THE UNIVERSITY OF HULL

SEA TROUT (*Salmo trutta* L.) STUDIES OF THE RIVER TYWI, SOUTH WALES

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Dafydd Martin Evans B.Sc.

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SEA TROUT STUDIES OF THE RIVER TYWI, SOUTH WALES

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Summary

The biological characteristics of the adult sea trout stock in the River Tywi were examined between 1988 and 1992 by sampling catches from a temporary main-river trap, and the commercial seine, commercial coracle and recreational rod fisheries. Radio tracking of adults during the spawning season investigated behavioural activities and their possible effects on stock characteristics. Catch effort data from angler log books assessed the annual performance of the rod fishery between 1990 and 1993. The effects of ameliorating the impact of acidification on water quality, benthic macro-invertebrates and juvenile salmonids, through reservoir liming, was studied.

During the five-year period 2880 sea trout were sampled. Large fish run early in April, May and June, with whitling appearing from June onwards. A late run of large fish occurs in October-December. One to four-year-old smolts were recorded, with an overall mean smolt age (MSA) of 2.24. Annual MSA declined over the study period and was related to the water temperature regime in the two years prior to smolting. This stock displayed fast growth rates and longevity, with sea trout surviving up to six spawning migrations. Adults were predominantly female but the sex ratio varied between stock components.

Female sea trout spawned over 1-3 days, with a preference for spawning during darkness, before leaving rapidly as kelts. Dominance in males was size related; males spawned with more than one female resulting in longer residence times.

Angler log books indicated diurnal variation in sea trout catch rates. River flow was linked to catch and effort for salmon, but a similar relationship was less evident for sea trout; angling method was suggested as a contributing factor.

Liming ameliorated the acidity and high trace metal levels observed below the reservoir prior to liming, resulting in improvements in the numbers of acid-sensitive macro-invertebrates and juvenile salmonids, especially salmon.

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CHAPTER 1

INTRODUCTION

1.1 Introduction

Sea trout (*Salmo trutta* L.) is an important fish species in the United Kingdom, sustaining substantial recreational and commercial fisheries. It is a particularly important salmonid in Wales since it contributes equally with salmon to the estimated economic value (£12 m) of fishing rights (excluding the River Wye) for migratory salmonids (Cresswell, 1989). The reported Welsh rod catch of sea trout is 50% greater than for England, similar to that of Ireland and only 40% less than that of Scotland (Cresswell, 1989). When the geographical area of the countries is considered the importance of sea trout in Wales becomes clear.

The River Tywi, with a mean annual declared rod catch of 5194 fish (1987-1991), supports the most prolific recreational sea trout fishery in Wales and England. For this five-year period the declared rod catches from the River Tywi represented 26.2% of the declared Welsh catch, and 15.8% of the declared catch of England and Wales combined. The River Tywi also supports commercial estuary netting with 12 coracle net licences in the upper estuary and 9 seine net licences in the lower estuary. The mean annual declared commercial catch (1987-1991) was 1563 sea trout.

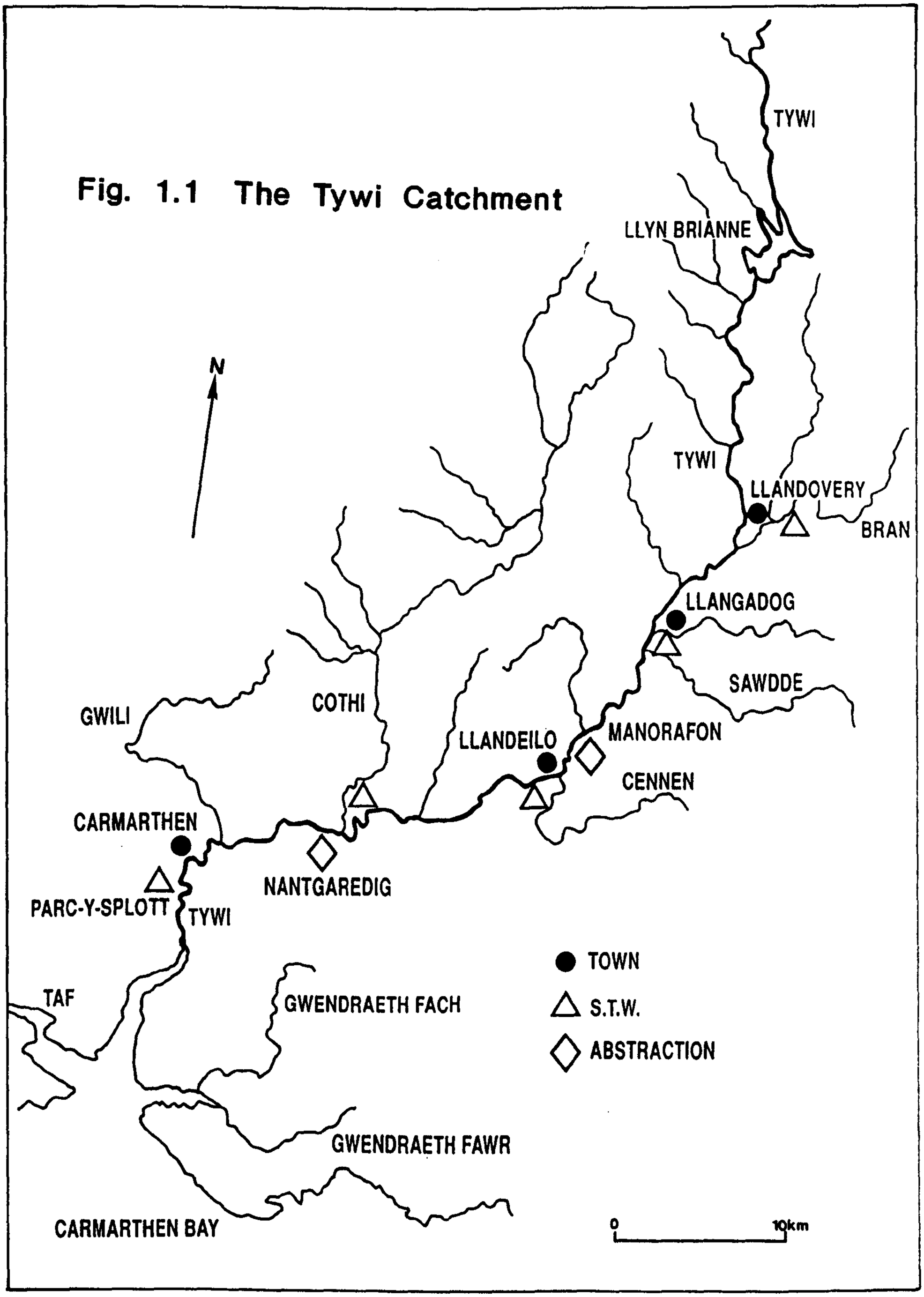
Historically the management of the Atlantic salmon (*Salmo salar* L.) has received considerable research, but increased recognition of the importance of sea trout in recent decades, together with well publicised concerns in Ireland (Anon., 1990,1991), have highlighted inadequacies in the information available to manage the sea trout stocks. Compared to the salmon, the polymorphic nature of the anadromous and resident forms of sea trout complicates investigations of its life history strategy and reproductive cycle (LeCren, 1985). The longevity and multiple spawning of sea trout adds further complexity to the management of stocks.

1.2 Study site

1.2.1 The River Tywi

The River Tywi (Fig. 1.1) rises at an altitude of 425 m AOD in an afforested and moorland area of mid-Wales. It is 111 km in total length and has a catchment area of some 1376 km^2. The headwaters have been regulated since 1972 by a major reservoir, Llyn Brianne, which augments

Fig. 1.1 The Tywi Catchment



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flow for potable abstraction at Nantgaredig (64 km downstream of the impoundment) and also at times of high demand at Manorafon (39 km downstream). Major tributaries include the Rivers Gwili, Cothi, Sawdde and Llandovery Bran. Water quality is generally good (NWC Class 1A), apart from the upper reaches which suffer from acidification (see Section 1.2.3 and Chapter 5).

Average daily flow for the River Tywi is 45 $m^3 s^{-1}$, although flows as low as 2 $m^3 s^{-1}$ have been recorded at Nantgaredig (D. Mee pers.comm.). The River Tywi flows into Carmarthen Bay, which is shared with the Rivers Taf and Gwendraeth. At its widest point the estuary is 2.5 km wide but narrows rapidly upstream. The tidal cycle is asymmetric, with a prolonged ebb and shortened flood period. Tidal range is large at the estuary mouth mean spring 6.6 m, neap 3.7 m, decreasing to 2.6 m and 0.4 m for spring and neap tides, respectively, at Carmarthen which is 16 km inland.

Most of the catchment has shaly rock of the Silurian and Ordovician age (Howells & Jones, 1972; WWA, 1986). In the upland areas in the North the soils are typically podzols with brown podzolic soils on the floors of the valleys and ferric stagnopodzols on the higher ground. The low lying areas have predominantly brown earths and stagnogley soils, with alluvial soils close to the river.

Mean annual rainfall in the catchment varies from 2400 mmy^{-1} on the Black Mountains in the east to <1200 mmy^{-1} in the estuary, with a mean of 1550 mmy^{-1} for the whole catchment (NRA, 1994).

The catchment is predominantly rural with sheep farming and large expanses of conifer plantations to the North and on higher ground. The middle reaches have mixed dairy and livestock rearing whilst intensive dairy farming is practised in the lowlands.

This rural predominance of the catchment results in a sparse population (c. 41,600), with a large proportion (47%) centred around the towns of Carmarthen, Llandeilo and Llandovery (NRA, op cit).

1.2.2 Llyn Brianne reservoir

Llyn Brianne was built after authorisation by the West Glamorgan Water (Llyn Brianne) Order 1968. The reservoir was filled in 1972 and

thereafter abstraction at Nantgaredig supplied potable water to parts of South Wales, including Llanelli and Bridgend.

The reservoir operates on a hypolimnial release regime (discharging to the river via an aerator system), with an overspill capacity. It maintains a minimum compensation flow in the River Tywi below the impoundment of 0.787 $m^3 s^{-1}$, for abstraction at Nantgaredig. The dam wall is 88.5 m high and the reservoir has a gross capacity of 62,140 Ml, and covers an area of 210 hectares.

As the reservoir was constructed in the headwaters of the River Tywi, a fisheries protection scheme was initiated (Howells & Jones, 1972). This consisted of trapping adults below the reservoir and transporting them by road before releasing them to spawn upstream of the reservoir. Two smolt traps were constructed on the major tributaries above the reservoir to trap downstream migrating smolts. The smolts were also to be transported by road to below the impoundment. In the 1970s and early 1980s production evident from the smolt trap results was poor and provoked questions regarding the water quality of the upper river (Gee, 1990).

1.2.3 Water quality of the upper catchment

Water quality investigations from 1981 onwards demonstrated the presence and detrimental effect of acidification in the upper catchment around Llyn Brianne (Stoner *et al.*, 1984). In 1985 the river above Llandovery was downgraded to NWC Class 2, on the basis of low pH (WWA, 1986).

Acidification of the surface water in the upper catchment, as throughout many areas of upland Wales, is influenced by geology, soils and land use (Hornung *et al.*, 1990). Much of the Tywi catchment has shaly rock of the Silurian and Ordovician age (Howells & Jones, 1972; WWA, 1986) with low or no buffering capacity. In the upper catchment the overlying soils also lead to a base deficiency providing poor neutralising capacity.

Whilst some agricultural land use may increase the neutralising capacity of soils due to lime addition (Hornung *et al.*, 1990), other land uses such as conifer afforestation may further increase acidity (e.g. Bird *et al.* 1990) and mobilise toxic metals such as aluminium. Much of the area surrounding Llyn Brianne reservoir has been afforested with conifers since the late 1950s.

Fig. 1.2 illustrates the predicted occurrence of acid-sensitive waters from agricultural land use, soils and geology (Hornung *et al.* 1990). The source of the River Tywi and its uppermost tributaries are clearly located within the area of mid-Wales typically affected by acidification.

1.3 Salmonid fisheries

1.3.1 History of licensing and regulation

Licensing of netsmen and anglers was introduced to the River Tywi area after 1867, when a Board of Conservators was assembled to attempt to reduce the alleged over exploitation of the salmonid stocks (Stewart, 1979). This semi-autonomous board became the River Tywi Fishery District which remained responsible for fisheries until the introduction of the 1948 River Board Act which instigated the South West Wales River Board in 1951. This amounted to an amalgamation of the River Tywi and River Teifi Fishery Districts. A further act of parliament in 1963 saw the introduction of the South West Wales River Authority which remained responsible for fisheries until the Welsh Water Authority was formed in 1974. The National Rivers Authority (Welsh Region) was introduced in 1989 and is currently responsible for fisheries regulation.

Assessing historical catch records from the above authorities is fraught with uncertainty and unreliability due to factors such as incomplete records, changes in data collection methods, reporting procedures and variation in fishing effort. Variation in fishing effort alone may cause differences in catches. Small (1991) described how angling effort has changed on the Rivers Wye and Dee since 1867. He showed a gradual increase in effort between 1867 and 1950 but thereafter a rapid increase until about 1980. Since 1980 no discernable trends were apparent. Incomplete records (South West Wales River Authority, 1974) suggest that a similar pattern in angling effort may have been followed on the River Tywi.

1.3.2 Current River Tywi Fisheries

The migratory salmonid rod fisheries, i.e. salmon and sea trout, mainly extend from the upper estuary throughout the main river and also to the larger tributaries, the Rivers Cothi and Gwili. Some migratory salmonid

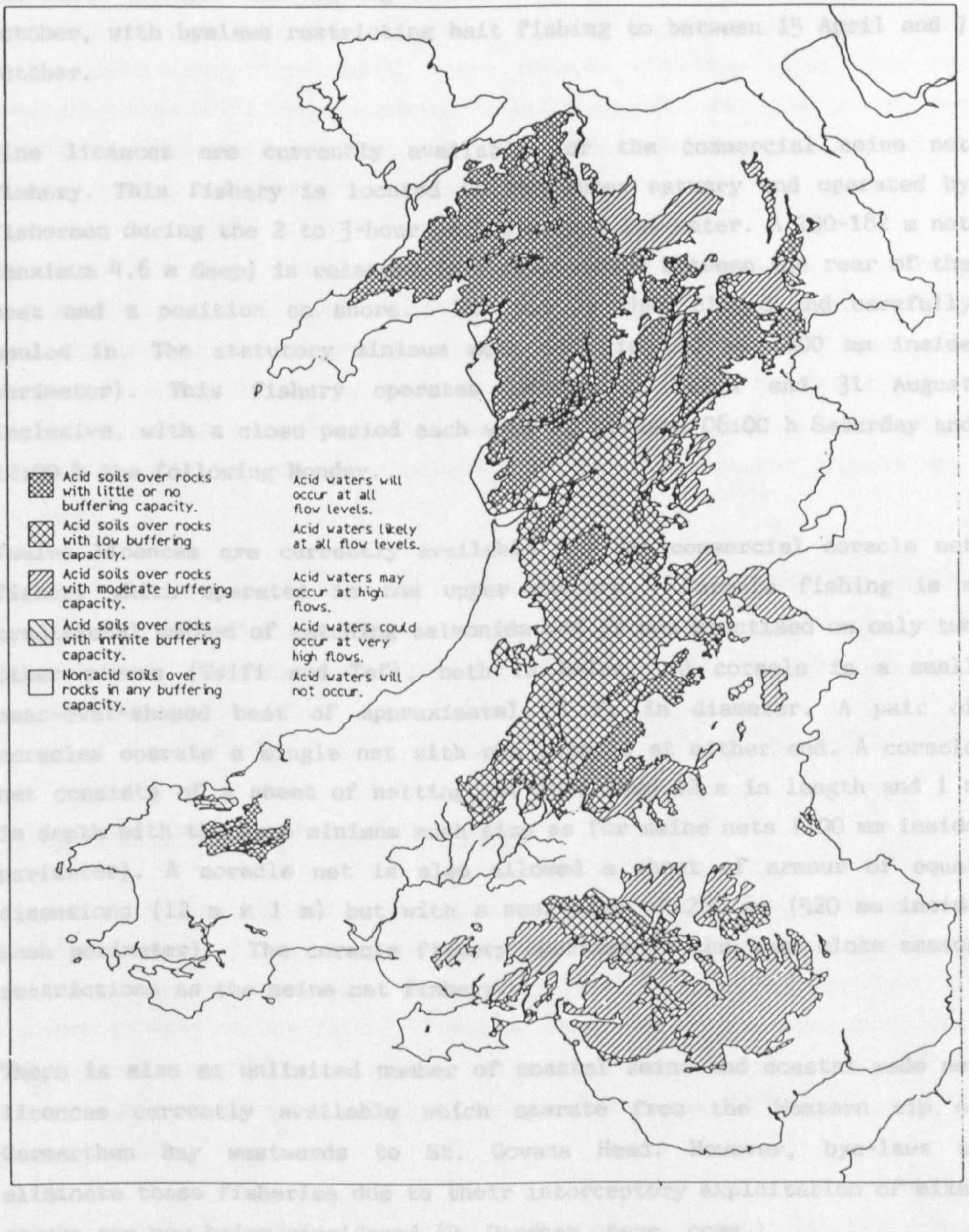


Fig. 1.2. The predicted occurrence of acid-sensitive waters in Wales from agricultural land use, soils and geology.

Source - Hornung et al. (1990).

fishing also takes place in smaller tributaries, viz. Llandovery Bran and the River Sawdde. The fishing season runs from the 20 March to the 17 October, with byelaws restricting bait fishing to between 15 April and 7 October.

Nine licences are currently available for the commercial seine net fishery. This fishery is located in the outer estuary and operated by fishermen during the 2 to 3-hour period around low water. A 100-182 m net (maximum 4.6 m deep) is released in a semi-circle between the rear of the boat and a position on shore. The net is then closed and carefully hauled in. The statutory minimum mesh size is 100 mm (200 mm inside perimeter). This fishery operates between 1 March and 31 August inclusive, with a close period each week of between 06:00 h Saturday and 12:00 h the following Monday.

Twelve licences are currently available for the commercial coracle net fishery which operates in the upper estuary. Coracle fishing is a traditional method of catching salmonids and is now practised on only two other rivers (Teifi and Taf), both in Wales. A coracle is a small near-oval-shaped boat of approximately 5 ft in diameter. A pair of coracles operate a single net with one coracle at either end. A coracle net consists of a sheet of netting not more than 12 m in length and 1 m in depth with the same minimum mesh size as for seine nets (200 mm inside perimeter). A coracle net is also allowed a sheet of armour of equal dimensions (12 m x 1 m) but with a mesh size of 260 mm (520 mm inside mesh perimeter). The coracle fishery operates to the same close season restrictions as the seine net fishery.

There is also an unlimited number of coastal seine and coastal wade net licences currently available which operate from the Western tip of Carmarthen Bay westwards to St. Govans Head. However, bye-laws to eliminate these fisheries due to their interceptory exploitation of mixed stocks are now being considered (D. Gardner, pers. comm.).

Carmarthen Bay is also the location of a drift-net fishery for sea bass (*Dicentrarchus labrax*). Whilst this fishery is prohibited from landing salmonids, they are captured as a 'by-catch' (Gardner, 1990). All salmonids captured should be released, but the opportunity for illegal take and the unlikely survivorship of meshed fish results in many of the salmonids caught being effectively lost to the in-river stock.

1.3.3 Catch statistics

Declared catches from the commercial fishery are available from 1956, whilst consistent recreational catch records from the River Tywi are available from 1972, when compulsory catch returns for sea trout were instigated. The sea trout declared catch (1956-1991) from the commercial and recreational fisheries are shown in Fig. 1.3.

The River Tywi is a productive river with declared rod catches typically between 3500 and 6000 fish. A decline in rod catches was associated with the drought year of 1976 and the following two years. The last five years have seen violent swings in declared rod catches from 9580 fish in 1987, the highest ever recorded annual catch, to 2243 sea trout in 1990. 1990 was associated with drought conditions and followed another drought year in 1989.

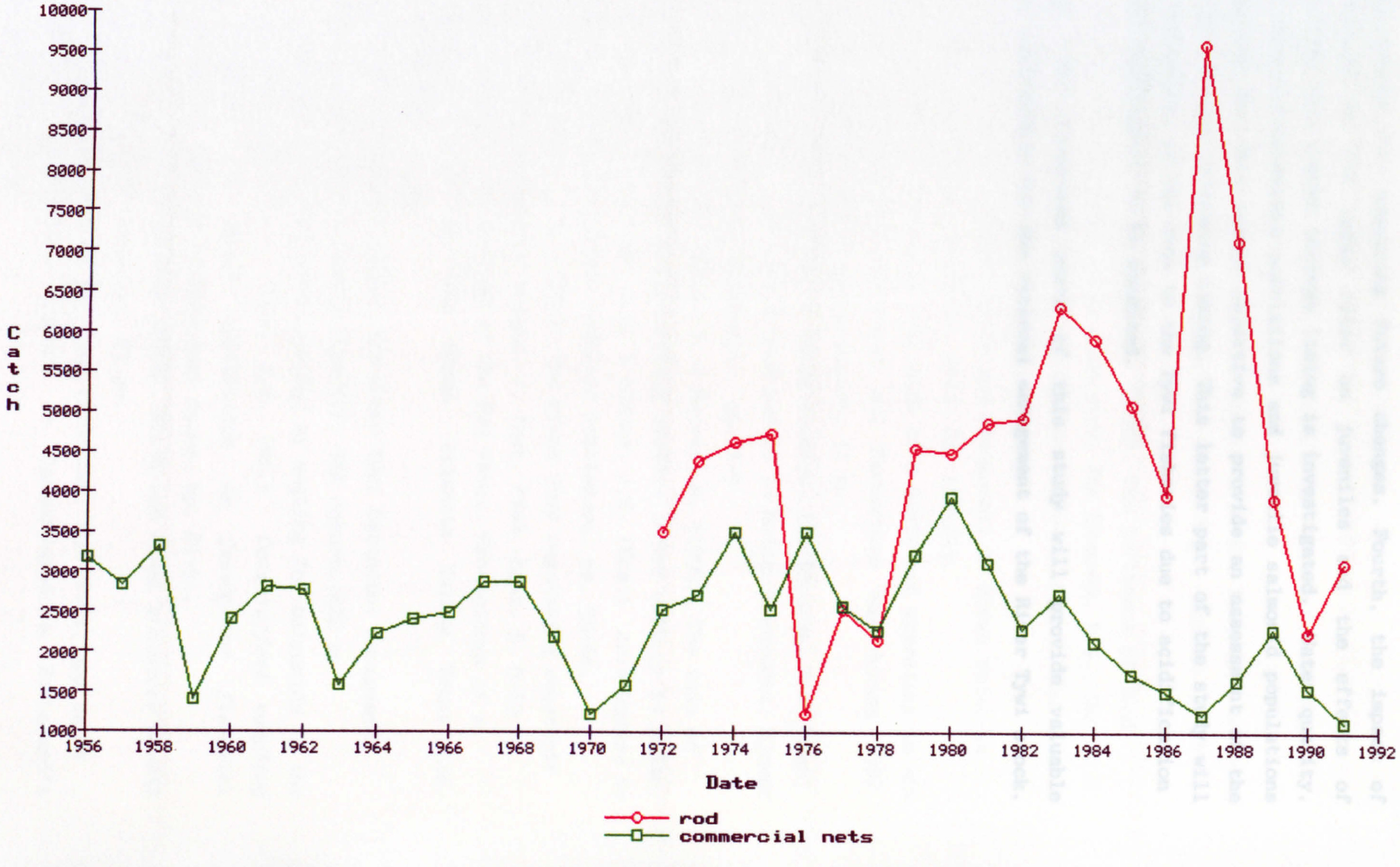
The declared commercial catches were typically 2000 to 3000 fish during the period prior to 1980, but an apparent downward trend with consistently lower catches has occurred since, only interrupted by a modest increase in 1989.

1.4 Aims and objectives of this study

This study was instigated as a CASE award between Hull University and the NRA (Welsh Region). It attempts to expand existing knowledge relating to the sea trout by describing the adult population of the River Tywi. It further focuses on one factor affecting salmonids in the catchment, viz. the deleterious effects of acidification in the upper main river, and the attempt by the NRA (Welsh Region) to ameliorate these through liming of Llyn Brianne.

These overall objectives are achieved by first describing the biological characteristics of the adult stock through sampling of the fisheries. Second, a behavioural study of adults during the spawning period is undertaken to investigate the possible implications on the biological characteristics of the stock. Third, the use of angler log books to assess the annual performance of the rod fishery and to determine any spatial differences within the catchment. This component of the study will provide a base-line understanding of the rod fishery and provide a

Fig. 1.3 Declared sea trout catches from the Tywi catchment



possible method for measuring future changes. Fourth, the impact of acidification in the upper river on juveniles and the effects of ameliorating this impact through liming is investigated. Water quality, benthic macro-invertebrate populations and juvenile salmonid populations are examined for this latter objective to provide an assessment of the ecological changes following liming. This latter part of the study will enable the extent of the loss to the Tywi fisheries due to acidification below the impoundment to be examined.

Together, the integrated parts of this study will provide valuable baseline information for the rational management of the River Tywi stock.

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CHAPTER 2

ADULT SEA TROUT BIOLOGICAL STUDIES

2.1. Introduction

Information on the biological characteristics of a sea trout stock enables rational management decisions to be made for that particular stock. Detailed understanding of factors such as run timing, growth and age class structure assist both quantitative and subjective decisions regarding the regulation of exploitation through close seasons and size limits. Biological characteristics may also describe the juvenile life history and this can assist in interpreting the early fresh-water phase of the life cycle.

The most common source of biological characteristics is derived from interpretation of the age and growth of fish from scales, which are related to length and weight. One of the earliest studies of sea trout was undertaken by Lamond (1916), but the extensive work of G.H.Nall during the 1930s is more widely recognised (Nall,1930; 1933; 1938). In Wales, the studies of G.S.Harris during the late 1960s and the early 1970s (Harris, 1970) are widely acknowledged. The River Tywi was one of the rivers studied by Harris during that period, and will thus provide comparative information for data collected during this investigation.

This study examined a large number of sea trout taken by various methods between 1988 and 1992. It is an opportunistic study gathering information from three sources. First, as a by-product of a radio tracking project (1988-1990; Clarke *et al.* 1990, 1991), large numbers of sea trout were available for examination from the lower estuary seine and jumper nets, and from a temporary main river trap. Second, data from a sample of fish caught by the upper estuary coracle fishery (1990-1992) allowed this method to be examined. Finally, information from anglers (1989-1992) enabled the recreational fishery to be evaluated.

The objectives of this chapter are twofold. First, to describe the biological characteristics of sea trout from the River Tywi for the period 1988-1992. Second, to distinguish the differences between catches of the various gears and any relative bias these may impose on population assessment.

2.2 Sampling sites

Fig. 2.1 shows the location of the lower estuary netting sites, the upper estuary coracle fishery and the location of the trapping site. The rod fishery extends from the upper estuary throughout the main river and in to the main tributaries, viz. Cothi and Gwili (see Fig. 1.1).

2.3 Materials and Methods

2.3.1 Capture Methods

Seine and Jumper nets

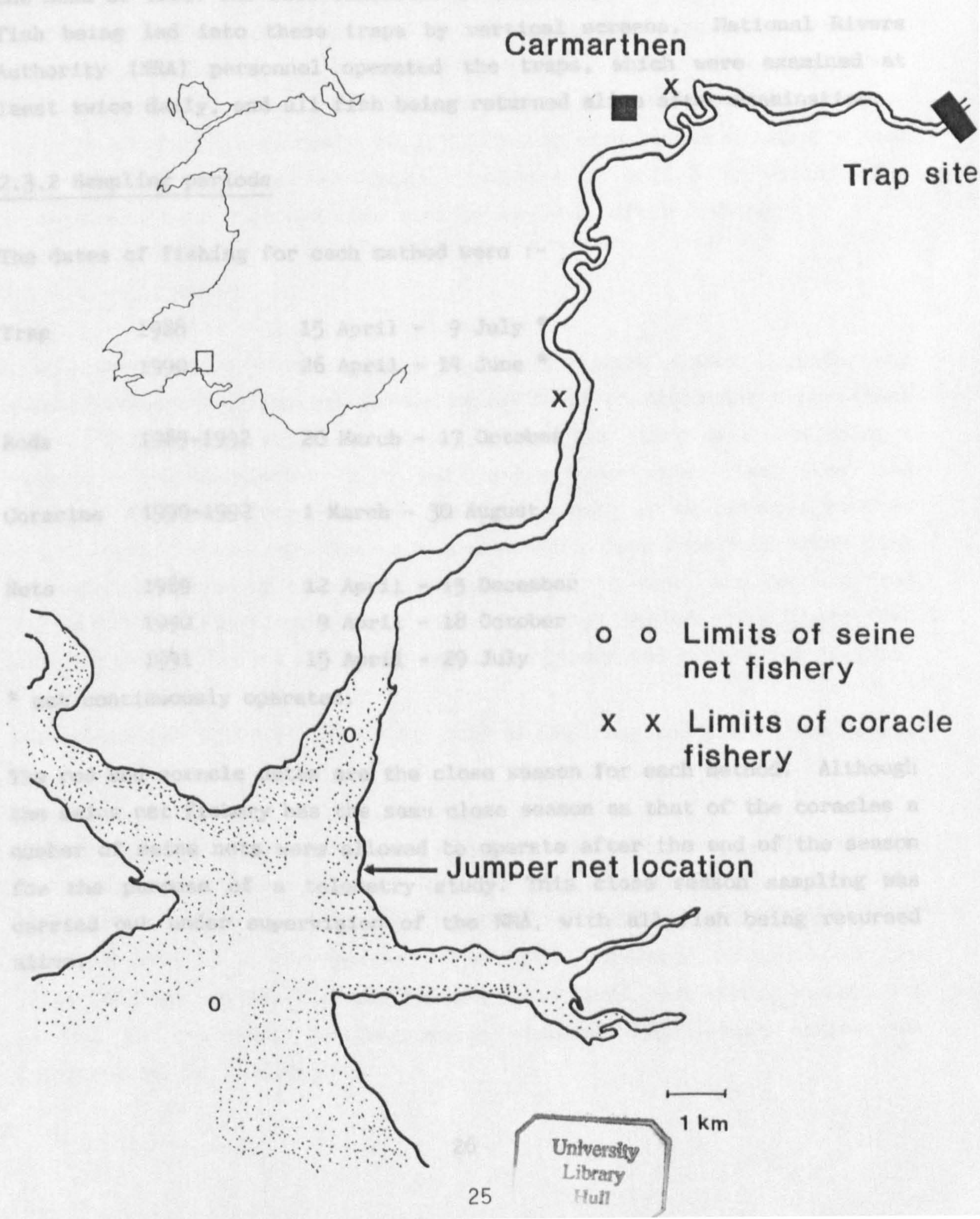
These methods, from here on denoted the 'Net' fishery, provided sea trout from the lower estuary commercial seine nets and the National Rivers Authority (NRA) operated jumper nets. Data from each method were pooled, since analysis showed no significant gear related differences in samples between the two operating methods (Mee *et al.*, 1990). During 1989 and 1990 these nets were allowed a 50-mm mesh size bunt for scientific purposes, but in 1991 a 102-mm (4 inch) mesh size was used (statutory minimum 100 mm).

The seine net fishery (method described in Chapter 1) was sampled in the 1989-1991 seasons. Jumper nets were used in 1989 and 1990. They consist of a triangular 'bag' net (8 m long x 2.5 m deep) fixed in position by ropes and scaffolding poles. A leading net (30-50 m long x 4 m deep) guides fish into the 'bag', from which they are unable to locate the exit. As the tide recedes the contents of the bag are then examined. The jumper nets were positioned such that the leading net was perpendicular to the tidal flow.

Coracles

Sea trout from the coracle fishery (method described in Chapter 1) were sampled by a local fishmonger (also one of the coracle fishermen) for the years 1990-1992. Random sub-sampling of the catches was achieved by sampling all the fish caught on 1-2 days of the week.

Fig. 2.1 The Tywi estuary showing fishery and sampling sites



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Rods

A number of anglers were asked to keep details of sea trout caught and provided scales for age determination. Details were provided from all legal methods, including fly, spinning and bait fishing.

Trap

The main river was trapped 19.8 km upstream from the river mouth, above the head of tide. The site consisted of three separate box traps with the fish being led into these traps by vertical screens. National Rivers Authority (NRA) personnel operated the traps, which were examined at least twice daily, and all fish being returned alive after examination.

2.3.2 Sampling periods

The dates of fishing for each method were :-

Trap	1988	15 April - 9 July *
	1990	26 April - 14 June *
Rods	1989-1992	20 March - 17 October
Coracles	1990-1992	1 March - 30 August
Nets	1989	12 April - 15 December
	1990	9 April - 18 October
	1991	15 April - 29 July

* not continuously operated.

The rod and coracle dates are the close season for each method. Although the seine net fishery has the same close season as that of the coracles a number of seine nets were allowed to operate after the end of the season for the purpose of a telemetry study. This close season sampling was carried out under supervision of the NRA, with all fish being returned alive.

2.3.3 Fish details

As well as a sample of scales for age determination the following parameters were recorded for sea trout sampled: date of capture, fork length (to the nearest 5 mm), weight (g), sex (gutted fish from the rod fishery). Not all parameters were collected for all methods of capture, notably the weights of fish caught in the trap and the nets during 1990 were not recorded.

In 1991 the girth measurements of a sub-sample of 100 fish from the seine net fishery were recorded using a tailor's measuring tape. The operculum girth (Fig. 2.2) and the maximum dorsal girth (anterior to the dorsal fin) were recorded as well as the girth of any mesh mark left by the net. The mesh size of 50 randomly selected meshes were measured using a push wedge (MAFF 90-130 mm net gauge) attached to a 2.5 kg weight. The measurements were recorded when the net was wet, after fishing.

2.3.4 Age and growth

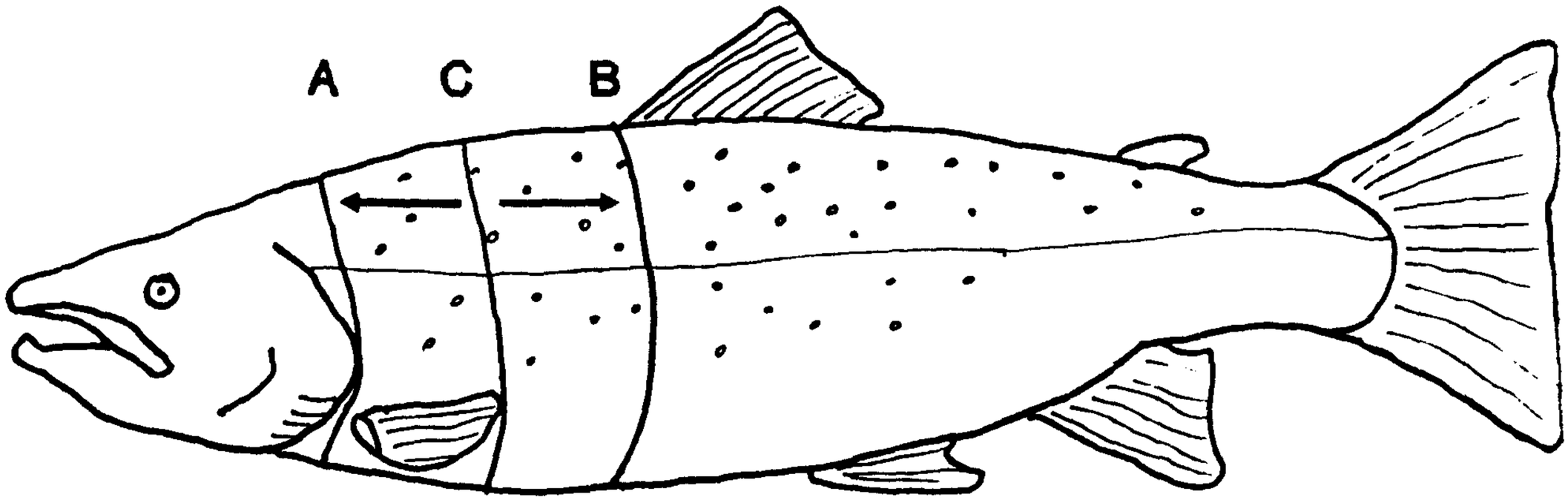
Scales that needed cleaning prior to reading were placed in water and rubbed between the fingers, before being dried on absorbent paper (Anon 1984). Scales from the first three years of the study were read using a Projectina microprojector (x30) and samples from latter years were read using an Olympus SZ40 microscope (x40). The sets of scales were read at least twice. Where possible, six scales were read from individual fish to evaluate constancy of ageing among scales. However, the samples from the radio-tracking project only contained three scales since these fish were returned alive to the water and undue damage and stress was avoided.

The convention of Nall (1930) was used to describe the age of sea trout, i.e.

X. Y+

where X refers to the smolt age (in years) and Y the post-smolt history. The + sign denotes a year of incomplete growth since the end of the previous winter. A spawning mark is shown by the letters SM. Thus a fish aged 2.1+2SM+ is a five-year-old fish which migrated to sea after two years in fresh water, did not spawn in its first post-smolt winter, but spawned in the next two successive winters. Replacement scales are described by the letter R.

Fig. 2.2 Positions of girth measurements taken from sea trout from the seine net fishery



A = Opercular girth

B = Maximum dorsal girth

C = Mesh mark girth

Although various names may be used locally for sea trout at different life stages the following are used here.

Whitling	Sea trout returning in the same year as their smolt migration. (.+)
Maidens	Sea trout with no apparent spawning mark on the scales but at least 1 post-smolt winter. (.1+,.2+)
Previous spawners	Sea trout with one or more spawning marks present on their scales. (.+SM+,.1+SM+,.1+3SM+)

Sub samples of scales from 20 fish were randomly picked from each annual sample of aged whitling and .1+ maidens. Previous spawners were not used due to inaccuracies from re-absorption of scale material. Scales were displayed using a Projectina microprojector (x30) and all measurements were taken along the anterior radius.

Length at the end of a freshwater year and at smolting were estimated by linear back-calculation of growth (Lea, 1910; Harris, 1970; L'Abee-Lund *et al.*, 1989; Okland *et al.*, 1993). The position of smolting was determined from the change in growth pattern on the scale (Jonsson, 1985).

Growth rate in the second year of life was used as an index of parr growth (Jonsson *et al.*, 1993). Analysis was undertaken separately for whitling and maidens owing to the possible change of the relationship between scale growth and fish growth with age (Carlander, 1981).

2.4 Results

Section A Biological characteristics

2.4.1 Data sources

In the five-year period, 2880 sea trout were examined. The number of fish sampled in each year from each method is shown in Table 2.1.

Table 2.1 The number of sea trout sampled in each year.

Year	Method				
	Trap	Net	Coracle	Rod	Total
1988	595	-	-	-	595
1989	-	438	-	275	713
1990	92	319	147	194	752
1991	-	317	115	160	592
1992	-	-	70	158	228
					2880

Replacement scales prevented the age determination of all fish recorded. From the total of 2880 sea trout sampled 2689 were assessed for sea age only and of these 1847 for both sea and river age.

2.4.2 Run timing

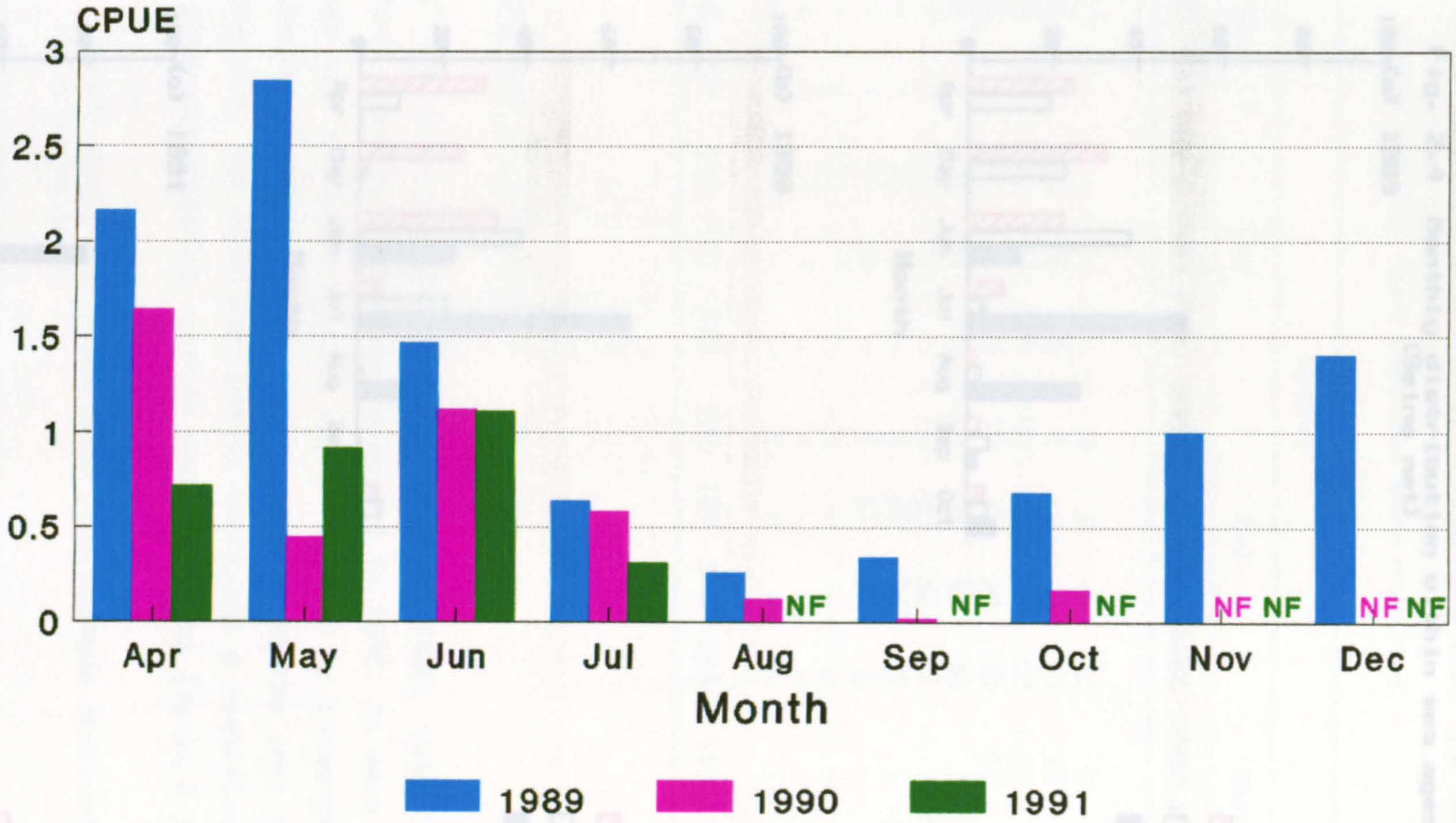
The majority of fish were caught between April and July although an experimental period of extended netting into November and December of 1989 showed that fresh fish entered during this period (Table 2.2).

However, total monthly catches mask the temporal variation in effort and therefore catch per unit effort data have been used to minimise this potential bias in effort. Catch per unit effort (CPUE) was available for the seine net fishery as catch of sea trout per net haul. The monthly CPUE for the seine net fishery for 1989 to 1991 (Fig. 2.3) demonstrated that the main run of sea trout was early, between April and June. However, the catches between October and December 1989 highlight that part of the run enters late in the year.

Analysis of the sea age distribution from the seine net fishery (Fig. 2.4a-c) revealed that many of the multiple spawners return early, followed by 1SW maidens, with the whitling group captured from June onwards. The high value for whitling in June in 1991 (Fig. 2.4c) is caused by the low number caught and the cessation of sampling in July.

Fig. 2.3 Seine net CPUE 1989-1991

CPUE (Catch per haul)



NF No Fishing

Fig. 2.4 Monthly distribution within sea ages (Seine net)

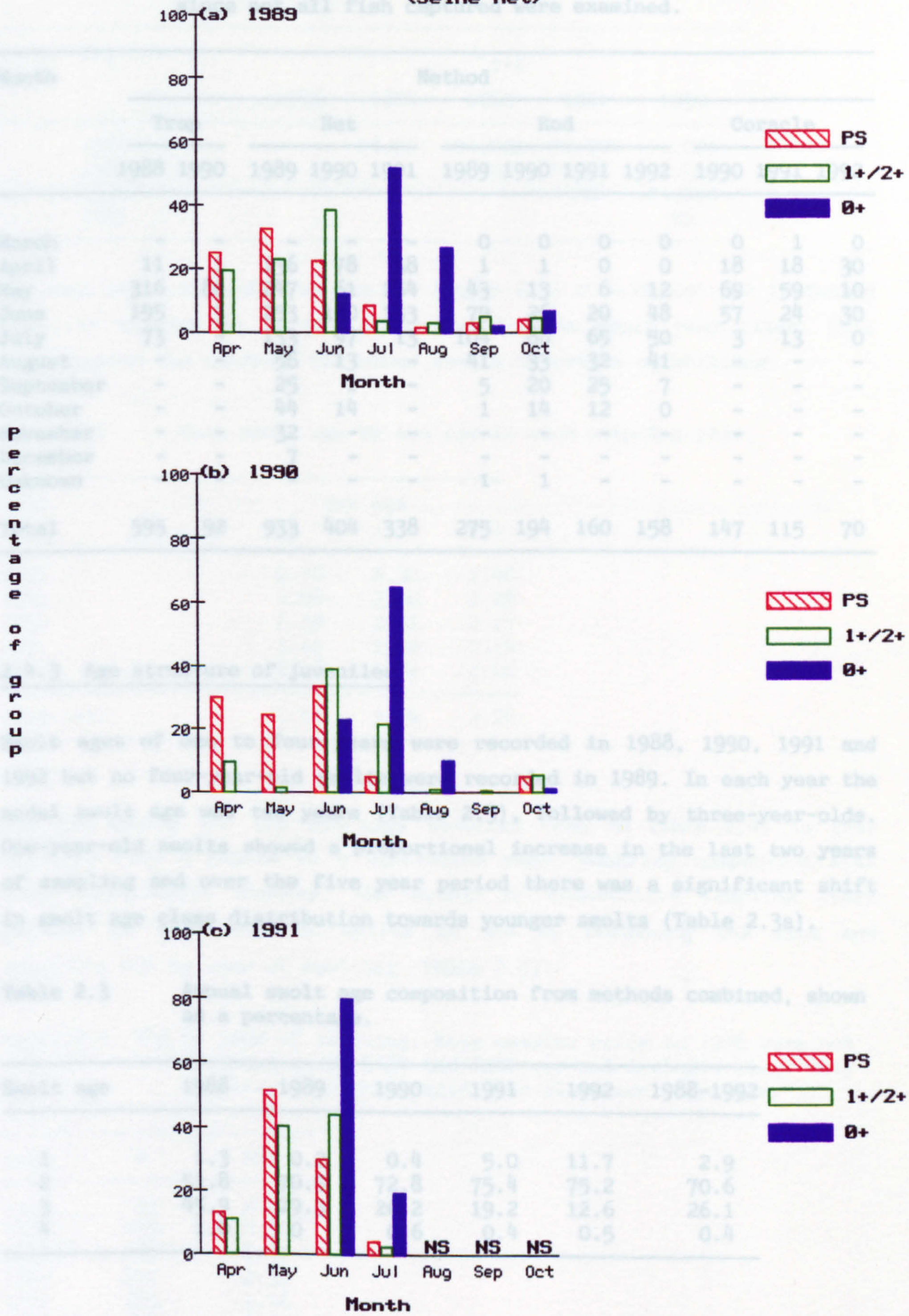


Table 2.2 Monthly distribution of catches by method (1988-1992). Note that the number of sea trout recorded from the Net fishery is larger than quoted in the sample size in section 2.4.1 since not all fish captured were examined.

Month	Method											
	Trap		Net			Rod				Coracle		
	1988	1990	1989	1990	1991	1989	1990	1991	1992	1990	1991	1992
March	-	-	-	-	-	0	0	0	0	0	1	0
April	11	1	136	78	38	1	1	0	0	18	18	30
May	316	86	247	61	154	43	13	6	12	69	59	10
June	195	5	253	140	133	79	26	20	48	57	24	30
July	73	-	133	97	13	104	86	65	50	3	13	0
August	-	-	56	13	-	41	33	32	41	-	-	-
September	-	-	25	1	-	5	20	25	7	-	-	-
October	-	-	44	14	-	1	14	12	0	-	-	-
November	-	-	32	-	-	-	-	-	-	-	-	-
December	-	-	7	-	-	-	-	-	-	-	-	-
Unknown	-	-	-	-	-	1	1	-	-	-	-	-
Total	595	92	933	404	338	275	194	160	158	147	115	70

2.4.3 Age structure of juveniles

Smolt ages of one to four years were recorded in 1988, 1990, 1991 and 1992 but no four-year-old smolts were recorded in 1989. In each year the modal smolt age was two years (Table 2.3), followed by three-year-olds. One-year-old smolts showed a proportional increase in the last two years of sampling and over the five year period there was a significant shift in smolt age class distribution towards younger smolts (Table 2.3a).

Table 2.3 Annual smolt age composition from methods combined, shown as a percentage.

Smolt age	1988	1989	1990	1991	1992	1988-1992
1	1.3	0.2	0.4	5.0	11.7	2.9
2	51.8	70.5	72.8	75.4	75.2	70.6
3	45.9	29.3	26.2	19.2	12.6	26.1
4	1.0	0	0.6	0.4	0.5	0.4

Table 2.3a Significance of Chi^2 test for changes in the relative percentage of 1/2 and 3/4 year-old smolts between years (from Table 2.3), using the hypothesis Ho of $\%OBSYr_i = \%EXPYr_{i+1}$ (1 df).

Yr_i	Yr_{i+1}				
	1988	1989	1990	1991	1992
1988		P<.001	P<.001	P<.001	P<.001
1989			NS	P<.05	P<.001
1990				NS	P<.001
1991					NS

The mean smolt age (MSA) by sea age (Table 2.4) showed that MSA increased from .1+ maidens to .+ sea trout for a given smolt year class. This demonstrated the tendency for older smolts to return as whitling.

Table 2.4 Mean smolt age by sea age in each sampling year.

Year	Sea age		
	.+	.1+	All
1988	2.76	2.30	2.46
1989	2.58	2.20	2.29
1990	2.39	2.21	2.27
1991	2.31	2.10	2.14
1992	2.18	1.91	2.02
1988-1992	2.45	2.14	2.24

The MSA quoted for the sea ages combined (ALL in Table 2.4) is that obtained from a sample in a particular year, enabling comparison with other literature. However, the number of consecutive sampling years permits an evaluation of shifts in MSA by combining the data and analysing MSA by year of smolting (Table 2.5).

Table 2.5 MSA by year of smolting. Note samples prior to 1986 were not included since N<20 and 1992 was not included since only whitling from that smolt year class are available.

Year	N	MSA
1986	93	2.28
1987	258	2.30
1988	485	2.27
1989	371	2.30
1990	388	2.16
1991	168	2.07

A decreasing MSA was observed for latter years (Tables 2.4 and 2.5). This was most apparent in the whitling group where a 21% reduction in MSA (2.76-2.18) was observed between 1988 and 1992.

2.4.4 Age structure of adults

The number of different sea age classes obtained were 19, 14, 15, 13 and 12 in 1988, 1989, 1990, 1991 and 1992, respectively. Combining the five years data gave 20 different sea ages. Due to the four different smolt age classes, from the overall scale samples for each year, 30, 23, 31, 25 and 22 different age categories were recorded in 1988, 1989, 1990, 1991 and 1992, respectively. Combining the five years of data gave 44 different age categories (Table 2.6).

Table 2.6 The different age classes recorded for the overall sample 1988-1992. (n=2689)

Sea age	River age			
	1	2	3	4
.+		*	*	*
.1+	*	*	*	*
.+SM+	*	*	*	
.2+	*	*	*	*
.1+SM+	*	*	*	
.+SM+1+		*	*	
.2SM+		*	*	
.2+SM+	*	*	*	
.1+2SM+		*	*	
.+3SM+		*	*	
.2+2SM+	*	*	*	
.1+3SM+		*	*	
.+4SM+		*	*	
.2+3SM+		*		
.1+4SM+		*	*	
.+5SM+		*		
.2+4SM+			*	
.1+5SM+		*	*	
.+6SM+			*	
.2+6SM+		*		

Sea age

Sea ages ranging from .+ to .2+6SM+ were observed during the 5 years (Table 2.6). The inclusion/exclusion of gears and the changes in selectivity of some gears makes it difficult to evaluate the relative abundances of different sea ages between years. This particularly affects the assessment of .+ sea trout not taken in the commercial fisheries. To offset this complication the composition of annual samples (Table 2.7) has two sets of results. Firstly the composition of sea ages from all the fish sampled and secondly (values in parenthesis) the composition from samples when all .+ fish were removed.

The dominant sea age in the samples was that of .1+ maidens (Table 2.7) followed by .+ sea trout and .1+SM+ fish. The .+SM+ class of 1988 was a strong group and also the .1+ class in 1989 (.1+SM+ in 1990).

Table 2.7 Percentage composition of samples by the major sea ages. Values in parenthesis represent composition with .+ removed.

Sea age	1988	1989	1990	1991	1992	1988-1992
.+	17.1	18.6	18.2	9.81	20.1	16.4
.1+	35.1(42.4)	50.2(61.6)	36.3(44.3)	52.3(57.9)	49.1(61.5)	43.8(52.4
.+SM+	12.1(14.6)	7.0(8.6)	2.7(3.4)	6.6(7.4)	6.7(8.4)	6.9(8.2
.1+SM+	15.7(18.9)	13.0(15.9)	19.5(23.9)	14.0(15.5)	5.8(7.3)	14.8(17.8
Others	20.0(24.1)	11.2(13.9)	23.3(28.4)	17.3(19.2)	18.3(22.8)	18.1(21.6

2.4.5 Maturation

The majority of sea trout sampled matured as .1+ maidens, followed by .+ sea trout (Table 2.8). Comparison of age at first maturity between years was affected by the strong sea age groups described above and the relative contribution of gears. The strong .+SM+ group in 1988 resulted in high values of maturation at .+ for 1988 and 1989 whilst the strong .1+ group in 1989 resulted in high maturity estimates at .1+ for 1990 and 1991. The high value of maturation at .+SM+ in 1992 (41.1%) is caused by a high relative contribution from the rods (see section 2.4.9)

Table 2.8 Age at first maturity (%) of previous spawning fish and the percentage of 1SW fish bearing spawning marks.

Sea age	1988	1989	1990	1991	1992	1988-1992
.+	36.8	37.1	19.5	31.8	41.1	30.9
.1+	58.9	58.5	69.3	64.7	55.4	62.9
.2+	4.3	4.4	11.2	3.5	3.5	6.2
% SM of 1SW	25.6	12.2	6.9	11.3	12.0	13.6

The majority of sea trout had spawned only once (Table 2.9) and the overall percentage of sea trout surviving three spawnings varied from 4.9% to 18.0%. This variation between years depended on individual year class strength, as described above for maturation.

Table 2.9 Frequency (%) of spawning marks from previous spawners and the percentage of previous spawners in each sample.

No. SM's	1988	1989	1990	1991	1992	1988-1992
1	62.4	74.3	61.0	62.6	51.8	63.7
2	19.6	20.8	25.9	20.2	32.2	22.5
>=3	18.0	4.9	13.0	17.2	16.0	13.8
% PS	45.5	28.6	42.3	34.6	25.0	36.7

2.4.6 Sex composition

The sex of sea trout sampled from the rod fishery was determined from gutted fish (Table 2.10) and displayed a predominance of females. The ratio of males to females by sea age group demonstrated an overall decline from .+ to .1+/.2+ sea trout to previous spawners. The overall annual sex composition will therefore depend on the relative composition of each sea age group.

Table 2.10. Ratio (R) and sample size (N) of males to females from the rod fishery. (Based on gutted fish)

Sea age		1989	1990	1991	1992	Overall
.+	R	2.22 : 1	1 : 1.38	1 : 1.14	1 : 1.36	1 : 1.10
	N	11 : 5	13: 18	14: 15	14: 19	52 :57
.1+/.2+	R	1 : 2.52	1 : 1.60	1 : 3.05	1 : 2.13	1 : 2.33
	N	31: 78	20: 32	11: 35	22: 51	84: 196
PS	R	1 : 4.29	1 : 1.78	1 : 2.78	1 : 3.17	1 : 2.74
	N	7 : 30	18: 32	8 : 26	6 : 19	39: 107
Overall	R	1 : 2.40	1 : 1.65	1 : 2.71	1 : 2.02	1 : 2.05
	N	47: 113	53: 87	34: 76	45: 91	179:367

2.4.7 Growth rates

Juveniles - Back calculation

Back calculation of length at age from whitling and .1+ maidens demonstrated that the mean length at smoltification increased significantly with age ($p<0.05$ Student's t-test; Figs. 2.5a & b), with faster growing juveniles smolting earlier.

Development rates of juveniles have been found to be highly dependent upon water temperature (Elliott, 1975; Crisp, 1992) thus a measure of temperature will help to explain any changes in growth rates. Mean monthly water temperatures (1984-1991) from Nantgaredig (see Fig. 1.1) were used as an index of water temperature in the catchment (Fig. 2.6); this demonstrated that the winter periods of 1988-1990 were extremely mild and that the summers of 1989-1990 were very hot.

Increased winter temperatures particularly affect the incubation rate of eggs (Crisp, 1992) and variation in summer temperatures affect the growth rate of the free-swimming stage and parr. To assess the effect of the warmer winters, each mean monthly temperature for the winter period (November to April inclusive) was summed to give a cumulative winter temperature and then plotted against the percentage of one year old smolts (incubated as eggs during that winter period) observed from the

Fig. 2.5a Whitling back-calculated lengths from 2 and 3 year old smolts. Error bars give the ranges observed.

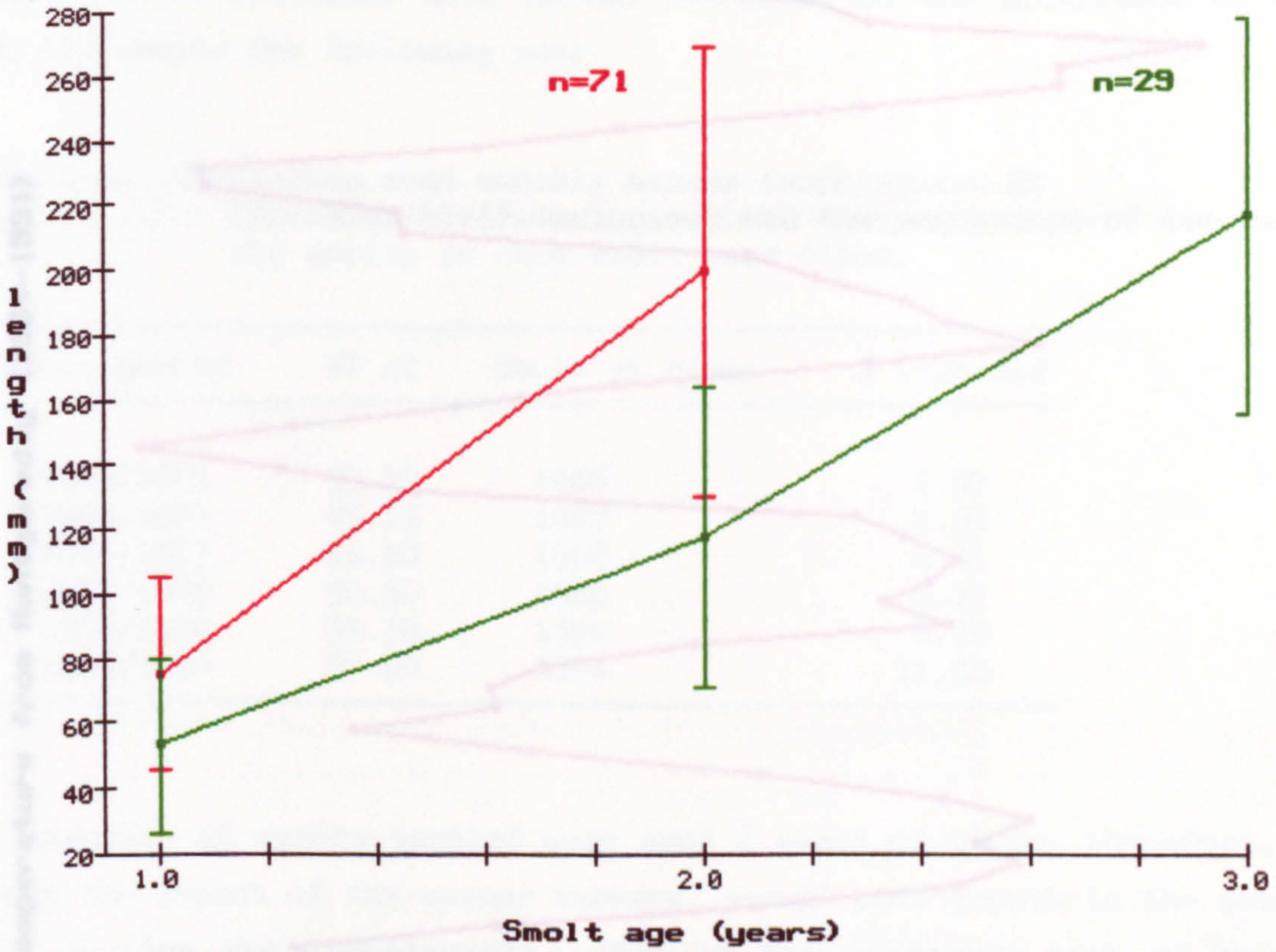
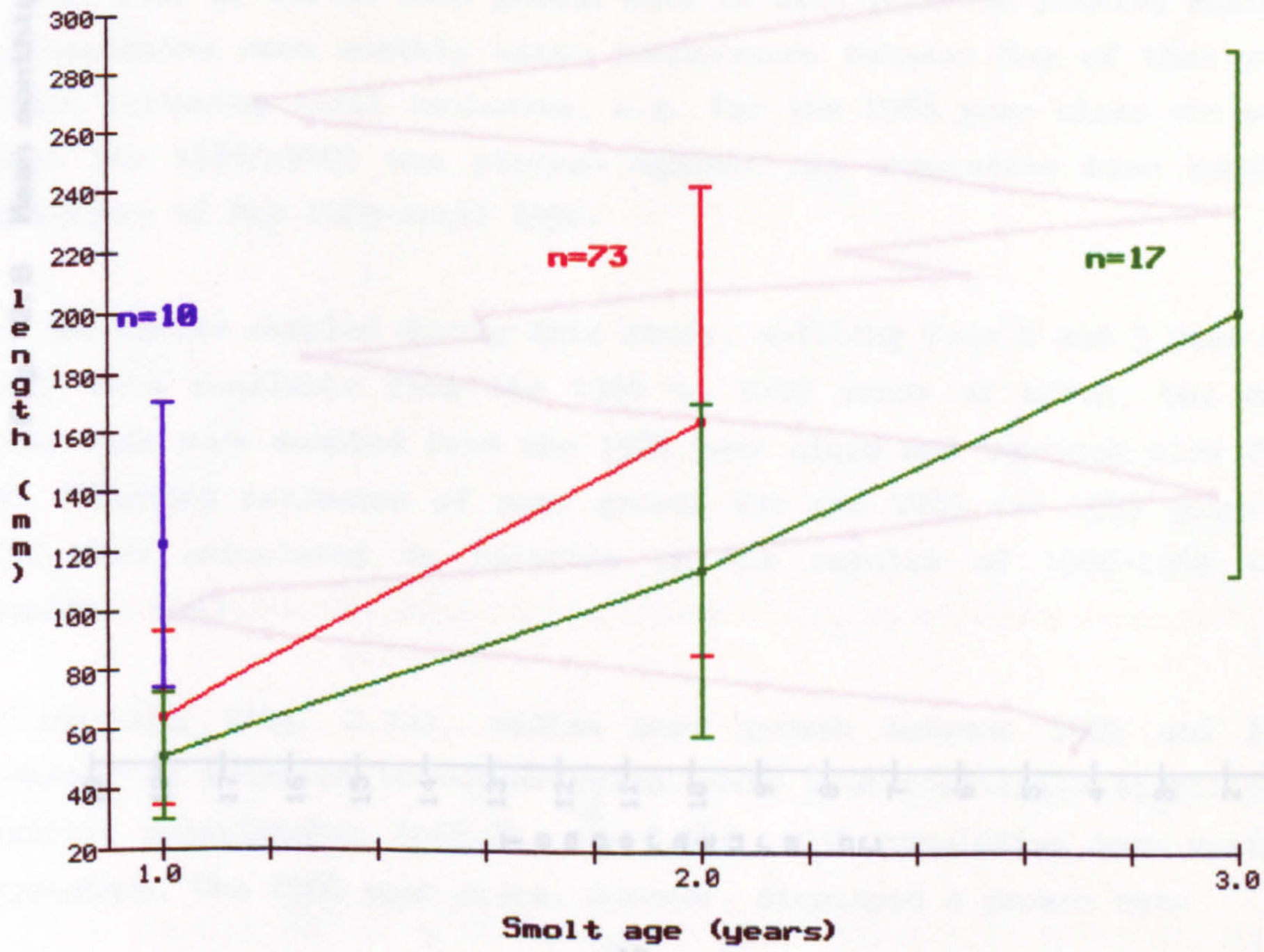


Fig. 2.5b Maiden back-calculated lengths from 1, 2 and 3 year old smolts. Error bars give the ranges observed.



