake is located relatively far from the middle of the water body. This is the case at LPS, where the intake is approximately 160m from the shore, in a section of estuary about 1km wide. Moore *et al.* (1995) observed smolts holding position in an area of low flow near a road bridge following passive upstream transportation with a flood tide, and Shearer (1992) states that smolts "rest in areas of low water velocity when currents are not in the direction in which they wish to travel". It is conceivable that smolts in the Forth Estuary seek similar areas, which are generally found at the margins of the water body, and increase likelihood of contact with the power station intake when in the vicinity of LPS. The large number of smolts impinged at HW on 3 May 2000 may have been due to downstream movement on the previous ebb tide, followed by lateral movements to the edges of the estuary to avoid upstream displacement with the flow of the incoming tide. Such a migration would have brought the smolts into the vicinity of the LPS intake and resulted in removal in abstracted CW.

4.4.3. Impingement of river lamprey and eel at LPS, 1999 – 2000

The estimates of impingement of river lamprey during 1999 – 2000 were very similar to the Figure of 4.68×10^4 individuals estimated to have been impinged at LPS in 1996 (Maitland, 1997). River lamprey differed from species such as cod and whiting, in that though younger individuals were impinged at LPS, the majority of the impinged abundance was adolescents and adults, as previously noted by Maitland *et al.* (1984) and Maitland (1997). The reason for such a trend may have been that juvenile lampreys undergoing a downstream migration following approximately 5 years in freshwater utilised the ebb currents of the estuary in a similar way to that of Atlantic salmon smolts discussed in section 4.4.2. Since the strongest currents would have been likely to be near the centre of the estuary, this would have facilitated avoidance of the LPS intake, which is relatively near to the shore. Impingement of river lamprey at KPS tended to include a much greater proportion of seaward migrating juveniles than older fish, believed to be caused by the CW intake being nearer to the centre of the water body than that of LPS (Figure 1.1), resulting in greater likelihood of removal of juvenile lamprey in abstracted CW (Maitland *et al.*, 1984). Clupeids, gadoids, and possibly to a lesser extent smelt, have all been noted as prey for river lamprey (references in Hardisty and Potter, 1971). Enhanced abundances of the first two of these groups of prey species in the estuary over the late summer – early spring period (as shown by impingement at LPS; see Chapter 2) would have provided significant resources for adolescent lampreys which were in the parasitic phase of the life cycle and may have returned to the estuary from the sea for feeding purposes during this time (Maitland *et al.*, 1984). Seeking prey in the mid-lower estuary would have caused these individuals to encounter the LPS CW intake more often than if they had been resident in the Firth of Forth or North Sea. Several whiting and herring collected during impingement sampling at LPS were noted to possess scarring characteristic of lamprey attachment (personal observations). In combination with larger lamprey that were likely to be sexually mature adults undergoing upstream migration in late summer and autumn, it was clear that LPS impinged primarily adolescent and adult river lamprey, with minimal abundances being noted in March – May of both years, the time of downstream migration of juveniles (Maitland *et al.*, 1984).

River lamprey are the only species of fish that was regularly impinged at LPS that is listed by the International Union for Conservation of Nature and Natural Resources (IUCN) and possesses Lower Risk (near threatened) status (Hilton-Taylor, 2000). The definition of this classification is as follows (IUCN, 2000):

VULNERABLE (VU) - A taxon is Vulnerable when it is not Critically Endangered or Endangered but is facing a high risk of extinction in the wild in the medium-term future.

LOWER RISK (LR) - A taxon is Lower Risk when it has been evaluated, and does not satisfy the criteria for vulnerable. Taxa included in the Lower Risk category can be separated into three subcategories:

1. Conservation Dependent (cd). Taxa which are the focus of a continuing taxon-specific or habitat-specific conservation programme targeted towards the taxon in question, the cessation of which would result in the taxon qualifying for one of the threatened categories above within a period of five years.

2. Near Threatened (nt). Taxa which do not qualify for Conservation Dependent, but which are close to qualifying for Vulnerable.
3. Least Concern (lc). Taxa which do not qualify for Conservation Dependent or Near Threatened.

Maitland (1997) noted that river lamprey are contained in the European Community (now European Union) Habitat and Species Directive, and that member states thus have a duty to protect and conserve this species. The prospects of reduction in quantities of river lamprey impinged at LPS are discussed in Chapter 5. Agassiz trawl data from the study detailed in Chapter 3 showed river lamprey to be caught during trawling rather infrequently (Figure 4.7), with no significant trends in abundance.

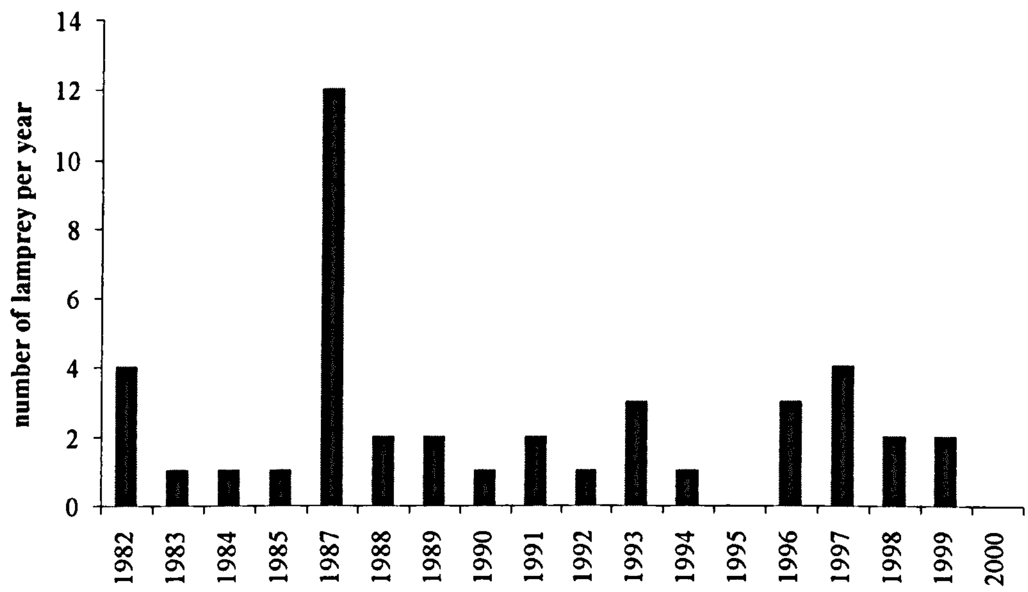


Figure 4.7. Abundance of river lamprey captured in Agassiz trawls of lower Forth Estuary (see Chapter 3 for details).

Thus, as noted by Maitland (1997), uncertainty exists over determining whether or not the environmental impact of LPS on lamprey populations is significant, at least in the sense of the population being able to sustain itself without risk of extinction. Under the most conservative of definitions of adverse environmental impact, that of any impingement occurring, it is clear that LPS has impinged river lamprey throughout its years of operation. Decline of river lamprey in other river and estuary systems that may be impacted with additional anthropogenic sources of mortality are likely to have resulted in designation of the LR (nt) status above. River lamprey in the mid-Severn Estuary, for example, have declined to approximately half of their 1970s level, possibly due to impediments to adult upstream migration and unfavourable alterations in silt habitats of the larval stages (Potter *et al.*, 2001). In terms of the population of river lamprey from the Forth it is unclear if power station impingement creates an unacceptable risk to the population's ability to sustain itself. River lamprey impingement at KPS between December 1961 and November 1962 amounted to approx 5700 individuals (Sharman, 1969). This was an order of magnitude less than the mean annual quantities impinged at LPS in 1999 – 2000 (this study) and 1996 (Maitland, 1997), and presumably was composed of mostly juveniles migrating downstream, assuming a similar pattern to the study of lamprey impingement at KPS by Maitland *et al.* (1984). The 1961 – 1962 study was undertaken while KPS was abstracting just under 15 m^3s^{-1} of CW (a power output of 360 MW), and additional screening was in place at the intake that were designed primarily to reduce removal of smolts (Sharman, 1969). The impingement of juvenile lamprey likely to have been during downstream migration in the present study was in the order of 4900 and 1375 individuals in 1999 and 2000 respectively. This presents a complex situation when attempting to infer changes that may have taken place in the population between the 1960s and the present study. While the impingement of downstream juvenile migrants in the present study at LPS was of the same order as the total suggested from KPS data in the 1960s, the present study consisted of samples from far greater quantities of CW than KPS. The observation that juveniles tend to be impinged at LPS in relatively low proportions compared to upstream migrating adults or adolescents (Maitland *et al.*, 1984; this study), whereas the opposite was true at KPS, could mean that, per unit volume of water in the estuary, little has changed with regard to abundance of river lamprey. It seems that the river lamprey population in the Forth catchment area initially experienced loss of primarily juvenile lamprey with the commencement of generation at KPS, which was then augmented to include loss of adolescents and adults when LPS came online, and, with decommissioning of KPS, changed to a situation where primarily older juveniles and adults are removed from the population, whilst downstream migrant juveniles still experience mortality to a similar or slightly less extent than when KPS was in its infancy. Chapter 5 discusses more fully implications of long-term power station operation on fish populations in theoretical terms.

A definite decline in eel abundance in the Forth over the past 40 years seems to be a more straightforward conclusion to reach in comparison to the uncertainty over lamprey population abundance discussed above. An estimated 11000 eels were impinged at KPS during the same 1961 – 1962 study mentioned above (Sharman, 1969). This was an order of magnitude greater than levels estimated from data of the present study, despite the fact that total water volume abstracted by LPS annually in 1999 – 2000 was approx 4.5 – 6 × the amount of the KPS study. Maitland (1997) estimated 4392 silver eel to have been impinged at LPS in 1996, a total abundance 2 – 3 × that of the present study, despite the fact that total quantity of electricity generated, and hence CW abstracted, in 1996 was somewhat less than the annual values for the present study (see Figure 5.2). The Agassiz trawl dataset from 1982 – 2000 was of no use in assessing trends in abundance of the species, for only four individuals were captured throughout the 19 years of the study (Table 3.1). The decline in eel populations in the Forth, as inferred above from declines in capture by power stations, appears to be a Europe-wide phenomenon. This is suggested by the fact that, for the period 1980 – 1999, elver and glass eel recruitment observed downward trends in 13 of 14 European river systems, yellow eels declined in 6 of 10 studied areas, and silver eel decreased in all five river areas examined (EIFAC, 1999). The systems not observing a downward trend in abundance showed no apparent trend in recruitment over the time series. Impingement data from Severn Estuary power stations suggested no significant change in abundance of eel between the mid-1970s and the late 1990s in the mid-estuary at Oldbury (Potter *et al.*, 2001), while Henderson and Seaby (2001) noted a decline at the outer estuarine Hinkley Point site between 1981 and 2001. Reasons for this almost universal decrease in eel abundance are uncertain, with possibilities including overfishing of elvers and habitat destruction (Henderson and Seaby, 2001), as well as the introduction of *Anguillicola crassus*, a swimbladder parasite that was introduced into Europe from Asia in the 1970s (Evans and Matthews, 1999). Henderson and Seaby (2001) noted that the decline in eel populations was not likely to be attributable to power station operation, but that impingement may come under greater scrutiny were conservation measures to be introduced in the future. Eighteen British and continental European power stations in the NE Atlantic were estimated to have removed approximately 2.4×10^5 yellow and silver eels per annum from 1980 – 1990 (Henderson, 2000); the mean of this figure is just over 1.3×10^4, indicating that the value obtained for LPS in the present study seems reasonable and may indicate further decline over the past decade.

4.5. Conclusions

The monetary value of commercially important species above MCLS lost directly to impingement at LPS in January 1999 – December 2000 was very low, and primarily attributable to impingement of sprat, a species for which no restrictions on landing size exist. Wet masses of whiting, cod, plaice, and herring that could have existed as adults were they not impinged as juveniles were much greater, though still relatively minor in the context of landings by fishing vessels in the North Sea. Similar trends have been observed at other power stations, and the comparison of incidental power station loss to directed fisheries generally suggests low values of loss due to impingement. Chapter 5 will discuss the potential limitations of reliance on the EAV method in assessing environmental impacts of LPS on the fish populations in the Forth Estuary. LPS was, based on the results of the present study, the largest single source of removal of commercially important species amongst British coastal and estuarine power stations at which impingement research has been undertaken. This was not wholly unexpected, bearing in mind that the station is the largest estuarine or marine generating plant in the UK. Estimates of impinged abundance of threatened river lamprey and declining eel, and recreationally and socio-politically important salmonid smolts showed that LPS could be regarded as displaying an adverse environmental impact, based simply on a conservative definition whereby any impingement would be regarded as detrimental to these populations. The difficulty in assessing whether or not LPS has an impact in terms of affecting the populations' abilities to sustain themselves was clear. The final Chapter of the present study will discuss general points regarding this difficulty.

Chapter 5. General Discussion

5.1. To what extent might LPS impact upon the fish populations of the Forth Estuary?

5.1.1. Introduction

LPS is the largest British marine or estuarine generating station and not surprisingly it seems likely that it has the greatest fish impingement mortality among UK power stations (Figure 2.4). Of paramount importance in the present study however is an attempt to discuss to what extent population-level effects on the ichthyofauna of the Forth Estuary may be occurring as a result of the impingement of numerous individuals.

