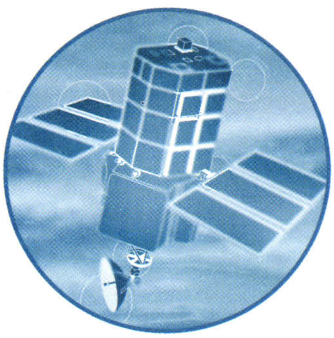
Eel and Elver Stocks in England and Wales - Status and Management Options



Research and Development

Technical Report
W248

MAFF

ENVIRONMENT AGENCY

Eel and Elver Stocks in England and Wales – Status and Management Options

R&D Technical Report W248

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Website: www.environment-agency.gov.uk

ISBN 1 85705 400 8

This report is the result of work jointly funded by the Environment Agency and the Ministry of Agriculture Fisheries and Food (MAFF).

Dissemination Status
Internal: Released to Regions
External: Released to Public Domain

Statement of Use
This report assesses current eel stock status in England and Wales and possible reasons for observed distributions and differences in density and biomass. It should be used by Environment Agency or MAFF staff in making decisions on monitoring needs and procedures, in assessing issues for further study and in promoting local, national and European management proposals.

Keywords
European eel, *Anguilla anguilla* (L): population dynamics; recruitment; migration; stocks; mortality; fisheries conservation; management; monitoring.

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EXECUTIVE SUMMARY

Export statistics indicate glass eel recruitment in England and Wales (as elsewhere in Europe) has declined, by ~55% between peak years in 1972-82 and 1983-99. Catches and CPUEs of yellow/silver eels peaked in the late-1980s and mid-1990s but have declined markedly since 1997. These changes possibly reflect lags of about 16-17 years, equivalent to the average freshwater life span, after particularly good recruitment years, but catches and effort have tended to closely follow market values and these have declined because of competition from farmed eels.

Assessing changes in eel stocks has been hampered by a lack of good-quality data series, due to natural variability and differences in fisheries survey methods, sites and records. However, >40 datasets analysed and intensive R&D resurveys of the River Severn (despite its estuarine glass eel fishery) and River Dee did not reveal any significant overall decline in stocks of yellow over the last 20-25 years. Resurveys did reveal significant declines in the Rivers Piddle and Frome (Dorset), but low recruitment and the impacts of Poole Harbour as a habitat sink for immigrants are implicated, not fishing mortality. Information on spawner escapement is lacking and data on stocks in estuarine, coastal and still freshwater environments is poor relative to that for rivers. However, the weight of evidence shows that the scattered and low-intensity commercial fisheries in England and Wales have had minimal impacts on stocks and spawner escapement, especially when compared to the extent of unexploited populations. Recent glass and yellow/silver eel catches are estimated to be < 0.5% and < 5.0% respectively of natural yields. Current recruitment is probably similar to that during the 1920s-1960s and oceanic factors are briefly discussed as possibly the main cause.

Shorter rivers and lower reaches of large catchments generally appear to receive sufficient recruitment to meet carrying capacities and have relatively high density populations dominated by small eels, the majority maturing and emigrating as males. Density tends to decline with distance from tidal limits (biomass to a lesser extent because of increasing average individual size) due to density-dependent migration and habitat segregation; upper reaches are dominated by larger females. Major declines in populations tend to occur at >50 km from tidal limits. Densities (but not necessarily biomass) are generally greater in west and south coast rivers compared to east coast ones, due to distances from Atlantic migration pathways and differences in river lengths, channel slopes and productivity. Low initial recruitment and migration barriers explain apparently restricted distributions in the upper reaches of large catchments such as the Thames, possibly exacerbated by estuarine and coastal habitat sinks for immigrants.

No evidence was found that habitat changes have significantly impacted stocks, spawner escapement or fisheries over the last 20 – 25 years. Eel habitat preferences are very catholic. Distance from tidal limits, altitude, natural and anthropogenic migration barriers, habitat productivity and, especially, availability of day-time refuges were found to be the main determinands of density, biomass and population structures in specific sites. Because of the adaptability of eels, enhancing stocks by eel-specific habitat improvements would not be cost-effective, but the effects of water-quality and physical migration barriers need to be minimised. Habitat suitability indices and biomass-based indices (as used in the Fisheries Classification Scheme) are currently of limited use for setting management reference points and in monitoring and further modelling studies are required to develop such approaches. A dual biomass-density classification scheme for general coarse-scale spatio-temporal comparisons is, however, proposed. Graphing and mapping are recommended for representing population characteristics and distributions (especially against distance from the tidal limit) and for highlighting spatial and temporal differences and anomalies.

More research is required to inform management on the relative effects of fishing *v.* natural mortality (especially density-dependent effects). Integrated studies of all freshwater life stages should be conducted, comparing fished and unfished waters or, preferably, using controlled experimental manipulations. No specific R&D programme can be recommended but suggestions for studies are made in relation to risks, resource implications and possibilities of integrated studies, partnerships or external funding. Exploratory studies are, however, suggested on glass eel and elver/juvenile recruitment and on the use of fish counters to quantify spawner escapement, to inform management and monitoring and with a view to possible integration into more extensive studies. Stochastic life-stage compartmental modelling approaches are recommended as a cost-effective means to aid future R&D, setting of management reference points and monitoring.

A National Eel Management Plan for England and Wales is proposed. In the short term, the precautionary principle should be applied in England and Wales to prevent fisheries impacting any further on stocks and escapement. Thus fisheries on all life stages should not be allowed to expand in effort or geographical range beyond the averages pertaining over the last five years. For long term management, reference points must be set to promote escapement of spawners and other life stages. Because of the problems of quantifying spawner escapement and/or glass eel recruitment, it is recommended that population indices (density, biomass and length class frequencies) are the best surrogate measures of stock status and escapement. As stocks in England and Wales have not changed significantly, interim and pragmatic management reference levels should be set on the optimum baseline status of stocks (and hence, indirectly escapement of each life stage) at local, regional and national levels determined over the last ten years. For temporal comparisons between (ideally, at least 20-25) sites, criteria signaling non-compliance with these reference points are if (a) a statistically significant difference in population indices is detected and/or (b) the mean of the percentage changes in site densities and/or biomass is $\geq$ -25%. Changes in average eel size, proportions of silver eels, sex ratios and intra-catchment distribution also provide evidence of possible critical changes.

If non-compliance with these management objectives is detected, any errors in monitoring should be checked first. If fishing mortality is implicated, exploitation should be controlled by one or more of closed area measures or restrictions on gear number, types and usage. Licences that include relevant conditions (including the provision of catch and effort data for the previous year) should be issued annually to individual fishermen. Various bye-law licence conditions to minimise by-catch mortalities are also recommended.

Careful monitoring is essential to aid in setting management reference points, measuring compliance and deciding on remedial actions. Because of their cost-effectiveness, annual import-export and fishing-effort data should be analysed to provide predictive information on recruitment and exploitation. However, electric fishing surveys in selected rivers are recommended as the best direct monitoring approach. Advice is given on survey methods, timings, site numbers and selection and sampling frequency. Eel-specific surveys should be conducted in primary indicator catchments at least every 5 years, i.e. in the Severn, Piddle, Frome and Dee because of their geographical distribution, existence of good historic data and presence/absence of fisheries. Because of distance from the Atlantic migration pathways, recruitment is relatively lower (and yellow/silver eel fisheries more important) on the east coast. Therefore recommendations are also made to use at least one east coast catchment as a primary indicator system (e.g. the Darent or Mole in the Thames catchment and the Essex Stour). Second priority catchments and methods are suggested, where eel data should be collected during routine multi-species or other surveys. Possible integration with R&D mortality studies are also discussed.

Standardised monitoring and data collection and processing and any resultant management interventions should be carried out locally under Regional direction but coordinated by a national centre. Because of the panmictic nature of eels, integration with European management, monitoring and R&D initiatives need to be taken into account.

KEY WORDS
European eel, *Anguilla anguilla* (L.); population dynamics; recruitment; migration; stocks; mortality; fisheries; conservation; management; monitoring

1. PROJECT DESCRIPTION

1.1 Project Objectives

This R&D project was instigated by the Environment Agency and the Ministry for Agriculture Fisheries and Food for England and Wales. The overall objectives of the project were to gather information and reach recommendations for sustainable management of eel stocks in England and Wales. The specific objectives were:

(a) To assess in-river stock status
- by identifying and using reliable historic data sets
- establishing the level of change that can be detected
- repeating surveys that gave rise to the historic data sets

(b) To establish a recommended methodology for future monitoring of stocks and fisheries (freshwater and coastal/estuarial)

(c) To design and cost a realistic research programme on the mortality of eel in freshwater and to investigate the habitat requirements of the eel and the changes in habitat availability that have taken place in the last 25 years

(d) Review the management options to ensure sustainable eel stocks and fisheries

1.2 Background to the Project

The European eel *Anguilla anguilla* L. is unusual in being panmictic and having a catadromous multi-stage life cycle, with long distance migrations and ubiquitous distribution (see Section 1.4). It is an important component of fish communities and aquatic ecosystems throughout Europe but is also of economic importance. It is commercially exploited during the glass eel/elver stages (primarily now as seed stock for aquaculture, but also for human consumption in Spain and Portugal) and during the yellow and silver eel stages, making potentially valuable socio-economic contributions to rural and fishing communities (Moriarty, 1996; Knights, 1997a). There have, however, been declines in glass eel recruitment and catches since the 1970s and there is anecdotal evidence for subsequent declines in yellow and silver eel stocks and fishery yields (McKinnon & Potter, 1994; White & Knights, 1994; Mann, 1995; Knights *et al.*, 1996; Moriarty, 1996; Moriarty & Dekker, 1997). Various causes have been suggested (Knights, 1997a,b), including overfishing of glass eels and elvers. This situation has been exacerbated in recent years by demands from the Far East (especially China), due to shortages of Japanese eel seed stock.

Concerns about eel stocks and fisheries in Europe led the European Commission in 1998 to request advice from the International Council for Exploration of the Sea (ICES) on the management of European eel. Specifically, information was requested on stock status and possible escapement targets for potential spawners and suggestions for management actions. European eel scientists considered these issues in an EC Concerted Action AIR Programme (Moriarty, 1996; Moriarty & Dekker, 1997). It was concluded that fisheries were generally small-sized, scattered and artisanal in nature but their socio-economic importance was emphasised, especially for many rural and fishing communities. It was estimated that glass eels catches were sufficient to provide stocking material to restore or enhance the European capture fisheries whilst ensuring the conservation of the species. In the light of high export demands in the late 1990s, the EC was urged to consider ways and means of maintaining an adequate supply of glass eel for Europe. Coordinated baseline surveys and monitoring programmes were also recommended. ICES/ACFM (Advisory Committee for Fisheries Management) responded in 1998 by stating that the European eel stock *'is outside safe biological limits and the current fishery is not sustainable'* and that *'fishing mortality is probably high'* on all life stages (Coop. Res. Rep. 229). The precautionary approach was invoked, with emphasis on setting provisional escapement targets to maintain or enhance spawner escapement. Advice from the Working Group on Eel was requested. At the Group's meeting in Silkebörg in September 1999 the lack of reliable long-term data on catches, effort and stock status were recognised as major impediments to estimating escapement targets and making optimal management decisions. Applying the precautionary principle, the Working Group proposed threshold reference levels (conservation limits) to help maintain or improve spawner escapement and restore recruitment and stocks (ICES CM 2000/ACFM:6, 1999). ACFM subsequently responded in November 1999 to EC by saying that fisheries *'impact the eel stock significantly'* and management should reduce fishing mortality at all life stages *'to the lowest possible level'* until a full recovery plan can be agreed upon and implemented. Because most eel fishing takes place under national jurisdiction, a *'common approach by all stakeholders to eel fisheries management is required'*. It was further recommended that member states should set targets for the escapement of glass, yellow and silver eels on a catchment area basis to achieve a 'recovered state'.

In England and Wales, the Environment Agency has statutory duties to maintain, improve and develop fisheries. The Agency produced Consultation Documents on a Proposed National Eel Management Stategy and on Eel Licences, Duties and Byelaws in spring 2000. In the light of discussions above, European issues have also had to be considered in this R&D project, as have recommendations relating to eels in the Salmon and Freshwater Fisheries Review (2000).

1.3 Project Plan

Consistent and logical management decisions (including possible legislative changes) must be based on good science but, as throughout Europe, there are gaps in knowledge of eel stocks and fisheries. This project was designed to address key issues with particular reference to:-

- Comparing the current status of stocks in England and Wales with historical data sets to clarify any possible changes (and causes) over the last 20-25 years (Chapter 2)

- conducting repeat surveys of selected rivers and sites where good quality historical data sets for eels could be identified. This led to detailed eel-specific resurveys of sites in the Rivers Severn, Dee, Piddle and Frome (Dorset) and exhaustive analyses of any changes and their causes (Chapter 3, plus raw data in Appendix B).

- Because of the possible importance of habitat loss and degradation on stocks, evidence for possible changes in habitat availability over the last 20-25 years has been explored and assessed. This also involved reviewing the habitat requirements of eels (Chapter 4)

- Knowledge of the relative impacts of natural *v.* fishing mortality in England and Wales is poor, evidence from the literature is therefore reviewed (Chapter 5) and a life-cycle model proposed and discussed (Appendix A), leading to;-

- Designing future research on mortality was a project aim but it was concluded earlier in the project that it was not possible to design and cost a specific research programme. Instead, suggestions and recommendations for various other research and modelling approaches are made in Chapter 6.

- Management issues, including possible application of the precautionary principle and the setting of management targets and conservation limits, had to be reviewed and pragmatic and attainable recommendations made. These are covered in Chapter 7, along with options for controlling fisheries (and by-catches and the use of passes and stocking as pro-active management measures). International as well as national issues are considered.

- High quality and consistent monitoring is required to help set and monitor management reference points and decide on appropriate management interventions. These issues are covered in Chapter 8, but interactions with stock-habitat relationships and the use of habitat and monitoring indices are also discussed in Chapter 4. Survey methods and resources issues are further reviewed and recommendations made in Chapter 3 and Appendix C.

1.4 Background Information on the Life Cycle of *Anguilla anguilla*

Below is a brief introductory review of some key features of the life cycle and biology of the European eel, drawn from White and Knights (1994). Further details, are given in the text where relevant, especially in relation to new data revealed by the R&D study.

The European eel is noted for its catadromous life cycle, breeding (as far as is known) in the Sargasso Sea in the SW Atlantic but spending much of its life feeding and growing in freshwater or estuarine/coastal waters (Fig 1.1). As far as is known, breeding occurs indiscriminately between spawners derived from different European waters, leading to a panmictic stock with minimal, if any, genetic separation of sub-stocks. It uses a reproductive strategy more typical of marine than freshwater fish, i.e. very large numbers of small eggs are produced that hatch into (leptocephalus) larvae that must feed and grow in the plankton before metamorphosing into the next life stage. High fecundity is necessary to compensate for variable and, potentially, very high natural mortality during the oceanic planktonic stage which may last ~3 years, although some authors claim migration may be completed in less than a year. Eels face further high natural mortality as they concentrate on the Continental Shelf (and during the estuarine and freshwater migratory stages. They metamorphose on the Continental Shelf into transparent glass eels and then migrate into a wide geographical range of European waters and the Mediterranean, during which time density-dependent mortality could increase markedly. Glass eels enter estuaries in the spring in England and Wales, as temperatures rise above 4-6°C. They utilise selective tidal stream transport but because of limited energy reserves, they must commence feeding before fully metamorphosing, developing pigmentation and becoming actively-migrating elvers. These will be called 0+ eels to indicate they are in their first year of non-oceanic life.

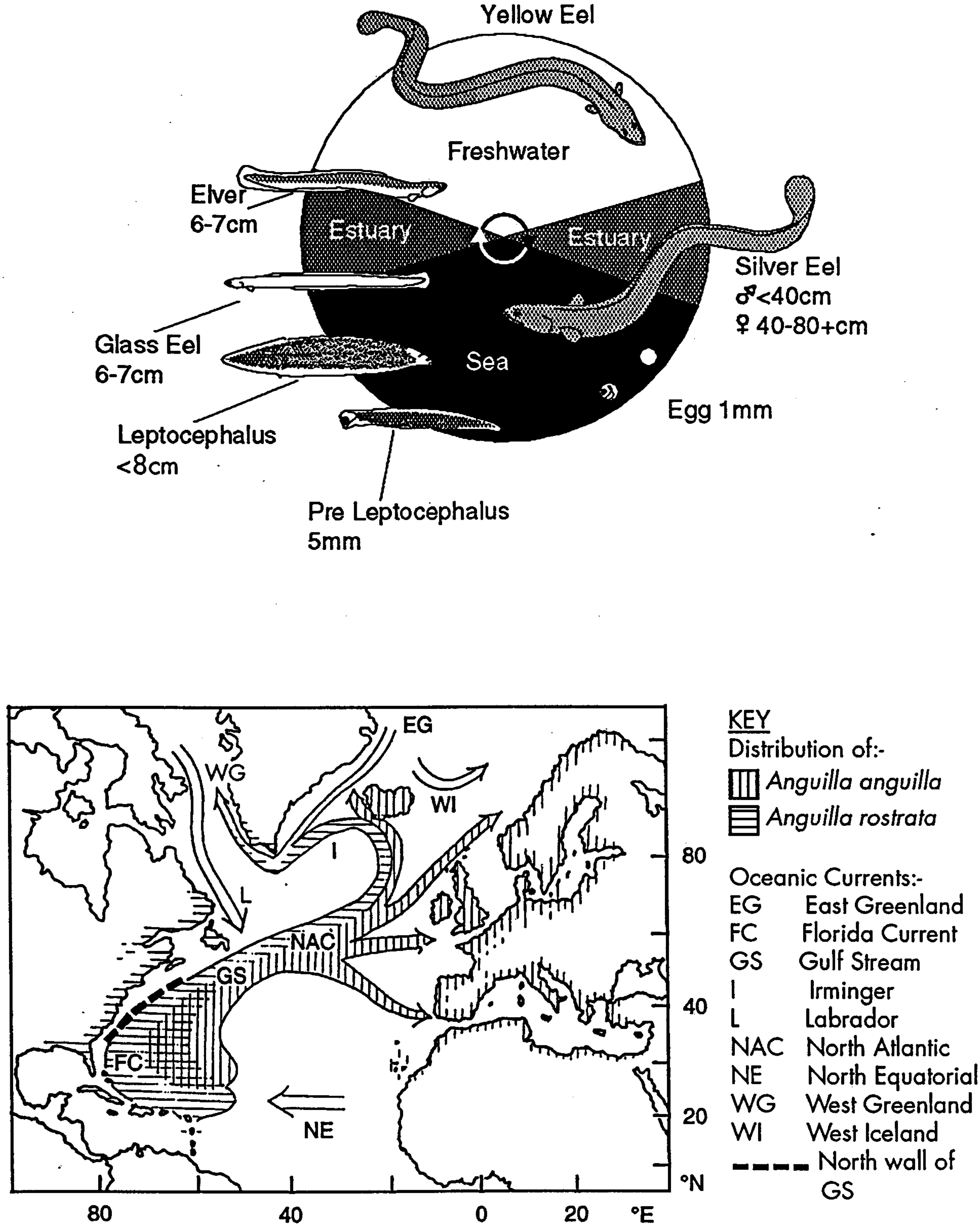


Fig 1.1 Life cycle and oceanic currents and migration pathways for the European eel *Anguilla anguilla* (and the American eel, *Anguilla rostrata*) (from White & Knights, 1994)

Some elvers will actively migrate up-river as temperatures rise above 14-16°C, but many may stop in the estuary or lower river stretches. Increasing density and competition promote up-river migration and habitat segregation, but this can be inhibited by natural and man-made barriers. High recruitment and low fishing or natural mortality would be expected to produce high density populations, dominated by smaller eels and earlier-maturing males. Declining recruitment of glass eels and elvers would be expected to be reflected by lower stock densities and an increasing preponderance of larger eels and relatively more female eels. This picture is most obvious in catchments on the east coast which are most distant from Atlantic migration pathways and the edge of the Continental Shelf, where metamorphosis from the leptocephalus occurs. Males mainly mature into 'silver' eels at <45 cm at >4-6+ years, females at 6 to 15+ years, before emigrating back to the sea to breed.

2. ASSESSMENT OF STOCK STATUS: SURVEY OF HISTORIC DATA SETS

2.1 Introduction

The specific contract objectives for this part of the study are:-

To assess in-river stock status by;-
- *identifying and using reliable historic data sets*
- *establishing the level of change that can be detected*
- *repeating surveys*

Identification of particularly robust historic data sets led to detailed repeat surveys of the Rivers Severn and Dee and the Dorset Frome and Piddle. These are briefly described, but further details are included in the next chapter. The bases for detecting significant levels of change are also discussed in detail in Chapters 3 and 7 and in Appendix C.

2.1.1 Data requirements

In identifying historic data sets, the following requirements were taken into account;-

- availability of ex/intensive commercial, fishery survey and other relevant data
- suitable time scales, preferably spanning the last 20-25 years, since the decline in recruitment of glass eels to Europe from peak values in the late 1970s and encompassing the average length of time spent by females in rivers in England and Wales
- reproducibility (sites surveyed, sampling month, methods used, types of data collected, etc)
- reliability of sampling methods and data obtained (via electric fishing, netting, trapping, mark-recapture and analysis of commercial fishery information)
- types and extent of data (density, biomass, length-class frequency, age, growth rates, numbers of silver eels, sex ratios)
- comparisons of exploited *v.* non-exploited catchments, where possible
- different geographical regions and habitat types
- absence/knowledge of confounding factors, e.g. migration barriers

In addition to using historic data sets, other more restricted ones were analysed to provide further information on intra- and inter-catchment differences in stocks and population structures and on possible fishery or other deleterious factors.

2.1.2 Sources of information

Information was sought from former Water Authority and NRA/EA fisheries and CMP/LEAPS, MAFF, CEFAS, ITE, IFE, RSPB, English Nature, British Waterways, the former CERL, publications in the literature, angling organisations and personal contacts (including eel fishermen and dealers, especially regarding the Thames, Solent, Stour (Dorset), Essex, Dee and Severn fisheries). Most older data sets were obtained directly from these sources, mainly in hard-copy. Some more recent eel data have been abstracted from the electronic-format (multi-species) data sets being collected by the Agency for processing on Oracle at ITE Monks Wood, as part of the development of the Database and Atlas of Freshwater Fisheries (DAFF) programme.

2.1.3 Deficiencies in data sets

Few robust time-series data sets were found that adequately met the listed criteria and were amenable to statistical analyses of temporal trends. Key deficiencies were;-

- lack of reliable commercial fisheries data, because of poor return rates, accuracy of catch returns and poor quality of other records located and variability of fishing effort

- lack of consistency of sampling sites, times and methodologies in fisheries surveys over suitable time scales, plus variations in data collection, collation and presentation. This was (and still is) noticeable between areas within Regions, let alone between Regions, compounded by policy and personnel changes and losses of data during privatisation and setting up of the NRA and then the Environment Agency. The majority of pre-1990 eel data had to be abstracted from hard-copies of tables, reports and field data sheets of varying format and quality, some key ones were mislaid on privatisation. Major impediments to developing robust time-series were reductions in fishing effort (e.g. to single electric fishing runs without stop nets) and the lack of consistency in survey sites used and in timings of surveys between years

- lack of consistent data for the late 1970s-early 1980s before routine electric fishing surveys came into full use

- lack of quantitative data specifically on eels (and other coarse fish), many surveys only collecting quantitative information on salmonids (especially in the NE, NW and Wales)

- there is a particular lack of information on estuarine and coastal stocks. This contrasts to the estimates of McKinnon and Potter (1993) that 41% of commercial fishermen fish partly or wholely in estuaries and 23% in coastal waters

Recommendations to overcome some of these shortcomings are given in the chapter on monitoring (Chapter 8).

2.1.4 Deficiencies in sampling

The following are common confounding factors due to the basic biological attributes of eels (Knights *et al.*, 1997);-

- the efficiency of electric fishing is affected by the body form and daytime cryptic behaviour of eels. They are therefore not as easily attracted to/stunned by electrodes as other species and catches often increase on successive runs. They (particularly small eels) are also not so easily seen and netted, especially when other species (e.g. salmonids) are being targeted. Furthermore, efficiency is reduced in deeper, turbid and more saline waters and dense vegetation.

- in this context, it must be emphasised that in this Chapter, most of the density and biomass figures from electric fishing surveys are underestimated. The current R&D resurveys have shown (Chapter 3) that sampling specifically for eels over three or four runs produces estimates up to three or four times greater than those achieved during standard multispecies three-run surveys.

- eels select sites during the day according to their suitability for burrowing or hiding in. Diurnal and seasonal migrations then occur to and from feeding areas and deeper waters for over-wintering (Tesch, 1977; Deelder, 1984). Thus catches are not necessarily absolute measures of habitat suitability at a specific survey site over extended time periods (see further discussions in Chapter 4). Additional upstream recruitment migrations by eels (dependent on water temperature) of up to 30cm also produce fluctuations in local populations. Large females in upper catchments or still waters may, however, be much more sedentary as growth slows (e.g. see Baras *et al.*, 1998) and they build up fat reserves for oceanic emigration and eventual transfer to the eggs.

- netting (e.g. by active seining or passive fyke netting) and trapping are generally inefficient and size-selective of larger eels and can only reveal general population trends via CPUEs (Naismith & Knights, 1990; Knights *et al.*, 1997).

2.1.5 Data sets analysed

Given the above drawbacks, a large number of individual and area data sets were located and examined. These are summarised Table 2.1, but fuller details of the Severn, Dee and Dorset river resurveys are included in Chapter 3. In some cases, individual data sets, including more recent ones, have been analysed to see if there are any abnormalities in distributions or population characteristics that might be related to recruitment declines or overfishing.

Discussions begin with a review of historical trends in glass eel recruitment (in comparison with other European data) and then in commercial yellow-silver eel fishery time-series data sets. Further sources and more detailed information are being sought as part of the Environment Agency's Economic Evaluation of Eel Fisheries R&D programme. Data sets to assess temporal (and spatial) stock and population trends are then presented in geographical order, beginning with the Severn/Bristol Channel and moving around the Regions of England and Wales in an anti-clockwise direction. The logic for this is that south-west facing catchments are expected to be better positioned relative to Atlantic currents and hence to glass eel migration and recruitment. This is illustrated in Fig 2.1, showing the progressive delay in the time of arrival of glass eels off the UK coasts after metamorphosis from the leptocephalus larva on the Continental Shelf. Not only have the runs of glass eel declined since the 1970s, but so has their condition (Moriarty & Dekker, 1997). Thus any delays would further increase natural mortality until they migrate into freshwater as temperatures rise above 4-6°C. This differential supply of recruits is why the main glass eel fisheries are located in the south-west, yellow-silver eel fisheries in the east and south.
All data sets cited will be provided in electronic format (or as paper copies of historic (multi-species) fishery surveys where these exist) to the DAFF programme and archived at the National Coarse Fish Centre.

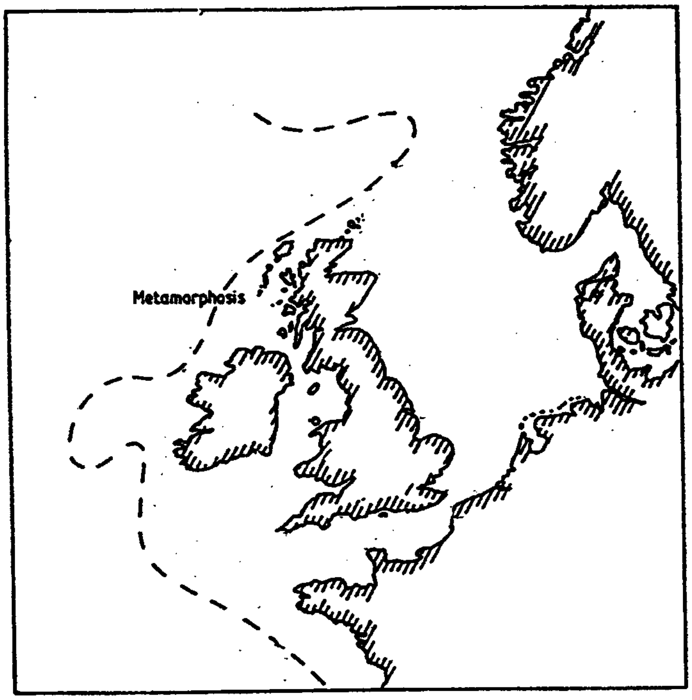


Fig. 2.1 Map showing the approximate position of the edge of the Continental Shelf where metamorphosis of leptocephali to glass eels occurs (from Tesch, 1977)

Table 2.1 Summary of stock assessment information in Sections in Chapter 2

KEY: Possible significant declines (⇩) (and possible periods of stability ⇔) shown in italics for D=density, B=biomass, LCF=length class frequency (see Chapter 3 for further details of resurveys of the Rivers Severn, Dee, Frome and Piddle); ?? = changes possibly due to interfering factors, e.g. differences in survey methods or migration barriers.

2.1	Introduction	
2.2	***GLASS EEL RECRUITMENT***	***⇩ 1974,78-79 ⇩1983 ⇔1984-99?***
	YELLOW-SILVER CATCHES	***⇩1987-88, 95-97 OR ⇔??***
2.3	Hinkley Point B Power Station 1981-97	
2.4	Bristol Avon	
2.5	N. Wessex rivers (1991-95)	
2.6	***TAW & TORRIDGE (1977-87)***	***⇩? (LCF)***
2.7	Rivers of N. Devon and Cornwall	
2.8	Fowey catchment (1978-87)	
2.9	Tamar (1978-97)	
2.10	***TAVY (1983-98)***	***⇩? (B, LCF)***
2.11	Avon & Erme (1978-91)	
2.12	Dart (1984-87)	
2.13	Teign (1979-90)	
2.14	Otter (1978-86-94)	
2.15	Exe, Culm & Creedy (1978-81)	
2.16	***AXE (1979-90)***	***⇩?? (D, LCF)***
2.17	***FROME/PIDDLE RESURVEYS (1973-90)***	***⇩ (D, B, LCF)***
2.18	Poole Harbour & The Fleet (1988-96)	
2.19	***STOUR (DORSET) (1992-8)***	***⇩?? (N)***
	(v. Hampshire Avon (1978-91)	
2.20	Conclusions regarding SW Region rivers	
2.21	S. Region rivers and fisheries (1978-98)	
2.22	Thames Estuary and freshwater catchment (1974-97)	
2.23	Rivers of eastern England (1979-98)	
2.24	NE rivers (Tweed, 1977-89 & Dee, 1977-81)	
2.25	NW Region rivers (Lune, 1991-97, Leven silver traps, 1942/4-94/5)	
2.26	Welsh region rivers (Dee, 1983-99, Afon Cwm, 1984-98)	
2.27	Midlands Region rivers [Severn resurveys, Section 3]	
2.28	Other sources of information	
2.29	Conclusions	

2.2 Historic Trends in Glass Eel Recruitment and Eel Fisheries

2.2.1 Glass eel recruitment; European perspectives

In view of the panmictic nature of *A. anguilla*, time-trends for the whole of Europe need to be considered to put any data for England and Wales into context. Declines since the late-1970s have been noted in all studies, from the Mediterranean (Tiber, Italy, for 1975-99 and Minho, Portugal, for 1974-94) to Scandinavia (e.g. Imsa, Norway, for 1983-96) (Moriarty, 1990, 1996; Moriarty & Dekker, 1997; EIFAC/ICES, 1999). The majority of data sets are, however, based on commercial catch data and are therefore influenced by variations in effort and inaccurate reporting. The longest fishery-dependent time series is for the Loire (1924-99), that for fishery-independent sampling is for Den Oever in the Netherlands (1937-99) (Moriarty, 1980, 1996; Dekker, 1999, on Internet source http://homr.wxs.nl/-hwdenie/rivoanp.htm; EIFAC/ICES, 1999). The latter time-series is based on standardised hand-netting of glass eels by fisheries scientists below the sluices to the IJsselmeer and is the most reliable available. Glass eels and elvers are trapped as they enter the R. Bann in N. Ireland to stock the Lough Neagh fishery, historical data for which has recently been presented by Rossell (1999). Fig. 2.2 shows there have been major declines from peaks in the late 1970s, with a tendency to stabilise in the 1990s and the changes at each location are significantly correlated with one another (P = 0.001) over 1972-98. Large fluctuations have occurred in the past and the Loire and Den Oever data for 1938-99 are not significantly correlated, but Fig. 2.2 shows that recruitment was relatively low between the 1920s-1960s

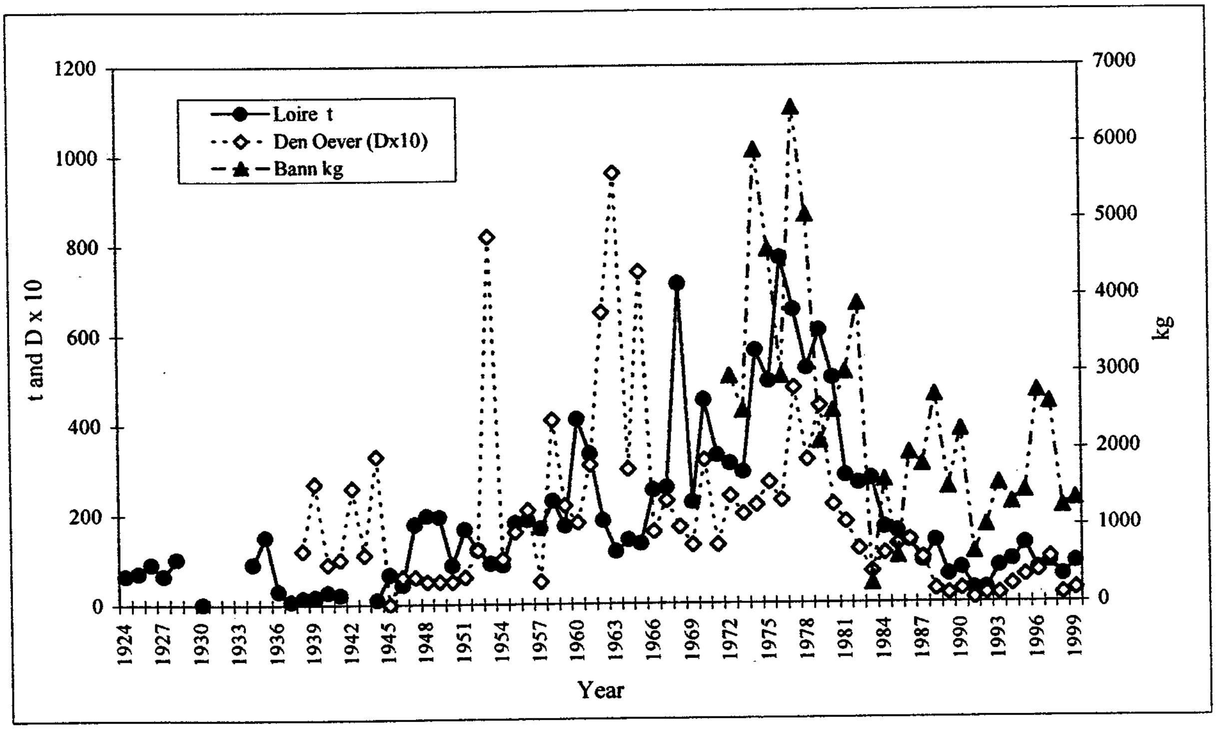


Fig 2.2 Long-term glass eel annual recruitment trends in the R. Loire, France (commercial catch, tonnes), Den Oever, Netherlands (relative index as April density, D x 10) and R. Bann, N. Ireland (trap catch, kg) (for sources, see text)

and similar to the current situation. It may be that current levels are more 'normal' and that the peaks in the 1970s-1980s were unusually high. Much larger fluctuations may have occurred over longer time scales and there are historical accounts of exceptionally large runs, e.g. even in the Thames (Naismith, 1992).

2.2.2 Glass eel recruitment in England and Wales

No long-term direct quantitative measurements of glass eel recruitment have been made and fisheries catches are the only means of assessing trends. Catch-return information was considered to be unreliable by White and Knights (1984) and Knights *et al.* (1997). Estimates were made by MAFF, the former SWWA and NRA and by the Environment Agency from various sources and, after introduction of licensing, from catch-return data (Knights *et al.*, 1999a). Derivations from Customs & Excise import/export data are the only other currently available source (McKinnon & Potter, 1993; White & Knights, 1994; Knights *et al.*, 1997). Problems can occur, however, in interpretation because of incomplete or mis-recordings and grouping of sources of imports and destinations of exports. Furthermore, excess catches may sometimes be dumped. For example, the MAFF and export estimates for 1979 were 67.2 and 40.1 t respectively (Fig. 2.3) but other evidence suggests the real catch was in the region of 120 t but prices were so low (~ £3 kg^{-1}), many eels were released. Although export statistics are for the whole of the UK, they have been assumed to cover only Great Britain because the only fishery allowed in N. Ireland is that on the R. Bann (trapping glass eels and elvers for stocking the L. Neagh fishery) and Scottish catches have been very small (A. E. Seymour, former owner of Sargasso Ltd eel farm, personal communication). The main catches are made in the estuaries around the Bristol Channel and in N. Wessex, but high market values have encouraged fishermen to fish a wider range of new sites in the late 1990s, as discussed later.

Import-export statistics for 1993-99 have been purchased from agents (Business & Trade Statistics Ltd, Lancaster House, More Lane, Esher, Surrey KT10 8AP) to update the time-series from 1979 first presented by White and Knights (1994). Further research is being conducted as part of the Economic Evaluation of Eel Fisheries R&D programme. Also, there have been some imports in recent years, these represent glass eels being transhipped from the French fisheries to the Far East (Knights 1999a). Actual catches have then been calculated as total exports minus total imports per year. The catch in 1992 was estimated in a questionnaire survey and analysis of export data by McKinnon and Potter (1994) to be between 8-15 and 19 tonnes respectively, this compares well with the estimate of 17.7 tonnes from exports made by White and Knights (1994). These values are larger than the total catch return of 5.04 t. However, MAFF and other agency estimates correlate well over 1979-98 with export figures (0.776, $P = 0.001$). Following the introduction of licensing, catch return data correlate with export estimates over 1980-98 (0.644, $P = 0.005$) and the means ± SD have been 21.46 ± 16.01 and 20.69 ± 8.54 t respectively. Catch returns therefore show more variability between years and although their accuracy may be queried, it appears they are useful in revealing time trends, as is indicated in Fig. 2.3a.

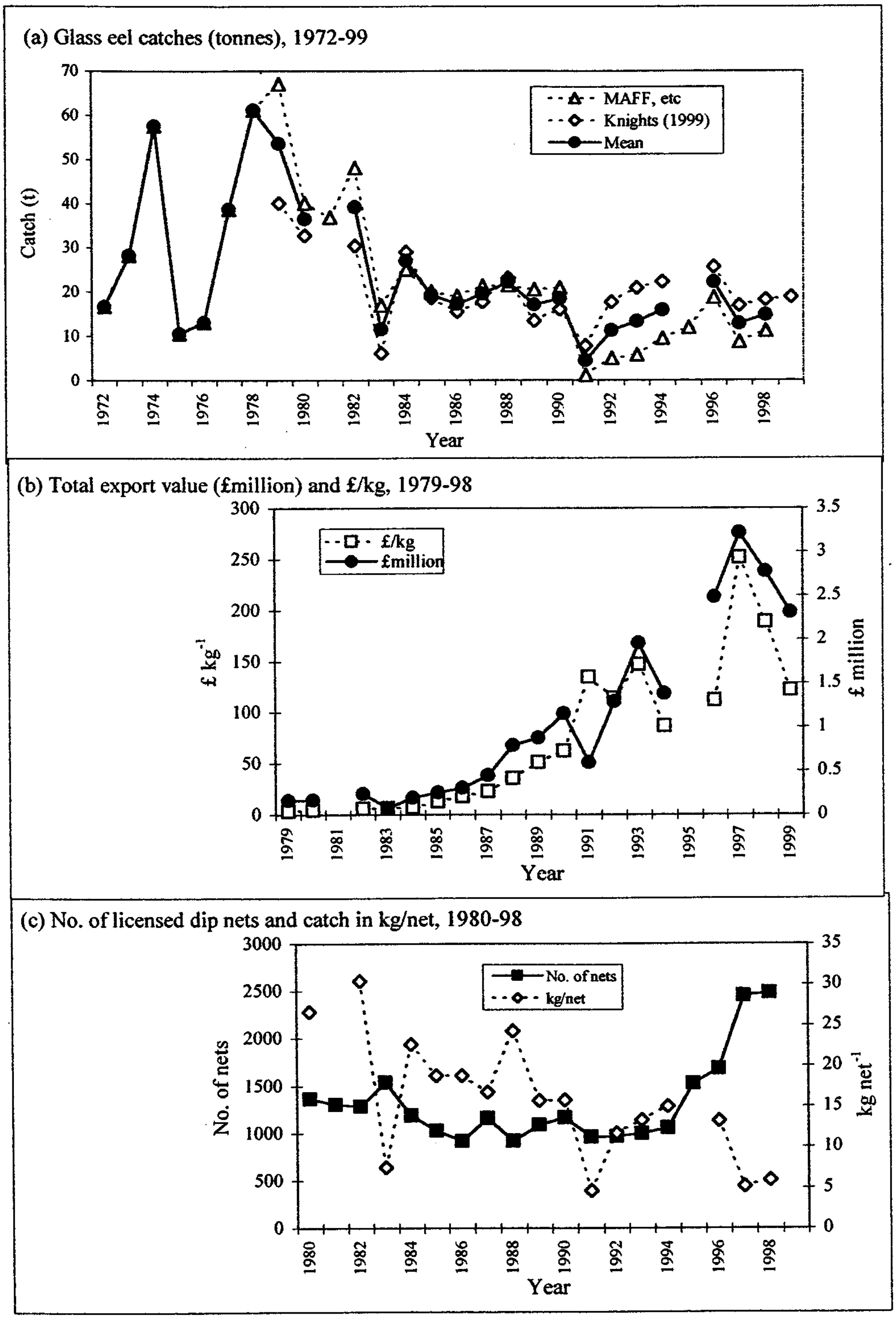


Fig. 2.3 Glass eel catches, export values and fishing effort in Britain, 1972-98 (see text for further explanation)

Some of the variations in MAFF/SWWA/NRA/EA and export data estimates (and their means) shown in Fig. 2.3a may be due to local factors, such as dry spring months (as in 1976) or low water temperatures, unfavourable winds and spates (as in 1991, White & Knights, 1994). The export picture for 1999 is confused (as discussed below) and catch return data were not available at the time of publication. According to export data, however, there has been a decline between the averages for peak years in the late 1970s-early 1980s and the 1990s of about 55%.

All the estimates correlate significantly (P = 0.001) over 1972-98 with the Loire, Den Oever and Bann time series. Fig. 2.3b shows that the value of catches were relatively low in the early 1980s but then rose steeply due to the shortfall in glass eels needed to meet demands from the Spanish direct-consumption and Scandinavian/E. European restocking markets – but, especially due to the increasing demand for seed stock from European eel farms. The steep increases in values in recent years are due to demands from eel farmers in the Far East, mainly China, reflecting buoyant markets for eel products but shortages of *A. japonica* glass eels. However, Japanese glass eels catches increased four to five fold in 1999, to about 130 t. This is approaching the level estimated needed to meet aquaculture demands, given improvements in survival in transit and in Chinese production methods (see UK Glass Eel web site, www.glasseel.demon.co.uk). Demand from China was also reduced in 1999 due to local government import quotas (set at 150 t), imposed in attempts to combat overproduction. Catches in France increased by 50 - 70% over 1998-99 and these combinations of factors depressed prices as the season progressed and a number of dealers have gone out of business.

The export-import situation for English-Welsh catches is very confused for 1999. Some 22 t of elvers were imported early in the year from France and Spain for trans-shipment to the Far East, at a value of about £150 kg^{-1}. The estimated British catch was 18.9 t, worth on average about £122 kg^{-1} (as shown in Fig. 2.3b), although prices fell during the season to below £60 kg^{-1}. However, some later monthly exports to Germany, Italy, Belgium and Sweden had values of between £10 – 32 kg^{-1}, implying these, even if only in part, comprised residual elvers being sold cheaply. If all these exports represented home-caught eels, the total catch could have been as large as 50 - 60 t. This appears an unlikely increase over the estimated catch of 18.3 t in 1998. If European catches had increased by up to 70%, as claimed by UK Glass Eels, then British catches could have been as high as 30 t.

Given these provisos for 1999, Fig. 2.3a suggests that recruitment has stabilised during the 1990s. Increased fishing effort could have influenced the number of dip net licences sold per year (Fig. 2.3c), correlating with the rise in total export catch values (0.692, P = 0.001) and value per kg (0.610, P = 0.005). This assumes that there was a direct correlation between prices paid to fishermen and export prices, which include dealer's profit-margins, transport costs, etc. However, between 1979 and 1998 there were no significant correlations between total catches, numbers of dip net licences sold and CPUE as kg per instrument. Thus, as is suggested by the Den Oever fishery-independent time series, recruitment has indeed stabilised, with a possible increase in the 1998-9 season.

In relation to the 1999-2000 season, the Chinese import quota has been reduced to 70 t and although Japanese glass eel catches have fallen by some 30% compared to 1999 and early French catches were reasonable, prices were relatively low (about Fr1000 kg^{-1}). In England, prices in the spring 2000 season started at about £35 kg^{-1}, later averaging about £60 kg^{-1}, but rising as high as £180 kg^{-1} on the later spring tides in May. These prices reflect relatively poor catches, however, because the wet spring caused high river flows and levels and difficult fishing conditions. It is believed that the higher prices were designed to encourage a late effort to make up for the low catches earlier in the season, but that few glass eels were actually caught (Alan Starkie, Environment Agency, personal communication). Licence sales were about 20% less than in the previous year.

Concerns still remain, however, that overfishing of some ecosystems has and still may be occurring and that the precautionary principle needs to be applied. These issues are discussed further in later chapters, but they also will be considered during the review of stocks in specific systems in England and Wales.

2.2.2 Yellow and silver eel fishery data for England and Wales

Export data have been used to estimate catches and their values for comparison with catch estimates, licence sales and catch return data collated by MAFF and submitted to EIFAC (Knights *et al.*, 1999). The catch estimates shown in Fig. 2.4a follow the same trends but are not significantly correlated. Examination of the MAFF estimates Region-by-Region show the apparent relationships occur by chance due to lack of data or single high estimates in different Regions in different years. No MAFF estimate for 1988 is shown in Fig. 2.4a because the data set includes an exceptional outlier value of 213.19 t for South West/Wessex, >10 times the size of any other Regional estimate. Export estimates are generally about five to seven times the total annual ones made by MAFF, implying they are generally underestimated.

Interpretation of raw export data is complicated by apparently erroneous data entries (e.g. occasional excessively high or low tonnages and values in £ per tonne per shipment), mixed elver/yellow eel shipment records, etc., as was discussed earlier in relation to estimating glass eel catches. Another problem is that the export data are for the whole of the UK. To account for the major Lough Neagh fishery in N. Ireland, any exports via Belfast Airport and/or annual catch data (derived from Rossell, 1999) were subtracted from the UK totals. The remaining figures possibly still include some catches made in N. Ireland (e.g. from Lough Erne). Key eel merchants estimate English and Welsh exports are only about 150 t year^{-1} compared to 402 ± 117 and 66 ± 24 t year^{-1} from Customs and Excise and MAFF data respectively. Export figures do not include catches sold locally to the London jellying market, smokers, shops and hotels and eels consumed by fishermen. McKinnon & Potter (1993) estimated that 34% of the 1992 catch went to local processors or Billingsgate market, 60% went to merchants. Many eels are also taken by road direct to the Continent via the Channel ports and Channel Tunnel and may escape recording as exports. However, eel catch estimates for 1992 from questionnaire survey and export data were 368.8 t and 433 t respectively (McKinnon and Potter, 1993), compared to the Authors export estimate of 234 t and that of 35.63 t in data collected by MAFF. This supports the validity of using export data in estimating and tracking temporal trends in commercial catches.

Fig. 2.4 shows that catches of yellow and silver eels have fluctuated in Britain since 1980, with increases in some periods but with no overall long-term major decline. Some fluctuations may relate to past patterns of glass eel recruitment, but the picture is a confusing one. For example, catches increased in 1987-89, 1991 and 1995-97, each following lags of about 13 - 19 years after the peak recruitment years of 1974 and 1978-79 and, to a lesser extent, 1991 (Fig. 2.3). Troughs in recruitment and catches may also be separated by the

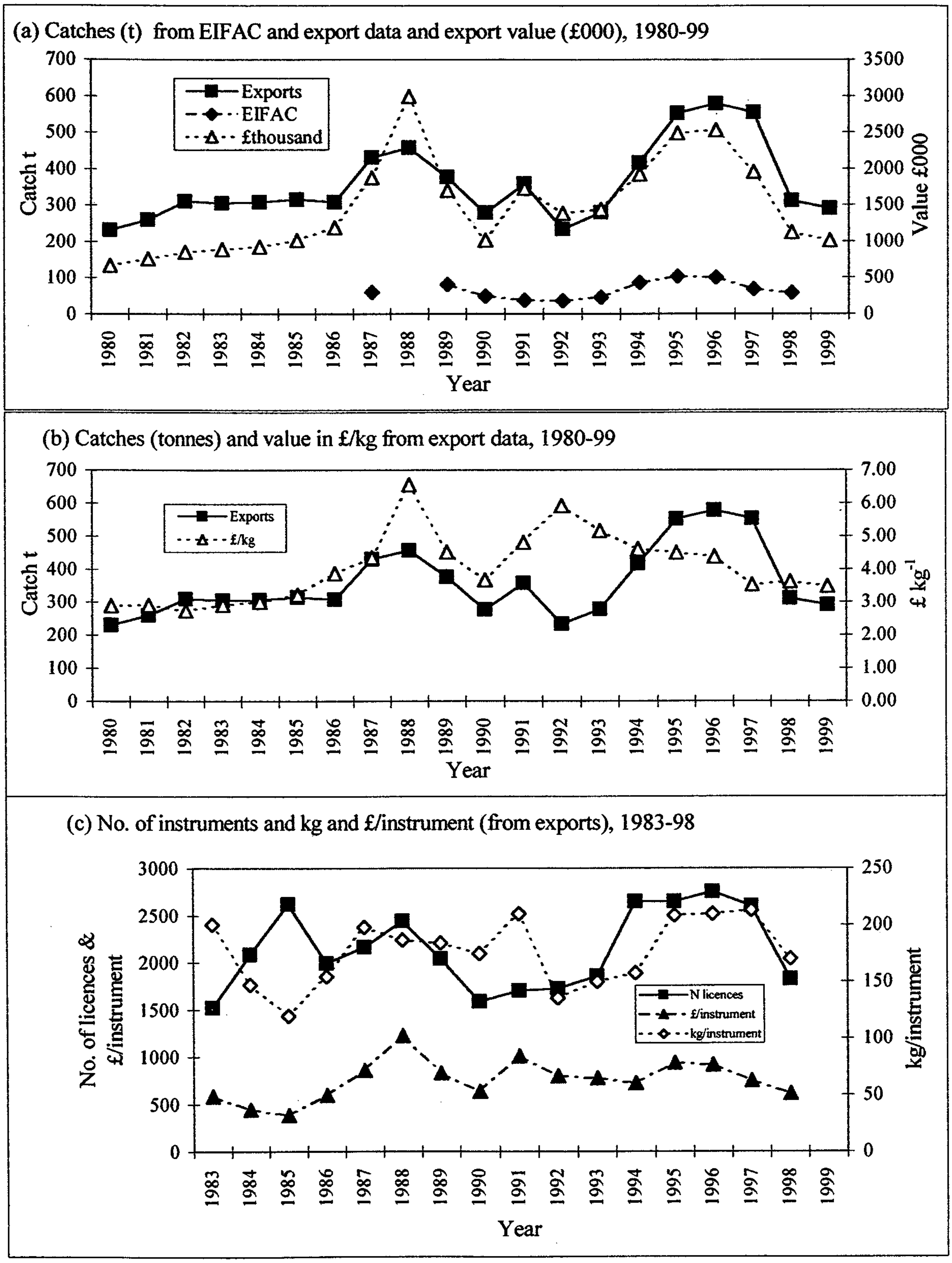


Fig. 2.4 Catch and effort data for yellow and silver eel fisheries in England and Wales, 1980-98 (see text for further explanation)

same order of years. This range of years typically covers the span of the growth stages of larger female eels in many British waters (e.g. see Chapter 3 and Knights, 1999b). Moriarty and Dekker (1997) quote a similar figure for Europe of a 10 to 20 year lag, covering the time differences to maturation in males and slower-growing females. Feunteun (personal communication) has developed a model which estimates population changes with time following a unit of glass eel recruitment and assuming instantaneous mortality rates (Z) of 0.17 and 0.65. This showed lags of 7 and 14 years respectively between changes in recruitment and adult stock levels. Rossell (1999) has found a statistically significant lag of 16-17 years between glass eel catches in the R. Bann (N. Ireland) and their seeding into Lough Neagh (plus some supplementary stockings) and the subsequent catch tonnages of yellow and silver eels. Total catch weights would have been biased by silver eels caught in the traps downstream of the Lough, because of their relatively high individual weight compared to yellow eels caught by long-lining. It was not possible to show any similar significant relationships for Britain as a whole because glass eel catch data only exist from 1972 and thus too few data points are available for meaningful statistical analysis. For example, a lag of 17 years is only covered by 10 data points, compared to Rossell's (1999) set of 19 data points from his 1959 - 1999 set. The British yellow/silver eel catches in the late 1990s have also possibly been strongly affected by market factors, as discussed further below.

Although catches might be expected to relate to stock availability, possible influences of increased fishing effort in response to price fluctuations need to be taken into account, on the assumption that prices paid to fishermen correlate with export prices. Peak export tonnages and total export values are significantly correlated with effort in terms of the number of instrument licences sold ($P = 0.001$) but not, however, with CPUE £/instrument or £ kg^{-1}. Peaks in export prices per kg in 1987-89 and 1991-93 could reflect lack of supplies due to the poor stock levels and catches in the traditional European mainland fisheries (Moriarty, 1996; Moriarty & Dekker, 1997). These could have encouraged greater individual fishing effort and hence catches in Britain. However, only 18% and 26% of 101 eel fishermen questioned in 1992 by McKinnon and Potter (1993) claimed to have increased the amount of time or area they fished respectively, 31% and14% respectively claimed decreases. Furthermore, 56% said catches had not changed significantly since 1987, 22% said they had increased and only 18% said they had decreased. This may reflect local variations in recruitment and stocks, but overall, catches and effort did not decline between 1980 and 1997, although values in £ kg^{-1} have declined steadily since 1992.

The post-1992 declines in unit price may have been caused by the increasing output of eels from warm-water farms, especially in Scandinavia and the Netherlands. The situation has worsened in recent years because of overproduction by Chinese eel farms and dumping of cheap eels in European markets. The 1999 catch appears to be particularly low, at about 290 t. Even this figure may be inflated by inclusion of some cheap exports of glass eels, as discussed in the preceding Chapter. The export value shown in Fig. 2.4b of about £3.40 kg^{-1} is also possibly inflated. Some export records show prices < £1.00 kg^{-1} and market prices quoted by fishermen and dealers later in 1999 have been around £2.00 for even good quality silver eels – and difficulties have been experienced at times in finding any buyers. Fresh, chilled and frozen eels continue to be imported at <£2.00 – 3.00 kg^{-1} from New Zealand and N. America, as in the past, but now with cheap eels also coming from Ireland and the Far East. Given these confusions, it is still clear that values kg^{-1} over the last three years are similar to those pertaining in the early 1980s, i.e. allowing for inflation, they have declined markedly in real terms.

The downward trend in CPUE in kg instrument^{-1} since 1997 is possibly due to the cumulative impacts of the declines in glass eel recruitment in the late 1970s-early 1980s. The impacts may be more severe in some parts of England and Wales than others, depending on geographical differences in recruitment and past fishing mortality. No clear picture emerges from examination of Regional catch estimates made by MAFF, but such issues are considered further in later Chapters. Licence sales and total catches have also fallen drastically, but this is probably due to declines in prices in real terms than to declining stocks. Rossell (1999) states that such competition is a major threat to the future economic viability of the Lough Neagh fishery. This may lead to further self-regulation in England and Wales as fishermen reduce effort or stop fishing completely. Some may, however, try to make up catches by keeping smaller eels. This would, however, not be economically viable if efficient aquaculture production continues to supply eels at cheap prices of the larger sizes and fat contents preferred by the markets.

2.2.3 Conclusions regarding temporal trends in commercial eel catches in England and Wales

The data reviewed above are far from perfect, but the overall conclusions are that:-

- glass eel recruitment peaked in 1974 and 1978 (and to a lesser extent in 1992) but then declined by 75 – 90%, tending to stabilise in the 1990s. It is possible that glass eel catches have been maintained to some extent by increased effort but CPUEs have varied and have not reflected increased licence sales in recent years. The time- trends mirror those seen elsewhere in Europe, although declines appear to have been less severe and have occurred later than in more northerly Scandinavian and southerly Mediterranean areas (Moriarty, 1996; Moriarty & Dekker, 1997). This could well be because (as with the French Biscay fisheries) of the favourable position of Britain (especially south-western estuaries) relative to Atlantic migration pathways.

- there has not been a major nation-wide decline in yellow or silver eel commercial fishery catches between 1980 and 1997 due to lack of recruitment or overfishing. Reported catches and exports have fluctuated but peaked in 1988, 1991 and 1995-97. This may reflect peaks in catchable large eels some 13-19 years after each of the peaks in glass eel recruitment of 1974 and 1978 and 1982. This time lag approximates to the length of the growth stages of female eels in many British waters. However, the peak catches coincided with years when licence sales and effort were relatively high because of (at least in 1988 and 1991) good prices.

- export tonnages, licence sales and CPUE in kg/instrument were particularly high in 1995-97, despite declining catch values in £ kg^{-1} (due to competition from cheaper farmed eels).

- catches have fallen since 1997, possibly due to the cumulative effects of the poor glass eel recruitment since the late 1970s-early 1980s and the loss by emigration of slower-growing and larger female eels. However, market factors have probably had the greater impact. The price of yellow and silver eels in real terms has fallen steadily since 1992 due to competition from farmed eels, the situation being exacerbated recently by overproduction in China and dumping of cheap eels in Europe. Falling prices correlate with the declines in overall fishing effort as measured by licence sales.

- glass eel and yellow/silver eel catches, values, licence sales and CPUEs need careful monitoring to inform management. This is especially important over the next few years when changes in stocks and fisheries might occur due to the long-term effects of recruitment failures or market forces or a combination of both. MAFF and catch return data are of poor quality, but export data appear to offer a useful means of tracking catch trends over time.

Monitoring is discussed further in Chapter 8, the following sections review possible historical changes in stocks in individual waters moving in an anti-clockwise direction around England and Wales from the Severn (resurveys in the Severn are considered in detail in Chapter 3). This helps reflect geographic variations in habitat types and productivities and distances of catchments from the Atlantic migration pathways, as well as differences in fishery exploitation (i.e. between glass eel fisheries in the Bristol Channel/Severn Estuary area and yellow-silver eel fisheries in Southern and Anglian Regions).

2.3 Estuarine Eel Data from Hinkley Point B Power Station Screen Catches (1981-97)

There is little information available on eel stocks in coastal waters and estuaries. However, eels are trapped along with other fish on power station intake screens and this is potentially a useful fishery-independent means of sampling estuarine and coastal eels. Catches are, however, dependent on siting of intakes, pump throughputs and screen mesh sizes and operational characteristics and on tidal and seasonal influences (e.g. see Henderson, 1989). A number of power station catches have been studied by the former CEGB and other organisations, but only the time series for Hinkley Point B and West Thurrock (Thames Estuary) cover more than a few years. The latter is discussed in detail in comparison to other data for the Thames in Section 2.2.2.

The data set for Hinkley Point B covers 16 years, from 1981 to 1997, the years of major change in recruitment according to Section 2.2.2. It was purchased from PISCES Conservation Ltd. (IRC House, The Square, Pennington, Lymington, Hants SO41 9GN). Hinkley Point B is a base-load nuclear power station situated in Bridgwater Bay in the outer Bristol Channel, east of the R. Parrett, at 51° 20'N, 3° 15'W. Sampling has been standardised to collections on two screens over a tidal cycle for 6 h (equivalent to 3.24 x 10 m gallons of intake water) on one day per month per year. The data should, in theory, provide information of relevance to coastal/estuarine stocks for comparison to the stocks and fisheries of the Bristol Channel/Severn Estuary. However, only 269 eels were caught over 16 years (204 months), averaging 2.9 ± 5.2 month. Initial analyses of total annual catches against year suggested a marked decline in stocks since the early 1980s. However, Fig. 2.5a shows this to be an artefact due to four relatively very large catches of 30 in March 1981, 27 in March 1982, 32 in November 1982 and 10 in January 1984. The sizes of eels in these

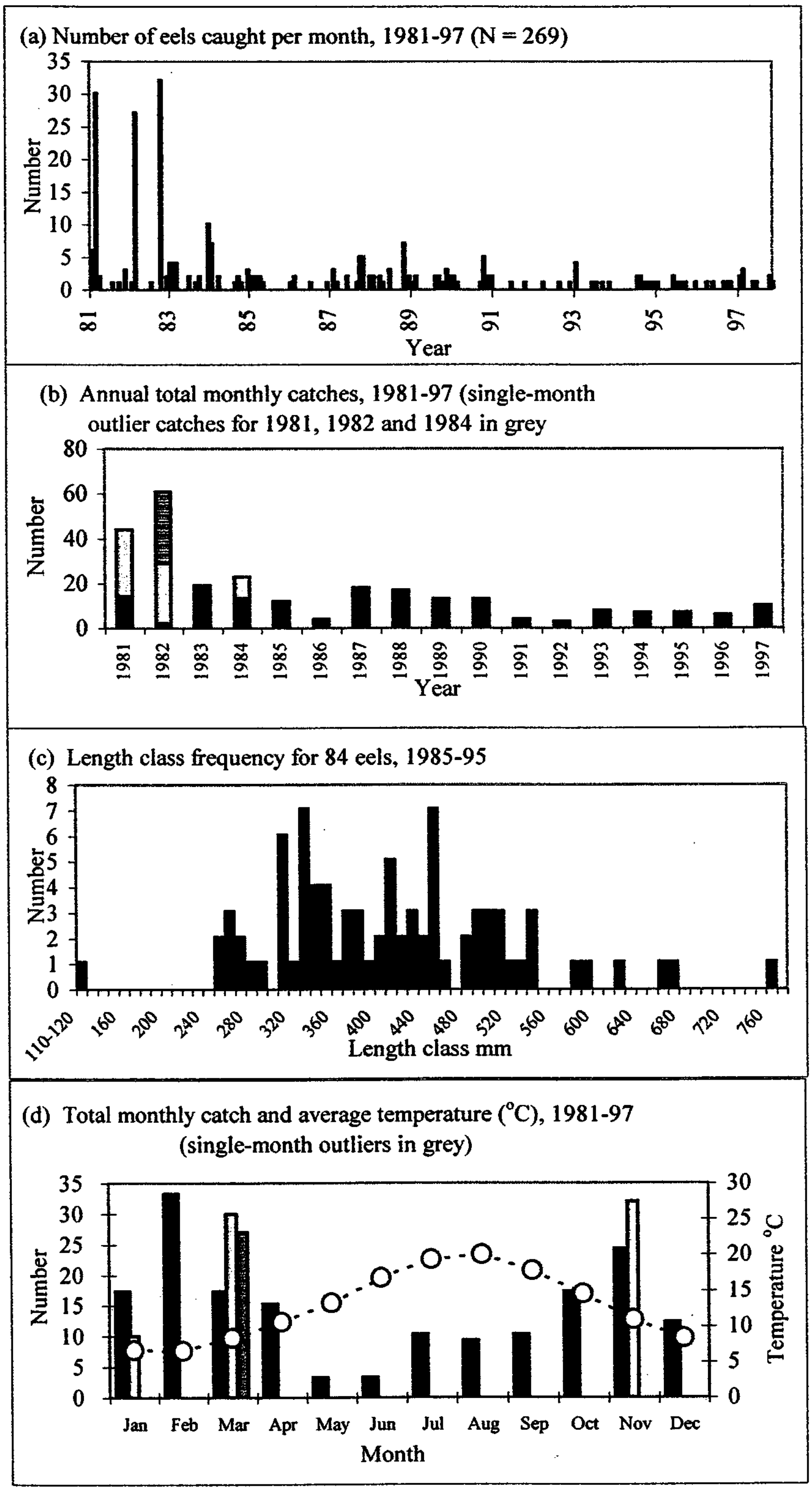


Fig. 2.5 Data from eel catches over 6 hours per month from two screens at Hinkley Point B Power Station, 1981-97

samples were not recorded and the reasons for these extreme outlier results is unknown. Overall, monthly sample sizes are too small for meaningful statistical analysis but if outliers are taken into account, Fig. 2.5b shows no obvious temporal trends in catches.

The screens might be hoped to sample immigrant glass eels and any resident juveniles, but from size determinations of 84 eels between 1985-95, only one elver/small juvenile was caught, all other eels were 250-760 mm (Fig. 2.5c). No data on silver eels were recorded but many of the eels were probably silver emigrants, the majority being over 35-40 cm with at least 37% being females >450 mm. Salinity was fairly constant over 16 years at 26.5 ± 2.3 ppt but Fig. 2.5d shows that catches tended to be larger at temperatures of $\leq 10°C$ in the late spring and autumn. Size of individuals also tended to be larger at these times and samples were probably mainly of migrants, rather than local resident eels.

In conclusion, the Hinkley Point B Power Station screen data do not reveal any obvious changes in coastal eel stocks in the outer Bristol Channel between 1981 and 1997.

2.4 Bristol Avon

The Bristol Avon discharges into the Bristol Channel and would be expected to receive good glass eel recruitment. Although runs have been noted, the estuary is unsuitable for commercial fishing because of soft, muddy and steep banks. Unfortunately, no suitable historical data sets for assessing time trends in eel stocks for comparison with the Severn are available. However, eel stocks are claimed to be large in the Upper Avon Catchment Management Plan (1994) and Lower Bristol Avon CMP (1995). Eels are said to be abundant up to the Chew Valley Lake dam on the R. Chew (+15 to +25 km from the tidal limit) and in the Frome to Frome (+49 km from the tidal limit). In the upper catchment, they are abundant to Trowbridge and Devizes, then decrease in frequency upstream of Chippenham (~ +80 km). Some eels also penetrate into the Kennett and Avon Canal (British Waterways, personal communication).

The above claims are partly borne out by limited data for eels for 1989 and 1990 included in the Fisheries Classification System database (see further discussion in later Chapters). Densities and biomasses were as high as 27.4 eels and 1552.5 g $100m^{-2}$ respectively in the R. Chew but in middle and upper catchments sites, the mean densities were < 1 eel $100m^{-2}$ at biomasses of ~ 400 g $100m^{-2}$. These population estimates are probably on the low side because of low sampling effort (e.g. 1 and 2 fishing runs only) and lack of focus on eels as a target species. However, they are similar to those obtained in routine fishery surveys at comparable distances from the tidal limits in the Severn (Chapter 3), despite the lack of any significant glass eel fisheries in the Avon.

2.5 N. Wessex Rivers

The N. Wessex area encompasses the catchments of the rivers (especially the Parrett, Tone, Brue and Kings Sedgemoor Drain) and wetlands of the Somerset Levels and the peatland valleys of the Somerset Moors. The relatively short rivers have Atlantic-facing estuaries, encouraging recruitment of glass eel. The shallow, warm and productive waters should support high productivities and yields, although problems are posed by migration barriers. Tidal barriers inhibit glass eel/elver immigration whilst closure of inland sluices in the warmer summer months to maintain inland water levels inhibits upstream migration of older juveniles (Knights, 1997). Loss or degradation of habitat due to drainage and agricultural developments have also affected aquatic habitats. Of particular concern, however, are the possible impacts of the glass eel fisheries based in the river estuaries, especially the tidal Parrett. Exact catch sizes are unknown but Morrice (1989) estimated the catch in 1982 at about 18 tonnes (~ 40% of the total for England and Wales). Dip net licence sales in the area are typically 20-25% of the total for England and Wales. There are, however, few fyke net or other eel fishing licences issued for the N. Wessex waters.

There are only a few sporadic eel stock records before 1991, widescale electric fishing and netting were not used extensively because their efficiency for capturing eels was believed to be too poor, especially in brackish conditions and the larger rivers and drains. Survey data for 1991-95 using modern electric fishing gear has been processed by Francis Farr Cox (EA Fisheries, Bridgewater, unpublished report). Eel lengths and weights were recorded but most surveys only involved one or two runs. Estimates of density and biomass are therefore largely based on actual catches and underestimate the true levels. No detailed density data have been made available.

Despite the lack of historical data, information on current population biomass and body sizes can be compared with other catchments to see if there are any abnormal patterns indicative of recruitment failure or overfishing. Biomass per site data are plotted in Fig. 2.6, using the Fisheries Classification Scheme (FCS) classes. Given that eel populations have probably been underestimated, this shows that stocks are good, with nearly 60% of the sites where eels occurred falling in the top two classes, against an expected 40% (Table 2.2). The sites where no eels were found were in upper reaches, due to the impacts of major migration barriers. High obstructions prevent recruitment above the dams of Sutton Bingham Reservoir in the Yeo catchment and Nutscale Reservoir in the Horner Water catchment. Declines in populations also occur upstream of high weirs at Lydford and Lovington in the upper River Brue. A barrier at the confluence with the Brue explains the fact that eels were only found in one site out of eight on the R. Alham (Knights, 1997).

Biomasses are relatively high in unobstructed upstream reaches of rivers such as the Brue, showing that eels can penetrate well inland in these short rivers. However, catches probably also reflect local variations in suitability for diurnal burrowing/hiding and in suitability for electric fishing. At most sites, high densities of eels were associated with high biomass, the exceptions possibly reflecting differences in local habitat and sampling efficiency. The Horner River (W. Somerset), for example, was unusual in having a low biomass relative to density compared to other rivers. This steep-gradient and fast-flowing river is, however, dominated by boulders and rocks, offering an ideal habitat and crevices for small eels but probably not for larger ones.

No impacts of the glass eel fisheries are obvious from the data. Fishing is concentrated on the tidal Parrett yet eel populations in the upper Parrett and its tributary, the Isle, are high relative to the Brue where exploitation is much lower. Furthermore, the tidal doors at Highbridge Clyce and other barriers probably affect immigration into the Brue. Thus physical obstructions to migration (and the availability of habitat) are probably at least as important as fishing mortality in determining the resident eel population. There appears, however, to have

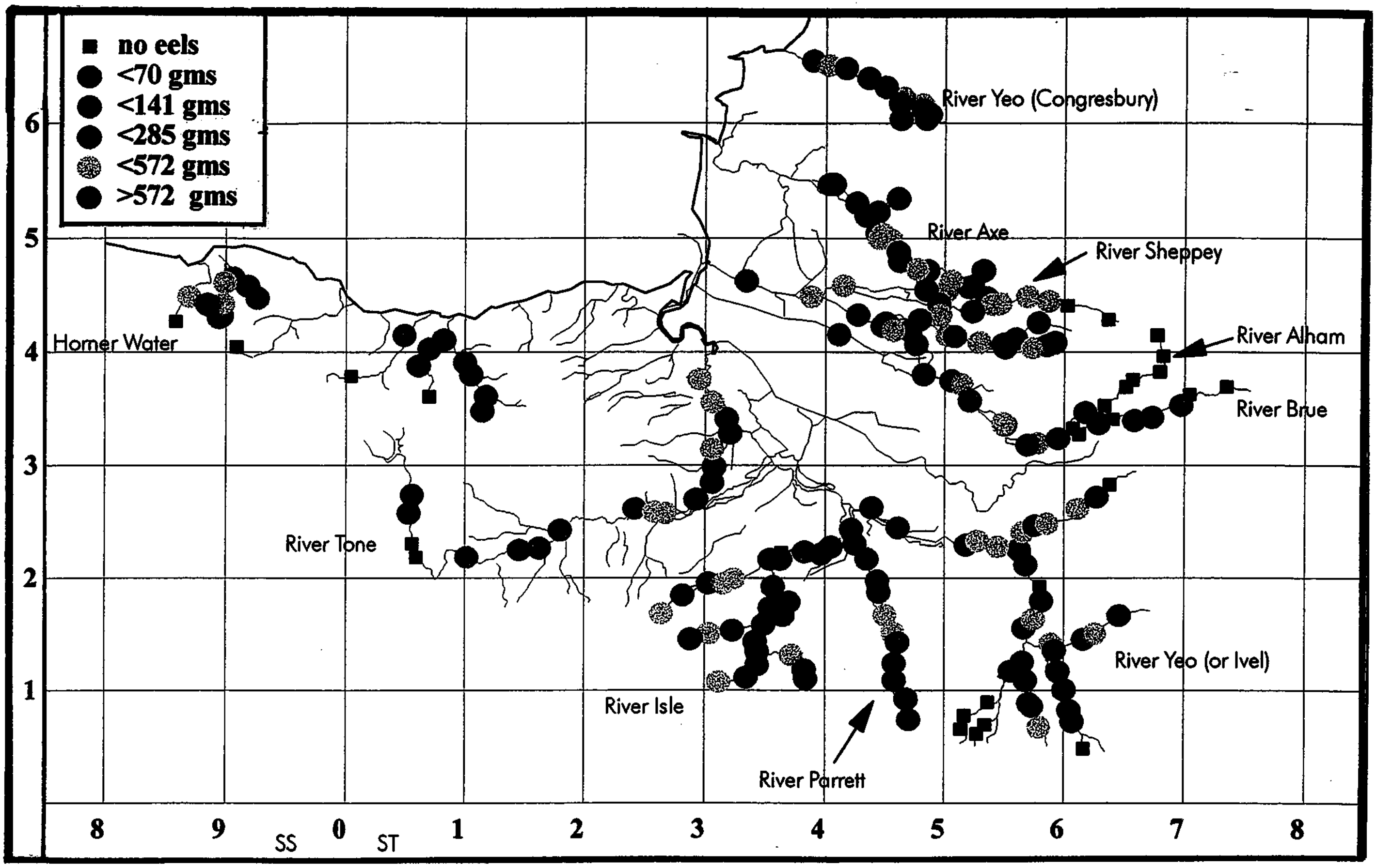


Fig. 2.6 Map of the North Wessex river catchments, showing relative distribution of eel biomass (Fisheries Classification Scheme biomass classes, Table 2.2) at different sites in the mid 1990s

Table 2.2 Biomass (minimum estimates) of eels in N. Wessex river sites (N = 208); number and percentages of sites falling into different Fisheries Classification Scheme (FCS) classes

FCS eel biomass class (g $100m^{-2}$)	Number of sites	Percentage of sites (N = 179) with eels present
No eels	29 (13.9% of total)	-
< 70	23	12.8%
<141	15	8.4%
<285	36	20.1%
<572	49	27.4%
>572	56	31.3%

been a marked fall in the mean density for 10 sites in the Isle between 1992 and 1997 surveys, from 11.2 to 5.3 eels $100m^{-2}$. The 1992 data were distorted by exceptionally large catches at three sites and it is suspected that a local elver dealer may have dumped excess stock into the river in that year. Populations vary between other rivers (e.g. the Yeo (Congresbury), Axe and Sheppey) but the reasons are unclear. Elvers have been seen to ascend the Yeo as far as the dam at Blagdon Reservoir. Silver eel traps have been run below the Reservoir since 1987 but eels cannot ascend the dam in any numbers and populations can only be maintained by stocking. No silver eel catches have been possible in drought years when there has been no overspill (Chris Klee, Bristol Water plc, personal communication).

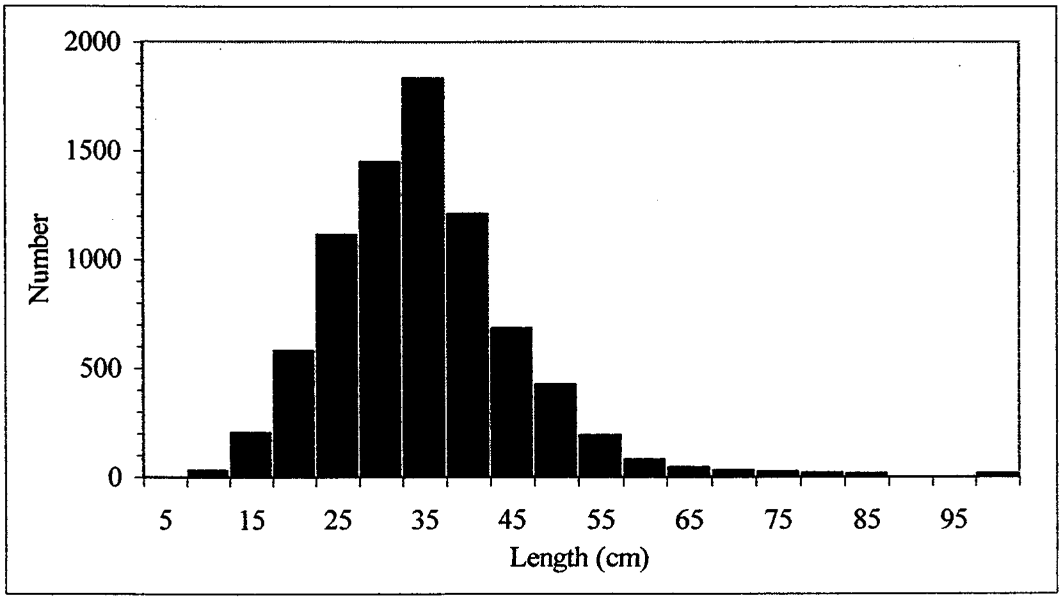


Fig. 2.7 Length-class frequency for N. Wessex eel samples, 1990-98 (N = 7733)

The evidence available indicates that stocks in the N. Wessex rivers have not been impacted by declines in recruitment and/or overfishing of glass eels. Average biomass and density are about 300-500 g and 5 – 10 eels 100m^{-2}. Given that populations have been underestimated and the impacts of tidal gates and other migration barriers, biomasses, densities and size-distributions are similar to the unexploited shorter rivers of Devon and Cornwall, discussed in succeeding Chapters. Average length and weight are 32.2 ± 9.8 cm (range 5.5 - 100 cm, N = 7733) and 81.1 ± 94.2 g (1.0 - 2700 g, N = 7645) in N. Wessex samples measured between 1990-8. Thus populations are dominated by smaller and earlier maturing males, as shown in (Fig. 2.7), with only 10.3% of the eels measuring > 45 cm. Larger eels were found sporadically, mainly in upper catchment sites. This picture is again similar to that seen in short rivers in Devon and Cornwall, given that upstream migration in the N. Wessex system is inhibited by numerous water level control structures. The picture contrast, however, with that for rivers in the east of the country, such as the Essex rivers (Section 2.23), where densities tend to be lower but biomasses relatively high because populations are dominated by larger (female) eels. The most obvious explanation is that the south-western rivers receive relatively more glass eel recruits direct from the Atlantic (Figs. 2.1 and 2.8). Even though overall recruitment may have declined since the late 1970s-early 1980s, numbers of immigrants probably exceed the natural carrying capacity of the shorter rivers and natural density-dependent mortality is probably very high in relation to fishery mortality. High densities in tidal and lower freshwater reaches then enhance tendencies to migrate upstream. Because the rivers are relatively short, densities still remain high and this in turn leads to a preponderance of smaller and earlier maturing males. These hypotheses are further discussed when reviewing population time trends and geographical differences in other waters in England and Wales.

2.6 Taw and Torridge Catchments

The common estuary mouth of the Rivers Torridge and Taw on the N. Devon coast face the Atlantic and might be expected to receive good glass eel recruitment (Fig. 2.8). Unfortunately, salmonids have been the target species for any quantitative studies. Anecdotal evidence suggests, however, that eel stocks used to be good but that relatively few glass eels or elvers have been seen in recent years. Some (very limited) subsidiary information on eels in the Torridge was collected during an R&D project on salmonid stocks in the late 1980s. Only presence-absence has been recorded in routine fishery surveys, showing eels are generally widespread in the catchments, except at more upstream and upland sites. Some relative abundances recorded in the Taw in 1993 show stocks in a core area of the main river (to about +20 km above the tidal limit) are 'abundant' (> 100 eels per site catch) but eels are only 'present' (1-10) through much of the upper catchment.

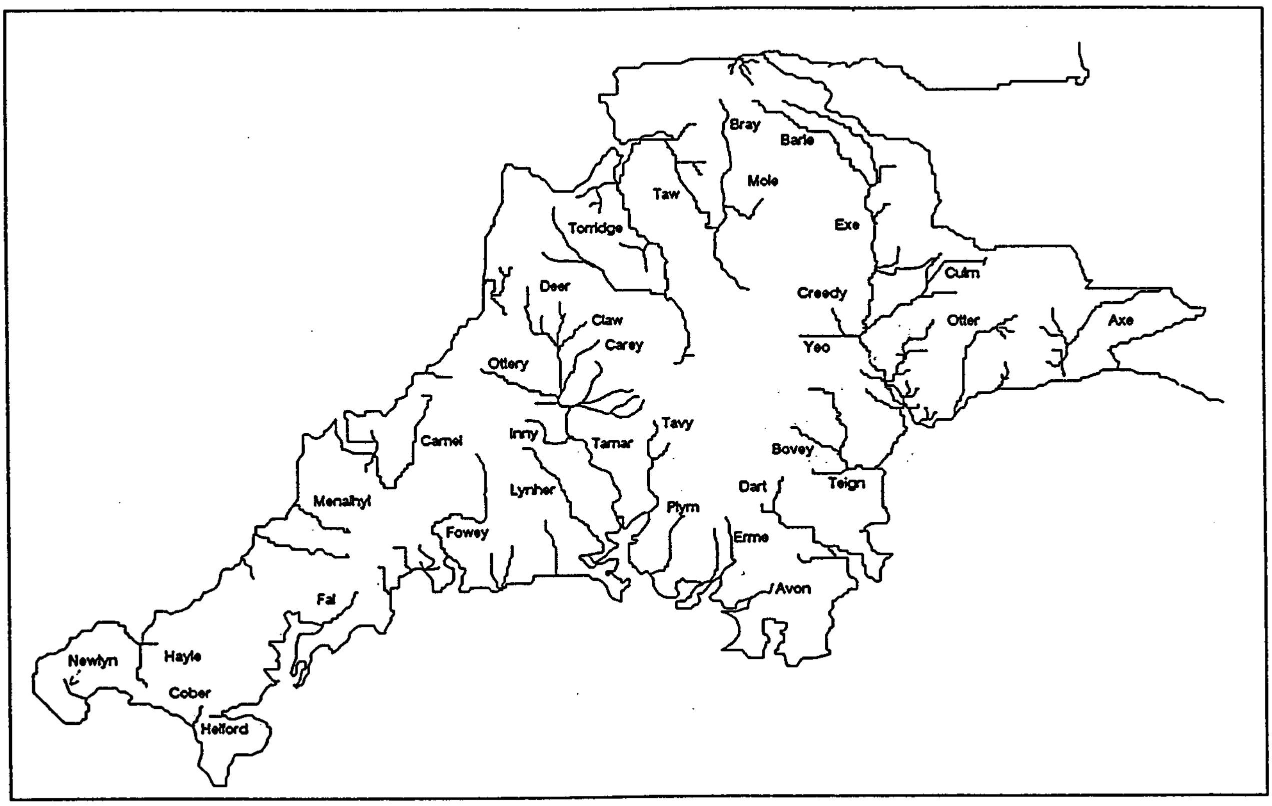


Fig. 2.8 Map of the main rivers of south-west England

Although there are no time-series records for actual density or biomass, some historical data on length class frequencies in samples from various sites in the Taw catchment are available for 1977-1991. Few sites can be matched over time, however, and some samples are very small, even low down the river, possibly reflecting the suitability of sites for eels or differences in sampling methods and efficiency. Little change in length class frequencies can be distinguished in these data sets. Two sets of site data can be combined, however, that are representative of lower and mid-to-upper river sites and which cover a good time span. The

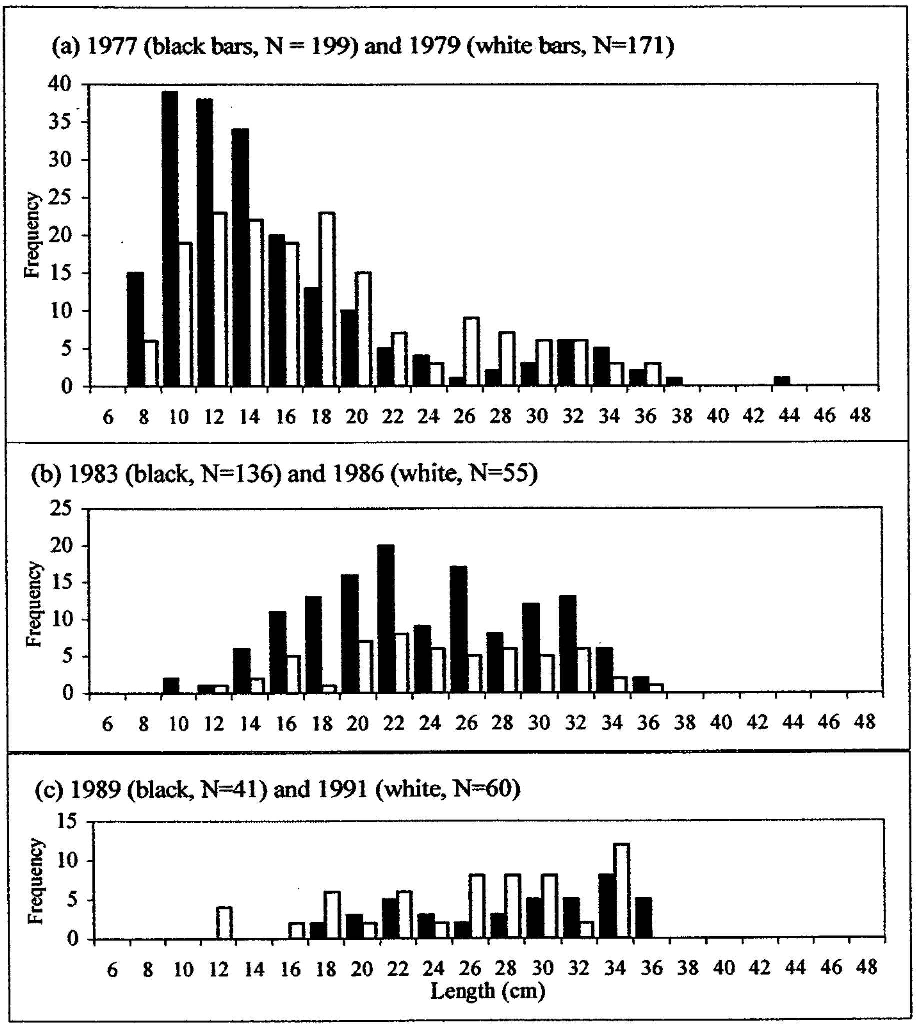


Fig. 2.9 Length class frequency data (and N) for combined samples of eels from the lower Taw catchment (Rock Bridge and Filleigh Bridge sites on the Bray tributary, approximately +15 km from the tidal limit) for 1977, 1979, 1983, 1986, 1989 and 1991

combined data for Rock Bridge and Filleigh Bridge (R. Bray, a lower tributary, at approximately +15 km) for six years between 1977 and 1991 are shown in Fig. 2.9. It appears that the number of eels per combined sample declined markedly after 1979, whilst average size increased, the mode shifting from 10 cm in 1977 to 22 cm in 1983 to 34 cm in 1989-91. This implies that recruitment was high in 1977 (and the immediately preceding years) and 1979 but declined thereafter, this agrees with the conclusions about nationwide trends reached in Section 2.2. The increases in average size in the 1980s and 1990s would accord with a relatively high initial density population at these low river sites, followed by changes as upstream migration occurred and as the population matured and eels emigrated as males at < 40 cm.

The other site samples combined (for 1979, 1983, 1988 and 1991) are those for N. Tawton and Bondleigh, at approximately + 45 km on the main river Taw (Fig. 2.10). Catch sizes are

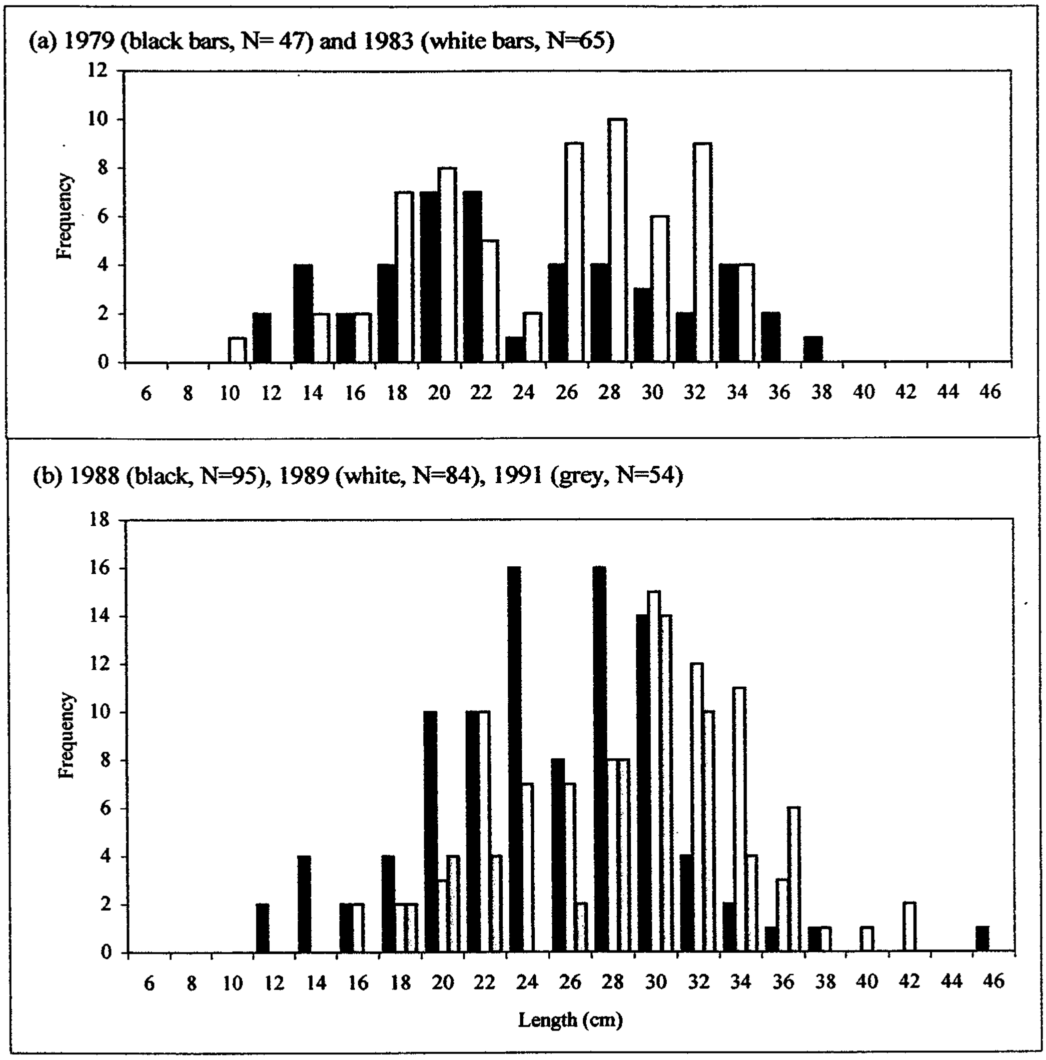


Fig. 2.10 Length class frequency data (and N) for combined samples of eels from the mid/upper Taw catchment (N. Tawton and Bondleigh sites on the Taw, approximately +45 km from the tidal limit) for 1979, 1983, 1988, 1989 and 1991

smaller and average sizes larger than those for the Bray, as would be expected for more upstream sites. The sample sizes did not change so much over time and increases in modal size were not so obvious, possibly due to enhancement of local populations by migration from the lower river reaches. Densities were still probably relatively high, there only being a few (female) eels > 40 cm. According to local fisheries officers, larger females are caught in some upland sites much further inland, but relatively sporadically.

Although the data available for the Taw and Torridge are not very robust, the derivations made suggest that recruitment may have declined since the late 1970s. It appears, however, that the impacts on more upstream populations may not have been great, certainly in comparison with catchments such as that of the Thames discussed later. This may be because the Taw and Torridge still receive reasonable glass eel runs because of their position, facing the Atlantic migration pathways. The large common estuary and areas of salt marsh may also act as habitat reservoirs for recruits. Neither river is commercially exploited to any significant extent.

2.7 Rivers of N. Devon and Cornwall

Although they are well positioned for recruitment from the Atlantic (Fig. 2.8), the short, steep, flashy and hard substrate rivers on this coast, west of the Taw and Torridge, do not provide very suitable habitats for eels. However, stocks can be quite large, e.g. the density of eels at the lowest survey site on the East Lyn in 1983 (Myrtleberry, +2.5 km) was 16.82 eels $100m^{-2}$ (mean length 26.68 ± 4.31 cm). In contrast, waterfalls and rapids inhibited migration such that densities fell to 0.81 $100m^{-2}$ at Rochford (+6 km), average size being 60.20 ± 4.53 cm. Eels were absent further upstream. A similar situation was apparent in the West Lyn at and above +6 km. The Lyn catchment has been surveyed more recently but only presence-absence of eels has been recorded.

Good sets of data for other SW rivers have been collected in the past by the former South West Water Authority. Unfortunately, resource restrictions in ensuing years and an emphasis on collecting quantitative data only for salmonids means that more recent electric fishing information on eels (and other coarse fish) is sparse and often only exists as relative abundance or presence-absence records. Changes in sites surveyed and reductions in effort (e.g. to single runs or dip netting) have also affected historic comparisons. Attempts have been made by the Authors to assess possible trends in eel stocks in south west England by comparing presence-absence data. This has proved fruitless because either (a) absences are due to major permanent barriers, such as those described for the Lyn, or (b) capture of an occasional eel in any one year in upper catchments can be largely a matter of chance.

The best historic data sets for Devon and Cornwall are reviewed in the following Sections.

2.8 The Fowey Catchment (1977-87)

Extensive electric fishing surveys were carried out in the Fowey catchment during 1977-87, spanning the period of major declines in glass eel recruitment (Fig. 2.3). The studies were carried out to assess possible impacts (with the primary emphasis on salmonids) of the planned Colliford Reservoir scheme. Incidentally-collected data on eel density and biomass have been obtained from H. Sambrook (South West Water plc). Average lengths have been derived from average weight (estimated from unit biomass ÷ unit density) via the relationship $L = \text{antilog}(\log W + 2.8447) \div 3.0377$, determined from measurements on 392 Fowey eels. Many sites were surveyed but there are relatively few continuous data sets for individual sites over the 10 years. There are also no matching data sets for the 1990s to make longer-term comparisons.

The most continuous records are for the density of eels at two sites on the St. Neot River, below the present Colliford Reservoir. Site S8 (Pengelley, +15.6 km) is nearest to the confluence with the Fowey, site S5 (St. Neot) is +19.1 km from the tidal limit. Histograms (Fig. 2.11a) reveal variations in density over 10 years, but no evidence of any clear trends, except perhaps for a slight increase between 1980 and 1983. This might reflect the high levels of recruitment in preceding years (Section 2.2). The peaks in 1983 are, however, biased because they comprise relatively more September-October samples than other years, and catches were always largest in these months (see below). The average size of eels was about 19 and 23 cm respectively, i.e. these higher density populations were probably dominated by males, as would be predicted.

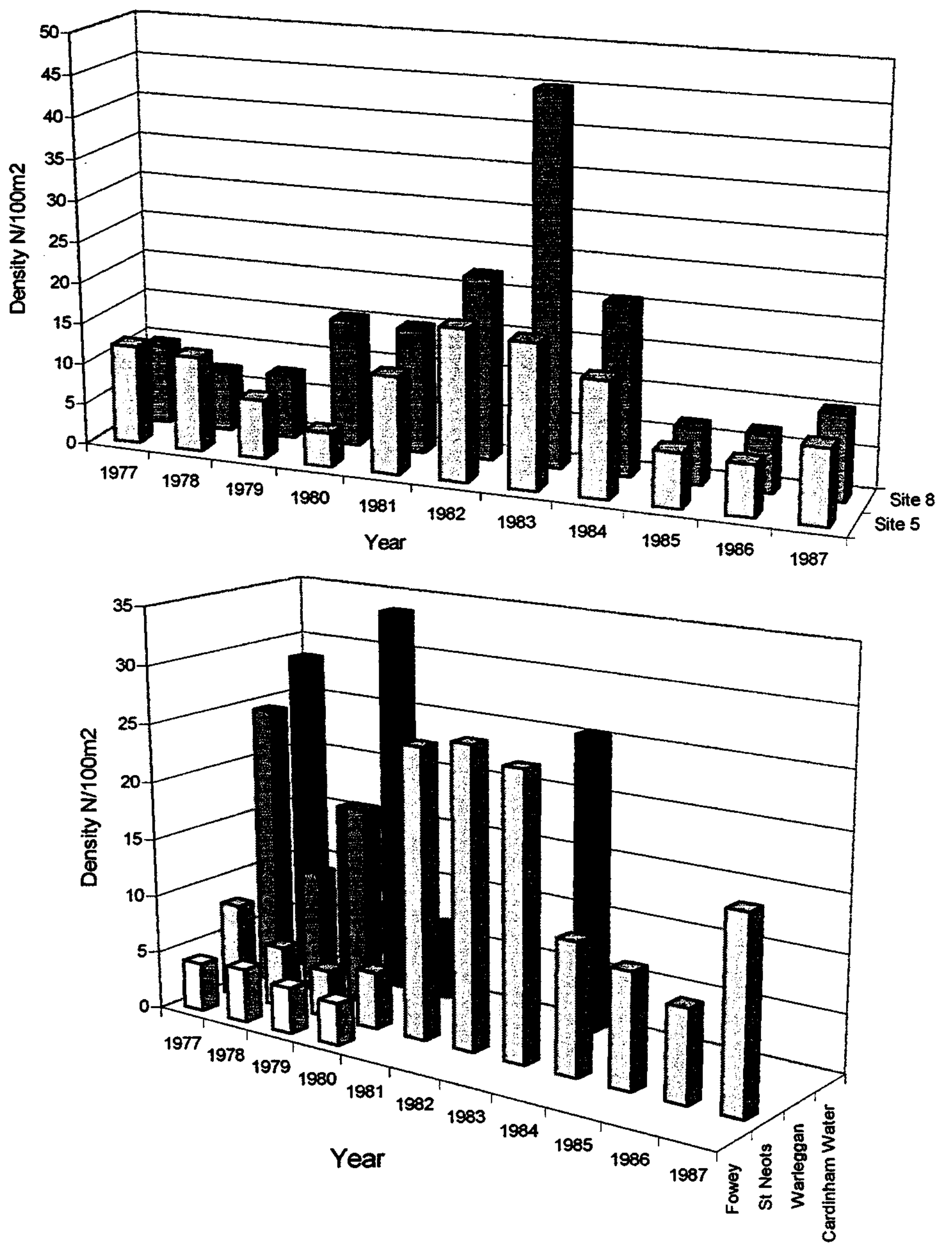


Fig. 2.11 Mean annual density (eels 100m^{-2}) between 1977-1987 for (a) two matching sites on the St. Neot River (Site 5, St. Neot, +19 km from the tidal limit, and 8, Pengelley, +15.6 km) and (b) for all matching sites on the Fowey and tributaries

The mean annual density for combined data from the other best matching sites per year between 1977-87 in the upper Fowey and in the St. Neot, Warleggan and Cardinham tributaries are shown in Fig. 2.11(b). Again, there is a suggestion of some increase in the St. Neot in the early 1980s, but the rest of the data sets are too short or intermittent to reveal any trends.

Overall, despite spanning the period of declining recruitment from the peaks of 1978-79, the 10 years of survey data do not reveal any clear trends. However, the extent of information on Fowey populations and their intra-catchment distribution make them worthy of further discussion for comparison with catchments discussed in subsequent Sections. Such baseline data could be useful in setting stock and escapement reference points for managing stocks and fisheries (see Chapter 7). Also, the catchment could be considered a suitable candidate for future monitoring (see Section 8.4.6).

Remembering that multi-species surveys would have produced underestimates, mean densities and biomasses per site were relatively high at downstream sites (15-20 eels 100m^{-2} and 100-300 g 100m^{-2}) (Fig. 2.12a,b). Density decreased with distance upstream to about +25 km but biomass was maintained relatively more constant (density-biomass correlation, excluding September outliers (see below), R^2 = 0.342). This occurred because average individual size increased with distance (Fig. 2.12c). Densities then decreased markedly above +25 km, to less than 1 eel 100m^{-2} and biomasses reached minimum values and lengths maximum values. Figs. 2.12d,e and f show these changes were related to major changes in altitude of sites, where the rivers rise steeply up on to Dartmoor and where habitat productivities would be relatively much lower. However, waterfalls also interfere with

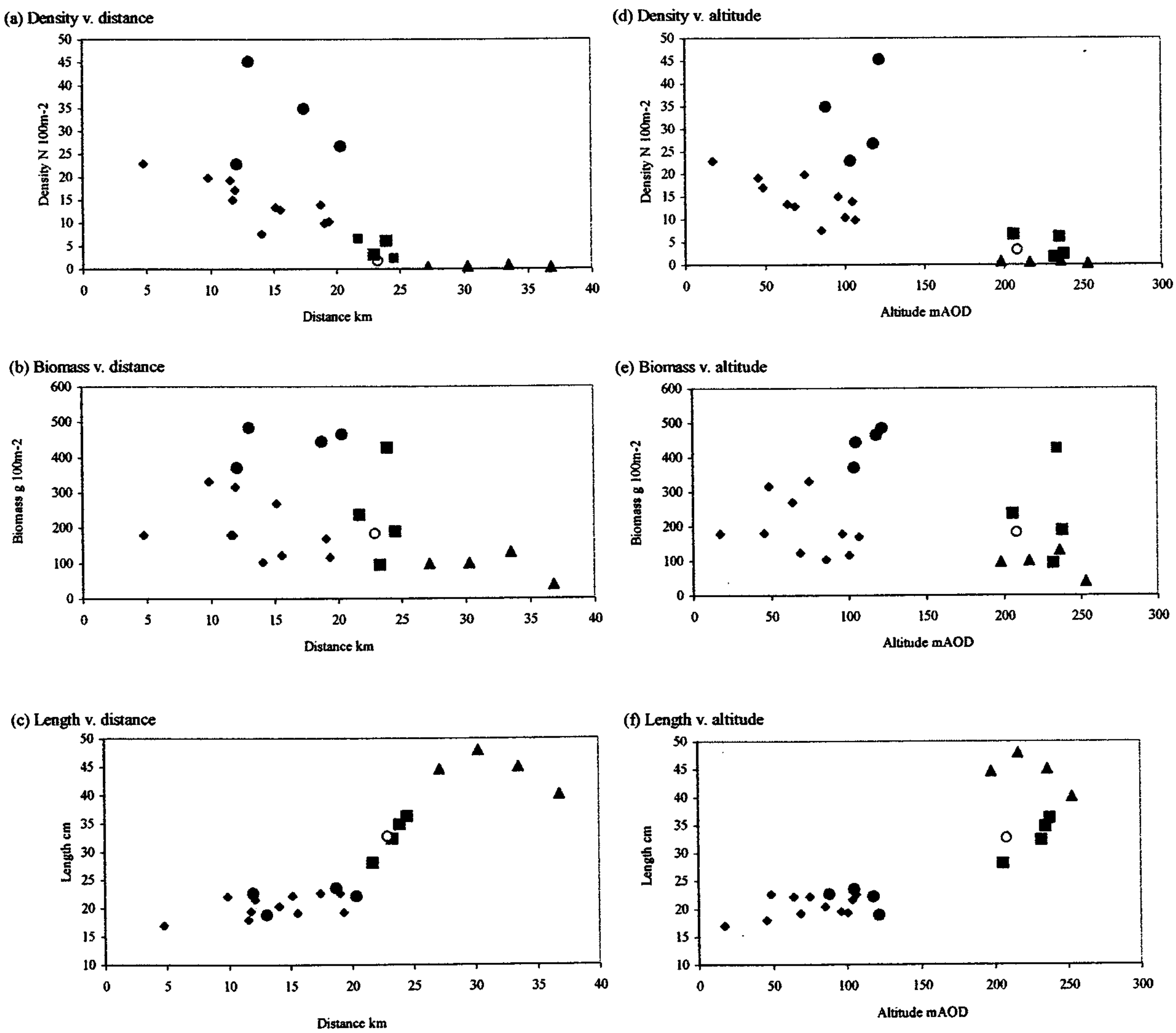


Fig. 2.12 Relationships between mean density, biomass and length per site and distance from the tidal limit and altitude for Fowey catchment sites, 1977-83 (triangles = Fowey sites above Golitha Falls; squares = St. Neot sites above Colliford Falls; closed circles = September samples; open circles = Trenant site above falls)

migration, especially above Treverbyn Bridge-Golitha Falls on the Fowey and, prior to construction of Colliford Dam, above Colliford Falls on the St. Neot. Catches in downstream sites were occasionally very large, especially in September (outliers shown as closed circles in Fig. 2.12), as discussed further below. Females > 40 cm dominated in the low density upstream sites but catches were often very variable, due to the occasional chance captures of isolated individuals. The impacts on eel stocks of dam construction and the impoundment of Colliford Reservoir in 1983 are unknown.

An important feature of the survey data was that there were often very marked seasonal differences in catches at the same sites. This is illustrated in Fig. 2.13, which shows data from electric fishing surveys over 1977-80 in matching sites in downstream zones (< +20 km) and upstream sites in the St. Neot River. Densities are relatively higher but biomass and average individual size smaller at the lower sites. The patterns of seasonal change agree with those found in the small number of published studies embracing a wide range of months, e.g. East Stoke Mill Stream on the Dorset Stour (Ibbotson *et al.*, 1994), Rivers Brett and Chelmer, Essex (Barak & Mason, 1992) and the River Tweed, E. Scotland (Mills & Hussain, 1985).

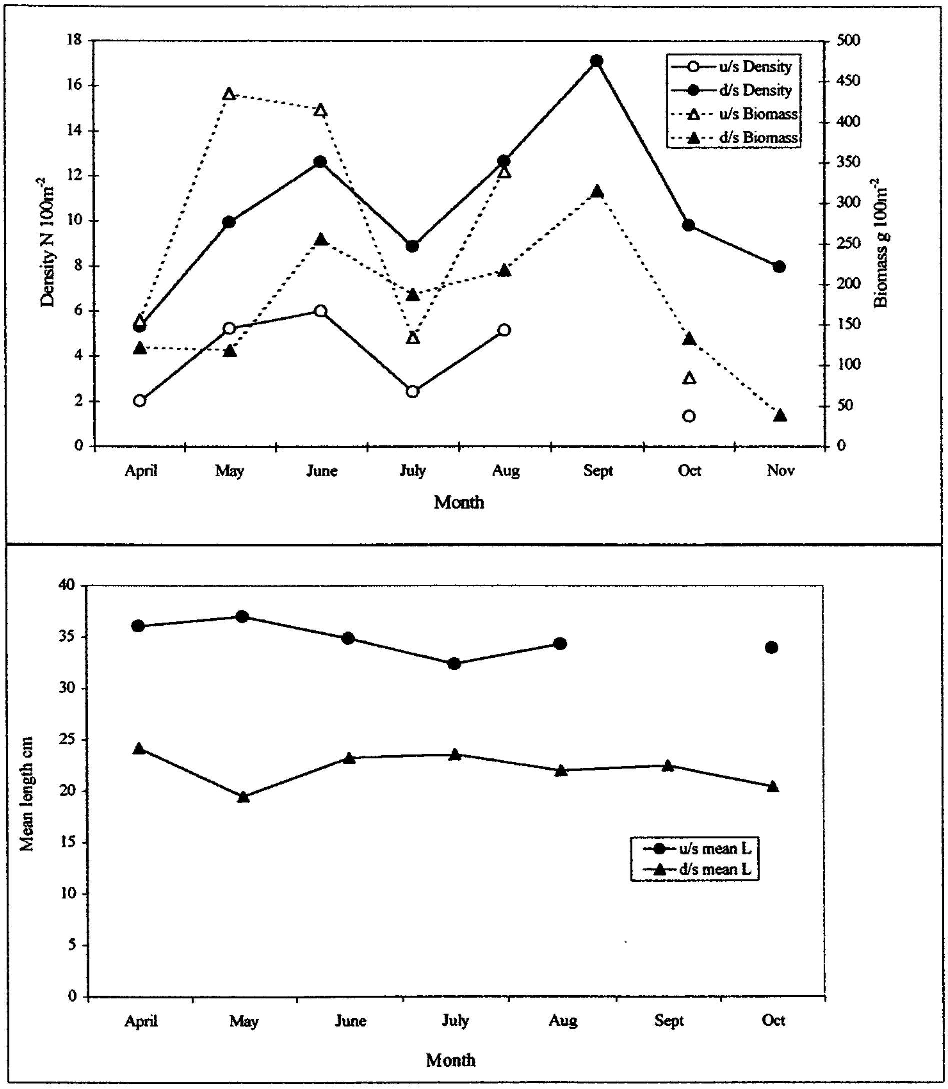


Fig. 2.13 Seasonal changes in mean density, biomass and mean length (from biomass ÷density) in the St. Neot River between 1977-80, comparing combined data for matching upstream sites (10 sites > +25km) and downstream sites (6 sites < +25 km)

Spring increases are due to movement of eels from deeper water overwintering areas, those in late summer-early autumn to peaks in upstream migration and activity levels, plus silver eel emigration. Numbers and rates of migration are particularly sensitive to temperatures > 14°C (White & Knights, 1998). Variations tend to be most extreme at low river sites because of migration of elvers and small yellow eels from estuarine and coastal waters. For example, total catches of immigrant elvers in traps at the tidal limit of the Severn varied from 4006 in 1991 to 33,539 in 1992 to 12,778 in 1993 (White & Knights, 1994). Juvenile eel catches also varied greatly, between 9841, 25,329 and 1705, over the same years. Even at upstream sites, annual variations were large, e.g. trap catches of yellow eels up to 30 cm at Lincombe Weir (+42.5 km) varied between 22,875 and 373 over the three years of study.

Such natural variability needs to be taken into account in interpretation of historical survey data and in future monitoring, as further discussed and emphasised in later Chapters of this report.

As an overall conclusion for this section, it appears that the Fowey system did not suffer any major changes in stock levels between 1977 and 1987, despite the possible nationwide declines in glass eel recruitment after 1978-79 discussed in Section 2.2. Recruitment appeared sufficient to maintain stocks at relatively high levels and probably near to the carrying capacity, at least in the lower parts of the system (below waterfall barriers), comprising about 80 km of river habitat.

2.9 River Tamar

The Tamar and a number of other rivers discharge into the Tamar Estuary/Plymouth Sound and would be expected to share glass eel recruitment and exchanges with populations in the Sound. Some historical quantitative data were located for the Tamar and Tavy, but not for the Tiddy, Lynher or Plym.

Cornish River Board information shows that eels were culled and elvers were caught on the Tamar in the early 1960s. Elver catches were '½ t' in 1961, '274 lb' in 1966 and '2 cwt + 672 lb' in 1968. No elver run was observed in 1969, '2 cwt' were caught in 1970 but no noticeable runs have been reported in recent years. A data set for density and biomass (plus derivations of average length per sample) was located for 35 Tamar catchment sites surveyed in 1978. Subsequent surveys have, however, only recorded presence-absence. In 1978, density was 10.12, 22.32 and 27.84 eels $100m^{-2}$ at the three lowest sites below +25 km but then fell to < 6 $100m^{-2}$ upstream. The lowest values < 1 (and absences) were found in the upper reaches of the Kensey and Ottery (draining from Bodmin Moor) and Thrushel and Wolf (draining from Dartmoor), declines being less marked in the main river than in these tributaries. These declines would be expected because of the rapid increases in slope, flashiness and hardness of substrates on the slopes of the moor uplands, plus associated declines in productivity. Biomasses varied between 6 and 408 g $100m^{-2}$ and eels contributed means of 1.96% (range 0.65-9.21%) and 9.38% (0.97-36.24%) to overall fish density and biomass.

Only presence-absence comparisons can be made with the most recent survey data. In 1978, eels were found in 31 out of 35 sites (88.6%) whereas in 1995, they were only found in 24 out of 58 sites (41.4%). The 1995 survey, however, focussed on tributaries upstream of those surveyed in 1978 and only five approximate matches between sites could be found. Eels were found in 1997 at 14 out of 15 sites on the Wolf compared to four out of six in 1978. In the Lynher and Tiddy combined, seven out of 29 sites yielded eels in 1997, the Plym, 24 out of 30. None of these results are conclusive but suggest there have been no major declines in stock distributions between 1978 and the late 1990s.

2.10 River Tavy

Fisheries surveys were carried out in 1983 in relation to plans for water transfers as part of the Roadford Reservoir scheme. Densities, biomass and lengths of eels were recorded at 35 sites. A survey in 1998 covered 51 sites, these include eel biomass for 22 sites that match the 1983 survey, plus length data for 35 sites. Unfortunately, 15 of these only used single-run electric fishing and two were only dip-net surveys, therefore 77% of matching sites in 1998 have only yielded minimum estimates of biomass. Fig. 2.14 shows some biomasses were very high at downstream sites in 1978 but that there may have been a general decline over 15 years. The mean per site has fallen from 667.4 ± 732.7 to185.1 ± 37.3 g $100m^{-2}$ (range 29.6-1638.8 and 30-744 respectively). This change is statistically significant ($P < 0.01$), but is probably largely due to lower fishing effort and sampling efficiency in 1998. Some lower main river and Lupp tributary catches in 1978 were also exceptionally large, three yielding estimated densities of 116, 200 and 114 eels $100m^{-2}$ and biomasses of 1638, 2744 and 3362 g $100m^{-2}$ respectively. Fig. 2.14 also compares mean lengths with distance from the tidal limit at Lopwell Dam. These have tended to increase, this could have correlated with a decline in density. However, the overall mean (± SD) increase (from 25.6 ± 4.3 to 28.9 ± 6.0 cm) is not statistically significant. Furthermore, single-run electric fishing may have biased catches towards larger eels because they were easier to catch.