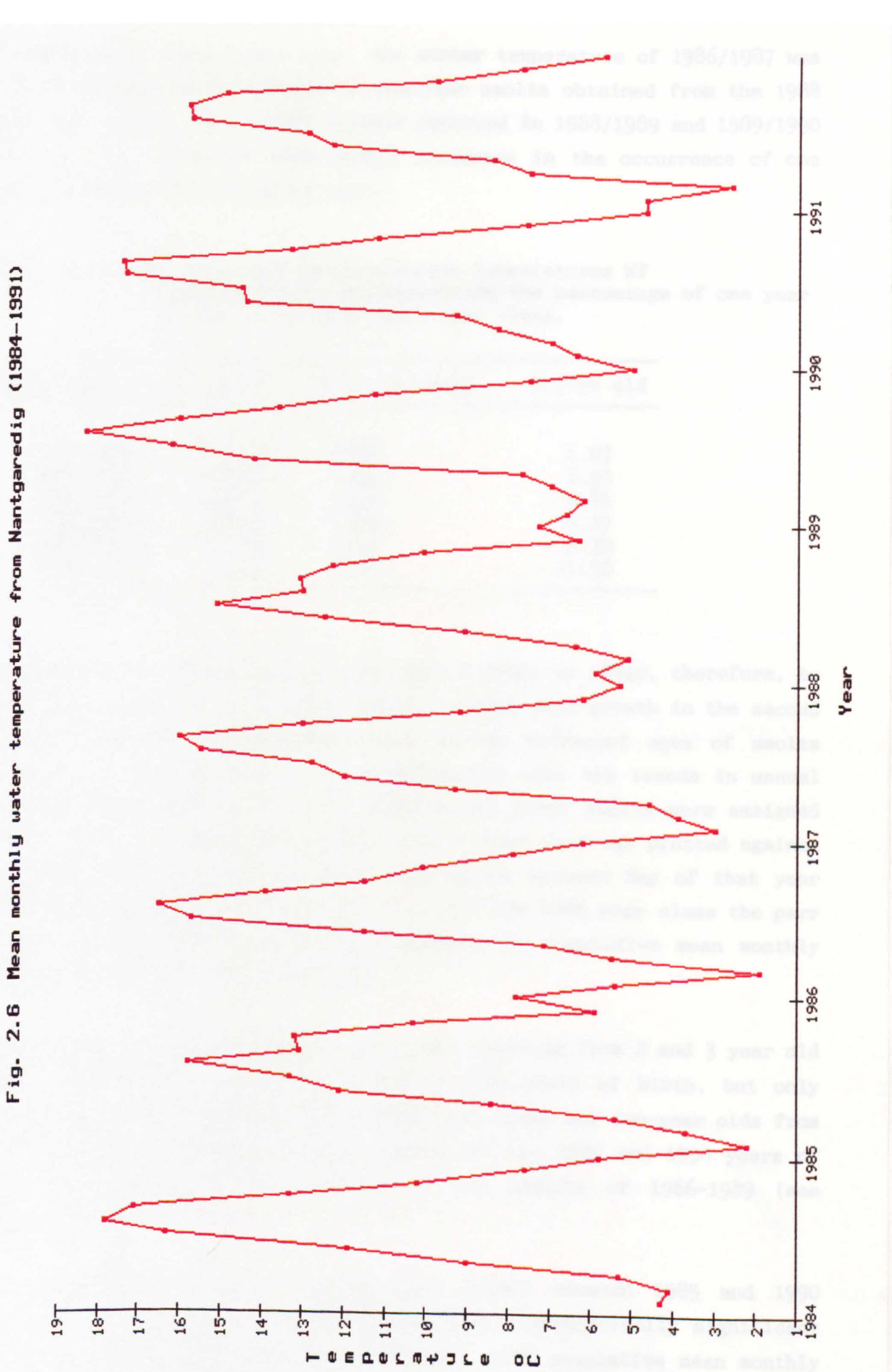


Fig. 2.6 Mean monthly water temperature from Nantgaredig (1984–1991)
Temperature oC
Year
1984
1985
1986
1987
1988
1989
1990
1991
1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19

following smolt year class, i.e. the winter temperature of 1986/1987 was plotted against the percentage of one-year smolts obtained from the 1988 smolt year class. The warmer winters observed in 1988/1989 and 1989/1990 (Table 2.11) coincided with marked increases in the occurrence of one year old smolts the following year.

Table 2.11 Cumulative mean monthly winter temperatures WT (November-April inclusive) and the percentage of one year old smolts in each smolt year class.

Winter period	WT oC	Smolt yr class	% 1-yr old
1984/1985	33.10	1986	1.07
1985/1986	45.13	1987	1.55
1986/1987	46.90	1988	0.41
1987/1988	50.50	1989	0.27
1988/1989	55.30	1990	6.70
1989/1990	57.90	1991	11.30

The majority of smolts sampled were aged 2 years or older, therefore, to assess the impact of the warmer summers, annual parr growth in the second year of life was investigated. Due to the different ages of smolts within a smolt year class, the investigation into the trends in annual back-calculated parr growth were carried out after smolts were assigned to their year of birth. Mean growth rate in each year was plotted against the cumulative mean monthly water temperature between May of that year and the following April inclusive, e.g. for the 1988 year class the parr growth (in 1989/1990) was plotted against the cumulative mean monthly temperature of May 1989-April 1990.

From the adults sampled during this study, whitling from 2 and 3 year old smolts were available from the 1986 to 1989 years of birth, but only 3-year olds were sampled from the 1985 year class and two-year olds from 1990. Weighted estimates of parr growth for the 1985 and 1990 years of birth were calculated in relation to the results of 1986-1989 (see Appendix. 2.1).

For whitling (Fig. 2.7a), median parr growth between 1985 and 1990 increased in relation to warmer years, with a statistically significant parabolic relationship ($p<0.01$, $r^2 = 0.98$) with cumulative mean monthly temperature. The 1988 year class, however, displayed a growth rate

Fig. 2.7a Median growth rate (whitling) in year 1-2 (yr birth) vs cumulative mean monthly temperature

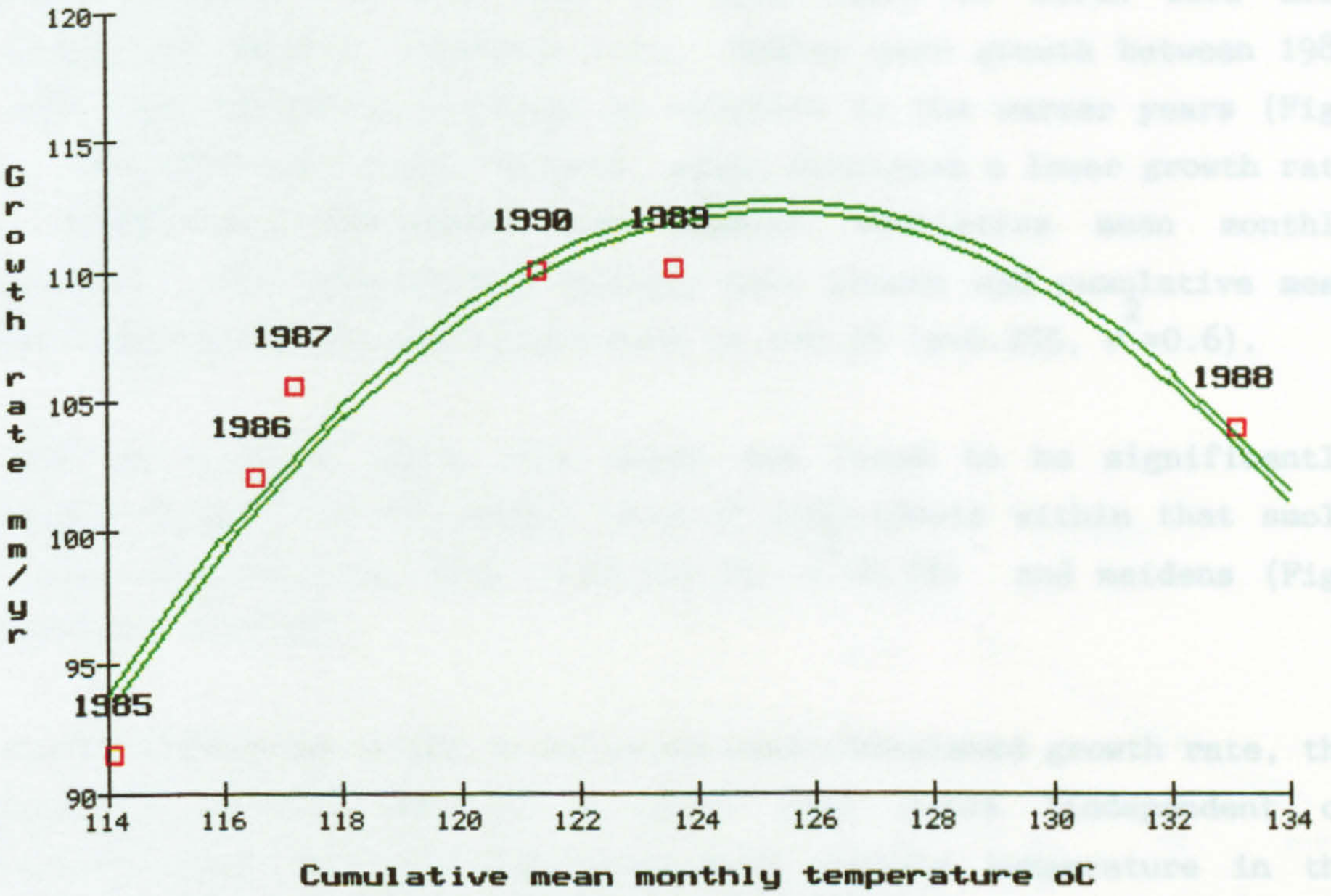
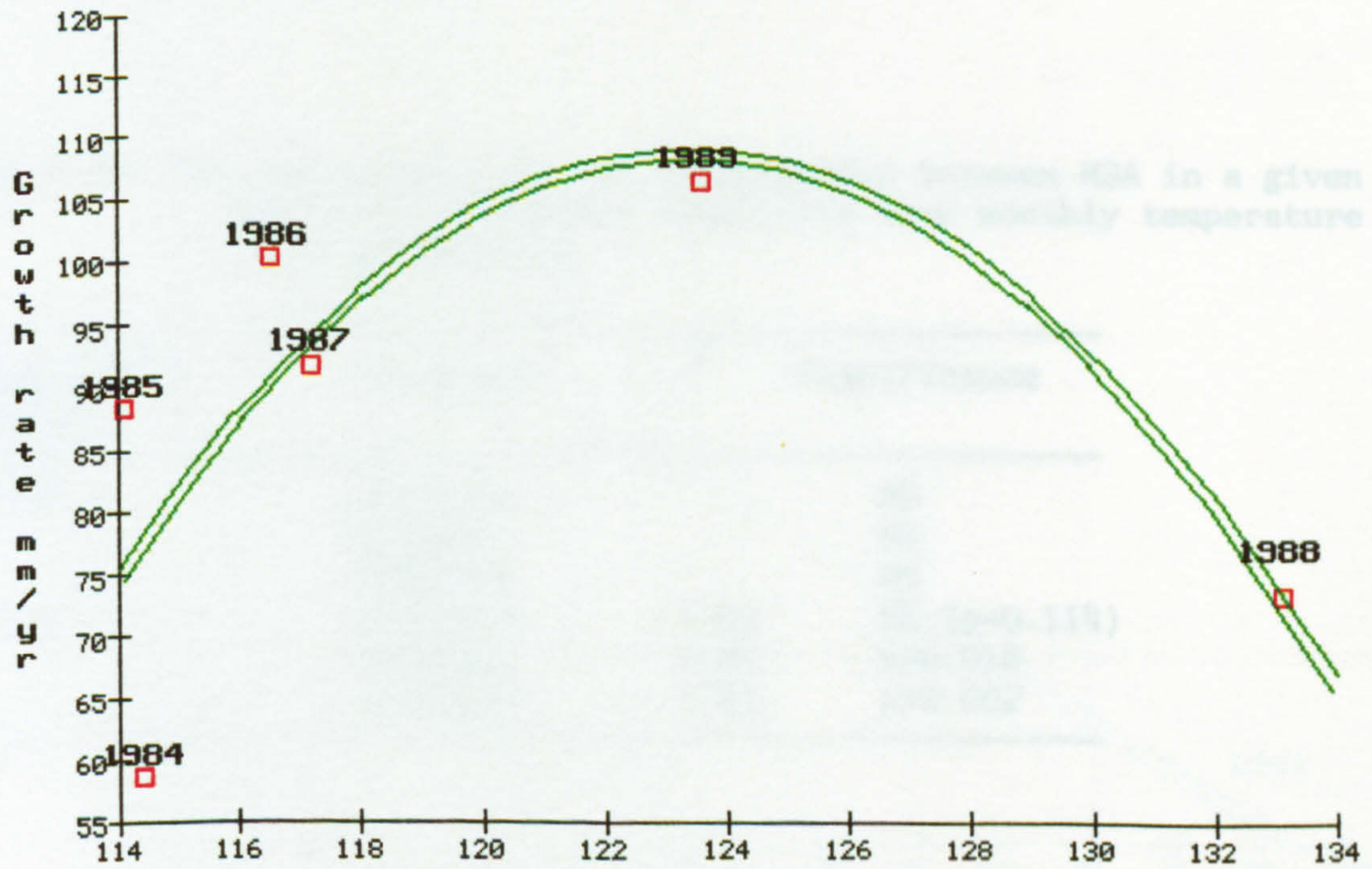


Fig. 2.7b Median growth rate (maidens) in year 1-2 (yr birth) vs cumulative mean monthly temperature



similar to the 1986/1987 year classes even though it experienced the highest cumulative temperature.

Weighted estimates for the 1984 to 1989 years of birth were also calculated for maidens (Appendix 2.1). Median parr growth between 1984 and 1989 also showed an increase in relation to the warmer years (Fig. 2.7b). The 1988 year class, however, again displayed a lower growth rate even though it experienced the highest cumulative mean monthly temperature. The relationship between parr growth and cumulative mean monthly temperature was not significant at $p<0.05$ ($p=0.255$, $r^2=0.6$).

The MSA of a given smolt year class was found to be significantly negatively related to the growth rate of individuals within that smolt year class for whitling (Fig. 2.8a; $p<0.05$, $r^2=0.78$) and maidens (Fig. 2.8b; $p<0.01$, $r^2=0.96$).

Since temperature was found to influence back-calculated growth rate, the relationship between MSA in a given year class (independent of back-calculation) and the cumulative mean monthly temperature in the periods one and two years prior to smolting were investigated (Table 2.12). MSA from a given smolt year class was inversely related to the cumulative mean monthly temperature in the two years prior to smolting ($p<0.05$; Figs. 2.9a-c).

Table 2.12 The significance of the relationship between MSA in a given smolt year class and cumulative mean monthly temperature prior to smolting.

Period before smolting	Age group	r^2	Significance
1 year	whitling		NS
	maidens		NS
	combined		NS
2 year	whitling	0.62	NS (p=0.114)
	maidens	0.88	p=0.018
	combined	0.93	p=0.002

Fig. 2.8a Whitling MSA by year of smolting vs median growth rate (smolt length/age)

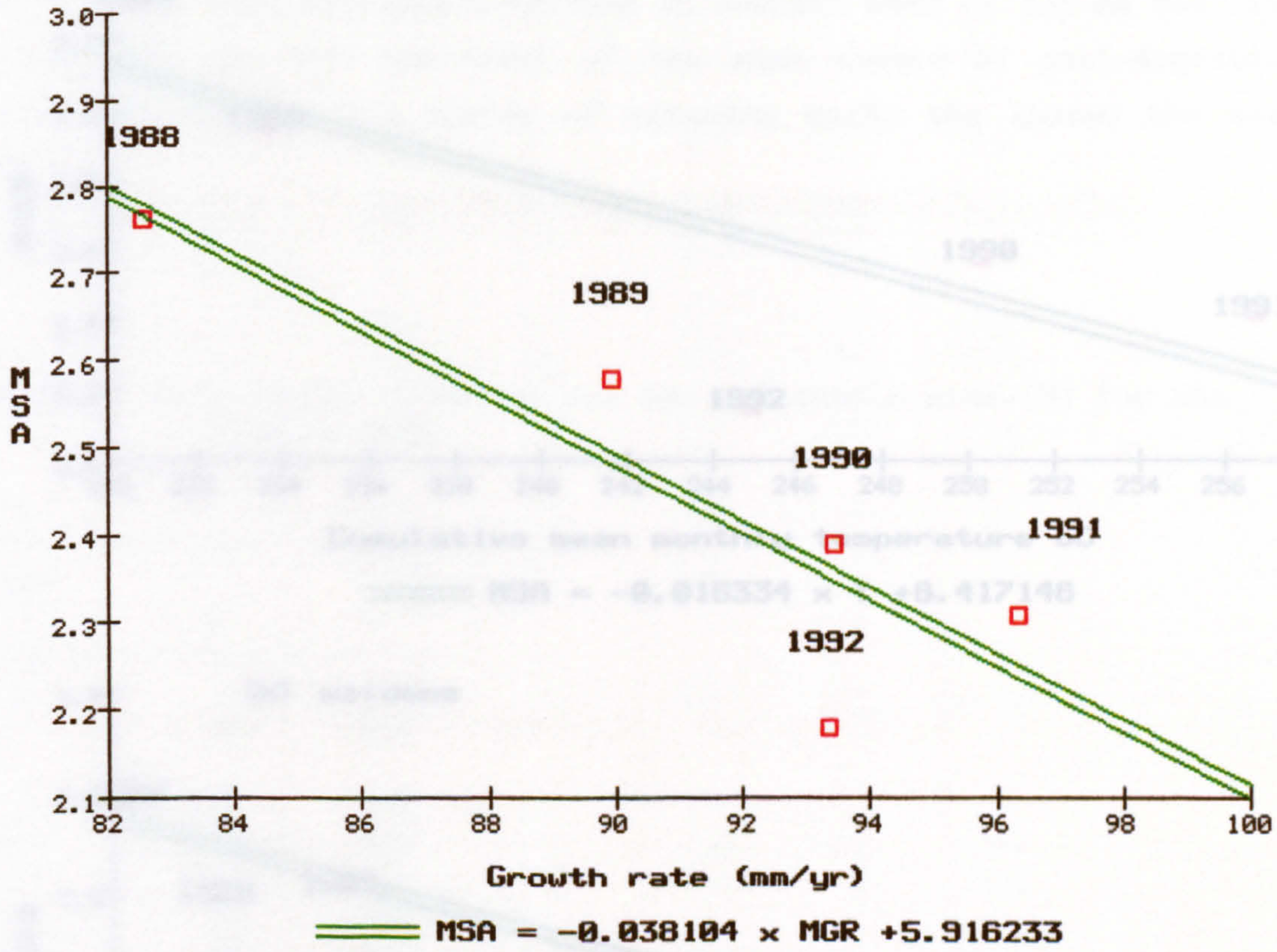


Fig. 2.8b Maiden MSA by year of smolting vs median growth rate (smolt length/age)

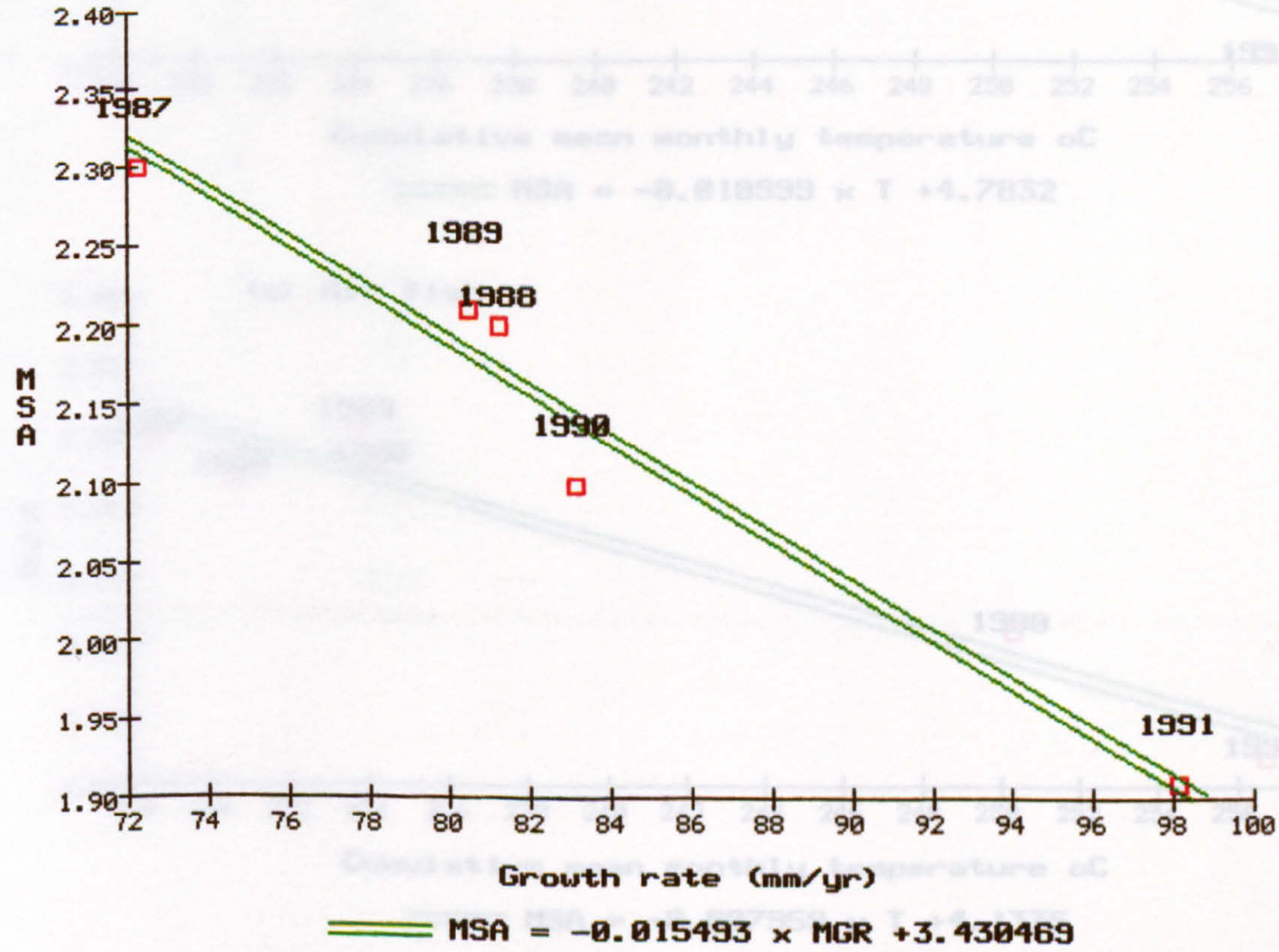
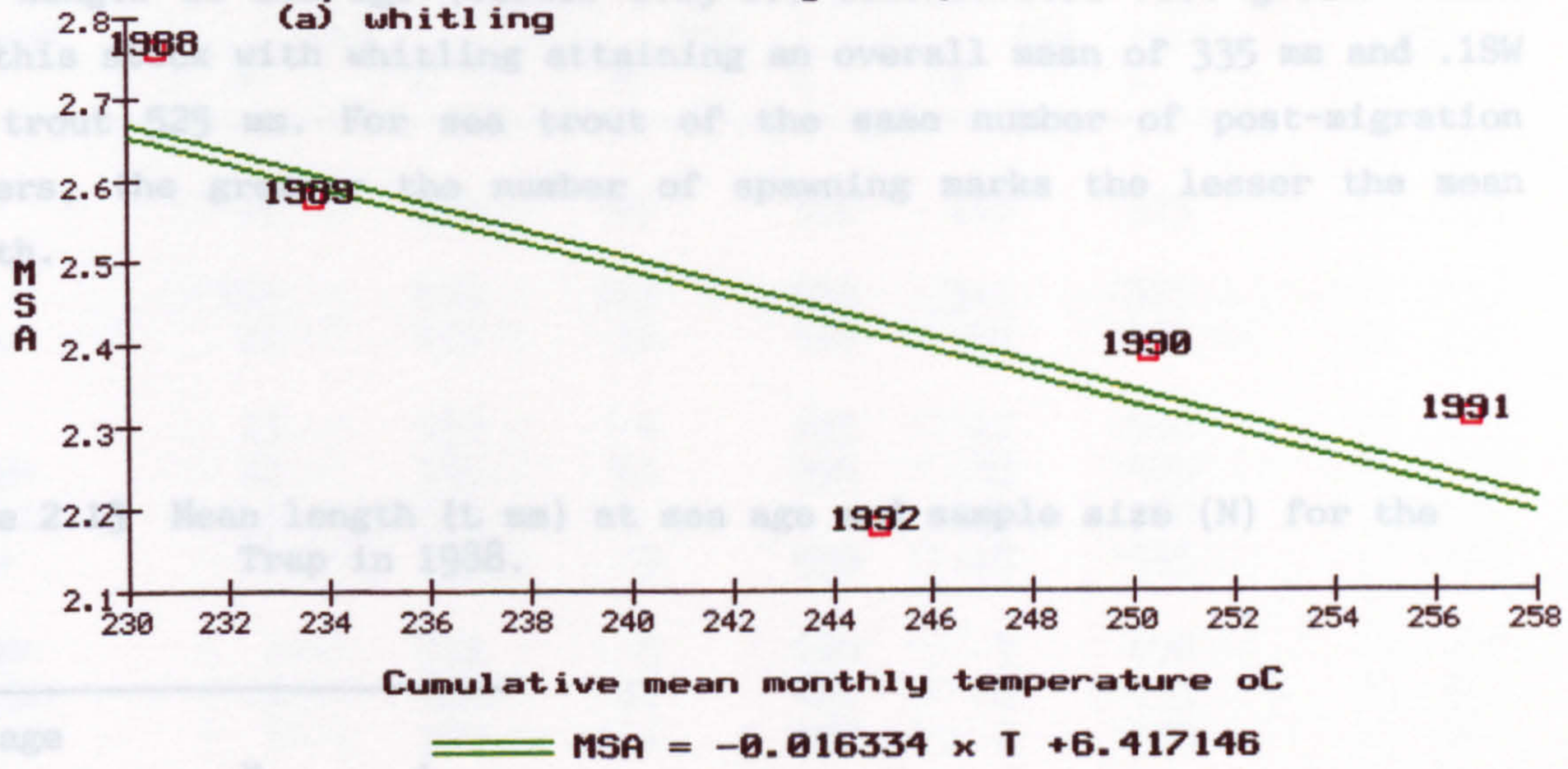
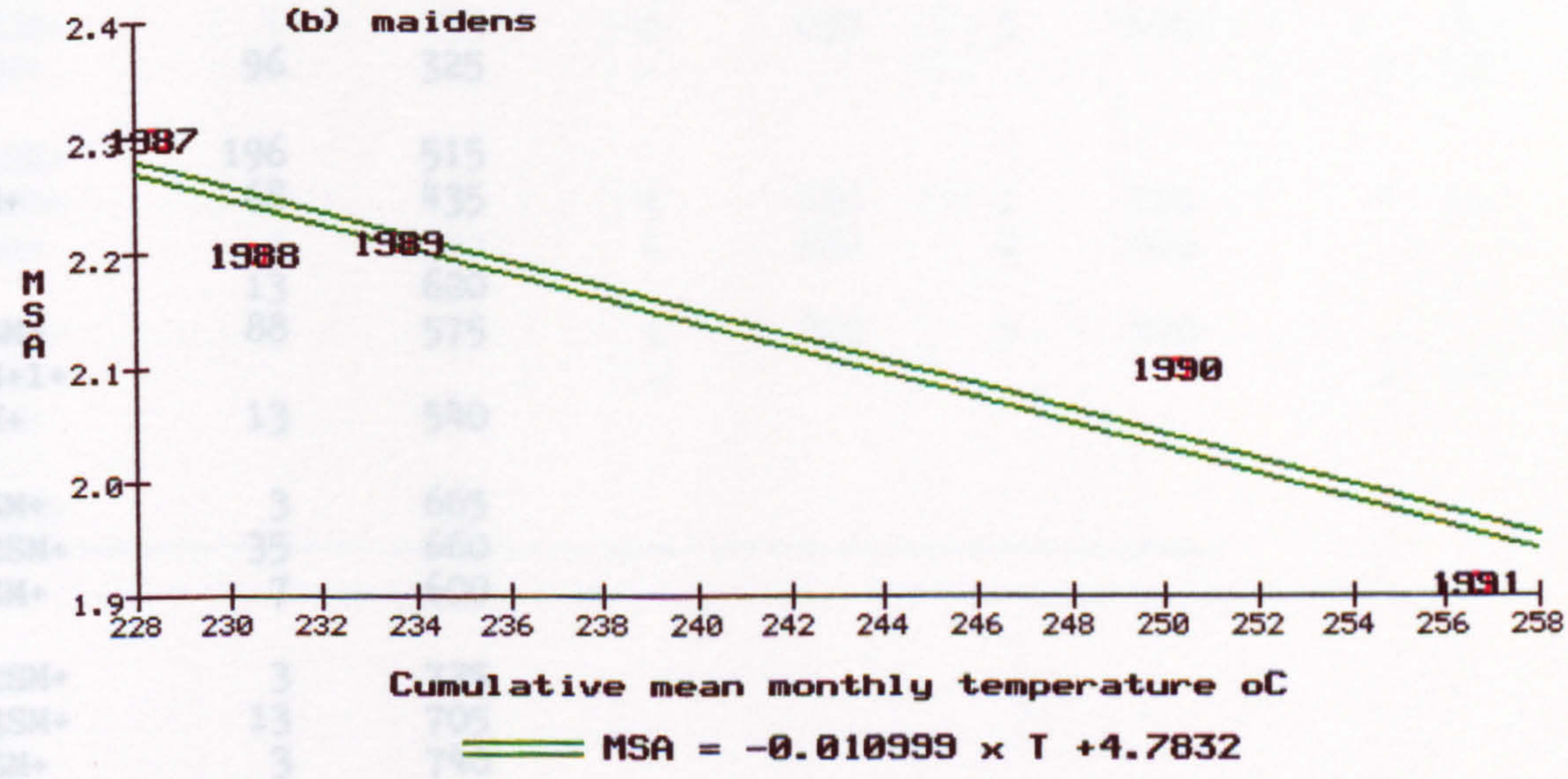


Fig. 2.9 MSA by year of smolting vs cumulative mean monthly temperature in the two years prior to smolting

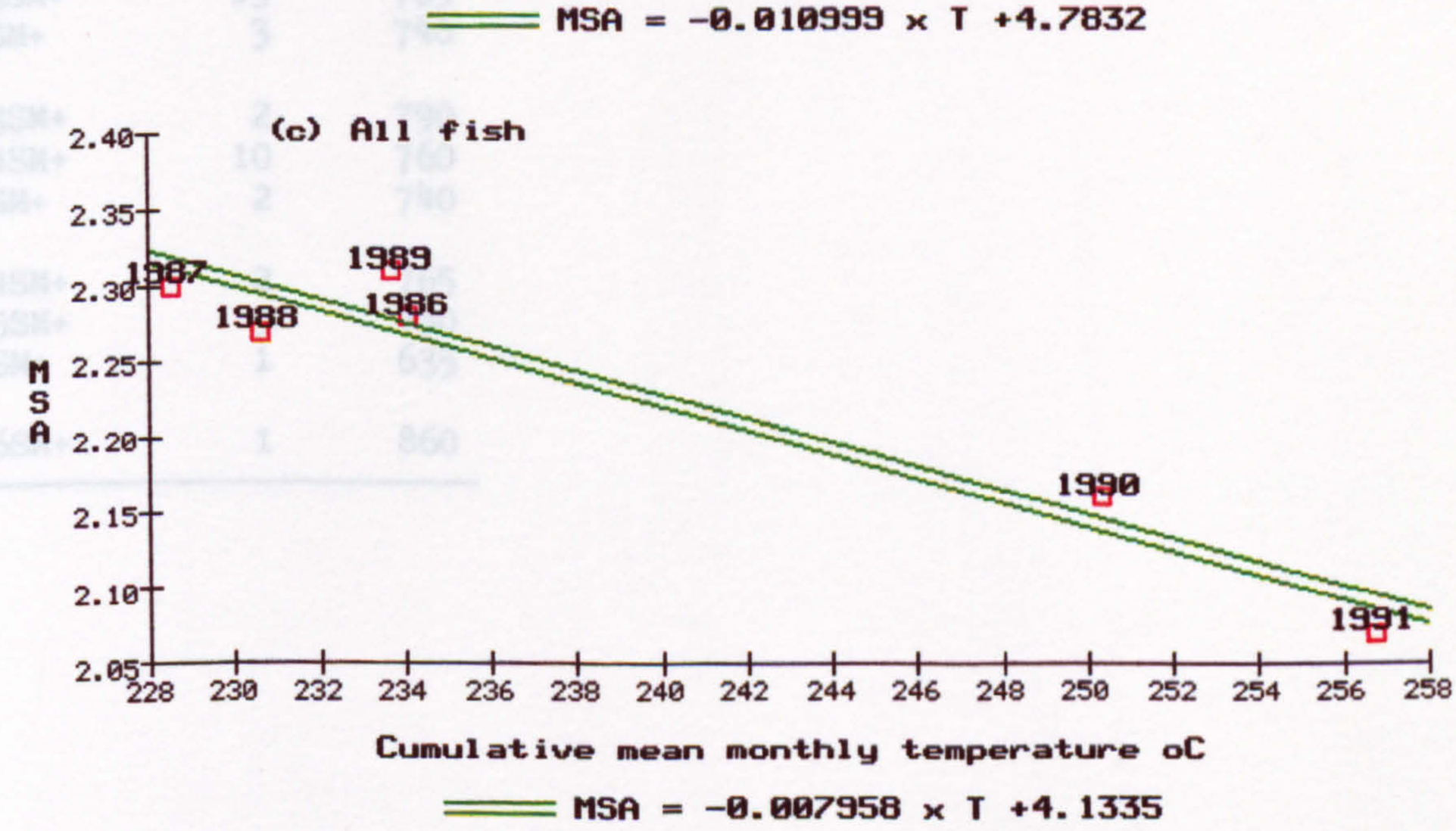
(a) whitling



(b) maidens



(c) All fish



Adults - Length at sea age

Mean length at sea age (Tables 2.13-18) demonstrated fast growth rates for this stock with whitling attaining an overall mean of 335 mm and .1SW sea trout 525 mm. For sea trout of the same number of post-migration winters, the greater the number of spawning marks the lesser the mean length.

Table 2.13 Mean length (L mm) at sea age and sample size (N) for the Trap in 1988.

Sea age	N	L
.+	96	325
.1+	196	515
.+SM+	68	435
.2+	13	620
.1+SM+	88	575
.+SM+1+		
.2SM+	13	540
.2+SM+	3	665
.1+2SM+	35	660
.+3SM+	7	600
.2+2SM+	3	725
.1+3SM+	13	705
.+4SM+	3	740
.2+3SM+	2	790
.1+4SM+	10	760
.+5SM+	2	740
.2+4SM+	2	765
.1+5SM+	5	750
.+6SM+	1	635
.2+6SM+	1	860

Table 2.14 Mean length (L mm) at sea age and sample size (N) for the Rod and Net samples in 1989.

Sea age	Rod		Net		All	
	N	L	N	L	N	L
.+	29	330	89	320	118	325
.1+	153	525	163	530	316	525
.+SM+	27	475	18	460	45	470
.2+	13	580	4	635	17	595
.1+SM+	22	565	61	595	83	585
.+SM+1+	1	580			1	580
.2SM+	9	535	9	550	18	545
.2+SM+	2	715	5	680	7	690
.1+2SM+	9	610	11	660	20	635
.+3SM+	1	570	1	590	2	580
.2+2SM+						
.1+3SM+	1	735	2	630	3	665
.+4SM+						
.2+3SM+						
.1+4SM+			1	720	1	720
.+5SM+	1	540	1	660	2	600
.2+4SM+			1	820	1	820
.1+5SM+						
.+6SM+						
.2+6SM+						

Table 2.15 Mean length (L mm) at sea age and sample size (N) for the Rod, Net, Coracle and trap samples in 1990.

Sea age	Rod		Net		Coracle		Trap		All	
	N	L	N	L	N	L	N	L	N	L
.+	48	350	77						125	330
.1+	56	545	116	535	62	535	16	535	249	535
.+SM+	7	485	5	515	4	485	3	525	19	500
.2+	10	595	6	625	2	595	2	620	20	605
.1+SM+	32	600	41	600	38	590	23	600	135	595
.+SM+1+	1	690	1	530					2	610
.2SM+	6	535	4	535	3	550	4	530	17	535
.2+SM+	6	665	9	640	3	650	7	650	25	650
.1+2SM+	6	640	14	640	20	645	10	635	50	640
.+3SM+	4	595	4	570	2	595	5	565	15	580
.2+2SM+			4	695	1	680	3	715	8	700
.1+3SM+	1	690	7	685	3	700	3	660	14	685
.+4SM+					3	695			3	695
.2+3SM+										
.1+4SM+			1	740	3	770			4	760
.+5SM+										
.2+4SM+										
.1+5SM+			2	730					2	730
.+6SM+										
.2+6SM+										

Table 2.16 Mean length (L mm) at sea age and sample size (N) for the Rod, Net, and Coracle samples in 1991.

Sea age	Rod		Net		Coracle		All	
	N	L	N	L	N	L	N	L
.+	43	375	10	380*	2	390*	55	375
.1+	57	525	179	525	59	520	295	525
.+SM+	13	480	21	455	3	480	37	465
.2+	5	615	12	595	2	625	19	605
.1+SM+	12	605	45	600	23	580	80	595
.+SM+1+								
.2SM+	5	570	2	505	2	500	9	540
.2+SM+	3	650	1	670	2	670	6	660
.1+2SM+	5	675	18	660	8	665	31	665
.+3SM+	4	645	6	585	2	610	12	610
.2+2SM+								
.1+3SM+	1	800	8	695	2	650	11	695
.+4SM+	1	690	1	690	2	585	4	635
.2+3SM+			1	765			1	765
.1+4SM+	1	740	3	720	2	730	6	725
.+5SM+								
.2+4SM+								
.1+5SM+								
.+6SM+								
.2+6SM+								

Table 2.17 Mean length (L mm) at sea age and sample size (N) for the Rod and Coracle samples in 1992.

Sea age	Rod		Coracle		All	
	N	L	N	L	N	L
.+	41	355	4	395*	45	355
.1+	72	530	38	535	110	530
.+SM+	10	480	5	465	15	475
.2+	9	610	4	610	13	610
.1+SM+	9	595	4	625	13	605
.+SM+1+						
.2SM+	4	545	2	520	6	535
.2+SM+	1	635			1	635
.1+2SM+	3	690	8	680	11	680
.+3SM+						
.2+2SM+			1	750	1	750
.1+3SM+	4	700	2	720	6	710
.+4SM+	2	680			2	680
.2+3SM+						
.1+4SM+			1	750	1	750
.+5SM+						
.2+4SM+						
.1+5SM+						
.+6SM+						
.2+6SM+						

Table 2.18 Mean length (L mm) at sea age and sample size (N) for the the five year period 1988-1992.

Sea age	N	L	Minimum	Maximum
.+	439	335	210	440
.1+	1166	525	370	725
.+SM+	184	460	355	675
.2+	82	605	510	690
.1+SM+	399	590	445	720
.+SM+1+	3	600	530	690
.2SM+	63	540	470	640
.2+SM+	42	660	570	770
.1+2SM+	146	655	525	780
.+3SM+	36	595	465	690
.2+2SM+	12	710	715	770
.1+3SM+	47	695	625	825
.+4SM+	12	685	585	770
.2+3SM+	3	780	765	810
.1+4SM+	22	750	660	815
.+5SM+	4	670	540	760
.2+4SM+	3	785	750	820
.1+5SM+	7	745	705	800
.+6SM+	1	635	635	635
.2+6SM+	1	860	860	860

Adults - Effect of smolt age and juvenile growth rate

Within years smolt age was observed to affect the mean length of returning adults in some samples. Whitling from 3-year-old smolts were consistently larger than from 2-year olds, and this was significant for the 1988-1990 samples ($p<0.05$; Table 2.19).

This trend was not as clear for .1+ sea trout (Table 2.19) and no significant difference ($p<0.05$) in mean lengths from 2 or 3 year old smolts were observed for .1+ fish in 1988, 1989 or 1992. However, in 1990 and 1991 3 year old smolts did produce significantly larger .1+ sea trout ($p<0.05$).

Table 2.19 Mean length (mm) at sea age from 2 and 3 year old smolts.

Smolt age	Sea age .+					Sea age .1+				
	1988	1989	1990	1991	1992	1988	1989	1990	1991	1992
2	310	305	325	375	360	515	525	530	525	535
3	325*	340*	360*	380	360	515	535	555*	550*	565

* Significantly different ($p<0.05$) using the Student's t-test.

For the combined years, whitling length was positively related to back-calculated smolt length (Fig. 2.10, $p<0.0001$, $r^2=0.38$). However, back-calculation demonstrated a compensatory factor for smaller smolts in that sea growth increased with decreasing smolt length (Fig. 2.10). Thus any increase in smolt length resulted in a smaller proportional increase in whitling length.

Back-calculation also showed that mean smolt length increased between the 1988 and 1990 smolt year classes but decreased thereafter. Mean whitling length of the sample used for back-calculation did not follow the same pattern (Fig. 2.11); it increased in 1991 and only decreased slightly in 1992. This therefore adds to the scatter in Fig. 2.10 with the overall pattern of increasing whitling length with smolt length. The mean annual sea growth therefore increased between 1988-1992 (Fig. 2.11), possibly indicating better sea feeding. However, a factor already established was the decrease in MSA between 1988-1992, with less three year old smolts in latter years. When mean sea growth of the sample was plotted against the MSA of the sample (Fig. 2.12) a highly significant relationship was observed ($p<0.001$, $r^2=0.99$), indicating that sea growth of whitling may be influenced by smolt age as well as smolt length.

For 1+ maidens, the overall sample again displayed a significant positive relationship between smolt length and the length of maidens (Fig. 2.13; $p<0.0001$, $r2=0.29$) but with more noise in the data. Sea growth was also inversely related to smolt length ($p<0.0001$, $r2=0.18$) for .1+ maidens. Annual mean smolt length was lower in later years (Fig. 2.14) but the mean length of whitling did not decrease, thus sea growth was higher in latter years. No relationship was evident between MSA of the sample and the sea growth observed (Fig. 2.15).

Fig. 2.10 Relationship between back-calculated smolt length (SL) and both whitling length (WL) and sea growth (SG). (n = 100)

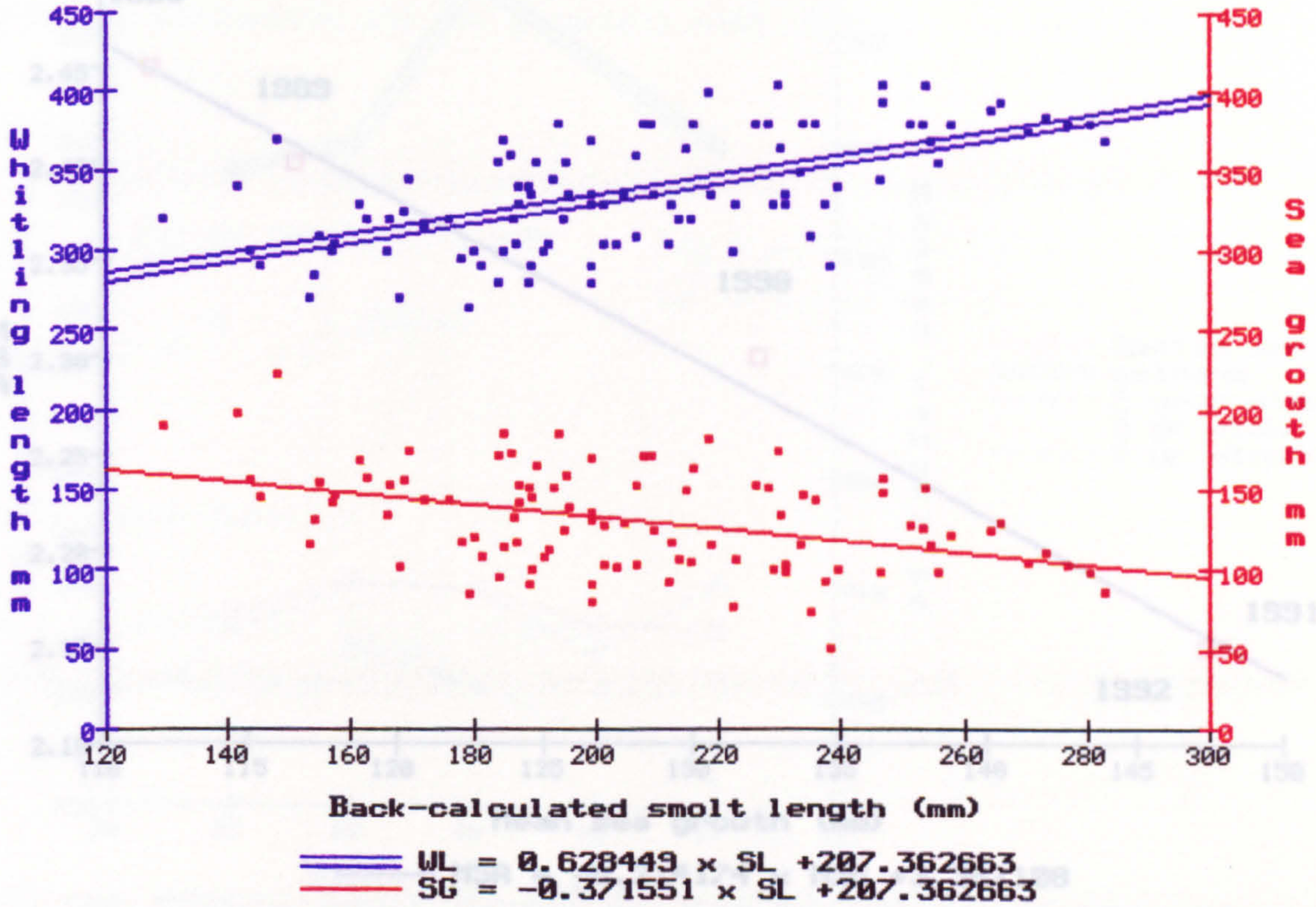


Fig. 2.11 Mean smolt and mean whitling lengths (mm) by smolt year class

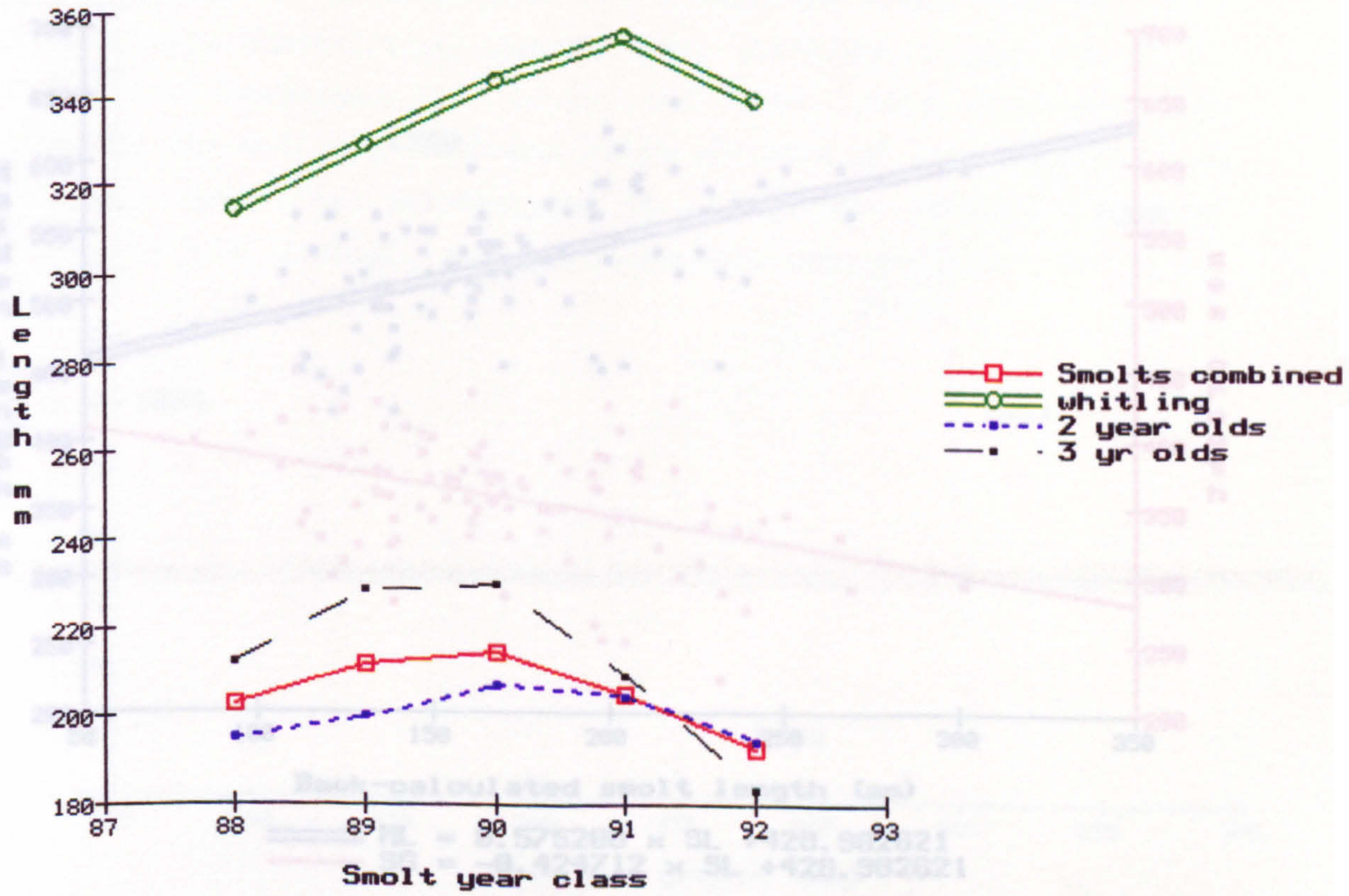


Fig. 2.12 MSA of the whitling smolt sample vs the mean sea growth (MSG)

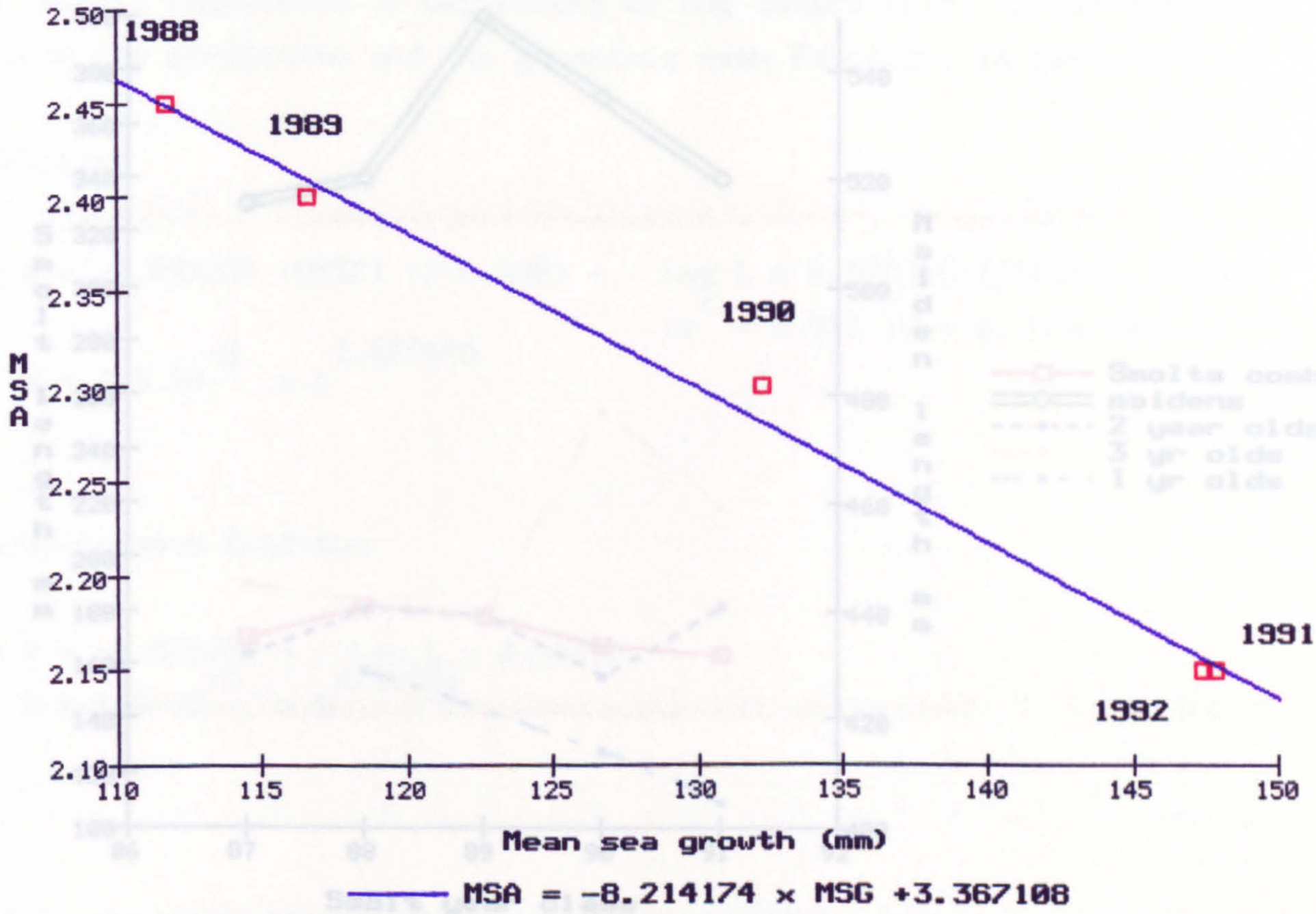


Fig. 2.13 Relationship between back-calculated smolt length (SL) and both maiden length (ML) and sea growth (SG). (n = 100)

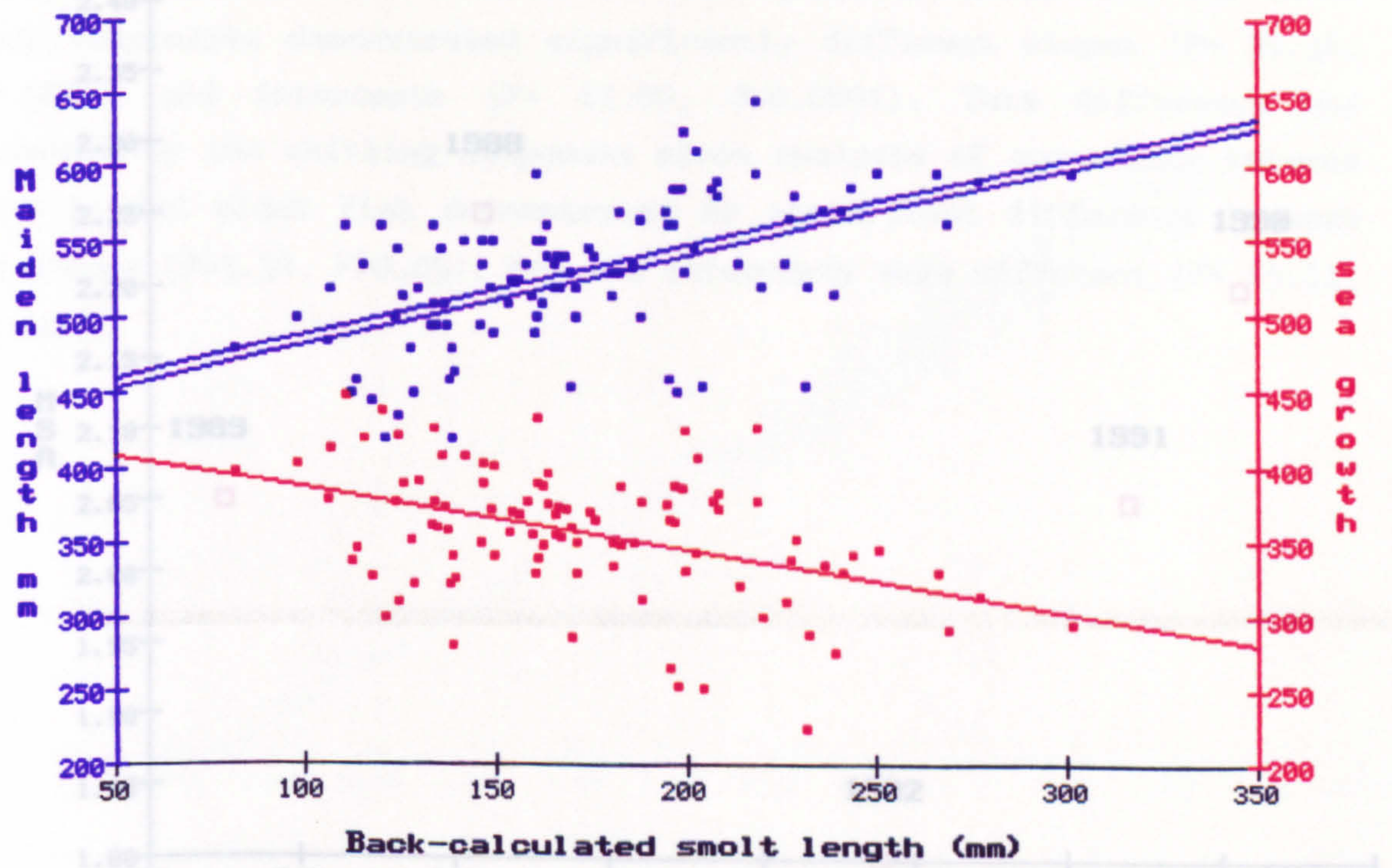


Fig. 2.14 Mean smolt and mean maiden length (mm) by smolt year class

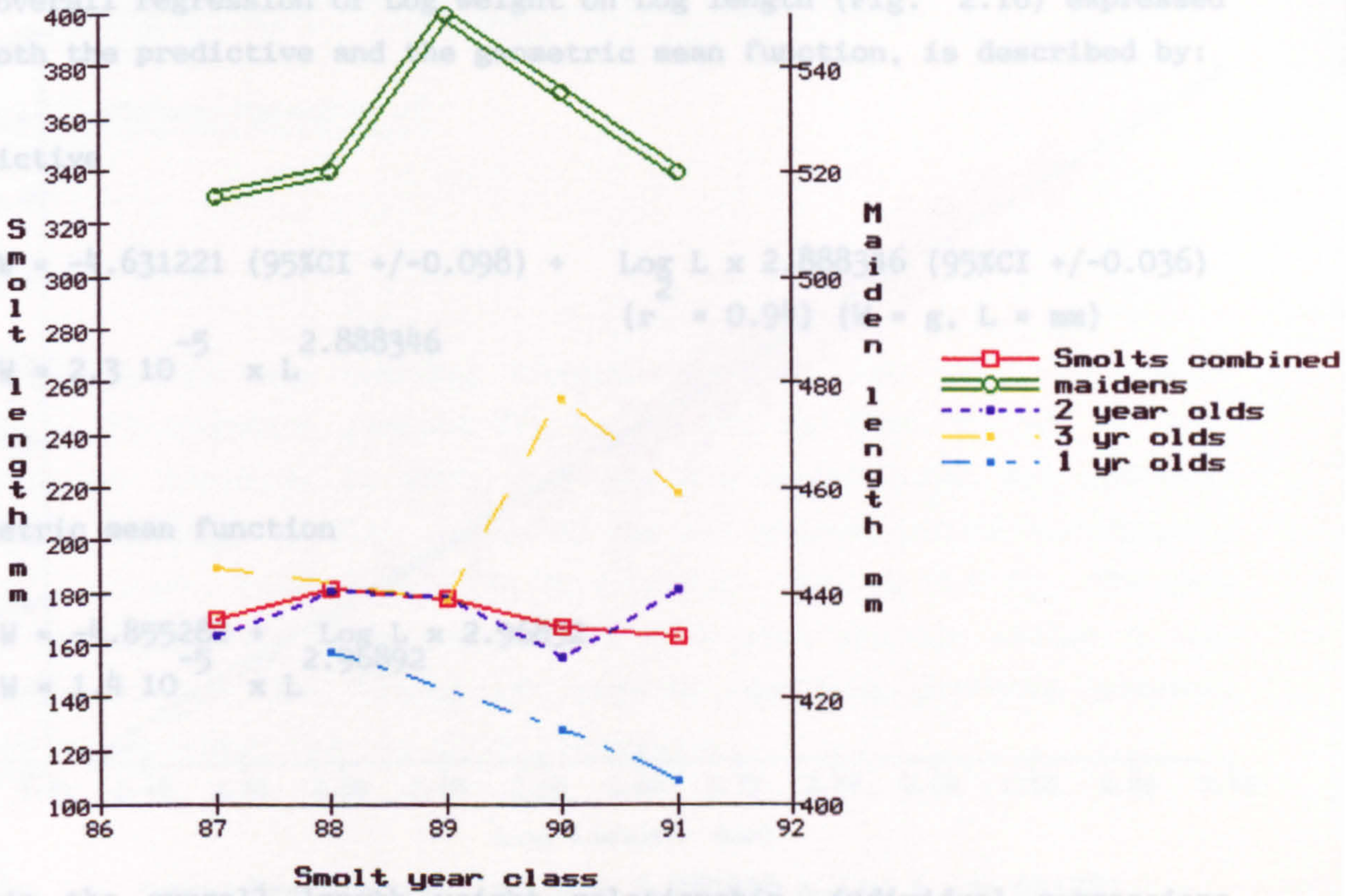
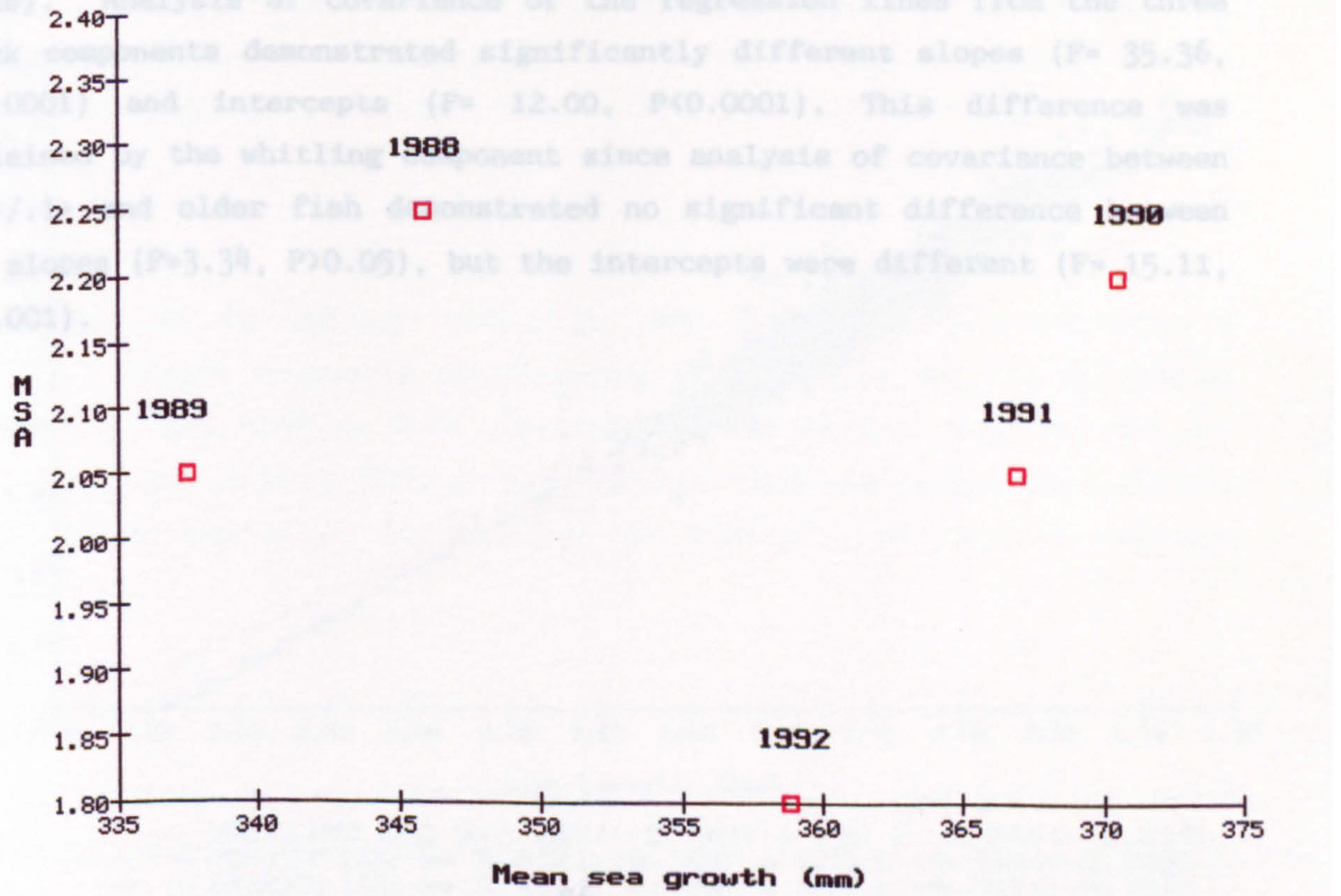


Fig. 2.15 MSA of the maiden smolt sample vs the mean sea growth (mm)



Length Weight relationship

Weights were obtained for a total of 1493 sea trout from 1989 to 1992. The overall regression of Log weight on Log length (Fig. 2.16) expressed as both the predictive and the geometric mean function, is described by:

Predictive

$$\text{Log } W = -4.631221 \ (95\%CI +/-0.098) + \text{Log } L \times 2.888346 \ (95\%CI +/-0.036)$$

$(r^2 = 0.94)$ (W = g, L = mm)

$$\text{or } W = 2.3\ 10^{-5} \times L^{2.888346}$$

Geometric mean function

$$\text{Log } W = -4.855282 + \text{Log } L \times 2.96892$$

$$\text{or } W = 1.4\ 10^{-5} \times L^{2.96892}$$

Within the overall length weight relationship, individual regressions derived from whitling, .SM+/.1+ and fish older than one winter after smolting, demonstrated higher slope values for non-whitling (Fig. 2.16b). Analysis of covariance of the regression lines from the three stock components demonstrated significantly different slopes ($F = 35.36$, $P<0.0001$) and intercepts ($F = 12.00$, $P<0.0001$). This difference was explained by the whitling component since analysis of covariance between .SM+/.1+ and older fish demonstrated no significant difference between the slopes ($F=3.34$, $P>0.05$), but the intercepts were different ($F = 15.11$, $P<0.001$).