Controversy exists over the impact of CW extraction on fish populations. The Electric Power Research Institute (EPRI) of the USA stated, in comments on the proposed US EPA CW legislation (US Federal Register, 2000):

"EPRI and some of the best fishery scientists in the world have never identified a site where definitive or conclusive aquatic population or community level impacts have occurred from operation of CWIS [cooling water intake structures], ..." (EPRI, 2000, p.2).

A similar view was given by Dey *et al.* (2000, p. S17):

"To the authors' direct experience and knowledge, there have been no demonstrated adverse impacts on populations or the balanced indigenous community by CWS [cooling water systems] operation, suggesting that entrainment/impingement losses are typically small in relation to the compensatory capacity of the involved populations".

The "compensatory capacity" referred to in the above quotation is the various means by which populations avoid extinction despite natural mortality being supplemented by additional agents of mortality such as fishing or power station impingement. Thus Boreman (2000) implies that the proportional reduction in abundance of juveniles from a particular cohort due to power station mortality is greater than the proportional decrease in abundance of adults originating from that year class. Compensatory mechanisms, also known as direct density-dependent mechanisms, limit population growth at high abundance or increase numerical growth at low abundance. Rose and Cowan (2000) suggest slower growth, lower survival and increased emigration as examples of the former, with more rapid growth, better survival and increased immigration as agents of increased population growth rates. Attempts to quantify compensatory responses of fish populations to anthropogenic mortality have often relied on stock-recruitment models, exemplified by the classic Ricker (1954) curve (Figure 5.1). The "surplus" of recruits that theoretically exists in such a relationship means that the stock can be thinned from equilibrium level (where number of recruits matches abundance of stock, *i.e.* parent:progeny ratio is 1:1) to a lower level, and this may actually increase the abundance of recruits that exist (Figure 5.1).

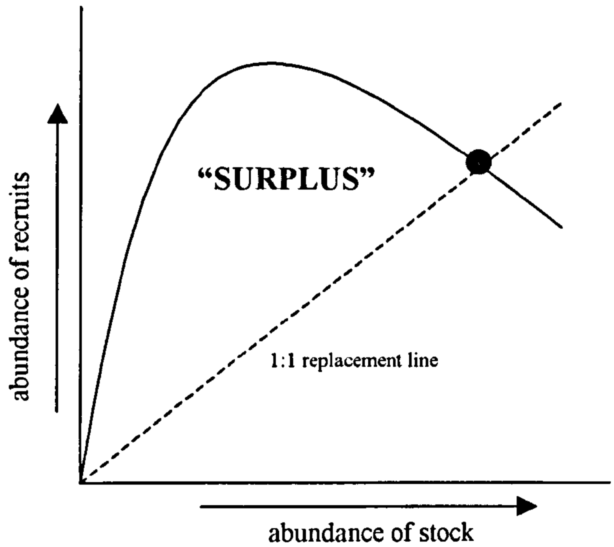


Figure 5.1. Ricker (1954) curve of theoretical relationship between stock size and number of recruits into the stock. Equilibrium size of the stock is where number of parents = number of recruits (denoted by ●). After Boreman (2000).

The EAV approach used in Chapter 4 does not take into account the possible compensatory mechanisms that may be operating in the fish populations analysed. The method is density-independent, *i.e.* no allowance is made for compensation that may occur through reduction in resource constraints with thinning of the population (Turnpenny and Taylor, in press). Instead, just a single value of loss is generated. The EAV method has a further disadvantage: no account is taken of the contribution the impinged animals may have made to the ecosystem had they died from natural causes and remained within the water body. Thus although only approx. one in 400 35-day old whiting might be expected to survive to maturity at age 2 based on EAV calculations (Turnpenny, 1989), 399 fish would become food for predators, scavengers or decomposers. Boreman (2000) highlights the fact that removal of 'surplus' by power plant operation will result in another component of the food web being out-competed for that resource, and that the term 'surplus' is rather a misnomer since there will be no wastage of the 'extra' recruits entering the system. The removal of fish from the system due to power station mortality is thus likely to have a general impact on the ecosystem, not just individual fish populations. This is a topic of great interest requiring further study in the context of ecosystem level impacts (see section 5.4.5).

5.1.2. Use of short term data

The populations of each fish species in the Forth are reduced in abundance by differing proportions due to the abstraction of CW at LPS. Had estimates of benthic and demersal species' absolute abundances been calculated solely by extrapolation of Agassiz trawl data obtained in 1999 – 2000, > 100% of fish assessed to have been present in the estuary would have been estimated to have been impinged at LPS in several cases (Table 5.1). These simple calculations are based on the assumption of fish being distributed uniformly between two arbitrary points within the estuary when, as discussed in section 3.4.3, there is likely to be tidally facilitated movement between the estuary and inshore areas of the Firth of Forth. Assessment of short term absolute abundance can therefore be seen to be prone to substantial uncertainty, and so an assessment of long-term relative abundance was undertaken.

Table 5.1. Estimated total abundance of benthic and demersal fish in the Forth Estuary (extrapolated values based on mean area of 5316 ha between the Forth Bridges and Dunmore; see Figure 1.1) and percentage of these values estimated to be lost to power station impingement. Trawl abundances shown are arithmetic annual mean of monthly summation of 6 trawls ± 95% CI, impingement abundances as in Table 2.4). Trawl efficiency of 33% assumed.

	mean annual trawl-calculated abundance		**total estimated impingement at LPS**		**LPS impingement as % of trawl abundance**	
	1999	**2000**	**1999**	**2000**	**1999**	**2000**
whiting	$1.84 \pm 2.05 \times 10^5$	$2.14 \pm 1.37 \times 10^5$	1.21×10^6 ($7.92 \times 10^5 - 2.02 \times 10^6$)	2.22×10^6 ($1.41 - 3.54 \times 10^6$)	**658%**	**1037%**
cod	$2.18 \pm 3.40 \times 10^5$	$7.23 \pm 8.56 \times 10^5$	1.05×10^5 ($6.48 \times 10^4 - 2.13 \times 10^5$)	2.62×10^5 ($1.80 - 4.37 \times 10^5$)	**48.2%**	**36.2%**
plaice	$3.13 \pm 2.18 \times 10^5$	$9.40 \pm 9.16 \times 10^5$	3.08×10^5 ($1.93 - 5.69 \times 10^5$)	1.19×10^6 ($7.04 \times 10^5 - 2.20 \times 10^6$)	**98.4%**	**126.6%**
flounder	$2.95 \pm 3.30 \times 10^5$	$3.43 \pm 2.52 \times 10^5$	2.73×10^5 ($1.99 - 5.29 \times 10^5$)	1.74×10^6 ($1.23 - 2.56 \times 10^6$)	**92.5%**	**507%**
pogge	$3.61 \pm 4.70 \times 10^5$	$4.09 \pm 4.08 \times 10^5$	5.36×10^4 ($1.61 \times 10^4 - 2.45 \times 10^5$)	1.02×10^5 ($4.40 \times 10^4 - 2.96 \times 10^5$)	**14.8%**	**24.9%**
fatherlasher	$6.64 \pm 6.98 \times 10^4$	$3.17 \pm 1.46 \times 10^5$	2.22×10^2 ($2 - 2.91 \times 10^4$)	7.15×10^2 ($5 - 1.09 \times 10^5$)	**0.33%**	**0.23%**
gobies	$1.11 \pm 2.05 \times 10^4$	$5.53 \pm 5.37 \times 10^4$	4.88×10^5 ($2.83 \times 10^5 - 1.02 \times 10^6$)	1.02×10^6 ($5.47 \times 10^5 - 3.07 \times 10^6$)	**4396%**	**1844%**
eelpout	$1.03 \times 10^5 \pm 8.19 \times 10^4$	$1.70 \times 10^5 \pm 7.13 \times 10^4$	4.81×10^2 ($3 - 7.99 \times 10^4$)	1.14×10^3 ($24 - 1.01 \times 10^5$)	**0.47%**	**0.67%**
sea snail	$3.69 \pm 4.58 \times 10^4$	$6.27 \times 10^4 \pm 1.01 \times 10^5$	4.52×10^4 ($2.11 \times 10^4 - 1.42 \times 10^5$)	2.88×10^4 ($6.72 \times 10^3 - 1.42 \times 10^5$)	**81.6%**	**45.9%**

5.1.3 Use of long term data

The long term relative abundance study assumed that the Agassiz trawling method did not change significantly from year to year, and that the efficiency of the gear for catching each species was constant. Thus for species x the assumption was that a constant y % of the individuals encountering the trawl would be captured.

How can the long-term Agassiz trawl data set discussed in Chapter 3 be used to evaluate potential impacts of LPS on the fish populations of the Forth? Dey *et al.* (2000) suggest that operating power plants would be deemed to have had no adverse impact on fish populations if there are no long-term declines in fish abundance. The mid-lower Forth Estuary dataset suggested that declines had occurred in two out of ten assessed species (Chapter 3), but that there was no long-term trend in abundance of the remaining eight species. The decline in eelpout abundance seems likely to have been caused by warming of the climate initiating a northward shift in the species' distribution, while there is no clear reason for decreases in whiting abundance (section 3.4.2). Assuming that the trawl data accurately reflect relative abundance, the populations of cod, plaice, sole, gobies, flounder, sea snail, pogge, fatherlasher in the Forth Estuary appear to have remained stable from 1982 – 2000 (Figure 3.4). This lack of decline could be taken as indicating no impact of CW abstraction on the fish populations in the Forth. But, as previously noted, LPS commenced operations ten years before the Agassiz dataset used in Chapter 3 began, and can be seen to have been extracting substantial quantities of CW throughout its operational existence (with the exception of the 1984 – 85, the year of the UK coal miners' strike) (Figure 5.2), assuming that the total quantity of electricity generated may be taken as a reasonable proxy for the quantity of CW abstracted.

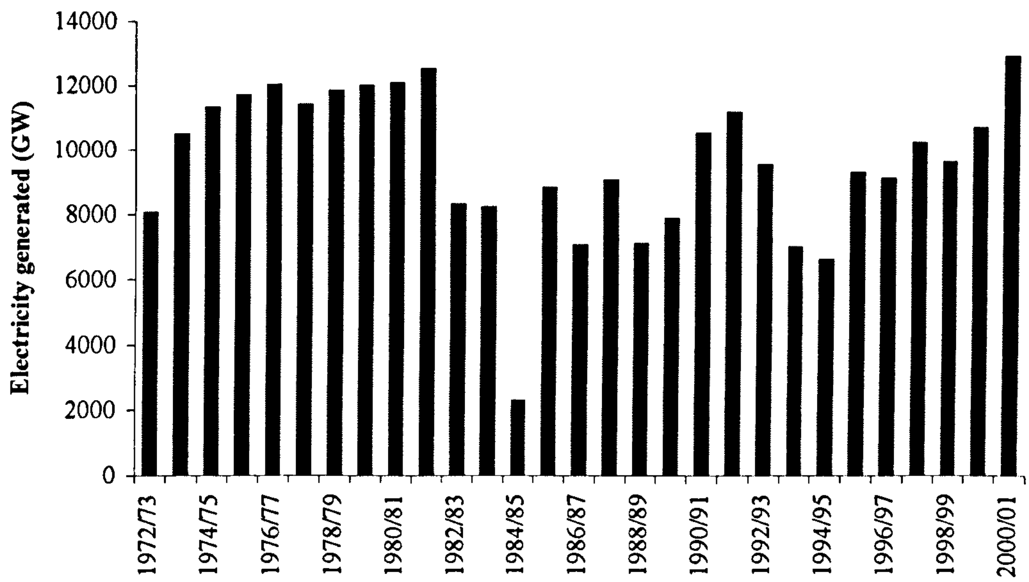


Figure 5.2. Electricity generation by LPS, April 1972 - March 2001. Totals are shown for financial years (April - March). Data from D. Robertson, ScottishPower plc. (personal communication).

Thus the Agassiz trawl dataset is of fish populations that had already experienced a decade of impingement mortality prior to commencement of the 19-year time series. It is impossible to know what equilibrium densities prior to LPS operation may have been without field data having been collected in the same way. The issue is complicated by the Forth Estuary having undergone various other changes, including closure of KPS, substantially decreased effluent inputs, and possible climatic change. The potential effects of LPS on the fish populations of the Forth Estuary may be examined theoretically using a simple model of the interaction of density-dependent (DD) and density-independent (DI) mortality given by Sinclair (1989) (Figure 5.3).