It is not possible to say with certainty that eel stocks in the Tavy catchment have declined. It appears that populations in all but the headwaters have been dominated by males, implying that recruitment has been good, probably with seasonal supplementation by local migrations from lower sites and Plymouth Sound. Lopwell Dam off Plymouth Sound does not appear to be a major migration barrier. Furthermore, densities were unchanged (415 g $100m^{-2}$ in 1978 *versus* 413 g $100m^{-2}$ and mean length 14.6 ± 4.9 cm in 1998) in Milton Coombe Stream which arises directly from Plymouth Sound, seawards of Lopwell Dam. Local factors may, however, have had strong influences on populations, due to other migration barriers and to low and variable flows. The Tavy is potentially fast flowing, the main river dropping fairly uniformly from 560 mAOD over 35.2 km with an average gradient of 15.9 m km^{-1}. It is, however, subject to major overnight water abstractions upstream to supply Mary Tavy and Morwellham HEPSs and to abstraction for public water supply just above the tidal limit at Lopwell Dam. Associated dams could interfere with immigration, ascent being exacerbated by low flows in July-August and in drought years, such as 1989 and 1990. However, the overall picture of length class frequency distributions with distance from the tidal limit for 1998 samples is similar to that seen in other rivers in south Devon, as discussed below.

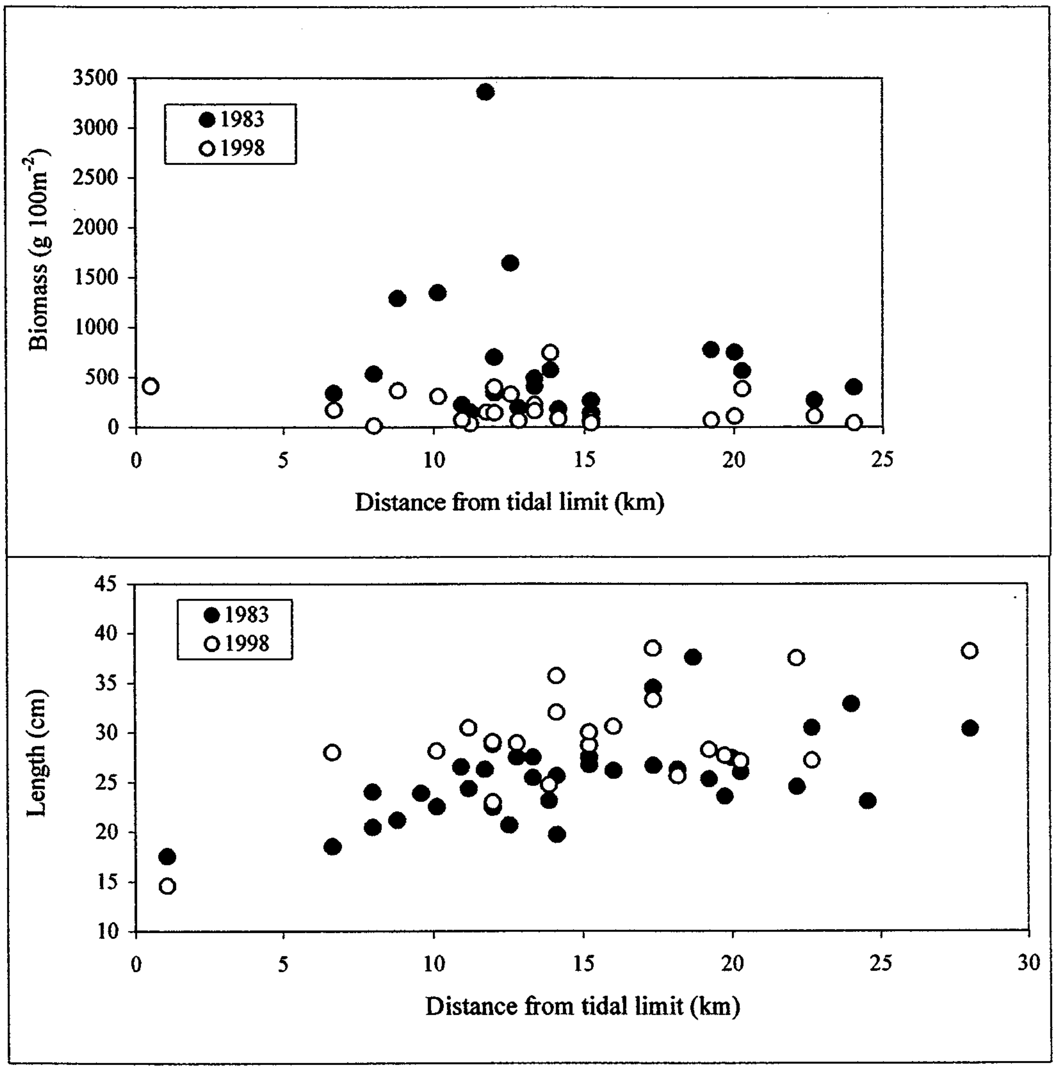


Fig. 2.14 Comparisons of 1983 and 1998 eel biomass (for 22 matching sites) and mean lengths (for 35 sites in each year) in the Tavy catchment, plotted against distance (km) from the tidal limit

2.11 Rivers Avon and Erme (S. Devon)

Five upper river sites in the Devon Avon were electric fished in 1978, two sites yielded no eels, the others densities of 0.85, 2.01 and 7.6 eels $100m^{-2}$. Twenty eight sites were surveyed in 1991, but none matched with those of 1978, and a survey in 1997 only recorded presence-absence. Absences were common to sites in each of these surveys in the upper river (> +20 km), between Didworthy and the Avon Dam as the gradient increases on the edges of Dartmoor. Eels were absent at two sites relatively close to the tidal limit, these sites were at Yabbacombe and at Cornmill, in the upper parts of the Knapmill and Tor Brook tributaries respectively. Other than these results, the general trends in density with distance from the tidal limit (Fig. 2.15) are similar to those for other south-western rivers discussed in this report.

No historical data for the Erme have been located but densities were estimated in the 1991 survey. The results in Fig. 2.15 are similar to those for the Avon, with low river and off-estuary sites having high densities, except that densities fall sharply > +6 km. This must be due to the migration barrier formed by Strowford Weir.

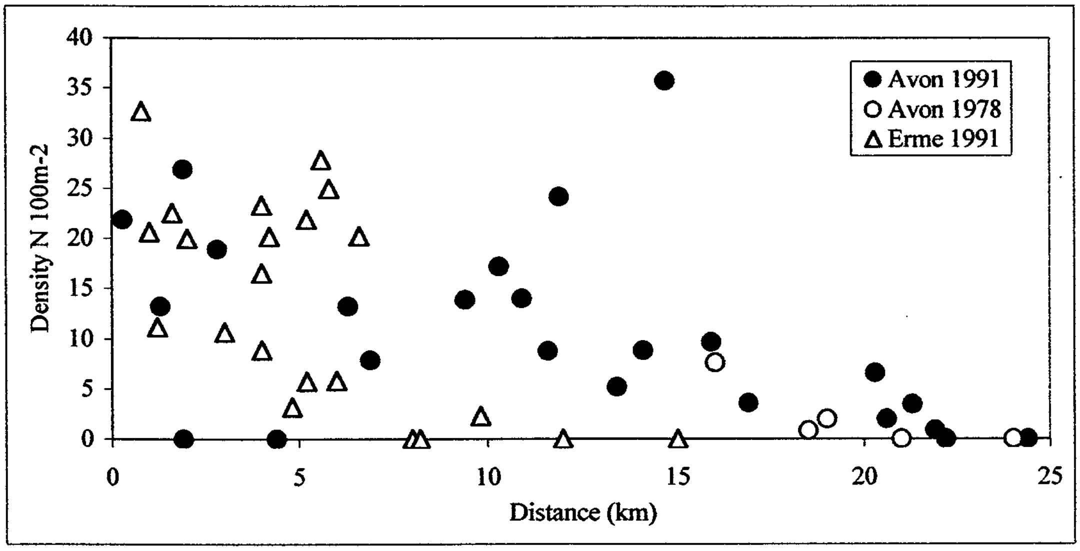


Fig. 2.15 Density of eels in sites at different distances from the tidal limit in the River Avon (S. Devon) in 1978 and 1991 and the River Erme in 1991

2.12 River Dart

A comprehensive data set for eel densities in 1987 exists for the Dart. General fish population information for 44 sites was used to help derive the Fisheries Classification Scheme and information on habitat variables (gradient, substrate, etc.) for 13 was used in the HABSCORE programme. Further analyses and discussion in relation to eel habitat requirements and change are given in Chapter 4.

Subsequent surveys (e.g. in 1997) have only recorded presence-absence and the only other data on eels located was some 1984 information on eel populations in three sites in the Cherry Brook tributary (> 35 km and > 300 m AOD). Records for a survey in 1980 could not be found. Densities in 1984 ranged between 0.99 – 1.15 eels 100m^{-2} and lengths between 43.55 - 48.65 cm. The densities for Cherry Brook sites in 1987 appeared higher, at 2.59 – 5.89 eels 100m^{-2}, but these came from surveys conducted in June-September whereas those in 1984 were carried out in October.

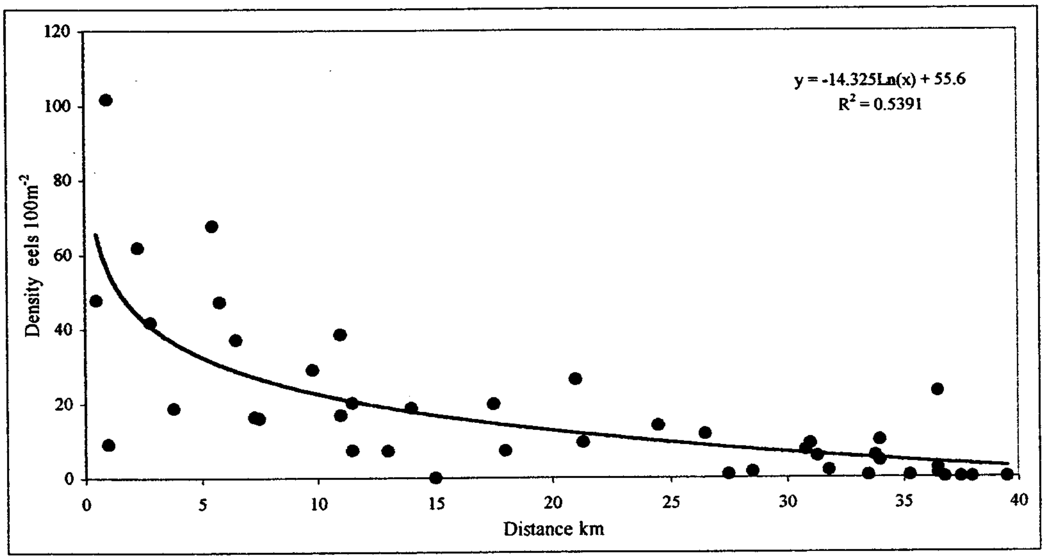


Fig. 2.16 Relationships between eel population density (N $100m^{-2}$) and distance from the tidal limit (km) in the R. Dart catchment in June-September 1987

Eels were found throughout the catchment, in lower, more neutral pH stretches and in Dartmoor moorland streams (except in inaccessible waters, such as Devonport Leat). The very high densities at lower river sites probably relate to ease of movements between the river and estuary (Fig. 2.16). The exceptionally high density of > 100 eels $100m^{-2}$ was found at the Dartmouth STW site in the Bidwell Brook tributary, probably due to proximity to the estuary and main river, to ideal substrate for burrowing in and to locally high productivity. Conversely, the relatively low densities of < 20 eels $100m^{-2}$ at < +10 km are explained by the influences of migration barriers (e.g. the A38 crossing at the Tuckenny site, R. Wash). Poor water quality is implicated in some low or zero densities (e.g at +15 km at Harbourneford on the R. Harbourne). Similar impacts on salmonid density were noted in the 1987 fisheries report. The general picture of population density distribution in the Dart in 1987 looks very similar to that for other short south-western rivers in earlier and later years and shows no obvious signs of poor recruitment.

2.13 River Teign

This small spate river is similar to the Dart, arising on Dartmoor granite at an altitude of 302 m AOD. It is fairly steep and has a bedrock-boulder-sand substrate until reaching the flood plain in the lower reaches. A three-run electric fishing survey of 15 sites was carried out in August 1979, assessing salmonid, coarse fish and eel densities. Eels were found to contribute significantly to fish biomasses and were only absent from two upland sites. Lengths of eels in samples were measured and weights (and thence biomass per site) calculated from length-weight regression coefficients of 0.0086 and 3.1658 (determined on 37 eels from the middle R. Lumburn in 1979). Salmonid data showed little change compared with those from 1963, 1972 and 1975 surveys, but no figures for eels were available for these years. The 1979 eel density data can, however, be compared with those obtained for 13 matching sites from a survey of 36 sites (using similar methodologies) carried out between June and September in 1990.

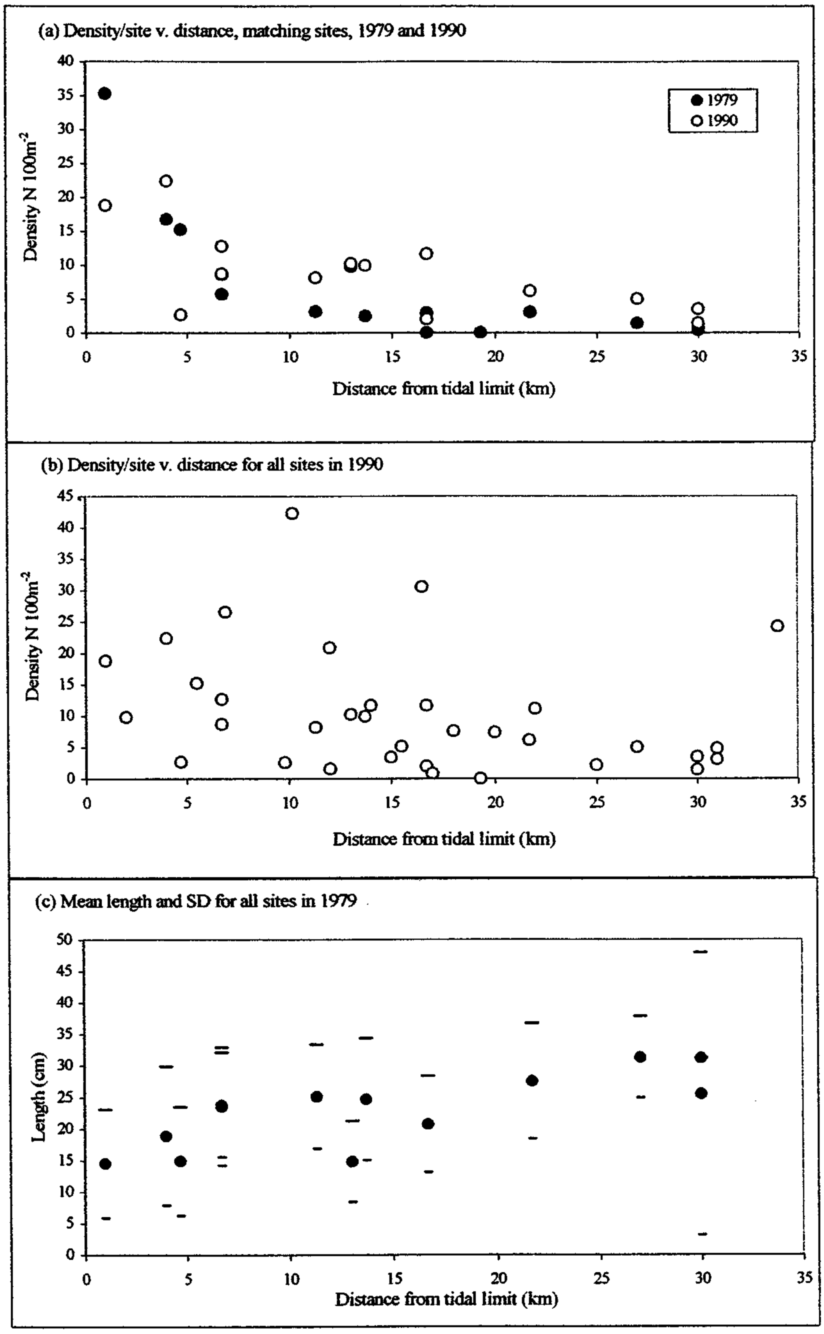


Fig. 2.17 Eel data for R. Teign (S. Devon) surveys with distance (km) above the tidal limit; (a) density/site for matching sites in 1979 and 1990; (b) density/site for all sites in 1990; (c) mean eel length/site for all sites in 1979

Fig. 2.17a shows that density declined from high values of 15-20 eels $100m^{-2}$ with distance from the tidal limit at matching sites in 1979 and 1990. The scatterplot suggests that densities may have been higher in June-September 1990 than in August 1979, although there was an unusually low value in 1990 at + 4.7 km (Little Bovey, on the R. Bovey) and high value at + 4.0 km (New Bridge, on the main river). Low densities and absences at +17-20 km (upper R. Bovey sites) were probably due to the migration barrier formed by Parke Weir, upstream of Bovey Tracy. The longer time span of collections in 1990 may have introduced unknown variations in catches. However, the mean densities (± SD) in 1979 and 1990 were 8.10 ± 9.79 and 8.79 ± 6.20 eels 100m^{-2} respectively and these differences are not significantly different.

Plotting the data for all 36 sites surveyed in 1990 reveals a similar pattern of change in density with distance (Fig. 2.17b). Fig. 2.17c shows that mean length per site increased with distance upstream in 1979, helping to maintain biomass. Only at the highest sites did the average size suggest there were many females present in local populations.

Comparisons do not indicate any significant trends in eel population densities in the Teign over a period of 11 years spanning the major declines in recruitment after 1978-79. It is possible size structures of populations may have changed, but the similarities in density distributions between 1979 and 1990 would suggest this is unlikely and it appears the Teign populations are dominated by relatively small eels in all but the most upstream sites.

2.14 R. Otter

The Otter is another short (< 40 km) river, rising at ~200 m AOD on Greensand in the Blackdown Hills, with three main tributaries (Tale, Wolf and Love (or Lupp)). The upper reaches are narrow and shallow, gradients are fairly gentle and the river widens and becomes deeper below Honiton. The system is mainly spring-fed and of good quality, with pH 7.0 - 9.0 all year round. Habitat data have been used in the HABSCORE R&D programmes, relationships for eels are discussed in later sections. Eel density data were located which bridged the years of recruitment declines, i.e. 1978 (17 sites), 1983 (31 sites) and 1986 (29 sites). Later surveys (e.g. in 1998) have only recorded presence-absence but eels have maintained a presence throughout the catchment (87% of 77 site surveys), except for sporadic absences in some upper sites above weirs (Fig. 2.19). On average, eels contributed 35.5% and 25.4% of total fish biomass in the tributaries and main river respectively in 1978. Otterton Weir (just above the tidal limit) and Head Weir (confluence of the Tale and the main river) do not pose significant barriers to eels, although they may have impacts on salmonid migration.

Density data can be partially matched for 12 sites for each of 1978 (September-November), 1983 and 1986 (sampling months not recorded). Fig. 2.18 possibly indicates some decline over eight years, but inter-year variation is high. For example, the mean density and SD for the eight closest matching sites over the years are 15.4 ± 9.4, 10.5 ± 14.7 and 11.6 ± 11.8 eels 100 m^{-2} respectively and these differences are not statistically significant. Density decreases and average body length increases (and hence biomass tends to remain more stable) with distance upstream, although boat sampling at lower sites was probably less efficient than at other sites. The consistently higher densities in the lower Tale (Fig. 2.18) were due its favourability as a soft substrate habitat, a more uniform U-shaped channel being dredged in 1977.

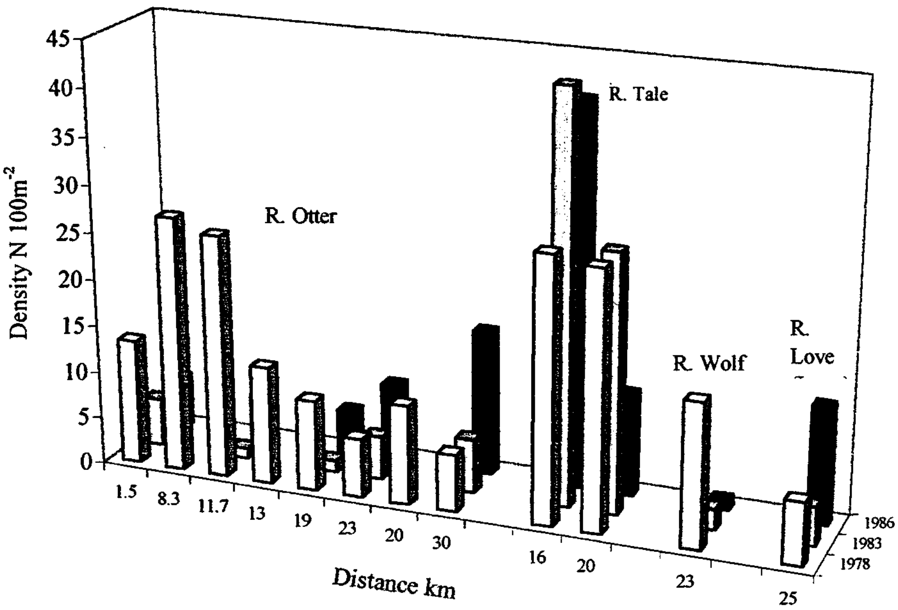
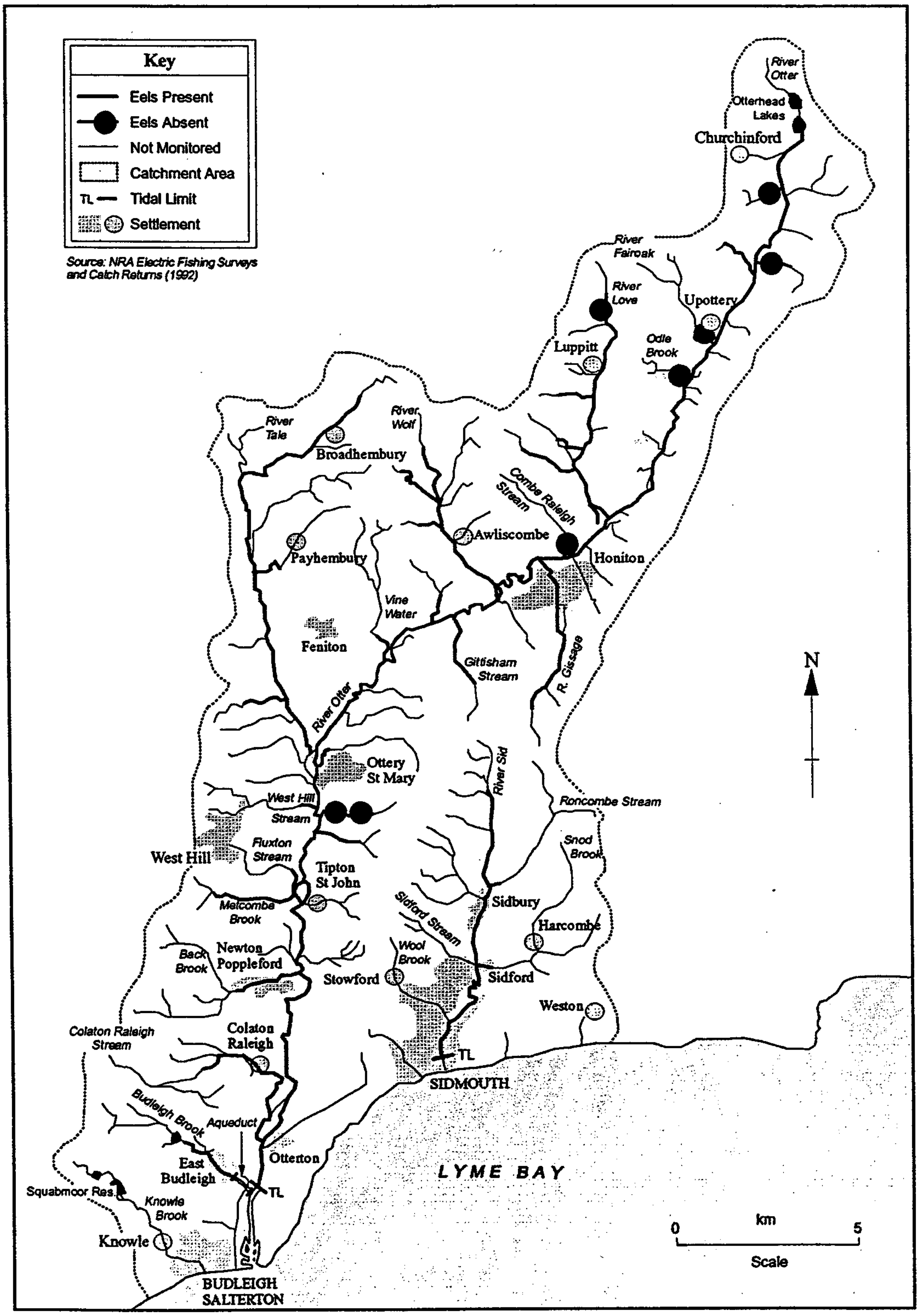


Fig. 2.18 Eel densities for nearest matching sites against distance (km) from the tidal limit in the main R. Otter and tributaries in 1978, 1983 and 1986



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The Rivers Sid and Otter Catchment Management Plan
NRA South Western Region

Fig. 2.19 Eel distribution in the Otter catchment (from Rivers Sid and Otter Catchment Management Plan, 1996)

2.15 Rivers Exe, Culm and Creedy

The Exe is a major river rising at 450 m AOD on Exmoor, with the Culm (rising on the Blackdown Hills) and Creedy forming major tributaries upstream of Exeter, main river lengths being 83, 45 and 24.3 km respectively (Fig. 2.20). This system is potentially of interest for comparison with other shorter southwest English rivers and because recruitment from the large estuary would be to be shared between the three watercourses, ie. a situation similar to that for the Tamar and other Plymouth Sound rivers, but unlike that for the short rivers like the Dart and Otter.

2.15.1 River Exe

Density data (and field sheets for lengths) have been examined for 12 Exe sites surveyed in July-August and 10 sites (including eight repeats) in September-October 1977. These were focussed on the upper tributaries (mainly in the Haddeo and Withiel Florey Stream), at sites that would be most affected by the development of Wimbleball Reservoir. Densities and lengths have also been analysed from field sheets for 22 sites surveyed in 1981. These include a much wider range of non-matching sites than the 1977 survey and exclude three that were lost on impoundment of Wimbleball. Post-1992 surveys in the Exe have only recorded presence-absence, but 1977 and 1981 data have been combined to help assess any differences between the Exe, Culm and Creedy and other Devon rivers.

The lower Exe cannot be electric fished, so the density-distance picture is truncated. However, Fig. 2.21 shows that density to about +40 km was relatively high and compared well with the other southwestern rivers. The low density of 0.66 eels 100 m^{-2} at +25 km occurred in the one Dart tributary site surveyed, those of 2.41 and 5.39 100 m^{-2} occurred in the Lowman tributary. In all these cases, downstream weir barriers are implicated. On the main river, a discontinuity is apparent after about + 40 km. This would suggest there is a migration barrier between Homingham and Oakford Bridges. This could be formed by Chain Bridge Weir, but this is not considered to be an important barrier by fisheries officers and is not mentioned in the Exe Local Environment Agency Plan (LEAP) (see Fig. 2.20). An histogram of length class frequency for samples from above and below this Weir (where densities are above and below 5 eels 100 m^{-2} respectively) shows a distinct difference (Fig. 2.22). The typical pattern of increasing individual size and higher proportions of females with declining density is also shown by this figure. Fig. 2.21c gives a more confused picture for mean lengths because density and biomass relationships with distance upstream are possibly affected by weirs on lower tributaries. It is not possible to say whether there have been any changes in eel stocks in the Exe during the late 1980s and 1990s, because of the lack of recent quantitative data. A survey in 1998 found no eels in 18 out of 95 sites (18.9%), compared to two out of 14 (14.3%) in 1977 and zero out of 22 in 1981. Absences were, however, mostly in upper tributaries, relatively more of which were surveyed in 1977 and 1998 and eels were apparently absent at seven sites in July-August 1977 but then found in four of these in October 1977. This emphasises that presence-absence data in upstream sites are unreliable, being prone to seasonal variations and to the chance capture of individual eels.

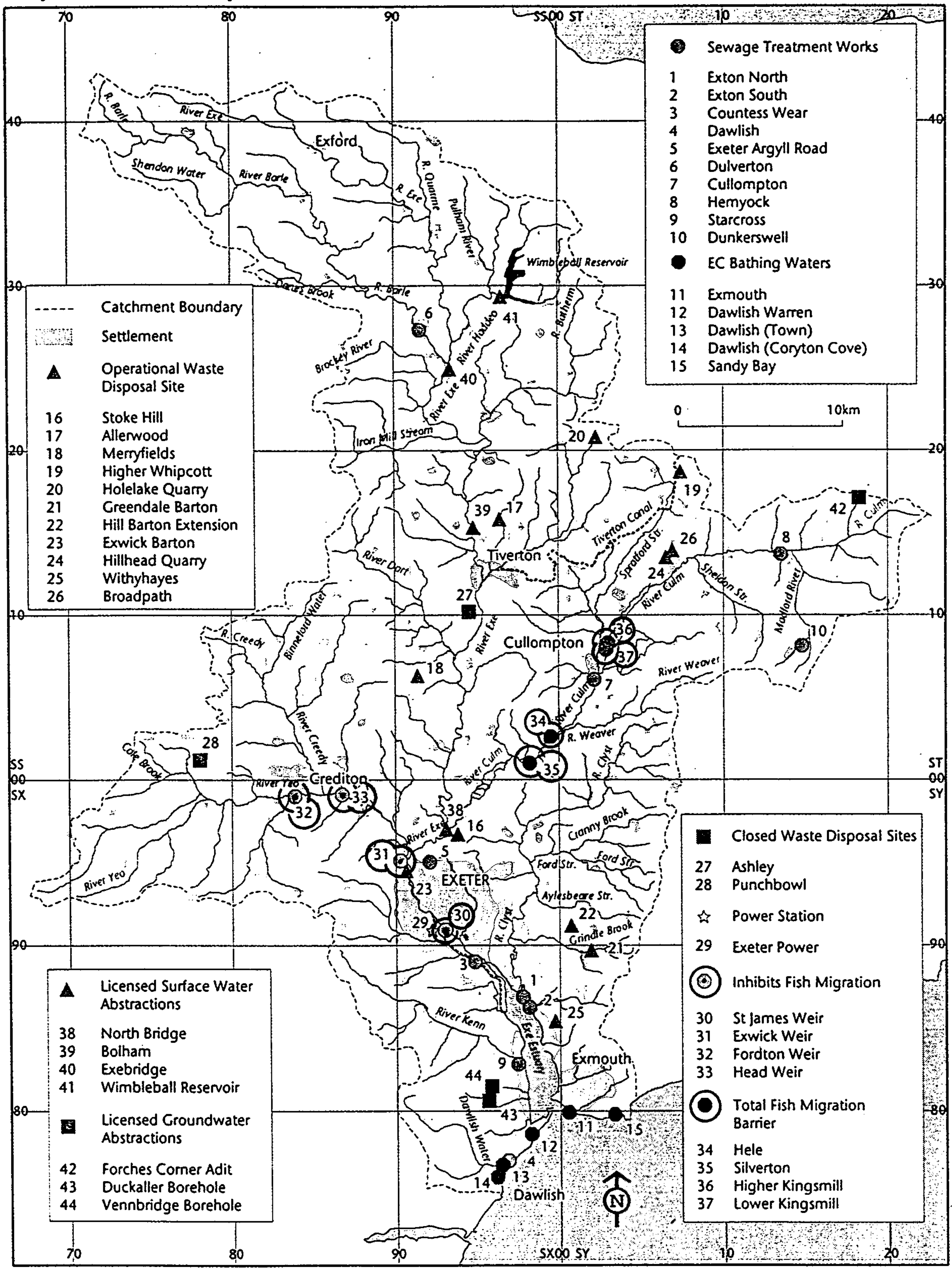

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Fig. 2.20 Map of the Exe catchment (from the Exe LEAP Consultation Draft, 1999)

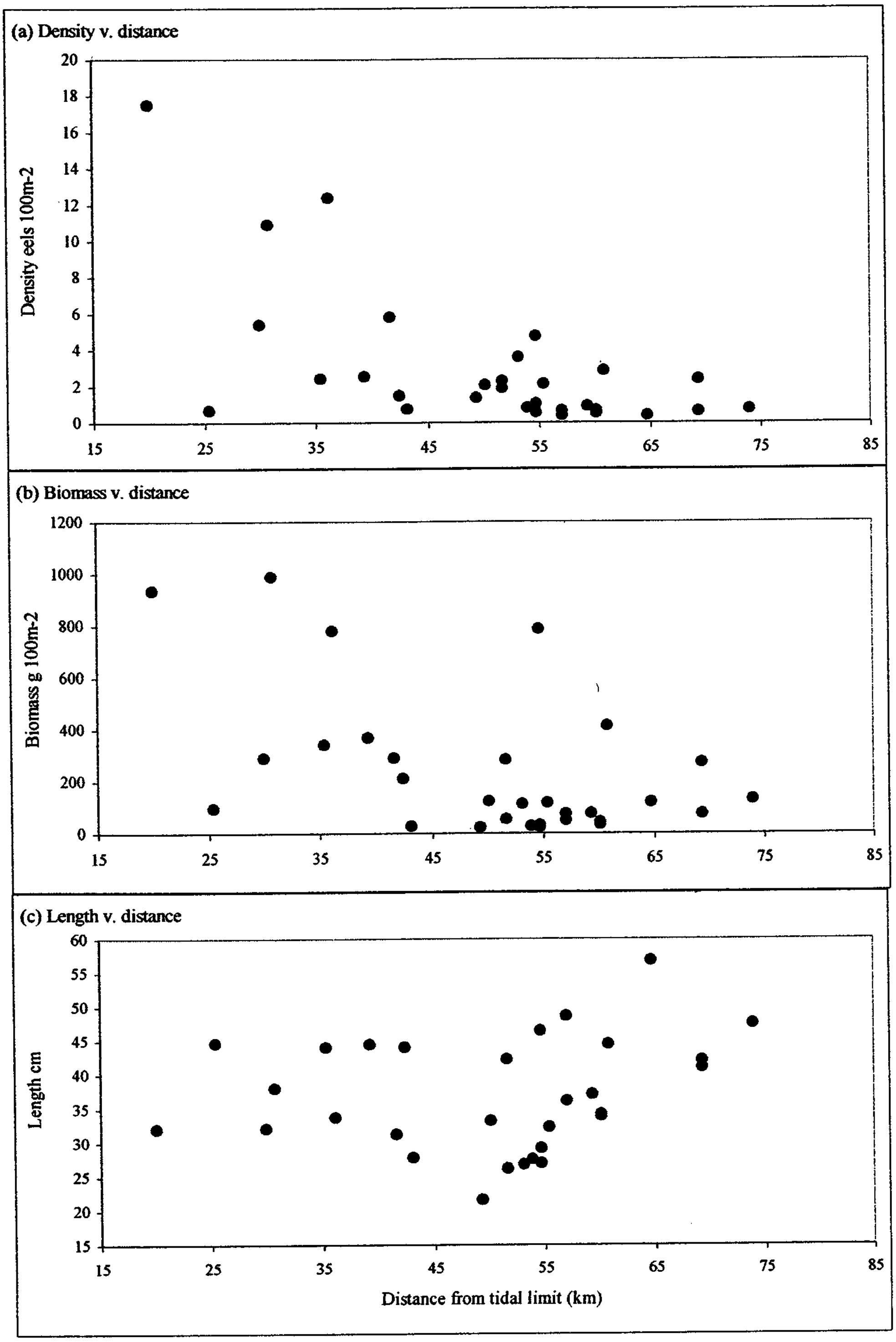


Fig. 2.21 Eel data for R. Exe surveys (1977 and 1981 combined) with distance from tidal limit (km) for matching sites; (a) density, (b) biomass and (c) average length

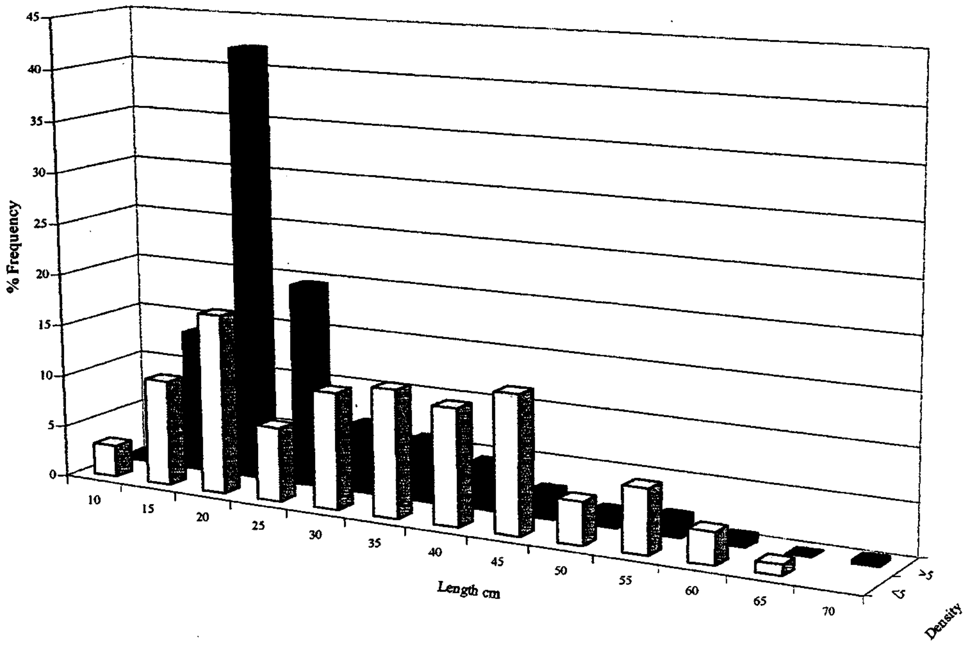


Fig. 2.22 Length-class frequencies of eels (1977 and 1981 data combined) for R. Exe sites with densities > 5 eels 100m^{-2} (6 sites downstream of Chain Bridge Weir at ~ +40km, N = 98) and < 5 eels 100m^{-2} (24 sites upstream of the weir, N = 271)

2.15.2 River Culm

Data for density and biomass were located for surveys in 1977 (15 upper main river and one Spratford Stream site), 1978 (10 main river sites, including five repeats from 1977), 1981 (12 main river, four Spratford Stream, six other tributaries) and 1989 (record sheets for density only for 16 sites). Later surveys (e.g. in 1997) have only recorded presence-absence. Only five sites can be matched over 12 years, these are shown in Fig. 2.23 plotted (together with all sites for the most comprehensive survey of 1981) against distance from the tidal limit. Outlier density and biomass values of 66.67 and 211.65 eels 100m^{-2} and 2926.67 and 6448.96 g 100m^{-2} from 1981 have been omitted. Both of these sites were on the R. Weaver, a lower tributary > 35 +km, where rich silty substrates at Tye and Garlandhayes provide ideal daytime habitat, close to the main Exe. The relatively high densities and biomasses of > 25 eels 100m^{-2} and > 700 g 100m^{-2} at +35-55 km shown in Fig. 2.23 are again probably associated with ideal habitats in the middle Culm and lower Spratford Stream. Low densities and biomasses > +45-50 km are found in upper Culm and Madford River sites.

Comparisons of the five sites matching over 12 years do not reveal any clear trends. Mean lengths of eels (derived from biomass ÷ density and application of weight-length conversion) indicate that, as expected, high density sites are dominated by smaller eels (mean ~ 35 cm), upstream low density sites by females (mean > 40 cm). The similarities of distribution of densities in the Culm and Exe suggests, however, that recruitment is shared fairly evenly between the two rivers, given that the habitats at some sites on the R. Weaver, lower Culm and Spratford Stream are particularly favourable to eels.

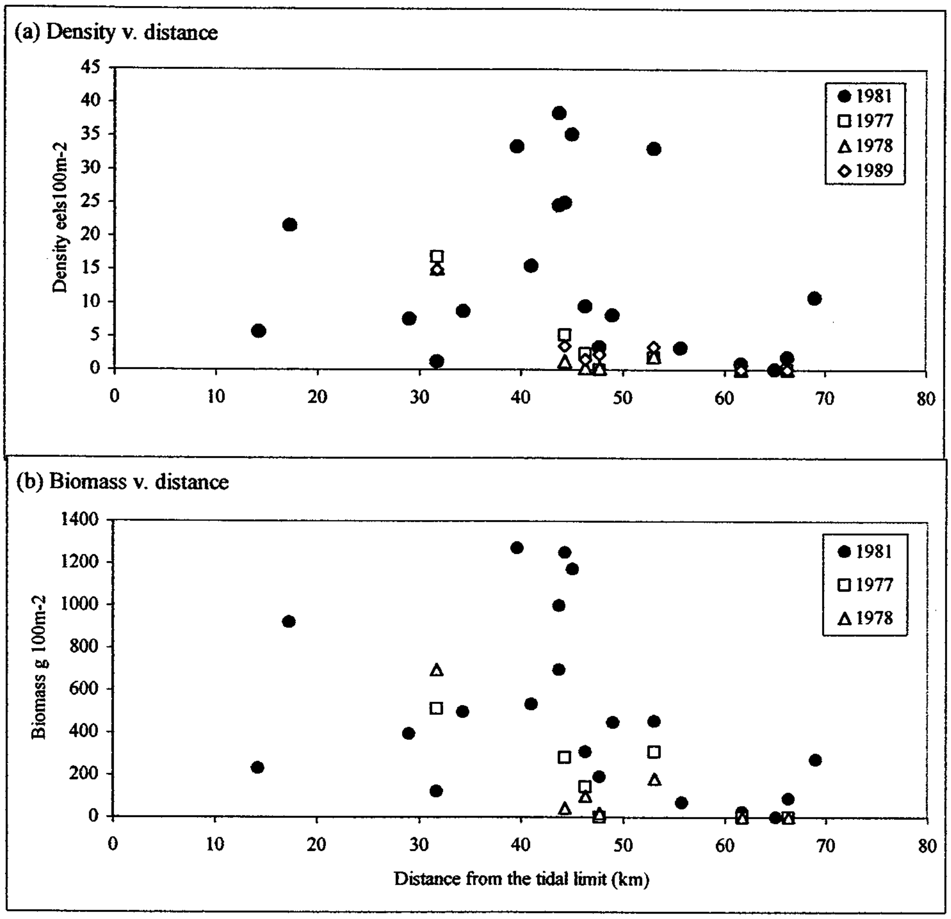


Fig. 2.23 Eel data for R. Culm surveys (all sites for 1981 plus matching sites for 1977, 1978 and 1989) with distance from tidal limit (km), (a) density and (b) biomass

2.15.3 River Creedy

The Creedy has its confluence with the Exe just upstream of Exeter, +5 km from the tidal limit. It is generally a mesotrophic river of good biological and chemical quality, but the substrate mainly comprises large, flat stones with occasional deposits of sand and gravel which is not ideal as daytime habitat for eels. Unfortunately, quantitative data were only located for one survey of 12 sites in 1978, later surveys (e.g. in 1997) only recording presence-absence.

Plotting density and mean lengths against distance from the tidal limit for 1978 (Fig. 2.24) shows that densities are relatively low in comparison with those for the Exe and Culm, except for the density of 26.6 eels 100m^{-2} at Codshead Bridge just above the confluence with the Exe at +14.1 km. Average lengths also tend to be slightly greater. These differences are probably due to habitats being less favourable for eels, but also to the impacts of migration barriers. Exwick Weir (no. 31 on Fig. 2.20) could be a barrier but Higher Marsh Weir just upstream of the confluence with the Exe is possibly more important. Gunstone Weir at +15 km on the R. Yeo, upstream of Crediton, is also possibly a barrier to eel migration, explaining some of the low densities and absences (including those found in 1997) in the upper Yeo and its tributaries, the Troney and Cole Brook, at > +25 km. The situation in 1978 may have been exacerbated by the droughts of 1975-76, these upper streams being particularly vulnerable to low flows. Head and Fordton Weirs downstream of Gunstone are considered to impact salmonid migration (Fig. 2.20), but do not appear greatly affect that of eels.

The data available cannot directly reveal any historic trends in the Creedy, and the differences between the Creedy and the Exe and Culm appear to be due to habitat differences and migration barriers. However, the relatively small number of sites involved, their positions relative to the main Exe and tidal limit and the wide range of densities found mean that the Culm could be a useful system for more detailed and quantitative monitoring in the future.

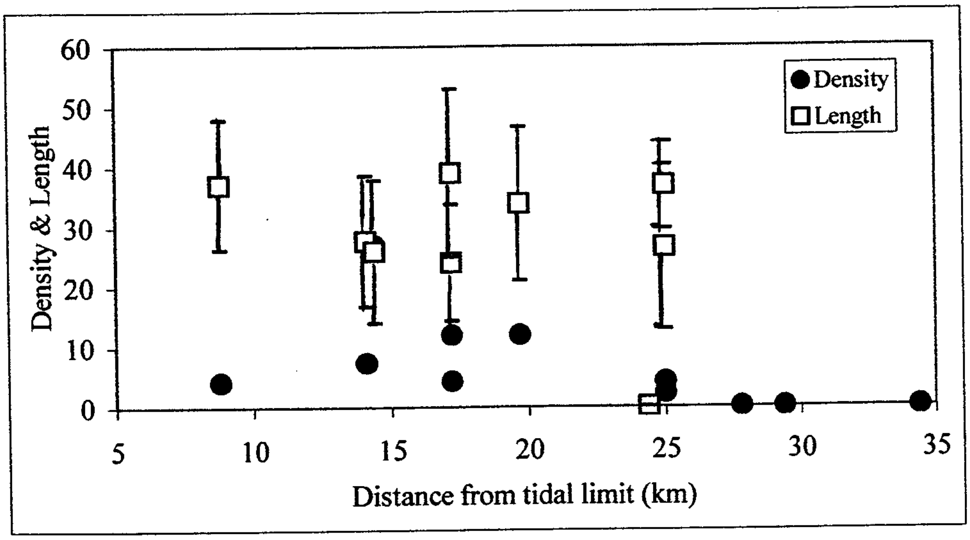


Fig. 2.24 Density ($100m^{-2}$) and mean length ± SD (cm) of eels from River Creedy sites surveyed in 1978

2.16 River Axe (S. Devon)

The Axe is a productive, good quality river, largely of gentle gradient and soft substrates. Reasonably quantitative and comparable historic data sets for density were found for 13 sites (plus seven qualitative site surveys) in September 1979, 11 in July 1986 and 30 sites (plus six qualitative site surveys) in August-September 1990. Length data were collected for 12 comparable sites in 1979 and 1990. Eels were found at all sites in all years, densities being generally much higher than in other south Devon rivers, as shown in Fig. 2.25 for sites with at least two records from 1979, 1986 and 1990. Average biomass per site was estimated in 1986, showing eels contributed 35% (range 6–86%) to fish biomass.

Density was only directly comparable for eight sites for each of the three years. Comparisons in Table 2.3 imply an increase in density between 1979 and 1986 and then a decrease in 1990, but only the 1990 result shows a statistically significant change ($P = 0.05$). The data are, however, distorted by one exceptionally large catch in 1979 and two in 1986 (see Fig. 2.25). Furthermore, surveys were conducted in July in 1986, whereas those in 1979 were

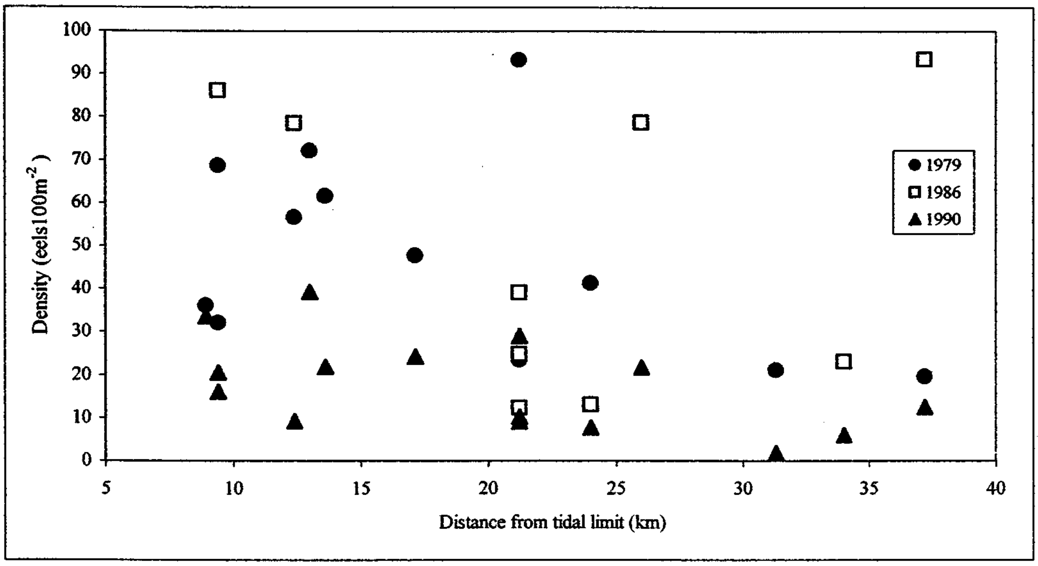


Fig. 2.25 Eel density data for R. Axe surveys in 1979, 1986 and 1990 (for sites with at least two matching surveys) with distance from the tidal limit (km)

carried out in September. In contrast, there is apparently a very marked decrease of ~ 50-60% in overall density for all sites combined between 1979 and 1990. Length class frequencies for all site samples combined in Fig. 2.26 show there was a marked shift towards a higher proportion of larger eels between 1986 and 1990. The distributions tend to be bimodal in both years, mainly because lower sites are dominated by large numbers of small eels, upstream ones by large eels. This is shown in Fig. 2.27 for representative low, middle and upper river sites. There appears to have been a major reduction in (small) eels at lower sites in 1990 compared to 1986. The results suggest declines in recruitment since the late 1970s-early. However, other local factors may have played a role, seeing that between 1979 and 1990, recorded salmonid densities declined by 43% at 12 matching sites and total coarse fish densities by 69%. This implies either that there had been major catchment-wide changes affecting all fish (possibly due to droughts and low flows) or, more likely, there were major differences in sampling methodology and/or efficiency. This could explain the lack of smaller eels found in 1990.

Table 2.3 Eel population data for the River Axe catchment (S. Devon, England), comparing mean density (eels 100m^{-2} ± SD) in 1979, 1986 and 1990 for all sites per year and for eight matching sites

YEAR	1979 (September)	1986 (July)	1990 (August-September)
Total no. of sites	12	11	36
Mean density for all sites per year (eels 100m^{2})	47.9 ± 23.1	61.0 ± 39.1	24.6 ± 23.1
Mean density per 8 comparable sites (eels 100m^{-2})	47.0 ± 23.9	66.3 ± 42.8	14.5 ± 6.7
Mean number of eels per site for all sites	29.8	38.8	13.0

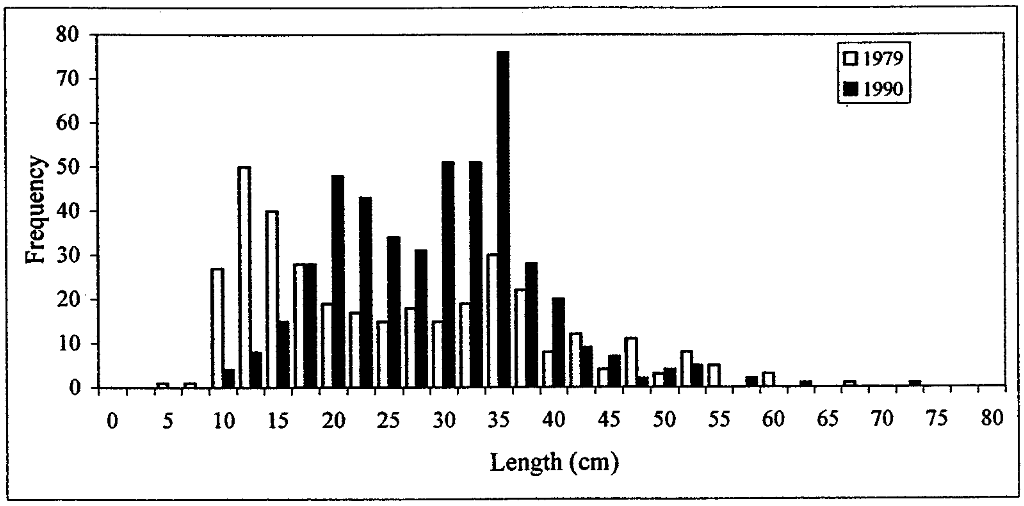


Fig. 2.26 Length class frequency distributions for all sites combined for the R. Axe in 1979 (N = 896) and 1990 (N = 555)

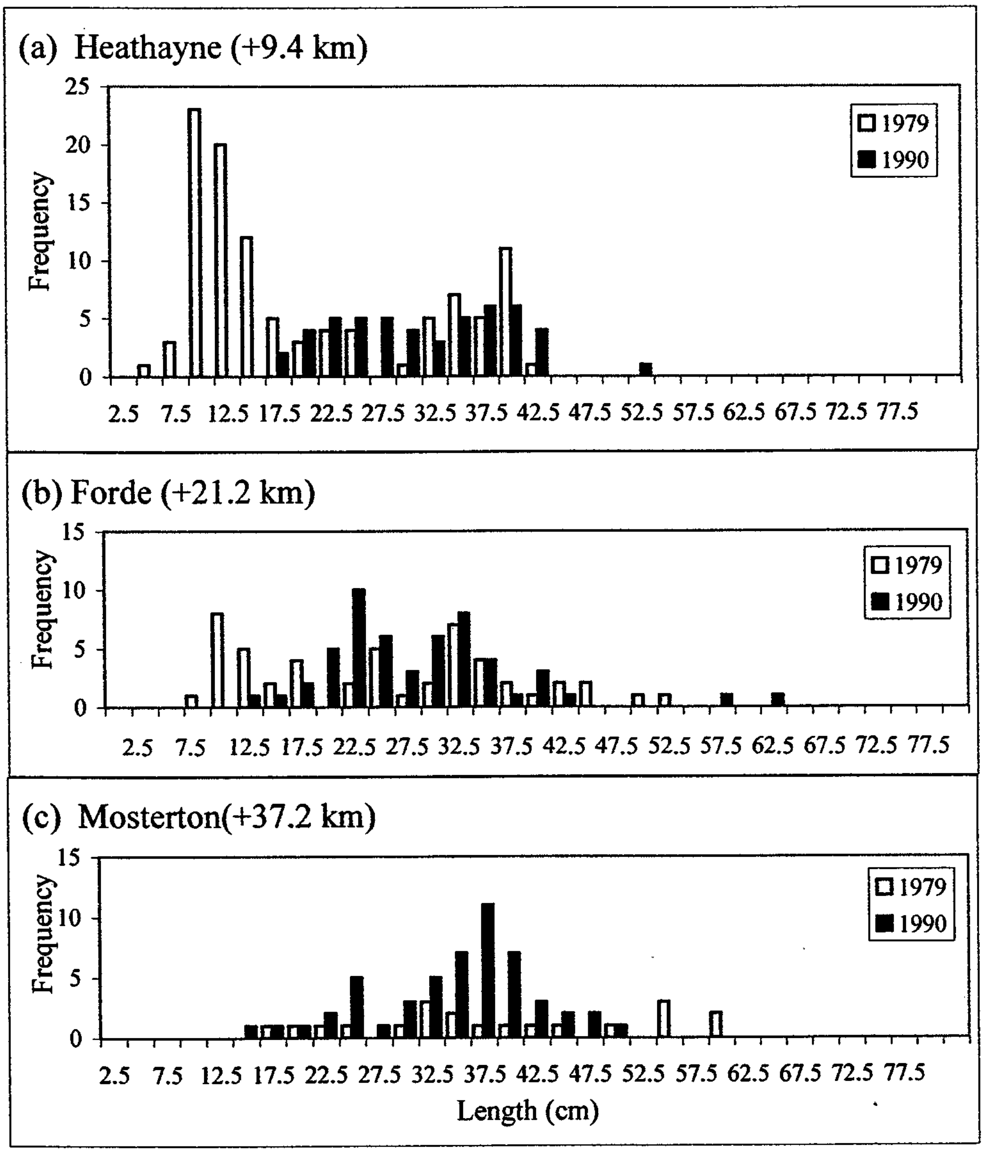


Fig. 2.27 Length-class frequency distributions for representative lower, middle and upper river sites in the Axe for 1979 and 1990

2.17 Rivers Frome and Piddle

Good historical data sets covering 1973-1990 have been analysed, these suggest declines in stocks in these rivers have occurred, as summarised in Table 2.4 below. These rivers have therefore been the subject of resurveys, full details are given in Chapter 3. It is possible that any declines could be associated with changes in populations in Poole Harbour, into which they discharge. This is discussed in the next section.

Table 2.4 Data for eels stocks for 1973, 1977-8, 1991-2 and 1990 in the River Frome (with special reference to the Tadnoll Brook tributary and to East Stoke Millstream) and in the River Piddle, Dorset.

River	Sampling Season	No. of Sites	Density N $100m^{-2}$	Biomass g $100m^{-2}$	Mean size[1]
Tadnoll Brook	1973[2]	16	68.8 ± 21.3	210 ± 100	3.0g
	1980[2]	16	32.7 ± 2.6	120 ± 0.02	3.7g
East Stoke Millstream	1990[3]	1	16.7 ± 3.3	137 ± 0.02	8.2g
	1990[3]	1	16.0 ± 1.3	103	6.4g
	1991-2[4]	5	Range = 5-14	-	-
R. Frome	1990[3]	6	11.2 ± 2.0	1200 (Range 810-1530)	107.1g 35-40cm
R. Piddle	1976-7[5]	All 19	26.8 ± 12.0	2570 ± 1200	95.9g
	1990[3]	All 3	6.92 (4.8-9.71)	760 (290-1500)	109.8g
	1976-7[5]	Site 19 Culeaze	19.0	2100	110.5g
	1990[3]	Throop	4.58 ± 1.75	290	63.3g
	1976-7[5]	Site 10 Trigon	22	2000	90.9g
	1990[3]	Trigon	6.48 ± 3.70	480	74.1g

([1] relative mean individual weight calculated by dividing biomass by density estimates; [2] derived from Fig. 7 and Table 4 in long-term study by Mann & Blackburn, 1991; [3] NRA unpublished one-off 1990 eel survey data; [4] from Fig. 3 in Ibbotson *et al.*, 1994; [5] MAFF eel study report, Morrice *et al.*, 1987)

2.18 Poole Harbour and The Fleet

2.18.1 The Fleet

It has been claimed by local fishermen that the fishery in The Fleet (Weymouth) has declined. According to NRA/Environmental Agency catch returns for 1988-1996 (ignoring an aberrantly low return for 1991), catches in The Fleet have been relatively constant with a mean of 1.8 (range 1.1-1.7) t year^{-1}. This is similar to the figure of 2 t year^{-1} quoted by Morrice (1989) for the 1980s. These are probably underestimates, but suggest stocks have not declined since the 1980s.

2.18.2 Poole Harbour

Poole Harbour is an unusual habitat in England in that, with its restricted connection to the sea, it forms a large coastal lagoon of some 38km^2 at HWS, with approximately 80% of this comprising intertidal areas and mudflats. It is relatively shallow and sheltered, has a small tidal range (with double high tides) and low fresh water inputs and therefore has a relatively slow flushing time and fairly constant and high salinity. It is generally warm and primary productivity (of salt marsh, macroalgal and phytoplankton communities) and secondary productivity (based mainly on detritus) are high. These characteristics, plus accessibility to emigration of glass eels from the sea, mean that it should support highly productive eel populations and fisheries. However, local eel fishermen claim that stocks and fisheries have declined and that the average size of eels in catches has fallen over the last 20-30 years. The situation was investigated for the Environment Agency (Knights, 1997) and a summary of key points and conclusions is given below.

According to Roger Castle (eel fisherman, unpublished report to Environment Agency S. Wessex Area), yields have declined by 80-85% since the 1960-70s. Fyke nets were introduced into the Harbour fishery in the 1950s and catches in the 1960-70s were about 30 t year^{-1}, with excellent by-catches of other species. This accords with estimates made by Morrice (1989). At the end of the 1960s, five 'canoes' were laying up to 650 ends per night. Increasing effort (by as much as a tenfold increase in numbers of ends deployed per day) was needed to maintain catches in the early 1980s. Subsequent declines in catches have, it is claimed, led to reduced numbers of fishermen and hence total effort since the mid-1980s. Assuming the area of Poole Harbour is 38 km^{-2} and extrapolating from estimated total commercial yields, Knights (1997) calculated that catches could have fallen from about 7.9 in the 1960-70s to 1.3 kg ha^{-1} year^{-1} in the 1990s, a decline of 80-85%. In comparison, yields in warm productive coastal lagoons could be expected to be as high as 10-15 kg ha^{-1} year^{-1} (Tesch, 1977; Moriarty & Dekker, 1997). Although firm evidence is lacking, size-overfishing is implicated, probably exacerbated by reduced recruitment. Such long-term trends in Poole Harbour would mirror the declines in catches of 65-80% reported in, for example, Lake IJsselmeer (Netherlands), Mediterranean lagoons in France and Italy and in the Baltic Sea off Sweden (Moriarty, 1996; Moriarty & Dekker, 1997).

In relation to more recent changes, quantitative information on fishery yields was first collected by the former NRA in 1988, licensing of the fishery only began in 1991 and catch returns are probably inaccurate, as discussed in Section 2.1. In 1988, the declared catch was 17,045 kg (including 1270 kg caught by a single tidal seine net), but there are no records for fishing effort. Between 1991-96, the average catch was 19.6 (range 10.9-28.0) kg end^{-1} year^{-1}, shared between 2 to 8 licencees. There did not appear to be any definite trend of change over this short time span and the annual declared catch CPUEs were similar to those for the Thames Estuary fishery over the same time period (Section 2.22.3).

Although it is claimed that the average size of eels has declined in Poole Harbour and size overfishing may have been implicated, historic quantitative data are lacking. Fyke net samples of 42 eels were collected in 1996 by the Environment Agency and lengths and ages measured. The data do not reveal any useful information about small juveniles because fyke nets of the dimensions used (11 mm mesh cod end) only reliably catch eels > 31-32 cm (Naismith & Knights, 1990a). However, Knights (1997) showed that the sample was dominated by small immature eels, nearly 70% being < 38cm, with the mode at ~ 34 cm. There was a sharp decrease in number of eels > 34 cm, this could have been due to selective removal by fishing. However, in high density populations, eels above this size would be expected to be progressively maturing as males and emigrating, e.g. as found in the Thames Estuary by Naismith & Knights (1990b, 1993) and in the R. Frome by Mann & Blackburn (1991).

Only 31% of the Poole Harbour sample were females of >45 cm, up to a maximum length of 65.2 cm (Table 2.5). The sample may have been distorted by eels migrating between the Harbour, the rivers and coastal waters. However, allowing for the small sample size, the overall size-frequency distribution is very similar to that found in eels fyke netted in the outer and middle reaches of the Thames Estuary (Naismith & Knights, 1993). Similar pictures are seen in comparable European waters (EIFAC/ICES, 1995; Moriarty & Dekker, 1997).

Table 2.5 Summary of Poole Harbour eel sample data, collected in August-September 1996 (means ± SD, ranges in brackets)

	N	% of sample	Body length (cm)	Age (years)	Growth rates (cm $year^{-1}$)
Females (> 45 cm)	10	31%	54.0 ± 7.0 (45.0 – 65.2)	8.6 ± 3.2 (3 - 16	6.34 ± 2.20
Other eels (< 45 cm)	32	69%	34.5 ± 4.7 (24.4 – 42.9)	3.9 ± 1.3 (3 – 7)	7.58 ± 3.58 (2.60 – 11.97)

Average growth rates for female eels from Poole Harbour (6.34 cm $year^{-1}$) appeared very similar to means for other similar European waters, although this conclusion was only based on a sample of ten eels >45cm (range 45-65cm). The average growth rates for Poole eels < 45cm appear relatively high at 7.58 cm $year^{-1}$. They are higher than those found in the Piddle and Frome of 2.6 – 2.8 cm $year^{-1}$ and those in the Thames Estuary (4.67-6.07 cm $year^{-1}$) and enclosed freshwater sites in the Thames catchment (6.19-6.62 cm $year^{-1}$) (Naismith & Knights, 1993). They are, however, similar to values found in warm productive estuaries/coastal lagoons in Europe (Moriarty & Dekker, 1997).

In conclusion, it appears that overfishing may have occurred in the past and that recruitment to Poole Harbour (and hence presumably to the rivers entering it) has probably fallen in the long term but that in the 1990s, the situation has been more stable. The proportion of females is possibly relatively high compared to, say, the middle Thames Estuary, but the Harbour population appears to be still relatively dominated by faster growing males that mature and emigrate at < 40-45 cm. It may be that the carrying capacity of the Harbour is sufficient to support a high proportion of current recruitment and that the competitive pressures to migrate into the rivers has lessened. It is also possible that large catches in the past were not only made up of high densities of resident male eels, but also of larger eels migrating daily and seasonally back and forth between the rivers and Harbour. These hypotheses are discussed further when reviewing population levels, structures and dynamics in the in-fluent Piddle and Frome (Section 3.9) and in large river-estuary systems such as the Thames and those found in Essex.

2.19 Rivers Stour (Dorset) and Avon (Hampshire)

No historical data are available for the Stour. Recent salmonid surveys have only involved single run electric fishing, but surveys in 1998 suggest stocks are fairly good in the two lowest tributaries, the Allen and Crane. At 58 sites on the Allen, eels were 'abundant' (>10 eels caught per single run) at 48 (83%) sites, `frequent' (3-10 eels) at 5 (8.5%) sites and absent from 5 sites. At 28 sites on the Crane, they were abundant at 21 (75%) sites, frequent at 6 (21.5%) sites and absent at 1 (3.5%) site. Although these results for lower tributaries are encouraging, local fisheries officers have expressed concern about an apparently major decline in overall eel stocks in the Stour between routine surveys of 1992 and 1998, despite, it is claimed, using the same methodology and equipment. Total numbers of fish caught per single run for 26 comparable sites for the whole catchment, including upper river and tributary sites, are shown in Table 2.6.

Table 2.6 Total number of fish caught in surveys of 26 comparable sites (and percentage decline) in the Dorset Stour in 1992 and 1998

	Eels	Roach	Dace	Chub	Pike	Perch	Barbel	Bream	Carp	Tench	Grayling	Bleak
1992	1150	4360	558	379	843	193	22	87	7	41	4	0
1998	69	2103	321	133	305	102	14	60	3	18	0	218
% decline	94.0	51.8	42.8	64.9	63.8	47.1	36.4	28.7	57.1	56.1	-	-

Although eel numbers have apparently declined drastically, the data show that catches of other species were also much lower in 1998 than in 1992. This suggests there may have been some common cause for reductions in all fish stocks or differences in sampling and efficiency.

However, evidence from a local eel fisherman is that eel stocks, at least in middle and upper parts of the catchment, have declined by about 70% since the early-1990s (Roger Castle, unpublished report to the Environment Agency, South Wessex). Silver eel catches at the Corfe Mullen (Coventry Arms) trap upstream of Wimborne are stated to have declined from approximately three tonnes in the early-1990s to one tonne. Catches at the Longham trap further downstream (below the confluence of the Allen but above that of the Crane) have declined from about one t to an all-time low of 0.3 t in 1999 (despite this being the only trap in operation in that year). Yellow eels upstream of Blandford Forum have also declined, according to angling and fyke net catches (although no figures are given in the report). Low stocks and the paucity of silver eels in the late-1990s could have resulted simply from falls in recruitment since the early-1980s (Fig. 2.3). This would correlate with the average age of 15 years of emigrant eels in the Stour (Brian Knights, unpublished results). However, silver eel catches in the Hampshire Avon have been maintained, the 1999 catch being the highest for many years (even using fyke nets, which are inefficient in the spate conditions when emigration peaks occur). The differences between the two rivers occur despite the fact that they join together about one km above a common estuary and both discharge into Christchurch Harbour. Comparisons of Agency electric fishing data for surveys in the Hampshire Avon in 1978 and 1991 also suggest little change in eel stocks, although detailed analyses are not possible because of the poor quality of the 1978 data records and lack of any matches between sites. A yellow eel fishery used to exist in the Harbour but no fyke net licences have been issued in recent years.

The unpublished report by Roger Castle suggests that, whilst recruitment of glass eels to both rivers has probably declined, water and habitat quality in the lower Stour have negatively impacted upstream immigration, especially that of elvers and small yellow eels during the early summer. The lower Stour was extensively dredged (down to 3 – 4 m) in the mid-1980s and he has noted that the summer current velocities are low, the bottom silty and devoid of cover. DO sags possibly occur, exacerbated by general catchment eutrophication and by local sewage works discharges and water abstractions/low flows. These poor water and habitat quality factors, he claims, have reduced populations in the dredged zone and pose a migration barrier. This has then led to a tendency for emigrants to preferentially move up the Avon, leaving the upper Stour catchment sparsely populated. In support of his theory that water quality is particularly bad in the dredged zone, he states that eels stored in holding nets in the affected stretch of the Stour showed a 75% mortality, whilst mortality was zero during storage over the same period further up the Stour and in the Avon.

The Environment Agency (S. Wessex Area) has proposed studies to clarify differences in stocks in local rivers and possible causes as Action 13q in the Second Annual Review (January-December 1999) of the Dorset Stour LEAP.

2.20 Conclusions Regarding the South West Region Rivers

Before reviewing historic data sets for other Regions, it is worth discussing the conclusions arising from those gained for rivers in the South West Region because they have generally proved to be the most comprehensive. Many past electric fishing surveys, especially in south Devon and Cornwall, have been relatively efficient and well documented and long-serving fisheries staff have been able to locate old hard-copy reports and data sheets. Even so, multi-species surveys will have underestimated eel populations in comparison with those aimed specifically at sampling eels, according to the resurveys reported in Chapter 3. Glass eel recruitment to the Parrett and other N. Wessex estuaries is still high enough to support commercial fisheries glass eel fisheries, probably resulting from proximity to the main Atlantic migration pathways. However, according to licence sales and catch return data, exploitation of yellow and silver eels is lower in more south-westerly rivers than that in southerly and eastern ones. Only The Fleet and Poole Harbour and some S. Wessex rivers (e.g. the Dorset Stour and Hampshire Avon) support fyke net and silver eel fisheries. This is because of the unsuitability of many of the south-westerly rivers for fyke netting, competition with game fishing, the general predominance of smaller and less-valuable males and distances from London and Continental markets.

Although some past records (especially for south Devon and Cornwall rivers) are good, resource constraints and an increasing focus on salmonid surveys in recent years have led to reduced fishing effort, changes in survey sites and a lack of quantitative surveys of eels and other coarse fish species. Given these drawbacks, the available evidence suggests that in the shorter rivers exposed more directly to the Atlantic-southern English Channel glass eel

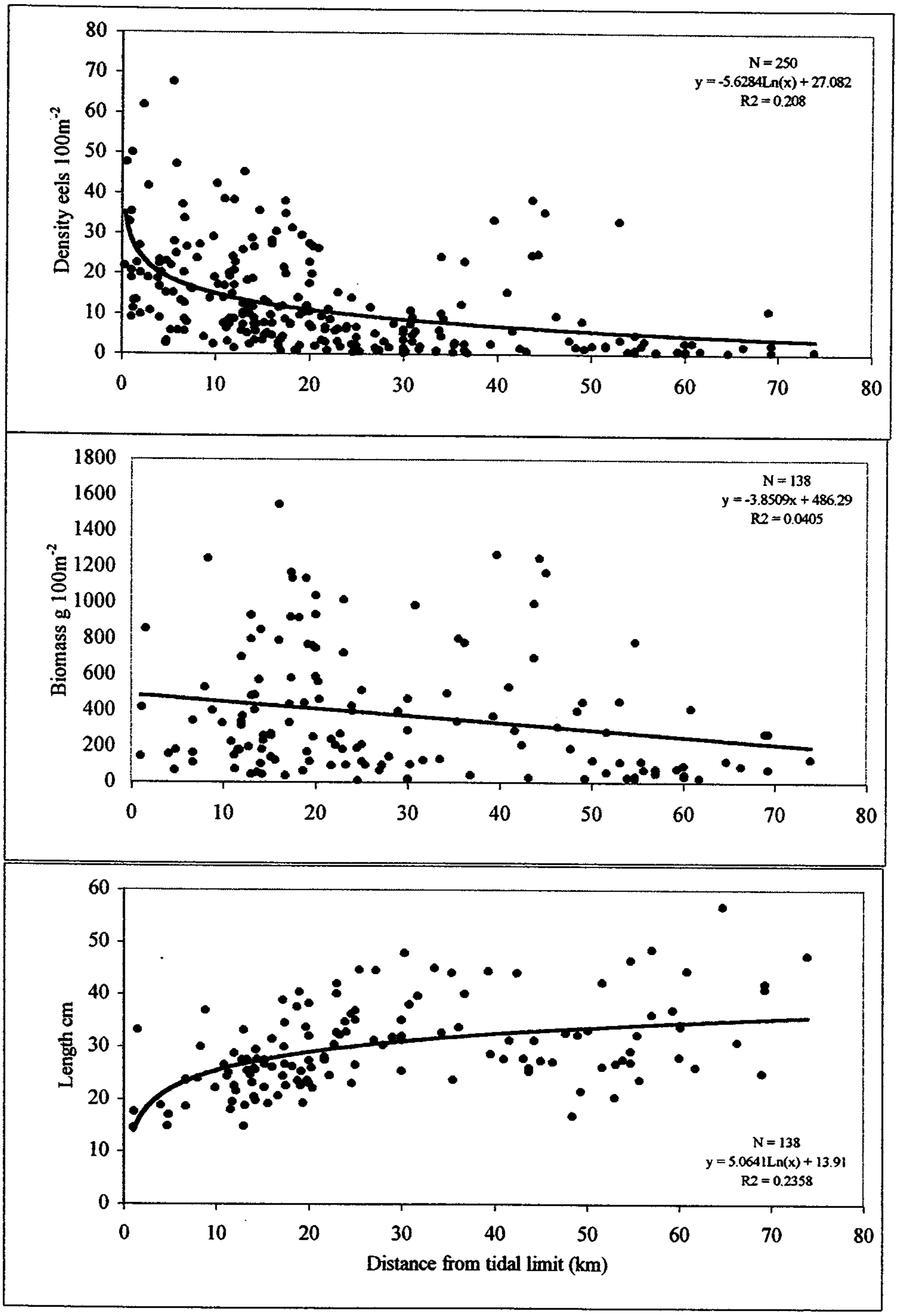


Fig. 2.28 Relationships between eel density, biomass and mean length and distance from the tidal limit in rivers in the south west of England, 1978-91 (combined data for R. Fowey, Tamar, Tavy, Teign, Otter, Exe, Culm and Creedy, plus density-only data for the Avon, Dart and Erme)

migration routes, recruitment has been sufficient to maintain relatively high density populations, even after the declines in glass eel recruitment to Britain in the late 1970s-early 1980s. Fig. 2.28 combines the most comprehensive data sets for south Devon and Cornwall rivers for 1978-91. These scatterplots show that (given high variability between sites, seasons and years), densities are indeed relatively high, the mean (± SE) being 12.50 ± 0.77 eels $100m^{-2}$ with values commonly exceeding 20 eels $100m^{-2}$ in lower river and middle tributary sites. Biomass also tends to be high (mean 376.39 ± 29.54 g $100m^{-2}$) and compares favourably with other European waters of similar types (Tesch, 1977; Moriarty & Dekker, 1997). Biomass tends to be maintained relatively more constant than density with distance upstream because average size of individual eels increases (Fig. 2.28). Populations are largely dominated by eels maturing as males of less than 40 cm after 6 - 8 years, the mean length being 26.55 cm at sites < +25km, increasing to a mean of > 40 cm further upstream. Changes in density due to poor recruitment after the late 1970s should have shown up by at least the late-1980s, with the situation stabilising during the 1990s (Section 2.2). However, no significant changes are apparent in the shorter rivers of Devon and Cornwall. Furthermore, similarities of biomass throughout such catchments (given relative differences in habitat suitability, productivity and distances from tidal limits) suggest that their carrying capacities are still being achieved. There is, however, much seasonal migration occurring between the lower reaches of rivers and tributaries and associated estuaries. These issues are addressed more fully when discussing habitat requirements and change in Chapter 4.

The picture for the short south-western rivers is very similar to that seen in the R. Fremur, near St. Malo in northern Brittany, France (Feunteun *et al.*, 1998). The Fremur comprises 17 km of main river, plus 28 km of tributaries, with a gentle gradient and a range of habitat types. In 1995, 33 sites were electric fished and eels found to be heterogeneously distributed throughout the system (given some influences of migration barriers). Mean density and biomass (± SE) were 47 ± 51 eels $100m^{-2}$ (range 0 – 197) and 1880–1530 g $100m^{-2}$ (range 0–5500) respectively. Small eels (mode 6 – 8 cm) dominated lower river sites, the mode tended to shift towards >30 cm upstream but there were very few females >40 cm (except in Bos Joli Reservoir, sampled with fyke nets).

It appears that good recruitment to the relatively short N. Wessex rivers helps to maintain stocks at quite high levels despite the glass eel fishery in the Parrett and other local estuaries. Possible impacts of the fishery in the Severn Estuary are discussed further in Chapter 3 and reach a similar conclusion for River Severn stocks. Data for the Warwickshire Avon are too sparse to reveal historic trends and in any case, stocks have been badly affected by pollution and poor habitat quality in the past (A. Starkie, Environment Agency, personal communication). Although data for the Bristol Avon are also sparse, stocks appear to be similar to those in the Severn even in the absence of a significant glass eel fishery. Density-dependent natural mortality would be high during the long tidally-assisted migration up the Severn, as discussed in Chapter 5. The large extent of estuarine habitat available could also act as a major habitat sink for current levels of recruitment. Similar possibilities are discussed later with respect to the Thames catchment (Section 2.22) and Essex rivers (Section 2.23). Poole Harbour may also form a sink that has led to reduced recruitment to the Frome and Piddle since the late 1970s-early 1980s (Chapter 3). This situation might have been exacerbated by overfishing of yellow and silver eels in the Harbour in the past, this could have reduced competition and hence the tendencies for young yellow eels to migrate upstream into the rivers. Christchurch Harbour may also form a habitat sink, influencing emigration into the influent Dorset Stour and Hampshire Avon. Fyke netting in the Harbour in the past might also have had some impacts. However, stocks in the Stour may have declined since the early-1990s due to poor water and habitat quality in the lower reaches forming a migration barrier. Conversely, stocks in the Hampshire Avon have not been so impacted, according to the data and hypotheses discussed in Section 2.19.

Although south-western rivers are not significantly affected by commercial fisheries, there is also some evidence that eel densities may have declined (and the proportions of larger and female eels have increased) at more upstream sites in some of the longer rivers. This is especially so in those which have large extents of estuarine habitat that can act as sinks for recruitment (e.g. the Taw-Torridge, Tamar, Exe and Poole Harbour systems). The possible differences between such large river/estuary systems and smaller, shorter rivers is returned to in later discussions.

2.21 Southern Region Rivers and Eel Fisheries

Many waters in this Region provide favourable habitats and eels are known to comprise a high proportion of the total fish density and biomass in some catchments, wetlands, estuaries and coastal waters. McKinnon and Potter (1993) estimated that in 1992, 23% of all NRA yellow and silver eel instrument licences were sold in the Region and that it yielded 33.2% (122.2 t) of the declared British catch. Licence sales over 1996-98 averaged 19.0% of country-wide totals.

Detailed information is required to make comparisons with the South West Region, but very few historical data sets for eels could be located. Few quantitative surveys including eels appear to have been carried out in the past or records cannot be located, whilst more recent surveys have focussed on salmonids. Similarly, although some fishermen claim catches have been declining, commercial fishery returns have only been collected in more recent years and these are of doubtful quality. Silver eel traps have been run for many in years in rivers such as the Test and Itchen, but owners are either unwilling to release historical data or the records are too incomplete to be of much use (Adrian Fewings, Environment Agency, personal communication). Furthermore, effort and efficiency have varied greatly over time, declining particularly in recent years because of the falling market values of silver eels. Some information is, however, available from an Environment Agency fish counter at Nursling on the River Test, showing counts of 327 eels moving downstream during 1996 and 666 during 1997 at various times between May and November. Many immigrants can, however, leave the river by other channels. Such counters could be beneficial in future R&D studies or monitoring, as discussed in Chapters 5 and 8.

The only means left to assess any trends in eel stocks are to examine more contemporary data from electric fishing surveys of representative rivers for any signs of abnormalities in density, biomass and population structure characteristics. Representative data sets are reviewed for the R. Adur, which is one of the few including some historical data, and for the Rother, Medway and Darent/Cray

2.21.1 River Adur

Data were provided by Southern Region fisheries staff, these were augmented by some records provided in the past to the Fisheries Classification Scheme R&D programme. The data were collected in 1978, 1882, 1984, 1991, 1993, 1997 and 1998 but come from different months and numbers of sites. Surveys also recorded varying combinations of density, biomass and, by derivation, lengths. To make best presentation of the information available, Fig. 2.29 shows the data for 1978 separately and those for the 1980s, early-1990s and late-1990s combined. No trends over 20 years can be distinguished. Densities in the Adur actually

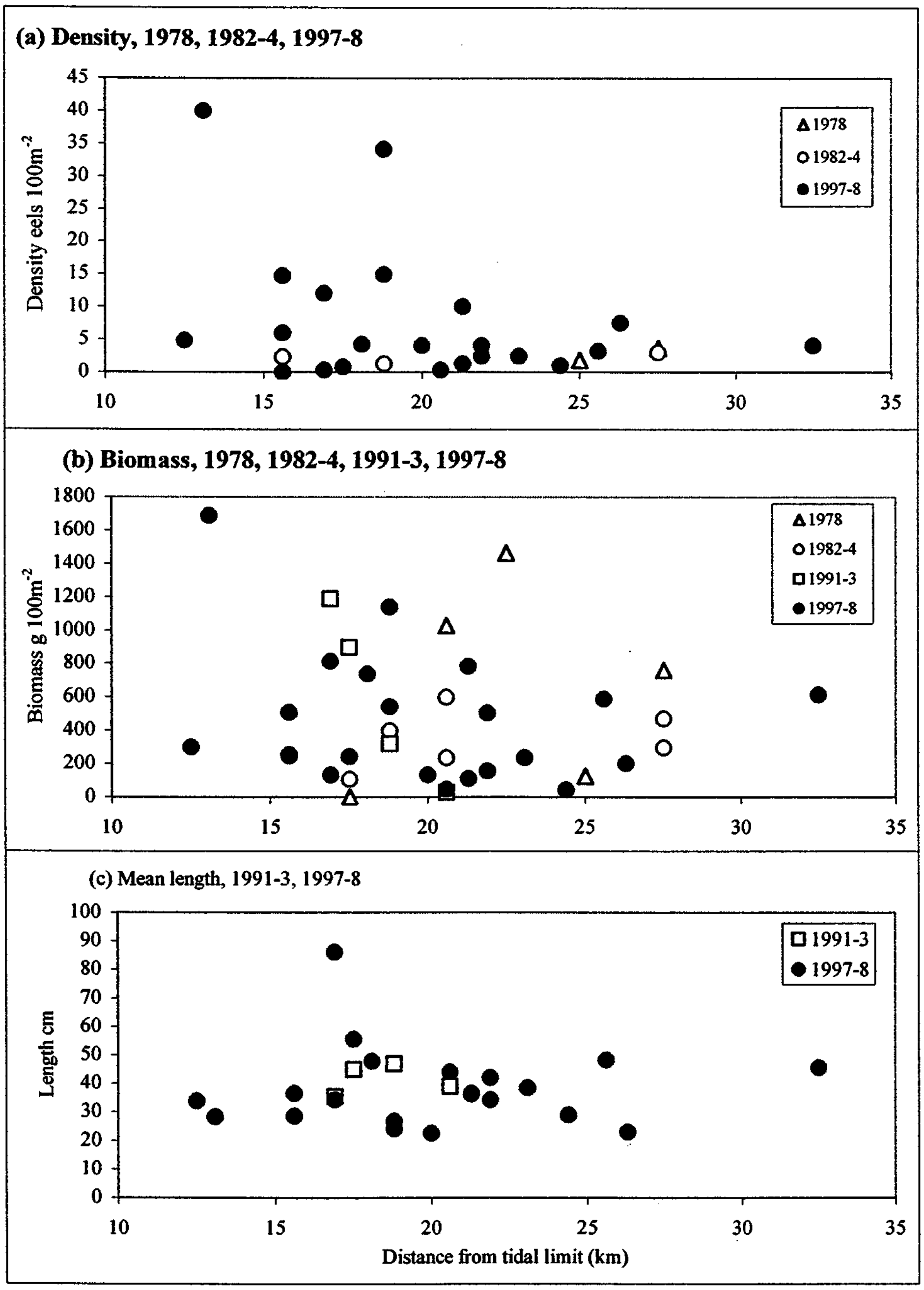


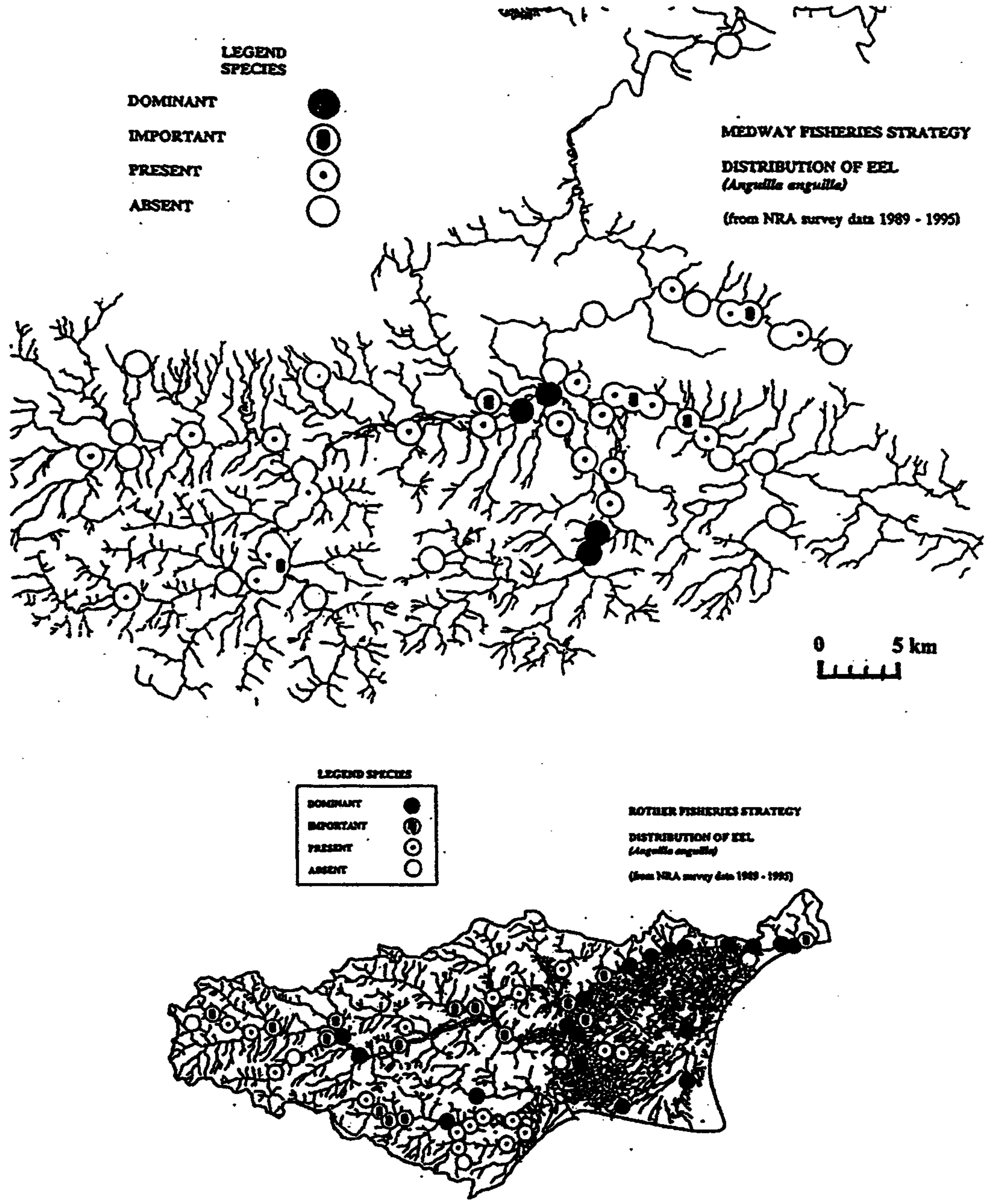
Fig. 2.29 Relationships between density, biomass and mean length of eels against distance from the tidal limit, from survey site data available for the R. Adur in 1978 and 1982-84 (combined) and 1991-93 (combined) and 1997-98 (combined)

appear higher in the 1990s than they did in 1978 and 1982-4, but this could relate to improvements in sampling, the greater number of sites surveyed and differences between month of survey. Biomasses and estimated mean lengths do not appear to be very different either, or to have changed significantly over time. The lower river is too deep to sample efficiently, but the densities found > +15 km are generally not very different from those shown for south-western rivers in Fig. 2.29, given that there are relatively fewer sites in the Adur with very high densities. The overall mean density (± SE) is relatively lower, at 6.81 ± 1.86 eels 100m^{-2} (N = 27), compared to the south-west overall mean of 12.50 ± 0.77 (N = 250), but the mean biomass is actually larger, at 581.17 ± 107.54 *v.* 376.39 ± 29.54 g 100m^{-2}. This implies that the carrying capacity of the Adur is, on average, greater than that of south-western rivers but that stocks tend to comprise fewer but larger (and relatively more female) eels. There has been no significant commercial exploitation in the Adur according McKinnon and Potter (1993) and local fisheries officers. The most likely explanation for the differences is that whilst overall habitat productivity is greater in the lowland Adur, recruitment to southern England has been lower than in the south-west over the time-span of the surveys due to distances from the Atlantic migration pathways.

2.21.2 Kent Area catchments

No historical records for eels were located, except those from electric fishing and migration studies on the Darent and Cray in the 1980s (Naismith & Knights, 1998, 1993), when they were included in the Thames Water Authority area. Discussion of these two rivers is included in the next section on the Thames catchment. Kent area survey results are expressed according to a four-category classification system based on relative biomass, density and number caught per minute compared to long-term averages, as tabulated in Fig. 2.30. Recent raw catch data have been inspected and the Rother and Medway chosen as examples to illustrate some tentative conclusions. The maps in Fig. 2.30 (covering the last three or four years of survey) can help indicate some possible trends that need further monitoring;-

(a) River Rother catchment: Eels are clearly found at their highest densities in the Rother catchment in the Romney Marsh drains, but especially in the encircling Royal Military Canal (Fig. 2.30a). Ease of entry for immigrants and the nature of the habitat favour high densities of small eels. Densities at some Canal sites can exceed 100 eels 100m^{-2} (biomass > 2000 g 100m^{-2}, derived mean length < 20-25 cm) and eels often comprise a relatively large proportion of local fish community density and/or biomass. Although eels (mainly larger ones) are present throughout most of the upper Rother system, densities are often < 1 eel 100m^{-2}, including the Tillingham and Brede tributaries relatively close to the sea in the south west of the catchment. Local eel fishermen have complained that catches have declined, but no robust quantitative data are available to support this view. Although migration barriers (due to tidal sluices and water-level management structures) probably have negative impacts, the densities deeper in the catchment appear low compared to the N.Wessex rivers and wetlands discussed in Section 2.5. It is possible that because the Rother is more remote from the main Atlantic migration routes, recruitment is relatively much lower than in the south west of England – and that the Marsh drains and, especially, the Royal Military Canal, now act as habitat sinks for immigrants, leaving the main catchment relatively impoverished.
The importance of local estuarine and coastal habitats for eels is unknown, but it is interesting to note that local stocks are large enough to have encouraged the development of a fishery in recent years, setting fyke nets at low tide off Greatstone Beach near Dungeness. Catch return estimates for this fishery have involved considerable under-reporting at times, according to off-the-record discussions with some fishermen.



CLASSIFICATION GRADE	QUALITY		
	BIOMASS (WEIGHT /M^2)	DENSITY (NO/M^2)	TEMPORAL DENSITY (NO/MINUTE)
GOOD	>20	>1	>8
ABOVE AVERAGE	6-20	0.16-1	2-8
BELOW AVERAGE	<6	<0.16	<2
POOR	Where more than one expressed quality was classification grade 3		

Fig. 2.30 Maps to illustrate the relative densities and biomasses of eels at different sites over recent years in the Rother and Medway catchments (from Kent Area, Environment Agency Southern Region reports, using the Kent Area fisheries classification system tabulated)

(b) River Medway catchment: This is a relatively large and productive lowland catchment which should be easily colonised from the estuary. Fig. 2.30b shows, however, that densities are relatively low compared to comparable-sized systems discussed earlier. There is anecdotal and circumstantial evidence (e.g. from old eel traps on mills) that stocks were historically very much larger than they are today. Insufficient recruitment is implicated, possibly exacerbated by the large amount of habitat available to emigrants in the Thames and Medway Estuaries and N. Kent Marshes. This, it is suggested in the next Section, has had similar impacts on the Thames catchment.

2.22 The Thames Estuary and Freshwater Catchment

The Thames is a major catchment and must have supported good eels stocks in past centuries. For example, there are records in the Domesday Book of 1086 for many eel traps deep in the catchment (Naismith, 1992; Naismith & Knights, 1993). Records of eel bucks and large glass eel runs show it supported good stocks prior to the gross pollution of the Estuary during the Industrial Revolution. This acted as a barrier to glass eel recruitment and eels were generally absent or scarce above the tidal limit until sewage effluent and other pollution problems were overcome in the late 1960s (Andrews *et al.*, 1982; Naismith & Knights, 1993). Stocks in the Estuary increased and a commercial yellow-silver eel fyke net fishery developed in the 1980s. Detailed studies were conducted on stocks by electric fishing and fyke netting during 1981-88 and revealed a restricted distribution throughout the catchment (Naismith, 1992; Naismith & Knights, 1988a and b, 1990 a and b, 1993, 1994). This section deals with historical information on freshwater stocks derived from electric fishing and fyke net survey data, then with power station screen and commercial catch data for the Estuary.

2.22.1 Fishery survey data

TWA, NRA and Environment Agency survey data were collated by G. Armstrong (EA, formerly Thames Region) and analysed for Thames Region Fisheries by WRc plc in 1998 (Contract No. 11096, Naismith, 1998). The contract report includes survey data from 1974 to 1997, involving 1415 individual electric fishing sites from river source to the tidal limit. Details of the study are given in the WRc report and only the key features of importance to the current R&D programme are summarised below. It must be emphasised that no studies have been able to efficiently sample deeper waters and small eels. However, some more recent information has been gained from reports of boom boat electric fishing surveys of the lower freshwater Thames (Environment Agency Thames Region Fisheries-South East internal report).

(a) Presence-absence data

The data available are limited in their ability to accurately quantify temporal trends. Many parts of the catchment have only been surveyed for the first time in recent years and only a small proportion have been resurveyed, many of these sites being ones containing no eels. However, from comparative mapping over time (Fig. 2.31), it appears that distributions have changed little since 1974 (Naismith, 1998).

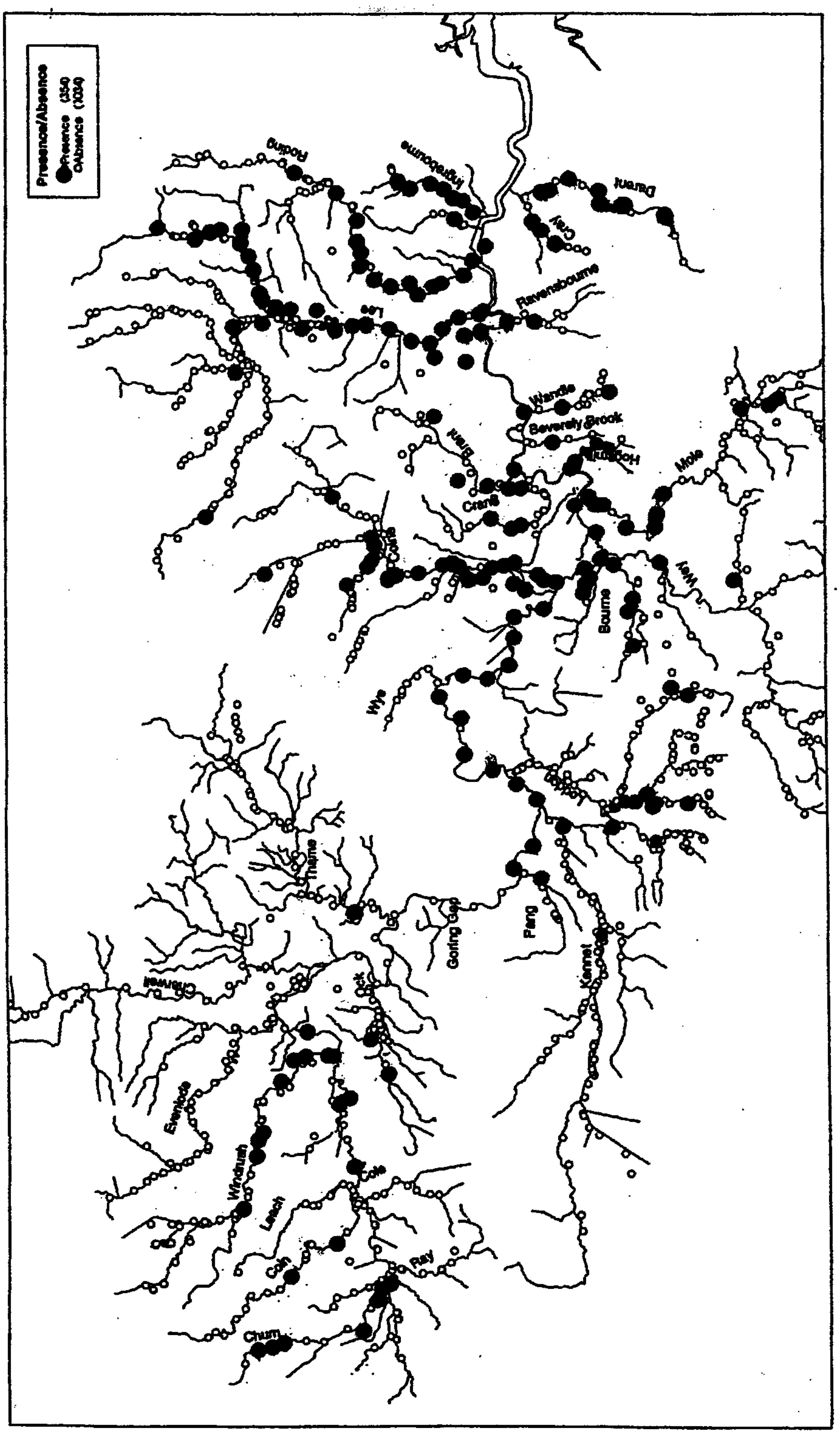


Fig. 2.31 Presence-absence of eels in the Thames Catchment, from all sites surveyed between 1974 and 1997 (from Naismith, 1998)

The key conclusions from the data shown in Fig. 2.31 are that:-

- eels are present throughout the estuary but have only been found in 25% of freshwater sites surveyed. This is a much lower incidence than in other large catchments

- such as the Severn, where Aprahamian (1988) found eels in 93% of estuary tributary and 69% of river sites surveyed.

- they are well distributed in a core zone (encompassing the lower freshwater Thames and tributaries), but largely absent above the Goring Gap.

- distribution would be expected to be wider (even if at low density and sporadic occurrence) in some waters were it not for physical barriers to migration at different distances upstream of the tidal limits. Key pinch points are on the main Thames at Whitchurch (just above the confluence with the Kennet), the upper Roding, middle Mole, lower Wey, Colne tributaries, lower Wye and lower Loddon. The distributions on the Thames, Colne and Lee are similar, to about +70 to 95 km and upstream of 20 to 26 weirs. A suspected barrier on the Mole impacts upstream migration at +25 km

- small isolated populations of eels are found deep into the catchment (e.g. in the Windrush and Churn). However, those looked at in detail have been shown to comprise eels of similar (large) sizes and ages and the majority are probably the result of unrecorded small isolated stockings in the past (Naismith & Knights, 1993)

- there is no evidence for any substantial trends of change in the overall distribution from the 1970s to the late 1990s. In some areas, there is some evidence of an extension of range (e.g. in the Wandle, Ingreborne and Mole) but generally, it appears that the penetration into unused habitat has been slow to negligible. This probably possibly reflects the overall declines in glass eel recruitment since the late 1970s-early 1980s (Section 2.2), leading to relatively low densities in downstream river stretches and hence to reduced tendencies to migrate upstream. The possible impacts of the estuarine fishery are discussed in the next Section

- thus it is appears that the ~90 km of main river and colonised tributaries (plus the Estuary, see below) represent the total habitat necessary to accommodate current recruitment.

(b) Population characteristics

Small eels are less efficiently sampled by electric fishing and fyke nets generally only catch eels > 25 cm and results can only be expressed in relative CPUE terms (Naismith & Knights, 1990b; Knights *et al.*, 1996). However, catch data (remembering that routine surveys are less efficient than eel-specific ones) show that density and biomass are often variable at similar distances from the tidal limit. This probably reflects differences in habitat suitability and catch efficiency, especially in the sampling of smaller eels generally and sampling of all eels in deeper waters.

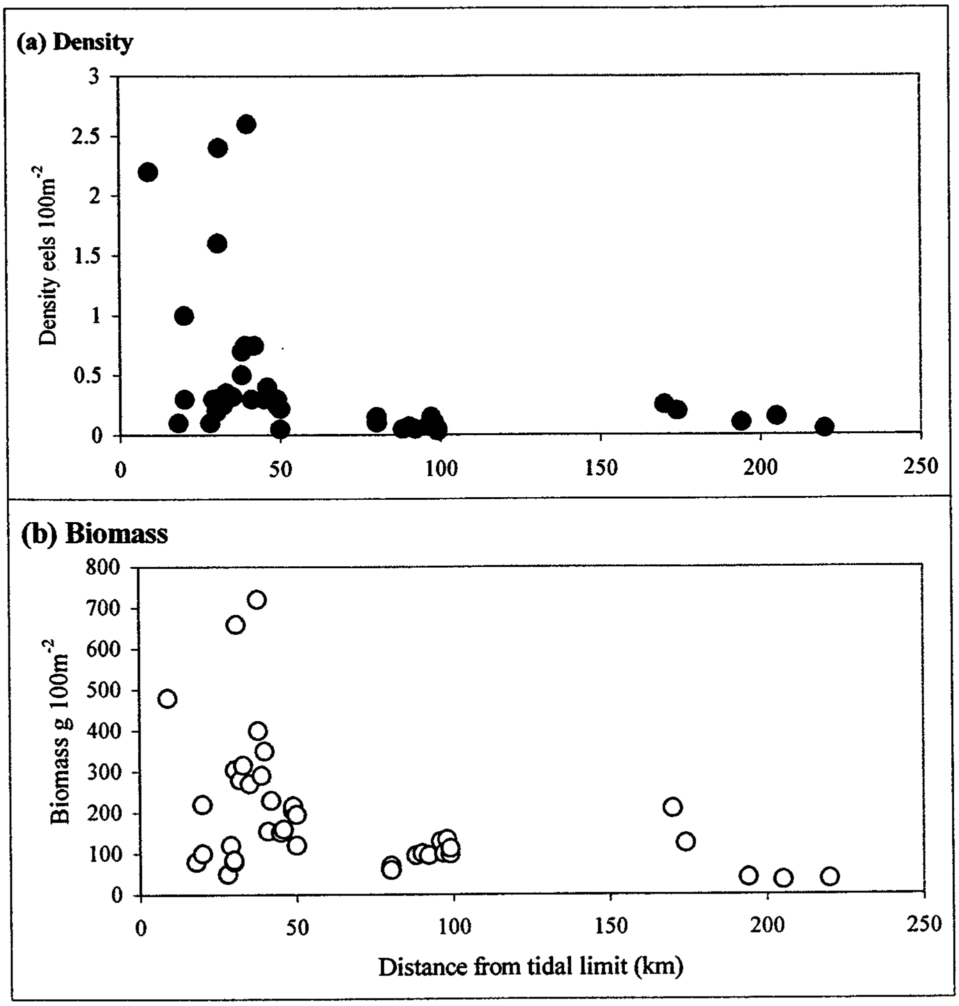
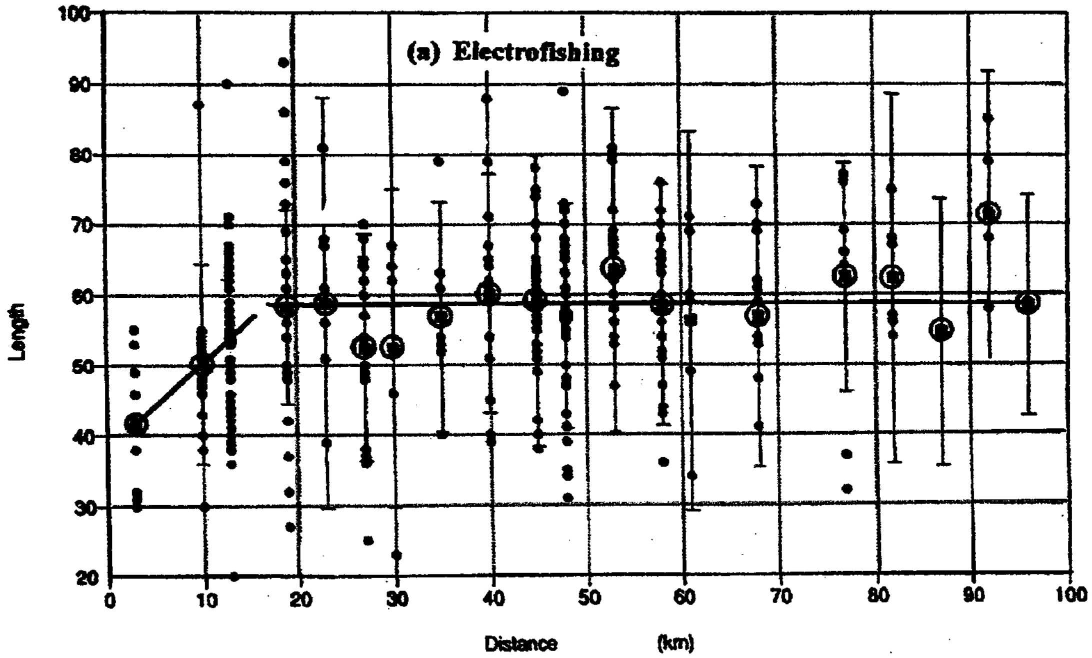


Fig. 2.32 (a) Density (eels 100m^{-2}) and (b) biomass (g 100m^{-2}) of eels (at sites where found) with distance (km) from the tidal limit in the fresh water Thames and tributaries (from Naismith & Knights, 1993)

Key trends are;-

- density and biomass decline with distance upstream of the tidal limit (Fig. 2.32), from ~ 0.5 eels 100m^{-2} and 200 g 100m^{-2}, with the decline being most marked beyond +50 km from the tidal limit at Teddington There are, however, hotspots, especially in some lower tributary sites nearest to the tidal limit. Those found around +200 km are due to localised past stockings, as discussed above

- boom boat electric fishing has revealed that large eels are found mainly in backwater and impounded habitats in the lower freshwater main river, rather than weir pools. Hotspots are occasionally found in the main channel in margin habitats. Eels comprise about 7-11% of the total main river fish biomass and 5% of the density, giving estimates for eels of 56-126 g 100m^{-2} and 0.13-0.28 eels 100m^{-2}. Overall, densities and biomasses are low relative to other southern rivers reviewed earlier.



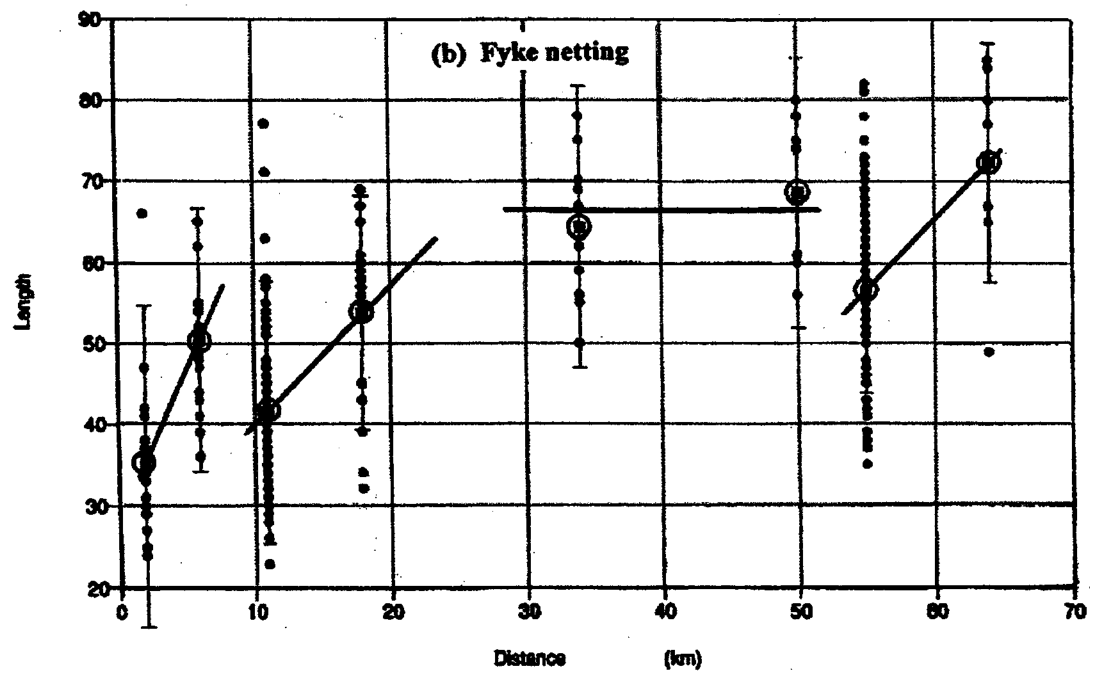


Fig. 2.33 Distribution of mean eel lengths (circles) and 5-95% range estimated from spatial sampling errors from raw data (dots) in different parts of the freshwater Thames: (a) from boom boat electric fishing surveys (distances from the tidal limits measured to mid-point between weirs) and (b) from fyke net survey data gathered in 1986 (adapted from Naismith, 1998)

- average size tends to increase and larger slower growing females >40 cm predominate upstream of the tidal limit (Fig. 2.33), this tends to maintain biomasses at relatively high levels despite declining densities.

- fyke net CPUE data (Fig. 2.34) show a similar pattern for freshwaters, but reveal that the Estuary supports high densities of mobile eels, especially in sites such as the outfall of Beckton STW.

- in the Estuary, the middle reaches (-40 to -55 km, below the tidal limit) are dominated by smaller eels mainly maturing as males (modal lengths about 32 and 37 cm respectively), whereas the outer reach (> -55 km) populations comprise almost all larger females. Netting in inner reach sites occasionally caught larger female silver eels, probably on their seaward migration (Naismith & Knights, 1990a).

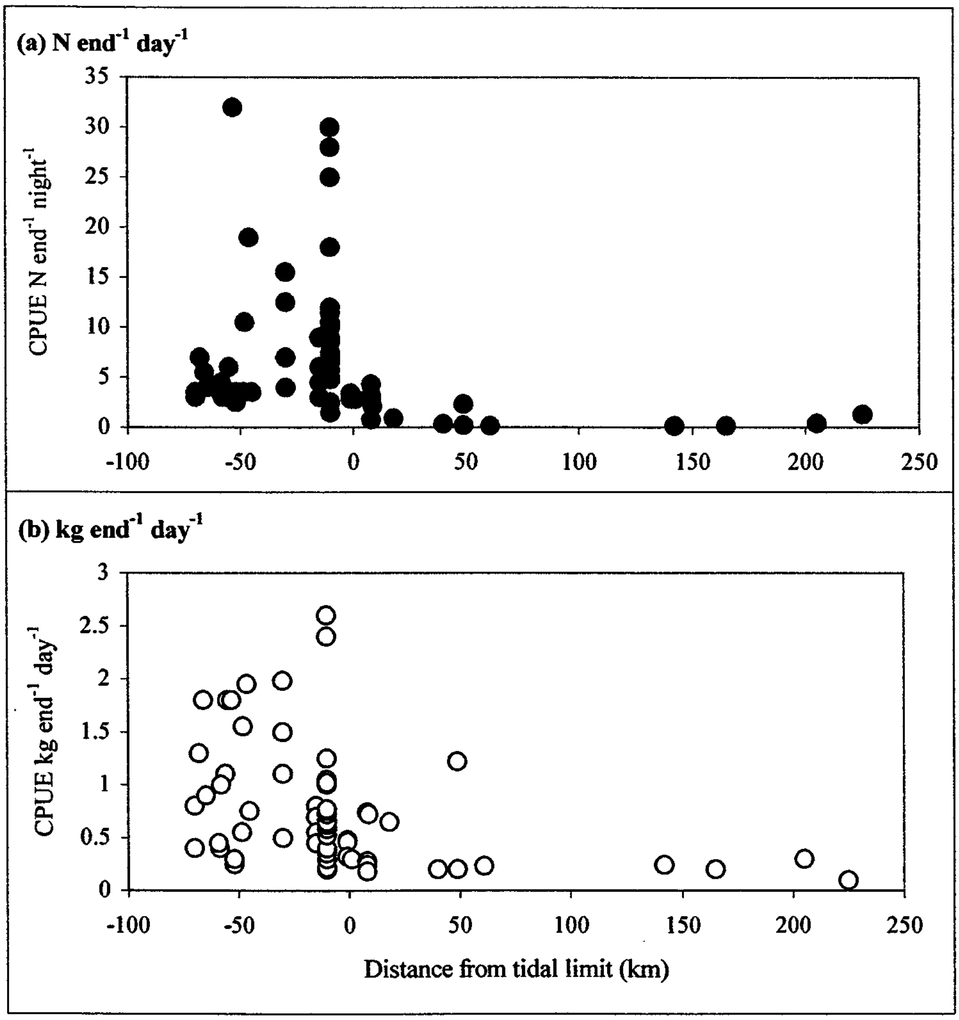


Fig. 2.34 Fyke net CPUE (where eels were found) as (a) eels end^{-1} day^{-1} and (b) kg end^{-1} day^{-1} with distance downstream (–km) and upstream (+km) of the tidal limit at Teddington Weir in the Thames catchment and estuary (after Naismith & Knights, 1993)

- growth rates tend to be highest in the Estuary (range of means per reach 4.67-6.07 cm yr^{-1}), relatively slower in the main river (3.89-3.80 cm yr^{-1}), but up to 5.09-5.24 cm yr^{-1} in some tributaries (Naismith & Knights, 1993). These differences would reflect the productivity and more favourable temperature regimes and length of the growing season in the different habitats. This in turn correlates with density, i.e. eels in high density populations tend to grow faster and mature and emigrate at 6-8 years. Females tend to mature at 6-13 years in the Estuary and > 15 years in freshwaters.