Fig. 2.16 Tywi sea trout
overall length vs weight relationship

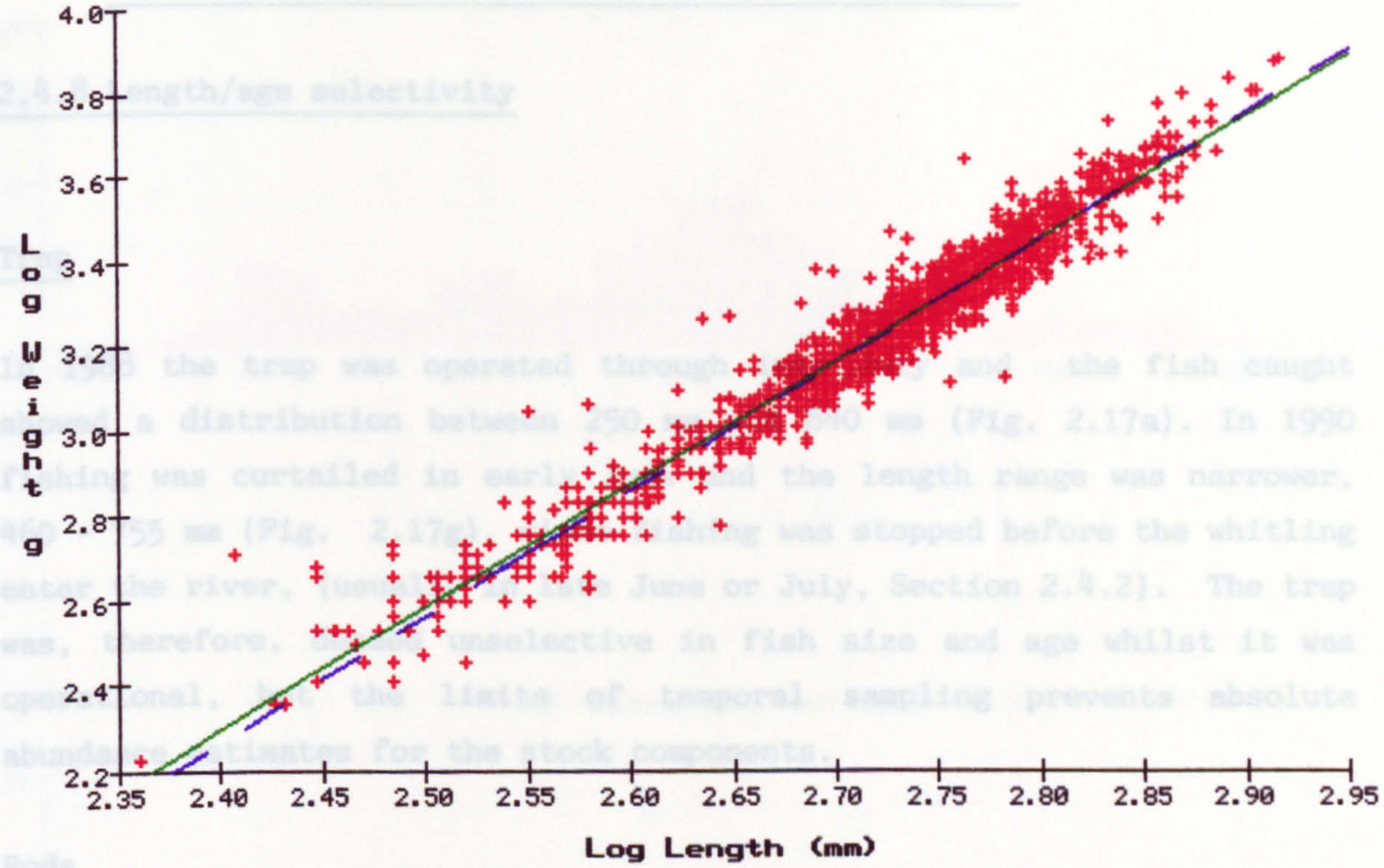
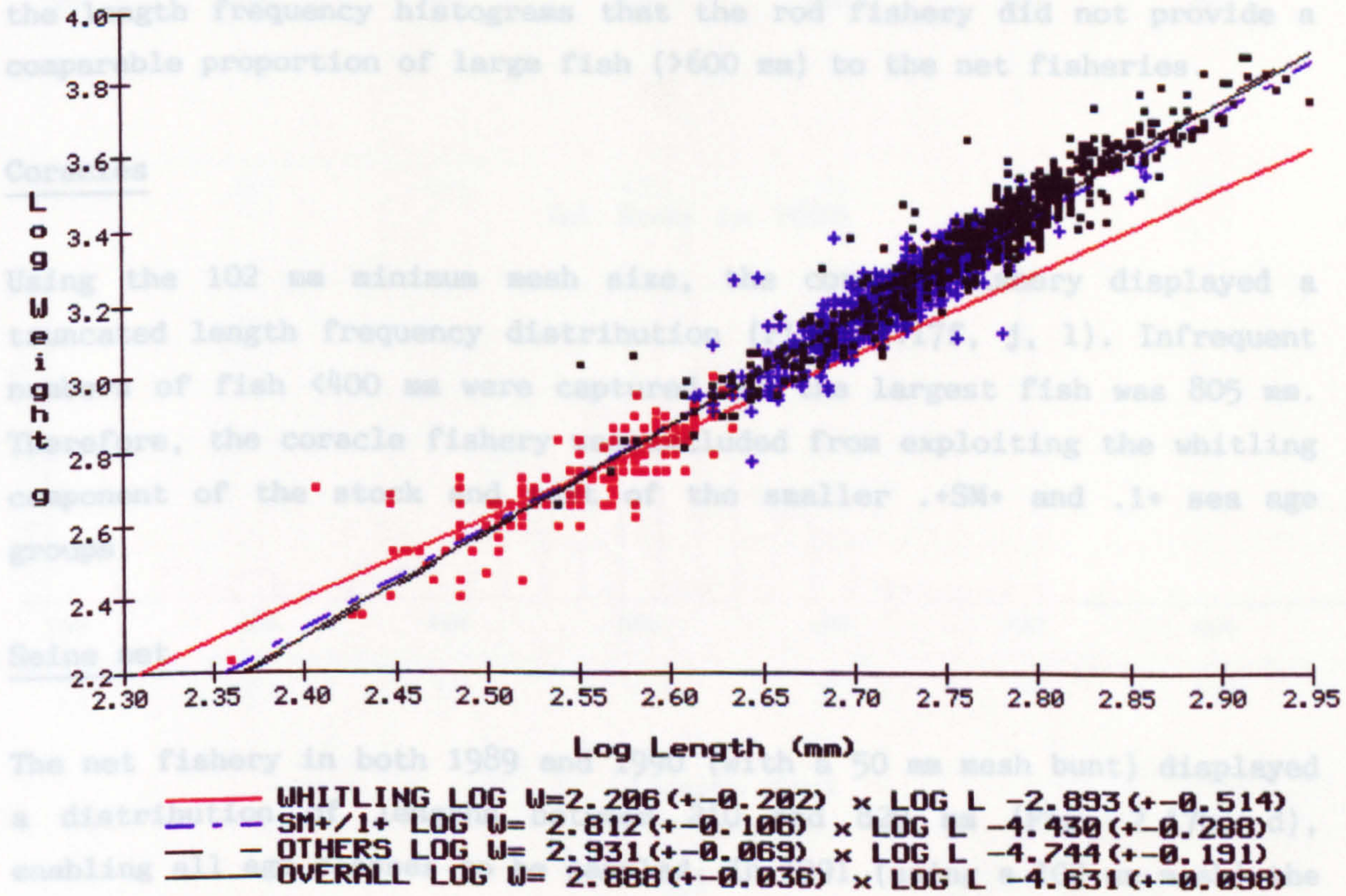


Fig. 2.16b Length v weight relationships for whitling, (F=466, P<0.0001, R2=0.73), SM+/1+ (F=2744, P<0.0001, R2=0.80) and older fish (F=7009, P<0.0001, R2=0.92)



Section B

Gear selection in the sea trout fisheries of the River Tywi. Effects on population and biological parameters.

2.4.8 Length/age selectivity

Trap

In 1988 the trap was operated through into July and the fish caught showed a distribution between 250 mm and 840 mm (Fig. 2.17a). In 1990 fishing was curtailed in early June and the length range was narrower, 460 - 755 mm (Fig. 2.17g), since fishing was stopped before the whitling enter the river, (usually in late June or July, Section 2.4.2). The trap was, therefore, deemed unselective in fish size and age whilst it was operational, but the limits of temporal sampling prevents absolute abundance estimates for the stock components.

Rods

Fish caught in the rod fishery demonstrated a wide range of lengths over the four years (210-800 mm; Fig. 2.17c, e, i, k) and the rods apparently exploited all age classes. Apart from 1989, there was no indication from the length frequency histograms that the rod fishery did not provide a comparable proportion of large fish (>600 mm) to the net fisheries.

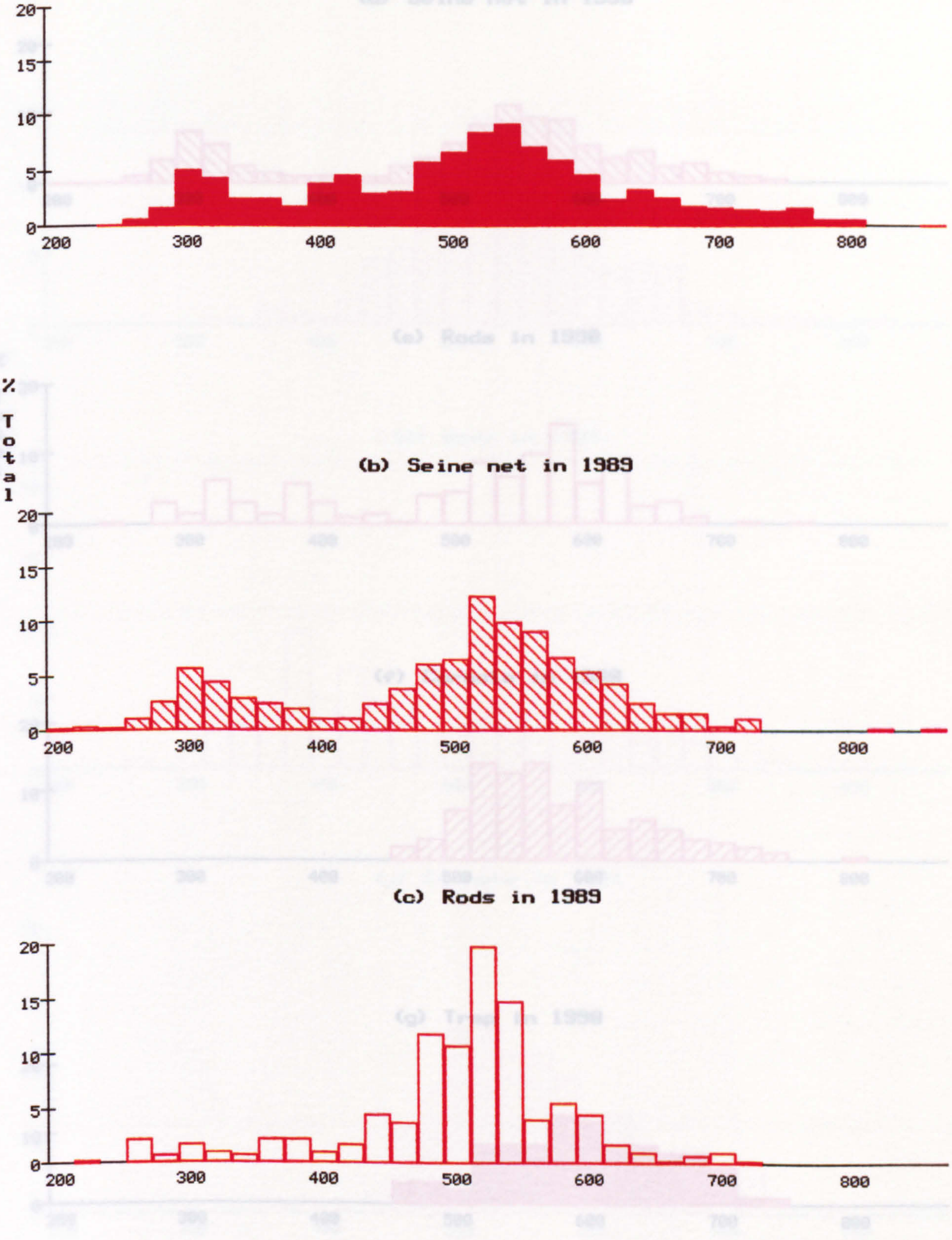
Coracles

Using the 102 mm minimum mesh size, the coracle fishery displayed a truncated length frequency distribution (Figs. 2.17f, j, l). Infrequent numbers of fish <400 mm were captured and the largest fish was 805 mm. Therefore, the coracle fishery was excluded from exploiting the whitling component of the stock and part of the smaller .+SM+ and .1+ sea age groups.

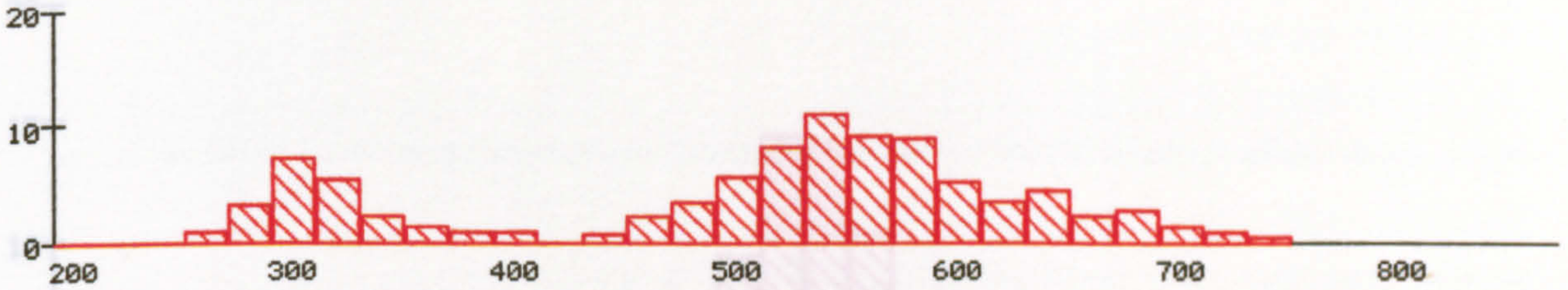
Seine net

The net fishery in both 1989 and 1990 (with a 50 mm mesh bunt) displayed a distribution of lengths between 210 and 820 mm (Fig. 2.17b, d), enabling all age classes to be sampled. In 1991 (using a 102 mm mesh) the

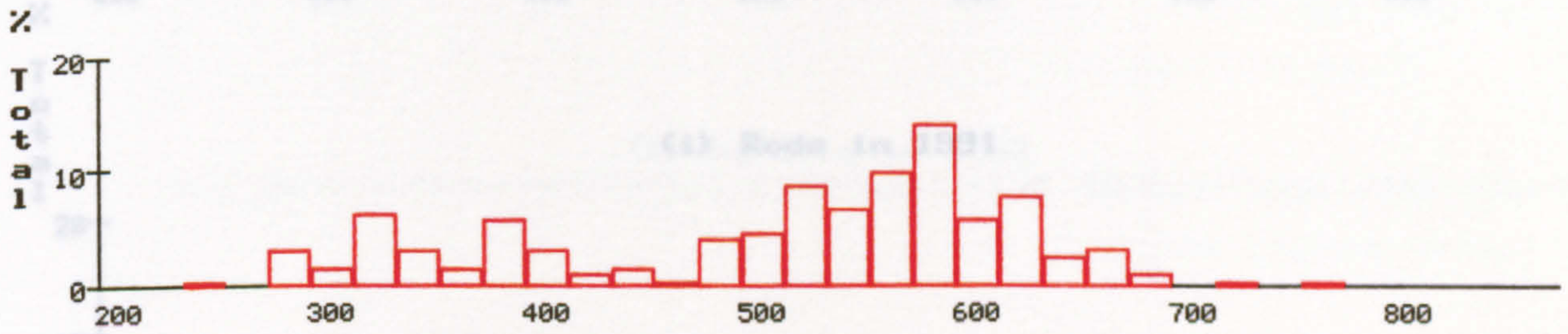
Fig. 2.17 Length frequency Histograms of Sea Trout



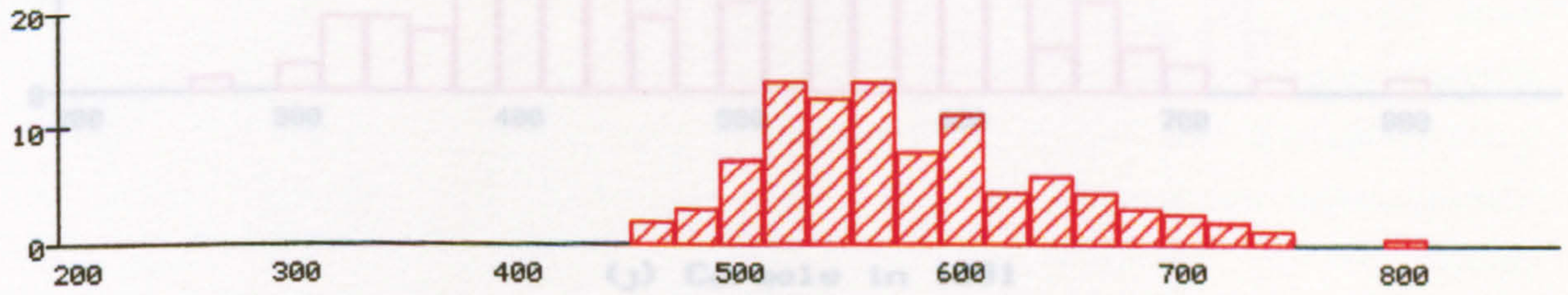
(d) Seine net in 1990



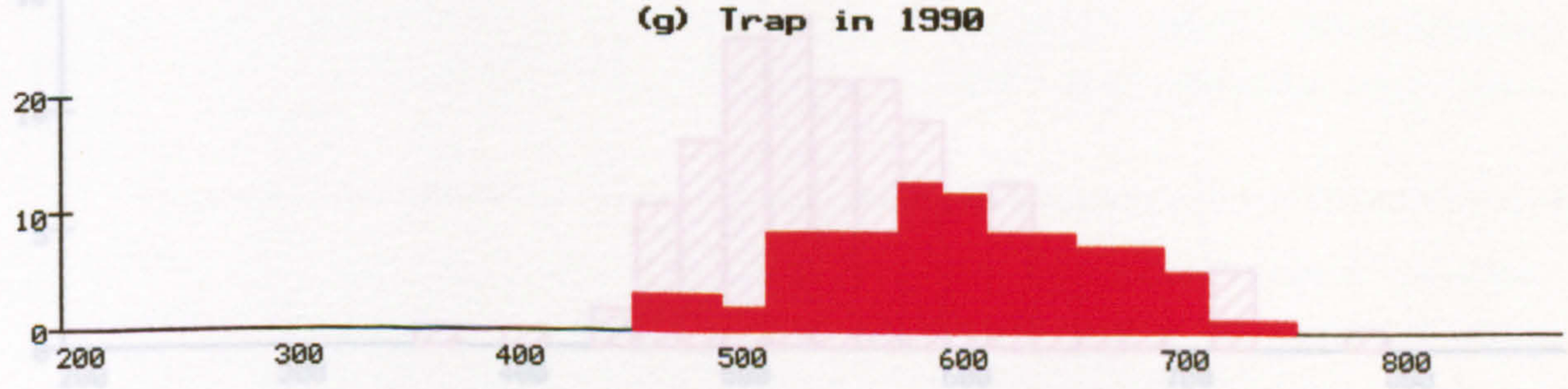
(e) Rods in 1990



(f) Coracle in 1990

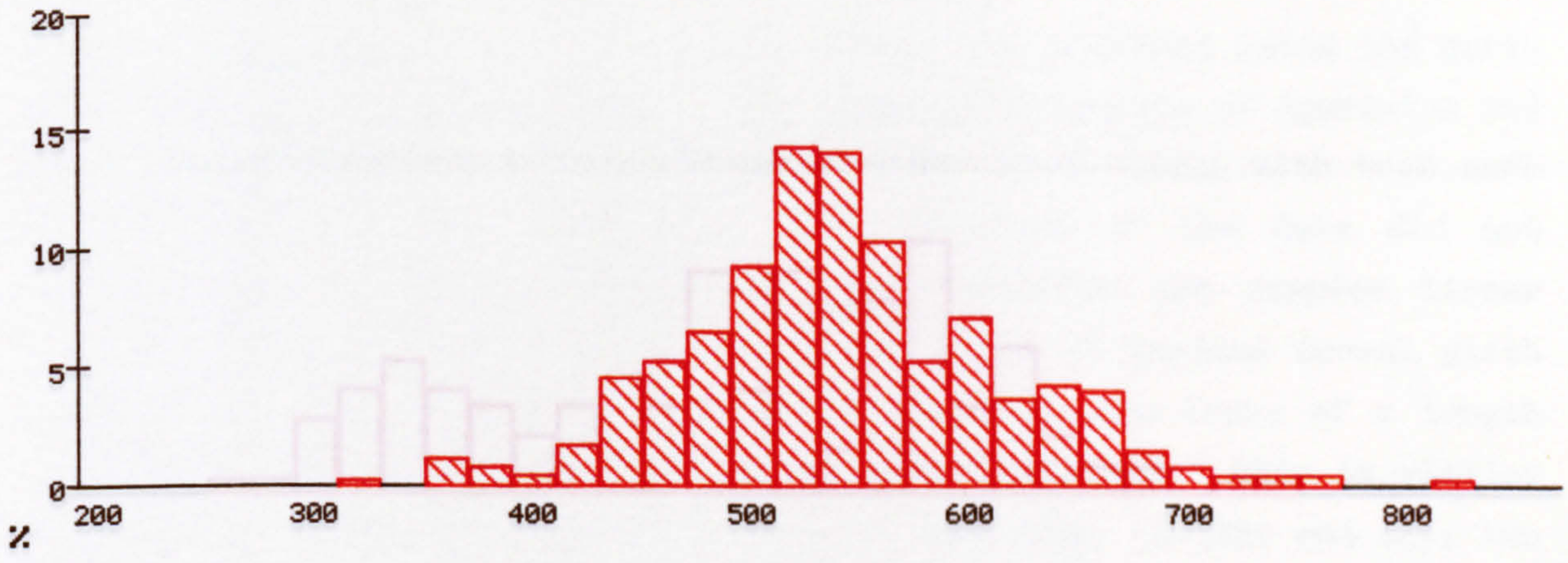


(g) Trap in 1990

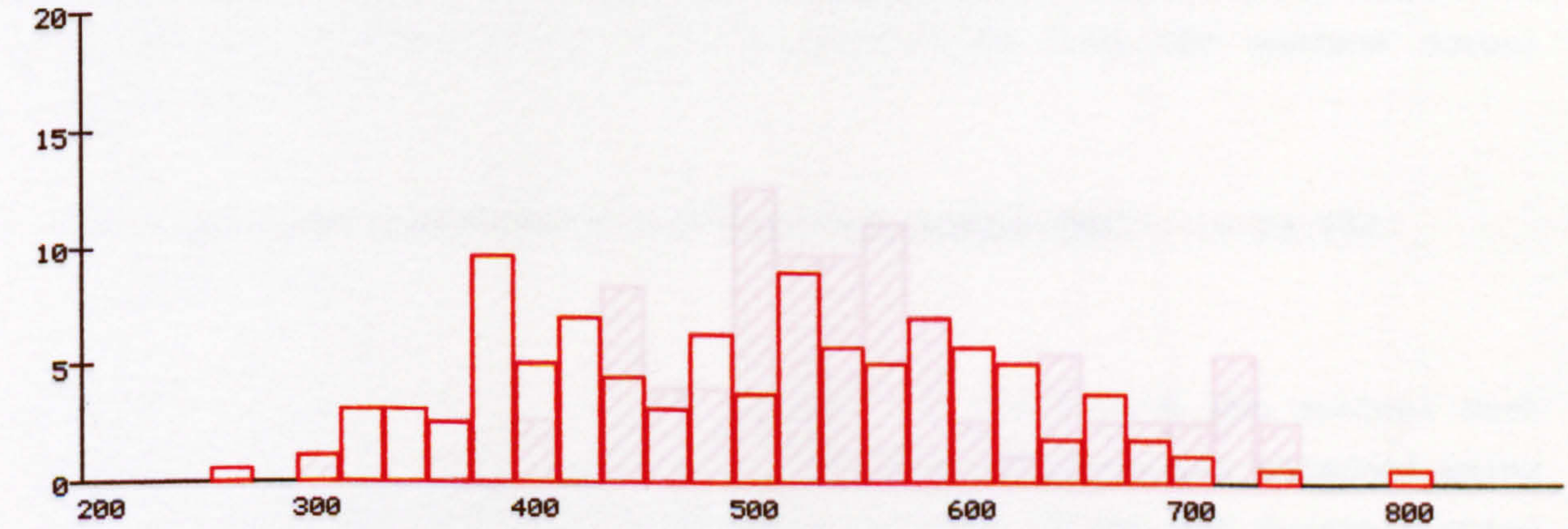


Length (mm)

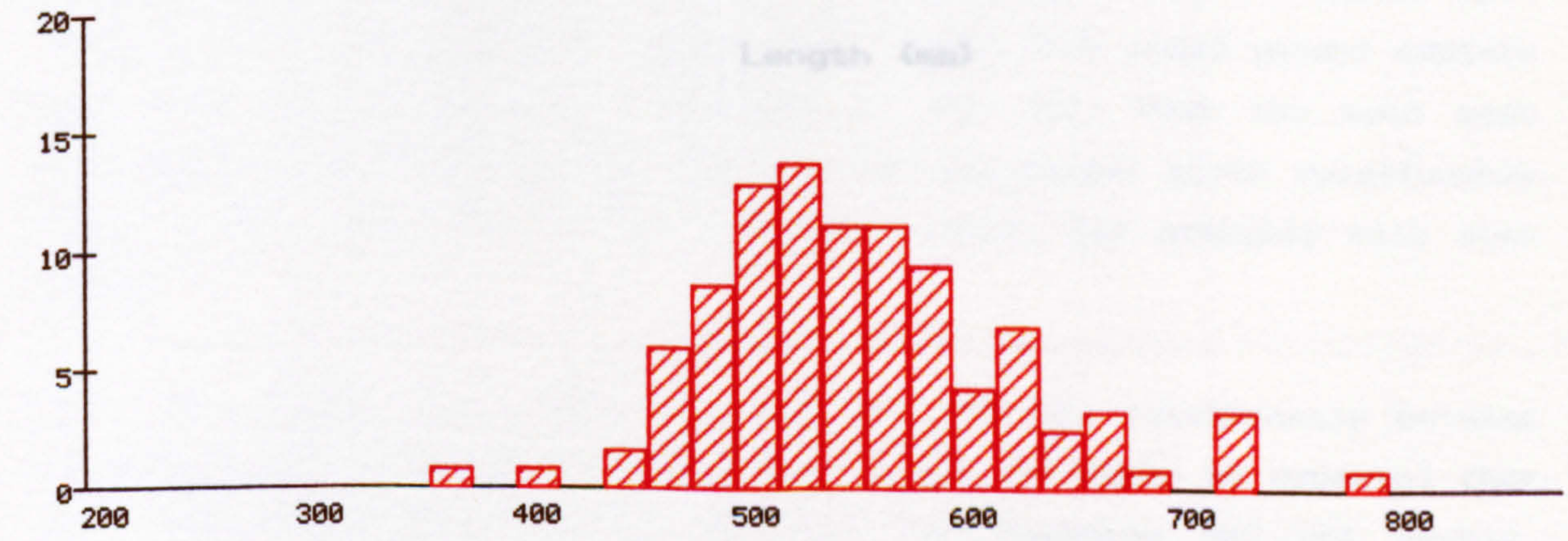
(h) Seine net in 1991



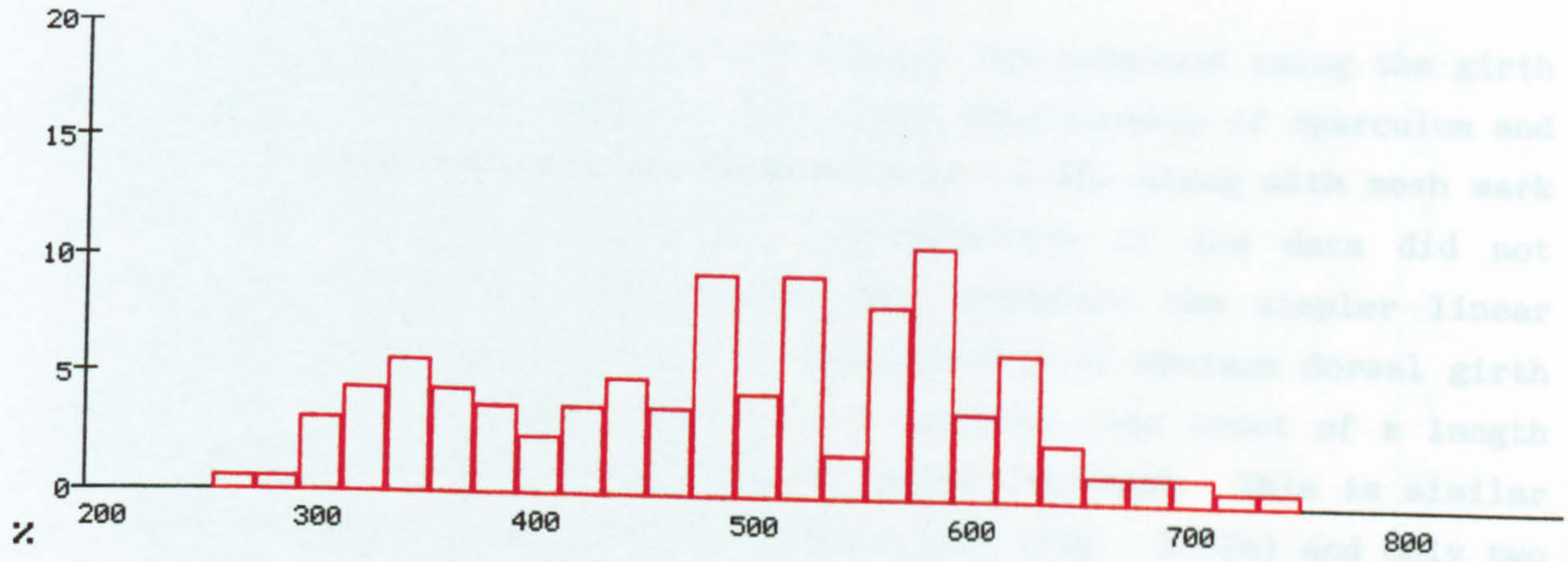
(i) Rods in 1991



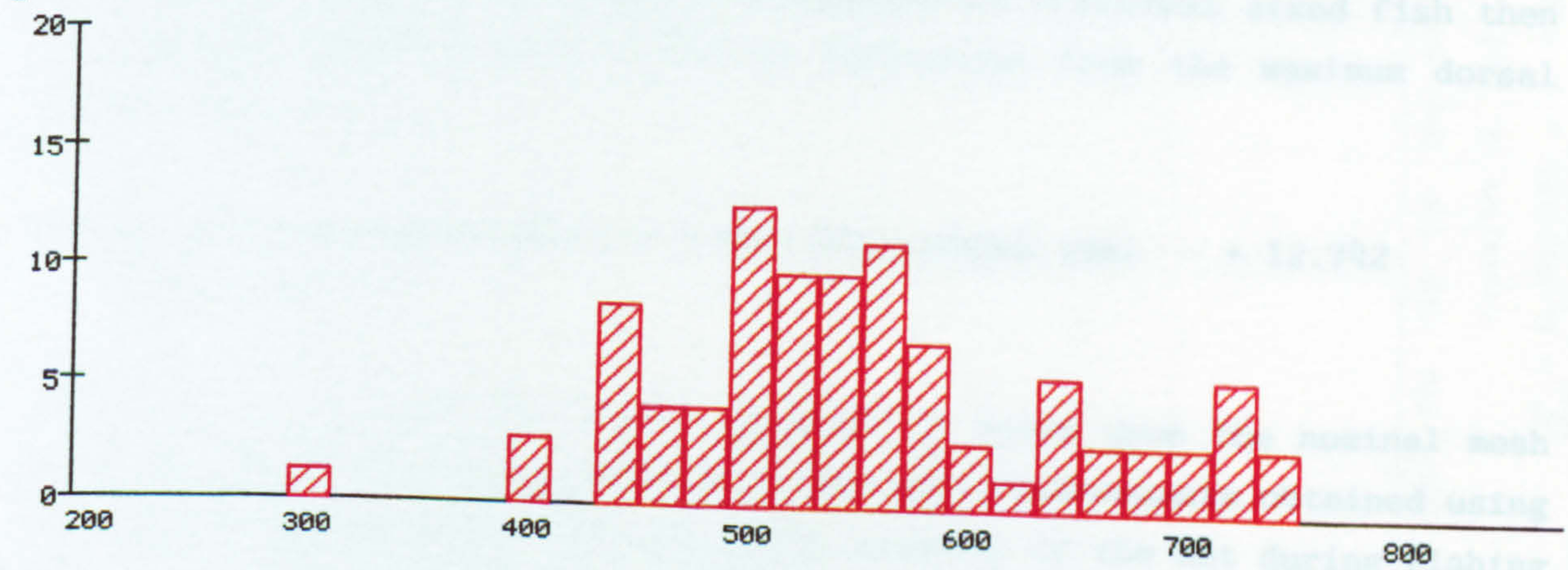
(j) Coracle in 1991



(k) Rods in 1992



(l) Coracle in 1992



distribution was mainly between 400 and 820 mm (Fig. 2.17h), with isolated fish of below 400 mm and none below 320 mm. Using the 102 mm mesh size therefore excluded nearly all whitling and will also have excluded some of the smaller .SM+ and .1+ fish.

The mesh selection in the seine net fishery was examined using the girth measurements collected in 1991. The linear relationship of operculum and dorsal girth measurements are shown in Fig. 2.18, along with mesh mark girths from 37 fish (logarithmic transformation of the data did not improve the correlarion coefficients and therefore the simpler linear equations have been used). From the relationship of maximum dorsal girth and nominal mesh perimeter (203 mm), in general, sea trout of a length less than 373 mm would be expected to escape unharmed. This is similar to the empirical distribution observed in 1991 (Fig. 2.17h) and only two sea trout were meshed below the range of mesh measurements recorded.

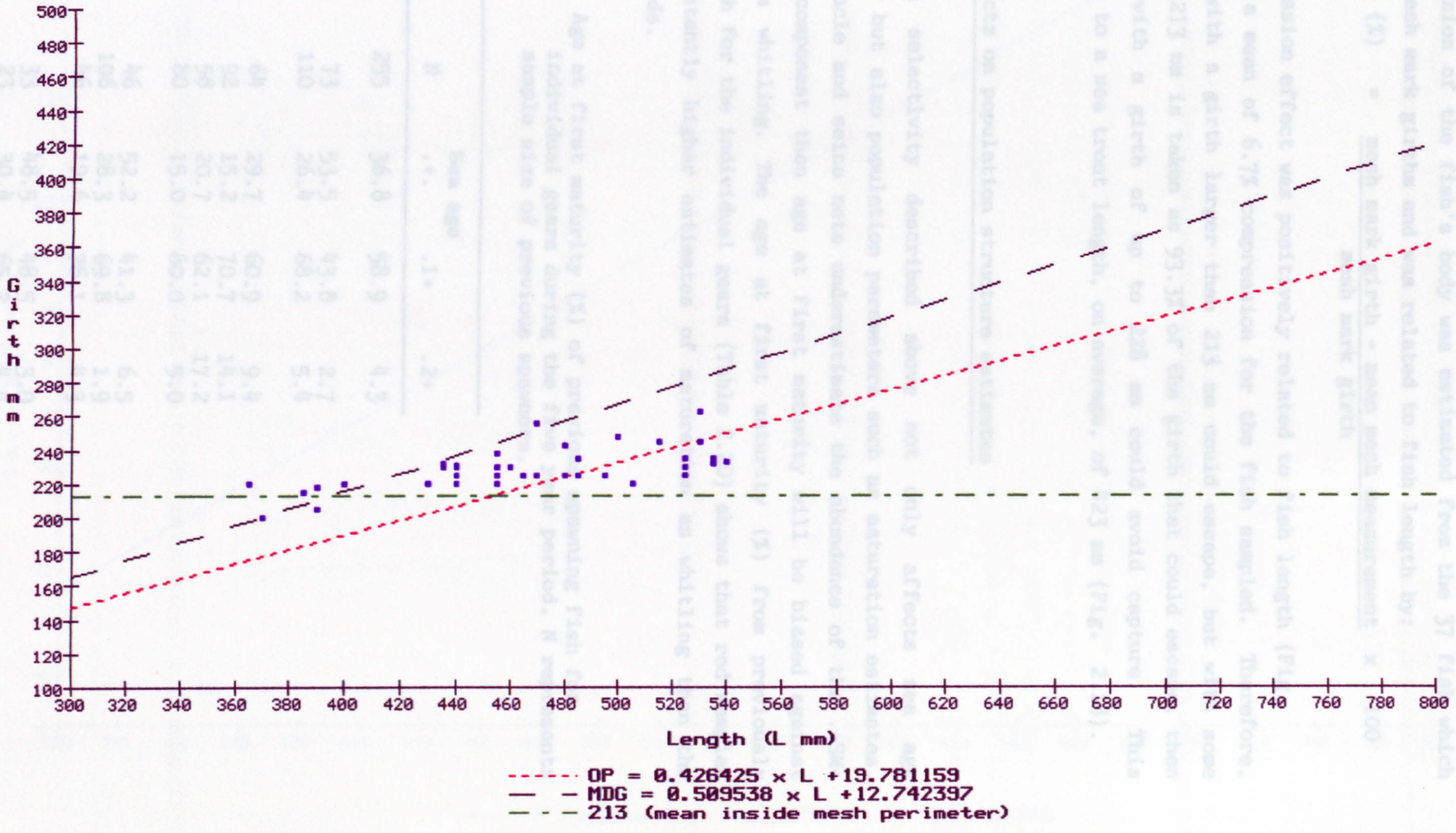
If stock management required the escapement of different sized fish then the appropriate mesh size could be determined from the maximum dorsal girth relationship:

Inside Mesh perimeter (mm) = 0.510 x Fish Length (mm) + 12.742
(Dorsal Girth)

The above assessment of fish escapement is based upon the nominal mesh size used (203 mm inside perimeter). The mesh measurements obtained using the push wedge assesses the stretching property of the net during fishing and thus the modification upon nominal size. The push wedge measurements were doubled and an average inside mesh perimeter of 213 mm (range 206-220 mm) was recorded, therefore displaying a 4.9% increase upon nominal size. This therefore means that larger fish could escape capture but they would probably be damaged by the net. From the mean mesh measurement recorded (213 mm) and the maximum dorsal girth relationship (Fig. 2.18), sea trout of 395 mm could escape, but probably with some damage.

From the average mesh perimeter (213 mm) and the relationship between opercular girth and fish length (Fig. 2.18), it would be expected that sea trout >455 mm would be captured free-swimming and not meshed. However, the compression of the fish's body allows meshing of larger girths.

Fig. 2.18 The relationship between fish length (mm) and maximum dorsal girth (MDG; p<0.001, r2=0.90) and opercula girth (OP; p<0.001, r2=0.91). Observed mesh mark girths are included (dots) as well as the mean inside mesh perimeter (213) mm



The compression of the fish's body was estimated from the 37 fish which displayed mesh mark girths and was related to fish length by:

$$\text{Compression (\%)} = \frac{\text{mesh mark girth - mean mesh measurement}}{\text{mesh mark girth}} \times 100$$

This compression effect was positively related to fish length (Fig. 2.19) with a mean of 6.7% compression for the fish sampled. Therefore, sea trout with a girth larger than 213 mm could escape, but with some damage. If 213 mm is taken as 93.3% of the girth that could escape, then sea trout with a girth of up to 228 mm could avoid capture. This corresponds to a sea trout length, on average, of 423 mm (Fig. 2.18).

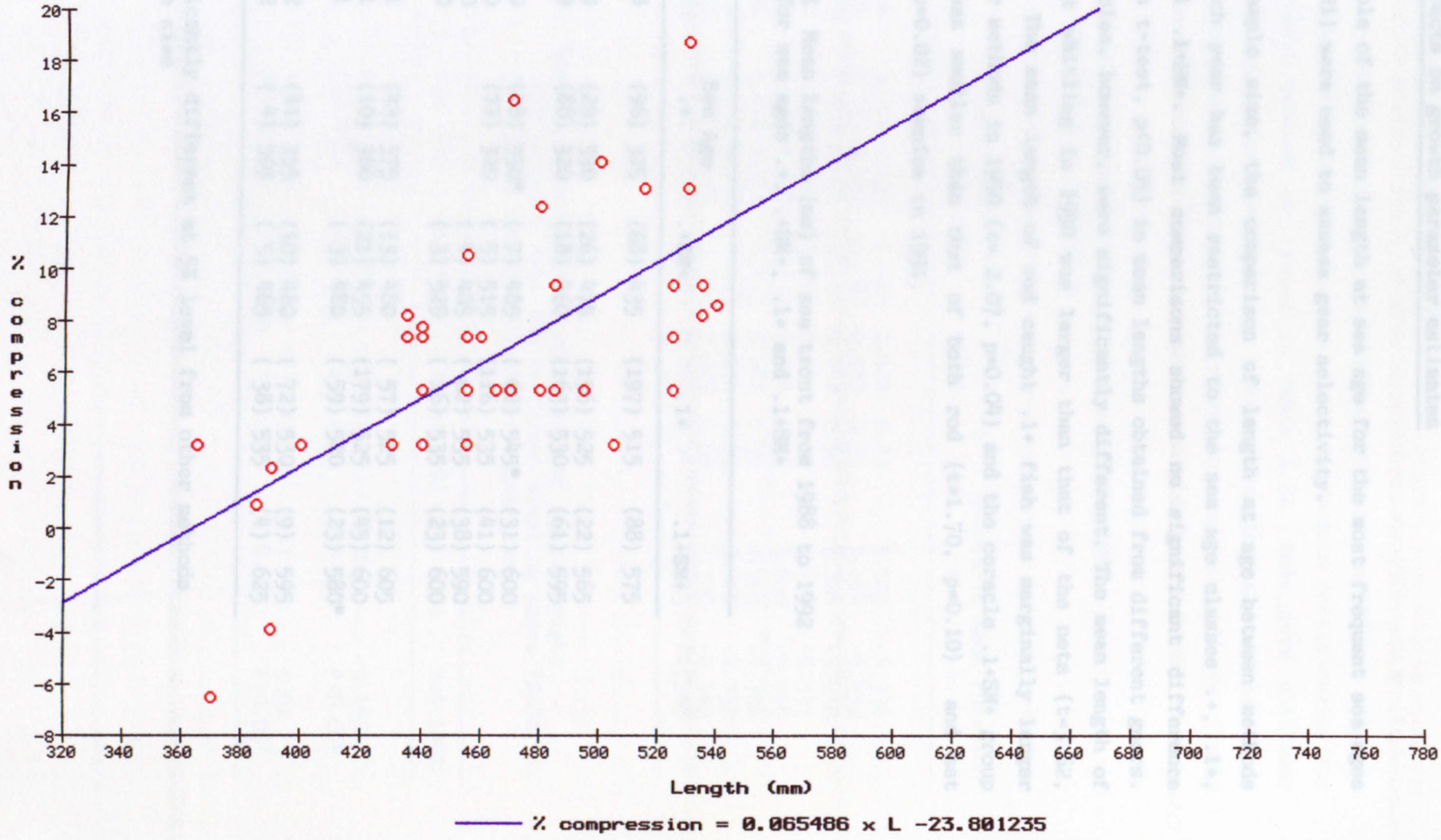
2.4.9 Effects on population structure estimates

The length selectivity described above not only affects sea age selectivity but also population parameters such as maturation estimates. If the coracle and seine nets underestimate the abundance of the .+SM+ sea trout component then age at first maturity will be biased against maturity as whitling. The age at first maturity (%) from previously spawned fish for the individual gears (Table 2.20) shows that rod samples gave consistently higher estimates of maturation as whitling than the other methods.

Table 2.20 Age at first maturity (%) of previous spawning fish for individual gears during the five year period. N represents sample size of previous spawners.

Method		N	Sea age .+.	.1+	.2+
Trap	88	255	36.8	58.9	4.3
Rod	89	73	53.5	43.8	2.7
Net	89	110	26.4	68.2	5.4
Rod	90	64	29.7	60.9	9.4
Net	90	92	15.2	70.7	14.1
Trap	90	58	20.7	62.1	17.2
Coracle	90	80	15.0	80.0	5.0
Rod	91	46	52.2	41.3	6.5
Net	91	106	28.3	69.8	1.9
Coracle	91	46	19.6	76.1	4.3
Rod	92	33	48.5	48.5	3.0
Coracle	92	23	30.4	65.2	4.4

Fig. 2.19 The compression of sea trout (%) as the difference between the mesh mark girth and the mean inside mesh perimeter, divided by the mesh mark girth. ($p<0.001$, $r2=0.52$)



2.4.10 Effects on growth parameter estimates

A sub-sample of the mean length at sea age for the most frequent sea ages (Table 2.21) were used to assess gear selectivity.

Due to sample size, the comparison of length at age between methods within each year has been restricted to the sea age classes .+, .1+, .+SM+ and .1+SM+. Most comparisons showed no significant difference (Student's t-test, $p<0.05$) in mean lengths obtained from different gears. Three samples, however, were significantly different. The mean length of rod caught whitling in 1990 was larger than that of the nets ($t=3.62$, $p<0.01$). The mean length of rod caught .1+ fish was marginally larger than other methods in 1990 ($t= 2.07$, $p=0.04$) and the coracle .1+SM+ group of 1991 was smaller than that of both rod ($t=1.70$, $p=0.10$) and net ($t=2.35$, $p=0.02$) samples in 1991.

Table 2.21 Mean lengths (mm) of sea trout from 1988 to 1992 for sea ages .+, .+SM+, .1+ and .1+SM+

Method		Sea Age .+	Sea Age .+SM+	Sea Age .1+	Sea Age .1+SM+
Trap	88	(96) 325	(68) 435	(197) 515	(88) 575
Rod	89	(29) 330	(26) 475	(156) 525	(22) 565
Net	89	(88) 320	(18) 460	(163) 530	(61) 595
Rod	90	(48) 350*	(7) 485	(62) 545*	(31) 600
Net	90	(77) 320	(5) 515	(116) 535	(41) 600
Coracle	90		(4) 485	(62) 535	(38) 590
Trap	90		(3) 525	(16) 535	(23) 600
Rod	91	(43) 375	(13) 480	(57) 525	(12) 605
Net	91	(10) 380	(21) 455	(179) 525	(45) 600
Coracle	91		(3) 480	(59) 520	(23) 580*
Rod	92	(41) 355	(10) 480	(72) 530	(9) 595
Coracle	92	(4) 395	(5) 465	(38) 535	(4) 625

* Significantly different at 5% level from other methods
() Sample size

2.4.11 Length weight relationships

Weights were obtained for a total of 1493 sea trout from 1989 to 1992 and the predictive and geometric mean functional regression relationships for the individual methods are shown in Table 2.22. Rods gave shallower regression lines than the net or coracle samples and analysis of covariance demonstrated that these differences were significant (Table 2.22a).

Table 2.22. Length Weight Relationships for 1989 to 1992
Log W = a + b x Log L

Method		Predictive b	95%CI	a	95%CI	r^2	Geometric Mean Function b	a
Coracle	90	3.376	0.154	-5.985	0.424	0.93	3.503	-6.336
Net	91	3.133	0.068	-5.280	0.187	0.97	3.184	-5.432
Net	89	3.090	0.171	-5.190	0.468	0.92	3.224	-5.530
Coracle	91	3.005	0.180	-5.280	0.494	0.91	3.156	-5.337
Coracle	92	3.002	0.144	-4.937	0.394	0.96	3.060	-5.098
Rod	91	2.878	0.097	-4.633	0.261	0.96	2.905	-4.704
Rod	92	2.871	0.083	-4.601	0.224	0.97	2.919	-4.732
Rod	90	2.638	0.084	-3.974	0.227	0.95	2.700	-4.142
Rod	89	2.629	0.050	-3.932	0.266	0.91	2.753	-4.267

Table 2.22a Results of analysis of covariance of length/weight relationships from the different sampling gears (Rods, R; Net, N and Coracle, C). * Not significant

	RvN 1989	RvC 1990	RvNvC 1991	NvC 1991	RvC 1992
Slope					
F	22.41	8.73	51.39	2.46	8.193
P	P<0.0001	P<0.001	P<0.0001	P>0.05*	P<0.005
Intercept					
F	16.65	51.94	8.763	3.301	2.02
P	P<0.0001	P<0.0001	P<0.0005	P>0.05*	P>0.05*

2.5 Discussion

Section A Biological Characteristics

2.5.1 Age Structure of adults

The difference between the total number of age classes obtained in the five year period (22-31) is partly accounted for by the increase in the smolt age groups recorded, but the number of sea age groups also differed (12 in 1992 and 19 in 1988). This was due to the inclusion/exclusion of uncommon sea age groups in the latter. The overall figure of 44 different age classes is more than the 28 recorded for the Tywi by Harris (1970), but these are not directly comparable since the probability of finding different age classes is a function of sample size, and also possibly the range of gears used. This study examined considerably more fish (2689) than the 347 fish sampled by Harris (op cit) and therefore a higher number of different age classes would be expected.

The large number of age classes demonstrates the plasticity of the age structure/life strategy of the Tywi sea trout stock, but in terms of numbers the stock is still highly dependent on a small number of age classes. From the 2689 fish sampled, 60.2% were maidens of .+ or .1+ sea age, and this represented only 7 different age classes. However, the remaining 39.2% of older fish may still account for a large proportion of the spawning stock due to their increased size and proportion of females.

The majority of fish sampled were maidens, and .1+ fish were more abundant than .+ sea trout, indicating a low tendency to return as whitling. However, the gear selectivity against whitling in the net samples prevents quantitative abundance estimates between sea age groups. From the overall sample, 63.3% of sea trout were maidens (.+,.1+,.2+) and concomitantly 36.7% were previous spawners. This compares with 53.1% maidens and 46.9% previous spawners recorded for the Tywi by Harris (1970). Whilst this may indicate a reduction in survival or increase in exploitation between the two sampling periods, it may also be explained by the variation in the abundances of individual sea age classes affecting statistics, since maiden/previous spawning proportions observed in 1988 and 1990 were similar to those found by Harris (op cit).

The annual variation in individual sea age class abundances within

samples affected the figures for relative contribution of sea ages within previous spawners, such that the percentage of previous spawners with >=3SM's ranged from 4.9%-18.0%, with an average for the five-year period of 13.8%. This is similar to the 14.7% quoted for the Tywi previously by Harris (op cit). The contrasting figures between 1988 and 1992 illustrate the need for sampling over several years to eliminate the effect of annual variation when trying to describe a sea trout stock. The proportion of spawners surviving three spawning migrations on the Tywi is relatively high compared with other rivers: c.f. 4.2% for Teifi, 8.3% for the Dyfi, 19.5% for the Rheidol (Harris, 1970), 8.5% for the Dee (I. Davidson pers. comm.), 18.4% for the Tawe (D. Mee pers. comm.), 5.1% from the Connemara, Ireland (Fahy & Nixon, 1982) and 10.5% for the Tay, Scotland (Walker, 1984).

Evidence from fish which exhibited spawning marks suggests that once maturity is reached Tywi sea trout spawn annually, a result consistent with other studies (Lamond 1916, Nall 1930, Pratten & Shearer 1983). Nevertheless, three fish were found which had apparently matured as post-smolts, but had not spawned in the following winter. This uncommon behaviour has been recorded from the Tweed and Coquet (Solomon, 1993).

The age at which sea trout first become mature suggests that .1+ maidens are the most important group with 62.9% of previous spawners having matured after one year at sea. The proportion of fish maturing as .+ over the two years was 30.9%. This conflicts with the premise that .+ may be more abundant. This may be due to three main reasons.

1. Not all whitling mature and spawn. (LeCren, 1985)
2. The spawning mortality of whitling is greater than that for fish maturing at an older sea age.
3. The .SM+ group were undersampled due to mesh size selectivity.

2.5.2 Age structure of juveniles

Four smolt ages were found in total (1, 2, 3 & 4) but the 1- and 4-year-old smolts constituted only 3.3% of the overall five-year sample. However, the increase in 1-year olds in 1991 and 1992 demonstrated the within-stock variation that can take place and again highlights the need to sample over several years to describe the MSA of a stock. The overall MSA of 2.24 was slightly lower than the 2.31 recorded for the Tywi by

Harris (1970). This does not necessarily mean a shift in smolt age composition since environmental factors, shown to influence MSA (section 2.4.7), during Harris's studies are unknown, and a MSA of 2.31 is within the range of annual estimates in this study.

The decrease in MSA in latter years (1991-1992) has implications for other studies undertaken during the same period. Many sea trout studies were carried out in the early 1990s (Solomon, 1993), partly in response to the need for information following crashes in Ireland in 1989 (Anon., 1990, 1991). Care should be taken when interpreting the MSA values obtained during this period since they may be lower than 'normally' expected due to the climatical factors demonstrated in this study. Indeed, Solomon (1993) observed that on several English and Welsh rivers the MSA's from 1990 onwards were lower than earlier studies on the same river.

The variation in smolt ages also has implications for other stock assessment methods. Each year electrofishing surveys are undertaken in the Tywi catchment (RJSMP, Chapter 7) to establish juvenile densities. If parr smoltify as 1-year olds they will not be present during the survey period as 1+ parr, and therefore parr densities may be lower than expected, leading to the assumption that production was lower.

The smolt age composition of each smolt year class in this stock, and in many other studies, are described from scale reading. It should be recognised that this may not reflect the exact smolt age composition of the smolt run itself if there is a variation in post-smolt survival rates between different smolt ages, but Harris (1970) stated that the two smolt composition estimates can be similar. Smolt ages from scale reading for the stock may also differ from that of the smolt run due to the variation of smolt ages between returning sea ages. The tendency for older smolts to return as whitling rather than maiden .1+/.2+ fish in this study was demonstrated for the Tywi by Harris (1970), and has been shown for other rivers (Walker, 1987; Shearer, 1987). If smolts returning as whitling demonstrate higher survivorship than for .1+ maidens, due to a much lower sea residence, this may over-represent the proportion of older smolts. Similarly, there is evidence to suggest that older and larger smolts have better survivorship than their smaller counterparts (Potter, 1985) and this may be due to reduced sea water tolerance of younger smolts (Morgan & Paveley, 1993). Thus the proportion of one-year-old smolts in

returning adults may be lower than that of the smolt run.

The differing MSA between sea age groups, i.e. whitling and non-whitling, means that determining the relative abundance of each sea-age group is necessary to truly describe the stock MSA in an unbiased manner.

2.5.3 Growth Rates

The River Tywi sea trout stock is amongst the fastest growing in the UK (Harris, 1970, Fahy, 1978, Harris and Winstone 1987, Solomon, 1993). Stocks from the N.W.England, N.Wales, S.W.England and W.Scotland display lower mean length for age, with a mean length of .1+ sea trout typically between 360-440 mm. River Tywi fish are amongst the largest for their age and are comparable to those of the Tweed and Coquet systems, which are also noted for their rapid growth (Nall, 1930; Solomon, 1993).

The mean lengths of the major sea age groups in this study, viz., .+,.1+ and .1+SM+ (320 mm, 530 mm and 595 mm, respectively) were slightly larger than those found by Harris (1970) with 299 mm, 512 mm and 562 mm, respectively. The length weight relationship showed that sea trout were short for a given weight (Fahy, 1978) and there was no evidence in either year of the thin, emaciated or sea lice infested fish found contemporarily in Ireland (Anon., 1990, 1991).

It is recognised that smolt age is primarily regulated by freshwater growth rate (Okland *et al.*, 1993; Jonsson & L'Abee-Lund, 1993), which in turn is primarily regulated by the density-independent factors of water temperature and egg/alevin size (Elliott, 1984, 1985a, 1985b). Growth rates may also be influenced by the water chemistry or productivity of a stream, and this may differ within a river catchment (Egglishaw & Shackley, 1985). There seems to be little evidence of a relationship between trout growth and trout density (Egglishaw & Shackley, 1985; Elliott, 1985b), although some authors have indicated that trout growth may be related to juvenile salmon density, as well as temperature (Egglishaw & Shackley, 1977, 1985).

Many authors have indicated the importance of temperature in modifying smolt age. Saltveit (1990) demonstrated how slower growing parr in the lower section of the River Surna, below a hypolimnetic reservoir input,

smolted one year later than parr upstream of the cold water discharge. Mann *et al.* (1989) demonstrated that trout parr downstream of a chalkstream spring, with a constant 10 °C water temperature, continued to grow over winter, whereas no growth increments were recorded from fish in neighbouring streams. Jonsson & L'Abee-Lund (1993) demonstrated how MSA increased with latitude, and Elliott *et al.* (1993) inferred that this was due to differences in temperature regimes. These studies show how smolt age composition can change between rivers, but this study takes this one step further and demonstrates the within river variations in smolt age composition from variations in annual temperature regimes.

The relationship between MSA and the temperature regimes encountered by the smolts in the years prior to smolting was also observed by Fahy (1980), who related smolt production to the 'growing season'. From this study it is hypothesised that the mild winters and warm summers of 1988-1990 increased the development rates of incubating eggs (Crisp, 1992) and parr such that smolt ages decreased. The impacts of the abnormal climatic regime of 1988-1990 upon the environment is described by Cannell & Pitcairn (1993), and Weatherley *et al.* (1991) demonstrated the increased growth rates of trout in mild winters and summer droughts in the upper reaches of the River Tywi. These latter authors found that trout growth increased when water temperatures were up to 3 °C above 'normal' but that growth would be retarded at 4.5 °C above 'normal'. This is probably related to the optimum temperature preference of trout since the temperature would be towards the higher range of trout feeding (4-19 °C; Elliott, 1975) and above the optimal temperature of 13 °C (Elliott, op cit). The depressed parr growth observed for the 1988 year class, even though they experienced the highest cumulative temperature, is probably explained by the high temperatures of the 1989 drought being above the optimum growth temperature.

2.5.4 Sex Ratios

It is common in adult sea trout populations to find a large bias towards females (Nall 1930, Harris 1970, LeCren 1985, Campbell 1977). However, the observation that different components of the stock displayed different sex ratios, i.e. whitling have a sex ratio close to unity and older sea ages were dominated by females, makes it difficult to give a sex ratio for the stock unless the relative abundances of the different

sea age groups are known. Clearly, different life history strategies may be employed by the sexes for spawning (Chapter 3) and this can influence the sex ratio of the stock. The overall male:female ratio obtained in this study of 1:2.05 (20% whitling) is similar to the 1:2.012 quoted for the Tywi by Harris (1970).

2.5.5 CPUE and Abundance

The CPUE data show that Tywi sea trout enter the river early in the year, forming an early spring run of large previous spawners and maiden .1+ and .2+ fish in April and May. This is relatively common in Wales, where several rivers are considered to have good runs of sea trout in the first two weeks of May. In June the proportion of previous spawners declines with the introduction of the first post-smolts in the catch and the proportion of these post-smolts increases during the summer. The Tywi run of sea trout is therefore as early, if not earlier, than any other stock where information is available (Solomon, 1993). The early timing of the run may reflect the longevity of the stock, with the high proportion of large multiple spawning sea trout.

Peak catches were recorded in the rod fishery in July of each year, some 1-2 months after the estuary fishery. Whilst some delay may be expected due to the time needed for fish to ascend the river, the primary reason for the peak catches for the rods in July is the increased effort during this period, fishing method and also the arrival of whitling (Chapter 4).

A late run of sea trout was observed after both the commercial netting and rod seasons in 1989 and 1990 (although sampling was curtailed earlier in 1990 and a full assessment was not possible). The majority of these were fresh 'silver' fish, although the epidermis had thickened considerably. Many still bore sea lice (Lepeophtheirus salmonis), considered to be a good indicator that the fish had not been in fresh water for any length of time (FFLP, 1990). Whilst this shows that a late run of fish occurs there was strong evidence of delayed entry in 1990 and 1991 associated with low flows (Clarke *et al.*, 1990, D. Clarke pers. comm.), with several fish radio tagged in April and May not entering fresh water until increased river flows in October. This suggests that part of the late run may therefore be delayed spring migrants and the extent of the late run may depend upon river flows in the preceding

spring/summer. The number of fish entering during the fishing seasons (commercial and recreational) may not therefore reflect the number of fish spawning and this stresses the need for temporal sampling of the sea trout run. On the Tywi the run extends at least from March to December, with unknown, possibly small, numbers of fish entering in January/February.

Section B

2.5.6 Gear selection

Describing the biological characteristics of any stock necessitates sampling across the range of age classes. Different sampling gears may not exploit all age classes and this may be due to forced regulation, e.g. the use of mesh size to restrict sizes exploited by nets. This work showed that differences occurred between the different gear in the size distribution sampled.

The trap retained fish of >250 mm in 1988 and was therefore deemed unselective while it was operating. The trap results of 1990 illustrate the bias which can be introduced by short-term sampling, when particular age classes may be unavailable.

The nets in 1989 and 1990 were less obviously selective in their length distribution due to the small meshed bunt, but the relative abundance of the smaller sea-age classes (.+,.SM+) were probably under-represented due to their escape through the large-meshed wings of the net. The net sample of 1991 and the coracle samples were selective due to mesh size, which is a standard method of restricting exploitation (Fahy, 1984; Solomon, 1993). Pratten and Shearer (1985) found a similar exclusion of sea trout below 350 mm when using a 90 mm mesh size (180 mm perimeter) on the North Esk.

Further factors, in addition to mesh size *per se*, affect the size of fish retained. The stretching properties of the net and the compression of the fish's body allow variation in escapement and mesh selection models have been proposed accounting for these factors (Sechin, 1969; Clarke & King, 1986). Solomon (1993) stated that the stretching properties of the meshes are particularly important with monofilament gill nets but the less elastic twine (nylon) used in seine nets provide a more defined size. Although the compression of the fish's body could allow the escapement

of some fish with a girth larger than the stretched mesh size, in practice, the base of the dorsal fin often prevented passage through the mesh.

In this study the mesh measurements were taken from seine nets with a uniform nominal mesh size (102 mm) and twine thickness (4z) throughout the length of the net. A different twine thickness may be used at the fishermen's discretion, and varying twine thickness (z) may alter the stretching properties of the meshes and hence the chance of escapement for meshed fish. The choice of twine z used by the fishermen is a balance between increased twine thickness, providing increased strength, and minimisation of the drag effect of the net. Some fishermen may use a stronger twine in the bunt section of the net compared to the wings, since this is the section where fish may struggle most as the net is brought ashore.

Sea trout may also be retained through methods other than meshing or free-swimming, notably tangling by the teeth in a mesh (Hamley, 1975) or through the rapid retrieval of a net resulting in entrapment by meshes stretched obliquely (Fahy, 1984). These latter two methods result in the capture of sea trout that would usually pass through the meshes. Several of the whitling from the seines in 1991 were tangled by the teeth.