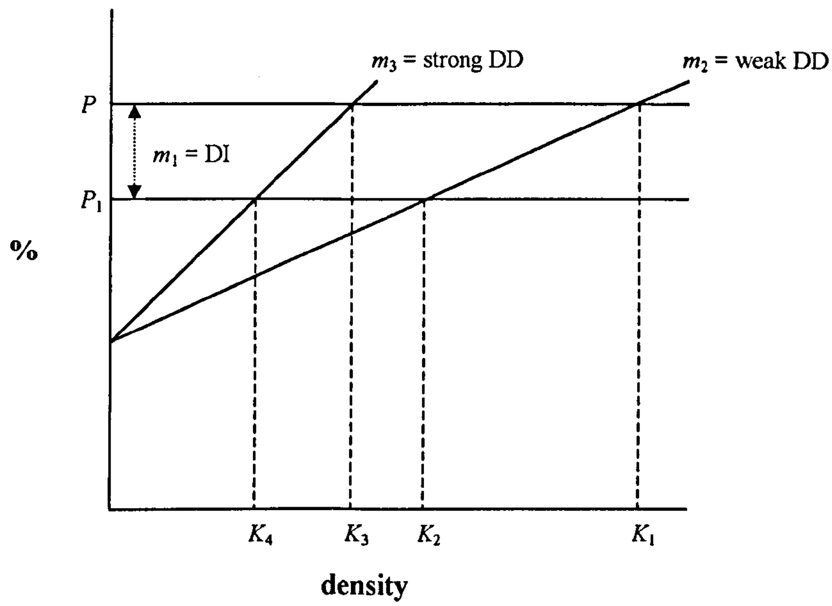


Figure 5.3. Model of interaction of density-dependent (DD) and density-independent (DI) processes. % represents the percentage of population size that is represented by production (P) and loss due to DI mortality (m_1) and DD mortality (m_2 and m_3). Equilibrium population densities are K_1 – K_4. After Sinclair (1989).

The model assumes that production (*P*, the input of births and immigration to the population) is held constant at all population densities. A DI mortality, m_1, would produce a uniform proportional decrease in population size across all densities of population, resulting in a new level of production, P_1. Mortality induced by a power station is of this type, for it is insensitive to population densities (Boreman, 2000). The resultant change in equilibrium density as a result of the DI mortality depends very much on the strength of any DD mortality operating on the population. With relatively weak DD mortality (m_2), there is a relatively large change in equilibrium density from K_1 to K_2. The interaction of a relatively strong DD mortality, m_3, with DI mortality m_1 gives a proportionally smaller decrease in equilibrium density from K_3 to K_4. Though very much a simplification, the Sinclair (1989) model produces an important conclusion: equilibrium densities are determined by **both** DD and DI factors. It may be that fish populations in the mid-lower Forth Estuary, having experienced DI mortality through power station impingement for almost 30 years, may be at equilibrium sizes below those that would be predicted were mortalities caused by CW impingement not occurring. This would be dependent on the extent to which DD factors operate to regulate the abundance of Forth fish populations.

The much-studied ichthyofauna of the Severn Estuary and Bristol Channel may offer the first indication of impacts on fish populations caused by direct-cooled power stations. The region appears to have become a more favourable environment for fish in recent years, with total abundance increasing by 3 – 4 × (Henderson and Seaby, 2001; Potter *et al.*, 2001). The latter authors attributed the increase in abundance to improved reproductive success caused by lessening of sublethal effects due to reductions in toxic metal emissions. Metal refining in the Severn had previously led to the estuary possessing concentrations of the trace metals cadmium, copper, lead, zinc and nickel several times higher than in the Forth (Balls *et al.*, 1997). Increasing temperatures were assumed to be responsible for increased species richness due to greater prevalence of warmer water species (Henderson and Seaby, 2001). An additional improvement in the Severn environment may be the reduction in CW abstraction through closure of several power stations. Seven direct-cooled power stations are or were operational in this area, including the world's first full-scale commercial nuclear power station at Berkeley (opened in 1962; Langford, 1983). Of potential significance to the Severn ecosystem are closures of four of these power stations since 1989 (Table 5.2). The total estimated annual impingement of fish > 3 cm SL from the region is likely to have been approximately halved to 3.44×10^6 individuals (Henderson and Seaby, 2001).

Table 5.2. Direct-cooled power stations on the Bristol Channel/Severn Estuary. Data from Henderson and Seaby (2001).

	operational status	annual impingement mortality (fish > 3cm SL) pre-1989	current annual impingement mortality (fish > 3 cm SL)
Hinkley Point A	closed (2000)	1.3×10^6	-
Hinkley Point B	working	9.9×10^5	9.9×10^5
Oldbury	working	2.5×10^5	2.5×10^5
Berkeley	closed (1989)	2.5×10^5	-
Uskmouth	closed (1995)	2.9×10^5	-
Aberthaw B	working	2.2×10^6	2.2×10^6
Pembroke	closed (1997)	1.6×10^6	-
total		6.9×10^6	3.4×10^6

Preliminary evidence of increased abundance in fish and other taxa that may partially be due to power station closures exists for several species, though results must be treated with caution until statistical analyses can be undertaken; to this end a sufficiently long period of time from the closure of Hinkley Point A in 2000 is required, *i.e.* in 2002 – 2003. Changes in climate and the water quality of the estuary may also have influenced conditions, though the latter was probably already contributing to increasing fish abundances in the 1970s (Potter *et al.*, 2001). Long term monitoring of fish impingement at Hinkley Point B station, *loc. cit.*, has shown increases in common shrimp, common prawn (*Palaemon serratus*), sprat, whiting, flounder and sand goby abundances since the initiation of power station closures (Henderson and Seaby, 2001). In most cases the populations exhibited great stability in number over the first decade of monitoring, from 1981 – 1990; the extreme example is of common prawn, with stable abundances until 1998, following which an almost exponential increase has occurred (Henderson and Seaby, 2001). The authors suggest that a decrease in total fish impingement of > 3 × 10^6 individuals per annum would provide detectable abundance changes in a total population numbering 10^8 – 10^9 individuals, providing sufficient sampling is undertaken. In conclusion, Henderson and Seaby (2001, p.11) state:

"...the SEDS [Severn Estuary Data Set] will offer over the coming 2 years the best opportunity available in the world to test for the impact of direct-cooled power stations."

The implications of this future research for the situation observed at LPS may be significant. Both the Severn and the Forth estuaries have undergone improvements in water quality, and both are experiencing increasing water temperatures. Some species are decreasing in abundance at each location as a result of the latter phenomenon (sea snail in the Severn, eelpout in the Forth), whether as a result of movement offshore or increased mortality. The overall trend in species abundance differs, however: a substantial increase in numbers has occurred in the Severn, whereas a significant decrease has occurred in the Forth. Part of the increase in abundance in the Severn is attributable to increases in abundance of sprat and herring; these pelagic species were not included in the long-term analysis of Forth Agassiz trawl data. Pelagic species cannot be said to account for all of the increase in abundance, for major increases in whiting have also occurred in the Severn. This has not occurred in the Forth, with a significant decrease being noted (Chapter 3). Changes in climate would, if anything, have tended to decrease whiting abundance more in the Severn than the Forth since the latitude of the Severn is nearer to the southern limit of the species' range (see Table 3.7). The increases in abundance of sand goby and flounder at Hinkley Point in the 1990s (though not at Oldbury in the mid-lower Severn estuary in the case of flounder; Potter *et al.*, 2001) were not exhibited by these species in the Forth (lack of any trend over 19 years of Agassiz trawling was noted in Chapter 3). These observations may indicate that operation of LPS, generally stable since 1984 (Figure 5.2), may indeed be significantly limiting fish abundance in the Forth Estuary. If the seven power stations operating throughout the 1980s contributed to lower equilibrium densities of fish species in the Severn region through impingement of approximately 6.9×10^6 fish annually, it could reasonably be argued that the possibility of LPS exhibiting a similar effect in the Forth is considerable. This is for two reasons. First, annual impingement of fish at LPS is perhaps three times greater than total impingement of fish was estimated to be at the seven Severn stations (Table 5.3). Second, the Forth Estuary is between 6 – 7 × smaller in size than the Severn (Table 5.3). Intuitively one would assume that the total abundance of fish within the Forth estuary would be proportionally smaller. Thus a greater magnitude of impingement loss in a smaller estuary would be expected to produce significant reductions in the ichthyofaunal equilibrium densities of the Forth, should such a case be proven in the Severn.

Table 5.3. Comparison of size and fish impingement by direct-cooled power stations in the Forth and Severn estuaries. Severn impingement data refer to total impingement estimated to have occurred prior to initiation of power station closures (see text). Area data from Buck (1993) (Forth) and Davidson *et al.* (1991) (Severn); impingement data from this study (Forth) and Henderson and Seaby (2001) (Severn).

	total area (ha)	**estimated total annual impingement at power stations**
Forth Estuary	8401	2.19×10^7
Severn Estuary	55700	6.44×10^6

5.2. Potential for mitigation of impingement losses

5.2.1. Applying the precautionary principle to mitigation of fish impingement at LPS

It may thus be prudent to consider a precautionary approach to mitigation of impingement losses at LPS. The 'Precautionary Principle' is open to a variety of interpretations (see O'Riordan and Cameron, 1994), but a reasonably concise and relevant definition is given in the proposed changes to the US Magnuson-Stevens Fishery Conservation and Management Act, known as the Fisheries Recovery Act (MFCN, 2001):

"The term "precautionary approach" means –

(A) exercising additional caution in favor of conservation in any case in which information is absent, uncertain, unreliable, or inadequate as to the effects of any existing or proposed action on fish, essential fish habitat, other marine species, and the marine ecosystem in which the fishery occurs".

Obviously this statement is contained in legislation proposed in relation to conservation of fisheries, but the elements are relevant in the case of LPS, since it remains uncertain to what extent CW abstraction in the Forth impacts the ichthyofauna at a population level. Given the extent of impingement losses in purely numerical terms, a reasonable precautionary approach would err on the side of investigating mitigation of impingement that would not entail excessive cost. Strict application of the precautionary principle at LPS would involve cessation of generation, obviously an inviable option entailing considerable economic and societal costs (Figure 5.4).

Henderson and Seaby (2000) argue that best available technology (BAT) to reduce impingement of fish is installation of closed cycle cooling, but this is the most costly mitigation option (Figure 5.4). Section 5.2 outlines potential methods to mitigate fish removal in CW abstracted at LPS.

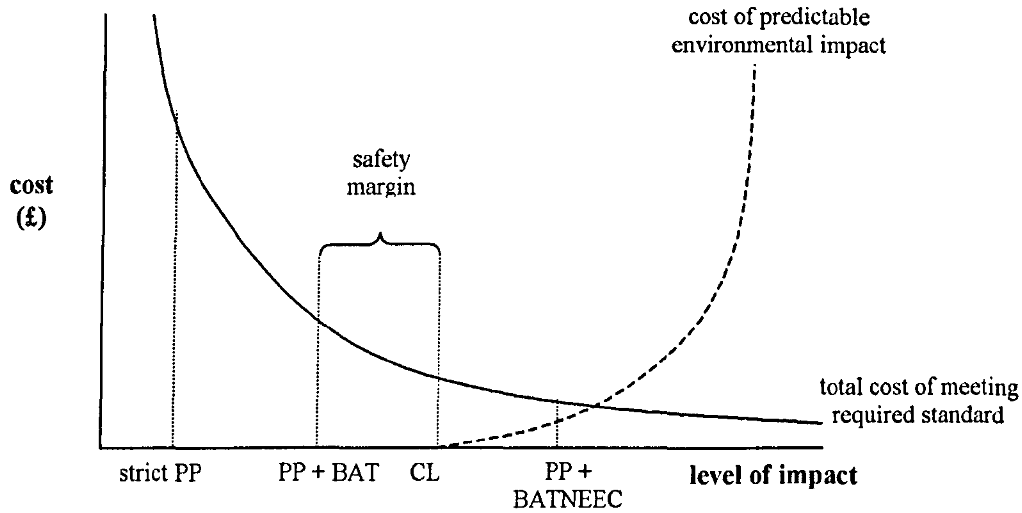


Figure 5.4. Theoretical representation of costs of ameliorating environmental impact through differing levels of application of the precautionary principle. PP = precautionary principle, BAT = best available technology (to ameliorate effects of impact), CL = critical load, BATNEEC = best available technology not entailing excessive cost. After Pearce (1994).

5.2.2. Determination of expenditure on mitigation techniques

Determination of an appropriate level of expenditure on technologies to mitigate fish impingement mortalities is not a simple procedure. Bailey *et al.* (2000) formulated a framework based upon balancing of costs and benefits to the environment and fisheries. They suggested that "The maximum value of an intake technology is the value of the loss [of fish and/or other species]. The value [of the technology] can never be more than the initial loss even if the technology cost virtually nothing and was 100% effective". If applied to the situation observed at LPS, two obvious problems arise: should the value of a 'loss' be calculated as the actual loss of commercial species that are above MCLS (section 4.3.1.) or as the potential loss of equivalent adults through actual loss of juveniles (section 4.3.2.), since mostly juveniles are impinged? Based on mean figures for 1999 – 2000, this would represent an expenditure of £1,644 in the former case, compared with £247,000 in the latter. Also, how to evaluate the loss of species that are not commercially important and so cannot be evaluated in monetary terms?

In an effort to overcome these difficulties, the State of Maryland, USA, promulgated environmental laws in 1975 in the Code of Maryland Regulations (COMAR) that introduced a simple cost-benefit analysis to assess the monetary level of mitigation required to be installed if impingement occurred due to cooling water intake structures. Each fish impinged was assigned a length-based dollar value (Table 5.4), the sum of which then had to be multiplied by a weighting factor depending on whether, for example, they were classified as commercial species (weighting of 1.0) or forage species (weighting of 0.75).