(c) Conclusions from fishery surveys

Eels are numerous in the Thames Estuary but populations in freshwaters are relatively impoverished. The mean density and biomass at < +50 km are below 0.6 eels $100m^{-2}$ and 250 g $100m^{-2}$. This biomass is relatively low compared to the mean values of 300 to 400 g $100m^{-2}$ pertaining in the south-western and other rivers reviewed.

Upstream recruitment appears relatively low, only ~ 10,000 migrant elvers and juvenile yellow eels (but no glass eels) being trapped at six weir sites on lower river sites during 1983-88 (Naismith & Knights, 1988). Numbers of migrants fell with distance upstream and with number of barriers, lengths and ages were up to 20-30 cm and 4-8 years respectively. Few < +1 migrants were trapped, most appearing to spend at least their first year in the Estuary or lower river regions. Overall, recruitment patterns were similar to those observed in the Severn and Avon by White and Knights (1994), but numbers of migrants were much lower. This lack of upstream migration recruitment is probably the main reason why the distribution range in the Thames has not increased to the extent that might be expected following removal of pollution barriers in the Estuary. The Thames flows into the southern North Sea and receives lower natural recruitment than the south-west facing rivers (Tesch, 1977; Deelder, 1988). However, east coast rivers, such as those of Essex (Section 2.23), have higher densities. It is possible that the Thames estuary and lower river and tributaries provide sufficient habitat for the recruitment levels pertaining since the cleaning up of the estuary. Recruits from the North Sea also have the choice of entering the Medway and estuarine creeks and marshlands, before reaching the main Thames. Differences in the numbers and types of migration barriers could also be important factors.

The sporadic survey data reviewed above are not ideal for determining any long term time trends in actual numbers of eels in the Thames Estuary and freshwater catchment. The next sections deal with monthly and annual data sets for power station screen and commercial catches in the Estuary.

2.22.2 Data for eel catches on the screens of West Thurrock Power Station

West Thurrock Power Station was situated on the north bank of the Thames Estuary, 35.5 km downstream of London Bridge. The screens caught samples of fish comparable to those found by beam-trawling in the same region (Andrews *et al.*, 1982; Araujo *et al.*, 1998). Samples were collected fairly regularly between 1994 until the Station's closure in 1993, from 0.5 h before low water on spring tides to 3.5 h after. Catches were then standardised to a CPUE of eels per 100 million gallons, to allow for variations in pump and screen operations.

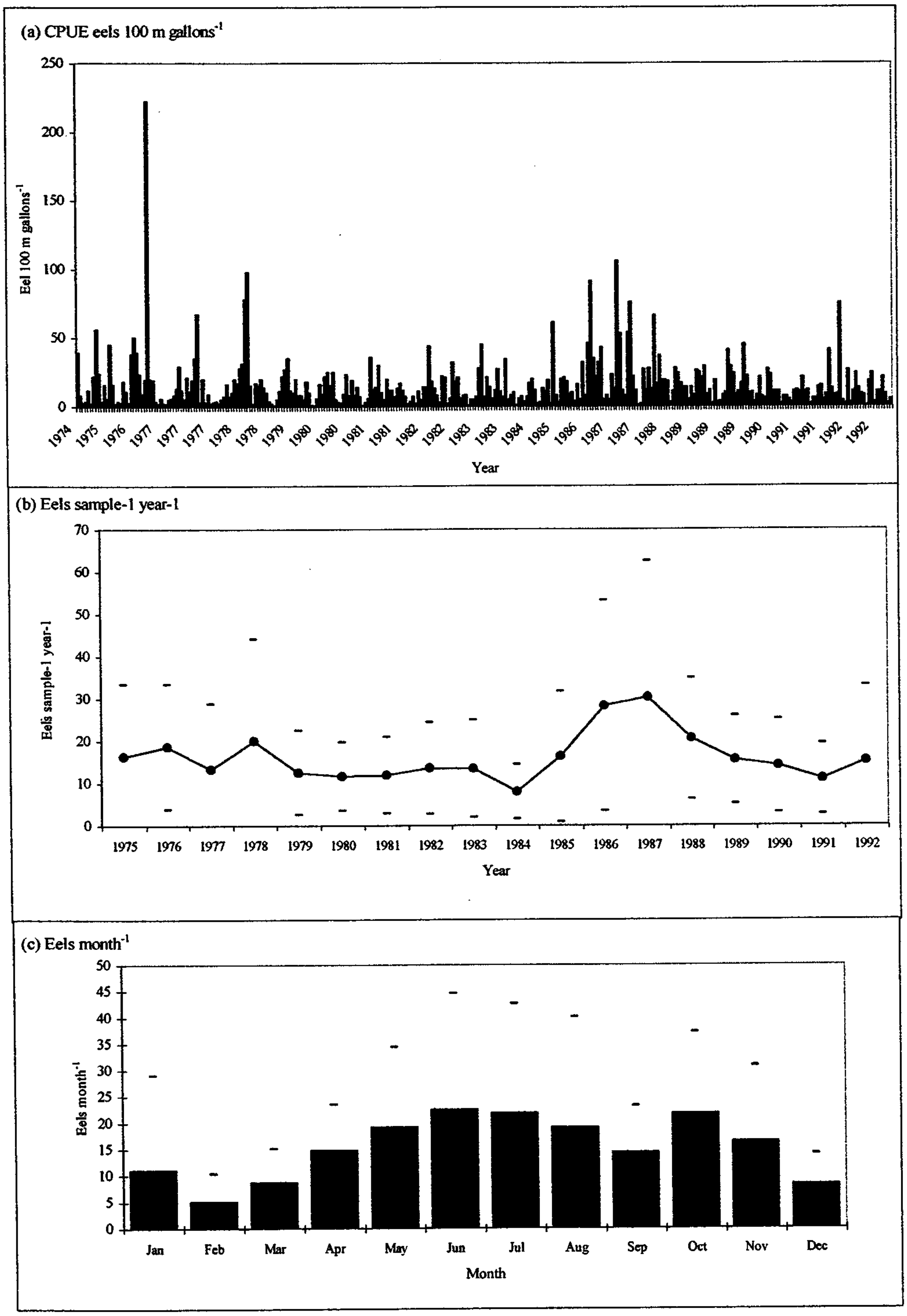


Fig. 2.35 West Thurrock Power Station Screen eel sampling: (a) CPUE as eels 100 million gallons^{-1} sample^{-1} (301 sampling sessions) between August 1974 and March 1993, (b) mean eels sample^{-1} year^{-1} ($\pm$ SD) for complete years between 1975-92 (minus extreme outlier of 222 for October 1979), (c) mean N/month between 1975-92 (minus individual sample outliers > 60). See text for further explanation.

Eels were the fifteenth most abundant species (mean CPUE of 5.3 $month^{-1}$) between 1980-89 according to the detailed surveys of Araujo *et al.* (1998). Over the 10 years, eels showed statistically significantly different peaks of abundance in June and October and troughs in February, but no one year mean was highly significantly different from the others, unlike all other species except dab, *Limanda limanda* (L.). A similar seasonal pattern is seen in the complete data set of 307 samples covering August 1974 to March 1993 obtained from M. Thomas (Environment Agency Thames Region) (Fig. 2.35). Data for nearby sites and some approximately matching dates show salinity was generally about 10-15 ppt but varied between 1-21 ppt, probably due to changes in river discharge. Average temperatures were lowest in January-February (7-9°C) and highest in late summer (20-23°C). Peaks of relative abundance of eels occurred in June (Fig. 2.53c), as was also found by fyke netting by Naismith and Knights (1988). These were possibly associated more with local prey availability (especially small crustacea, see Andrews *et al.*, 1982) than temperatures (June average ~ 16-20°C). The October peaks possibly relate to captures of silver emigrants or to local stock migrations to over-wintering areas.

Multivariate analysis by Araujo *et al.* (1998) showed relative abundance of eels increased significantly in 1985-87, following a stable period between 1980-84. No length or age data were collected for eels but those caught in fyke nets in the same area of the Estuary (see previous Section) were mainly females > 37-45 cm and mean ages of 5-6 years. It is tempting to seek possible correlations between the peak screen catches in 1985-7 to the peaks in glass eel recruitment in 1978-79, suggested by the data reviewed in Section 2.2. However, individual CPUEs were very variable, regardless of month. Some were extremely large relative to the mean ± SD over 1974-93 of 16.4 ± 19.7 eels/100 million gallons/sampling session. There are a number of confounding factors which distort the data and prevent meaningful statistical analysis and interpretations:-

- there was variability in week of sampling and the total number of samples taken varied between years (from 10 in 1984 to 21 in 1978 and 1989) and for months (from 19 samples overall for February to 31 for October)

- capture efficiency was probably variable and errors were introduced when converting actual catches to eels per 100 m gallons. For example, Andrews *et al.* (1982) noted 71 adult eels were trapped in 2 hours on 28 October 1976. The final catch was 89 and this value was converted into a CPUE of 222 per 100 m gallons. This extreme outlier is very obvious in Fig. 2.35a and has been excluded in Figs. 2.35 b and c. Other peaks were subject to the same influences and Fig. 2.35c is plotted omitting 10 statistical outliers > 60, i.e. those for October 1977 (222), August 1978 (98 and 78), June 1985 (61), July 1986 (91), June 1987 (106), October 1987 (76), May 1988 (66) and January 1992 (75).

Repitition of ANOVA tests for significance of month and year variations for the complete data set confirmed the results of Aruajo *et al.* (1998), i.e. that monthly CPUEs were significantly different ($F = 5.5$, $P < 0.01$) but not inter-year differences, with or without removal of outliers in both cases. Repitition of multivariate analyses following removal of outliers showed no significant trends in annual relative abundance. Variability of sampling over time prevented meaningful application of time series analysis. Local temporal and spatial factors may have produced the occasional very high CPUEs. However, the available salinity, temperature and dissolved oxygen data are insufficient to assess any inter-relationships with CPUEs.

In conclusion, the West Thurrock data are not amenable to robust statistical analysis because of high variability in catches, due to unknown natural causes or to sampling influences and CPUE conversion errors. However, given these provisos, there is no clear evidence of any major declines in eel stocks in the Thames Estuary between 1974-93. Incidentally, the results also show that entrapment on the screens would have had little impact on overall stocks or migrants in the Estuary.

2.22.3 Data for eel catches in the commercial eel fishery in the Thames Estuary

According to licence data and local fisheries officers, exploitation in the freshwater rivers of the Thames catchment has been very low. A commercial fishery developed in the Estuary below Tower Bridge in the early 1980s and the former National Rivers Authority started collecting catch returns in 1982 for fyke nets, traps and pair trawls. Licensing was instituted in 1984. Declared catches have comprised about 10% of the totals for England and Wales. Catches are mainly sold for local processing, with some being exported to the Netherlands but these may not have been recorded in UK export data. Local markets for jellied eels have, however, been shrinking slowly over time, as have exports, according to local fishermen, merchants and processors.

Fyke netting (catching eels > 30-33 cm, particularly from small boats over mudflats in the middle-lower Estuary, where females dominate) has supplied the largest component of total catches in most (~ 90%) years (Fig. 2.36a). Licence information shows a core of fishermen have worked most seasons, with new ones entering but then often leaving after a few years. Returns claimed total catches of approximately 10 t y^{-1} until 1986, CPUEs being maintained at ~ 0.3-0.4 kg end^{-1} day^{-1} by more licensees but with decreasing effort (as fyke end days $year^{-1}$, Fig. 2.36b). The number of licensees, effort and catches appeared to decline in the late 1980s-early 1990s but then increase again in the mid-1990s. The declines in the late 1980s contrast to the possible increases in West Thurrock PS catches, discussed above. However, any conclusions about any impacts overfishing and/or poor recruitment between 1987 and 1993 are influenced by a number of confounding factors;-

- catch returns are known to be incomplete and of poor quality, due to poor record keeping and unwillingness to divulge financially sensitive details. Fyke net CPUEs of ~ 0.4-0.5 kg end^{-1} day^{-1} would be expected in the Estuary from the studies of Naismith and Knights (1990b) and total catches are probably under-estimated mainly because of under-claiming of total effort and omission of some exceptionally large individual catches. McKinnon and Potter (1993), for example, estimated the catch at 13.2 t in 1992, compared to a declared catch of 4.08 t. Furthermore, Thames Region fisheries officers suggest that the total catch in 1994 were inflated by one ‘highly suspect’ large return from one fisherman. If this return is excluded, fyke net catches in 1994 would total 5.63 t. However, a fisherman has admitted to making an accurate (and relatively large) return at about that time. He was subsequently advised by other fishermen not to do this in the future. The apparently high CPUEs in 1982 (derived from voluntary returns prior to the introduction of licensing) and 1994 may be relatively true reflections of actual catches.

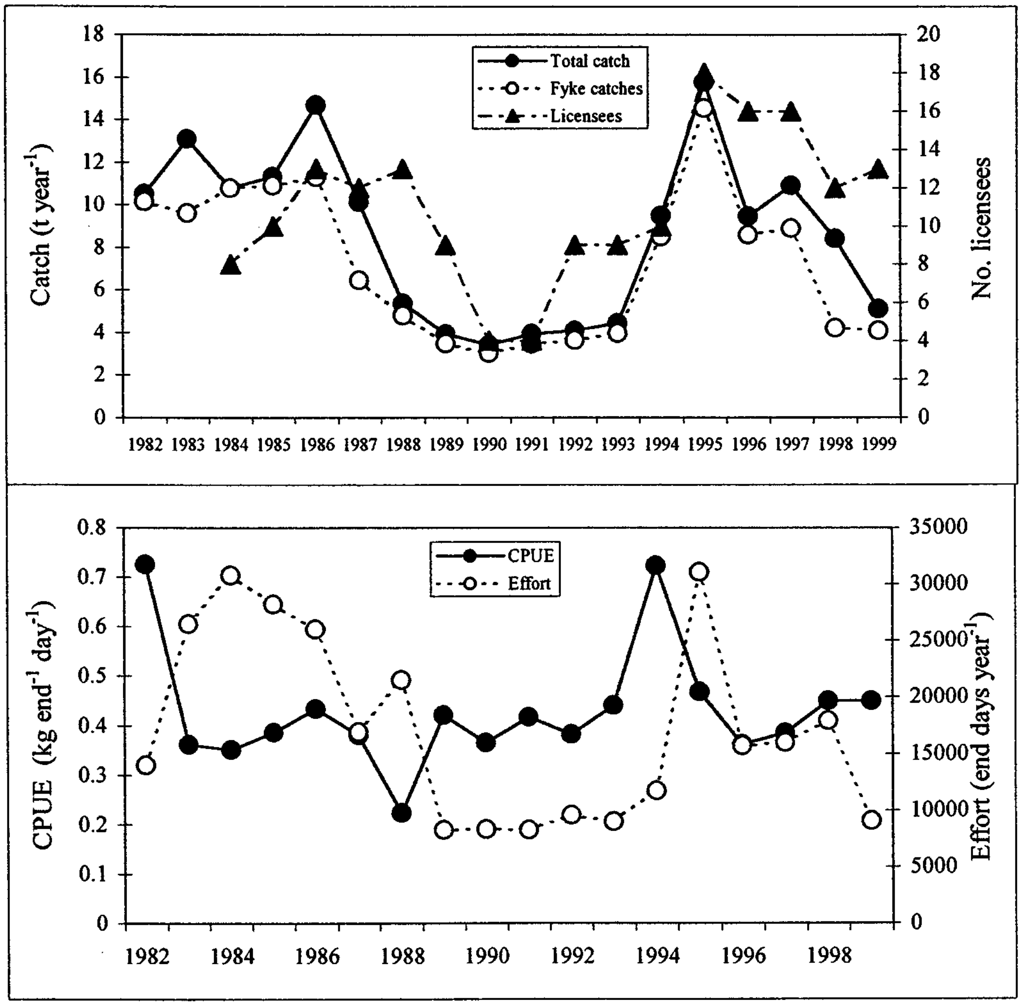


Fig. 2.36 Annual data for the Thames Estuary eel fishery, 1982-99; (a) total (all methods) and fyke net catches (t year^{-1}) and number of licensees, (b) fyke net CPUE (kg end^{-1} day^{-1}) and fishing effort (fyke net days year^{-1})

- the apparent declines in 1987, 1988 and 1989 were significantly affected by poor return rates to Thames Region.

- dieldrin residue levels found in Thames eels led to warnings being issued in 1989 by local Environment Health Officers to limit weekly consumption. This had negative impacts on local sales and exports to the Netherlands for a number of years, and hence on effort and catches

- according to anecdotal evidence, unlicensed nets are not uncommon and the actual total catch may have been as much as ten times the reported ones in good years. It has not been possible to check the catch returns made by the few licensed pair trawlers and the catches made by the occasional illegal boat are unknown.

There have been major declines in total catches since 1995 (Fig. 2.36), but these are not due to shortages in availability of eels but to major reductions in fishing effort (in end days^{-1} year^{-1}). CPUEs have not fallen and fishermen contacted say that although they have taken out licences, market demands and prices have often been at such low levels, it has not been economic to go fishing.

In conclusion, it appears that although catch return data suggest a decline in eel stocks in the Thames Estuary in the late 1980s-early 1990s, this is probably explained by a combination of interfering factors, particularly low return rates and the impacts of the dieldrin scare in 1989. The falls in effort and catches after 1995 mirror those for the rest of England and Wales discussed in Section 2.2.2, where market forces have been key factors. The steady decline in the sale of jellied eels and imports by processors of relatively cheap frozen eels from New Zealand, N. America and, more recently, from China have particularly affected London markets.

2.22.4 Modelling of fishery mortality in the commercial eel fishery

Naismith and Knights (1990b) did not note any changes in their Estuary fyke net catches between 1985-88. Some variations in local catches were put down to short-term over-fishing of local areas, plus natural migrations of populations due to food availability, and to movements of salinity and temperature fronts and oxygen sags. The study was designed to assess the possible impacts of fishing mortality on Estuary stocks. Mark-recapture studies were ineffective because of very poor recapture rates. Instead, a compartmental length-based cohort model was developed to assess impacts of natural and fishing mortality and migration on different life stages of males and females, comparing fyke net catch data from fished and unfished regions. It was concluded that fishing mortality effects were low and masked by those due to natural mortality and, especially, to emigration/immigration and local migrations. Density-dependent mortality appeared to have minor effects in such an open and dynamic ecosystem. It was further concluded that although the stock was not being over-exploited, glass eel recruitment was probably relatively low.

2.22.5 Conclusions

Although the data sets available for the Thames catchment and Estuary are extensive and cover reasonable time scales, there are disadvantages to each because of lack of consistency and accuracy and interferences by other factors. The picture is one of greatly reduced freshwater stocks in comparison with pre-Industrial Revolution times, but overall there have been no major changes since the late-1970s. After removal of pollution barriers to immigration since the 1960s, the range and density of freshwater stocks would have been expected to have increased more than it has. This contrasts to what appear to be relatively healthy and stable estuarine populations.

The overall conclusion is that as recruitment of glass eels to Britain has declined since the late 1970s (Section 2.2.2), but the Thames flowing into the North Sea has suffered an impoverished recruitment relative to that for Atlantic-facing river systems. The numbers of immigrants appear to be sufficient to meet the carrying capacity of the Estuary and ~90 km of lower freshwater river and tributaries, but that density-dependent competition pressures are insufficient to cause more juveniles to migrate upstream in successive years after first arrival (Naismith & Knights, 1988). It is possible this situation has been exacerbated by removal of eels by the estuarine fishery but as discussed above, the fishing mortality appears low relative to natural mortality.

In relation to the non-tidal stocks, the tendency for low density populations to produce more and larger females of high fecundity can be seen as a evolutionary strategy for such a catadromous species, helping to compensate for low recruitment to continental waters from the Atlantic. Incidentally, this suggests that low density stocking of waters in the upper Thames catchment could be a useful management option to enhance production and escapement of spawners. A simple calculation shows that there are some 2000 km of under-utilised river which at an average width of 5 m represents an area of 1 x 10^{-7} m^2. At a stocking density as low as 1 eel $100m^{-2}$, this area could potentially yield 1 x 10^5 females, carrying a potential 1 x 10^{11} eggs. Stocking is discussed further in Chapter 7.

Stocks in the freshwater catchment should continue to be monitored to assess possible changes in distribution as an indirect measure of recruitment and/or overfishing. This will also help to assess the impacts of migration barriers and the possible benefits of passes (and of upstream stocking). Key sites on the boundaries of the present core distribution area recommended by Naismith (1998) include ones on the upper Roding, middle Mole, lower Wey, tributaries of the Colne, the lower Wye, the lower Loddon and the River Thames above Whitchurch. Uses of Thames sites for monitoring is discussed further in Section 8.4.5.

Despite the drawbacks noted above, instrument licence sales and catch return data should continue to be collected for the Estuary fishery to help monitor general trends and to help predict future changes. As discussed in Section 2.22.3, falling prices are having negative impacts on the fishery, but this situation might change in the future. More and better information is particularly required on pair trawl catches (both licensed and unlicensed). Periodic repetition of fyke net surveys in key sites (e.g. at Beckton Outfalls) might also be of value in monitoring changes in stocks and possible fishery impacts. Such surveys are, however, very resource costly for the precision and accuracy achieved (see Naismith & Knights, 1990a,b and Knights *et al.*, 1996, plus further discussions in Section 8.4.3).

Although the resource implications are great, monitoring of screen catches at an estuarine power station should be considered. Catches of eels at Lots Road (Chelsea) are insignificant but jointly funded surveys at a lower estuary power station (e.g. Tilbury or Barking) would also provide information on juveniles of commercially important species using the Estuary as a nursery ground.

Finally, any monitoring of Thames stocks should ideally be integrated with that of other easterly-facing rivers, such as those in Essex (see Section 2.23) – and if possible with the Medway, North Kent Marshes and other waters which share recruitment from the southern North Sea.

2.23 The rivers of eastern England

2.23.1 The rivers of Essex

Extensive electric fishing survey records were obtained from the Environment Agency Anglian Region for Essex rivers covering 1984-98. The deeper and more saline lower river reaches are not easily fished but an initial presentation of average biomass for each of the major rivers for different survey years (Fig. 2.37) does not reveal any consistent trends in stocks over 14 years. There was an apparently major and consistent decline between 1984 and 1996 in the Holbrook. Stocks are, however, probably prone to large short-term fluctuations because the Holbrook is only 1.5 km long and easily influenced by migrations to and from the estuary and, possibly, by discharges from Alton Water. Survey data for other short coastal rivers was combined into year groups to compensate for small sample sizes. Mean density has appeared to decline since 1984-85 (from 2.08 to 1.65 eels $100m^{-2}$), biomass to remain fairly steady (at 288.22 to 245.42 g $100m^{-2}$) and mean weight to increase (from 140.2 to 201.9 g). However, significant changes could not be demonstrated because of the small overall sample sizes (8 – 23) and high variability of the data.

Looking in more detail at data for different stretches of each of the main rivers did not reveal any significant trends either. Fig. 2.38, for example, shows the changes between survey years for mean density, biomass and mean weight (derived from biomass ÷ density) in the main river Stour, the Brett and Box tributaries and the upper tributaries. There are no consistent trends between years, although a clear decline in density and biomass occurs with distance from tidal limits, whilst average weight increases fairly steadily. It appears that populations are dominated by females of >100 g, >40 cm. This is confirmed by an intensive two year study of eel populations between November 1985 and November 1987 at sites below (+10 km) and above (+20 km) Hadleigh on the R. Brett and below (+17 km) and above (+30 km) Chelmsford on the Chelmer (Barak & Mason, 1992). Modal lengths were > 40-65 cm and modal ages 8–11 years, at densities of 0.5 – 5.18 eels $100m^{-2}$ and biomasses of 350-2100 g $100m^{-2}$. Although growth rates in the two rivers were similar (4 to 5 cm $year^{-1}$), eels aged between 9 and 13 years in the upstream sites were significantly larger in both length and weight, relatively more of them being older females > 60 cm.

Densities and biomasses (± SD) for 171 sites in the Essex rivers, at 1.51 ± 1.90 eels $100m^{-2}$ and 367.82 ± 373.75 g $100m^{-2}$, are higher than those in the Thames catchment. This is possibly due to the fact, as discussed earlier, that recruits entering the Thames Estuary have a relatively large area of habitat and more influent rivers in the estuary to colonise before entering the main freshwater river and more distant headwaters. Although densities are generally lower in the Essex rivers than in south-western rivers (Section 2.20), overall biomasses are in fact not significantly different. Thus although the Essex rivers may not be receiving the same amount of recruitment, habitat carrying capacities appear to still being met, albeit by relatively large (and many female) eels. Small eels seem very scarce, although this is probably partly due to sampling problems in deeper waters in downstream sites. It is also possible that many eels spend their earlier years in the extensive marshlands and shallow estuarine and coastal areas along this part of England (as discussed further in the next section). Stocks in the coastal areas must be large, e.g. a local colony of coastal-nesting cormorants has been estimated to feed preferentially on eels and to consume in the region of 50 tonnes per year (David Carss, ITE, personal communication).

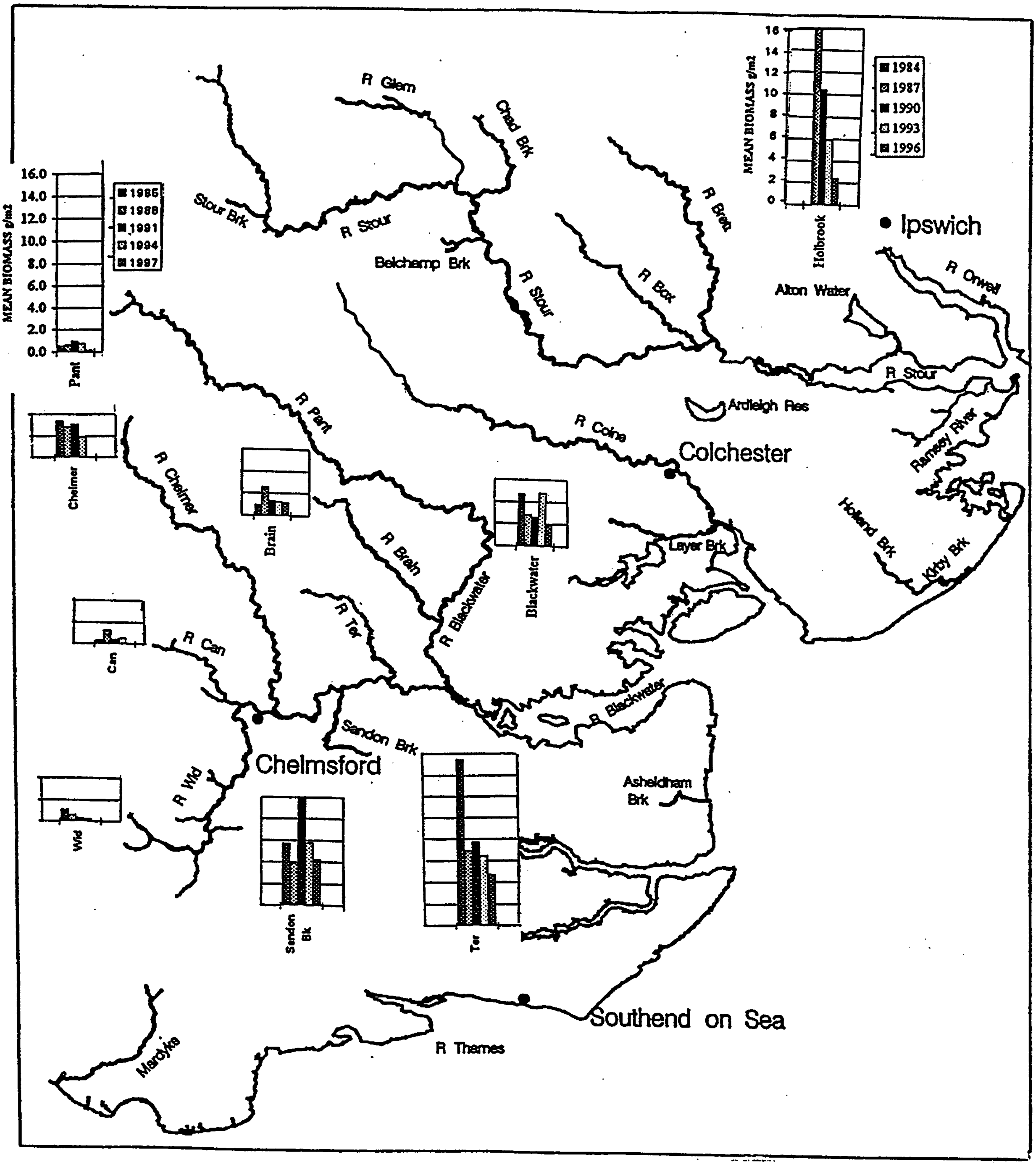


Fig. 2.37 Histograms showing mean annual biomass for combined sites for each of the main rivers of Essex between 1984 and 1997

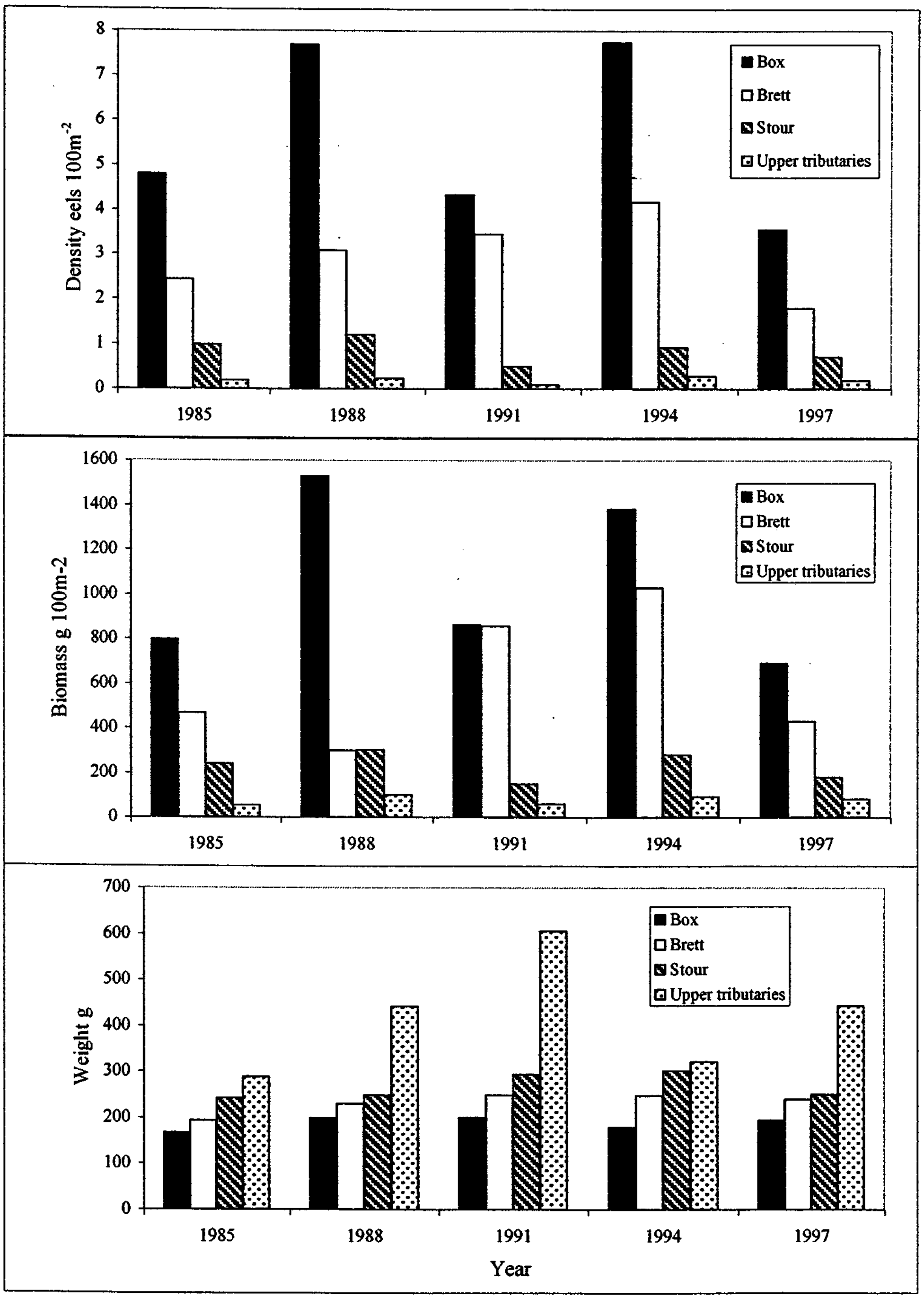


Fig. 2.38 Mean density, biomass and weight per year (1985-97) for sites in the Stour (Essex) catchment; lower tributaries (R. Box, 6-7 sites, and Brett, 10-12), main river (32-36 sites) and upper tributaries (5-12 sites)

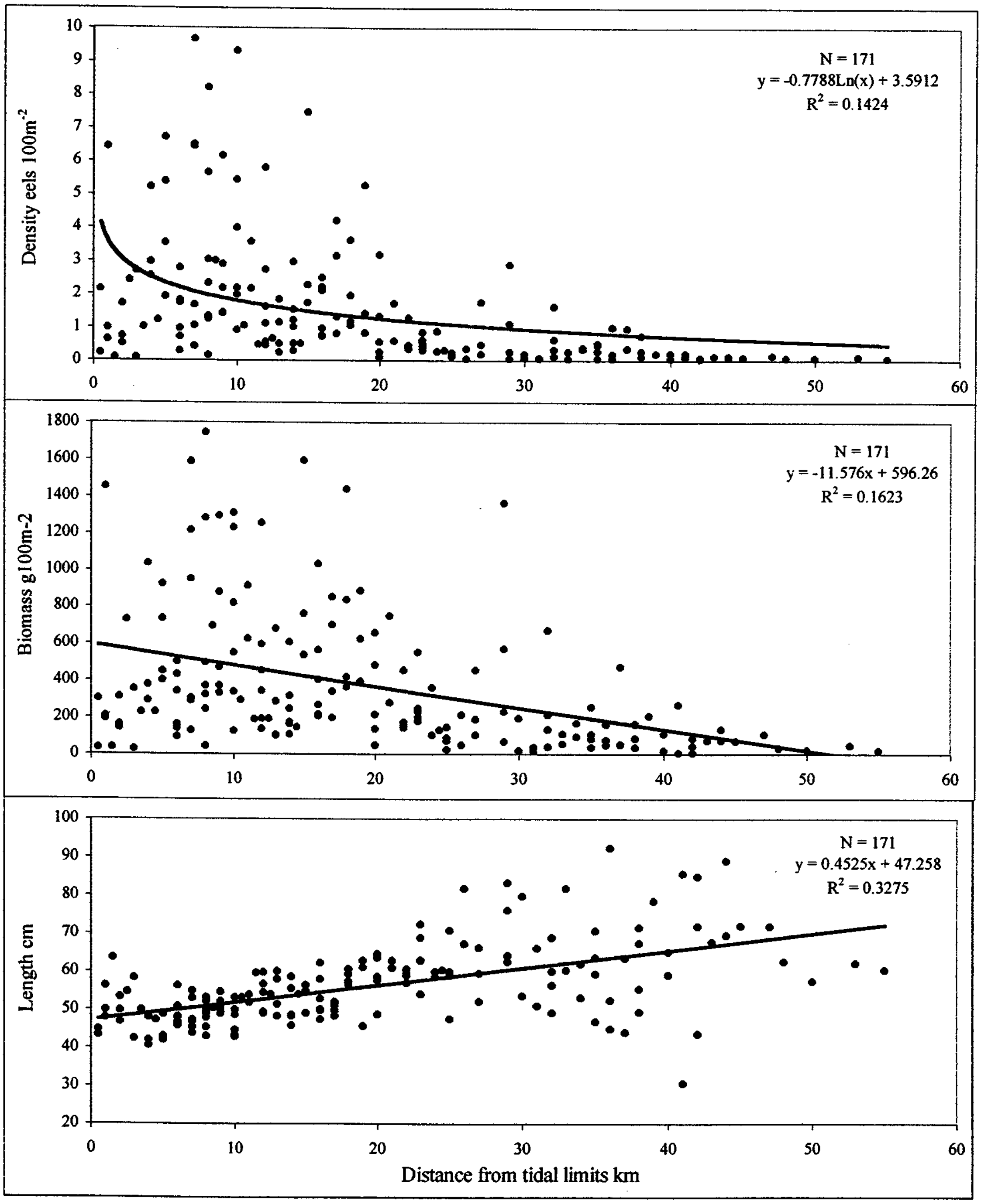


Fig. 2.39 Relationships between density, biomass and individual weight and distance from the tidal limits for the main Essex rivers (Stour, Colne, Blackwater and Chelmer) 1993-95 (171 sites)

According to data presented by Barak and Mason (1992), Essex river eels at 10 - 30 km from the tidal limit were generally maturing at about 50+ cm in 1985-87, equivalent to ages of about 10-15 years. Such silver eels would have been derived from glass eels entering the area in the peak recruitment years in the early to late 1970s. Stocks should have subsequently begun to decline in the 1990s, but this is not apparent from the data reviewed above. It is possible, however, that the major period of low recruitment that began in 1983 could be starting to exert an effect, coupled to fishing mortality. More monitoring is needed to track any trends in the future.

2.23.2 Other east coast waters

Anglian Region is the area supporting the largest number of fyke net eel fishermen, favoured waters being the deeper broadlands, brackish water drains and tidal estuaries. These can potentially support large numbers of eels and there is a strong eel fishing tradition in the Region, catches supplying local, London and Continental markets. Eel fishermen have complained that catches have declined in recent years in the Essex rivers and estuaries and elsewhere in the Anglian Region, especially of seasonal runs of `green' eels between feeding areas (e.g. to where other fish are breeding) and deeper waters (where the eels over-winter). Instrument licence sales have declined steadily in the Anglian Region from 1345 in 1994 to 760 in 1998 but, according to MAFF estimates, overall catches in the Region increased over 1995-98 to an average of 25 t (~ 25 to 30% of the national totals). This figure is probably greatly underestimated. For example, the MAFF estimate for 1992 was 10 t, compared to a questionnaire survey one of 123 t (McKinnon & Potter, 1993). Recent falls in catch values could be having an effect on effort, as discussed in Section 2.2.3.

Netting is the only means of sampling eels in deeper tidal and still waters. No good quantitative data on fyke net surveys were found but seine net catches in surveys of the lower R. Witham off the Wash were 141 in 1981, 38 in 1985, 37 in 1988 and zero in 1991, implying a marked decline in stocks (Forbes & Wheeler, 1997). The average size was about 160 g (46 cm).

Electric fishing is also not very efficient in many east coast waters but Table 2.7 summarises the best fisheries survey data located, plus information from two unpublished studies. One of these was an unpublished IFM Diploma Project by Nick Bromage (Environment Agency Anglian Region) on eels in the non-tidal Yare upstream of Norwich, conducted in 1987. Although a specific and detailed study, no directly related historical trend data were available, except a list of annual mean biomasses over 1981-1987 for various Norfolk-Suffolk rivers from Anglian Water Authority surveys. The Author concluded that migration barriers influence population sizes and structures, but these barriers all comprise mills and other structures that have been present for a long time.

A PhD thesis on eels in tributaries and drains in the lower Trent catchment in the late 1970s-early 1980s has been examined (Carpenter, 1983). Much of the work was on diet, with eels being sampled by fyke netting as difficulties were experienced in most habitats with electric fishing because of water depth, weed growth and salinity. A rotenoning cull of all fish in one blocked-off drain in 1979 yielded a density estimate of 346 g $100m^{-2}$, the eel size range being 50-146 g. The River Meden (a tributary of the Idle) was the only water electric fished (as part of a general cull of coarse fish), but details of fishing methods and efficiency are not clearly

Table 2.7 Representative mean population data for eels from surveys conducted in non-tidal reaches of east coast rivers of England between 1980 and 1998

River	Year	Survey sites	Density N 100m^{-2}	Biomass g 100m^{-2}	Length cm	Source
Meden (Lwr Trent)	1980	3 low river sites	2.00	273	38.5	Carpenter (1983)
Waveney	1981	Whole river	-	211.6	-	Bromage (1987)
Gipping	1982	"	-	432.0	-	"
Whitewater	1983	"	-	216.4	-	"
Tiffey	1984	"	-	56.0	-	"
Deben	1984	"	-	618.3	-	"
Alde	1984	"	-	1606.0	-	"
Blyth	1984	"	-	216	-	"
Wang	1984	"	-	406.0	-	"
Burn	1984	"	-	195.0	-	"
Glaven	1984	"	-	471.2	-	"
Stiffkey	1984	"	-	677.5	-	"
Waveney	1985	"	-	101.0	-	"
Bure	1985	"	-	344.2	-	"
Wensum	1986	"	-	435.0	-	"
Tas	1987	"	-	970.4	-	"
Yare	1987	+18km, 7 sites	3.85	554.6	44.5	"
Ancholme	1989	30 sites	1.37	280.1	49.5	NRA/EA
Lower Trent	1989-1998	Total of 86 sites	0.56	132.3	51.5	NRA/EA

stated. A limited experimental netting survey (of variable annual effort) was made of elver runs in the Idle and Trent. The 1980 catch was16.8 kg in 1980, 2.58 kg in 1981.

Information on Environment Agency fisheries surveys for 1989-98 in the Lower Trent and tributaries has been analysed. Grouping eel density and biomass data by year did not reveal any clear temporal trends because of variations in numbers of samples per year, month of sampling and survey sites used. Overall, eels were found in 86 out of 109 (78.9%) site records examined, but densities (where eels were found) appeared low, the mean (± SD) being 0.56 ± 0.91 eels 100m^{-2} (range 0.1 - 5.9). Mean biomass was 132.3 ± 156.5 g 100m^{-2} (range 10 – 940), average length 51.5 cm. Difficulties in sampling at some sites and the fact that only two run fishing was generally used mean that these must be considered as minimum estimates.

Overall, the data are too variable to reach any firm conclusions about temporal trends, but Table 2.7 indicates that densities are generally less than 3 eels 100m^{-2} (and commonly < 0.5 in the Lower Trent). Mean biomass from the table is about 400 – 450 g 100m^{-2} and mean length > 40 cm. Thus the picture of low density but relatively high biomass and a bias towards large individual size seen in other east coast rivers is repeated.

The numerous tidal and water level control structures in the area could be pose barriers to migration. Stocks in lower tidal river stretches and estuarine areas, especially of smaller eels, are probably large. For example, an estimated 80,000 eels died following a DO crash incident below the Dog in Doublet Sluice at Whittlesey on the River Nene in 1991 (Knights, 1992). Generally, however, the habitats available to eels are not dissimilar to those found in N. Wessex, where river biomasses are comparable but densities and average size appear lower. The differences must be due to the poorer recruitment to North Sea rivers.

There is little quantitative information on fen drains, broadland stillwaters and reedbeds. However, studies indicate that eel densities in Broadland open waters are low, whilst those in the littoral zones are relatively high, at about 20 and 910 g $100m^{-2}$ respectively (Perrow *et al.*, 1996). Furthermore, using specialised electric fishing surveys, eels were found in 93% of 14 reedbeds. In contrast, eels were caught in only 31% of 49 fen dyke sites surveyed, although this must be partly due to the problems of sampling such waters. In a range of British reedbeds, stocks and population structures depend on ease of access for recruits but of special significance is that eels are the commonest fish inhabitants of such habitats and that the great majority are < 20cm (for further details on eel habitats, see Chapter 4). This differential distribution helps explain the apparent lack of small eels in the surveys of poorly-recruiting rivers which have large areas of associated brackish or freshwater marshland habitat.

2.23.3 West Beck (Upper R. Hull) 1988-89

In contrast to the preceding examples, restricted studies found relatively high stocks of eels in the late 1980s in the West Beck (Upper R. Hull), a chalk stream north of Hull and downstream of Driffield. Two eel-targeted three-catch depletion studies were conducted at two sites in November 1988 and at the same sites plus seven others (including two upstream and four downstream) in November 1989. The aim was to assess stock differences upstream and downstream of Wansford Trout Farm at Whinhill, at +36 km and +39 km respectively. Table 2.8 shows densities were relatively very high at some sites downstream of the trout farm effluent and comparable to those seen in some SW England rivers. Smaller eels dominated all but the most upstream site and eels comprised up to 55% of the local fish community biomass.

Table 2.8 Mean eel population data for West Beck (upper R. Hull) in November 1988 and November 1989 upstream (Site A) and downstream (Site B) of a trout farm effluent entry point (see text for further explanations)

Date	Site(s)	Distance from tidal limit (+ km)	Density (eels $100m^{-2}$)	Biomass (g $100m^{-2}$)	Mean individual weight (g)
1988	A	+39	4.7	587	124.7
	B	+36	41.7	3847	92.3
	u/s of A (2 sites)	> +39	5.8-10.0	541-2410	93.3-241.0
1989	A	+39	1.2	180	150.0
	B	+ 36	94.1	7152	76.0
	d/s of B (7 sites)	< +36	25.3-59.6	2113-3862	64.8-84.4

These results appear to contradict assertions made above that recruitment to east coast rivers is relatively impoverished. There are, however, a number of explanations for the high values obtained;-

- the surveys were focussed on eels and this in itself can raise catches by three to five fold compared to routine multi-species surveys
- water depths (< 1m) and clarities were conducive to good sampling efficiency
- silty substratum and weeds formed ideal habitats for daytime hiding
- sampling took place in November when eels might be seeking ideal over-wintering conditions
- productivity was locally high downstream of the trout farm effluent entry point, lower populations upstream possibly reflecting those more typical of less eutrophicated chalk stream conditions
- upstream migration was possibly inhibited by weir barriers during the drought of 1989
- emigration might be enhanced because the river connects directly with the productive Humber Estuary

The declines in density and biomass at matching sites between 1988 and 1989 could have been due to reduced capture efficiency but also perhaps to delayed recolonisation following removal of all eels caught in 1988 for laboratory studies.
The overall results for West Beck appear unusual compared to those for other east coast rivers, possibly because of its high productivity compared to rain-fed, sand/clay substrate rivers of nearby catchments. A resurvey of the same sites during November in the near future would be useful to see if any changes have occurred due since the late 1980s due to recruitment failure or other factors. Additional surveys in other months would also help clarify patterns of annual population change and the river could be useful for general monitoring (see Section 8.4.6).

2.23.4 Angling match data for eels for Yorkshire rivers

Using angling match data to determine temporal trends in stocks has drawbacks because of variations in effort and accuracy of records over years and, for eels, the influences of temperature on catchability. However, data sets covering 1971-98 have been analysed for six stretches of the Rivers Derwent, Wharfe, Ouse, Nidd and Swale discharging to the Humber Estuary. These datasets are considered to be of good quality (e.g. involving similar effort per year but given the absence of data for three stretches in 1993) by David Hopkins, Environment Agency, Yorkshire Area.

Results for percentage rating (i.e. percentage of eels in total match catch returns) per river stretch per year plotted in Fig. 2.40a show that the relative importance of eels tends to decline with distance from the tidal limit (see title), as might be expected. Fig 2.40b shows the mean of combined data per year, indicating an increase in percentage ratings between 1971 and 1987 and then a decline with a recovery in 1994. These results might reflect an increase in stocks following the high glass eel recruitment years in the late 1970s-early 1980s and the subsequent decline, with a lag of about five years, i.e. the probable time taken to mature to a catchable size. However, there are a number of confusing factors involved. For example, improving water quality in some lower stretches of these rivers may have favoured recolonisation by eels in the earlier years. However, other species have since recovered, especially in the Wharfe (stretch B in Fig. 2.40a) after cleaning up of a brewery effluent. Thus the declining percentage of eels in catches may reflect changes in overall fish community structure rather than any significant changes in absolute densities of eels. It is possible, however, that commercial yellow eel fishing may have caused some declines in stocks in the Wharfe in the 1990s (David Hopkins, personal communication). Recruitment to the Derwent (stretch D) may have been impacted since the late 1970s by the operation of Barmby Barrage. Removing the Wharfe and Derwent data does not, however, change the overall trends shown in Fig. 2.40b.

Although not providing robust proof of changes since 1971, the angling match results suggest that stocks in these Yorkshire rivers are quite good, even to +100 km. There is no clear evidence of major declines in stocks that could be explained by lack of recruitment or to overfishing.

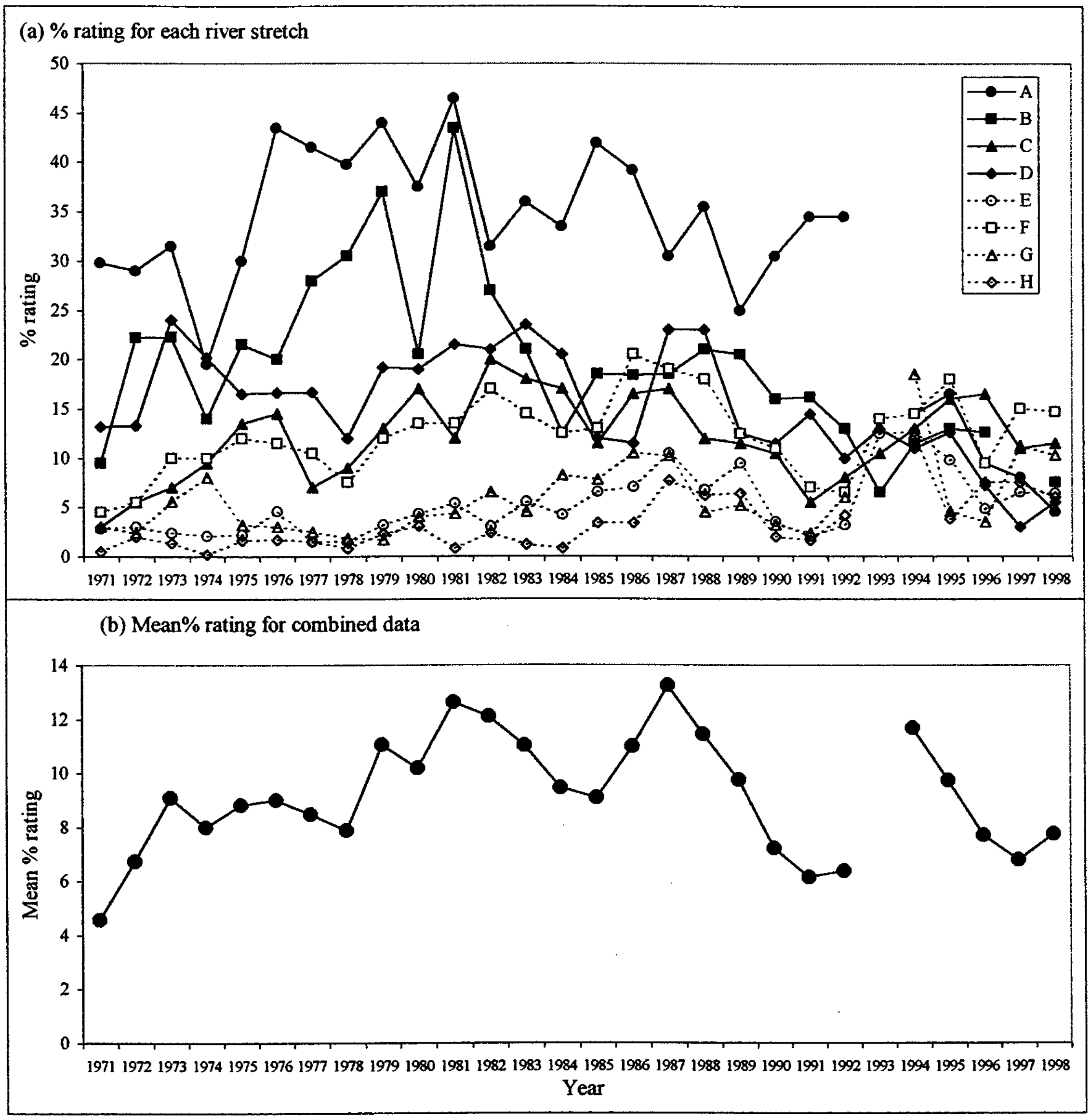


Fig. 2.40 Percentage of eels in angling match catch returns (1971-98) for various stretches of some Yorkshire rivers, spanning different distances from the Humber Estuary. (Stretch A, Derwent +2 to +30 km; B, Wharfe +29 to +36 km; C, Ouse +32 to +53 km; D, Derwent +35 to +55 km; E, Nidd +52 to +71 km; F, Nidd +53 to 61 km; G, Nidd +65 to +71 km; H, Swale +81 to +103 km)

2.24 North East Region and Eastern Scottish Rivers

No quantitative historic data sets were located for the North East Region, fisheries surveys having focussed on salmonids, only recording the presence-absence or relative abundance of other species. Some historical records of possible use (e.g. surveys relating to the Tyne-Tees-Wishe water transfer scheme) cannot be located. There are no historical accounts of glass eel/elver runs, but small runs have been recorded at the eel pass on the Tees Barrage. The trap captured 0.42 kg (~ 1400 individuals) in 1991 and 1.40 kg (~ 4700) in 1999. Eels have been found by fisheries surveys in all NE rivers and they are often recorded as abundant to frequent in lower stretches. Occurrences in the upper reaches are, however, patchy and variable and these do not form meaningful patterns because of chance captures of individual eels between years. Studies in 1983 of the possible impacts of acidification on fish stocks found no eels in 20 brooks in the upper Trent and Don (and the westward-flowing Mersey and Weaver) at 200 – 400 m AOD in the Peak District of the Pennines (Turnpenny *et al.*, 1987). Low upstream stocks in the many NE (and NW) catchments might be expected because they are relatively long and channel slopes steep (2 – 10 m km^{-1} or greater), with waterfalls and rapids posing barriers to migration. Physical and pollution barriers in the lower stretches of the more industrialised rivers would also be of importance.

More detailed quantitative data is needed to clarify the recruitment-stock situation. NE coast rivers would be expected to have suffered low glass eel recruitment relative to those directly facing the Atlantic. However, the more northerly a river is, the closer it is to the migration pathways in the NE Atlantic (see Fig. 2.1 and Tesch, 1977). This hypothesis was assessed by examining data for the Tweed (Mills & Hussein, 1985) and the Dee (Carss *et al.*, 1999) in eastern Scotland.

2.24.1 River Tweed

Some historical population data have been located for the River Tweed catchment, part of which lies in England. Subsidiary to salmonid studies, eel density and biomass data were collected at 30 sites surveyed in 1977 and 89 sites surveyed in 1984 by Mills & Hussein (1985), i.e. the period over which glass eel recruitment peaked and then began to decline (Section 2.2.2). Data obtained from the Tweed Foundation for other surveys in 1972, 1988 and in the 1990s only show presence or absence of eels, although some length measurements were made, as discussed below.

Although no precise details of site locations are given by Mills and Hussein (1985), approximate distances from the tidal limit have been calculated and data are plotted in Fig. 2.41 for 30 matching main river and tributary sites for 1977 and 1984. Eels were absent in three sites in 1977 and five matching sites in 1984, all these sites being situated in the Upper Tweed at higher altitudes and > 100 km from the tidal limit. However, unlike other more southerly long river systems, such as the Thames, densities were occasionally relatively high even further upstream. There appears to have been a decline in mean (± SD) density between 1977 and 1984, from 13.5 ± 21.0 to 8.8 ± 10.1 eels $100m^{-2}$, as well as in biomass, from 599.1 ± 743.8 to 443.3 ± 401.1 g $100m^{-2}$. The changes were not however significant. Furthermore, Mills and Hussein (1985) claim that stocks in the lower Tweed and the Teviot and Till tributaries may have been impacted between 1977 and 1984 by an outbreak of 'cauliflower disease' (a viremial tumour disease). The 1977 data for the lower Tweed (especially Eden Water) and Teviot were also distorted by two exceptionally large catches (Fig. 2.41) and

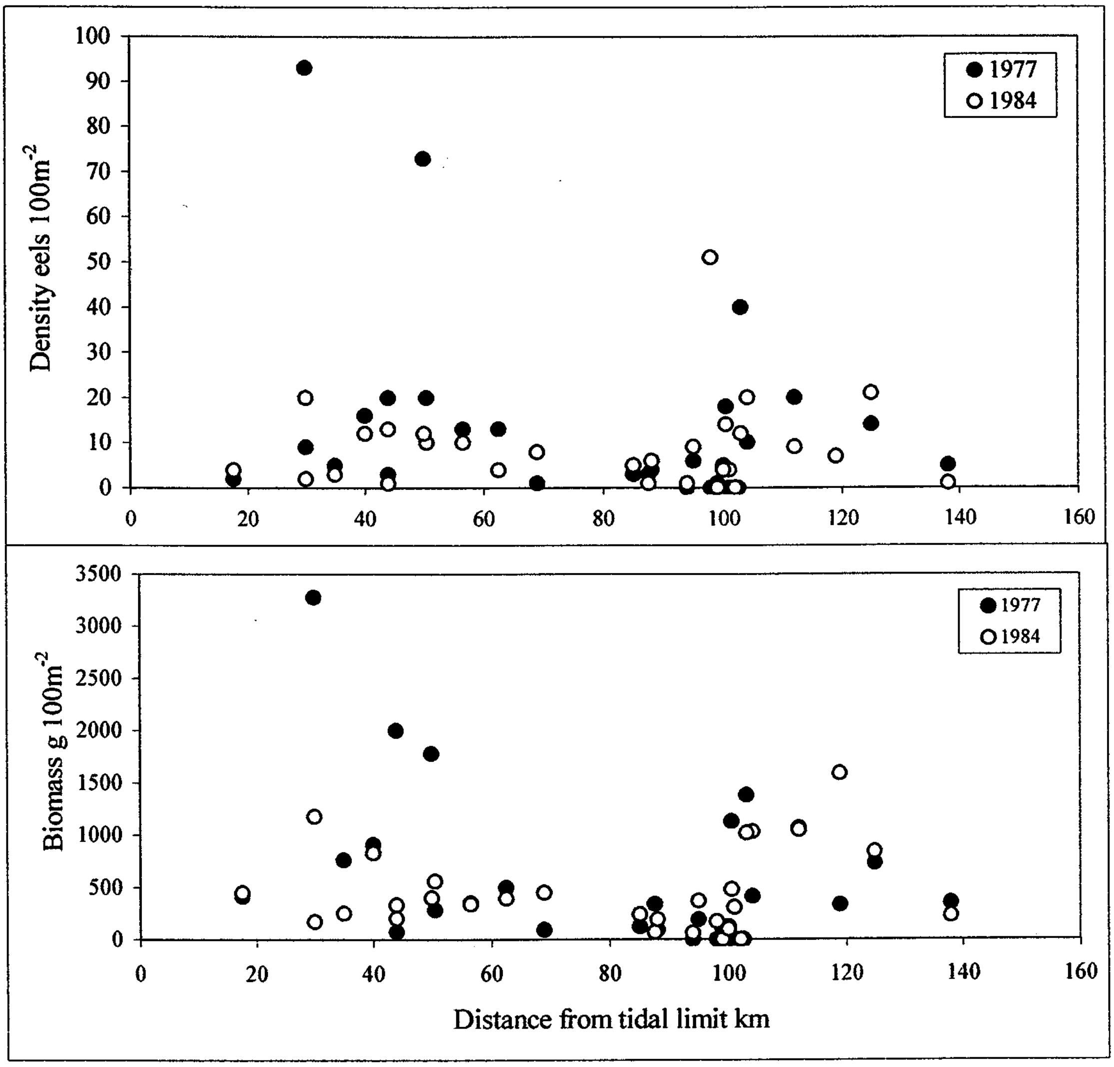


Fig. 2.41 Population density and biomass with distance from the tidal limit (km) in the Tweed catchment, 1977 and 1984.

there were only two site matches for the Till between the years. If these two outliers are discounted, the mean (± SD) density and biomass for 1977 become 8.5 ± 9.2 eels and 461.1 ± 506.6 g $100m^{-2}$.

It appears that even if glass eel recruitment declined after 1978-79, there were no significant changes in Tweed stock size in the seven years between 1977 and 1984. There may, however, have been changes in size and age frequency distributions. Although the Tweed surveys were primarily aimed at salmonids, the paper reports on detailed studies of eel sizes, sex ratios, age and growth, carried out by Hussein in 1977. These involved monthly sampling in four contrasting tributaries, the Leet (approximately +20 km), Eden (+35 km), Leader (+60 km) and Eddleston (+105 km) Waters. Metamorphosed elvers of < 10 cm were only found in the Eden and Leet Waters (Fig. 2.42), appearing in March-June in the former tributary but not until September in the latter. Subsequent upstream migration of small yellow eels led to increases in proportions of sexually-undifferentiated eels in all the tributaries in the spring and summer, especially of 3+ to 5+ eels in Leader and Eddleston Waters. There were relatively high percentages of smaller undifferentiated eels in these waters, despite their

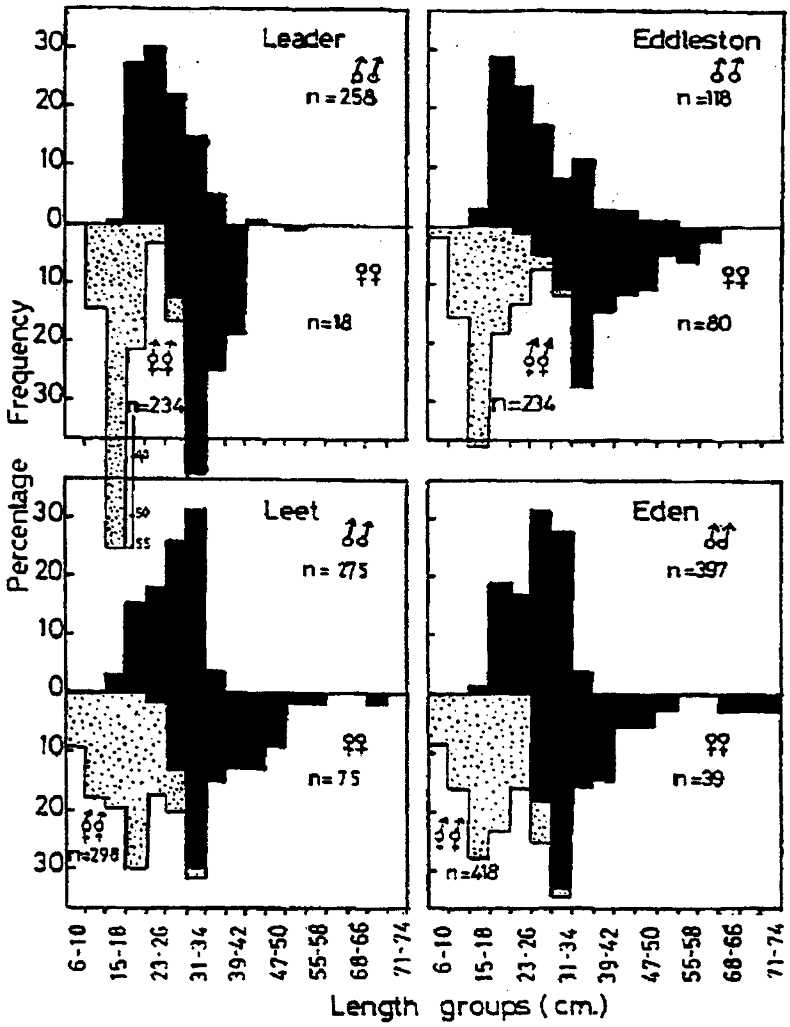


Fig. 2.42 Length-class frequency distributions of undifferentiated, male and female eels in the Leader (+60 km), Eddlestone (+105 km), Leet (+20 km) and Eden (+20 km) Water tributaries of the River Tweed, NE Scotland, in 1977 (from Mills & Hussein, 1985)

distance from the tidal limits, there being few significant barriers to migration. Sex ratios were skewed towards males of 20 – 35 cm, the modal age for emigration generally being 6 – 10 years. Modal length and age for females were generally 30 - 40 cm and 10 - 14 years respectively, although many males and females in Eddleston Water were between 16 and 21 years old. This might be expected as this tributary is situated upstream of Peebles, arising from high ground between the Pentland and Moorfoot Hills.

These results are indicative of relative high densities and that the Tweed was well-colonised by eels, although relatively few 0+ elvers penetrated far upstream. If recruitment had declined by 60 – 70% between 1978 and 1984 (Fig. 2.2), changes in population density and average size might have been expected, especially due to emigration of earlier maturing males. Unfortunately, size frequency data were not collected in the 1984 survey, but average size (derived from biomass ÷ density) did increase from 63.7 ± 47.1 g (66.6 ± 48.0 g, if the two outliers discussed earlier are discounted) to 76.0 ± 58.4 g in 1984. This suggests a decline in small eels. However, this view is not supported by other length data for 1977, 1988 and 1993, shown in Table 2.9 (Tweed Foundation, Edinburgh, unpublished information). These are derived from limited samples for the Till, Teviot Water and Ettrick Water, spanning lower to upper catchment sites. The changes between years are not significant and the low proportion of females is indicative of relatively high population density, even in the late 1980s and mid-1990s.

Table 2.9 Mean lengths of eels and percentage of females (> 45 cm) in the Tweed catchment, from samples taken from Ettrick Water (August 1977, 1988 and 1992), Teviot Water (July/August 1977 and 1988) and the Till (June/July 1988 and 1993).

Site/Year	1977		1988		1993	
	Mean L cm ± SD	% >45 cm (N)	Mean L cm ± SD	% >45 cm (N)	Mean L cm ± SD	% >45 cm (N)
Ettrick Water	32.4 ± 9.2	6.5% (46)	24.9 ± 6.3	2.0% (49)	26.5 ± 7.6	0% (50)
Teviot Water	38.1 ± 10.4	17.1 (41)	27.8 ± 7.2	4.0% (50)	-	-
Till	-	-	22.4 ± 11.3	2.9% (34)	32.8 ± 8.7	9.5% (42)

Therefore the historical data taken overall suggest the Tweed is well colonised, with population characteristics intermediate between those of south-western and south-eastern England (Section 2.20 and 2.23). This might be expected from the prevailing mild climate and sedimentary geology, plus the lack weir and other migration barriers Growth rates were relatively slow, however, at about 2 cm year^{-1}, possibly because of a shorter growth seasons. Eel populations appear to be meeting carrying capacities and not to have been affected by recruitment declines. This could relate to the favourable position of the north east of Scotland in relation to North Atlantic migration pathways, as discussed further in the next section.

2.24.2 River Dee

Carss *et al.* (1999) studied eel populations in two shallow freshwater lochs and associated burns in the Dinnet sub-catchment, some 55 km from the North Sea, over 1990-6. Densities and biomasses in burns above the lochs averaged 2–4 to zero eels 100m^{-2} and 80-150 g 100m^{-2}, those in Mondavian Burn below the lochs were 10-16 100m^{-2} and 350-510 g 100m^{-2} respectively. Figures were even higher in Clarack Burn downstream, 80 eels 100m^{-2} and 3360 g 100m^{-2}. Median length in burns was 24 cm (range 14-52 cm, 7% silver eels). In the lochs, the annual mean densities and biomasses over six years were 2-13 100m^{-2} and 137-403 g 100m^{-2}, mean size 29-38 cm. In all years, densities were higher, particularly of smaller eels, in rocky areas (and in reeded burns) than in soft substrates in the lochs. Populations appeared to decline significantly between 1990 and 1992 but then increased to maximum levels in 1996. The time scale is too short to indicate definite temporal trends and Carss *et al.* (1999) concluded that inter-year recruitment varied depending on annual migrations of juveniles, mortality due to otter predation also being an important factor. Although not statistically sigificant, negative relationships between density and individual size suggest density--dependent regulation of eel populations occurred.

Recruitment to NE Scotland is aided by the relatively short distance from the edge of the Continental Shelf, where metamorphosis from the leptocephalus to glass eel occurs (see Fig. 2.1). North-easterly currents also possibly aid migrations into the northern North Sea compared to those up the English Channel (Bowman, 1913). Recruitment may not have declined as drastically as it has in the southern North Sea because of (a) northerly shifts in the Gulf Stream/N. Atlantic Drift currents between 1974-78 and 1983-85 (Knights *et al.*, 1997) and (b) changes in the N. Atlantic Oscillation, associated with stronger westerly flows of relatively warm air in the N. Atlantic. Both these phenomena have correlated with the penetration of warmer water into the northern North Sea and changes in associated plankton populations (Sparks & Reid, 1999). These trends have continued into the 1990s.

2.25 North West Region Rivers

The paucity of historical quantitative data mirrored that found for the NE Region. Some past studies have been conducted on eels in the Lake District, but these were designed mainly to look at diet and competition with salmonids and no data on density or biomass were collected (e.g. see Tesch, 1977). The only quantitative fisheries survey data sets found for a whole single catchment were for the R. Lune, with 74 comparable sites surveyed in 1991 and 1997. Eels were present in 70.3% of sites in 1991 and 63.5% in 1997, mean densities (± SD) being 1.20 ± 2.47 and 0.96 ± 1.99 respectively. The catches are too variable and time scales to short to indicate any significant time trends but densities appear relatively low.

The majority of recent surveys in the NW have only used single-run electric fishing without stop nets. Such sampling data on eels for 415 sites in S and SW Cumbrian rivers between 1992-96 has been compared in an unpublished Environment Agency report by E.P.K. Watson and D.F.J. McCubbin. Eels were present in relatively high numbers in estuaries, river mouths and in lower river reaches draining lakes (often at 100-200 eels per run over about 50 m). Numbers tended to decrease with distance upstream (especially > +30 km) and above certain migration barriers, but eels were found in reasonable numbers (1-10 eels per ~ 50 m) in main rivers and larger upstream tributaries above and below Windermere, Esthwaite, Rydal, Grasmere, Coniston Water and Wastwater. Eels are known to be present in Elterwater, above Skelwith Falls on the R. Brathay which are impassable to salmonids.

Although not providing any idea of historical trends, these data sets imply stocks in south and south west Cumbria are reasonably healthy. This is supported by fully quantitative multi-species surveys conducted in 1999 on the lower Leven below Windermere and on the Eamont, between Ullswater and the confluence with the Eden (Environment Agency, unpublished data). These are the first of a ten-year series of surveys, being conducted to set baseline data prior to any possible drought order applications for these waters. In the Eamont (16 sites between +68 and +79 km), mean densities (± SD) were 2.68 ± 1.46 (range 0 (at one site) to 5.1). In the Leven, two sites at Low Wood (+2.5 km) had densities of 17.6 and 27.4 eels $100m^{-2}$

Also of interest is a review of data on silver eel migration data by Watson and McCubbin gathered by the Logie fish counter at Backbarrow on the R. Leven, 3 km downstream of Windermere. A total of 369 emigrant silver eels were counted during October-December in 1994 and 3194 in 1995. In comparison, large-scale experimental traps in similar locations caught 793 eels in 1942, 1487 in 1943 and 2010 in 1944 (Lowe, 1952). This suggests that eel stocks in the Leven-Windermere system were similar in the 1940s to those in the 1990s. Recruitment levels in the 1940s were probably similar to those pertaining now, according to data presented in Fig. 2.2. Levels were low relative to the peaks of the 1960-70s, suggesting there may be long-term cyclic trends, independent of fishing pressures.

The only other quantitative data found for the NW Region are those for electric fishing and immigration studies into Leighton Moss RSPB Reserve, connected by Red Barn Dyke/Quicksand Pool to Morecombe Bay. Average densities in open waters and reedbed zones in 1998 were 3 and 15 eels $100m^{-2}$, biomasses 65 and 327 g $100m^{-2}$ respectively. These values are within expected limits and do not indicate any severe limitations in past recruitment. Fishing pressures on glass eels do, however, appear to be high, although recent studies suggest recruitment of later arriving pigmented elvers and juvenile is sufficient to maintain stock levels. These issues are discussed in more detail in relation to habitat preferences and natural and fishing mortality in Chapters 4 and 5.

Overall, it appears that the NW Region rivers receive reasonable numbers of glass eel recruits, either via the North Channel or St. George's Channels to the Irish Sea or via both. Any recruitment from the north would have been expected to mirror the declines seen in the R. Bann in N. Ireland (Fig. 2.2), that from the south to mirror the declines in the main Severn area fisheries (Fig. 2.3).

2.26 Welsh Region rivers

No quantitative historical routine fisheries survey data were located. The lack of data for the South Wales rivers subject to glass eel fishery pressures, such as the Wye and Usk, is especially disappointing. Previous detailed studies on the Dee by Aprahamian are discussed in detail in relation to resurveys carried out as part of this R&D programme in Chapter 3.

The only historical data set for eels, collected secondarily to a long-term salmonid study in relation to afforestation, has been purchased from Dr Trevor Crisp. This has been focussed on 10 sites in the Afon Cwm, a tertiary tributary of the Dyfi in mid-Wales, at approximately +20 km and 280-415 m AOD. Eel catches between 1984-1998 have been relatively small (561 in total) and Fig. 2.43 shows the total catch for all sites combined per month of survey for each year. There were no obvious changes in eel numbers and hence, prsumably, density over 14 years, given that the 1984 set was distorted by two exceptional catches in April and May and the 1998 set by exceptionally low catches in April and September. Eels were only consistently caught (and in larger numbers) at the most downstream site, probably because of accumulations below a migration barrier formed by a gauging weir and sill. Densities varied between 1 and 7 eels $100m^{-2}$, standing crop between 40-230 g $100m^{-2}$ and size range 8-68 cm. Catches declined upstream, probably because of the barrier and the rapidly increasing gradient. High inter- and intra-year variability of all parameters were probably due to temperature variations and the impacts of spates, although efficient and reproducible sampling of eels in such upland rivers is not easy. The population parameters, although varying between years, are within the ranges typical of rivers in the south west of England at the same distance from the tidal limit and at similar altitudes, channel slopes and hard substrates.

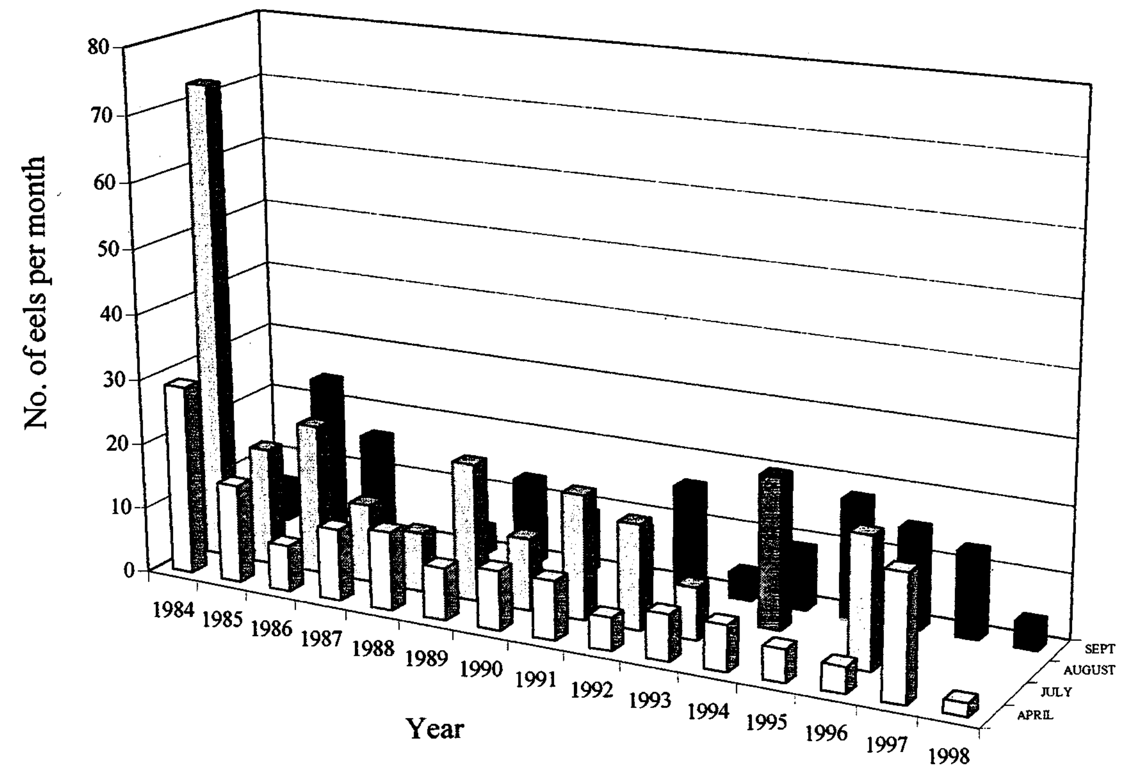


Fig. 2.43 Total number of eels electric fished per sampling month per year (10 sites combined) between 1984 and 1998 in the Afon Cwm, a tributary of the R. Dyfi, mid-Wales

In conclusion, quantitative data for Welsh rivers are very sparse but stocks are probably fairly good in lowland waters. Low densities and absences in sites nearer to headwaters would be expected, due to distance from the tidal limits, exacerbated by increasing gradients, decreasing productivity and migration barriers posed by rapids. The lack of evidence for significant declines in stocks in the Dee is discussed in Chapter 3. It is possible that glass eel recruitment has been maintained at relatively high levels in Welsh rivers compared to east coast ones because of their proximity, albeit via the Irish Sea, to the Atlantic glass eel migration routes.

2.27 West Midlands Region Rivers

No quantitative historical data sets were located for rivers in the Midlands Region discharging to the west coast, except for the Severn studied by Aprahamian and discussed in detail, along with R&D resurvey data, in Chapter 3. Some results for the Warwickshire Avon have been examined but these are very intermittent in temporal and spatial coverage. Any interpretation of temporal trends is in any case confused by changes following recovery of the upper reaches from chronic pollution and to restocking over the last 10 years or so (e.g. see White & Knights, 1994).

2.28 Other Sources of Information

2.28.1 Eel distribution in canals

Information has been supplied by the Fisheries Managers of the Southern, Midlands/North East and North West Regions of British Waterways. The information comprises non-quantitative estimates of eels seen during drain-downs, but the picture emerging is well-illustrated by that for the Grand Union Canal, with many small eels near to the tidal limit with a major decline about 25-30 km upstream when only occasional very large eels have been seen. No eels have been recorded in drain-downs in the North Oxford Canal until it meets the Grand Union where eels can enter both systems via the Stratford-on-Avon and Worcester & Birmingham Canals. The decline in stock densities but increase in size with distance from the sea agrees with the picture seen in rivers in earlier sections. Few canal reservoirs are known to contain eels because of dam barriers and because most are pump-filled. These conclusions are supported by anecdotal evidence from canal anglers. Members of the Barbel Society and other night-time anglers state that the number of large eels hooked has decreased in the middle and upper Severn and the Midlands canals compared with past experience.

2.28.2 Information from eel angling

Some anglers are of the opinion that stocks of eels have declined, especially those of larger females in waters distant from tidal limits All the specialist eel angling groups were contacted but none were able to provide quantitative information on possible changes. Records of eel catches from the published literature were examined but these only revealed seasonal and inter-year changes relating to water temperature and not to any long-term temporal trends in stocks. One useful dataset was, however, located for various Yorkshire rivers covering 1971-98, this has been reviewed in Section 2.23.4 but did not indicate any significant declines in eels stocks.

2.29 Assessment of Historic Data Sets on Stocks: Conclusions

Assessments of possible changes in eel stocks in England and Wales and robust statistical comparisons have been hampered by the paucity of good historical information. There have been large inter-year differences in fisheries survey methods and their efficiencies and in sites used. More recent surveys have commonly been reduced to single electric fishing runs or dip netting, with only presence-absence or relative abundance data recorded. Other sources of information found have generally been restricted in space or time, are not amenable to statistical analyses or can only offer non-quantitative and indirect evidence as to past or current status of populations (e.g. via density-biomass-body size relationships). Large variations in populations are also common, due to inter-seasonal and inter-annual variations in local migrations. Changes in local environmental factors can also explain some changes over time. The best historic data exist for the Severn, Dee and Dorset Frome and Piddle and these have therefore been the subject of detailed resurveys in this R&D programme, as discussed in the next Chapter.

Despite these provisos, the available evidence shows there has been a decline in glass eel recruitment to Britain, following peaks in 1974 and 1978-79, stabilising through the late

1980s and 1990s. This broadly mirrors the situation seen elsewhere in Europe. Some commercial fishermen claim that catches have been declining but without supplying quantitative proof. Indirect evidence from exports suggests that catches actually increased in the late 1980s and again in the mid-1990s, i.e. with a lag of about 16-17 years after the good recruitment years. This would correlate with the normal span of the growth stage of females in many British rivers. Catches appear to have declined markedly since 1997 which could reflect falling stocks due to poor recruitment beginning in the late 1980s. Catches and licence sales have, however, tended to follow prices throughout the 1980-90s. Demand for eels in Europe has been falling in recent years, according to some European dealers and processors. A major cause of collapsing prices has, however, been the increasing availability of cheaper farmed eels, the situation being exacerbated in recent years by dumping of cheap eels from the Far East because of overproduction in Chinese eel farms. Many dealers, processors and fishermen are very pessimistic about the future economic viability of eel fishing in England and Wales.

From the historic and other data sets examined, it is not possible to prove that a significant overall decline in stocks due to recruitment failure or fishing mortality has occurred throughout England and Wales (or NE Scotland). Instead, there appear to be important geographical variations in population densities and biomass, depending on the position of rivers relative to the main Atlantic migration pathways of glass eels. This is well illustrated by comparing data for the rivers of south-west England with those in the east and south-east. Populations in the lower reaches of shorter rivers in the south-west of England do not appear to have changed much since the late 1970s. They still have high densities and biomass of eels, means typically being ~ 15 eels $100m^{-2}$ and 400 g $100m^{-2}$. Given that these values are probably underestimated by factors of three to five (according to resurvey studies discussed in Chapter 3), they are comparable to the upper limits of European averages (Tesch, 1977; Moriarty & Dekker, 1997). Values fall upstream of about +25 to +30 km to 5 or less eels $100m^{-2}$ but biomasses tend to be maintained at about 300 g $100m^{-2}$ because individual mean weight increases from about 30 g (25 cm) to about 60 g (35 cm). Overall, means are ~ 12 eels $100m^{-2}$ and 370 g $100m^{-2}$ respectively and only in the most upstream sites of long rivers (or where migration barriers occur) are eels absent or rare. In comparison, densities in the productive easterly-facing Essex rivers (and the Thames) are commonly less than 4-5 eels $100m^{-2}$ (overall average ~ 1) but biomasses are again maintained at relatively high levels, mean ~ 360 g $100m^{-2}$ because average sizes are > 200-400 g (50-60 cm). These results, taken with those from other rivers discussed above and in Chapter 4, indicate that the carrying capacity of productive British rivers for eels, as determined in multi-species surveys, typically lies between 300-400 g $100m^{-2}$. As mentioned above, true values are probably three to five times larger than these.

At the level of individual rivers, few 0+ elvers penetrate far upstream. Instead, density-dependent migration of 0+ juvenile yellow eels occurs, leading to segregation of sizes and sexes so that smaller and earlier maturing males tend to dominate lower river stretches, larger and later maturing females the lower density upper reaches. Comparing geographical locations, rivers in the south west, west and north east (at least in the Tweed and Dee, Sections 2.24.1-2) appear to be receiving sufficient recruitment to easily meet carrying capacities. They tend to have high density populations dominated by small eels, the majority of which mature and migrate as males. In the longer rivers, increasing gradients and declines in habitat productivity and suitability for eels lead to falling densities and a greater preponderance of females.

In contrast, eel stocks in the easterly rivers, such as those in Essex, are also meeting their (similar) carrying capacities, but the fact that eel densities are so much lower but average sizes so much larger implies that recruitment is relatively low. Recruitment to southern North Sea rivers is recognised as being relatively impoverished (Tesch, 1977). Some productive east coast rivers with suitable habitats can, however, support relatively high stocks, e.g. West Beck (upper Hull) discussed in Section 2.23.2. Poor recruitment as a major factor in waters further away from Atlantic migration pathways is implied by the lack of increases in distribution in the Thames catchment since the removal of pollution barriers in the estuary in the 1960-70s. Other catchments with large estuaries and lower marshlands to absorb recruits may also have experienced declines in populations, e.g. the Taw-Torridge, Tamar and Exe systems and, as discussed in Chapter 4, the Dorset rivers discharging into Poole Harbour. Man-made migration barriers have also influenced expected patterns of stock distributions and population structures with distance from tidal limits in the more heavily regulated river systems.

The final conclusion is that, despite major declines in glass eel recruitment since the late 1970s-early 1980s and some high localised fishery pressures (mainly on glass eels in the Severn area), there is no proof that these have had any significant impacts on overall stocks in England and Wales. No significant changes in river stocks were found in the intensive resurveys of the Severn and Dee reported in Chapter 3. However, low recruitment appears to have had negative impacts in the Piddle and Frome, probably exacerbated by the habitat sink offered by Poole Harbour, plus possible impacts of the harbour yellow/silver eel fishery.
Some of the commercial fishery evidence reviewed could imply that declining recruitment may be impacting yellow and silver eel fisheries, as older females derived from earlier peak recruitment years are maturing and emigrating. The picture is confused, however, by declining market values and hence fishing effort. No specific information on spawner escapement has been gained, but overall it appears that the scattered and low intensity commercial fisheries in England and Wales have had minimal impacts. No significant fisheries specifically exploiting silver eels have been located in England and Wales.

More detailed information is required to further test the hypotheses discussed above about density-dependent effects on mortality, migrations and sexual differentiation, geographical differences, etc. These issues are considered when discussing possible future research into natural *v.* fishing mortality in Chapter 5. Careful monitoring in the next few years is essential to track trends in recruitment, stocks and fisheries. This is essential to inform the short- and long-term management of both glass eel and yellow/silver eel fisheries, as discussed further in Chapters 7 and 8.

3 CURRENT AND HISTORIC EEL STOCK STATUS - REPEAT SURVEYS OF HISTORIC SURVEY SITES.

3.1 Introduction

The specific objectives for this part of the study are:-

- *to undertake a series of eel surveys on river systems where there is verifiable historic data in order to quantify temporal changes in eel stock status;*
- *to combine the historic and re-survey (R&D project) data into a unified database to provide a baseline for future monitoring of eel stocks in those rivers.*

Practical and resource constraints dictated that the re-survey programme should be limited to running freshwaters amenable to electric fishing, even if data from other habitats could be located. The selection of river systems for repeat surveys was dictated by the availability of historic data. It was considered that, ideally, historic datasets would have the following attributes:

- data should relate to west, south and east coast rivers and include both 'exploited' and 'non-exploited' catchments;
- historic surveys should be quantitative, have employed reproducible methodology and be of sufficient scale to provide a basis for meaningful statistical comparison;
- the historic data should include information on population density, length/frequency, length/weight relationship, population age structure and growth rate;
- data should relate to the 1970s and/or early 1980s, so as to bridge the periods of a), maximum glass eel recruitment in the late 1970s and early 1980s and subsequent decline and b), maximum glass, yellow and silver eel exploitation in the late 1990s (Sections 2.2.2 - 3)
- original data should be available, rather than only edited, abridged or pooled data from reports or published papers.

It became apparent at an early stage that routine Environment Agency (and former NRA and RWA) fish stock surveys could not provide reliable quantitative historic data on eels. Three main shortcomings with Agency data on eels were apparent:

- Agency Regions (and Areas within them) vary in their treatment of eels during fish surveys. In some cases, eels are individually weighed and measured and population densities are estimated. More often, eels caught are only bulk weighed, or eels are simply recorded as present/absent. Furthermore, the treatment of eels within a given Agency Region or Area tends to reflect the current priorities and policies of fishery staff and therefore varies over time.
- Eels are invariably under-represented in the catch during multi-species or salmonid oriented fish surveys. Thus even where full catch records for eels are available, the resulting population estimates are unlikely to be quantitatively reliable. This point is discussed by example and in more detail in Section 3.11.2.

- There appears to be a widespread difficulty within Agency Fisheries sections with the identification and retrieval of pre 1990s datasets, where, as is usually the case with eels, the data have not been incorporated into key reports.

3.2 Availability and Nature of Historic Data

Only three historic datasets could be located that were of sufficient quality and extent to make resurveys worthwhile. These datasets relate to the Severn, the Welsh Dee and the Piddle/Frome system in Dorset. No reproducible quantitative historic data could be located for any east coast river.

River Severn
A very extensive investigation of eel distribution, abundance, age and growth in the River Severn was undertaken by Aprahamian in 1983 and 1984 (Aprahamian 1986, 1988). This survey covered 109 tributary sites spanning Zones 1-10 of the river (Figure 3.1). Surveys were based on electrofishing of typically 100 m lengths of channel with population estimates undertaken by catch depletion using 2, 3 or 4 fishings. Large numbers of eels from each zone were aged enabling growth rates to be compared in different parts of the catchment. The original catch (length and weight of each individual) and age data were available in hard copy.

Welsh Dee
In addition to his main study on the Severn, Aprahamian (1988) also surveyed 8 sites on the River Dee, spanning the catchment from close to the tidal limit to above Bala Lake (Figure 3.2). Surveys were again based on electrofishing with population estimates undertaken by catch depletion. A large proportion of the eels caught were aged, enabling both growth rate and typical age of arrival at different parts of the catchment to be determined. The original catch (length and weight of each individual) and age data were available in hard copy.

Dorset Piddle and Frome
The Piddle and the Frome form a linked system. Both rivers have tidal limits at Wareham and both enter the western end of Poole Harbour some 2 km further downstream. Although the River Frome is larger than the Piddle, both rivers are chalk streams and are essentially similar in character. Several data sets relate to these two rivers.

MAFF undertook a detailed study of eel populations in the Piddle in 1976 and 1977 (Morrice *et al* (undated)). The study covered a 13.7 km continuous length of channel upstream from the tidal limit at North Mills Weir, Wareham (Figure 3.3). The channel was divided into 21 sections ranging in length from 399-809 m and individual population estimates were made for each section. Eels were surveyed by the electrofishing of entire sections with populations estimated by the Petersen mark/recapture equation (population data relates only to eels >250 mm as smaller eels were not marked). Some sections were fished in August and September 1976 (t_1 and t_2) but 19 contiguous sections were fished in both May/June 1977 (t_3) and July/August 1977 (t_4) with the t_3 and t_4 catches used for the overall population estimates. A random sample of 84 yellow eels from the July/August 1977 fishing were aged from otoliths.

The MAFF Piddle study also attempted to quantify the silver eel run during Autumn 1976 and Autumn 1977 by the installation of an eel trap at the tidal weir. Although not entirely successful in terms of quantifying the run, the exercise provided good information on the length frequency and sex ratio of migrating silver eels and 279 silver eels were aged. The MAFF study has not been published, no final edited report appears to have been produced and the original data are no longer available. However, a draft report was located and this is sufficiently detailed to allow several key parts of the data to be reconstructed and cross checked. However, omissions and anomalies in the draft report impose some significant limitations on the scope of the retrievable data.

The Freshwater Biological Association (now IFE) (Mann and Blackburn 1991) undertook a detailed study of the biology of the eel and its interactions with juvenile trout and salmon in an approximately 2 km section of the Tadnoll Brook, a tributary of the Frome (Figure 3.3), spanning the period 1973 to 1984. The original data are not available but the published paper contains population estimates (biomass and density) derived by catch depletion based on electrofishing the entire study stretch in March 1973 and August 1980. A length/frequency plot is provided for August 1980. Length/age plots for 588 eels captured in August 1982 and August 1983 are also included. Sex was determined by gonad examination of a total of 1126 eels of 250 mm or more captured between August 1980 and August 1984. The sex data is summarised in table form. The Tadnoll Brook paper, (Mann and Blackburn 1991) also provides eel length frequency data for the East Stoke Mill Stream, a branch of the River Frome (Figure 3.3), collected in March at irregular intervals between 1963 and 1984. Eel population estimates in the East Stoke Mill Stream, based on monthly samples from June 1991 to February 1992 are provided in Ibbotson *et al.* (1994).

In August 1990, the former NRA undertook a small scale eel survey covering six sites on the Frome and three sites on the Piddle (see Figure 3.3 for site locations). The distribution of sites provides a useful catchment overview and the brief report provides density and biomass estimates for each site and composite length frequency plots for each river. The original data are no longer available.

3.3 Resurvey Strategy and Site Selection

The primary objective of the resurvey programme was to provide the optimum match to the original surveys in terms of site location and survey timing, and, where practical, survey methodology, within the constraints of available resources. Other fundamental objectives of resurveys were that they should be of sufficient scale to allow useful statistical comparison between historic and contemporary data and that they should cover, as far as possible, periods of apparent change in glass eel recruitment and fishing pressure.

River Severn

For the River Severn, it was possible to match sites, survey methodology and timing relatively precisely. However, it was deemed both impractical and of doubtful cost effectiveness to repeat the entire 1983/84 survey. In 1988, 24 of the original 109 Severn sites were resurveyed, namely:

Zone 1	4 of the original 6 sites;
Zone 2	7 of the original 11 sites;
Zone 3	10 of the original 11 sites;
Zone 5	all 3 of the original sites.

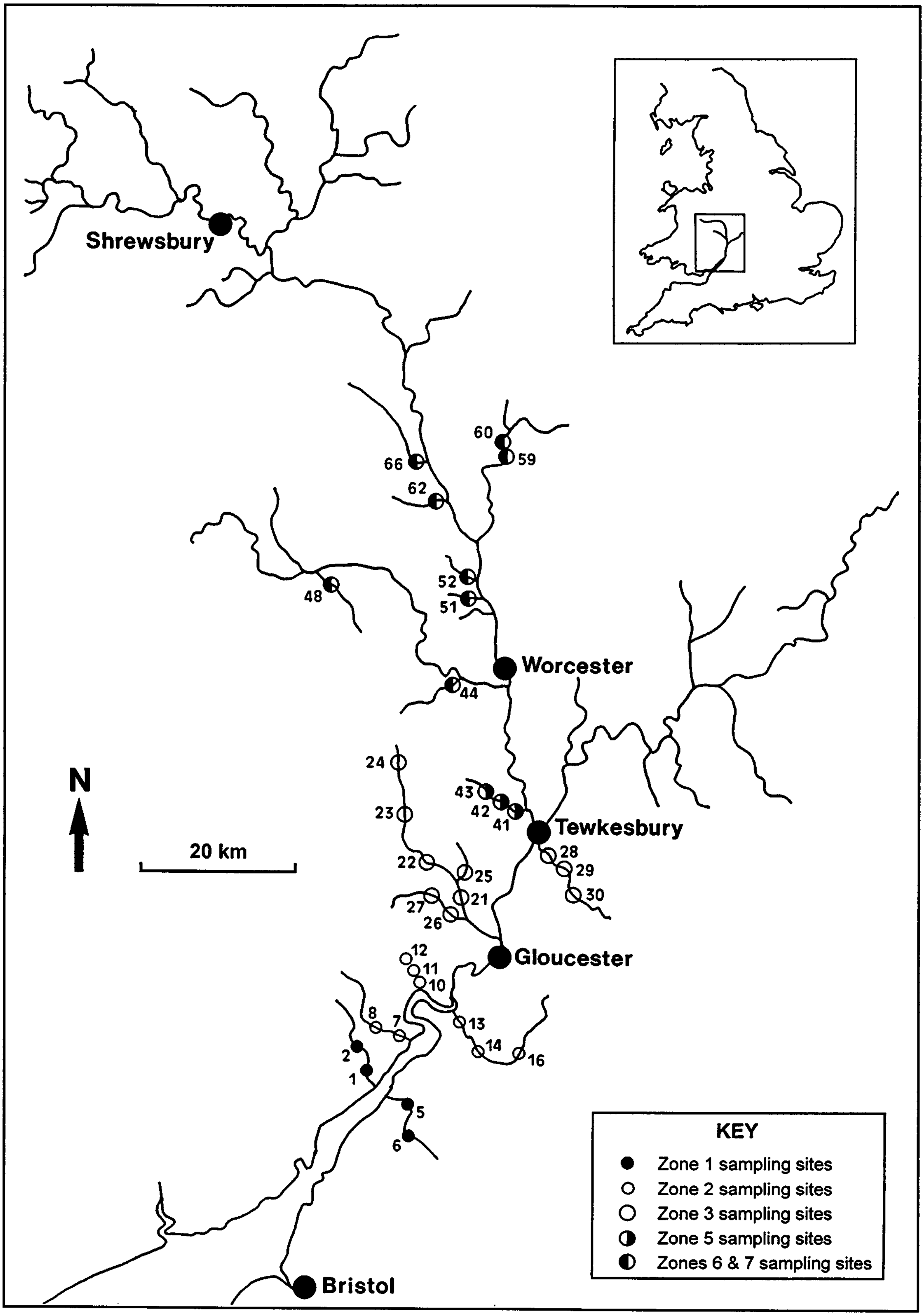


Figure 3.1 River Severn: 1983/84 and 1998/99 eel survey sites

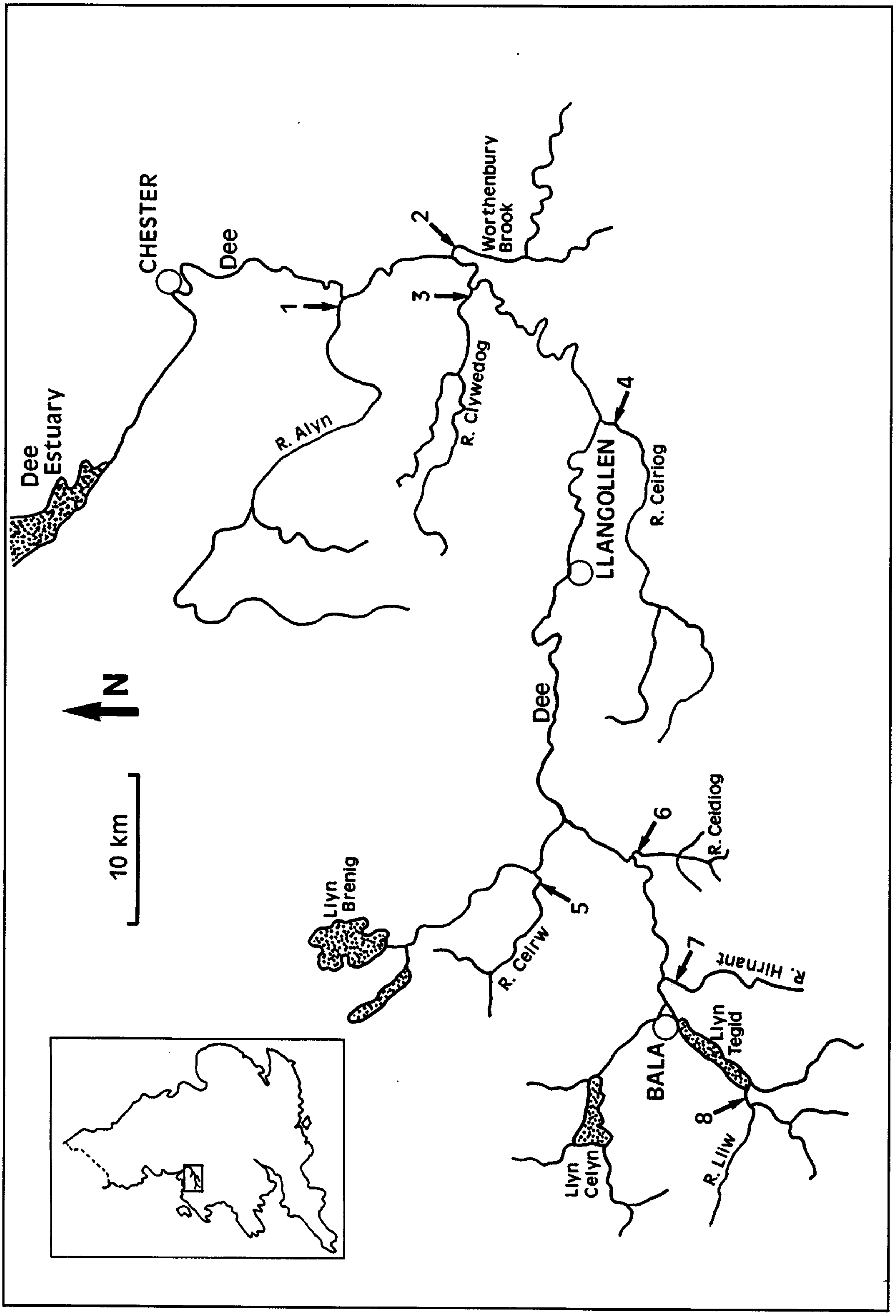


Figure 3.2 River Dee: 1984 and 1999 eel survey sites

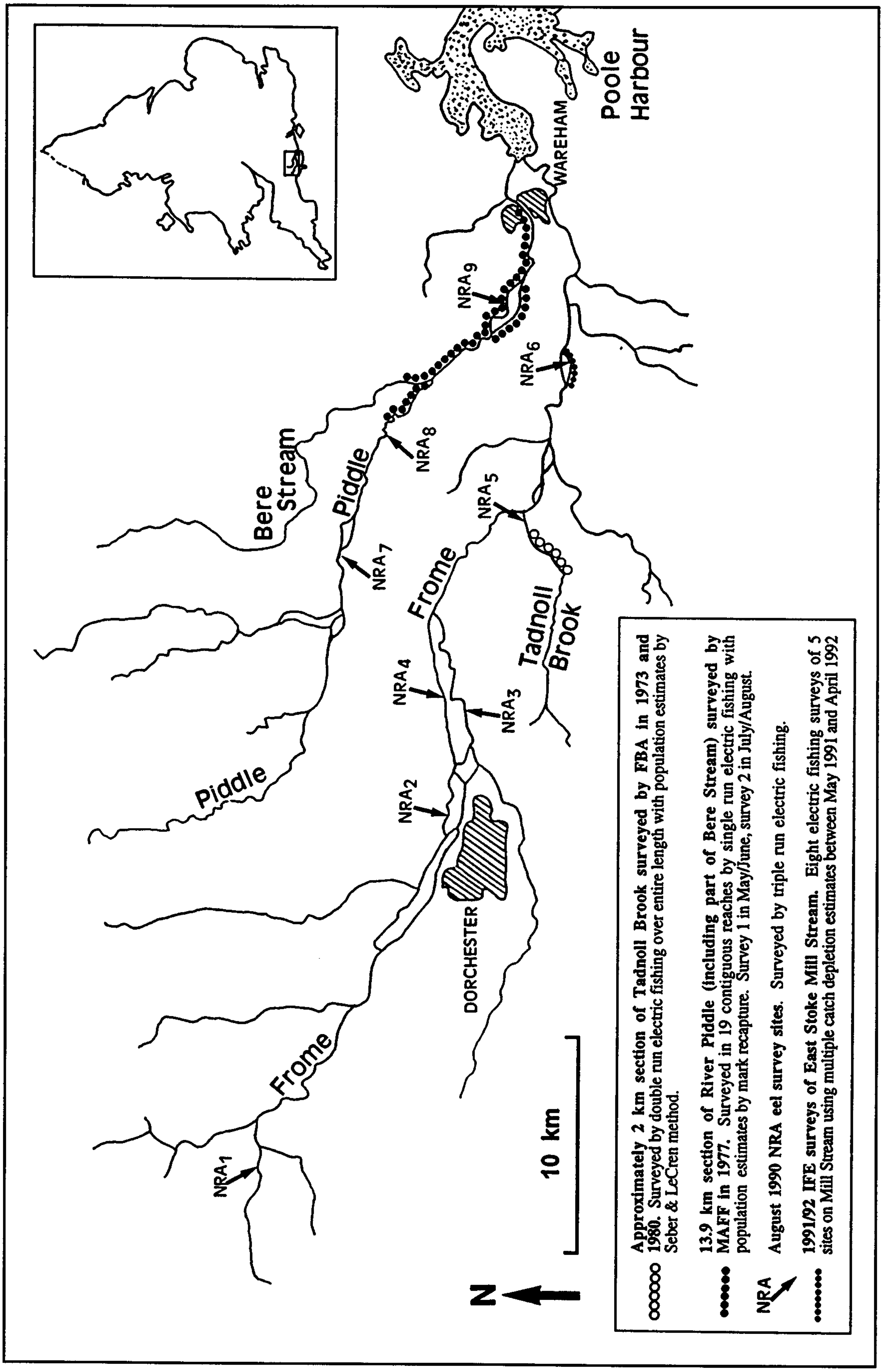


Figure 3.3 River Piddle and River Frome: 1973, 1977, 1980, 1990 and 1991/92 eel survey sites and reaches

No sites in Zone 4, the main exploitation (elver fishery) zone were resurveyed, as sites above and below this zone would be expected to yield adequate information on any impacts of the fishery on stocks.

In 1999, 8 of the original 26 sites in Zones 6 and 7 were resurveyed. The 1983/84 survey also included 40 sites in Zones 8, 9 and 10. However, eel densities were generally very low this far up the catchment and eels were apparently absent from a high proportion of sites. It was therefore concluded that resurveying a sample of sites from the uppermost zones would be unlikely to facilitate meaningful temporal comparisons. Instead, it was deemed more useful to examine Environment Agency fish survey data from the 1990s. These surveys record eels and have included many of the tributaries covered by the 1983/84 eel survey. Whilst not allowing a direct quantitative comparison with the historic data, the approach was expected to highlight any major changes in overall range within the catchment.

Also in 1999, 16 (of 24) of the 1998 survey sites in Zones 1-5 of the Severn were surveyed for a second time together with one additional site in Zone 2. The principal objective of the 1999 surveys on the Severn was to determine whether changes observed in 1998 were carried through into 1999.

River Dee

The 1984 River Dee eel survey covered 8 sites spanning the catchment. The 1999 survey was therefore a straightforward replication of the earlier survey.

Rivers Frome and Piddle

The 1990 NRA Piddle and Frome survey covered 9 sites so the 1999 survey was a straightforward repeat of the earlier survey. Effectively matching the FBA (Tadnoll Brook) and MAFF (Piddle) studies was more difficult because the original data relate to relatively long river reaches as opposed to the more usual 100 m survey sites and in the case of the Piddle, because the original survey was based on mark/recapture.

For the Tadnoll Brook, 8 sites (each approximately 100 m) were fished in 1999 along the original circa 2 km study reach. The intention was that these 8 sites should be more or less uniformly distributed and therefore representative of the whole reach, thus allowing meaningful temporal comparisons. However, access restrictions prevented any sites being located on the upper-central section of the reach. This necessitated the selection of 6 sites in close proximity on the lower half of the study reach and 2 sites in close proximity on the uppermost part of the study reach. Although a less than ideal compromise, the distribution of habitat along the study reach suggested that in practice, the 8 sites were probably relatively representative of the overall study area.

For the Piddle, 9 sites were fished in 1999. The general area of each site was initially selected so as to give a relatively uniform distribution along the 13.7 km length of river that formed the basis of the 1976/77 study (Figure 3.3), However, final site location was largely determined by access opportunity so that the distribution of sites became essentially semi-random. The objective of site location was that each site would be representative of the 1976/77 survey reach within which it was located and that cumulatively, the 9 sites would be representative of the overall 13.7 km study length of the Piddle. This would therefore allow temporal trend assessment, both on an individual site/reach basis and on an aggregate all sites/whole study length basis.

The strategy for the 1999 Piddle and Tadnoll Brook surveys was deemed to represent the most cost effective deployment of available resources. However, it was recognised at the outset that the approach would inevitably result in some incompatibilities between the historic and contemporary datasets and that this would be likely to impose limitations on the statistical validity of temporal analysis.

3.4 Field Survey Methodology

In general, sites were electric fished using three upstream fishing runs to enable triple catch depletion population estimation. In a few instances four runs were employed, either because of erratic catch depletions, or simply because four runs had been used for the historic survey. In a few instances only two runs were employed, but this typically only occurred in muddy slow-flowing sections requiring an excessive clarification period between fishings.

In most cases, stop nets were employed to isolate the stretch being fished, normally with a run length of approximately 100 m. In some cases, natural discontinuities or barriers obviated the need to place stop nets. However, it was apparent that the deployment of stop nets when fishing for eels is of questionable benefit. This is firstly because of the species' highly sedentary daytime behaviour and secondly because of the practical difficulties in actually setting a net that effectively stops eels.

The fishing equipment comprised a 220 v Allum generator coupled to an Electracatch control box supplying 50 or 100 hertz pulsed direct current to hand-held anodes. At the majority of sites, the generator, control box and fish holding tank were placed in a small flat bottomed boat which was pushed along the stretch behind two wading electrode operators. At deeper sites, electrode operators were also located in the boat. At most deeper sites it was still possible to push the boat, but at two sites, the boat was pulled along by a rope to each bank. At very small or shallow sites, a single long cabled anode was operated from a bankside generator and control box.

In order to draw eels from their burrows and hideaways and to achieve an acceptable capture efficiency, it is necessary to employ relatively high current and voltage settings and to proceed very slowly along the channel. With considerable single-mindedness of approach and if other fish species are essentially ignored, it is possible to obtain adequate quantitative eel data, at least in relatively shallow channels. However, eels are not generally sampled effectively in multi-species electric fishing surveys (this point is discussed by example and in more detail in Section 3.11.2).

On completion of each run, the entire eel catch was anaesthetised with benzocaine (this was found to be more effective than 2-phenoxyethanol). All eels were individually measured to the nearest millimetre and weighed to the nearest gram. A selection of eels across the size range (all eels if the catch was low) was taken for ageing at each site. Heads were removed and placed in labelled jars containing sufficient 50% alcohol to cover the head. All other eels were placed in an aerated recovery tank prior to release to the river at the end of each site survey.

At each site fished, photographs were taken and standard site record sheets, based on the HABSCORE system, were compiled. These provide information on width, depth, flow patterns, substrate, habitat, submerged and emergent vegetation and bankside vegetation. Width and depth transects were taken at each 20 m interval along the fished stretch. Between each transect point, the percentage of in-stream and overhanging vegetation and tree root systems, different bed types and flow patterns (pools, runs, riffles etc.) were estimated by visual inspection and recorded. Overall values for the whole fished stretch could then be obtained by averaging the estimated values for each sector. A sketch plan of each fishing stretch was also produced.

In addition to the electric fishing surveys, random samples of migrating silver eels were obtained from commercial eel fishermen operating traps on both the Piddle (trap at Trigon NGR SY884885 - 173 eels) and the Frome (trap at East Burton NGR SY824874 - 168 eels) during Autumn 1999. Further frozen samples of eels from these two traps collected in Autumn 1997 (Piddle 41 eels and Frome 41 eels) were obtained from the Blandford Forum office of the Environment Agency. Summary raw data is included as Appendix B to this report.

3.5 Laboratory Analysis

All length and weight data from the 1998 and 1999 surveys were entered to Excel 97 spreadsheets. Population estimates were undertaken using the recently produced "Population Estimation by Removal Sampling" package produced by Pisces Conservation Ltd., which can import/export data with Excel 97. For the 1983/4 Severn and Dee surveys, the original length/weight and age data were also entered to Excel 97 and population estimates were recomputed. For the various Piddle and Frome surveys, datasets were reconstituted (to the extent that this was realistic and verifiable) from the published/printed figures and tables and entered into spreadsheets. Thus both contemporary and historic data were brought to a common format for further processing and output. The complete historic and re-survey data set has been compiled on CD and held centrally by the Environment Agency.

Table 3.1 The number of eels aged from each river or reach (in resurveys of the Rivers Servern, Dee, Piddle and Frome

River or Reach	Number of eels aged
River Severn - Zone 1	94
River Severn - Zone 2	129
River Severn - Zones 6 & 7	27
River Dee - below Llangollen	101
River Dee - above Llangollen	65
River Piddle	137
River Frome - Tadnoll Brook	147
River Piddle - silver eels from trap	74
River Frome - silver eels from trap	73
Total all sites	847

For age determination, both left and right sagittal otoliths were dissected out from each preserved eel head. The pairs of otoliths were then dried at 60-65 °C for at least 48 hours prior to further preparation. Various methods of otolith preparation can be employed including clearing whole otoliths (e.g. Mann and Blackburn 1991), embedding and grinding (e.g. Svedang 1999) and burning and cracking (e.g. Naismith and Knights 1993). The method used in this study was burning and cracking, with cracked otoliths mounted in clear silicone between slides and coverslips. This method was selected primarily because it is employed by the Environment Agency's National Fisheries Laboratory (NFL) at Brampton. The NFL staff provided both training in the use of the method and subsequent external quality control. Quality control was based on random samples being sent to NFL for independent re-ageing.

Ages from ring counts were 'read' independently by two operators using a light microscope with reflected light against a dark background. Any differences in interpretation were then reconciled by discussion, aided by a third 'reader' if necessary. Table 3.1 shows the number of eels aged from each river or zone.

3.6 Survey Results for Individual 1998/99 Re-Survey Sites

Data for each of the 75 site surveys conducted in 1998 and 1999 have been compiled on a standardised two page site report. Each report contains a site photograph, a 1:12,500 scale map showing the exact location of the fishing stretch and a sketch plan of the fished stretch showing key habitat features. Catch statistics are summarised in terms of the number of eels captured on each run, the resulting population estimate, density (mm^{-2}), biomass (gm^{-2}) mean length (mm) and mean weight (g). A length frequency plot for the eels captured is also included in the site report.

The site reports also include quantitative information on habitat, including channel width and depth, nature of substrate, type and availability of cover etc. Other information in the reports includes fishing method, an assessment of flow conditions at the time of the survey and details of riparian ownership.

The site reports have been compiled as an unpublished annexe to this report which is held centrally by the Environment Agency. Thus any future monitoring of eel stocks in the Severn, Dee, Piddle and Frome can replicate the 1998/99 surveys precisely in terms of method and site location. The inclusion of habitat information will enable any major temporal changes in habitat character to be identified.

3.7 River Severn - Results and Initial Discussion

3.7.1 Introduction

The River Severn resurvey programme of 33 sites was the most extensive of the three river-system surveys undertaken for this R&D project (Section 3.3 and Figure 3.1). The lower Severn survey was the only one of the three that combined both a relatively large number of sites (N = 24, 1998; N = 17, 1999) and a very close match (site location and survey method) between the historic and current surveys. The resulting Severn dataset is thus probably the only one that affords a sound statistical basis for temporal analysis of biomass and density.