The mesh size in the Tywi fishery allows the escapement of whitling but from examination of the age structure of previous spawners, appreciable maturation is at the maiden .1+ sea age (section 2.4.5). Maiden .1+ fish are therefore likely to be important to the spawning stock of the River Tywi due to their increased size (fecundity), increased female ratio and proportion of maturation. Although over exploitation in the fishery could impact significantly upon the spawning stock, current exploitation estimates (D. Mee pers. comm.) do not indicate that this is the case; a case supported by the longevity of the Tywi stock.

The rod fishery recorded sea trout up to 800 mm and was therefore able to sample all age classes, although some bias in the abundance of different sea age groups may take place. Several anglers taking part in this study reported the releasing of some .+ sea trout unsampled, consequently the relative abundance of whitling was underestimated. It has been noted elsewhere (LeCren, 1985) that rods may underestimate the numbers of large fish present. This may be because larger fish are harder to catch but it

may also be due to the angling method (Evans, 1990). A further reason may be the location of the fishery within the river system. On the River Fowey (H. Sambrook pers. comm.) the major rod fishery is concentrated in the middle and lower river and many large sea trout migrate rapidly passed these fisheries. Radio tracking of large early season fish (D. Clarke pers. comm.) on the River Tywi has shown that many fish migrate to the middle and upper river over a short time period. On the Tywi, the middle and upper river coincides with the location of a major part of the rod fishery, allowing exploitation of these fish.

The majority of comparisons between mean length at sea age were not significantly different and therefore any method may be used to gain this information as long as the age class is exploited by that gear. The difference in mean length of the whitling samples of 1990 was explained by the rod sample having a higher ratio of three to two-year-old smolts than the net sample (Section 2.4.7 shows that three-year-old smolts produced larger whitling than two-year olds.)

The difference in regression equations of length and weight obtained from the various gears is explained by the age classes exploited. The net and coracle samples include few or no whitling and therefore have a high proportion of large multiple spawning sea trout, approaching their asymptotic length. Consequently data are biased towards larger fish giving a higher gradient. The rod sample has a large proportion of whitling which tend to bias the data towards smaller lengths. The lower weight of sea trout for a given length from the rod fishery may also be influenced by the period of time spent in-river, since weight loss would accompany the cessation of feeding in fresh water (Harris, 1970).

As well as the gear-related differences in catches *per se*, there may also be some spatial and temporal effects that alter the catches between the gears. The delaying of migrants in the spring due to low flows could result in the estuary gears sampling fish that may not be available to the rod fishery or a temporary main river trap. This does not necessarily mean that the estuary nets give better results since if the fish displayed oscillatory behaviour with the tide, i.e. they did not migrate in to fresh water on a single tide, then the nets may over-represent some of the stock component due to several chances of capture. These effects will result in biased relative abundances of the different stock components. It should also be recognised that the estuary seine net

fisheries are restricted to certain states of the tide, and if there is a difference in migratory patterns between stock components this could also bias the results. A further complicating factor with estuarial sampling is the possible effect of a mixed stock fishery, such that fish captured may not be from the targeted river. The radio-tracking studies on the Tywi suggested that very few sea trout from the seine net fishery were of non-Tywi origin (D. Clarke pers. comm.).

From the gears described in this study the best approach to a quantitative assessment of adult stocks would be to trap the river above the head of tide, ensuring that the trap can effectively capture all sizes of fish. Second, a small meshed net in the estuary may be used if there is no evidence that a mixed fishery exists. Both methods would require sampling throughout the run period, which on the River Tywi would mean operating from March to December. These methods would not necessarily require continuous operation from March to December as long as sampling was temporally stratified.

Assessment by other gears may provide biased quantitative estimates of sea age abundances and therefore qualitative data only. The rod fishery sampled all the common age classes and would therefore be a third choice of sampling, preferably using an angler log book scheme to note the abundance of fish released. The sampling of commercial fisheries would provide good qualitative data for the sea ages exploited and may indicate whether the rod fishery is sampling the large fish in a population.

The importance of stratified temporal sampling has been stressed previously, and sampling by the rods and commercial nets would therefore introduce some bias, but the extent of this needs to be taken into context. On the Tywi the majority of fish enter between March and July, within the fishing seasons. This may not, however, be the case on other rivers e.g. Coquet where peak entry of fish is between September and November (Solomon, 1993).

Qualitative data such as length at age is available from all of the gears described. However, given the operational resources needed for trapping or netting, the qualitative data from rods or commercial fisheries may be obtained with apparent ease and little manpower. These qualitative data provides useful information, e.g. to distinguish stocks of different juvenile and adult growth rates and longevity.

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Appendix 2.1 The back-calculated growth rates of parr (year 1-2) to accompany section 2.4.7.

Each fish was categorised according to its year of birth and from the samples of aged whitling and 1+ maidens the median growth rate of parr during each calendar year was calculated (Table A2.1).

Table A2.1 Median back-calculated growth rates.

Year class	Median growth rate (mm/yr) whitling	1+ maidens
1984		46 *
1985	62 *	88
1986	102	100
1987	106	92
1988	104	73
1989	110	1124**
1990	118 **	

* Only 3-year olds sampled during study period
** Only 2-year olds sampled during study period

Due to the absence of age classes at either end of the sampling period the calculated growth rates for the 1984/1985 year classes would be lower than expected since none of the faster growing 2-year olds were included. Similarly, for the 1989/1990 year classes the median growth rate is higher than expected since none of the slower growing 3-year olds were sampled.

Weighted estimates of median parr growth were calculated in relation to the results of years with complete age class structure. The worked example for whitling is given here:

Whitling overall median growth rate of 2 year olds (n=71) = 124 mm
Whitling overall median growth rate of 3 year olds (n=29) = 65 mm

For the 1985 year class 3 year old growth rate = 62 (Table A2.1) but 2 year olds were missing:

Estimated median growth rate of 2 year olds = 62 x (124/65)
= 118 mm

To obtain the median growth rate of the year class an assumption was made that the proportion of 2:3 year olds was similar to the smolt age composition observed for the river in 1988 (52% 2s, 46% 3s).

Therefore the overall median growth rate for the 1985 year class was:

$$MGR = \frac{(118 \times 52) + (62 \times 46)}{98} = 92 \text{ mm}$$

For the 1990 year class 2 year old growth rate = 118 mm (Table A2.1) but 3-year olds were missing:

Estimated median growth rate of 3 year olds = 118 x (65/124)
= 62 mm

To obtain the median growth rate of the year class an assumption was made that the proportion of 2:3 year olds was similar to the smolt age composition observed for the river in 1992 (75% 2s, 13% 3s).

Therefore the overall median growth rate for the 1990 year class was:

$$MGR = \frac{(118 \times 75) + (62 \times 13)}{88} = 110 \text{ mm}$$

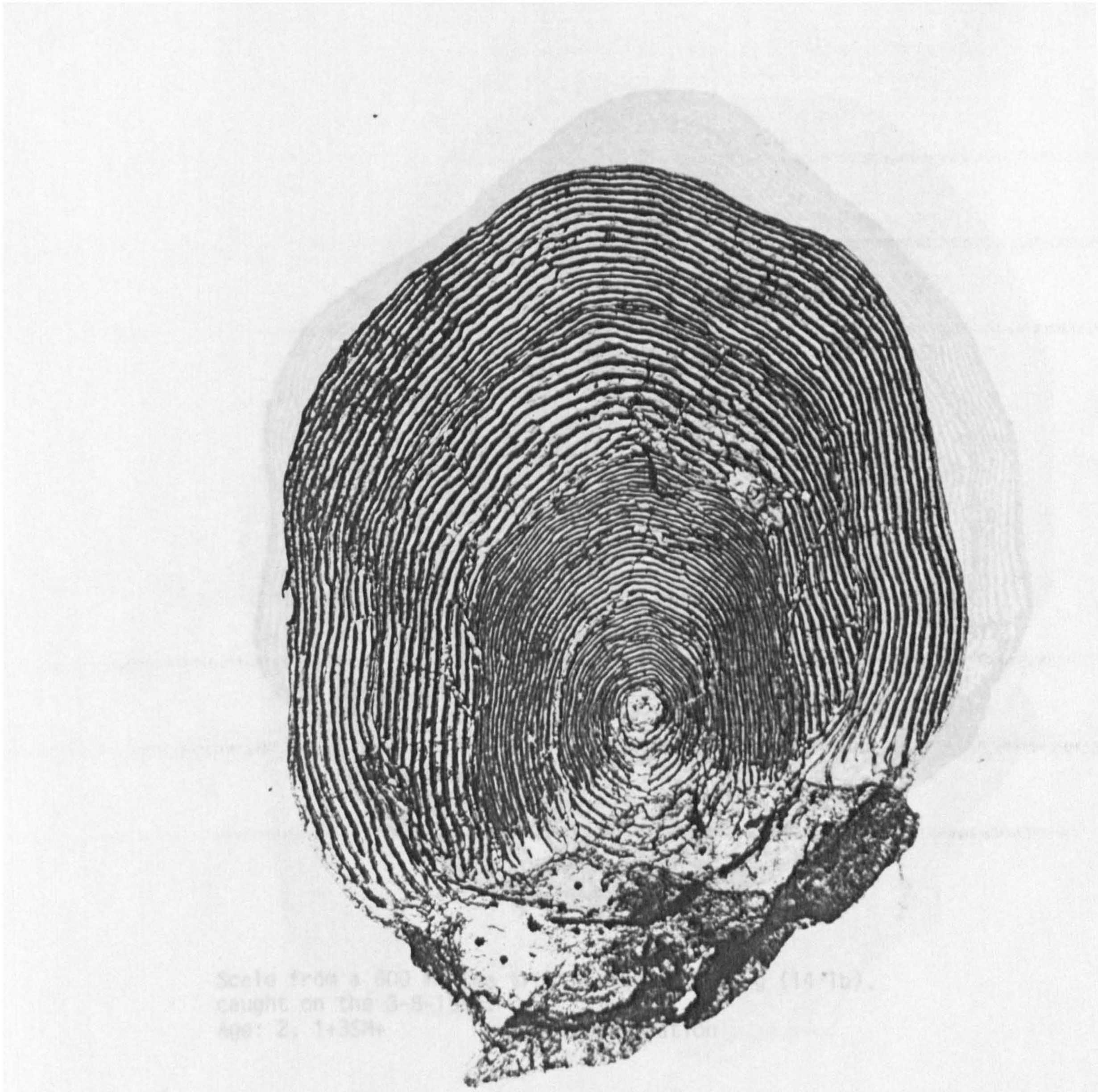
The median growth rates, weighted for year class sampling, used in the graphs in section 2.4.7 are given in Table A2.2.

Table A2.2 Median back-calculated growth rates corrected for weighted estimates of absent year classes.

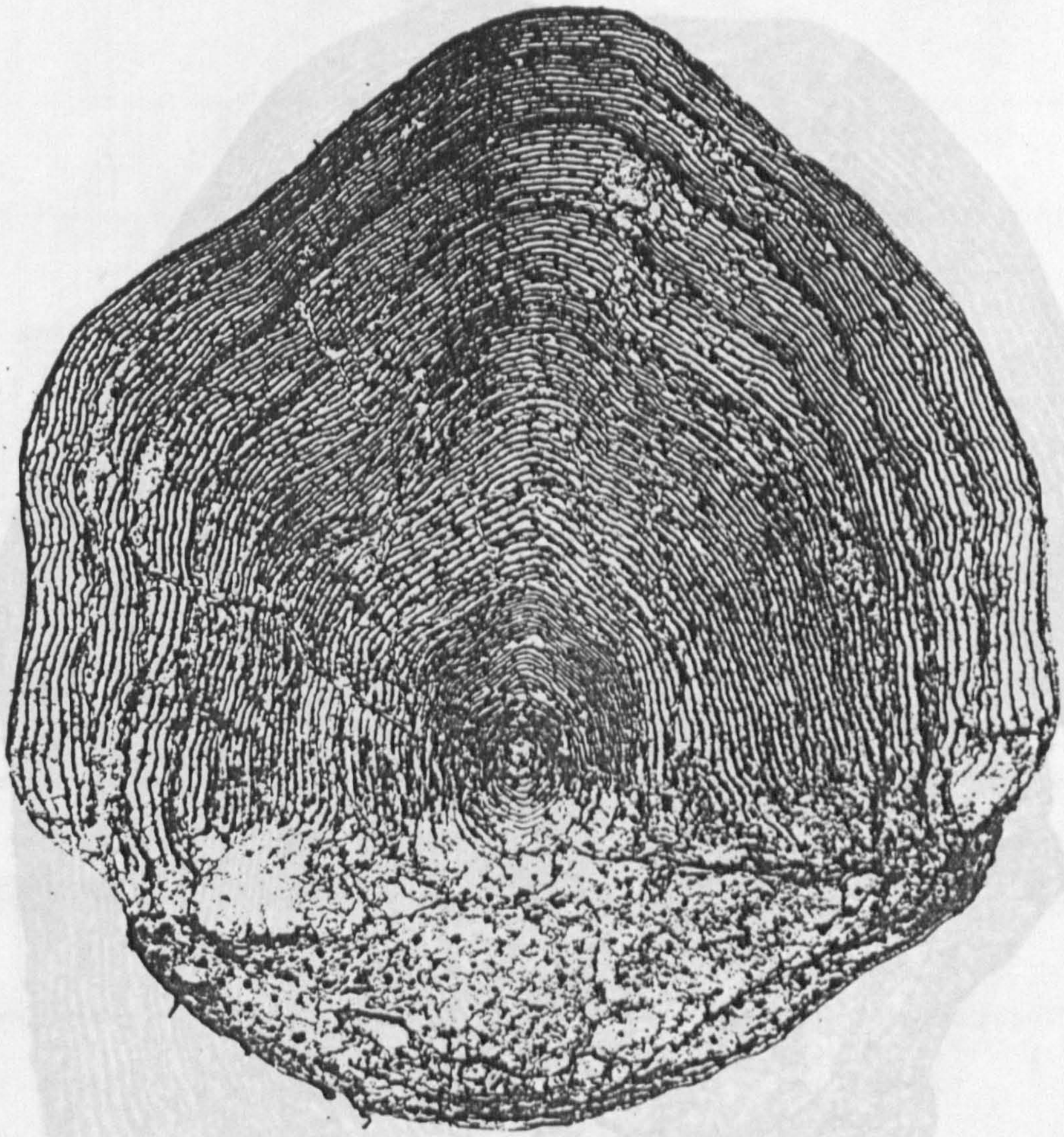
Year class	Median growth rate (mm/yr) whitling	1+ maidens
1984		59
1985	91	88
1986	102	100
1987	106	92
1988	104	73
1989	110	106
1990	110	

Appendix 2.2

Examples of sea trout scales



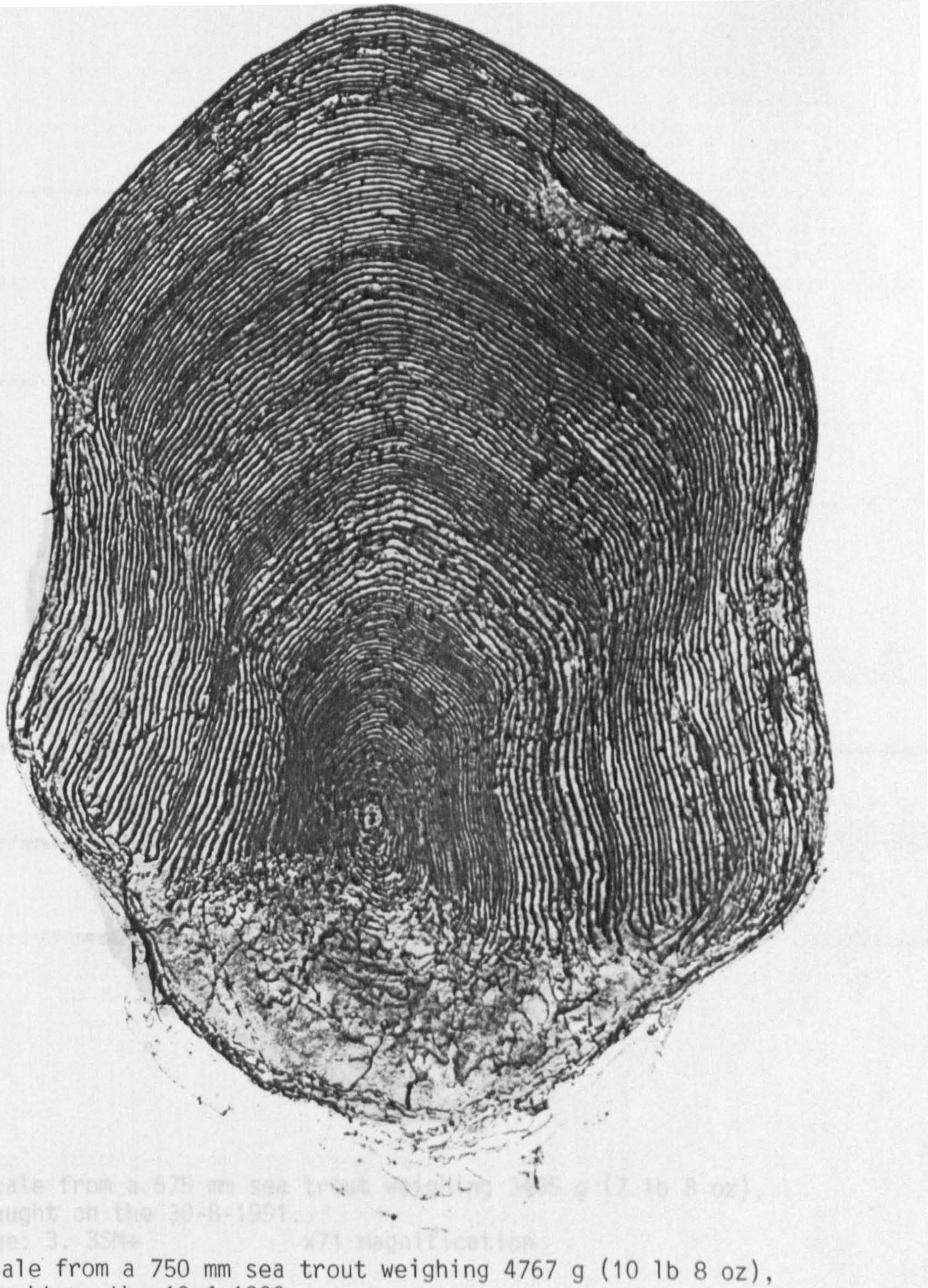
Scale from a 490 mm sea trout weighing 1305 g (2 lb 14 oz),
caught on the 28-6-1989.
Age: 2. +SM+ x85 magnification



Scale from a 800 mm sea trout weighing 6356 g (14 lb),
caught on the 3-8-1991.
Age: 2. 1+3SM+ x42 magnification

Scale from a 700 mm sea trout weighing 3859 g (8 lb 8 oz), caught on the 22-6-1989.
Age: 2. 1+2SM+ x56 magnification

Scale from a 645 mm sea trout weighing 3065 g (6 lb 12 oz),
caught on the 17-7-1990.
Age: 3. 1+ x71 magnification



Scale from a 750 mm sea trout weighing 4767 g (10 lb 8 oz),
caught on the 19-4-1990.
Age: 3. 1+4SM+ x56 magnification

Scale from a 675 mm sea trout weighing 3405 g (7 lb 8 oz), caught on the 30-8-1991.
Age: 3. 3SM+ x71 magnification

Scale from a 695 mm sea trout weighing 3632 g (8 lb),
caught on the 4-5-1992.
Age: 2. 2+SM+ x56 magnification

Scale from a 480 mm sea trout weighing 1192 g (2 lb 10 oz), caught on the 13-6-1992.
Age: 1.1+ x71 magnification

Scale from a 330 mm sea trout weighing 511 g (1 lb 2 oz), caught on the 7-7-1992.
Age: 2.+ (whitling) x85 magnification

Scale from a 660 mm sea trout weighing 3235 g (7 lb 2 oz), caught on the 2-6-1992.
Age: 2.+4SM+ x71 magnification

CHAPTER 3

OBSERVATIONS ON THE SPAWNING BEHAVIOUR OF MALE AND FEMALE ADULT SEA TROUT USING RADIO-TELEMETRY.

3.1 Introduction

Compared with the Atlantic salmon (*Salmo salar* L.) the polymorphic nature of the anadromous and resident forms of *Salmo trutta* L. has confounded the understanding of the life strategy and reproductive cycle (Le Cren 1985).

The spawning actions of *Salmo trutta* on the redds have been described previously (Jones & Ball 1954; Cambell 1977; Crisp & Carling 1989), but the behaviour during the pre- and post-spawning periods is unclear. Whilst the technology to track sea trout using telemetry has been available for the last decade (Solomon and Storeton-West 1983), the studies to date have concentrated on the tracking of upstream migrants, on, or after their initial entry to fresh water (e.g. Clarke *et al.* 1992) and focussed less on aspects such as the spawning behaviour, as has been carried out for Atlantic salmon (Webb & Hawkins 1989; Baglinière *et al.* 1990).

This study examined the short-term tracking of a small number of adult sea trout in the vicinity of the redds during the spawning period, and discusses how the observations may affect the biological characteristics described in Chapter 2.

3.2 Study site

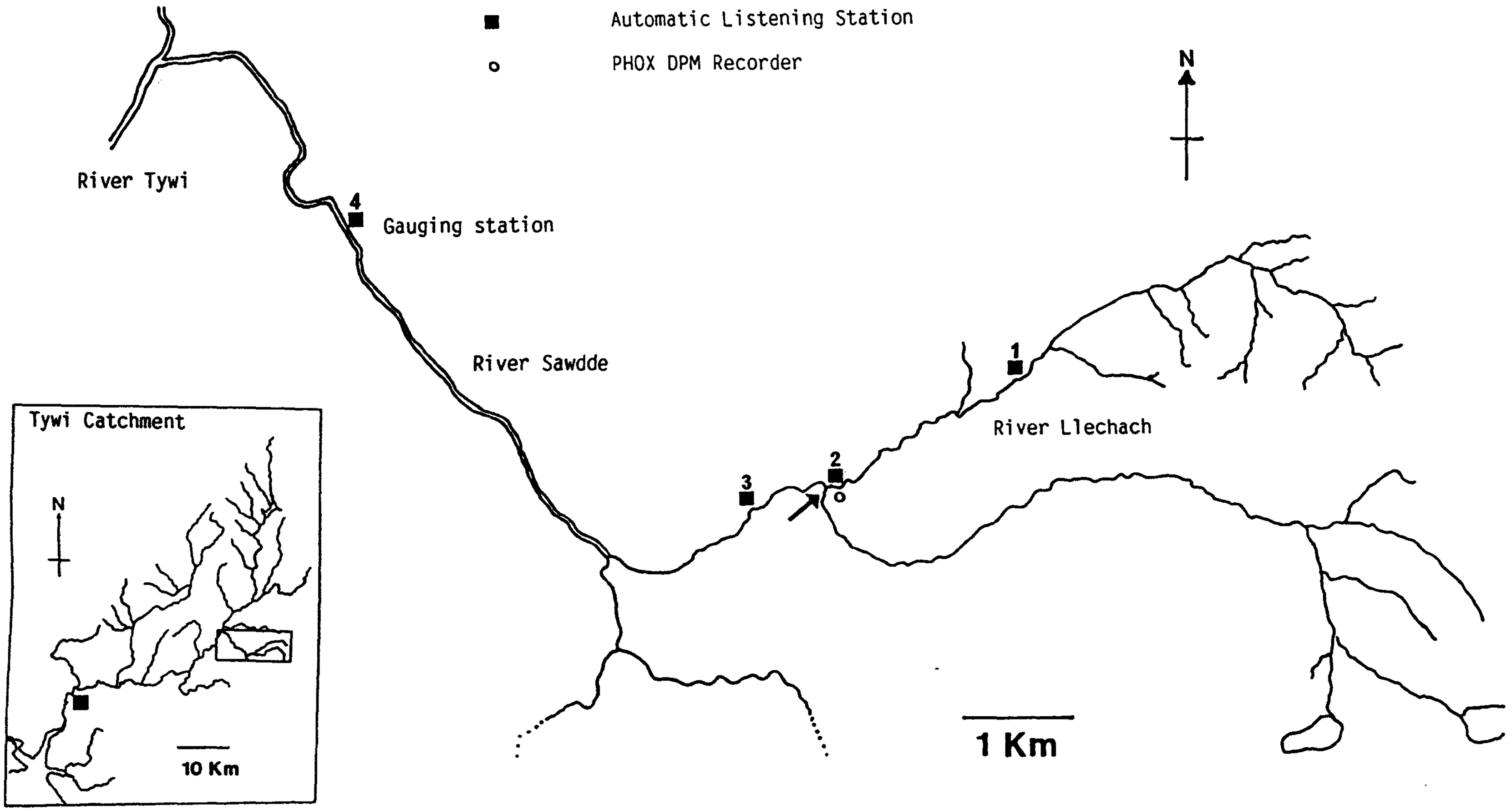
The River Llechach rises at 400 m in moorland (National Grid Reference :SN 805 255), and flows through deciduous woods and grazing land for some 7 km until it enters the River Sawdde, a major tributary of the River Tywi (Fig. 3.1). The River Llechach is some 4 m wide at its confluence with the River Sawdde and this junction is 52.7 km from the head of tide. The site was chosen as it is a major spawning area for sea trout.

3.3 Materials and Methods

3.3.1 Fish capture

In 1990 and 1991 sea trout were captured by electrofishing a pool 75 m downstream of the River Llechach confluence using pulsed DC 50 Hz equipment (Fig. 3.1). Two fish were also caught by the same method in the River Llechach.

Fig. 3.1 Map of the River Llechach and location of monitoring equipment



In 1992 an alternative method of capture was used to try to increase the numbers of sea trout available for tagging. A temporary box trap, with an inscale, was placed 75 m into the River Llechach. This trap was a partial river trap since a gap of 0.5 m was left between the banksides and the outermost 'leading' screens, to allow the unhindered downstream passage of kelts. The trap was inspected at least twice a day, pre and post-darkness, but usually four times. After the sea trout were tagged the trap was closed and the gates removed to prevent immediate re-entry. Trapping recommenced when the tagged fish had left the vicinity.

3.3.2 Tagging and tracking

Once captured the sea trout were quickly assessed for suitability for tagging. Only fish greater than 525 mm, and not unduly stressed during capture or badly infected with the fungus Saprolegnia *spp*. were tagged. The fish were anaesthetised in the tagging bags containing fresh water using approximately 100ppm 2-phenoxyethanol. After anaesthesia the fish were sexed, measured (fork length mm) and two scales taken for age determination. A radio tag was then inserted into the stomach, via the oesophagus, using a tube and plunger. An external floy tag was attached below the dorsal fin and the area treated with Cicatrin powder. The fish were allowed to recover in a section of the pool, out of the main current, until they swam off of their own accord.

The radio tags used were designed by MAFF (Potter, 1988) and comprised of a uniquely identifiable transmitter (173.805-173.850 MHz) encased in a polycarbonate cylinder measuring 52 x 15.5 mm. Each tag weighed 13 g and was small in comparison to the size of the fish. A modified Yaesu receiver (type FT 290R, pre-tuned to the tag frequencies) with a half wavelength aerial enabled active tracking on foot or by car. Automatic listening stations (ALS) placed on the bankside (Fig. 3.1) enabled further recording of movements. An automatic listening station was also situated at the head of tide in 1990 and 1991 (Fig. 3.1), to identify the time of travel for kelts. Exact fish positioning was achieved by first locating the proximity of the fish using the pre-tuned receiver and half wavelength aerial, and then de-tuning the receiver so that maximum signal strength occurred within 1-2 m of the fish.

Locations of tagged fish were kept in a field book and related to the distance from the confluence of the River Llechach using a 1:10 000

Ordnance survey map. Detailed notes were kept of the spawning behaviour of both tagged and untagged fish and a video camera was used to film spawning behaviour. Some untagged sea trout were identifiable on more than one occasion due to individual markings and infection of Saprolegnia *spp*. This study was carried out by the author alone so continuous tracking was not possible. Observations were generally restricted to between 07:00 and 00:00 h. Darkness and the wooded nature of the tributary resulted in visual observations being difficult after 17:00 h.

3.3.3 Environmental data

In 1990 the water height and temperature of the River Llechach were recorded daily, upstream of the confluence with the River Sawdde, using a temporary stage height and thermometer. In 1991 and 1992 water temperature and conductivity of the River Llechach were measured continuously (15 min interval) using a pHOX 100 DPM unit (Fig. 3.1). A decrease in conductivity was used as a means of demonstrating an increase in river discharge with rainfall events.

3.4 Results

The numbers and details of sea trout tagged in each year are shown in Table 3.1. Each fish was given a code letter for descriptive reference.

3.4.1 Tracking of sea trout

During the three years appropriate data were obtained for 6 of the 17 fish. Detailed information describing the movements and spawning behaviour of 2 tagged females (B and C) and 4 tagged males (E, N, O and P) was obtained. Two untagged females and one untagged male were also observed during their spawning period. Many untagged female and male sea trout were also observed during redd cutting; note many of the males were whitling (.+ sea age). The remainder of the tagged sea trout were only tracked for a short period due to predation, tag regurgitation or early departure from the tracking zone.

Examples of tracks obtained from male and female sea trout and are shown in Figs.3.2-5.

Table 3.1. Details of the radio-tagged sea trout. The convention of Nall (1930) for describing the age of sea trout has been used i.e. X.Y+, where X denotes the smolt age and Y the post-smolt history. A spawning mark is denoted by SM and R represents replacement scales. Thus fish C is a 4+ year old fish that migrated to sea as a two year old smolt and matured in the second winter after smolting.

Date of capture	Code	Sex	Length	Age	Tagging location (Km from confluence)
29 Oct. 1990	A	F	580	R. R	-0.075
29 Oct. 1990	B	F	540	R. +SM+	-0.075
29 Oct. 1990	C	F	620	2. 1+SM+	-0.075
29 Oct. 1990	D	F	555	R. 1+	-0.075
17 Nov. 1990	E	M	620	2. +2SM+	0.510
6 Nov. 1991	F	F	570	2. 1+	-0.075
6 Nov. 1991	G	M	550	R. R	-0.075
9 Nov. 1991	H	M	580	2. 1+	-0.075
15 Nov. 1991	I	F	820	2. 1+3SM+	0.050
15 Nov. 1991	J	F	740	R. 1+2SM+	-0.075
15 Nov. 1991	K	F	580	2. 2+	-0.075
3 Nov. 1992	L	F	630	R. 1+SM+	0.075
5 Nov. 1992	M	F	590	2. 2+	0.075
5 Nov. 1992	N	M	590	R. R (at least 1 SM)	0.075
5 Nov. 1992	O	M	560	2. 1+	0.075
5 Nov. 1992	P	M	570	R. 1+	0.075
10 Nov. 1992	Q	F	680	2. 2+SM+	0.075

3.4.2 Pre-spawning behaviour of females

All tagged fish held concealed positions after recovering from the anaesthetic. The two females tracked throughout their spawning period showed a downstream movement of between 0.6 and 1.3 km after tagging, to known holding pools. These movements did not occur immediately after tagging since both fish waited until darkness before moving downstream. The short term-tracks obtained for the other tagged females also showed that they remained stationary after tagging, until darkness. The exception was fish M which continued 125 m upstream in daylight within 2 h of tagging.

Females B (Fig. 3.3) and C (Fig. 3.2) remained at their downstream positions for 10 and 3 days, respectively. Upstream movements took place following an elevation in flow (fish B) but also at night during falling flow (fish C). For fish C entry into the River Llechach was continuous from this first movement. Fish B remained below the confluence pool for 6

Fig. 3.2 Track of Fish C. Fish position (solid line with dots) on Y axis, gauge height (solid line) on Y2 axis and confluence position (dotted line)
Sp - spawning activity p - paired with another sea trout

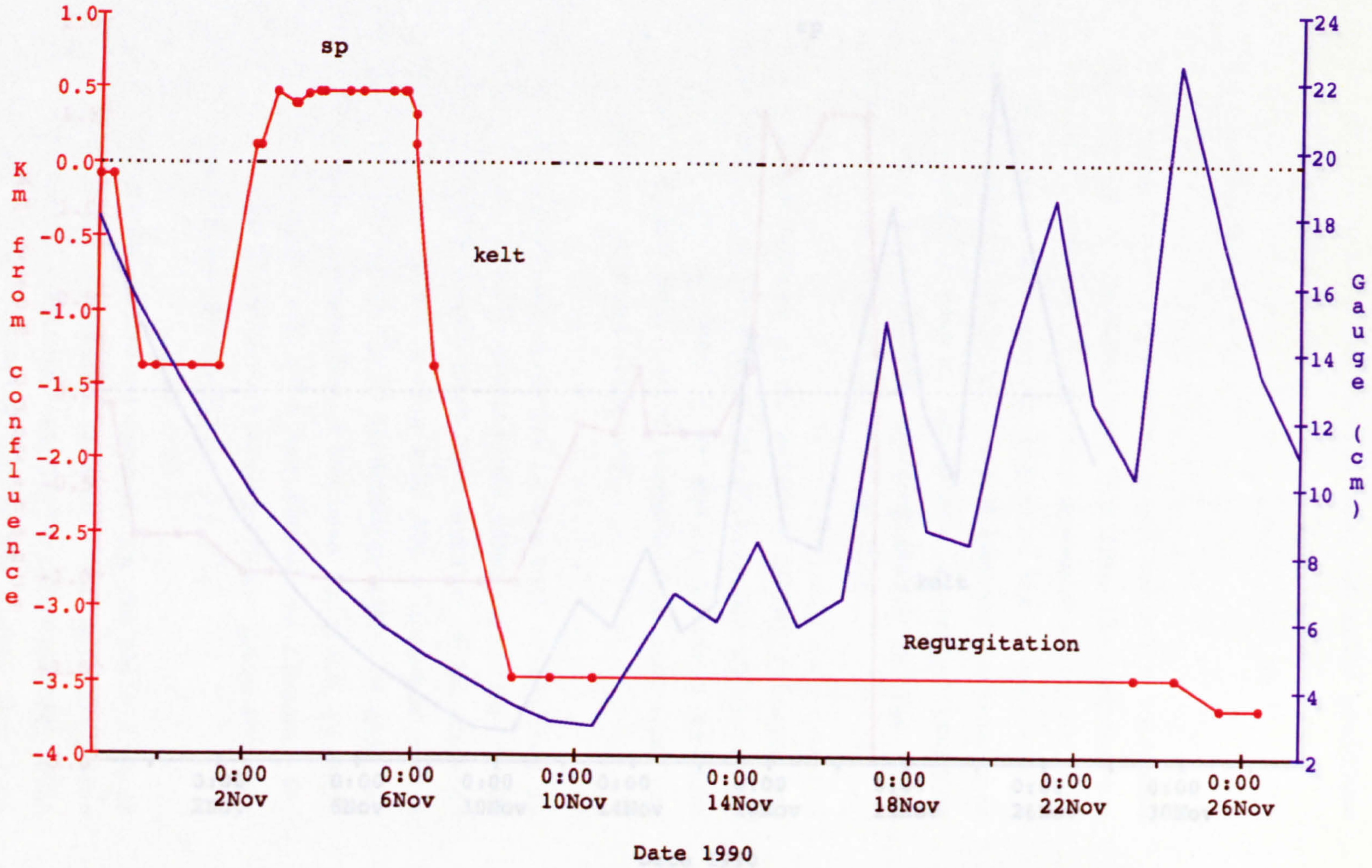
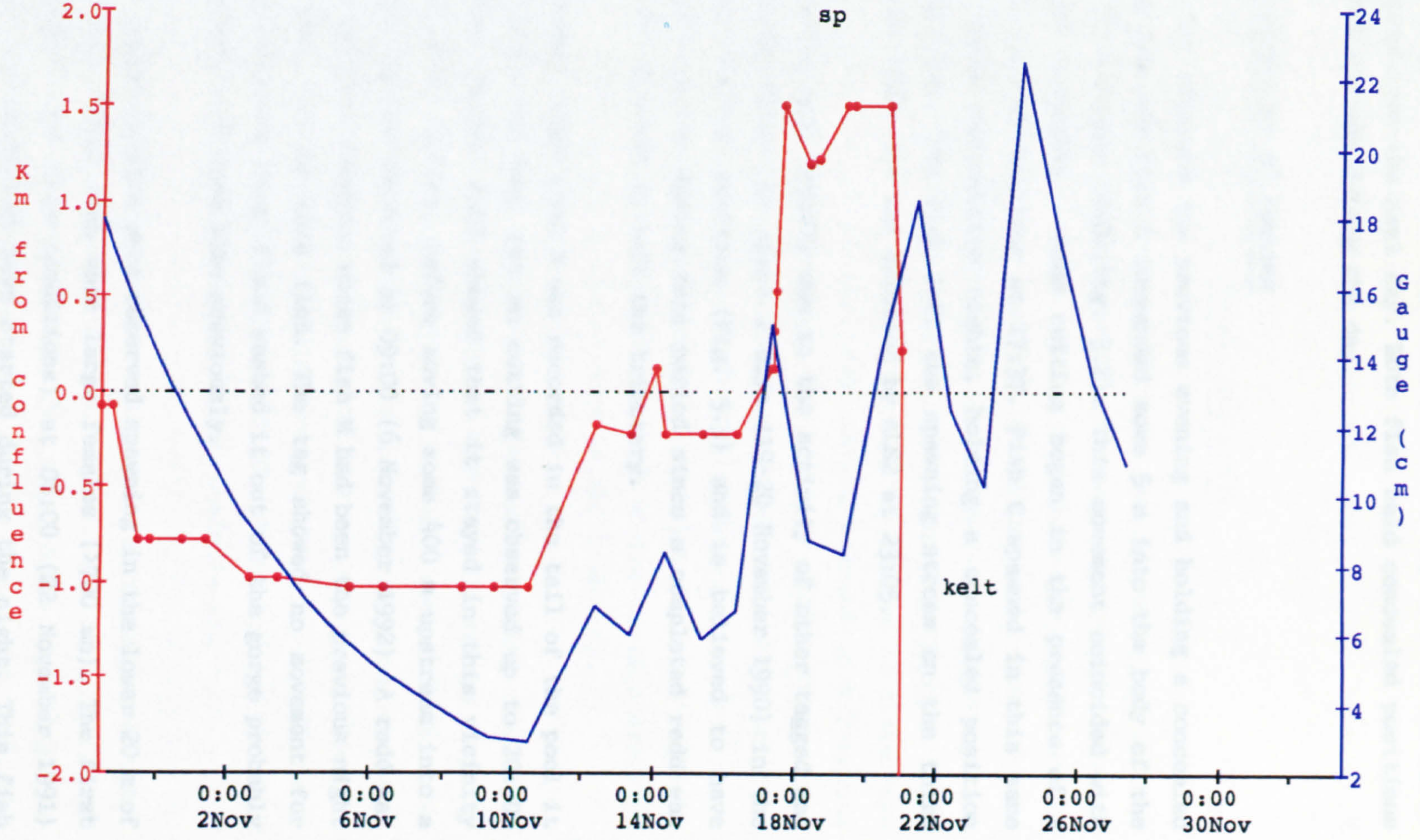


Fig. 3.3 Track of Fish B. Fish position (solid line with dots) on the Y axis gauge height (solid line) on the Y2 axis and confluence position (dotted line)
Sp - spawning activity p - paired with another sea trout



days but did make a short excursion into the River Llechach on the 2nd night following a small rise in flow. Final entry into the tributary for fish B was made on a third rise in water, to a distance 1.58 km above the confluence. A downstream movement was made by both fish (fish B, 300 m; fish C, 70 m) during the first night in the tributary but returning to the uppermost positions the next day. Both fish held concealed positions beneath tree root systems during the day.

3.4.3 Spawning behaviour of females

After entering the Llechach the previous evening and holding a concealed position during the day fish C descended some 5 m into the body of the pool at 16:55 (3 November 1990)(Fig. 3.2). This movement coincided with decreasing light intensity. Redd cutting began in the presence of a large male and several whitling at 17:37. Fish C spawned in this same location over three consecutive nights, holding a concealed position after 09:30 each day. The fish left the spawning stream on the third night (5 November 1990) and was detected by ALS2 at 23:05.

Fish B was observed only briefly due to the activity of other tagged sea trout and elevated flows. It spent 2 days (19-20 November 1990) in the vicinity of its highest position (Fig. 3.3) and is believed to have spawned in this locality during this period since a completed redd was observed the morning after it left the tributary.

At 19:00 (5 November 1992) Fish M was recorded in the tail of the pool it had occupied during the day, but no cutting was observed up to 20:30, when observations ceased. ALS2 showed that it stayed in this vicinity until 05:00 the next morning, before moving some 400 m upstream into a gorge, where the tag was detected at 09:00 (6 November 1992). A redd was observed in the precise location where fish M had been the previous night and may have been cut by this fish. The tag showed no movement for another 16 days before a large flood washed it out of the gorge probably having been regurgitated some time previously.

Both of the un-tagged females were observed spawning in the lower 20 m of the River Llechach. These fish were large females (>700 mm). The first was observed (under low flow conditions) at 08:00 (22 November 1991) still cutting a redd that had been started during the night. This fish then dropped out of the River Llechach and was followed 75 m downstream

to the large pool in the River Sawdde. That evening, as light intensity lowered, the fish was seen back upstream in the River Llechach on the same redd. This fish spawned over a three-day period completing one large redd. The second fish (11 November 1992, under medium flow conditions) was observed for the first time on a partly-cut redd, shortly before darkness. Observations the following day showed this fish dropping back onto the same redd at 16:00, from a pool some 25 m upstream.

3.4.4 Overview of observations of female sea trout

During the three years approximately 180 redds were recorded and some 50 females were observed spawning. From all the female sea trout observed, both tagged and un-tagged, several points could be noted. Redd siting was carried out by the female; one female was observed 'nosing' the gravel downstream of the subsequent redd site. Exploratory cutting took place with and without males(s) present. Some exploratory cuts would seemingly be abandoned. Sea trout were observed to take up to three days to complete spawning. Much of the redd cutting started with lowering light intensity and carried on during the night. Females often ceased cutting during the day and moved to concealed positions above or below the redd site, sometimes outside the tributary. Spawning also took place during daylight, with some redds started and completed during daylight (Female paired to Fish P on the 7/11/92). Most of the daytime spawning took place when large numbers of adults were in the tributary.

Redd cutting was accomplished by the female's repeated lateral flexing of the tail in a concentrated area. The female would move upstream slightly and drop back assessing the depression with her tail and anal fin. Ova were deposited and the female would then cut rapidly upstream to cover the eggs. Gaping of the mouth was observed during ova deposition but sometimes only slight opening of the mouth was noticeable. Several such pockets may be deposited in a completed redd. The depression at the head of the completed redd would often have a large cobble in it.

No sea trout was directly observed to construct more than one redd, although some observations suggest this may occur. For example Fish M moved upstream, before regurgitating the tag, after spawning was suspected. A large untagged female was observed to deposit at least one pocket of eggs in a redd but this redd was not further enlarged. Spawning

in another location was suspected since the redd would be too small to contain all the ova of a fish of that size. Some female kelts were observed moving downstream and apparently 'cutting' at the upstream edge of the depression of completed redds. One untagged kelt was observed, in the presence of a male, 'cutting' haphazardly in a redd completed some 4 days previously. Some 6 hours later the same pair were 110 m downstream with the female again 'cutting' in the depression at the head of a previously completed redd. Actual deposition of ova was not suspected in either case.

3.4.5 Pre-spawning behaviour of male sea trout

All males held concealed positions after tagging for between 1 and 6 h and later moved downstream between 0.050 and 0.700 km. Fish N and P descended from the tributary into holding pools in the River Sawdde. Males E, N and P returned upstream within 3.5 to 24 h, all under non-elevated flow. Fish N did not return into the River Llechach but ascended the River Sawdde a further 0.625 km.

3.4.6 Spawning behaviour of males

Males E, O and P were observed in detail during their spawning periods. Less detailed observations were obtained of Fish N due to the activities of other tagged fish.

Male P (Fig. 3.4) entered the lower 150 m of the River Llechach at 21:10 (5 November 1992), 11 hrs after tagging, and was recorded in the vicinity of spawning females throughout the night. Darkness prevented the identification of individual females. It was then recorded 25 m below the confluence the following morning but returned at 16:40 to a position 50 m above the confluence. The fish was observed in the vicinity of a female nearing the end of completing a redd 25 m downstream, but at 17:00 ascended a further 50 m and was observed as the dominant male with a spawning female. Again darkness made identification of individual females difficult. This position was maintained until 23:35 when it moved further upstream passing ALS2 by 00:40 (6 November 1992). At 07:35 (6 November 1992) it was observed 0.750 km above the confluence paired with a female of about the same size as itself, cutting a new redd. Spawning continued during daylight and the redd was completed before both male and female

Fig. 3.4 Track of Fish P. Fish position (solid line with dots) on the Y axis conductivity (solid line) on the Y2 axis and confluence position (dotted line)
Sp - spawning activity p - paired with another sea trout

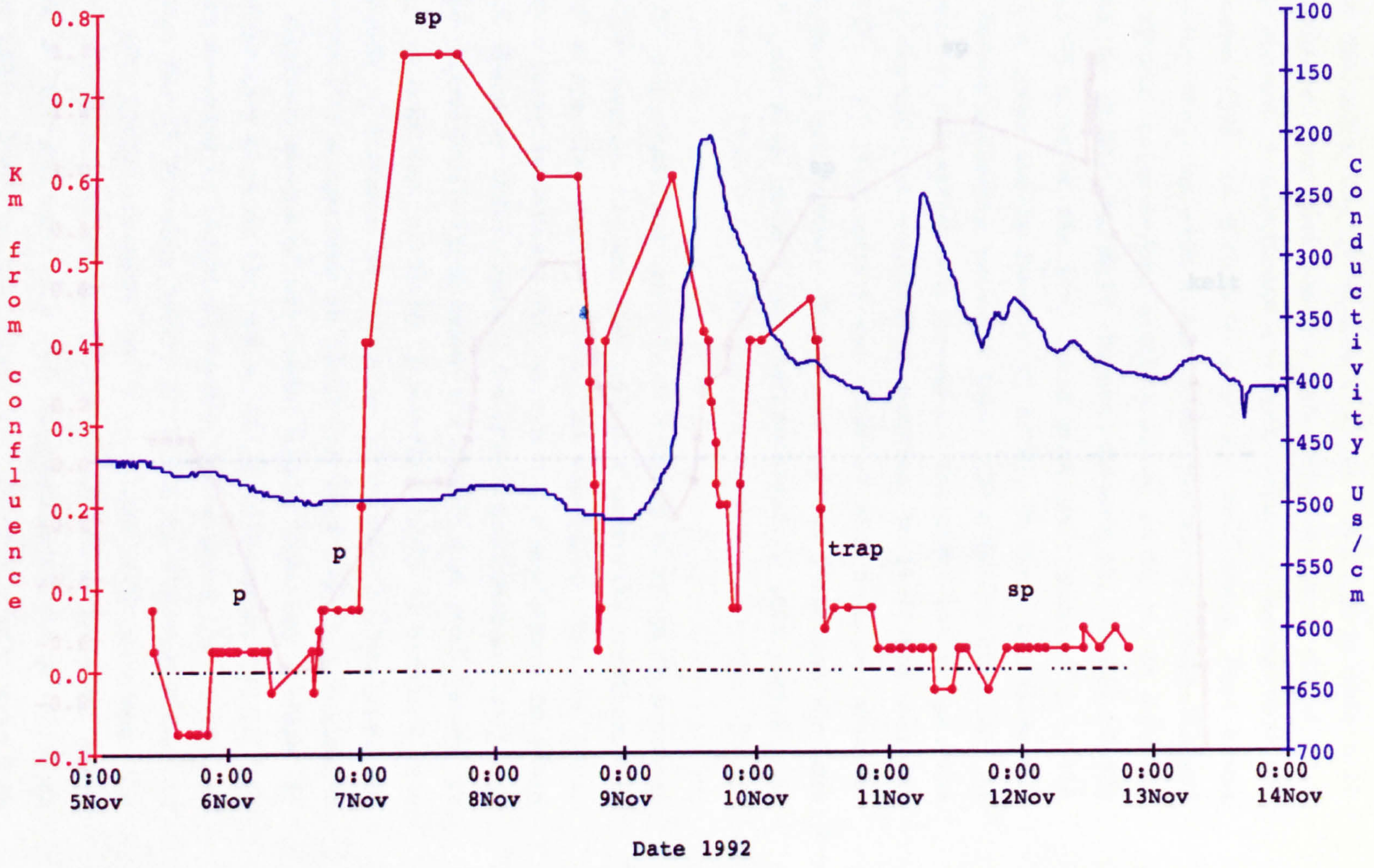
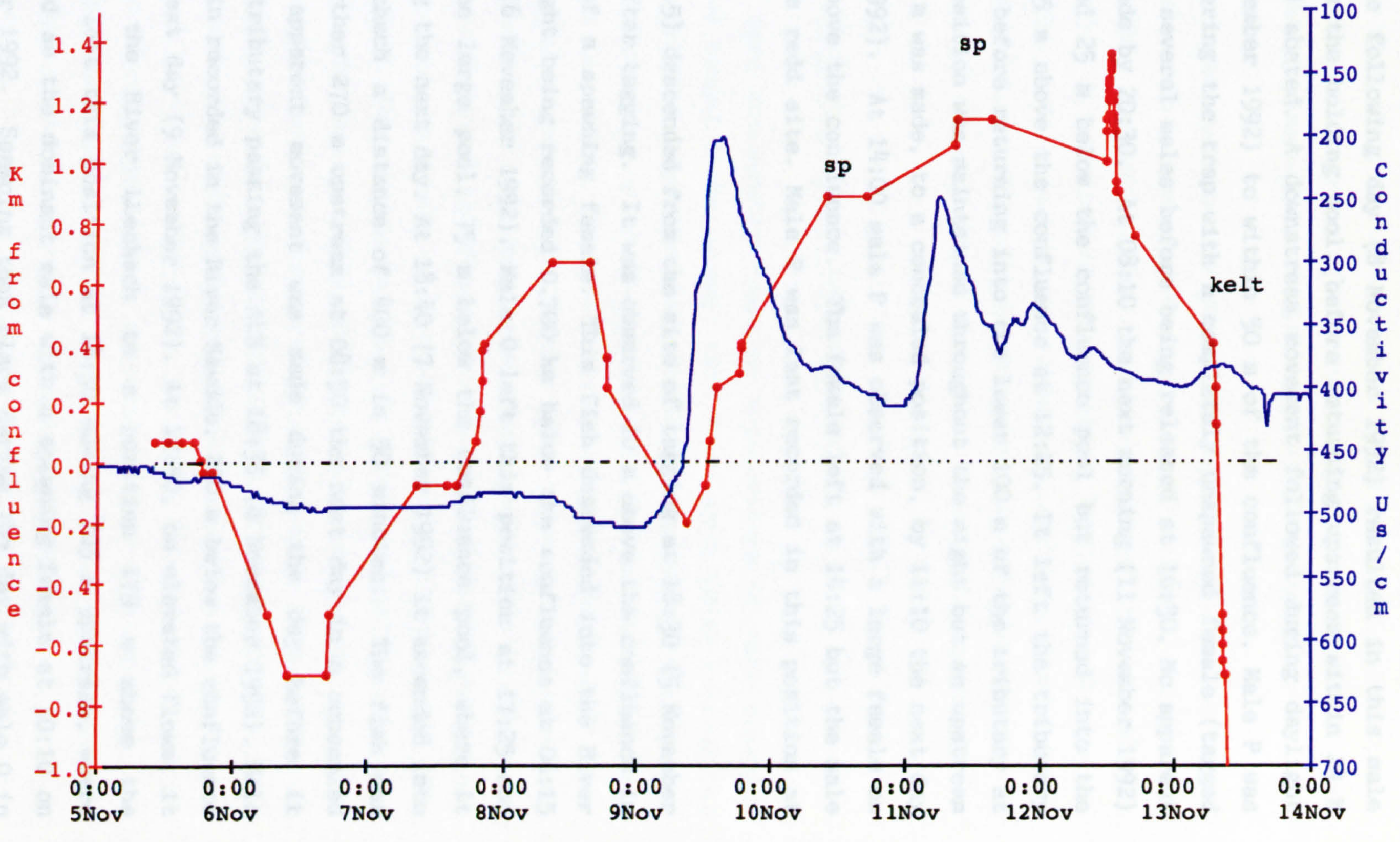


Fig. 3.5 Track of Fish O. Fish position (solid line with dots) on the Y axis conductivity (solid line) on the Y2 axis and confluence position (dotted line)
Sp - spawning activity p - paired with another sea trout



descended from the redd site. At 08:30 the next morning male P was observed in a large pool, 150 m below the redd site, where no apparent movement occurred during daylight. At 17:10 it descended the River Llechach to within 50 m of the confluence but returned upstream immediately, to the same holding pool occupied during the day. A large flood event the following day (8 November 1992) resulted in this male descending from the holding pool, before returning upstream within 24 h when the flood abated. A downstream movement followed during daylight (10:35, 10 November 1992) to within 50 m of the confluence. Male P was recaptured entering the trap with a completely unspawned female (tagged as fish Q) and several males before being released at 16:30. No apparent movement was made by 20:30. At 08:10 the next morning (11 November 1992) it was recorded 25 m below the confluence pool but returned into the tributary to 25 m above the confluence at 12:25. It left the tributary again at 17:50 before returning into the lower 100 m of the tributary at 23:15. This position was maintained throughout the night but an upstream movement of 25 m was made, to a concealed position, by 11:10 the next day (12 November 1992). At 14:00 male P was observed with a large female on a redd, 25 m above the confluence. The female left at 14:25 but the male remained at the redd site. Male P was last recorded in this position at 19:10.

Male O (Fig. 3.5) descended from the site of tagging at 18:30 (5 November 1992), 8 hrs after tagging. It was observed 10 m above the confluence in the vicinity of a spawning female. This fish descended into the River Sawdde that night being recorded 0.700 km below the confluence at 06:15 the next day (6 November 1992). Male O left this position at 17:25 and ascended to the large pool, 75 m below the confluence pool, where it remained during the next day. At 18:40 (7 November 1992) it ascended into the River Llechach a distance of 400 m in 50 minutes. The fish was recorded a further 270 m upstream at 08:50 the next day in a concealed position. No apparent movement was made during the day before it descended the tributary passing the ALS at 18:35 (8 November 1992). Male O was once again recorded in the River Sawdde, 200 m below the confluence at 08:50 the next day (9 November 1992). At 12:50, on elevated flows, it returned into the River LLechach to a position 175 m above the confluence. It left this position at 18:30 moving 705 m upstream, where it was observed as the dominant male with a spawning female at 10:10 on the 10 November 1992. Spawning took place during the day with male O in the same location at 17:30. At 09:00 the following morning the fish was

observed moving upstream. It ascended 135 m in 6 minutes to pair with a female beginning a new redd and its position was unchanged at 15:20. At 11:47 the next day male 0 was recorded 140 m downstream, but again actively ascending. It ascended 350 m in 60 minutes, before immediately descending 450 m in 58 minutes. It then stopped on a redd at this position before descending a further 150 m by 16:55. Male 0 left the River Llechach the next morning (13 November 1992) at 06:55 and covered the 6.3 km to the gauging station in 11 hrs, during daylight.

Male E moved upstream the second night after tagging and was observed as the dominant male with a spawning female at 09:30 the next morning (19 November 1992; This female had been observed 'nosing' the gravel the previous evening). It remained in this location for the next 2 days, as the dominant male, before leaving the tributary with the female kelt. Fish E then ascended the River Llechach again, in daylight (12:50, 21 November 1992) under elevated flows, but left the tributary at 07:25 the next morning.

3.4.7 Overview of observations of male sea trout

Several trends were discernible from the observations of male spawning behaviour. Males tended to be present when the female began cutting but on some occasions males appeared from downstream to attend the female. During cutting there were up to 10 males present but rarely would there be more than two large fish; the remaining males being whitling or parr. The dominant male was always the largest male present, with dominance among the remaining males also related to size. The dominant male would remain in the vicinity of the female for the duration of spawning but some males appeared to leave near the completion of the redd. The dominant male would often remain on the redd when the female moved away, and a secondary male would do thus if the dominant male moved away. The dominant male would often move downstream with the spent female (Fish E, P and the partner to female C), sometimes out of the tributary, but returned to the tributary with or without another female.

Both tagged and untagged males were observed to pair with more than one female and this resulted in males travelling several kilometres and spending more time in the vicinity of the spawning beds.

During spawning the dominant male would be positioned alongside the

female and the other males some one metre downstream of the pair. Quivering of the male was observed and the male often crossed over the back of the female, seemingly pressing the female into the depression. Other attendant males would be repelled if they approached the female and the dominant male either orientated itself to attack with its mouth or 'slap' other males with its tail. On some occasions the secondary males were seen simultaneously darting in upstream alongside the male and female, but this did not always appear to accompany ova deposition.

3.4.8 Response to environmental stimuli

The tracking data illustrate the use by some sea trout of elevations in flow, but movements into the lower River Llechach were common under low flows, mostly at night. During very high flows sea trout (Fish P) may move downstream to seemingly quieter water, returning after the spate had abated.

The temperature records show that in 1990 (Fig. 3.6) the first sea trout redds observed were associated with low water temperatures (c. 6^{o}C). The number of adults spawning in 1991 was poor and no clear trend was apparent. In 1992 the majority of redds were cut in one week, and this coincided with a mild weather period, with water temperatures between 8 and 11^{o}C (Fig. 3.7). The common facet between years during the main redd cutting period seemed to be falling or stable water levels.

3.4.9 Movement of kelts

Many kelts left the spawning stream during darkness (fish B, C and 0). Some fish continued out of the River Sawdde the same night covering the 6.3 Km to the gauging station in 3 hrs (fish B). Other fish (fish C, J) descended to known holding pools 1-2 Km below the River Llechach confluence, where they stopped during the day (fish C was observed in the exact location where it had moved to after tagging, 7 days previously). Subsequent movement passed the gauging station occurred during darkness, 32 to 54.5 hrs after first leaving the spawning beds. One fish (fish 0) left the River Llechach in the early hours (06:55) and covered the 6.3 Km to the gauging station in 11 hrs, during daylight and under non-elevated flow. Three fish were recorded passing the head of tide ALS (52.7 Km away), between 6 and 38 days after first leaving the spawning beds.

Fig. 3.6. Temperature (dashed line) on the Y axis and gauge height on the Y2 axis during 1990. Dotted line indicates spawning period

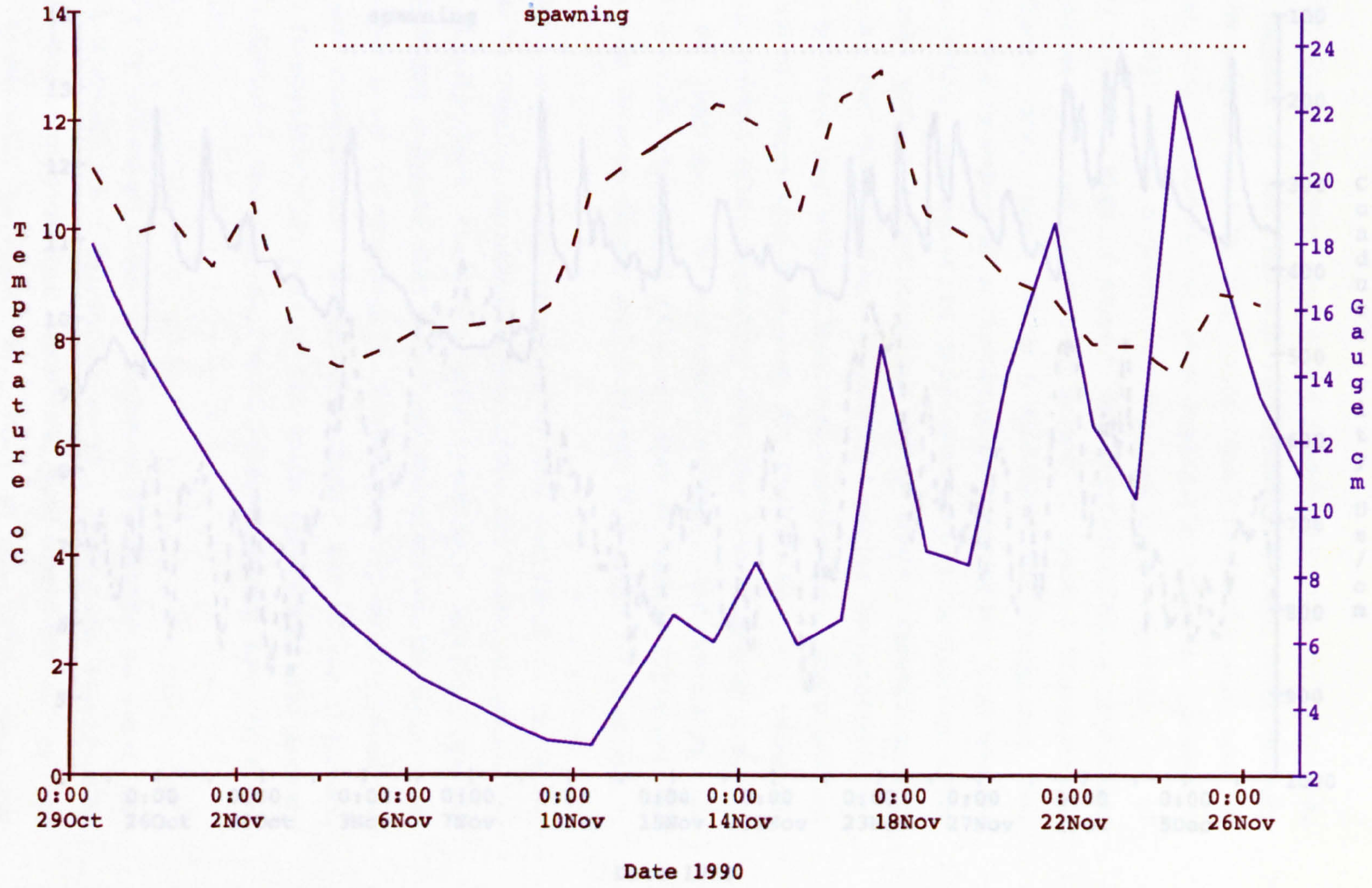
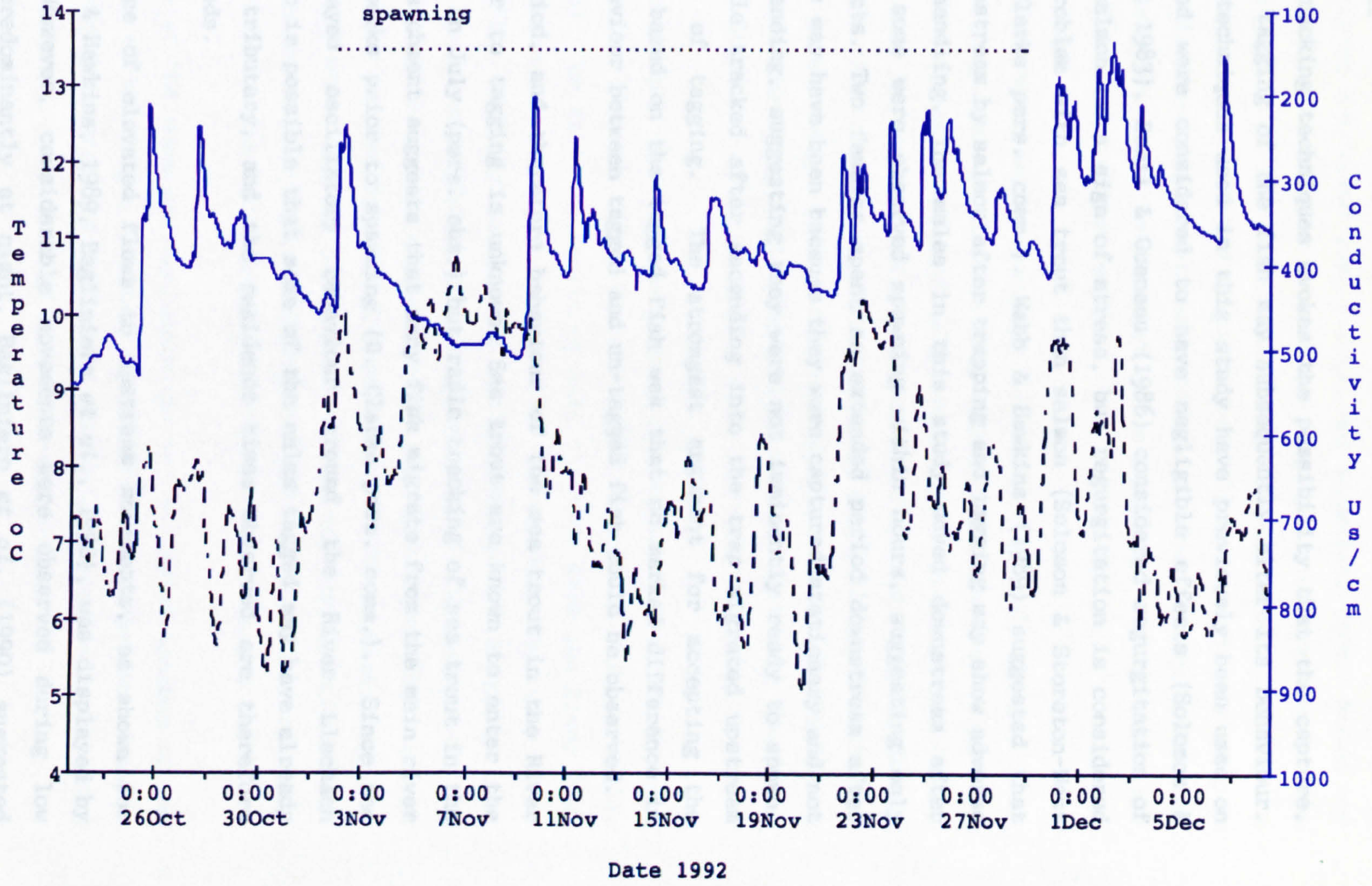


Fig. 3.7 Temperature (dashed line) on the Y axis and conductivity (solid line) on the Y2 axis during 1992. Dotted line indicates spawning period



The small number of kelts tracked does not allow a constructive comparison between males and females.