Table 5.4. Examples of dollar values assigned to fish species to estimate monetary value of species lost to impingement at power stations. Forage fish defined as "All fishes that are not listed elsewhere in this table and that are used as food by predatory fishes". Excerpted from section 08.02.09.01 of the Code of Maryland Regulations (W.A. Richkus, Versar Inc., personal communication).

(a) Saltwater (Ocean) Food and Sport Fishes

Species	**Under 4"**	**4" – 6"**	**6" – 8"**	**8" – 10"**	**10" – 12"**	**Price per Pound over 12"**
Cod *Gadus sp.*	.10	.20	.30	.40	.50	.50
Herring, Sea *Clupea sp.*	.01	.05	.09	.12	.15	.20

(b) Bait and forage fishes

	Under 4 inches	**Under 4 inches**
Forage fish: includes minnows, shiners, daces, chubs, silversides, anchovies, blennies, sculpins, gobies.	$1 per thousand	$2 per thousand

The regulation contained in COMAR 26.08.03.05.D(1) required that, based on the dollar value of fish lost to impingement, plant operators should install and implement mitigation technologies not exceeding the total value of organisms lost in a five year period, or five times the value of organisms lost in a single year. The Maryland regulations thus enabled monetary valuations of all species to be obtained. The regulations exist in the same format to the present day, as evaluations were carried out in the 1970s and early 1980s and so updating of the dollar values to account for factors such as inflation was unnecessary (W.A. Richkus, Versar Inc., personal communication). Were one to apply the values listed in Table 5.4 to cod, herring, and gobies impinged at LPS in 1999 – 2000, the total 1975 dollar value in the 2 year period would amount to approximately $388,595, suggesting a mitigation expenditure of just over $970,000. This crude analysis obviously ignores all other species impinged at LPS, for most of the species listed are not found in the eastern Atlantic, and the determination of worth would be likely to differ considerably if attempted for British species. Nevertheless, the analysis illustrates the extent to which the Maryland authorities regarded protection of Chesapeake Bay aquatic resources to be an important environmental issue.

5.2.3. Techniques for mitigation of fish impingement mortality

A variety of options of differing feasibilities exists to mitigate fish mortality caused by impingement at direct-cooled power stations (Table 5.5). These techniques are site-specific and cannot universally be applied to all power stations. Any mitigating device located in a water body is susceptible to biofouling and damage by large debris. The nature of some mitigation techniques makes them inapplicable to cooling water intake systems of existing facilities. A fish return system was incorporated in the design of the Sizewell B nuclear power station, for example, while the existing Sizewell A station could not have such a device successfully retrofitted (Turnpenny and Taylor, in press).

Table 5.5. Summary of selected fish impingement mortality mitigation techniques.

Method of mitigation	Main advantages	Main disadvantages
Fish collection and return systems	Return fish to water body following impingement	Difficult to retrofit to existing facilities; differential survival of species (Turnpenny and Taylor, in press)
Fish diversion systems, *e.g.* angled screens with pumped return to water body	Reduction in likelihood of impingement; return of fish to the water body	High cost; differential survival of species following return to water body (Taft, 2000); susceptibility to blockage by debris (Langford, 1983)
Physical barriers, *e.g.* barrier nets	Likelihood of fish entering immediate vicinity of CWIS is reduced	Unsuitable for areas of considerable tidal flows and high debris loading
Behavioural barriers, *e.g.* light or acoustic	Fish are deterred from entering the region of CW abstraction	Interspecific differences in response, some actually being attracted to intake (Taft, 2000)
Closed cycle cooling (cooling towers)	Possibly the best means to reduce losses of organisms through impingement and entrainment (Henderson and Seaby, 2000)	High cost of construction and maintenance (Bergen, 1988); increased emissions of greenhouse gases (Veil, 2000); size of structure (area occupied/visual pollution) (Barnthouse *et al.*, 1988b)
Stock enhancement	Augments abundance of stocked species (McLaren *et al.*, 1988)	Not feasible for all species due to inability to rear in captivity; disruption of genetic composition of wild stocks
Plant outages/flow reductions	Can be synchronised with times of peak abundance, *e.g.* of juvenile stages (Englert *et al.*, 1988)	Reduction of generation potential; periods of peak abundance differ between species
Reduction in intake velocity of CW	Increases likelihood of escape from intake current	Only relevant to species with recognised sustainable swimming capabilities, less effective in turbid waters (Turnpenny, 1988a); high cost and difficulty in redesigning existing intakes (Langford, 1983)

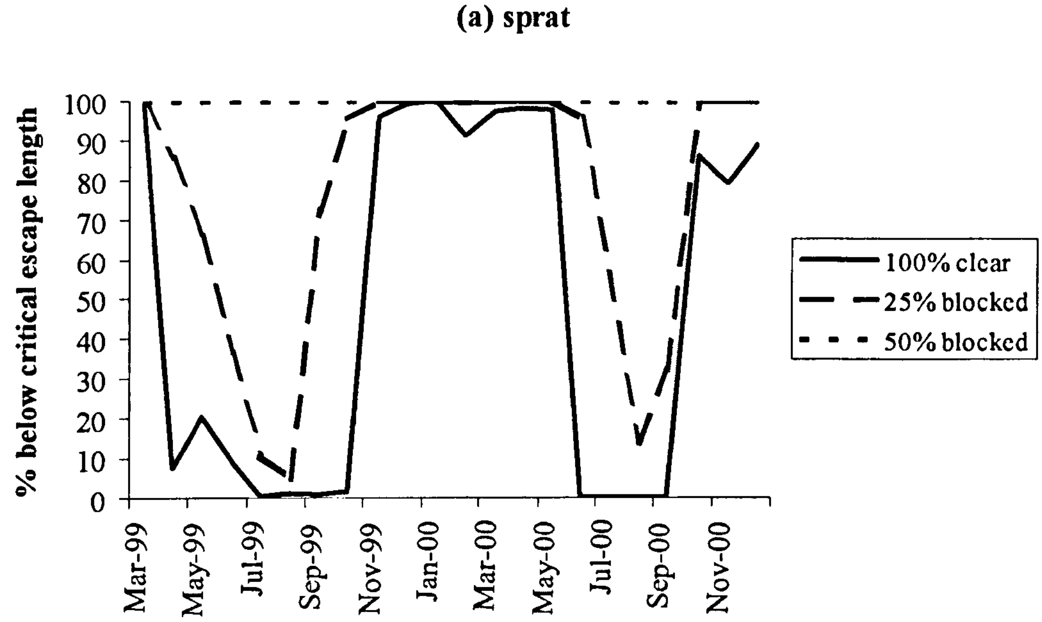
Whilst widening of the LPS intake to reduce intake velocity is likely to be unfeasible both from economic and engineering standpoints, the significance of intake velocity is still of relevance. Average intake velocity at the surface of the coarse screens was calculated as being 57.7 cms^{-1} with 4 CW pumps operational (section 2.2.1) and 43.3 cms^{-1} with 3 pumps working. Ability of fish to escape from water intakes is dependent, amongst other factors, on the approach velocity of the intake (*i.e.* velocity of the water entering the intake), the size and species of the fish, and the temperature of the water. The critical escape length was modelled by Turnpenny (1988a) as:

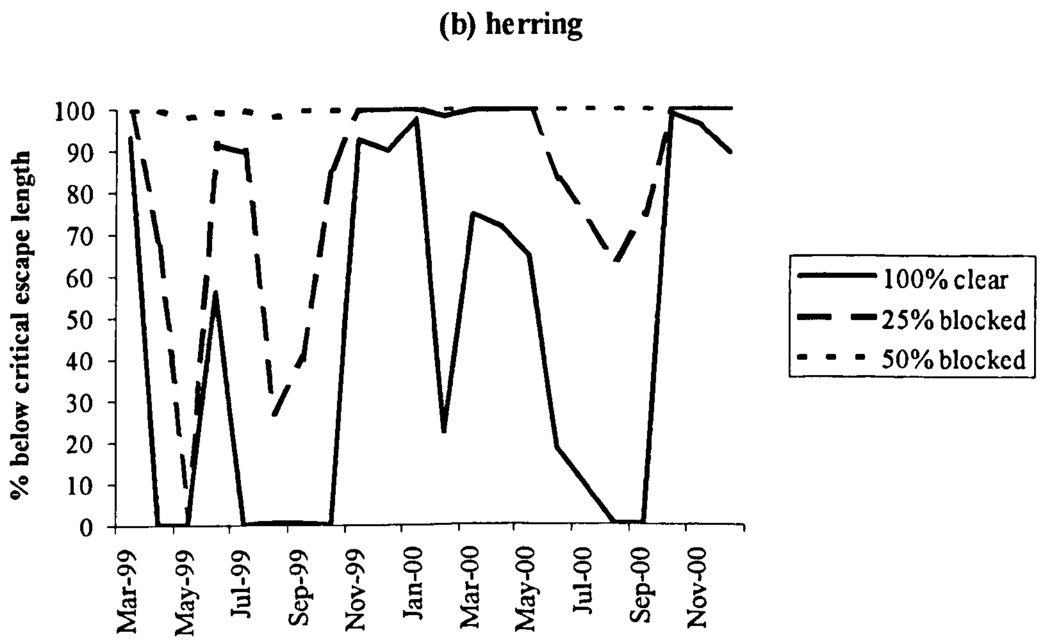
$$L_{crit} = [(V_a/(a = k_t.T)]^{1/0.6}$$

where L_{crit} = critical escape length, V_a = approach velocity, a & k_t are species-specific constants, T = temperature (°C). The critical escape lengths of four commonly impinged species at LPS were calculated based on the intake velocity dependent on whether three or four CW pumps were operational and mean water temperature recorded each month at LPS, assuming three conditions of intake debris blockage that would have influenced intake velocity to differing degrees: 100% clear intake surface area, 25% blocked (*i.e.* a 25% decrease in surface area, with corresponding increase in intake velocity) and 50% blocked. The rationale for the blockage percentages was based on personal observations of the coarse screens. For each month of sampling at LPS between January 1999 and December 2000 (excluding July 2000, as no sampling took place, see above), the proportion of fish impinged that were below theoretical critical escape lengths was calculated.

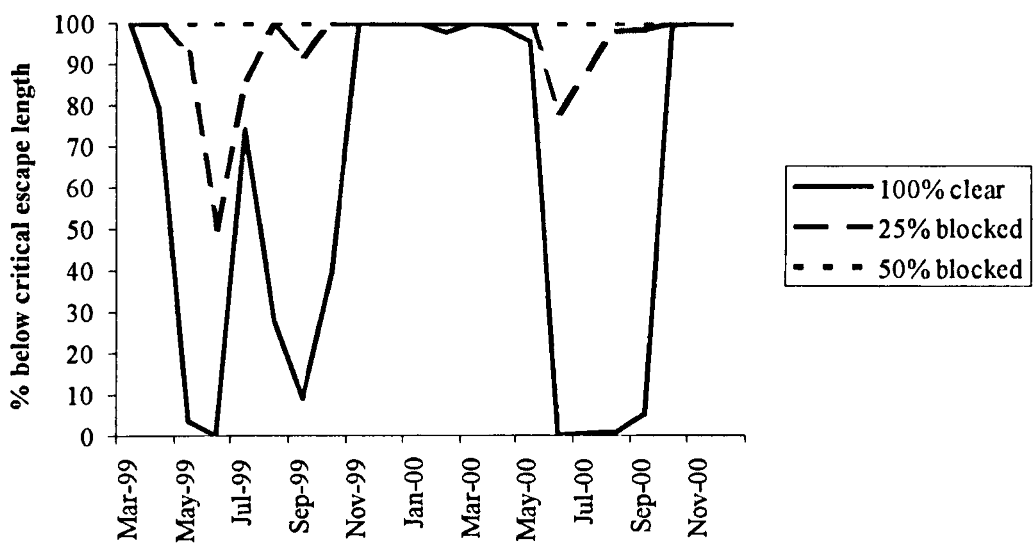
For all species it was noted that the percentage of fish below L_{crit} at LPS increased in the colder months of the year, as expected with the corresponding decrease in water temperature (Figure 5.5). When the intake surface area was halved, *i.e.* at 50% blockage, only a small percentage (< 0.5%) of herring were above L_{crit} and so capable of escaping the intake flow (Figure 5.5b), whereas the size classes of all other three species impinged at LPS were below the L_{crit} required to escape no matter the water temperature (Figure 5.5a,c,d).

Figure 5.5. Proportion of fish below critical escape length, LPS, January 1999 – December 2000: (a) sprat; (b) herring; (c) whiting; (d) plaice. 100% clear = 57.7 or 43.3 cms^{-1} (——); 25% blocked = 76.1 or 57.7 cms^{-1} (— —); 50% blocked = 115.0 or 86.6 cms^{-1} (- - -).