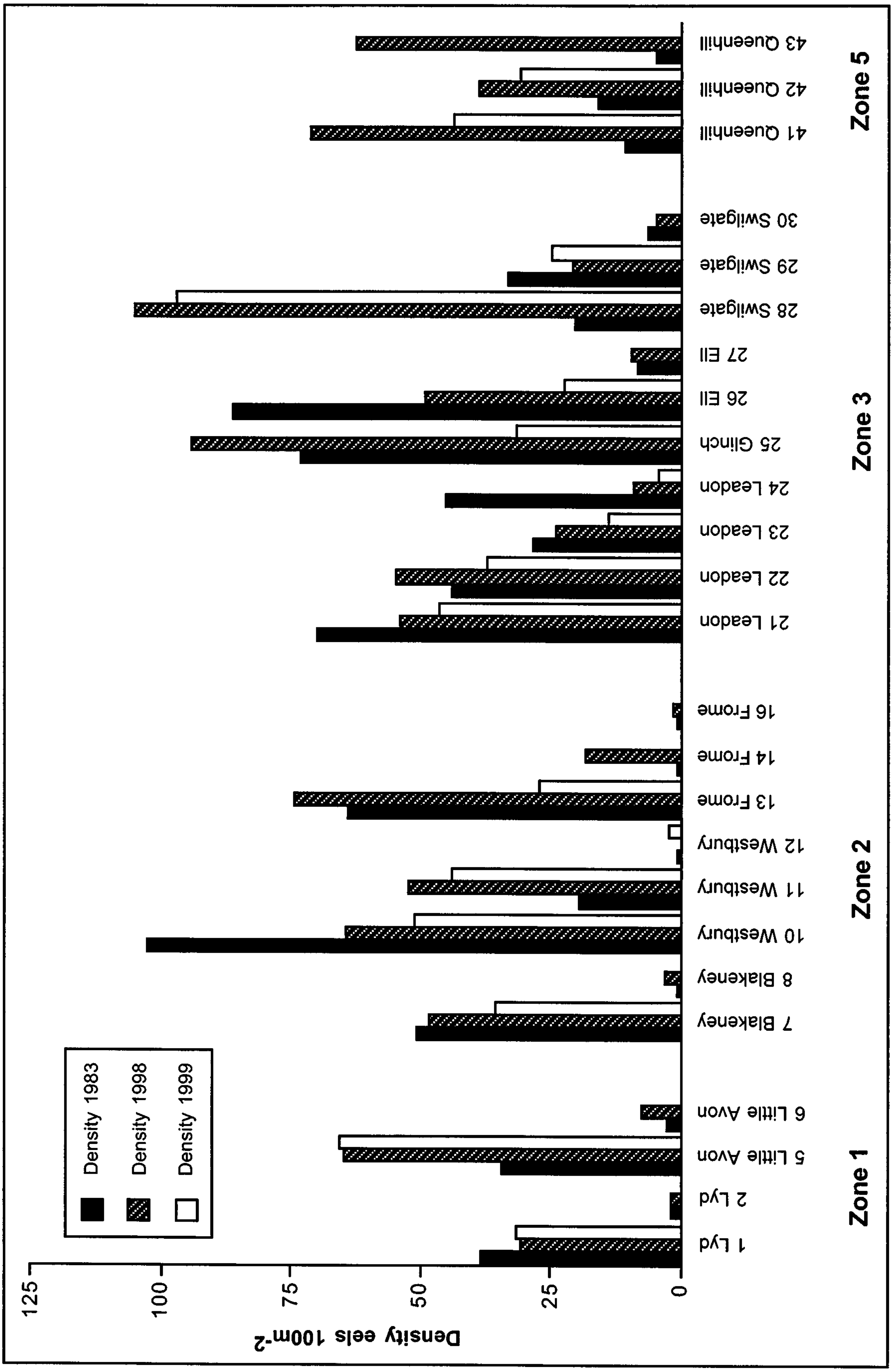


Figure 3.4 **Eel density at the Lower Severn survey sites (Zones 1, 2, 3 & 5) in 1983, 1998 and 1999**

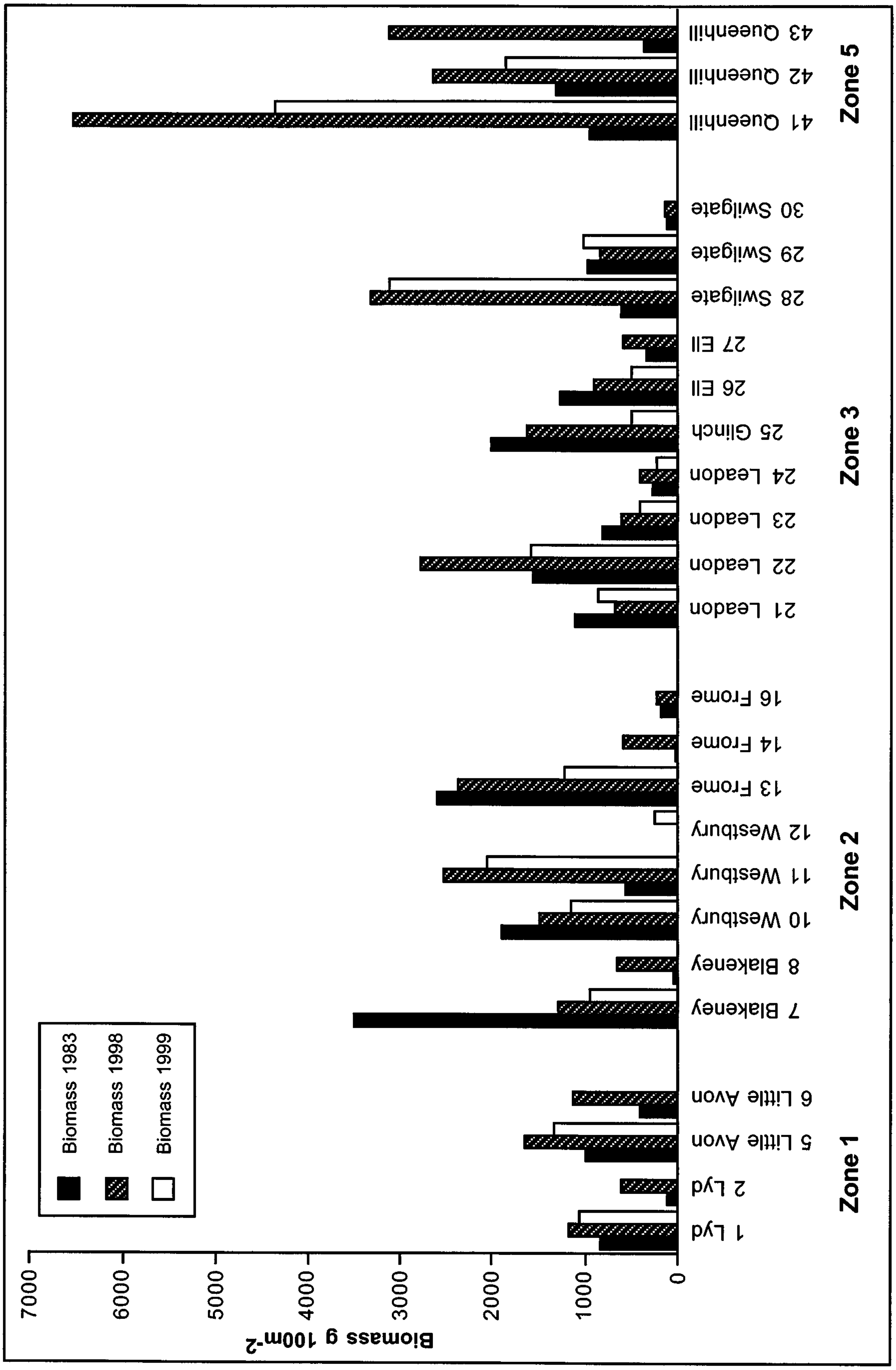


Figure 3.5 **Eel biomass at the Lower Severn survey sites (Zones 1, 2, 3 & 5) in 1983, 1998 and 1999**

Sites and survey dates for the 1983, 1989 and 1999 eel surveys are shown in Appendix B. The 1998 survey dates for individual sites were typically between 1 and 3 weeks later than those in 1983. The 1999 survey was a very close temporal match to the 1998 survey with few site sampling dates differing by more than 2 days.

3.7.2 Density and biomass

(a) Lower Severn, Zones 1, 2, 3 and 5

Eel density data for the 1983, 1998 and 1999 surveys of the lower Severn are presented in Figure 3.4 and the corresponding biomass data are shown in Figure 3.5. As can be seen from Figure 3.4, eel densities vary greatly from site to site. This pattern of inter-site variation is typical of large complex river systems and can be attributed to a range of environmental factors such as differences in habitat quality and the presence or absence of upstream migration barriers, as discussed in Chapter 2. Importantly, there is no obvious suggestion of long-term change in density. Typically, those sites that supported either low, moderate or high densities in 1983 support broadly similar densities now. However, there are some clear exceptions to this generality. For example, the three sites on the Queenshill Brook (41, 42, 43) and Site 28 on the Swilgate show a marked increase in density. Conversely, Site 24 on the Leadon and Site 26 on the Ell appear to support far fewer eels than they did in 1983

Even in a situation of no overall change, it would be expected that there would be some major, but localised, changes in eel populations over a 15 year period, such as those highlighted in the previous paragraph, affecting the whole or parts of individual tributaries. Habitats may have changed as a result of positive or negative management activities (or recovery from earlier land drainage operations). Migration barriers may have been modified or removed.

In the case of Sites 41, 42 and 43 on the Queenhill Brook (Zone 5), the low (relative to 1998 and 1999) density and biomass of eels in 1983 are probably attributable to habitat impoverishment following channel dredging and bed regrading undertaken during the previous year. Although detailed records of the nature and extent of channel works are no longer available, they are known to have encompassed Sites 41 and 42 and may have extended to Site 43.

At the great majority of sites, biomass (Figure 3.5) for the three (or two) survey years broadly reflects density. As with density there are some exceptions, the most obvious being the Blakeney Brook. At both sites on this stream (Site 7 and Site 8), there was a relatively small change in density from 1983 to 1998 but a much greater change in biomass. At Site 7, mean eel weight decreased from 69 g in 1983 to 27 g in 1998 whereas at Site 8, mean eel weight increased from 42 g in 1983 to 210 g in 1998. Again however, localised changes of this sort are almost inevitable between samples, as indicated by the data reviewed in Chapter 2.

The critical question is whether the data provide any evidence for catchment-wide long-term trends in eel population density, biomass or size structure, relating to either changes in glass eel recruitment or changes in fishing pressure. Basic catch and population statistics for each site are presented in Appendix B. Tables 3.2 and 3.3 provide a temporal comparison of mean density for each Zone, mean density for Zones 1-3 and the mean density for all lower Severn Zones for the 1983 and 1998 surveys and the 1983 and 1999 surveys based on common sites.

Table 3.2 Comparison of eel population density (eels $100m^{-2}$) for the 1983 and 1998 lower Severn surveys, based on common sites.

	Zone 1	Zone 2	Zone 3	Mean Zones 1-3	Zone 5	Mean all zones
1983	18.0	34.2	41.5	34.9	10.3	31.8
1998	26.3	37.5	42.6	37.8	57.0	40.2

Table 3.3 Comparison of eel population density (eels $100m^{-2}$) for the 1983 and 1999 lower Severn surveys based on common sites.

	Zone 1	Zone 2	Zone 3	Mean Zones 1-3	Zone 5	Mean all zones
1983	33.7	47.5	50.1	41.8	13.2	43.4
1999	48.5	32.0	34.8	35.7	37.2	35.9

The 1983 versus 1998 data suggest that there has been an increase in eel population density in each of the four zones and that there has been a mean increase for all sites from 1983 to 1998 of approximately 26%. This apparent increase is not statistically significant (Paired *t*-test, N = 24, $P = 0.18$). However, the data also suggests that there has been relatively little change in density in Zones 1-3 and that most of the apparent increase is attributable to Zone 5, where observed density increased by 553%. As indicated above, the temporal difference in density in Zone 5 may simply reflect the effects of channel works a year prior to the first (1983) survey

The 1983 versus 1999 data suggests an apparent decrease in overall density of approximately 17%, but again this is not statistically significant (Paired *t*-test, N = 17, $P = 0.40$). A major increase in density (282%) over the period is still apparent for Zone 5, however.

A comparison of the mean eel density for 1999 with that for 1998 (based on the 16 sites common to both surveys) suggests a decline in mean density of approximately 29% from 53.5 eels $100m^{-2}$ in 1998 to 38.0 eels $100m^{-2}$ in 1999. Furthermore, this difference is highly statistically significant (Paired *t*-test, N = 16, $P = 0.0034$). Examination of Figure 3.4 shows that estimated eel population density has increased slightly at one site, is effectively unchanged at two sites but decreased relatively substantially at the other 13. The high level of statistical significance is therefore not surprising. The decline in eel density in Zone 5 of 32.1% is very close to the overall mean.

Tables 3.4 and 3.5 provide a temporal comparison of mean biomass for each Zone and the overall mean biomass for all lower Severn Zones for the 1983 and 1998 surveys and the 1983 and 1999 surveys, based on common sites. The 1983 versus 1998 data suggest that there has been an increase in eel biomass in each of the four zones and that there has been a mean increase for all sites from 1983 to 1998 of approximately 66%. The biggest increase (469%) occurred in Zone 5. The overall observed biomass increase borders on statistical significance (Paired *t*-test, N = 24, $P = 0.053$). The 1983 versus 1999 data suggest a very small increase in overall eel biomass of approximately 5.5% but this is not statistically significant (Paired *t*-

test, N = 17, $P = 0.84$). However, the apparent overall biomass increase is essentially limited to Zone 5, where observed biomass increased by 274% from 1983 to 1999.

Table 3.4 Comparison of eel biomass (g 100m^{-2}) for the 1983 and 1998 lower Severn surveys.

	Zone 1	Zone 2	Zone 3	Mean Zones 1-3	Zone 5	Mean all zones
1983	596.7	1261.5	904.0	964.6	873.8	953.3
1998	1138.7	1303.7	1187.9	1217.9	4097.8	1579.1

Table 3.5 Comparison of eel biomass (g 100m^{-2}) for the 1983 and 1999 lower Severn surveys.

	Zone 1	Zone 2	Zone 3	Mean Zones 1-3	Zone 5	Mean all zones
1983	921.2	1716.0	1071.7	1117.4	1128.3	1250.2
1999	1195.2	1127.6	1024.9	1082.0	3098.0	1319.1

A comparison of the mean eel biomass for 1999 with that for 1998 (based on the 16 sites common to both surveys) presents an almost identical picture to that for density, suggesting a decline in overall mean eel biomass of approximately 31% from 1925 g 100m^{-2} in 1998 to 1319 g 100m^{-2} in 1999. As with density, this difference is highly statistically significant (Paired *t*-test, N = 16, $P = 0.0031$). Also, the decline in eel biomass in Zone 5 of 33.2% is very close to the overall mean.

The apparent and statistically significant decline in both biomass and density between 1998 and 1999 is unexpected, given that the one year time span is very small relative to the life cycle of an eel. Survey sites, survey team, equipment, equipment settings, methodology and survey timing were effectively the same for the two surveys so it is unlikely that an explanation lies here. River conditions were good and apparently similar for both surveys at all but two sites. Water clarity was noticeably poorer in 1999 than in 1998, to the extent that capture efficiency may have been potentially impaired, at Site 21 on the Leadon and Site 26 on the Ell. However, whilst Site 26 was one of four sites showing a substantially above average reduction in estimated eel population density from 1998 to 1999, the decline at Site 21 was quite small. Finally, if the observed reduction in population size was simply a result of reduced fishing efficiency, then this would be expected to be most apparent for the smaller size classes of eels, as these are the least amenable to capture. The fact that the overall 1998 to 1999 reductions in density (-29%) and biomass (-31%) are virtually identical implies that there has been no overall change in the size structure of the catch. The similarity in size structure of the population is shown particularly clearly by the cumulative length frequency plots in Figure 3.11 for Zones 2 and 3

It thus seems unlikely that the apparent decline in population density and biomass between 1998 and 1999 is a sampling effect and it is therefore difficult to avoid the conclusion that the observed differences are real. However, a major change in the underlying eel population density and biomass within the Severn catchment as a whole, within one year, is also highly unlikely. The underlying cause of the observed change is enigmatic but the most likely explanation may lie in differences in seasonal migration patterns and hence eel distribution within the catchment between the two years. Factors such as water temperature have been shown to affect upstream migration of eels (White and Knights, 1997) and seasonal cycles of density and biomass have been observed on some rivers. An example is provided in Figure 2.14 of this report, which shows mean monthly density and biomass at sites on the Fowey from 1977 to 1987. Density and biomass increased almost threefold between April and September although average body length changed little. A similar but less pronounced summer maximum was observed for eel catches on the screens at West Thurrock Power Station on the Thames Estuary (Figure 2.35), where fish were sampled from 1974 to 1993.

Figure 3.6 shows daily mean water temperature for the Severn in 1998 and 1999 recorded at Saxons Load near Upton on Severn (NGR SO 863387). It has been assumed that water temperature at this site will be broadly indicative of temperatures at the lower Severn survey sites for the two years. During the July surveys (19 - 23 July) average water temperature was approximately 20 °C in 1999 and 18 °C in 1998. During the month preceding the surveys, water temperature averaged approximately 3 °C warmer in 1999. Water temperature dropped sharply in early August 1999 and by the time of the August survey (18 - 20 August), water temperature was some 3 °C cooler than in 1998. Thus there were marked differences in water temperature between the two years, both at the time of and preceding the surveys.

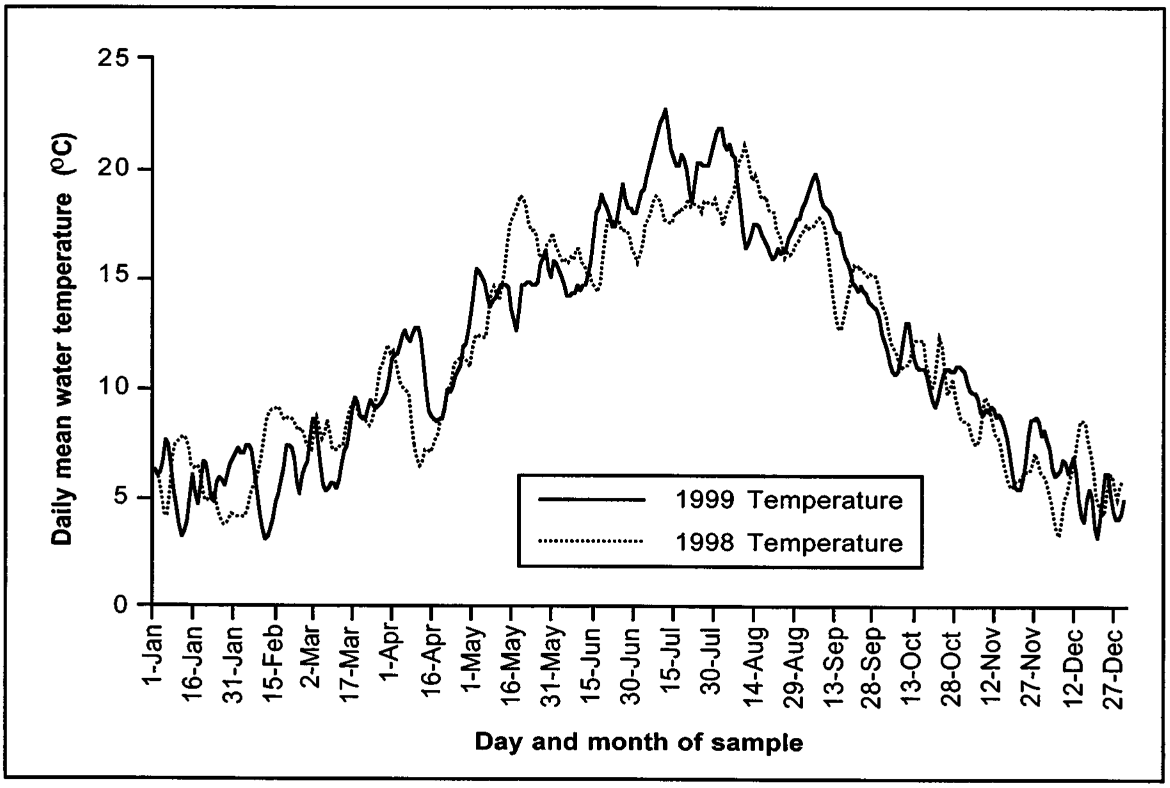


Figure 3.6 A comparison of daily mean water temperature at Saxon Lode on the Lower River Severn in 1998 and 1999

The change in population density from 1998 to 1999 varies greatly between sites, ranging from +20% at Site 29 to -67% at Site 25. However, there is some evidence that this variation may not be entirely random. The mean 1998-1999 population density change for the July surveys was -21% but it was -45% for the August surveys, although with the August sample set comprising only 4 sites, this is not statistically significant. Figure 3.7 shows percentage change in eel density in the lower Severn between 1998 and 1999 plotted against distance from the tidal limit. Although the correlation is weak, there is a possible pattern of greater density decline with increasing distance from the tidal limit.

Although the foregoing discussion is highly speculative, it does offer some potential insights into factors affecting eel population density and distribution.

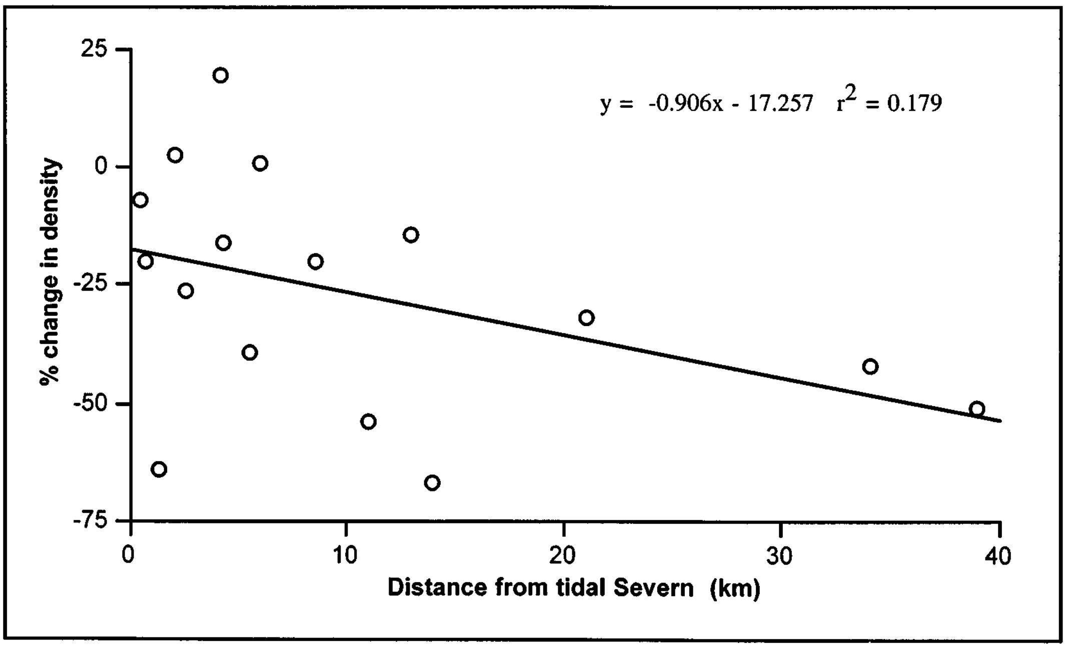


Figure 3.7 Percentage change in eel density in the lower Severn between 1998 and 1999 in relation to distance from the tidal limit

In summary, the data suggests that reduced elver recruitment between mid 1980s and late 1990s has not had a major effect on eel stock density or biomass within the lower Severn. The data provides no real indication as to whether changes in glass eel fishing pressure over the same period have been of significance because the observed Zone 5 (above the main fishery in Zone 4) population density in 1983 is likely to have been heavily influenced channel works in the previous year. However, the statistically highly significant difference in observed biomass and density between 1998 and 1999 provides a very clear illustration of the difficulties in making temporal comparisons of eel populations.

(b) Upper Severn, Zones 6 and 7

A catch summary for the 8 upper Severn Sites (Zones 6 & 7) is given in Table 3.6 which shows density and biomass estimations for 1983 and 1999. As both the number of sites and the total number of eels captured were small (total catch 1983 - 18 eels; total catch 1999 - 37 eels), statistical analysis of population density is effectively precluded and data interpretation is therefore limited to essentially observational comments.

Table 3.6 Eel population density and biomass for the 1983 and 1999 Upper Severn surveys

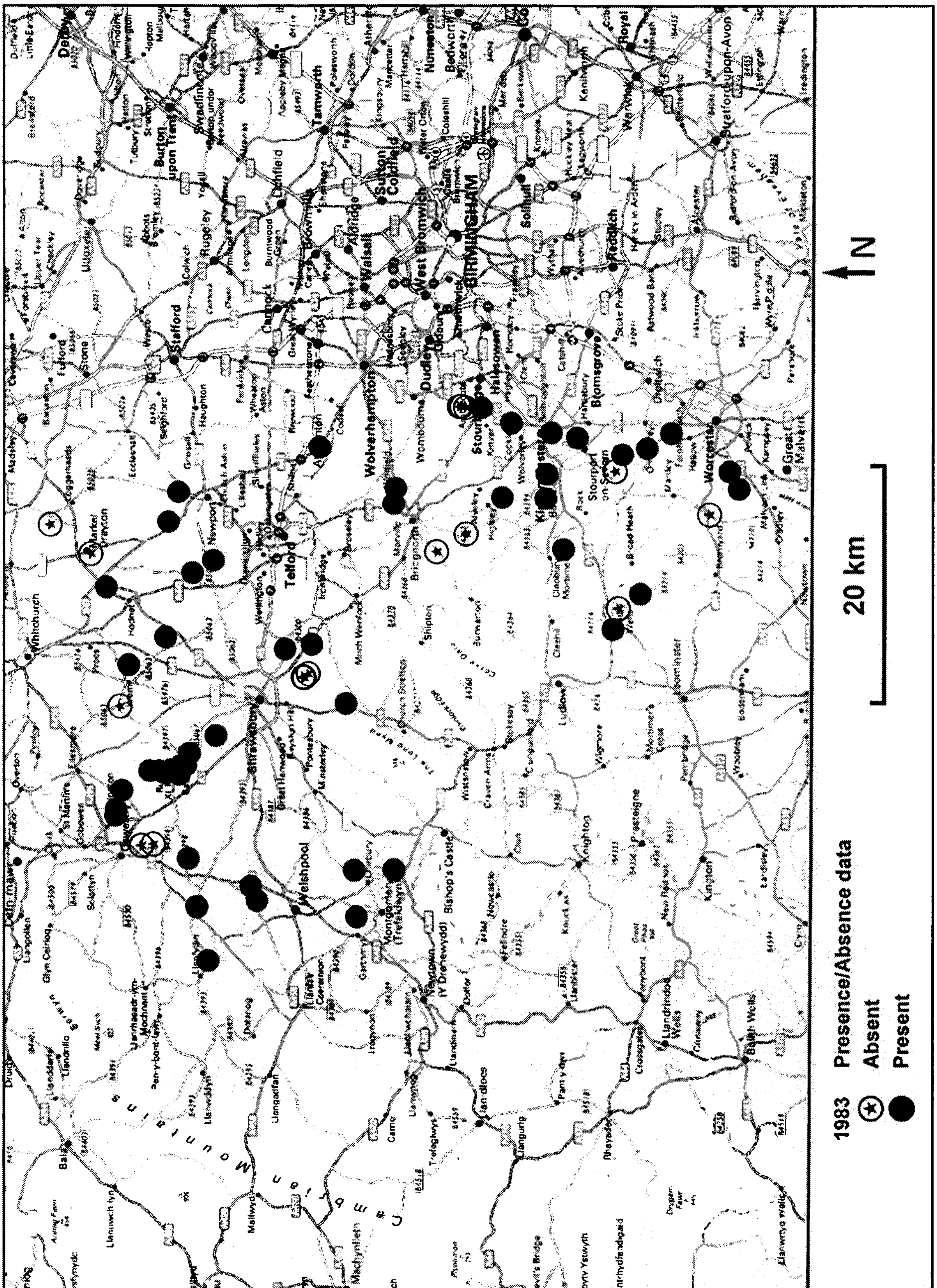
Site	Density 1983 eels $100m^{-2}$	Density 1999 eels $100m^{-2}$	Biomass 1983 g $100m^{-2}$	Biomass 1999 g $100m^{-2}$
44 Leigh Brook	2.3	3.6	54.8	217.4
48 Kyre Brook	0	1.0	0	57.5
51 Grimley Brook	0.3	0.8	37.1	79.7
52 Shrawley Brook	2.9	1.9	78.8	14.1
59 Smestow Brook	0	0	0	0
60 Smestow Brook	0	0	0	0
62 Dowles Brook	0.4	0.7	35.9	66.2
66 Borle Brook	0	0	0	0

Eels were present in both years at 4 sites, absent in both years at 3 sites and at one site, the Kyre Brook, eels were absent in 1983 but present in 1999. The mean population density in 1999 was 1.0 eels $100m^{-2}$ compared to 0.74 eels $100m^{-2}$ in 1983. Biomass was 54.4 g $100m^{-2}$ in 1999 compared to 25.8 g $100m^{-2}$ in 1983. The data clearly implies that there has been no decline in eel stock status in these parts of the Upper Severn. On the contrary, there may have been a slight improvement but the putative increase in numbers could equally be due to random variation.

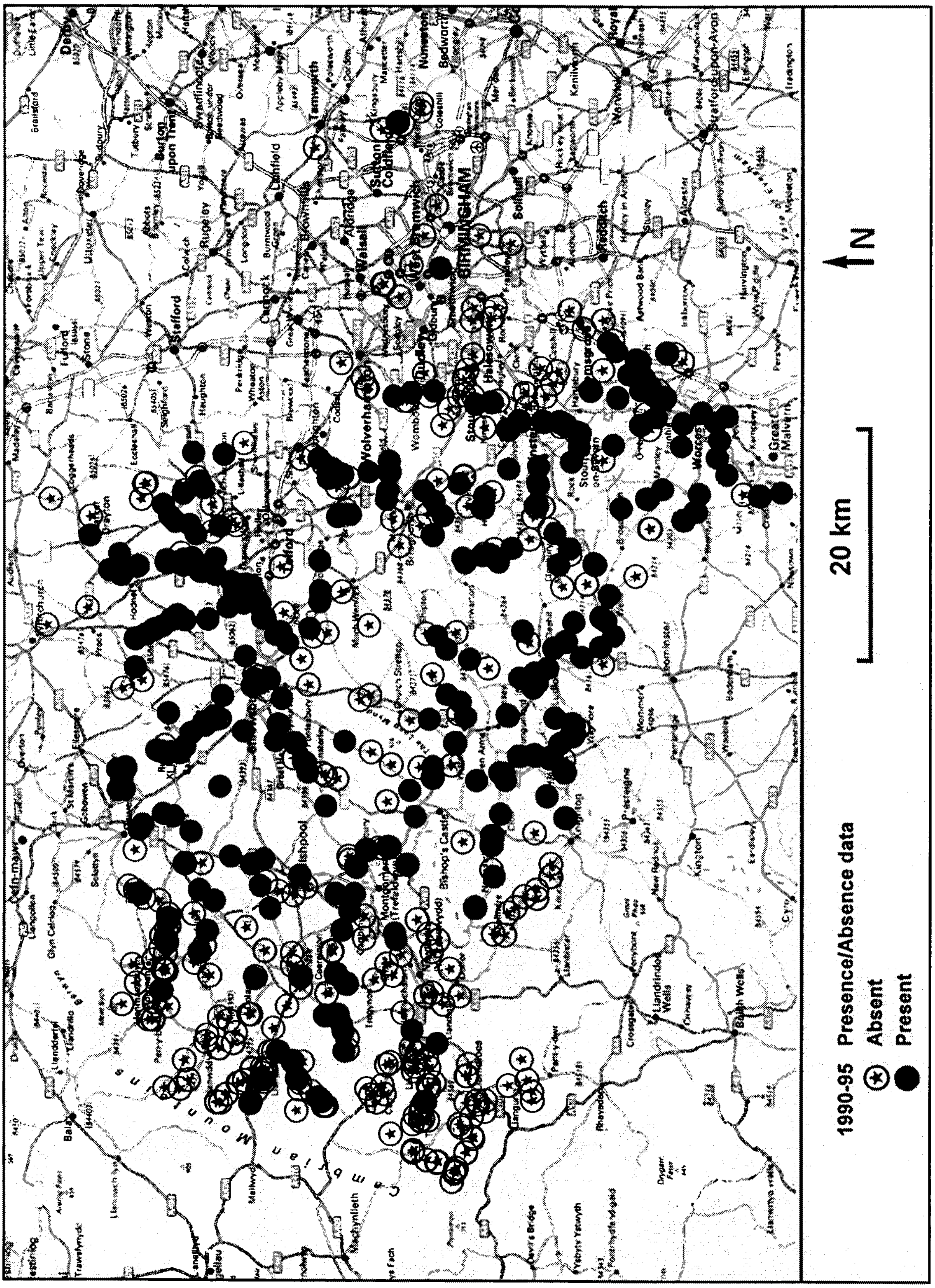
There is a suggestion from the data that the average size of eels in Zones 6 and 7 may have been greater in 1999 than in 1983. Mean recorded length for the 1983 eel catch was 254 mm but this increased to 299 mm in 1999. However, because of the wide range of lengths and relatively small sample size, the observed difference in mean length is not statistically significant.

3.7.2 Eel distribution in the upper catchment based on Environment Agency survey data for 1991-1995.

The 1983 eel survey covered 66 sites within the upper Severn catchment (Zones 6-10). Routine Environment Agency surveys carried out between 1991 and 1995 covered many of the same tributaries as the 1983 eel survey and eel catches were routinely recorded in the EA surveys. This potentially facilitates a 10±2 year temporal comparison. However, there are inherent difficulties in making quantitative (density, biomass) comparisons between eel data from multi-species surveys and data from mono-specific eel surveys. These difficulties are discussed in detail in Section 3.11.2. In practice, the probable quantitative incompatibility of the datasets can be avoided if a simple presence/absence comparison of distributional range is undertaken. Eel population densities are typically very low in the upper catchment and many sites/tributaries are likely to be at or beyond the limits of upstream migration. Distribution is therefore likely to be the most useful indicator of changing stock status in the upper Severn.



Figures 3.8 Location of 1983 eel survey sites for the upper Severn with eels recorded on a presence absence basis.



Figures 3.9 Location of 1991-1995 routine fish stock survey sites for the upper Severn with eels recorded on a presence absence basis.

Figures 3.8 shows the location of the 1983 eel survey sites and Figure 3.9 shows the location of the 1991-1995 routine fish stock survey sites for the upper Severn, with eels recorded on a presence absence basis in both cases. 1995 is the latest year for which routine fish stock survey data could be obtained from the EA regional database. There is no apparent reduction in eel distribution over the 10±2 year period between the two surveys. Virtually all eel-positive sites from the 1983 survey are matched exactly, or very closely, by eel-positive sites in the 1990s. However, for several of the 1983 eel-negative sites, there are identical or very closely matching sites which were eel-positive in the 1990s.

Overall, eels currently appear to be well distributed through the upper catchment, although few appear to penetrate the source streams arising from the Cambrian Mountains. Eels also appear to be largely absent from streams draining the Birmingham conurbation.

3.7.3 Size structure of the Severn eel population

Combined length frequency distributions for eels from each of the lower Severn survey zones are shown as conventional bar chart length frequency (percent) plots in Figure 3.10 and as cumulative length frequency plots in Figure 3.11. These figures relate to the 1983, 1998 and 1999 surveys. Figure 3.11 shows a clear shift in population structure towards larger eels in both 1998 and 1999 in Zones 2 and 3. This change in population structure towards larger individuals is also apparent in Zone 1 in 1998 but is much less apparent in Zone 1 in 1999 or in Zone 5 in either year. It should be noted however, that because of relative sample sizes, consistent trends in population structure are much more likely to be apparent in Zone 2 (7 sites 1998, 5 sites 1999) and Zone 3 (10 sites 1998, 8 sites 1999) than in either Zone 1 (4 sites 1998, 2 sites 1999) or Zone 5 (3 sites 1998, 2 sites 1999).

Pooled length frequency data for the four zones are plotted in the bottom graph of Figure 3.11. This illustrates both the clear overall shift towards larger eels from 1983 to the late 1990s and the marked similarity in population size structure between 1998 and 1999.

Figure 3.12 examines the proportion of juvenile eels <150 mm long, (the size band approximately corresponding to the 0+ to 3+ cohorts), for the 1983, 1998 and 1999 surveys in Zones 1, 2, 3 and 5 of the Lower Severn. The inclusion of 4 cohorts in the comparison is useful because it smoothes out the effects of unusually high or low rates of elver recruitment in any one year. Figures 2.2 and 2.3 (Chapter 2) indicate that glass eel recruitment was relatively good in Europe in 1980, 1981 and 1982 but very low in 1983.

In 1998, the proportion of small eels in Zones 1 - 3 was approximately one half that of 1983. The difference between years in Zone 5 was much less marked. Overall there was a 52% reduction in small (<150 mm) eels between 1983 and 1998. The 1999 situation was generally similar to that for 1998, apart from the greater proportion of <150 mm eels in Zone 1. Zone 1 was only represented by two sites in 1999 however. Overall there was a 42% reduction in small (<150 mm) eels between 1983 and 1999.

The observed reduction in eels of less than 150 mm after 1983 is statistically significant in both 1998 (Paired *t*-test, N = 24, $P = 0.029$) and 1999 (Paired *t*-test, N = 17, $P = 0.055$). There is no significant difference in the proportion of small eels between 1998 and 1999 (Paired *t*-test, N = 16, $P = 0.79$)

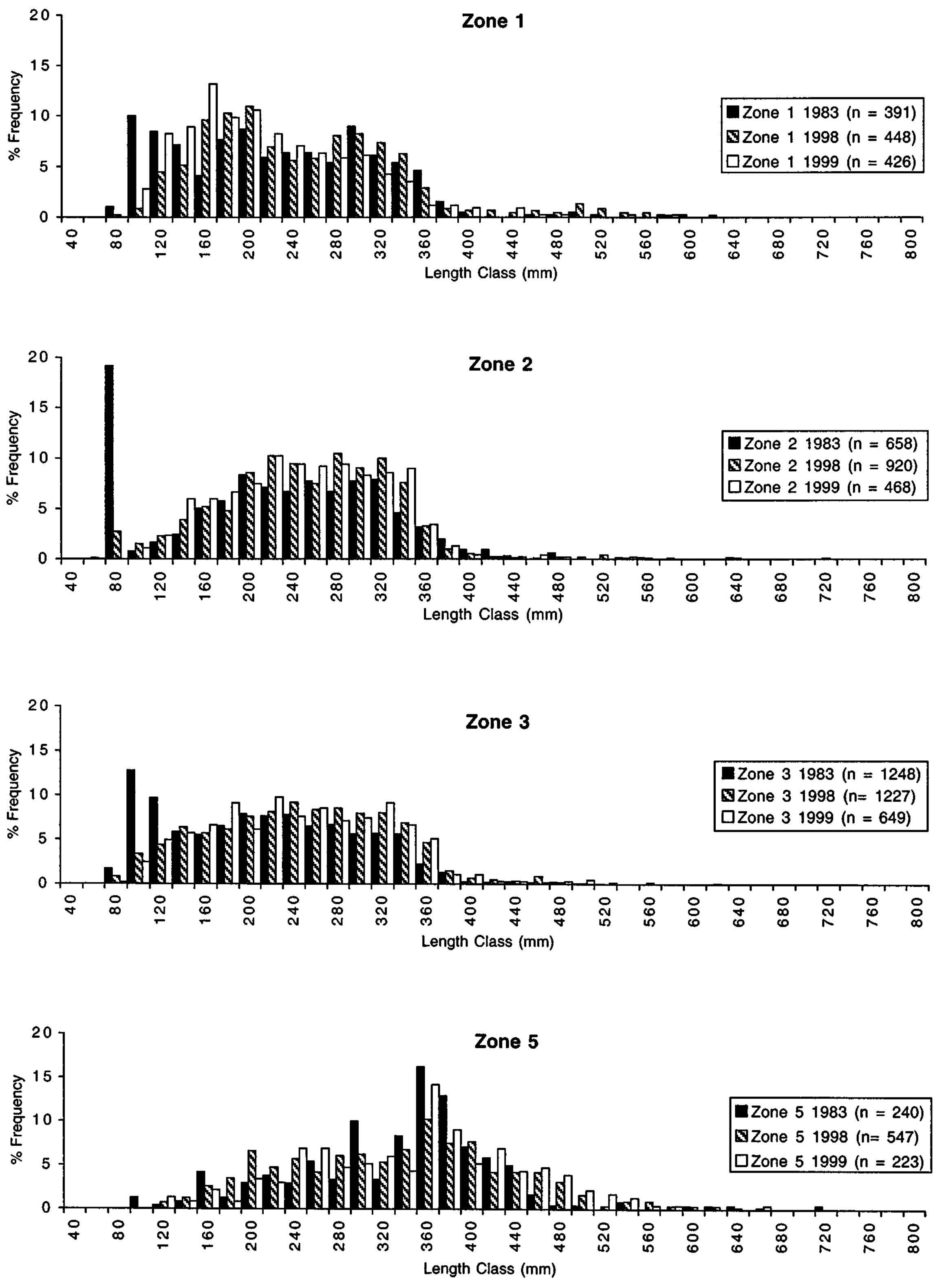


Figure 3.10 Comparison of percentage length frequency for the 1983, 1998 and 1999 eel surveys of the lower Severn

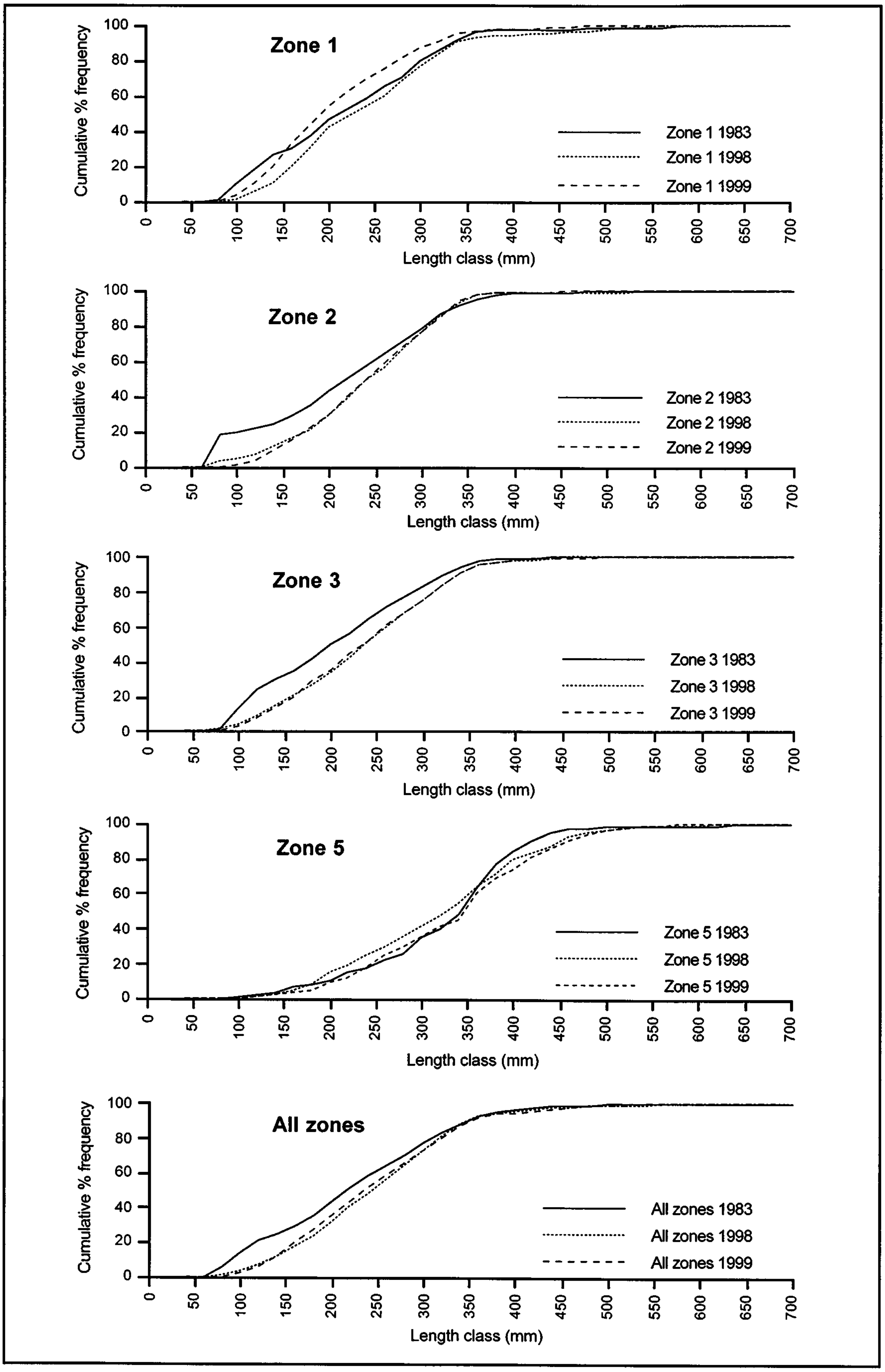


Figure 3.11 Comparison of cumulative percentage length frequency for the 1983, 1998 and 1999 eel surveys of the lower Severn

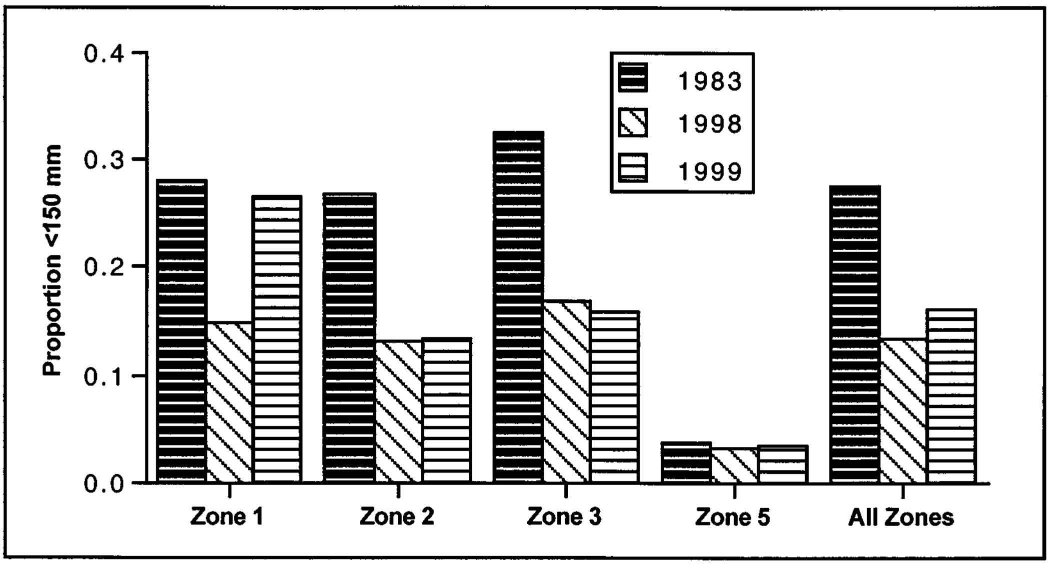


Figure 3.12 The proportion of Severn eels <150 mm long in 1983, 1998 and 1999

One of the more notable features of Figure 3.12 is the relatively low proportion of small eels in Zone 5 and the apparent very small change between 1983 and 1998/99. This almost certainly relates to the distance of the Zone 5 sites from the tidal limit, with the consequence that the primary source of recruitment is the upstream migration of older juveniles rather than 0+ eels. Whether the Zone 4 glass eel fishery also affects recruitment patterns to Zone 5 cannot be determined from the data. However, the current density and biomass in Zone 5 do not suggest any negative effects from the elver fishery on overall eel stocks.

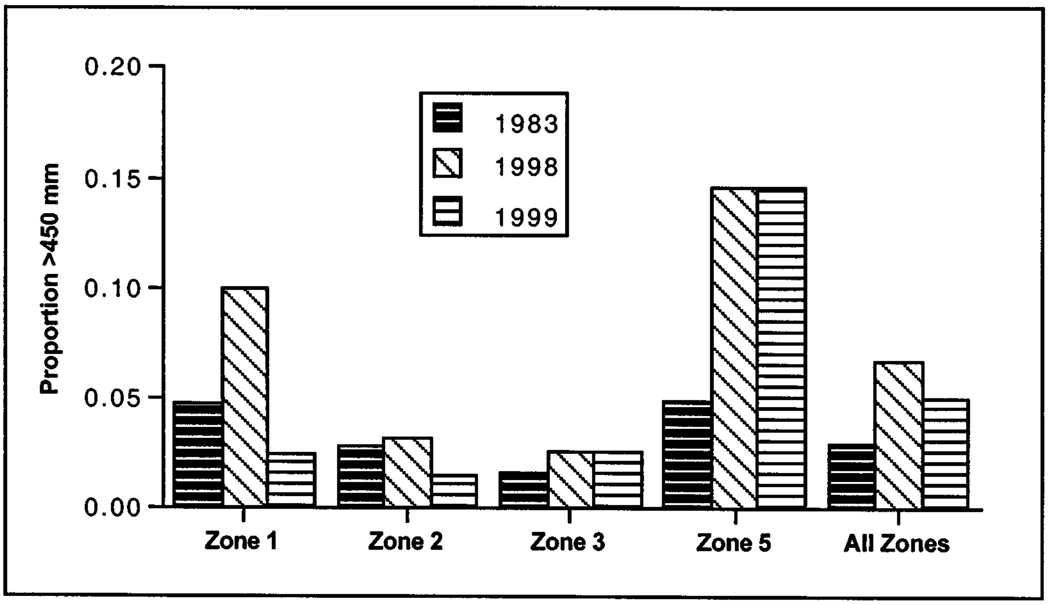


Figure 3.13 The proportion of Severn eels >450 mm long in 1983, 1998 and 1999

Figure 3.13 examines the proportion of large female eels > 450 mm in the >250 mm component of the population in Zones 1, 2, 3 and 5. As might be expected from the relative proportion of small eels, the highest proportion of large females occurred in Zone 5, although a relatively high proportion also occurred in Zone 1. Whilst the 1998 data suggests that the proportion of large females had increased since 1983, this was less obviously so in 1999. The temporal pattern for large female eels is thus rather less clear cut than for the small eels. Overall there was a 128% increase in the proportion of large female eels (>450 mm) between 1983 and 1998. Using the 1999 data, the suggested increase was 70%.

There is only relatively weak statistical support for the putative increase in the proportion of large female eels in the population after 1983 with P values falling short of the normally accepted level of significance (1998 Paired t-test, N = 24, $P = 0.13$; 1999 Paired t-test, N = 17, $P = 0.091$).

3.7.5 Mean eel population density, biomass and size in relation to distance from tidal limit

Figures 3.14 and 3.15 below show the relationship between eel population density and biomass (1998 Zones 1, 2, 3 & 5; 1999 Zones 6 & 7 and corresponding sites for 1983) and the approximate distance from the tidal limit (the confluence of tributary and tidal main river for Zones 1- 3 and distance upstream of Upper Lode Weir, Tewkesbury, for Zone 5, 6 & 7).

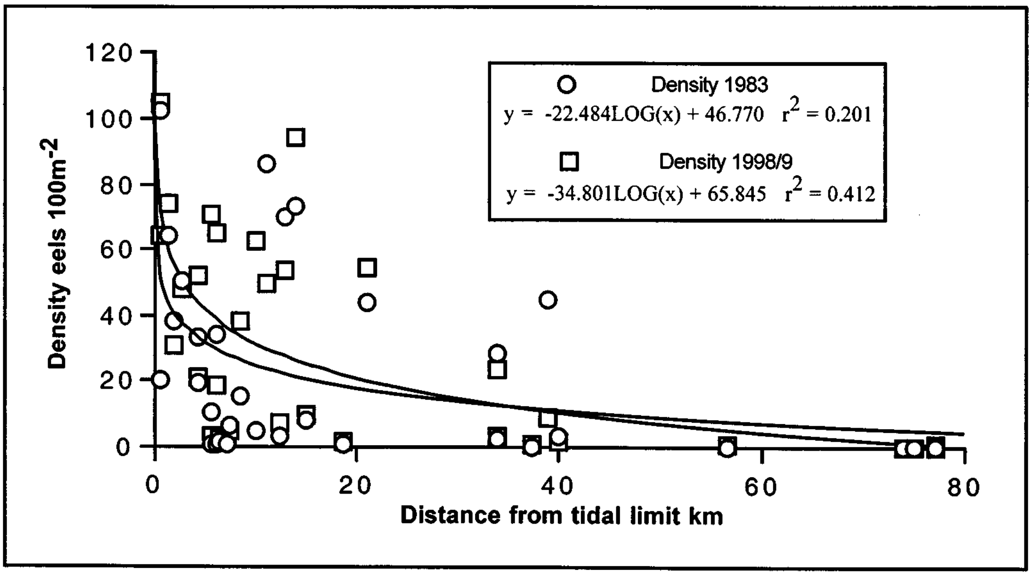


Figure 3.14 Eel density in the River Severn in relation to distance from the tidal limit for the 1983 and 1998/99 surveys

There is a general trend of declining density and biomass with distance from the tidal limit. Close to the tidal limit, biomass and density are highly variable and population size is likely to be a function of such factors as local habitat quality and the presence/absence of migration barriers. At sites which are distant from the tidal limit, populations are invariably low. The general pattern is very similar for both the historic (1983) and contemporary (1998/99) surveys. The observed eel distribution in the Severn therefore closely matches that seen in other large river systems reviewed in Chapter 2.

Figure 3.16 displays mean eel length (1998 data for Zones 1, 2, 3 & 5; 1999 data for Zones 6 & 7 and corresponding sites for 1983) in relation to distance from the tidal limit. There is no clear indication of a relationship between mean eel size and distance from tidal waters. The Severn data therefore do not obviously support the classic size distribution of a lower catchment dominated by smaller eels and an upper catchment dominated by large females.

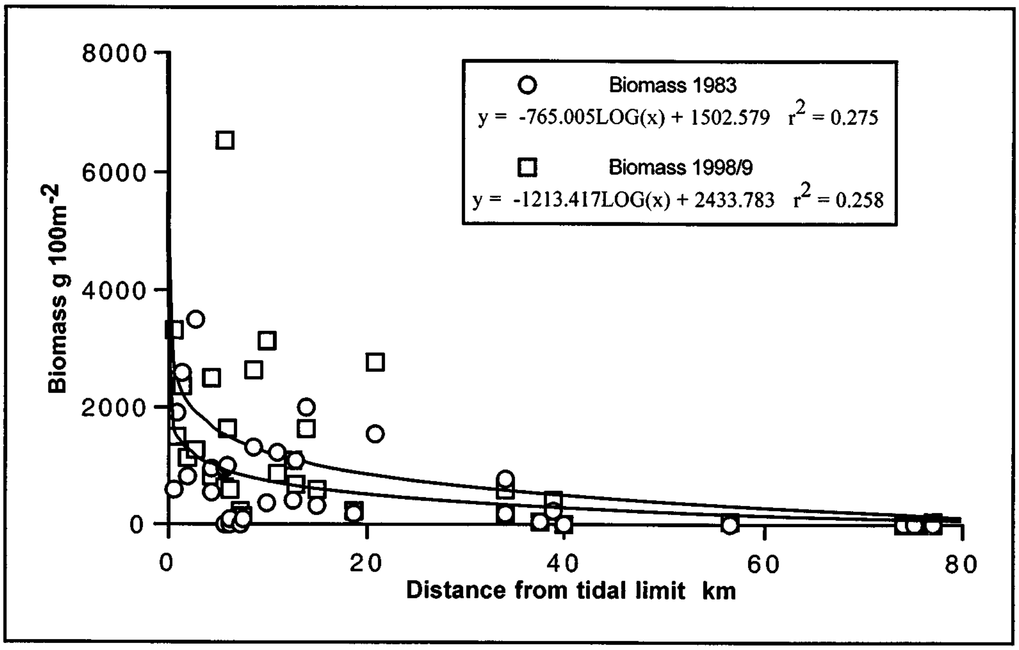


Figure 3.15 Eel biomass in the River Severn in relation to distance from the tidal limit for the 1983 and 1998/99 surveys

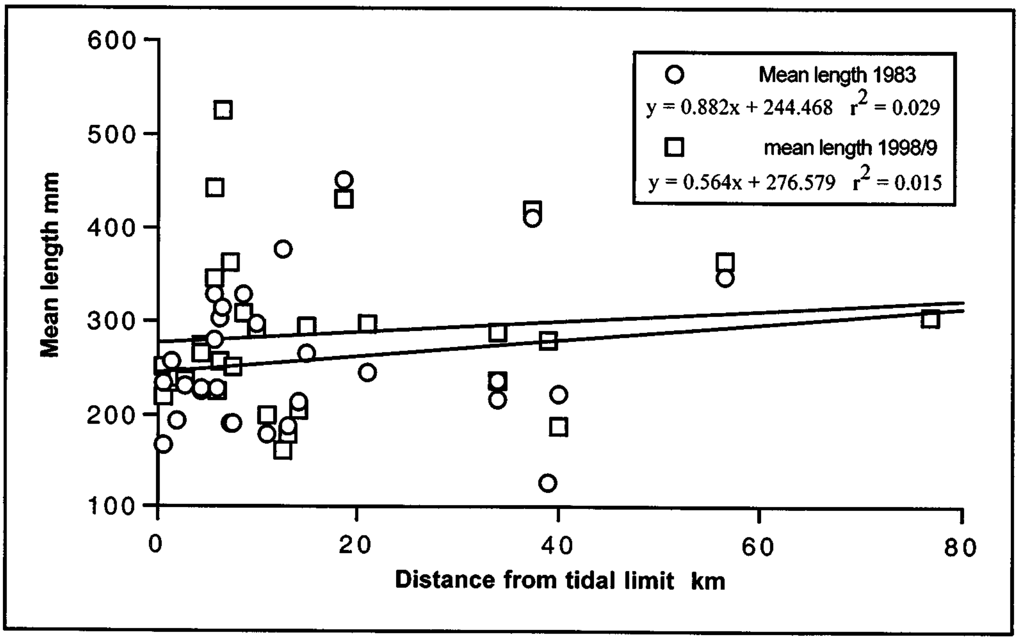


Figure 3.16 Mean eel length in the River Severn in relation to distance from the tidal limit for the 1983 and 1998/99 surveys

3.7.6 Age and growth rate of the Severn eel population

Eel age data for the Severn system is presented in Figure 3.17 (Lower Severn, Zone 1), Figure 3.18 (Lower Severn, Zone 2) and Figure 3.19 Upper Severn, Zones 6 &7). In each figure, the two graphs are organised as follows:

upper graph length against age with regression line for the 1983 eel survey;

lower graph length against age with regression line for the 1998/99 eel surveys;

Table 3.7 gives calculated annual growth rates ((length -70 mm) / age) for each zone as means and standard deviations. Calculated growth rates differ slightly from those implied by the slope of the regression lines in Figures 3.17 - 19, primarily because of the variable position of the intercept on the length axis.

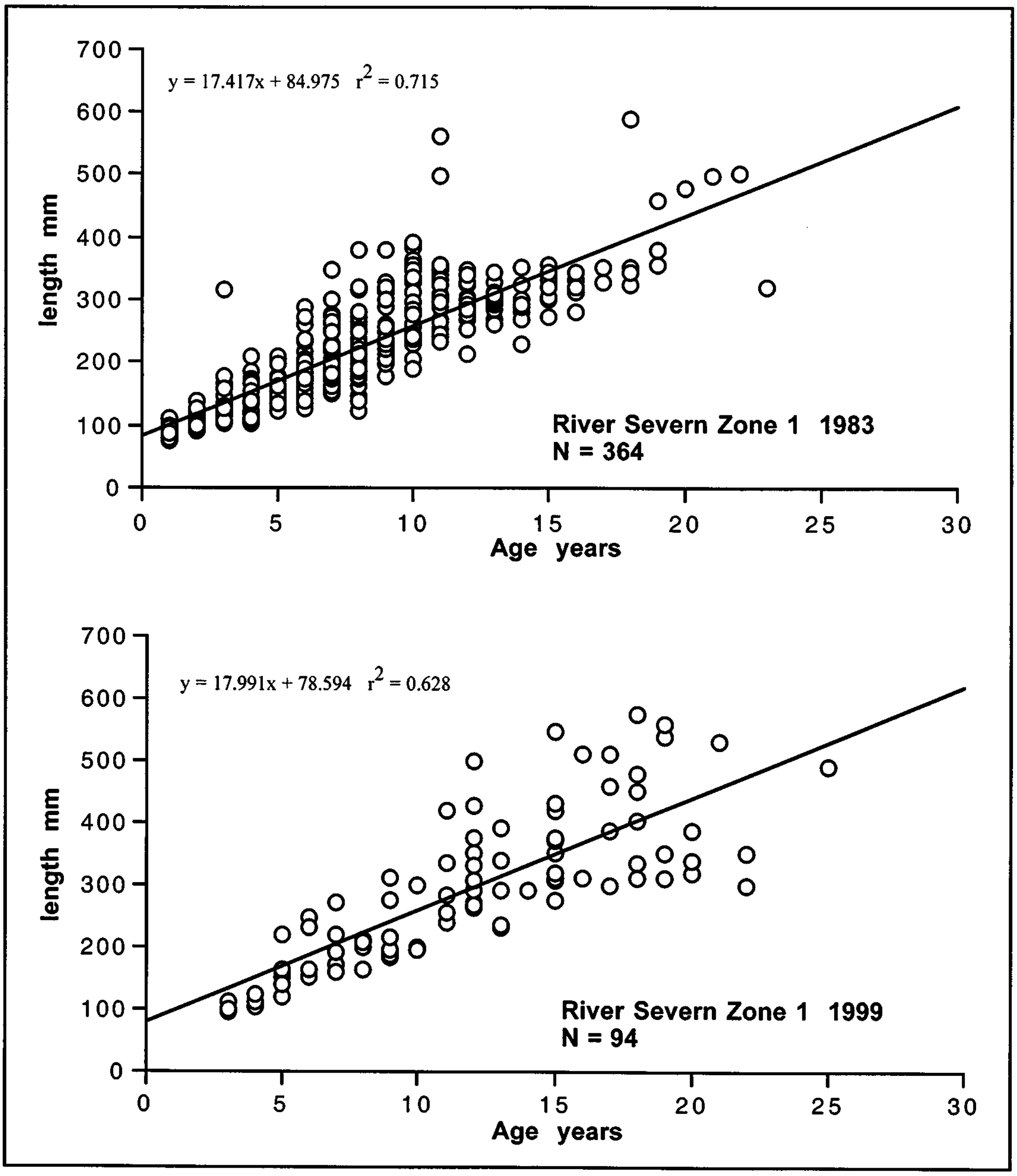


Figure 3.17 Growth rate and age for eels captured in the River Severn Zone 1 in the 1983/84 and 1998/99 eel surveys

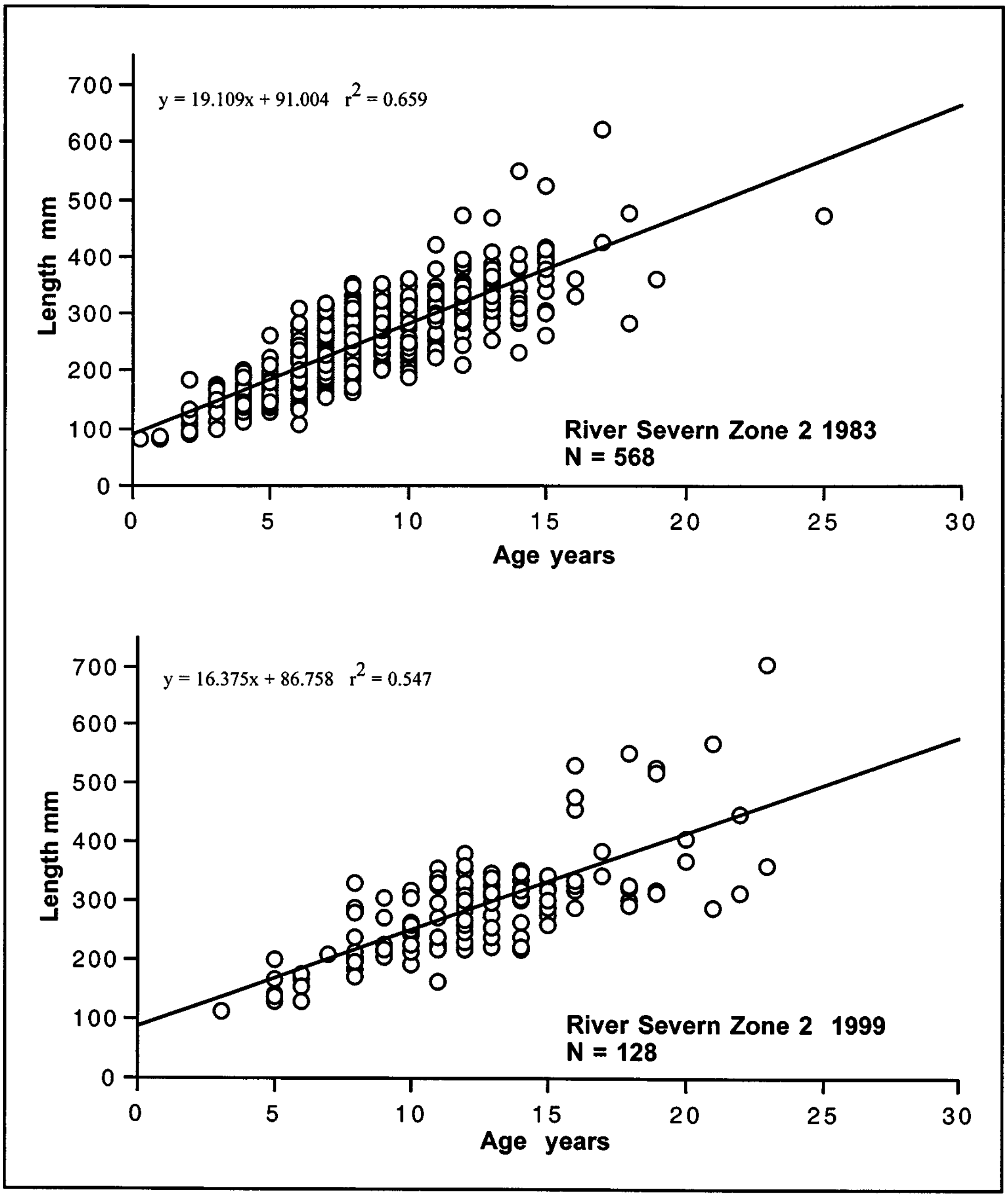


Figure 3.18 Growth rate and age for eels captured in the River Severn Zone 2 in the 1983/84 and 1998/99 eel surveys

Table 3.7 Mean annual growth rate (mm y^{-1}) and standard deviation for eels in Zones 1, 2, and 6+7 of the River Severn in 1983/4 and 1999

	Zone 1	Zone 2	Zones 6+7
1983/4	19.7 ± 7.0	21.9 ± 6.1	25.9 ± 11.8
1999	18.4 ± 5.9	17.8 ± 4.7	20.5 ± 5.2

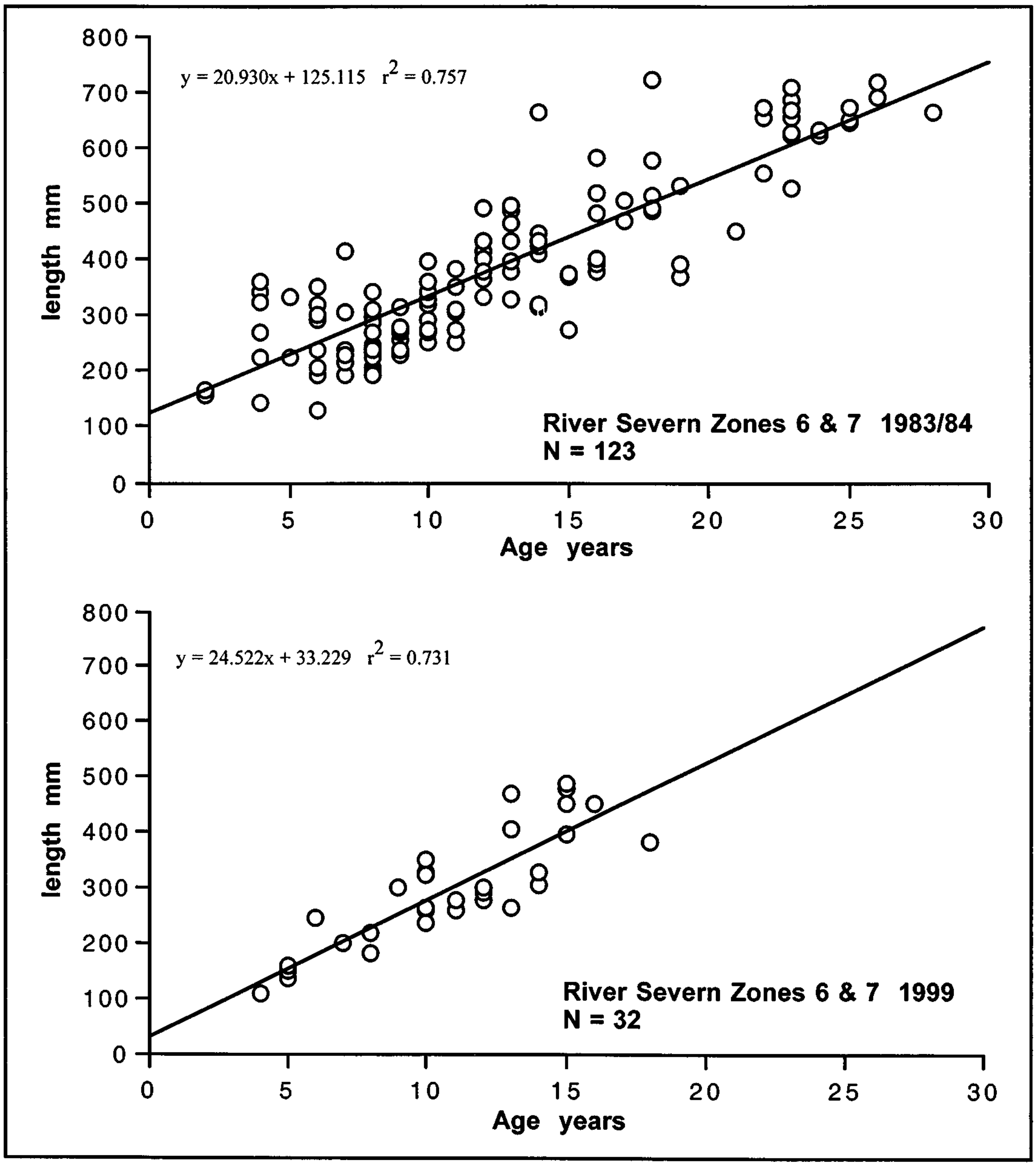


Figure 3.19 Growth rate and age for eels captured in the River Severn Zones 6 and 7 in the 1983/84 and 1998/99 eel surveys

Taken at face value, there appears to have been a slight drop in growth rates between 1983 and 1999. For all three zones (1, 2 and 6+7) the apparent decline in mean growth rate is statistically significant. However, it is important to recognise that there are inherent difficulties in comparing age and growth rate data for eels when datasets have been derived from different workers. These difficulties stem primarily from the inevitable subjectivity in the interpretation of the supernumerary growth bands that frequently occur on eel otoliths.

Thus apparent differences (or similarities) in growth rate may be genuine, but could equally be due to differences in otolith interpretation. This problem should not arise when datasets have been produced by the same workers because any interpretational bias should apply equally to each data set. Thus the implication from Table 3.7 that growth rates in Zones 6+7 were greater than those in Zones 1 and 2 in both 1983 and 1998/99 (Table 3.7)is probably correct. The problems of comparing data sets of eel age are discussed further in Section 3.9.5

in relation to the Piddle and Frome data and in Section 3.11.3..

It is also apparent from Figures 3.17, 3.18 and 3.19 that there is a large variation in the growth rate of individual eels within a population, so that at a given age, length may vary by a factor of two or more. Whilst the use of large sample sizes can overcome this problem in comparing growth rates, large sample sizes do not overcome the inherent risk of bias in ageing. Thus although growth rates determined in 1999 for Zones 1, 2, and 5 of the Severn appear to be generally similar to, but slightly lower than those determined in 1983, a more prudent conclusion would be that there is no evidence that growth rates have changed.

3.7.7 Summary of conclusions relating to the Severn

In spite of its relatively extensive scale, the Severn re-survey programme has provided no substantive evidence for a major change in eel population density or biomass within the Severn catchment. Estimated densities for 1998 and 1999 were generally similar to those for 1983. If there have been changes in overall biomass or density within the catchment between the early 1980s and the late 1990s, then it is likely that these changes are relatively minor.

The data indicate that there has been a significant and marked reduction (circa 50%) in the proportion of small eels <150 mm in Zones 1-3 of the lower Severn population between 1983 and 1998/99. This correlates with the apparent marked pan European decline in glass eel runs since the early 1980s There may have been a corresponding increase in the proportion of large female eels (>450 mm) in the population but statistical support for such an increase is weak.

There appears to be little if any difference in the extent of catchment colonisation between 1983 and the 1991-1995 period.

Whilst there is clear evidence of a reduction in glass eel recruitment to the Severn system since the early 1980s, it would appear that, to date, recruitment has still been adequate to maintain the eel population to at least its 1983 level. However, in view of the potential residence time of eels within the catchment, the effects of reduced recruitment during the late 1980s and 1990s may not yet be fully apparent and may not become so for perhaps a further five years.

The 1998/99 resurvey programme has produced no evidence to suggest that the current levels of glass eel exploitation have any significant impact on eel stocks in the Severn catchment. This is in line with the conclusions of Aprahamian (1986) and White and Knights (1984). Stocks also appear similar to those in the Bristol Avon (Section 2.4) which would also be expected to receive good recruitment but which lacks a significant glass eel fishery.

Potentially one of the most significant observations from the River Severn resurvey programme is the relatively large (circa 30%) apparent difference in stock density and biomass between 1998 and 1999. Although statistically significant, it is unlikely that the observed decline represents a genuine underlying change in eel stocks within the overall catchment. The most likely explanation is that the observed decline resulted from differences in eel distribution within the catchment between the two summers, possibly attributable to differences in temperature regime. Whatever the underlying cause, there are clearly important implications for the design, execution and interpretation of any future monitoring programme for eels.

3.8 River Dee - Results and Initial Discussion

3.8.1 Density and biomass of the Dee eel population

Eight matched sites were sampled on the Dee in 1984 and 1999 so that direct comparisons between years are possible. Figure 3.20 displays eel population density data for the two survey years and Figure 3.21 comprises a corresponding plot for biomass. As might be expected, there is considerable variation in density and biomass between years at some sites. In particular, at Site 2 on the Worthenbury Brook, an extraordinarily high density of 384 eels 100m^{-2} occurred in 1984. This density is well in excess of any other density recorded for the historic or R&D surveys on any of the study rivers.

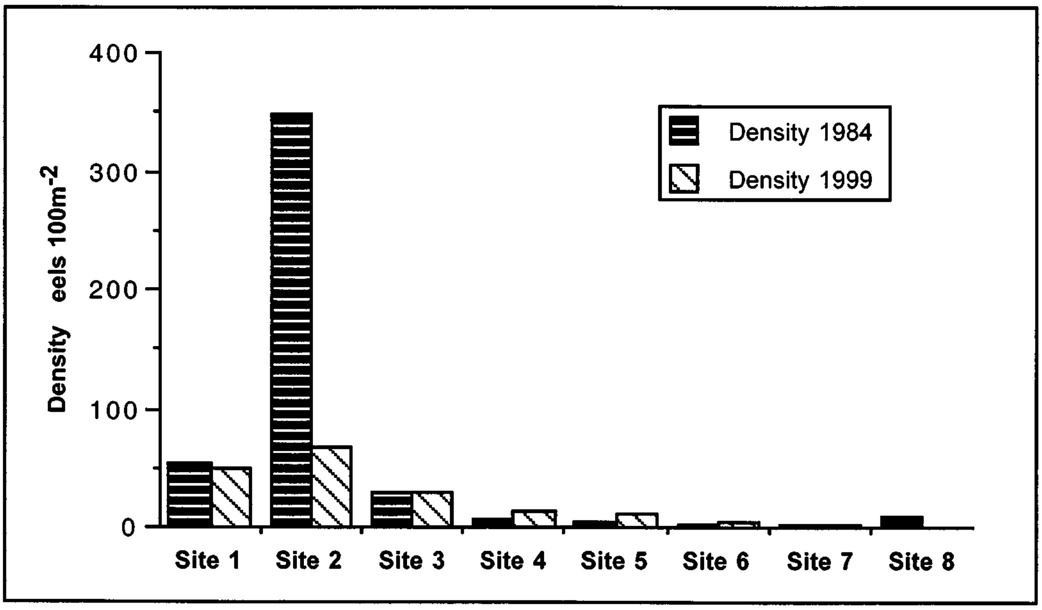


Figure 3.20 Eel population density at survey sites on the River Dee in 1984 and 1999

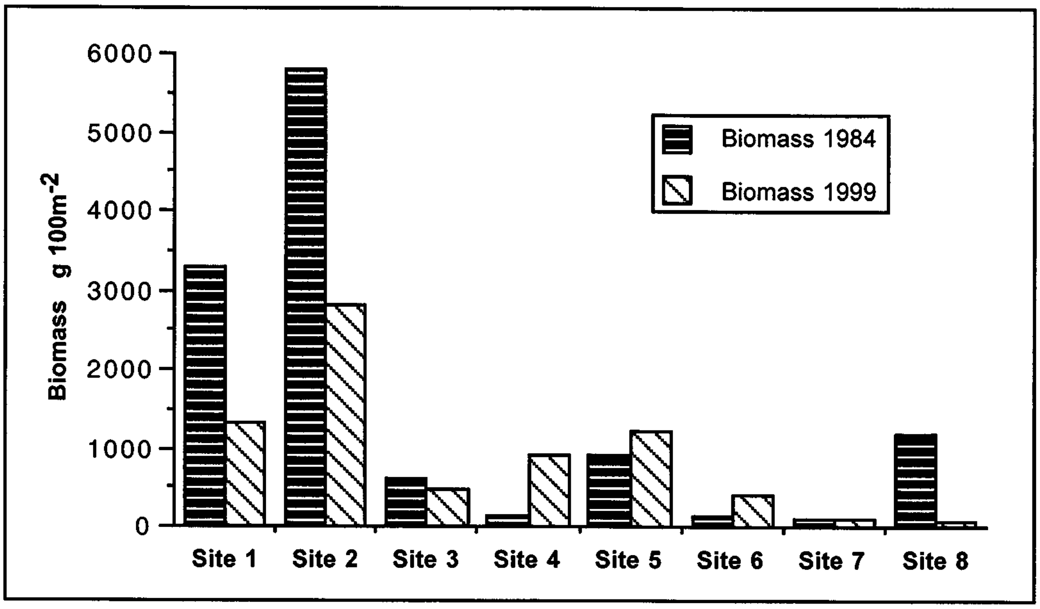


Figure 3.21 Eel biomass at survey sites on the River Dee in 1984 and 1999

Ignoring the clearly atypical 1984 Site 2 density, the overall pattern of density and biomass along the river is generally similar for the two years. Paired *t* tests indicate no significant difference in either density or biomass between years but in view of the small number of sites this is unsurprising.

Mean density for the four lower Dee sites in 1999 was 40 eels $100m^{-2}$ and mean biomass was 1386 g $100m^{-2}$. Mean density and biomass for the Lower Severn sites (Zones 1,2,3 & 5) in 1999 was 43.4 eels $100m^{-2}$ and 1319 g $100m^{-2}$ respectively. Thus eel stocks in the two catchments appear to be markedly similar.

Mean density for the four upper Dee sites in 1999 was 4.43 eels $100m^{-2}$ and mean biomass was 453 g $100m^{-2}$. Mean density and biomass for the upper Severn sites (Zones 6 & 7) in 1999 was 0.74 eels $100m^{-2}$ and 54.4 g $100m^{-2}$ respectively. Thus eel stocks in the surveyed upper Severn zones are much poorer than those in the upper Dee, despite the greater distance from the tidal limit and higher elevation of the Dee sites. Whilst the difference in eel stock level could be related to the effects of the Severn glass eel fishery, it would seem more likely that it is a reflection of the number and types of migration barriers in the Severn system.

3.8.2 Size structure of the Dee eel population

Figure 3.22 displays individual length frequency plots for all eight sites for each survey year. Substantial variation in the length frequency distribution between the survey years is apparent at individual sites. However, the overall pattern of length frequency distribution is clearly similar for the two years. Smaller eels tend to dominate the lower river (Sites 1 - 4, downstream of Llangollen) and larger eels tend to dominate the upper river (Sites 1-5, upstream of Llangollen). Figure 3.23 displays cumulative length frequency plots for the lower Dee and upper Dee. The cumulative length frequency distribution suggests that there may have been a reduction in the proportion of smaller eels in the lower Dee between 1984 and 1999.

Table 3.8 compares the proportion of eels <150 mm at Sites 1 - 4 and for the lower Dee as a whole. The pattern is not consistent between sites but there is a suggestion that there may have been a small decline in the proportion of small eels in the population. However, the sample size (4 sites) is too small to provide any convincing evidence of temporal change.

Table 3.8 Percentage of eels <150 mm in the lower Dee

Year	Site 1	Site 2	Site 3	Site 4	Average
1984	30.1	37.7	42.8	31.7	35.6
1999	19.7	37.5	59.7	2.2	29.6

Table 3.9 compares the proportion of large female eels in the population at each of the Dee sites. This is expressed as the percentage of eels >450 mm in the >250 mm component of the population. Although there are substantial variations between survey years at individual site level, there appears to be no overall difference in the proportion of large female eels in the population.

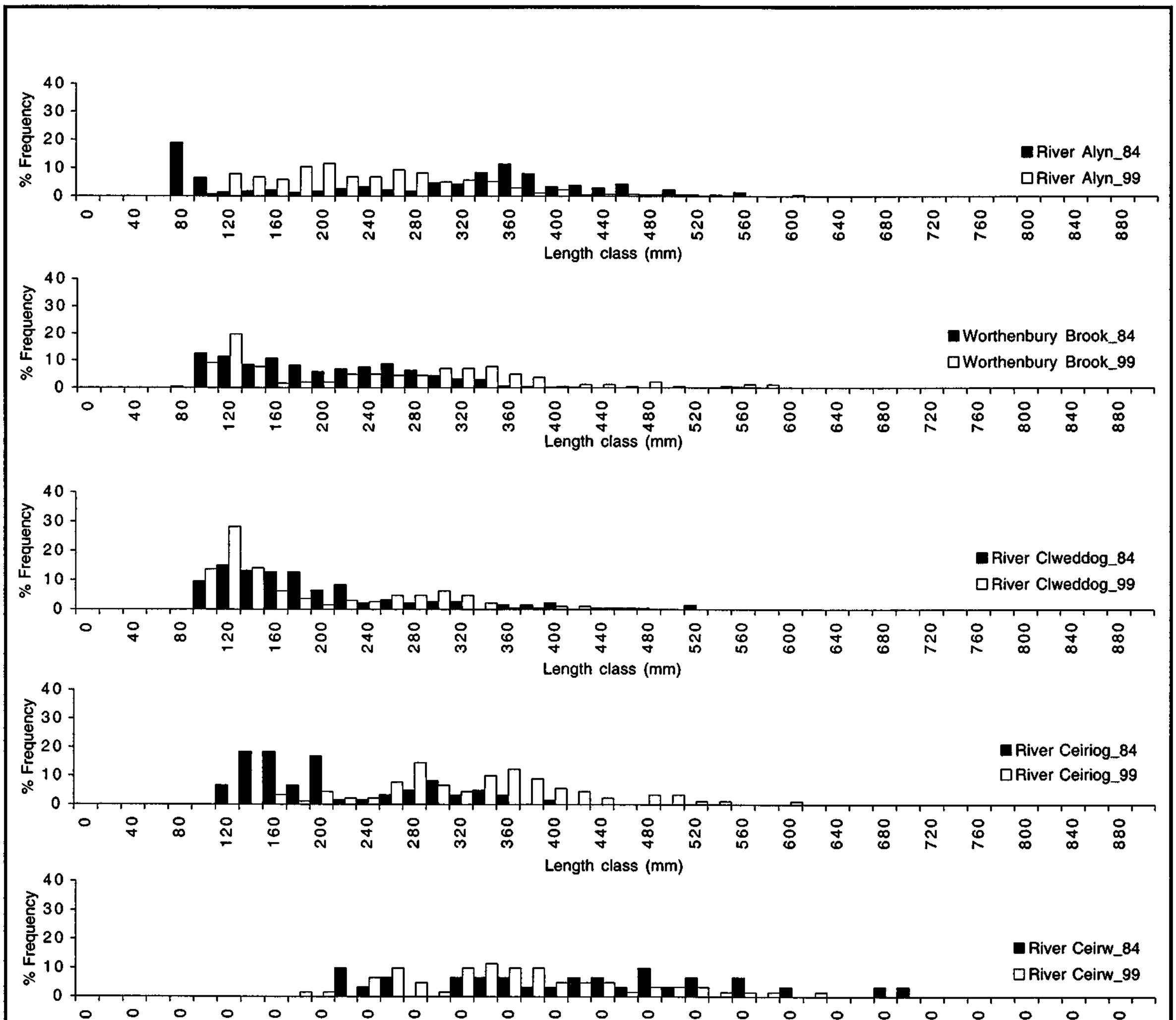
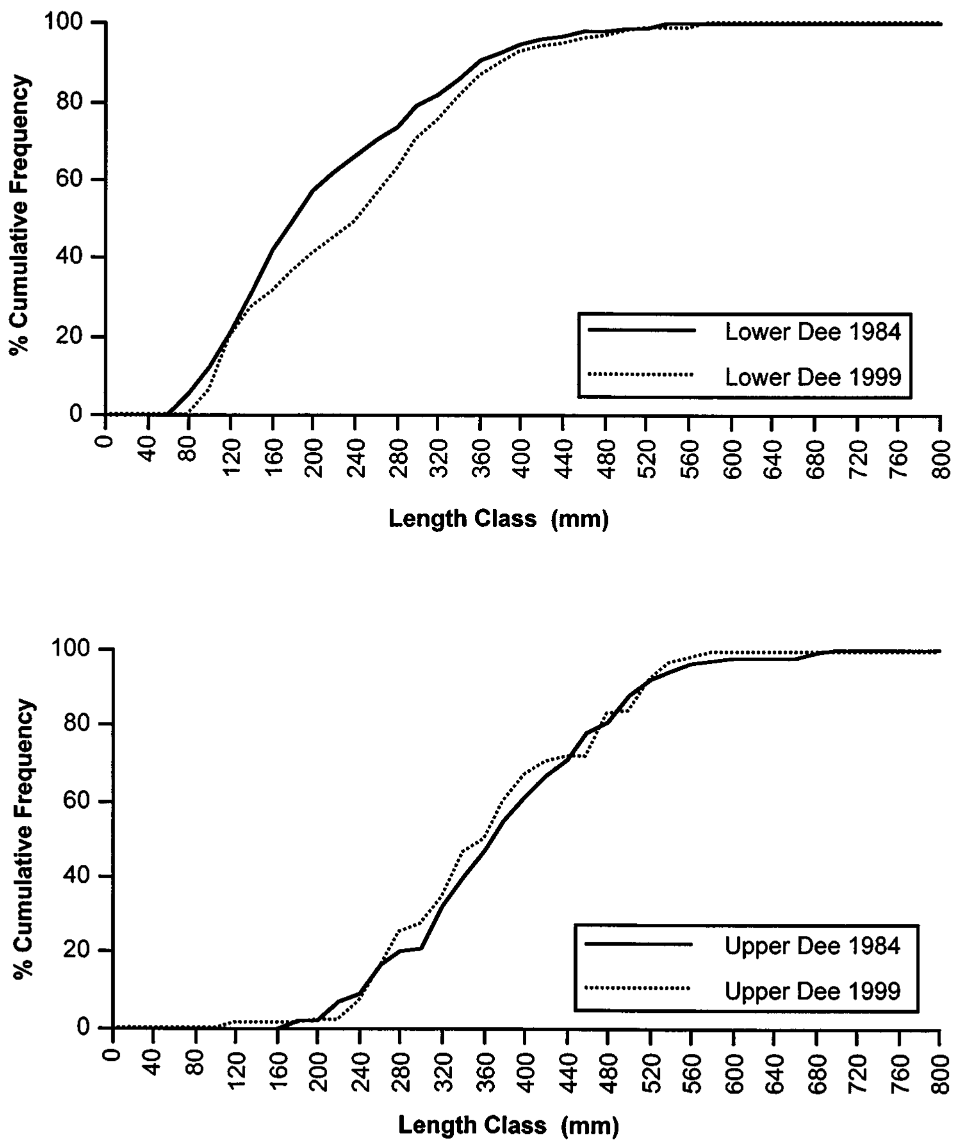


Figure 3.22 Percentage length frequency of River Dee eels in 1984 and 1999



The lower Dee includes the rivers Alyn, Worthenbury, Clweddog and Ceiriog.
The upper Dee includes the rivers Ceirw, Ceidog, Hirnant and Lliw.

Figure 3.23 Cumulative percentage length frequency of River Dee eels in 1984 and 1999

Table 3.9 Percentage of eels > 450 mm in the Dee

Year	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Site 8	Average
1984	11.9	1.2	11.1	0.0	50.0	10.0	14.3	44.0	12.8
1999	4.0	11.6	2.2	11.7	20.8	30.4	11.1	66.7	11.8

Although there are no clear temporal differences in eel population structure between 1984 and 1999, the two data sets show very clear spatial relationships. Figure 3.24 illustrates mean eel length in relation to approximate distance from the tidal weir at Chester and shows the classic picture of increasing average size with distance from the tidal limit. The 1984 and 1999 relationships are markedly similar.

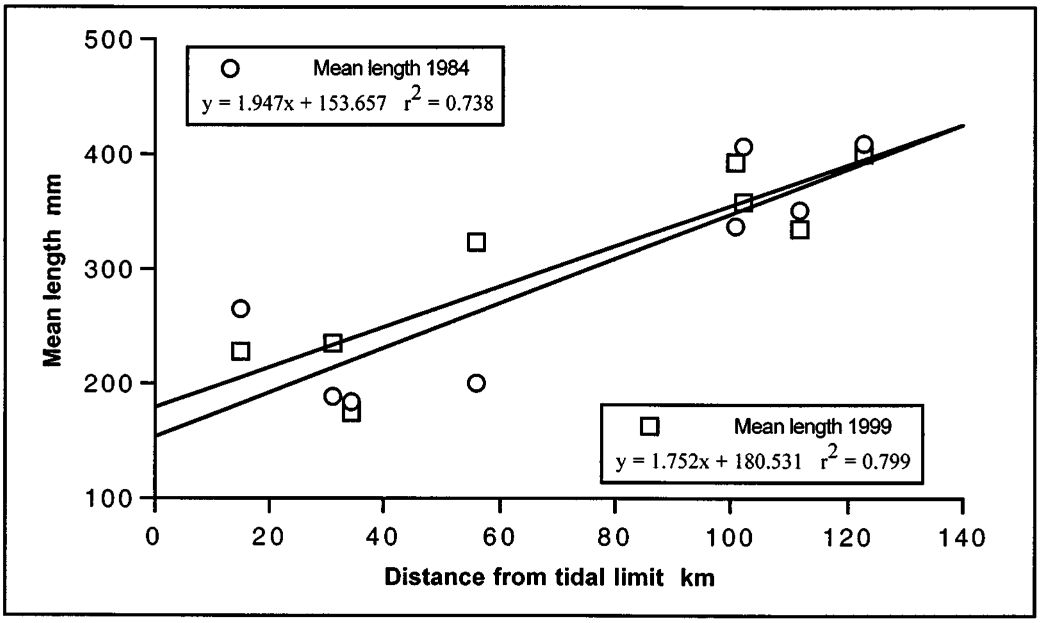


Figure 3.24 Mean eel length in relation to distance from the tidal limit of the Dee in 1984 and 1999

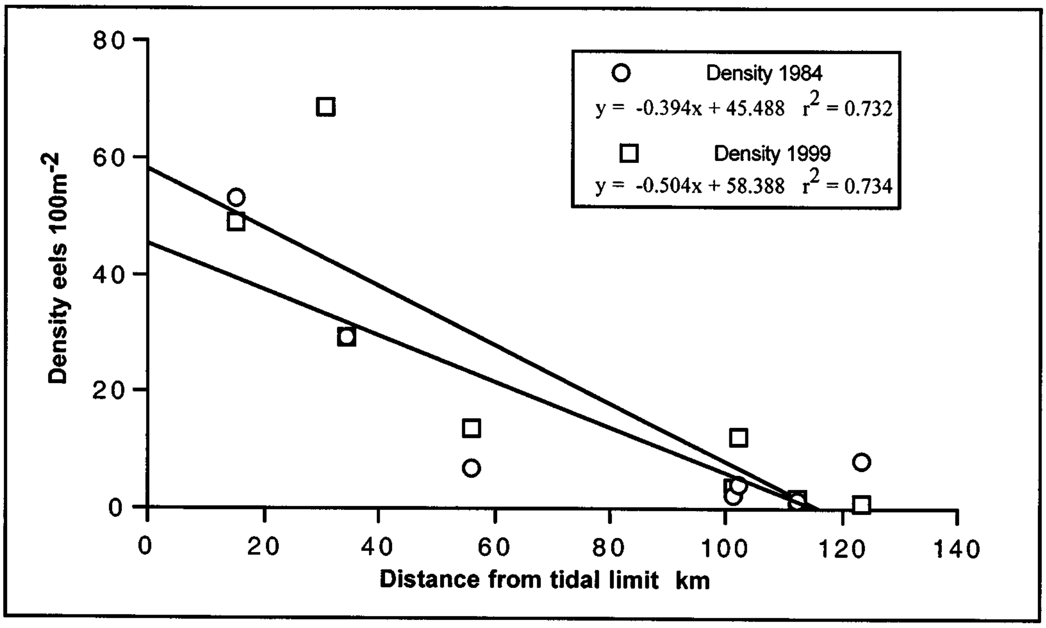


Figure 3.25 Mean density of eels in relation to distance from the tidal limit of the Dee in 1984 and 1999 (Site 2, 1984 excluded)

Figure 3.25 shows the relationship between eel density and approximate distance from the tidal weir at Chester in 1984 and 1999, but with the highly atypical density at site 2 in 1984 excluded. Although individual site densities show marked random variation and the sample size is small, the expected pattern of decreasing population density with increasing distance from the estuary is apparent. The pattern is generally similar for the two survey years.

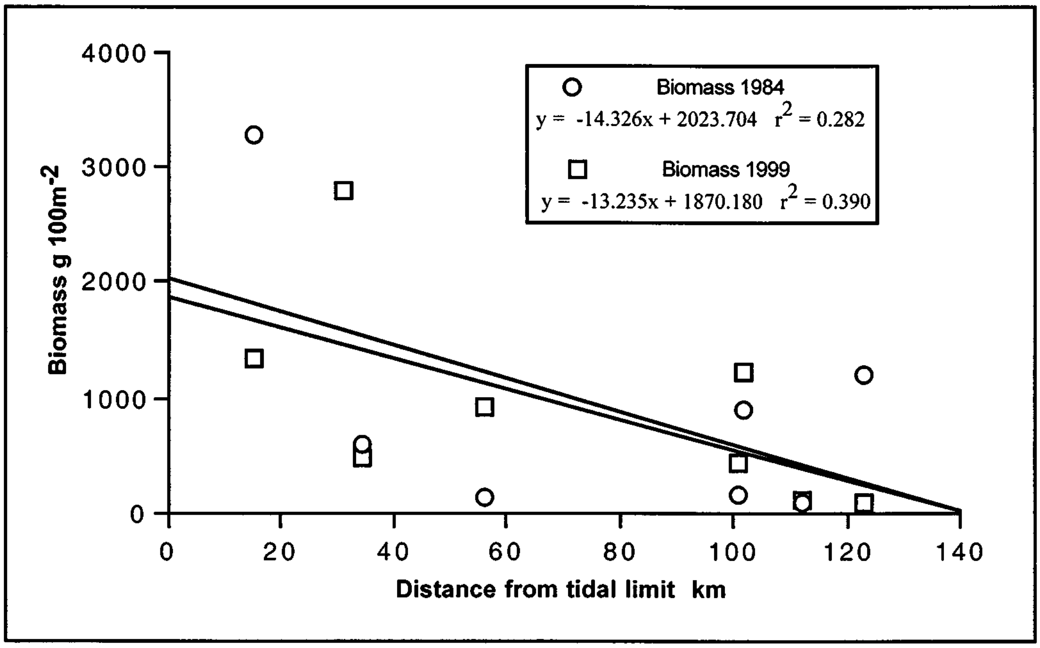


Figure 3.26 Mean biomass of eels in relation to distance from the tidal limit of the Dee in 1984 and 1999

Figure 3.26 shows the relationship between eel biomass and approximate distance from the tidal weir at Chester in 1984 and 1999 but with the highly atypical biomass at site 2 in 1984 excluded. Although individual site biomass shows marked random variation and the sample size is small, the expected pattern of decreasing population biomass with increasing distance from the estuary is apparent. The pattern is markedly similar for the two survey years.

3.8.3 Age and growth rate

Length versus age data together with linear regression lines for eels in the Dee for 1984 and 1999 are displayed graphically in Figure 3.27. Calculated annual growth rates ((length -70 mm) / age) for the 1984 and 1999 surveys as means with standard deviations were as follows:

- 1984 survey 18.46 ± 7.2 mm y^{-1}
- 1999 survey 18.86 ± 4.2 mm y^{-1}

Calculated growth rates differ slightly from those implied by the slope of the regression lines in Figures 3.27, primarily because of the variable position of the intercept on the length axis. There is no indication that mean eel growth rate in the Dee has changed between 1984 and 1999. At approximately 18.5 mm per year, growth rates appear to be similar to those for the Severn.

3.8.4 Summary of conclusions relating to the Dee

The relatively small sample size of eight sites could be expected to detect major changes in eel population density or population structure but a much larger sample size would be required to detect minor changes with any degree of certainty. There is no evidence from the data to indicate major changes in either population density or biomass. There is a suggestion that there may have been a decline in the proportion of small eels of <150 mm, approximately corresponding to the 0+ - 3+ cohorts, within the population between 1984 and 1999. However, this is not statistically significant and the probability that the observed change is simply a random sampling effect is clearly high. If there has been a reduction in the recruitment of small eels to the population of the Dee, then this does not appear to be reflected in the older age classes and there is no apparent change in the relative proportion of large female eels.

The Dee data present a classic picture of eel population distribution with density and biomass decreasing and mean length, age and the proportion of females increasing with increasing distance from the tidal limit.

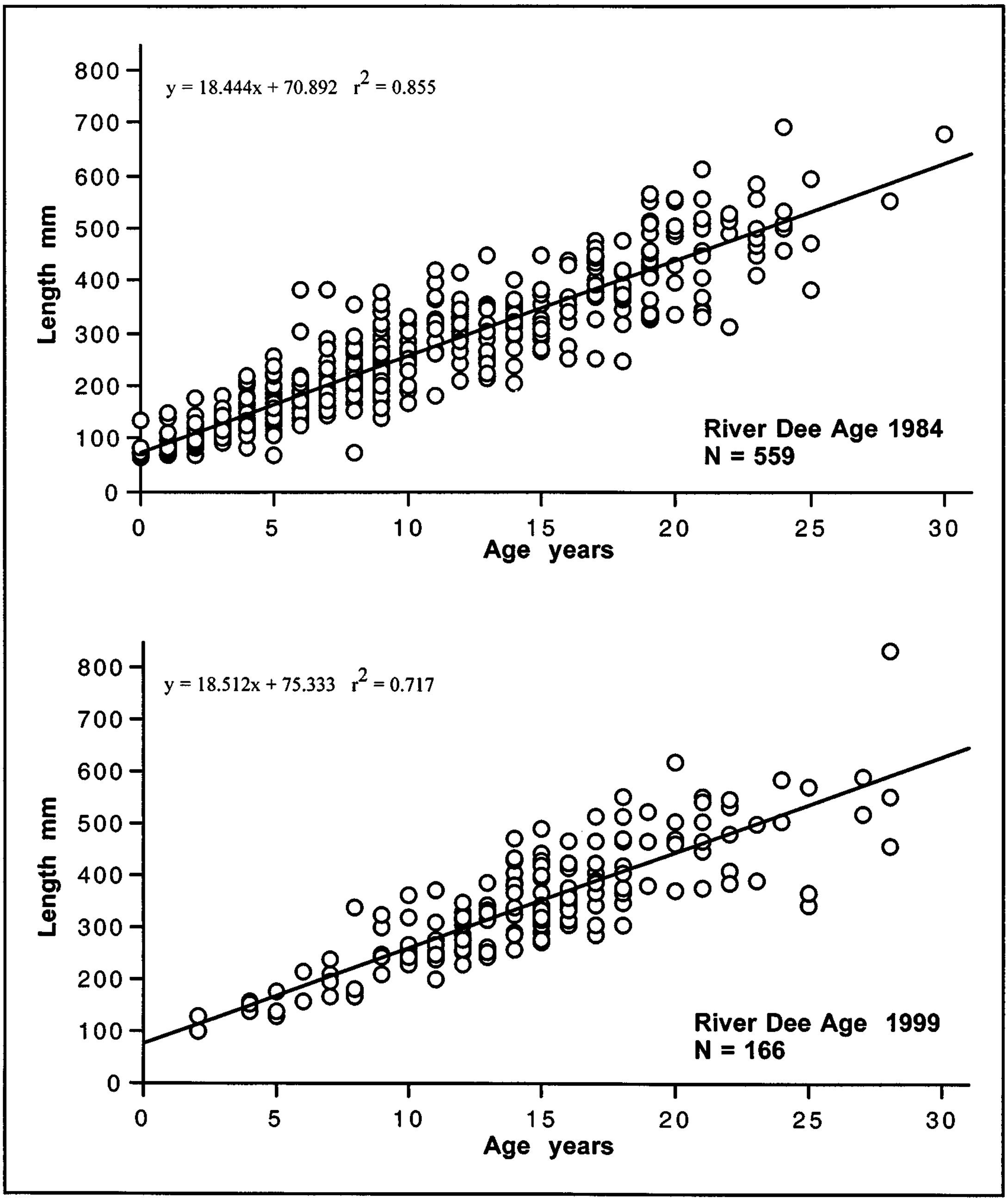


Figure 3.27 Comparison of growth rates for eels captured in the River Dee in 1984 and 1999

3.9 Rivers Piddle and Frome - Results and Initial Discussion

3.9.1 Introduction

The extent of the historic and contemporary data for the Rivers Piddle and Frome potentially allows wide ranging spatial and temporal comparisons to be made. However, this potential is somewhat curtailed by the absence of original data for the historic surveys. For the Tadnoll Brook study (Mann and Blackburn 1991), it has been possible to reconstruct length/frequency, length/age, density and biomass data from the published figures with a substantial level of confidence. Although of small size, the figures are drawn to a high standard and there are no apparent contradictions or anomalies between figures, tables and text.

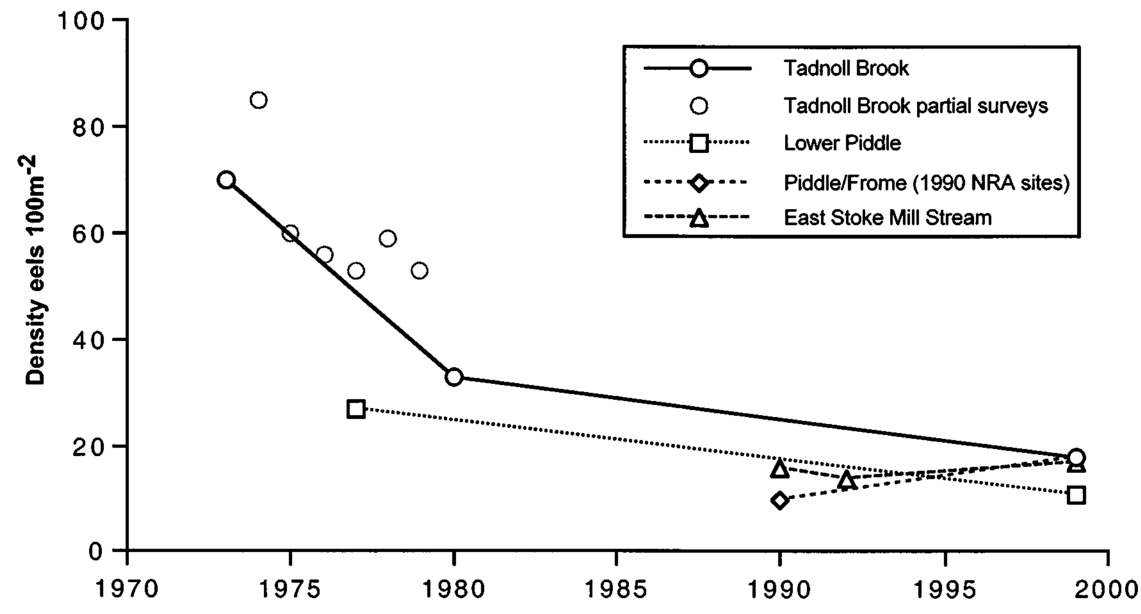
Only limited data are contained in the brief NRA report on the 1990 Piddle and Frome eel survey but interpretation of the figures and tables is very straightforward. Major interpretational difficulties arise with the 1976/77 Piddle study report however. Density, biomass and length frequency data are straightforward and unequivocal. In other cases, e.g. with age and growth, figures are inconsistently or incorrectly labelled and annotated and there are apparent anomalies and contradictions both within the text and between the text and figures or tables. Great caution is therefore required in using information from the draft report. This is unfortunate, given the scale of the MAFF study and its great potential value.

3.9.2 Density and biomass

Figure 3.28 summarises the historic and contemporary eel population density data for the various parts of the Piddle and Frome systems in graphical form. It should be noted that density and biomass data for the Lower Piddle exclude eels of less than 250 mm for both survey years as no information is available for smaller eels from the 1976/77 survey.

There is a relatively strong suggestion from Figure 3.28 that population densities have declined, possibly quite markedly, from the 1970s to the 1990s. Through the 1990s, population densities appear to have been stable or perhaps increased slightly. However, caution must be exercised in interpreting the density data for the Piddle and Frome and it would be unwise to draw more than tentative conclusions.

As indicated in Section 3.2, there are differences in the way in which the various data sets were collected and the extent to which the data are directly comparable is therefore unclear. The lower Piddle (1977) population estimates were based on mark recapture although survey timing for 1977 and 1999 was similar. The Tadnoll Brook (1973, 1980) population estimates were obtained by catch depletion but were based on the entire study stretch rather than discrete sites as used in 1999. Furthermore the historic Tadnoll data relate to March 1973 and August 1980 whereas the 1999 surveys were conducted in June. The 1992 Mill Stream data (Ibbotson *et al* 1994) are based on 5 sites sampled in August, whereas the 1990 (NRA) and 1999 data relate to a single site sampled in August. Because of compatibility doubts, no attempt has been made to quantify the apparent decline in population density, or to assess its statistical significance.



Notes on comparisons:

Tadnoll Brook: 1973 and 1980 densities, for whole (circa 2 km) study area in March (1973) and August (1980) (from Mann and Blackburn 1991).
1999 density, mean for 8 sites within study area in June.

Lower Piddle: 1977 density, mean for 9 reaches determined by mark/recapture May/June (from Morrice *et al.* undated.
1999 density, mean for 9 sites, one within each reach, estimated by catch depletion in June.

Piddle/Frome: 1990 density, mean for 6 sites on Frome and 3 on Piddle in August (from NRA)
1999 density, mean density for same 9 sites in August.

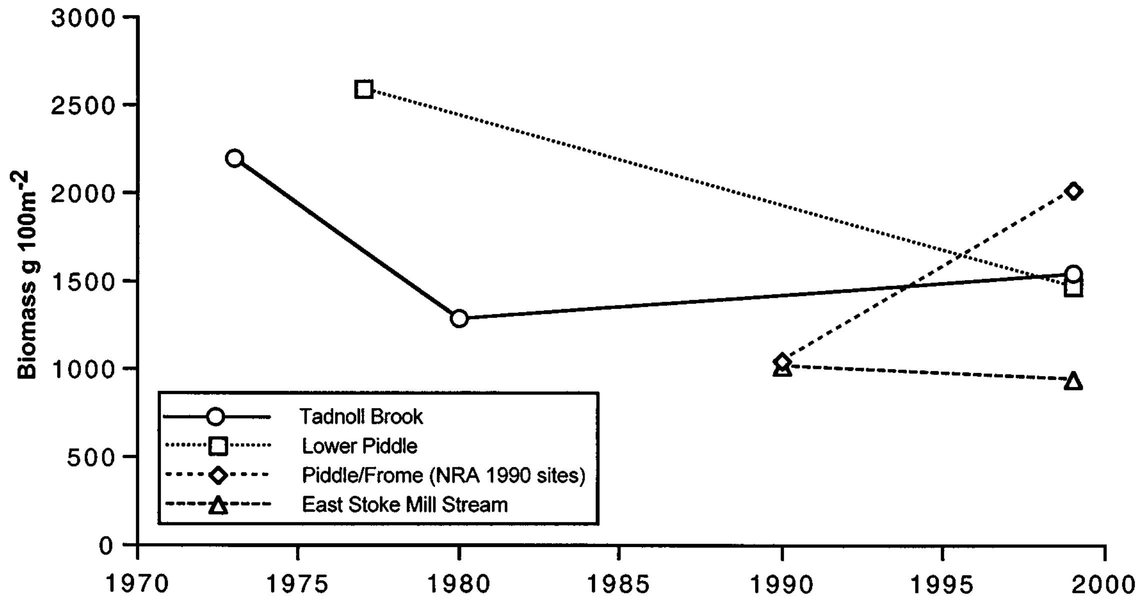
Mill Stream: 1990 density, one site in August (from NRA)
1991/2 density, mean of monthly samples June 91 to February 92 (from Ibbotson *et al.* 1994).
1999 density, one site in August

Figure 3.28 Temporal trends in eel population density for the Piddle and Frome

Figure 3.29 summarises the historic and contemporary eel biomass data for the various parts of the Piddle and Frome systems. The temporal pattern for biomass shows some similarities to that for density in that there is a possible suggestion of decline over the time frame. However, this is much less clear than for density. Thus whilst eel biomass in the Piddle and Tadnoll Brook may have been higher in the 1970s than it is now, any change would appear to be relatively small.

3.9.3 Length frequency distribution

Contemporary and historic length/frequency datasets for the Piddle and Frome are based on large or very large, essentially random, samples of eels obtained by electric fishing. Thus comparisons between datasets should have a more robust statistical basis than comparisons



Note:

See Figure 3.28 for origins of data.

Figure 3.29 Temporal trends in eel population biomass for the Piddle and Frome

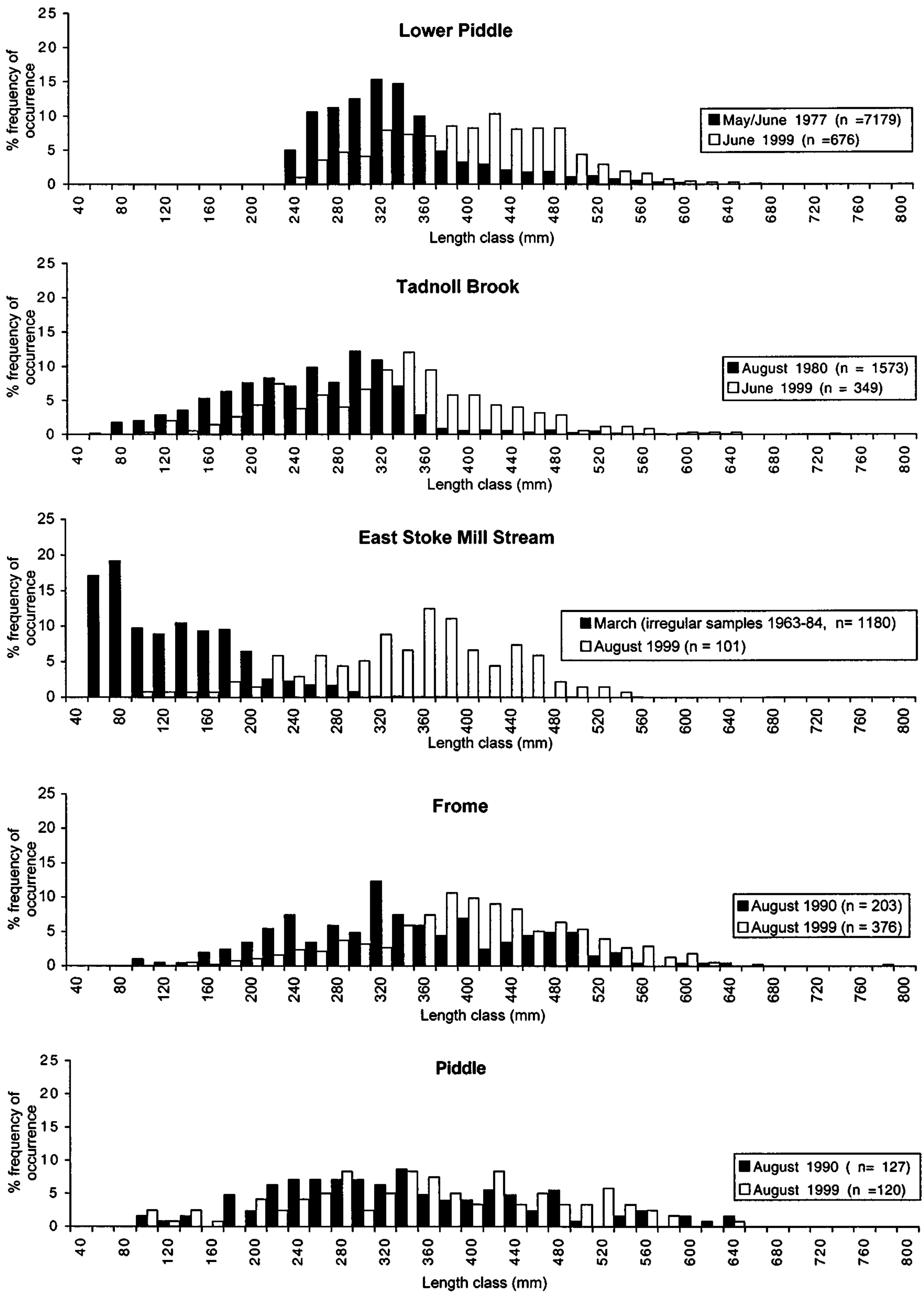
of density and biomass. However, there are some differences in survey timing (see below) which implies that some caution is still required when interpreting apparent temporal trends.