3.5 Discussion

The use of tracking techniques evokes the possibility that the capture, handling and tagging of the fish may subsequently alter its behaviour. The capture techniques used in this study have previously been used on sea trout and were considered to have negligible effects (Solomon & Storeton-West 1983). Baril & Gueneau (1986) considered regurgitation of the tag by salmon as a sign of stress, but regurgitation is considered more of a problem with sea trout than salmon (Solomon & Storeton-West 1983; D. Clarke pers. comm.). Webb & Hawkins (1989) suggested that movement downstream by salmon after trapping and tagging may show adverse effects of handling. The males in this study moved downstream after tagging but some were observed spawning within hours, suggesting only limited effects. Two females spent an extended period downstream after tagging. This may have been because they were captured stationary and not actively ascending, suggesting they were not imminently ready to spawn. The one female tracked after ascending into the trap migrated upstream within hours of tagging. The strongest argument for accepting the observations based on the tagged fish was that no marked difference in spawning behaviour between tagged and un-tagged fish could be observed.

The time period, and therefore behaviour of the sea trout in the River Sawdde, prior to tagging is unknown. Sea trout are known to enter the River Sawdde in July (pers. obs.) but radio tracking of sea trout in the River Tywi catchment suggests that many fish migrate from the main river in the few weeks prior to spawning (D. Clarke pers. comm.). Since the males displayed oscillatory behaviour around the River Llechach confluence it is possible that some of the males tagged may have already entered the tributary, and the residence times observed are therefore minimum periods.

The importance of elevated flows to upstream movements, as shown for salmon (Webb & Hawkins, 1989, Bagliniere *et al.*, 1990), was displayed by some fish. However, considerable movements were observed during low flows, but predominantly at night. Bagliniere *et al.* (1990) suggested that increases in water temperature triggered spawning for salmon. Whilst

this may seem apparent for the sea trout in this study during 1992, the observations of 1990 suggest the opposite. A falling or stable flow regime is suggested as an important factor.

The distances moved upstream by males prior to and during spawning suggests that some cue, other than visual, may stimulate them. Moore and Scott (1991,1992) showed that pheremones may play an important role during spawning time in attracting males, but further work is needed to fully establish the behavioural significance. If olfaction does play a significant role then the downstream movement of male sea trout between spawning events could be interpreted as a strategy to increase the opportunity of finding another female.

The observation that males occupied the spawning beds longer than females, and displayed greater active movement within the tributary, leads to a greater possibility of male mortality. Jonsson *et al.* (1991) demonstrated that the energetic cost of spawning for male Atlantic salmon was higher than that of females. This increased probability of mortality for males is in accordance with with the higher sex ratio of females to males in previous spawning fish, compared to maidens, entering the River Tywi (Chapter 2).

Several authors have reported sex ratios in favour of females for sea trout populations (Harris, 1970; Cambell, 1977; Pratten & Shearer, 1983; Le Cren, 1985). The ratio of female to male whitling from the River Tywi (Chapter 2) shows a more equal distribution as reported in other works (Le Cren, 1985). What is unclear is the proportion of whitling that mature. Many mature male whitling were observed on the spawning beds but very few female whitling were encountered in the River Llechach. Although the male whitling were seldom dominant males, due to their size, they may have fertilised some eggs by darting in during egg laying, similar to parr. Thus the presence of male whitling ensures a pool of mature males capable of spawning with large females in the absence of larger males.

The structure of the spawning population was not estimated because of the partial nature of the trap. However, the oscillatory behaviour of males around the trap site suggests that care should be taken with estimating spawning populations, to avoid duplication in sampling.

Sea trout are known to survive many spawning migrations (Harris, 1970; Le Cren, 1985) and the rapid descent from the spawning beds shown in this study is a probable factor in determining this survivorship.

This study has drawn upon results from a small number fish, which is often the case with radio-tracking studies. Hence it is difficult to analyse the data statistically and conclusions are drawn from interpretation of behaviour, supporting hypotheses. However, without radio-tracking, behavioural studies over such a time period would be difficult if not impossible.

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CHAPTER 4

RESULTS OF THE RIVER TYWI ANGLER LOG BOOK SURVEY (1990-1993).

4.1 Introduction

It has long been recognised that total annual catch returns may not reliably indicate trends in fishery performance due to variation in factors such as fishing effort and reporting rates. Log books are increasingly used to gather detailed information to improve the quality of catch data, both for annual trends and also for temporal trends within seasons.

This chapter aims to examine the temporal and spatial trends in the performance of the rod fishery within the Tywi catchment, over the four year period 1990-1993. It will therefore provide a baseline upon which future surveys can be compared, and possibly allow some measure of the effect of liming Llyn Brianne (Chapter 7) on the number of returning adults.

4.2 Materials and Methods

At the beginning of each season (1990-1993), anglers selected at random from Angling Association membership lists or license counterfoils were sent log books and asked to record details of fishing trips in the Tywi catchment. Details requested for each trip are shown in Fig. 4.1; night time was defined as the period during which anglers considered that car headlamps would be required for driving. At the end of each season the anglers were sent pre-paid envelopes to return their log book. Relevant details from each completed book were extracted and the log books were then returned to the anglers.

The data were spatially classified into relevant sections of the river including the lower river (downstream of Dryslwyn bridge), the middle river (Dryslwyn bridge to Llangadog bridge), the upper river (above Llangadog bridge), the River Gwili and the River Cothi (Fig. 4.2).

Mean daily flows for the four year period were obtained from the Capel Dewi (Nantgaredig) gauging station.

Fig. 4.1 An example page from a log book

REMEMBER: ONLY ONE TRIP TO BE RECORDED ON EACH PAGE

RIVER		LOCATION	

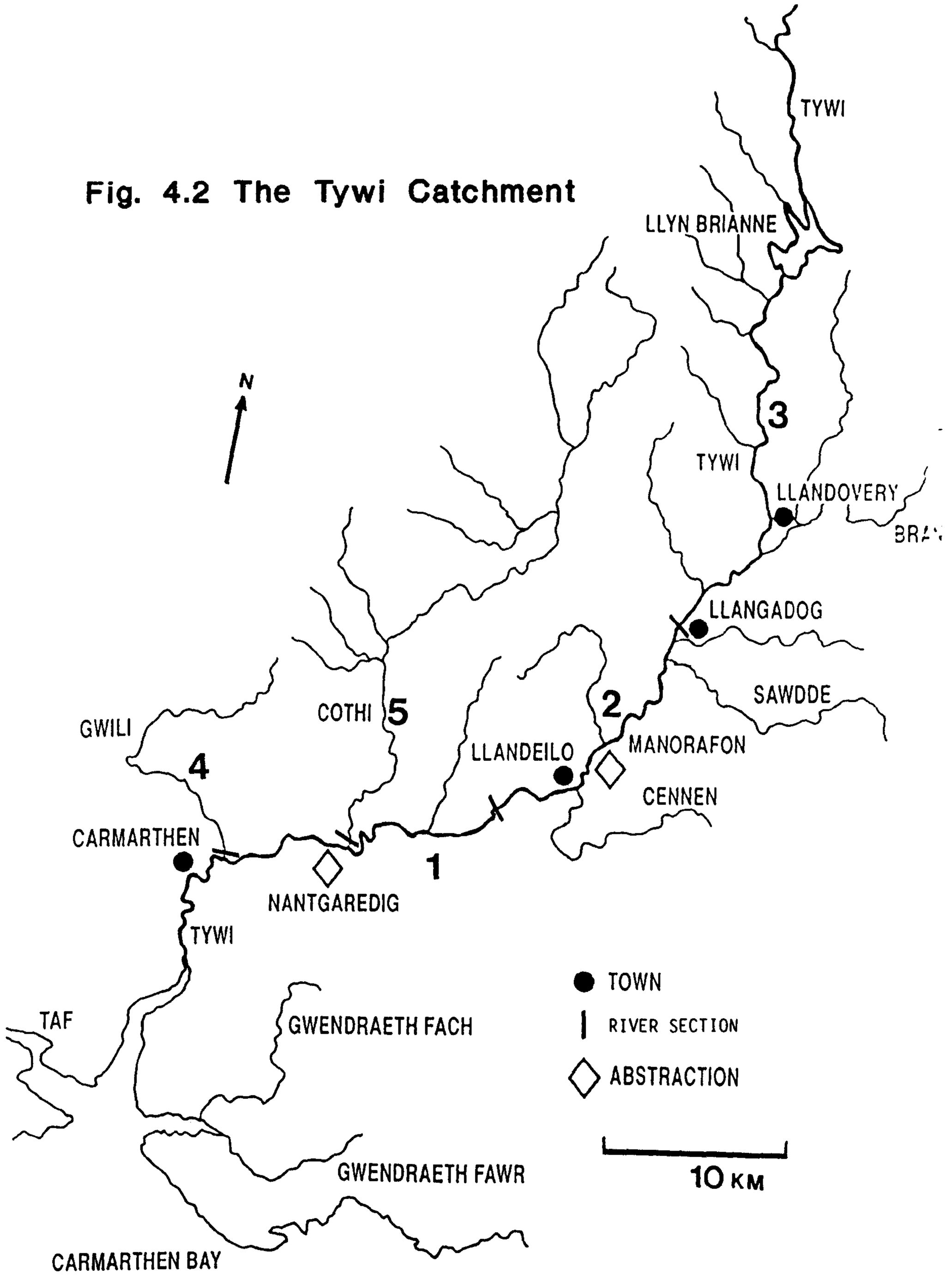
DATE PLEASE CIRCLE APPROPRIATE BOXES											
DATE	1	2	3	4	5	6	7	8	9	10	11
	12	13	14	15	16	17	18	19	20	30	31
	21	22	23	24	25	26	27	28	29		
MONTH	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	

FISHING EFFORT	HOURS
HOURS FISHING FOR SALMON ONLY	
HOURS FISHING FOR SEA TROUT (SEWIN) ONLY	
HOURS FISHING FOR BOTH SPECIES AT THE SAME TIME	
HOURS FISHING AT NIGHT	

CATCH			IF NIL TICK BOX ——————>							
FISH No.	SPECIES (TICK BOX)		WEIGHT		CAPTURE TIME (TICK)		METHOD			TAG No.
	SALMON	SEWIN	LBS	OZ	DAY	NIGHT	FLY	SPNR	BT	
1										
2										
3										
4										
5										
6										
7										
8										
9										
10										

COMMENTS .....

Fig. 4.2 The Tywi Catchment



4.3 Results

4.3.1 Anglers response

The return rate of books varied between 37% and 51%, but not all books were properly completed (Table 4.1). The summary statistics of catch and effort in each year for the Tywi catchment combined are shown in Table 4.1.

Table 4.1. Summary of catch and effort statistics for the Tywi catchment combined, 1990-1993.

Statistic	1990	1991	1992	1993
Books out	297	343	330	325
Books in (useable)	151 (116)	155 (105)	144 (104)	120 (94)
% returned	51%	45%	44%	37%
Total trips	2156	2261	2265	1769
Total trips per angler	18.6	21.5	21.8	19.2
Salmon				
No. anglers	104	99	98	86
Total trips	1192	1149	1270	953
Total hours	4541.25	4190.75	4731.00	3358.75
Mean length of trip (hrs)	3.81	3.65	3.73	3.52
Trips per angler	11.5	11.6	13.0	11.1
Catch	62	73	50	46
Catch per angler	0.60	0.74	0.51	0.53
Catch/trip	0.052	0.064	0.039	0.048
Catch/hour	0.0137	0.0174	0.0106	0.0137
Average weight (kg)	3.78	3.91	3.83	3.49
Sea trout				
No. anglers	115	104	102	90
Total trips	1907	2054	1991	1592
Total hours	6399.75	6763.5	6735.25	5188.5
Total hrs at night	1580.5	1739.75	1559.75	1027
Mean length trip	3.36	3.29	3.38	3.26
Trips per angler	16.6	19.8	19.5	17.7
Catch	546	696	941	596
Catch per angler	4.75	6.69	9.23	6.62
Catch/trip	0.286	0.339	0.473	0.374
Catch/hour	0.0853	0.1029	0.1397	0.1149
Average weight (kg)	1.23	1.04	1.09	1.39

From the total number of trips recorded and the total number of anglers, between 18.6 and 21.8 total trips were made per season (Table 4.1). The number of trips for salmon varied between 11.1 and 13.0 per season, and for sea trout the number of trips per season varied between 16.6 and 19.8.

The mean length of trips for salmon in each season (mean range 3.5-3.8) were longer than trips for sea trout (mean range 3.3-3.4).

4.3.2 Catch and effort for salmon

The overall season catch per hour from the Tywi catchment for salmon was 0.0137, 0.0174, 0.0106 and 0.0137 in 1990, 1991, 1992 and 1993 respectively (Table 4.1; Fig. 4.3). Therefore, for the years 1990, 1991, 1992 and 1993 it took 73, 57.5, 94.3 and 73 hours to catch a salmon, respectively. This implies that the salmon fishery performed best in 1991, followed by equal performance in 1990 and 1993, and poorest catch rates in 1992.

The monthly catch per hour for each year (Fig. 4.4) demonstrated the temporal trend in CPUE with the highest CPUE recorded during the last two months of the season, except in 1991 when a high CPUE was also observed in March. This temporal increase in CPUE was similar to the pattern of anglers' effort for salmon (Fig. 4.5).

The daily trend in CPUE for salmon (Figs. 4.6a-9a) demonstrated that in some years, especially 1991 (Fig. 4.7a), salmon were captured in March and April. However, it is not known whether some or all of these early fish were kelts. Sporadic catches of salmon occurred from June onwards, but consistent daily catches did not occur until August-October when autumnal freshets occurred. Daily effort for salmon was clearly influenced by river flow for the period July-October (Figs. 4.6b-9b), with increased effort coinciding with increases in flow. This relationship was not as clear during flow increases in the early part of the season (Figs. 4.6b & 9b).

There was a marked difference in the spatial CPUE within the catchment (Fig. 4.3). Within the main river the lower river displayed the highest CPUE, followed by the middle and upper river sections respectively.

Fig. 4.3 Annual salmon CPUE from the Tywi catchment

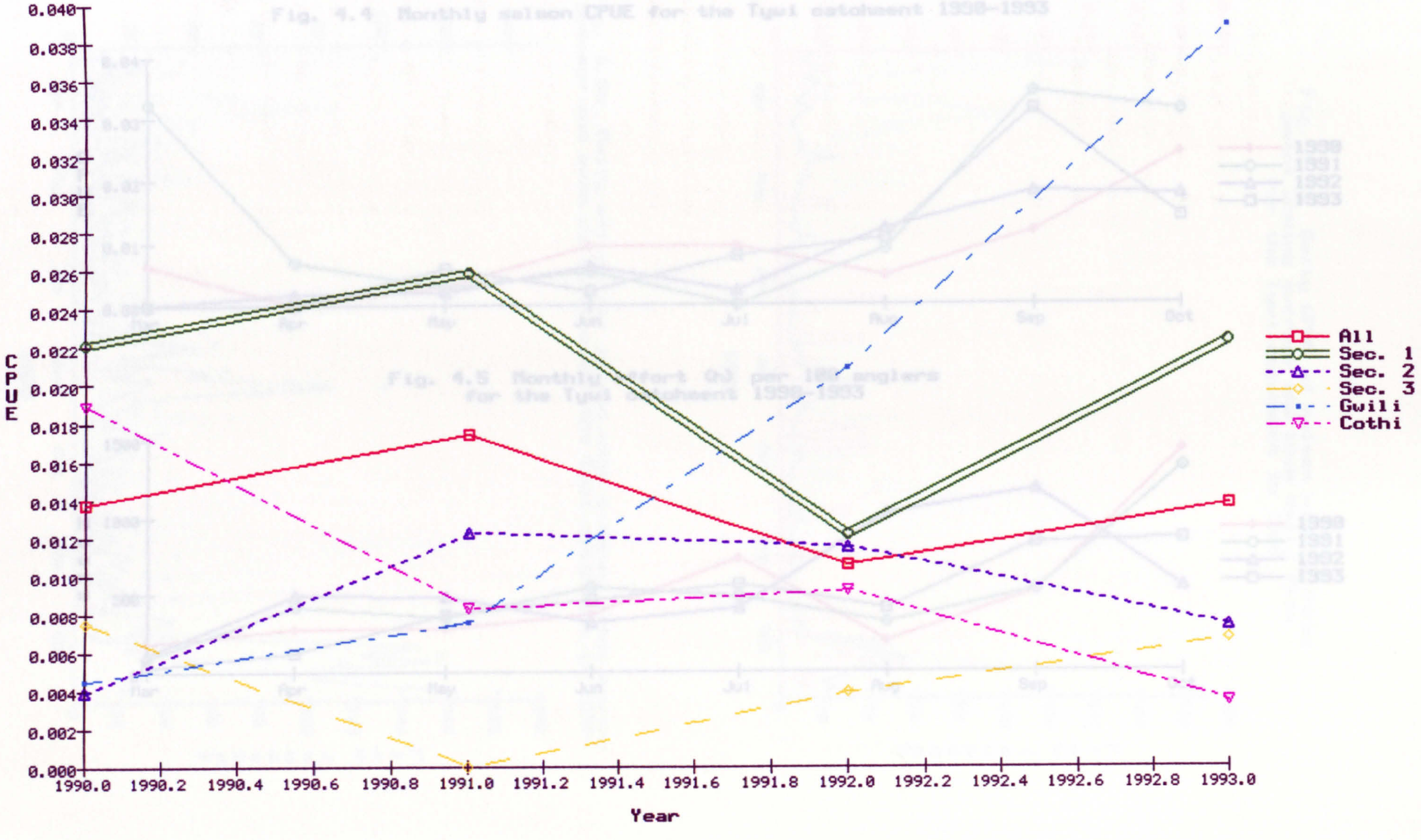


Fig. 4.4 Monthly salmon CPUE for the Tywi catchment 1990-1993

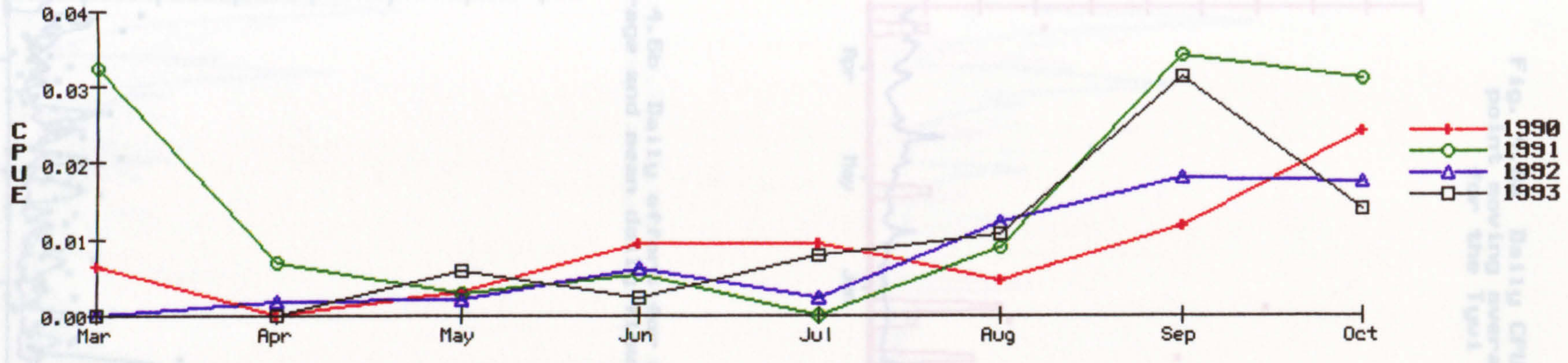


Fig. 4.5 Monthly effort (h) per 100 anglers for the Tywi catchment 1990-1993

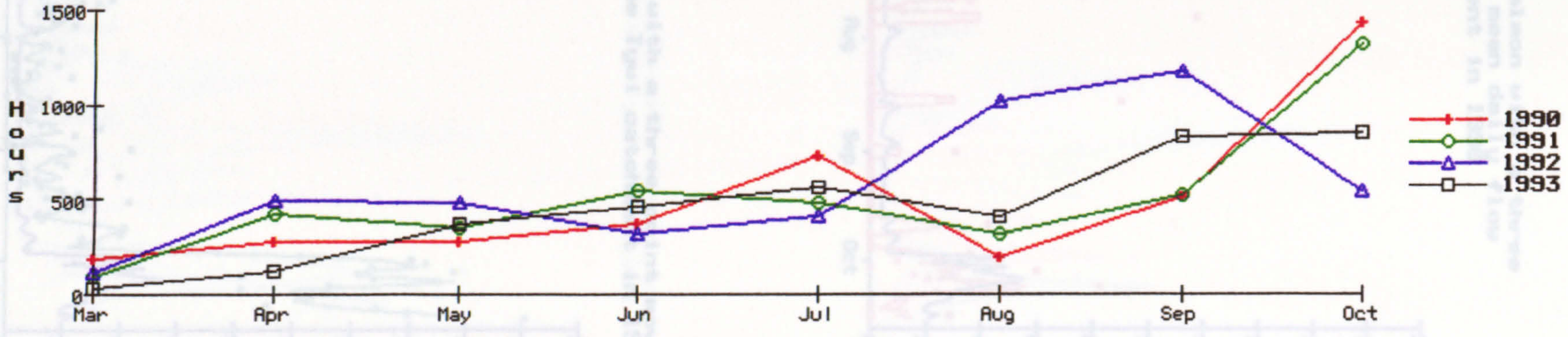


Fig. 4.6a Daily CPUE of salmon with a three point moving average and mean daily flow for the Tywi catchment in 1990

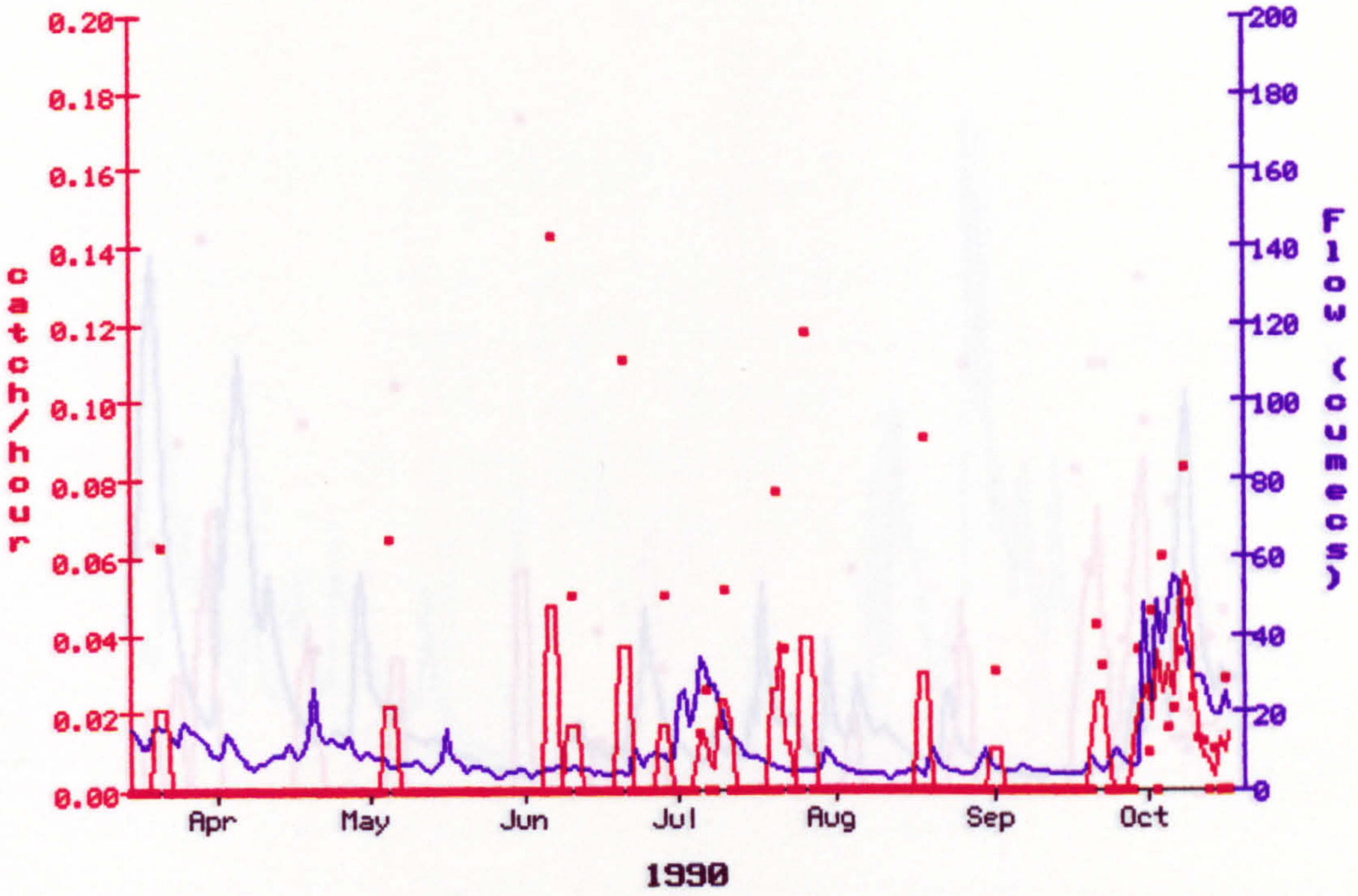


Fig. 4.6b Daily effort for salmon with a three point moving average and mean daily flow for the Tywi catchment in 1990

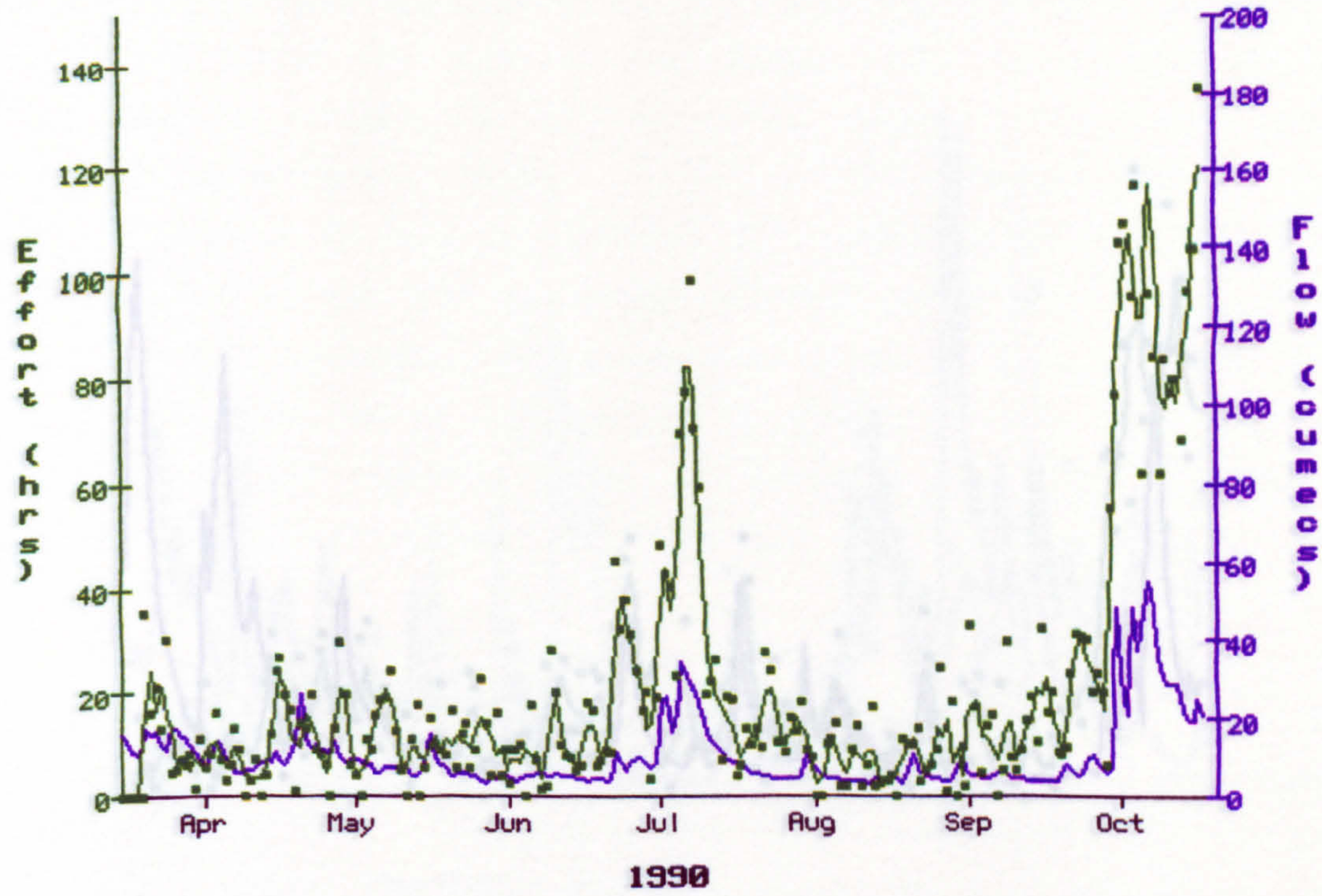


Fig. 4.7a Daily CPUE of salmon with a three point moving average and mean daily flow for the Tywi catchment in 1991

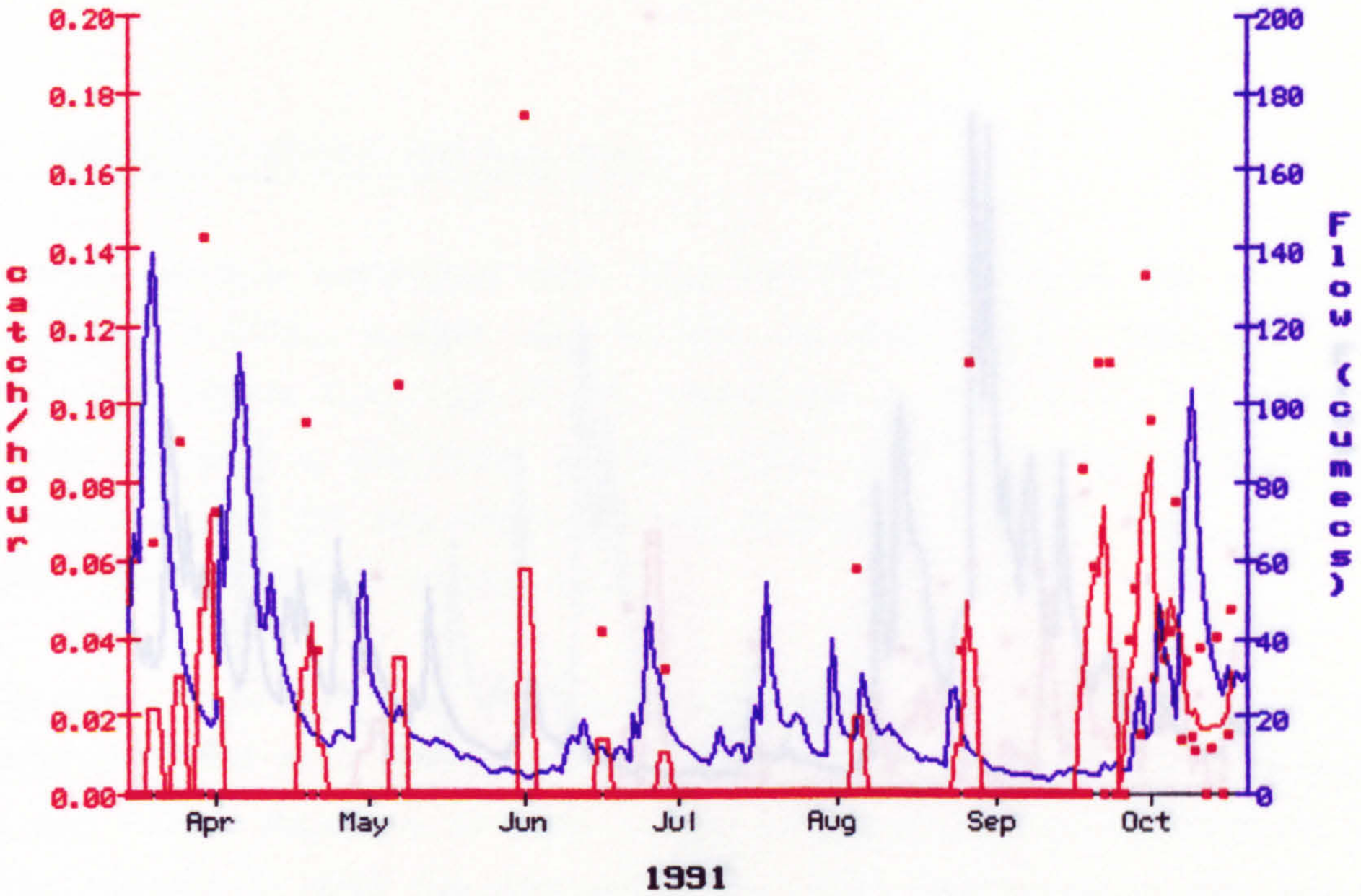


Fig. 4.7b Daily effort for salmon with a three point oving average and mean daily flow for the Tywi catchment in 1991

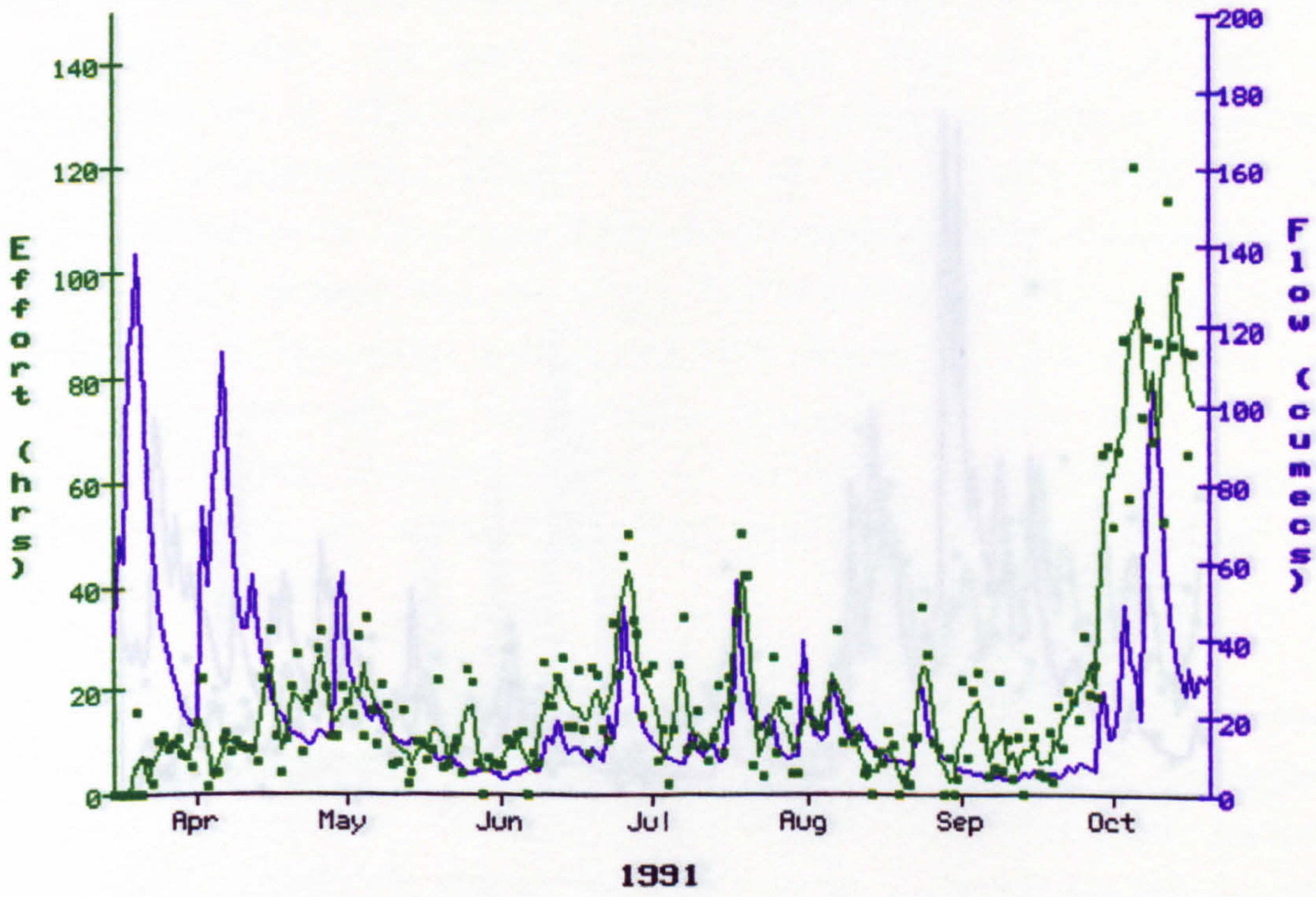


Fig. 4.8a Daily CPUE of salmon with a three point moving average and mean daily flow for the Tywi catchment in 1992

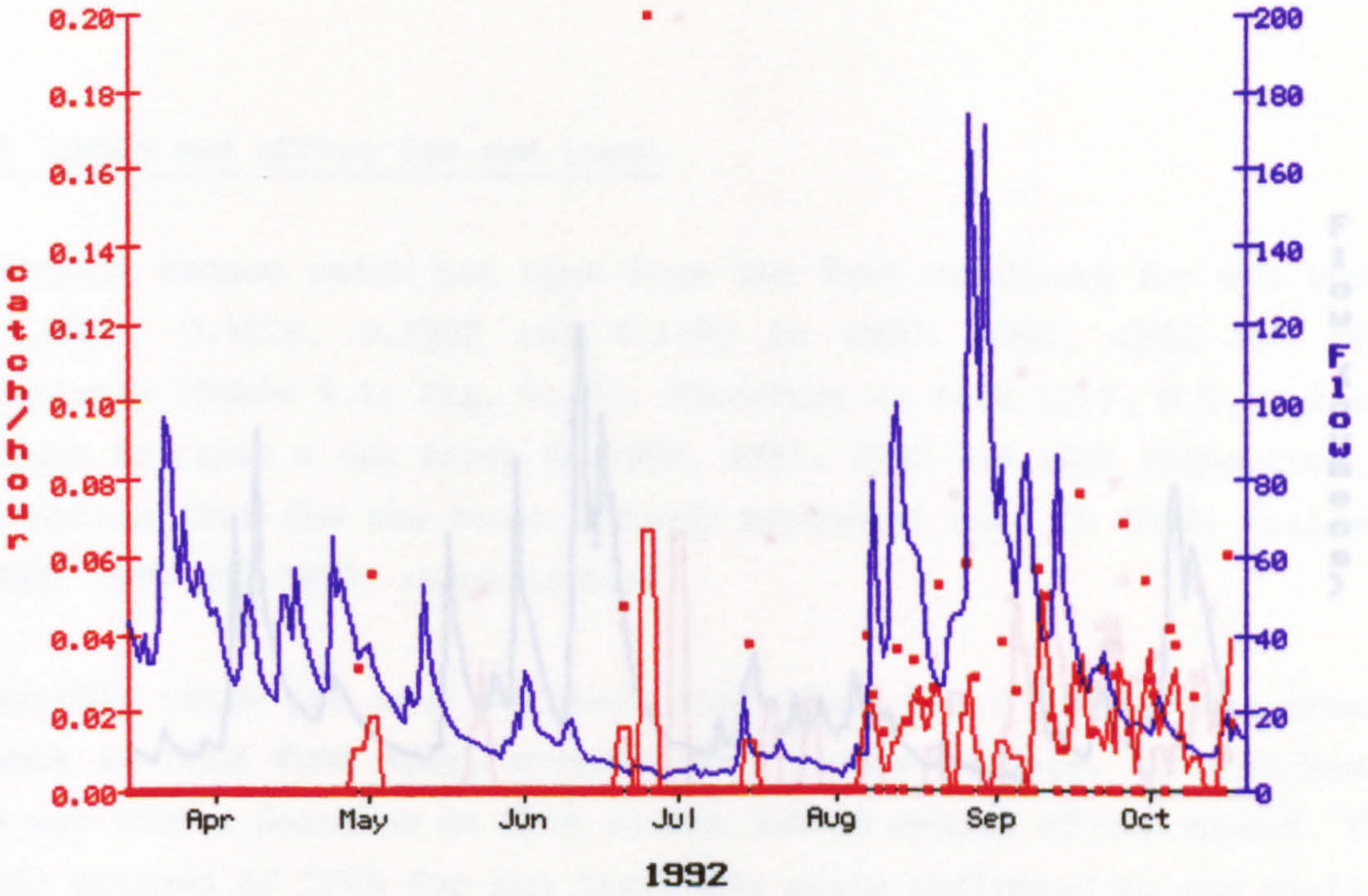


Fig. 4.8b Daily effort for salmon with a three point moving average and mean daily flow for the Tywi catchment in 1992

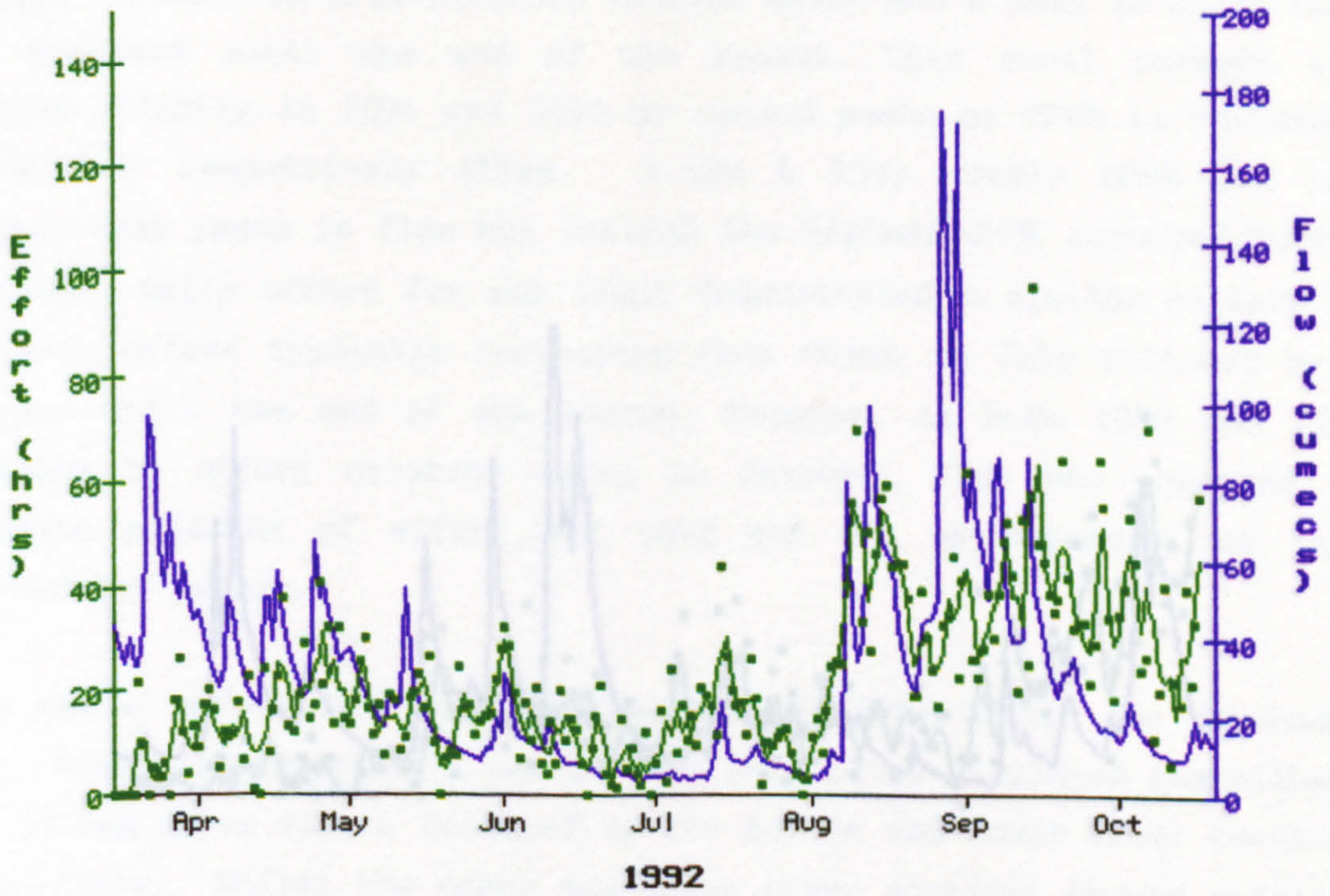


Fig. 4.9a Daily CPUE of salmon with a three point moving average and mean daily flow for the Tywi catchment in 1993

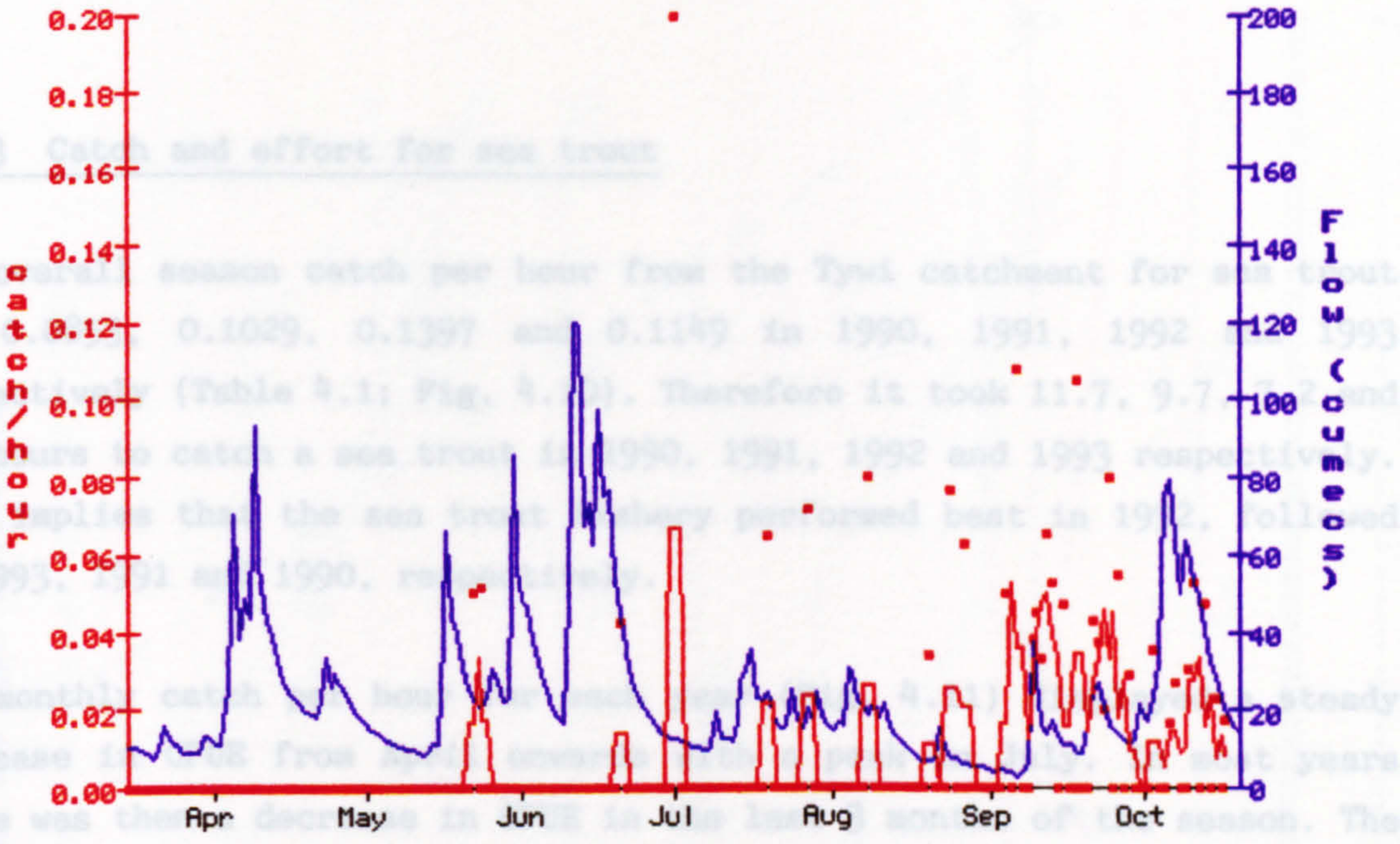
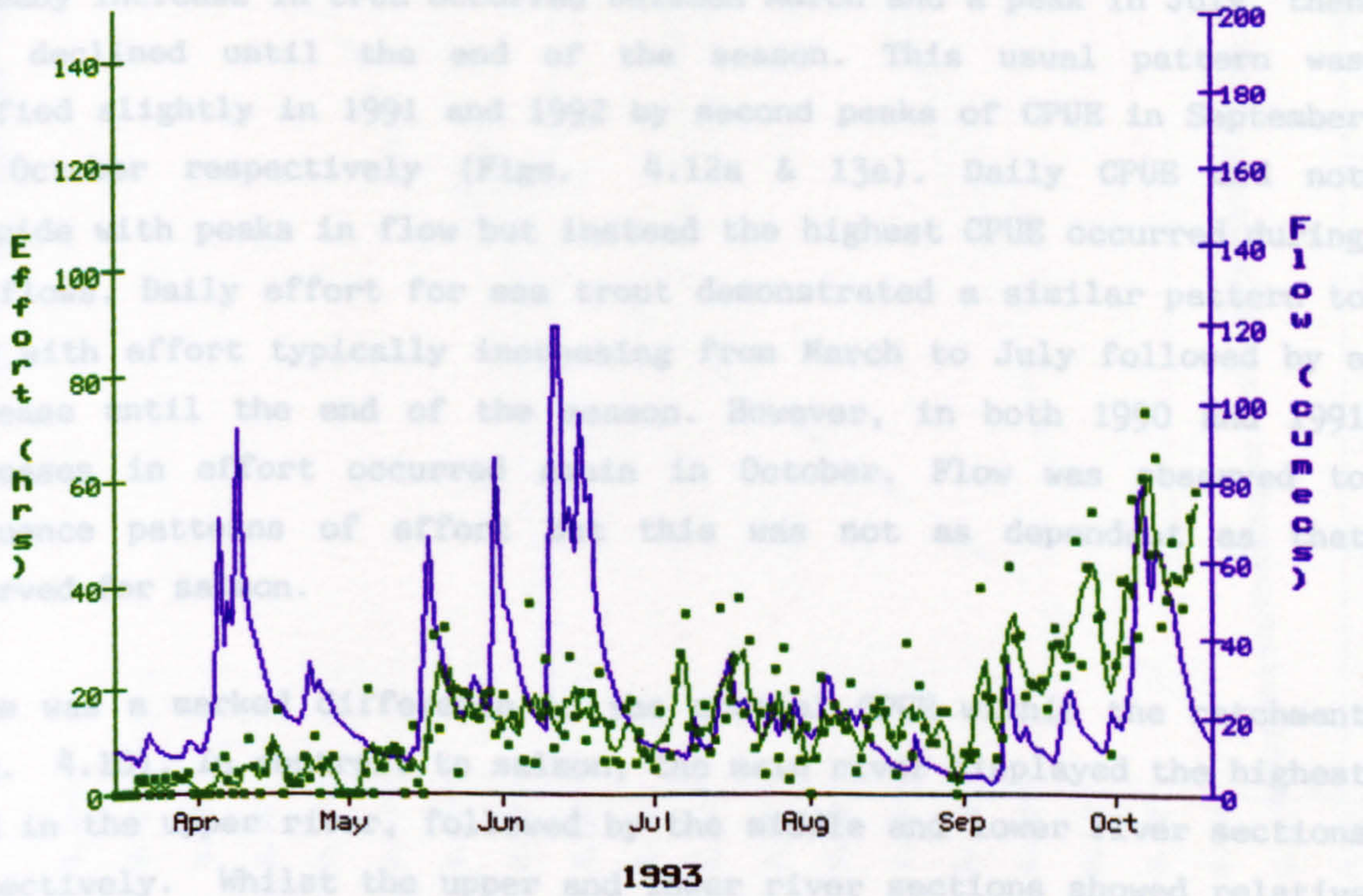


Fig. 4.9b Daily effort for salmon with a three point moving average and mean daily flow for the tywi catchment in 1993



The Cothi displayed a decreasing trend over the four years with a CPUE slightly less than the middle river section. The Gwili displayed a low CPUE in the dry years of 1990 and 1991 but increased in the wet years of 1992 and 1993.

4.3.3 Catch and effort for sea trout

The overall season catch per hour from the Tywi catchment for sea trout was 0.0853, 0.1029, 0.1397 and 0.1149 in 1990, 1991, 1992 and 1993 respectively (Table 4.1; Fig. 4.10). Therefore it took 11.7, 9.7, 7.2 and 8.7 hours to catch a sea trout in 1990, 1991, 1992 and 1993 respectively. This implies that the sea trout fishery performed best in 1992, followed by 1993, 1991 and 1990, respectively.

The monthly catch per hour for each year (Fig. 4.11) displayed a steady increase in CPUE from April onwards with a peak in July. In most years there was then a decrease in CPUE in the last 3 months of the season. The monthly pattern of CPUE for sea trout was again reflected by the similar pattern of effort for sea trout (Fig. 4.12).

The daily CPUE analysis for each year (Figs. 4.13a-16a) demonstrated the early timing of the sea trout run, with fish captured in March and April. A steady increase in CPUE occurred between March and a peak in July, then CPUE declined until the end of the season. This usual pattern was modified slightly in 1991 and 1992 by second peaks of CPUE in September and October respectively (Figs. 4.12a & 13a). Daily CPUE did not coincide with peaks in flow but instead the highest CPUE occurred during low flows. Daily effort for sea trout demonstrated a similar pattern to CPUE with effort typically increasing from March to July followed by a decrease until the end of the season. However, in both 1990 and 1991 increases in effort occurred again in October. Flow was observed to influence patterns of effort but this was not as dependent as that observed for salmon.

There was a marked difference in the spatial CPUE within the catchment (Fig. 4.10). In contrast to salmon, the main river displayed the highest CPUE in the upper river, followed by the middle and lower river sections respectively. Whilst the upper and lower river sections showed relative stability over the four years, the middle river displayed a marked

Fig. 4.10 Annual sea trout CPUE from the Tywi catchment

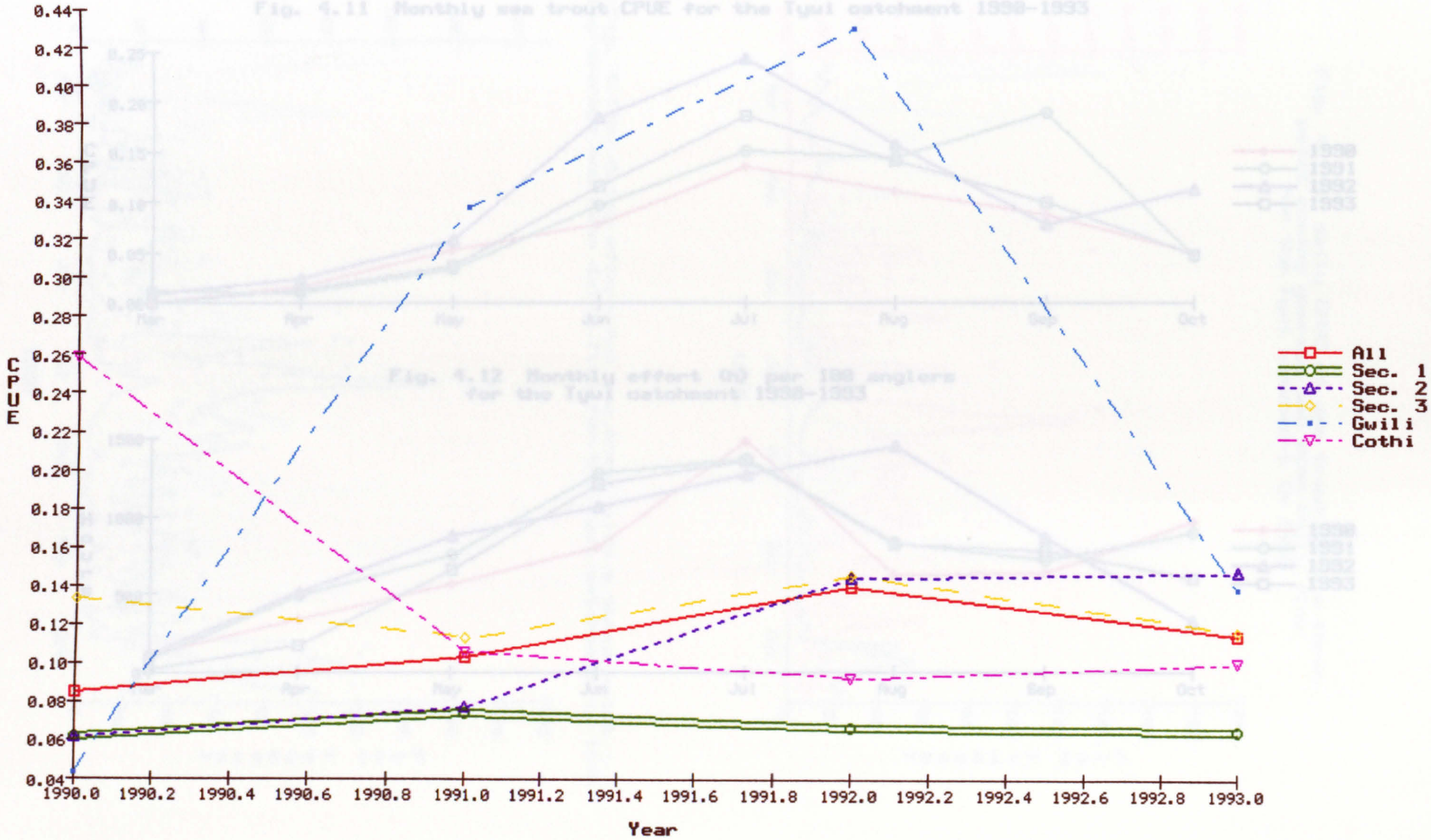


Fig. 4.11 Monthly sea trout CPUE for the Tywi catchment 1990-1993

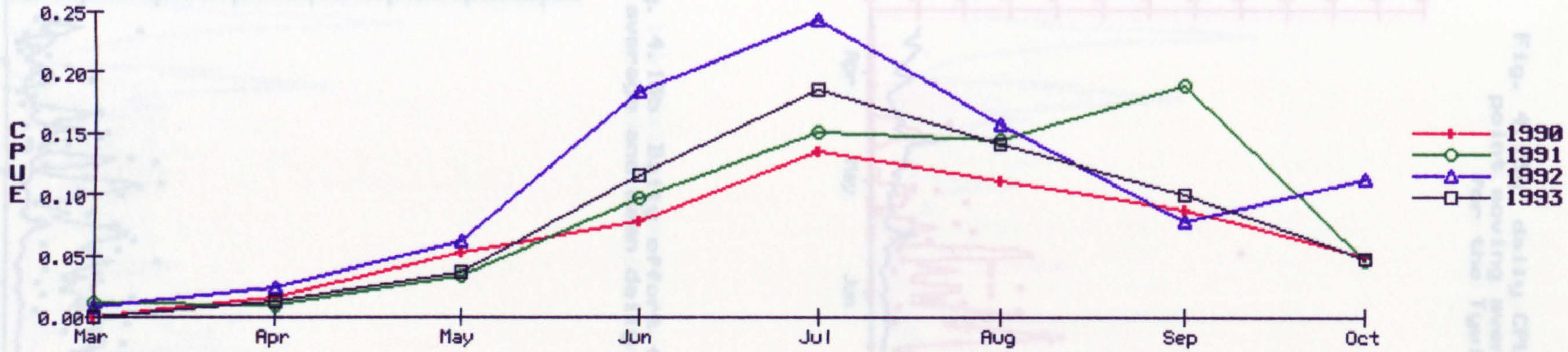


Fig. 4.12 Monthly effort (h) per 100 anglers for the Tywi catchment 1990-1993

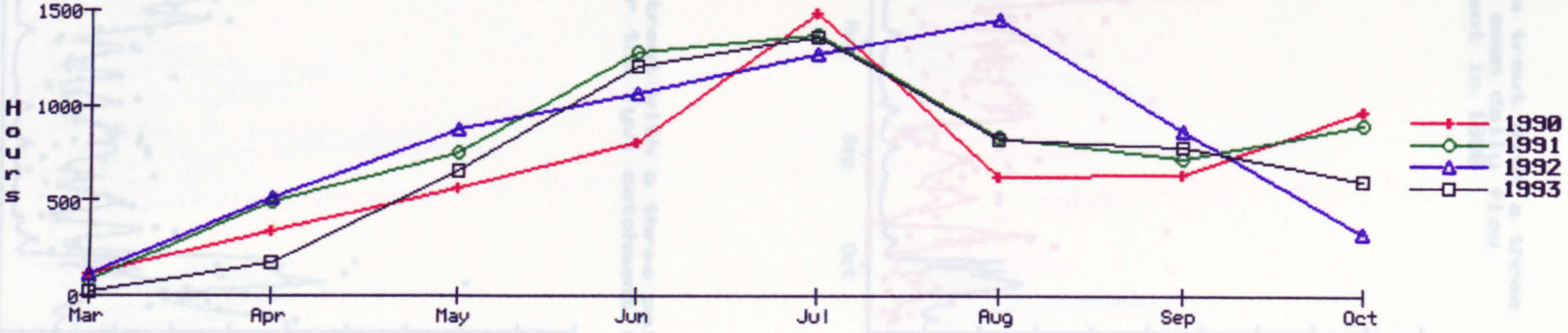
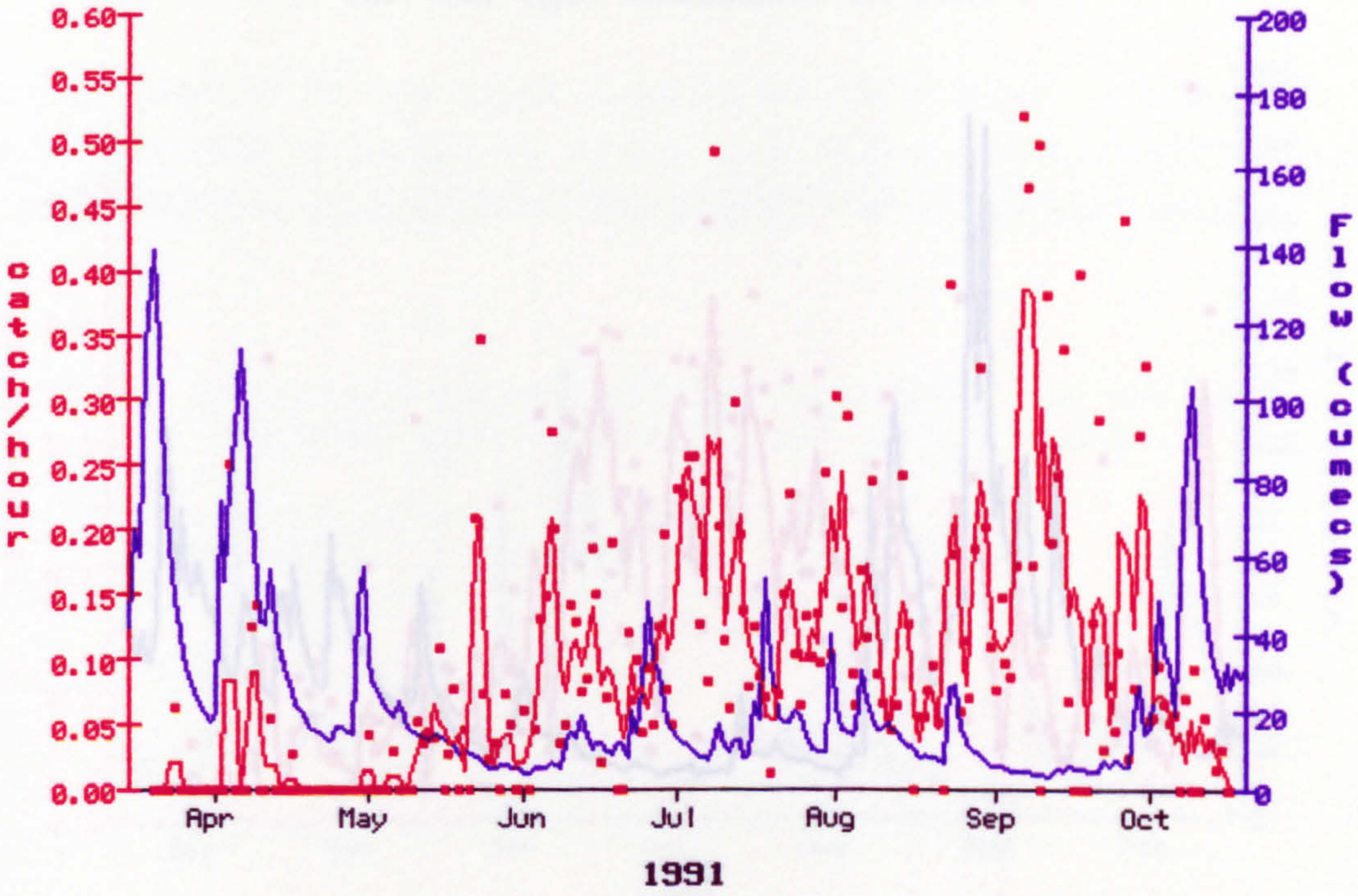