(c) whiting



(d) plaice

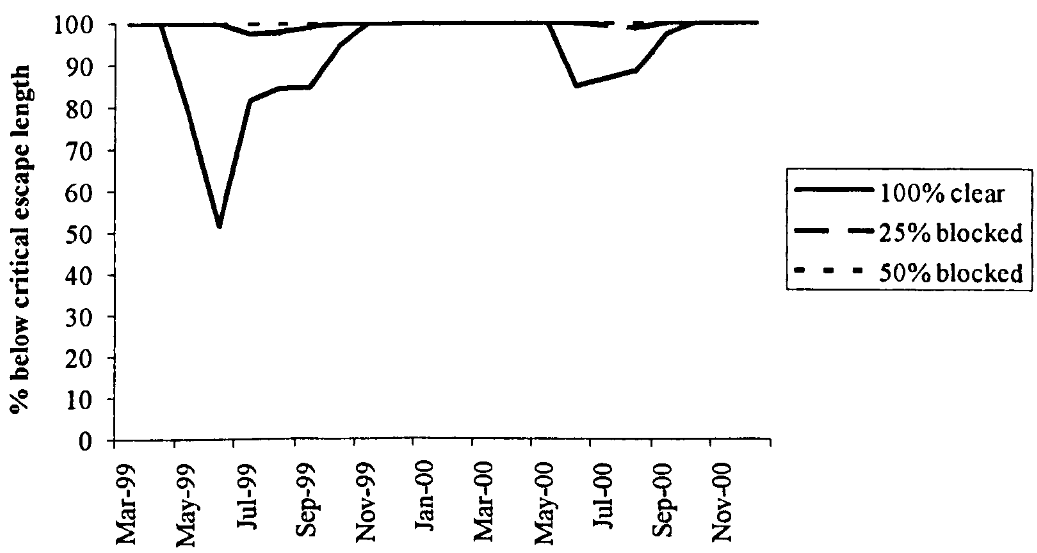


Figure 5.5. cont.

This simple analysis suggested that blockage of the LPS intake screens could theoretically be of some considerable importance in determining the proportion of fish removed in the CW flow, and that some potential exists for reducing losses by increased frequency of coarse screen debris removal. This would be likely to be less effective during the colder months when even a 100% clear intake would possess water velocities greater than the reduced fish swimming ability could overcome. It should be noted, however, that swimming ability alone may not suffice to aid escape from a CW intake, as fish must be able to see the intake in order to orientate away from it and facilitate escape (Turnpenny, 1988a). In turbid waters such as those of the lower Forth Estuary this ability may be reduced and so removal in the CW flow could occur irrespective of magnitude of water velocity in relation to swimming ability.

Acoustic fish deterrent systems (AFDS) have been tested at various locations in Europe, and are advantageous in that fish are discouraged from approaching the immediate vicinity of CW intakes (Table 5.6). The systems have been successfully retrofitted at Hartlepool (Turnpenny *et al.*, 1995) and Doel Nuclear Power Stations (Maes *et al.*, 1999b). AFDS have been shown to significantly reduce likelihood of impingement of the majority of the most abundant species estimated to have been impinged at LPS in the present study (Table 5.6).

Table 5.6. Percentage reductions in impingement of fish species at Hartlepool and Doel Nuclear Power Stations attributable to installation of acoustic fish deterrent systems. Data from Turnpenny *et al.* (1995) and Maes *et al.* (1999b).

species	Hartlepool	Doel
sprat	60.1%	88%
herring	79.6%	95%
whiting	53.5%	infrequent
gobies	infrequent	34%
smelt	infrequent	64%
all species combined	55.9%	51%

Species with swimbladders tended to exhibit greatest reduction in impingement, while species such as flatfish with reduced or absent swimbladders did not generally show significant decreases in abundance. Such species are often able to survive impingement, so that installation of AFDS in combination with a fish return system may diminish the differential protection offered by AFDS. This technique has been implemented at the recently opened Shoreham-on-Sea Combined Cycle Gas Turbine station. As mentioned above, difficulties exist in retrofitting return systems, so this may not be a viable option at LPS, but to what extent might installation of a successfully functioning AFDS reduce the extent of fish impingement at LPS? Assuming the values for the Hartlepool and Doel stations (Table 5.6), the total abundance of fish impinged at LPS could be decreased from the 1999 – 2000 mean of 2.19×10^7 fish to 9.66×10^6 – 1.07×10^7 per annum. The reduction in mass of sprat, herring and whiting impinged at Hartlepool with the AFDS operational was 69% (Turnpenny *et al.*, 1995). Such a decrease at LPS in 1999 – 2000 would have potentially reduced the mass of these three most commonly impinged species from 182.1 t to 56.5 t. Similar AFDS performance at LPS as at Hartlepool and Doel in reducing impingement of juvenile whiting, cod and herring would have the potential to reduce monetary losses of equivalent adults of these species from the mean annual 1999 – 2000 value of approx £217,000 to £57,000 - £69,000 (see section 4.3.2), assuming that reduction in cod impinged abundance was of the order of 54.7%, as suggested for 'other swimbladder species' by Turnpenny *et al.* (1995). Acoustic deterrence of larger clupeids and gadoids from the vicinity of the LPS CW intake could theoretically facilitate decreased removal of the parasitic phase of estuarine-inhabiting adolescent river lamprey (section 4.4.3), if these creatures associated with their intended prey and followed their avoidance responses closely. This would not be likely in the case of non-feeding upstream migrating adults or downstream migrating juveniles. The potential for acoustic deterrents reducing impingement of salmonid smolts was shown by brown trout (*Salmo trutta* L.) exhibiting steady movements away from a sound stimuli in laboratory experiments (Turnpenny *et al.*, 1993). This was probably due to the fish responding to vibrations near to the sound source, for the poor connectivity between the inner ear and swimbladder means that true hearing is relatively insensitive (Turnpenny *et al.*, 1998). There was no statistically significant response from eels in acoustic deterrence experiments (Turnpenny *et al.*, 1993), as confirmed by Maes *et al.* (1999b) during field trials at Doel, though in the field study a statistically insignificant reduction of some 65% was noted.

5.3. The suitability of impingement collections as a sampling device for fish of the Forth Estuary

5.3.1. Use of CW intakes to sample fish

Since power stations are generally constantly operating, and therefore constantly requiring to withdraw water, they are a very useful device to sample aquatic organisms with. Knowledge of pump capacity combined with duration spent sampling impinged and/or entrained materials allows standardisation of organisms sampled to unit volume, as undertaken in the present study (Chapter 2). Power station cooling water intakes have proved to be the best means of sampling in areas where the sedimentary or tidal regime diminishes the efficiency of other gears, *e.g.* in the Severn Estuary, where the tidal range may be up to 14.5m and most forms of fishing are not possible (Henderson *et al.*, 1992). The West Thurrock Power Station (now closed) on the Thames Estuary was a useful sampling tool on an estuary that is busy with boat traffic and thus awkward to sample by boat-based methods (Thomas, 1998). The main criticism of power station intakes as a sampling device is the fact that they are fixed points, providing information on only the small area near to the intake (Thomas, 1998). Henderson (1989), however, suggests that even a fixed source such as a power station intake may in fact sample the equivalent of a 20km stretch of coastline, dependent on tidal range and mobility of the species in the water body. Estuaries such as the Severn and the Hudson (NE USA) possess several power stations on them, thus allowing fish migrations to be studied at different times and locations along the water body (*e.g.* Claridge and Potter, 1984; Henderson and Holmes, 1989). Power station intakes have been noted to sample mostly smaller individuals compared with trawl studies, due to the relatively weak swimming ability of small fish compared to larger conspecifics (*e.g.* Thomas, 1998; Maes, 2000).

It has been shown that power station impingement samples may contain three times the abundance of organisms per unit volume compared with simultaneously-taken stow net samples (Maes, 2000). The general conclusion is therefore that using power stations as a sampling method is very efficient, though that there is a bias for smaller fish to be taken as they are more susceptible to the intake flow compared with stronger-swimming large fish.

5.3.2. Comparison of efficiency of LPS intake sampling with trawling

The present study attempted to compare efficiencies of power station intake sampling and trawling by conducting the two simultaneously on several occasions. This was rather unsuccessful due to lack of fish obtained by trawling however, so in lieu of comparison of simultaneously collected samples, a simple analysis of all data collected between January 1999 – December 2000 (April 1999 – January 2001 in the case of pelagic trawling; see section 3.2.2) is included in the present discussion. The number of species obtained by Agassiz and pelagic trawling, 28, was somewhat less than that due to LPS sampling (39), a difference readily attributable to the fact that the latter technique involved sampling an approximate total of 1.14×10^7 m^3 of CW, while trawling with both gears sampled an estimated 7.19×10^5 m^3 of estuarine water. The discrepancy in total sampling effort between LPS intake sampling and trawling reflected the fact that power station sampling was logistically easier to undertake, needing only two workers and not being subject to weather or pressure of work in other areas.

Abundances of pelagic species captured per unit volume by LPS CW intake sampling compared favourably with pelagic trawl hauls for sprat and herring, to approximately the same order of magnitude, while smelt were very much underrepresented in trawl catches (Table 5.7). Simultaneously conducted intake and stow net sampling by Maes (2000) in the Zeeschelde estuary (see below) suggested somewhat different tendencies: the ratio of intake to net abundances were 0.61, 0.62 and 1.09 for sprat, herring and smelt respectively. The only similarity with the Forth data was that the clupeids were sampled to the same order of magnitude per unit volume.

Table 5.7. Mean abundances of selected fish species sampled from LPS intake screens and from pelagic trawling in the lower Forth Estuary. For details of LPS and pelagic trawl sampling see sections 2.2.2 and 3.2.2.1 respectively. LPS data used were from January 1999 – December 2000 (total volume sampled ≈ 1.14×10^7 m^3); pelagic trawl data used were from April 1999 – January 2001 (total volume sampled ≈ 6.66×10^5 m^3).

	abundance per 10^3 m^3 of water sampled at LPS CW intake	abundance per 10^3 m^3 of water sampled by pelagic trawling
sprat	5.32	6.63
herring	4.18	2.61
smelt	1.80	7.66×10^{-5}

To facilitate a meaningful comparison between LPS intake sampling and Agassiz trawling, essentially a 2-dimensional sampling technique where abundance of fish per unit area is relevant, it was necessary to attempt to express abundance of impinged fish sampled per unit area. It was assumed that the velocity across the whole of the CW intake surface was 57.7 cm s^{-1} (see section 2.2.1). Sampling was only ever undertaken at one trash basket at any one time, so it was further assumed that fish obtained during impingement collections had passed through half of the total CW intake width of 51.7 m. Impinged materials were almost always sampled from two CW pumps, so the rate of CW sampling over the period 1999 – 2000 was assumed to have been 45.5 m^3s^{-1} (2 pumps × 22.75 m^3s^{-1} per pump). The total duration of sampling at LPS over the two year period was thus given by the calculation:

$$11399563 \text{ m}^3 / (2 \times 22.75 \text{ m}^3\text{s}^{-1}) = 250540 \text{ s}$$

where 11399563 m^3 represented the total volume of water sampled at the CW intake.

The 'area' therefore represented as having been sampled by the CW intake, as if it had been a stationary 'trawl' with water moving towards it, was therefore:

$$0.577 \text{ ms}^{-1} \times 25.85 \text{ m} \times 250540 \text{ s} = 3.74 \times 10^6 \text{ m}^2.$$

Using this unusual approach, it was clear that the power station intake sampled some benthic and demersal species at similar orders of magnitude compared with Agassiz trawling, *i.e.* whiting and plaice, whereas most species were collected at 1 – 3 × lower orders of magnitude (cod, flounder, pogge, eelpout, sea snail, fatherlasher), and gobies were an order of magnitude more abundant per unit volume in intake samples (Table 5.8).

Table 5.8. Mean abundances per unit area of selected fish species sampled from LPS intake screens and from Agassiz trawling in the lower Forth Estuary. For details of LPS and Agassiz trawl sampling see sections 2.2.2 and 3.2.1.1 respectively. LPS data used were from January 1999 – December 2000 (total 'area' sampled ≈ 3.74×10^6 m^2); Agassiz trawl data used were from the same period (total area sampled ≈ 9.60×10^4 m^2).

	abundance per 10^3 m^2 'area' sampled at LPS CW intake	**abundance per 10^3 m^2 area sampled by Agassiz trawling**
whiting	2.49	1.13
cod	0.28	2.66
plaice	1.15	3.54
flounder	0.44	1.79
pogge	9.74×10^{-2}	2.18
eelpout	8.30×10^{-3}	0.77
gobies	2.16	0.19
sea snail	5.24×10^{-2}	0.28
fatherlasher	5.89×10^{-3}	1.08

All of the species except flounder and sea snail tend to be found at greatest abundance at Port Edgar (section 3.3.1.2); it could therefore be argued that a possible reason for greater prevalence of pogge, eelpout, cod, fatherlasher and plaice in trawl samples compared with intake collections might be the fact that the analysis summed all trawl catches across all stations, thereby enhancing trawl numbers with the greater abundances from Port Edgar. How then to explain the greater abundance of impinged whiting and gobies, when these species were also shown to be present in greatest abundance at the site furthest from LPS? It may be that these two species tend to be present in higher densities nearer to the margins of the estuarine channel in the vicinity of LPS, thus are caught in low densities by mid-channel hauls at the Longannet trawl station. The opposite might be true of flounder and sea snail, both seemingly most abundant at the Longannet station than anywhere else, but being sampled at comparatively low densities. Flounder, in particular, may congregate on the opposite side of the channel to LPS, in order to exploit the large areas of intertidal flat upon inundation at HW.