Figure 3.30 comprises length frequency histograms for the various historic and contemporary surveys as follows:

- Lower Piddle - all MAFF reaches 1977 (May/June) and 9 sites 1999 (June);
- Tadnoll Brook - whole FBA study area 1980 (August) and 8 sites within study area 1999 (June);
- East Stoke Mill Stream - composite FBA data collected at irregular intervals between 1963-86 (March) and single site 1999 (August);
- Frome - 5 site NRA survey 1990 (August) and replicate 5 site survey 1999 (August);
- Piddle - 3 site NRA survey 1990 (August) and replicate 3 site survey 1999 (August).

It should also be noted that the lower Piddle data relates only to eels of greater than 250 mm.

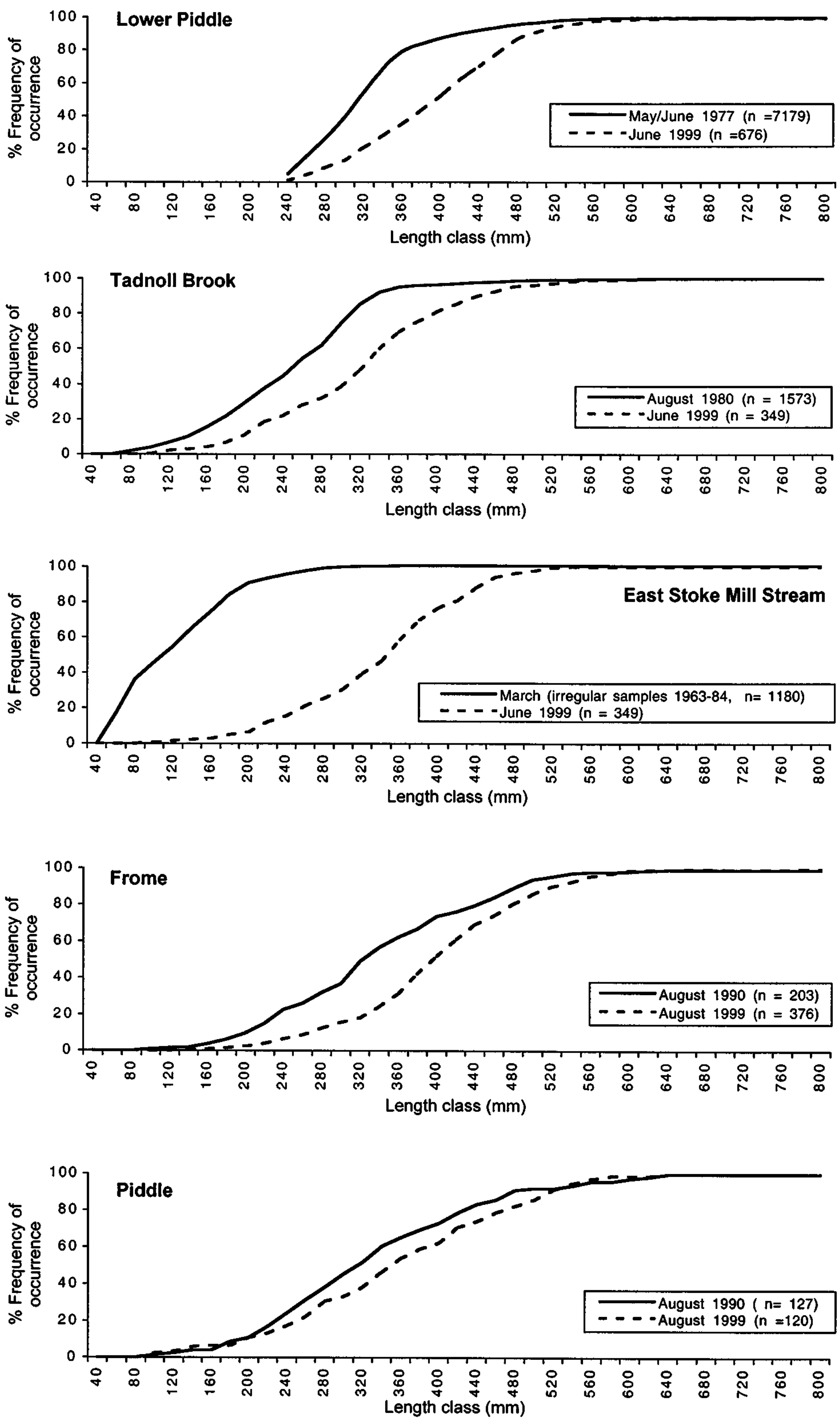
In all cases, the same temporal trend can be seen, i.e. an apparent decrease in the proportion of smaller eels and an apparent increase in the proportion of larger eels. This trend is very clear in the lower Piddle and Tadnoll Brook where the time span is 22 and 19 years respectively. The trend reaches an apparent extreme in East Stoke Mill Stream where the historic data stretches back to the 1960s. Even for the Frome and Piddle catchment overview surveys, where the time span is only 9 years, the trend to increasing size still appears to be present. Figure 3.31 displays the same data plotted as cumulative length frequency. Plotting the data in this simplified manner makes the nature and magnitude of change more readily apparent.



Note:

Lower Piddle data (top graph) relates only to >250 mm eels

Figure 3.30 Length frequency distributions for eels in the Piddle and Frome catchments based on historic and current data

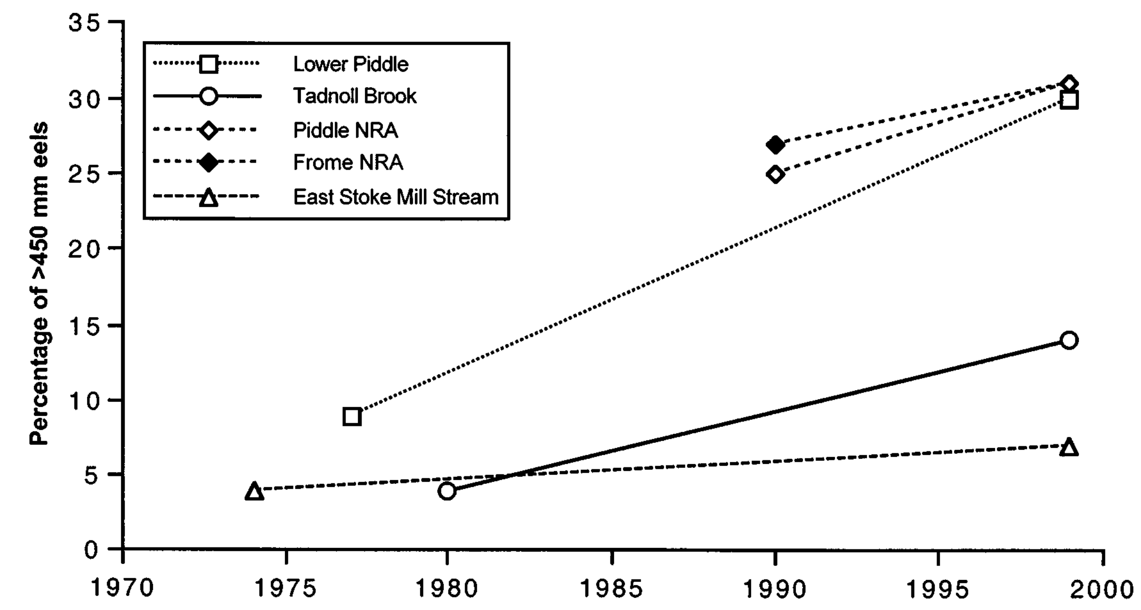


Note:

Lower Piddle data (top graph) relates only to >250 mm eels

Figure 3.31 Cumulative length frequency distributions for eels in the Piddle and Frome catchments based on historic and current data

The aforementioned changes in population structure are potentially important and thus merit further exploration. Figure 3.32 examines the proportion of large female eels in the river population. This figure relates only to eels of 250 mm or longer and it assumes that all eels over 450 mm are female. The justification for this assumption regarding female eels is discussed in Section 3.9.6 below. The lower population cut-off of 250 mm has been used primarily because of the lack of population data for smaller eels for the 1976/77 Piddle study. However, restricting the comparison to the 'semi-mature' component of the population also has the potential benefit of eliminating both the effects of possible changes in the underlying proportion of very small eels and the possible compounding effect of the lower and variable catchability of small eels.



Notes:

i All eels greater that 450 mm are assumed to be female

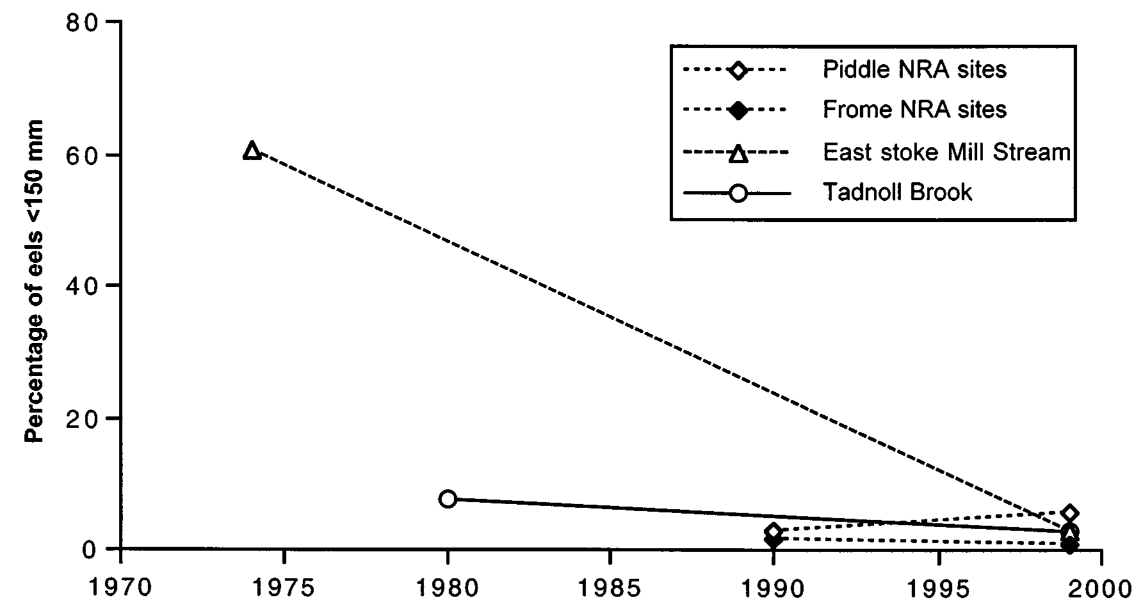
ii Figure 3.32 relates to river eels >250 mm because eels below this size were not recorded for the 1977 lower Piddle survey

iii The East Stoke mill Stream data relates to eels sampled between 1963 and 1986, therefore the median year of 1975 is used.

Figure 3.32 The percentage of large female eels in the non juvenile (>250 mm) river population

Figure 3.32 shows an apparently very clear cut trend to an increasing proportion of large female eels in the population from the 1970s to the present day. This trend is apparent in all of the temporal comparisons on the Piddle and Frome systems. Given the scale of the historic and contemporary surveys and the consequent large sample sizes, e.g. the 1999 catch of eels >250 mm from the Piddle was 676 individuals, there is fairly compelling evidence, (even allowing for potential bias due to mismatch of survey timing for two of the five temporal comparisons) for a real change in population structure. This is discussed further in Section 3.9.4.

Figure 3.33 displays the proportion of juvenile eels, <150 mm in length, in the population, which approximately corresponds to the 0+ to 3+ cohorts. Two points are apparent from this figure. Firstly, the proportion of small eels appears to have declined, possibly quite substantially, between the 1970s and the 1990s. The most convincing support for this conclusion might have come from the lower Piddle but unfortunately no data on small eels are available for the 1976/77 Piddle study. This is particularly frustrating because the comment "all data for eels below 25 cm were discounted" in the draft report (Morrice *et al.* undated) implies that information was probably collected. Secondly, irrespective of the extent to which the proportion of small eels may have declined, it appears to be currently very low in all parts of the Piddle and Frome catchment examined.



Notes:

i Figure 3.33 relates to eels up to 150 mm in length which corresponds approximately to the 0+ - 3+ year cohorts

ii Proportion of <150 mm eels for the lower Piddle in 1999 was approximately 1% - no data is available for eels <250 mm for 1977 for the lower Piddle.

Figure 3.33 The percentage of juvenile eels (<150 mm) in the Piddle and Frome populations

Table 3.10 compares the proportion of <150 mm eels observed in the Piddle/Frome population in 1999 with the proportion observed in the Dee and Severn in 1999.

Table 3.10 The percentage of <150 mm eels in the Piddle/Frome the Dee and Severn in 1999.

Tadnoll Brook	2.9 %	Dee (all sites)	29.6 %
Lower Piddle	0.8 %	Severn (Zones 1,2,3 &5)	16.1 %
Frome (NRA sites)	0.53%		

The relative proportion of small eels in the Piddle/Frome system appears to be about an order of magnitude lower than in the Severn or Dee. For the majority of sites, the distance from tidal limit is within a broadly similar range for the Piddle, Frome, Severn (Zones 1-5) and Dee.

The Tadnoll Brook is approximately 19 km from the tidal limit, while the other Frome sites ranging from 9 - 58 km. The Piddle sites range from approximately 0.5 to 18 km from the tidal limit. It might therefore be expected that substantial numbers of juvenile eels would have migrated to almost all of the Piddle/Frome sites within a year or so of entering freshwater and this is clearly not the case.

Whilst some elvers clearly do still enter the rivers, elvers are still seen in most years on the low weir on the East Stoke Mill Stream (M. Ladle, IFE, personal communication), it is difficult to avoid the conclusion that relatively few elvers (either in absolute terms or relative to the Severn or Dee) have moved into the freshwater part of the Piddle and Frome systems in the last few years. At least two factors may have contributed to this apparent decline and currently low level of recruitment. Firstly, the general (pan European) reduction in glass eel recruitment might be expected to have a relatively greater effect on southern than on western British rivers because of their respective locations relative to the main migration routes. Secondly, the very large expanse of available habitat within Poole Harbour may have soaked up an increasingly high proportion of potential river immigrants.

3.9.4 Eel population characteristics in relation to distance from the tidal limit in the Piddle and Frome.

In their report on the 1976/77 eel study of the Piddle, Morrice *et al.* (undated) make the following statement. "The mean length of eels over 25 cm appears to be independent of the distance of the sampled section from the river estuary. This does not agree with the widely held view that the abundance of females increases upstream in rivers (Tesch 1977)." Figure 3.34 shows mean eel length for the Piddle sites surveyed in 1999 in relation to distance from the tidal limit at Wareham. Figure 3.35 is a similar plot for the Frome.

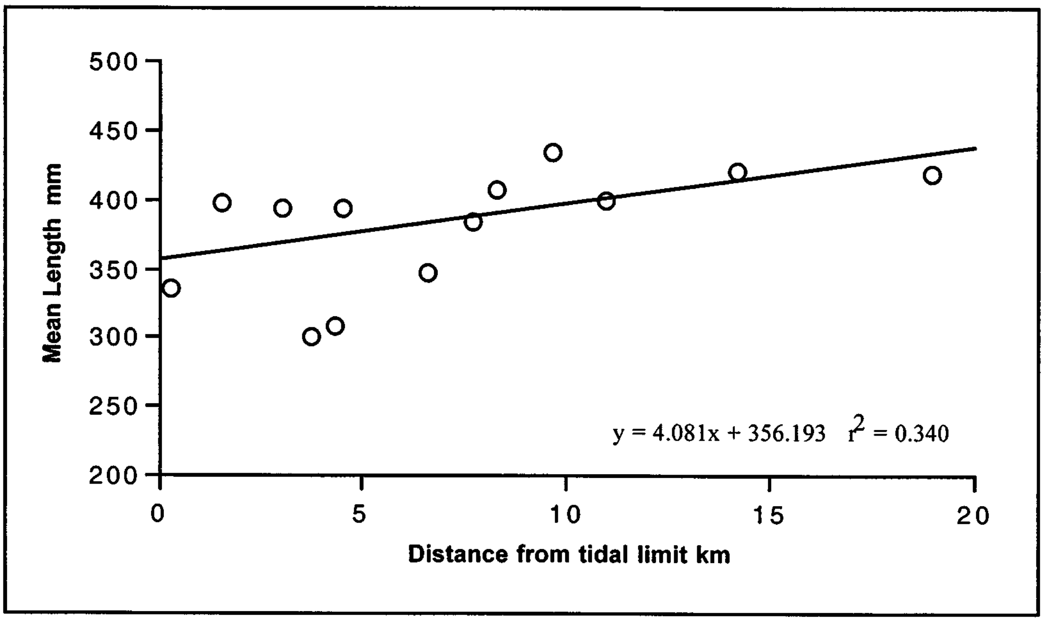


Figure 3.34 Mean eel length in relation to distance from the tidal limit in the Piddle in 1999

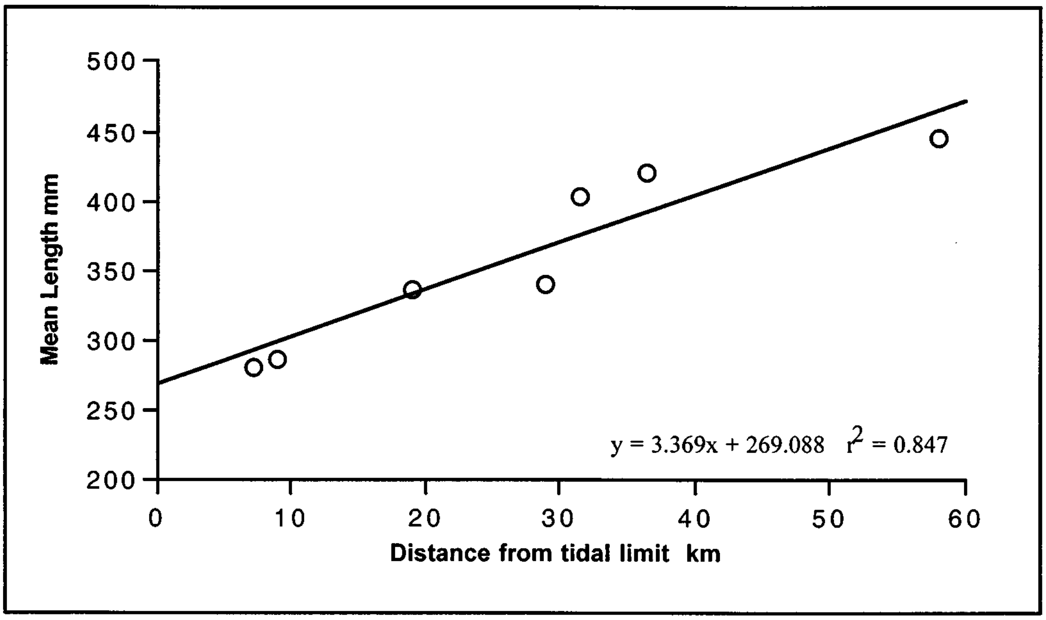


Figure 3.35 Mean length of eels in the River Frome in relation to distance from the tidal limit in 1999

The 1999 Piddle data suggest that, despite the relatively short distance involved, there probably is a slight increase in the mean length of eels with increasing distance from the tidal limit, although this trend is not particularly marked or clear cut. In the case of the Frome, increase in mean size with increasing distance from the estuary is much more readily apparent. However, the length/distance regression appears to be quite similar for the two rivers. The more clear cut relationship for the Frome probably simply reflects the substantially greater maximum distance from the tidal limit (Frome, 58 km; Piddle, 18 km).

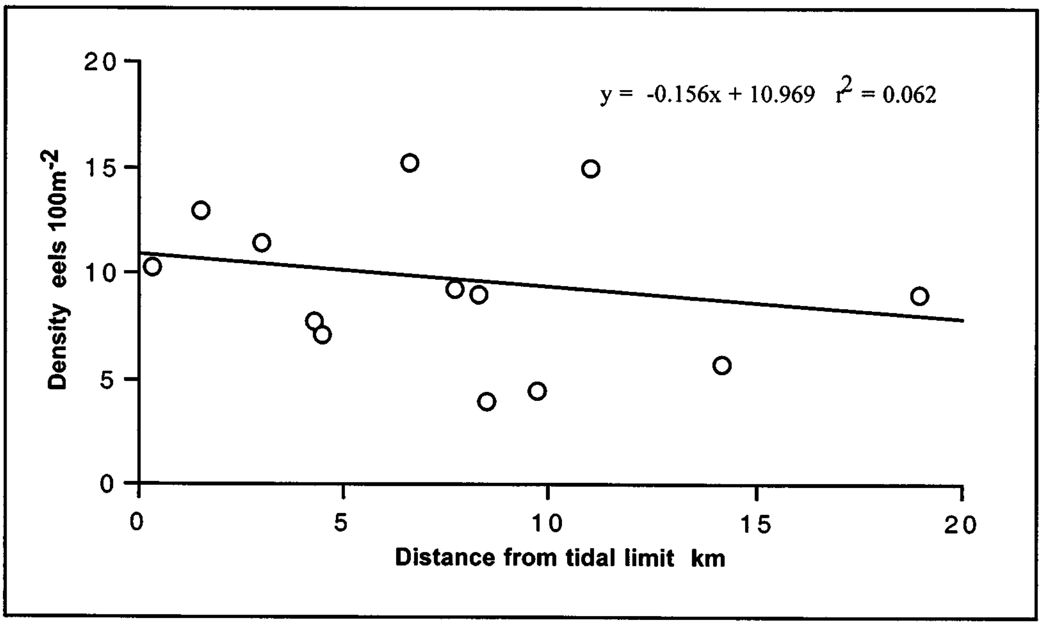


Figure 3.36 Eel population density in the River Piddle in relation to distance from the tidal limit in 1999

Figures 3.36 to 3.39 show eel biomass and density in the Piddle and Frome in relation to distance from the tidal limit. There is no clear suggestion of a relationship between either biomass or density and distance from the tidal limit in either river. Rather, the data suggest that eels may be relatively uniformly distributed through the two catchments. Eel population densities in the Piddle and Frome are relatively low compared to the Severn and Dee (Figures 3.14 and 3.25) at similar distances from the tidal limits but biomasses (Figures 3.15 and 3.26) are broadly similar in all three river systems.

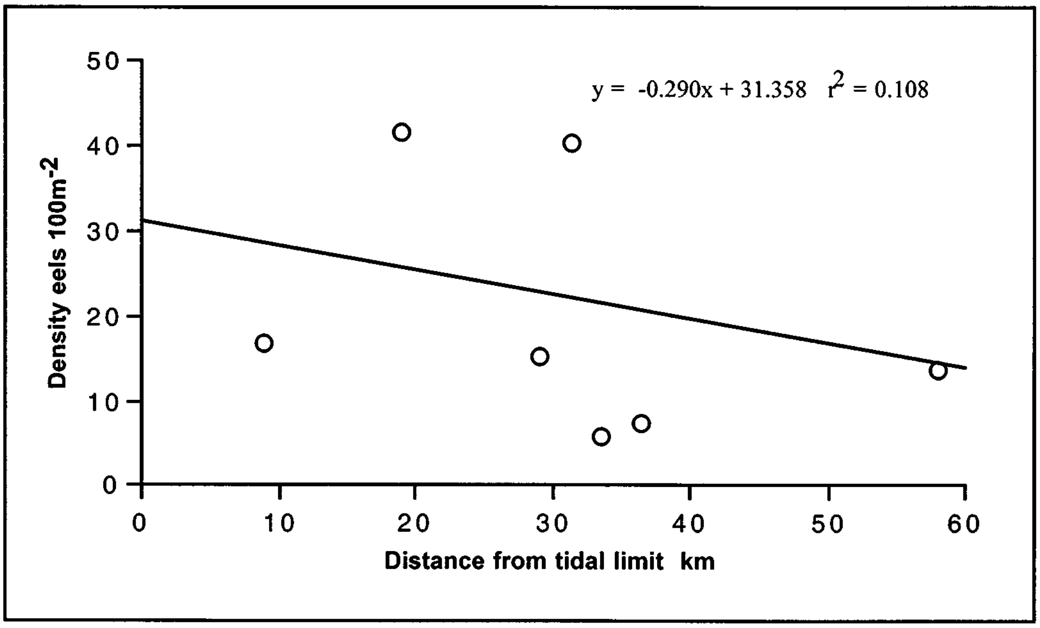


Figure 3.37 Eel population density in the River Frome in relation to distance from the tidal limit in 1999

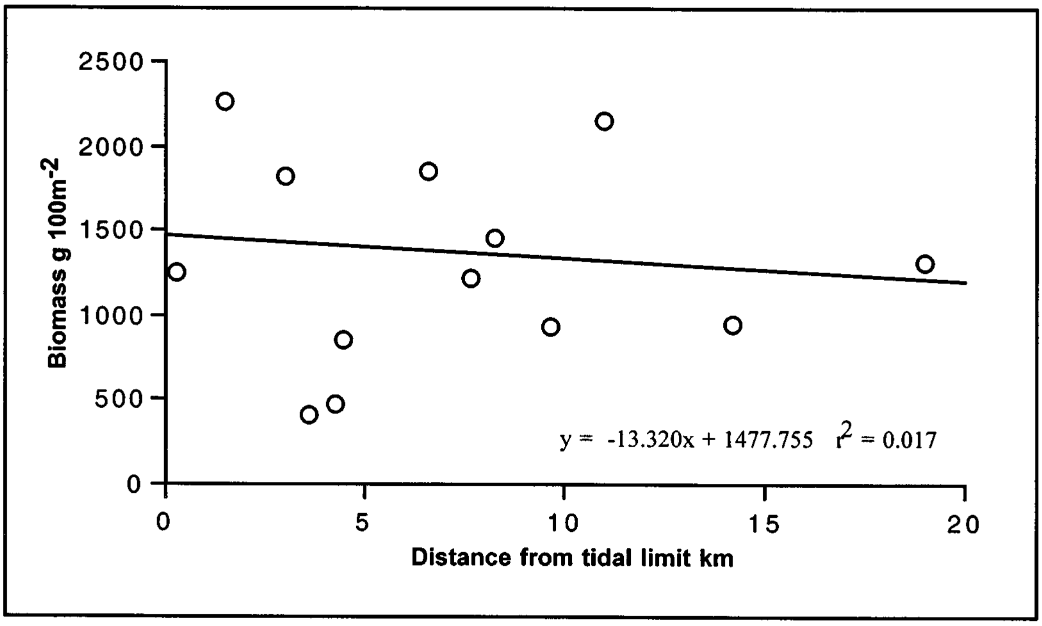


Figure 3.38 Eel biomass in the River Piddle in relation to distance from the tidal limit in 1999

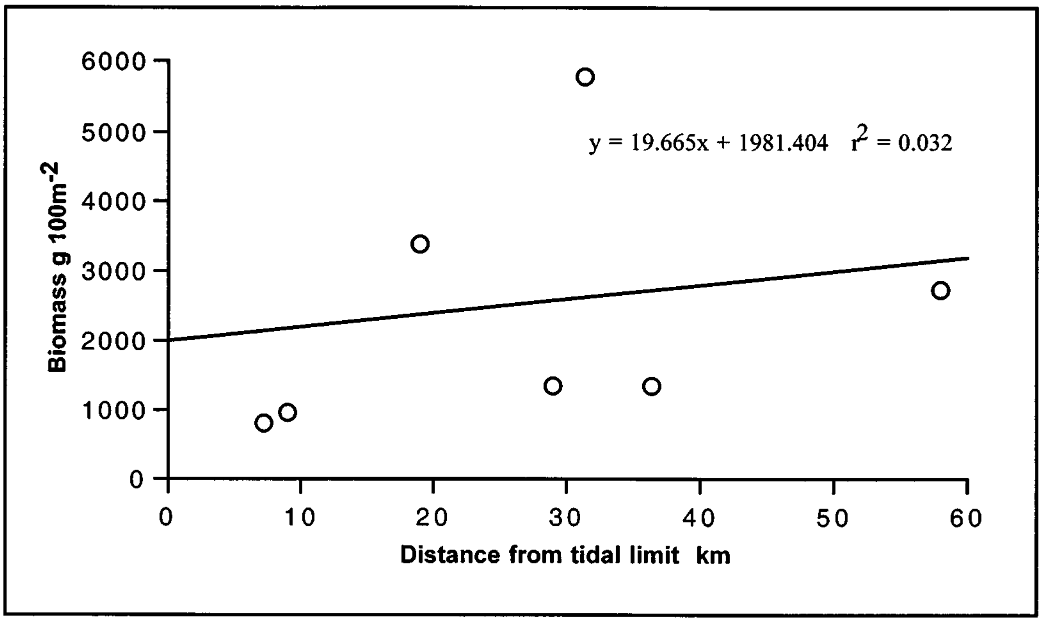


Figure 3.39 Eel biomass in the River Frome in relation to distance from the tidal limit in 1999

3.9.5 Age and growth rate

Historic and contemporary length age data for the Piddle (1976/77 and 1999) and Tadnoll Brook (1982/83 and 1999) are presented as scatter plots with linear regression lines in Figures 3.40 and 3.41 respectively. As was the case for the Severn and Dee, eel growth for the Piddle and Tadnoll Brook appears to fit reasonably well to a linear model and therefore for simplicity, growth rate has been determined by least squares regression.

For both rivers, the historic data have been reconstructed from the reported/published length age histograms. The reported regression for the Piddle data (Morrice et. al., undated) of $y = 26.6x + 123$ closely matches the currently computed regression of $y = 27.58x + 125.5$ for the recreated data, indicating successful data reconstruction.

The Piddle and Tadnoll Brook data provide a clear example of the difficulties that may arise when comparing eel age data from different authors. This arises in part from the method of otolith preparation employed. The results obtained by different preparation methods may vary considerably (Vollestad *et al.* 1988) and it has been reported by various authors e.g. Vollestad (1985), that the burning method, used for the current study, yields slightly greater ages (i.e. slightly lower growth rates) than the clearing method, employed for the historic Piddle and Tadnoll studies. Possibly more important is the frequently reported problem (e.g. Knights & White 1997)) of apparently supernumerary zones in eel otoliths, especially from older eels, and the consequent subjectivity in their interpretation.

There is thus a very real possibility that one group of workers will consistently over or underestimate eel ages relative to another group. The fact that few fish scientists have more than relatively limited experience of eel ageing makes differences in interpretation even more likely. Whilst the use of the Environment Agency's Brampton Fisheries Laboratory for quality control will have helped to ensure consistency for the 1999 eel ageing (and also hopefully for any future eel ageing), it does not eliminate the potential comparability problem with historic data. Caution is therefore required in assessing, interpreting and comparing the historic and contemporary eel age data for the Piddle and Frome systems. However, comparing different datasets produced by a single team should be straightforward, because any inherent bias resulting from interpretation or preparation method is likely to be more or less constant.

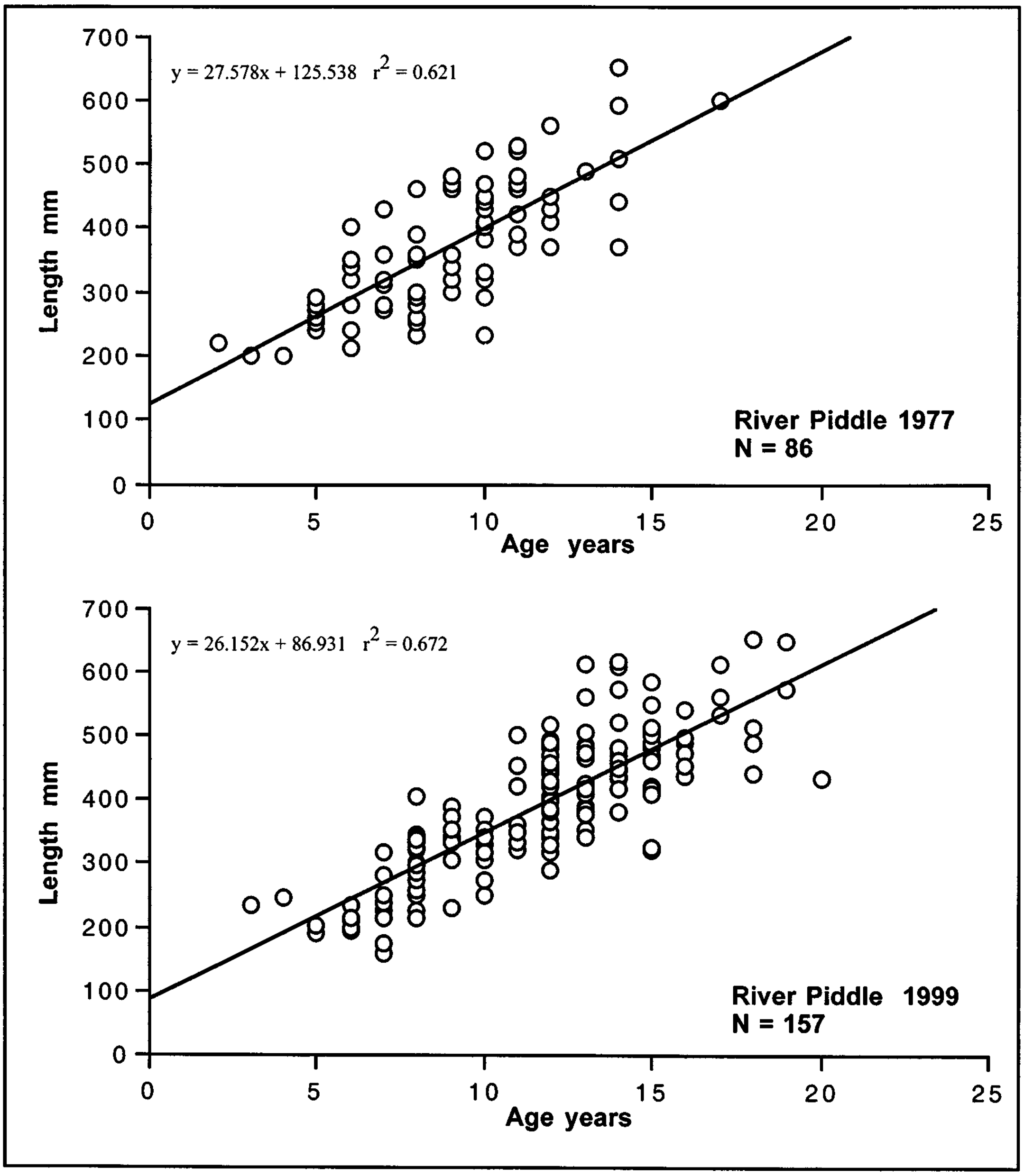


Figure 3.40 Historic (1976/77) and contemporary (1999) eel growth rates in the River Piddle

Table 3.11 shows calculated annual growth rates ((length -70 mm) / age) for the Piddle and Tadnoll Brook as means and standard deviations. Calculated growth rates differ from those implied by the slope of the regression lines in Figures 3.40 and 3.41, because of the variable position of the intercept on the length axis. The historic Tadnoll Brook data (Figure 3.41) indicates a mean annual growth rate of 27.3 mm per year. The very large sample size (N = 687), the range of ages and the intercept of 72 mm (close to the typical size of elvers entering freshwater) suggests reliability in the data.

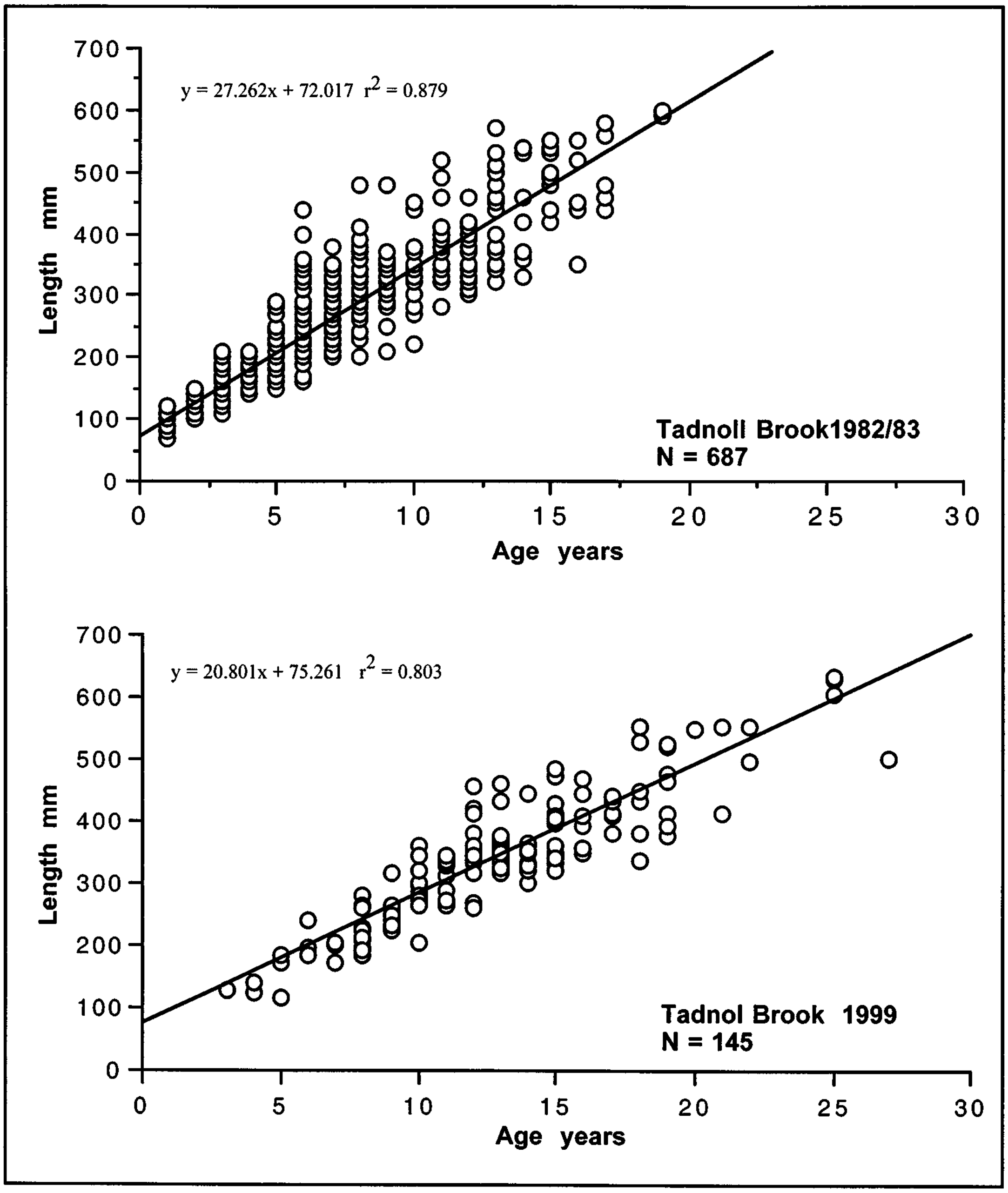


Figure 3.41 **Historic (1982/83) and contemporary (1999) eel growth rates in the Tadnoll Brook**

Table 3.11 **Mean annual growth rate (mm y^{-1}) and standard deviation for eels in the Piddle (1976/77 & 1999) and Tadnoll Brook (1982/83 &1999)**

	River Piddle		Tadnoll Brook	
Historic	35.3 ± 7.2	(1976/7)	27.6 ± 6.7	(1982/3)
Current	27.7 ± 6.2	(1999)	21.1 ± 2.9	(1999)

The slope of 27.6 for the historic Piddle data appears to imply, at least superficially, a virtually identical growth rate to the Tadnoll Brook. Closer examination of the Piddle length/age data however, suggests that, apart from the relatively small sample size (N = 86), there are two potential problems with the data. Firstly, the larger eels, those in the 400 - 650 mm length range, appear to be relatively young, perhaps by 2 - 4 years, compared to those in other data sets for river eels presented in this report, or published in the literature, e.g. Svedang et al (1996). It is also apparent from Figure 3.40 that no Piddle eels shorter than about 200 mm were aged, i.e. only the fastest growing individuals from the younger age classes would have been aged. Although the reported regression for the historic Piddle age data is given as y = 26.6x + 123, Morrice *et al.* (undated) state that the mean annual growth rate for yellow eels above 200 mm was 34 mm per annum. This figure was presumably derived by back calculation to 70 mm at age 0, although this is not stated. Recomputing mean annual growth rate from the reconstructed Piddle data (Table 3.11) yields a figure of 35.3 ± 7.2 mm y^{-1}.

Mean growth rate in 1999 as indicated by linear regression is 26.2 mm per year (27.7 ± 6.2 (length -70 mm) / age)) for the Piddle and 20.8 mm per year (21.1 ± 2.9) for the Tadnoll Brook. These growth rates are therefore somewhat lower than the historically determined annual growth rates of 35.3 mm per year for the Piddle and 27.3 mm per year for the Tadnoll Brook and the temporal differences are statistically significant.

The most likely explanation for the apparently reduced growth rates in 1999 is that they simply reflect differences in otolith preparation method and the relative interpretational bias of the different groups of agers, as discussed above. There are no obvious reasons for believing that growth rates would have been greater in the late 1970s and early 1980s than they are now. If anything, the reverse trend in growth rate would appear to be more plausible, given that population density appears to have declined. Moriarty (1973) provided some evidence for a negative relationship between eel growth rate and population size, although confirmatory evidence from later studies, e.g. Aprahamian (2000), is generally lacking.

The apparently greater eel growth rate in the Piddle than the Tadnoll Brook (Frome) is statistically significant. In contrast, measured growth rates of silver eels from the Piddle and Frome traps indicate that there is no significant difference in eel growth rates in the two rivers for either sex (see Table 3.12 and Section 3.9.6). This latter observation is in line with expectation given the inherently similar character and trophic status of the Piddle and Frome. This therefore suggests that eel growth rates in the Tadnoll Brook may not be typical of the Frome as a whole. Although its source is from chalk springs, a significant part of the Tadnoll catchment comprises heathland. It is thus quite feasible that the brook might be less productive and support lower eel growth rates than the main River Frome.

Eel growth rates in the Piddle and Frome are generally higher than those in the Severn and Dee. This is in line with expectation given the respective trophic status of the various rivers. However, eel growth rates in the Dorset rivers are much lower than those reported for eels in Poole Harbour, presumably reflecting the higher productivity of estuarine/marine systems. Growth rates determined by the NRA for a sample of Harbour eels in 1996 were 63.4 ± 22 mm y^{-1} for female (>450 mm) eels and 75.8 ± 35.8 mm y^{-1} for <450 mm eels.

3.9.6 Silver eel migration, sex ratio and growth rate

Length frequency distributions for samples of migrating silver eels collected from the Piddle (257 eels from funnel net at tidal limit in 1976/7 and 173 eels from rack at Trigon on 10/11/99) and the Frome (168 eels from rack at East Burton on 24/10/99) are shown in Figure 3.42. Each of these plots shows a very clear bimodal distribution with a lower frequency maximum around 380 mm, a pronounced minimum in the region of 440-460 mm and upper frequency maximum somewhere in the 480-600 mm length range. The twin peaks almost certainly represent males (lower peak) and females (upper peak) respectively, with most silver eels <450 mm being male and almost all silver eels >450 mm being female. This fits the pattern observed in other rivers as discussed in Chapters 2 and 4. Also, in their study of eels in the Tadnoll Brook (River Frome), Mann and Blackburn (1991) determined the sex by gonad examination of 1126 eels (> 250 mm) culled between 1980 and 1984. Of the 162 eels above 450 mm in length in the sample, all but one were female.

Age data for migrating silver eels from the Piddle in 1976/77 and in 1997 & 1999 are presented in Figure 3.43. Age data for migrating silver eels from the Frome in 1997 & 1999, are presented in Figure 3.44. The scatter plots of length against age show male and female eels separately and thus allow average size, age and hence growth rate to be determined separately for each sex. Growth rates for male and female silver eels for both rivers are summarised in Table 3.12. As discussed in Section 3.9.5 in relation to the historic data on Piddle yellow eels, the spread of determined silver eel ages from 1976/77 suggests an element of under-ageing.

Table 3.12 Growth rates (mm y^{-1}) for male and female silver eels from the Piddle and Frome traps

	Piddle	Frome
Historic	34.5 ± 12.9 (male)	-
1976/77	48.1 ± 12.9 (female)	-
Current	24.4 ± 3.7 (male)	21.2 ± 3.6 (male)
1997/99	27.7 ± 6.2 (female)	28.2 ± 5.6 (female)

Silver eel traps also provide information on the sex ratio of the population, provided that samples are randomly taken and that the traps catch both sexes with equal efficiency. During the 1976/77 Piddle study, eels were trapped throughout the migration season using a funnel net designed to catch males and females. The overall ratio of trapped males : females was 2.8 : 1 in 1976 and 2.7 : 1 in 1977, based on the assumption that eels up to 450 mm were male.

The observed sex ratio of silver eels from the Piddle fits well with the observed length frequency distribution of the river population in 1977. Figure 3.30 shows a sharp fall in the frequency of occurrence between 320-340 mm and about 420 mm with very few eels above 450 mm. The 340-420 length band corresponds to the typical male silver eel size range (see Figure 3.43) and thus suggests that the majority of eels in the population were developing as males.

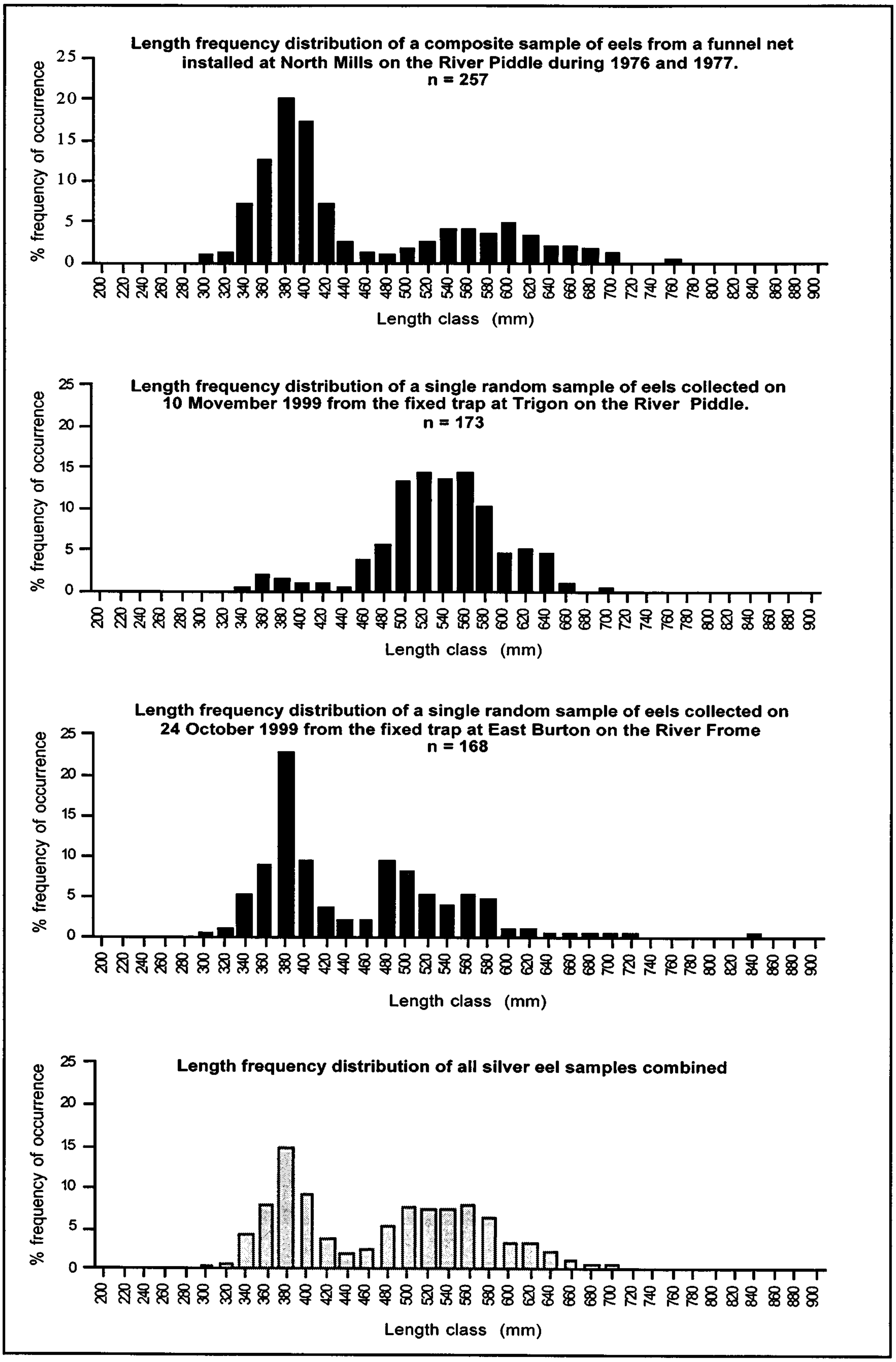


Figure 3.42 Length frequency distributions for samples of migrating silver eels collected from the Piddle (1976/77 and 1999) and the Frome (1999)

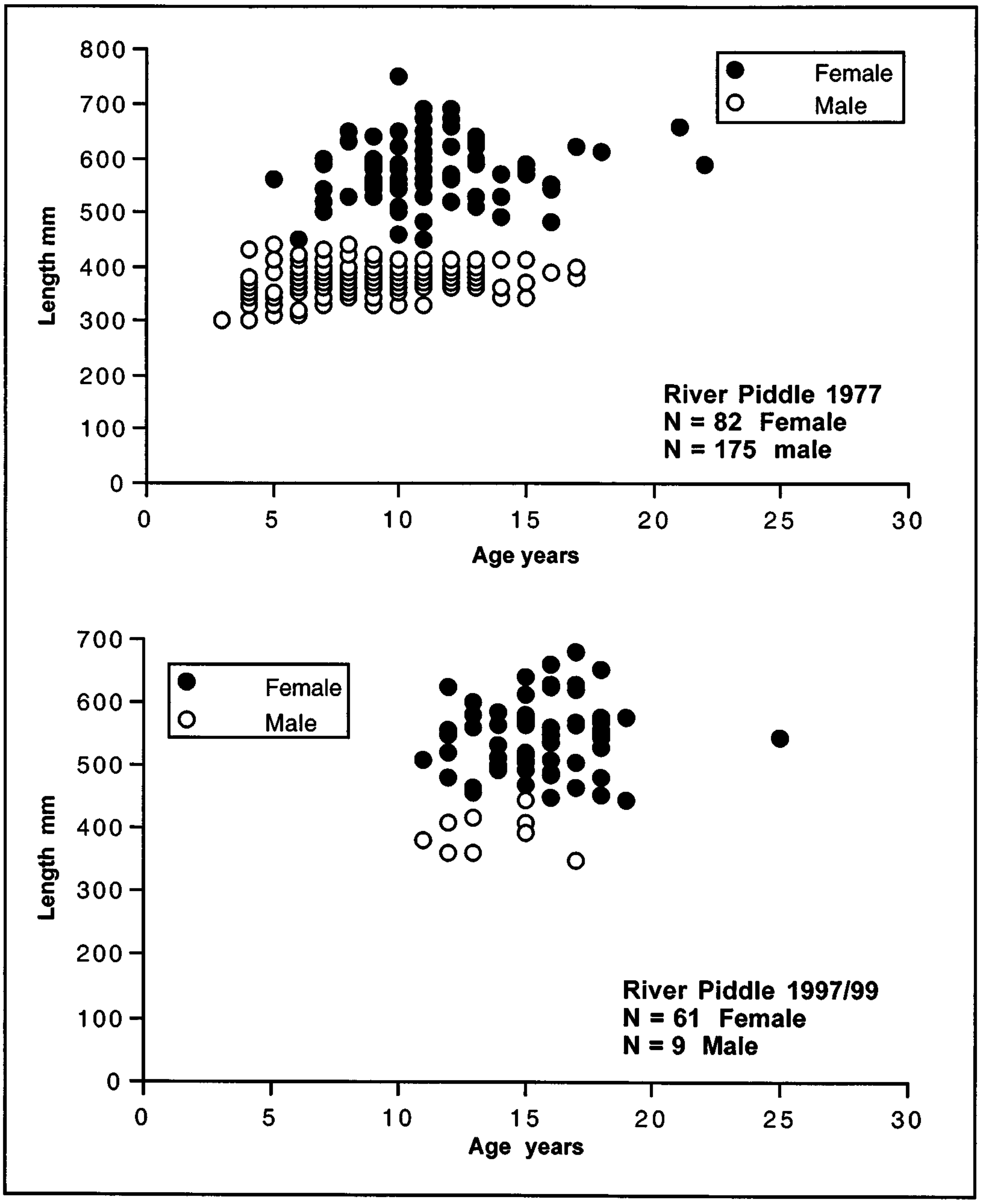


Figure 3.43 Length and age for migrating silver eels from the Piddle (1976/77 and 1997/99.

Recalculated annual growth rates for male and female eels in the Piddle in 1976/77, based on length and age at migration, were 34.5 ± 12.9 mm y^{-1} for males and 48.1 ± 12.9 mm y^{-1} for females. As stated in Section 3.9.4, the recalculated annual growth rate for Piddle yellow eels in 1976/77 was 35.3 ± 7.2 mm y^{-1}. The annual growth rate for all yellow eels is thus considerably closer to that of males than of females, again suggesting a male dominated population. Overall therefore, there is quite convincing evidence for a substantially male dominated eel population in the Piddle during the latter half of the 1970s.

Figure 3.30 shows a very different length frequency distribution for Piddle eels in 1999 to that for 1977, with a near constant frequency of occurrence of eels in the size classes from 320 and 480 mm. There is then a steady decline in frequency of occurrence in the river population from 480 - 650 mm, the typical female silver eel size range. This suggests a female dominated population that would be expected to give rise to only a relatively small male silver eel run. Samples of silver eels obtained from the Trigon rack (some 4 km upstream of the tidal limit) in 1997 (41 eels) and 1999 (173 eels) had male:female ratios of 0.07:1 and 0.08:1 respectively. However, although the Piddle samples were random in their collection, the Trigon trap is almost certainly partially selective for females, at least to some degree, because of the relatively wide spacing of the bars of the rack. A significant, but unknown, proportion of the males undoubtedly escapes. Thus whilst the trap data strongly suggests that the silver eel run is now female dominated, it is unlikely to be female dominated to the extent that the observed sex ratio implies.

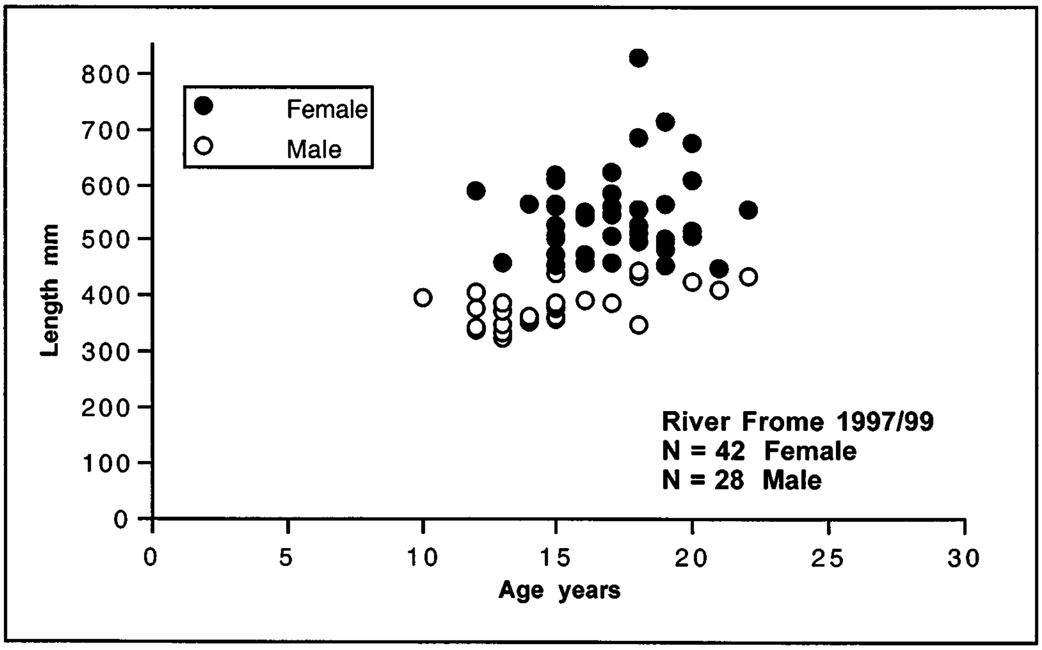


Figure 3.44 Length and age for migrating silver eels from the Frome (1997/99)

For the reasons discussed in Section 3.9.5, it is unrealistic to compare growth rates obtained in 1976/77 with those obtained in 1999. However it is realistic to compare relative growth rates calculated for male and female silver eels from 1997/99 (Figure 3.43) with those for the river yellow eel population 1999 (Figure 3.40). Calculated annual growth rates for male and female eels in the Piddle in 1999 were 24.4 ± 3.7 mm y^{-1} for males and 31.1 ± 6.8 mm y^{-1} for females, based on length and age at migration. The calculated annual growth rate for Piddle yellow eels in 1999 was 27.7 ± 6.2 mm y^{-1}. The relative growth rates of yellow and silver eels are therefore compatible with, and may be suggestive of, a shift towards a more female dominated population in 1999.

For the Piddle, therefore, the putative temporal trends are compatible with one another. Population densities appear have been reduced whereas biomass appears to have been largely maintained (or only slightly reduced), as a result of a shift in sex ratios to a greater proportion of larger female eels.

As is the case with the Piddle, the Frome and Tadnoll Brook length frequency data (Figure 3.30, Section 3.9.3) imply an apparently marked shift over the last 20 years to a more female dominated eel population. However, fewer comparative windows on the dynamics of the

Frome population are available. Furthermore, both historically and contemporarily, the data appears to suggest a somewhat greater proportion of males in the population of the Frome than of the Piddle.

Silver eel samples were obtained from the trap at East Burton in 1979 (41 eels) and 1999 (168 eels). The rack bars are very closely spaced on the Frome trap so that representative samples are produced. The two samples produced male:female ratios of 0.52:1 and 1.15:1 respectively, suggesting a typical sex ratio may be close to unity. However, there is no historic data on silver eel sex ratios for the Frome.

There is historic data for yellow eel sex ratios within the Frome system. Mann and Blackburn (1991) determined the sex of a large sample (1126 eels) culled from the Tadnoll Brook between 1980 and 1984. Sex was determined by gonad examination of eels over 250 mm in length. The sex data is presented as numbers of male and female eels in each 50 mm length class within the overall sample. It is therefore straightforward, using the length frequency distribution for the Tadnoll Brook in 1980 (Figure 3.20), to determine the ratio of males to females in the >250 mm component of the population. This gives a male:female ratio of 3.7:1. In terms of expected yield of silver eels, this figure will underestimate the proportion of males because females stay in the river for longer. Based on potential residence times, the output male:female ratio for the Tadnoll Brook in 1980 could be greater than 5:1.

The length frequency distribution for the Tadnoll Brook in 1980 (Figure 3.20) indicates a very sharp fall in the frequency of occurrence between 320 and 400 mm, within the length range of male silver eels, and very few eels >450 mm were present, the exclusively female size range. This observed length frequency pattern could potentially arise because the local habitat favours small eels and large individuals are simply moving to another part of the river. The sex ratio data discussed above is therefore useful because it supports the alternative explanation, i.e. that the majority of eels are maturing as males and migrating to sea.

A direct comparison of the 1997/99 silver eel male:female ratio (0.52:1; 1.15:1) from the East Burton trap with the 1980 Tadnoll Brook sex ratio (3.7:1 (possibly >5:1)) is somewhat problematic. Although the comparison suggests a major change in sex ratios between 1980 and the present, the true change, if it has occurred, is likely to be smaller. Whilst the Tadnoll Brook sex data may well be representative of eels in the middle reaches of the Frome, the trap draws eels from all areas upstream. There is quite clear evidence that the average eel length, and hence proportion of females, increases, though not greatly, with distance upstream (see Section 3.9.4). However, the proportion of large female (>450 mm) eels has increased more than threefold from 4 - 13% between 1980 and 1999.

No yellow eels were aged from the River Frome sites and it is therefore not possible to compare growth rates of male and female silver eels with those of yellow eels as was done for the Piddle. Although eels were aged from the Tadnoll Brook, growth rates in this tributary appear to be lower than for the Frome as a whole (see Section 3.9.5). Great caution is required in interpreting the Tadnoll/Frome age data because the validity of direct comparisons is very uncertain. Neither the Piddle or Frome data sets enable temporal comparisons of absolute growth rate, but the Piddle, despite uncertainties in the quality of the historic data, provides a better basis for determining relative change on growth rate.

A comparison of the historic (Tadnoll Brook (1980), Frome (1990) and Mill Stream (1963-86)) and 1999 length frequency distributions all suggest a shift to a less male dominated population in the Frome. However, this trend is possibly less marked than in the Piddle and as discussed in the preceding paragraphs, the limited growth rate data is unhelpful. The Frome and Tadnoll Brook length frequency distributions (Figure 3.30) for 1999 still show a decline in frequency of occurrence within the critical 320 - 440 length classes, although the decline is much less marked than historically. This suggests that there is still a significant exodus of male silver eels, which is the pattern observed on the East Burton trap.

3.9.7 Summary of key points relating to the Piddle and Frome

Analysis of the historic and contemporary eel data for the Dee and Severn is essentially straightforward and permits quantitative comparisons that are amenable to statistical testing. In contrast, the historic eel data for the Piddle and Frome, although extensive, are fragmentary and disparate. Few direct quantitative comparisons between the historic and contemporary situation are possible and statistical analysis is largely precluded. Nevertheless, the data provide a diversity of windows from which possible changes in eel population characteristics can be assessed. Even if it is sometimes opaque, the picture from almost all of these windows is essentially similar. Eel population structure appears to have changed quite substantially in the last 20 years. The evidence for this may be largely circumstantial and unquantifiable, and arguments may be somewhat circular, but overall, the evidence is compelling.

The following changes appear to have occurred in the Piddle and Tadnoll Brook (Frome) eel populations between the late 1970s and 1999.

- There appears to have been a major decline in eel population density in both rivers (Tadnoll Brook 1973 - 1980 - 1999; Piddle 1976/7 - 1999)
- There may also have been a decline in biomass in the two rivers but this is much less marked than the change in population density.
- There appears to have been a marked decline in the number of elvers entering the rivers. The recruitment of small eels (<150 mm, approximately corresponding to 0+ to 3+ years old) is now apparently very low. As larger eels are still relatively abundant, this may suggest that many eels are spending up to several years in the estuarine environment of Poole Harbour before entering freshwater, or that the decline in recruitment is relatively recent..
- There appears to have been a major change in sex ratios. Twenty years ago, the eel population of both rivers appears to have been heavily male dominated Female eels now appear to make up a much larger proportion of the population in both rivers and in the Piddle, females are probably now the dominant sex.

Possible changes to the eel populations of Poole Harbour, which comprises the estuarine component of the Piddle and Frome systems, were reviewed in Section 2.18. It was concluded that recruitment to Poole Harbour had probably declined, possibly substantially, in the long term but may have been relatively stable during the 1990s. It was also noted that the Harbour population, although dominated by males, appeared to contain a relatively high proportion (31%) of large females. Growth rates also appeared to be higher (75.8 mm y^{-1} for <450 mm eels and 63.4 mm y^{-1} for >450 mm eels in 1996) in the Harbour population than in the riverine populations (21-31 mm y^{-1} in 1999). The led to the suggestion that the carrying capacity of the harbour may be sufficient to support a relatively high proportion of the current recruitment, thus lessening the competitive pressure to migrate into the rivers. These hypotheses concerning the Harbour population are compatible with the apparent changes to the Piddle and Frome populations.

3.10 Overall Summary and Conclusions from the Re-Survey Programme of the Severn, Dee, Piddle and Frome

Eel population data (density, biomass, length frequency and growth rate) were collected in 1998 and 1999 for three river systems, the Dee, Severn and Piddle/Frome, for which quantitative historic data were available.

For the River Dee, which was surveyed in 1984 and 1999, the data provide no evidence to indicate major changes in either population density or biomass. Whilst there is a slight suggestion that there may have been a decline in the proportion of small, <150 mm, eels in the population, there is no evidence of any change in the relative proportion of large female eels. However, the Dee survey comprised only eight sites. Whilst a survey of this limited scale could be expected to detect major changes in eel population density or population structure, a much larger sample size would be required to detect more subtle changes with any degree of certainty

The River Severn surveys in 1983/84 and 1998/99 provided the largest data set (24 sites lower Severn 1998, 17 sites lower Severn 1999, 9 sites upper Severn 1999). The large number of sites and the close matching of site locations and survey methods between the historic and repeat surveys meant that this data set was amenable to statistical analysis

In spite of its relatively extensive scale, the Severn re-survey programme has provided no substantive evidence for a major change in eel population density or biomass within the Severn catchment. Estimated densities for 1998 and 1999 were generally similar to those for 1983 although there is some evidence that biomass may actually have increased slightly between 1983 and the late 1990s. The data indicate that there has been a statistically significant and marked reduction (by circa 50%) in the proportion of small eels <150 mm in the lower Severn population. There may have been a corresponding increase in the proportion of large female eels (>450 mm) in the population but statistical support for such an increase is weak. There appears to be little if any difference in the extent of catchment colonisation between 1983 and the 1990-1999 period.

Glass eel recruitment appears to have been as low in 1983 as in 1998 and 1999 according to evidence reviewed in Section 2.2.1. The much larger proportion of <150 mm (0+ - 3+) eels in 1983 in tributaries below Tewkesbury probably therefore reflects the relatively high recruitment in the preceding years (Figure 2.3). Although glass eel recruitment has been low in the late 1980s and in the 1990s, it appears to have been sufficient to maintain the overall river population to at least the 1983 level. Both density and biomass at the Zone 5 survey sites, upstream of the main glass eel fishery, have shown a large increase since 1983, but the relatively low populations recorded in 1983 can probably be attributed to habitat impoverishment following major channel works in the preceding year. However, densities recorded in 1998 and 1999 were above average for the lower Severn, despite increasing fishery pressures over the last 5 years. This supports the view of Aprahamian (1986) that the glass eel fishery cannot be proven to impact significantly on Severn eel stocks.

There is some evidence that reduced juvenile recruitment has been mirrored by an increased proportion of large female eels within the lower Severn tributaries but statistical support for this suggestion is weak and the overall proportion of eels >450 mm, at around 6%, is still quite low.

The historic eel data for the Piddle and Frome, although extensive and generally relatively robust, are somewhat fragmentary and disparate. There are some problems of data comparability due to the manner in which data were collected or reported. Also, the timing of historic and repeat surveys differs in some cases. The ability to make direct statistically validated quantitative comparisons between the historic and contemporary situation is therefore restricted and statistical analysis is effectively precluded. Nevertheless, the data provide compelling evidence that eel population density, size structure (LCF) and probably sex ratio have changed substantially in the last 20 years.

There appears to have been a major decline in eel population density in both the Piddle and Frome between the late 1970s/early 1980s and 1999. There may also have been some decline in biomass in the two rivers but this is probably not great. There is a strong suggestion of a major decline in the number of elvers entering the rivers and the proportion of small eels (<150 mm) in the population (typically < 1%) is now very low. As larger eels are still relatively abundant, this could suggest that many eels may be spending up to several years in the estuarine environment of Poole Harbour before entering freshwater. There appears to have been a major change in sex ratios. Twenty years ago, the eel population of both rivers appears to have been heavily male dominated Female eels now appear to make up a much larger proportion of the population in both rivers and in the Piddle, females are probably now the dominant sex. The limited data from silver eel traps on the Piddle and Frome provide no information as to the relative size of historic and contemporary silver eel runs, although it is implicit from the conclusions concerning population density that silver eel escapement will have declined. However, the trap data support the conclusion that the proportion of females in the population has increased. Thus it appears that in the Dorset rivers, declining recruitment has resulted in reduced population density, but that biomass has been largely maintained by an increased proportion of large female eels in the numerically reduced population.

Overall therefore, the data from the re-survey programme support the conclusions in Section 2.2.1 that there has been a reduction over the last 20 years in the level of recruitment of glass eels to UK rivers. For the Dee and the Severn, and by inference possibly other west coast rivers, rates of elver recruitment would still appear to be adequate to maintain river eel populations at historic levels. These levels are probably at or near to carrying capacity. In the Piddle and Frome, juvenile recruitment may no longer be adequate to fully meet local carrying capacity and substantial changes to population structure appear to have occurred. It has not been possible to undertake comparable studies of any east coast rivers. This is unfortunate because, being furthest from the Atlantic migration routes, eel stocks in eastern rivers are likely to be the most vulnerable to the consequences of a general shortfall in glass eel numbers.

3.11 Quality assessment of the re-survey programme and implications for future eel stock monitoring programmes.

3.11.1 Effectiveness of 1998 and 1999 eel surveys as judged by catch depletions and capture probabilities

Eels are amongst the more difficult fish species for which to undertake stock assessments (Knights *et al.* 1997). Even in relatively small shallow streams, such as the majority of sites in the re-survey programme, capture efficiencies when electric fishing may be low and failure to achieve adequate depletions on repeat fishing runs is a common phenomenon (Knights *et al.* 1997). These difficulties stem from the cryptic burrowing behaviour of eels during daytime, compounded by the relatively long period of exposure to the electric field required to draw eels from their hiding places. Furthermore, they are frequently less easy to see when stunned than other fish species and they typically recover very rapidly from electro-narcosis. In view of these difficulties, it is important to assess critically the quality of the re-survey data, because the quality of the data is fundamental to the validity of the resulting conclusions.

The primary requirements for population estimation by catch depletion are as follows:

- single run capture probabilities of > 30%;
- equal catchability of all members of the population;
- no immigration to, or emigration from, the survey stretch;
- equal sampling effort for each capture run.

The re-survey programme has therefore been examined in respect to each of these requirements.

Capture probabilities for the re-survey programme were typically high. Mean calculated single run capture probabilities (Zippin 1956, Carl and Strube 1978) were as follows:

Lower Severn 1998	0.512
Lower Severn 1999	0.496
Dee 1999	0.567
Piddle and Frome 1999	0.498

A total of 80 sites in the programme contained eels. At 5 sites, population densities and biomasses had to be computed as minimum estimates due to failure to achieve effective depletions and capture efficiencies could not be calculated. At a further 9 sites, calculated single run capture efficiency was below 0.3 but at only one of these was it below 0.2. Thus 64 of the 80 eel-containing survey sites (80%) were judged acceptable in terms of capture probability and depletion pattern. Thus in terms of catch statistics, the overall re-survey programme can be judged as markedly successful.

For the historic data, it has only been possible to compute capture probabilities for the Severn and Dee, because of the abridged nature of the Piddle and Frome data. For both the Severn (matched 1998/99 sites) and Dee, single run capture probabilities were 0.42.

The issue of equal catchability of all members of the population encompasses two problems. Firstly, small fish are less responsive to the electric field and also less visible when stunned and secondly, some individuals may be inherently uncatchable because of, for example, their location within the survey stretch (see below)

Despite considerable effort during the re-survey programme to capture eels of all sizes, capture efficiencies were generally lower for smaller eels. This point is illustrated in Table 3.13 with reference to the 1999 lower Severn catch data. Calculated mean capture probability for eels >300 mm was 0.627 and for eels of 150-300 mm it was 0.543. This difference is statistically significant (Paired T-test, $n = 16$, $P = 0.012$). For very small eels, e.g. <150 mm, the difference becomes more marked. The likely outcome is that the true contribution of small eels to the population will be underestimated. Whilst this could significantly depress density estimates, at least at down river sites where small eels may make a substantial contribution to overall population density, biomass estimates are unlikely to be more than minimally affected.

Table 3.13 Capture probability and contribution to number and weight of catch by different size classes of eels, based on the 1999 lower Severn catch data.

Size range	Capture probability	Number %	Weight %
<150 mm	0.470	15.6	1
150-300 mm	0.543	57.5	32
>300 mm	0.672	26.9	67

The issue of reduced capture efficiency of small eels is an important consideration because one of the central findings arising from the re-survey programme is the apparent reduction, at many sites, in the relative proportion of small eels between the 1970s/early 1980s and the late 1990s. The 1998/99 re-survey team were assiduous in their efforts to ensure that small eels were properly represented in the catch. However, although it has been assumed when making temporal comparisons that equal effort was expended in capturing the smallest eels during the historic surveys, the authors cannot be certain that this was the case. It seems reasonable to assume, albeit subjectively, that the historic capture efficiencies for juvenile eels were not greater than those of the re-surveys but may, in some cases have been lower. Thus the true temporal reduction in the proportion of small could be greater than the observed reduction, but appears unlikely to be smaller.

It is undoubtedly the case that some eels within a population will be of inherently lower catchability, purely because of their location, although this can only be judged subjectively. This may lead to substantial underestimates of the true population at 'difficult' sites. Eels are often difficult to draw out when they are within gabion basket or unmortared blockwork banks, beneath large boulders or within marginal reed beds or tree root systems. In all of these places, eels may be stunned *in situ* and thus remain unseen. There is at least some prospect however, that such eels will have shifted their position at least slightly and thus may be successfully drawn into open water on subsequent fishing runs. The problem of unequal catchability is probably most acute at sites with extensive marginal reed beds. Those eels deepest within the reed beds, especially the smaller individuals, are often too distant from the anode to be affected by the electric field, and simply maintain station throughout the fishing exercise. Thus even where an apparently ideal depletion is obtained, this may only relate to a part of the population.

At the majority of sites in the re-survey programme, non-catchability of more than a small proportion of the population was not perceived to be a major problem. At a few sites, however, non-catchability undoubtedly led to relatively substantial underestimates of population density and biomass. However, it is reasonable to assume that this will also have been the case for the historic surveys (though not necessarily at the same sites because of habitat change), so the overall effect on temporal comparisons of population density and biomass is likely to be small. Any resulting effects on temporal comparisons of size/age structure are almost certainly insignificant.

The prevention of immigration and emigration from the survey stretch is perhaps the one requirement of depletion population estimation that is inherently readily met in the case of eels. This is despite the fact that, in practice, it is probably almost impossible at many sites to set and maintain an eel-proof stop-net barrier. If a fine mesh net is used, water pressure tends to lift the net from the bed at the channel margins, the very point where eels are most likely to move if they do so. A coarse mesh net is easier to maintain in place but small eels will pass straight through. In practice, eels are highly sedentary during the day and tend to remain resolutely in their hiding places despite disturbance in the immediate vicinity. There is therefore very little likelihood of either immigration into the survey stretch, or of driving eels out from the top of the survey stretch, as is likely to occur with other fish species.

Any eels which are drawn from their refuges while electric fishing, but not successfully netted out, make rapidly for cover as soon as they are free of the effects of the electric field. They rarely drift down river in the current for more than a very few metres before finding cover and going to ground. Even in the absence of stop nets, it would be safe to assume that virtually no eels would enter or exit the fishing stretch within the time span of a typical survey and only a very few would be lost from the bottom of the fishing stretch as a result of downstream drift while semi stunned. Maintaining an essentially closed population for the duration of the fishing exercise is therefore not a significant issue for eels and the employment of stop nets is probably of doubtful benefit.

In relation to the final survey requirement, equal sampling effort for successive fishing runs was achieved by maintaining the same equipment settings, by the same survey team personnel undertaking the same tasks and by a similar rate of progress through the survey stretch for each fishing at a site.

In the light of the foregoing discussion, the authors are confident that a robust data set has been obtained from the re-survey programme. The data are therefore appropriate for making both temporal comparisons with historic data and also for use as a baseline for the future monitoring of eel stock status.

One problematic aspect of the re-survey data is the relatively large (approximately -30%) and statistically significant difference in the mean density and biomass obtained from the 1998 and 1999 surveys of the lower Severn (see Section 3.7.2 for discussion of potential causes) Whilst there seems little doubt that the observed changes in density and biomass (at the survey sites) were real, the probability of a pan-catchment change of the observed magnitude within the Severn system in the space of one year is clearly low. This therefore suggests that there may have been a temperature induced difference in distribution pattern within the catchment in the two summers (see Section 3.7.2 for more detailed discussion of the effects of season and water temperature on eel population density). Whatever the underlying explanation, the apparently conflicting survey results, which occurred despite careful matching of survey times, clearly have potentially significant implications for any long term monitoring of eel stocks, even if survey timings are closely matched.

3.11.2 The relative value of multi-species and single-species eel surveys for the assessment of eel stocks

An important consideration is the relative usefulness of multi-species and eel-specific surveys for the assessment of eel stocks in rivers. The Severn data provides an opportunity to examine this issue because the Environment Agency regularly records eel catch data whilst undertaking routine fish stock surveys on the Severn system. For 15 of the 1998 R&D re-survey sites, there were routine surveys in summer 1996 or 1997 at almost exactly matching sites. Whilst some differences in eel population density might be expected (see preceding paragraph), very large differences would be unlikely to occur over the 1-2 year time span.

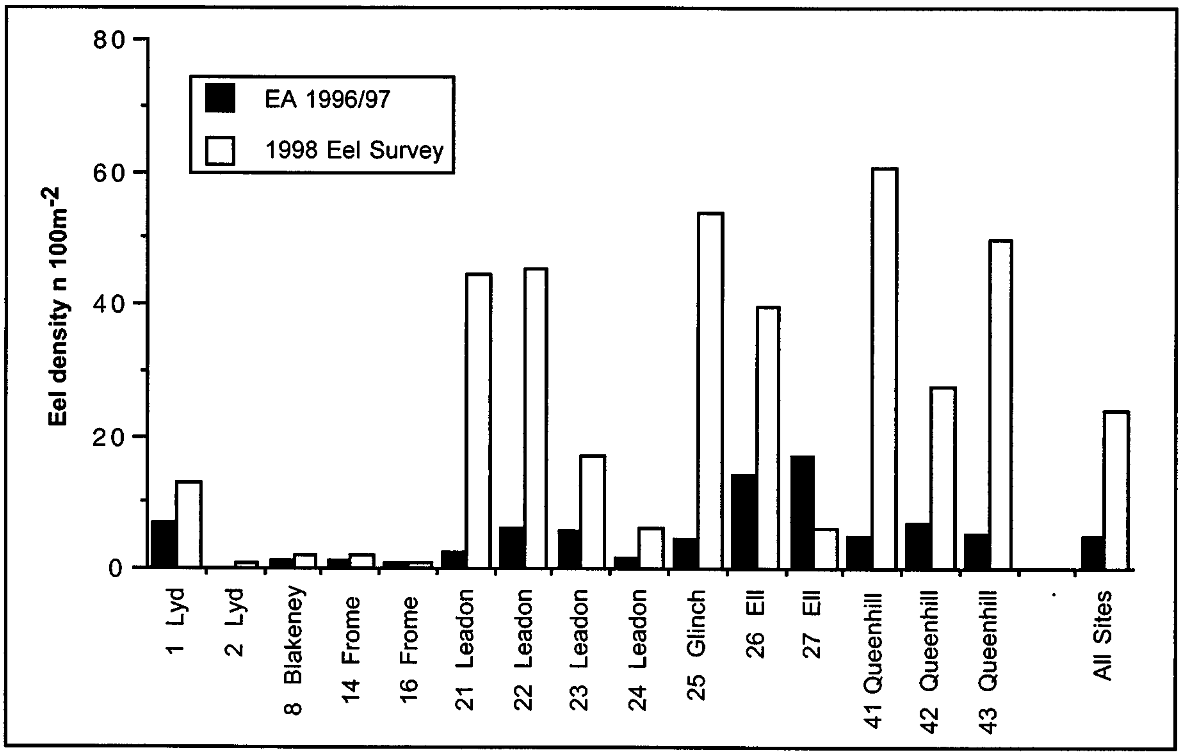


Figure 3.45 Comparison of eel densities derived from multi-species (1996/97) and single species (1998) surveys at matching sites on the lower Severn

Figure 3.45 shows Environment Agency determined densities for 1996/97 and R&D eel survey determined densities for 1998 at the 15 matching sites. For both data sets, densities have been based on 2 fishing runs (standard for the EA surveys) in order to maximise comparability. In all but one case, the R&D eel survey density is higher than the multi-species survey density. Overall mean density was 4.8 eels 100m^{-2} for the 1996/97 multi-species surveys and 23.8 eels 100m^{-2} for the 1998 eel specific surveys, an approximately 5 fold difference. The difference is highly statistically significant (Paired *T*-test, n = 15, P = 0.0038). It is very unlikely that an apparent change of this magnitude could be explained on the basis of simple inter-year variation.

Table 3.14 Mean R&D re-survey density and biomass values for the Severn, Dee and Piddle/Frome

River system	Density eels 100m^{-2}	Biomass g 100m^{-2}
River Severn 1998 - Zones 1-5 (24 sites)	40.2	1579
River Dee (8 sites, 4 high in catchment)	22.2	920
Rivers Piddle & Frome (26 sites)	15.2	1728

Table 3.14 shows mean eel density and biomass for the R&D re-surveys of the three river systems. If it is assumed that the Severn, Dee and Piddle/Frome, three widely differing river systems, are generally representative of the range of relatively productive English and Welsh rivers, then the data suggest that typical density and biomass values could be expected to be in the region of 15-40 eels $100m^{-2}$ and 1000-2000 g $100m^{-2}$ respectively.

Chapter 2 reviews available data on river eel stocks for the two countries, most of the data having been derived from multi-species fish surveys. As indicated in Section 2.29, these data suggest that typical density and biomass values for relatively productive English and Welsh rivers would be in the range of 5-12 eels 100 m^{-2} and 300-400 g $100m^{-2}$ respectively. The R&D resurvey data suggest that these figures are probably underestimated by at least threefold.

The foregoing brief assessment provides a very clear indication that multi-species surveys are likely to seriously under-estimate eel populations in probably the majority of cases. Thus whilst multi-species surveys undoubtedly provide useful data on differences in spatial distribution (presence/absence) and possibly also on temporal trends, they do not, in general, provide a good basis for the assessment of absolute density and biomass. The relative inefficiency for eels of multi-species surveys is an inevitable consequence of their sedentary burrowing day time behaviour and the consequently rather different and single minded strategy required for their capture, coupled with the fact that in multi-species surveys, eels tend to be very much a secondary or even incidental target species.

It is clear that single-species surveys are required for specific and accurate eel stock assessments, although this is not to say that eel-specific surveys do not also underestimate populations at some sites (Section 3.11.1).

3.11.3 The accuracy and value of eel age and growth rate determination.

The most time consuming element of the resurvey programme was the eel ageing. It is therefore important to consider both the quality of the data produced and its value in the overall assessment of eel stock status. As discussed in Sections 3.7.6 and 3.9.5, difficulties are likely to arise when comparing eel growth rates obtained by different workers because of the inherent uncertainty and potential bias in otolith interpretation and the potential compounding effects of different otolith preparation methods.

From the 940 eels aged for the R&D programme a random selection of 124 mounted otolith pairs were sent to the Environment Agency's National Fisheries Laboratory (NFL) at Brampton for independent ageing for quality control purposes. Figure 3.46 examines the correlation between the R&D team's age determinations and those of the NFL for the AQC set of 124 otolith pairs. The comparative data provides a useful insight into the problems of accuracy and precision inherent in eel otolithometry. As indicated by the regression line, the mean level of disagreement is about 6% with the R&D team tending to attribute the slightly younger ages. However, there is greater divergence at the individual eel level, with R&D team and NFL ages differing by up to three years.

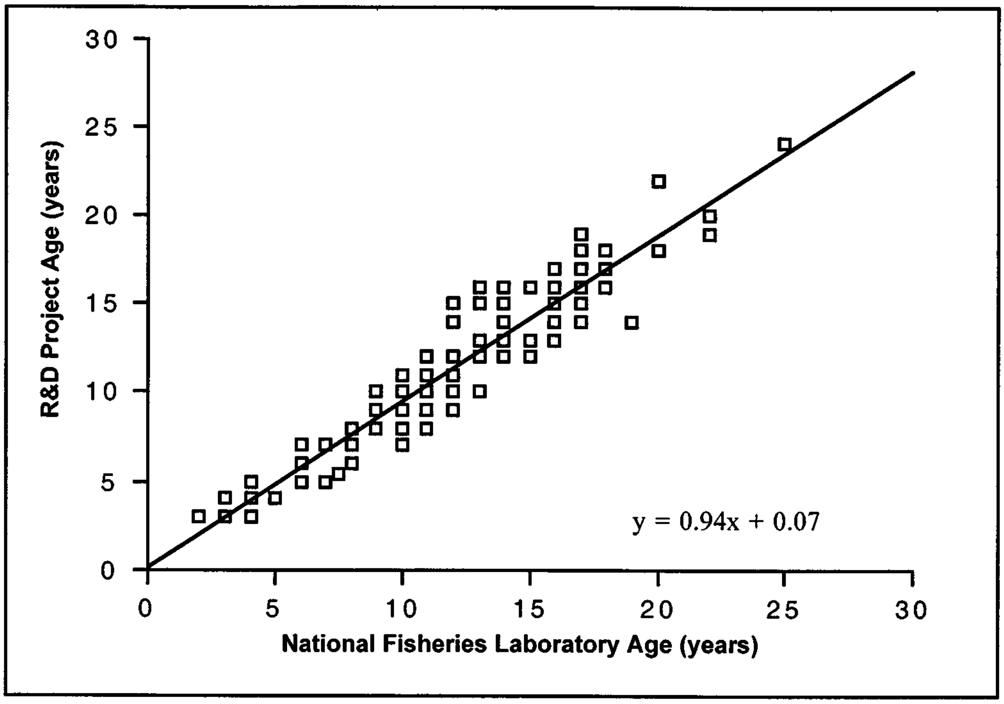


Figure 3.46 Comparison of R&D Team and National Fisheries Laboratory determined ages for the AQC set of 123 eels.

When ageing eels, it is conventional to state a level of uncertainty. The level of uncertainty for the NFL age determinations was:

1 year uncertainty	27 eels;
2 years uncertainty	60 eels;
3 years uncertainty	35 eels;

For the R&D team assigned ages, 12 out of the 124 (10%) lay outside of the uncertainty limits of the NFL age but only one by more than one year. The foregoing demonstrates very clearly the problem of bias and uncertainty in eel ageing. To these difficulties must be added the very substantial variations in individual growth rates within a given population. Defining the mean growth rate for a population within narrow confidence limits may be, from a practical view point, effectively unachievable.

Table 3.15 Annual growth rate of eels from various catchments within Europe (from Aprahamian 2000)

Water Body (Country)	Growth rate (mm y^{-1})
Burrishoole (Ireland)	13.8 - 20.8
Caven Lakes (Ireland)	18.9 - 40.5
Ballynahinch (Ireland)	19
Koge-Lellinge A (Denmark)	23.1
Giber A (Denmark)	24.5
Barrow Ireland	33.0
Thames (England)	33.8 - 66.2
Quadalquivir (Spain)	37.0 - 152.0
Rhine (Germany)	55.0
Isma (Norway)	62.0

Reported eel growth rates from different locations vary substantially, ranging from 14 to 152 mm per year, as shown in Table 3.15, taken from Aprahamian (2000). This table suggests that bias and uncertainty are likely to be small in comparison to the true differences in eel growth rate in very dissimilar habitats such as a highly productive estuary and an upland stream. or even a moderately productive river. Thus for example, mean growth rates within Poole Harbour (determined by the NRA in 1996 at 63.4 - 75.8 mm y^{-1} are clearly substantially and unequivocally greater than those determined for the Piddle and Frome in 1999 which ranged from 21.1 mm y^{-1} (Tadnoll Brook yellow eels) to 31.1 mm y^{-1} (Piddle female silver eels).

However, whilst determining growth rates for different populations in different habitats may be of considerable scientific interest, a more likely objective for stock monitoring purposes would be to analyse temporal or spatial trends in growth rate within a single system. In this latter case, it is highly likely that bias and/or uncertainty in ageing, coupled with high random variation in individual growth rate, would be large or very large relative to any true differences in growth rate. For example, Aprahamian (2000), despite ageing a total of 1980 eels, was unable to demonstrate any significant relationship between growth rate and density (12 - 114 eels $100m^{-2}$) or biomass (256 - 2524 g $100m^{-2}$) in tributaries of the lower Severn.

Given the statistical requirement for very large sample sizes for ageing and consequently high resource input, and, irrespective of sample size, the inherent uncertainty and potential bias in the data produced, eel otolithometry cannot be regarded as a useful stock monitoring tool.

4. ASSESSMENT OF HABITAT REQUIREMENTS AND CHANGE

4.1 Introduction

Project objectives are;-

- ***to investigate the habitat requirements of the eel and any significant changes in habitat availability that have taken place in the last 25 years.***

The principal aims are to;-

- determine key habitat and environmental parameters that influence intra- and inter-catchment distribution and population structures in space and over time
- detect whether changes in the extent of habitat and related environmental characteristics over the last 25 years could have affected stocks, fisheries and spawner escapement (especially in the light of known declines in glass eel recruitment and any possible impacts of fishing mortality)
- aid in the setting of management targets to maintain spawner escapement and hence sustainable stocks and fisheries (Chapter 7)
- provide information for the planning and execution of monitoring programmes and the interpretation of data gained in relation to management targets (Chapter 8)
- to determine whether habitat improvements and protection are required for eels

This Chapter reviews information in the published literature and other sources, including issues mentioned in NRA and Environment Agency Catchment Management Plans and LEAPs. HABSCORE, Fisheries Classification Scheme (FCS) databases and habitat information collected in this R&D project (especially from resurveys discussed in Chapter 3) are also reviewed to help determine key parameters. R&D project temporal data are examined to determine any impacts of habitat change over the last 20-25 years.

Correlations between population indices and environmental characteristics are explored to see if predictive habitat suitability indices (HSIs) can be devised to aid monitoring and management. Intra- and inter-catchment patterns of population distributions and structures are also reviewed and relevant classifications for spatial and temporal comparisons of eel stocks considered.

It must be emphasised that when discussing density and biomass data from Chapter 2, the data were gained from multi-species electric fishing surveys. They therefore generally represent underestimates because R&D resurveys have shown that eel-specific sampling yields values about three to four times greater than those from routine surveys (Section 3.11.2).

4.2 Key Environmental Parameters

Parameters reviewed were those identified in Chapters 2 and 3, from models for eels proposed by Schouten (1992) and Klein Breteler (1996), plus others from the general literature and from NRA/Environment Agency Catchment Management Plans and Local Environment Agency Plans (Table 4.1). Use is made of habitat information from the HABSCORE and FCS databases. Other key changes that might be linked to recruitment and stock declines have been reviewed by Knights *et al.* (1997) and Moriarty and Dekker (1997).

A composite list and brief conclusions about their possible impacts, drawn from these sources and other publications, follows;-

4.2.1 Physico-chemical parameters (Table 4.1, Issues 1 and 2)

Eels are tolerant of low **dissolved oxygen** and **unionised ammonia** (Knights, 1981, 1995) and these should only be limiting under extreme chronic or episodic pollution conditions. Chronic pollution sources are now rare, although DO certainly has been a limiting factor in the past, e.g. in the Thames and Mersey Estuaries, industrialised north eastern rivers and in upper reaches of the Warwickshire Avon and other rivers in the Midlands. Oxygen sags can have drastic and short-lived local impacts, e.g. as in the eel kill at Whittlesey in the Nene described in Section 2.23.1. It has also been suggested that poor water (and habitat) quality may form a partial migration barrier in the Dorset Stour (Section 2.19). Otherwise, no evidence was found to show that DO and ammonia have had major impacts in England and Wales over the last 20-25 years.

Key **temperatures** for eel migrations are 4-6°C and > 16°C, the optima for glass eel immigration into freshwater and the upstream migration of 0+ elvers and > 0+ juvenile yellow eels respectively (White & Knights, 1997). Inter-year variations in temperatures can produce marked differences in upstream recruitment, for example, glass eel catches in 1991 in the Severn were very low because of low water temperatures in the spring (White & Knights, 1994). Eels are basically warm-water species and growth tends to increase linearly with the number of days above 10°C per year and activity levels and growth rates are highest above 25°C. All waters in England and Wales are suitable in these respects, although growth would be faster in, for example, southern stillwaters than in more northerly upland streams. However, average temperatures in England and Wales have, if anything, increased over the last few decades due to climate change Table 4.1, Issue 3) and this should have benefited eels.

Concerns have been expressed about increasing **acidification** of poorly buffered upland waters over recent decades (Table 4.1, Issue 2). Turnpenny *et al.* (1987) found eel stocks in mid- and North Wales declined below pH 5.5-6.0. However, this might be expected because of the distance of most of their survey sites from the sea, their relatively high altitude, intervening migration barriers and low ecosystem productivity (see Section 2.26). Furthermore, Jessop (2000) found stocks of *A. rostrata* were still quite high in rivers in Nova Scotia, Canada, at pH 4.7-5.0 and 0.19-0.48 mg l^{-1} total dissolved aluminium, although he quotes other studies that show elver mortality can be enhanced under such conditions.

Table 4.1 Summary of habitat-related issues applicable to eels given in NRA/Environment Agency Catchment Management Plans, Local Environment Agency Plans, Action Plans and Annual Reviews (published up to April 2000)

1. Acute pollution impacts
- Gross localised pollution
- Accidental discharges

2. Chronic pollution impacts
- Over-enrichment (low DO, high ammonia)
- Eutrophication
- Bioaccumulation/magnification (e.g. of organochlorines)
- Acidification
- Acid mine drainage

3. Long-term climate change
- Increases in average temperatures and changes in rainfall patterns

4. Biological Factors
Infestation of eels by *Anguillicola crassus*

5. Flood control and water level management
- Tidal barrages
- Tidal flaps, doors and sluices
- Weirs and sluices
- Pump drainage
- Water transfers
- River regulation

6. Habitat Loss & Degradation
- Land drainage

7. General Habitat Quality
- Canalisation
- Culverting
- Siltation
- Concretion
- Loss of /change in in-river vegetation

8. Low Flows
- Droughts and over-abstraction

9. Water abstraction
- Entrapment losses
- Losses to pumps, HEP, etc.

10. Eel fishery issues (dealt with in Chapter 8)
- Impacts on eel stocks
- Illegal fishing
- By-catches
- Enforcement problems
- Impacts of eels as predators
- Impacts of predation on eels

It is unlikely that acidification in England and Wales has had significant effects on eel stocks. The same is true of pollution due to **acid mine drainage** affecting some local catchments.

In view of the euryhalinity of eels, **conductivity** is an unimportant factor, except when this is linked to low nutrient status, pH and habitat productivity.

Eutrophication has been an important trend in many waters over recent years (Table 2.1, Issue 2). This should, however, have benefited eels because of increases in vegetation cover and enhanced trophic status, especially in view of the wide diet breadth in eels (Tesch, 1977; Deelder, 1984).

4.2.2 Biological factors

One biological factor of unknown importance is infection of eels by the **swim-bladder nematode, *Anguillicola crassus*** (Table 4.1, Issue 4). This is a parasite endemic in *Anguilla japonica*, introduced in the 1970s from the Far East and now widespread in Europe, including England and Wales. Infection stresses have been linked to some eel-kills in central Europe (although oxygen crashes may have been the primary cause) and damage to the swim-bladder could affect the oceanic migration of silver eels but there is no concrete evidence for this (e.g. see Knights in Moriarty & Dekker, 1997).

4.2.3 Anthropogenic factors

Concerns have been expressed about bioaccumulation and biomagnification of **heavy metals** and liposoluble xenobiotic **organochlorine compounds** (such as pesticides and PCBs) (Table 4.1, Issue 2). Accumulated compounds could affect survival and could be mobilised from fat stores during migration of silver eels and residues passed on to the yolk in eggs. Effects on reproductive success are unknown but a review by Knights (1996) concluded there is no clear evidence of major effects on survival. Furthermore, high levels of contamination are only found in a limited number of waters and declines in recruitment are not clearly correlated with periods of maximum contamination or following reductions in pollution due to improved controls and the banning of chemicals such as DDT, dieldrin and PCBs. Thus although levels may be elevated in eels (especially older ones) in particularly highly polluted waters, overall effects on stocks have probably not been significant.

Man-made **migration barriers** have been shown in earlier sections to be of significance in some stretches of some rivers (Table 4.1, Issue 5). However, most of the major river control structures that impede migration have been in place since well before the declines in recruitment seen after the late-1970s. Recently-constructed or refurbished barriers have often been required to fit fish passes (Knights, 1998). Overall impacts on stocks nation-wide are minor according to Knights (in Moriarty, 1996, and Moriarty & Dekker, 1997) and Knights & White (1998). In some catchments, earlier discussions have shown that barriers may even have some benefits because low densities upstream can encourage the development of females rather than males.

Loss and degradation of aquatic habitats, due to land drainage, water management engineering, agricultural practices and urban developments are cited as fisheries issues in many CMPs and LEAPs and other publications (Table 4.1, Issues 6 and 7).

Losses of reed beds, fen and marshland drainage ditches, lakes and similar habitats to agriculture and to urban development could have had impacts on eel stocks. A range of publications consulted show that losses were particularly great after the Second World War, mainly to agricultural land reclamation, drainage and intensification, encouraged by government grants and subsidies. For example, between 1945 and the late 1970s, the number of ponds and lakes in England fell by 30% and the extent of East Anglian fens by 90%. However, rates of loss have subsequently fallen by < 5% over the last 20 years or so. Reductions in habitat would have been expected to have had effects on eel stocks between 1945 and the late 1970s, but this is not compatible with the peaks of recruitment in the late 1970s-early 1980s demonstrated in Section 2.2. No clear evidence of any major impacts on eels stocks in the last 20-25 years were located in reviewing data in Chapter 2.

Poor habitat management practices could also be important. For example, Growns *et al.* (1998) found populations of the long-finned eel *A. reinhardtii* declined significantly in stretches of an Australian river catchment subjected to removal of bankside and in-stream vegetation and woody material. Populations were not affected by eutrophication of the system. Dredging can also alter the form of rivers and change trophic structures and weed-cutting can injure eels. No proof of significant impacts of such management practices on eel stocks has been found. Some changes may indeed be beneficial to eels, e.g. Section 2.14 discusses how dredging a tributary of the R. Otter enhanced local eel stocks. Furthermore, the major losses of habitat and damage pre-date the late-1970s peaks and the later declines in glass eel recruitment.

Low river flows, over-abstraction, water transfers and water-intake mortality could be important (Table 4.1, Issues 8 and 9), but no evidence of deleterious effects on eel stocks were found. Effects could be locally important in some seasons or years, however. The example of the complex system of impoundments and transfers in the upper R. Tavy (Cornwall) has been discussed earlier in Section 2.10. Low flows could actually benefit eels in making it easier to climb barriers (Knights, 1998) and because low flow conditions are often associated with elevated water temperatures.

4.2.5 Oceanic habitat factors

Although not within the R&D project aims, these ideally need to be considered because of the catadromous life cycle of eels. Although proof is lacking, changes in N. Atlantic current systems and trophic status have been suggested to be a major cause of recruitment declines. These possibilities are discussed in more detail in Chapter 5.

4.3 Key Local Habitat Features and Habitat Suitability Indices

From the preceding discussion, general water quality and anthropogenic factors do not appear to have had major impacts on eel distribution and population characteristics in England and Wales over the last few decades. Local habitat features may, however, be of importance, acting alone or in combination, in determining baseline stock distributions and characteristics. Key ones suggested in earlier Chapters include substrate, gradient, altitude and distance from tidal limits. Habitat suitability indices (HSIs) are a means of modelling and quantifying relationships between sets of such variables and population measures which can be useful in;-

- identifying key habitat features and combinations which influence distribution, standing crops and population structures and dynamics
- clarifying past and predicting future impacts of habitat change (especially in comparison to possible impacts of natural *v.* fishing mortality)
- informing monitoring and management

The Schouten/Klein Breteler model for eels mentioned earlier attempted to identify key habitat factors and then to (subjectively) allocate scores for each on a scale of 0 (totally unsuitable for eels) to 10 (maximum suitability). Summing the values for a particular habitat produces an aggregate score to assess relative suitability. Actual *v.* predicted scores can be compared to highlight waters not meeting their potential and then reasons for the short-fall sought. This model has, however, not been tested and validated. It also includes some factors of minor importance to stocks in England and Wales (e.g. mortalities on passage through hydropower turbines and drainage pumps) but omits others of importance (e.g. river gradient).

The use of single-species HSI (and related IFIM and PHABSIM) models for salmonids have been reviewed by other NRA/EA R&D projects and shown to have drawbacks in derivation, validation and application (Milner *et al.*, 1993; Wyatt *et al.* 1995; Barnard & Wyatt, 1995). Deriving models from localised stream ecosystems produced good predictions when the models were applied to the same or very similar ecosystems, but were found to be restricted in their application to other geographical regions and less comparable habitats. The development of such models also requires considerable amounts of information and hence allocation of resources. Adequate data for eel habitats are very limited or not available for rigorous derivation or testing of such models, especially given their apparently ubiquitous distribution, migratory behaviours and catholic habitat preferences, especially compared to juvenile salmonids. Efficient and reproducible sampling of eels is also problematic. However, the potential for developing even a coarse-scale model for eels to achieve the above aims needs to be explored. This could in turn inform development of HSI models for other fish species.

The HABSCORE database contains habitat and salmonid information for 409 sites but none on eels. Sites included are mainly derived from NE, NW and other salmonid rivers and many only include a restricted number of sites per catchment. The only complete and matching contemporaneous data on eel populations found were for the River Dart in Devon, these are reviewed below. The FCS database includes additional records to the HABSCORE one, giving a wider spread of river types from different geographical regions. It includes 458 sites, 339 of which could be matched with both density and biomass data for eels. Habitat information is, however, restricted to width, gradient and conductivity.

The HABSCORE habitat features (for further details of asterisked items, see Wyatt *et al.*, 1995 and Barnard & Wyatt, 1995) include;-

- **Catchment site features**: catchment, river and site names, sampling area dimensions, depths*, NGR, distance from principal source and from tidal limit, link number (total number of first order streams upstream of site), downstream link number (link number of next river downstream)

- **General site features:** altitude, site gradient, 1085 slope (average slope of river between 10% and 85% of the distance between the site and principal source)*, flow range code*, flow rate and type codes*, riparian shading (type and extent)*, sources of cover for young salmonids*, relative area of deep water*, substrate category (bedrock, silt, etc.) codes*, substrate embeddedness score (High = 3, Moderate = 2 to Low = 1, where 3 = high level of clogging of boulder/cobble interstices with fine silt or sand accretion, reducing cover for young salmonids)

Because datasets in Chapters 2 and 3 indicate that densities, biomass and population structures are correlated with distances of sites from the tidal limit, this parameter was measured where necessary and added to the FCS database.

4.4 Assessment of Applicability of HSI Approaches

The Dart is reviewed in detail because it was the only river found to provide enough site information on eel density (but not biomass) to match with habitat data in the HABSCORE database. The Dart is a relatively small S. Devon catchment (475 km^2) with a short main river (~ 35 km) and simple tributary system. It rises at 550 m AOD on Dartmoor, flattening out below Buckfastleigh and running through areas of rich red soils. Sites therefore encompass a range of lowland and upland habitat types but it is unfortunate that no biomass data were available – or any comparable eel data for bigger river systems.

Table 4.2 Significant relationships between eel density and habitat variables in the River Dart (S. Devon) 1987 electric fishing survey of 44 sites (habitat data from the HABSCORE and FCS databases, eel and distance data from the current R&D programme, see text and HABSCORE manuals for details)

EEL DENSITY (N 100 m^{-2}) Significant correlations ($p < 0.025$)	HABSCORE database (15 matching sites, 13 with eels present)	FCS database (44 matching sites, 38 with eels present)
Negative correlations	Altitude Distance from tidal limit Percentage of deeper water	Altitude Distance from tidal limit
Positive correlations	Conductivity Cover for > 20 cm trout Overhanging and emergent vegetation providing cover for young salmonids	Conductivity

Aspects of eel distribution in the Dart have already been discussed in Chapter 2.12. The 1987 fisheries survey of 44 sites recorded information on eel density which are matched by 44 FCS but only 15 HABSCORE site data sets (with absences at six and two sites respectively). Multivariate techniques and transformations (e.g. as used in the HABSCORE programme analyses) were not applied to the data as it became clear that Pearson correlation matrix and simpler regression analyses met the aims of the present study and the restricted data available. Key relationships are given in Table 4.2 and selected ones illustrated in Fig. 4.1, plotting all available relevant data combined and showing the trend lines producing the highest R^2 values.

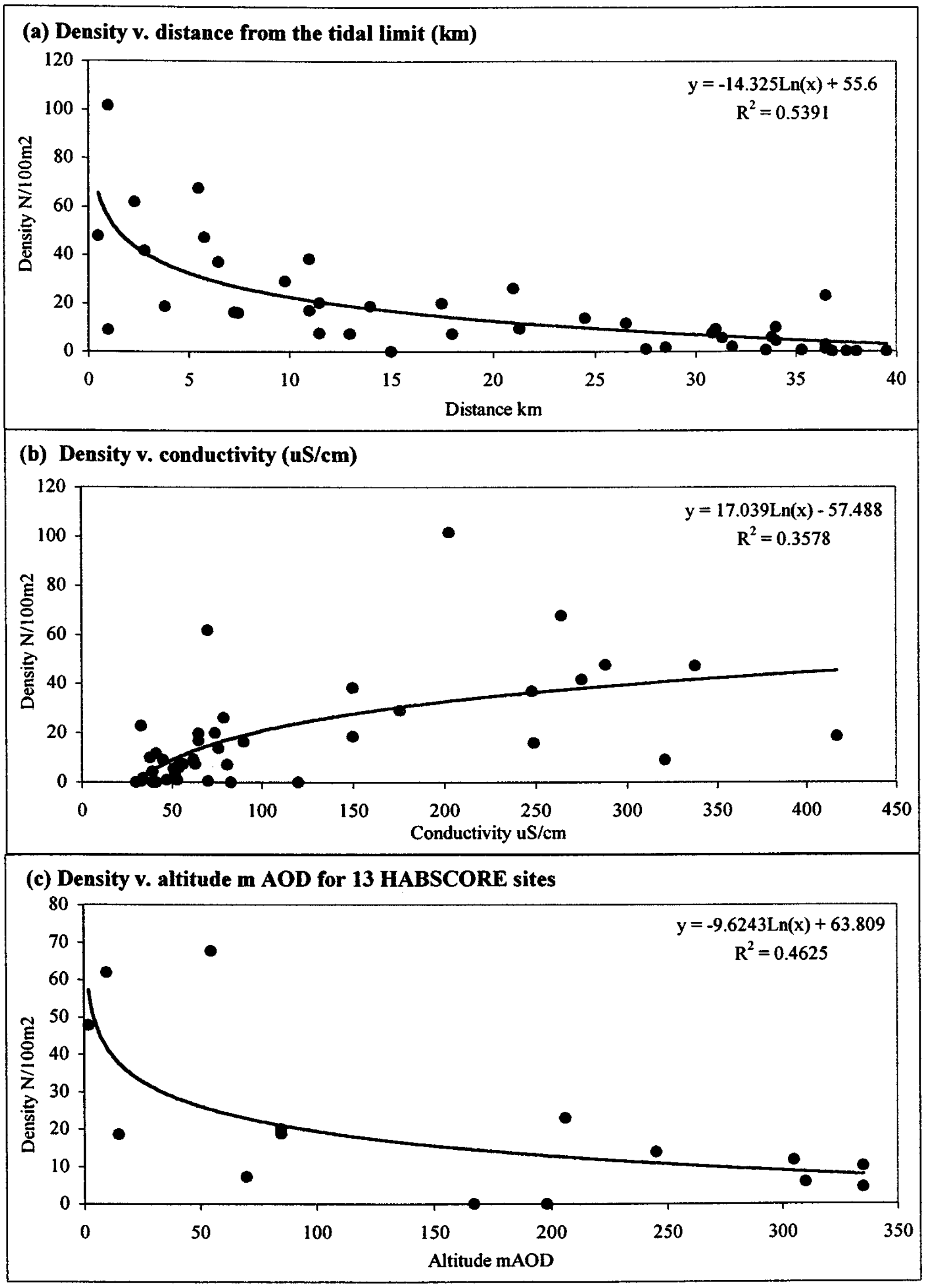


Fig. 4.1 Relationships between eel density and selected habitat variables for sites on the River Dart (S. Devon) (combining HABSCORE and FCS database habitat information and R&D eel population information for the survey of 1987, see text for further details)

4.4.1 General inter-relationships

Some specific local habitat characteristics were significantly correlated with one another ($p < 0.05$) but these would have been expected to be inter-related. For example, positive correlations were found between embededness, moss cover on rocks and extent of cobble and gravel substrate and between vegetation overhang and shading by herbs.

4.4.2 Altitude, gradients, conductivity, flow and water depths

Density declined significantly with increasing altitude and falling conductivity (Table 4.2 and Fig. 4.1b and c). However, distance was the prime factor because altitude and conductivity were more strongly correlated with distance upstream than with density. Furthermore, the negative correlations with conductivity would be expected because the Dart arises on granite and then flows in its lower course through areas of later, non-igneous substrate.

No significant relationships were found with site gradients (local or 1085) or with flow rates, implying these factors do not inhibit colonisation of the catchment. The local site gradient (measured between 5 m AOD contour lines up- and downstream of a site) is normally low because of the selection of relatively flat stretches for electric fishing. However, neither gradient measure takes account of sudden changes between sites due to waterfalls and rapids that can interfere with upstream migration.

The negative correlation found in the Dart between density and percentage of deeper water is probably due to reduced capture efficiency in such waters rather than any particular preference. Deeper waters may, however, be selected for over-wintering because temperatures remain more stable and eels in burrows and crevices are safer from predators, such as otters (Tesch, 1977; Deelder, 1984).

The FCS database covers a wider range of rivers than the HABSCORE one, but the only habitat parameters included in the former are gradient, mean width, conductivity and altitude. Analysis of 339 suitable sets of site data with both density and biomass information reveal that the only statistically significant relationship ($p < 0.001$) is between density and distance from the tidal limit.

4.4.3 Substrate and vegetation cover

Significant positive correlations were found with vegetation cover but none with substratum types. Interpretations of such a small data set are confused, however, because cover and substrate preferences vary widely in eels of different sizes. Furthermore, comparisons with the River Otter suggest there are differences between rivers of similar type geographically close to one another. Analyses of two datasets for the Otter in 1978 showed the only statistically significant correlations ($p > 0.05$) were all positive ones between (a) density and distance from the tidal limit and non-vegetation cover, (b) biomass and 1085 gradient and (c) mean eel length and embeddedness. Most of these relationships would be expected, but the sample sizes were very small because density data could only be matched for five sites to HABSCORE habitat data and no information was available for some potentially important tributaries.

No clear relationships were revealed when the current R&D resurvey data and HABSCORE-type habitat records were compared for different sites. For example, the highest recorded densities in the Piddle and Dee were sometimes associated with a predominance of sand-silt-clay and in-stream vegetation at a site, in others, river beds were very gravelly. Low densities were also found at sites with plenty of in-stream vegetation and sand-silt-clay. The former result may have been influenced by sampling problems in dense vegetation. Similar variability has been recorded in other studies reviewed by Tesch (1977) and Deelder (1984). The only common associations they mention are that lower densities or absences are generally associated with bed-rock, gravel and hard sands. Such sites tend to occur, however, in head waters at higher altitudes, where rivers flow over igneous bedrock. This helps to explain the apparently poorer stocks in the upper Dee and Severn, and in the Pennines and similar British uplands.

More recent examples also show high variability. For example, whilst eels (of various sizes) in the R. Lee (Thames catchment) preferred wider, deeper and lentic channels with muddy and silty substratum (Pilcher & Copp, 1997), higher densities of eels were found in rocky areas in NE Scottish lochs and streams than in muddy-bottom ones (Carss *et al.*, 1999). Baras *et al.* (1998) tracked large radio-tagged eels and found the highest preferences in a small inland stream were for suitably-sized crevices in stone walls, followed in order by gaps between weir boulders, within rootwads, burrows in undercut banks, between rocks and then burrowing in open silts. Smaller eels may favour reedbeds and vegetation (e.g. see Perrow *et al.*, 1996). They are difficult to sample in such habitats, which helps explain why small eels appear so under-represented in many surveys.

In relation to possible habitat/prey relationships, eels are catholic feeders on a very wide range of prey (e.g. see Tesch, 1977, and Barak and Mason, 1992). Even large eels feed on many different types of small invertebrates, although some may become piscivorous. Small eels may form an important component of the diet (Lowe, 1951; Sinha & Jones, 1967; Knights, 1983).

In conclusion, it appears that eels do not show distinct day-time habitat preferences, excepting that their negative phototactic and positive thigmotactic behaviours will lead them to select any site with any type of shelter or substrate suitable for burrowing or hiding in, relative to their body size. Distance upstream is a key variable, as discussed further below.

4.4.4 Distance from tidal limit

For the Dart, density was significantly negatively correlated ($p < 0.001$) with distance from the tidal limit. Graphing relationships shows there were some low and high outliers (Fig. 2.16). The exceptionally high density of 101.61 eels 100m^{-2} was found at a site close to Dartington STW where the substratum was silty and where good feeding grounds were available in the main river just downstream. Sites with > 55 eels 100m^{-2} were located in lower tributaries where substrates were probably particularly favourable for day-time burrowing. The low densities in some lower catchment sites appeared to be associated with migration barriers and poorer water quality (Section 2.12). Electric fishing was also probably less efficient in deeper waters in lower reaches, where conductivity also tended to be relatively high because of saline influences near the tidal limit.

Similar negative relationships between density and distance from tidal limits (but less pronounced ones with biomass) have been demonstrated in Chapter 2 and resurveys in Chapter 3 and also in many other European rivers (Tesch, 1977, Deelder, 1983). The data in earlier chapters shows that the most marked cut-offs in populations tend to occur at on average about > +50 km. The explanations for these relationships are that glass eels are attracted by freshwater discharges to enter rivers (via single or combined estuary mouths, sounds, bays, etc.) and then must metamorphose to the elver stage and commence feeding before being able to actively migrate upstream. Some elvers may migrate long distances in their first year in fresh water, but many will settle in suitable habitats below or just above the tidal limit. As the local population density increases over time, density-dependent factors come into play. Agonistic interactions will become more common which enhance tendencies to migrate upstream (Knights, 1983). Biomass will also increase as eels grow (at rates depending particularly on local habitat productivity and length of growth season). As the carrying capacity is approached, competition for resources (both for food and shelter) could then further enhance feeding foray distances and longer distance migrations upstream, though some eels may also drop downstream. Habitat segregation of different sizes and ages will be further enhanced by increased aggression in larger eels and cannibalism (Aprahamian, 1983, 1985). The result is that shorter (< 40-50km) rivers and downstream reaches of long rivers will tend to achieve similar (relatively high) densities and biomass related to carrying capacity. Higher densities will tend to enhance earlier maturation and emigration as males at < 40-45 cm. Conversely, lower density populations upstream will tend to produce mainly females, maturing at a later age.