Fig. 4.14a Daily CPUE of sea trout with a three point moving average and mean daily flow for the Tywi catchment in 1991



ig. 4.14b Daily effort for sea trout with a three point moving average and mean daily flow for the Tywi catchment in 1991

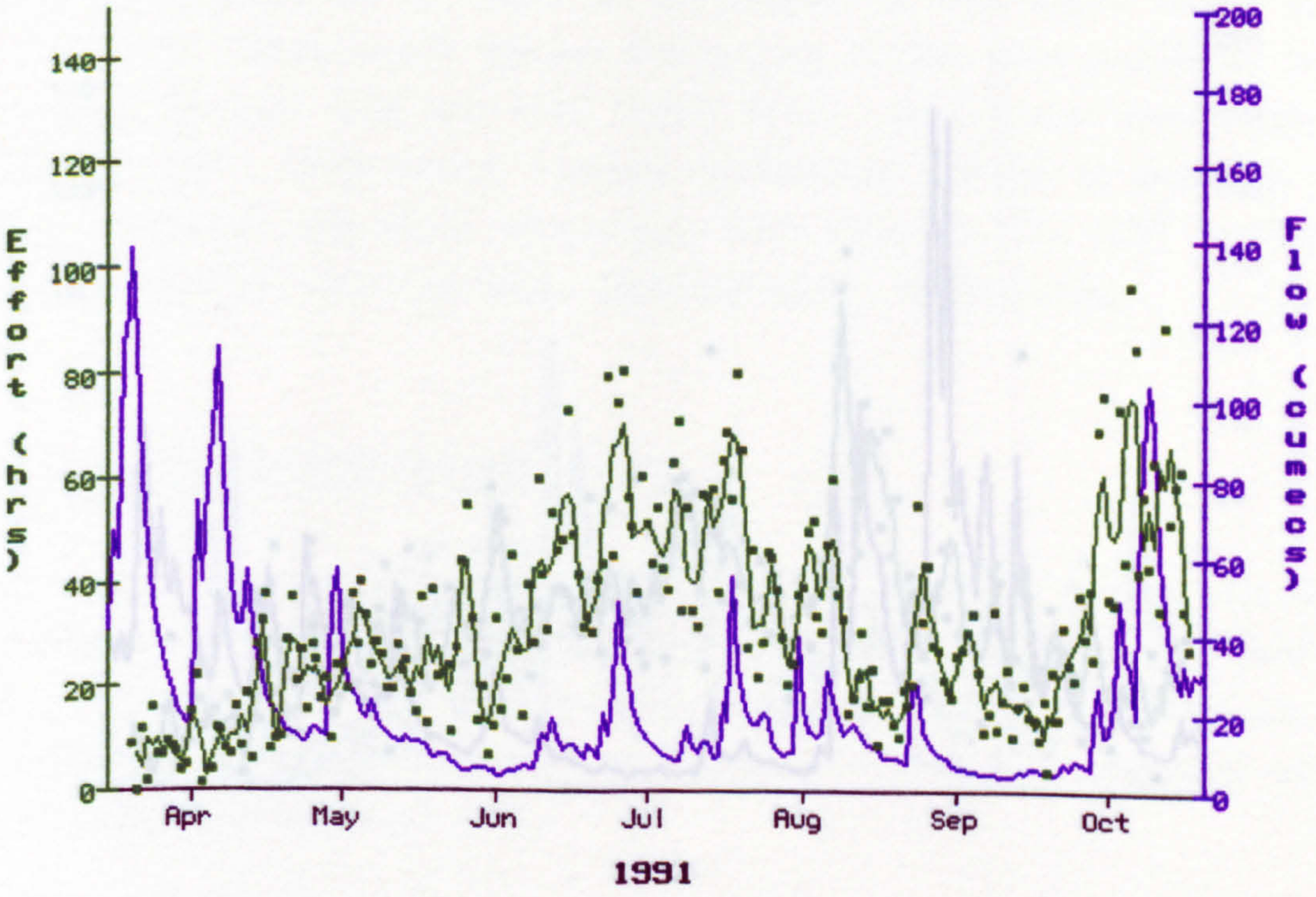


Fig. 4.15a Daily CPUE of sea trout with a three point moving average and mean daily flow for the Tywi catchment in 1992

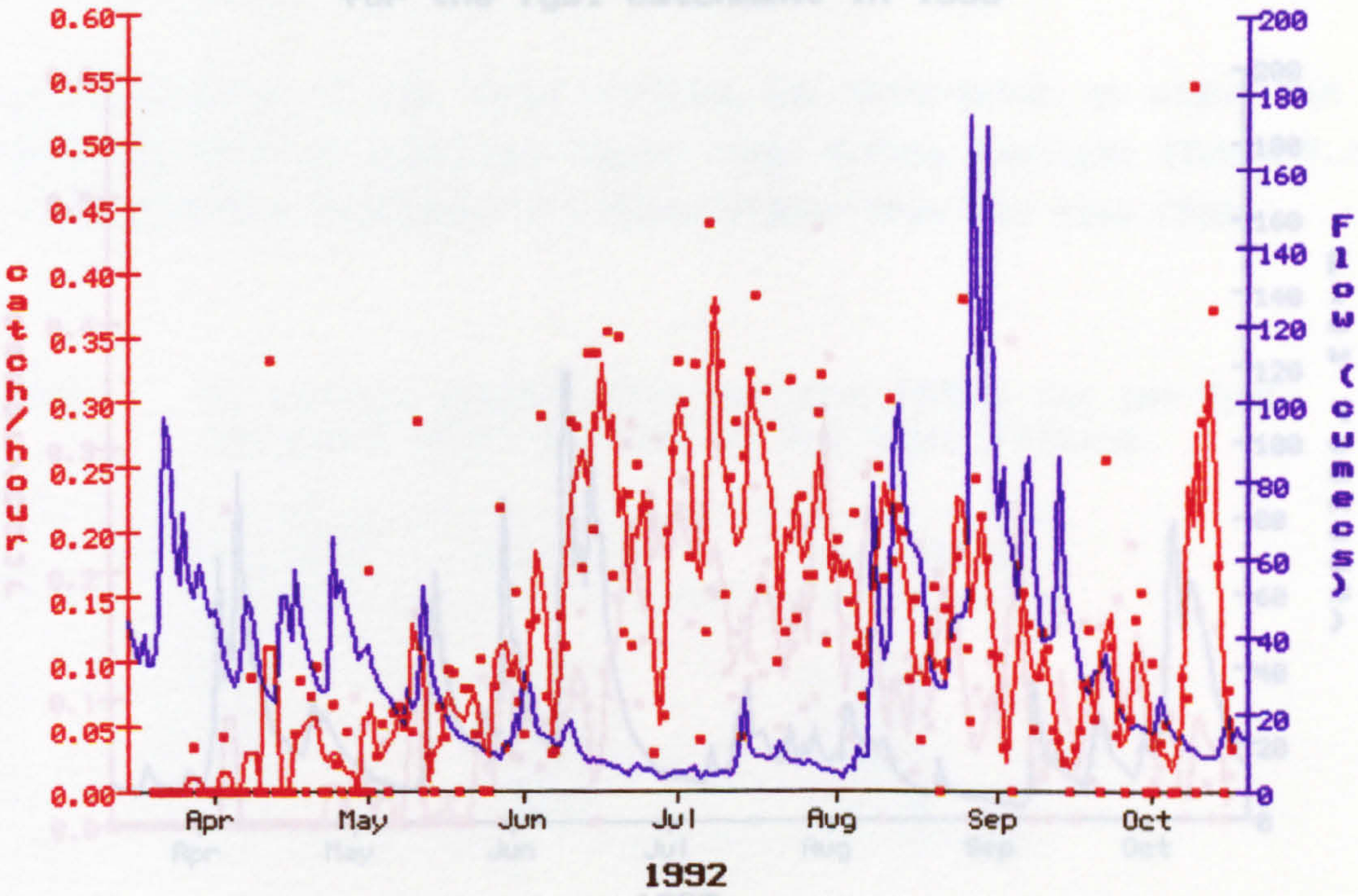


Fig. 4.15b Daily effort for sea trout with a three point moving average and mean daily flow for the Tywi catchment in 1992

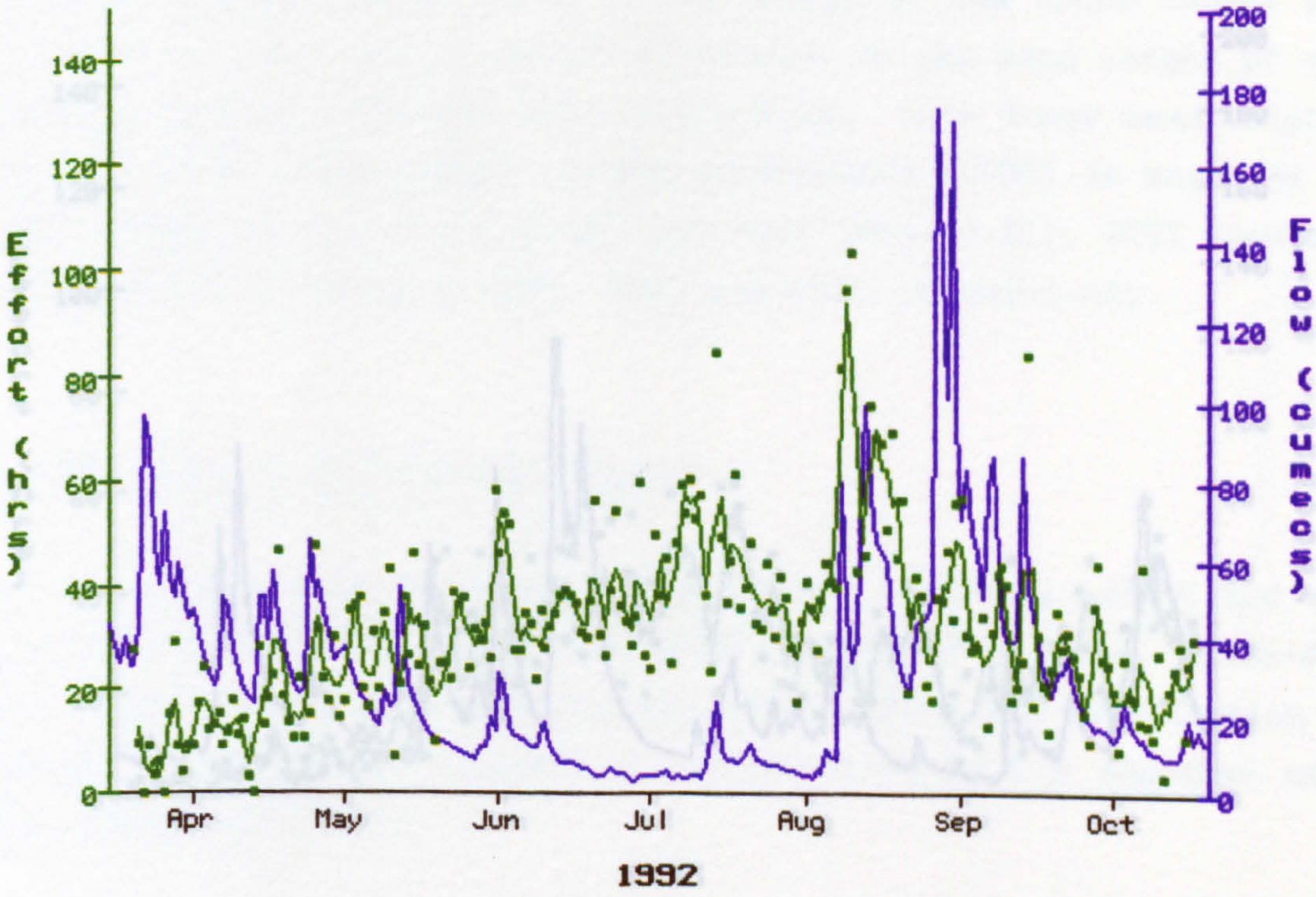


Fig. 4.16a daily CPUE of sea trout with a three point moving average and mean daily flow for the Tywi catchment in 1993

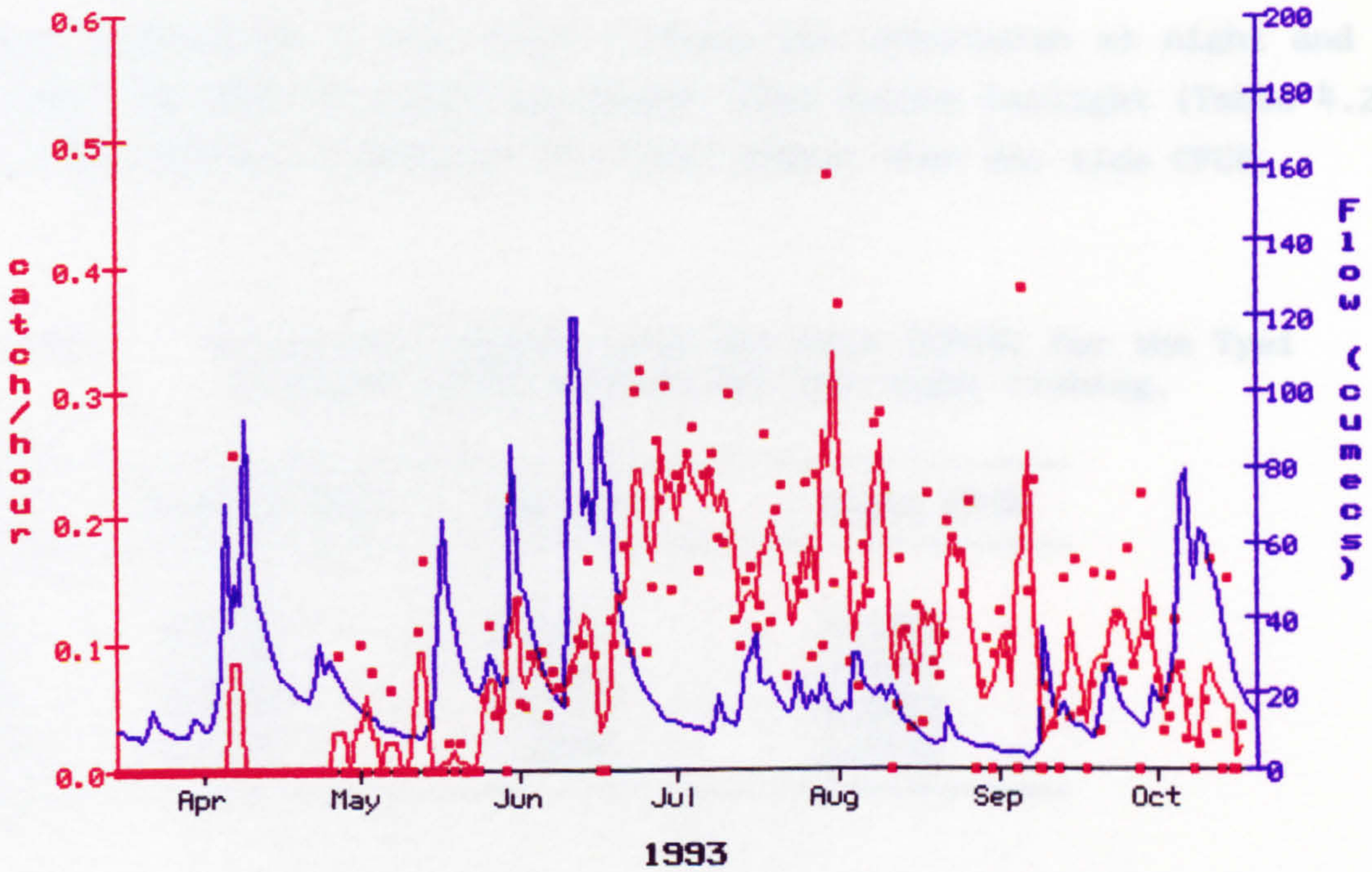
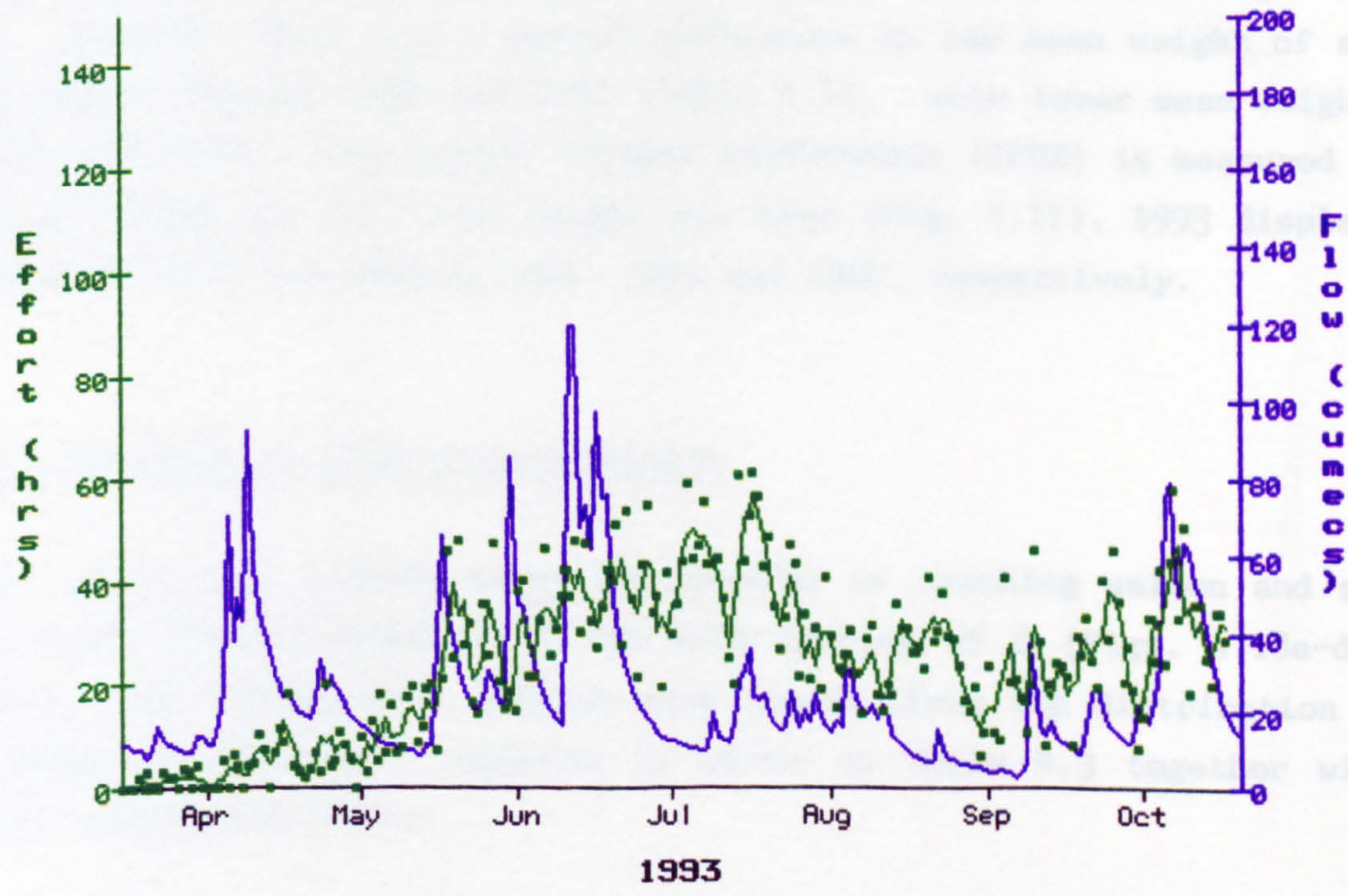


Fig. 4.16b daily effort for sea trout with a three point moving average and mean daily flow for the Tywi catchment in 1993



increase between 1990 and 1993. The Cothi displayed a decreasing trend over the four years with a CPUE generally less than the middle river section. The Gwili displayed a low CPUE in 1990 and 1993 but large increases in CPUE in 1991 and 1992.

A large proportion of sea trout fishing was undertaken at night and in each year the CPUE at night was higher than during daylight (Table 4.2). Night time CPUE was typically 2-3 times higher than day time CPUE.

Table 4.2. The overall season catch per hour (CPUE) for the Tywi catchment split between day and night fishing.

Year	Overall CPUE	Day CPUE	Night CPUE
1990	0.0853	0.0542	0.1803
1991	0.1029	0.0802	0.1684
1992	0.1397	0.1018	0.2654
1993	0.1149	0.0906	0.2132

4.3.4 Sea trout CPUE in terms of weight

The CPUE given above is in terms of the number of sea trout caught per hour. However, there was a marked difference in the mean weight of sea trout caught between 1990 and 1993 (Table 4.1), with lower mean weights in 1991 and 1992. When annual fishery performance (CPUE) is measured in terms of weight of sea trout caught per hour (Fig. 4.17), 1993 displays the highest CPUE followed by 1992, 1991 and 1990, respectively.

4.3.5 Variation in CPUE between anglers

In all years most anglers were unsuccessful in catching salmon and sea trout with a modal annual CPUE for both species of 0 (Figs. 4.18a-d & 4.19a-d). The overall mean CPUE in each season (from the distribution of the means of individual anglers) is given in Table 4.3 together with poisson confidence limits.

Fig. 4.17 Tywi catchment catch effort data for sea trout, CPUE in terms of number of fish/hour and weight of fish/h

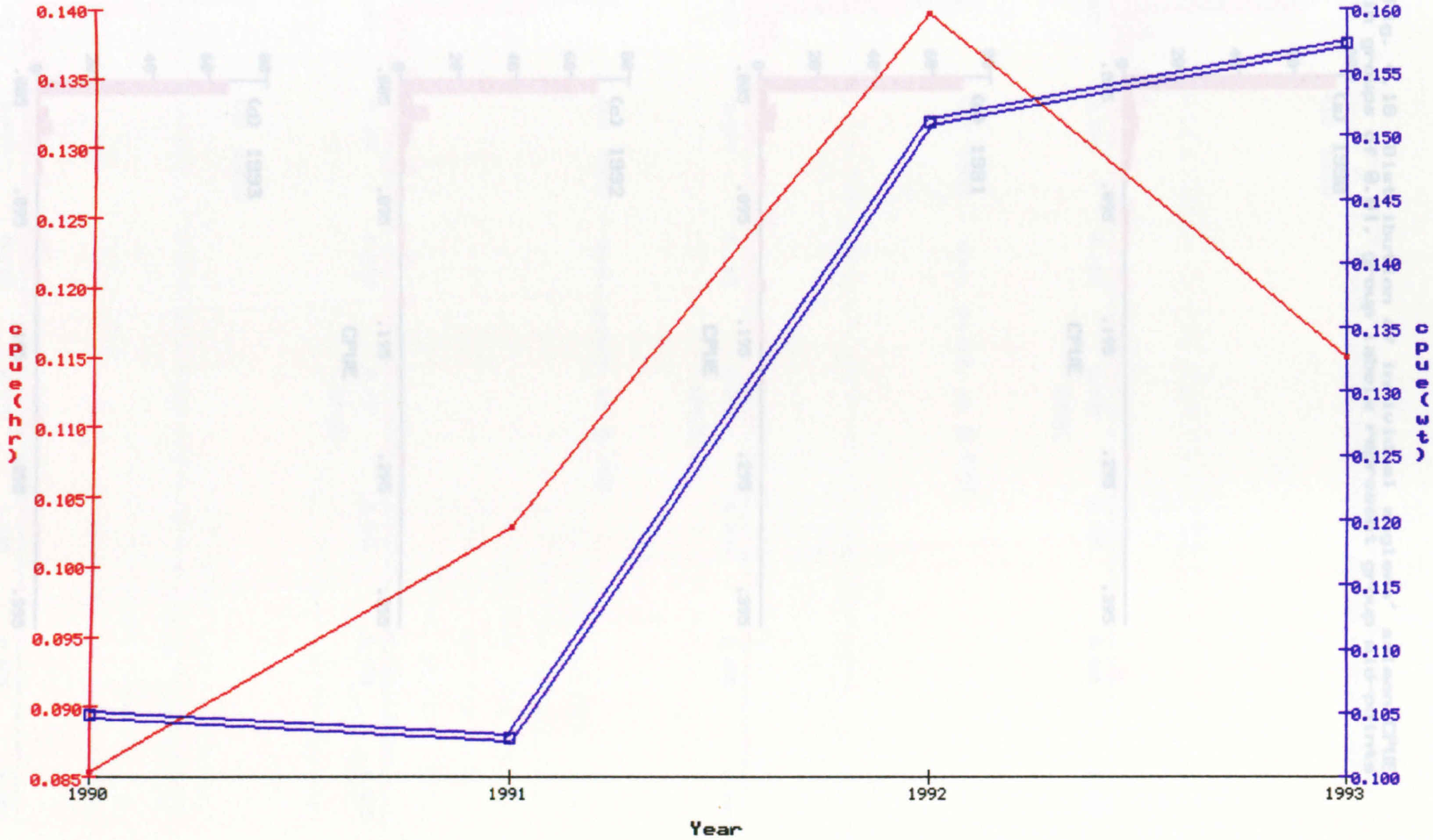


Fig. 4.18 Distribution of individual anglers' salmon CPUE in groups of 0.01, group labels represent group mid-points

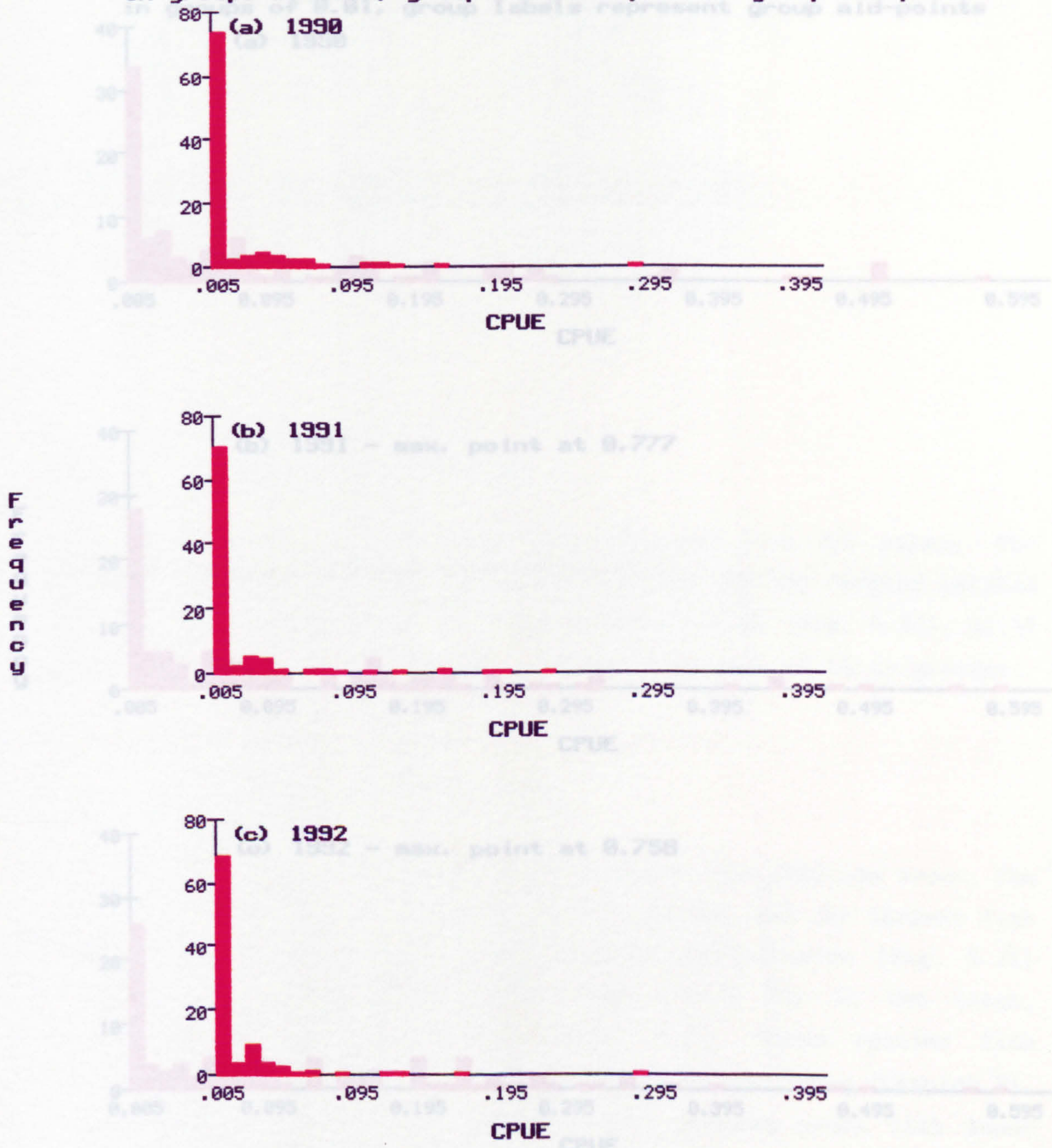


Fig. 4.19 Distribution of individual anglers' sea trout CPUE in groups of 0.01, group labels represent group mid-points

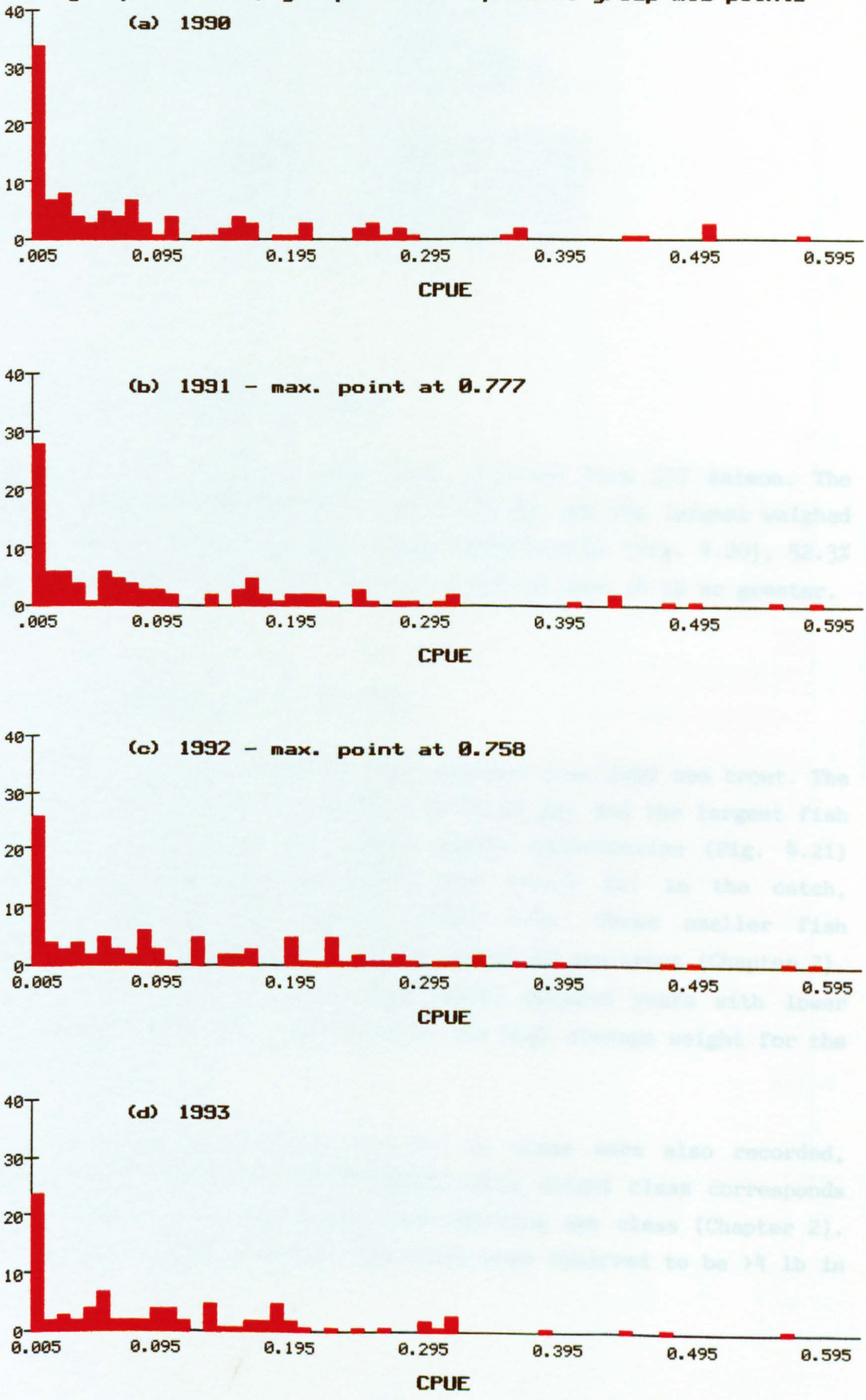


Table 4.3. The mean CPUE from individual anglers for salmon and sea trout in 1990-1993. Poisson 95% CI as mean +/- 1.96(SQR(mean/no. observations))

Year	Sea trout		Salmon	
	mean	95% CI	mean	95% CI
1990	0.1063	+/- 0.0596	0.0186	+/- 0.0262
1991	0.1111	+/- 0.0641	0.0200	+/- 0.0279
1992	0.1286	+/- 0.0696	0.0150	+/- 0.0242
1993	0.1093	+/- 0.0683	0.0164	+/- 0.0271

4.3.6 Weight distribution of salmon

Over the four year period weights were obtained from 277 salmon. The smallest salmon recorded weighed 2.2 lb (0.99 kg) and the largest weighed 22 lb (9.98 kg). From the overall weight distribution (Fig. 4.20), 52.3% of salmon were <8 lb, 39.7% were 8-14 lb and 8.0% were 14 lb or greater.

4.3.7 Weight distribution of sea trout

Over the four year period weights were obtained from 2959 sea trout. The smallest sea trout recorded weighed 6 oz (0.17 kg) and the largest fish weighed 15 lb (6.81 kg). The overall weight distribution (Fig. 4.21) demonstrates the dominance of small fish (=<1.5 lb) in the catch, representing 43.8% of the sample (Table 4.4). These smaller fish correspond to the whitling or .+ sea age group of sea trout (Chapter 2). The relative abundance of these fish varied between years with lower relative abundance in 1993, reflected in the high average weight for the year (Table 4.1).

Appreciable numbers of fish in the 3-4 lb class were also recorded, representing 35.5% of the overall sample. This weight class corresponds primarily to the .1+ or one winter post-smolting age class (Chapter 2). From the overall sample 20.7% of sea trout were observed to be >4 lb in weight.

Fig. 4.20 Weight distribution of Tywi catchment salmon from anglers catches 1990-1993 (n=277)

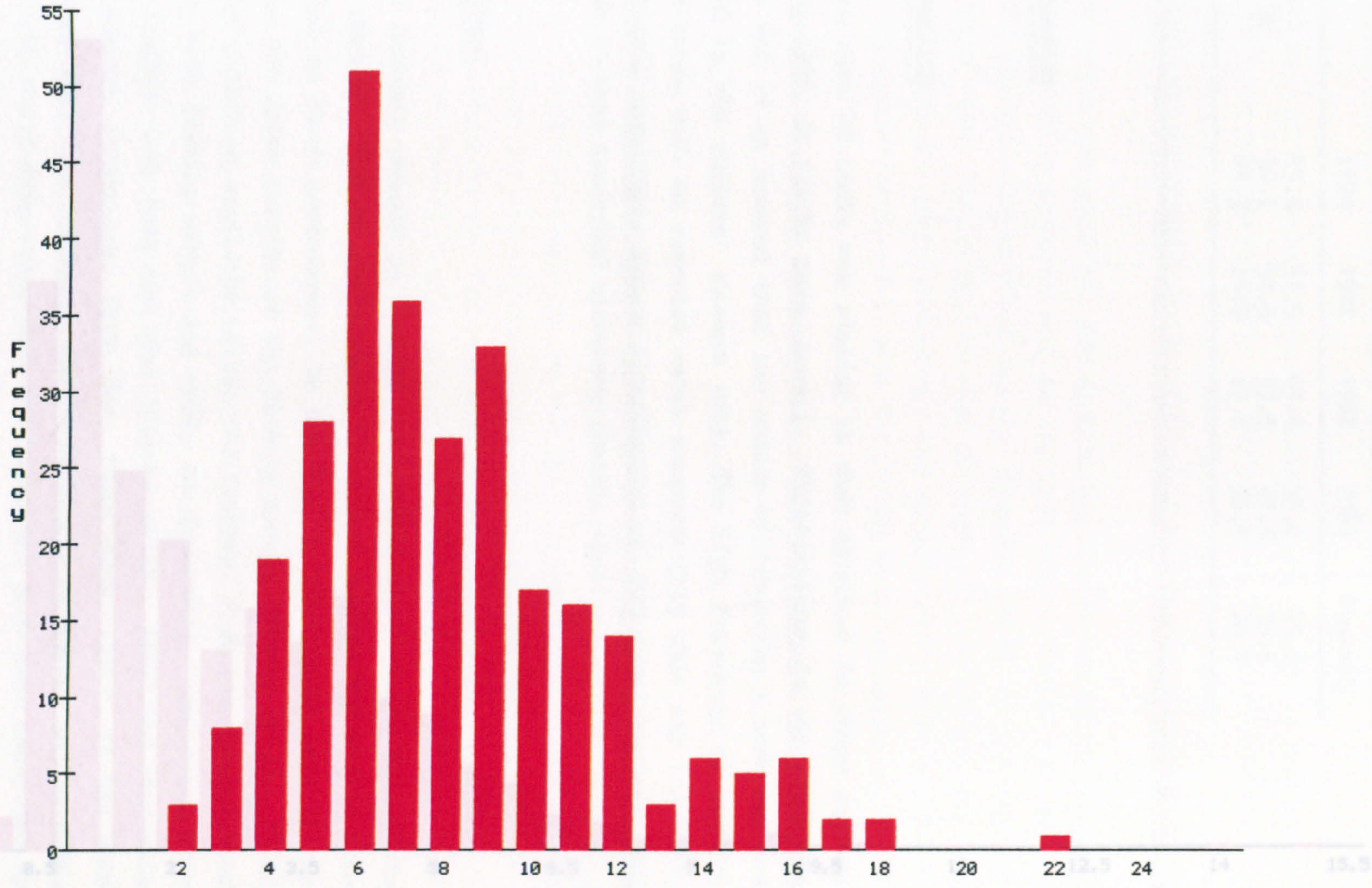


Fig. 4.21 Weight distribution of Tywi catchment sea trout from anglers catches 1990-1993 (n=2959)

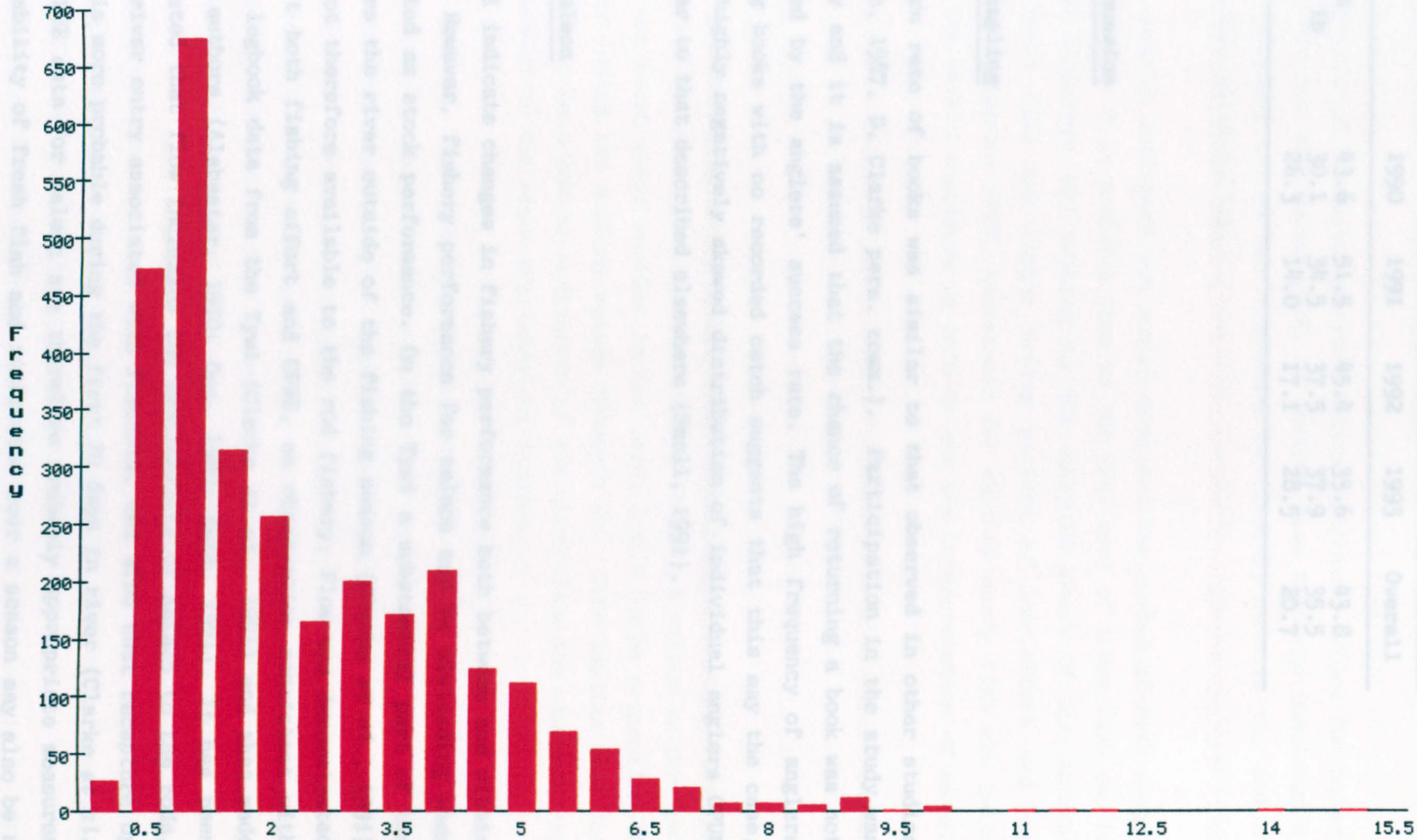


Table 4.4. Weight distribution (%) of sea trout in the classes =<1.5 lb, >1.5 lb-4 lb and >4 lb. (Overall n=2959)

Class	1990	1991	1992	1993	Overall
=< 1.5 lb	43.6	51.5	45.4	33.6	43.8
1.5 lb-4 lb	30.1	34.5	37.5	37.9	35.5
> 41b	26.3	14.0	17.1	28.5	20.7

4.4 Discussion

4.4.1 Sampling

The return rate of books was similar to that observed in other studies (Wightman, 1987, D. Clarke pers. comm.). Participation in the study was voluntary and it is assumed that the chance of returning a book was not influenced by the anglers' success rate. The high frequency of anglers returning books with no recorded catch suggests that this may the case, and the highly negatively skewed distribution of individual anglers CPUE is similar to that described elsewhere (Small, 1991).

4.4.2 Salmon

CPUE will indicate changes in fishery performance both between and within seasons. However, fishery performance for salmon may be misleading when interpreted as stock performance. On the Tywi a substantial part of the run enters the river outside of the fishing season (Clarke *et al.*, 1991) and is not therefore available to the rod fishery. Flow was demonstrated to affect both fishing effort and CPUE, an observation consistent with previous logbook data from the Tywi (Clarke *et al.*, 1991) and that made by many authors (Alabaster, 1970; Gee, 1980; Bunt, 1991). It has been demonstrated that flow increases the availability of salmon to the rods, through river entry associated with freshets, and also that recapture by anglers is more probable during the first 20 days in river (Clarke *et al.* 1991). CPUE data for salmon are therefore probably appropriate measures of availability of fresh fish and integrated over a season may also be a useful measure of in-season availability. However, angling CPUE may be a

poor indication of stock performance on the Tywi, and where used should be interpreted with respect to flow and an assessment of the out of season run.

The observation of a similar pattern between anglers' effort for salmon and flow, and the associated CPUE, is derived from historic knowledge by anglers. Angling effort may therefore be directly related to, and an indicator of, catchability.

A factor seen to influence the annual data was the number of fish caught in March/April. It is probable that on the Tywi many of these fish may be kelts and therefore add nothing to the spawning stock of the current season. These fish are caught during periods of low effort and can increase the apparent CPUE. Assessment for whether early fish are kelts or fresh run should therefore be made to aid the interpretation of annual CPUE.

4.4.3 Sea trout

As for salmon, sea trout CPUE will indicate trends in fishery performance. The main difference between sea trout and salmon on the Tywi is that sea trout enter earlier in the year, and a large proportion of fish enter during the angling season (Chapter 2). This implies that sea trout CPUE may be a better indication of the stock from the viewpoint of the proportion of the stock available for capture.

Although reduced flow has been demonstrated to reduce the success of entry to river by larger sea trout (D.Clarke pers. comm.), many of the smaller sea trout (whitling) enter in July-August when flows may be lowest. This confuses any relationship between flow and availability. It has also been shown that sea trout remain available to capture throughout the season independent of flow (D.Mee pers. comm.), therefore confusing contemporary relationships of flow and availability.

The month of the year was found to have a greater influence upon fishing effort and CPUE than river flow, both effort and CPUE generally increasing from March to July and then decreasing to October. The CPUE data suggest that flow may in fact have a negative relationship to CPUE since CPUE was typically highest during the lower flows of July. This is

where angling method further confuses the issue of sea trout capture and flow. The principle method used during low flows is fly fishing at night, and night time CPUE was demonstrated to be 2-3 times higher than day time CPUE. This implies that sea trout are either more catchable at night or that the more successful anglers fish at night.

4.4.4 CPUE and abundance

The relationship between catch, effort and abundance is given by:

Catch = Effort x Catchability x Abundance eqn 1 (Paloheimo & Dickie, 1964)

CPUE = Catchability x Abundance eqn 2

The relationship between CPUE and abundance is therefore influenced by catchability. If catchability is constant between years, or follows the same temporal pattern between years, then CPUE is linearly related to abundance. However, other studies indicate that this may not be the case and that catchability increases with decreasing stock levels (Mills *et al.* 1986). Small (1991) suggests a relationship of CPUE with abundance as proportioned to between the cube and square roots of the number of fish entering a river; implying that to double the CPUE would require a 4-8 fold increase in the sea trout run. A better understanding of the relationship between angling CPUE and fish abundance will be available through studies where abundance can be assessed through trap or counter systems (Aprahamian, 1993; I. Davidson pers. comm.)

Anglers in this study used all legal fishing methods viz. fly, spinning and bait. As well as the variation in anglers skill there may be a difference in the efficiency of the different gears *per se*, and this will affect catchability and therefore the catch for each unit of effort. Whilst the effort recorded in this study was not separated between gears, changes to future recording methods could enable this. If differences do occur between methods then this could be used to assess any changes in future patterns of fishing methods or any regulatory restriction such as that currently used for spring salmon (Mawle, 1992).

Log books allow a measure of fishery performance accounting for the variation in effort not afforded by total catch figures. However, this

principle assumes that each unit of effort is independent and it does not consider the possible 'congestion externalities' that may occur. Some association fisheries are heavily fished and it is therefore possible that gear saturation occurs and anglers 'interfere' with each other, reducing the probability of capture for the next angler through disturbance of the water. This would reduce CPUE within a given area with increased angling effort; thus with the same abundance of fish in a given area, reducing the angling effort (e.g. private/syndicate waters) may increase CPUE. This has implications for annual CPUE data if major changes occur in the pattern of usage over time. A combined index of CPUE and an index of the relative changes in the pattern of fishing effort may therefore indicate long term trends better than CPUE alone, though assessing the total number of anglers may be difficult.

A factor that also needs to be considered is the constancy of availability of fish within the rod fishery throughout the season. This is particularly important when assessing the proportion of fish caught or catchability from abundance estimates using traps or counters. Often it is assumed that once fish have entered the river they will remain within the fishery, but in latter months, especially for sea trout, fish may migrate into larger tributaries where rod fisheries do not occur. This effectively decreases the abundance of sea trout within the rod fishery.

The number of different sea ages at return is a peculiar facet of sea trout biology. It has been stated that rods may not represent the proportion of large fish present (LeCren, 1985), and therefore a knowledge of whether different catchabilities exist between different sea aged fish is needed. This will aid the interpretation of the changes in annual CPUE with the relative differences of these sea ages within the stock.

4.4.5 Effect of the mixed fishery

During many trips anglers reported fishing for both species at the same time and for the analysis this effort was allocated to both salmon and sea trout. Fishing for both species at the same time is most common with one particular method of fishing, i.e. spinning (pers.obs.), where anglers perceive a chance of catching either species. However, due to the different temporal availability of the species this may not be a true

reflection of effort and this can bias the overall trend of effort and CPUE. This particularly affects the effort pattern for salmon; whilst sea trout may be available between April and June and thus spinning effort during this period will be productive, the majority of salmon do not enter until the latter months and thus the duplicated effort for salmon is during a period of negligible abundance. If it were not for the sea trout availability then no effort may have been expended for salmon. This has the effect of reducing annual CPUE for salmon and therefore where mixed fisheries exist this needs to be considered when comparing performances with other rivers.

4.4.6 Spatial differences in CPUE

The difference in spatial CPUE within the catchment has important implications for constancy in annual surveys. Clearly for future surveys the same proportion of anglers from each section should be used since bias towards more anglers from one or other of the sections may bias the annual catchment CPUE. Spatial catch and effort information can provide patterns of usage throughout the catchment though factors such as flow may influence spatial availability, especially for salmon.

4.4.7 CPUE for sea trout in terms of weight

CPUE is measured to assess changes in stock abundance and therefore infer changes in spawning stock and deposition rates. Due to the different traits in age at return/maturation of sea trout, an index of abundance in terms of numbers may not reflect changes in the spawning stock. Since deposition rates rely upon the size of fish, CPUE in terms of weight assesses changes in spawning stock more effectively. This is particularly important when sex ratios are considered since whitling from the Tywi have a female:male ratio close to unity and non-whitling have a sex ratio of 2.4:1 in favour of females (Chapter 2).

4.4.8 Future surveys

Log book studies should continue on the Tywi since they provide more detailed data than catch statistics and longer term results may provide a

better understanding of annual variation. Log books are also the most reliable method currently available to assess any changes in returning adults due to the lining of Llyn Brianne, and any impact of changes in management policy, e.g Net Limitation Orders or changes in byelaws.

Some key points for future surveys include:

(1) Ensuring a similar balance of effort from each section of the river to enable comparisons with this study.

(2) Possibly modifying the log book to include a section where the angler can indicate which method of fishing was used, so that the efficiency of different gears may be assessed.

(3) Including an assessment of whether salmon caught at the beginning of the season are kelts or fresh run.

(4) Establishing an assessment of the total number of anglers fishing the river, so that any large changes in the pattern of usage may be determined.

(5) Implementing an assessment of weight in ranking annual results for sea trout CPUE, so that more applicable interpretation of spawning stock can be made.

(6) The dissemination of results back to anglers to stress the importance of log book studies.

(7) Running a parallel collection of scale material with log books since the vehicle for distribution is already available and this would provide valuable biological information.

Further gaps in the knowledge include the possible differences in catchability between different sea age groups of fish viz. whitling and older fish for sea trout and the grilse and MSW component of salmon stocks. This information will aid the interpretation of overall catches between years but a knowledge of the abundance of the stocks in needed, which may be available from studies undertaken with river counters or traps.

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CHAPTER 5

THE EFFECT OF LIMING ON WATER QUALITY

5.1 Introduction

The acidity of rain was first described over a hundred years ago in Manchester by Smith (1872). It is only in recent decades, however, that the effects of acid rain have been of increasing concern to scientists and politicians in Europe, Scandinavia and North America. The major acidifying pollutants in the atmosphere include sulphur dioxide (SO_2) and oxides of nitrogen (NO_x) from the burning of fossil fuels in power stations and industrial plants, with motor vehicles also contributing to the latter. The doubling of the consumption in fossil fuels from 1940 to 1970 in Europe (Gee, 1990) resulted in increases in the emissions of these pollutants during this period.

The term 'acid rain' is usually used to encompass all the pollutants from the air contributing to acid deposition, but only about one third of deposition is in direct precipitation, with the remainder as dry deposition (Mason, 1989). While rainwater is naturally acidic (pH 5.6) in the presence of atmospheric CO_2 (Howells, 1990), the presence of SO_2 and NO_x reduces pH further. In Wales, upland areas receive acidic deposition with a mean pH of 4.4-4.6 (Donald *et al.* 1990) but have episodes with pH <4.0 associated with easterly winds. When this deposition occurs on thin soils overlying slow-weathering rocks, offering little buffering capacity, acidic conditions can prevail in the watercourses. Land management such as conifer afforestation leads to increased 'scavenging' of dry deposition (Gee, 1990; Whitehead *et al.* 1990) and a resultant increase in loading of pollutants.

Management strategies therefore have to be found to ameliorate the effects of acidification. The most ideal strategy would be to cut the emissions of pollutants at source. To date, legislation to accomplish this has produced an apparent downward trend in emissions since 1980 (Donald *et al.*, 1990), but a reduction in deposition of some 50% upon current levels is required to reverse acidification in Wales (Whitehead *et al.*, 1990). The economics involved in reducing emissions at source and the large geographical area incorporated requires this to be a long term strategy. An alternative strategy to reduce the loading of pollutants on vulnerable soils would be the de-afforestation of conifer plantations. Hornung *et al.* (1990) describe this as a long term strategy since decreases in acidity can take two years or more to materialise, and could actually increase in the short-term after felling.

Many of the upland areas of Wales have already been affected by acidification and therefore lost much of the buffering capacity of their soils (Edwards *et al.*, 1990). It is in these most acid sensitive sites that liming is required to prevent further detrimental effects from acid deposition to aquatic biota, as discussed in Chapters 6 and 7.

The headwaters of the River Tywi is one such area of low buffering capacity where damage to aquatic biota has been established (e.g. Stoner *et al.*, 1884; WWA, 1987). In 1990, Llyn Brianne reservoir was at its most acidic since 1982 (NRA, 1990) and so the NRA decided to lime the reservoir in February 1991. The objective of this operation was to rehabilitate the aquatic biota below the impoundment, specifically the juvenile salmonid stocks.

The aim of this chapter is :
(i) to describe the water quality from the reservoir prior to liming, and any longitudinal changes in the main river due to the input of tributaries.
(ii) to describe the changes in water quality below the impoundment brought about by liming the reservoir.

This study looks solely at the changes downstream of the reservoir. Investigations within the impoundment itself were, however, undertaken by the NRA and are reported elsewhere (Rogers & Evans, 1991; A.Rogers pers. comm.). Changes in the biota downstream of the impoundment are discussed in Chapters 6 & 7.

5.1.1 The liming operation

The liming method used was that of dosing the reservoir with lime slurry from a purpose-built boat. The lime dose was calculated from a Swedish liming model (Warfvinge *et al.* 1984) and 850 tonnes of lime were applied over a 3-week period, beginning on the 27th February 1991. The limestone used was a by-product of roadstone quarrying and brought to the reservoir by road transport. The boat held 5 tonnes of lime and was pressure loaded from the bulk lorries. During application the liming boat drew in water from the reservoir via a pipe and propelled this water out in front of the bow of the boat, with the Venturi effect drawing lime from the hold to form the slurry. The reservoir was treated in three sectors to ensure

even distribution of lime. The first reliming took place in October 1991 (beginning 7th) and consisted of 500 tonnes of lime. This was to complete the dose originally calculated from the model (1350 tonnes). Further re-applications of 500 tonnes of lime were carried out in April 1992 (27th), November 1992 (2nd) and March 1993 (24th). The autumn applications of lime were undertaken to combat the reacidification anticipated from winter dilution.

5.2 Materials and methods

5.2.1 Sampling strategy

The examination of the effects of liming the reservoir on the water quality of the River Tywi below the impoundment was carried out in two phases, investigating the extent and geographical limits of acidification in the pre-liming phase, and then examining changes in the post-liming period. Two sampling methods were used:

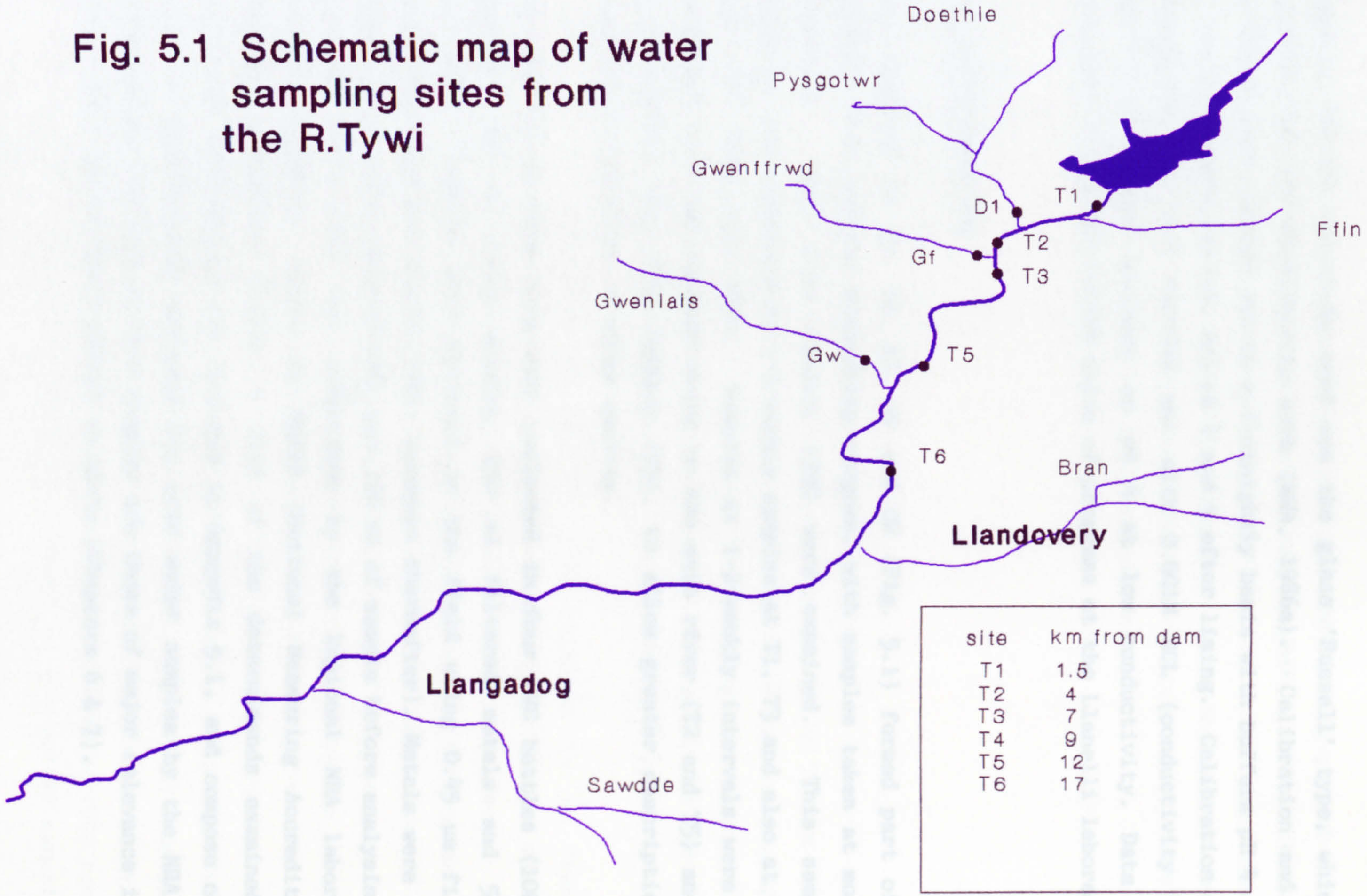
(i) Continuous monitoring (15-30 min. intervals) of pH, conductivity and temperature from the reservoir outlet, and further downstream at the point where acidity was suspected to have minimal impact. This latter position was determined from personal communications with NRA personnel. Continuous monitoring was employed to fully describe the time-scale over which changes occurred and to 'capture' acidic episodes.

(ii) Spot sampling at 1-2 weekly intervals for a wider range of determinands than continuous monitoring, and at a greater number of sites. A sampling frequency of 2-week intervals was shown by Weatherley and Ormerod (1991) to be acceptable in describing mean conditions for pH. Spot sampling enabled the description of key determinands including metal concentrations.

5.2.2 Continuous monitoring

Monitoring was carried out at site T1 and T6 (Fig. 5.1) for the period November 1990 to December 1992. Short-term monitoring of the River Doethie (D1) was carried out between February and May 1991, to investigate the episodic nature of acidic events in this tributary. The apparatus used were pHOX 100 DPM units measuring pH (temperature corrected), temperature and conductivity with data recorded on Tinylog

Fig. 5.1 Schematic map of water sampling sites from the R.Tywi



site	km from dam
T1	1.5
T2	4
T3	7
T4	9
T5	12
T6	17

loggers. The pH electrode used was the glass 'Russell' type, which is reliable for low conductivity work (WWA, 1986a). Calibration and data retrieval were carried out on a fortnightly basis with buffers pH 4 and 7 in the pre-liming period, and pH 4 and 9 after liming. Calibration of pH electrodes was also carried out with 0.001M HCL (conductivity 56.16 $Uscm^{-1}$) to ensure accuracy at pH 4 at low conductivity. Data were processed on the NRA LOGSYS suite of programs at the Llanelli laboratory.

5.2.3 Spot sampling

Spot sampling at T1, T3, D1, GF and GW (Fig. 5.1) formed part of the NRA/NWC river quality monitoring program, with samples taken at monthly intervals. Data post January 1990 were examined. This sampling frequency was increased to 1-2-weekly samples at T1, T3 and also at a new site (T6) from July 1990. Samples at 1-2-weekly intervals were also collected from two further sites on the main river (T2 and T5) and the River Doethie (D1) from January 1991, to allow greater description of longitudinal profiles in water quality.

The samples at each site were collected in four PVC bottles (1000 ml general, 250 ml total metals, 250 ml filtered metals and 50 ml nutrients). Samples were filtered in the field using 0.45 µm filters (micropore-type pre August 1991, acrocaps thereafter). Metals were fixed with 1 ml of molar AristaR HNO_3 per 100 ml of sample before analysis. The analysis (with AQC) was undertaken by the Regional NRA laboratory (South), Llanelli, which is NAMAS (National Measuring Accreditation Service) accredited (1137). A list of the determinands examined and analytical methodology are included in Appendix 5.1, and compose of the suite of determinands analysed for acid water samples by the NRA. The determinands reported in this chapter are those of major relevance in the River Tywi, due to their effect on biota (Chapters 6 & 7).

5.2.4 Data analysis

For this study the pre-liming period includes all sampling prior to 27th February 1991. Sampling after 1st March 1991 is included in the post-liming period analysis.

Statistical comparison between the pre- and post-liming period for each determinand was carried out using either the parametric Student t-test or the non-parametric Mann-Whitney test. The Student t-test was used with determinands displaying normal distributions (Wilk-Shapiro or Pearson chi-square test of normality) and the Mann-Whitney test used with determinands displaying non-normal distributions.

5.3 Results

5.3.1 Continuous monitoring

Mean daily pH from November 1990 to December 1992, at sites T1 and T6, are shown in Figs. 5.2a-b. Prior to liming the mean pH at T1 was 5.45 for the period November 1990 to 27th February 1991, with values ranging from 4.95 to 6.37. Mean daily pH was higher at T6 during this period (6.47), with a range of 5.84 to 6.86. This downstream increase in pH was chiefly due to the input of unacidified tributaries (section 5.3.2).