The study by Maes (2000) mentioned below found that the Doel CW intake sampled all benthic and demersal species more efficiently than stow nets when compared on a volumetric basis, though the differences were usually within the same order of magnitude. Treating the Agassiz trawl as a 3-dimensional sampling device, by multiplying area swept by the minimal height of the trawl opening (0.55m) would have resulted in the LPS intake yielding less fish per unit volume for all benthic and demersal species except gobies (Table 5.9).

Table 5.9. Mean abundances per unit volume of selected fish species sampled from LPS intake screens and by Agassiz trawling in the lower Forth Estuary. For details of LPS and Agassiz trawl sampling see sections 2.2.2 and 3.2.1.1 respectively. LPS data used were from January 1999 – December 2000 (total volume sampled ≈ 1.14×10^7 m^3); Agassiz trawl data used were from the same period (total volume sampled ≈ 5.28×10^4 m^3).

	abundance per 10^3 m^3 sampled at LPS CW intake	**abundance per 10^3 m^3 sampled by Agassiz trawling**
whiting	0.82	2.05
cod	9.24×10^{-2}	4.83
plaice	0.38	6.44
flounder	0.14	3.26
pogge	3.19×10^{-2}	3.96
eelpout	2.76×10^{-3}	1.40
gobies	0.71	0.34
sea snail	1.71×10^{-2}	0.51
fatherlasher	1.93×10^{-3}	1.97

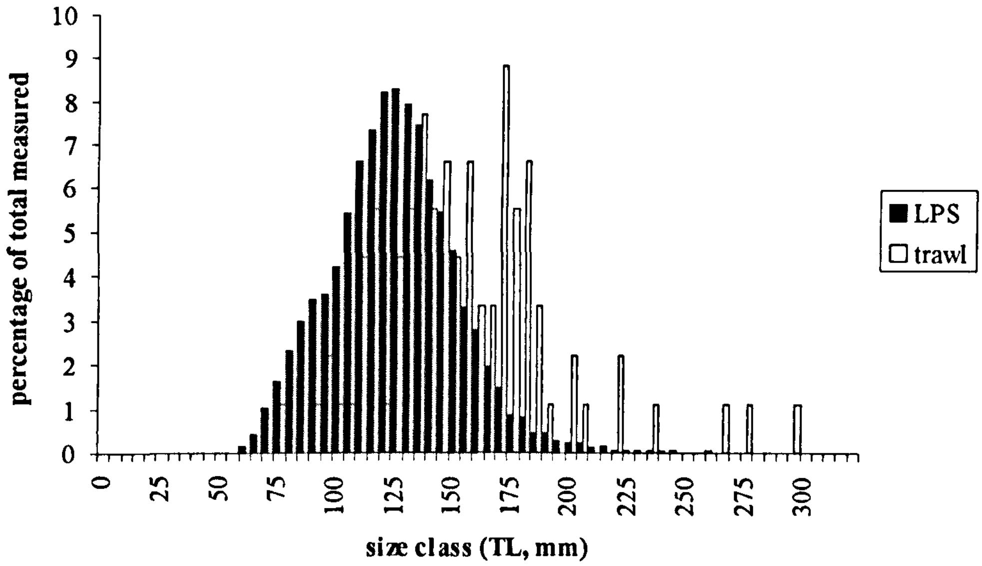
Comparison of the lengths of common fish obtained by LPS intake sampling and trawl sampling reveals a trend that has often been observed at other locations: fish sampled at power station CW intakes tend to be smaller than those taken by other means. This was clearly the case in the Forth Estuary for whiting, cod and plaice (Table 5.10; Figure 5.6a,b,e). The mean lengths of sprat captured by both LPS intake and pelagic trawl were very similar (Table 5.10), though it can be seen that the distribution of length classes sampled by the latter method was somewhat more even than in the case of the former technique (Figure 5.6c). Herring sampled at the LPS intake possessed a similar overall modal length as the fish taken by pelagic trawling (Figure 5.6d), but the greater proportion of fish of longer lengths meant that the mean length of intake fish was smaller than that obtained by trawling (Table 5.10). The difference was, however, somewhat less than in the MJ gadoids and plaice.

Table 5.10. Mean lengths (± SD) of fish sampled from intake screens at LPS and by pelagic or Agassiz trawling in the lower Forth Estuary, 1999 – 2000. Quantity of measured fish is included in parentheses.

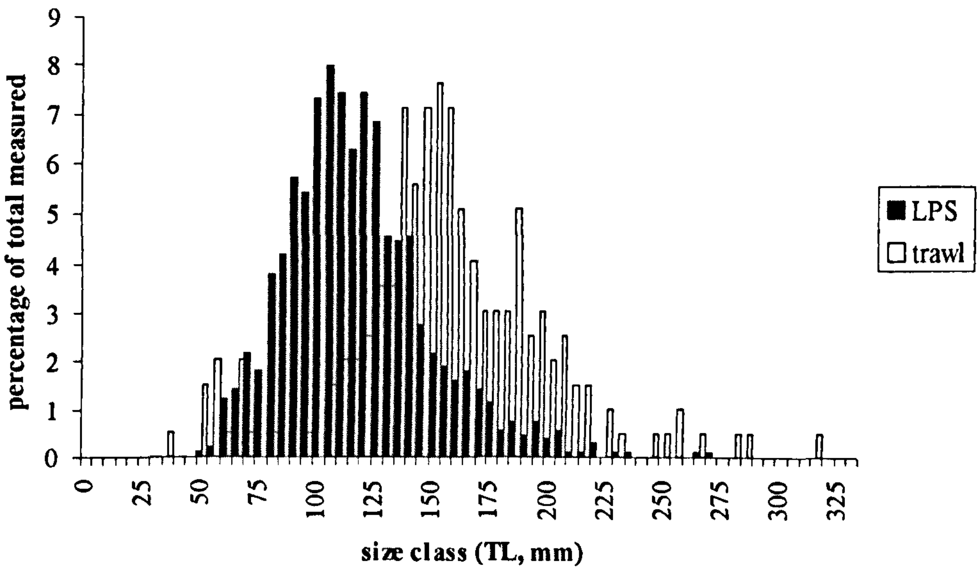
	LPS CW intake	trawls
whiting	122.6 ± 26.4 mm (n = 9194)	152.5 ± 38.4 mm (n = 91)
cod	115.2 ± 30.9 mm (n = 1089)	152.2 ± 44.7 mm (n = 197)
sprat	81.4 ± 16.8 mm (n = 37804)	81.5 ± 24.1 mm (n = 241)
herring	94.3 ± 21.4 mm (n = 25946)	104.1 ± 37.9 mm (n = 232)
plaice	74.2 ± 24.1 mm (n = 4311)	106.3 ± 34.1 mm (n = 275)

Figure 5.6. Comparison of length-frequency distributions of all measured fish sampled from LPS CW intake and trawling in the mid-lower Forth Estuary, 1999 – 2000. Trawl data used to plot graphs were from Agassiz hauls in the cases of whiting, cod and plaice, and from pelagic hauls for sprat and herring.

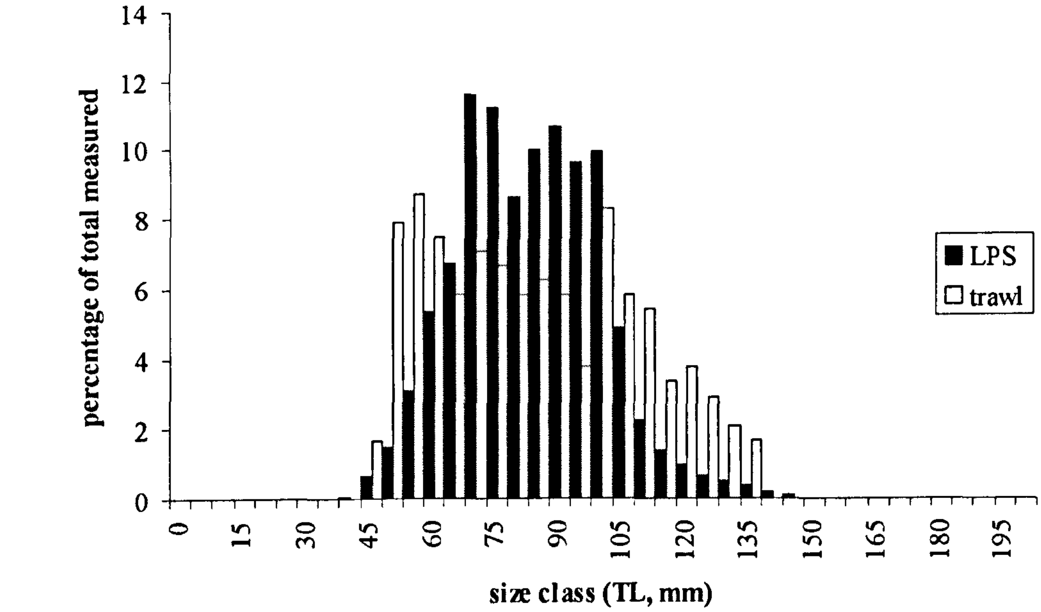
(a) whiting



(b) cod



(c) sprat



(d) herring

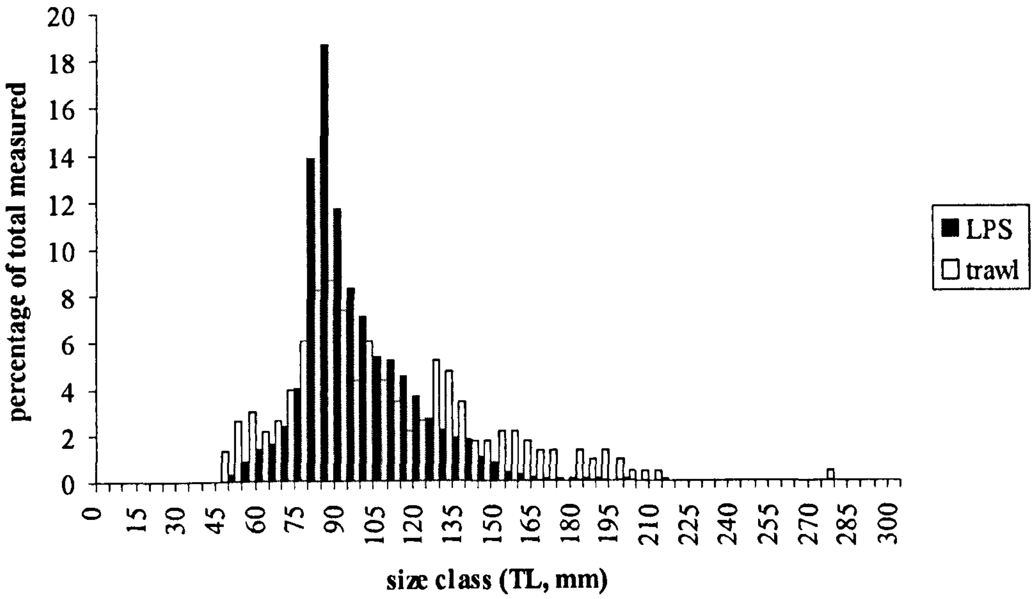


Figure 5.6 cont.

(e) plaice

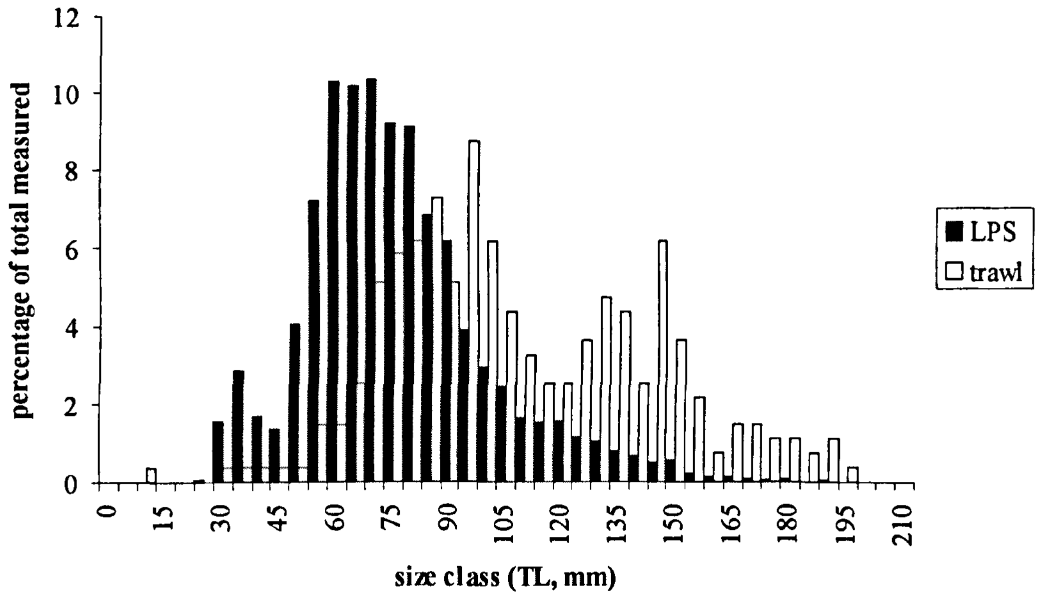


Figure 5.6 cont.