Deviations from these relationships with distance from tidal limits are of value in interpreting monitoring data and informing management because these could be indicative of overfishing, recruitment failure or other causes, as discussed further in Chapter 7 and 8. However, no examples of fishing mortality impacts have been identified in the datasets reviewed. Migration barriers are, however, of obvious importance in some instances, e.g. sluices and weirs in parts of the catchments in N. Wessex (Section 2.5). Natural barriers (e.g. Golitha Falls in the Fowey catchment, (Section 2.8) and steep gradients and low productivity of upland reaches on igneouss substrata (e.g. in the upper reaches of the Dart, Severn and Dee) are also important.

There are, however, some long and productive lowland rivers where distributions appear more truncated than expected. For example, in the Thames relatively few eels penetrate more than 50 km upstream and the upper catchment is poorly utilised, certainly in relation to historical records for past centuries (Section 2.22.1). Such changes over very long time scales are probably due to major changes in rivers, primarily the imposition of increasingly severe barriers for water level and flood control and for navigation purposes. Low-river barriers have major impacts because they reduce tidal penetration (and hence tidal-stream transport of glass eels upstream) and interfere with upstream migrations of elvers and > 0+ yellow eels. Problems are compounded by channelisation and losses of smaller stream, ditch and wetland habitats. However, fuller colonisation might have been expected in the Thames following improvements in estuarine water quality and the removal of related pollution barriers in the 1970s. Even with reduced recruitment, eels can still being found at long distances from the sea. Baras *et al.* (1996) have, for example, trapped eels migrating up the River Meuse (S. Belgium) in relatively large numbers at 227 km from the tidal limit. For example, Baras *et al.* (1998) found local streams off the Meuse at +225-230 km supported relatively high density and biomass populations (11.8 eels $100m^{-2}$ and 2540 g $100m^{-2}$ respectively). It must be noted, however, that eels can reach this area not only by the Meuse but also via the Albert Canal, which is +150 km from the Schelde Estuary. Thus there are two migration routes available, both comprised of large, low-gradient canalised systems with locks capable of taking large ships. The size of the locks and the fact that they are opened frequently, even during the night when eels would be migrating, means they do not form major barriers to immigrants.

The evidence shows that the apparently reduced distributions in upper regions of long rivers in England and Wales are probably mainly due to combinations of low recruitment (especially to east coast rivers distant from the main Atlantic migration routes), migration barriers and, in some, to increasing gradients and unfavourable habitats. Fishing mortality does not appear to have had any major influences, at least over the last 20-25 years. It is possible that without the glass eel fishery in the Severn Estuary, overall eel abundance and distribution range might be greater. However, as concluded in Section 3.10, much of the available habitat appears to be occupied at or near carrying capacity, given distances from the tidal limits and the suitability of sites.

The extent of estuarine habitat may also be an additional factor influencing the extent of colonisation in some systems. The average river width from the HABSCORE and FCS databases is 5.5 m (SD ± 3.1 m), therefore on average, a river length of 50 km represents an area of approximately 27.5 Ha. Short and narrow estuaries offer a very small area of habitat that might be colonised by elvers or > 0+ yellow eels. In contrast, the areas available are very much larger in productive estuaries and adjacent shallow coastal habitats, such as those associated with the Thames, Severn and Essex rivers. Such areas have high potential carrying capacities and could act as habitat 'sinks' for initial recruits and any eels dropping back downstream. Although firm data are lacking, it is possible that eel stocks in upper parts of the Taw and Torridge (Section 2.6)and in rivers discharging into Plymouth Sound (Section 2.9) and into The Solent (Section 2.2) are affected by the extents of estuarine-coastal habitat 'sinks' available, situations exacerbated by low initial recruitment. Low stocks seen in many other rivers in Europe, especially those off the Baltic Sea (Moriarty & Dekker, 1997), can be explained in the same way.

According to the R&D resurvey studies (Section 3.9), the Piddle and Frome show the strongest evidence for declines in stock density since the late 1970s, together with increases in average size of eels and shifts in sex ratios towards a greater proportion of females. However, these rivers are only subject to limited exploitation via silver eel traps. They discharge into what is effectively a large coastal lagoon, Poole Harbour, and there is no significant freshwater outflow to the sea via the restricted mouth to attract glass eels. There is anecdotal evidence from fishermen that eel stocks in the Harbour have declined over the last 20-30 years, leading to marked reductions in effort and fyke net catches of eels (Section 2.18). It is possible that growth overfishing has occurred in the past, but such fishery pressures on yellow and silver eels cannot explain the changes seen in riverine stocks. Instead, the impacts of Poole Harbour as a habitat sink coupled to declines in recruitment offer the best explanation for the changes seen. The total lengths of the Piddle and Frome is ~ 45 km, i.e. about 25 Ha of riverine habitat, whereas the Harbour comprises some 3800 Ha at HWS of productive shallow habitat (Knights, 1997b). This therefore offers a very much larger area of suitable habitat of relatively high carrying capacity to immigrants. Coupled with the declines in glass eel recruitment, relatively fewer elvers and 0+ yellow eels have therefore been entering the rivers, leading to lower population density. Biomasses appear to have been maintained near to carrying capacity, however, by increases in modal sizes and shifts in sex ratios in favour of females eels (Section 2.18).

4.5 Population-Habitat Relationships and HSIs: Conclusions

Although there is currently insufficient data to develop and rigorously test a Habitat Suitability Index for eels which might be useful in monitoring and management, it appears that any such models would not be applicable because eels are too adaptable to different habitat conditions within rivers, let alone between adjacent or widely geographically separated ones. Furthermore, apparent relationships of density and biomass to habitat characteristics can be distorted because electric fishing only samples eels during daylight when they are sheltering in burrows and crevices. Trapping and netting are inefficient means of sampling and again would only give a restricted view of overall habitat preferences. Daily, seasonal and annual migratory behaviours also interfere with meaningful applications of any HSI approach

These conclusions agree with the criticisms aimed at HABSCORE model approaches for salmonids which are much less ubiquitous than eels, i.e. that models derived from one ecosystem can produce good predictions when applied to very similar ecosystems but are restricted in their application to less comparable habitats and other geographical regions. The same is true for logistic regression models of relationships between the distribution of long-fin eels (*A. dieffenbachii*) and 26 biotic and physicochemical variables for 99 sites in the Taieri River, New Zealand (Broad *et al.*, 2000). The probability of occurrence of long-fin eels declined with factors such as altitude and increased with certain types of riparian vegetation. The models were generally good at predicting from habitat factors whether eels would be present at a specific site, but applicability to other New Zealand rivers were relatively poor.

In conclusion, the only predictable relationships are that (a) eels will tend to be most common at sites where they can burrow or hide during the day (e.g. sites with soft substrate, suitable crevices or vegetation) and (b) that density tends to decline with distance from the tidal limits.

There are also major trends in eel population characteristics due to the geographical locations of rivers in England and Wales, as discussed in the next section.

4.6 Geographical Variations

Table 4.3a gives summary statistics for density and biomass at sites for different Regions from the HABSCORE and FCS databases. These show that mean densities decrease markedly, but biomasses vary much less, with distance from the Atlantic (noting that the majority of Wessex FCS sites are from upper reaches only in the Bristol Avon and Somerset Frome). There are no matching quantitative data on eels for Wales.

The results in Table 4.3b for South West Region and Essex rivers from Chapter 2 yield similar pictures of relatively lower densities in easterly rivers with smaller relative differences in biomasses. Density and biomass are, however, strongly significantly correlated ($p < 0.001$) in both the FCS and current R&D datasets.

Table 4.3 Descriptive statistics for sites with eel data for both density (eels 100 m^{-2}) and biomass (g 100 m^{-2}) by Region (pre-reorganisation) included in (a) the FCS and HABSCORE data bases and (b) from analyses in Chapter 2

(a) FCS and HABSCORE data base derivations

Mean ± SD and Range	South West (N=73)	Southern (N = 29)	Wessex (N=27)	Severn Trent (N = 79)	Anglian (N = 106)	Thames (N = 25)	ALL REGIONS (N = 339)
Density	15.52	4.48	3.42	2.03	1.37	0.62	2.03
± SD	2.07	1.43	1.27	2.93	1.60	0.58	3.99
Range	0.5 – 101.61	0.10–32.1	0.10–27.4	0.05– 20.57	0.08–9.52	0.10-2.1	0.05- 101.61
Biomass	-	442.25	559.71	352.60	280.07	247.34	346.49
± SD	-	649.76	530.98	309.03	312.66	235.66	401.34
Range	-	25.2-2952	20.3-2020	3.36- 1391.21	3.00- 1757.00	4.40- 914.6	3.00- 2952.00

(b) Data from Chapter 2 on South West and Essex rivers

Mean ± SD and Range	South West (N = 146)	Essex rivers (N = 171)	COMBINED DATA (N = 317)
Density	17.03	1.51	8.68
± SD	29.70	1.90	21.63
Range	0.3- 211.65	0.03-9.67	0.03- 211.65
Biomass	510.11	367.82	433.77
± SD	739.99	373.75	576.59
Range	9.44- 6448.96	4.83- 1745.11	4.83- 6448.96

The analyses in this and the preceding section confirm the view expressed in Section 2.28 that, given local site differences and productivities and factors such as migration barriers and gradients;-

(a) the intra-catchment distribution and population structures of eels tends to vary predictably with distance from tidal limits. This is mainly dependent on density-dependent factors in relation to carrying capacity which influence upstream migration and habitat segregation
(b) inter-catchment differences are mainly due to distance from the Atlantic recruitment pathways and the edge of the Continental Shelf where metamorphosis from the leptocephalus to the glass eel occurs.

4.7 Conclusions

The main conclusions from the current Chapter are that eels are very catholic in their habitat preferences and are tolerant of a wide range of environmental conditions. Overall, the evidence reviewed shows that habitat changes and fishing mortality have not been severe enough over the last 20 – 25 years to have significantly impacted recruitment, stocks, spawner escapement or fisheries. However, some studies in Chapters 2 and 3 indicate that recruitment declines may have affected stocks and population structures in some catchments in England and Wales, especially those with large areas of estuarine habitat sinks, such as the Piddle/Frome/Poole Harbour system (Section 4.4.4). It must be recognised, however, that our knowledge of stock-habitat relationships in estuarine, coastal and still freshwater environments is poor relative to that for rivers. Problems of efficiently sampling small eels also means that knowledge of their habitat preferences and any impacts of change are deficient. Oceanic factors affecting the migration of leptocephali and glass eels (and perhaps of silver eels) are possibly the most likely cause of recruitment declines. Issues of natural *v.* fishing mortality acting on all life stages are discussed in more detail in the next Chapter.

Athough biomasses tend to be similar in river sites of comparable productivity and, presumably, carrying capacity, there are clear tendencies for densities to decline and for average size and proportions of females to increase with distance from tidal limits and Atlantic migration pathways. However, because eels are so very adaptable to different habitat conditions, the uses of habitat suitability indices and modelling approaches are severely limited and their further development and use in setting management targets and monitoring cannot be recommended (Section 4.5). Graphing of density, biomass and size data against distance from the tidal limit is the simplest and most applicable approach to summarising distributions and populations structures in relation to habitat – and to highlighting spatial and temporal differences and anomalies. These issues are discussed further in relation to setting management targets in Chapter 7 and in relation to monitoring them in Chapter 8. The use of simple comparative indices based on density and biomass for general broad intra- and inter-catchment site comparisons is reviewed further in Section 8.8.5.

As eel stocks do not appear to have been severely affected by any recent changes, habitat restoration and enhancement do not have the same urgency as they do for salmonids, for example. It is unlikely that stocks could be cost-effectively enhanced by eel-specific measures, but that they can benefit from other general fisheries habitat improvements carried out by the Environment Agency and by the prevention of any losses, degradation or pollution of aquatic habitats. The only key habitat management issues specific to eels are that the impacts of barriers to immigration need to be minimised (especially if their effects are exacerbated by low flows and droughts), by removing artificial barriers or providing passes or alternative routes. Such management issues are further discussed in Section 7.5.

5. REVIEW OF NATURAL v. FISHING MORTALITY AND ESCAPEMENT

5.1 Introduction

This review was conducted to inform and supplement other project objectives and to underpin the specific objective *to design and cost a realistic research programme on the mortality of eels in freshwater* (see Chapter 6)

It was written in the context of the findings of Chapters 2 and 3, i.e. that glass eel recruitment has declined on average by ~ 55% from the peak years of 1972-82 to the low levels pertaining during 1983-99 - but that this, and eel fishing mortality, do not appear to have had significant widespread effects on stocks or spawner escapement in England and Wales. Fishing mortality might, however, exceed natural mortality in the future and the National Eel Management Plan proposed in Chapter 7 recommends that management reference points should be set, based principally on current population indices. If stocks then show signs of significant decline relative to these reference points, protective management actions would need to be instituted. However, more detailed knowledge of the relative effects of natural and fishing mortality are needed to help mangers decide whether any fisheries are having significant impacts and hence need controlling.

Many rivers examined support good biomasses of eels, often at what appear to be the local habitat carrying capacity, but baseline stocks show two key spatial trends. These are that density tends to decrease but average individual size (often associated with higher proportions of females in populations) with (a) distance of catchments from Atlantic larval migration pathways and hence recruitment and (b) distance of sites from the sea in individual catchments (Chapter 4). These differences correlate with the foci of exploitation, i.e. glass eel fisheries are concentrated in the south-west (especially the Bristol Channel/Severn Estuary area), yellow/silver eel fisheries in the east, in Southern and Anglian Regions. It would be expected that natural mortality of glass eels is enhanced by the longer migration distances from the edges of the Continental Shelf to the English Channel/southern North Sea coasts. Conversely, evidence reviewed in earlier Chapters indicates that the numbers of recruits in the lower reaches of some rivers could well exceed habitat carrying capacities, especially in shorter rivers in the south-west, but even in those off the major glass eel fishery estuaries. In these cases, density-dependent mortality would be expected to be an important limiting factor, probably exceeding the impacts of fishing mortality. The differential distribution within catchments is probably due to density-dependent stimulation of annual upstream migrations of elvers and small yellow eels (Chapter 4). It is also appears that the expression of phenotypic sex is density-dependent, with high densities causing relatively more eels to mature and emigrate earlier as males.

The specific **aims** of this Chapter are to clarify some of these issues further by assessing the relative impacts (with particular reference to density-dependent factors) of natural and, where it occurs, fishing mortality on;-

- Leptocephali and glass eels in Atlantic and offshore waters and hence recruitment to freshwaters
- glass eels/elvers in estuaries and hence riverine recruitment
- yellow and silver eels and hence survival to maturity

- silver eel spawner escapement
- silver eels during migration to and spawning in the Sargasso Sea.

The **objectives** are to clarify;-

- whether and where recruitment or growth overfishing are occurring or might occur in the future
- to what extent these might be causes of declining glass eel recruitment as opposed to natural factors
- the relative impacts of natural *v.* fishing mortality on stocks and on fisheries themselves

Knowledge of these issues will aid the setting of meaningful management targets in the future. Habitat availability and quality are not major issues, except where barriers inhibit upstream migration (Chapter 4). In Chapter 7 it is suggested that proactive management measures (e.g. stocking and passes) could be used to enhance stocks. Knowledge of natural *v.* fishing mortality is essential in this context, e.g. to decide what densities will result in maximum survival and growth rates.

5.2 Expression of Mortality

Mortality can be estimated in two principal ways;-

(a) in **percentage terms** between initial recruitment and some point in time or when eels reach a certain age or size (e.g. a commercial catchable size at > 30-35 cm) or at silvering and emigration.

(b) via estimation of **instantaneous mortality rates** over time, calculated as:-

$$Z = \frac{(\ln Nt_1 - \ln Nt_2)}{(t_2 - t_1)}$$

where Nt_1 = number of eels of a particular age group at time t_1 and Nt_2 = the number at time t_2, usually after an interval of one year. To include all possible components, Z can be considered to comprise:-

$$Z = M + F + E - I$$

where M = natural mortality, F = fishing mortality, E = losses due to emigration and -I = gains due to immigration (e.g. to new recruits migrating into a population or to silver eels entering a population, even if temporarily, during breeding migration)

5.3 Assessment of M and F

Input-output, cohort and mark-recapture analyses and life-stage models have been used in relation to eel populations, often in combination with one another;-

- **Input-output estimates of whole-life mortality** between natural recruitment and escapement can be derived for waters where it is possible to account for all immigration into a system, subsequent F (e.g. from catch data) and all emigration out (e.g. via silver eel traps). Some such studies are in progress (e.g. see Pederson, Table 5.1) but most of this type have been of closed ponds or lakes because they can be stocked with known numbers and sizes of eels, survival of which can be determined later by netting or drain-down or via eel traps at the outlet (see Table 5.1). Furthermore, resources have been more readily available for such studies because of the pressures to determine the cost-benefits of stocking *per se* (Knights & White, 1997).

- **Direct determinations of life-stage and size or age-specific mortality** can be made by stocking followed by periodic resurvey. This has been done in ponds and in eel-free rivers (e.g. by Berg & Jørgensen (1994) using electric fishing), sometimes involving mark-recapture (see below). More recently, large-scale and long-term studies of the Fremur and Vilaine catchments in Brittany have started which involve estimating recruitment (via traps at the head of the estuary) and subsequent mark-recapture, electric fishing and up-stream trapping surveys (Feunteun *et al.*, 1998; Briand *et al.*, 2000, unpublished report).

- **age-structure cohort analyses** have been applied to natural populations, e.g. Vøllestäd & Jonsson (1988) used standard age-structured models in studies of populations in the Imsa river system in SW Norway, in addition to input-output approaches. Other studies have involved estimating age from length in commercial catches and the use of catch–curve analyses. Svedäng (1999), for example, used this approach to estimate Z values for fished and unfished areas along the west coast of Sweden, leading to estimates by subtraction of 0.23 for M and 0.31 for F in eels between 37 and 65 cm (Table 5.1). Sparre (1979) applied similar approaches to eel fisheries of the German Bight and produced an estimated F of 0.2 for the most-exploited sizes (Table 5.1).
  The general applications and accuracy of age-structured approaches are, however, limited because of the inaccuracies inherent in otolith ageing (especially of older eels, Appendix C.2.4) and back-calculation of growth rates, variable growth rates in eels (including males *v.* females) and the problems of sampling smaller/younger eels. Immigration and emigration are further confounding factors.

- **length-structure cohort analyses** have been applied by, for example, Dekker (1998) to the IJsselmeer fishery, via a length-length-transition matrix approach to compensate for variable growth rates. Naismith and Knights (1993) used a length-sex compartmental model to compare fished and unfished reaches of the Thames Estuary (see below)

- **mark-recapture** studies can be used to estimate population numbers and changes due to natural *v.* fishing mortality, as discussed further in Section 5.4.3. However, the effects of marking on behaviour and survival are unpredictable and recapture rates often very low (Knights *et al.*, 1996). Like other methods, interpretations are complicated by variations in sampling efficiency, difficulties in accurately quantifying commercial fishing effort and catches and in distinguishing changes in populations due to immigration and emigration. Migration can involve long-distance upstream movements (occurring after glass eel metamorphosis and throughout the yellow eel growth stages) and more localised short-term or seasonal movements up- and downstream (possibly between fresh water and estuaries) and, finally, the emigration of silver eel spawners (Tesch, 1977).

Table 5.1 Natural and fishing mortality values for European eels at different life stages and in closed stillwaters, rivers, estuaries and coastal waters

Habitat Location	Natural mortality %	M	Fishing mortality %	F	Comments	Reference sources
Stillwaters and rivers						
IJsselmeer (Netherlands)				1.0	Yellow and silver eels heavily exploited	Dekker (1999a)
Lough Neagh (Ireland)	82%		75%		Stocked system heavily exploited	Knights & White (1997a)
Shannon lakes (Ireland)			20%		Low exploitation rates	McCarthy *et al.* (1994a); Moriarty (1987,1990c)
L.Derg &Ree (Ireland)	68-74%	0.3-0.38			Low exploitation rates	McCarthy *et al.* (1994b)
Swedish lakes	<90%				Stocking study	Wickström *et al.* (1996)
Finnish lakes	28-73%				Stocked, variable exploitation	Pursianen & Toivenen (1984)
Byelorussian lakes	95%				Stocked	Leopold (1976)
Small lake (Denmark)	41-78%	0.13-0.04			Stocking study after seven years	Pedersen (1999)
Ponds (Netherlands)	30-77%	0.17-0.65			Fingerling stocking experiments	Klein Breteler (1994)
Ponds (Denmark)	89%				Stocking study over I year	Dahl (1967)
Coastal lagoons (Italy)		0.25-0.72			Commachio stocked lagoons	Cicotti, in Moriarty & Dekker (1997)
Rivers in E. England	75%				From age-cohort analyses	Barak & Mason (1992)
River Imsa (Norway)	50-84% (Mean 73%)	0.088-0.225 (Mean 0.167)			Recruitment & escapement measured over 13 years	Vøllestad & Jonsson (1988)
River Gudena (Denmark)	77-82%				Post-stocking studies	Berg & Jørgensen (1994)
Stocked streams (Denmark)	97%	0.23-1.79			Post-stocking studies	Pederson (in Moriarty & Dekker, 1997)
Unstocked streams (Denmark)	60%				Post-stocking studies (still in progress)	Pederson (in Moriarty & Dekker, 1997)
Danish stream		0.36-0.65			From age-cohort analyses	Rasmussen & Therkildsen (1979)
River Tiber (Italy)		0.8			From age-cohort analyses	Cicotti, in Moriarty & Dekker (1997)

Table 5.1 (cont.)	%	M	%	F	Comments	Reference source
Coasts and estuaries						
Coast of W. Sweden		0.23	96.5%	0.16-0.42	Coastal fykenet fishery data	Svedäng (1999)
Coast of S.E. Norway		0.04-0.46		0.02-0.08	Coastal fykenet fishery data	Vøllestad (1988)
German Bight				0.2	Age-structured cohort analysis	Sparre (1979)
Thames Estuary (England)	>95%	Z = 0.5-0.7		Z = 0.5-0.7	Compartmental length-structured cohort model, comparing exploited v. unexploited zones	Naismith & Knights (1993)
Glass eel fisheries						
Severn, England	>99?		<1.0%		Mark-recapture	This report
Vilaine (France)			96-99%		Mark-recapture & trapping below barrage	Briand & Fatin (1999)
Small river, Nova Scotia	> 99%	>0.99	30-50%		Mark-recapture & trapping	Jessop (2000)
Creek, NW England			>70-90%		Trapping study	Mower (1998-99)
Small river, Taiwan			44-75%		Mark-recapture	Tzeng (1984)
Silver eel fisheries						
Coastal waters, Sweden	80%	0.18	27%		Mark-recapture	Wickstrom & Hamrin (1997) and Wickstrom *et al* (1996)
Limfjord, Denmark			43-88%		Mark-recapture	Pederson in Moriarty & Dekker (1997)

- **mortality-per-life-stage models** have been developed to cover exploited components or the whole life-cycle. Dekker (1999) has done the former for all European waters *versus* those in the Biscay area where glass eel recruitment is highest and those in other waters where recruitment has been lowest (Section 5.4.7). Knights (1999a) has also developed a life-stage model to compare the relative impacts of M and F for England and Wales, as discussed further below and in Appendix A.

The following detailed discussions deal with studies of M and F and models relating to different life stages and exploitation, concluding with an overview of whole life cycle mortality, leading to suggestions in Chapter 6 for further research in England and Wales.

5.4 Natural and Fishing Mortality during Different Life Stages

5.4.1 Natural mortality during the leptocephalus and early glass eel stages

Evidence indicates that eels may breed all year round in the Sargasso Sea area (20–30° N, 50–75°W), then pelagic leptocephali drift passively or actively via the advective Gulf Stream and North Atlantic Drift towards Europe (McCleave *et al.*, 1998). Sources of nutrition are unknown but suggestions include dissolved or particulate organic matter, faecal pellets of copepods or Appendicularian thecae (e.g. see Westerberg, 1990). The migration could take three years, although some workers argue the leptocephalus stage only lasts about six months and the subsequent metamorphosed glass eel stage on the continental shelf only three months (e.g. see Desaunay & Guerault, 1997). Glass eels move landward, probably using localised current systems and selective tidal stream transport, but during this time they do not appear to feed.

Natural mortality during the leptocephalus marine larval stage is impossible to quantify accurately because of our lack of knowledge of breeding success in the Sargasso Sea and numbers of glass eels surviving to colonise continental waters. Tesch, however, estimated that in the 1970s, natural annual mortality between spawned egg and leptocephalus (stage I-IV) was 99.1% and that between leptocephalus (stage I-IV) and glass eel (stage V-VI) was 95.6% (Tesch, 1980). In the absence of firm data, his estimates were based upon:-

- the assumption that the breeding stock comprised 30 x 10^6 (15,000 t) migratory females, each producing 1 x 10^6 eggs, i.e. 30 x 10^{12} eggs in total

- extrapolating from the mean density of larvae per unit volume of water sampled during sea cruises in 1971-7 in the N. Atlantic, he estimated the total number of leptocephalus larvae (stage (I-IV) on the European continental shelf to be of the order of 2.7 x 10^{11}

- the assumption that the total final glass eel recruitment to continental waters was 1.2 x 10^{10} individuals (4000 t, including a commercial catch of 2000 t)

Although Tesch did not estimate mortality during continental growth stages, if glass eel recruitment was 0.6 x 10^{10} (i.e. recruitment minus his estimate of 50% fishing mortality) and escapement of males and females was 60 x 10^6 (i.e. twice the estimated female escapement), total mortality would be > 99.99%. This compares to estimates of mortality between metamorphosed elver and silver eel stage made in the 1990s of ~75% (Moriarty & Dekker, 1997). This would imply a very high proportion of the mortality occurs between arrival of glass eels on the Continental Shelf and the early elver/juvenile stages. Although the glass eel life stage may be relatively short, density-dependent mortality would be expected to increase markedly as they become concentrated during migration towards estuaries as water temperatures rise above 6°C. Furthermore, they have to spend one or two weeks completing metamorphosis to the pigmented elver in the oligohaline zones before being able to resume active migration upstream into freshwater, when temperatures rise above 10 – 11°C (e.g. see Gascuel *et al.*, 1995; White & Knights, 1997). During this time, they would again be prone to high density-dependent mortality.

To put these natural mortality values into context, other highly fecund marine species with small early larvae have similarly high rates. For example, Kristiansen *et al.* (2000) have shown by mark-recapture that M for 0-group cod larvae of mean length 8 cm was >98%

during their first three months in coastal waters, falling to ~ 25% for 12 cm larvae over a similar time period. These differences are explained by (a) as the numbers of cod decline in an area, so do the impacts of density-dependent mortality and (b) the larger the cod, the less vulnerable they become to small predators.

Tesch's estimates for eels for the 1970s are very speculative and cannot be validated, particularly given the lack of knowledge of silver eel oceanic migration, survival and reproductive success. However, the lower density-dependent mortality and higher growth rates in more productive continental fresh, estuarine and coastal waters compared to those in the ocean accord with the perceived evolutionary value of the catadromous life strategy of eels (e.g. see Gross *et al.,* 1988). Also, the high mortality rates during marine larval stages would accord with the reproductive strategy of eels, where females have evolved a very high compensatory fecundity. For example, silver eels sampled from the Langham trap (R. Stour, Dorset, England) in 1998 show potential egg production of 0.5-3 million for 50-75 cm eels, rising to ~ 5 million for 95 cm eels (Knights & Lindsay, 1999, unpublished).

5.4.2 Impacts of natural and fishery mortality on glass eel recruitment

Tesch (1980) estimated that the annual commercial catch in the 1970s was 2000 t (and fishing mortality was about 50% of the total recruitment). This appears high compared to the maximum annual catch estimates of 1200 t in 1979 (Desaunay & Guerault, 1997) and that of ~ 920 t in the early 1990s (Moriarty, 1996). The demand in the 1970s would have come from the direct consumption and restocking markets, the demand for aquaculture seedstock increasing from the 1980s onwards (mainly for European farms, with extra demands for culture in the Far East in the late 1970s and again in recent years). It is possible that increasing exploitation rates could help explain the declines in recruitment and continental stocks since the late 1970s-early 1980s, although recruitment has tended to stabilise during the 1990s (Section 2.2.1). These correlate with declines in the marine larval stock in the Bay of Biscay between 1971-9, 1982-5 and 1985-9 surveys (Tesch *et al.*, 1986; Tesch & Niermann, 1991).

An alternative view is that increases in natural mortality during the earlier larval oceanic migration phase could be a major cause of declines in recruitment (Knights *et al.*, 1996). These could perhaps follow cycles, as suggested by the long-term data series presented in Fig. 2.2. These show that current levels of recruitment are similar to those pertaining between the 1920s to 1950s, with increases to peaks in the late 1970s. The subsequent declines are mirrored by those seen in the abundance of *A. rostrata* in the US and Canada, suggesting common factors may be acting during the oceanic life stages of both the North Atlantic species of eel (Richkus & Whalen, 1999). Desaunay and Guerault (1997) reviewed evidence to show that not only have numbers of glass eels entering various European estuaries declined, but so have mean lengths and weights – by up to 14% and 35% respectively in the 1990s compared to maxima in the 1920s and 1980s. Such synchronous changes throughout Europe (and similar ones occurring in American eel, *Anguilla rostrata*) support the view that large-scale temperature, circulatory or other changes have occurred in the North Atlantic and these could have impacted larval food sources, growth and survival. Correlations are also seen with changes in populations and distributions of plankton and other components of N. Atlantic communities (Knights *et al.*, 1996). Even in the long-established Severn (England) fishery, the Severn Fishery Board Annual Report for 1916 stated 'numbers (of elvers) showed a great falling off as compared to 18 or 20 years ago'. It can also be argued that more recent exploitation cannot explain the declines in eel stocks first noted in Scandinavian and Baltic fisheries and in Italy in the late 1960s (e.g. see Devall, 1998). Although Fig. 2.2 suggests glass eel recruitment increased through the 1960s towards peaks in the late 1970s, this may not have been a common to all European waters. Recruitment may have actually decreased to those waters furthest from the Atlantic migration routes in the 1950s – 1960s, even in east coast rivers of England (Devall, 1998).

In the Far East, it also appears that recruitment of glass eels of *Anguilla japonica* varies cyclically, independent of fishery pressures (Knights *et al.*, 1996). The marine life stages are the most vulnerable to natural mortality factors (especially the glass eels as they begin to congregate prior to migration into freshwaters), but a species as fecund as the eel should easily be able to compensate for high mortality during the freshwater growth stages (Desaunay & Guerault, 1997). Furthermore, declines in density of yellow eels would be expected to encourage changes in sex ratios towards the production of relatively more and larger females of higher fecundity. At the moment, however, our lack of knowledge of the oceanic larval stages and possibly key oceanographic factors means there is no unequivocal evidence for cause-effect relationships to help settle the continental *v.* marine and natural *v.* fishing mortality arguments. Research into these issues is needed, but no recommendations are made in the following chapter because this is beyond the remit of the R&D project

5.4.3 Glass eel fishing mortality

Mark-recapture techniques could help clarify the total number of glass eels immigrating into a river, the numbers caught and the numbers escaping (Knights *et al.*, 1996). This involves marking and releasing glass eels downstream of an estuarine fishery and then assessing the numbers caught by fishermen and the number escaping upstream and then applying mark-recapture estimation techniques. However, many methodological problems are involved, especially finding ways of reliably marking eels and then resampling (see examples given below). The latter problems tend to invalidate the basic assumptions underpinning the use of mark-recapture estimation techniques such as the Peterson method (White & Knights, 1994). Separating different waves of migrants using selective tidal stream transport on major spring tides is a further problem and some migrants drop back seawards. It is also necessary to distinguish between the initial runs of true glass eels and the later up-river immigration of metamorphosed elvers, as discussed further below.

Few mark-recapture studies on glass eels have actually been carried out. White & Knights (1994) discussed an unpublished study conducted by Bristol Channel Fisheries in 1991. Exact details are not available but it is believed that two batches of glass eels (one of 5 kg, one of 10 kg, i.e. about 45,000 fish in all at 3000 glass eels/elvers kg^{-1}) were dyed with neutral red and released into the Bristol Channel/Severn Estuary, England. Using the Peterson mark-recapture estimation ($N = MC/R$), M (number of eels marked and released) = 45×10^3, C (total number of eels captured and brought into the elver station) = 23.4×10^6 (from commercial catch estimates of 7.8 t in 1991, White & Knights, 1994) and R = 225 (0.5% of M), then $N = 4.7 \times 10^9$ glass eels/elvers. At 3000 individuals kg^{-1}, this is equivalent to about 1.41×10^3 t and compared to the estimated commercial catch of about 7.8 t, suggests a very low fishing mortality of < 0.5%. The study did not, however, adequately meet many of the criteria for accurate prediction using the Peterson method, especially constancy of fishing effort and unknown losses due to emigration and immigration out of the fishing zone.

White and Knights (1994) also assessed trap catches on weirs at the tidal head of the Severn and Avon, upstream of the fishery. These sites are not favoured by commercial fishermen because of low catches. Few eels were caught in traps until several months after the main glass eel runs had commenced and these comprised actively migrating post-metamorphosis pigmented elvers and older juveniles. Numbers were < 1% of the estimated commercial catch and four orders of magnitude less than the total estimated run. Given the imprecision of these studies, it still appears that that natural mortality was very high in the long run up the estuary to the tidal head (some 43 km between Gloucester and Tewkesbury). This would have been exacerbated by density-dependent effects as eels became concentrated by the narrowing of the estuary and by the time-lag during metamorphosis. Taken in conjunction with the glass eel study, the results strongly suggest that fishing mortality was relatively very low in comparison with natural mortality. Furthermore, resurvey data (Chapter 3) and the studies of Aprahamian (2000) indicate that the large variations in glass eel recruitment and in fishing effort since the early 1980s appear to have had no significant impacts on stocks in the rivers off the Severn Estuary or above the tidal limit. It is therefore probable that elver/juvenile recruitment has still been more than adequate to maintain riverine stocks.

A few mark-recapture studies have been carried out on shorter estuaries than the Severn but still with high levels of commercial exploitation. One such study is that in the Vilaine (Brittany, France), which supports an intensive fishery in a 2 km zone truncated by the large d'Arzal barrage. Fishing is carried out using two 1.2 m diameter nets suspended by poles from the sides of boats. Large numbers of boats circle around the estuary on high spring tides, catching eels accumulating just below the barrage dam. Briand *et al.* (2000) estimated that during one night with 100 boats fishing, the water volume below the dam was filtered 13 times. Dyed eels have been released below the fishery and the numbers escaping estimated from those captured by an eel pass trap on the barrage (Briand *et al.*, 2000). Escapement appeared to be very low, the number of migrants subsequently moving actively towards freshwater being estimated over 1996-1998 at only 0.3 - 4% of the commercial catches (i.e. figures similar to those in the Severn and Avon study). However, results may have been confused by abnormal migratory behaviour of marked eels (many being found to drop back downstream) and low trap efficiency. Furthermore, as in the Severn-Avon study, traps caught actively migrating elvers, mainly arriving after the principle fishing season, rather than freshly-arrived glass eels. Therefore, as discussed earlier, natural mortality of glass eels could be high and trap catches cannot adequatley distinguish between this and the impacts of fishing mortality. Recruitment into the Vilaine catchment was effectively cut off by the dam after its construction in 1970 but the fish pass now allows entry. Eel distributions and populations are now building up and recent studies, via electric fishing and trapping, suggest that the numbers of 0+ elvers and small > 0+ eels entering may be sufficient to colonise the catchment to a reasonable level.

A study involving a system with much smaller runs than those seen in SW England and the French Biscay fisheries but subject to heavy exploitation was carried out by Tzeng (1984). This was carried out on *Anguilla japonica* in the Shuang-Chi River in northern Taiwan. It involved the marking and release of 4225 glass eels and examination of recaptures by 70 commercial fishermen. The total number of immigrants was estimated to be about 65,400 individuals (< 200 kg) and the total commercial catch was 38,661 individuals. Thus the total exploitation rate was F ~ 60%, ranging between 44 and 75% over seven waves of immigration. F appears exceptionally high but so was the fishing effort, with 70 men hand-trawling discrete shoals of glass eels as they congregated in coastal waters and in the short (5 km) estuary. This fishery is very different from those operating in larger, deeper and more open European estuaries, using dip nets or nets suspended from boats – and where runs are measured in tonnes and tens of millions of individuals rather than kilogrammes. Unfortunately, there are no data on up-stream populations in the Shuang-Chi River to show whether sufficient late-arriving elvers recruited to maintain stocks.

Jessop (2000) studied recruitment of *A. rostrata* in another relatively small system, East River (Novia Scotia, Canada), via mark-recapture and trapping. The lower river contains good stocks (6-12 eels $100m^{-2}$ of eels up to 20+) and annual glass eel runs over 1996-8 were estimated by mark-recapture to be between 0.43 to 2.39 million. Migrants in May-October took > 6 weeks on average to cover the 1.3 km between study sites. Overall mortality rates were estimated at Z = 0.9945 by trap count and 0.9968 by mark-recapture, similar to those discussed above for the Severn Estuary. Daily Z values were estimated at 0.0612 and 0.0675 respectively. These are larger than those of 0.0107-0.0233 estimated during the 3 months following stocking with glass eels/elvers of a small Danish stream (Berg & Jørgensen, 1994). Low pH (4.7-5.0) may be a limiting factor in East River, but predation and, especially, cannibalism by large resident eels was probably important, exacerbated by low elver growth rates. Densities were high at about 4.7 glass eels/elvers m^{-2}, accentuating density-dependent mortality. Jessop (2000) has also estimated that exploitation by a two-man dipnet fishery in East River varied between 30-52% over the 1996-8 seasons. The data possibly indicate relative exploitation impacts increase with decreasing size of run. Supporting data could not be gained in 1999 as fishing was not commercially viable because of the fall in market prices. Jessop (2000) estimated that glass eel supply greatly exceeded the carrying capacity of the East River and other low pH Maritime Province rivers. He suggested that a fishing mortality of 30-50% could be biologically justified in that it would not significantly affect recruitment into freshwater, especially given that F would lead to compensatory reductions in density-dependent M.

High prices for glass eel over the last few years have encouraged exceptionally high fishing efforts in small, previously unexploited, rivers in Europe. Licence sales have increased in England and Wales and fishermen have been working new sites, according to reports from many fisheries officers. 'New' sites are often below sluices or other barriers to wetlands where immigrants congregate in sufficient numbers to yield a catch worth the fishing effort and the costs of transport to collecting stations. These could potentially have very large impacts on local recruitment.

An example where there are some quantitative data is Red Barn Dyke/Quicksand Pool, just downstream of Leighton Moss SSSI on the fringes of Morecombe Bay between Silverdale and Warton, S. Cumbria (Environment Agency NW, 1997, unpublished internal report; Mower, 1998, 1999). Leighton Moss is a SPA (EC Birds Directive) and Ramsar Site managed by the Royal Society for the Protection of Birds. This 125 ha wetland is of particular interest because it is one of the few extensive shallow *Phragmites* wetlands in Cumbria, it is the only site for breeding bittern outside of East Anglia and it is also one of the few known breeding sites for otter in Lancashire. Although bittern appear to be generalist feeders in shallow (< 20 cm) wetland margins, small eels are a particularly important component of the local fish community in littoral and inner reedbed areas at Leighton Moss, more so than in many other RSPB wetland reserves, especially those in eastern England (Perrow *et al.*, 1996). Eels can easily enter from Morecombe Bay and mean densities in littoral reed zones in 1998, up to 12 m into inner reedbeds, in dykes and in open shallows were about 15, 6, 7.5 and 3 eels $100m^{-2}$ respectively. Comparable biomasses were 327, 212, 126 and 65 g $100m^{-2}$. The great majority of eels were small (~ 60% < 20 cm), larger ones preferring deeper dykes. Eels are also a key component of the diet of otters and studies of spraints at the reserve have shown eels to comprise 95% of the diet. A three-board drop-sluice controls water levels in the reserve and eel passes have been operated since 1997 to encourage entry to sustain eel populations. A trap has been included in the pass to assess levels of recruitment and by comparisons of catches with visual observations of eels passing directly over the pass, capture efficiency has been estimated at about 40% (Mower,1999). Commercial fishermen have been fishing glass eels for a number of years (being licensed since 1996) as they accumulate about 1 mile downstream below the tidal flap sluice at Red Barn Dyke, in the Quicksand Pool drainage channel connecting to Morecombe Bay. Fishermen have negotiated exclusive fishing rights from the local landowner. According to catch returns, the minimum commercial catch on the main February – April spring tides in 1999 was about 30 kg (87,550 eels) (Mower, 1999). This compares with an estimated recruitment to the Reserve of about 900 eels during this time, i.e. an apparent escapement of < 5% during the main glass eel runs. Overall recruitment from trapping between February and October was estimated to be between 15 000 and 45 000 eels, i.e. relying mainly on later-arriving pigmented elvers, plus juveniles up to 15 cm long moving upstream from Morecombe Bay. The overall annual fishing mortality was therefore estimated at the time to be about 50 – 85% (Mower, 1999). The need to control this particular eel fishery has been noted as an issue (Issue 25) in the Environment Agency South Cumbria 1998 LEAP and Action Plan. The RSPB has recently been successful in purchasing the fishing rights from the landowner and the fishermen have been cooperating in studies of recruitment. They have voluntarily stopped fishing on a number of key high tides in the spring of 2000, but this has not resulted in any significant increases in eels caught in the sluice trap (David Mower, personal communication). This means that the initial assumptions that the lack of immigrants in spring were virtually all due to fishing mortality is not tenable. As discussed earlier, it is possible that recruitment into freshwaters is primarily dependent on pigmented elvers and not glass eels *per se*. An unpublished report for the RSPB by B. Knights indicates that elver/juvenile recruitment is sufficient to sustain eel populations in the Leighton Moss wetlands, independent of glass eel fishing mortality. Densities and biomass in more open waters appear quite low but length class frequency distributions show most eels mature as male < 40 cm. This is indicative of a relatively high density stock and it may be that sampling efficiency was low in more open waters. Further studies are in progress to assess relationships between fishing effort and actual recruitment.

In conclusion, despite methodological problems and limited evidence, it appears that natural mortality of glass eels is relatively high and exacerbated by density-dependent effects during the time between their entry to estuaries and the metamorphosis and migration of elvers in to freshwaters. Density-dependent M would be exacerbated if migration is delayed in long estuaries or inhibited by barriers, e.g. as in the Severn and Vilaine respectively. Local factors such as low water temperatures and spates exacerbate the situation in different seasons (White & Knights, 1994). Predation by fish and birds and cannibalism by larger eels are probably of major importance. Starvation can also be a factor, as the glass eels are living on stored energy reserves until they begin first feeding and complete metamorphosis to the fully pigmented elver, when active migration can commence. In contrast, fishing mortality may be relatively high in smaller estuaries and creeks where runs can be exploited more efficiently, especially where sluices or other barriers cause migrants to become concentrated. Even small catches may be economically viable if prices are high enough.

Small systems (such as Red Barn Dyke/Quicksand Pool, and similar ones, such as those below sluices on the Romney and N. Kent Marshes and Pevensey Levels in southern

England) may need protection from overfishing, e.g. by designating them as closed areas. This may still be justified by application of the precautionary principle but the studies reviewed above suggest that in reality, glass eel fisheries may have only minor, if any, impacts on recruitment into catchments.

5.4.4 Natural and fishing mortality of yellow and silver eels

There have been relatively few detailed studies of natural post-elver mortality in rivers because of the methodological and resource problems discussed earlier. The majority of estimates come from long-term studies of stockings in closed stillwaters (Table 5.1). Results vary because of differences in methodology, stocking density, recapture efficiency and unknown losses to emigration and immigration.
As discussed in the preceding Section, elver and early juvenile stages are particularly vulnerable to predation and cannibalism because of their small size, especially when they are present in high densities after initial arrival. For example, Rasmussen and Therkildson (1979) studied an eel population in a small Danish stream (the Køge-Lellinge å) and from otolith-aged electric fishing catch samples, estimated M to be 0.83 for 0+ to 3+ (river age) eels (10-18 cm) and 0.36 for 4+ to 6+ eels (26-34 cm). It is possible, however, that the population was affected by unknown emigration and immigration and by damming of the river late in the study. The authors estimated M to be 0.60 for 7+ and 8+ (38-40 cm) and 0.50 for 10+ to 20+ (38-54 cm) eels, but losses from these size classes would certainly have been distorted by emigration of male and silver female eels. Emigration during all life stages is a major drawback to interpretation of all the riverine mortality estimates in Table 5.1.

From such evidence for higher mortality in the earlier stages, it is commonly thought that it is more cost-effective to stock eels grown on in small ponds or warm-water aquaculture rather than glass eels/elvers, despite the feed and other costs involved in on-growing (Knights & White, 1997). However, Klein Breteler (1994) found minimal differences in survival and growth rates between elvers and juveniles in short-term pond studies. Key long-term post-stocking studies in Sweden and Denmark are still in progress, but the results in Table 5.1 show M varying between 30-97% over the first 7-14 years.

Long-term estimates for annual fishing mortality from catch-landing data (Table 5.1) are only available for the IJsselmeer (Netherlands), Lough Neagh (N. Ireland) and Shannon lake system (Ireland). F should also be high in Italian lagoon systems, if they are well managed, because they are closed systems where yellow and silver eels can be fully exploited, following natural recruitment and/or stocking (De Leo & Gatto, 1995). F is high in the IJsselmeer and L. Neagh, but these are relatively closed and poorly-recruiting systems but are subject to unusually highly exploitation. Overfishing in the IJsselmeer (via fyke netting and trawling) occurred after closure of the Zuider Zee from the Waddensea, exacerbated by declining glass eel recruitment (Tesch, 1977; Dekker, 1989, 1991). The L. Neagh fishery is run as a cooperative and is dependent on stocking with elvers from the River Bann, plus imports. Yellow eels are exploited by long-line fishermen and then emigrant silver eels are trapped in the River Bann. Rossell (1999) compared stocking and catch data and estimated that smaller males can take about six years to mature but that main catches of larger eels occurs after about 17 years and that some limited natural recruitment occurs (i.e. some immigrants by-pass the river traps). Rossell also notes that competition with aquaculture supplies is threatening the economic viability of the L. Neagh fishery.

No closed waters in England and Wales have been subject to such high levels of exploitation. Few are large or productive enough or receive sufficient natural recruitment or stocking. The unusually high levels of exploitation and the use of stocking in the examples given above have also come about because they have unique historical traditions of eel fishing. These have contributed to local economies and become part of the way of life of local people, a situation not seen nowadays in England and Wales.

In open estuarine and river systems, immigration (and on-growth of uncaught eels) tends to occur in any fished river to replace the eels lost (Naismith & Knights, 1990, 1993). Experienced eel fishermen know this as a constraint to the intensity and frequency of fishing effort. It has also thwarted eel culling, such as that attempted by removal of all eels electric fished in Tadnoll Brook (Dorset) by Mann and Blackburn (1991).

Using a size-class cohort-analysis rather than a fishery-dependent approach, Naismith & Knights (1990) have constructed a compartmental model to assess the potential impacts of the fyke net fishery in the Thames Estuary, England. They estimated Z for each 3 cm length class of eels > 30 cm and compared average values (0.37) with those for immature and mature males and females. The lack of clear differences in Z between exploited and unexploited reaches indicated that fishing mortality effects were low and masked by M, seasonal variations in distribution and, especially, by emigration, immigration and local short-term migrations. Mean growth rates in the Estuary were quite high (4.35-6.07 cm $year^{-1}$) and at these rates, average derived Z per year would be about 0.5-0.7. Naismith & Knights (1990) further estimated that mortality between the elver and the 34-41 cm size stage was > 95%. These values appear high compared to published values for closed waters (Table 5.1) but, like the results from Køge-Lellinge å (Rasmussen & Therkildson, 1979), are very strongly influenced by losses to upstream emigration (E) and to seaward immigration (I).

5.4.5 Coastal yellow and silver eel fisheries

No suitable data have been located in Chapter 2 to assess coastal eel fisheries in England and Wales, although there are some for Scandinavian waters and the German Bight (Table 5.1). In such areas, localised shallow coastal waters and semi-enclosed fjords form productive habitats that are easily fished using fyke or pound nets. Exploitation rates can therefore be relatively high, as shown for example in the coastal fisheries along the west coast of Sweden by Svedäng (1999) (Table 5.1). Although based on many assumptions and indirect calculations, F was estimated to be > 96% between the 37 and 65 cm size classes and from length-class frequency relationships, growth-overfishing appeared to be occurring. Comparability of the fished and unfished sites and losses to emigration are especially questionable. The study did, however, show that recruitment had been sufficient to maintain CPUEs between 1976-97, probably because of the favourable position of the coast the relative to Atlantic migration pathways (Westerberg, 1999).

Such F values could possibly occur in England and Wales, but there is a lack of comparable sheltered sites and a tradition of coastal fishing, except perhaps in the Solent where one major vessel fishes with large fyke nets. Pair-trawling could also have high localised impacts, e.g. in the outer Thames Estuary. Even then, the overall impacts nationwide on spawner escapement would be expected to be minimal because of the relatively small number and localised nature of such fisheries. Even on the Swedish west coast, overall fishery impacts would be expected to be relatively low as there are only eight localised fisheries along well over 250 km of highly indented coastline (Svedäng, 1999).

5.4.6 Mortality *v.* escapement of silver eels

The EC Concerted Action could not find any clear data specific to silver eel fishing mortality and impacts on escapement of breeding stock (Moriarty & Dekker, 1997). It concluded, however, that capture efficiencies were generally low, principally because of the difficulties in completely blocking a migration pathway in a river (especially during spate runs) and in open estuarine and coastal water. Even the series of traps in the R. Bann below Lough Neagh were estimated to be only 60% efficient. In open waters off the Baltic coast of Sweden, mark-release studies reviewed by Wickström and Hamrin (in Moriarty & Dekker, 1977) showed mean recapture rates by commercial fishermen to be between 35 and 49%, but with figures as high as 69-76% in some experiments. This implies a potentially high level of exploitation is possible using pound nets at certain traditionally important sites, especially those where silver eels become concentrated as they migrate through the narrow seaways out of the Scandinavian rivers and Baltic Sea.

As discussed in other Chapters, there are very few silver eel traps still worked in England and Wales and current levels of exploitation appear to be very low. Effort per year and accuracy of records vary too much to draw any useful conclusions about historical changes. One commercial fisherman working a trap on the Dorset Stour has been catching reasonable numbers over the last few years but an unknown number of silver eels by-pass the trap via a nearby weir, varying with river discharge.

The consensus from the EC Concerted Action was that escapement from silver eel fisheries was in the order of 90%, except in Italian extensive aquaculture systems where lagoons are designed to retain all eels. It was also concluded that current European escapement, unless unprecedented increases in exploitation efforts and efficiencies occur, are sufficient to achieve a Minimum Biologically Acceptable Level of spawner escapement. However, the 1999 ICES/EIFAC Working Group on Eel meeting concluded that the precautionary approach should be applied and ACFM have recommended more proscriptive actions, as detailed in Chapter 7.

5.4.7 Whole-life mortality models

The data and models discussed above show there are important differences in fishing mortality throughout Europe between (a) heavily and lightly exploited systems and (b) principal fishery targets (glass eel recruits, yellow eel growth stages or silver eel escapers). These differences need to be taken into account when proposing management initiatives and targets to sustain local fisheries. In relation to the European stock as a whole, however, mortality throughout the complete life cycle has to be considered to assess the relative impacts of M and F.

Given the problems in accurately assessing M *v.* F and the variable estimates shown in Table 5.1, eel biologists in the EC Concerted Action Programme accepted that natural mortality was generally in the order of 75% over the continental life span in all European waters (Moriarty

& Dekker, 1997). They also concluded that eel fisheries tended to have their main effects locally, in systems most amenable to high levels of exploitation.

Dekker (1999a) has subsequently proposed a model for all continental life stages, based on the estimated natural mortality of 75% spread evenly over 18 years of continental life. In the absence of any better evidence, M was assumed to be constant at 0.21 per annum during every catchable life stage. This cannot be true, according to the data presented in Table 5.1 and discussions above which indicate that mortality (especially density-dependent mortality) is greatest early in freshwater life. The model then estimates M over the specific duration of each life stage (e.g. 0.25 years for glass eels). Each M value is then compared with known F values. In the absence of better data, these are derived mainly from heavily exploited systems (Moriarty, 1996). The conclusion is that over 18 years, cumulative M is 2.52, compared to F values of 5.21 in the 'Biscay area' (where glass eels are relatively numerous) and (without any restocking), 1.65 in other regions and 3.86 overall. Dekker admits that the model only serves as a very preliminary one, based on insufficient information and that it is best viewed as a reformulation of existing assertions, re-emphasising differences between the major fisheries which target different life stages. It is acknowledged that the model cannot suffice for developing rational management and targets at the continental level, but can help clarify where the knowledge base is especially deficient. The model is cited, however, by Dekker (1999b) when claiming that 'available evidence suggests an overall overfishing of the European stock'.

Knights (1999a) has also proposed a life-stage model, including the unexploited marine larval stages, to compare the possible impacts of M and F for stocks in England and Wales. Details are given in Appendix A, but key points are that the model involved;-

- calculation of habitat availability and eel density/sex ratios distributions, leading to:-
- estimation of the numbers of emigrant female spawners from fresh, estuarine and coastal waters and their fecundity and thence:-
- estimation of the number of eels per life stage per environment, allowing for losses due to M, through a complete life cycle, leading to:-
- estimation of female spawner production and escapement in the next generation
- from discussions earlier in this chapter, M was assumed to be 99.99% for oceanic larval stages, 90% for continental stages between immigrant glass eel and pigmented elver/early juvenile stages and 75% for the yellow to silver eel stage.

The main conclusions were;-

- the model was found to balance to the same order of magnitude through a complete life cycle (the relative fishing impacts are considered below)
- the importance of marine larval mortality is emphasised by the fact that to produce one order of magnitude change in spawner escapement, only a 0.1% change in natural mortality would be required during the leptocephalus oceanic or glass eel continental

shelf migration (e.g. due to unfavourable oceanic environmental and/or feeding conditions)

- natural mortality of glass eels during estuarine migration and the first-feeding and pigmentation stages could have major impacts on recruitment to a catchment in any one season, e.g. due to the impacts of density-dependent predation and to local factors such as low water temperatures or spates)

In relation to F, fishing mortality in fresh and estuarine/coastal waters would have negligible impacts on spawner escapement in comparison to M in England and Wales where exploitation rates are relatively low, as follows;-

- a glass eel catch of ~20 t per year represents < 0.5% of estimated total number of glass eel migrants

- a yellow/silver eel catch of ~ 150 t $year^{-1}$ represents ~ 0.7% of total escapement – even if estimated at 150 t $year^{-1}$ over 8 years of catchable size life span (> 1200 t), this represents ~ 5% fishing mortality losses

It must also be emphasised that the model includes many assumptions based on incomplete knowledge, especially of marine reproduction and life stage mortalities, of eel populations in estuarine and coastal habitats and inter-migrations with freshwaters. It is also does not specifically account for the effects of density-dependent factors on mortality, although it is reasoned above that the M and F values are compatible with expected density-dependency relationships. Also, the model is not a closed one for England and Wales and (a) glass eels could be derived from spawners originating from other UK and European waters and (b) some glass eels derived from British spawners could recruit to other European waters. Furthermore, eels have a compensatory life strategy in that if mortality (M or F) is high, densities in freshwaters will tend to fall, probably leading to (a) reductions in density-dependent mortality and (b) the production of relatively more, larger and fecund females than would otherwise occur.

Despite the above caveats, the model is useful in highlighting the most vulnerable life stages and adaptive strategies. An Excel spreadsheet version is available into which new data can be entered as it becomes available and it can also be used to test 'what-if' scenarios.

5.5 Conclusions

It is clear from preceding discussions that the information base on the impacts of M and F and density-dependent factors is poor, especially with regards to the probable major influences of oceanic M. Three key questions need answering in relation to the management of stocks and fisheries and setting targets for continental life stages at the local, regional and international levels;-

5.5.1 Is glass eel recruitment overfishing occurring?

Glass eel recruitment has definitely declined since peaks in the late 1970s (especially towards the more northerly and southerly parts of the species range), tending to stabilise during the 1990s (Section 2.2). However, there is currently no evidence from reviews above and in Chapter 2 to suggest that stocks have been significantly affected in catchments in England and Wales that are at the greatest risk from overfishing of glass eels. Indeed, in exploited rivers such as those off the Bristol Channel and in southwest England, the number of recruits available appears to greatly exceed the potential carrying capacity of available habitats. Densities and biomass decline with distance from the tidal limits, but similar patterns are seen in other unfished rivers. Therefore density-dependent M and ecosystem carrying-capacities are probably more limiting than F, exacerbated by the effects of natural and man-made constraints on upstream migration. Furthermore, the relatively high levels of recruitment and hence densities tend to produce to a preponderance of earlier maturing males, except at sites furthest upstream. It could then be argued that harvesting glass eels has relatively low impacts, and that recruitment failure or overfishing could be counterbalanced by the use of 'excess' glass eels for restocking (Moriarty & Dekker, 1997). It could also be argued that their use in aquaculture production (where survival rates tend to be much higher than in the wild) helps reduce fishing pressure on wild stocks and indirectly promotes spawner escapement.

5.5.2 Is growth-stage overfishing occurring?

Overfishing of yellow eels has been proven to have affected length-class frequency distributions, sex ratios and spawner escapement in a few heavily-exploited systems, such as the Ijsselmeer and in localised areas on the west coast of Sweden. The picture for England and Wales is far less clear because fisheries tend to be small scale, widely scattered and generally unsuitable for continuous high levels of exploitation. Thus nationally (and throughout Europe as a whole), yellow eel fisheries probably have minor effects overall, given their relatively low number and distribution compared to the extent of unexploited waters. In Chapter 7 (Section 7.2.3), for example, it is estimated that recent glass eel and yellow/silver eel catches in England and Wales represent < 0.5% and < 5.0% respectively of natural yields, and the equivalent estimates for the whole of Europe are 2.7% and 6.7% respectively.

5.5.3 Are silver eel fisheries having significant impacts on spawner escapement?

Again, F may be relatively high in heavily exploited waters, such as the River Bann and Baltic where silver eels become highly concentrated and are easier to catch, but overall effects are unclear, given the extent of unexploited waters. Silver eel fisheries in England and Wales are sparse and their efficiencies probably low and thus their combined effects on spawner escapement minimal.

5.5.4 National and international issues

Scientific uncertainties regarding stock status and fishing mortality were recognised by the Working Group on Eel (ICES/EIFAC, 1999). It was concluded that the precautionary principle should be applied and that lack of firm data should not preclude management initiatives to protect eel stocks and maintain sustainable fisheries, but ACFM/ICES subsequently concluded that that national stock ‘recovery plans’ should be instituted. These issues are discussed in more detail in Chapter 8, where it is argued that the knowledge of stock levels and change over the last 20–25 years in England and Wales are such that management reference points can be recommended as part of a National Eel Management Plan. However, if stocks show significant changes, discussions above show that it will be difficult to determine whether and to what extent fishing rather than natural mortality is implicated. Further research on F and M are therefore required, as discussed in the next chapter.

6. RESEARCH PROGRAMMES ON MORTALITY

6.1 Introduction

The specific project aims are;-

- ***To design and cost a realistic research programme on the mortality of eels in freshwater.***

The objectives are to;-

- build on the review of natural *v.* fishing mortality and escapement (Chapter 5) to provide clearer pictures of the impacts of fisheries and, possibly, habitat change in relation to natural mortality (Chapter 4) in England and Wales

- provide further information for setting management targets and for planning and executing monitoring programmes and interpreting data obtained (Chapter 7)

- aid in making decisions about managing fisheries, especially in relation to possible needs and methods to control exploitation (Chapter 8)

- address Recommendation 85 of the Salmon and Freshwater Fisheries Review (2000) in relation to eels, that 'research should be continued in particular to elucidate the population dynamics of the species and to improve the monitoring of eel stocks. This work should be integrated with studies on eels in other European countries'.

This chapter takes cognisance of the requirement to improve knowledge of eel biology, management methods and fisheries, as laid down in the Environment Agency's Consultation Document on a Proposed National Eel Management Strategy (March, 2000). This specifically mentions 'elucidation of natural and fishing mortality values' and places emphasis on seeking partnership funding for future R&D.

Research needs, approaches and risks are reviewed first. Possible integrated whole-life and individual life-stage studies (with and without controlled stocking and culling) are then discussed for selected rivers and for closed, estuarine and coastal waters. Modelling approaches are also reviewed, before reaching conclusions and making recommendations.

6.2 Research Needs, Approaches and Risks

6.2.1 Key objectives

The key objectives are to provide the best possible information for rational management and monitoring, especially in distinguishing whether any deviations of stocks from management targets are due to exploitation and hence what controls might need to be applied (Section 5.5). In Chapter 4, it was concluded that no significant changes in freshwater habitats had occurred in the last 25 years that could be associated with increases in natural mortality. However, more information is need to distinguish the relative impacts of natural (especially density-dependent *v.* density-independent) mortality (M) and fishing mortality (F) during the following life stages, how they influence recruitment to the next stage and, eventually, spawner escapement;-

(i) glass eel
(ii) glass eel to elver stage
(iii) pre-exploited yellow eel (<30 cm)
(iv) exploited yellow and pre-migratory silver eel
(v) emigrant silver eel

In England and Wales, glass eel and fyke net fisheries are much more significant than those for silver eels emigrants, therefore mortality during stages (i) and (iv) are of greatest importance. Information on mortality during the other stages is, however, of importance in providing information about relative survival rates and overall impacts on final spawner escapement, as discussed in the preceding chapter. Ideally, the oceanic life stages and reproduction also need further study, but this is beyond the scope of the R&D project aims.

6.2.2 Research approaches

As pointed out in Chapter 5, the main means applied to studying eel population dynamics and assessing the relative impacts of natural *v.* fishing mortality (individually or, more commonly, in combination) are;-

- input-output estimates of whole-life mortality
- direct determinations of life-stage and size or age-specific mortality
- age- or length-structure cohort analyses
- mark-recapture studies
- mortality-per-life-stage models

These approaches are reviewed in later sections.

6.2.3 Risk assessment

The ideal approach would be to conduct integrated comparative or controlled studies, using combinations of the above approaches, on all life stages in specific selected habitats over the duration of the continental life span. This, however, entails considerable resource implications and there are many methodological and other problems inherent in studying the population dynamics of eels (e.g. as outlined blow in Section 6.6). The main risks in any R&D programmes are;-

- Insufficient or inaccurate information on fishing effort and catches are available
- Unforeseen changes occur in fishing effort due to external factors, e.g. market values fall and commercial fishing is reduced or is no longer viable
- Natural environmental changes or unforeseen events (e.g. pollution incidents) interfere with the studies
- Methodological problems (especially those related to quantitative sampling) interfere with the studies
- The availability of funds changes in the future

Improved monitoring of eels stocks recommended in Chapter 8 will add to the overall knowledge base for selected rivers, but in view of the resource implications and the risks listed above, suggestions are also made for more restricted studies and for the development and testing of predictive models. Because of the difficulties in assessing priorities for the Environment Agency in relation to resource costs, no specific single and costed research programme is suggested. However, various recommendations are made, especially for studies that could integrate with others (e.g. environmental impact assessments aimed at salmonids) or involve partnerships and joint funding. Prioritisation is left to the Environment Agency based on resource availability and perceptions as to what other integrated studies, partnerships or external funds are most worth pursuing.

6.3 Studies on Rivers

6.3.1 Introduction

To obtain meaningful results throughout the riverine life stages to inform management on the relative impacts of M and F on stocks, studies would entail a long-term commitment to cover the normal freshwater life-span of eels, i.e. a minimum of about five to seven years but preferably a period of up to15-20 years. Whilst the ideal situation would be to compare one catchment subjected to glass eel fishing mortality and one subject to yellow and silver eel fishing mortality with comparable unfished ones, this would be costly in relation to the risks outlined above. More restricted studies of M and F in relation to localised glass eel/elver recruitment, yellow/silver fisheries and spawner escapement M and F are suggested below. For integrated whole-catchment/life-stage studies, in view of the risks listed above, it would be better to use catchments where no commercial fishing occurs but where fishery impacts can be mimicked and controlled experimentally. These would have to involve systems effectively closed to uncontrolled or unmeasurable migration because of the confounding effects of immigration and emigration into the whole system and between reaches/sub-sections. Annual upstream migration of juvenile yellow eels, for example, prevented clear results being obtained from removing all eels caught by electric fishing in the studies of Mann and Blackburn (1991) in Tadnoll Brook, Dorset.

6.3.2 Ideal research approaches

Such studies would involve (using as appropriate the monitoring methods discussed in Chapter 8);-

- Assessment of glass eel recruitment (see Section 8.6)

- The installation of traps to catch all 0+ elver and > 0+ juvenile recruits near to the tidal limit (Section 8.6.3)

- The installation of traps on weirs and other barriers to catch all emigrant immature yellow eels moving through the system (Section 8.6.3d)

- The installation of traps or counters near to the tidal limit to assess emigration/escapement of yellow and silver eels (Section 8.7)

- Once base-line recruitment is determined, in-river populations could be manipulated in controlled ways by stocking different amounts of glass eels/elvers or on-grown juveniles to imitate the impacts of glass eel exploitation or 'good' and 'bad' recruitment years

- Marking of immigrants (and any stocked eels), e.g. via tetracycline labelling of the otoliths or, if large enough, via sub-epidermal injections of acrylic paint or PIT tags.

- Annual seasonal eel-specific electric fishing surveys at selected sites (Section 8.4)

- Assessment of changes in density, biomass, population structure, and from marked fish, length data and sex ratios, age and growth rates (Section 8.4)

- Estimation of natural mortality at different life stages across the whole freshwater life span

- Measurement of key environmental parameters (habitat characteristics and temperature) to assess relationships to survival, growth and immigration/emigration (Section 8.4.13)

- Controlled culling of commercial-size yellow and silver eels to imitate the impacts of yellow/silver eel fishing and assessment of changes in populations

- Using culled eels to assess sex ratios by dissection and age and growth rates via otolithometry (Appendix C)

- The development of predictive models (as discussed further in Chapter 6)

6.3.3 Research locations

Possibly suitable catchments on the basis of their manageable size, possibilities for erecting traps at their bases and availability of some information on stocks and population structures include the following rivers;-

- R. Leadon in the Severn system (see Chapter 3 and White and Knights, 1994)
- R. Darent and/or R. Mole, lower and middle tributaries respectively of the Thames (Section 22 and Naismith & Knights, 1993). The Roding (another lower tributary) might also be suitable, although the knowledge of stocks is less extensive, an elver trap has recently been constructed on the tidal barrage (see Section 6.5.3)
- R. Piddle, Dorset (see Chapter 3)
- R. Stour, Dorset (Section 2.19)
- the Test (Hampshire) and Leven and Eden (Cumbria) are possible candidates because they have fish counters near their bases that might be suitable for measuring spawner escapement (Section 6.3.4).

Because of the high resource implications, any possibilities of combining research on mortality with monitoring and other research programmes should be explored. For example, some data on stocks are already available for the Leven and there are plans to carry out long term fisheries surveys to set baselines in relation to drought-order abstractions (Sections 8.4.6 and 8.7). Although salmonids are currently the principle target, a combined programme could include studies of eel stocks and population dynamics and, via the fish counter, spawner escapement.

Partnerships (e.g. with RSPB in studies of Leighton Moss, Section 6.5.2b) and external funding should also be sought. In the latter context, it must be pointed out that there is a risk that the EU or other funding bodies might deem that such whole catchment studies in England and Wales are not of the highest priority because;-

- Similar studies have already been carried out in the past, e.g. on the Imsa in Norway, by Vøllestad & Jonsson (1988)

- Similar extensive and long-term studies are already in progress in Europe, e.g. on the Vilaine (Briand *et al.*, 2000) and other Brittany rivers, including the Fremur (Feunteun *et al.*, 1998).

6.3.4 Silver eel escapement

Limited local studies of glass eel/elver recruitment, M and F are discussed below (Section 6.5.2) but other restricted ones on silver eel emigration should be carried out if at all possible. The Agency could construct silver eel traps, use ones already available or commercial fishermen could be involved. The risks entailed are discussed in Section 8.7 and it is concluded that, at least initially, it would be more cost-effective to fully explore the use of fish counters, e.g. such as those on the Test (Hampshire) and the Leven and Eden in (Cumbria) (for further details, see Chapter 2). Sampling to assess sex ratios and fecundity should also be carried out (see Section 6.5.3). However, the possibilities of combining studies on spawner escapement with other aspects of population dynamics and impacts of F and M in single catchments are worth exploring further, e.g. in the Leven, as discussed above.

6.4 Controlled studies on semi- or fully enclosed stillwaters

6.4.1 Introduction

These potentially offer advantages over catchment studies in using smaller and more manageable and controllable systems. However, these would probably require stocking and would be prone to the same risks listed above in terms of unforeseen changes in habitats and natural mortality and possible repetition of past or replication of on-going studies, as exampled by studies discussed in Chapter 5.

An example of a study in a semi-closed water body (adapted for the present R&D programme) is that previously proposed to the Environment Agency for Poole Park Boating Lake as a more feasible alternative to large scale studies in Poole Harbour, Dorset (Knights, 1997). This can be summarised as follows;-

6.4.2 Poole Park Boating Lake study

- The Lake has an area of 22.5 ha and is similar to Poole Harbour in being shallow (averaging 0.75 m and not more than 1 m), with restricted connections and long retention times. Temperature, salinity and dissolved oxygen are generally quite high, averages of 12°C, 15 - 20 ppt and 113.5% DO respectively being quoted in various studies.

- Productivity is high, growth of macrophytes and phytoplankton appears to be good and there are verbal reports of high densities of mysids, shrimp and brackish water fish, such as mullet.

- Traps could be placed in the narrow channel connecting the Lake and Poole Harbour to assess natural recruitment

- If necessary, the Lake could be stocked at different rates in different years to imitate the effects of fishing mortality. With an area of 22.5 ha and at an ideal stocking rate of 300 (0.1 kg) glass eels/elvers ha^{-1}, this would entail at a maximum the purchase of 2.25 kg (7200 fish). At an estimated £60 kg^{-1}, this would cost £135.

- nets or traps could be mounted in the channel to capture emigrating silver eels. Nets and traps could, incidentally, provide extra information on the movement of other species of fish (and macroinvertebrates) between the Lake and the Harbour.

- Electric fishing to sample populations will be limited by salinity, depth and, possibly, turbidity. However, the size of the Lake lends itself to fyke net surveys of eel stocks, including mark-recapture studies, to assess natural mortality, growth rates and potential production. Difficulties would be experienced in sampling small eels, however. Very fine-mesh fyke nets are not effective (Knights *et al.*, 1996) but studies using, for example, small baited traps, could be tried

- Additional benefits might accrue from supplementary studies of the ecology of the Lake in providing information on (a) the causes of blue-green algal blooms and (b) possible biomanipulation remedies.

6.4.3 Poole Park Boating Lake pilot studies: risk assessment

- The Lake is prone to influences of combined sewer overflows, sedimentation and dilution and eels might be affected by, for example, drastic oxygen crashes.

- It has changed in recent years from a macrophyte-dominated to a phytoplankton-dominated one, possibly associated with nutrient over-enrichment. This has resulted in occasional blooms of nuisance and toxic blue-green algal blooms (e.g. of species of *Oscillatoria*, *Anabaena* and *Phromidium*). It is unlikely that eels are directly affected by such blooms but changes in underlying trophic structures might affect growth and production.

- Opening and closing of sluice gates to flush out the Lake causes periodic changes in the habitat and could allow uncontrolled exchange of naturally-recruiting eels and loss of stocked eels to Poole Harbour. Tidal flushing was carried out at HWS approximately once per month by Poole Borough Council until 1995 but no regular programme has been followed since Easter 1996.

The above approaches could be adapted to suitable enclosed and stocked stillwaters and wetlands. Such studies would be of interest to organisations such as RSPB, as discussed further in the next Section.

6.5 Studies in Estuaries and Coastal Waters

6.5.1 Extensive integrated studies

Ideally, studies of representative fished *v.* unfished coastal habitats should be conducted but the practical difficulties of accurate sampling and resource costs would be very high. Such studies are in any case outside the remit of the project aims which only cover mention 'freshwater'. However, some research on glass eel/elver mortality during the estuary-to-freshwater transition stages is essential, as follows.

6.5.2 Studies of glass eels, 0+ elvers and >0+ juvenile eels

It was recommended in Section 8.6 that any resource-intensive initiative to monitor glass eel recruitment should be held in abeyance until the CFP/EU Concerted Action Glass Eel Monitoring Programme issues its conclusions. However, it was also recommended that support be given to other studies that are already underway, this could be extended to other research on key issues, as follows;-

(a) Glass eel and elver/juvenile recruitment and mortality in the Severn

Previous discussions have emphasised that although the initial recruitment of glass eels into estuaries is of obviously of great importance, actual recruitment to freshwater is dependent on the subsequent upstream immigration of elvers and juveniles (e.g. see Sections 5.4.2-3). The most important studies carried out on these issues in England and Wales have been (a) a mark-recapture study of glass eel recruitment in the Severn Estuary by Bristol Channel Fisheries (now UK Glass Eels) and (b) trapping studies of elvers/juveniles at the tidal limit at Tewkesbury (White & Knights, 1994). However, the latter study had to be conducted independent of the former because specific experimental details and results were not released by Bristol Channel Fisheries. Thus only estimates could be made of fishing and natural mortality.

It is **recommended** that these studies should be repeated but in an integrated fashion, preferably over at least two to three years to account for natural inter-year variability. This will entail gaining the cooperation UK Glass Eels and the use of the permanent Agency traps at Tewkesbury (augmented, if necessary, by temporary pass traps such as those used by White and Knights, 1994). The possibilities of UK Glass Eels funding the glass eel studies should be explored, more information on sampling methods is given in Sections 8.6. Methods can be adapted to coordinate with those being used in studies proceeding in the Vilaine (Brittany) fishery (Briand *et al.*, 2000). This will aid comparisons of results gained, including those from the studies at Leighton Moss suggested below. It must be pointed out, however, that the resources and risks involved would be high. For example, marking can also affect survival and migratory behaviour (Briand *et al.*, 2000: Jessop, 2000), and high river flows and levels can have major impacts on individual or total glass eels runs (as occurred in the Severn in 1991 and 2000). Also, Jessop (2000) estimated that a Peterson population estimate with a 95% CL of < ±1, based on an expected run of 1 million and an expected recapture rate of 15-20,000 elvers, would require the marking of least 25,000 eels (e.g. using Bismarck Brown Y and/or fin-clipping).

(b) Glass eel and elver/juvenile recruitment and mortality at Leighton Moss
A key study which is at present underway in England is that designed to assess the impacts of the glass eel fishery in Red Barn Dyke/Quicksand Pool on recruitment to Leighton Moss SSSI/RSPB Reserve, Cumbria (Section 5.4.3). As discussed in Section 8.6.5, such studies should be actively supported and possibilities of joint initiatives with RSPB explored. Furthermore, it is **recommended** that the possibilities be explored of carrying out similar partnership studies on east coast wetland sites, such as Minsmere. This would provide valuable information on possible differences between sites at widely varying distances from the main Atlantic migration routes.

(c) Risk assessment
The above studies entail risks as outlined in Section 6.2.2.

6.5.3 Studies of yellow and silver eels

Quantitative sampling is particularly problematic in open estuarine and coastal waters, further difficulties arising from the unknown effects of immigration and emigration. Fyke netting is the only feasible means of sampling in such environments but according studies in the Thames Estuary by Naismith and Knights (1990a), its cost-effectiveness is very low for the level of sensitivity to change achieved. Svedäng (1999) was able to use historical commercial catch data from Swedish west coast fisheries (Section 5.4.5). Both of these studies relied upon catch-curve assessment of total mortality (Z) values, the relative effects of F being derived by comparison with unfished sites where Z was assumed to be equal to M. Although Svedäng's (1999) results were suggestive of some growth overfishing, neither study was able to reveal any impacts of changes in recruitment or overfishing on stock numbers. It is unlikely that such detailed and accurate fisheries data would be available in England and Wales and intensive/extensive fyke net surveys would be prone to the risks outlined in Section 6.2.2. Thus, whilst it is worth continuing to monitor the Thames eel fishery (and stocks in the freshwater catchment) as recommended in Section 8.4.6, more detailed studies cannot be strongly recommended. However, some useful comparative information might become available from environmental impact assessment studies using fyke nets following construction of the Roding Barrage (R. Roding) (Steve Thomas, EA Thames Region, personal communication). These EIA surveys could, possibly, be combined with studies of the elver pass erected on the Barrage and more detailed surveys of riverine stocks (see Section 6.3.3).

Ideally, research is needed on silver eel production and spawner immigration out of estuaries and coastal waters, across the Continental Shelf and hence to the (assumed) breeding grounds in the Sargasso Sea. Similarly, reproduction and egg production and survival are key unknowns in the life cycle. These are beyond the scope of the project aims, but it is **recommended** that studies of silver eel emigrants (see Section 6..3.4) should involve studies of sex ratios and fecundity, as well as numbers escaping. The latter, for example, was essential to the development of the mortality model in Appendix A which, as discussed in Section 5.4.1, emphasises the importance of natural mortality during the oceanic leptocephalus and coastal/estuarine glass eel life stages.

6.6 Modelling Studies

Given the risks involved in instituting extensive and long-term research programmes, the use of modelling has much to commend it, even if only in helping to clarify key parameters and hence informing and guiding future cost-effective research. The problems of developing and applying standard fish population dynamic models to *A. anguilla* have been discussed in the previous chapter with respect to assessing mortality. Chapter 8 also highlights difficulties with respect to recruitment, spawner escapement and stock management. Key complications are;-

- The catadromous life style of the eel, with oceanic components similar to marine fish, involving high fecundity to compensate for high M during the long Atlantic planktonic migration phase, high density-dependent mortality as glass eels congregate in coastal and then estuarine waters on their migration towards fresh water and during the metamorphosis to the actively migration elver. Conversely, M probably decreases in yellow eels because as they grow, there are fewer predators large enough to catch and consume them.
- High variability in initial recruitment and then in annual and seasonal migrations and differences in catchability during the yellow eel phase
- Difficulties in ageing by otolithometry and high variability in growth rates and age-at-length in eels mean that modelling based on year classes or age-cohorts are not as applicable as they are to other fish species
- Variability in growth rates during different life stages in different habitats and hence in age-at-maturity and emigration, interacting with variations in sex ratios with population density
- Fisheries are seasonal and targeted in different locations at different life stages, i.e. glass eels and elvers, yellow eels > 30 cm and emigrant silver eels
- The dearth of quantitative information on many of these factors and how they are interrelated

The development and use of stochastic probabilistic models could help take into account these uncertainties, optimising the use of what little firm data are available but also being amenable to insertion of different values to explore 'what-if' scenarios. The mortality-per-life-stage compartmental approaches used by Naismith and Knights (1990), Dekker (1999) and Knights (1999, see Appendix A) have much to commend them, i.e. to break the life cycle down into compartments based on individual life stages. These correspond more to size than to age groupings, and have the advantage of relating to the different fishery target life stages listed above. Such models consider 'recruitment' to each stage, the possible impacts of density-dependent and density-independent M and F (and emigration and immigration) and thence 'escapement' to 'recruit' to the next life stage. Each compartmental overview can then be joined to the next to try to produce a picture for the whole life cycle.

Naismith and Knights (1990) only modeled the situation in the yellow/silver eel fishery in the Thames Estuary but Dekker (1999) considered all continental life stages. His model was directed at broad comparisons across the whole of Europe, with emphasis on comparing the impacts of F in Biscay area glass eel fisheries with those in yellow/silver eel fisheries in other areas (especially in NW Europe and the Mediterranean), plus the effects of stocking. He initially applied a dynamic pool model approach (Beverton & Holt, 1957) to assess the change in number of fish between each life stage compartment, inserting estimated values of recruitment, F and M and silver eel escapement. F values were derived from commercial landing data, M values were derived from estimates for the duration of the whole continental life span. These were assumed to remain stable such that values per discrete stage were simple divisors, e.g. M was estimated at 2.52 over 18 years and over three months of the glass eel stage, M was assumed to be 0.04. Dekker then applied Virtual Population Analyses (VPA) to each compartment, using the Gulland (1965) iterative method, to assess the number of eels left to recruit to the next compartment, together with sensitivity analyses. This model has a number of major drawbacks. Firstly it relies on very broad continent-wide estimates of M and F, based on information from relatively few, widely-scattered and often intensive fisheries in different countries (see Table 5.1). Secondly, M (and, probably F) are not stable throughout all life stages. For example, density-dependent mortality would be expected to be highest during the peak periods of glass eel/elver immigration. Finally, in estimating the impacts of F on the whole European stock, insufficient allowance was made for the large areas of unfished or lightly fished waters compared to those subject to high fishing pressures.

Many of these problems could be overcome in developing a national model, thereby being able to focus on the data for England and Wales accumulated in the present R&D project. This would then provide useful information for application to other European countries, because the east-coast/west-coast divide between glass eel and yellow/silver fisheries, due to the differences in distance from the Atlantic migration pathways, reflects those seen between the Biscay area and the rest of Europe. However, the present R&D programme has not produced a very extensive body of information, nor has it been able to demonstrate significant overall impacts of recruitment failure, F or M or habitat change on the national scale. Indeed, recruitment still appears to be meeting or even exceeding the carrying capacity of many rivers, particularly in the shorter ones in the south west. The few examples of declines in stocks (e.g. the Piddle and Frome) and examples of restricted distribution (e.g. in the Thames catchment) appear to be due to low recruitment, but not to fishing mortality. Thus any model based simply on currently available data is likely to produce balanced relationships overall between recruitment, stocks and spawner escapement. Therefore, it would be particularly valuable to explore, develop and run VPA-type and other (e.g. life table and energetic) models, not only using available data, but also inserting hypothetical values for recruitment, M and F and to include density independency/dependency, emigration/immigration and other factors to test 'what if' scenarios. Such models would help clarify which are the most critical parameters acting during different life stages (including possible habitat changes), provide information to guide future cost-effective research, aid in the interpretation of monitoring information and help inform future stock management.

6.7 Conclusions and Recommendations

6.7.1 Introduction

Key R&D objectives are to provide information on the relative impacts of fishing *v.* natural mortality to aid in the setting of management targets, planning monitoring programmes and interpreting data gained to help maintain sustainable fisheries and stocks. No specific single and costed research programme is suggested but various recommendations are made. Prioritisation is left to the Environment Agency, based on resource availability and perceptions as to what other integrated studies, partnerships or external funds are most worth pursuing.

6.7.2 Risk assessment

The main risks facing comparative or controlled studies on all life stages in specific selected habitats over the duration of the continental life span are;-

- Insufficient or inaccurate data on fishing effort and catches are available
- Unforeseen changes occur in fishing effort due to external factors, e.g. market values fall and commercial fishing is reduced or is no longer viable
- Natural environmental changes or unforeseen events (e.g. pollution incidents) interfere with the studies
- Methodological problems (especially those related to quantitative sampling) interfere with the studies
- The availability of funds changes in the future
- Studies could involve unnecessary replication of others underway in Europe

6.7.3 Studies on rivers and stillwaters

Ideally, integrated studies of all life stages over the duration of the continental growth phase should be carried out in specific catchments or stillwaters, comparing fished and unfished waters or using controlled experimental manipulations to mimic the impacts of changes in recruitment, F and M. However, resource implications and risks are considered high. Choices will therefore depend on funds available and the possibilities of combining research with other studies, partnership research and external funding. Suggestions for locations and integration with other studies are made in Sections 6.3 and 6.4.

6.7.4 Studies in estuaries

Although the project aims are to focus on mortality in freshwaters, two restricted studies are **recommended** to clarify M and F and the relationships between glass eel recruitment into estuaries and the effective recruitment to freshwater stocks by 0+ elvers and >0+ juveniles. The first is to institute a study in the Severn involving mark-recapture of glass eels (requiring the cooperation of UK Glass Eels) and trapping studies at Tewkesbury (Section 6.5.2a). The second is to explore joint studies on glass eel fishing and recruitment with RSPB at their reserve at Leighton Moss (Cumbria), with possible extensions to east coast sites, such as Minsmere (Section 6.5.2b). If such methods are proven to be reliable and cost-effective, they could be integrated into catchment-wide studies.

6.7.5 Studies on silver eel escapement

It is **recommended** that the possibilities of collecting eel data in the long term from fish counters on the Test (Hampshire) and Leven and Eden (Cumbria) should be explored further (Sections 6.3.4 and 6.5.3). If such methods are proven to be reliable and cost-effective, they could be integrated into catchment-wide studies (see Section 6.7.3).

6.7.6 Modelling studies

In view of the resource implications and risks involved in extensive and long-term research programmes, it is **strongly recommended** that summative and predictive modelling studies for stocks in England and Wales be instituted as soon as possible. These should be based on a life-stage compartmental approach, involving the development and running of VPA-type and other (e.g. life table and energetic) models. These could not only use available data from the current R&D report, but also hypothetical values can be inserted for recruitment, M and F and include density independency-dependency, emigration-immigration and other factors to test 'what if' scenarios (Section 6.6). Such models would help clarify which are the most critical parameters acting during different life stages and, by summation, the whole life cycle. They would also cost-effectively provide information to guide future research, help inform future stock management (Chapter 7) and aid in the interpretation of monitoring information (Chapter 8).

7. REVIEW OF MANAGEMENT OPTIONS

7.1 Introduction

7.1.1 Project objectives

The specific project objectives are;-

- ***To review management options to ensure sustainable eel stocks and fisheries***

Management of eel stocks and fisheries are discussed below in the context of the precautionary approach and determining management reference points, and a National Eel Management Plan is proposed. This involves consideration of Environment Agency duties and policies and its Consultation Documents on a Proposed National Eel Management Strategy (March, 2000) and on Eel Licences, Duties and Byelaws (April 2000). European issues are also considered because of the panmictic nature of eels and declines in recruitment and stocks. Recommendation 28 of the Salmon and Freshwater Fisheries Review (2000) also emphasises the need to integrate actions in England and Wales with any over-arching EC policies and recommendations. This accords with Advisory Committee for Fisheries Management (ACFM)/International Council for Exploration of the Sea (ICES) recommendations to the European Union, calling for the development of European and national stock programmes (ACFM/ICES, 1999). At present, however, no other country in Europe has made any proposals for application of the precautionary approach or for setting management reference points and means of attaining these. France has been exploring the development of indices of recruitment, stock status and escapement, but this has not reached the stage of addressing the management issues raised by ACFM/ICES (Lambert & Rigaud, 1999). Current management systems vary greatly between countries (Section 7.4). These are generally based on local practices or non-statutory regulations that have evolved in *ad hoc* ways to maximise catch values, rather than as biological conservation measures. For example, glass eels can only be caught in the River Bann (N. Ireland) for stocking Lough Neagh and minimum size restrictions are imposed in the Neagh long-line fishery to maximise catches of larger yellow and silver eels that command higher prices. Many control measures on eel fishing have evolved primarily to protect other fish species (e.g. salmonid migrants or estuarine by-catches). Very few studies have been conducted to assess the efficacy of current management practices. Discussions and recommendations in this chapter will therefore focus on a National Eel Management Plan for England and Wales. European issues will be addressed where appropriate, good practices reviewed and suggestions will be made for international actions and sharing of the conservation burden for the total European stock.

7.2 Proposals for a National Eel Management Plan

7.2.1 Introduction

The **principal aims of the Plan** are similar to those of the Agency's Draft Management Strategy, i.e.;-

To maintain, improve and develop eel fisheries to permit the sustainable exploitation of the eel stock for recreational and commercial benefit, taking into account the unique biology and life cycle of this species

The key objectives are, via consistent and standard regulatory and management frameworks, to;-

- **Ensure sustainability of stocks and fisheries by maintaining spawner escapement relative to exploitation**
- **Obtain and use best available information to inform management**
- **Maximise long term socio-economic benefits of commercial eel fisheries**
- **Minimise potentially damaging effects of commercial fisheries on the environment**

7.2.2 Application of the Precautionary Approach

Application of the precautionary approach to commercial fisheries has been promulgated by FAO, UN, ICES and NASCO, issues specific to eel fisheries have been reviewed by Russell and Potter (1999). Basically, this approach states that if there is evidence of threats or irreversible damage to eel stocks due to overfishing, lack of scientific information should not be used as a reason for postponing or failing to take conservation and management measures. The precautionary approach should also be applied in a more general sense to counteract other damaging anthropogenic impacts, such as migration barriers, habitat degradation or pollution. Application of the precautionary approach requires that the best scientific evidence available should be used to set management reference points. These comprise threshold reference levels (**conservation limits**) below which stocks should not be permitted to fall. In view of uncertainties about eel stock dynamics and abilities to measure them accurately, **management targets** should then be set above these limits, using the best available information, to provide guide values to aim at to ensure that controls on fisheries are applied before stocks could begin to be negatively impacted.

The Working Group on Eel (1999) proposed to ACFM/ICES that, because of the lack of reliable long-term data on stock status and dynamics and on catches and effort, it was not possible to set management targets and that conservation limits should be set at current levels of exploitation. ACFM/ICES (1999) subsequently recommended to the EC that;-

(a) a common approach to eel fishery management under national jurisdictions should be agreed
(b) a recovery plan should be implemented for the European eel stock, including plans for all areas as well as monitoring of fisheries and stock development
(c) management should reduce fishing mortality to the lowest possible level until such a plan is agreed upon and implemented
(d) a recovery plan should contain a definition of when the stock is considered to be in a recovered state
(e) A recovery plan should contain targets for escapement of glass, yellow and silver eel on a catchment area basis

7.2.3 Management approaches for England and Wales

This R&D project has shown that stocks and escapement in England and Wales have not changed significantly overall during the last 20-25 years, despite marked declines in glass eel recruitment. There is anecdotal evidence for declines in a few fisheries over the last few decades, such as that in Poole Harbour (Section 2.18), but the only statistically significant changes in stocks detected have been in the Piddle and Frome (1977-99), Lower Severn (1998-99), Devon Axe (1979-86) and Dorset Stour (1986-90). In the Piddle and Frome, an unusual combination of low recruitment and the effects of Poole Harbour as a habitat sink are implicated (Sections 2.18 and 3.9.7). In the Lower Severn, natural inter-year variations in intra-catchment distribution are implicated (Section 3.7.7). The apparent changes in the Axe are probably due to survey sampling differences (Section 2.16). Those in the Stour catchment might also be due to survey differences, but could have resulted from poor water and habitat quality factors in the lower reaches that inhibit upstream migration (Section 2.19). Resurveys are needed to clarify causes in these examples. Population distributions appear restricted in some catchments distant from the Atlantic migration pathways, such as the Thames (Section 2.22.1), but there is no clear evidence that this is a recent phenomenon.

This situation makes it difficult to propose recovery targets for England and Wales, other than ones based on the current status of stocks, fishing mortality and spawner escapement. A possible target for stock recovery is to control fishing mortality to return recruitment to the peaks applying in the 1970s, but this is probably unrealistic because these levels were high relative to those pertaining between the 1920-1960s (Section 2.2.1). During this period, exploitation rates were probably as high or higher than they are now (e.g. see Tesch, 1977). Furthermore, the weight of evidence from this R&D project indicates that the scattered and low-intensity fisheries in England and Wales have not had significant impacts on stocks or escapement. For example, it has been estimated (Section 5.4.7 and Appendix A) that recent glass eel catches represent < 0.5% of annual national recruitment and yellow-silver eels catches < 5.0% of escapement over 8 years of catchable life span.

Overall, rivers in England and Wales appear to be recruiting well enough to maintain stocks close to predicted carrying capacities, given (a) inter-catchment differences due to position relative to Atlantic immigration pathways and (b) intra-catchment differences due to habitat productivity, suitability for day-time cryptic hiding, altitude, distance from tidal limits and the impacts of migration barriers. The same picture appears to apply to rivers in France and to the coastal waters and rivers of south-west Sweden (Svedäng, 1999; Hakkan Wickstrom, personal communication). The glass eel fisheries in French estuaries of the Biscay coast are very much larger than those in England and Wales but even these do not appear to have affected inland stocks to the extent that might have been expected (see discussion of Lambert & Rigaud, 1999, in Section 7.3.4). The general Europe-wide picture is that the impacts of recruitment declines have been greatest in the most northerly and southerly extents of the range of *A. anguilla*. Recruitment (at least in more northerly regions) has possibly always been historically lower than in Biscay Bay areas. In the Baltic areas, for example, it appears that recruitment is mainly dependent on post-glass eel life stages migrating via the Skagerrak-Kattegat and that declines have been affecting yellow and silver eel fisheries since the 1960s. Finally, reviews of mortality factors (Section 5.4) have also emphasised that historical trends in glass eel recruitment and similarities with declines in the American eel (*Anguilla rostrata*) could be primarily due to changes in oceanic factors and early larval survival. The oceanic and estuarine factors may greatly outweigh the impacts of fisheries (Sections 5.4.1 and 5.4.2).

In conclusion, the evidence suggests that the British contribution to European spawning stocks has remained at a satisfactory level. Furthermore, it is likely that any severe national restrictions on the currently low levels of fishing mortality would be counteracted by enhanced density-dependent natural mortality. This is especially likely in the main glass eel fishing areas where the number of potential recruits greatly exceeds the carrying capacity of local rivers. Severe restrictions would also be difficult to implement quickly and might in some cases lead to socio-economic hardship. However, although fisheries appear to be having low impacts and, currently, to be in decline because of market forces, the situation could change in the future. Therefore the precautionary approach should be applied to counteract this possibility. Thus it is **recommended that in the short term and pending the development of any over-arching EU management requirements, fisheries in England and Wales should not be allowed to expand in effort or geographical range beyond the averages pertaining over the last five years, unless and until this is proven to be unnecessary.** This accords with Recommendation 30 of the Salmon and Freshwater Fisheries Review (2000) for glass eels, but it is further recommended that as a precautionary measure, this should be extended to other life stages

7.2.4 Review of possible management reference points

The discussions above lead to acceptance of a *status quo* scenario for stocks and fisheries in England and Wales. However, appropriate management reference points are needed to guide decisions on interventions at the international, national and local levels and future monitoring requirements. These should ideally be based on stock-recruitment relationships. For other anadromous species such as the salmon, the number of recruits and returning mature spawners derived from those recruits can be measured, correlations sought and relationships modeled. This approach is used by the Environment Agency for deciding on reference points for the management of salmon populations in catchments. It cannot be directly applied to eel because of the lack of knowledge of the size of the spawning stock and spawner escapement or of mortality during the marine life stages.

Spawning stock biomass per recruit (SPR) models have been explored, as an extension of yield-per-recruit analyses, using data on maturity at age/size (Dekker, 1999b). Equating SPR to the average recruitment per spawner biomass yields an estimate of levels of F at which the stock replaces itself on average. Relevant historical data are too scarce to reach firm conclusions, but prevailing SPR levels of the exploited European stock can be related to estimates of unexploited stock SPR and expressed as %SPR. Dekker used European-wide estimates of M (at two levels), size at maturity and maximum body size and produced provisional replacement %SPRs of 23-33%. This leads to a possible reference point that F (or other anthropogenic impacts) should not be allowed to reduce %SPR below ~30%.

Dekker (1999b) also explored simple stock production models. These can be used to predict an optimal production (and hence spawner escapement) at a biomass equal to half the biomass of an unexploited population, when F = M. In this scenario, the overall natural mortality for the continental life stages was assumed to be 75% (from Moriarty & Dekker, 1997). This implies that fisheries should not be allowed to raise total mortality above 75% of levels pertaining in unexploited systems. It is interesting that spawner escapement from the stocked and well-managed but highly exploited Lough Neagh fishery is estimated to be about 30% and that F $\approx$ predicted M (Robert Rossell, personal communication). The fishery has maintained consistency of yields without evidence of size overfishing over several decades.

A number of problems arise in using the above approaches. Firstly, is very difficult to monitor recruitment, mortality and escapement and baseline reference data are not available. Furthermore, setting reference points on spawner escapement alone does not help a manager distinguish between the impacts of glass eel, yellow or silver eel fisheries. Ideally, escapement reference points need to be set for each life stage, as advised by ACFM/ICES (1999). More research is required to derive robust recruitment-stock-escapement and/or maximum sustainable yield models. Some possible approaches have been discussed in Chapters 5 and 6 and monitoring requirements are discussed in the Chapter 8. No country in Europe has currently set management reference points. It is **recommended that a suitably pragmatic and short-term approach for England and Wales is to use indices based on stock levels, population structures and distributions as surrogate reference points for recruitment to and escapement from each continental life stage.**

7.3 Population Indices for Setting Management Reference Points

7.3.1 Introduction

The discussion above leads to the **recommendation that interim management reference points for England and Wales should be set to maintain stocks and spawner escapement at or above current levels. This would help to ensure that national eel production and escapement makes the best possible contribution to the European stock.** Furthermore, if it can be shown that exploitation is not significantly affecting achievement of management objectives, there may be grounds for encouraging more fishing.

Population indices upon which to set management reference points could be based on optimum levels of one or more of density, biomass, length-class frequency distribution, average individual size, growth rate, age distribution, proportion of silver eels, sex ratio and catchment distribution pattern of stocks. National reference points could be set (e.g. see spawner escapement estimates made in Appendix A). However, large regional differences exist, especially on a west-to-east continuum, relating to distances from Atlantic migration pathways (e.g. see Table 4.3). Therefore, reference points need to be set on data gained from individual catchments and Chapters 2 and 3 (and the R&D Database Collection) provide useful baseline information. However, there are uncertainties involved in using population indices. Firstly, there is a lack of knowledge about estuarine and coastal populations and in rivers, as shown in Chapters 2 and 3 (and by Lambert & Rigaud, 1999), natural variability in population characteristics of eels can be very high and accurate monitoring and detecting real changes difficult. Therefore, the need for management intervention in a catchment can only be signalled when stocks fall significantly below the management reference point.

7.3.2 Growth rate and age distribution indices

A lack of younger age classes in a population could be a sign of overfishing of glass eels and/or recruitment failure. Conversely, a preponderance of fast growing young eels in a population could imply high fishing mortality of older eels < 30 cm, the minimum commercially-valuable size. However, growth rates and age-at-size relationships can be naturally very variable in eels, problems compounded by inaccuracies in sampling and otolithometry (Section 3.9.5 and Appendix C2.4). Periodic recruitment of yellow eels into more upstream waters can complicate the picture because immigrants often exhibit low growth rates relative to resident eels (Naismith & Knights, 1988; White & Knights, 1994).

Given the need for large samples, the resource costs, methodological problems and the lack of clear predictive precision, changes in age distributions and growth rates cannot be recommended as general measures for setting management reference points.

7.3.3 Density and biomass indices

Biomass and density are the most widely available spatial and temporal measures of stock status and can be collected during routine or specialised electric fishing surveys (Appendix C). Spatial and temporal variability can, however, be high, as shown in many of the examples discussed in Chapters 2 and 3.

Glass eel recruitment failure or overfishing could lead to declines in density and biomass. However, even if density declined relatively quickly, reduced competition will enhance growth rates and the production of females > 40 cm. Thus high biomass could be maintained for many years, even following drastic declines in recruitment (Section 8.3.1 and Appendix C.2.3). This has been demonstrated in modelling studies of glass eel recruitment and stocks in the R. Loire (France) (Eric Feunteun, personal communication). Using data for glass eel recruitment (derived from catches in the Loire fishery) and modelling resultant eel stocks in the river at two different mortality rates, he showed that density fell as recruitment declined, with time lags of about two years at a Z of 0.65 and about five years at a Z of 0.15. Biomasses in the model actually increased in the short term because of enhanced growth rates at low population densities, only beginning to decline after about six and two years at the low and high Z values respectively. Despite these predictions, there have actually been no significant declines in density or biomass in the Loire, i.e. a picture mirroring that seen in England and Wales (Chapters 2 and 3) and other French rivers (Lambert & Rigaud, 1999). It appears that the number of glass eels, despite high fishing mortality, has been sufficient to produce enough pigmented elvers and 0+ yellow eels recruits to meet the carrying capacity of rivers on the Biscay coast and the southwest of England which face the Atlantic migration pathways.

These discussions suggests that biomass will generally not be a sufficiently sensitive index on its own for setting reference points, although the impacts of high levels of exploitation of yellow eels might be detectable over shorter time scales. Population density (at least in the easily catchable size range of > 20 cm) should be more sensitive. However, immigration and enhanced survival of eels at lower densities are likely to mean that the impacts of recruitment failure or glass eel overfishing may not become apparent for up to 10 years (Section 8.3.1). The effects of exploitation of yellow eels might be detectable over shorter time scales if fishing mortality is high enough (Appendix C.2.2). As discussed in previous chapters, such effects have been noted in relatively poorly-recruiting but highly-exploited systems, such as the IJsselmeer.

Given all these provisos, combined measures of density and biomass can be used to determine management reference points. These can be applied to individual sites or groups of sites in, for example, a whole catchment (such as the Dee, Section 3.8.1), a tributary (e.g. Tadnoll Brook, Section 3.9.2) or group of tributaries (e.g. different 'zones' in the Severn, Section 3.7.2).

Whether stocks are at or near carrying capacity can be judged from data from similar sites, catchments and regions given in Chapters 2 and 3 and in Table 4.3 (see also European data in Moriarty & Dekker, 1997). Having set a management reference point, comparisons can be made between one survey and the next and any declines quantified. In the light of uncertainties due to natural variations or measurement errors, management intervention may be triggered by a significant reduction in the stock below the reference level. The data available from this R&D project allow estimates to be made of reductions that may occur before a significant decline could be detected. The simplest and most robust test for critical declines between surveys due to fishing mortality (or other causes) is to compare appropriate data sets using a standard paired *t*-test, e.g. as used to compare resurvey sites in Chapter 3. The ideal number of sites (approximately 20-25) and methods and frequency of surveys needed to reliably detect significant changes are discussed further in Chapter 8 and Appendix C.

If the data are not amenable to such a test or as an additional indicator, a critical percentage level of change could be used to trigger management action. To explore such levels, the mean percentage of the change in density and biomass at each matching site between sampling years was calculated for a number of rivers. These data sets were selected because they comprised long time series and were judged to be suitably robust for a range of river types, as follows;-

- The St. Neot (Fowey catchment, Section 2.8) and Otter (Section 2.14), representative of productive and well-recruiting south-west rivers, with relatively high densities and biomass, with intervals between surveys of 1, 2, 3, 4, 5, 6, 7, 8 and 9 years between 1977-87. Unfortunately, only 1, 2 and 3 year inter-survey periods were available for biomass.

- The Stour catchment (Essex, Section 2.23.1), representative of less well-recruiting east coast rivers with relatively low densities but high biomass and average individual size. Data sets for density and biomass were available for inter-survey of periods of 3, 6, 9 and 12 years over 1985-97

- Data for the Lower Severn (1, 15 and 16 year intervals, 1998-99 and 1983-98-99) and Dee (15 years, 1984-99) and Lower Piddle (22 years, 1978-99) were used because of the long time spans covered and the comparable nature of the original and repeat surveys (Chapter 3).

Comparing the mean percentage changes in density and biomass of combined samples per river can be misleading because of high inter-site variability, as illustrated by the extreme outlier values commonly seen in data presented in Chapter 2. Taking the mean of the percentage change for each matching site in a survey programme helps to overcome such distortions. The means of the percentage changes in density and biomass calculated in this way were greater than -5% for all survey intervals at all sites except for those for the Lower Severn (1998-99) and Piddle (1977-99). In these data sets, mean changes in density were –26.9% and -50.3%

respectively, those for biomass were –25.0% and –29.7% respectively. These were the only data sets where changes have been shown to be statistically significant. Thus critical changes in populations appear to be signaled by a mean decline below –25% in density and/or biomass over all matching sites sampled. The number of sites per river used in these calculations varied between 9 and 36, compared to the 25 sites recommended in Appendix C.4.

From other survey data reviewed in this R&D project, the only other examples of temporal changes which exceed this level of decline are for the Devon Axe and Dorset Stour. Mean percentage density in six matching sites on the Axe declined by -62.6% between 1979-86 and by –66.9% between 1986-90 (Section 2.16) and catch numbers at 26 comparable sites on the Stour declined by >90% between 1992-98 (Section 2.19). The causes may be due to differences in survey methods and efficiency or, in the latter case, to migration barriers. Such declines should trigger immediate checks and possibly further surveys.

A decline of –25% in density and/or biomass (as relative measures of carrying capacity) is expected to be detectable and this bears similarities to the figures of ~30% for %SPR and silver eel escapement criteria discussed in Section 7.2.4. This adds support to the **recommendation** that;-

- **interim and pragmatic management objectives for England and Wales should be set so as to maintain current optimum stock status**

- **management reference levels should be based upon maintaining optimum density and biomass (as relative measures of carrying capacity) for sites and catchments, utilising electric fishing survey data given in other chapters**

- **significant reductions in the stock below the management reference level should be assessed on one or both of the following criteria (preferably relating to a minimum of 20-25 contiguous sites between one survey and the next);-**

  (a) there is a statistically significant decline (using, for example, a paired *t*-test) in density and/or biomass, and/or;-

  (b) the mean of the percentage changes in density and/or biomass is ≥ -25%.

Where there are no data available from previous surveys, estimates of baseline densities and biomasses for sites can be derived from comparable geographical regions and river zones from those given in other chapters. For example, sites about 20 km from the tidal limits on tributaries of rivers in mid-Wales at altitudes of 200–400 m AOD would be expected to support similar densities and biomasses of eels between April and September as the Afon Cwm (R. Dyfi) (see Section 2.26). It is not possible to set very specific management reference points using such spatial comparisons. However, if abnormally low densities and biomasses are found relative to those predicted, this should stimulate fuller and more frequent surveys until possible causes of have been clarified, especially any possible impacts of fishing mortality. An example of this comes from survey data for the River Exe, Devon (Section 2.15). Densities and biomasses at sites less than 40 km from the tidal limit (Fig. 2.21) were comparable to other rivers in SW England (Fig. 2.28), but mean values further upstream tended to be about 30-50% lower than predicted. No fisheries are believed to have contributed to these differences and it is possible that migration barriers (e.g. Chain Bridge Weir) are important (Section 2.15). This example illustrates a situation where further surveys are needed to clarify whether the differences are significant and if so, the possible causes. Although it is not possible to be sure when a significant reduction in the stock will be detectable, it is likely to be of the order of a mean difference for all sites of $\geq$ -25% compared to expected management values, as derived above for temporal changes between repeat surveys. Therefore it is **recommended** that;-

- **for sites where no historical data are available, management reference levels should be estimated from baseline density and biomass information for comparable site survey data from data in this R&D project. Major differences (e.g. means of $\geq$ -25%) between comparable sites should trigger further studies to determine whether fishing mortality is implicated and if controls need to be applied**

In conclusion, it must be emphasised that the primary management aim should be to ensure that stocks do not fall significantly below the management reference levels, based upon population indices. However, earlier discussions have shown there are many uncertainties involved in accurately monitoring eel populations and factors such as migration barriers can have major impacts on stocks. Therefore, it is **recommended that, in practice, if a significant decline is observed, the first response should be to check that sampling methods were not at fault. The role of anthropogenic influences other than fishing (e.g. migration barriers, habitat change or pollution incidents) should also be clarified as quickly as possible.** Determination of length class frequency (LCF) should also be carried out (if not already done) as an additional check, as discussed in the next Section.

7.3.4 Population structure indices

Declines in the proportions of small eels (< 20 cm) and increases in those of larger eels in off-estuary and low river sites could be indicative of recruitment failures or overfishing of glass eels. Statistically, changes in LCF distributions might be more sensitive indicators of such fishing mortality than density or biomass (e.g. see Section 3.9.3 and Appendix C2.1). Overfishing of yellow/silver eels might be expected to lead to declines in larger size classes. However, LCF distributions are often variable in estuaries and in low density sites in upper reaches of rivers and stillwaters where commercial sized eels >30 cm are most common. This is because sample sizes available are often small and distributions are commonly multi-modal and variably skewed because of inter-seasonal and inter-annual variations in immigration and emigration (e.g. see Sections 2.6 and 2.10). Furthermore, natural variations in recruitment cause substantial inter-annual variations in the mean size of eels in upstream populations. Francis and Jellyman (1999) conducted simulation experiments on New Zealand freshwater fisheries for *A. australis* and *A. dieffenbachii*. They concluded that because of such variations (compounded by typical levels of sampling error), biomass would need to drop by more than 40% before changes in the mean size of eels could be reliably detected. Changes in LCF and in modal and average size (determined directly or from biomass ÷ density) in upstream habitats may therefore be useful indicators of high yellow/silver eels fishing mortality, but they are not as statistically robust as those for lower river populations.

Another approach using electric fishing survey data of standing stocks is to use indices based on size groups that relate to different life stages. This has been attempted in France, the only other European country that has conducted nation-wide integrated electric fishing surveys comparable to those in England and Wales (Lambert & Rigaud, 1999). Such surveys have, however, only been carried out intensively in the mid- to late-1990s, and only a few comparable data sets exist for the 1980s for rivers in Lower Normandy. Length data have been collected in all surveys, but single fishing runs have generally been used, thus yielding semi-quantitative data only on density. Samples were divided into size groups representative of different life stages, i.e. 50-150 mm (glass eels and migrant pigmented elvers), 150-300 mm (sexually-immature yellow eels), and so on up to the largest size groups, female silver eels of 600-1200 mm. The relative abundances of each size class per site sample were then examined relative to distances from tidal limits in different catchments, regions and nationally. It was hypothesised that these studies would reveal indices that could be used to clarify impacts of fisheries, escapement and habitat differences, although setting management reference points was not mentioned as such. It was concluded, however, that the uses of such indices were severely limited by high levels of variability. Historical comparisons suggested that the relative proportions of the largest eels had declined in catchments in Lower Normandy, but these were not statistically significant. Generally, densities (given these were only semi-quantitative) and relative distributions of size classes and males *v.* females with distances from tidal limits were similar to those observed in England and Wales, independent of river basin size. There was no obvious evidence of impacts of glass eel fishery mortality.

From the above discussions (see also Appendix C2.1), it can be concluded that changes in modal and average size and size class distributions can provide information about recruitment failures and glass eel fishing mortality in off-estuary and low main river and tributary sites. However, natural variability can be very high and such data are less robust for detecting changes due to growth overfishing in other river sites. They may be able to give indications of growth overfishing in poorly-recruiting but highly-exploited systems such as the IJsselmeer (Dekker, 1996), but comparable fisheries do not exist in Britain. Therefore it is **recommended that as measures additional to density and biomass, population structures should be assessed via length class frequency analyses. Any statistically significant change between one survey and the next for a group of contiguous sites (e.g. for a catchment or tributary) should trigger further confirmatory studies and possible management intervention. Changes in average size (e.g. estimated from biomass ÷ density), relative proportions of silver eels and sex ratios can also give some estimates of stock status and changes, although these would be generally of low precision**.

7.3.5 Spatial distributions of populations

Another measure of the status of stocks and possible fishery impacts is the distribution of populations in catchments in terms of presence/absence. As discussed in Section 4.4.4, population densities naturally decline with distance from tidal limits as habitat segregation occurs, depending on the number of initial recruits and relative densities and carrying capacities in lower river stretches. As pointed out above, density and biomass may be particularly variable in upper reaches and not give good indications about fishery impacts. However, at least in longer rivers, changes in the distance upstream above which, on average, no eels at all are found could be indicative of the relative number of initial recruits or fishing mortality.

Detailed examination of historical presence-absence data for rivers in England and Wales show eels to be largely absent at high altitudes on moorlands and mountains and where there are major migration barriers, as would be expected. However, eels are absent from upper parts of the main river and tributaries in large lowland catchments such as the Thames although they would be expected to be present (Section 2.22.1). Strictly, this means that stocks are below management reference levels but this is in comparison with non-quantitative information on stocks in past centuries. No clear changes have been detected by electric fishing, fyke netting or in power station screen catches since surveys began in the late 1970s. Similar presence-absence results were found in some of the rivers in France surveyed by Lambert and Rigaud (1999). The greater extension (up to +800 km) in very large rivers such as the Loire and Rhone is probably due to higher levels of initial recruitment, because of the favourable positions of their mouths relative to Atlantic migration pathways. Furthermore, upriver migration may be easier because of the size of these navigable rivers and the existence of large ship-locks, compared to heavily-regulated systems like the Thames, with its many migration barriers.

Changes in distribution range can be monitored with little extra effort during routine fishery surveys. Any contraction of range in the future should trigger investigations to check whether this is due to fishing mortality downstream or locally. Range of distributions should be monitored in other upper catchment zones in the Severn and the rivers of N. Wessex because they support known glass eel fisheries. No catchments have been identified where yellow eel fisheries have clear impacts on distributions. It is possible that the Essex rivers might be useful in this respect and they are recommended as important systems for future monitoring in Chapter 7. As an example of a river with minimal fishing pressure, the Dee should be chosen for comparison, especially in view of the availability of historic data (Chapter 3). It should become easier to track any changes in distribution in the future in a wide range of catchments via routine survey data being collected by the Database and Atlas of Freshwater Fisheries for GIS presentation.

In summary, it is **recommended that presence/absence data be collected for the upper parts of all catchments during routine or eel-specific surveys (especially for the Thames, Severn, Dee and N. Wessex and Essex rivers) and contractions in range should trigger investigations to check whether this is due to fishing mortality. If overexploitation is implicated, appropriate control measure should be instituted without delay.**

7.3.6 Management reference points in the European context

The recommendations made above must be viewed as interim ones for England and Wales because of the eventual need for international actions to protect the whole European stock. In other countries, there is stronger evidence for deleterious impacts of recruitment declines and fishing mortality. In these cases, more conservative reference points need to be set and more stringent fishery controls may be necessary. Ideally, further research on recruitment-stock-escapement relationships and fishing *v.* natural mortality is required to provide more robust models on which to base rational and integrated management at the international level. Appropriate monitoring systems then need to be instituted. These issues are discussed for England and Wales in Chapters 6 and 8. However, no other countries in Europe are currently making any concerted efforts to meet ACFM/ICES requirements. The only related international and integrated approach being pursued at the moment is a Common Fisheries

Policy/EC Concerted Action research programme (No. 98/076), aimed at establishing a Europe-wide recruitment monitoring system for glass eels.

In conclusion, the recommendations made above for setting management reference points based on the *status quo* are currently applicable for pragmatic purposes for England and Wales but may need to be changed in the future. The recommendations will help ensure the best national contribution is made to the recovery and sustainability of the total international stock. In the absence of better means of setting management reference points, similar approaches based on using population indices as surrogate measures of glass, yellow and silver eel escapement could be adopted in other countries. However, few other countries have such extensive and integrated fish survey programmes as England and Wales. Whilst there have been many isolated research studies which yield information on local and regional baselines for eel density, biomass, etc., monitoring in many countries would have to be greatly improved and better targeted to provide adequate information for management purposes. Monitoring is discussed in detail and recommendations made in the next chapter.