Liming of the reservoir in February 1991 resulted in an immediate increase in pH at T1 to circumneutral levels (Fig. 5.2a). This was maintained throughout the summer and mean daily pH showed little variation (pH 6.7-7.3). This is expected since the water released from the reservoir came from the hypolimnion, which is a large mass of mixed water buffering temporal variability. The first re-liming was in October 1991 and although there was a slight decrease in pH during the subsequent winter, conditions were maintained above pH 6.6 through to the second re-liming in April 1992 (Fig. 5.2a). This lime application in spring 1992 once again increased pH to circumneutral levels. The high rainfall experienced during the summer and autumn of 1992 resulted in overspill of the reservoir in September 92, which corresponds to the decrease to pH 6.6 at that time (Fig 5.2a). A third re-liming in October 1992 resulted in the pH being maintaining above 6.8 through to the end of sampling in December 1992.

The initial liming in February 1991 increased pH at site T6 by c.0.5 pH units, to circumneutral levels (Fig. 5.2b). The pH levels during the summer of 1991 were similar to that observed at T1, but a higher variation in mean daily pH was observed at T6. This increased variation was attributed to episodic depressions in pH during rainfall events, from

Fig. 5.2a Mean daily average pH and flow
at T1 from continuous monitoring

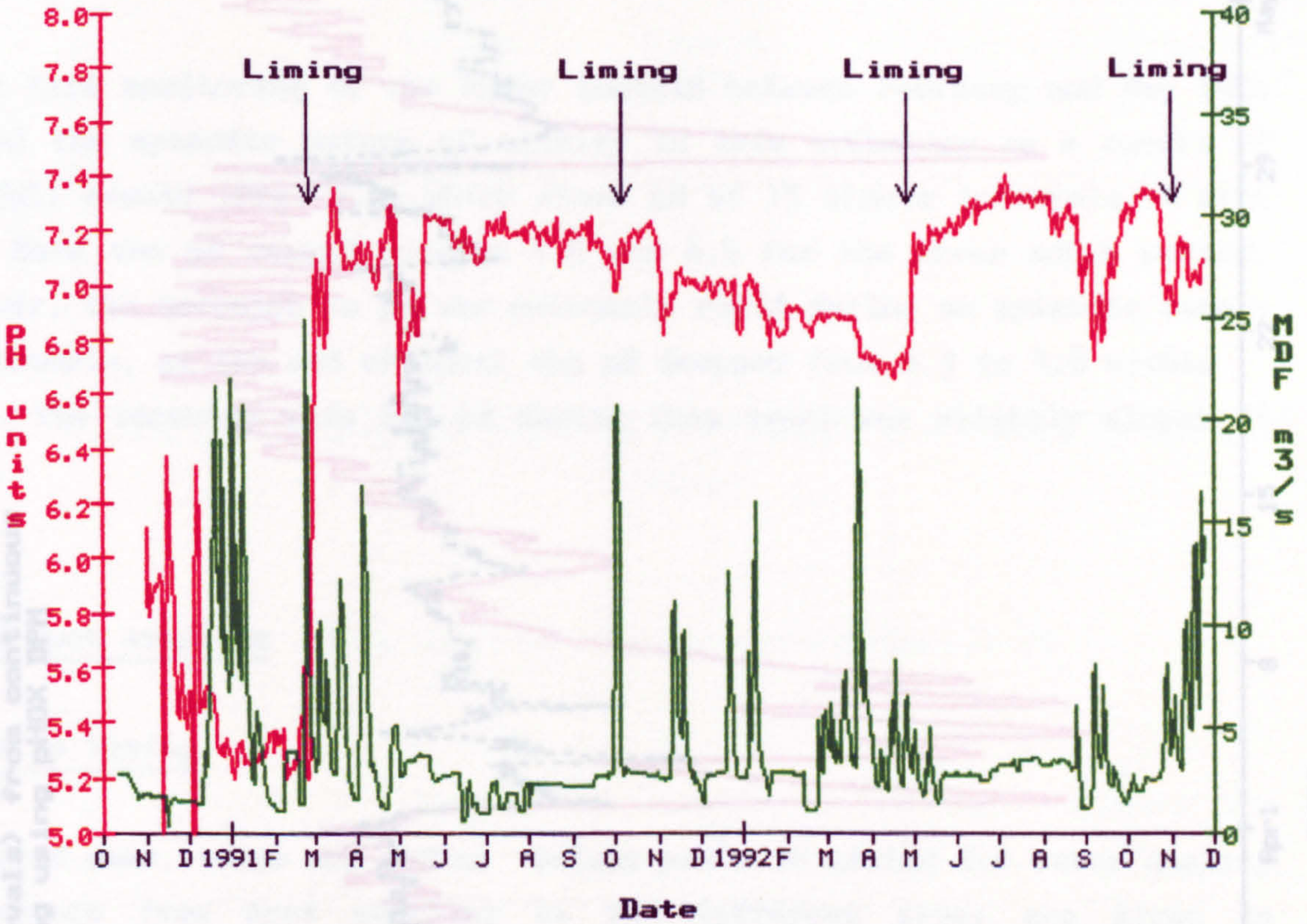


Fig. 5.2b Mean daily average pH and conductivity
at T6 from continuous monitoring

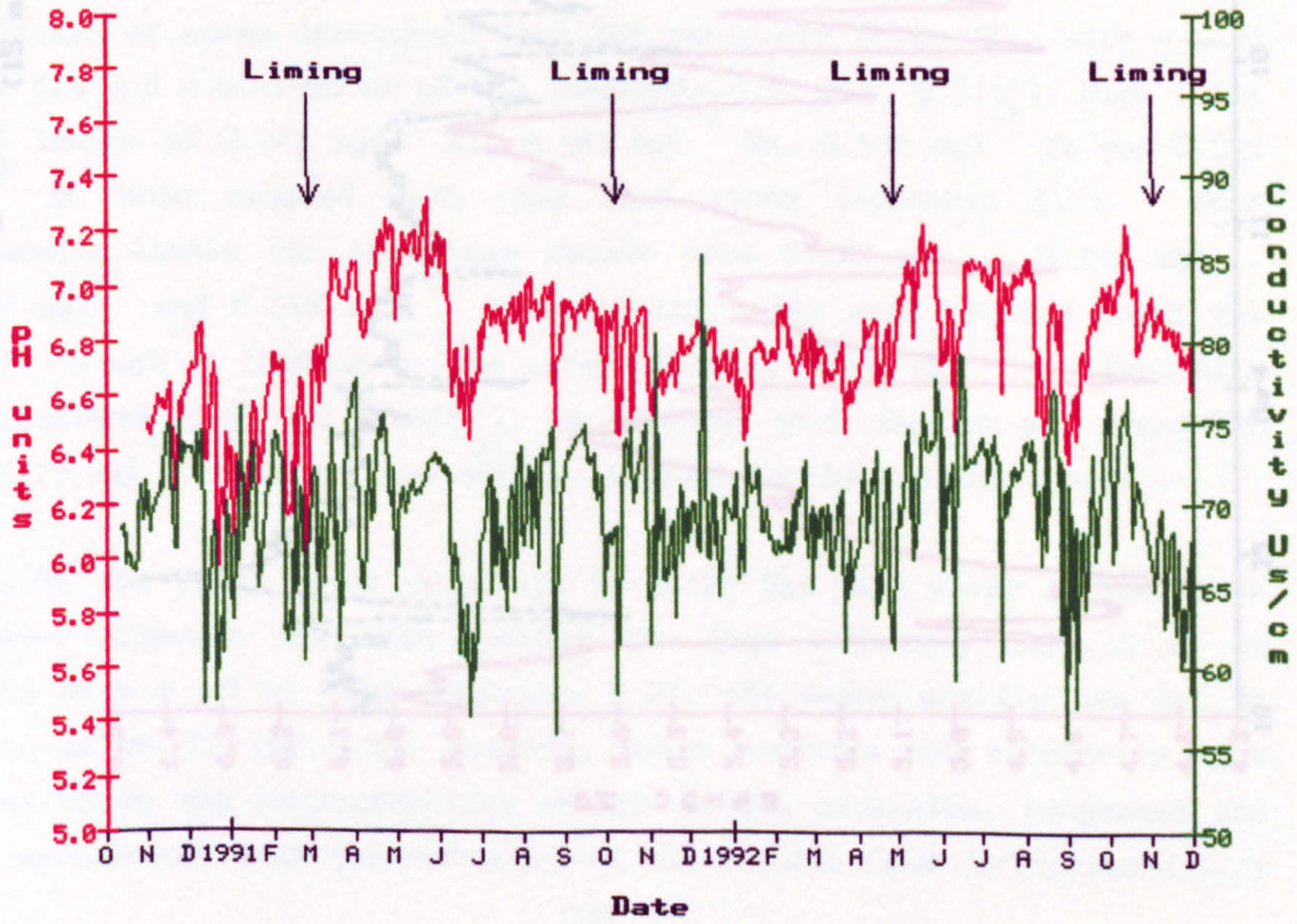
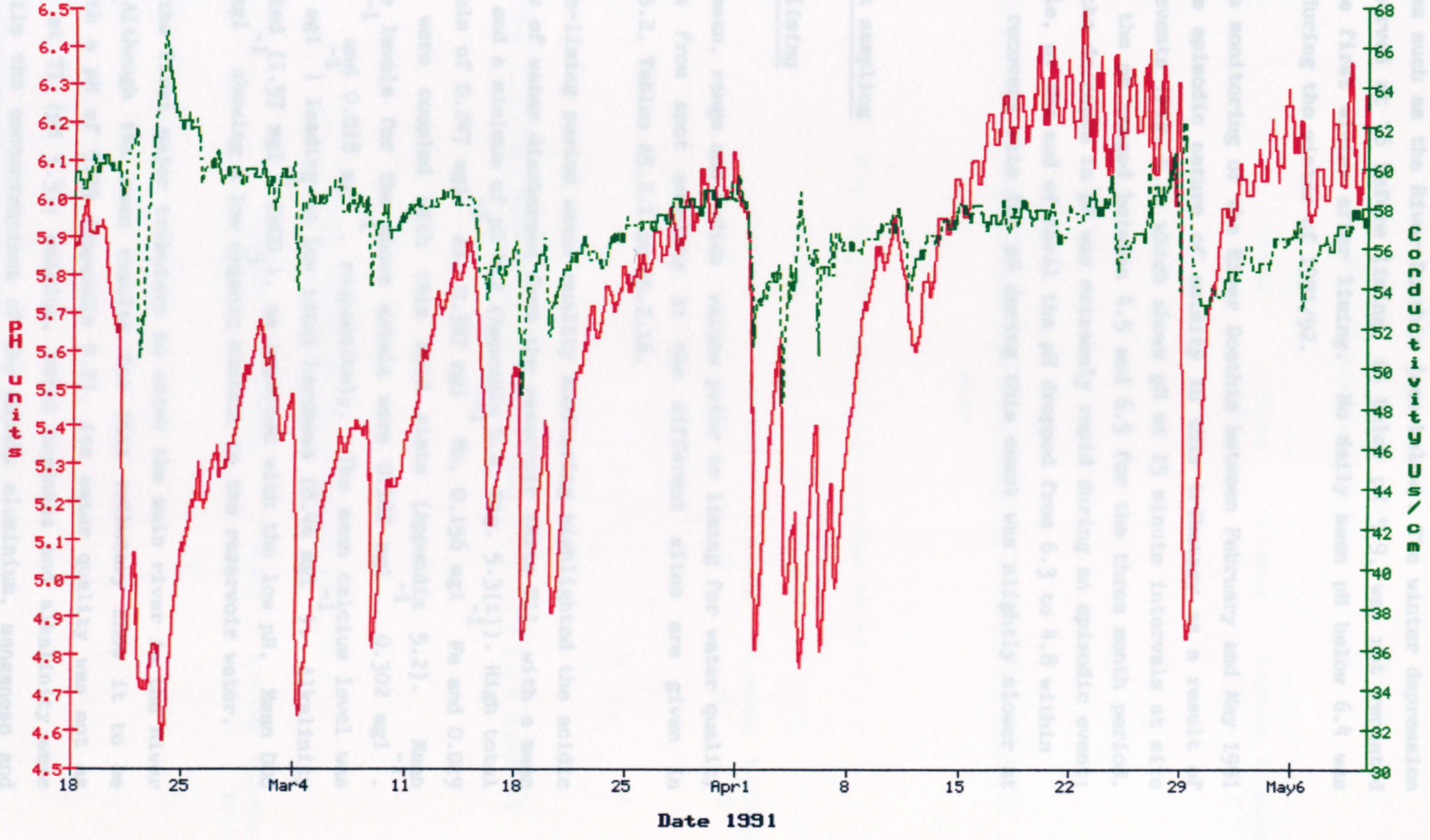


Fig. 5.2c pH and conductivity from the River Doethie (15 min. intervals) from continuous monitoring using pHOX DPM



tributaries such as the River Doethie (see below). The winter depression in pH observed at T6 before liming, to below pH 5.9, was not repeated during the first winter after liming. No daily mean pH below 6.4 was recorded during the winter of 1991/92.

Short term monitoring of the River Doethie between February and May 1991 showed the episodic nature of acidity in this tributary as a result of rainfall events (Fig. 5.2c which shows pH at 15 minute intervals at site D1). Here the pH ranged between 4.5 and 6.5 for the three month period. However, the decrease in pH was extremely rapid during an episodic event; for example, at the end of April the pH dropped from 6.3 to 4.8 within 14 h. The recovery rate for pH during this event was slightly slower at 75 h.

5.3.2 Spot sampling

Prior to liming

Detailed mean, range and median values prior to liming for water quality parameters from spot sampling at the different sites are given in Appendix 5.2, Tables A5.2.1 to A5.2.16.

In the pre-liming period water quality monitoring highlighted the acidic conditions of water discharged from the reservoir (Site T1), with a mean pH of 5.3 and a minimum of pH 4.2 (Appendix 5.2, Fig. 5.3(i)). High total metal levels of 0.247 mgl^{-1} Al, 0.347 mgl^{-1} Mn, 0.156 mgl^{-1} Fe and 0.019 mgl^{-1} Zn were coupled with this acid state (Appendix 5.2). Mean filterable levels for the above metals were 0.182 mgl^{-1}, 0.302 mgl^{-1}, 0.075 mgl^{-1} and 0.018 mgl^{-1}, respectively. The mean calcium level was low (1.73 mgl^{-1}) leading to low total hardness (8.46 mgl^{-1}). Alkalinity was depleted (1.57 mgl^{-1} $CaCO_3$), as expected with the low pH. Mean DOC was 1.79 mgl^{-1} showing a low organic content to the reservoir water.

Below T1 the first major tributary to enter the main river is the River Doethie. Although the mean results for this tributary show it to be acidic with a pH of 5.79 (Appendix 5.2), its water quality was not as acidic as at T1 (pH 5.3); calcium, total hardness and alkalinity were higher while the concentrations of the metals aluminium, manganese and zinc were lower. This was reflected in the results from T2 (Appendix 5.2)

Fig. 5.3(i) pH at T1

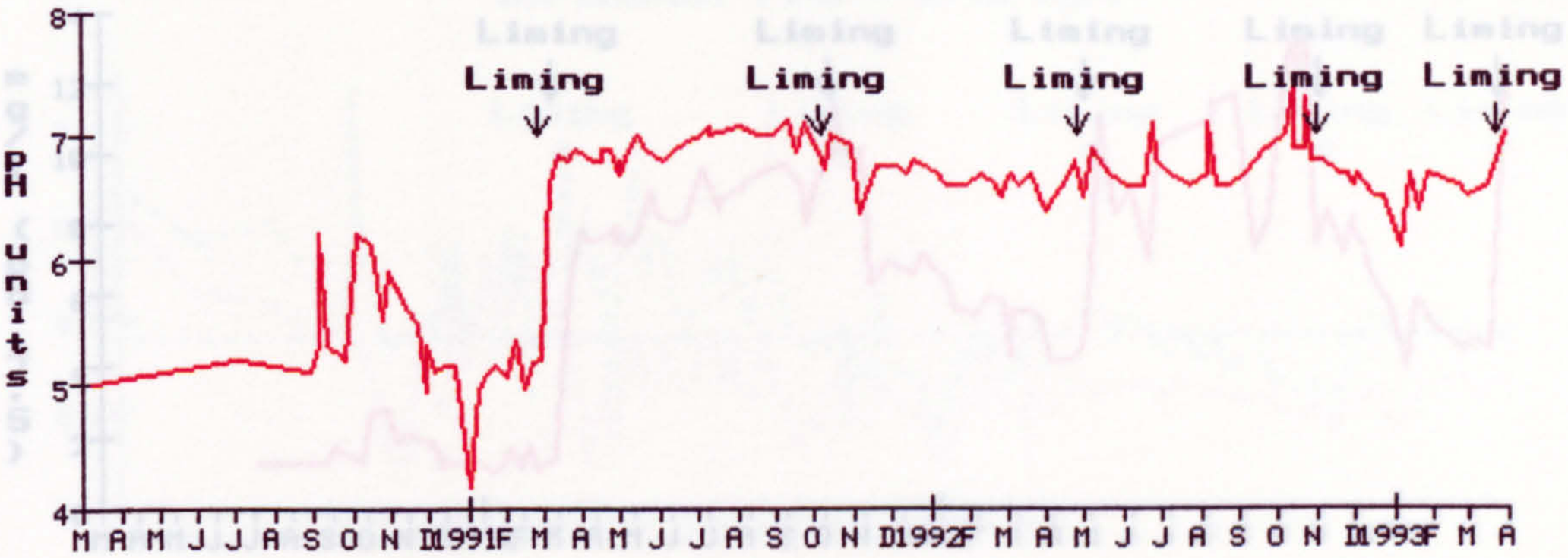


Fig. 5.3(ii) Dissolved calcium at T1

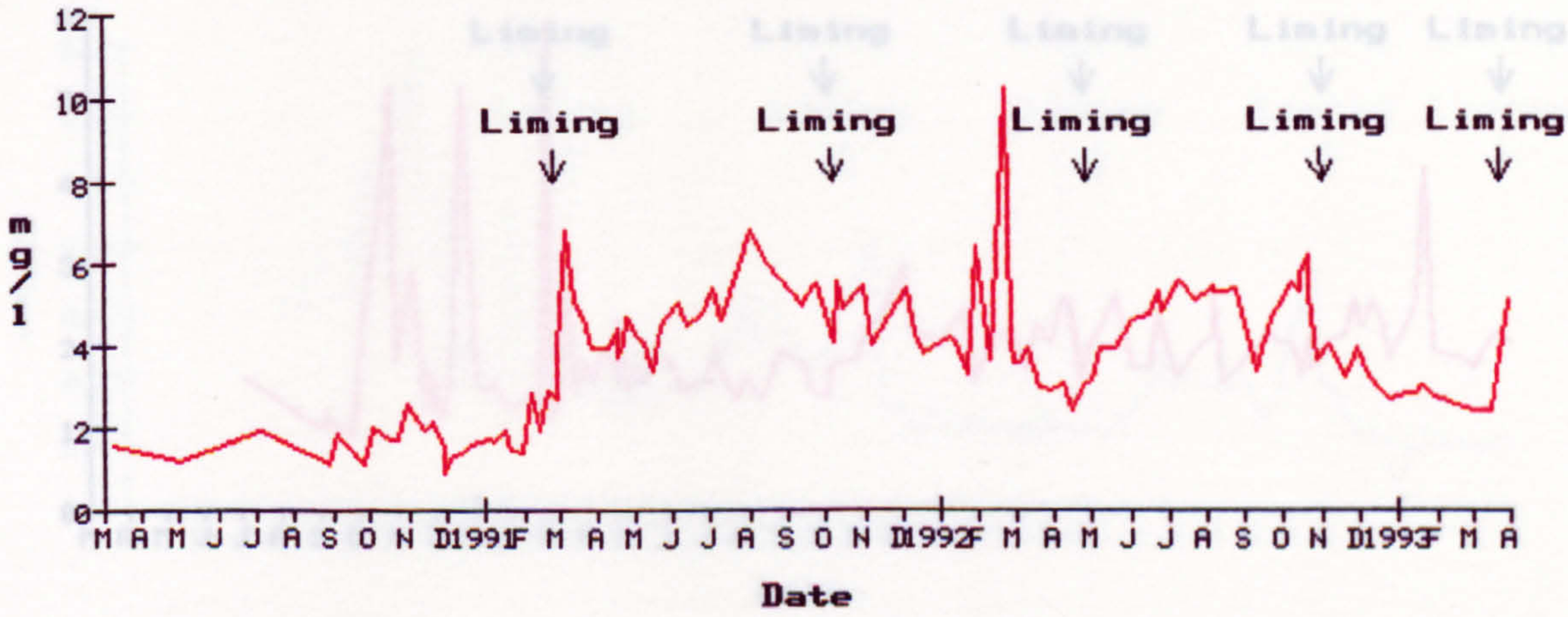


Fig. 5.3(iii) Hardness at T1

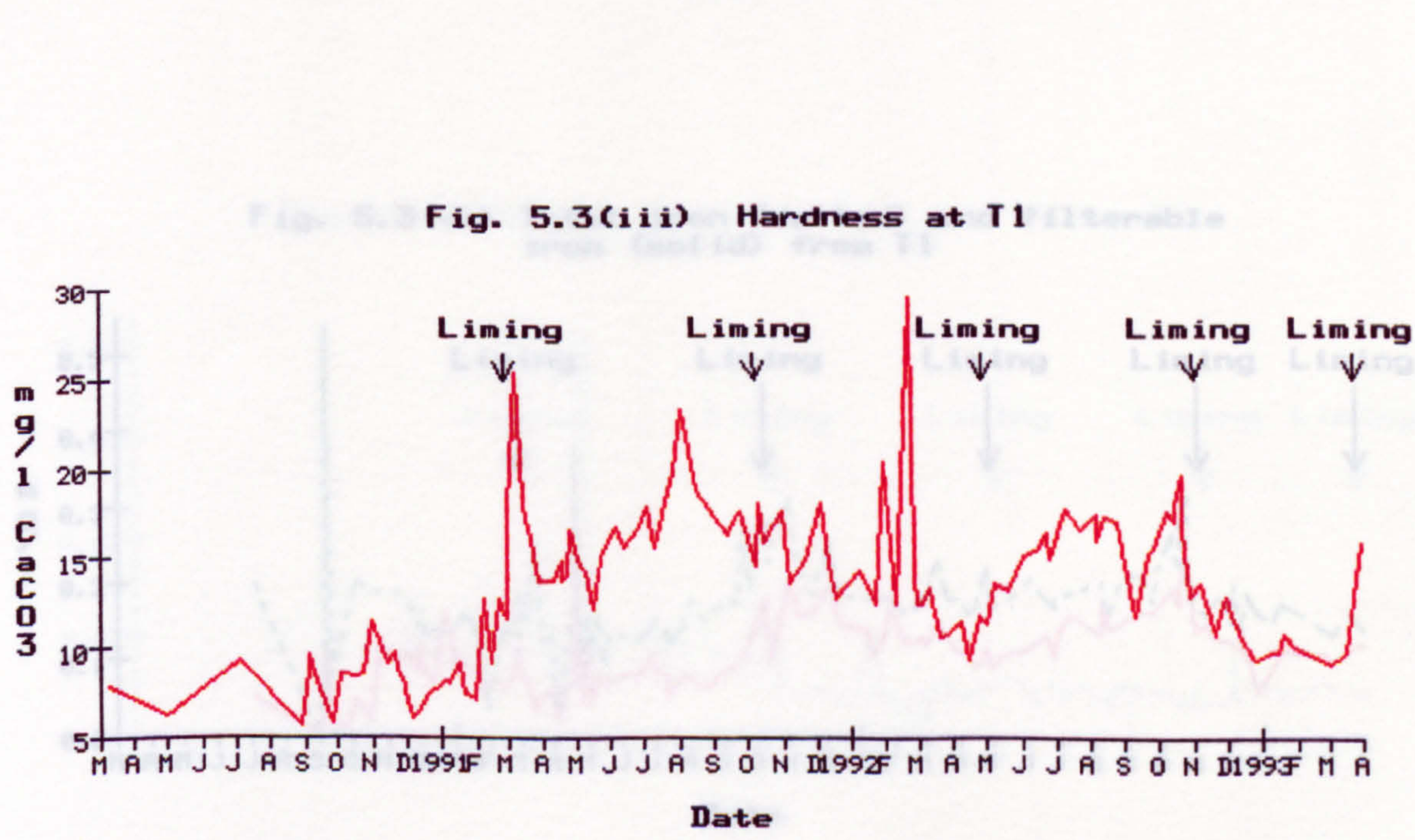


Fig. 5.3(iv) Alkalinity at T1

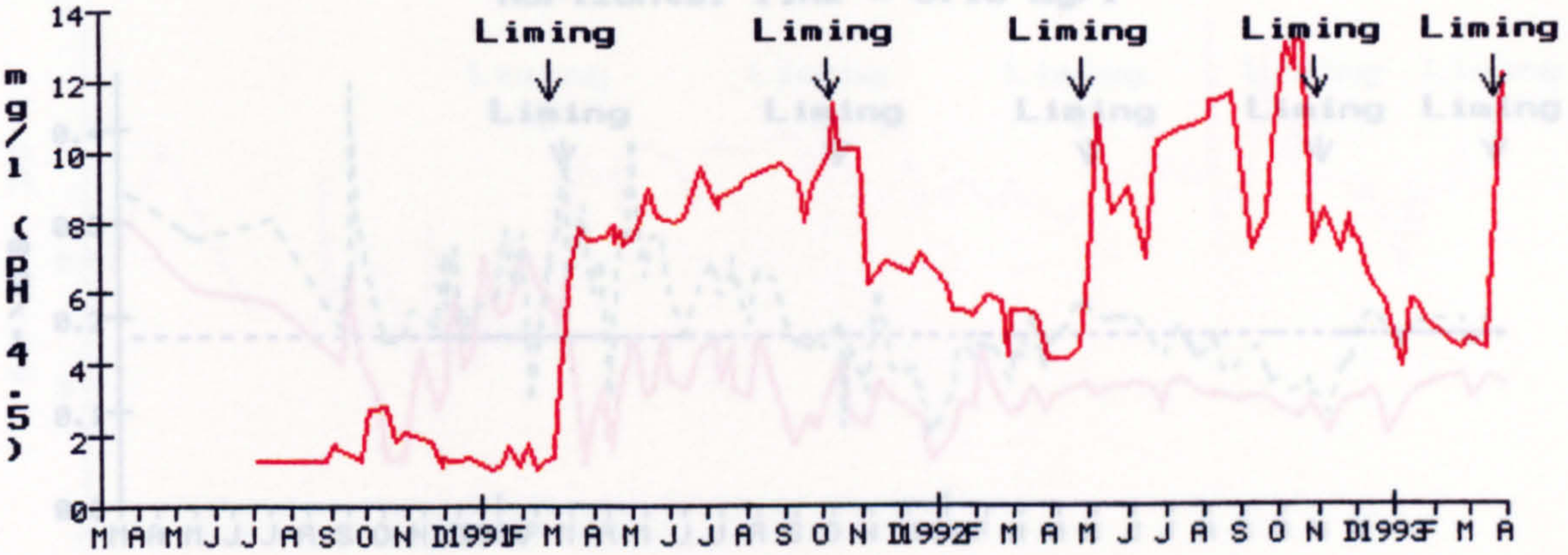


Fig. 5.3(v) Dissolved Organic Carbon at T1

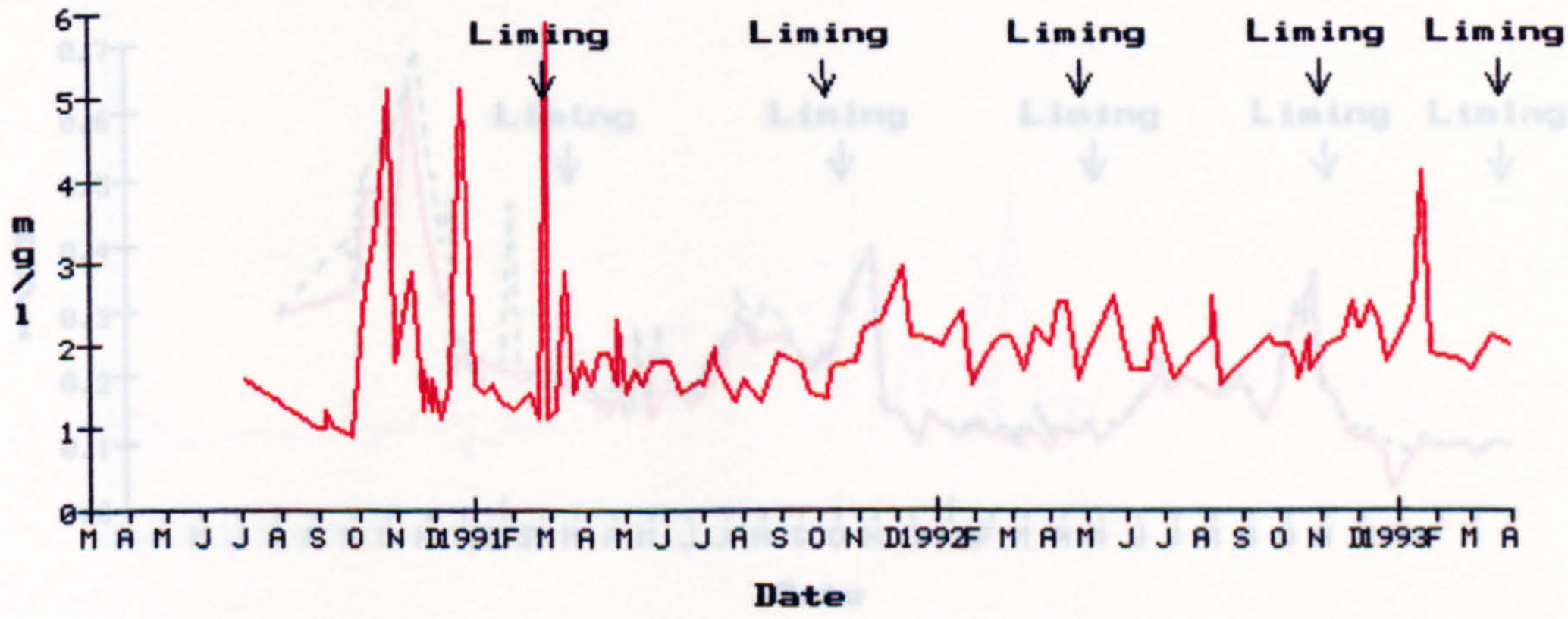


Fig. 5.3(vi) Total iron (dotted) and filterable iron (solid) from T1

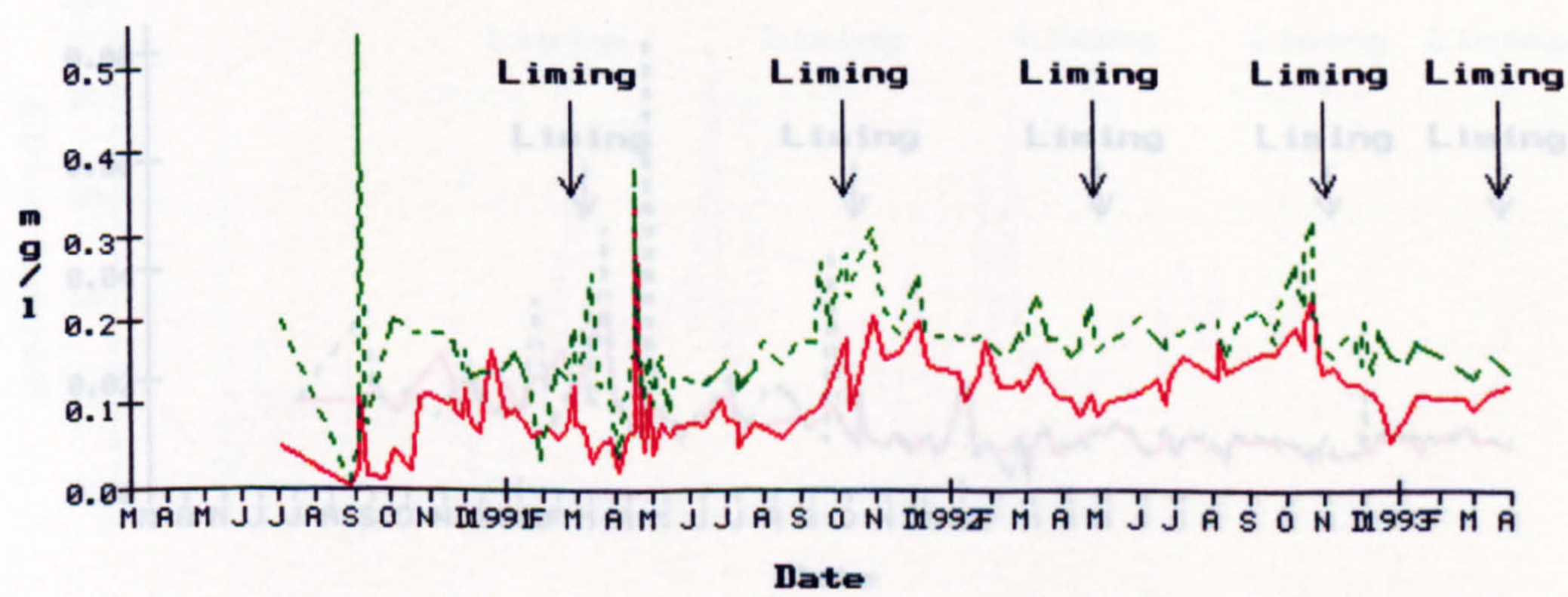


Fig. 5.3(vii) Total aluminium (dot) and filterable aluminium (solid) from T1
Horizontal line = 0.18 mg/l