This rather crude comparison has some limitations. The trawl data represent the summation of all measured fish from hauls at all three mid-lower estuarine stations. Thus a trawl undertaken at Port Edgar would be unlikely to sample the same shoal of herring as was being simultaneously sampled at LPS, for example. For the purposes of this simple analysis, the assumption was obviously all size classes of fish would be as likely to encounter trawls as the LPS intake, and that the preponderance of different size classes in the two sampling methods is a reflection of differences in avoidance of capture in the two methods. Despite the simplicity of the comparison, the conclusions are similar to a much more rigorous test of the same phenomenon by Maes (2000). He showed that stow nets, a passive fishing technique facilitated by tidal currents, tended to simultaneously catch the same species as the CW intake of the Doel power station, but that the nets caught larger fish than the intake (Table 5.11). The two gears were located only 50 m apart and were sampled simultaneously, so were likely to be encountering fish from the same micro-area of estuary.

Table 5.11. Mean lengths of fish sampled at Doel Power Station CW intake and by two stow nets located 50 m from the intake. Samples collected during 10 different 1-h periods. Statistical probabilities of lengths significantly differing are shown, based on a Mann Whitney U test. Adapted from Maes (2000).

	stow net	Doel CW intake	number measured	probability of significant difference
whiting	139.8 mm	122.5 mm	96	$p = 0.039$
sprat	66.2 mm	61.4 mm	1036	$p < 0.001$
herring	79.0 mm	71.9 mm	1056	$p < 0.001$
plaice	126.1 mm	91.6 mm	20	$p = 0.021$

A simultaneous comparison of trawling with intake sampling at the adjacent West Thurrock power station, Thames Estuary, also confirmed that larger fish were present in the water body than were appearing on intake screens (Thomas, 1998), though no formal statistical analysis was undertaken.

Sampling of impinged fish at the LPS CW intake screens offers a cost-effective and relatively straightforward means of monitoring fish populations of the Forth Estuary. A good example of this is the case of sprat, noted to be taken at a marginally greater rate per unit volume by pelagic trawling than LPS impingement sampling. The important point is that sampling at LPS supplies fish from 45.5 m^3s^{-1} of abstracted CW, assuming two CW pumps are operational and 'feeding' the trash basket where sampling is occurring. Pelagic trawling at 2.5 knots with gear of opening 17.75 m^2 (section 3.2.2.1) therefore samples approx. 22.9 m^3s^{-1}. To obtain 100 sprat would therefore take on average approx. 413 s by impingement sampling, and 659 s by pelagic trawling. Given the premium on time that usually exists in sampling programmes, intake sampling offers a good means to obtain specimens relatively quickly. The considerable quantities of

CW abstracted mean that inefficiencies of the method compared to Agassiz trawling may be offset by the ease of obtaining samples.

5.4. Future research

5.4.1. Studies of impingement of species of particular interest

The present study investigated all impinged species collected during routine sampling, the timing of which was based on predetermined tidal and diurnal characteristics. In the future it may be desirable to concentrate limited resources into sampling at certain critical times. The best example of this would be an investigation of impingement of salmonid smolts during the downstream migration season. The present study attempted to generate estimates of Atlantic salmon and sea trout impingement in 2000 (section 4.3.2). To improve upon these estimates it is necessary to sample with a greater intensity during the months of April – June, something that has already been attempted in 2001 by SEPA (A.S. Hill, Scottish Environment Protection Agency, personal communication). The most thorough method would be search and removal of all smolts from skips prior to disposal to landfill, a very demanding task. Aprahamian and Jones (1997) enumerated all Atlantic salmon smolts captured in one month at Uskmouth Power Station in 1989. An abstraction rate at LPS of three times that of Uskmouth means that far greater quantities of fish and debris would need to be processed in order to identify smolts.

5.4.2. Acoustic studies of fish within the Forth Estuary

Potentially movements of individual fish within the Forth Estuary could be studied by fitting fish with miniature acoustic transmitters, as carried out on Atlantic salmon smolts in the River Conwy Estuary, Wales, by Moore *et al.* (1995). The main purpose of such a study would be to elucidate tidal and diel influences on the movements of individual fish. The particular hypothesis that enhanced impingement at LPS during spring tides may have been a result of increased tidal currents and tidal excursion causing a greater quantity of fish to enter the region of the CW intake per unit time than during neap tides could be tested by this method (section 2.4.3.2). The acoustic transmitters employed by Moore *et al.* (1995) required a number of acoustic relay buoys and automatic listening stations to record signals from the buoys. Such transmitters are 17mm long × 8mm diameter, have a mass of about 0.35g in water, and are surgically placed within the body cavity of the fish, thus limiting the size and species that could have the devices implanted. Maximum range of transmitters was about 100m, usually 50 – 75m because of the environmental effects of changing turbidity, salinity, etc. that could influence propagation of the signal (Moore *et al.*, 1995). The approximate battery life of 30 d for the transmitter would be sufficient to allow monitoring of fish over at least one spring-neap tidal cycle.

Small-bodied and delicate species such as sprat and juvenile herring encountered in the Forth Estuary could not readily be fitted with acoustic transmitters. The potential use of sonar employed from a research vessel for estimation of the abundance of clupeids in the Forth was already suggested (section 3.4.5), and an extension of this may be to attempt to track the movements of clupeid schools by pursuit in a vessel employing such acoustic equipment.

5.4.3. Mark and recapture of fish in the Forth Estuary

Capture of live fish by trawling (or by impingement collections for more robust species such as flatfish) followed by marking (*e.g.* with dye), release and subsequent recapture could facilitate useful further study of the fish populations of the Forth Estuary. The discrepancy in estimates of fish population sizes within the Forth Estuary and abundance of fish impinged at LPS has already been discussed (section 5.1). Mark and recapture experiments may offer a means to generate more accurate estimates of population size. Population size, n, may be estimated by using the equation:

$$n = \frac{MC}{R}$$

where M = number of marked fish released, R is number of marked fish recovered among C total fish recaptured (Pawson and Eaton, 1999). Using this technique, Pawson and Eaton (1999) released 322 alcian dye-marked 0-group sea bass close to Kingsnorth Power Station, Medway Estuary. Eighteen of these marked fish were recovered in impingement samples up to a month following release, out of a total of 1.65×10^4 bass collected from the intake screens. This suggested the total population of bass living near the intake to be approx. 2.95×10^5 individuals. Given that only 48% of fish were impinged after entering the CW system (the remainder being damaged or eaten by crabs; see section 2.2.3.3), the mortality due to impingement was approx. 17.4% (5.10×10^4 of 2.95×10^5) (Pawson and Eaton, 1999). To undertake such an exercise would be potentially labour-intensive, since accurate estimates of total impingement are necessary and much time needs to be devoted to sorting through all debris displaced into trash baskets. The requirement to examine fish closely for marks also increases handling time during the data collection process. Marking of fish may assist in assessing the spatial extent of marked species, *e.g.* to what extent the species move beyond the boundaries of the estuary and into the Firth of Forth. This would require additional trawl sampling beyond the boundaries of the estuary, again demanding of resources. As with all mark and recapture studies, it would be assumed that marked fish could always be identified upon recapture, and that the marking process did not alter marked fishes' susceptibility to predation or impingement.

5.4.4. Fish entrainment studies at LPS

The present study concentrated solely on routine sampling of impinged fish at LPS. Entrainment of fish, meaning the passage through the condensor system of life stages too small to be impinged (*i.e.* eggs and larvae), was not investigated. Study of entrainment at LPS would potentially be of great interest. From a purely scientific standpoint assessment of eggs entrained at LPS would provide information on species that may be spawning in the estuary. In addition, estimates of mortality of eggs and larvae may also be incorporated into calculations of losses of equivalent adults (section 4.3.2) for commercial species. This was undertaken at the Sizewell Power Stations and for some species losses due to entrainment were estimated to be greater than those caused by impingement (Turnpenny and Taylor, in press; Table 5.12).

Table 5.12. Estimated annual equivalent adult tonnages of commercially important species lost at Sizewell Nuclear Power Stations due to impingement and entrainment mortalities. Data from Turnpenny and Taylor (in press).

	Sizewell A station		**Sizewell B station**	
	impinged	**entrained**	**impinged**	**entrained**
plaice	0.41 t	2.34 t	0 t	4.54 t
sole	4.0 t	19.5 t	0.20 t	37.8 t
dab	2.30 t	0 t	0.41 t	0 t
cod	0.84 t	0 t	0.10 t	0 t
whiting	41.0 t	0 t	32.7 t	0 t
herring	51.0 t	13.8 t	73.2 t	26.7 t
sea bass	0.35 t	0 t	0.07 t	0 t
total	100 t	35.6 t	107 t	69.0 t

Ichthyoplankton surveys in the Firth of Forth from March – July 1986 yielded fish eggs of 23 species and larvae of 12 species; dab (42.2%), whiting (13.4%) and flounder (11.1%) were most common of the eggs examined, while sandeel (65.3%) and clupeids

(29.3%) dominated larval abundance (Poxton, 1987). Egg and larval densities were greatest in the centre of the Firth and least in the inner Firth, suggested by the author to be due to the eggs being primarily of marine as opposed to estuarine origin. This might also suggest that equivalent adult losses attributable to entrainment may be low at LPS, and studies could investigate the hypothesis that losses caused by entrainment would be expected to be lower in the Forth Estuary than at marine sites such as Sizewell. An entrainment study at the 2000 MW Fawley Power Station on the estuarine Southampton Water estimated that direct losses of eggs and larvae of fish were relatively small and that, even with conservative assumption of survival to adulthood of 0.01% of eggs and 0.1% of larvae, equivalent adult losses would be low (Dempsey, 1988; Table 5.13). The author described the low diversity and abundance of entrained early life stages of fish as 'typical' of estuarine waters; a similar study at LPS would be of interest given that the salinity regime at Fawley (28 – 32 PSU; Turnpenny, 1988b) is very similar to that in the mid-lower Forth Estuary. Identification of herring eggs in samples of entrained materials at LPS could also elucidate the extent of herring spawning in the Firth in spring (section 4.4.1).

Table 5.13. Estimated losses of ichthyoplankton (and adult equivalents) due to entrainment at Fawley Power Station (September 1986 – September 1987). Direct losses ± S.D. Adult equivalent values calculated based on assumption of 99.99% mortality of eggs and 99.9% mortality of larvae. Reproduced from Dempsey (1988).

	direct loss	**adult equivalents**
larvae:		
sandeels (*Ammodytes* spp.)	$8.8 \times 10^4 \pm 8.8 \times 10^4$	0 – 180
sand smelt	$6.3 \times 10^4 \pm 6.3 \times 10^4$	0 – 120
fatherlasher	$5.4 \times 10^4 \pm 5.4 \times 10^4$	0 – 110
flounder	$1.1 \times 10^6 \pm 2.8 \times 10^5$	820 – 1380
gobies	$1.8 \times 10^7 \pm 2.9 \times 10^6$	15100 – 20900
sprat	$3.6 \times 10^6 \pm 9.2 \times 10^5$	2680 – 4520
great pipefish	$6.3 \times 10^5 \pm 3.1 \times 10^5$	320 – 930
lesser pipefish	$4.1 \times 10^5 \pm 5.4 \times 10^4$	60 – 160
long-spined sea scorpion (*Taurulus bubalis*)	$1.4 \times 10^5 \pm 1.4 \times 10^5$	0 – 280
eggs:		
lesser weever	$5.4 \times 10^4 \pm 5.4 \times 10^4$	0 – 11
sole	$3.3 \times 10^5 \pm 3.3 \times 10^5$	0 – 6
sprat	$8.5 \times 10^5 \pm 8.5 \times 10^5$	0 – 19

5.4.5. Impingement and entrainment studies of other taxa at LPS

Fish are only one part of the Forth Estuary ecosystem, and impingement and entrainment of other organisms present in the water body also occurs. Substantial quantities of common shrimp, *Crangon crangon* L., were observed to be impinged during the course of the present study, and estimation of the extent of their impingement is of relevance given their importance as food items for fish such as whiting (Henderson and Holmes, 1989). Assessment of potential ecosystem-level effects by LPS would require sampling of additional trophic levels of the estuarine food web. Thus it would be relevant to undertake entrainment studies of removal of primary producers (*e.g.* phytoplankton), holoplanktonic crustacea such as calanoid copepods, and larval fish, in addition to the sampling of impinged fish that is already well established. Such sampling could be undertaken using a pump sampler fitted with a fine mesh employed in the intake forebay, as carried out in 1992 – 93 at the Sizewell power stations to investigate entrainment of commercial fish species (Turnpenny and Taylor, in press; see section 5.4.4). The potential importance of removal of organisms in the lower levels of food webs was discussed by Henderson and Seaby (2000) with the example of a simple theoretical four level pelagic food chain. Transfer of production (g of carbon) was assumed to be 10% efficient between successive trophic levels, so that 1 g of carbon fixed by primary producers resulted in 0.001 g of carbon produced by predatory fish. A theoretical decrease in standing crop due to entrainment of organisms in the first three levels of the simple chain resulting in transference efficiency decreasing to 9% would give a decrease in predatory fish production of 27% (Figure 5.7). Entrainment studies could be supplemented with sampling in the outflow channel in order to assess survival rates of organisms passing through the condensor circuit system.

trophic level	**g of carbon produced at 10% transference efficiency**		**g of carbon produced at 9% transference efficiency**
1.primary producer	1		1
	↓	1 % decrease in	↓
2. planktonic crustacean	0.1	→	0.09
	↓	transference efficiency	↓
3. larval fish	0.01		0.0081
	↓		↓
4. predatory fish	0.001		0.00073

Figure 5.7. Theoretical example of potential importance of decreased transfer efficiency caused by decrease in standing crop of organisms in trophic levels 1 – 3 due to power station entrainment. Developed from Henderson and Seaby (2000).