Although these pragmatic approaches based on population indices must be used in the short-term, it is **recommended that research should continue into stock-recruitment-escapement relationships to produce more robust models on which to base national and international management reference points in the future.**

7.4 Management Control Options

7.4.1 Introduction

If there are significant reductions in stocks below the management reference levels, management controls must be brought into play as part of the National Plan. This section reviews various management options, some of which may be easy to institute in the short term using byelaw provisions whilst others could be used in the future after changes in supporting legislation. Discussions are also expanded to encompass proactive measures (eel passes and stocking) which can be used to overcome anthropogenic impacts such as migration barriers and help compensate for poor recruitment. These could help augment production and spawner escapement and hence the British contribution to the European eel stock.

International eel markets are very volatile, leading to large short-term fluctuations in fishery pressures (as witnessed by the recent rapid changes in the markets for glass eel as seed-stock for aquaculture in Europe and China). Therefore fishery legislation needs to be capable of rapid responses, between years and, if at all possible, within years. It may be possible to combine controls on eel fishing with other requirements, e.g. controls on fyke netting to protect migratory salmonids or prohibition of in-river traps because of flood-prevention requirements, as has been done elsewhere in Europe (see examples below). However, for maximum effectiveness, legislation should be designed specifically to protect eels.

7.4.2 Management responsibilities

As already recognised by the Environment Agency, regulation and management need to be more consistent between areas and Regions. Under the National Plan, setting and monitoring management reference points and imposition of any fishery controls need to be coordinated centrally. Therefore, it is **recommended that;-**

- **overall management of the Plan should be under the direction of the Agency's National Eel Group or delegated to, for example, the National Coarse Fisheries Centre.**
- **management reference points need to take into account advice from ACFM/ICES and any future pan-European actions required by the EU.**
- **geographical and local differences in recruitment, stocks and fisheries mean that reference points will have to be set for individual catchments and areas at the Regional level in England and Wales to achieve national targets.**
- **monitoring of management reference points should be prioritised, planned and coordinated within a national context, with responsibilities for local and area programmes delegated to the Regions (see Chapter 8).**
- **fishery exploitation targets may have to set (e.g. to reduce fishing effort by 50%) or closed seasons imposed to assure compliance with management objectives at the local level. Such management interventions could possibly rely on local byelaws, but these need to be imposed consistently within Regional and national plans and legislative frameworks. All such actions will need to be integrated with any future European requirements.**

To simplify processes, it is **recommended that any primary legislation should be based upon giving the Environment Agency the necessary enabling powers, with details being dealt with in secondary legislation via byelaws.**

Other general issues include the need to adapt sea fisheries legislation to allow management of estuarine and coastal eel fisheries by the Environment Agency, coordinated as necessary with other responsible agencies. Also, enforcement powers and penalties need to be more consistent.

Various fishery control measures are already applied to eel fisheries in Europe (e.g. see Table 2.4, Moriarty & Dekker, 1997). In many cases these reflect local traditions and practices or have evolved primarily for the protection of other species, e.g. salmonid migration. Some of the measures are designed to enhance eel immigration or spawner escapement but no country has attempted or been able to set quantitative escapement targets because of the lack of knowledge of stocks, numbers of migrants, etc. There is very little quantitative evidence for the effectiveness or cost-benefits of any control measures in use. Given these uncertainties, management control options include the following;-

7.4.3 Prohibition of fishing

This R&D study indicates that bans on commercial fishing in England and Wales, at the levels currently practiced, would be unlikely to produce any significant changes in individual catchment populations in many cases, overall stocks or spawner escapement. Furthermore, fisheries appear to be declining because of competition from farmed and imported eels. However, **as a precautionary measure and to protect systems that are potentially vulnerable to fishery exploitation, it is recommended that fishing for glass, yellow and silver eels should be restricted to traditional fishing waters (as stated as Recommendation 30 for glass eels in the Salmon and Freshwater Fisheries Review (2000).** The Agency's Byelaws Review (April 2000) consultation document expresses a similar view, but **'traditional' fishing waters need defining, e.g. as where a fishery has existed continuously over at least the last five years.**

The situation may change in the future, especially if recruitment should decline further. Commercial glass eel fishing is already banned in countries where supplies have been perceived to be relatively low (Sweden, Denmark, Germany, N. Ireland, Ireland, Netherlands, Belgium). In some of these countries (e.g. Denmark, Belgium and the Netherlands), some fishing is conducted by government agencies or under special licence for restocking only. Closure of traditional glass eel and other fisheries in England and Wales (especially in inter-tidal and coastal areas) would pose legislative and enforcement problems and could result in socio-economic hardship in some cases. Such fishing can provide important part-time and seasonal incomes for individual fishermen and many have made capital investments in boats, nets and traps. Eel fisheries also provide wider employment to netmakers, glass eel merchants, transporters and to those purchasing the eels, such as eel farmers and processors. Possible delays in implementing controls due to substantial objections to byelaw changes therefore need to be considered, as do possible demands for financial compensation for loss of historic fishing rights. **Prohibition on a catchment, Regional or national basis might however become essential according to the National Plan if stocks decline. Government funding may then have to be sought if compensation becomes a legal issue for loss of traditional fishing rights.**

7.4.4 Total allowable catches/quotas

These approaches are only used in some localised and long-established European eel fisheries. Limits are set (and enforced) by co-operatives, such as that based on Lough Neagh, or by other types of fishermen's organisations where they have delegated fishing rights on state-owned waters, e.g. as in some areas of the Netherlands. Catch limits are, however, designed to allocate the resource (i.e. to maintain as stable an income as possible for member fishermen), rather than restrict exploitation on conservation grounds. Limits have tended to be adapted over time in *ad hoc* fashions in the face of changing recruitment and/or market forces. No comparable systems exist in England and Wales and this R&D project has not been able to gather sufficient data to recommend catch size restrictions applicable to widely-dispersed and small-scale fisheries. It would in any case be very difficult to measure catch sizes and to enforce TACs. **It is therefore concluded that Environment Agency or MAFF-imposed TAC or quota approaches are not feasible or appropriate.** This accords with the Salmon and Freshwater Fisheries Review (2000) conclusions regarding salmon fisheries, and that gear and effort controls are preferable.

7.4.5 Gear number controls

In some European eel fisheries (e.g. Lough Neagh) gear number controls are imposed However, as with TACs and quotas, these are generally imposed and enforced by fishermen's organisations to help maintain individual incomes rather than as a stock conservation measure. Such organised fisheries and self-regulation do not currently exist in England and Wales.

For England and Wales, it is **recommended that the power to restrict gear numbers should be sought to allow better control of fishing effort if this is necessary to restrict exploitation.** Gear controls are included in Recommendation 103 of the Salmon and Freshwater Fisheries Review (2000). Furthermore, **legislation should allow for immediate and/or year to year responses, with licences being sold on a calendar year basis before the main fishing seasons start.** A degree of pre-planning might be possible from estimating possible mismatches from one year to the next between supply and demand. For example, demand for glass eels from China and Europe might exceed catches of *A. japonica* in the Far East and of *A. anguilla* in France late in one year, hence putting extra pressure on British fisheries as they start up in the spring in the following year. There is a degree of correlation between the Severn and, for example, the Loire catches, making it possible to make some estimates of likely British recruitment from French catches (White & Knights, 1994). Such issues are being examined in the on-going Environment Agency Economic Evaluation of Eel Fisheries R&D programme.

7.4.6 Gear type and usage controls

Gear controls are used in some European eel fisheries, e.g. the banning of bottom trawling in the IJsselmeer in 1970, restrictions to baited long-lining in Lough Neagh and regulations regarding net sizes and meshes in glass eel fisheries in Spain and Portugal. These measures have, however, not primarily evolved to protect eel stocks but as means of protecting other species and to promoting and stabilising catches and incomes of individual fishermen. In England and Wales, different gear restrictions are enforced via byelaws in some Environment Agency Regions, but these have generally been developed for the protection of other species (e.g. salmonid migration).

For England and Wales, it is **recommended that where fishing mortality leads to stock falling to unsatisfactory levels, controls on gear types should be imposed specifically to enhance escapement of glass, yellow and silver eels**. Some specific details, including ones on 'traditional' gears used in local areas (e.g. putts and pots), are already included in the Agency's Byelaw Review (April, 2000) document. Some additional key points arising from the current R&D are as follows;-

- **fishing for glass eels should continue to be restricted to the use of hand-held dip nets which are used continuously by one person from the bank and not from a stationary or moving boat. The maximum opening and depth and handle length should be 0.8 m^2, 0.6 m and 2.5 m respectively**. These restrictions help prevent the spread of fishing to other waters unsuitable for such a fishing method (e.g. those, like the Bristol Avon, with steep muddy banks). Restricting fishing to bank-side shallows should also allow for escapement of eels in deeper waters in mid-stream, especially actively migrating elvers and juveniles. There is, however, no quantitative evidence available to prove this. Such restrictions also have commercial benefits in that hand-netting is more selective and glass eels suffer less stress and hence lower mortalities during capture, compared to fishing from moving boats (as practiced in many of the French fisheries). A strong degree of self-regulation occurs because the dealers reject catches showing evidence of stress and damage.

- From earlier discussions, it appears that the extent of silver eel fisheries and the types of gear used in England and Wales currently have minimal impacts on overall spawner escapement. However, gaps should be required between instruments to enhance escapement from individual catchments. On the basis of evidence of 50-70% escapement in Rivers Bann and Shannon (e.g. see McCarthy *et al.*, 1993; Moriarty & Dekker, 1997), the Eel Fishing Regulations (N. Ireland) (1979) require that weir traps using stow (coghill) nets should not cover more than 90% of the width of a river. Similar regulations apply in the Republic of Ireland. A 10% gap is probably sufficient because spate flows carrying emigrants downstream would be fastest where the currents are least obstructed. Efficiencies are in any case greatly reduced in spate-runs because of blockages caused by debris trapped in nets. Other countries require larger free gaps of 30-50% (e.g. in Sweden) where nets are used in still or slow flowing waters to catch actively swimming silver eel migrants. These are generally local regulations, commonly tied to time controls on fishing to protect other species (e.g. migrating salmonids). Wolf-type grid traps can be mounted to cover the whole of the width of a river and could be much more efficient and therefore an escapement gap should also be left. On the basis of the evidence available and applying a precautionary approach, it is **recommended that fishing for silver eels should not be allowed to extend beyond the average levels of exploitation pertaining over the last five years. To promote escapement from individual catchments, no more than 50% of the total width of a channel should be blocked by any silver eel catching devices.** This agrees with the Agency proposal for a minimum 50% coverage by any instrument. Coverage can be greater if there are other suitable escapement routes nearby (e.g. a by-pass channel or weir just upstream).

- Fyke net designs can be stipulated and enforced to control overall exploitation rates and landing sizes, as discussed below.

7.4.7 Minimum and maximum landing size

Landing size limitations would help minimise any overfishing of smaller yellow eels, in accord with Recommendation 31 of the Salmon and Freshwater Fisheries Review (2000). Landing size limits are enforced in Sweden, Denmark, the Netherlands and Italy via setting of minimum mesh sizes for nets, although the primary aim is commonly to minimise capture of other species. Size limits are also imposed on the long-line fishery by the Lough Neagh Fisheries Cooperative, undersize eels being rejected at its central collecting station as eels are sorted into the sizes preferred by different smoking markets. The aim of all these limitations is, however, to maximise the catch of larger, more valuable eels, rather than as a true conservation measure. It would not be easy to monitor landings in the dispersed fisheries of England and Wales and sorting catches would be time consuming. **The best approach is therefore to enforce minimum mesh sizes for fyke nets and spacings between meshes, bars, etc. in other instruments such as eel pots and silver eel traps.** Fyke nets are the favoured catching method in England and Wales, these can be of different patterns, depending on the depth of water and currents pertaining in a fishing area. The key feature is that the minimum cod end mesh size should be set at a minimum of 19 mm stretched internal dimensions (12 mm for knotless nets) to ensure preferential capture of eels > 30–35 cm (40–

50 g) (Naismith & Knights, 1990a). It must be emphasised, however, that eels smaller than this generally have little commercial value, as in most other countries in Europe. Therefore, fisheries tend to be self-regulating in this respect, but enforcing minimum mesh sizes will help to minimise by-catches of other species. An appropriate gap between bars in silver eel traps is about 20mm to maximise capture of large silver eels whilst promoting escape of immature yellow eels (Tesch, 1977; Knights, 1982).

Recommendation 31 of the Salmon and Freshwater Fisheries Review (2000) states that consideration should also be given to imposing maximum size limits to promote escapement of larger females of high fecundity. The Environment Agency's draft National Eel Management Strategy and Byelaws review also mentions this issue. No country in Europe sets upper size limits for eels. Exclusion netting, grids or rings could be mounted in fyke net entrances or cod ends, suitable mesh or gap sizes can be found in Knights (1982) and in Naismith and Knights (1990). However, such restrictions could discourage eels of all sizes from entering and would exacerbate blockage problems, as was found by Koed and Dieperink (1999) with river fyke nets when using front-net guards of mesh size as large as 75mm. Emigrant silver eels are not commonly caught in fyke nets (Knights *et al.*, 1997) but they and larger yellow eels have a relatively high market value and can thus provide an important economic return for many fishermen. Given the additional potential problems of reductions in overall catches and net blockages, the use of such exclusion devices cannot be recommended. Requiring fishermen to size-sort catches after capture and to release large silver eels could be required but would be time-consuming.

In relation to fixed traps, the same argument applies about the important contribution made by large silver eels to the value of fishermen's catches. It is therefore recommended, as discussed in the preceding section, that escapement be encouraged by leaving parts of channels open to emigrants rather than trying to prevent entry of larger eels to or promoting their escapement from fixed traps.

7.4.8 Close seasons or time periods

These are currently in operation in most European countries, but only rarely are they designed as specific eel conservation measures. Most are based on traditional or practicable fishing season, on conflicts with fishing for other species (e.g. as in Ireland) or are primarily related to requirements to allow unhindered migration of salmonids (e.g. Denmark and N. Ireland). There have been no rigorous assessments of their effectiveness. It is clear, however, from earlier discussions that large inter-seasonal and inter-annual variations occur in migrations and activity levels and hence catches of all life stages. Partial close time periods have not proved to be effective in glass eel fisheries, e.g. weekend closures of the Vilaine (Brittany) fishery commonly lead to enhanced catches on the next open night (Briand *et al.*, 2000). Temperature is a key factor in stimulating migration of glass eels and has been used experimentally to set limits on the Vilaine fishery. The fishing season used to be restricted to between 15 November and 15 April and escapement was estimated from pass trap catches at the d'Arzal Dam to be 30 – 500 kg, i.e. between 0.3% and 3.1% of the annual commercial catch. In 1998, fishing was prohibited when temperatures rose above 12°C and escapement then increased to about 1 tonne, 5.9% of commercial catches (Feunteun, 2000). In comparison, annual trap catches at the head of the Severn varied markedly during1991-93, from 6.63 – 13.3 kg, i.e. about 0.8% of the estimated average annual commercial catch of 15 t and less than four orders of magnitude of the estimated total run in 1991 (Section 5.4.3). It must be re-emphasised, however, that such trap catches almost exclusively comprise actively-migrating pigmented 0+ elvers and 1+ and 2+ juveniles and not true glass eels. All life stages would be easily caught by the mid-water nets attached to rapidly moving boats in the French Biscay fisheries, but not by the dip net fisheries in larger estuaries in England and Wales. These preferentially catches glass eels moving upstream near the bank as tides begin to ebb. The majority of the trap catches are also achieved after the end of the main glass eel runs, thus many of the elvers and juveniles represent later-arriving migrants from outer estuary or coastal habitats. Trap catches in the Severn varied markedly from year to year mainly is response to water temperature, the threshold for migration being 14 – 16°C (White & Knights, 1997).

Close seasons for silver eels would also only be effective if they covered long time spans. For example, downstream runs measured via an electric counter in the R. Test (Hampshire) occur in bursts (probably associated with spates) but these occur at varying times between May and November (Section 2.21).

In conclusion, close seasons could be used to constrain fishing periods and thus prevent expansion of a fishery or permit restriction, but they will only be fully effective if they span relatively long time periods, e.g. the total known duration of local glass eel immigration or silver eel emigration runs. Therefore, **whilst close seasons could aid controls on fishery pressures, further studies are needed to fully assess the efficacy of short-duration or temperature-delimited glass eel or silver eel close fishing seasons in England and Wales before they can be recommended.**

7.4.9 Closed areas

Imposition of closed areas are recommended because these would be very effective locally in preventing fishing in waters where stocks have declined and where eels are particularly vulnerable to capture. The example of the high fishing pressure exerted on glass eels in Red Barn Dyke/Quicksand Pool leading to Leighton Moss SSSI in Cumbria has already been mentioned in this context. However, even in this example, the true impacts of the glass eel fishery on actual recruitment to freshwater are unclear (Section 5.4.3). Closed areas would, however, be a useful proactive management approach, i.e. for preventing extension of fisheries into new areas where glass eel or silver eel runs might be vulnerable in the future, as recommended earlier.

7.4.10 Licensing of fishermen

Fishermen should be licensed individually, with conditions set for the number and types of instruments and fishing location(s) they can use in any one year. This would help in ensuring long-term fishermen can be given precedence, whilst excluding any who have wilfully broken byelaws. **A further condition of renewal of a licence should be the provision of annual catch returns, including fishing effort data (e.g. as numbers of dip net fishing nights or number of fyke net-end layings).** This helps provide information on catch sizes, CPUEs and market statistics. This would help to improve return rates and although the quality of such information has generally been poor in the past and underclaiming of effort common, licensing and returns provide a channel of communication between the Agency and fishermen. This could be used to conduct questionnaire surveys, as recommended in Section 8.5.1.

7.4.11 Licensing of eel dealers, transporters and processors

No country in Europe operates a dealer licensing system for eels. Various countries (e.g. Italy) require information on catches and sales from dealers and markets, but this is very difficult to enforce and data quality is poor. Licensing of and requiring data from specialist dealers, transporters and processors in England and Wales could provide a means of (a) controlling exploitation (e.g. via TAC limits at which they must cease buying or exporting eels) and (b) gaining data on catches, stocks, market trends and the effectiveness of any fishery control measures being applied. This approach is suggested for eels in Recommendation 104 of the Salmon and Freshwater Fisheries Review (2000). The Environment Agency has responded by suggesting that this issue might be better addressed by licensing of fish farms and premises (Recommendation 118) and documentation and registration of fish movements (Recommendation 124). There are however no eel farms operating in England and Wales.

At present there are two main dealers/exporters of glass eels, and one main transporter/exporter of yellow/silver eels. Only the former operate centralised buying, holding or processing facilities. The provenance of glass eels brought in to 'elver stations' is, however, not revealed by fishermen. Therefore such licensing might be feasible in relation to general monitoring and control of glass eel catches, but not to management of recruitment and stocks in specific catchments. In relation to yellow and silver eels, unknown quantities are purchased *ad hoc* direct from fishermen by many small fish transporters, smokers, restaurateurs and other operators. The only quantitative estimate available was made by McKinnon and Potter (1993) who estimated that 34% of the yellow/silver eel catch in 1992 went to local processors or Billingsgate market. The latter market is now almost defunct and the few jellied eel processors left in the London area rely mainly on imported eels (Section 2.2.2). It would be difficult and costly to identify and police all these small operators. It would also be difficult to impose and enforce appropriate legislation and to get all dealers and processors, even the few major ones, to give full and accurate information because of commercial sensitivities. **Unless and until means can be found to overcome these problems, dealer, processor and transporter licensing cannot be recommended.**

7.4.12 By-catches

Whilst not related to protecting eel stocks as such, by-catches need to be considered because these could lead to undesirable impacts on other species. Recommendation 32 of the Salmon and Freshwater Fisheries Review (2000) states that the Environment Agency should review means by which by-catches of other fish can be reduced to the minimum (e.g via design and placement of fyke nets). Entrapment of other animals (e.g. otters and birds and invertebrates such as crayfish) also need to be considered.

As pointed out earlier, many regulations for eel fishing in Europe are primarily directed at protecting other species, although there is little hard evidence for their efficacy. The only data on by-catches in England and Wales comes from extensive fyke netting (total of 1593 end days) in rivers, stillwaters and the estuary in the Thames catchment (Naismith & Knights, 1994). This showed that by-catches and mortalities are not generally of great significance in regularly attended nets. However, it is **recommended that fyke nets and other capture devices should not be left unattended for more than one day or tidal cycle so that any by-catches can, if at all possible, be released alive.** Despite the results of the above study (and supporting data in other work cited), it is possible that impacts on other species may be greater in some waters and that the precautionary approach might need to be applied. By-catches would be difficult to prevent by changing the design of fyke nets, except by using mouth-exclusion devices to prevent the entry of large fish, such as those discussed earlier for excluding large eels. It is therefore **recommended that if by-catches of other species are found to be a problem, the only effective controls are to preclude their use in places and/or at times when such species may be endangered.** Such controls are already practiced in some Regions, e.g. for the protection of salmon smolts.

Problems can be experienced in catching surface-swimming waterfowl if the tops of fyke nets are exposed above the water surface. It is **recommended that fyke nets used in non-tidal waters should never be set with the tops of hoops or leader nets exposed above water levels.** In reality, this is a fairly rare occurrence as fishermen do not wish to make their nets obvious because of the dangers of theft and vandalism. In tidal waters, fyke nets can become fully exposed at low tide and whilst eels can often survive in air, other fish species may not be able to do so. Therefore it is **recommended that fyke nets used in non-tidal waters should not be set and left so that they become fully exposed at low tide.**

In relation to possible entrapment of diving birds or mammals, provision of guards equivalent to those already in use to exclude otters should be used. Otter guards come in various forms, such as a rigid 85 x 85 mm square grid mounted in the at the fyke entrance (Jefferies *et al.*, 1988). Such guards can, however, lead to blockage problems in water with high laminar velocities and debris loadings, e.g. in estuaries like the Thames (unpublished information, Naismith and Knights, 1993) and in fast-flowing rivers (Koed & Dieperink, 1999). Otters are unlikely to be found in such habitats but application of the precautionary principle leads to the **recommendation that otter guards should be used in all waters.**

7.5 Other Management Options

Whilst not providing controls on fisheries to answer the requirements of ACFM/ICES, proactive management options can be used to compensate for recruitment failures and any fishing mortality and to enhance production and hence spawner escapement. They may also be required where stocks are affected by non-fishery impacts, e.g. by migration barriers. Such measures include the use of stocking (perhaps involving levies on glass eel fisheries) and the construction of passes on migration barriers to enhance production in under-utilised waters (Knights and White, 1997, 1998). Screening of water intakes could reduce mortality, but such losses are probably minimal in England and Wales (Chapter 5). Although habitat loss and degradation over recent decades have not proven to be of major importance nation-wide (Chapter 5), protection and enhancement of local habitats currently used by eels (or that could be opened up by stocking or passes) must also be considered. Key issues relating to passes and stocking are considered in the next sections.

7.5.1 The use of passes to enhance recruitment into freshwater catchments

Passes can enhance migration and upstream recruitment of young eels into freshwater where stocks are locally restricted because of migration barriers. Passes to unfished but under-utilised habitats would augment the national contribution to European stocks. Thus this should form part of the National Eel Management Plan for England and Wales. The provision of passes is covered in Recommendation 127 of the Salmon and Freshwater Fisheries Review (2000), although this should also mention migration of post-glass eel/elver life stages.

Local schemes may judged to be of benefit, e.g. in the Severn and Avon (White & Knights, 1994), but research is required to accurately predict the additional upstream recruitment that might be achieved in other rivers (Knights & White, 1998). Also, possible impacts of changes in density on sex ratios and the effects on aquatic communities generally need to be assessed beforehand. However, **as a general fisheries management approach, all opportunities should be taken to prevent weirs and other structures being a barrier to migration of eels and all other fish species. Passes for elvers and yellow eels can be of positive benefit, but the first priority is to determine the needs for passes in a particular catchment and to derive sustainable and cost-effective management objectives and to subsequently monitor these** (Knights & White, 1998).

Important catchment and site-specific factors are;-

- the number, types, purposes, management and distributions of obstructions
- estimation of the significance of each in terms of passability and the availability of alternative routes
- the extent and quality of habitat that would be opened up to migrants,
- the location and effectiveness of any existing fishways and
- the extra numbers and sizes of eels that might be produced
- the effects of increased density on growth rates, sex ratios and production of spawners
- the effects on aquatic communities (e.g. stocking would not be advisable where possible competition or predation could have negative impacts on other species of fish or on invertebrates, such as crayfish)

To quantify the impacts of barriers, commercial and fishery survey data should be analysed or data gained from catches in simple pass-traps mounted on the barrier(s) in question. The latter will also give an indication of how much a pass could augment upstream populations (Knights & White, 1998).

The wide variety of pass designs available has been reviewed by Knights and White (1998). Basic requirements are;-

- a flow of water to attract eels towards a pass,
- suitable design and placement of the entrance and exit to encourage climbing and to aid escapement upstream
- suitable water velocities down a pass and/or the provision of some form of climbing material to aid ascent.

The simplest and cheapest solution to render barriers climbable is to provide a rough surface (e.g. to form a 'rock wall'). Eel ladders can be provided up which eels can climb. These can consist of by-pass channels or pipes or troughs attached to barriers, provided with suitable climbing material, e.g. geotextiles, brushes or horticultural netting.

Long-term post-construction monitoring is essential to ensure passes are efficiently maintained and managed and to provide information for other schemes. Fitting traps on passes will provide useful information on eel migration, recruitment, population distributions and dynamics. Traps can also be used to provide eels for stocking, as discussed below.

7.5.2 Stocking strategies

Stocking can be a cost-effective means of restoring or maintaining yields in fisheries and enhancing spawner escapement (Knights & White, 1997). It is essential in catchments with barriers where passes are ineffective and in isolated waters suitable for eels. The cautionary notes mentioned in the Salmon and Freshwater Fisheries Review (2000) (Recommendations 46 – 57) should, however, be taken into account, such as the need for an assessment of possible environmental effects of re-introductions (e.g. enhanced inter-specific competition). It is also **essential to assess the needs for and to monitor the effectiveness and benefits of eel stocking,** as discussed below.

There have been relatively few successful detailed studies of stocking according to Knights and White (1997). None have been of sufficient duration to cover the whole of the freshwater life stage, although some long-term studies in Scandinavia are still in progress (e.g. Pederson, 1999). Ideally, stocking material should be derived from within the same catchment or from one with a similar orientation relative to Atlantic migration pathways. This is because there is some evidence that long-distance transfers (e.g. of French elvers to the Baltic) can affect the ability of the resultant silver eels to find their way back towards the Atlantic migration pathways (Westin, 1990). Moriarty & Dekker (1997) recommended further research into this, because it could counteract the benefits of transfer of glass eels/elvers from areas of high supply in Europe to those of low supply. Stocking material could be trapped at the tidal limits of a particular (or nearby) catchment. If catches were too low or variable, extra stocking material would probably have to be purchased at market value. Some system of levy on catches could be devised, but this would entail legislative difficulties. Another approach would to purchase fingerlings on-grown in eel farms or other culture facilities. There is some evidence that this is beneficial because natural mortality is relatively much lower for larger eels, because of the declining numbers of predators big enough to catch and consume them (Knights & White, 1997). This approach has been used to stock the Severn and Avon systems in the past, using on-grown eels purchased from the former Bristol Channel Fisheries. Significant local supplementation of naturally-recruiting stocks have been noted by Aprahamian (1988) and White and Knights (1994). The only sources of farmed eel in the future are likely to be from the European mainland. In Denmark, a system is already in place whereby eel farmers are required to offer part of their stock for sale to government agencies for use in restocking. Fisheries organisations make requests for such stocking material and this is overseen by government fisheries managers.

If stocking is considered to be a viable part of a recovery plan for England and Wales and stocking material can be made available, then the first issue is to decide who is to pay for stocking. If the principle of full-cost recovery is applied, fishermen themselves would have to pay, directly or indirectly. This should also apply if stocking is to be used as a compensatory stock management approach, although it could be argued that conservation of eels as an integral component of aquatic communities and biodiversity warrants central government funding.

Assuming stocking is only used as a conservation measure and no exploitation is allowed, suitable waters must be identified. Stocking densities need to be adjusted to ensure optimum survival, growth rates and sex ratios are achieved. Lower densities would be preferable because this will generally promote the development of relatively more later maturing and larger females of high fecundity. Knights and White (1997) concluded that in warmer and more productive still waters, these generally appear to be about 0.1 kg ha^{-1} (i.e. about 300 glass eels/elvers ha^{-1} or the equivalent weight of juveniles). The potential yield is about 20 kg ha^{-1} at 40-50 g $recruit^{-1}$. Data on survival rates are sparse but the evidence reviewed in Chapter 5 suggests a typical figure to be 20-30% between elver stocking and final exploitation. To maintain the same yields in colder and less productive lakes, stocking rates should be reduced to 150-200 eels ha^{-1}. In both cases, numbers ha^{-1} can be reduced if on-grown or wild-caught yellow eels are stocked but this will increase initial stocking costs.

In rivers, Knights and White (1997) recommended that eels should be scatter-stocked (to minimise density-dependent mortality) in the summer, when temperatures are high enough to encourage dispersal. Typical target densities used in previous stocking programmes have been 1-2 juvenile yellow eels m^{-2} in low productivity waters, rising to 4-5 m^{-2} in warmer ones with plenty of bottom cover and/or marginal vegetation and high productivity of macro-invertebrate prey.

Unfortunately, there is a lack of information on still waters in England and Wales to suggest which might be most suitable for stocking and what benefits might be achieved in terms of enhancing spawner escapement. Stronger cases can, however, be made for some of the poorer-recruiting east coast rivers. For example, although precise stock information is lacking, eels are often absent from upper catchments or densities appear to be low in NE coast rivers relative to similar rivers (e.g. in NE Scotland). More data are required for these rivers, but an example of a prime candidate for stocking is the Thames. Very few eels are found in the upper parts of the catchment, with only a few isolated populations (e.g. in the Windrush and Churn) that have apparently resulted from single local stockings in the past (Section 2.22). Simple calculations made in Section 2.22.5 showed that stocking the Upper Thames at low density (~1 eel $100m^{-2}$) to preferentially encourage the production of large female spawners could yield 1 x 10^5 eels (~ 20 t), carrying a potential 1 x 10^{11} eggs. If the total long-term annual catch of yellow-silver eels in England and Wales is about 300 t (Fig. 2.3, Section 2.2.2) and the sex ratio is about 1:1, this represents ~150 t of females. Therefore, the extra production from the upper Thames would comprise a minimum compensatory escapement, relative to this 150 t, of ~ 13%. It appears that **although stocking could have local benefits, it would have to be done in many rivers and on a very large scale to make significant contributions to the overall stock.**

7.6 Implications for Monitoring of Eel Stocks

For the National Eel Management Plan to succeed, **monitoring will be essential to help determine baseline stock indices where these are not known and to measure temporal and spatial changes in relation to management reference points. Monitoring of commercial fisheries will also clarify the possible roles of overfishing in stock decline.**

Monitoring information will also aid in selecting suitable options for controlling exploitation and in the assessment of their eventual efficacy. It is also essential in maximising the biological and cost-benefits of proactive measures, such as stocking or provision of passes

There is a pressing need for careful monitoring over the next few years because stocks and fisheries may well be entering a period of major change. As discussed in earlier sections, it is possible that the cumulative effects of recruitment declines since the late1970s-early 1980s might begin to affect stocks and fisheries. In addition, the indications are that market demands and hence fishing pressures will continue to decline in the near future. Therefore monitoring of any effects of reductions in exploitation could provide very useful information about past impacts of fishing mortality on stocks and about possible future needs for restrictive management controls and their efficacy. These issues are considered further in Chapter 8.

7.7 Summary of Recommendations for a National Eel Management Plan for England and Wales

7.7.1 Aims and Objectives

The principal aims of the Plan are;-

To maintain, improve and develop eel fisheries to permit the sustainable exploitation of the eel stock for recreational and commercial benefit, taking into account the unique biology and life cycle of this species

The key objectives are, via a consistent and standard regulatory and management framework, to;-

- Ensure sustainability of stocks by maintaining spawner escapement relative to exploitation
- Obtain and use best available information to inform management
- Maximise long term socio-economic benefits
- Minimise potentially damaging effects on the environment
- Support eel fishing through effective communications and provision of advice

7.7.2 Management responses in the short term

Recommendation 1
In the short term and pending the development of any over-arching EU management requirements, the precautionary approach should be applied in England and Wales to prevent fisheries impacting any further on current levels of escapement of glass, yellow and silver (spawner) eels. To achieve this, fisheries should not be allowed to expand in effort or geographical range beyond the averages pertaining over the last five years, unless and until this is proven to be unnecessary (Sections 7.2.2).

7.7.3 Setting management reference points

Recommendation 2
There is no clear evidence for any overall declines in eel stocks or fishery impacts in England and Wales over the last 25 years. Furthermore, there is a lack of adequate information on recruitment, natural and fishing mortality and spawner escapement upon which to base normal management targets and conservation limits. Therefore, pending any Europe-wide management initiatives, interim and pragmatic management objectives need to be set for England and Wales. It is recommended that these should be based upon maintaining current stock status, thus optimising escapement of all life stages, and hence the national contribution to the overall European eel stock (Sections 7.2.3, 7.2.4, 7.3.3).

Recommendation 3
Interim and pragmatic management reference points for maintaining optimum stocks should be based upon population indices, i.e density, biomass (as relative measures of carrying capacity) and population structures and distributions for sites and catchments, utilising electric fishing survey data detailed in this R&D report (Section 7.3).

Recommendation 4
The need for management intervention in a catchment would be signalled when stocks fall significantly below the management reference point. This should be assessed on one or both of the following criteria (preferably relating to a minimum of 20-25 contiguous sites) between one survey and the next (Section 7.3.3);-

- *(a) there is a statistically significant decline (using, for example, a paired t-test) in density and/or biomass, recognising that biomass may be a less sensitive indicator in the short term than density, and/or;-*
- *(b) the mean of the percentage changes in density and/or biomass is $\geq$-25%.*

Recommendation 5
For sites where no historical data are available, management reference levels should be estimated from baseline density and biomass information for comparable site survey data from data given in this R&D report. Major differences (e.g. means of $\geq$ -25%) between comparable sites should trigger further studies to determine whether fishing mortality is implicated and if controls need to be applied (Section 7.3.3).

Recommendation 6
In practice, if significant declines are detected, the first response should be to check that sampling methods were not at fault. The role of anthropogenic influences other than fishing (e.g. migration barriers, habitat change or pollution incidents) should also be clarified (Section 7.3.3).

Recommendation 7
As measures additional to density and biomass, population structures should be assessed via length class frequency (LCF) distributions and, for general indicative purposes, average size, proportions of silver eels and sex ratios (e.g. from the assumption that all eels > 40 cm are females). Any statistically significant changes in LCFs or anomalous changes in other parameters between one survey and the next for a group of contiguous sites could be indicative of overfishing and should trigger further confirmatory studies (Section 7.3.4)

Recommendation 8
Presence/absence data should be collected for the upper parts of all catchments during routine or eel-specific surveys (especially for the Thames, Severn, Dee and N. Wessex and Essex rivers) and contractions in distribution range should trigger investigations to check whether this is due to fishing mortality (Section 7.3.5).

Recommendation 9
Although management reference points need to be based in the immediate future on the population indices recommended above, research should continue into stock-recruitment-escapement relationships to produce more robust models on which to base national and international management targets and conservation limits in the future (Section 7.3.6).

7.7.4 Management responsibilities and legislative aspects (Section 7.4.2)

Recommendation 10
Overall management of the Plan should be under the direction of the Agency's National Eel Group or delegated to, for example, the National Coarse Fisheries Centre.

Recommendation 11
Management reference points need to take into account advice from ACFM/ICES and any future pan-European actions required by the EU.

Recommendation 12
Geographical and local differences in recruitment, stocks and fisheries mean that reference points will have to be set for individual catchments and areas at the Regional level in England and Wales to achieve national targets.

Recommendation 13
Monitoring of management reference points should be prioritised, planned and coordinated within a national context, with responsibilities for local and area programmes delegated to the Regions (see Chapter 8).

Recommendation 14
Setting exploitation targets and management interventions for fisheries may have to be taken locally, possibly using local byelaws, but these need to be imposed consistently within Regional and national plans and legislative frameworks. All such actions will need to be integrated with any future European requirements.

Recommendation 15
To simplify processes, it is recommended that any primary legislation should be based upon giving the Environment Agency the necessary enabling powers, with details being dealt with in secondary legislation via byelaws.

7.7.5 Management intervention options

Recommendation 16
The Environment Agency should be given the powers to prohibit eel fishing in any waters on a catchment, Regional or national basis if this becomes essential because stocks decline

significantly. Government funding may have to be made available if payment of compensation for loss of traditional fishing rights then becomes a legal issue (Section 7.4.3)

Recommendation 17
The Environment Agency should have the power, if this is necessary to restrict exploitation and maintain stocks and escapement, to limit the numbers and types of fishing instruments and the waters in which they can be used in England and Wales via conditions attached to individual licences (Sections 7.4.5, 7.4.6, 7.4.9).

Recommendation 18
Legislation should allow for immediate and/or year to year responses, with licences being sold on a calendar year basis before the main fishing seasons start (Section 7.4.5).

Recommendation 19
Fishing for glass eels should continue to be restricted to the use of hand-held dip nets of standard dimensions which are used continuously by one person from the bank and not from a stationary or moving boat (Section 7.4.6).

Recommendation 20
Minimum mesh sizes for fyke nets and spacings between meshes, bars, etc. in other instruments such as eel pots and silver eels traps should be set to minimise capture of undersize eels (Section 7.4.7)

Recommendation 21
Fishing for silver eels should not be allowed to extend beyond the average levels pertaining over the last five years. Where silver eel trapping devices are permitted, these should cover no more than 50% of the total width of a channel, unless there are other comparable nearby open routes (e.g. a weir) for escapement of spawners (Section 7.4.6).

Recommendation 22
Close seasons could aid controls on fishery pressures, but would need to span the whole period of glass eel or silver eel migration and maximum catchability to be fully effective. Further studies are needed to fully assess the efficacy of short-duration or temperature-delimited glass eel close fishing times/seasons in traditional fisheries in England and Wales (Section 7.4.8).

Recommendation 23
Fishermen should be licensed individually (and instruments identifiably tagged), with conditions set for the number and types of instruments and fishing locations and periods they can use in any one year. Provision of annual catch returns, including fishing effort data, should be a condition for renewal of a licence (Section 7.4.10).

7.7.6 By-catch control measures (Section 7.4.12)

Recommendation 24
Fyke nets and other instruments should not be left unattended for more than one day or tidal cycle so that any by-catches can be released alive.

Recommendation 25
Fyke nets used in non-tidal waters should not be set with the tops of hoops or leader nets exposed above water levels, those in tidal waters should not be set and left so that they become fully exposed at low tide.

Recommendation 26
Otter guards should be used in all waters, these can also help prevent entrapment of larger fish and other vertebrates.

Recommendation 27
If by-catches of other species are still found to be a problem, the only effective controls are to preclude fishing in places and/or at times when such species may be endangered.

7.7.7 Proactive eel management options

Recommendation 28
All opportunities should be taken to prevent weirs and other structures being barriers to the upstream or downstream migration of eels and other fish species (7.5.1).

Recommendation 29
Passes for elvers and yellow eels can be of positive benefit, but the first priority is to determine the needs for passes in a particular catchment and to derive sustainable and cost-effective management objectives and to subsequently monitor these (Section 7.5.1).

Recommendation 30
Stocking can be a cost-effective means of restoring or maintaining yields in fisheries and enhancing spawner escapement. However, it is essential to assess the needs for and to monitor the effectiveness and benefits and disbenefits of eel stocking (Section 7.5.2).

7.7.8 Monitoring requirements

Recommendation 31
Careful monitoring of stocks and fisheries is an essential component of the Management Plan (Section 7.6 and Chapter 8) to;-

(a) help, where necessary, to determine the baseline status of stocks and hence set management reference points via appropriate population indices
(b) measure compliance with reference levels and hence any needs for management intervention
(c) where necessary, aid selection of appropriate management options and exploitation targets
(d) check on the efficacy and cost-benefits of selected management options
(e) provide information on commercial fishery trends to inform management and help ensure sustainability and socio-economic benefits

8. MONITORING OF EEL STOCKS AND FISHERIES

8.1 Introduction

The overall Project aims are;-

- ***to establish recommended methodologies for future monitoring of stocks and fisheries (freshwater and coastal/estuarial) in England and Wales.***

Monitoring is required to provide the best information possible for the National Eel Management Plan for England and Wales proposed in Chapter 7 to achieve sustainability of eel stocks and fisheries. In the absence of adequate data to set management reference points on conventional stock-recruitment relationships or direct measures of spawner escapement (plus problems of assessing non-compliance), it was recommended in Chapter 7 that;-

- interim and pragmatic management reference points should be set so as to maintain stock status at carrying capacity and hence, indirectly, maintain escapement of spawners and other life stages
- specific reference points should be set to maintain optimum densities, biomass and size structures for catchments and groups of sites

Other population characteristics, such as mean size, proportions of silver eels and females and spatial distributions are also of potential use (as discussed further in this chapter).

Careful monitoring of stocks and fisheries is an essential component of the Management Plan (Section 7.6) to;-

- help determine the baseline status of stocks, where this is not already known, and hence set stock reference points for management
- assess compliance with reference points and hence any needs for management intervention
- where necessary, aid selection of appropriate management options and, if appropriate, exploitation targets
- check on the efficacy and cost-benefits of selected management options

In addition, monitoring should also be carried out to provide information on;-

- commercial fisheries to inform management of temporal and spatial trends and actual or potential impacts on stocks. This will aid management in ensuring sustainability and socio-economic benefits
- the needs for (and the subsequent efficacy of) management of other impacts, e.g. migration barriers and the use of passes and stocking

Note has been taken of the principal objectives of the Environment Agency's Monitoring Review and of Recommendations 69 and 70 of the Salmon & Freshwater Fisheries Review (SFFR), i.e. that monitoring should be structured and targeted to produce information effective in meeting management needs. Recommendations 29 and 70 of the SFFR have also been considered, i.e. that information on catches of eels and elvers should be improved and that monitoring of fisheries should be linked with monitoring of other components of aquatic habitats. European issues have also been taken into account, e.g. the ACFM/ICES and Working Group on Eel (1999) recommendations that integrated monitoring throughout Europe is needed. Of particular note is the CFP/EU Concerted Action Glass Eel Monitoring Programme which has been set up to produce recommendations for a Europe-wide network of monitoring stations. This is due to report in 2001.

Eel monitoring data may also have other uses, e.g. changes in migration patterns and recruitment could be indicative of climate change (Knights *et al.*, 1996). Knowledge of stocks is also an integral component of pollution monitoring (Knights, 1996).

Finally, it must be noted that developments in monitoring for the National Plan may be applicable to other countries, helping to promote international integrated efforts to monitor and conserve the whole European eel stock.

8.2 Aims and Objectives

These are to ensure that monitoring programmes provide the best local, regional and national information possible for managers to;-

- set management reference levels on the optimum status of stock density, biomass, population structure and spatial distribution
- measure as accurately as possible compliance with reference levels
- decide on appropriate management options and measure their subsequent cost-effectiveness

This chapter first discusses management of monitoring programmes by the Environment Agency and then the uses of commercial catch and effort data. Monitoring methods, locations and timings are discussed, with emphasis on electric fishing surveys of riverine populations. Discussions are, however, extended to estuarine and coastal stocks and to possible future developments in monitoring glass eel/elver recruitment and spawner escapement. Recommendations are made for recording, presenting and interpreting monitoring data. In Section 8.9, recommendations are made for the uses of monitoring in setting management reference points and assessing compliance

8.3 Management of Monitoring Programmes

To ensure the principal aims of the National Eel Management Plan are met, strategic, prioritised, cost-effective and consistent monitoring programmes need to be devised devised and coordinated centrally. It is **recommended that monitoring programmes should be directed by the Agency's National Eel Group or delegated to, for example, the National Coarse Fisheries Centre**. Similarly, **monitoring should be prioritised, planned and coordinated within a national context, with responsibilities for local and area monitoring programmes delegated to the Regions**. It is, however, **recommended that**

monitoring of selected catchments be funded centrally to ensure their long-term viability. Finally, **any management interventions in fisheries may have to be taken locally, but need to be imposed within Regional and national frameworks and plans**

Key area, Regional and national roles in coordinating and interpreting monitoring data and related management planning are discussed in later sections.

8.4 Monitoring of Resident Eel Stocks

8.4.1 Introduction

The pros and cons of assessing spatial and temporal trends in commercial fishery data and their uses in predicting actual or potential impacts on stocks are discussed first. To sample resident stocks, netting and electric fishing are the most common methods used on a regular basis and these will be reviewed in detail. Catches in baited traps and pots tend to be small and sampling efficiency and representativeness are unknown (Naismith & Knights, 1990a). Sonar techniques may have some utility in deeper waters, but only at night when eels are active and swimming in mid-water. Furthermore, the ability to distinguish and quantify smaller eels is poor and general applications of sonar are limited by the availability of specialised equipment, technology and expertise. Pass traps can provide supplementary information about upstream and downstream migration and on the impacts of barriers on catchments recruitment patterns.

After discussing electric fishing methods in more detail, this chapter will review and make recommendations for (a) survey locations, frequencies and timings and then (b) measurement of population parameters.

8.4.2 Commercial fishery monitoring

It was recommended in Section 7.4.10 that provision of catch returns be a condition of renewal of a licence. Although the quality of catch return data might continue to be poor, licensing can provide useful information on effort and CPUEs. The processes involved also offer a route for communication and interaction between the Agency and fishermen. Periodic newsletters and meetings will aid interactions, as practiced in the past by, for example, Southern Region. Therefore, it is **recommended that licence sales, numbers of licensees, catch returns and estimated CPUEs are monitored annually to help track spatial and temporal trends. These will provide information about recruitment changes, overfishing, market forces and socio-economic trends (e.g. due to market changes or impacts of fishery controls).**

Some authors have been able to locate sufficient commercial fyke net catch data of (presumed) high enough reliability to monitor long term trends in fisheries. For example, Svedäng (1999) demonstrated no apparent declines in CPUEs in fyke net fisheries on the west coast of Sweden between 1976-97. Similarly, no declines in stocks are apparent in the export data and returns from specific fisheries, such as that in the Thames Estuary, reviewed in Section 2.22.3. Although being effort-dependent and varying in response to market forces, the best source of information currently available on commercial catches is export-import data. Interpretations are complicated by errors and omissions in records and the influences of variable fishing effort, but analyses produce useful insights into exploitation trends and potential impacts on stocks, as shown in Section 2.2. Monthly EU and non-EU import and export data summaries can be purchased from agents for a subscription of about £150 per year. Therefore **annual export-import data analyses are recommended as cost-effective means of annually monitoring general trends in eel stocks and fisheries.**

Useful data about fishing practices, areas, temporal changes, etc. have also been gained from questionnaire surveys, a discussed in Chapter 2. It is therefore **recommended that a repeat survey be conducted, asking essentially the same questions as posed in 1992 by McKinnon & Potter (1993) and in 1993 by White and Knights (1994).**

8.4.3 Net sampling

Netting may be the only method of objectively sampling eels in deeper, turbid and saline waters. Beam-trawling and seine netting are, however, both inefficient, especially in sampling smaller eels, problems compounded by the eel's habit of burrowing into the bottom or hiding in crevices during daylight (Chapter 4). Pair- and other trawls will only sample larger eels in mid-water. Fyke nets are useful passive fishing devices for commercial purposes, but they are size-selective and the effective area sampled is unknown. Therefore catch results can only be used to give relative CPUE or as presence/absence for eels > 30 cm (Knights *et al.*, 1996). Their accuracy and precision in detecting changes in more open waters has been shown to be limited without very extensive and intensive studies by Naismith and Knights (1990a). For example, studies of natural *versus* fishing mortality in the Thames Estuary reached some tentative conclusions but involved > 1300 end nights of fishing and analyses of > 8500 eels caught (Naismith & Knights, 1990b). Mark-recapture techniques can be effective in smaller closed waters, but are severely limited by low recapture rates in more open waters with mobile populations, even at high levels of sampling and marking effort (Naismith & Knights, 1990a, b; Knights *et al.*, 1997).

In conclusion, experimental fyke netting cannot be recommended as a standard monitoring tool, although it may be the only means of sampling eels in deeper stillwaters, rivers, estuaries and coastal waters in discretionary studies. Instead, it is recommended that it is more cost-effective to use commercial fishery data, as discussed in Section 8.4.2.

8.4.4 Electric fishing sampling

As discussed in previous chapters and in Appendix C, sampling by electric fishing is affected by capture efficiency and high levels of natural variability. However, electric fishing is still the most widely available and cost-effective means of quantitative monitoring (Appendix C). It is therefore **strongly recommended that electric fishing is used as the principal means of determining stock status, setting management reference points and monitoring compliance.** Capture efficiencies (especially of smaller eels of < 200 mm) is higher in eel-specific surveys (Aprahamian, 1988; Lambert *et al.*, 1994). It was found in resurveys (Section 3.11.2) that routine multi-species surveys using three-run catch depletion methods are likely to underestimate eel populations by a factor of 3-4. Although data from widespread routine surveys can still be useful (see Section 8.4.6), more detailed data on population characteristics can be gained from eel-specific studies focussed on selected rivers. Therefore, it is **recommended that resources should be allocated at a national level for a regular core programme of detailed eel-specific surveys employing standardised electric fishing methodologies.**

Details of electric fishing methods for eel-specific surveys are given in Appendix C.5.1. Key points are that relatively high voltage and current settings, slow runs and long dwell periods are required, with continuous energisation of anodes. Stop-nets are not considered necessary (Appendix C.5.5). Surveys should be based on three fishing runs, or four if depletions are erratic. These methods would not be applicable in multi-species surveys because of time constraints and potential high voltage damage to other species. However, as much use as possible should be made of data from routine surveys. Although tending to underestimate eel stocks, these can be cost-effectively used, incidental to monitoring requirements for other species, to yield basic information on eel stocks. In such cases, comparability of methods between surveys is of key importance. Sites and timings of all surveys need to be standardised to ensure meaningful historical comparisons can be made. Details are given in Appendix C, along with discussions of resource implications, but key points are summarised below.

8.4.5 Sampling locations: primary monitoring catchments

Ideally, a large number of sites in many contrasting catchments should be regularly monitored specifically for eels, especially any that are known to support commercial fisheries. However, the cost-benefits of such extensive surveys are unclear because of the high levels of temporal and spatial natural variability in populations. For example, Lambert and Rigaud (1999) found that the uses of size distribution indices of eels were severely limited by high variability in electric fishing data from 1,895 sites, surveyed in 15 catchments over three years (1994-96). Instead, it is recommended that it is more cost-effective to select sites in a number of key primary catchments for regular detailed eel-specific surveys. Principal criteria for choices are;-

- representativeness of river types (short *v.* long, lowland *v.* upland), geographical region (east *v.* west coast rivers) and local sites (distance from tidal limit, altitude, etc.)
- rivers with and without fisheries
- robust historical baseline information on which to base management reference points is already available

The number of river systems selected will ultimately be limited by resource availability (Appendix C.5). It is **recommended that a minimum of four primary west/south coast catchments and at least one east coast one should be selected for long-term eel-specific monitoring. These should be surveyed on a rolling programme of one catchment per year**. The **Severn, Dee, Piddle and Frome** are of fundamental importance because of their geographical distribution and the quality and temporal span of the existing data sets. The **Severn** is also important because of the major glass eel fisheries located in the estuary, whilst the **Dee** serves as a 'control' west coast river with minimal fishing pressure. It is further **recommended** that the **Severn** monitoring programme should focus on **Zones 1-3**, above the glass eel fishery and tidal limits, fuller details being given in Appendix C.4. The **Piddle and Frome** systems are south coast rivers that both discharge into Poole Harbour, which supports a yellow eel fishery. Taken together, they offer a suitable number of sites of different types with good historical data to detect any future changes in stocks (Section 3.9). Furthermore, they have been revealed in this R&D study to have shown significant changes in population characteristics since the late 1970s. Any future changes need monitoring and causes clarified.

Both Chapter 2 reviews and the re-survey studies have indicated that eel populations in catchments more remote from the main glass eel-bearing Atlantic currents are likely to be more vulnerable to fluctuations in recruitment. They are also the ones that support fyke net fisheries. It is therefore **strongly recommended that at least one, and preferably two, eastern catchments be added to a formal eel-specific monitoring programme.**

The main candidates are **at least one of the lower tributaries of the Thames and at least one of the Essex rivers** (Blackwater, Colne, Stour and Chelmer, Section 2.23.1) because of the availability of good quality routine survey data covering reasonable time spans. Furthermore, the Thames supports an estuarine fyke net fishery and the Essex rivers, estuarine and coastal ones. They should be subject to the same single-species surveys as for primary catchments, using the intensive electric fishing methods recommended in Appendix C to establish the firmest baseline picture of stocks possible and then to monitor changes. Key indicator rivers or tributaries are;-

- the **Darent** and **Mole** tributaries are **recommended** in the Thames, because of their relatively small sizes and representativeness, positions relative to the estuary fishery and the tidal limit at Teddington and the extent of past and current knowledge of emigration, stocks and distribution (Naismith & Knights, 1993; Naismith, 1998). The Mole is possibly at an intermediate stage in recolonisation. By comparison with data from other on-going routine fish surveys of Thames tributaries, eel-specific surveys would provide information on whether recolonisation continues or stocks decline due to poor recruitment or other causes

- Of the Essex rivers, the **Stour** is **recommended** as a representative catchment (Section 2.23.1, Fig. 2.38) and because further detailed information on eel stocks is available on the Brett tributary for 1985-87 (Barak & Mason, 1992).

8.4.6 Sampling locations: secondary monitoring catchments

Generally, only presence-absence or relative abundance data have been collected in the routine electric fishing surveys conducted in the north-east, north-west and Wales, where attention has been focussed on salmonids. Although there is information for some NE Scottish rivers (Tweed and Dee, Sections 2.24.1-2), this leaves important gaps in our knowledge. The same is true for some rivers where fisheries might be having important impacts (e.g. South Wales rivers supporting glass eel fisheries). There are, however, other examples where there are some detailed but restricted historical routine survey data upon which to base management reference points and where resurveys using the same techniques could reveal possible reductions in stocks. In other cases, surveys for other purposes are planned where additional data on eels could be cost-effectively be gathered. **No strictly prioritised recommendations can be made for secondary monitoring catchments. Instead, it is recommended that the most cost-effective approach is that local area and Regional resources are devoted to ensure that the best quantitative (but most historically comparable) information is gained on eels during future routine fishery surveys.** To detect any significant temporal or spatial anomalies, comparisons must be made with management reference points previously determined via routine survey. Comparisons with eel-specific survey data would not be meaningful, because such surveys tend to yield population estimates up to four to five fold larger than those in routine surveys. It must also be emphasised that even presence-absence in large catchments can yield useful information in assessing possible changes in distribution range that might relate to changes in recruitment or fishing (see Section 8.4.12). All routine monitoring information should then be considered at a regional and national level to decide whether further eel-specific surveys might be warranted in the long term.

Catchments suggested for consideration for quantitative routine surveys in the future, together with reasons, are;-

- Fuller surveys should be undertaken of the **Warwickshire Avon** and, especially, at least one of the **Welsh rivers** entering the Bristol Channel and subject to glass eel fishery mortality, such as the **Wye**, because data are lacking for comparison with the Severn, and with other Welsh rivers, such as the Dyfi (see below) and Dee.

- A more comprehensive survey of the **Bristol Avon** catchment would provide a useful comparison with the Severn, because it forms a large catchment which should receive good glass eel recruitment but does not support a significant glass eel fishery (Section 2.4)

- regular monitoring of selected sites in the **N. Wessex rivers** should continue because they support important glass eel fisheries and some data on in-river stocks and distribution are already available (Section 2.5). There is also a need to clarify the roles of migration barriers and the possible benefits of installing passes on key examples.

- fuller and more careful routine monitoring of the distribution range in a large SW catchment system, such as the **Tamar and Tavy** (Sections 2.9-10), could act as relatively cost-effective monitor of changes in recruitment and stocks for comparison with similar studies of the Thames or other rivers on the east coast

- the **Fowey** warrants further monitoring because of the existence of good historical data sets covering 1977-87 but scarcity of comparable surveys in later years (Section 2.8)

- the **Test** (Hampshire) because of silver eel escapement data (available from 1996) from the fish counter at Nursling (Section 2.21)

- the **Yare** in Anglian Region, because it is one of the few east coast rivers where there is detailed information of populations, although this is for relatively few sites upstream of Norwich (Section 2.23.2)

- the **West Beck/Upper River Hull**, because of data from eel-specific surveys conducted in 1988-89 (Section 2.23.2)

- the **Tees** because of the existence of some data for eel migration via the Tees Barrage fishpass (Section 2.24)

- **Lower Leven and upper Eamont (Cumbria)**: surveys of 16 and 19 sites respectively were conducted in these river sections in 1999 and yielded incidental data on eel densities (Section 2.25). NW Region are planning this will extend to a 10 year programme of spring and autumn surveys to gain baseline data (with emphasis on salmonids), prior to any possible drought-order abstraction applications by North West Water. This would be a cost-effective means of gaining long-term data on eels stocks in north west England, incidental to the main purposes of the survey programme. It is therefore **strongly recommended that National support be given to this programme and sufficient resources be allocated for at least 10 years to ensure it also collects eel density, biomass, length class frequency and silver/female eel data.** This could perhaps be coupled to studies of silver eel escapement via fish counters (see Section 8.7).

- the **Lune** in the northwest, because of the availability of two fairly recent but comprehensive surveys (Section 2.25)

- the **Dyfi** in Wales, in view of the historical data set for the **Afon Cwm** tributary (Section 2.26)

8.4.7 Site selection criteria

As discussed in Chapter 4, eels are highly adaptable in terms of their habitat requirements, relative to distance from tidal limits and altitude. Almost any site that offers a reasonable mix of habitat and day-time refuges is therefore likely to be reasonably representative of that part of a catchment. Site selection criteria are considered in more detail in the Appendix (Sections C.2.4 and C.5.2), but it is **recommended that survey sites should be located within 50 km of the tidal limit and should be selected with representativeness and ease of sampling in mind.** Sites immediately above and below major obstructions should be avoided because they can interfere with migrations and distort both upstream and downstream population densities and structures. Ideal sampling sites are those on relatively small tributary streams of 3-6 m width, not exceeding 1 m in depth, with high clarity and a mixture of instream habitat including suitable refuges.

8.4.8 Number of sampling sites per catchment

The minimum number of sampling sites per catchment will need to be determined individually, taking into account the need to minimise inter-site variance. The analyses presented in Appendix C.3, based on inter-site variance and differing levels of statistical confidence and power, indicates a probable **minimum number of 25 sites per catchment.** The results of the R&D resurveys of the lower Severn and Piddle/Frome (Chapter 3) and other data sets reviewed in Section 7.3.3 help confirm this suggested minimum.

8.4.9 Survey frequency

For a long-lived species like the eel, which typically spends 6-20 years in freshwaters in England and Wales before migrating back to sea, changes in recruitment or overfishing will only be reflected relatively slowly in catchment density, biomass or population structure. For example, the modelling studies of Feunteun (2000) predicted time lags between changes in glass eel recruitment and those in density of yellow eels to be between two to five years, depending on differences in assumed Z mortality levels. Analysis of R&D resurvey data (Appendix C.2) suggests that readily detectable reductions in population density may not even become apparent for 5 – 10 years. The impacts of yellow and silver eel fisheries may also take 5 – 15 years to manifest themselves.

From the above (see Appendix C.3), it is **recommended that detailed eel-specific surveys of the primary and secondary monitoring rivers are conducted at least every 4-5 years.** If, however, **consecutive surveys show significant reductions in stocks or non-compliance with management objectives, confirmatory surveys may be necessary, perhaps at annual intervals, to provide a statistically sound basis for possible management decisions on how to restore stocks.** Equally, if there is evidence from any other sources (e.g. from commercial fishery data or other fisheries surveys) that changes might be occurring, shorter intervals between surveys must be used.

8.4.10 Survey timing

Peaks in density and/or biomass commonly occur in late spring-early summer and again in late summer-early autumn (Fig. 2.13). It is therefore **essential that resurvey dates for a given site are matched as closely as possible to the original, preferably between May and September.**

8.4.11 Population attributes as indicators of compliance with management reference levels

According to earlier discussions and recommendations, **population density, biomass and length class frequency distributions (LCFs) should be determined at all sites and on all sampling occasions in eel-specific surveys (Appendix C.2). LCFs should also be determined in multi-species surveys if resources allow. Relative proportions of silver eels and sex ratios can also provide useful subsidiary information.** Fuller details are given in Appendix C, but basically, numbers should be counted (distinguishing between yellow and silver eels), total weight of the catch recorded (to the nearest g) and lengths of all eels (anaesthetised with benzocaine) should be measured (to the nearest mm). Approximate sex ratios can then be distinguished by assuming all eels > 450 mm are female. The relative degree of sexual maturation of silver eels can be assessed by dissection or via the Pankhurst ocular index (Pankhurst, 1982). This index reflects the changes in eye size and other ocular changes that occur as adaptations to the under-sea visual environment. It is doubtful, however, that the time taken to determine maturity by these methods will add much information of use in monitoring stock reference levels.

8.4.12 Population distribution range

Because distribution range within a catchment is a potentially useful indicator of population change, it is **recommended that presence/absence of eels should be recorded in all surveys. This data can be assessed locally or Regionally, but also via country-wide**

spatial and temporal comparisons via the Database and Atlas of Freshwater Fisheries initiative, using GIS approaches.

8.4.13 Collection of habitat data

It is recommended that basic habitat data should be collected for each site on each eel survey occasion (Appendix C.5.6). If stocks should change significantly, this will help distinguish the effects of changes in habitat quality from factors such as fishing mortality and provide information on what habitat improvements are needed.

8.5 Monitoring of Estuarine and Coastal Stocks

Information on estuarine and coastal stocks is important where commercial fyke net and pair-trawl fisheries exist but also because of migratory interchanges between estuarine and freshwater populations. Furthermore, unknown numbers of eels spend most or all of their lives in brackish or marine environments (Tsukamoto *et al.*, 1998; Wickström *et al.*, 1999) and these possibly make major contributions to the breeding stock (Chapter 5 and Appendix A).

The only fishery-independent survey methods available are experimental fyke netting and, possibly, power station screen sampling. Fyke nets require very large amounts of sampling effort and are unlikely to be sensitive enough to reveal anything other than major changes, as discussed in Section 8.4.3. **Instead, as recommended in Section 8.4.2, commercial fishery data should be analysed.**

Useful long-term comparative data series of yellow eel stocks have been gained from power station screens at sites such as West Thurrock (1974-1993) on the Thames Estuary (Section 2.22.2). Although sorting catches is labour-intensive, such passive sampling has supplementary benefits in providing information on other estuarine species and on commercially important marine species using the estuary as a nursery ground. It is **recommended that a replacement for West Thurrock P.S. is required (e.g. Barking, Tilbury, Lots Road) and shared funding could be sought (e.g. in conjunction with MAFF).**

8.6 Monitoring of Glass Eel/Elver Recruitment and Fisheries

8.6.1 Introduction

Although the use of population indices determined by electric fishing are recommended as interim and pragmatic means of setting and monitoring management reference points, these should be replaced by stock-recruitment or other approaches if these can be made robust enough. The current inability to accurately quantify glass eel/elver recruitment and spawner escapement is a major impediment to developing appropriate models. However, means of monitoring need to be considered. This would also help assess correlations with population indices and hence their validity as management reference points. Monitoring of glass eel/elver recruitment will be considered first.

It is important to assess recruitment to local catchments and wider-scale long-term trends to provide information about natural changes *versus* possible overfishing, and to aid the setting of management reference points and, if needed, targets for exploitation. This is especially apposite, in view of the declines in the initial recruitment of glass eels seen throughout Europe over the last two decades (Section 2.2). Commercial catch and effort data can also provide information on trends in recruitment and fishery pressures that might influence production and escapement of later life stages in future years (e.g. see Section 2.2). Furthermore, they provide information on catches according to Recommendation 29 of the Salmon and Freshwater Fisheries Review. Commercial data sources will be discussed first, but direct quantitative monitoring of recruitment is the ideal approach. Appropriate sites and methods are being sought in England and Wales during 2000-2001 to integrate with a CFP/EU Concerted Action Glass Eel Monitoring Programme, the aims of which are to produce a coordinated Europe-wide network of monitoring stations.

8.6.2 Export-import data

Glass eel merchants are unwilling to divulge commercially sensitive information. Indirect assessment of individual fishermen's catches and comparisons of best-estimates of total commercial catches and CPUEs have been of some benefit in assessing recruitment (Section 2.2). However, virtually all glass eels caught in England and Wales are sold as seed stock for aquaculture, direct consumption or restocking in other countries and are therefore recorded in Customs and Excise export data, except for some purchased for restocking the Lough Neagh fishery in N. Ireland. Monthly EU and non-EU import and export data summaries can be purchased from agents for a subscription of about £150 per year. Interpretations are complicated by trans-shipments, errors and omissions in records and the influences of variable fishing effort. However, such information can be used cost-effectively to track long-term trends in recruitment and fisheries and comparisons can be made with other European data sets, as shown in Section 2.2.2. It can aid predictions of future changes in yellow eel stocks. Commercial operators contacted agree with the general trends revealed by export-import derivations, although they were unwilling or unable to confirm the details. It is therefore **strongly recommended that import-export data be collected and analysed annually to cost-effectively monitor trends in glass eel recruitment and fisheries.**

8.6.3 Passive netting and trapping of glass and juvenile yellow eels

Passive netting and trapping methods catch glass eels (and pigmented elvers) carried by currents or during swimming. These include fine-mesh stow or fyke nets and traps on barriers, other devices for collecting glass eels when they seek shelter on ebb tides are reviewed below. All such devices would have to be deployed over the same duration of months as active netting methods (Section 8.6.4) and although ostensibly requiring less man hours because they are passive devices, would need regular checking, emptying and maintenance. The potential use of power station screen catches is also reviewed below.

(a) Stow nets

These come in various forms but are basically similar to fyke nets in being funnel-like with in-scales and a cod end (Tesch, 1977). The main difference is that they usually have a rigid rectangular opening and are anchored or staked to the bottom or strung from boats in positions to catch glass eels carried passively by flood or ebb currents. They can be fitted with fine mesh wings, possibly right across a channel, to funnel glass eels into the cod-end. They can be relatively efficient and cost-effective in appropriate locations, where currents and channel dimensions are suitable. Problems arise, however, if currents are fast and debris is trapped, blocking or damaging nets. Theft and vandalism could also cause problems, as would by-catches of older eels and other fish and invertebrates. River drainage and navigation constraints need to be taken into account and nets would require continual attendance during night-fishing and careful sorting of catches.

(b) Traps and collectors

Various forms and sizes of basket and pipe traps have been used (Tesch, 1977; personal communications with various fishermen and European eel scientists). These are faced into currents so that glass eels are swept or swim into them, past in-scales. Glass eels seek refuge by burrowing or hiding in crevices during ebb tides during daylight and to avoid strong currents. Some traps (or open trays or boxes) have been stuffed with straw or other natural or artificial fibrous materials to encourage such avoidance behaviour. The sampling efficiency of such devices is unknown.

Another form of collector which also encourages hiding behaviour comprises large bunches of long filaments of nylon and similar material. These have received some attention in Australia and prototypes are being tested in different location during studies of the Vilaine estuary in France in the spring of 2000 (Cedric Briand, personal communication).

(c) Power station intake catches

During surveys of potential monitoring sites in the Severn Estuary in spring 2000, the possibility of collecting glass eels from screens power station intake screens was considered. However, it was discovered that the location of intakes, large mesh sizes and poor retention of glass eels meant that no power station in England and Wales was believed to be suitable (Colin Taylor, personal communication). This correlates with the lack of small eels seen in past screen catches at Hinkley Point and West Thurrock Power Stations (Sections 2.3 and 2.22.2 respectively). It is therefore unlikely that screen sampling will be viable for glass eel monitoring.

Another possibility is to suspend nets in intake channels. For example, an Isaac-Kidd Midwater Trawl has been deployed used for two-four nights per week between late January-April in the intake channel (with a flow of two knots) of the Ringhals nuclear power station on the west coast of Sweden (Westerberg, 1999). No similarly suitable intake channels have, however, been located in England or Wales.

(d) Eel-ladder traps

Traps built specifically to catch glass eels/elvers or included in eel passes on barriers in estuaries have been used successfully, e.g. on the River Bann in N. Ireland to supply seed stock for the Lough Neagh fishery. They have also been used in various scientific studies in Europe and N. America (Knights & White, 1998). These consist essentially of a sloping channel ('ladder') down which water flows to attract migrants. The channel is provided with a substratum (e.g. brushwood or plastic mesh or bristles) to aid climbing, the eels then falling into an upper trap box (Knights & White, 1998). Traps in low-mid estuary locations can catch glass eels using selective tidal stream transport (as in the R. Bann system). Those further upstream, beyond saline fronts, tend to be very selective for pigmented elvers and > 0+ juveniles migrating after the main glass eel runs, because they have adapted to fresh water and are more ready to swim and climb actively. Thus, for example, even traps in the short 2km stretch of the Vilaine estuary (Brittany) below the d'Arzal Dam, let alone those at the head of a long estuary such as the Severn, tend to catch very few glass eels (White and knights, 1994). A similar situation is seen in the Red Barn Dyke/Leighton Moss SSSI trap studies discussed in Sections 2.25 and 5.4.3. This means that such traps do not directly measure glass eel recruitment from the sea but are more indicative of effective dispersive upstream migration into a catchment, after the glass eel stage.

If low estuary sites could be located and correlations between glass eel runs and catches validated, ladder-traps could be cost-effective means of monitoring initial glass eel recruitment. If other traps were then placed above a commercial fishery, they could provide information about fishing mortality. Catches are, however, strongly influenced by river discharges and water temperatures and by natural density-dependent mortality between the glass eel and elver stage. White and Knights (1994), for example, found no correlation between trap catches at the tidal limit on the Severn and commercial catches during 1991-93.

Traps can be simple and cheap in design, being fed by gravity-flows or by pumps (Knights & White, 1998). Construction and trap-running costs can be low and traps do not need such continuous attendance and night-time working as stow nets or (as discussed below), active netting. However, tidal inundation and spate damage, blockages and vandalism are common problems, as found in surveys of recruitment to the Severn and Avon (White & Knights, 1994) and by the Environment Agency in running pump-fed traps at the tidal limits of the Severn Estuary (A.C. Churchward, Personal Communication). Thus regular inspection, maintenance and emptying of trap boxes are essential. Conflicts with commercial netsmen, theft and vandalism are other potential problems. Timings of operational use are discussed in Section 8.6.6(c).

8.6.4 Active netting techniques

A number of active netting methods can be used to sample glass eels, each has advantages and disadvantages;-

(a) Off-shore netting

Some useful data have been gained from studies such as the Young Fish Survey conducted in February since 1975 in the Skaggerrak and Kattegat areas of the Baltic/North Sea, using an Isaac-Kidd or Modified Isaac-Kidd Trawl (Westerberg, 1999). These studies have shown, for example, that failures of glass eels to penetrate into the Kattegat and Baltic may be related to a lack of westerly winds during the winter and early spring. Trawling and plankton netting of leptocephali and glass eels has been attempted on the Continental Shelf off Britain and in the Bristol Channel respectively, but catches relative to effort and costs are generally very small and relationships to final recruitment into freshwaters obscure (Section 5.2).

(b) In-shore boat netting

Various forms of towed or pushed nets are used in European fisheries (Tesch, 1977). Some European studies have monitored catches made by individual boats (e.g. Briand *et al.*, 2000).

There are, however, no such fisheries in Britain and to set up comparable systems would be very expensive and efficiencies difficult to determine.

(c) Hand dip-netting

Dip netting (e.g. using a standard Severn-type or shrimping-type net) from the shore or from a platform or boat could be carried out by Agency personnel. It might be possible to employ selected fishermen or buy their catches but this is not recommended because very close supervision would be required to ensure constancy of effort per season and between years. Efficiencies and representativeness will be dependent on finding appropriate sites and the amount of effort expended during the migration season. Results can only be expressed in CPUE terms. Competition with commercial fishermen for sites could also be a problem.

(d) Drop-trapping and lift- and drag-netting

Closable box-like drop-nets have been used to sample glass eels throughout the water column off the west coast of Sweden, but catches were very small relative to effort and costs (Westerberg, 1999). Another method for sampling the water column is to use lift-nets where glass eels accumulate. This involves lowering a large fine-mesh net with a rectangular supporting frame into the water (e.g. from a boat or suitable bank wall or jetty), leaving it to lie flat on the bottom or low in the water and then lifting it rapidly. Variants of this pattern have been used commercially (e.g. see Tesch, 1977). Lift-netting at the Den Oever sluices on the IJsselmeer in the Netherlands have provided some of the best long-term data sets available (Section 2.2). Dragging a rigid-entrance net may also be applicable, as used in a boat lock at the entrance of the R. Ysell in Belgium (Moriarty, 1996). Such netting methods are resource-intensive and their sampling efficiency needs validating. Night-working and safety issues are also important considerations

8.6.5 Recommendations for sampling methods for glass eel monitoring

Preceding discussions show that passive sampling methods may be more resource-intensive than active netting, but that methodological problems, variable reliability, resource costs and safe-working issues may be limiting in both cases. The cost-effectiveness of different methods will depend heavily on locating suitable sites (as discussed below). Catches will be liable to short-term and year-to-year fluctuations due to changes in local environmental factors, especially freshwater discharge and temperature and variable admixtures of 0+ pigmented elvers and older juveniles (Vøllestad & Jonsson, 1988; White & Knights, 1994). Therefore, to achieve any value, monitoring of glass eel recruitment will involve long-term commitments. These issues will be considered in detail by the Concerted Action Glass Eel Monitoring Programme and although some preliminary national research might be warranted in the short term, it is recommended that any resource-intensive initiatives are held in abeyance until the report of the Programme is published.

Although pass traps have the disadvantages discussed above, work is proceeding at the Red Barn Dyke/Leighton Moss SSSI complex, comparing commercial glass eel catches with trapping (Section 2.25 and 5.4.3). Similar studies on east coast reedbed sites (e.g. Minsmere) are being considered by RSPB. It is **recommended that the Agency support any appropriate partnership studies on monitoring recruitment**

8.6.6 Key sampling site characteristics and timings

(a) Local site characteristics

Ideal sites vary due to local conditions and will have to be individually located. Fishing or trapping sites must be as near as possible to the sea or saline front to sample newly-arriving glass eels or at the tidal head to sample elvers and juveniles. Ideal sites would be found where glass eels become concentrated as they are transported by tidal currents towards the head of a narrowing estuary channel up to a barrier. They are also attracted by fresh water flows (e.g. below a sluice or head of a ship-lock) and may concentrate in areas of quiet water as the tide ebbs. Netting is most efficient as the tide ebbs and the glass eels concentrate towards banks or walls or behind obstructions where water velocities are lower. Sites must be located to provide safe positions from which to fish or service traps.

(b) Possible monitoring sites

Glass eel recruitment varies in quantity and timing in different parts of England and Wales, depending on position relative to the principal migration routes originating from the Gulf Stream/North Atlantic Drift (Fig. 2.1). Ideally, at least one site should be located facing westwards (e.g. in the Bristol Channel/Severn Estuary) with another on the east coast. Ideally, fished and unfished systems need to be monitored and comparisons made between glass eel and elver recruitment to help clarify possible influences of fishing mortality. Sites in the Severn Estuary and Essex rivers were explored in March 2000 as part of the Concerted Action Programme, none were found ideal but others will be considered in the European context as the Programme proceeds.

(c) Timings

Sampling of glass eels needs to begin on night-time spring tides in February-March as temperatures rise above about 6°C (White & Knights, 1994). Standardised sampling protocols will have to be developed according to local site characteristics. These could, for example, involve studies across a whole tidal cycle, sweeping with a dip net for 10 minutes once every hour or deploying a lift net for 5 minutes once every two hours (e.g. as in the Den Oever monitoring programme). Severn-type dip netting could be carried out for 5 minutes every 15 minutes during the hour up to high tide and then during every subsequent hour until the glass eels drop to the bottom (White & Knights, 1994). Although peak runs tend to occur on only a few tides, netting would have to be carried out nightly during March-May on high spring tides, continuing until catches decline to some criterion level, e.g. total nightly catches of less than 50 glass eels over three consecutive nights.

Ladder-traps designed to catch glass eels need to be operational from early March, as water temperatures rise above 4-6°C, and should be inspected at least once per week. When the main runs occur on high spring tides, they will need to be inspected every day. The frequency of inspection can be reduced after the main runs have ended in April-May. Late-arriving elvers and older migrants will not appear in traps until late spring-early summer, when temperatures exceed 14-16°C, but may still be caught into September if temperatures remain high enough (White & Knights, 1997). However, resource constraints may mean trapping ceases once a certain criterion is reached, e.g. less than 10 eels are caught per week for two weeks running. Samples should be counted and weighed and results expressed in CPUE as catch per trap per night. Sub-samples should be collected, as detailed in Section 8.6.7.

8.6.7 Sub-sampling for length-class frequency and other analyses

Whilst catch weights will be the primary measure of recruitment, conversion into numbers (e.g. via the commonly used relationships of 3000 glass eels kg^{-1}) may be misleading. This is because the weight and stage of pigmentation of individuals changes during the season (e.g. see White & Knights, 1994 and Briand *et al.* 2000). More importantly, there is evidence that the individual weight and condition of glass eels have generally declined over the last decade (Desaunay & Guérault, 1997; Dekker, 1998). Therefore, it is recommended that sub-samples should be taken of 100 glass eels per night once per collection week in any monitoring programme and length-class frequencies, condition factors and pigmentation stages checked over the migration season. Samples should then be preserved in alcohol and added to the EIFAC Eel Working Group Specimen Bank, held at Trinity College, Dublin.

8.6.8 Conclusions regarding glass eel/elver monitoring

Sites, standardisation of methods, timings, etc. will be considered in detail during the Concerted Action Programme. Although some preliminary research might be warranted in the short term, it is **recommended that any resource-intensive national initiatives are held in abeyance until the report of the Programme is published.**

8.7 Monitoring Silver Eel Spawner Escapement

Ideally, management reference points should be set on spawner escapement but, as discussed earlier, this is not currently feasible. Some predictions on spawner production have been made for England and Wales (Chapter 5 and Appendix A), but it is unlikely that good data could be gained from the few commercial fisheries still in operation. Low prices in recent years have been a major deterrent to continuous fishing effort. It might be possible for the Environment Agency to pay fishermen to run traps on a regular basis or to use Agency staff, e.g to run the Agency-built trap at Horton Mill on the River Colne, Thames Region. Pound-nets or silver eel trap could be integrated into studies of population dynamics and natural *v.* fishing mortality, as discussed in Chapter 6. However, any long-term and widespread trapping programmes would entail considerable practical and resource challenges. It might, however, be possible to cost-effectively monitor silver eel emigration using fish counters, incidental to other studies such as ones on salmonid migration. Data on eels from the counters on the Rivers Test (Hampshire) and Leven (W. Cumbria) have been discussed in Sections 2.21 and 2.25 respectively. It is possible that further details might be gained from the counter at Corby Hill on the River Eden (N. Cumbria), where 1999 counts are being validated from video footage. It is **recommended that further studies be conducted into the feasibility of monitoring silver eel escapement via fish counter systems already in use or under development.**

8.8 Recording and Presentation of Monitoring Data

8.8.1 Introduction

Reviews in Chapter 2 have shown that much of the survey effort in the past has been dissipated not just because of inadequate planning and execution, but also because of poor and inconsistent recording and presentation. Recommendations are made below, with the emphasis on producing visual presentations of data and the use of monitoring indices. These are useful for making general intra- and inter-catchment and regional comparisons (including comparisons with other species). They can help to reveal temporal changes and to pinpoint whether these might be due to fishery exploitation or to recruitment failure or anthropogenic causes, such as migration barriers or habitat degradation. However, they are not adequate for rigorous assessments of compliance with stock reference levels, these issues are discussed further in ensuing Sections.

8.8.2 Initial data collation and presentation

Raw data should be entered into a standardised Environment Agency fish population database which can calculate sampling efficiencies, estimate densities, etc. and output data to Excel (or Access) and thence into standard presentation forms and possibly standard indices/classification. This information can then be transmitted electronically between area and Regional and national centres.

8.8.3 Graphs and histograms

Annual commercial catch, effort and related fishery data can be added to graphs in Sections 2.2.1 and 2.2.2 to track temporal trends. For electric fishing surveys, standard software should be developed so that all data points can be easily processed for plotting graphs of density and biomass and mean weight per site against distance from tidal limits (using Excel or Access). It has been shown in earlier Chapters that such graphs are particularly clear ways of making comparisons within and between catchments. These help pinpoint hot and cold spots of baseline stocks and local distributions (e.g. see Fig. 2.12). They can also give general indications about changes over time, e.g. as shown in figures in Chapter 3 that illustrate changes in the rivers resurveyed in the lower Severn.

Histograms are useful ways of comparing spatial and temporal trends in length-class frequencies, giving indications about impacts of fishing mortality, recruitment failure, sex ratios, age-at-maturation and migration, as exampled in Chapter 3 resurvey studies and discussed in Appendix C.2.1. Cumulative frequency graphs also help reveal temporal changes (e.g. see Fig. 3.31). Software should be developed for rapid processing of size data into such presentation forms.

Although they cannot be used for monitoring precise management reference points, **graphing and histogramming of data are recommended for visual presentation of population data to help reveal spatial differences and temporal changes requiring further attention.**

8.8.4 Mapping

Eel biomass and density per site in terms of relative class or as presence-absence can be represented on maps of different scale using different symbols or, as recommended by the FCS, dots of specific colours. Examples of these presentations are shown in Figs. 2.6, 2.30 and 2.31 respectively. Mapping can help pinpoint spatial variations due to, for example, migration barriers (e.g. as in Fig. 2.6), but they are limited, unless used in transparent overlays or GIS representations, to detect temporal changes. Presence-absence maps (e.g. see Fig. 2.31) can be particularly useful in tracking changes in distribution ranges that could be indicative of major changes in stock characteristics. Although it cannot be used to set and monitor precise management reference points, **mapping is also recommended for visual representation and clarification of stock characteristics (including spatial distributions) and change.** Standard software should be developed to aid mapping, e.g. using ArcView GIS.

8.8.5 Monitoring indices

Although of limited use for setting and monitoring quantitative reference points for eel stocks, indices based on biomass have been accepted by the Environment Agency for general comparisons of monitoring data for eels and other species. The Fisheries Classification Scheme (FCS) is based upon an expanded HABSCORE database. The origins and numbers of data sets used to derive indices for eels, and mean biomass and density, are shown in Table 4.3(a). This shows that site data tend to be biased towards lower biomass and density rivers and reaches, mainly because many data sets were originally collected from surveys aimed primarily at salmonids. It must also be remembered that eel-specific surveys tend to yield higher catches than do multi-species ones (Chapter 3 and Appendix C), therefore the indices discussed below are only applicable to routine survey data.

The current FCS system proposed for eels is based upon biomass quartiles, using the same values at both Levels One and Two, with the divisions (g $100m^{-2}$) between Classes being:-

- **572** **A (Excellent)-B (Good)**
- **285** **B (Good)-C (Fair)**
- **141** **C (Fair)-D (Fair)**
- **70** **D (Fair)-Class E (Poor).**
- **0** **F (Absent)**

The FCS divisions equate to approximate densities of 4, 2, 1 and 0.5 eels $100m^{-2}$ in the database (cf. Table 4.3). Although applicable in many Regions, these divisions do not produce clear discriminations in rivers such as those in the South West Region and in the lower reaches of other rivers, according to reviews in Chapter 2. From density data for the R. Dart in 1987, for example, 87% of all sites where eels were found would fall into Class A. That the FCS database and biomass quintile classifications are unrepresentative for detailed nation-wide comparisons to eels is further demonstrated by extra data accumulated in this report. For example (Table 4.3), the mean biomasses in SW and Essex rivers (130 sites each) were found to be very similar (381.4 ± 376.5 *v.* 367.8 ± 373.75 g $100m^{-2}$) but the average size of individuals is much smaller in SW rivers and hence density is higher (mean densities of 11.3 ± 11.57 *v.* 1.51 ± 1.90 eels $100m^{-2}$ respectively).

Despite these criticisms, biomass classes, because they can be related to habitat productivity and carrying capacity, may be judged suitable for general national spatial and temporal comparisons. Ideally, however, density data are also essential for meaningful comparisons of eel stocks and population characteristics. A quartile classification could be used, starting from, for example, 20 eels $100m^{-2}$. Densities above this are not common, even in SW rivers and this value is cited as being at the top end of the range for European waters generally (Moriarty & Dekker, 1997). This would yield quartile divisions at 20 (Class A-B), 10 (Class B-C), 5 (Class C-D) and 2.5 (Class D-E) eels $100m^{-2}$. This would still not produce clear distinctions in classes in low density rivers and there is value in adopting a more arbitrary scale of, for example, 20, 10, 2 and 1 eels $100m^{-2}$. This would have the added advantage of providing indications of the relative numbers of females present, as they tend to predominate in waters where densities are < 1 eel $100m^{-2}$.

In conclusion therefore, it is **recommended that consideration be given to the use of combined biomass and density indices and descriptors for general comparisons of eel stocks and distributions**, using joint classes of;-

Biomass Class (g $100m^{-2}$)		**Density Class (eels $100m^{-2}$)**	
A -	**≥570**	**a -**	**≥20**
B -	**280**	**b -**	**10**
C -	**140**	**c -**	**2**
D -	**70**	**d -**	**1**
E -	**>0**	**e -**	**>0**
F -	**Absent**	**f -**	**Absent**

This would allow for general nation-wide comparisons with biomass classes for other species, but add more meaningful comparative expressions of eel stock status. For example, populations in a lower river site in the south-west of England would commonly fall into Class Aa because of a relatively high density of small eels, those in an upstream site at, say, > +40 km, into Class Bc because of a relatively low density but relatively high biomass of larger female eels (cf Fig. 2.28). Comparable sites in an Essex river would commonly fall into Classes Ac and Cd (cf Fig. 2.39). As an example of a temporal change, a low-river site classification might change between surveys from Aa to Ab. This could signal that overfishing of glass eels was occurring, but that the subsequent reduction in density and competition was allowing faster growth of older eels and hence maintenance of biomass.

This classification scheme does not offer the precision of absolute measures of density and biomass and would not easily account for the high levels of inter-site, inter-seasonal and inter-annual variability that commonly occur. However, it offers distinct benefits for broad national spatial and temporal comparisons during routine multi-species suveys compared to the use of indices based on biomass alone and is therefore **recommended** for adoption nationally.

The above system is more limited in its application to eel-specific surveys because capture rates tend to be much higher (by up to three to five fold). For example, the mean biomass and density values for lower Severn R&D survey sites for 1983, 1998, and 1999 (Chapter 3) all fell into the Aa class. Some broad spatial differences are, however, illustrated, e.g. the means for upper Severn sites would all fall into class Ee, those for the Dee into class Dc (ranging from Aa to Ee with distance upstream). The means for the Piddle and Tadnoll Brook have intermediate values, both falling into class Cb in 1999. Temporal changes between sites are also illustrated to some extent, e.g. 57% of lower Severn sites fell into class Aa in 1983, 67% in 1998 and 71% in 1999. Some sites show particularly large changes, e.g. Sites 8, 12 and 14 in Zone 2 rose from class Ed in 1983 to Ac, Aa and Ab respectively in 1998-99. However, the class divisions are generally too coarse and too many site values exceed the Aa class criteria to clearly demonstrate temporal changes in populations in the lower Severn and Piddle-Frome comparable to those seen in Chapter 3.

These discussions serve to emphasise that the revised FCS classification system proposed can only be recommended for application to routine multi-species survey data. Although useful in highlighting spatial differences and temporal changes, such index classes will be too broad and imprecise to set and monitor management reference points.

8.9 Monitoring of Management Reference Points

Population status indicators for sites in a catchment, tributary or group of tributaries can be determined from previous survey data or from the first survey in a new programme. That these are close to optimum levels and are appropriate for setting management reference points can be determined by comparisons with data for comparable sites in Chapters 2 and 3.

As discussed in earlier chapters and in detail Appendix C.2, the following are potentially key indicators of significant change and non-compliance with stock reference levels;-

- overfishing of glass eels or recruitment failure could be indicated by declines in density and, especially, the relative proportions of smaller eels (< 200 mm) in lower catchments sites. Reductions in competition and density-dependent mortality would encourage faster growth of yellow eels and hence maintenance of biomass. Thus biomass may not be a very sensitive indicator of change in the short term.

- overfishing of yellow eels >30 cm should in theory lead to a decline in density, biomass and a shift towards a higher proportion of relatively smaller and immature eels. Resultant reductions in density-dependent habitat segregation might be indicated by a contraction in inter-catchment distribution.

Although data presentations discussed in Section 8.9 can give useful indications of such changes, specific analyses are needed to whether stocks have declined significantly relative to stock reference levels. If the data available on density, biomass and length class frequencies are suitably robust for a group of sites, standard statistical comparisons such as paired *t*-tests should be used to assess significance of changes, as in the resurveys in Chapter 3. The other criterion is if the mean of the percentage changes in density and biomass is greater than –25%. Additional, if less robust, indications of non-compliance could be revealed by changes in proportions of silver eels, sex ratios and catchment distributions.

If there is evidence of non-compliance, any errors in survey methods need checking first and, if needed, further confirmatory surveys conducted. The next step is to check from licence records (and from local fishery officer knowledge) whether there is any fishery that might be implicated. If this is found to be the case, catch return data should be examined to assess potential impacts. National commercial catch trends should also be examined, e.g. to determine whether prices have risen to levels that might be encouraging exceptionally high levels of exploitation. If the impacts of the fishery can be quantified, it might be possible to set exploitation targets (via future licence conditions) to maintain a sustainable fishery, e.g. by halving effort on a gear-number basis. However, it is doubtful that sufficient knowledge of natural *v.* fishing mortality, replacement by immigration, etc. is available and thus a precautionary approach may have to be taken, e.g. all fishing may have to be banned until stocks recover.

In the above scenario, if no fishery is implicated in non-compliance with stock reference levels, other causes must be sought. For example, glass eel catch data (including data for other European fisheries collected by ACFM/ICES) might reveal drastic declines in initial recruitment, independent of fishing mortality. This may then require national and international applications of the precautionary approach to all eel fisheries to protect the European stock. At the local level, examination of graphical data and mapping presentations (including those using general FCS-type indices for eels and other species) will also be helpful. For example, anomalies in spatial relationships might indicate a migration barrier is impacting upstream stocks. If FCS-type indices for all fish species show declines, then pollution or drastic changes in habitat quality might be implicated. In these cases, other management options must be selected to assure compliance with stock reference levels, as detailed in Section 7.5. After implementation of any fishery controls or other management options, regular monitoring should be carried out to assess their effectiveness.

8.10 Regional and National Roles

Overall planning and prioritisation of monitoring programmes should be a national responsibility, e.g. coordinated by an appropriate group at the National Coarse Fish Centre. As recommended earlier, the rolling programme of eel-specific resurveys of primary catchments and sites should be funded centrally. Central funds may also need to allocated to support other monitoring initiatives (e.g. partnership or R&D studies, as suggested in Chapter 6). Monitoring should be carried out locally, under Regional guidance. Initial data analyses, presentations and reports should be produced by local fisheries offices associated with specific surveys. Raw and processed data and reports should also be transmitted electronically to Regional and national centres (e.g. the National Coarse Fish Centre) for archiving and processing for the Database and Atlas of Freshwater Fisheries (DAFF). This will aid the production of regular Regional and national overview reports to inform the wider management of eel stocks and fisheries. Reports can also then be made to MAFF and thence to ACFM/ICES/ to inform Europe-wide management.

From data presentations and reports, management reference levels can be set Regionally for catchments and sites (if not already done so) and non-compliance with them identified. Regions should then consult with national offices to check that any problems are local ones and not country-wide or international (e.g. due to catastrophic recruitment failures). Needs for confirmatory studies and remedial measures (e.g. imposition of appropriate local, Regional or national fishery controls) should then be decided via consultation between local and Regional officers and the national centre. Monitoring information can also be used to assess the needs for any proactive compensatory management measures (e.g. passes or stocking). Decisions should also be made about further monitoring to assess the effectiveness of management measures selected.

8.11 Summary of Recommendations for a National Eel Monitoring Programme

8.11.1 Aims and Objectives

These are (Sections 8.1 and 8.2) to;-

- **develop the best methodologies for strategic, prioritised, cost-effective and consistent monitoring programmes to ensure that the objectives of the National Eel Management Plan (Chapter 7) are met**

This requires that monitoring programmes provide the best local, regional and national information possible for managers to;-

- set interim and pragmatic management reference points based on the optimum status of stock density, biomass, population structure and spatial distribution
- measure as accurately as possible compliance with stock reference levels
- decide if and what management interventions are needed and to measure their subsequent cost-effectiveness

Further useful information can be gained from monitoring commercial fishery data on the possible impacts of fishing mortality and the sustainability and socio-economic benefits of fisheries. The importance of other anthropogenic factors (e.g. migration barriers or habitat degradation) can also be clarified, as can interrelationships between migration and stock status with factors such as climate change and pollution monitoring.

8.11.2 Management of monitoring programmes (Sections 8.3 and 8.10)

Recommendation 1
Monitoring programmes should be directed by the Agency's National Eel Group or delegated to, for example, the National Coarse Fisheries Centre. They should be prioritised, planned and coordinated within national and international contexts, with responsibilities for local and area monitoring programmes delegated to the Regions.

Recommendation 2
It is, however, recommended that monitoring of selected primary catchments be funded centrally to ensure their long-term viability (see Recommendation 7)

8.11.3 Monitoring of resident eel stocks

Recommendation 3
Licence sales, numbers of licensees, catch returns and estimated CPUEs should be monitored annually to track spatial and temporal trends in commercial fisheries. This will provide information about recruitment changes, possible overfishing, market forces and socio-economic trends, e.g. due to market changes or impacts of fishery controls (Section 8.4.2).

Recommendation 4
Annual export-import data analyses are recommended as cost-effective means of annually monitoring general trends in eel stocks and fisheries (Section8.4.2).

Recommendation 5
Useful data about fishing practices, areas, temporal changes, etc. have been gained from questionnaire surveys. It is therefore recommended that a repeat survey be conducted, asking similar questions to those posed in 1992 by McKinnon & Potter (1993) and in 1993 by White and Knights (1994) (Section 8.4.2)

Recommendation 6
It is recommended that electric fishing is used as the principal means of determining stock status and management reference points and for monitoring compliance with stock reference levels (Section 8.4.4). For eel-specific monitoring, electric fishing methods recommended in Appendix C.5 should be followed.

Recommendation 7
Resources should be allocated at a national level for a regular long-term core programme of detailed eel-specific surveys on a rolling programme of one catchment per year, employing standardised electric fishing methods (Sections 8.4.4 and 8.4.9). This should include four primary west/south coast catchments (the Severn, Dee, Piddle and Frome) and at least one east coast one (e.g. the Darent and/or Mole in the Thames catchment and/or the Stour in Essex) (Section 8.4.5)

Recommendation 8
Secondary monitoring catchments should be chosen where eel data can be cost-effectively collected regularly via routine surveys to ensure that the best (but most historically comparable) information is gained. Fuller details and reasons for suggested choices are given in Section 8.4.6, but prioritisation is left to the Environment Agency according to resource availability. Suggested catchments include the Warwickshire and Bristol Avon, N. Wessex rivers, Fowey, Tamar/Tavy, Test, Yare, West Beck/upper River Hull, Tees, lower Leven, upper Eamont, Lune, Dyfi, and Wye.

Recommendation 9
A minimum of 25 sites per catchment should be surveyed if possible, located within 50 km of the tidal limit and selected for representativeness and ease of sampling (Sections 8.4.7 and 8.4.8). Resurvey dates for a given site should be matched as closely as possible to the original, preferably between May and September (Section 8.4.10).

Recommendation 10
If any consecutive surveys show non-compliance with stock reference levels, confirmatory surveys may be necessary, perhaps at annual intervals, to provide a statistically sound basis for possible management decisions on how to return stocks to the management target (Section 8.4.9).

Recommendation 11
In eel-specific monitoring (and wherever possible in other surveys) population density, biomass and length class frequency distributions should be determined at all sites and on all sampling occasions (Appendix C.2). Relative proportions of silver eels and sex ratios and basic habitat data can also provide useful subsidiary information (Appendix C.5.6).

Presence-absence data can also reveal changes in inter-catchment distribution that may be related to fishing pressures (Sections 8.4.11-8.4.13).

Recommendation 12
To assess possible changes in estuarine and coastal stocks and impacts of fisheries in the most cost-effective manner, commercial fishery data should be analysed annually. However, power station screen sampling has produced good historical data and it is recommended that a replacement for West Thurrock P.S. in the Thames Estuary is required. Screen sampling provides valuable information on other estuarine species and, especially, on commercially-important marine species using the estuary as a nursery ground. Therefore, shared funding should be sought (e.g. in conjunction with MAFF) because of the general value of such monitoring (Section 8.6.3c).

8.11.4 Monitoring of glass eel/elver recruitment and fisheries

Recommendation 13
Import-export data should be collected and analysed annually to cost-effectively monitor trends in glass eel recruitment and fisheries (Section 8.6.2).

Recommendation 14
Although monitoring of glass eels and elvers is considered in detail in Section 8.6, it is recommended that any resource-intensive national initiatives are held in abeyance until the report of the European CFP Concerted Action R&D Programme is published. In the interim, the Environment Agency should support any appropriate partnership (Section 8.6.5). The benefits of gaining information on glass eel and elver recruitment should also be considered as support for the recommendations made in Chapter 6 for catchment R&D programmes to study population dynamics and mortality factors.

8.11.5 Monitoring silver eel spawner escapement

Recommendation 15
Further studies should be conducted into the feasibility of monitoring silver eel escapement via fish counter systems already in use or under development (Section 8.7). These could be integrated into R&D studies of population dynamics in a selected study river, as recommended in Chapter 6.

8.11.6 Recording and presentation of monitoring data

Recommendation 16
Graphing, histogramming and mapping methods are recommended for visual presentation of population data to help reveal spatial differences and temporal changes requiring further attention (Sections 8.8.3-4).

Recommendation 17
Consideration should be given to the use of combined biomass and density indices for general comparisons of eel stocks and distributions. Such an adaptation to the Fisheries Classification Scheme could help highlight spatial differences and temporal changes requiring further attention (Section 8.8.5).

8.11.7 Monitoring of management reference points (Section 8.9)

Recommendation 18
Population status indicators for sites in a catchment, tributary or group of tributaries should be determined from previous survey data or from the first survey in a new programme. That these are close to optimum levels and are appropriate for setting management reference points can be determined by comparisons with data in Chapters 2 and 3 for comparable sites and sampling techniques.

Recommendation 19
If visual presentations of data suggest major changes have occurred, non-compliance with stock reference levels should be assessed further. If the data available on density, biomass and length class frequencies are suitably robust for a group of sites, standard statistical comparisons such as paired t-tests should be used to assess significance of changes. The other criterion is if the mean of the percentage changes in density and biomass is greater than –25%. Additional, if less robust, indications of non-compliance could be revealed by changes in proportions of silver eels, sex ratios and catchment distributions.

Recommendation 20
If there is evidence of non-compliance, any errors in survey methods need checking first and, if needed, further confirmatory surveys conducted.

Recommendation 21
Commercial fishing licence records and catch and effort data should be checked to ascertain the possible effects of fishing mortality. If the impacts of a fishery can be quantified, it might be possible to set exploitation targets to maintain a sustainable fishery. Otherwise, the precautionary approach will have to be applied, using management options reviewed in Chapter 7. If the whole national (and/or international) stock appears to be threatened, applications of the precautionary approach to all eel fisheries may be required.

Recommendation 22
If fishery mortality is not implicated in non-compliance with stock reference levels, other causes must be sought (e.g. drastic declines in recruitment, migration barriers, habitat change or pollution). In these cases, other management options must be selected to assure compliance with stock reference levels, as detailed in Section 7.5.

Recommendation 23
After implementation of any fishery controls or other management options, regular monitoring should be carried out to assess their effectiveness.

8.11.8 Regional and national roles (Section 8.10)

Recommendation 24
Overall planning and prioritisation of monitoring programmes should be a national responsibility, with the rolling programme of eel-specific resurveys of primary catchments and sites centrally funded. Central funds may also need to allocated to support other monitoring initiatives (e.g. partnership or R&D studies, as suggested in Chapter 6).

Recommendation 25
Monitoring and data management and analyses should be carried out locally, under Regional guidance but in coordination with a national centre. Regular Regional and national overview reports should be produced to inform management, these should be transmitted to MAFF and thence to ACFM/ICES/ to inform Europe-wide management.

Recommendation 26
Management reference levels should be set Regionally for catchments and sites and compliance with these monitored via local programmes. If non-compliance is identified, Regions should then consult with a national office regarding imposition of fisheries controls or use of other management options.

Recommendation 27
National and Regional offices should coordinate further monitoring to assess the effectiveness of any management measures selected.

REFERENCES

Andrews, M.J., Aston, K.F.A., Rickard, D.G. & Steel, J.E.C. (1982) Macrofauna of the Thames estuary. *The London Naturalist* **61**, 30-62.

Aprahamian, M.W. (1982) Age structure of eel, *Anguilla anguilla* (L.), populations in the River Severn, England, and the River Dee, Wales. *Aquaculture and Fisheries Management* **19**, 365-376

Aprahamian, M.W. (1986) Eel (*Anguilla anguilla* L.) production in the River Severn, England. *Polskie Archiwum Hydrobiologii* **33**, 373-389

Aprahamian, M.W. (1988) Age structure of eel, *Anguilla anguilla* (L.), populations in the River Severn, England, and the River Dee, Wales. *Aquaculture and Fisheries Management* **19**, 365-376.

Aprahamian, M.W. (2000) The growth rate of eel in tributaries of the lower River Severn, England, and its relationship to stock size. *Journal of Fish Biology* **56**, 223-227.

Araujo, F.G., Bailey, R.G. & Williams, W.P. (1998) Seasonal and between-year variations of fish populations in the middle Thames estuary: 1980-1989. *Fisheries Management and Ecology* **5**, 1-21

Barbin, G.P. & McCleave, J.D. (1997) Fecundity of the American eel *Anguilla rostrata* at 45° N in Maine, USA. *Journal of Fish Biology* **51**, 840-847

Barnard, S. & Wyatt, R.J. (1995) A guide to HABSCORE field survey methods and the completion of standard forms. *NRA R&D Note 401*: National Rivers Authority, Bristol, England 31 pp.

Baras, E., Jeandrain, D., Serouge, B & Philippart, J.C. (1998) Seasonal variations in time and space utilisation by radio-tagged yellow eels *Anguilla anguilla* (L.) in a small stream. *Hydrobiologia* **371-372**, 187-198.

Barak, N. A.-E. & Mason, C.F. (1992) Population density, growth and diet of eels *Anguilla anguilla* L., in two rivers in eastern England. *Aquaculture & Fisheries Management* **23**, 59-97.

Berg S. and Jorgensen J. (1994) Stocking experiments with 0+ eels (*Anguilla anguilla*) in Danish streams: post-stocking movements, densities and mortality. In *Rehabilitation of Freshwater Fisheries* (ed.I.G.Cowx) pp. 314-325. Fishing News Books, Oxford.

Boetius I. and Boetius J. (1980) Studies of the European eel, *Anguilla anguilla*. Estimates of fecundity and energy reserves for migration and spawning. *Dana* **1**, 1-28.

Bowman, A. (1913) The distribution of the larvae of the eel in Scottish waters. *Scientific Investigations of the Fishery Board for Scotland* **2**, 3-11

Briand, C., Fatin, D., Thomere, E., Fontenelle,G. & Fenteun, E. (1999) Late entry emigration estimation: mark-recapture studies. *ICES/EIFAC Working Party on Eel Meeting, Silkeborg, 1999.* EIFAC/ICES, 8pp.

Briand, C., Fatin, D., Thomere, E., Fontenelle,G. and Fenteun, E. (2000) Estuarine and fluvial recruitment of European glass eel in an Atlantic exploited estuary. Unpublished paper presented at the Environment Agency Eel Management Seminar, Birmingham, 8th March 2000, 10pp.

Broad, T.L., Townsend, C.R., Arbuckle, C.J. & Jellyman, DJ. (2000). A model to predict the presence of longfin eels (*Anguilla dieffenbachii*) in some New Zealand streams, with particular reference to riparian vegetation and elevation. *Journal of Fish Biology* (In Press).

Callaghan R.M. & McCarthy T.K. (1991) Variations in population structure and growth of eel in the river Dunkellin system, Western Ireland. *EIFAC Working Party on Eel, Dublin 1991.* Rome: EIFAC.

Carl, F.L. & Strube, M.L. (1978) A new method for estimating population size from removal data. *Biometrics* **34**, 621-630.
Carpenter, A.C. (1983) A study of the biology of the eel in tributaries of the lower River Trent. Unpublished PhD Thesis, University of Hull, 300pp
Carss, D.N., Elston, D..A., Nelson, K.C. &Kruuk, H. (1999) Spatial and temporal trends in unexploited yellow eel stocks in two shallow lakes and associated streams. *Journal of Fish Biology* **55**, 646-654.
Castelnaud G., Guerault D., Desaunay Y. & Elie P. (1994) Production et abondance de la civelle en France au debut des annes 1990. *Bulletin Francais de Peche et Pisciculture* **335**, 263-287
Castonguay M., Hodson P.V., Moriarty C., Drinkwater K.F. & Jessop T. (1994) Is there a role of ocean environment in American and European eel decline? *Fisheries Oceanography* **3**, 197-203.
Churchward A.S. (1996) Overview of the Severn eel fishery,England. *EIFAC (FAO) Working Party on Eel Meeting, IJmuiden, Netherlands.* FAO, Rome: 10pp.
Ciccotti, E., Ricci, T., Scardi, M., Fresi, E. & Cataudella, S. (1995) Intraseasonal characterization of glass eel migration in the River Tiber: space and time dynamics. *Journal of Fish Biology* **47**, 248-255.
Cochrane, W.G. (1977) Sampling Techniques (3rd ed.). John Wiley & Sons, New York.
Colombo G. & Grandi G. (1996) Histological study of the development and sex differentiation of the gonads in the European eel. *Journal of Fish Biology* **48**, 493-512.
Costa J.L., Almeida P.R., Assis C.A. Moreira F.M. & Costa M.J. (1991) A study of methods of estimating the size of eel populations in small streams. *Irish Fisheries Investigations Series A (Freshwater)* **36**, 68-69.
Dahl, J. (1967) Some recent observations on the age and growth of eels. *Proceedings of the British Coarse Fish Conference* **3**, 48-52
De Leo, G.A. & Gatto, M. (1996) Trends in vital rates of the European eel: evidence for density dependence? *Ecological Applications* **6**, 1281-1294.
Deelder C.L. (1984) Synopsis of biological data on the eel. *FAO Fisheries Synopsis* **No. 80.** FAO, Rome, 87 pp.
Dekker, W. (1986) Regional variation in glass eel catches: an evaluation of multiple sampling sites. *Vie et Milieu* **36**, 251-254.
Dekker, W. (1991) Assessment of the historical downfall of the IJsselmeer fisheries using anonymous inquiries for effort data. In *Catch Effort Sampling Strategies, Their Application in Freshwater Management* (ed. I.G. Cowx), pp.233-240. Fishing News Books, Oxford.
Dekker, W. (1996B) A length structured matrix population model, used as a fish stock assessment tool. In *Stocking and Introduction of Fish* (ed. I.G. Cowx), pp. 121-140. Fishing News Books, Oxford.
Dekker, W. (1998) Long-term trends in the glass eels immigrating at Den Oever, The Netherlands. *Bulletin Francais de Peche et Pisciculture* **349,** 199-214.
Dekker, W. (1999a) A Procrustean assessment of the European eel stock. *ICES/EIFAC Working Group on Eel, Silkeborg, 1999.* ICES/EIFAC, 8pp.
Dekker, W. (1999b) Is the European eel overfished? *ICES/EIFAC Working Group on Eel, Silkeborg, 1999.* ICES/EIFAC, 8pp.
Dekker, W. (2000) The fractal geometry of the European eel stock. *ICES Journal of Marine Science* **57**, 109-121.
Desaunay, Y. & Guerault, D. (1997) Seasonal and long-term changes in biometrics of eel larvae: a possible relationship between recruitment variation and North Atlantic ecosystem productivity. *Journal of Fish Biology* **51** (Supplement A), 317-339.
Devall, C.A. (1998) Maldon and the tidal Blackwater Volume 1: Heybridge Basin Eel Industry, 1928-1968. (ISBN 0 9533360 0 X). C.E. Devall, Chelmsford, Essex, 83pp.
Dill (1993) Inland Fisheries of Europe. *EIFAC Technical Paper* **52,** 281pp.

Donnelly R.E., Reynolds J.D. & Walsh T. (1993) The management of glass eel stocks in the Shannon Estuary, Ireland. EIFAC Working Party on Eel, Olsztyn 1993. Rome: EIFAC, FAO, 23 pp.
Elie, P., Cantrelle, I. & Charlon, N. (1982) Definition des limites differents stades pigmentaires durant la phase civelle d'*Anguilla anguilla* (L.). *Vie et Milieu* **3**, 149-157.
Environment Agency NW (1997) Water Level Plan for Leighton Moss. Unpublished report by the Environment Agency, Penrith CA11 9BP, 23pp.
Feunteun, E., Acou, A., Guillouet, J., Laffaille. & Legault, A. (1998) Spatial distribution of an eel population (*Anguilla anguilla* L.) in a small coastal catchment of northern Brittany (France). Consequences of hydraulic works. *Bulletin Francais de Peche et Pisciculture* **349**, 129-139.
Feunteun, E., Acou, A., Laffaille. & Legault, A. (2000) European eel (*Anguilla anguilla*): prediction of spawner escapement from continental population parameters. *Canadian Journal of Fisheries and Aquatic Science* **57** (In Press).
Forbes, I. & Wheeler, A. (1997) A comparison of late eighteenth century and present day fishery surveys on the River Witham, Lincolnshire, UK. *Fisheries Management & Ecology* **4**, 325-335.
Francis, R.I.C.C & Jellyman, D.J. (1999) Are mean size data adequate to monitor freshwater eel fisheries? *Marine & Freshwater Research* **50**, 355-366.
Gascuel, D. (1986) Flow-carried and active swimming migration of glass eel (*Anguilla anguilla*) in the tidal area of a small estuary on the French Atlantic coast. *Helgoländer Meeresuntersuchungen* **40**, 321-326.
Gascuel, D., Feunteun, E, & Fontenelle, G. (1995) Seasonal dynamics of estuarine migration in glass eels (*Anguilla anguilla*). *Aquatic Living Resources* **8**, 123-133.
Gross, M.R., Coleman, R.M. & McDowall, R.M. (1988) Aquatic productivity and the evolution of diadromous fish migration. *Science* **239**, 1291-1293.
Growns, I.O., Pollard, D.A. & Gherke, P.C. (1998) Changes in fish assemblages associated with vegetated and degraded banks, upstream of and within nutrient-enriched zones. *Fisheries Management and Ecology* **5**, 55-69.
Hussein S.A. (1981) The population density, growth and food of eels *Anguilla anguilla* L. in some tributaries of the River Tweed. *Proceedings of the 2nd British Freshwater Fisheries Conference, University of Liverpool*, 120-128
Henderson, P.A. (1989) On the structure of the inshore fish communities of England and Wales. *Journal of the Marine Biological Association UK* **69**, 145-163
Hvidsten N.R. (1985) Yield of silver eel and factors affecting downstream migration in the stream Emsa, Norway. *Report of the Institute of Freshwater Research, Drottningholm* **62**, 75-85
Ibbotson A., Armitage P., Beaumont W., Ladle M & Welton S. (1994) Spatial and temporal distribution of fish in a small lowland stream. *Fisheries Management and Ecology* **1**, 143-156
ICES (1999) ICES Cooperative Research Report No. 229, Report of the ICES Advisory Committee on Fisheries Management, 1998, Part 2, 446pp.
Jefferies, D.J., Cripps, S.J., Gorrod, L.A., Harrison, B., Johnstone, J.S. & Potter, E.C.E. (1988) *The effect of otter guards on the efficiency of eel fyke nets*. Vincent Wildlife Trust, London, 47 pp.
Jessop, B.M. (2000) Estimates of population size and instream mortality rate of American eel elvers in a Nova Scotia river. *Transactions of the American Fisheries Society* **129**, 514-526.
Koed, A. & Dieperink, C. (1999) Otter guards in river fyke-net fisheries: effects on catches of eels and salmonids. *Fisheries Management & Ecology*, **6**, 63-69
Kennedy G.J.A. & Vickers, K.U. (1993) The fish of Lough Neagh. Part A. A historical and taxonomic perspective of the fish fauna of the Lough Neagh catchment. In. *Lough Neagh: the ecology of a multipurpose water resource* (ed. R.B. Wood & R.V. Smith), pp. 381-395. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Klein Breteler J.G.P. (1994) Stockings of pre-winter and post-winter glass eels and of elvers from aquaculture: growth and survival in mesocosms. *EIFAC Working Party on Eel, Oviedo 1994*. EIFAC/ FAO, Rome: 7 pp.
Klein Breteler J.G.P. (1996) Application of a habitat suitability index model for eel *Anguilla anguilla* in a habitat evaluation procedure for the Netherlands in the 19th and 20th century. *EIFAC Working Party on Eel, Ijmuiden, The Netherlands 1996*. EIFAC/ FAO, Rome: 7 pp.
Knights, B. (1997) Risk assessment and management of contamination of eels (*Anguilla* spp.) by persistent xenobiotic organochlorine compounds. *Chemistry & Ecology* **14**, 171-212.
Knights, B. & White, E.M. (1993) Eel passes at the Dog in Doublet Sluice, River Nene. Unpublished Rep[ort to NRA Anglian Region, 12pp.
Knights B. & White E.M. (1997) An appraisal of stocking strategies for the European eel, *Anguilla anguilla* L. In *Stocking and Introduction of Fish* (ed. I.G. Cowx), pp. 121-140. Fishing News Books, Oxford.
Knights, B. & White, E.M. (1998) Enhancing immigration and recruitment of eels: the use of passes and associated trapping systems. *Fisheries Management & Ecology* **4**, 311-324
Knights B., White E. & Naismith I.A. (1996) Stock assessment of European eel, *Anguilla anguilla* L. In *Stock Assessment in Inland Fisheries* (ed. I.G.Cowx) pp.431-447. Fishing News Books, Oxford.
Koed, A. & Dieperink, C. (1999) Otter guards in river fyke-net fisheries: effects on catches of eels and salmonids. *Fisheries Management and Ecology* **6**, 63-69.
Kristiansen, T.S., Ottera, H. & Svasand, T. (2000) Size-dependent mortality of juvenile reared Atlantic cod released in a small fjord. *Journal of Fish Biology* **56**, 792-801.
Lambert, P. & Rigaud, C. (1999) Recherche d'éléments de gestion de la population d'anguilles sur la base des données produites par RHP (Convention d'études CSP-Cemagref No 97 420). *Cemagref Departement Gestion des Milieux Aquatiques, Études No. 49*, 63pp.
Lambert, P., Feunteun, E. & Rigaud, C. (1994) Études de l'anguille en marais d'eau douce. Première analyse des probabilités de capture observées lors des inventaires par pêches électrique. *Bulletin Francais Pêches et Pisciculture* **335**, 111-121.
Larsen, K. (1972) Studies on the biology of Danish stream fisheries. III On seasonal fluctuations in the stock density of yellow eel in shallow stream biotopes and their causes. *Meddelelser fra Danmarks Fiskeri- og Havundersøgelser* **7**, 23-46.
Larsson P., Hamrin S. & Okla L. (1990) Fat content as a factor inducing migratory behaviour in the eel (*Anguilla anguilla* L.) to the Sargasso Sea. *Naturwissenschaften* **77**, 488-490.
Leopold, M. (1976) Efficiency and prospectives of eel management, some economic approaches. *EIFAC/ICES Symposium on Eel Research & Management, Helsinki*. No. **17**, 1-17
Leopold M. (1980) Some remarks on eel management in lakes. *Aquaculture* **19**, 97-101
Leopold M. (1986) Problems in analysing the effectiveness of eel stocking in interconnected lakes. *Vie et Milieu* **36**, 291-293.
Leopold M. & Bninska (1984) The effectiveness of eel stocking in Polish lakes. *EIFAC Technical Paper* **42** (Supplement 1). 41-49.
Lobon-Cervia J., Bernat Y. & Rincon P.A. (1990) Effects of eel (*Anguilla anguilla* L.) removals from selected sites of a stream on its subsequent densities. *Hydrobiologia* **206**, 207-216
Lowe, R.H. (1951) Factors affecting the runs of elvers in the River Bann, Northern Ireland. *J. du Conseil International Exploration Mer* **17**, 299-315.
Lowe, R.H. (1952) The influence of light and other factors on the seaward migration of the silver eel (*Anguilla anguilla* L). *Journal of Animal Ecology* **21**, 275-309..
Mann R.H.K. (1995) Eel stock assessment in the UK. *Report to MAFF, Project No. SF0305, IFE Project No. T11063al.* Ministry of Agriculture, Fisheries and Food, London:. 86 pp.