5.4.6. Trawl studies

The present study originally intended to combine sampling of fish by trawling with simultaneous recording of water quality data using the same multiprobe as employed off the LPS intake jetty (section 2.2.2). The multiprobe was unavailable on sufficient occasions to make any meaningful use of the data thus collected, so no analysis was possible. In the future it may be desirable to collect such data in order to investigate associations between fish assemblages and environmental parameters, as undertaken in the Humber by Marshall and Elliott (1998) (see section 1.3.2). In the long term, it is desirable to continue sampling the mid-lower Forth Estuary using similar methods to those discussed in Chapter 3, particularly in the case of Agassiz trawling. The 20-year database of knowledge has already revealed some interesting changes in the ichthyofauna of the Forth. Will changes continue to occur as global temperatures continue to rise? Could a species such as eelpout become absent from the estuary proper due to these increases? Assuming Agassiz trawling is the best means of sampling the resident fish species, it will eventually be interesting to assess to what extent cessation of operations at LPS (scheduled for 2020; L. McSporran, ScottishPower plc., personal communication) might bring about changes in the Forth fish populations.

5.4.7. Alternative means of sampling at various locations within the estuary

Small scale fishing using stow nets occurs in the region of Kincardine on Forth. This passive fishing technique may offer an alternative to pelagic trawling, the latter being difficult to undertake in an estuary such as the Forth where tidal currents are strong. The ineffectual sampling in January 1999 served to illustrate occasional difficulties experienced with pelagic gear, for minimal numbers of fish were taken during six hauls in a month when fish impingement was reasonably pronounced and abundance would have been expected to be high. Stow net deployment at shallow and deep intervals in the water column with simultaneously conducted impingement sampling would allow further investigation of the efficiency of LPS as an alternative sampling device, as carried out in the Zeeschelde by Maes (2000) (see section 5.3.2). Fixed net fishing in the Severn yields species in similar to proportions to power station sampling (Henderson and Holmes, 1991).

5.5. Summary and Main Conclusions

- The study has investigated the fish populations of the lower Forth Estuary, in terms of species composition, diversity, and seasonal and spatial distribution.
- Both LPS impingement sampling and trawling yielded a quantity of fish species in the mid-lower Forth Estuary that was of the order expected for an inshore area of this geographical latitude.
- A preliminary analysis of the abundance of common clupeids (sprat, herring) by novel pelagic trawling suggested that abundance in the Forth Estuary was at least an order of magnitude greater than previously believed from bottom trawling. The estimates thus generated exhibited wide confidence intervals, suggesting evaluation by other means would be worth considering.
- An assessment of the 19-year dataset of Agassiz trawling of benthic and demersal species at three stations in the mid-lower Forth Estuary showed that while species richness did not show a long term trend, total abundance exhibited a significant decline. This was largely driven by significant decreases in whiting and eelpout abundance, the latter seemingly being related to increased temperatures caused by climatic change. Changes in the benthic and demersal ichthyofauna of the lower estuary are largely due to the decline of the aforementioned species and also the increase in abundance of cod and plaice.
- The common benthic and demersal species were generally most abundant at the most seaward site, presumably reflecting their marine origin, and were captured in greatest numbers at LW, likely to be due to their concentration in the estuarine channel and retreat from intertidal areas.

- Species caught in trawls showed seasonal trends in abundance typical of temperate inshore areas of the northern hemisphere, and these trends were generally mirrored in seasonal trends of impingement at LPS.
- The main factors influencing the presence of fish within the vicinity of the CW intake at LPS and the total rate of fish impingement are season and tidal range, typical of most marine and estuarine power stations. Light did not influence impingement rate in almost all common species investigated, supporting evidence from other locations that high turbidity levels diminish likelihood of escape by visual perception of intakes by day and night.
- The annual total of fish impinged at LPS in 1999 – 2000 was concordant with theoretical expectations based on an exponential relationship between CW abstraction and fish impingement, established from other locations in Britain and the North Sea coast of Europe. The estimated numerical extent of fish impingement at LPS is as predicted for a power station of this size.
- Extent of impingement of species such as salmonids, eel and river lamprey that are of special consideration due to recreational, socio-political or threatened status was determined but conclusions were unclear, given a lack of stock information and the nature of the routine impingement sampling protocol.
- The lack of data on relative abundance of fish in the Forth Estuary prior to commencement of generation at LPS means that equilibrium densities for fish for this period are not known. This results in a lack of certainty regarding the population level impact on the Forth Estuary ichthyofauna by LPS CW withdrawal.

- Mitigation of impingement losses may be considered as a precautionary measure to reduce mortalities caused by LPS CW abstraction. A variety of techniques are available, but differ widely in cost and retrofitting ability.
- A comparison of LPS impingement sampling and trawling revealed that impingement samples compared favourably to pelagic trawl samples in numbers of fish captured per unit volume, though differences in mean sizes of fish taken existed. Agassiz trawling tended to yield more fish per unit volume or area than impingement sampling, though the fundamental differences in the sampling techniques made comparisons difficult. The rate of water abstraction at LPS means that quantities of fish obtained per unit time may exceed those obtained by trawling even if impingement sampling is less efficient. This, coupled with relatively minor cost and ability to be carried out in almost all weather conditions, makes impingement sampling at LPS a particularly useful tool in monitoring fish populations of the Forth Estuary.
- Much potential research exists, both related to operation of LPS and in terms of trawl studies nearby. Alternative means of sampling the ichthyofauna may be worth consideration, including stow nets.

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Appendix 1 – scientific nomenclature of species sampled in the present study

family and species	ecological guild*	habitat†	sampling method‡
Petromyzonidae			
River lamprey, *Lampetra fluviatilis* (L.)	CA	B	L, A, P
Clupeidae			
Sprat, *Sprattus sprattus* (L.)	MS	P	L, A, P
Herring, *Clupea harengus* (L.)	MS	P	L, A, P
Salmonidae			
Atlantic salmon, *Salmo salar* (L.)	CA	P	L, P
Sea trout, *S. trutta* (L.)	CA	P	L
Osmeridae			
Smelt, *Osmerus eperlanus* (L.)	CA	P	L, A, P
Anguillidae			
European eel, *Anguilla anguilla* (L.)	CA	B	L
Syngnathidae			
Great pipefish, *Syngnathus acus* L.	ER	B	P
Lesser pipefish, *S. rostellatus* L.	ER	B	L, A, P
Atherinidae			
Sand smelt, *Atherina presbyter* Cuvier	MJ	P	L
Gasterosteidae			
Three-spined stickleback, *Gasterosteus aculeatus* L.	CA	P	L, P
Fifteen-spined stickleback, *Spinachia spinachia* (L.)	ER	D	L
Sternoptychidae			
Boreal pearlside, *Maurolicius muelleri* (Gmelin)	MA	P	L
Gadidae			
Whiting, *Merlangius merlangus* (L.)	MJ	D	L, A, P
Pollack, *Pollachius pollachius* (L.)	MJ	D	L, A, P
Saithe, *P. virens* (L.)	MA	D	L, A
Cod, *Gadus morhua* (L.)	MJ	D	L, A, P
Haddock, *Melanogrammus aeglefinus* (L.)	MA	D	L
Bib, Trisopterus luscus (L.)	MJ	D	L
Poor cod, *Trisopterus minutus* (L.)	MA	D	A
Silvery pout, *Gadiculus argenteus* Guichenot	MA	D	L
Ling, *Molva molva* (L.)	MA	D	L, A
Five-bearded rockling, *Ciliata mustela* (L.)	MS	B	L
Ammodytidae			
Lesser sandeel, *Ammodytes tobianus* L.	ER	B	L, P
Callionymiidae			
Common dragonet, *Callionymus lyra* L.	MA	B	L
cont.			

Trachinidae			
Lesser weever, *Echiichthys vipera* (Cuvier)	MA	B	L
Gobiidae			
Gobies, *Pomatoschistus* spp.	ER	B	L, A, P
Pholididae			
Butterfish, *Pholis gunnellus* (L.)	ER	B	L, A
Zoarcidae			
Eelpout, *Zoarces viviparus* (L.)	ER	B	L, A, P
Serranidae			
Sea bass, *Dicentrarchus labrax* (L.)	MJ	D	L
Percidae			
Perch, *Perca fluviatilis* L.	FW	P	L
Mugilidae			
Thick-lipped grey mullet, *Chelon labrosus* (Risso)	MS	D	L
Triglidae			
Grey gurnard, *Eutrigla gurnardus* (L.)	MS	B	L
Cottidae			
Fatherlasher, *Myoxocephalus scorpius* (L.)	ER	B	L, A
Agonidae			
Pogge, *Agonus cataphractus* (L.)	ER	B	L, A, P
Liparidae			
Sea snail, *Liparis liparis* (L.)	ER	B	L, A, P
Montagu's sea snail, *Liparis montagui* (Donovan)	MA	B	A
Scombridae			
Mackerel, *Scomber scombrus* L.	MA	P	L
Pleuronectidae			
Common dab, *Limanda limanda* (L.)	MJ	B	L, A
Flounder, *Platichthys flesus* (L.)	ER	B	L, A, P
Plaice, *Pleuronectes platessa* (L.)	MJ	B	L, A, P
Long rough dab, *Hippoglossoides platessoides* (Fabricius)	MA	B	A
Lemon sole, *Microstomus kitt*	MA	B	A
Soleidae			
Sole, *Solea solea* (L.)	MJ	B	L, A

* For ecological guild classifications see section 1.3.1.

† P = pelagic, B = benthic, D = demersal.

‡ L = collected by impingement sampling at LPS, A = collected by Agassiz trawling, P = collected by pelagic trawling.

Appendix 2 – CD-ROM of data collected in 1999 – 2000

Raw data obtained during all LPS impingement collections and the majority of trawling days in 1999 – 2000 are found as Microsoft Excel spreadsheets within the CD-ROM inside the back cover of the present study (files within folders 'Fish Data 1999' and 'Fish Data 2000'). These data include species common name (see Appendix 1 for scientific nomenclature), total lengths, masses and abundances. Impingement sampling sessions files take the form 16OCT1, 16OCT2, etc., signifying the first and second sessions on a particular date, 16 October 1999 in this case. LPS impingement files from the year 2000 possess an 'x' prefix, hence X27APR1, X27APR2, etc., for the first and second samples collected on 27 April 2000. All impingement files are divided into worksheets that represent each 3-minute subsample of impinged materials collected from the intake screens. These are labelled according to their location, e.g. WS1 and WS2 were the first and second subsamples collected at the west screens of the CW intake, while ES1 was the first subsample collected at the east screens. Raw data for 24-h sampling sessions in March and September 2000 (see section 2.2.3.2) are included (these are labelled XMAR24H1, XMAR24H2 etc. for March 2000, and XSEP24H1, XSEP24H2 etc. for September 2000).

All raw trawl data files can be identified by the suffix –tr; the form of the data files differs, some containing Agassiz and pelagic trawl data, others possessing data from only one gear type. In all cases the details of the trawling become evident upon opening the relevant file. Summary data files are included in monthly subfolders, taking the form xlgtnovsum for data collected at LPS in November 2000 (xlgtmar24hsum in the case of the 24-h sessions), and trwljulsum for trawl data collected in July 1999. Summary data for trawling undertaken in January 2001 are included (xtrwljan2001sum) in the Dec2000 subfolder of Fish Data 2000, these data having been used in section 3.2.2 to compensate for poor functioning of the pelagic trawl in January 1999.

The 1999 – 2000 impingement and trawl data are summarised in the file 'Data used in thesis', including abundance and biomass estimates of fish taken at LPS and abundances of trawled fish. Long-term Agassiz trawl data (1982 – 1998), as well as trawl data not included in the Fish Data 1999 and Fish Data 2000 folders, are not included, and enquiries regarding these data should be addressed to:

Tidal Waters Section

SEPA East Region

Clearwater House

Heriott-Watt Research Park

Avenue North

Riccarton

Edinburgh EH14 4AP.

In addition, a Microsoft Word 2000 document of the present study is also located on the CD-ROM (file 'whole thesis – final hardbound version').