Mann R.H.K. & Blackburn J.H. (1991) The biology of the eel *Anguilla anguilla* (L.) in an English chalk stream and interactions with juvenile trout *Salmo trutta* L. and salmon *Salmo salar* L. *Hydrobiologia* **218**, 65-76

McCarthy T.K., O'Farrell, M., Cullen, P. & O'Connor, W. (1993) Migration of silver eels in the River Shannon, Ireland: tagging experiments and monitoring of commercial fishing operations. *EIFAC (FAO) Working Party on Eel Meeting, Olsztyn, Poland.* FAO/EIFAC, Rome: 25pp

McCarthy T.K., Allison J., Callaghan R., Purcell P. & Copley L. (1994a) Yellow eel stocks in the River Shannon and the development of a fyke net fishery. *Report to Electricity Supply Board, Dublin, Ireland*

McCarthy T.K., Cullen K., Faherty K. & O'Farrell M.M. (1994b) River Shannon silver eels: Population biology, factors influencing downstream migration and commercial fishing. *Report to the Electricity Supply Board, Dublin, Ireland.*

McCleave, J.D. & Kleckner, R.C. (1982) Selective tidal stream transport in the estuarine migration of glass eels of the American eel (*Anguilla rostrata*). *J. du Conseil Permanent International pour l'Exploration de la Mer* **40**, 267-271.

McCleave, J.D., Brickley, P.J., O'Brien, K.M., Kistner, D.A., Wong, M.W., Gallagher, M. & Watson, S.M. (1998) Do leptocephali of the European eel swim to reach continental waters? Status of the question. *Journal of the Marine Biological Association, U.K.* **78**, 285-306.

McKinnon, S. & Potter, T. (1993) Survey of eel fisheries in the United Kingdom in 1992. Unpublished MAFF Internal Report, MAFF, Lowestoft, England

Milner, N.J., Wyatt, R.J. & Scott, M.D. (1993) Variability in the distribution and abundance of stream salmonids and associated use of habitat models. *Journal of Fish Biology* **43** (Supplement A), 103-119.

Moriarty C. (1984) Stock enhancement in Irish eel (*Anguilla anguilla* L.) fisheries. *EIFAC Technical Paper* **42** (Supplement 1), 50-57.

Moriarty C. (1986) Riverine migration of young eels (*Anguilla anguilla L.*) *Fisheries Research* **4**, 43-58.

Moriarty C. (1987) Eel stocks of the Shannon system and prospects for development. *Fishery Leaflet No. 139.* Department of the Marine, Dublin. 15 pp.

Moriarty C. (1990a) European catches of elver 1928-1988. *International Revue der Gesamten Hydrobiologie* **75**, 701-706.

Moriarty C. (1990b) Eel stocks, capture and culture. In *Fisheries in the Year 2000.* (ed. K.T. O'Grady, A.J.B. Butterworth, P.B. Spillett, & J.C.J Domaniewski) 14 pp. Institute of Fisheries Management, West Bridgford, England.

Moriarty, C. (1990c) Eel management practices in three lake systems in Ireland. In: W.L.T. van Densen, B. Steinmetz & R.H Hughes (eds.). *Management of Freshwater Fisheries.* Proceedings of a symposium organised by EIFAC, Goteborg, Sweden, 1988, pp. 262-269. Wageningen, Netherlands: Pudoc.

Moriarty C. (1996a) The European eel fishery in 1993 and 1994. (First Report of the EU Concerted Action AIR A94-1939). *Fisheries Bulletin No. 14 (1996).* The Marine Institute, Dublin,: 52pp.

Moriarty C. (1996b) Glass eel supply and demand in Europe in 1995 and 1996. *EIFAC (FAO) Working Party on Eel Meeting, Ijmuiden, Netherlands.* FAO/EIFAC, Rome: 4pp.

Moriarty, C. & Dekker, W. (eds.) (1997) Management of the European Eel. (Second Report of the EU Concerted Action AIR A94-1939). *Fisheries Bulletin No. 15 (1997).* The Marine Institute, Dublin,: 52pp.

Moriarty C. & Nixon E. (1990) Eel biomass in a small lowland stream. *International Revue der gesamten Hydrobiologie* **75**, 781-784

Moriarty C., Bninska M. & Leopold M. (1990) Eel, *Anguilla anguilla* (L.), stock and yield in Polish lakes. *Aquaculture and Fisheries Management* **21**, 347-356.

Morrice, C.P. (1989) Eel fisheries in the United Kingdom. *MAFF Internal Report No. 18*, 33pp. MAFF, Lowestoft, England
Mortensen E. (1982) Production of trout *Salmo trutta* in a Danish stream. *Environmental Biology of Fishes* 7, 349-356
Mower, D. (1998) Leighton Moss elver study, 1997-8. Unpublished report for Royal Society for the Protection of Birds.
Mower, D. (1999) Elver study at Leighton Moss 1999 in relation to commercial elver fishing. Unpublished interim report for Royal Society for the Protection of Birds.
Naismith, I.A. (1998) Status of eel in the Thames Catchment. *WRc Report 11069 to Environment Agency Thames Region.* 99pp.WRc, Medmenham, Bucks SL7 2HD.
Naismith I.A. & Knights B. (1988) Migrations of elvers and juvenile European eels, *Anguilla anguilla* L., in the River Thames. *Journal of Fish Biology* **33(Supplement A)**, 161-175
Naismith I.A. & Knights B. (1990a). Studies of sampling methods and of techniques for estimating populations of eels, *Anguilla anguilla* L. *Aquaculture & Fisheries Management.* **21**, 357-367.
Naismith I.A. & Knights B. (1990b) Modelling of unexploited and exploited populations of eels, *Anguilla anguilla* (L.), in the Thames Estuary. *Journal of Fish Biology* **37**, 975-986.
Naismith I.A. and Knights B. (1993) The distribution, density and growth of European eels, *Anguilla anguilla* L., in the River Thames catchment. *Journal of Fish Biology*, **42**, 217-226.
Naismith I.A. and Knights B. (1994) Fyke-netting and coarse fisheries in lowland Britain: practical advice for fishery owners and managers. *Fisheries Management and Ecology* **1**, 107-116.
Nielsen J. (1988) A review of marking and tagging methods applied to eel. *EIFAC Occasional Publications* **21**, 1-48.
NRA (1995) The survival and dispersal of stocked coarse fish. R&D Project Note 364. Bristol, England: National Rivers Authority (now the Environment Agency), 81 pp.
Pankhurst, N.W. (1982) Relation of visual changes to the onset of the sexual maturation in the European eel *Anguilla anguilla* (L.). *Journal of Fish Biology* **21**, 127-140.
Parsons J., Vickers K.U. & Warden Y. (1977) Relationships between elver recruitment and changes in sex ratio of silver eels *Anguilla anguilla* L. migrating from Lough Neagh, Northern Ireland. *Journal of Fish Biology* **10**, 211-229
Pilcher, M.W. & Copp, G.H. (1997) Winter distribution and habitat use by fish in a regulated lowland river system of south-east England. *Fisheries Management and Ecology* **4**, 199-215
Perrow, M, Howe, H. & Jowitt, A. (1996) A review of factors affecting the status of fish populations in the emergent plant zone of wetland habitats, particularly beds of reeds (*Phragmites australis*), with respect to the habitat requirements of bittern (*Botaurus stellaris*). Unpublished Report by ECON, University of Essex for Royal Society for the Protection of Birds, England.
Poole W.R., Reynolds J.D. & Moriarty C. (1990) Observations on the silver eel migrations of the Burrishoole River system, Ireland. *International Revue der gesamten Hydrobiologie* **75**, 807-815
Pursiainen M. & Toivenen J. (1984) The enhancement of eel stocks in Finland: A review of introductions and stockings. In: *EIFAC Technical Paper 42/Supplement 1.* Documents presented at the Symposium on Stock Enhancement in the Management of Freshwater Fish 1982, Budapest, Hungary. pp. 59-67. Rome: EIFAC, FAO.
Quigley D. & O'Brien T. (1993) The River Shannon eel fishery: A management review. EIFAC Working Party on Eel, Olsztyn, Poland 1993. Rome: EIFAC, FAO, 35 pp.
Rasmussen G. (1977) Production of eels in a small stream in Zealand. *Proceedings of the 8th British Coarse Fish Conference, University of Liverpool*, 61-68

Rasmussen G. (1983) Recent investigations of the population dynamics of eels (*Anguilla anguilla* L.) in some Danish streams. *Proceedings of the Third British Freshwater Fisheries Conference*, pp. 71-77. England: Liverpool.
Rasmussen G. & Therkildsen B. (1979) Food, growth and production of *Anguilla anguilla* L. in a small Danish stream. *Rapports et Proces-Verbaux des Reunions, Conseil International pour l'Exploration de la Mer* **174**, 115-123.
Richkus, W.A. & Whalen, K. (1999) American eel (*Anguilla rostrata*) Scoping Study: a literature and data review of life history, stock status, population dynamics and hydroelectric facility impacts. Contract TR-111873 Report to EPRI, Palo Alto, California, USA.
Rigaud C., Fontanelle G., Gascuel D. & Legault A. (1988) Le franchissement des ouvrages hydrauliques par les anguilles (*Anguill anguilla*): Presentation des dispositifs installes en Europe. *Les Publications di Department d'Halieutique* **9** 147pp. ENSA de Rennes, France.
Rossell, R. (1999) Key conclusions arising from Lough Neagh eel production 1959 to 1999. *ICES/EIFAC Working Group on Eel,* ICES/EIFAC.
Rossi R. (1979) An estimate of the production of the eel population in the Valli of Commacchio (PO Delta) during 1974-1976. *Bollitino di Zoologia* **46**, 217-223
Rossi R. & Cannas A. (1984) Eel fishing management in a hypersaline lagoon of southern Sardinia. *Fisheries Research* **2**, 285-298.
Rossi R., Carrieri P., Franzoi G., Cavallini G. & Gnes A. (1987-8) A study of eel (*Anguilla anguilla* L.) population dynamics in the Commachio lagoons (Italy) by mark-recapture method. *Oebalia* **XIV, N.S.**: 1-14.
Rossi R., Carrieri P & Rizzo M.G. (1988) Use of intensive rearing systems as back-up for coastal lagoon aquaculture: an experience with eels, *Anguilla anguilla* L., in the Commachio lagoons. *Aquaculture and Fisheries Management* **19**, 355-361.
Rossi R. & Colombo, G. (1976) Sex ratio, age and growth of silver eels in two brackish water lagoons in the northern Adriatic Valli of Commachio and Valli Nuova. *Archives of Oceanography and Limnology*, **18** (Supplement 3), 327-341.
Rossi R. & Corbari L. (1982) Biological analysis of the fish catch in the Acquitina lake (southwestern Adriatic Sea, Italy) for the 1976-9 period. *Memores Biologia Marinaria Oceanographia* **12**, 115-133.
Rossi R. & Papas P. (1979) Analisi del pescato di una 'valee' del Delta del Po (Valle Nuova) dal 1950 al 1978. *Annali Universite di Ferrara (Biologie I)* **9**, 123-148.
Rossi R. & Villani P. (1980) A biological analysis of eel catches, *Anguilla anguilla* L., from lagoons of Lesina and Verano, Italy. *Journal of Fish Biology* **16**, 413-423.
Russell, I. & Potter, T. (1999) Origin and principles of the precautionary approach and how these might apply to the European eel. *ICES/EIFAC Working Group on Eel, Silkeborg, 1999.* ICES/EIFAC, 8pp.
Schouten, W.J. (1992) Habitat geschiktheid model: de aal, *Anguilla anguilla.* Unpublished Internal Report, Organisatie ter Verbetering van de Binnenvissserij, Nieuwegein, The Netherlands, 21pp.
Sinha, V.R.P. & Jones, J.W. (1967) On the food of freshwater eels and their feeding relationships with salmonids. *Journal of Zoology, London* **153**, 119-137.
Skehan, B.W. & De Silva, S.S. (1998) Aspects of the culture-based fishery of the shortfinned eel, *Anguilla australis*, in western Victoria, Australia. *Journal of Applied Ichthyology* **14**, 23-30.
Sparre, P. (1979) Some necessary adjustments for using the common methods in eel assessment. In Eel Research and Management (ed. Thurow, F.) *Rapports et Procès-Verbaux des Réunions du Conseil International pour l'Exploration de la Mer* **174**, 41-44.

Svärdson, G. (1976). The decline of the Baltic eel population. *Reports of the Institute of Freshwater Research, Drottningholm* **55,** 136-143.
Svedäng, H., Wickström, H., Reizenstein, M., Holmgren, K. & Florenius, P. (1998) Accuracy and precision in eel age estimation, using otoliths of known and unknown age. *Journal of Fish Biology* **53**, 456-464.
Svedäng, H. (1999) Vital population statistics of the exploited eel stock on the Swedish west coast. *Fisheries Research* **40**, 251-265.
Svedäng, H., Neuman E. & Wickstrom H. (1996) Maturation patterns in female European eel: age and size at the silver eel stage. *Journal of Fish Biology* **48**, 342-351.
Tesch F.-W. (1977) *The Eel: Biology and Management of Anguillid Eels.* London: Chapman and Hall
Tesch, F.W. (1980) Occurrence of eel *Anguilla anguilla* larvae west of the European continental shelf, 1971-1977. *Environmental Biology of Fishes* **5**, 185-190.
Tesch, F.-W., Niermann, U. & Plaga, A. (1986) Differences in development stage and stock density of larval *Anguilla anguilla* off the west coast of Europe. *Vie Milieu*, **36**, 255-260.
Tesch, F.-W. & Niermann, U. (1991) Stock density of eel larvae (*Anguilla anguilla* L.) on the European continental slope, based on collections made between 1985 and 1989. *EIFAC (FAO) Party on Eel Meeting, Dublin, Ireland, 1991.* 5pp.
Tsukamoto, K., Nakai, I & Tesch, F.W. (1998) Do all freshwater eels migrate? *Nature* **396**, 635
Tzeng, W.-N. (1984) An estimation of the exploitation rate of *Anguilla japonica* immigrating into the coastal waters off Shuang Chi River, Taiwan. *Bulletin of the International Zoological Academy Sinica* **23**, 173-180.
Tulonen J. (1990) Growth and sex ratios of eels (*Anguilla anguilla*) of known age in four small lakes in southern Finland. *International Revue gestalt Hydrobiologie* **75**, 792-800
Turnpenny, A.W.H., Sadler, K., Aston, R.J., Milner, A.G.P. & Lynam, S. (1987) The fish populations of some streams in Wales and northern England in relation to acidity and associated factors. *Journal of Fish Biology* **31**, 415-434.
Vøllestad L.A. (1985) Age determination and growth of yellow eels, *Anguilla anguilla* (L.), from a brackish water. *Norway Journal of Fish Biology* **26**, 521-525.
Vøllestad L.A. (1992) Geographic variation in age and length at metamorphosis of maturing European eel: environmental effects and phenotypic plasticity. *Journal of Animal Ecology* **61**, 41-48.
Vøllestad L.A., Lecomte-Finiger R. & Steinmetz B. (1988) Age determination of *Anguilla anguilla* and related species. *EIFAC Occasional Publications* **21**, 1-28
Vøllestad, L.A. & Jonsson, B. (1988). A 13-year study of the population dynamics and growth of the European eel *Anguilla anguilla* in a Norwegian river: evidence for density-dependent mortality and development of a model for predicting yield. *Journal of Animal Ecology* **57**, 983-997.
Ward, R.C., Loftis, J.C. & McBride, G.B. (1990) Design of Water Quality Monitoring Systems. Von Nostrand Reinhold, New York.
Wardlaw, A.C. (2000) Practical Statistics for Experimental Biologists. John Wiley & Sons, New York.
Westerberg. H. (1990) A proposal regarding the source of nutrition of leptocephalus larvae. *Internationale Revue der gesamten Hyrobiologie* **75**, 863-864.
Westerberg, H. (1999) Oceanographic aspects of the recruitment of eels to the Baltic Sea. *ICES/EIFAC Working Group on Eel, Silkeborg, 1999.* 9pp.
Westerberg H., Haamer J. & Lagenfelt I. (1993) A new method for sampling elvers in the coastal zone. *ICES CM 1993/M:5*, 10pp.

Westin, L. (1990) Orientation mechanisms in migrating European silver eel (*Anguilla anguilla*): temperature and olfaction. *Marine Biology* **106**, 175-179.

White E.M. & Knights B. (1994) Elver and eel stock assessment in the Severn and Avon. *NRA R&D Project Record 256/13/ST.*: National Rivers Authority, Bristol, England 141 pp.

White, E.M. & Knights (1997a) Dynamics of upstream migration of the European eel, *Anguilla anguilla* L., with special reference to the effects of man-made barriers. *Fisheries Management & Ecology* **4**, 311-324.

White, E.M. & Knights (1997b) Environmental factors affecting the dynamics of upstream migration of the European eel, *Anguilla anguilla* L., in the Rivers Severn and Avon. *Journal of Fish Biology* **50**, 1104-1116.

Wickstrom H. (1984) The Swedish eel stocking programme. *EIFAC Technical Paper 42/Supplement 1,* 68-83. EIFAC/ FAO, Rome.

Wickstrom H., Westin L. & Clevestam P. (1996) The biological and economical yield from a long-term eel stocking experiment. *Ecology of Freshwater Fish* **5**, 140-147.

Wickström, H. Limburg, K., Svedäng, H. & Westin, L. (1999) Do freshwater eel and even stocked ones contribute to the spawning stock? *ICES/EIFAC Working Group on Eel, Silkeborg, 1999.* 2pp.

Wyatt, R.J., & Lacey, R.F. (1994) Guidance notes on the design and analysis of fishery surveys. *NRA R&D Note 292.* National Rivers Authority, Bristol, England 18 pp.

Wyatt, R.J., Barnard, S. & Lacey, R.F. (1995) Salmonid modelling literature review and subsequent development of HABSCORE models. *NRA R&D Project Record 338/20/W.*: National Rivers Authority, Bristol, England 189 pp.

Zar, J.H. (1996) Biological Analysis (3rd Ed.). Prentice-Hall Inc., New Jersey.

Zippin, C. (1956) An evaluation of the removal method of estimating animal populations. *Biometrics* **12**, 63-169.

Appendix A

EEL LIFE CYCLE MORTALITY MODEL FOR ENGLAND & WALES

A.1 Introduction

This model was presented to the ICES/EIFAC Working Group on Eels, Silkeborg, September 1999 by Brian Knights. It uses available data to predict;-

(a) the relative magnitudes of production of eels in England and Wales (from estimates of the extent habitat and stocks) and thence, via estimates of natural mortality at different oceanic and continental life stages;-
(b) the relative magnitudes of spawner escapement, recruitment and survival of growth stages through a complete life cycle.

Although the model does not specifically take into account density-dependent mortality, it provides a preliminary and speculative overview of the relative importance of natural mortality throughout the life cycle in comparison with fishing mortality.

A.2 Mortality model

A.2.1 Habitat available to eels in England and Wales

Assuming minimal impacts of barriers and using data from Dill (1993) and Water Research Centre information:-

- Length of rivers/canals = 70,000 km x 5m wide gives area = 350 km^2
- Areas of stillwaters suitable for eels = 685 km^2
- Area of estuarine/shallow coastal waters suitable for eels = 5000 km^2
- TOTAL AREA freshwaters = 1035 km^2
- TOTAL AREA of estuarine/coastal waters = 5000 km^2

A.2.2 Female eel density

Assuming (from density, biomass and length-class frequency data from Chapters 2 and 3) for rivers and stillwaters, plus estimates for estuarine/coastal waters:-

- Approximately 50% of river reach and stillwater areas have densities < 1 eel $100m^{-2}$ ($1 \times 10^4\ km^{-2}$) where females dominate
- This gives 535 km^2 of freshwaters and 2438 km^2 of estuarine/coastal waters dominated by females
- At densities of $1 \times 10^4\ km^{-2}$, total number of eels present are therefore 5.35×10^6 and 2.49×10^7 respectively

- If 60% of eels are maturing females > 45 cm (from length-class frequency data), the number of females is 3.21×10^6 and 1.49×10^7 respectively
- If females mainly mature at 10 years (50 cm) and 10% emigrate per year, then the number of female silver emigrants is 3.21×10^5 and 1.49×10^6 respectively

A.2.3 Mortality between different life stages

Mortality rates between each life stage have been estimated as follows (see next page for sources and definitions of habitat types);-

Table A1 Natural (M) v. fishing (F) mortality rates between life stages

Life stages			M	F
Egg	->	Continental shelf leptocephalus	99%[1]	-
Leptoceph.	->	Migratory glass eel	96%[1]	-
Glass eel in:-				
Estuary A[a]	->	Pigmented elver	99.5%[2]	<0.5%[2]
Estuary B[b]	->		Low/high?	-
Estuary C[c]	->		Low/high?	40-70%[2]
Coastal waters	->		Low?	-
			Ave. = 90%?	
FW yellow	->	FW silver	c.75%[3]	LOW
FW silver	->	SW silver eel	LOW?	Negligible[2]
SW silver	->	Egg	LOW?	-

[1] *Tesch (1977, 1980);* [2] *Knights (1999) unpublished;* [3] *Moriarty & Dekker (1997)*

[a] *ESTUARY A = long estuary with large runs, high density-dependent mortality and exploitation (e.g. Severn)*

[b] *ESTUARY B = relatively small estuary/creek with low runs and low density dependent mortality*

[c] *ESTUARY C = as for ESTUARY B, but one easily exploited because of tidal sluice or other immigration bottleneck (e.g. Red Barn Dyke/ Leighton Moss RSPB Reserve, England; East River, Nova Scotia; Shuang-Chi River, Taiwan)*

A.2.4 Relative numbers of eels per life stage

From the estimates above, focussing on female spawners, the table below shows relative numbers of eels related to natural mortality (M%) between one life stage and the next, assuming:-

- No shortage of males
- Natural mortality (M%) is 99.1% for oceanic larval stages between egg and glass eels immigrating into estuaries
- M is 90% for continental stages between immigrant glass eel and pigmented elver/early juvenile stages
- 3000 glass eels/elvers kg^{-1} ($3 \times 10^6\ t^{-1}$) and 5 mature eels kg^{-1} ($5 \times 10^3\ t^{-1}$)

Table A.2 Relative numbers of eels related to natural mortality (%) between life stages

Number of:- (*m≈% natural mortality*)	Fresh waters	Estuarine/ coastal waters	Total
Emigrant females =	3.21 x 10^5	1.49 x 10^6	1.81 x 10^6
$\Downarrow$			
(M≈zero?)			
$\Downarrow$			
Eggs/leptocephali = *(@1 x 10^6 eggs/female)*	3.21 x 10^{11}	1.49 x 10^{12}	1.81 x 10^{12}
$\Downarrow$			
(M≈99.99%)			
$\Downarrow$			
Glass eel migrants = *(@ 3 x 10^6 t^{-1})*	3.21 x 10^7 (9.63 x 10^1 t) (96.3 t)	1.49 x 10^8 (4.47 x 10^2 t) (447 t)	1.81 x 10^8 (5.43 x 10^2 t) (543 t)
$\Downarrow$			
(M≈90%)			
$\Downarrow$			
Recruited elvers = *(@ 3 x 10^6 t^{-1})*	3.21 x 10^6 (9.63 x 10^0 t) (9.63 t)	1.49 x 10^7 (4.47 x 10^1 t) (44.7 t)	1.81 x 10^7 (5.433 x 10^2 t) (54.3 t)
$\Downarrow$			
(M≈75%)			
$\Downarrow$			
Emigrant silver eels = *(@ 5 x 10^3 t^{-1})*	8.025 x 10^5 (40.125 x 10^2 t) (4013 t)	3.725 x 10^6 (18.625 x 10^3 t) (18625 t)	4.525 x 10^6 (22.625 x 10^3 t) (22625 t)
$\Downarrow$			
(M≈zero)			
$\Downarrow$			
If 50% are females:-			
$\Downarrow$			
Female spawner escapement =	4.013 x 10^5 (2007 t)	1.863 x 10^6 (9313 t)	2.263 x 10^6 (11313 t)

A.3 Conclusions

- The model balances to the same order of magnitude through a complete life cycle, although it must be emphasised that (a) it includes many estimates and assumptions based on incomplete knowledge (especially of marine reproduction, life stage mortality, of densities of eels in estuarine/coastal habitats and of inter-migrations with freshwaters) and (b) it does not specifically take density-dependent effects into account,

- The importance of marine larval mortality is emphasised by the fact that for every 0.1% increase in natural mortality during the leptocephalus oceanic or glass eel continental shelf migration (e.g. due to unfavourable environmental and/or feeding conditions), spawner escapement would be reduced by one order of magnitude

- The same applies to natural mortality of glass eels during the migration up estuaries and first feeding and pigmentation stages, where local factors (e.g low water temperatures and spates) could have major impacts on recruitment to a catchment in any one season

- Conversely, fishing mortality of glass eels would have negligible impacts on recruitment in comparison with natural mortality. For example, from export-import data, it has been estimated that glass eel catches in England and Wales in recent years have been about 20t (60×10^6) per year, i.e. < 0.5% of the estimated number of recruits.

- Similarly, yellow and silver eel fisheries in fresh and estuarine/coastal waters would have negligible impacts on spawner escapement in comparison with natural mortality, recent catches of about 150 t per year represents ~5% of estimated escapement over eight years of catchable life span.

- It must be noted that the model is not a closed one for England and Wales, glass eels could be derived from spawners originating from other UK and European waters and their eventual offspring could be dispersed to other countries.

- It must also be noted that eels have a compensatory life strategy in that if mortality (natural or fishing) is high, densities will tend to fall, leading to (a) reductions in competition and density-dependent mortality and (b) the production of relatively more, larger and high fecundity females and hence a relative increase in overall egg production rates

Appendix B

R&D RESURVEYS: RAW DATA

This appendix comprises raw data for the lower Severn (Tables B1 and B2), upper Severn (Tables B3 and B4), Dee (Tables B5 and B6) and Piddle/Frome (Tables B7 and B8).

For each river system,

the first table provides:

- water course, site number, NGR and survey date;
- catch for each electric fishing run at each site;
- population estimate (number of eels) for each site;
- method of population estimation (Zippin, Carl and Strube, Seber & LeCren, minimum estimate)
- probability (results from a goodness of fit of a chi-squared test of the observed and expected numbers captured;
- capture probability (the calculated probability that an individual will have been captured in a sample).

The second table provides:

- water course, site number, NGR and survey date;
- total sample area (m^2) at each site;
- mean eel length (mm) and mean eel weight (g) for each site;
- population density estimate (n m^{-2}) at each site with 95% confidence limits;
- biomass estimate (g m^{-2}) at each site with 95% confidence limits.

For the Severn and Dee, the data tables also include the historic (1983/84) data. Population estimates have been recalculated from the original catch statistics and the data are formatted in the same manner as the resurvey data. No raw data was obtainable for the historic Piddle and Frome surveys.

Table B.1 Population estimates for the surveys undertaken on the Lower Severn tributaries in 1983, 1998 and 1999.

MapID	Watercourse	NGR	Date	Run 1	Run 2	Run 3	Run 4	Total	Population Estimate	SE	Probability*	95%CI lower	95%CI higher	CaptureProbability**	Estimate method
1	Lyd	SO633033	19-Jul-99	74	28	26	*	128	154	12	0.040	130	178	0.449	Zippin
5	Little Avon	ST688956	21-Jul-99	201	59	38	*	298	317	7	0.012	304	331	0.607	Zippin
7	Blakeney Brook	SO681068	19-Jul-99	89	43	15	*	147	160	6	0.430	148	172	0.569	Zippin
10	Westbury Brook	SO718138	20-Jul-99	108	42	22	*	172	187	7	0.441	174	201	0.566	Zippin
11	Westbury Brook	SO697151	20-Jul-99	47	30	11	*	88	102	8	0.261	88	118	0.482	Zippin
12	Westbury Brook	SO678159	20-Jul-99	2	2	0	*	4	4	1	0.244	4	6	0.566	Zippin
13	Frome	SO756094	21-Jul-99	27	19	11	*	57	78	16	0.737	57	109	0.352	Zippin
21	Leadon	SO771271	22-Jul-99	29	35	17	*	81	165	73	0.044	81	308	0.201	Zippin
22	Leadon	SO732298	21-Jul-99	56	36	28	*	120	182	34	0.639	120	250	0.301	Zippin
23	Leadon	SO701365	22-Jul-99	28	19	4	*	51	57	4	0.089	51	65	0.539	Zippin
24	Leadon	SO697410	20-Jul-99	10	2	0	*	12	12	0	0.540	12	12	0.850	Zippin
25	Glinch	SO775277	19-Aug-99	50	17	8	*	75	79	3	0.598	75	86	0.619	Zippin
26	El	SO767248	19-Aug-99	49	15	*	*	64	71	5	*	*	*	*	Seber&LeCren
28	Swilgate	SO891323	22-Jul-99	87	60	50	*	197	342	69	0.546	206	478	0.248	Zippin
29	Swilgate	SO900298	22-Jul-99	33	14	2	*	49	51	2	0.216	49	54	0.681	Zippin
41	Queenhill	SO863364	18-Aug-99	94	45	*	*	139	180	21	*	*	*	*	Seber&LeCren
42	Queenhill	SO837369	19-Aug-99	44	38	12	*	94	118	13	0.027	94	144	0.414	Zippin

MapID	Watercourse	NGR	Date	Run 1	Run 2	Run 3	Run 4	Total	Population Estimate	SE	Probability*	95%CI lower	95%CI higher	CaptureProbability**	Estimate Method
1	Lyd	SO633033	21-Jul-98	33	22	26	10	91	132	24	0.113	91	180	0.254	Zippin
2	Lyd	SO618067	20-Jul-98	2	2	1	*	5	8	10	0.710	5	27	0.263	Zippin
5	Little Avon	ST688956	22-Jul-98	243	84	*	*	327	368	13	*	*	*	0.663	Seber&LeCren
6	Little Avon	ST728926	22-Jul-98	18	7	0	*	25	27	2	0.151	3	31	0.694	Carle&Strube
7	Blakeney Brook	SO681068	21-Jul-98	143	33	11	*	187	190	2	0.431	187	195	0.743	Zippin
8	Blakeney Brook	SO654078	21-Jul-98	3	4	1	0	8	12	8	0.256	8	27	0.319	Zippin
10	Westbury Brook	SO718138	23-Jul-98	117	62	21	16	216	228	5	0.280	218	239	0.517	Zippin
11	Westbury Brook	SO697151	21-Jul-98	67	35	17	7	126	134	4	0.882	126	142	0.511	Zippin
13	Frome	SO756094	23-Jul-98	135	50	21	5	211	215	2	0.658	211	220	0.632	Zippin
14	Frome	SO781059	22-Jul-98	90	72	*	*	162	162	*	*	*	*	*	Minimum Estimate
16	Frome	SO864027	23-Jul-98	7	1	2	0	10	11	2	0.321	3	15	0.556	Carle&Strube
21	Leadon	SO771271	24-Jul-98	114	44	22	6	186	191	3	0.603	186	197	0.595	Zippin
22	Leadon	SO732298	24-Jul-98	177	82	30	*	289	313	8	0.425	297	330	0.574	Zippin
23	Leadon	SO701365	23-Jul-98	36	26	10	*	72	88	10	0.233	72	108	0.435	Zippin
24	Leadon	SO697410	24-Jul-98	9	5	3	*	17	21	5	0.945	17	31	0.427	Zippin
25	Glinch	SO775277	20-Aug-98	54	48	26	20	148	179	18	0.459	148	214	0.370	Variable P
26	El	SO767248	20-Aug-98	88	53	16	*	157	175	8	0.072	159	192	0.528	Zippin
27	El	SO733262	20-Aug-98	11	5	5	1	22	24	3	0.493	22	29	0.451	Zippin
28	Swilgate	SO891323	28-Jul-98	153	76	41	25	295	321	9	0.761	304	338	0.467	Zippin
29	Swilgate	SO900298	27-Jul-98	18	8	6	*	32	38	6	0.537	32	50	0.449	Zippin
30	Swilgate	SO916279	28-Jul-98	6	2	1	*	9	10	1	0.819	9	12	0.615	Zippin
41	Queenhill	SO863364	19-Aug-98	204	77	*	*	281	328	16	*	*	*	0.623	Seber&LeCren
42	Queenhill	SO837369	19-Aug-98	72	38	14	*	124	138	7	0.409	124	151	0.536	Zippin
43	Queenhill	SO834385	28-Jul-98	98	44	*	*	142	178	18	*	*	*	0.551	Seber&LeCren

MapID	Watercourse	NGR	Date	Run 1	Run 2	Run 3	Run 4	Total	Population Estimate	SE	Probability*	95%CI lower	95%CI higher	CaptureProbability**	Estimate Method
1	Lyd	SO633033	28-Jun-83	59	41	23	26	149	205	25	0.280	156	255	0.276	Zippin
2	Lyd	SO618067	28-Jun-83	7	0	0	*	7	7	0	0.933	7	7	0.990	Minimum Estimate
5	Little Avon	ST688956	5-Jul-83	66	62	66	35	229	229	*	*	*	*	*	Minimum Estimate
6	Little Avon	ST728926	5-Jul-83	3	2	1	*	6	8	4	0.875	6	14	0.407	Zippin
7	Blakeney Brook	SO681068	30-Jun-83	86	31	23	21	161	251	87	0.680	161	421	0.182	Variable P
8	Blakeney Brook	SO654078	30-Jun-83	2	0	1	0	3	3	1	0.343	3	4	0.539	Zippin
10	Westbury Brook	SO718138	29-Jun-83	94	70	45	42	251	362	40	0.551	284	440	0.256	Zippin
11	Westbury Brook	SO697151	29-Jun-83	18	13	11	4	46	58	9	0.526	46	75	0.329	Zippin
12	Westbury Brook	SO678159	29-Jun-83	0	0	0	1	1	1	*	*	*	*	*	Minimum Estimate
13	Frome	SO756094	4-Jul-83	134	39	11	5	189	191	1	0.733	189	193	0.696	Zippin
14	Frome	SO781059	4-Jul-83	5	*	*	*	5	5	*	*	*	*	*	Minimum Estimate
16	Frome	SO864027	4-Jul-83	2	0	1	*	3	4	2	0.231	3	9	0.407	Zippin
21	Leadon	SO771271	18-Jul-83	92	43	35	25	195	280	58	0.757	195	394	0.233	Variable P
22	Leadon	SO732298	18-Jul-83	119	67	22	29	237	263	9	0.009	245	281	0.441	Zippin
23	Leadon	SO701365	19-Jul-83	24	12	13	15	64	133	66	0.223	64	263	0.151	Zippin
24	Leadon	SO697410	19-Jul-83	39	27	17	12	95	119	13	0.978	95	144	0.330	Zippin
25	Glinch	SO775277	11-Aug-83	101	25	9	15	150	182	29	0.034	150	239	0.267	Zippin
26	El	SO767248	12-Aug-83	83	103	69	34	289	343	20	0.324	303	382	0.409	Variable P
27	El	SO733262	12-Aug-83	5	12	2	5	24	28	5	0.052	24	38	0.442	Variable P
28	Swilgate	SO891323	6-Jul-83	24	27	14	35	100	100	*	*	*	*	*	Minimum Estimate
29	Swilgate	SO900298	6-Jul-83	26	38	13	5	82	85	2	0.873	82	89	0.644	Variable P
30	Swilgate	SO916279	6-Jul-83	5	4	2	1	12	13	2	1.000	12	17	0.500	Variable P
41	Queenhill	SO863364	29-Jul-83	191	*	*	*	191	191	*	*	*	*	*	Minimum Estimate
42	Queenhill	SO837369	29-Jul-83	21	14	*	*	35	63	35	*	*	*	*	Seber&LeCren
43	Queenhill	SO834385	29-Jul-83	13	1	*	*	14	14	0	*	*	*	*	Seber&LeCren

* Probability - results from a goodness of fit of a chi-squared test of the observed and expected numbers captured

** Capture Probability - the probability that an individual will be captured in the sample

Table B.2 Biomass and density estimates for the surveys undertaken on the Lower Severn tributaries in 1983, 1998 and 1999.

MapID	Watercourse	NGR	Date	Area (m^2)	Total Number	Population Estimate	Mean Length (mm)	Density ($n\ m^{-2}$)	95%Cl lower	95%Cl higher	Mean Weight (g)	Biomass ($g\ m^{-2}$)	95%Cl lower	95%Cl higher
1	Lyd	SO633033	19-Jul-99	488	128	154	221	0.315	0.266	0.365	34	10.637	8.970	12.304
5	Little Avon	ST688956	21-Jul-99	485	298	317	204	0.654	0.626	0.682	20	13.267	12.696	13.839
7	Blakeney Brook	SO681068	19-Jul-99	449	147	160	230	0.356	0.329	0.383	27	9.546	8.818	10.274
10	Westbury Brook	SO718138	20-Jul-99	365	172	187	229	0.513	0.476	0.550	22	11.542	10.714	12.370
11	Westbury Brook	SO697151	20-Jul-99	233	88	102	265	0.439	0.378	0.508	47	20.617	17.743	23.882
12	Westbury Brook	SO678159	20-Jul-99	178	4	4	363	0.024	0.022	0.036	104	2.548	2.339	3.747
13	Frome	SO756094	21-Jul-99	290	57	78	269	0.270	0.197	0.377	45	12.147	8.845	16.967
21	Leadon	SO771271	22-Jul-99	358	81	165	202	0.462	0.226	0.859	19	8.576	4.197	15.946
22	Leadon	SO732298	21-Jul-99	490	120	182	272	0.373	0.245	0.510	43	15.842	10.420	21.675
23	Leadon	SO701365	22-Jul-99	407	51	57	240	0.139	0.125	0.160	30	4.196	3.785	4.836
24	Leadon	SO697410	20-Jul-99	269	12	12	303	0.045	0.045	0.046	51	2.295	2.287	2.377
25	Glinch	SO775277	19-Aug-99	252	75	79	191	0.315	0.298	0.341	16	4.947	4.673	5.344
26	Ell	SO767248	19-Aug-99	313	64	71	221	0.225	•	•	22	4.962	•	•
28	Swilgate	SO891323	22-Jul-99	352	197	342	238	0.972	0.586	1.358	32	31.093	18.752	43.433
29	Swilgate	SO900298	22-Jul-99	204	49	51	266	0.249	0.241	0.266	41	10.082	9.755	10.768
41	Queenhill	SO863364	18-Aug-99	414	139	180	356	0.436	•	•	100	43.435	•	•
42	Queenhill	SO837369	19-Aug-99	381	94	118	308	0.308	0.246	0.376	60	18.522	14.797	22.596

MapID	Watercourse	NGR	Date	Area	Total	Population Estimate	Mean Length	Density	95%Cl lower	95%Cl higher	Mean Weight	Biomass	95%Cl lower	95%Cl higher
1	Lyd	SO633033	21-Jul-98	430	91	132	240	0.307	0.212	0.418	38	11.707	8.074	15.936
2	Lyd	SO618067	20-Jul-98	404	5	8	527	0.021	0.012	0.068	292	6.041	3.619	19.783
5	Little Avon	ST688956	22-Jul-98	568	327	368	226	0.648	•	•	26	16.570	•	•
6	Little Avon	ST728926	22-Jul-98	366	25	27	162	0.074	0.009	0.085	152	11.231	1.399	13.008
7	Blakeney Brook	SO681068	21-Jul-98	394	187	190	237	0.483	0.475	0.494	27	12.858	12.640	13.153
8	Blakeney Brook	SO654078	21-Jul-98	375	8	12	444	0.031	0.021	0.071	210	6.537	4.472	14.901
10	Westbury Brook	SO718138	23-Jul-98	354	216	228	219	0.645	0.616	0.675	23	14.969	14.284	15.654
11	Westbury Brook	SO697151	21-Jul-98	256	126	134	266	0.522	0.492	0.554	48	25.202	23.762	26.770
13	Frome	SO756094	23-Jul-98	289	211	215	233	0.744	0.730	0.760	32	23.587	23.156	24.113
14	Frome	SO781059	22-Jul-98	880	162	162	256	0.184	•	•	32	5.981	•	•
16	Frome	SO864027	23-Jul-98	787	10	11	431	0.014	0.004	0.019	164	2.294	0.701	3.060
21	Leadon	SO771271	24-Jul-98	354	186	191	180	0.540	0.525	0.556	13	6.808	6.625	7.014
22	Leadon	SO732298	24-Jul-98	571	289	313	286	0.549	0.520	0.578	50	27.648	26.182	29.113
23	Leadon	SO701365	23-Jul-98	367	72	88	236	0.239	0.196	0.293	26	6.135	5.027	7.514
24	Leadon	SO697410	24-Jul-98	227	17	21	279	0.092	0.075	0.137	44	4.071	3.304	6.039
25	Glinch	SO775277	20-Aug-98	190	148	179	205	0.942	0.777	1.125	17	16.289	13.447	19.468
26	Ell	SO767248	20-Aug-98	356	157	175	199	0.493	0.447	0.538	18	8.971	8.143	9.800
27	Ell	SO733262	20-Aug-98	258	22	24	295	0.094	0.085	0.114	63	5.885	5.349	7.143
28	Swilgate	SO891323	28-Jul-98	305	295	321	250	1.052	0.996	1.108	32	33.261	31.502	35.020
29	Swilgate	SO900298	27-Jul-98	185	32	38	274	0.208	0.173	0.273	40	8.313	6.919	10.918
30	Swilgate	SO916279	28-Jul-98	202	9	10	252	0.047	0.045	0.058	30	1.407	1.327	1.741
41	Queenhill	SO863364	19-Aug-98	462	281	328	346	0.710	•	•	92	65.315	•	•
42	Queenhill	SO837369	19-Aug-98	357	124	138	307	0.386	0.348	0.424	68	26.395	23.776	29.014
43	Queenhill	SO834385	28-Jul-98	285	142	178	290	0.624	•	•	50	31.224	•	•

MapID	Watercourse	NGR	Date	Area	Total	Population Estimate	Mean Length	Density	95%Cl lower	95%Cl higher	Mean Weight	Biomass	95%Cl lower	95%Cl higher
1	Lyd	SO633033	28-Jun-83	536	149	205	192	0.383	0.290	1.712	22	8.454	6.408	37.771
2	Lyd	SO618067	28-Jun-83	400	7	7	314	0.018	0.018	1.000	71	1.247	1.247	71.243
5	Little Avon	ST688956	5-Jul-83	668	229	229	227	0.343	•	•	29	9.969	•	•
6	Little Avon	ST728926	5-Jul-83	264	6	8	377	0.029	0.023	2.407	146	4.196	3.320	351.699
7	Blakeney Brook	SO681068	30-Jun-83	496	161	251	231	0.507	0.325	2.616	69	34.975	22.397	180.511
8	Blakeney Brook	SO654078	30-Jun-83	346	3	3	278	0.009	0.009	1.404	42	0.382	0.365	59.119
10	Westbury Brook	SO718138	29-Jun-83	352	251	362	167	1.028	0.808	1.751	18	19.020	14.945	32.388
11	Westbury Brook	SO697151	29-Jun-83	294	46	58	227	0.196	0.156	1.640	29	5.707	4.553	47.729
12	Westbury Brook	SO678159	29-Jun-83	176	1	1	191	0.006	•	•	10	0.057	•	•
13	Frome	SO756094	4-Jul-83	298	189	191	255	0.640	0.634	1.024	41	26.044	25.822	41.677
14	Frome	SO781059	4-Jul-83	850	5	5	301	0.006	•	•	63	0.370	•	•
16	Frome	SO864027	4-Jul-83	488	3	4	451	0.008	0.006	2.881	233	1.809	1.432	670.881
21	Leadon	SO771271	18-Jul-83	400	195	280	188	0.699	0.488	2.020	16	11.180	7.794	32.291
22	Leadon	SO732298	18-Jul-83	597	237	263	246	0.440	0.410	1.185	35	15.580	14.502	41.960
23	Leadon	SO701365	19-Jul-83	468	64	133	215	0.285	0.137	4.109	28	8.052	3.869	116.265
24	Leadon	SO697410	19-Jul-83	264	95	119	126	0.451	0.360	1.521	6	2.643	2.110	8.918
25	Glinch	SO775277	11-Aug-83	248	150	182	214	0.733	0.605	1.592	27	20.063	16.556	43.567
26	Ell	SO767248	12-Aug-83	396	289	343	178	0.865	0.765	1.324	15	12.569	11.110	19.220
27	Ell	SO733262	12-Aug-83	328	24	28	266	0.085	0.073	1.570	41	3.471	2.976	63.844
28	Swilgate	SO891323	6-Jul-83	492	100	100	234	0.203	•	•	30	6.024	•	•
29	Swilgate	SO900298	6-Jul-83	256	82	85	226	0.331	0.320	1.089	29	9.624	9.324	31.713
30	Swilgate	SO916279	6-Jul-83	210	12	13	191	0.062	0.057	1.422	19	1.190	1.098	27.334
41	Queenhill	SO863364	29-Jul-83	1800	191	191	327	0.106	•	•	89	9.448	•	•
42	Queenhill	SO837369	29-Jul-83	400	35	63	327	0.158	•	•	83	13.117	•	•
43	Queenhill	SO834385	29-Jul-83	300	14	14	296	0.047	•	•	78	3.648	•	•

Table B.3 Population estimates for the surveys undertaken on the Upper Severn tributaries in 1983 and 1999.

MapID	Watercourse	NGR	Date	Run 1	Run 2	Run 3	Run 4	Total	Population Estimate	S.E	Probability*	95%Cl lower	95%Cl higher	CaptureProbability**	Estimate method
44	Leigh	SO782535	13-Jul-99	12	8	2	*	22	23	1	0.339	22	31	0.75	Zippin
48	Kyre	SO607674	14-Jul-99	6	0	*	*	6	6	*	*	*	*	*	MinimumEstimate
51	Grimley	SO830608	13-Jul-99	2	0	*	*	2	2	*	*	*	*	*	MinimumEstimate
52	Shrawley	SO811638	14-Jul-99	2	1	*	*	3	3	*	*	*	*	*	MinimumEstimate
59	Smestow	SO862873	15-Jul-99	0	0	*	*	0	*	*	*	*	*	*	*
60	Smestow	SO862870	15-Jul-99	0	0	*	*	0	*	*	*	*	*	*	*
62	Dowles	SO777763	14-Jul-99	4	0	*	*	4	4	*	*	*	*	*	MinimumEstimate
66	Borle	SO677901	15-Jul-99	0	0	*	*	0	*	*	*	*	*	*	*

MapID	Watercourse	NGR	Date	Run 1	Run 2	Run 3	Run 4	Total	Population Estimate	S.E	Probability*	95%Cl lower	95%Cl higher	CaptureProbability**	Estimate method
44	Leigh	SO782535	24-Aug-83	5	2	2	1	10	11	2	0.854	10	16	0.40	Zippin
48	Kyre	SO607674	25-Aug-83	0	0	*	*	0	*	*	*	*	*	*	*
51	Grimley	SO830608	22-Aug-83	1	*	*	*	1	1	*	*	*	*	*	MinimumEstmate
52	Shrawley	SO811638	22-Aug-83	5	*	*	*	5	5	*	*	*	*	*	MinimumEstmate
59	Smestow	SO862873	14-Jul-83	0	0	*	*	0	*	*	*	*	*	*	*
60	Smestow	SO862870	14-Jul-83	0	0	*	*	0	*	*	*	*	*	*	*
62	Dowles	SO777763	1-Sep-83	0	1	1	*	2	2	*	*	*	*	*	MinimumEstmate
66	Borle	SO677901	18-Aug-83	0	0	*	*	0	*	*	*	*	*	*	*

* Probability - results from goodness of fit of a chi-squared test of the observed and expected numbers captured.
** Capture Probability - the probability that an individual will be captured during the sample.

Table B.4 Biomass and density estimates for the surveys undertaken on the Upper Severn tributaries in 1983 and 1999.

Site	Watercourse	NGR	Date	Area (m^2)	Total Number	Population Estimate	Mean Length (mm)	Density (n m^{-2})	95%Cl lower	95%Cl higher	Mean Weight (g)	Biomass (g m^{-2})	95%Cl lower	95%Cl higher
44	Leigh	SO782535	13-Jul-99	624	22	23	289	0.036	0.035	1.000	59.86	2.174	2.110	59.860
48	Kyre	SO607674	14-Jul-99	574	6	6	306	0.010	*	*	55.00	0.575	*	*
51	Grimley	SO830608	13-Jul-99	261	2	2	418	0.008	*	*	104.00	0.797	*	*
52	Shrawley	SO811638	14-Jul-99	156	3	3	186	0.019	*	*	7.33	0.141	*	*
59	Smestow	SO862873	15-Jul-99	473	0	*	*	*	*	*	*	*	*	*
60	Smestow	SO862870	15-Jul-99	716	0	*	*	*	*	*	*	*	*	*
62	Dowles	SO777763	14-Jul-99	544	4	4	367	0.007	*	*	90.00	0.662	*	*
66	Borle	SO677901	15-Jul-99	322	0	*	*	*	*	*	*	*	*	*

Site	Watercourse	NGR	Date	Area	Total Number	Population Estimate	Mean Length	Density	95%Cl lower	95%Cl higher	Mean Weight	Biomass	95%Cl lower	95%Cl higher
44	Leigh	SO782535	24-Aug-83	500	10	11	236	0.023	0.020	1.630	23.91	0.548	0.478	38.973
48	Kyre	SO607674	25-Aug-83	350	0	*	*	*	*	*	*	*	*	*
51	Grimley	SO830608	22-Aug-83	350	1	1	413	0.003	*	*	130.00	0.371	*	*
52	Shrawley	SO811638	22-Aug-83	170	5	5	222	0.029	*	*	26.79	0.788	*	*
59	Smestow	SO862873	14-Jul-83	500	0	*	*	*	*	*	*	*	*	*
60	Smestow	SO862870	14-Jul-83	400	0	*	*	*	*	*	*	*	*	*
62	Dowles	SO777763	1-Sep-83	508	2	2	348	0.004	*	*	91.15	0.359	*	*
66	Borle	SO677901	18-Aug-83	314	0	*	*	*	*	*	*	*	*	*

Table B.5 Population estimates for the surveys undertaken on the Dee catchment in 1984 and 1999

Dee 1999

Site	Watercourse	NGR	Date	Run 1	Run 2	Run 3	Total	Population Estimate	S.E	Probability*	95%Cl lower	95%Cl higher	CaptureProbability**	Estimate method
1	River Alyn	ST396562	16-Aug-99	148	77	41	266	311	15	0.836	219	340	0.475	Zippin
2	Worthenbury Brook	SJ418463	16-Sep-99	117	67	*	184	274	43	*	*	*	0.427	Seber&LeCren
3	River Clweddog	SJ396482	18-Aug-99	92	71	28	191	241	20	0.035	203	279	0.409	Zippin
4	River Ceiriog	SJ317393	16-Sep-99	34	27	29	90	90	*	*	*	*	*	Minimum Estimate
5	River Ceirw	SJ019445	17-Aug-99	24	22	15	61	125	54	0.582	61	249	0.200	Carle&Strube
6	River Ceidiog	SJ029382	17-Aug-99	7	12	6	25	25	*	*	*	*	*	Minimum Estimate
7	River Himant	SH949362	17-Aug-99	10	1	*	11	11	0	*	*	*	0.900	Seber&LeCren
8	River Lliw	SH873307	17-Aug-99	7	0	*	7	7	0	*	*	*	0.990	Seber&LeCren

Dee 1984

Site	Watercourse	NGR	Date	Run 1	Run 2	Run 3	Total	Population Estimate	S.E	Probability*	95%Cl lower	95%Cl higher	CaptureProbability**	Estimate method
1	River Alyn	ST396562	10-Aug-84	101	75	53	229	371	59	0.863	255	488	0.274	Zippin
2	Worthenbury Brook	SJ418463	9-Aug-84	357	260	152	769	1085	65	0.162	958	1213	0.337	Zippin
3	River Clweddog	SJ396482	20-Aug-84	138	49	*	187	214	12	*	*	*	0.645	Seber&LeCren
4	River Ceiriog	SJ317393	9-Aug-84	?	37	23	60	60	*	*	*	*	*	Minimum Estimate
5	River Ceirw	SJ019445	17-Aug-84	30	?	?	30	30	*	*	*	*	*	Minimum Estimate
6	River Ceidiog	SJ029382	13-Sep-84	13	*	*	13	13	*	*	*	*	*	Minimum Estimate
7	River Himant	SH949362	11-Aug-84	8	*	*	8	8	*	*	*	*	*	Minimum Estimate
8	River Lliw	SH873307	17-Aug-84	27	19	8	54	66	10	0.365	53	86	0.413	Zippin

* Probability - results from goodness of fit of a chi-squared test of the observed and expected numbers captured
** Capture probability - the probability that an individual will be captured during the sample

Table B.6 Biomass and density estimates for the surveys undertaken on the Dee catchment in 1984 and 1999

Dee 1999

Site	Watercourse	NGR	Date	Area (m^2)	Total Number	Population Estimate	Mean Length (mm)	Density (n m^{-2})	95%Cl lower	95%Cl higher	Mean Weight (g)	Biomass (g m^{-2})	95%Cl lower	95%Cl higher
1	River Alyn	ST396562	16-Aug-99	638	266	311	229	0.488	0.343	1.280	27	13.402	9.423	35.161
2	Worthenbury Brook	SJ418463	16-Sep-99	400	184	274	235	0.685	*	*	41	28.013	*	*
3	River Clweddog	SJ396482	18-Aug-99	826	191	241	176	0.292	0.245	1.461	16	4.722	3.972	23.667
4	River Ceiriog	SJ317393	16-Sep-99	671	90	90	324	0.134	*	*	69	9.302	*	*
5	River Ceirw	SJ019445	17-Aug-99	1026	61	125	358	0.122	0.059	*	100	12.161	5.935	*
6	River Ceidiog	SJ029382	17-Aug-99	764	25	25	393	0.033	*	*	128	4.190	*	*
7	River Hirnant	SH949362	17-Aug-99	683	11	11	336	0.016	*	*	66	1.079	*	*
8	River Lliw	SH873307	17-Aug-99	1150	7	7	400	0.006	*	*	116	0.709	*	*

Dee 1984

Site	Watercourse	NGR	Date	Area	Total Number	Population Estimate	Mean Length	Density	95%Cl lower	95%Cl higher	Mean Weight	Biomass	95%Cl lower	95%Cl higher
1	River Alyn	ST396562	10-Aug-84	698	229	371	267	0.532	0.365	2.130	62	32.965	22.642	132.086
2	Worthenbury Brook	SJ418463	9-Aug-84	311	769	1085	188	3.489	3.079	1.577	17	57.948	51.140	26.198
3	River Clweddog	SJ396482	20-Aug-84	736	187	214	185	0.291	*	*	21	6.022	*	*
4	River Ceiriog	SJ317393	9-Aug-84	878	60	60	200	0.068	*	*	20	1.362	*	*
5	River Ceirw	SJ019445	17-Aug-84	764	30	30	408	0.039	*	*	230	9.042	*	*
6	River Ceidiog	SJ029382	13-Sep-84	680	13	13	338	0.019	*	*	79	1.506	*	*
7	River Hirnant	SH949362	11-Aug-84	668	8	8	352	0.012	*	*	74	0.886	*	*
8	River Lliw	SH873307	17-Aug-84	820	54	66	410	0.080	0.065	1.592	149	12.022	9.654	237.736

Table B.7 Population estimates for the surveys undertaken on the Frome and Piddle catchments in 1999

Tadnoll Brook 1999

Section	Watercourse	NGR	Date	Run 1	Run 2	Run 3	Total	Population Estimate	S.E	Probability*	95%CI lower	95%CI higher	CaptureProbability**	Estimate method
1	Tadnoll Brook	SY805878	7-Jun-99	15	2	2	19	20	2	0.194	3	24	0.679	Carle&Strube
2	Tadnoll Brook	SY804877	7-Jun-99	15	17	10	42	101	82	0.299	42	261	0.164	Zippin
3	Tadnoll Brook	SY802877	7-Jun-99	14	11	*	25	65	86	*	*	*	0.214	Seber&LeCren
4	Tadnoll Brook	SY802876	8-Jun-99	24	15	7	46	56	8	0.663	46	71	0.443	Zippin
5	Tadnoll Brook	SY801875	8-Jun-99	22	34	7	63	63	*	*	*	*	*	Minimum Estimate
6	Tadnoll Brook	SY801874	8-Jun-99	44	22	8	74	81	5	0.579	74	91	0.579	Zippin
7	Tadnoll Brook	SY794872	8-Jun-99	19	13	3	35	39	4	0.183	35	47	0.527	Zippin
8	Tadnoll Brook	SY793869	8-Jun-99	19	18	8	45	67	19	0.228	45	104	0.313	Zippin

River Piddle 1999 (all eels)

Section	Watercourse	NGR	Date	Run 1	Run 2	Run 3	Total	Population Estimate	S.E	Probability*	95%CI lower	95%CI higher	CaptureProbability**	Estimate method
1	Wareham Common	SY921877	9-Jun-99	47	23	12	82	94	7	0.903	82	108	0.498	Zippin
2	Ferncroft Farm	SY909876	9-Jun-99	62	29	21	112	136	12	0.329	112	160	0.329	Zippin
5	Worgret Heath Farm	SY898881	9-Jun-99	33	25	16	74	114	28	0.737	74	168	0.296	Zippin
10	Trigon Estate	SY888885	10-Jun-99	43	17	*	60	71	8	*	*	*	0.605	Seber&LeCren
13	Forest Lodge House	SY873897	14-Jun-99	71	33	13	117	128	6	0.720	117	139	0.562	Zippin
15	Forest Lodge House	SY867906	14-Jun-99	42	18	9	69	76	5	0.796	69	86	0.546	Zippin
16	Bere Stream	SY858919	10-Jun-99	24	7	5	36	39	3	0.315	36	44	0.591	Zippin
17	Woodlands	SY864908	10-Aug-99	44	18	10	72	80	5	0.602	72	90	0.541	Zippin
19	Culheaze Farm	SY849923	12-Aug-99	84	33	12	129	137	4	0.868	129	145	0.617	Zippin

River Piddle 1999 (>250mm eels only)

Section	Watercourse	NGR	Date	Run 1	Run 2	Run 3	Total	Population Estimate	S.E	Probability*	95%CI lower	95%CI higher	CaptureProbability**	Estimate method
1	Wareham Common	SY921877	9-Jun-99	39	21	9	69	78	6	0.690	69	91	0.506	Zippin
2	Ferncroft Farm	SY909876	9-Jun-99	56	25	19	100	121	11	0.266	100	143	0.444	Zippin
5	Worgret Heath Farm	SY898881	9-Jun-99	33	24	14	71	100	20	0.670	71	138	0.338	Zippin
10	Trigon Estate	SY888885	10-Jun-99	40	17	*	57	66	6	*	*	*	0.620	Seber&LeCren
13	Forest Lodge House	SY873897	14-Jun-99	62	24	9	95	101	4	0.953	95	108	0.617	Zippin
15	Forest Lodge House	SY867906	14-Jun-99	39	17	8	64	70	5	0.902	64	79	0.552	Zippin
16	Bere Stream	SY858919	10-Jun-99	23	7	4	34	36	2	0.496	34	40	0.618	Zippin
17	Woodlands	SY864908	10-Aug-99	42	15	8	65	70	4	0.533	65	77	0.586	Zippin
19	Culheaze Farm	SY849923	12-Aug-99	79	32	10	121	128	4	0.594	121	135	0.628	Zippin

Frome/Piddle 1999 (NRA 1990 survey sites)

Site	Watercourse	NGR	Date	Run 1	Run 2	Run 3	Total	Population Estimate	S.E	Probability*	95%CI lower	95%CI higher	CaptureProbability**	Estimate method
1	Wraxhall Brook	ST577008	9-Aug-99	39	10	3	52	53	1	0.857	52	56	0.732	Zippin
2	Greys Bridge Dorcester	SY705909	9-Aug-99	25	8	1	34	37	1	0.497	37	39	0.763	Zippin
3	Carrier at Lewell Mill	SY380901	11-Aug-99	122	56	33	211	242	12	0.451	219	266	0.494	Zippin
4	N Stream at 9-htaches	SY749908	11-Aug-99	51	19	9	79	85	4	0.681	79	92	0.594	Zippin
5	Tadnoll Brook at Broomhill	SY813879	10-Aug-99	69	41	26	136	179	19	0.867	139	213	0.389	Zippin
6	Mill Stream at East Stoke	SY872867	11-Aug-99	58	29	14	101	115	8	0.941	101	130	0.507	Zippin
7	Southover Farm Tolpuddle	SY783941	10-Aug-99	13	8	7	28	45	20	0.678	28	84	0.278	Zippin
8	Throop	SY826932	10-Aug-99	20	9	2	31	33	2	0.475	31	36	0.637	Zippin
9	Trigon Estate	SY890885	12-Aug-99	37	17	7	61	67	4	0.863	61	75	0.558	Zippin

* Probability - results from goodness of fit of a chi-squared test of the observed and expected numbers captured
** Capture probability - the probability that an individual will be captured during the sample

Table B.8 Biomass and density estimates for the surveys undertaken on the Frome and Piddle catchments in 1999

Tadnoll Brook 1999

Section	Watercourse	NGR	Date	Area (m^2)	Total Number	Population Estimate	Mean Length (mm)	Density (nm^{-2})	95%Cl lower	95%Cl higher	Mean Weight (g)	Biomass ($g\ m^{-2}$)	95%Cl lower	95%Cl higher
1	Tadnoll Brook	SY805878	7-Jun-99	386	19	20	326	0.052	0.009	0.062	78	4.020	0.675	4.776
2	Tadnoll Brook	SY804877	7-Jun-99	345	42	101	342	0.293	0.122	0.756	87	25.490	10.600	65.842
3	Tadnoll Brook	SY802877	7-Jun-99	310	25	65	329	0.210	*	*	79	16.598	*	*
4	Tadnoll Brook	SY802876	8-Jun-99	378	46	56	350	0.148	0.122	0.187	96	14.261	11.714	17.967
5	Tadnoll Brook	SY801875	8-Jun-99	338	63	63	326	0.186	*	*	79	14.729	*	*
6	Tadnoll Brook	SY801874	8-Jun-99	480	74	81	345	0.169	0.154	0.189	116	19.621	17.925	21.978
7	Tadnoll Brook	SY794872	8-Jun-99	289	35	39	294	0.135	0.121	0.162	63	8.479	7.609	10.188
8	Tadnoll Brook	SY793869	8-Jun-99	384	45	67	312	0.174	0.117	0.270	80	13.901	9.336	21.547

River Piddle 1999 (all eels)

Section	Watercourse	NGR	Date	Area	Total Number	Population Estimate	Mean Length	Density	95%Cl lower	95%Cl higher	Mean Weight	Biomass	95%Cl lower	95%Cl higher
1	Wareham Common	SY921877	9-Jun-99	756	82	94	335	0.124	0.108	0.143	100	12.474	10.881	14.320
2	Ferncroft Farm	SY909876	9-Jun-99	933	112	136	397	0.146	0.120	0.171	155	22.635	18.647	26.590
5	Worgret Heath Farm	SY898881	9-Jun-99	880	74	114	394	0.130	0.084	0.191	141	18.245	11.843	26.897
10	Trigon Estate	SY888885	10-Jun-99	923	60	71	396	0.077	*	*	110	8.464	*	*
13	Forest Lodge House	SY873897	14-Jun-99	666	117	128	347	0.192	0.176	0.209	96	18.522	16.930	20.126
15	Forest Lodge House	SY867906	14-Jun-99	749	69	76	384	0.101	0.092	0.114	120	12.183	11.061	13.743
16	Bere Stream	SY858919	10-Jun-99	802	36	39	434	0.049	0.045	0.055	191	9.272	8.559	10.444
17	Woodlands	SY864908	10-Aug-99	772	72	80	408	0.104	0.093	0.116	141	14.660	13.194	16.472
19	Culheaze Farm	SY849923	12-Aug-99	855	129	137	400	0.160	0.151	0.170	134	21.543	20.285	22.820

River Piddle 1999 (>250mm eels only)

Section	Watercourse	NGR	Date	Area	Total Number	Population Estimate	Mean Length	Density	95%Cl lower	95%Cl higher	Mean Weight	Biomass	95%Cl lower	95%Cl higher
1	Wareham Common	SY921877	9-Jun-99	756	69	78	361	0.103	0.091	0.120	116	11.962	10.582	13.923
2	Ferncroft Farm	SY909876	9-Jun-99	933	100	121	420	0.130	0.107	0.153	172	22.312	18.439	26.331
5	Worgret Heath Farm	SY898881	9-Jun-99	880	71	100	401	0.114	0.081	0.157	146	16.575	11.768	22.951
10	Trigon Estate	SY888885	10-Jun-99	923	57	66	405	0.072	*	*	115	8.222	*	*
13	Forest Lodge House	SY873897	14-Jun-99	666	95	101	382	0.152	0.143	0.162	115	17.514	16.474	18.714
15	Forest Lodge House	SY867906	14-Jun-99	749	64	70	398	0.093	0.085	0.106	128	11.935	10.912	13.517
16	Bere Stream	SY858919	10-Jun-99	802	34	36	447	0.045	0.042	0.050	201	9.009	8.509	10.094
17	Woodlands	SY864908	10-Aug-99	772	65	70	428	0.091	0.084	0.100	155	14.025	13.024	15.473
19	Culheaze Farm	SY849923	12-Aug-99	855	121	128	412	0.150	0.142	0.158	142	21.248	20.086	22.450

Frome/Piddle (NRA 1990 survey sites)

Site	Watercourse	NGR	Date	Area	Total Number	Population Estimate	Mean Length	Density	95%Cl lower	95%Cl higher	Mean Weight	Biomass	95%Cl lower	95%Cl higher
1	Wraxhall Brook	ST577008	9-Aug-99	386	52	53	446	0.137	0.135	0.144	198	27.244	26.730	28.545
2	Greys Bridge Dorchester	SY705909	9-Aug-99	507	34	37	421	0.073	0.073	0.077	181	13.196	13.196	13.970
3	Carrier at Lewell Mill	SY380901	11-Aug-99	600	211	242	403	0.403	0.365	0.443	143	57.733	52.305	63.395
4	N Stream at 9-htaches	SY749908	11-Aug-99	564	79	85	340	0.151	0.140	0.164	89	13.384	12.440	14.537
5	Tadnoll Brook at Broomhill	SY813879	10-Aug-99	428	136	179	335	0.418	0.325	0.497	81	33.855	26.340	40.271
6	Mill Stream at East Stoke	SY872867	11-Aug-99	683	101	115	286	0.168	0.148	0.190	56	9.459	8.308	10.661
7	Southover Farm Tolpuddle	SY783941	10-Aug-99	494	28	45	419	0.091	0.057	0.169	144	13.078	8.138	24.311
8	Throop	SY826932	10-Aug-99	575	31	33	422	0.057	0.054	0.063	165	9.449	8.877	10.372
9	Trigon Estate	SY890885	12-Aug-99	855	61	67	308	0.078	0.071	0.088	61	4.746	4.321	5.322

Appendix C

MONITORING STRATEGY FOR RIVER EEL STOCKS BASED ON EEL-SPECIFIC ELECTRIC-FISHING SURVEYS OF KEY RIVERS

C1 Introduction

As indicated in Section 7.1, the primary management objective is to maintain eel populations in England and Wales at a favourable conservation status, by the setting of conservation limits (threshold reference limits and management targets) below which stocks and spawner escapement should not be allowed to fall. However, there are major practical difficulties and uncertainties in directly monitoring silver eel production and escapement. In contrast, monitoring of yellow eel stocks in rivers is inherently more straightforward. Several studies (e.g. Mann & Blackburn 1991; Aprahamian 1986, 1988; this R&D project, Chapter 3) have clearly demonstrated that eel specific electric fishing surveys in relatively small and shallow channels provides an effective and efficient method for the quantitative determination of river eel stocks. On the assumption that there is likely to be a direct relationship between yellow eel production and subsequent silver eel escapement, the direct monitoring of riverine yellow eel stocks has been recommended as an essential and principal component of the proposed eel stock monitoring strategy (Chapter 8)

This Appendix examines and makes recommendations for basic survey design criteria for monitoring of yellow eel stocks in key rivers based on catch depletion population estimation using electric fishing as the capture method.

Good survey design is essential to the success of a monitoring programme. In order to achieve this, monitoring objectives should be stated in specific, quantitative terms to facilitate the design of a study or programme and the estimation of sample sizes required to fulfil the objectives. An objective statement should include the following:

- the environmental attribute to be measured e.g. eel density or biomass;
- the target population, i.e., target season, waterbody type, size classification, location etc.;
- the desired level of confidence and the desired statistical power to detect a spatial or temporal difference or change, i.e. population trend or deviation from a preset management target or reference level;
- an acceptable maximum magnitude of error, or the acceptable minimum detectable magnitude of difference or change in a selected indicator.

In order to facilitate the development of an appropriate objective and method statement for eel monitoring, survey and monitoring requirements are considered under four section headings, namely:

- an assessment of population attributes to be measured and survey frequency;
- an assessment of the number of survey sites required to detect a given magnitude of change with a stated level of confidence;
- summary of key points and recommendations for monitoring programme design;
- recommendations for optimising the precision of electric fishing surveys of river eels.

C2 An Assessment of Population Attributes to be Measured and Survey Frequency

Before considering the number of sites required for a quantitative river eel survey by electric fishing, it is necessary to evaluate and define the population measures that are to be employed. The repeat surveys conducted in 1998 and 1999, together with the data reviewed in Chapter 2 of this report, provide an sound basis for assessing potential population measures. Quantifiable population attributes include length/frequency distribution, population density, biomass and age/growth rate. These are evaluated in turn in the following paragraphs. Geographic range within the catchment is considered separately in Section C.4.

C2.1 Length frequency distribution

The 1998/99 repeat surveys of the Severn, Dee and Piddle/Frome suggest that length frequency distribution potentially offers the most sensitive indicator of actual or potential eel population change in rivers, primarily because it measures the relative population density of different age classes. If glass eel recruitment rates to a system consistently fall below those required to maintain the population at or near to carrying capacity, then this should be detectable within about 5 years as a readily measurable reduction in the proportion of eels of <200 mm in the population. Even where changing recruitment rates remain within the range required to maintain populations, as would appear to be the case for the lower Severn (Section 3.7.3), changes in length frequency structure of the population will be indicative of changing population dynamics.

Statistically, population size structure can be expected to be a more sensitive measure than population density. For density (or biomass), sample size equates to the number of sites, whereas for length frequency, sample size will be much larger (unless a large number of size classes are employed and relatively few eels are caught). There is also some evidence (e.g. Severn eel surveys in 1998 and 1999, Section 3.7.3 and Figure 3.11) that length frequency distribution may be less susceptible than population density to the influence of factors such as temperature and season .

At lower river sites where overall population densities are usually relatively high and the smaller size classes of eels are typically abundant, and in terms of sensitivity and time scale, length frequency, in particular the proportion of small (<200 mm) eels, can be expected to be a very useful indicator of changing recruitment rates, and possibly also over fishing of glass eels. It potentially provides an early indicator that future changes in eel population density or biomass may follow.

However, because the optimum and minimum rates of elver recruitment to river systems are effectively unknown, there is no practical mechanism for setting quantitative thresholds in relation to length class frequency. Furthermore, length frequency is much less likely to be a sensitive or short term indicator of change at upper river sites where eel population densities are typically low and the population is dominated by the older age classes. Length class frequency may also not be a very sensitive indicator of yellow eel fishery mortality.

C2.2 Population density

Population density will obviously be a primary measure of yellow eel stock status, not least because, unlike length class frequency. it is amenable to the setting of management targets, or threshold levels of change. However, the sensitivity of population density as an indicator of changing eel population dynamics at typical river sites is at least partly dependant on the length frequency distribution of the catchable component of population. Figure C1 shows percentage length frequency and percentage cumulative length frequency for all eels captured during the 1999 surveys of the Severn, Dee and Piddle/Frome. Figure C.1 should therefore be generally representative of the catchable eel population at an average or typical river site. As implied by Figure C.1, the catchable population is dominated by eels in the 200-400 mm length class at the majority of sites Eels less than 200 mm comprise about 18% of the catch and eels larger than that 400 mm about 22%.

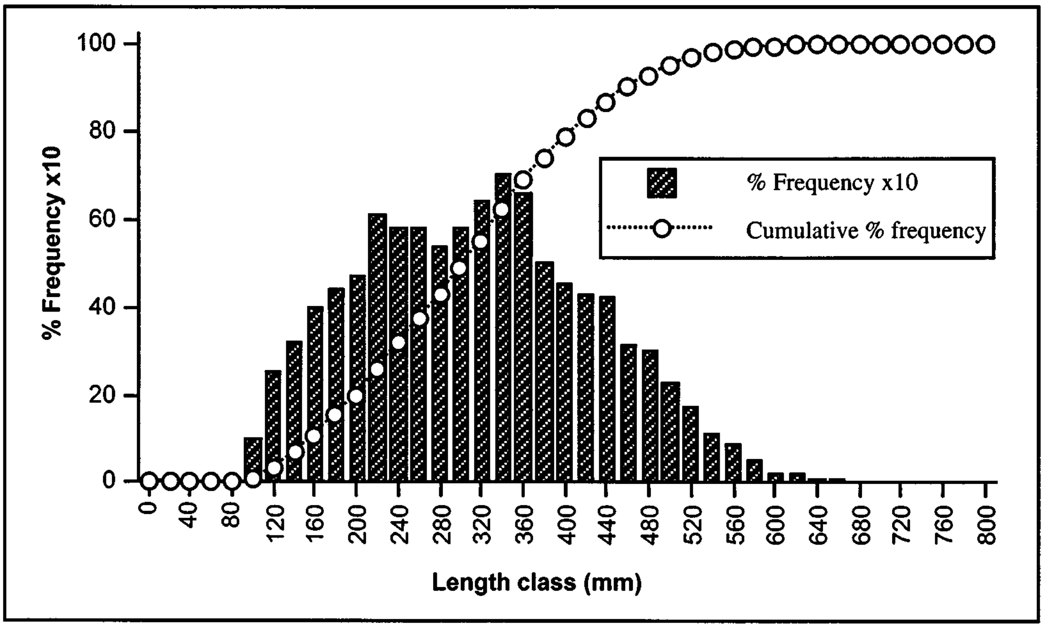


Figure C.1 Combined length frequency and combined cumulative length frequency plots for all eels captured during the 1999 eel surveys

At typical river sites, it therefore follows that detecting and measuring an overall change in absolute population density will be heavily dependent on detecting changes in the numbers of 200-400 mm eels. Even major changes in the population density of smaller eels could be expected to have a limited effect on overall population density (Figure C.1 suggests that even 5 years of zero glass eel recruitment to a system might typically reduce density by less than 20%). Such changes are likely to be masked by random spatial and temporal variance so that the density change is unlikely to be detected with an acceptable degree of statistical confidence (see Section C3.3 below).

The 200-400 mm size range might typically encompass eels in the 5-15 year age classes (Figures 3.17, 3.18, 3.19, 3.27 3.41, 3.42). The implication of this is that although population density is likely to be a very useful indicator of change on which to base management decisions, a reduction in population density due to recruitment failure (or over fishing of glass eels) may not become readily apparent or statistically confirmable for up to years after recruitment rates fall below critical levels. In contrast, however, a major increase in yellow eel exploitation in rivers could be expected to have a detectable effect on population density over a shorter time scale.

C2.3 Biomass

Ultimately, biomass is the only definitive measure for assessing whether a population is at or below carrying capacity. However, at high population densities, eels tend to mature and emigrate as males at a relatively small size, whereas at lower population densities, there is an increasing tendency to mature as older, much larger females. Thus potentially, high biomass levels could be maintained for many years following even a catastrophic decline in recruitment, with the population comprising ever larger but ever fewer individuals.

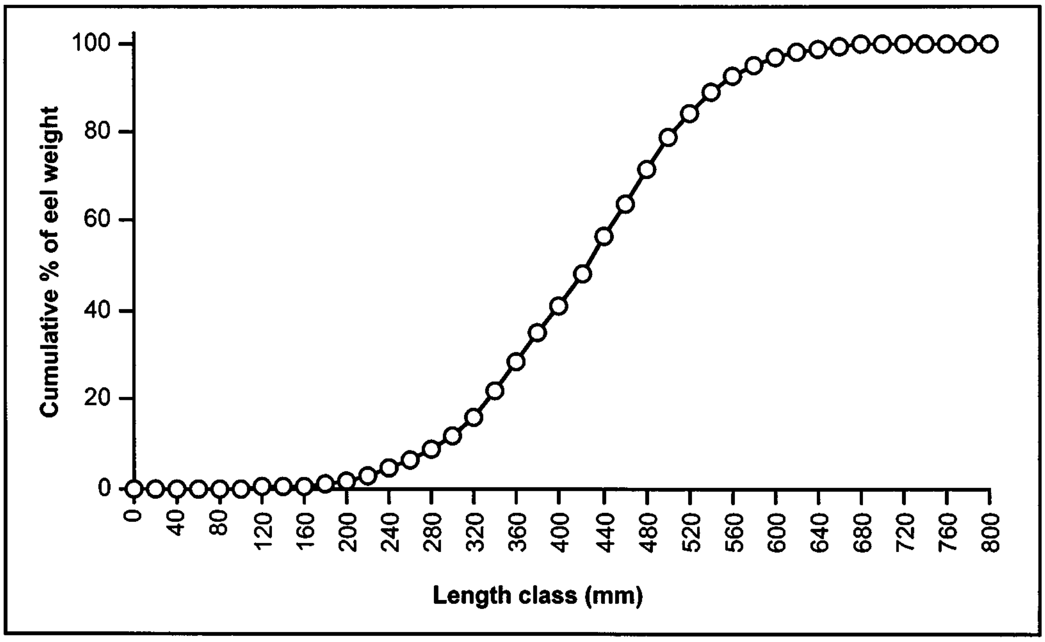


Figure C.2 Cumulative eel weight plotted against length frequency class based on all eels captured during the 1999 eel surveys

This point is illustrated by Figure C.2, which plots cumulative weight of eels against length frequency class for all eels captured in 1999. Whilst eels of >400 mm comprise only about 22% of the population density, they will typically comprise some 60% of the weight. (The relative contribution of >400 mm eels will be even higher at the lower population densities typical of more upstream sites.) Therefore a major drop in biomass would only occur when the number of large (>400 mm) eels begins to decline. In the early years following recruitment failure, the number of eels developing as females might actually increase so that biomass could go up rather than down, e.g. the glass eel recruitment and resulting eel stock model for the Loire developed by Feunteun (2000).

With typical female eels silvering and emigrating at an age of 15-20+ years, it is entirely possible that a readily measurable and statistically confirmable decline in biomass would not become apparent until 15 or more years after recruitment had fallen below the critical level. This suggests that for most rivers, biomass will not be a sensitive monitoring measure, other than in the very long term. However, in rivers which support yellow eel fisheries (largely confined to east coast rivers in Britain), major changes in exploitation rate could be expected to be reflected in changing biomass within a much shorter time scale.

C2.4 Age and growth rate

Changing recruitment rates and population density may be reflected in changes in population age structure or in average growth rate. Determining age and hence growth rate of eels is very time consuming and requires a very large sample because individual growth rates are highly variable and a typical population may contain 20 or more age classes. Thus the ability to detect the relatively small changes in growth rate that might be expected to result from changes in population density within a given river system will be limited. These difficulties will be compounded by the fact that otolith interpretation is distinctly subjective (see discussion in Section 3.11.3) and this subjectivity is likely to lead to at least some degree of operator bias. The consequent risk when comparing age/growth rates determined by different operators (as would be likely to occur in a long term monitoring programme) is that apparent and statistically significant differences in growth rate may simply reflect subjective bias in ageing.

Whilst of obvious scientific interest, it is questionable whether age/growth rate data would be of practical monitoring or management value. Incorporating ageing into an eel stock monitoring programme would be unlikely to be a cost-effective deployment of resources. Resources would almost certainly be better allocated by increasing the number of survey sites or the frequency of surveys.

C2.5 Conclusions

The foregoing and largely theoretical discussion leads to four conclusions regarding eel population status indicators and the timescales over which changes in these indicators might be expected to become detectable. These conclusions relate specifically to quantitative electric fishing surveys of rivers in England and Wales and not to stillwaters or estuaries or other capture methods.

- The most sensitive indicator of recruitment failure or over fishing of glass eels operating over the shortest time scale is likely to be length frequency, specifically a change in the relative proportion of smaller eels (e.g.<200 mm) in the population.
- Population density (particularly of eels in the 200-400 mm size range) will be a sensitive indicator, although readily detectable changes in population density resulting from a change in recruitment rate may not become readily apparent for up to 10 years. However, the effects of changing of yellow eel exploitation pressure might be detectable over shorter time scales
- Biomass is likely to be a relatively insensitive indicator with detectable changes possibly not occurring for up to 15 years or more following a change in recruitment rate. The effects of changing of yellow eel exploitation pressure should be detectable over much shorter time scales
- Measurement of age/growth rate is of doubtful value for monitoring.

These conclusions are generally born out if the comparison of the historic and 1998/99 resurvey data for the Severn, Dee and Piddle/Frome (Chapter 3) is considered in the light of declining glass eel recruitment during the 1980s and subsequently low recruitment through the 1990s, i.e..

- There appear to have been major changes in the proportion of small eels in the Piddle and Frome, clearly detectable changes in the Severn and possible changes in the Dee.

- There is fairly convincing evidence that overall population density has declined (possibly substantially) in the Piddle and Frome, but no evidence of change in the Severn or Dee.
- There is some suggestion that there may have been a decline in biomass in the Piddle and Frome but this is unclear. There is no evidence for a change in biomass in the Severn or Dee.
- Age determination has required a major resource input but has provided no direct information concerning possible changes in eel stocks.

It must be recognised that although the proportion of small eels in the population (at low to mid catchment sites) is likely to provide the best early warning of potential population change, this is only of actual significance if it subsequently leads to changes in overall population density or biomass. As minimum recruitment requirements are effectively unknown, it will not be possible to set target levels or action thresholds in relation to the numbers or proportion of the younger age classes in the population. Reference levels and management targets can be more readily set for population density (particularly eels >200 mm) and biomass, so these two population measures should form the basis for quantitative monitoring and subsequent stock management action. Measurement of all three (LCF, density and biomass) will provide the best information for management.

The likely time scales over which population changes might be expected to become apparent suggest that monitoring surveys should be conducted at a maximum interval of five years. This is a biologically based, rather than statistically based conclusion.

C.3 Survey Design and Sample Size

Several guides and software packages are available for estimating sample size in order to meet stated survey objectives e.g.

- "Sample Size and Sampling Frequency Estimator'', a software application developed by the U.S. Environmental Protection Agency Region 6 Watershed Management Section..
- Wyatt and Lacey 1994, Guidance notes on the design and analysis of fishery surveys, NRA R&D Note 292.
- Minitab's "Power and sample size calculator" (Minitab Inc)

In this case, the principle objective is to determine the number of sites required to detect a temporal change in eel population density (or biomass) and in particular, whether the level of change exceeds a predetermined management intervention threshold (or ideally, whether population density falls below an established reference or target level based on carrying capacity and/or favourable conservation status). The following calculations used Minitab's Power and sample size calculator.

The main difficulty is that real trends may be masked by natural variability, or apparent trends may be misleading (see Section 3.7.2 in relation to the 1998 and 1999 lower Severn surveys), if an insufficient number of samples are collected over time. Careful evaluation and planning is required to ensure that the monitoring programme will be sensitive enough to detect a desired magnitude of change within a specified period of time at a known level of confidence.

Some environmental attributes are extremely variable (as is clearly the case with eel population density) and large sample sizes may therefore be required to make estimates at an acceptably high level of confidence with a concomitant low probability of committing either a type I error or type II error:

- type I error is the risk of accepting a false positive or inferring a difference when none exist;
- type II error is the risk of failing to detect a difference when in fact the difference equals or is greater than the desired minimum detectable change;
- statistical power (1-b) is the probability of correctly rejecting the null hypothesis when it is false. Beta is the probability of making a type II statistical error.

The Severn data set provides information on both spatial and temporal variability and can therefore be used as a basis for estimating required sample size. Site density distribution for the lower Severn (1983, 1998 and 1999 data) sites are shown in Figure C.3. The data set is non-normally distributed, with a cluster of sites with very low population densities giving rise to a bimodal density distribution. This undoubtedly arises because a number of sites are above migration barriers, with the consequence that eel population densities are atypical of the normal range.

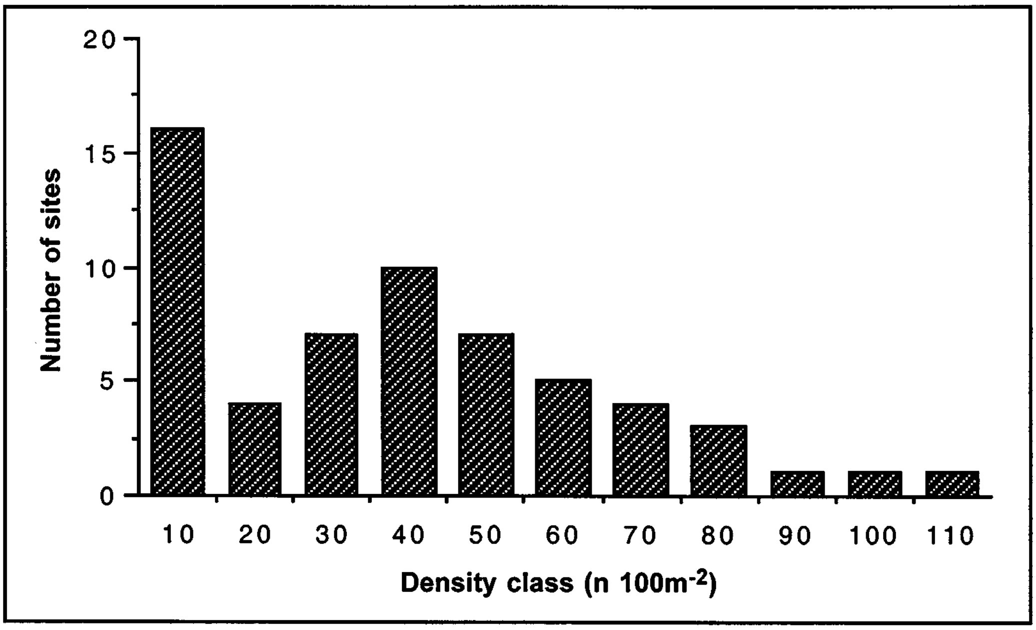


Figure C.3 Site density distribution for the lower Severn (1983, 1998 and 1999 data combined)

In order to take account of both spatial and temporal variance within the Severn data set, the following was undertaken:

i. The data set was reduced to the 16 sites common to all three survey years (1983, 1998, 1999). These are sites 1, 5, 7, 10, 11, 13, 21, 22, 23, 24, 25, 26, 28, 29, 41, 42. This has the added advantage of eliminating those very low density 'atypical' sites which are generally above migration barriers.

ii. For each temporal comparison, (83 v 98, 83 v 99 and 98 v 99) the data was plotted as in the example in Figure C4, i.e. the difference between the two surveys is plotted against the mean of the two surveys. The marked heteroscedasticity (fan shaped effect) indicates that high density sites tend to show a greater temporal variance than low density sites. Log_{10} transforming the data as in Figure C5 eliminates this effect by reducing the temporal variance of the high density sites.

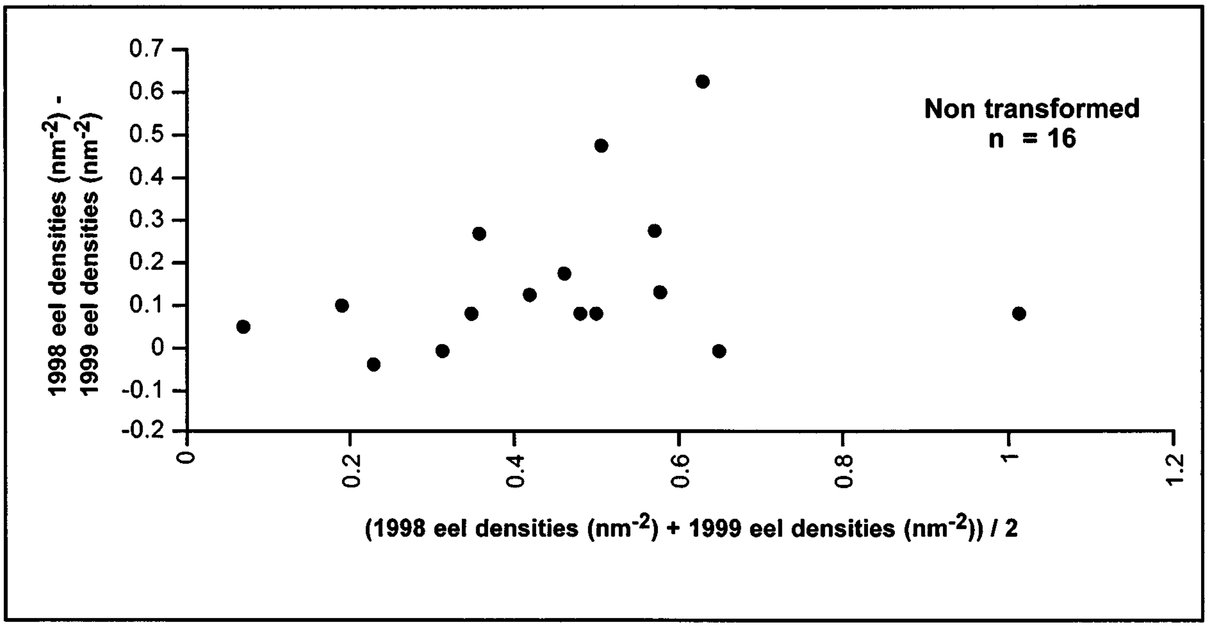


Figure C4 **1998 and 1999 lower Severn density data - the difference between the two surveys plotted against the mean of the two surveys.**

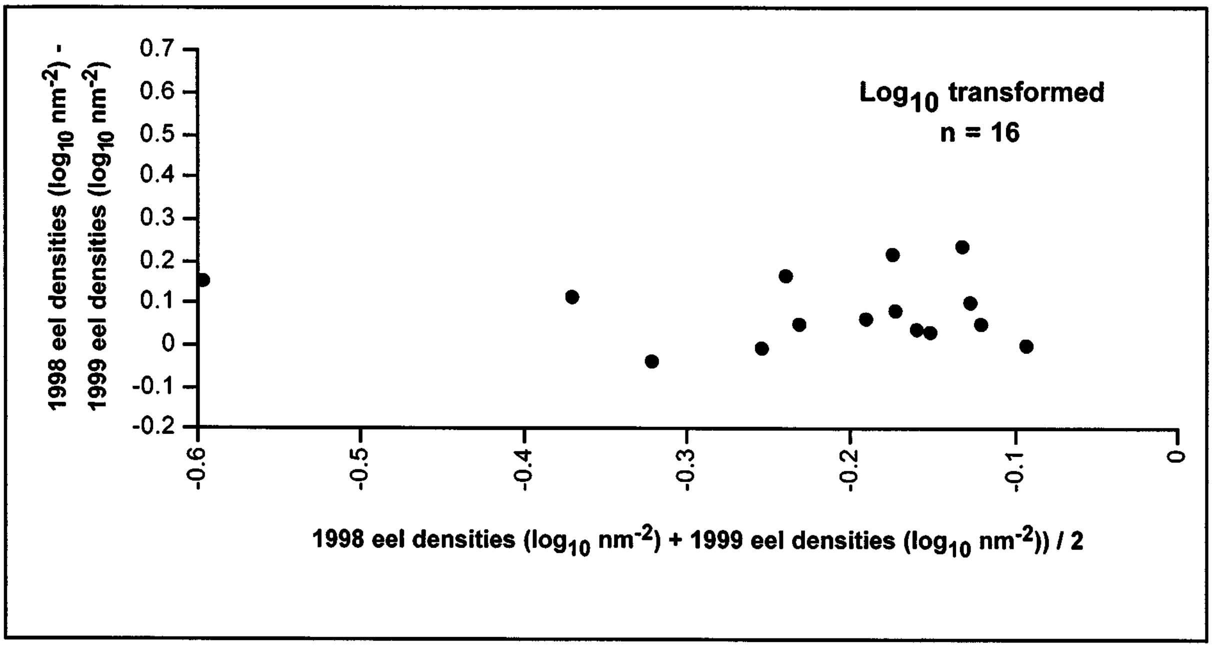


Figure C4 **1998 and 1999 lower Severn log_{10} density data - the difference between the two surveys plotted against the mean of the two surveys.**

iii. Using a paired one-tailed t-test, the difference between the two surveys is representative of any percentage change. The mean differences for $\log_{10}$ density for 1983-98, 1983/99 and 1998/99 were calculated and pooled standard deviation was derived. (in this case pooled SD = 0.348)

iv. Using Minitab Release 12.1, power and sample size calculator, sample sizes were calculated using a one sample t-test, one tailed alpha = 0.05, sigma = 0.348 and the difference to be observed ranging from 10-80%. Power levels (1-b) were 0.99. 0.95. 0.9, 0.85, 0.8, 0.7. The resulting sample sizes are shown graphically in Figure C6. This exercise was repeated for alpha values of 0.05, 0.10, 0.15, 0.20 and 0.25 and 1-beta values of 0.95, 0.90, 0.85, 0.8 and 0.7 for a percentage population change of 25%, 30%, 35% and 50%. The results are tabulated in Table 1a-c.

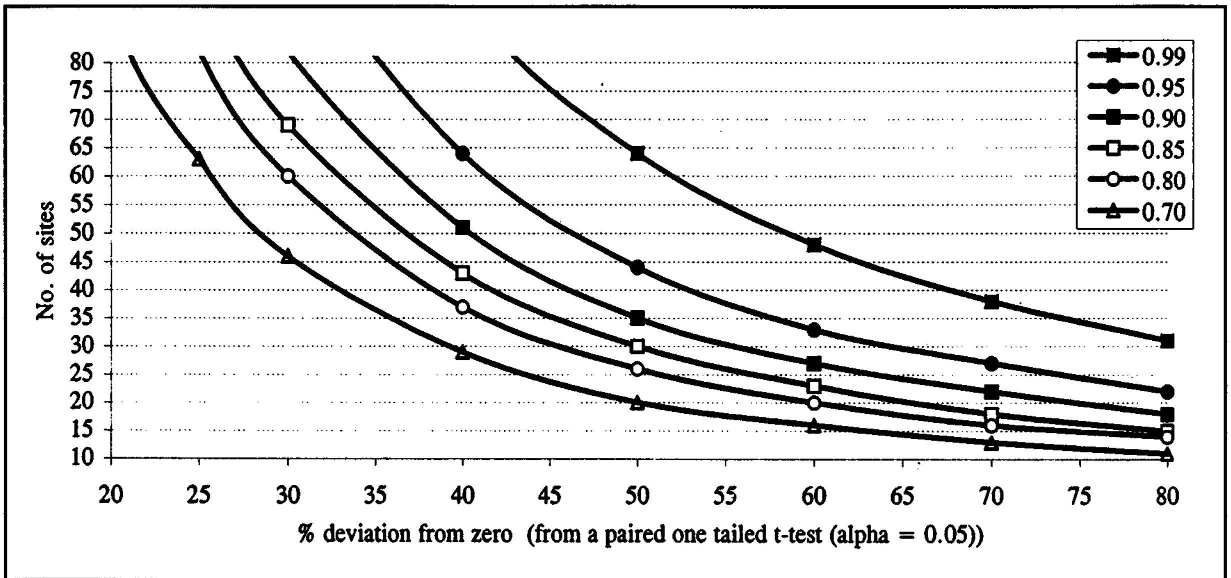


Figure C5 **Required number of sites for detecting different levels of difference in population density (% deviation from zero) in the lower Severn calculated using a one sample t-test, one tailed alpha = 0.05, sigma = 0.348 and 1-β of 0.99. 0.95. 0.9, 0.85, 0.8 and 0.7.**

As indicated in Table C1, the probable number of sample sites required to reliably detect a change in population density between repeat surveys is large. To detect a 50% change in density between two consecutive surveys with what might typically be regarded as acceptable levels of statistical confidence ($\alpha = 0.05$) and power ($1-\beta = 0.8$) would require in the order of 26 sites. Smaller changes in density, e.g. the proposed 'management intervention threshold' of -25% (Section 7.7, Recommendations 4.and 5) might be detected but with a correspondingly reduced level of certainty.

It should be noted that the foregoing projection relates only to the lower Severn (based on Zones 1, 2, 3 and 4 data from 1983, 1998 and 1999). In the absence of contra-indications, the most straightforward presumption is to assume that the survey site requirement for other river systems is likely to be broadly similar. However, the indications are that at least for the Piddle/Frome system, both temporal (Tadnoll Brook surveys 1973-84, Mann & Blackburn, 1991) and spatial (R&D surveys 1999) variance may be lower than that for the Severn, which would enhance the ability to detect change.

Table C1 Required number of sites for detecting various levels of difference in population density (% deviation from zero) in the lower Severn calculated using a one sample t-test, one tailed alpha and sigma = 0.348.

Table 1a 25% change

	Power (1-β)				
alpha (α)	0.95	0.90	0.85	0.80	0.70
α = 0.05	142	113	95	82	63
a = 0.10	112	86	71	60	44
a = 0.15	94	71	57	47	33
a = 0.20	81	59	47	38	25
a = 0.25	70	50	39	30	19

Table 1b 30% change

	Power (1-β)				
alpha (α)	0.95	0.90	0.85	0.80	0.70
α = 0.05	103	82	69	60	46
a = 0.10	82	63	52	44	32
a = 0.15	68	51	41	34	24
a = 0.20	59	43	34	27	18
a = 0.25	51	37	28	22	14

Table 1c 35% change

	Power (1-β)				
alpha (α)	0.95	0.90	0.85	0.80	0.70
α = 0.05	79	63	53	46	36
a = 0.10	63	48	40	34	25
a = 0.15	53	40	32	26	19
a = 0.20	45	33	26	21	14
a = 0.25	39	28	22	17	11

Table 1d 50% change

	Power (1-β)				
alpha (α)	0.95	0.90	0.85	0.80	0.70
α = 0.05	44	35	q30	26	20
a = 0.10	35	27	22	19	14
a = 0.15	29	22	18	15	11
a = 0.20	25	19	15	12	8
a = 0.25	22	16	12	10	6

The foregoing preliminary analysis suggests that the probable minimum number of eel monitoring sites for a river system will be in the order of 25. However, it is clear that a single repeat survey based on 25 sites may not be adequate to produce statistically robust evidence that the proposed -25% 'management intervention threshold' has been breached. Furthermore, the lower Severn data (1998 and 1999) implies that normal inter-year variation (even when sampling dates are closely matched) may exceed the proposed threshold. It will therefore be essential, should a single survey indicate that thresholds may have been breached, that the river system is resurveyed the following year in order to provide confirmatory evidence and to enhance statistical robustness. The inclusion of at least four river systems within a national programme will also increase the statistical sensitivity for detecting national trends

C.4 Summary of Key Points and Recommendations for Monitoring Programme Design

On the basis of the foregoing discussion and analysis, it is recommended that an eel stock monitoring programme based on electric fishing of selected river sites in primary monitoring catchments should measure three key population attributes:

- i length frequency of the population, as this is likely to provide the most sensitive early warning of potential or impending change in stock status due to recruitment failure or over fishing of glass eels;
- ii population density as the primary direct measure of yellow eel stock status (and potential silver eel escapement) - density will be a sensitive indicator of change in the medium term and, unlike length class frequency, it can be used in relation to management reference points, targets and thresholds;
- iii biomass (which follows automatically from length frequency and density) although it is likely to be a relatively insensitive indicator of change except in the very long term.

Age determination is unlikely to be a cost-effective option and cannot be recommended for routine monitoring purposes.

It is recommended that eel surveys should be conducted at maximum intervals of five years. As discussed above, this is a biologically based recommendation and matches the probable time scales (circa 5 years length frequency, up to 10 years for density and up to 15 years for biomass) over which population response to changing glass eel recruitment rates or over fishing can be expected to become readily detectable.

It is clear that management decisions require quantitative data with a known and acceptable level of certainty. Acceptable levels of statistical confidence and power for stock assessments of eels will require a relatively large number of sites per monitored river system. The required number of sites will need to be independently established for each catchment on the basis of temporal and spatial variance in population density, but is unlikely to be less than 25 sites per catchment. Resource constraints are thus likely to limit eel-specific stock monitoring to a small number of catchments.

It is recommended that at least 4 key catchments should be selected for long term eel stock monitoring. These primary monitoring catchments should include:

1. the lower Severn (focusing on Zones 1-3 and 5);

2. the Welsh Dee;
3. the linked Piddle and Frome system;
4. an eastern catchment such as the Darent, Mole or Essex Stour.

The Severn, Dee and Piddle/Frome are clear primary choices because of the quality and temporal span of the existing data sets. The Severn is also important because of the major estuarine glass eel fisheries, whilst the Dee serves as a control river with minimal fishing pressure. The inclusion of at least one eastern catchment is deemed essential as eastern catchments are likely to be more sensitive to fluctuations in glass eel recruitment as they are more remote from the main glass eel bearing currents. These recommended catchments, together with recommendations for second and third priority rivers, are discussed further in Section 8.4.

Eels are highly adaptable in terms of their requirements and they are thus typically very well dispersed throughout all accessible parts of the catchment . Almost any site within 40-50 km of the tidal limit that offers a reasonable mix of habitat and that is not upstream of major migration barriers is likely to be more or less representative in terms of population density. It is therefore recommended that survey sites are selected on this basis, in order to maximise the likelihood of population densities being normally distributed and spatial variance falling within reasonable limits.

An important consequence of adopting this approach to site selection is that whilst it maximises the potential for obtaining good quantitative population density data, it also effectively precludes the collection of data on distribution through the catchment. Contraction or extension of distribution range within a catchment is a potentially useful indicator of declining or expanding eel stocks.

Monitoring expansion or contraction of range requires monitoring towards and beyond the existing colonisation limit in the upper parts of a catchment, i.e. a significant number of currently eel-negative sites must be included within the site set. Population densities will inevitably be low, precluding meaningful population density estimates. Where densities are very low, it becomes a matter of chance whether a given site yields 0, 1 or 2 eels on any given sampling occasion. In effect, only presence/absence data can be obtained and a very large number of sites would be required to determine whether an apparent change in range was statistically significant. Using dedicated eel surveys to establish colonisation range within a large catchment would therefore be very unlikely to represent a cost effective deployment of resources.

It is therefore recommended that within the selected primary eel monitoring catchments, all multi-species fish surveys should record eels, even if only at presence/absence level and that the resulting data should be used to monitor eel distribution. Juvenile salmonid surveys in particular often cover a good range of middle and upper catchment sites and therefore fit well with the spread required for monitoring eel distribution. It is not anticipated that such surveys would provide robust quantitative data but they could clearly provide very effective distribution data for minimal resource input.

If results from the five yearly single-species eel surveys are indicative of a eel stock decline, i.e. they breach the predetermined alert (trigger) level, then survey frequency should be increased to annually, in order to increase the statistical probability of correctly identifying whether management intervention levels are breached. Eel survey frequency should also be increased to annually if routine multi-species surveys or commercial fishery data are suggestive of declining stocks.

Finally, consideration must be given to survey timing. It is evident that eels vary in their distribution on a seasonal basis and that as a result their local abundance (and perhaps also catchability) may change through the year. Good examples of this phenomenon are provided in Section 2.8 and Figure 2.13 for the St Neot River (Fowey catchment), in Section 2.22.2 and Figure 2.35 in relation to the West Thurrock Power Station screen catches from the Thames Estuary and in seasonal catches from the East Stoke Mill Stream on the Dorset Stour (Ibbotson et al 1994).

Therefore, in order to minimise the effects of seasonal variations in distribution (or catchability), it is essential that survey dates are consistent and strictly adhered to. However, the 1998 and 1999 re-survey data for the lower Severn (Section 3.7.2) strongly suggest that there may also be inter-year (possibly temperature driven) variations in distribution. If this is a commonplace occurrence, then it substantially increases the risk of drawing a statistically validated but none-the-less erroneous conclusion, e.g. that the population has declined when it is simply differently dispersed. This again emphasises the need to increase survey frequency in the event of a suspected population decline.

C5 Organisational Management and Resource Implications

Section C4 above (Section 8.11, Recommendations 7 - 9 in main report) recommends that eel specific-electric fishing surveys should be conducted on a minimum of four key catchments (Severn, Dee, Piddle/Frome and an eastern river) employing a minimum of 25 sites per catchment and a maximum sampling interval of five years. This section examines the management, implementation and resource implications of the proposed monitoring surveys.

C5.1 Organisation, management and funding

The four proposed primary catchments for yellow eel stock monitoring fall within four Agency Regions. Furthermore, these monitoring surveys comprise a part of the overall integrated Management (Chapter 7, main report) and Monitoring (Chapter 8 main report) strategies. It is therefore regarded as essential that both the overall programme and its component parts are co-ordinated and managed centrally (Section 8.11, Recommendations 1 - 8). It is further recommended that the eel-specific resurveys in the four primary catchments are funded centrally (Section 8.11 Recommendation 7) in order to ensure their long term viability. Data assessment and integration (from all strands of the overall programme) must also be undertaken as a central activity if a comprehensive national picture of stock status and trends is to emerge. This will also allow both data quality and the appropriateness of the monitoring programme to be kept under review and for major management decisions to be taken in the light of national and European trends and policies.

C5.2 Programme and logistics

It is recommended that a rolling programme of catchment surveys (one catchment per year) should be established rather than surveying all four primary catchments in the same year. The following programme is recommended.

Year 1 (2001) Eastern catchment - select catchment, identify and survey 25 sites to establish base-line data set.

Year 2 (2002) River Dee - expand base line data set by identifying and surveying a network of 25 sites on tributaries below LLangollen (to include sites 1-4 from the 1984 and 1999 surveys.

Year 3 (2003) River Severn - establish and survey a network of 25 sites from Zones 1, 2, 3 and 5 (in order to maximise long term continuity sites should be based as far as possible on sites surveyed in 1998/9 and/or 1983 but should exclude sites above known migration barriers and other 'difficult' sites such as Site.14.

Year 4 (2004) Piddle/Frome - establish and survey a network of 25 sites (in order to maximise both continuity and representativeness, the following sites are recommended:

9 sites surveyed by NRA in 1990 (also R&D sites 1999);
9 lower Piddle R&D sites surveyed in 1999;
2 Tadnoll Brook sites (1973-84 FBA study zone - also R&D sites 1999;
2 additional Piddle sites (near Affpuddle and near Puddletown);
3 additional Frome sites (upstream of Dorchester).

Year 5 (2005) Either: Resurvey eastern catchment to improve baseline data (the eastern catchment will have the least long term data);
Or: Resurvey one of Dee, Severn or Piddle/Frome if any of these were suggestive of change in years 2-4.

Years 6-10 Resurvey the four catchments , one per year in an appropriate sequence (dependant on year 5). This provides one year in hand to resurvey any individual catchment that may have shown unexpected trends or anomalies.

A rolling programme, such as that outlined above, is recommended for two key reasons. Firstly, it spreads resource demand uniformly across the years. Secondly, it provides a balanced data stream, thus enhancing the prospect of identifying putative trends at an early stage.

C5.3 Resource Implications

Based on experience gained during the R&D resurvey programme, a practised four person team can comfortably survey two or three sites using three electric fishing runs per day. The actual number of sites depends on travel time, ease of access and fishing, number of eels caught and water clearing time following sediment disturbance. Thus for 25 sites at an average of 2.5 sites per day, some 40 person days would be required for field work. Three to four days should be allowed for equipment preparation, site reconnaissance, establishing access arrangements etc. A further five to six days should be allowed for initial data entry and analysis and compilation to a standard format for transmission to the central project co-ordinator.

An annual resource allocation for the proposed electric fishing surveys of the four primary monitoring catchments should therefore provide for:

- approximately 50 person days of staff time;
- additional direct and indirect costs such as travel and subsistence, equipment provision and administration.

C.6 Methodological Recommendations for Optimising the Accuracy and Precision of Electric Fishing Surveys of River Eels.

Section C.4 above has made recommendations for monitoring programme design. In the light of experience gained during the re-survey programme, a number of methodological recommendations can be made concerning the conduct of electric fishing surveys of river eels.

C.6.1 Electric fishing method

It is essential that single-species eel surveys are undertaken as multi-species surveys are likely to underestimate eel populations by a factor of 3-5 (Section 3.11.2). Efficient eel capture demands the following:

- relatively high voltage and current settings in order to ensure an adequate attraction range and more than transitory stunning of eels;
- slow progress through the channel and a long dwell period adjacent to likely eel haunts because eels are often drawn very slowly from their refuges;
- anodes must be kept energised, even if stunned fish of other species are in the electrode vicinity, because any eels affected by the electric field but not yet fully stunned execute a very rapid recovery and escape;
- surveys should only be undertaken under conditions of optimum water clarity and a high level of concentration is required because eels tend to be stunned on the river bed and can be substantially more difficult to spot than other fish species (this is especially so for small eels);
- surveys should be based on 3 fishing runs, or 4 if depletions are erratic - although in practice, 2 runs may give adequate data on many occasions, eels often give erratic depletions and 2 runs do not permit adequate checks on data quality.

C.6.2 Site selection

In addition to the site selection criteria given in Section C.4, ease of sampling is a paramount consideration if good quantitative data are to be obtained. Ideal sampling sites are those on relatively small tributary streams of 3-6 metres width, with pools not exceeding 1 m in depth and a mixture of instream habitat including suitable refuges.

C.6.3 Electrical configuration of fishing equipment

The re-survey employed 50 hz pulsed dc at a current setting of approximately 5 amps through 2 anodes. This was effective at drawing and holding eels (lower settings were less effective) but this can be harsh treatment for more sensitive fish species if they remain in close proximity to an anode for more than a few seconds. A very useful small research project would therefore be to determine an optimum electric fishing configuration for eels, e.g. smooth or pulsed dc, pulse width, voltage, current etc., that maximises eel capture efficiency whilst minimising damage to other fish species.

C.6.4 Measuring and weighing eels

It is essential that eels are anaesthetised prior to measuring and weighing. Benzocaine (ethyl 4-aminobenzoate) was found to be more effective that 2-phenoxyethanol for this purpose. The following procedure was employed during the R&D resurvey programme.

The benzocaine was prepared as a stock solution in acetone (non toxic and water miscible) at approximately 50 gl^{-1}. At the end of each fishing run, the eel catch was transferred to a plastic dustbin containing 10-15 litres of water (the small volume and high sides prevented the escape of even the most determined eels) and a few drops of benzocaine solution were added. The required quantity of benzocaine appeared to vary and was determined by trial and error. An ideal concentration induces narcosis in the majority of eels within about 10 minutes. At this concentration, eels recover rapidly on transfer to an aerated recovery tank after at least 40 minutes continuous exposure to the anaesthetic. This allowed adequate time for a four person team to measure and weigh every individual from even a large first run catch of 200+ eels using one eel deliverer, two measurer/weighers, and one recorder.

C.5.5 Use of stop nets

As discussed in Section 3.11.1, the use of stop nets is of doubtful benefit for eel surveys. Furthermore, the deployment of stop nets can add considerably to on-site time, particularly in channels from which the turbidity from top net setting is slow to clear. It is therefore recommended that consideration should be given to the abandonment of stop nets for future eel surveys, or at least the abandonment of the top stop net. The use of additional fishing runs or the inclusion of additional sites would represent a more cost effective deployment of resources.

C.5.6 Collection of habitat data

At an individual site level, eel population densities can be heavily influenced by the availability of suitable daytime refuges. It is therefore strongly suggested that basic habitat data should be collected for each site on each survey occasion. The habitat survey procedure employed for the R&D eel resurveys provides an appropriate model. Local habitat characteristics can change substantially over a period of 5-10 years. If habitat information is available, then cases where local eel population changes may simply reflect changes in habitat quality can be readily identified and, if appropriate, such sites can be excluded from temporal comparisons.