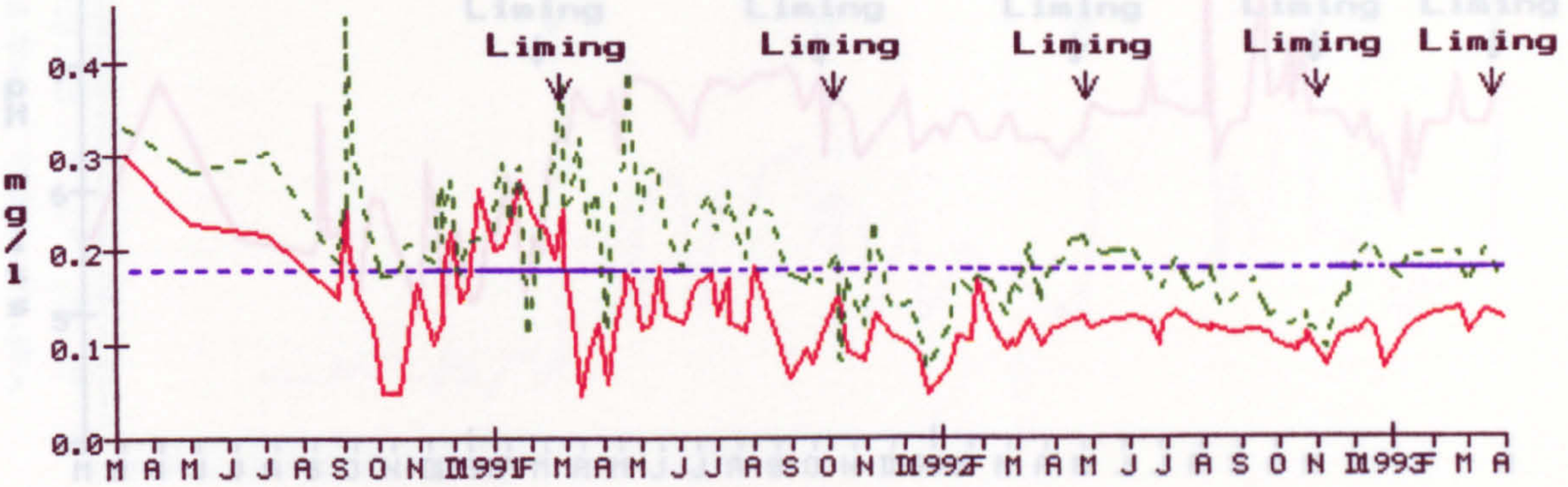


Fig. 5.3(viii) Total manganese (dashed) and filterable manganese (solid) from T1

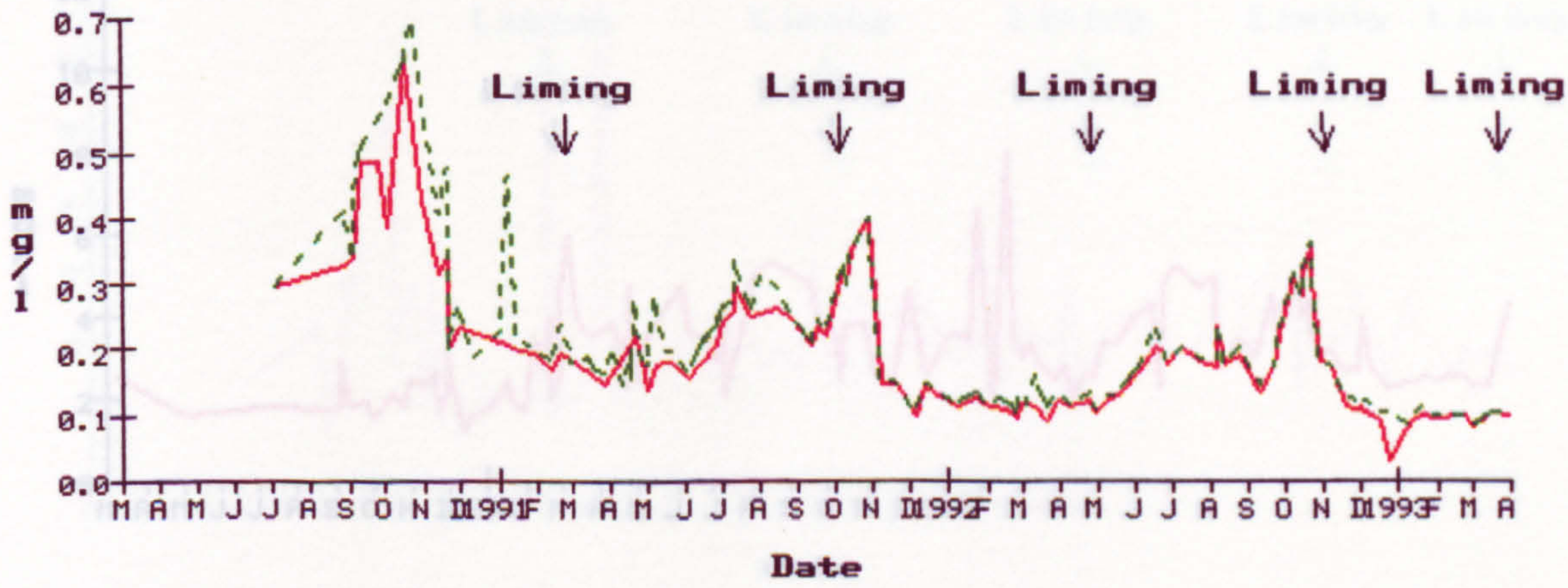


Fig. 5.3(ix) Total zinc (dot) and filterable zinc (solid) from T1

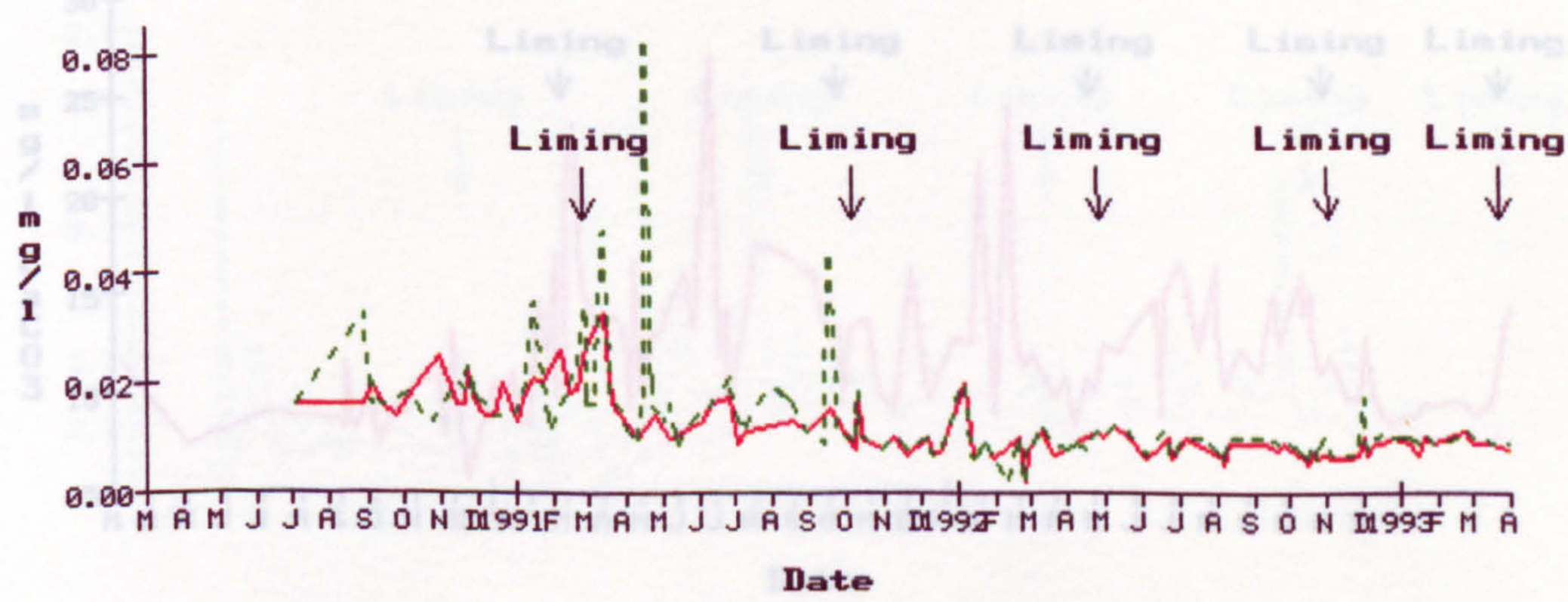


Fig. 5.3(x) pH from T3

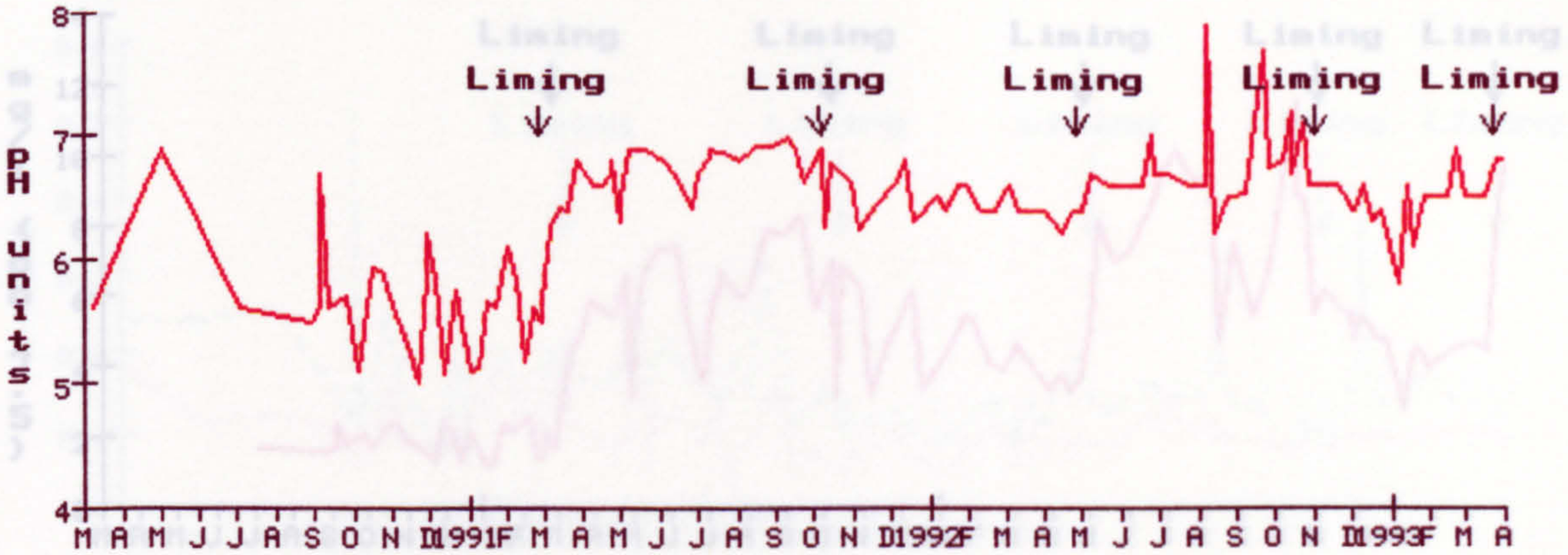


Fig. 5.3(xi) Dissolved calcium at T3

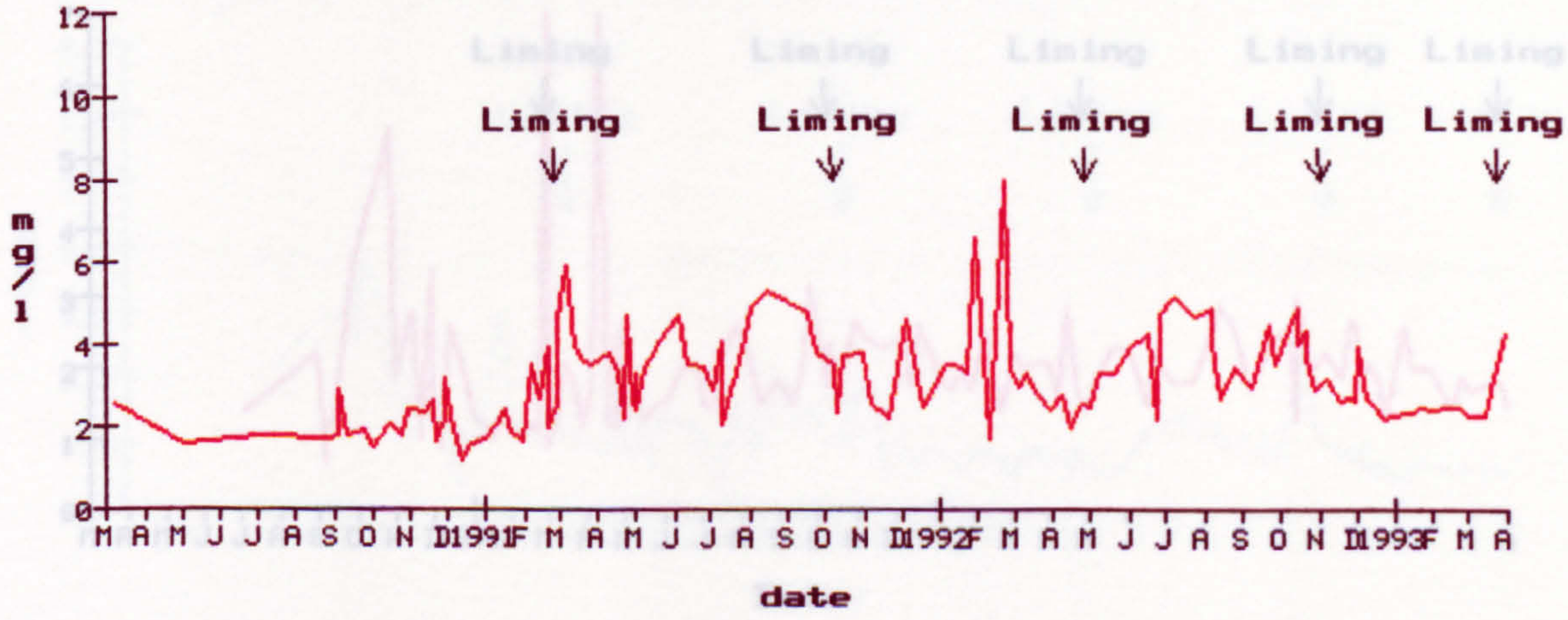


Fig. 5.3(xii) Hardness at T3

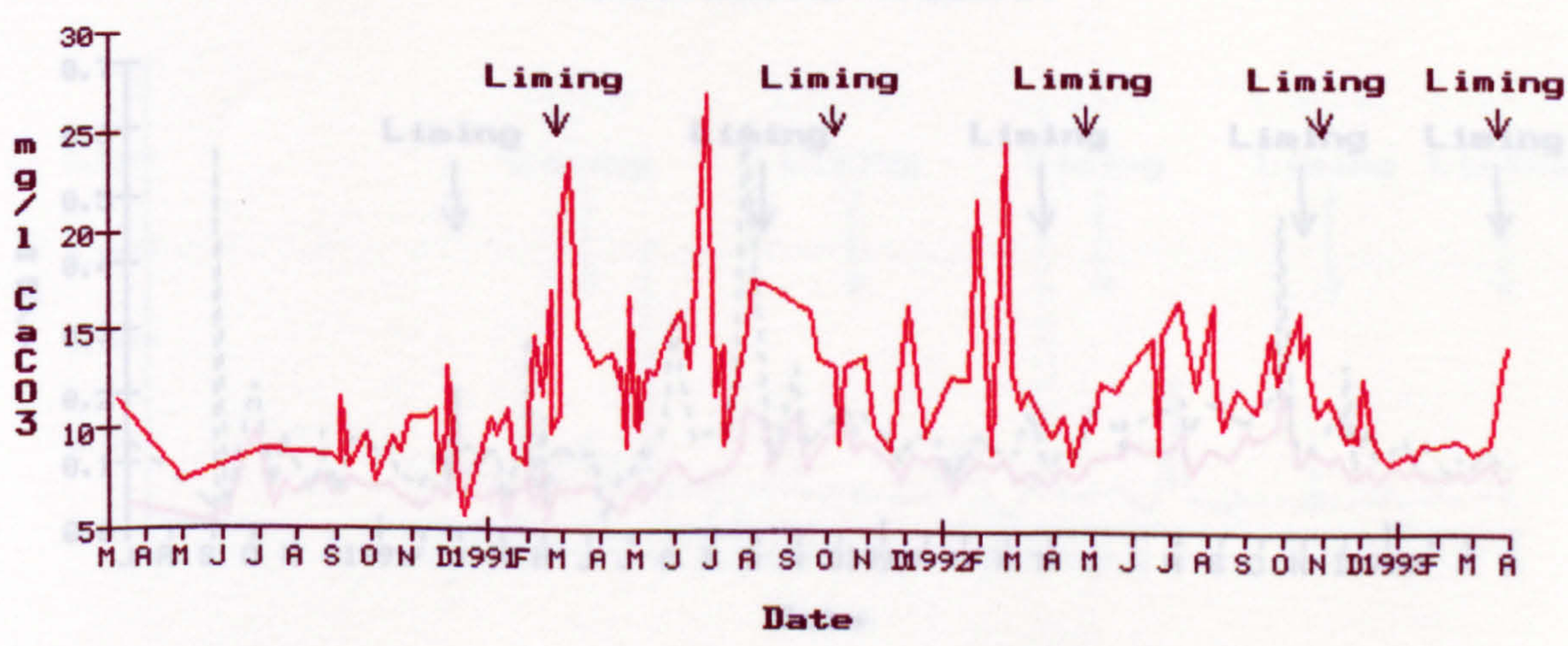


Fig. 5.3(xiii) Alkalinity at T3

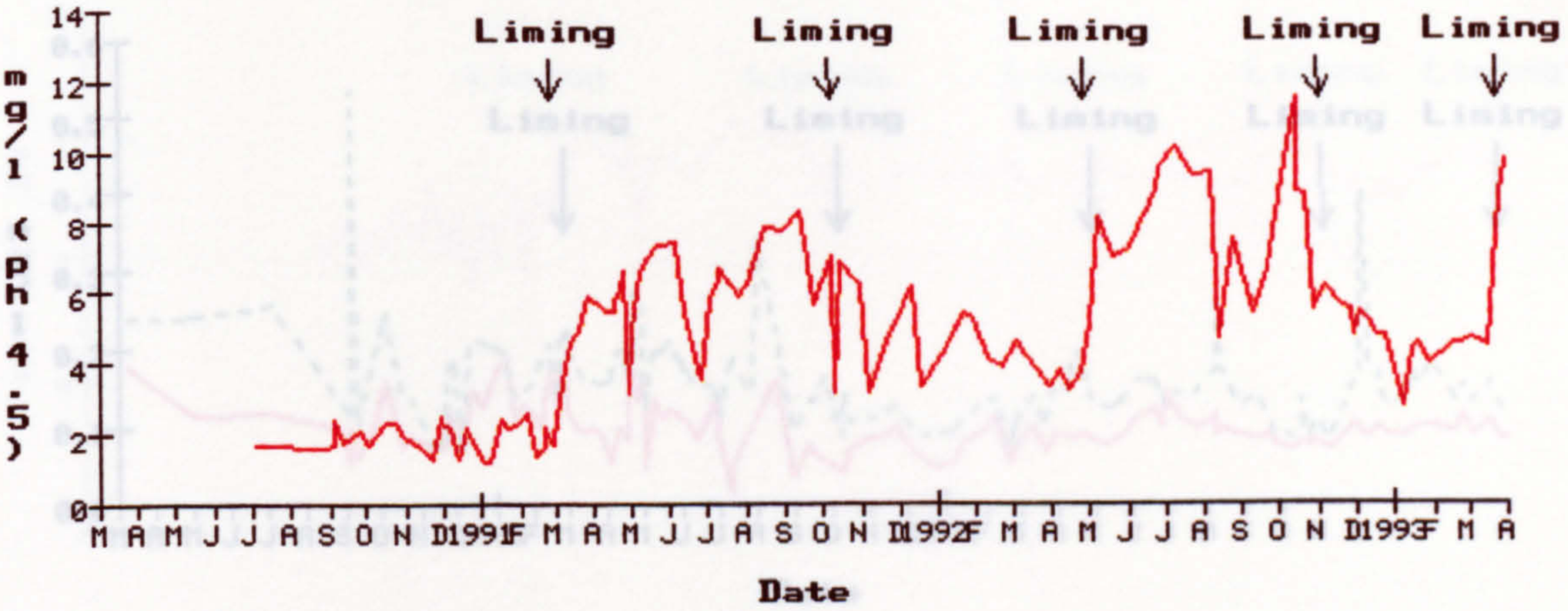


Fig. 5.3(xiv) Dissolved Organic Carbon at T3

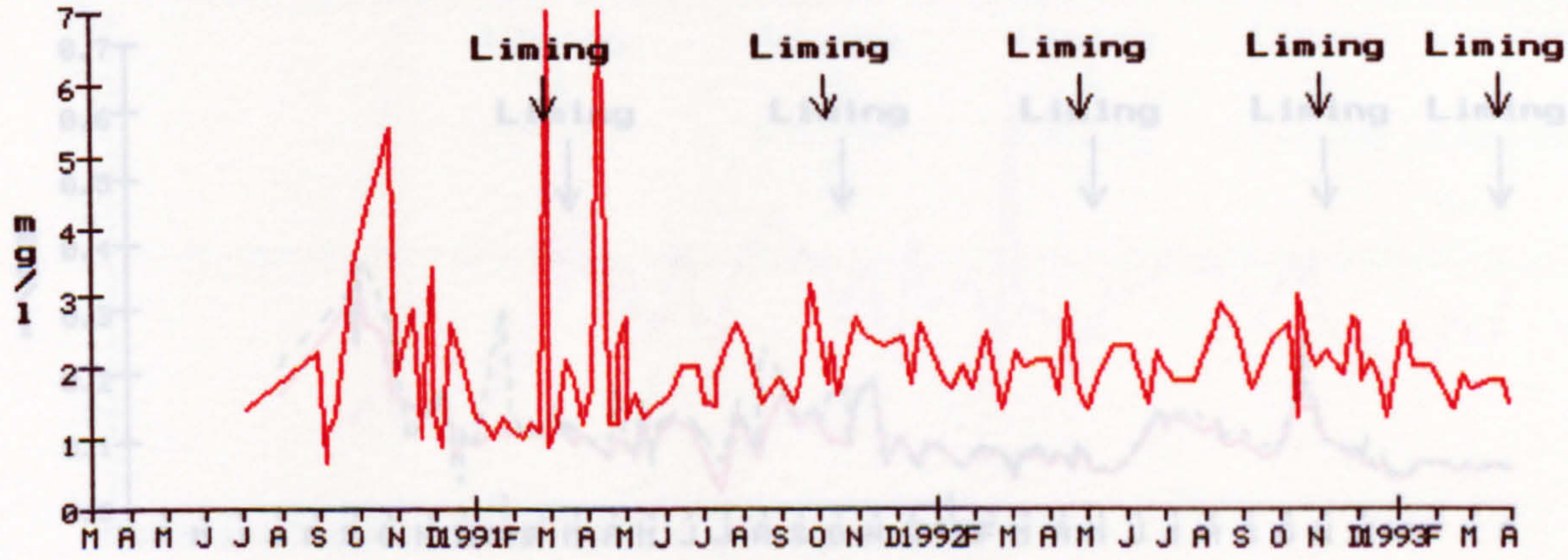


Fig. 5.3(xv) Total iron (dot) and filterable iron (solid) from T3

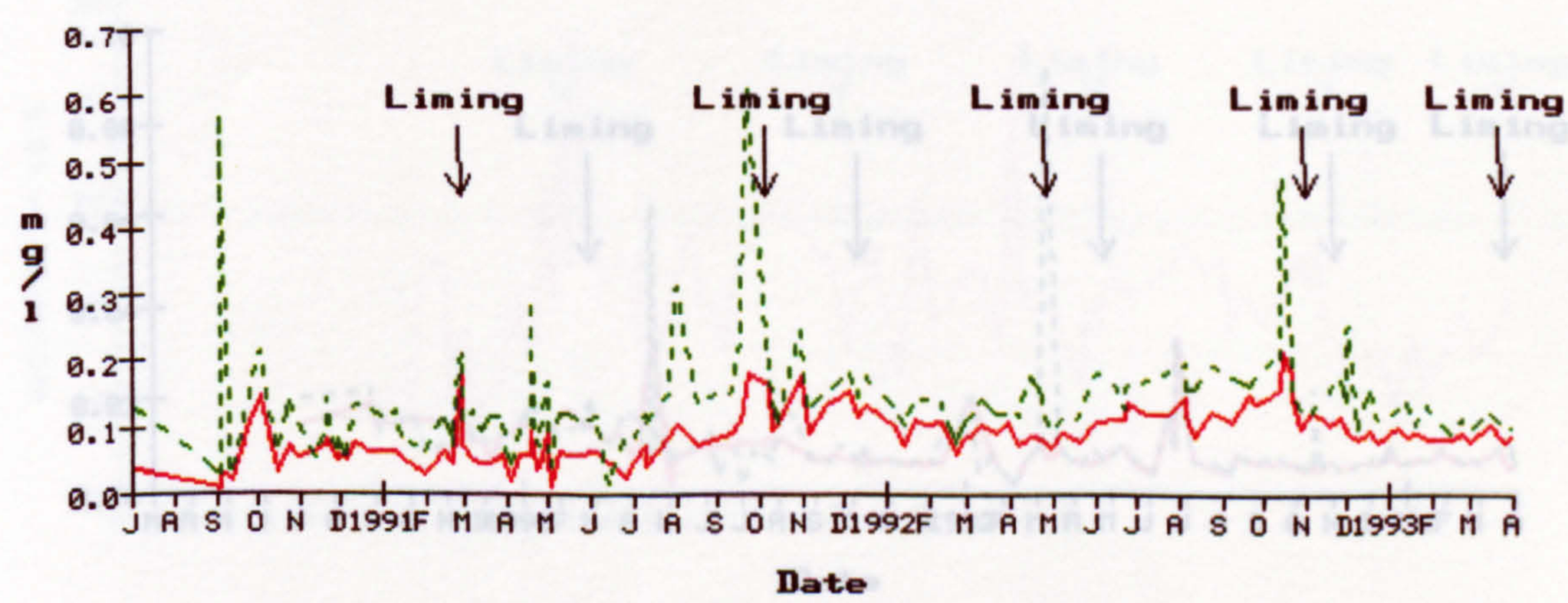


Fig. 5.3(xvi) Total aluminium (dot) and filterable aluminium (solid) from T3

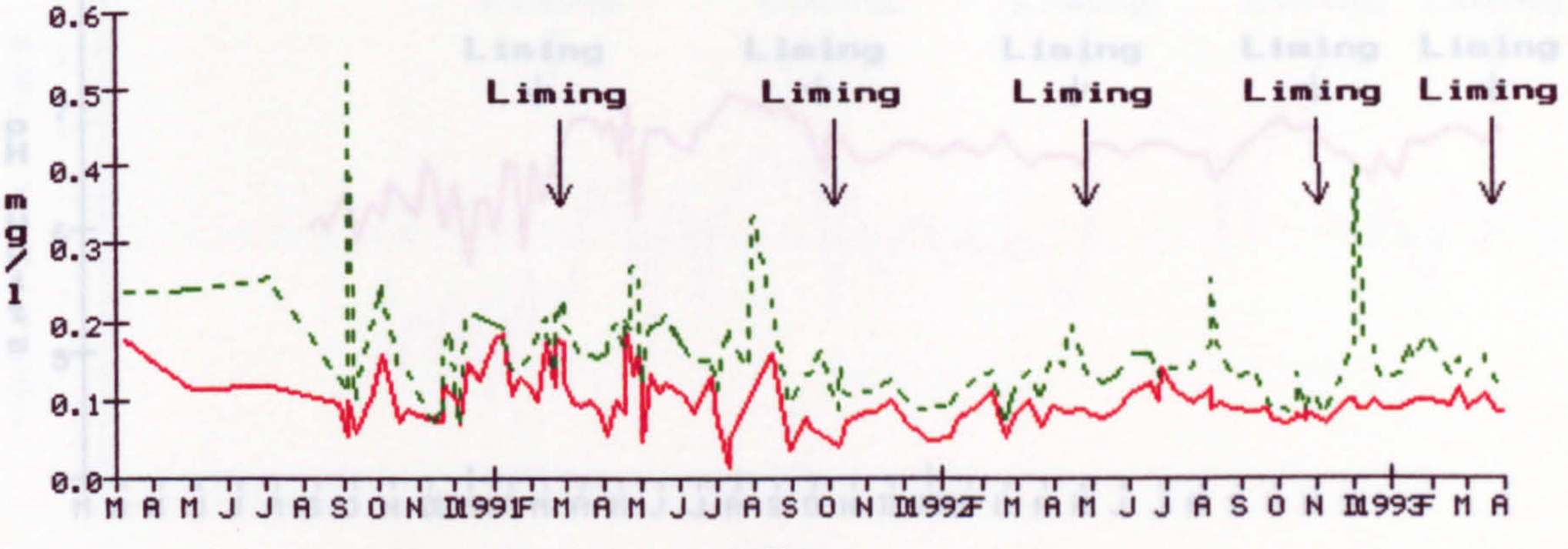


Fig. 5.3(xvii) Total manganese (dot) and filterable manganese (solid) from T3

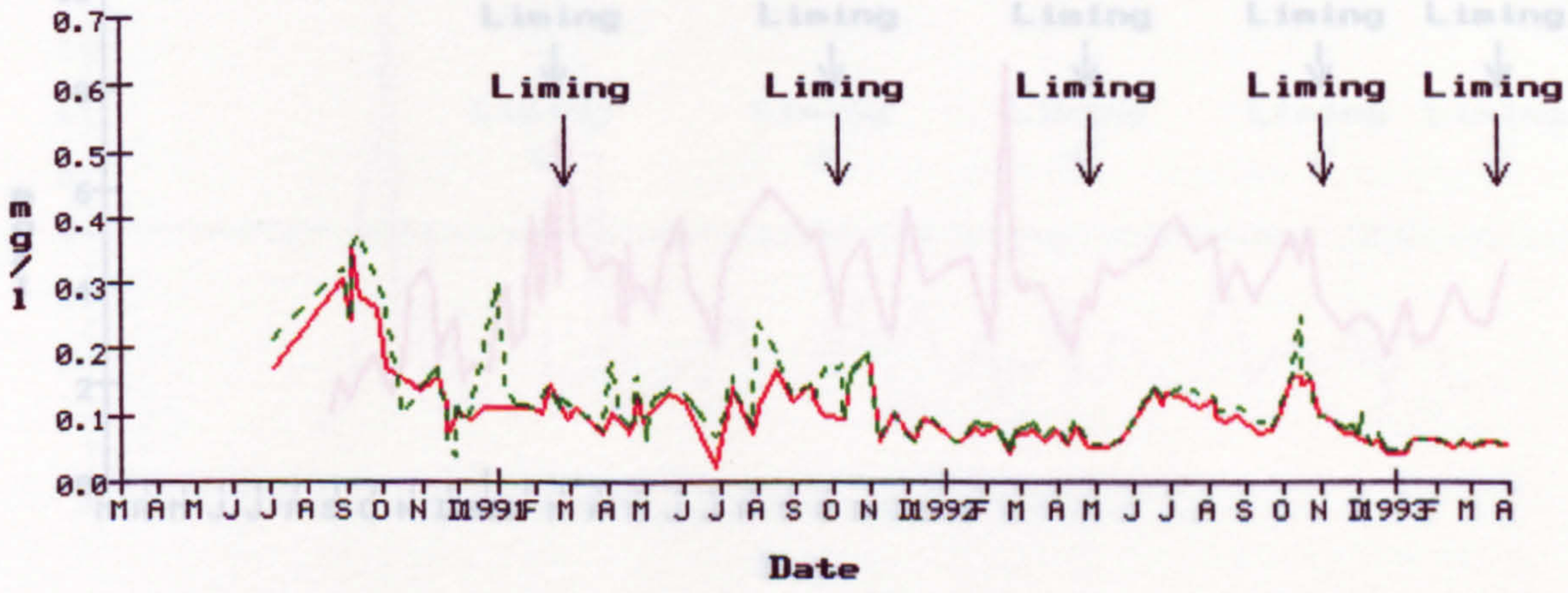


Fig. 5.3(xviii) Total zinc (dot) and filterable zinc (solid) from T3

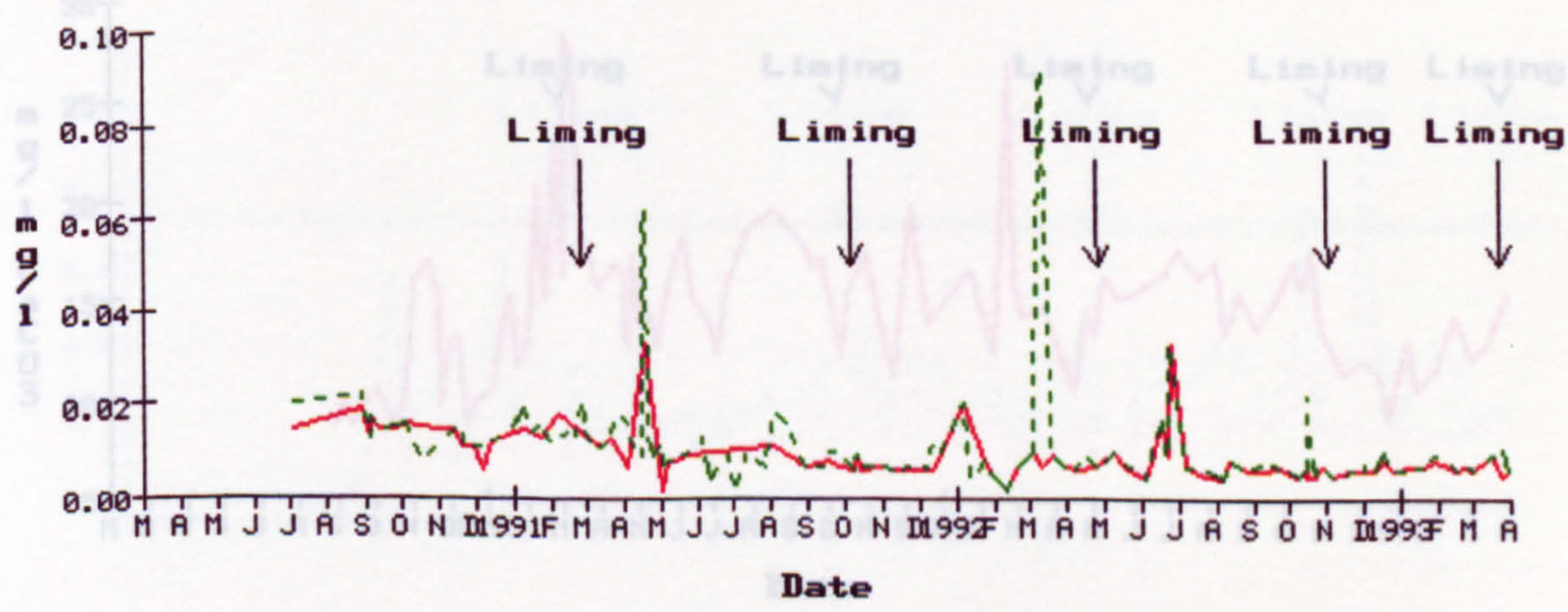


Fig. 5.3(xix) pH from T6

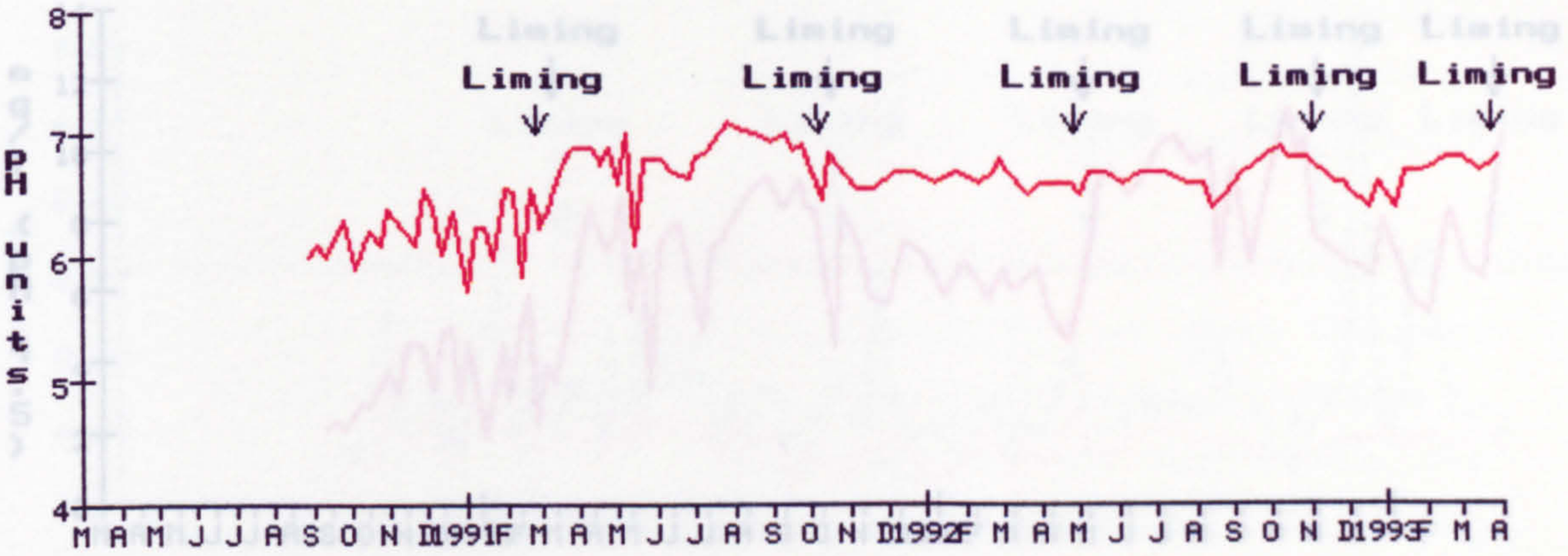


Fig. 5.3(xx) Dissolved calcium at T6

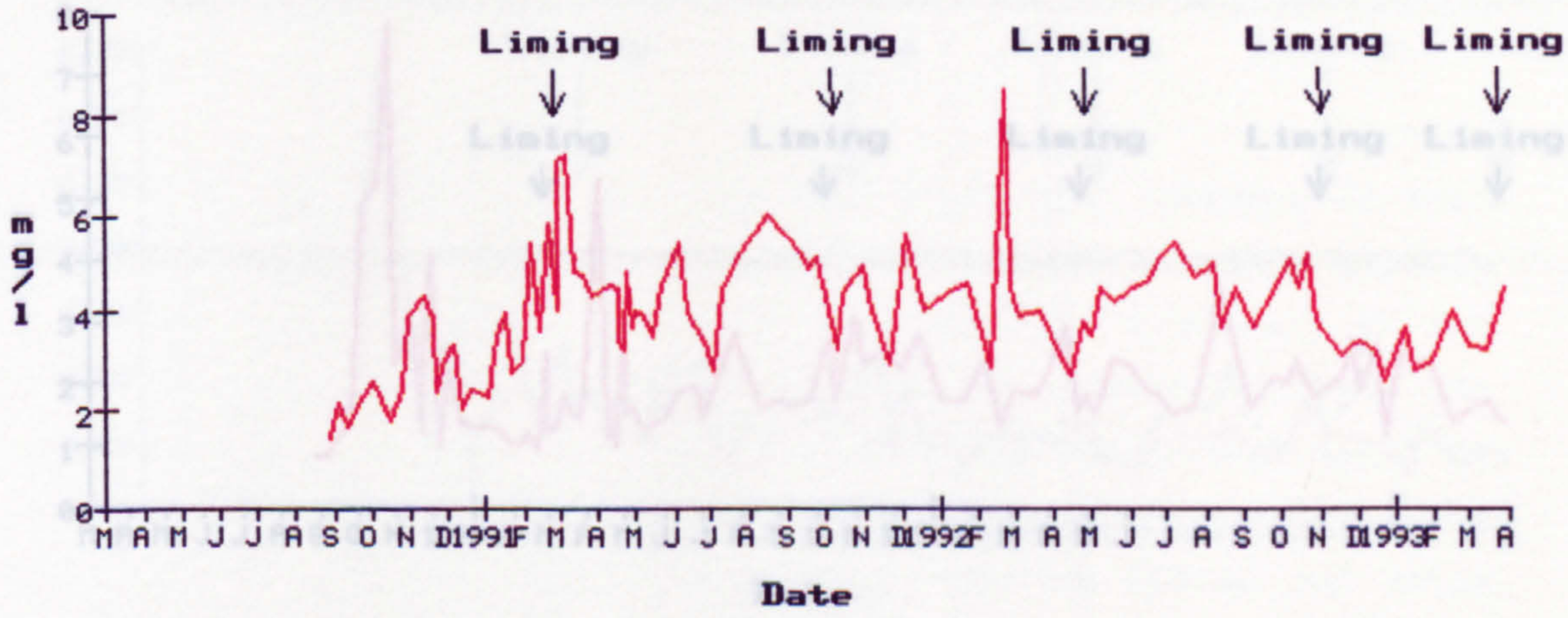


Fig. 5.3(xxi) Hardness at T6

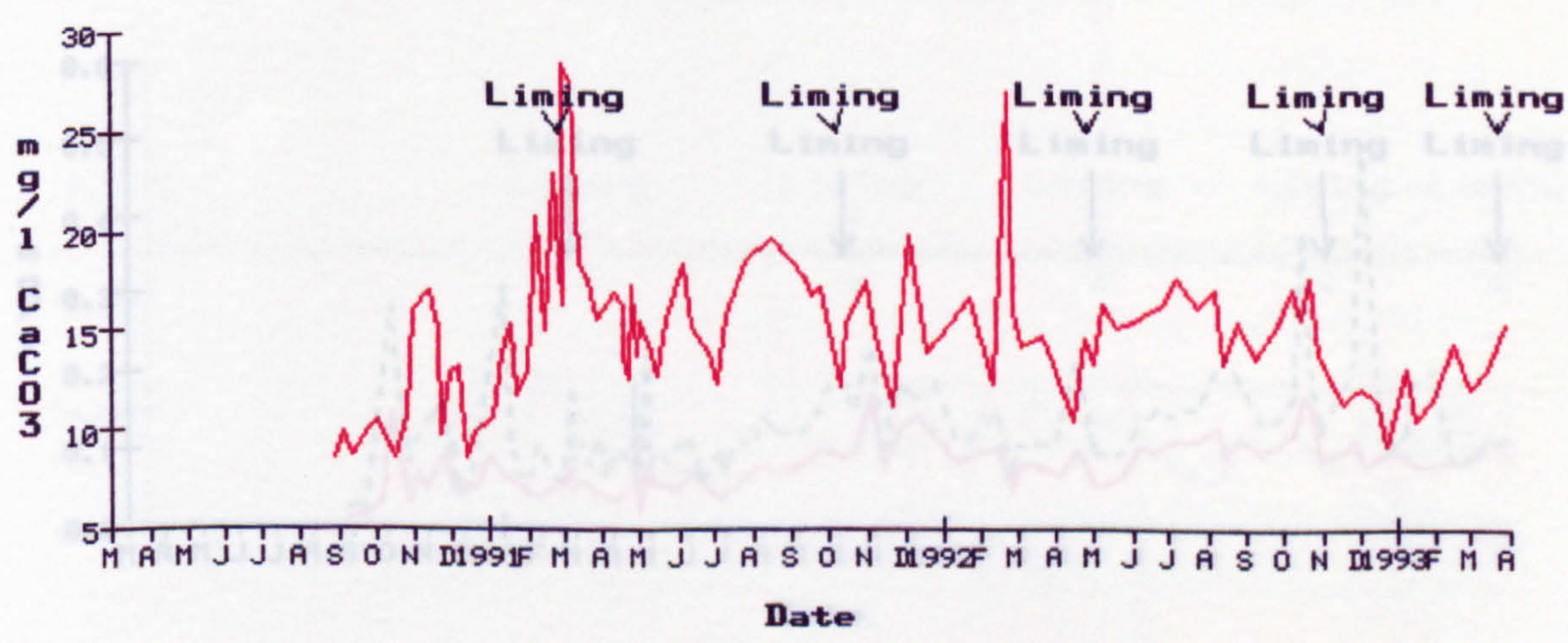


Fig. 5.3(xxii) Alkalinity at T6

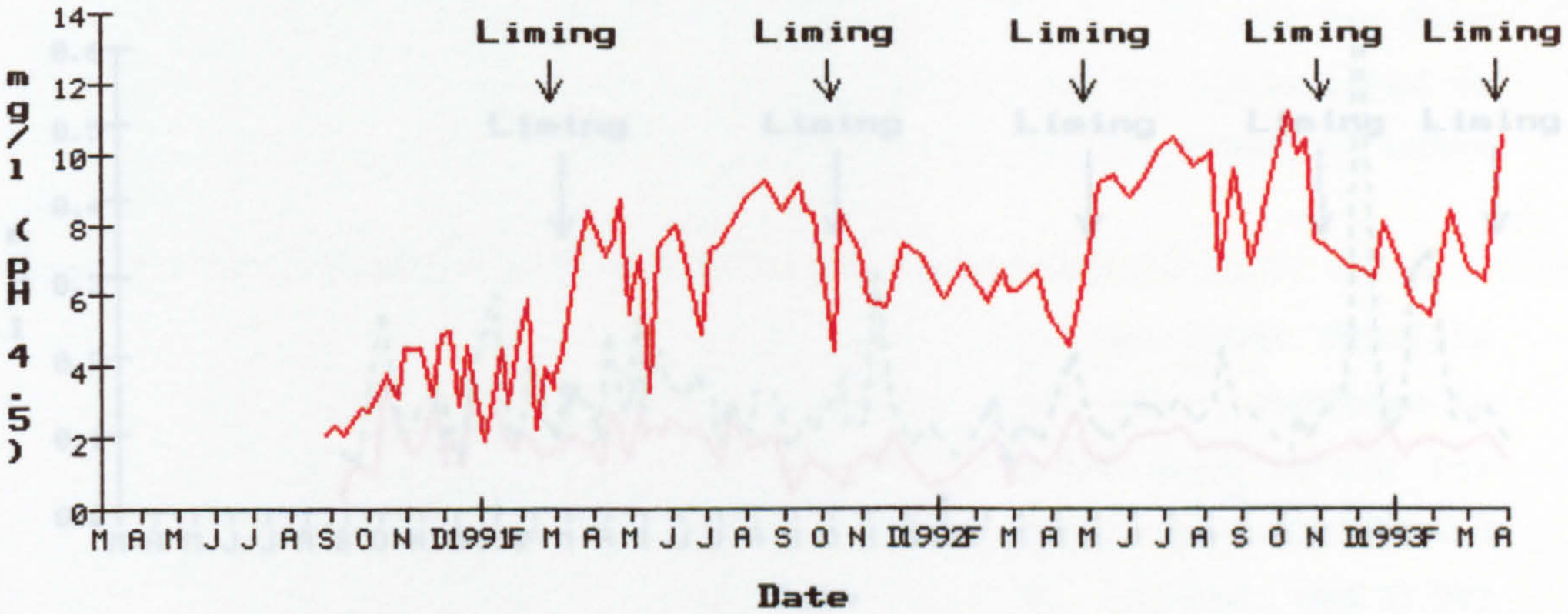


Fig. 5.3(xxiii) Dissolved Organic Carbon at T6

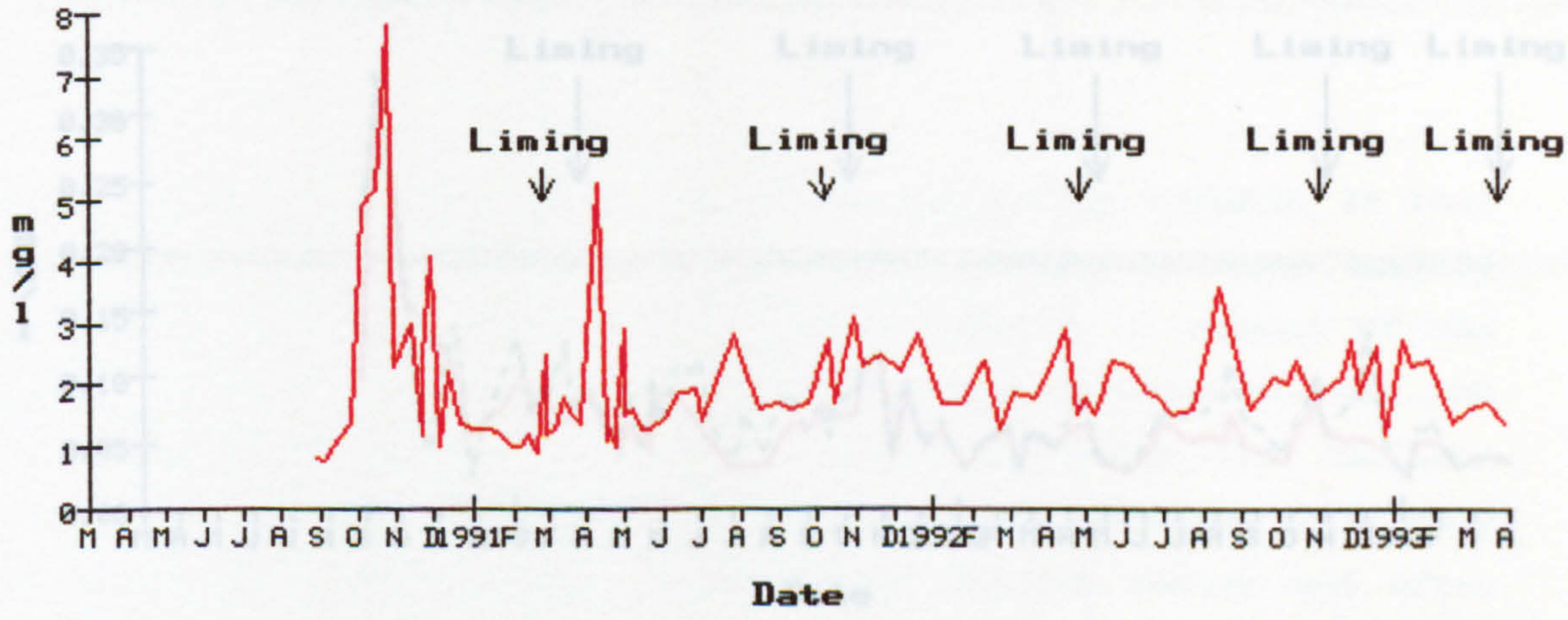


Fig. 5.3(xxiv) Total iron (dot) and filterable iron (solid) from T6

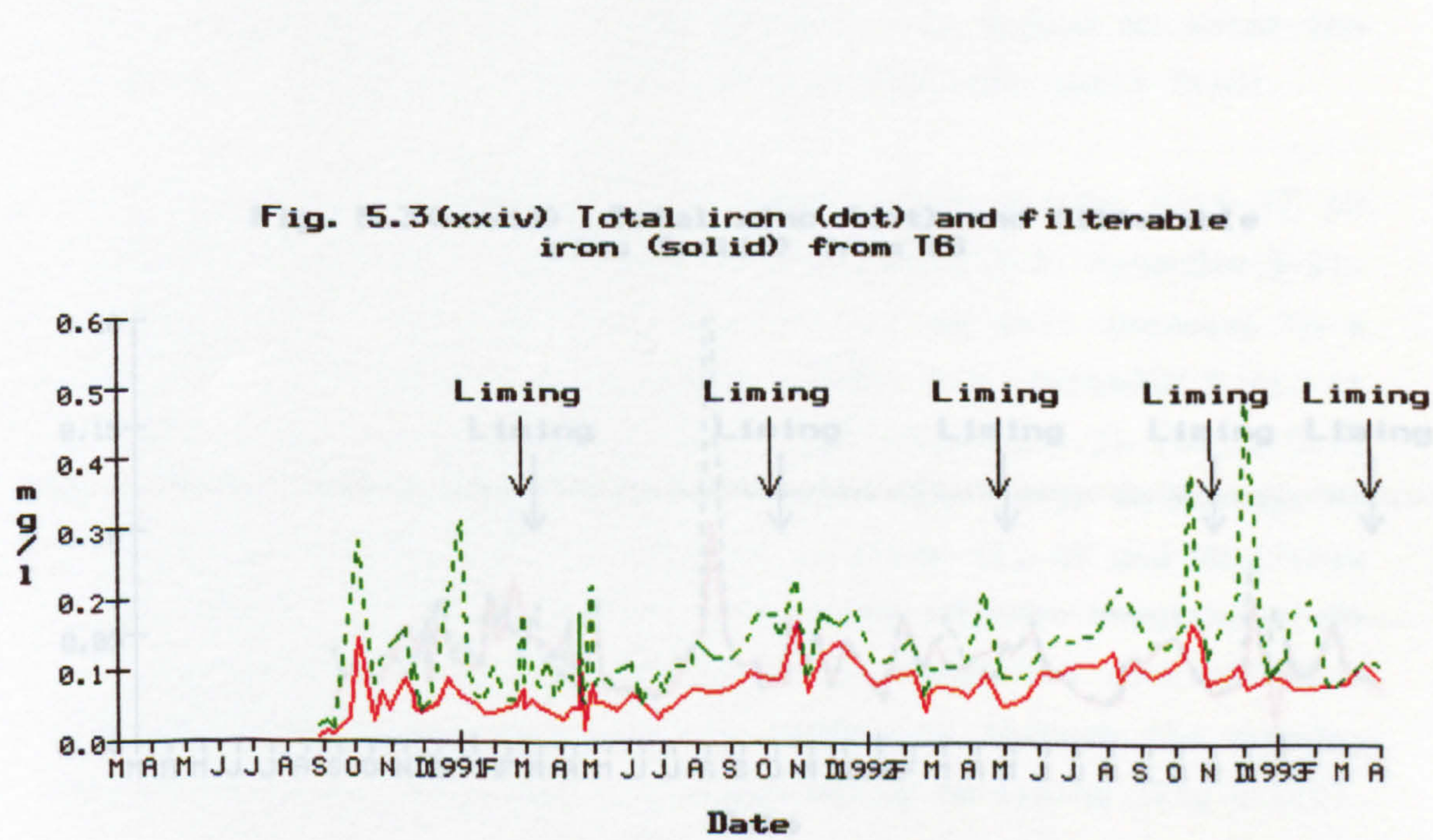


Fig. 5.3(xxv) Total aluminium (dot) and filterable aluminium (solid) from T6

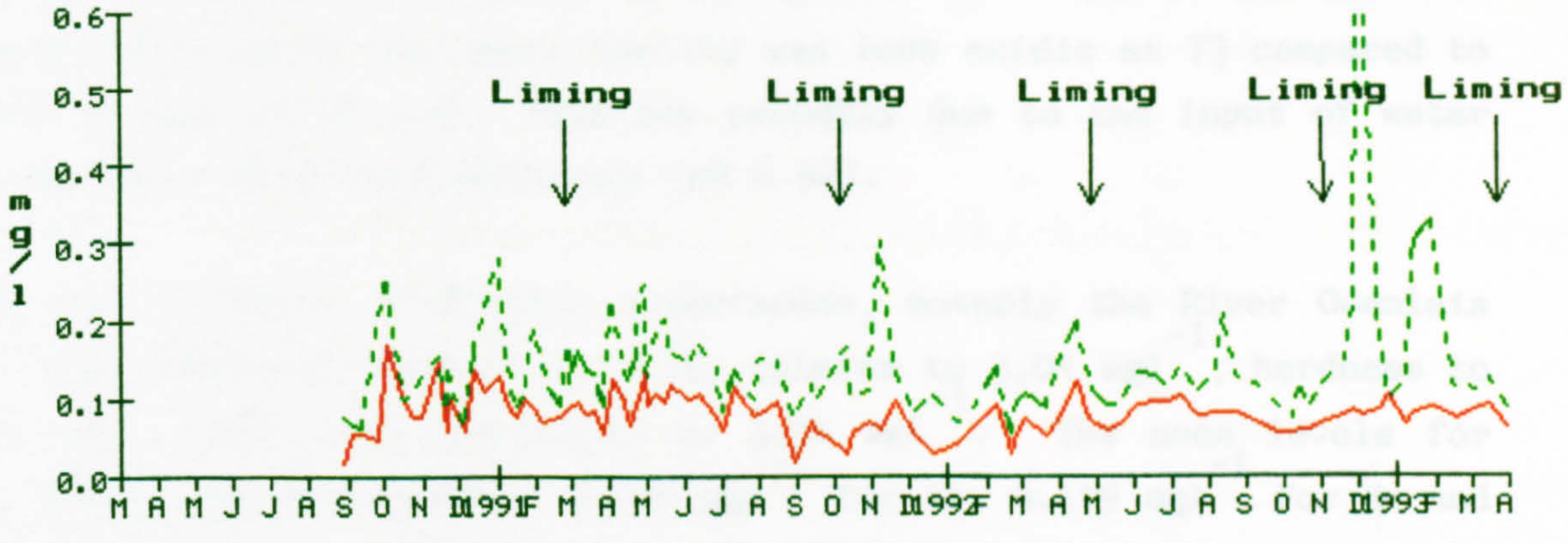


Fig. 5.3(xxvi) Total manganese (dot) and filterable manganese (solid) from T6

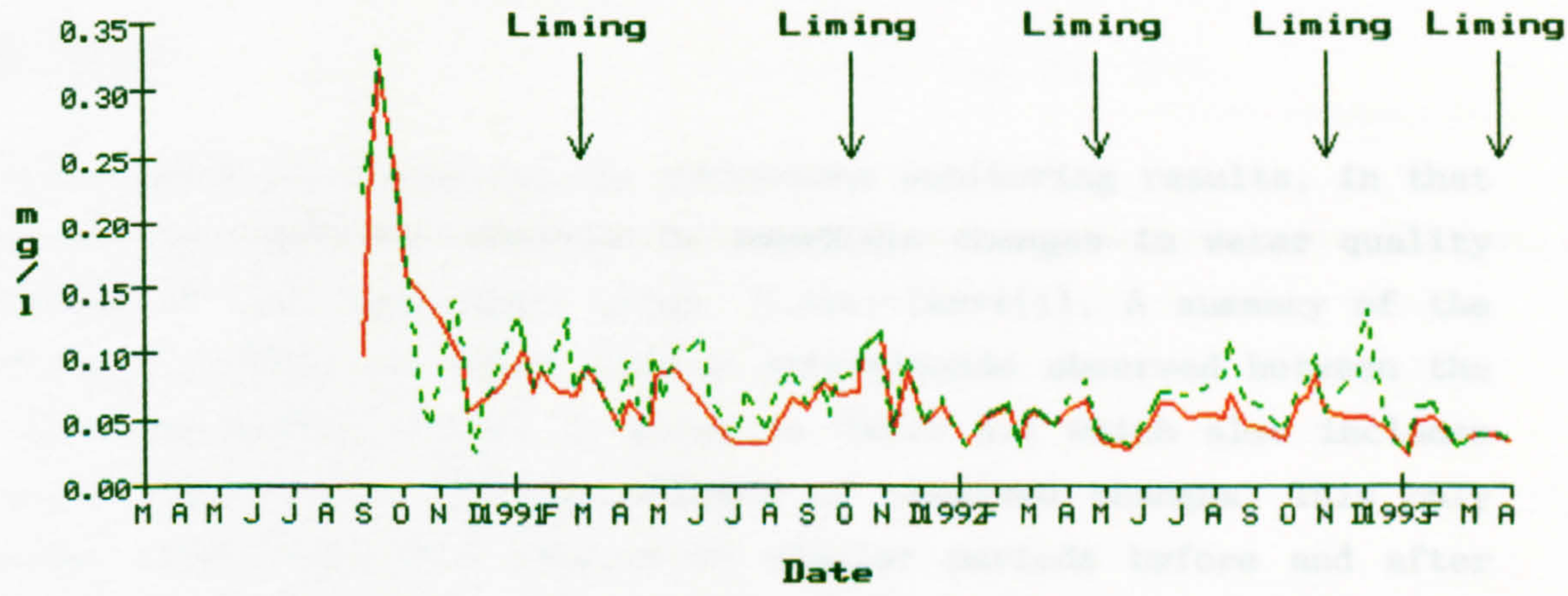
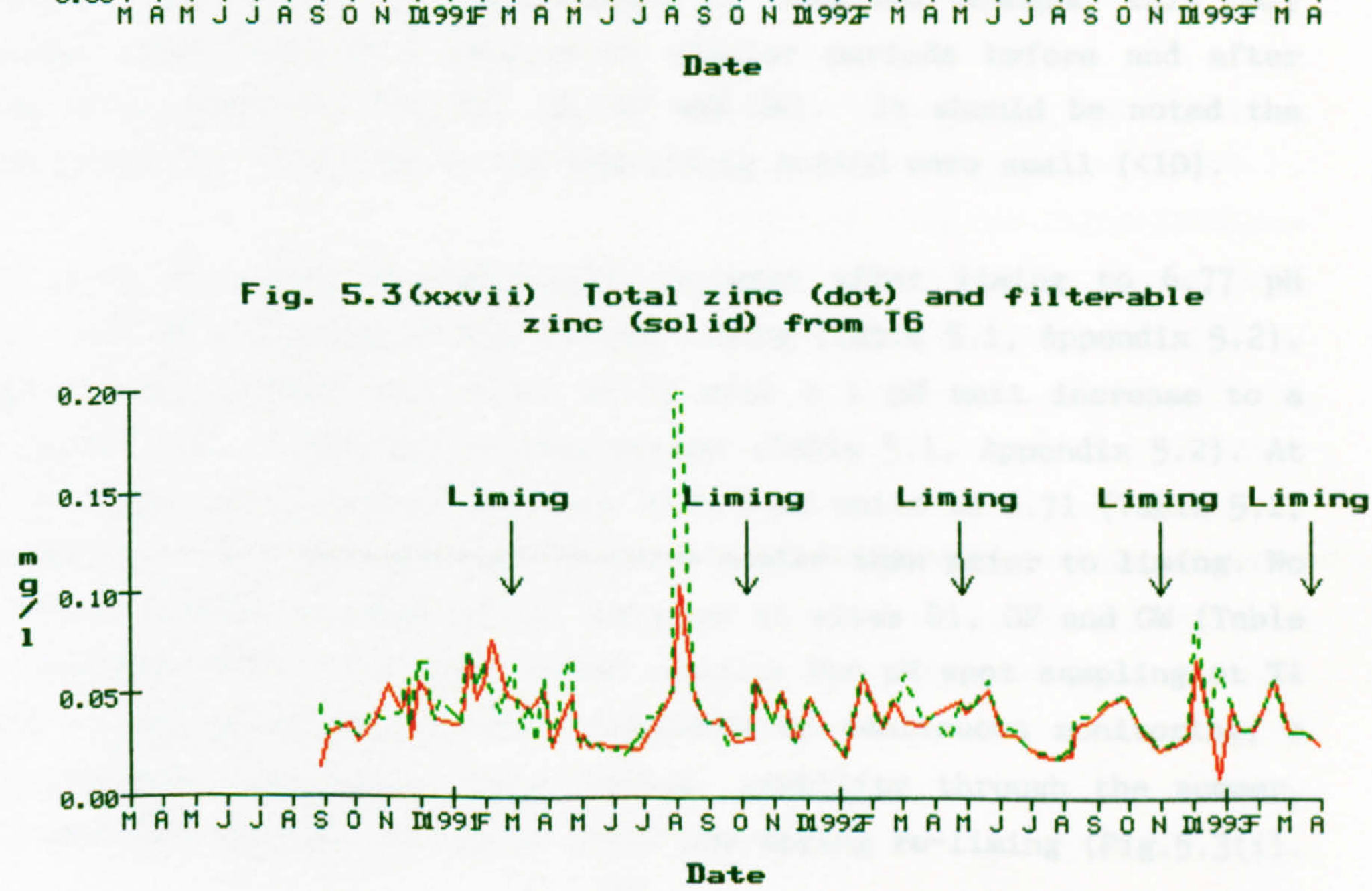


Fig. 5.3(xxvii) Total zinc (dot) and filterable zinc (solid) from T6



below the confluence of the River Doethie, although equivalent comparison is not possible since different sampling periods existed.

The temporal sampling period at T3 was similar to T1 and D1 and the data (Appendix 5.2) shows the water quality was less acidic at T3 compared to T1 with a mean pH of 5.67. This was probably due to the input of water from the River Gwenffrwd tributary (pH 6.52).

At T6 the influence of further tributaries, notably the River Gwenlais (pH 7.10), increased mean pH to 6.19, calcium to 3.04 mgl^{-1}, hardness to 12.73 mgl^{-1} $CaCO_3$ and alkalinity to 3.54 mgl^{-1}. The mean levels for total metals had decreased to 0.088 mgl^{-1} for Al, 0.119 mgl^{-1} for Mn and 0.098 mgl^{-1} for Fe. The mean concentration of zinc, however, was 0.047 mgl^{-1}, over twice the level at T1. This increase resulted from disused mine discharges entering the main river in the vicinity of site T4 (WWA, 1986).

After liming

The spot sample data supports the continuous monitoring results, in that liming of the reservoir resulted in immediate changes in water quality downstream of the impoundment (Figs. 5.3(i)-(xxvii)). A summary of the direction of change in water quality determinands observed between the pre- and post-liming period is given in Table 5.1 which also includes statistics quantifying the significance of observed changes. This only describes sites that were sampled at similar periods before and after liming (i.e. sites T1, T3, T6, D1, GF and GW). It should be noted the sample sizes for GF and GW in the pre-liming period were small (<10).

At T1 mean pH showed a significant increase after liming to 6.77 pH units, 1.47 pH units higher than before liming (Table 5.1, Appendix 5.2). A significant increase was found at T3 with a 1 pH unit increase to a mean pH of 6.62 in the post-liming period (Table 5.1, Appendix 5.2). At site T6 there was a smaller increase of 0.5 pH units to 6.71 (Table 5.1, Appendix 5.2), but this was significantly higher than prior to liming. No significant change in mean pH was observed at sites D1, GF and GW (Table 5.1, Appendix 5.2). The time series results for pH spot sampling at T1 showed a similar pattern to that displayed by continuous monitoring; a rapid increase immediately after liming, stability through the summer, and a decrease through the winter until the spring re-liming (Fig.5.3(i).

Table 5.1 A summary of changes in some chemical determinands between the pre- and post-liming periods. (+) increase (-) decrease.

Determinand	Site T1	T3	T6	D1	GF	GW
pH	+ ****	+ ****	+ ****	NS	NS	NS
Calcium	+ ****	+ ****ₘ	+ ****	- *ₘ	NS	NS
Hardness	+ ****	+ ****	+ **	NS	NS	NS
Alkalinity	+ ****	+ ****	+ ****	+ *	NS	NS
DOC	+ ****ₘ	+ *ₘ	+ *ₘ	+ **ₘ	NS	NS
Fe (filtered)	+ ****	+ ****ₘ	+ ****	+ *ₘ	NS	NS
Fe (total)	+ **	+ ***ₘ	+ *	+ *ₘ	NS	NS
Al (filtered)	- ****	- ****	NS	NS	- ***ₘ	NS
Al (total)	- ****	NS	NS ■	NS	NS	NS
Mn (filtered)	- ****	- ****	- ****	NS	NS	NS
Mn (total)	- ****	- ***ₘ	- ****	- ***	NS	NS
Zn (filtered)	- ****ₘ	- ****ₘ	NS	- **	- ***	NS
Zn (total)	- ****ₘ	- ****ₘ	NS	- ***ₘ	- ****	- ***

Significance level :- **** - $p<0.0001$, *** - $p<0.001$, ** - $p<0.01$
* - $p<0.05$, NS - not significant.

■ denotes that the Mann-Whitney non-parametric test was used.

At T3 and T6 the pH (Figs 5.3(x), 5.3(xix)) reflected the changes upstream, but also showed the effect of events from the River Doethie with a larger variation in pH, although levels below pH 6 were rare.

Calcium

At T1 the calcite addition significantly increased the mean calcium concentration by 2.67 mgl^{-1} following liming, to 4.40 mgl^{-1} (Table 5.1, Appendix 5.2). This also significantly increased calcium concentrations at T3 and T6 to 3.49 mgl^{-1} and 4.22 mgl^{-1}, respectively (Table 5.1, Appendix 5.2). Whilst D1 showed a small but significant decrease in calcium concentration over this period no significant changes were observed at GF and GW (Table 5.1, Appendix 5.2). The time series graph for calcium at T1 (Fig. 5.3(ii)) reflects that of pH, with an immediate increase after liming to typical levels of 4.0-6.0 mgl^{-1}. These levels were sustained throughout the summer but calcium concentrations then decreased with winter rainfall dilution, prior to re-liming in the spring. In both the winter of 1991 and 1992 the lowest calcium level remained above 2.0 mgl^{-1}.

Hardness

Due to the above change in calcium levels total hardness showed significant increases at T1, T3 and T6 to 14.81, 12.78 and 15.20 mgl^{-1} $CaCO_3$, respectively (Table 5.1, Appendix 5.2), compared with mean pre-liming concentrations of 8.46, 9.90 and 12.73 mgl^{-1}, respectively. No significant change was observed in the three tributaries (Table 5.1). The time series graphs (Figs 5.3(iii),(xii) and (xxi)) show the consistent similarity with that of calcium above.

Alkalinity

The mean value of alkalinity showed a 5-fold increase after liming to 7.84 mgl^{-1} $CaCO_3$, compared to 1.57 mgl^{-1} $CaCO_3$ before liming (Table 5.1, Appendix 5.2). Significant increases were also evident at T3 and T6 (Table 5.1), with 3- and 2-fold increases to 5.93 mgl^{-1} $CaCO_3$ and 7.46 mgl^{-1} $CaCO_3$ for mean post-liming values, respectively. D1 showed a slight but significant increase in this period but no significant changes were observed for GF and GW (Table 5.1). The alkalinity time series graph for T1 (Fig 5.3(iv)) shows an increase from 1.5 to c. 8.0 mgl^{-1} $CaCO_3$ immediately after liming in February 1991. The winter decrease is evident but values below 4.0 mgl^{-1} $CaCO_3$ were recorded only once in 1991 or 1992. At site T2, below the River Doethie, the minimum alkalinity recorded during the post-liming period was 2.79 mgl^{-1} $CaCO_3$, with a mean of 5.82 mgl^{-1} $CaCO_3$ (Appendix 5.2). Minimum alkalinity after liming at T3 and T6 were 2.74 and 3.30 mgl^{-1} $CaCO_3$, respectively.

Dissolved Organic Carbon

Dissolved organic carbon (DOC) showed a small but significant increase at T1 after liming (Table 5.1), as at T3 and T6. With the exception of D1, no tributary showed a similar result (Table 5.1, Appendix 5.2). The exceptional values of up to 6.0 mgl^{-1} observed at T1 prior to liming were not evident after liming, with only one detection >3.0 mgl^{-1} (Fig. 5.3(v)).

Aluminium

Aluminium was measured as total aluminium, and filterable aluminium at 0.45 µm. At T1 mean total aluminium showed a significant decrease after liming from 0.247 mgl^{-1} to 0.187 mgl^{-1} (Table 5.1, Appendix 5.2). There was no significant difference in total aluminium at any other site (Table 5.1). The time series graph for T1 (Fig. 5.3(vii) shows how total aluminium decreased over the summer post-liming, although a summer

decrease was also evident in the pre-liming period. The greatest difference in the post-liming period was the high winter and spring levels of aluminium before liming were not repeated during the winter/spring period in 1991/1992 or 1992/1993, although slight increases from summer levels were still evident. Filterable aluminium also showed a significant decrease at T1 following liming, from 0.182 mgl^{-1} to 0.115 mgl^{-1}. In contrast to total aluminium, however, filterable aluminium also showed a significant decrease at T3, although there was no significant difference at T6 (Table 5.1). No significant difference was observed in filterable aluminium at D1 or GW, but GF showed a significant decrease (Table 5.1).

Manganese

Total manganese decreased significantly at T1 after liming falling from 0.347 mgl^{-1} (Table 5.1) to 0.183 mgl^{-1}. Similar statistically significant decreases were also observed at T3 and T6. No significant change was evident in the tributaries apart from a decrease at D1. At T1 (Fig. 5.3(viii)), a major decrease was observed in the peaks of manganese during September and October observed before liming. Spring and summer levels also decreased. Since most of the total manganese is filterable at 0.45 µm (Fig. 5.3(viii)) results for filterable manganese were similar to total manganese.

Zinc

Total zinc levels showed a significant decrease at T1 and T3 following liming (Table 5.1), with an approximate 30% reduction at both sites (Appendix 5.2). This was a gradual change over the first year after liming (Figs. 5.3(ix) and (xvii)). No significant change was observed at T6 (Table 5.1), where zinc levels were higher due to the inputs in the vicinity of T4, described earlier. Filterable zinc was again equivalent to total zinc and so the main river observations were similar for both. Some of the tributaries showed decreases in zinc, though caution is needed in interpreting these results since levels in the tributaries were close to detectable limits.

Iron

In contrast to aluminium, manganese and zinc, total iron concentrations significantly increased in the post-liming period at T1 (Table 5.1) from 0.075 mgl^{-1} before liming to 0.118 mgl^{-1} after liming (Appendix 5.2). Significant increases were also evident at T3 and T6. D1 also showed

increases in iron but no change was observed at either GF or GW. This increase in iron at T1 (Fig. 5.3(vi)) occurred from September 1991 onwards. Filterable iron showed similar trends to total iron, with approximately 65% of total iron being filterable at 0.45 μm.

5.4 Discussion

The discussion will initially focus on the observations immediately downstream of the impoundment, i.e. at site T1. This will then be expanded to evaluate the changes in water quality further downstream of T1, including the effects of other tributaries of the main river.

5.4.1 General chemical characteristics

The NRA's primary objective in liming was to improve the survival of biota (see Chapters 6 & 7) by increasing pH, calcium and alkalinity levels and hence reducing toxic conditions from low pH and toxic metals. Levels of these chemical determinands have been shown to increase elsewhere after calcium carbonate application (e.g. Brocksen & Wisniewski, 1988; Howells & Dalziel, 1992).

The chemical reactions involved are:

$$CaCO_{3(s)} + H^{+} = Ca^{2+} + HCO3^{-} \qquad (pH<6)$$

$$CaCO_{3(s)} + H_2O = Ca^{2+} + HCO3^{-} + OH^{-} \qquad (pH>6)$$

The former equation describes the situation observed at the pH found in this study during the pre-liming period (c.5.0)(Olem, 1991). After the addition of calcium carbonate there is a reduction in H^{+} ions, and a concomitant increase in pH, calcium and bicarbonate (alkalinity) levels, as observed from the reservoir in the post-liming period. Rosseland & Hindar (1988) reported a similar increase in pH and calcium levels for the R. Audna, Norway following the liming of Lake Ytre Oydnavat. They noted an increase in pH from pH 5.0 to 6.8 and calcium levels from 1.5 mgl^{-1} to 3.5 mgl^{-1}.

During the summer period, when draw-down of the reservoir occurs, the compensation flow is taken from the large volume of the hypolimnion. The chemical composition of the water released during this period is therefore usually stable. Stratification occurs during the summer and

there is evidence that the incoming water to the reservoir after stratification only reacidifies the epilimnion (A. Rogers, pers. comm.) in the period prior to autumn overturn. This probably explains why during the wet summer/autumn of 1992, when the reservoir over-topped in September and the river below the impoundment consisted of the usual compensation flow and the overspill water, the pH downstream of the reservoir dropped to c. pH 6.6; pH increased again towards the end of September and during October when over-topping ceased and the outlet water only consisted of hypolimnetic compensation water.

From the results it is difficult to evaluate the effects of the autumn limings since this coincides with autumnal rainfall and therefore immediate dilution takes place. Parameters such as pH, calcium and alkalinity decrease soon after the autumn lime applications but without the autumn application of lime greater reacidification would probably have taken place. In both winters the neutralising capacity of the reservoir seemed able to buffer these inputs to maintain conditions of pH > 6.2, Ca > 2.5 mgl^{-1} and alkalinity > 4.0 mgl^{-1} through to the spring.

The one anomaly between continuous monitoring and spot sampling was the difference in mean values obtained for pH. The mean pH after liming was 6.8 from spot sampling but 7.0 for continuous monitoring. This 0.2 pH units difference in mean pH is greater than the 0.1 units difference given by Weatherley & Ormerod (1991). Assuming the continuous monitors to give a truer picture of events there are two explanations for the observed difference. First, due to convenience, the spot samples were invariably taken between 08:00 and 10:00 h and pH is known to vary diurnally (Mason, 1989) due to respiration and photosynthesis. Thus pH would be lowest at night due to CO_2 production and climb during the day. Consequently, sampling during such a consistently small time period might not be representative of daily means. Evidence of diurnal variation is shown in the pH trace of the River Doethie (Fig. 5.2c). Second, there was obviously a time lag from taking the spot samples to their analysis in the laboratory and Mason (1989) suggests changes can occur in samples during collection, transportation and storage.

The increase in DOC as observed in this study has previously been attributed to increased biological productivity after liming (Marcus, 1988; Broberg, 1988), possibly due to phosphorus release from the sediment and from the limestone (Olem, 1991).

5.4.2 Metals

Another major objective of liming is to reduce metal levels, especially aluminium. Olem (1991) lists the following explanations as to the mechanism involved: precipitation, oxidation, surface adsorption and ion exchange. Typically metal ion reductions are due to the first mechanism since the solubility of metal ions usually decreases with neutral or alkaline conditions.

Aluminium

As stated above aluminium is the primary trace metal targeted for reduction because of its detrimental effects on biota (see Chapters 5 & 6). However, aluminium is a complex metal since its amphoteric nature allows the formation of both inorganic and organic complexes, which tend to polymerise. In fresh water many species can be present representing inorganic monomeric forms, organic monomeric forms, polymeric complexes, aluminium hydroxide colloids, precipitates and clays (Howells *et al.*, 1990). These species are then collectively described as various fractions of aluminium (Driscoll, 1984) such that inorganic monomeric and organic monomeric aluminium constitute total monomeric aluminium, which combined with acid soluble aluminium gives total aluminium.

The inorganic monomeric fraction is regarded as potentially toxic in acidified waters and so this, along with total aluminium, are usually reported. Total aluminium in this study was that measured after acid digestion (Howells *et al.*, 1990): the filtered samples (0.45 µm) equate to total monomeric aluminium as described by Howells *et al.* (1990). Since the organic content of the River Tywi is low (mean DOC prior to liming, 1.79 mgl^{-1}) most of this filtered sample would be in the inorganic monomeric form (Driscoll *et al.*, 1989). Ormerod *et al.* (1989) carried out detailed analysis of aluminium in 13 of the streams entering Llyn Brianne and found that up to 90% of the total monomeric aluminium was inorganic monomeric aluminium. They also found that the concentration of inorganic monomeric aluminium was highly correlated with that of filtered aluminium. Consequently, in this study although the results for the filtered samples would give a slightly higher value for inorganic monomeric aluminium, they should be indicative of this fraction.

Within this inorganic monomeric fraction, the species found are highly pH

dependant (Lydersen, 1990). At a pH <4.0 aluminium is almost entirely in the trivalent state of Al^{3+}. As pH increases the cationic hydroxides $Al(OH)^{2+}$ and $Al(OH)_2^{+}$ predominate around pH 5.0 and 5.5 respectively. The hydroxide $Al(OH)_3$ occurs around pH 6.0 and as pH increases further the dominant form is $Al(OH)_4^{-}$. Lydersen (1990) stressed the effect of temperature on the relative concentrations of the hydrolysis complexes with the shift of the hydroxide species to a higher pH as temperature decreases.

Liming has been found to decrease aluminium levels (e.g. White *et al.*, 1984; Marcus, 1988; Driscoll *et al.*, 1989), due to the hydrolysis and precipitation of the inorganic monomeric forms of aluminium. Olem (1991) reported the liming of Lake Gardsjon in Sweden where total aluminium was found to decrease by some 33% from 0.300 mgl^{-1} to 0.200 mgl^{-1}.

Manganese, zinc and iron.

Other metals have also been shown to decrease after the addition of lime, e.g. manganese and zinc (Yan, 1983; White *et al.*, 1984). The reduction in manganese is thought to be caused by direct precipitation from the water column, while adsorption and/or coprecipitation is suggested as being responsible for the decrease in zinc levels (Driscoll *et al.*, 1989). In this study manganese still showed temporal peaks in September/October of each year, some 3 times the summer level.

The observed increase in iron after liming has been recorded elsewhere (Yan, 1983; White *et al.*, 1984), while other studies have shown no change in iron concentrations (Wright, 1985; Marcus, 1988). White *et al.* (1984) attributed part of the observed increase to the iron content of the limestone added. In this study the limestone contained 0.5-1.0 g/kg dry weight of iron and simple mass balance calculations that these concentrations could account for c. 40% of the increase observed.

Some of the decreases in metals in this study were gradual, occurring over a several month period. Dickson (1983) found that a period of 1-2 months passed before aluminium levels decreased when liming Swedish lakes at low temperatures (1-2 ^{o}C). Driscoll *et al.* (1989) also found a 6-8 weeks period for trace metals to decrease, and suggested that a low concentration of particulate matter may contribute to a slow removal rate.

5.4.3 Longitudinal changes in water quality in the upper River Tywi

The changes observed from immediately downstream of the reservoir at T1 would automatically change the chemical composition of the main river further downstream, though the extent of change depends largely on the dilution or relevant flows from the joining tributaries and any buffering of surface run-off. Prior to liming the tributaries acted in reducing the acidity, but after liming the mean pH at T1 was higher than at D1 and GF so mean pH in the main river may now decrease slightly downstream of these tributaries. The area of particular concern is the main river below the confluence with the River Doethie, which could still experience acidic conditions during an acidic event from the River Doethie. This obviously depends on the buffering capacity of the water from the reservoir, and the ratio of discharge of the River Doethie to the main river. Minimum levels of pH 5.9, Calcium 1.70 mgl^{-1} and alkalinity 2.79 mgl^{-1} were recorded after liming, downstream of the River Doethie confluence at site T2, but these events would have been of short duration and mean values indicate that the water quality from the reservoir can neutralise that of the River Doethie (Appendix 5.3).

Since the discharge of water from the reservoir is regulated then careful consideration is still needed during certain periods to maintain suitable water quality throughout the main river. During the autumn period, with high rainfall, spates may occur in the River Doethie and this can coincide with low flows from the reservoir when the reservoir is replenishing after the summer draw-down and only compensation flow is being released. Also, this has implications for any future changes in compensation flow since decreasing minimum compensation flow may adversely affect the buffering capacity of the main river.

Further downstream at T3, T5 and T6 the addition of other tributaries means that the effect of episodes from the River Doethie are further dampened, and so the increase in pH from the reservoir now ensures that periods of pH <6.0, Ca <2.5 mgl^{-1} and alkalinity <3.0 mgl^{-1} are rare.

5.4.4 The operational management of liming

Evidence available to date demonstrates that the neutralising capacity of the main river is able to cope with acidic episodes from the River Doethie, in maintaining the water quality downstream of the tributary

above that observed before liming. However, flood events in the River Doethie could result in acidic conditions below its confluence if the relative flow from the reservoir was low (Appendix 5.3). Consideration to alleviating this by increasing the flow from the reservoir during such periods could be considered, though it is acknowledged that this would be difficult due to operational needs of the reservoir owner (Welsh Water plc).

Currently the twice-annual lime application takes place in March/April and October/November. The timings of re-application are important to ensure efficient use of lime and the protection of biota. The spring liming needs to be completed as early as possible in February/March because salmonid eggs may be hatching during this period, and since this life stage is particularly sensitive (Chapter 7) missing this 'window' could result in the loss of a year class. This is also the period when the sensitive pre-smolt and smolt stage need protecting.

Since the spring liming takes place prior to stratification of the reservoir the whole water body is neutralised. After stratification, since it is the epilimnion that displays greater reacidification, the water quality of the hypolimnetic compensation flow is stable. However, in years of wet summers, e.g. 1992, re-acidified epilimnetic water can overspill the reservoir causing conditions of lower pH below the impoundment than if hypolimnial compensation flow alone was released. Destratification usually takes place in October/November of each year (A. Rogers pers. comm.) causing mixing of surface and bottom water to give a more uniform water quality. Driscoll *et al.*(1989) found that liming a stratified lake only neutralised the epilimnion but turnover enabled thorough mixing. Since Llyn Brianne has its second liming of the year around turnover time care should be taken in wet years when the reservoir is over-topping early, since much of the lime could be lost over the spillway.

The discussion above has considered the current operational regime of the reservoir, i.e. of a hypolimnial release of compensation water. Concerns about the effect of this release regime, specifically relating to altered thermal regimes, have raised the possibility of changing the operational regime to an epilimnial release, or multi-level draw-off. This would have important consequences for the liming operation. During the summer, instead of the stable conditions of the hypolimnion, the reacidifying

epilimnial water would be released, possibly resulting in less favourable water quality below the impoundment. This could be offset somewhat with a multi-level draw-off system by mixing hypolimnial and epilimnial water. In answer to this the NRA could investigate using a different dosing technique to combat the summer input of water, such as direct stream dosing of the two major inputs to the reservoir. In Norway, rivers of approximately 10 $m^3 s^{-1}$ are dosed in this way (Rosseland & Hindar, 1988). Dosing the outflow from the reservoir would not be as practical since this would allow the reservoir to reacidify, possibly releasing toxic metals back into the water body.

To summarise, liming has ameliorated the acidic and high trace metal level of the reservoir water quality observed prior to liming. The current liming operation has enabled conditions close to neutrality to be maintained since the original liming, with some changes due to dilution associated with precipitation and run off over winter. Reductions in the concentrations of the trace metals aluminium, manganese and zinc were observed after liming and have been maintained, with an increase in iron concentrations after liming.

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Appendix 5.1

Determinands included in the analysis of the spot samples and the analytical methodology.

Determinand	Analytical method
pH, pH units	Skalar automated system
Conductivity, Uscm-1	Manual Phillips conductivity meter
Colour, colour units (H)	Automated colourimetric analyser
Ammonia, mgl-1	" " "
Nitrogen total oxidised, mgl-1	" " "
Nitrite, mgl-1	" " "
Hardness total, mgl-1	Calculated as 2.5*Ca + 4.11*Mg
Alkalinity pH 4.5, mgl-1 $CaCO_3$	Mettler/Metrohm autotitrator
Chloride ion, mgl-1	Automated colourimetric analyser
Ortho-phosphate, mgl-1	" " "
Silica as Si, mgl-1 Si	" " "
Sulphate filtered, mgl-1	ICPOES
Sulphate total, mgl-1	ICPOES
Sodium filtered, mgl-1	ICPOES/ICPMS
Potassium filtered, mgl-1	ICPOES/ICPMS
Copper filtered, mgl-1	ICPMS
Copper total, mgl-1	ICPMS
Magnesium filtered, mgl-1	ICPOES
Calcium filtered, mgl-1	ICPOES
Zinc filtered, mgl-1	ICPOES/ICPMS
Zinc total, mgl-1	ICPOES/ICPMS
Cadmium filtered, mgl-1	ICPOES/ICPMS
Cadmium total, mgl-1	ICPOES/ICPMS
Aluminium filtered, mgl-1	ICPOES/ICPMS
Aluminium total, mgl-1	ICPOES/ICPMS
Dissolved organic carbon, mgl-1	Automated colourimetric analyser
Chromium filtered, mgl-1	ICPOES/ICPMS
Chromium total, mgl-1	ICPOES/ICPMS
Manganese filtered, mgl-1	ICPOES/ICPMS
Manganese total, mgl-1	ICPOES/ICPMS
Iron filtered, mgl-1	ICPOES/ICPMS
Iron total, mgl-1	ICPOES/ICPMS
Nickel filtered, mgl-1	ICPOES/ICPMS
Nickel total, mgl-1	ICPOES/ICPMS

ICPOES - Inductively coupled plasma optical spectroscopy
ICPMS - Inductively coupled plasma mass spectroscopy

Where the methodology ICPOES/ICPMS are both included for some determinands this relates to the laboratory changing methodology from the former to the latter in September 1991.

Appendix 5.2. Results of spot samples for selected determinands.

Table A5.2.1

Site	pH (pH units)				
	No.	Mean	Min	Max	Median
T1					
Pre-liming	26	5.30	4.19	6.20	5.19
Post-liming	90	6.77	6.10	7.40	6.75
D1					
Pre-liming	12	5.79	5.13	7.30	5.60
Post liming	92	5.86	4.51	7.10	5.90
T2					
Pre-liming	6	5.18	4.88	5.37	5.21
Post-liming	68	6.58	5.90	7.03	6.60
GF					
Pre-liming	6	6.52	5.70	7.10	6.45
Post-liming	19	6.61	6.20	7.00	6.60
T3					
Pre-liming	28	5.67	5.00	6.90	5.60
Post-liming	90	6.62	5.80	7.90	6.60
T5					
Pre-liming	No data				
Post-liming	69	6.62	6.10	7.03	6.60
GW					
Pre-liming	6	7.10	6.70	7.60	7.05
Post-liming	19	6.98	6.70	7.70	6.90
T6					
Pre-liming	22	6.19	5.72	6.56	6.21
Post-liming	72	6.71	6.10	7.10	6.70

Table A5.2.2

Site	Total Hardness (mgl^{-1} $CaCO_3$)				
	No.	Mean	Min	Max	Median
T1					
Pre-liming	28	8.46	5.71	12.80	8.35
Post-liming	87	14.81	9.08	29.50	14.90
D1					
Pre-liming	11	11.58	5.50	24.60	9.95
Post liming	91	7.98	3.06	16.70	7.69
T2					
Pre-liming	5	11.05	9.26	12.60	11.80
Post-liming	66	12.47	8.11	17.30	12.35
GF					
Pre-liming	5	11.07	7.74	15.30	10.80
Post-liming	19	10.76	4.47	24.30	10.20
T3					
Pre-liming	29	9.90	5.78	17.10	9.69
Post-liming	90	12.78	8.37	27.00	12.60
T5					
Pre-liming	No data				
Post-liming	68	13.60	9.19	20.20	13.70
GW					
Pre-liming	6	24.73	12.40	44.10	23.50
Post-liming	19	20.92	9.24	35.00	19.50
T6					
Pre-liming	23	12.73	8.54	22.90	11.80
Post-liming	70	15.20	9.12	27.60	15.40

Table A5.2.3

Site	Alkalinity pH 4.5 ($mgl^{-1}CaCO_3$)				
	No.	Mean	Min	Max	Median
T1					
Pre-liming	24	1.57	1.06	2.78	1.40
Post-liming	90	7.84	3.84	13.20	7.60
D1					
Pre-liming	10	1.75	1.12	2.60	1.87
Post liming	93	2.56	0.60	5.92	2.42
T2					
Pre-liming	6	1.32	0.84	1.60	1.38
Post-liming	68	5.82	2.79	10.40	5.45
GF					
Pre-liming	4	3.98	3.70	4.20	4.00
Post-liming	19	4.99	2.93	7.27	5.32
T3					
Pre-liming	26	1.93	1.20	2.60	2.00
Post-liming	90	5.93	2.74	11.50	5.50
T5					
Pre-liming	No data				
Post-liming	69	6.35	3.50	10.30	6.05
GW					
Pre-liming	4	8.48	4.90	13.10	7.95
Post-liming	19	11.75	6.46	23.10	10.70
T6					
Pre-liming	22	3.54	1.94	5.88	3.16
Post-liming	72	7.46	3.30	11.20	7.27

Table A5.2.4

Calcium (mgl^{-1})				
No.	Mean	Min	Max	Median
28	1.73	0.90	2.87	1.70
87	4.40	2.39	10.21	4.30
11	2.24	0.25	4.60	2.10
90	1.67	0.03	4.40	1.60
5	2.48	1.98	3.30	2.50
67	3.42	1.70	5.30	3.40
5	2.57	1.72	4.20	2.42
19	2.67	0.10	7.30	2.50
29	2.15	1.18	3.94	1.94
90	3.49	1.71	8.00	3.47
No data				
68	3.72	2.20	5.90	3.74
6	7.51	2.80	16.90	6.63
19	5.87	1.40	10.22	5.50
23	3.04	1.44	5.83	2.78
70	4.22	2.60	8.47	4.26

Appendix 5.2. continued. Results of spot samples for selected determinands.

Table A5.2.5 Aluminium -filterable (mgl^{-1})

Site	No.	Mean	Min	Max	Median
T1					
Pre-liming	27	0.182	0.048	0.301	0.199
Post-liming	89	0.115	0.049	0.173	0.095
D1					
Pre-liming	10	0.103	0.049	0.173	0.095
Post liming	80	0.078	0.008	0.194	0.076
T2					
Pre-liming	4	0.172	0.126	0.230	0.166
Post-liming	69	0.101	0.029	0.230	0.099
GF					
Pre-liming	5	0.083	0.040	0.147	0.082
Post-liming	16	0.029	0.004	0.073	0.030
T3					
Pre-liming	28	0.117	0.056	0.183	0.112
Post-liming	91	0.090	0.014	0.192	0.088
T5					
Pre-liming	No data				
Post-liming	67	0.087	.0004	0.171	0.084
GW					
Pre-liming	5	0.028	0.013	0.042	0.027
Post-liming	18	0.028	0.004	0.076	0.026
T6					
Pre-liming	22	0.088	0.017	0.168	0.080
Post-liming	73	0.077	0.023	0.134	0.077

Table A5.2.6 Aluminium -total (mgl^{-1})

Site	No.	Mean	Min	Max	Median
T1					
Pre-liming	24	0.247	0.109	0.449	0.252
Post-liming	91	0.187	0.073	0.389	0.182
D1					
Pre-liming	9	0.120	0.051	0.190	0.100
Post liming	93	0.107	0.024	0.334	0.099
T2					
Pre-liming	6	0.202	0.158	0.260	0.199
Post-liming	69	0.168	0.081	0.340	0.152
GF					
Pre-liming	5	0.099	0.044	0.189	0.061
Post-liming	17	0.063	0.004	0.212	0.034
T3					
Pre-liming	25	0.179	0.071	0.535	0.152
Post-liming	90	0.148	0.070	0.397	0.140
T5					
Pre-liming	No data				
Post-liming	69	0.146	.0004	0.394	0.129
GW					
Pre-liming	6	0.050	0.029	0.109	0.040
Post-liming	19	0.058	0.004	0.265	0.041
T6					
Pre-liming	19	0.127	0.062	0.281	0.110
Post-liming	73	0.145	0.056	0.687	0.127

Table A5.2.7

Site	Manganese -filterable (mgl^{-1})				
	No.	Mean	Min	Max	Median
T1					
Pre-liming	20	0.302	0.168	0.635	0.226
Post-liming	81	0.171	0.029	0.400	0.156
D1					
Pre-liming	7	0.110	0.053	0.399	0.056
Post liming	72	0.037	0.004	0.098	0.038
T2					
Pre-liming	5	0.122	0.059	0.186	0.112
Post-liming	62	0.109	0.049	0.228	0.102
GF					
Pre-liming	3	0.127	0.012	0.353	0.016
Post-liming	17	0.009	0.002	0.018	0.008
T3					
Pre-liming	18	0.169	0.070	0.353	0.141
Post-liming	85	0.091	0.023	0.193	0.083
T5					
Pre-liming	No data				
Post-liming	61	0.076	0.031	0.146	0.074
GW					
Pre-liming	3	0.007	0.002	0.011	0.008
Post-liming	17	0.011	0.002	0.039	0.009
T6					
Pre-liming	16	0.127	0.056	0.315	0.097
Post-liming	60	0.054	0.024	0.116	0.051

Table A5.2.8

Site	Manganese -total (mgl^{-1})				
	No.	Mean	Min	Max	Median
T1					
Pre-liming	22	0.347	0.180	0.694	0.283
Post-liming	91	0.183	0.084	0.400	0.173
D1					
Pre-liming	9	0.112	0.045	0.471	0.067
Post liming	93	0.041	0.004	0.271	0.039
T2					
Pre-liming	6	0.145	0.119	0.186	0.133
Post-liming	69	0.144	0.054	0.930	0.121
GF					
Pre-liming	4	0.104	0.014	0.353	0.024
Post-liming	19	0.011	0.002	0.019	0.012
T3					
Pre-liming	23	0.169	0.038	0.375	0.120
Post-liming	92	0.103	0.044	0.250	0.096
T5					
Pre-liming	No data				
Post-liming	69	0.088	0.033	0.191	0.089
GW					
Pre-liming	4	0.010	0.002	0.015	0.011
Post-liming	19	0.012	0.002	0.043	0.012
T6					
Pre-liming	19	0.119	0.025	0.332	0.090
Post-liming	73	0.065	0.024	0.133	0.063

Table A5.2.9

Site	Zinc -filterable (mgl^{-1})				
	No.	Mean	Min	Max	Median
T1					
Pre-liming	21	0.018	0.013	0.026	0.017
Post-liming	73	0.010	0.002	0.033	0.009
D1					
Pre-liming	6	0.014	0.007	0.020	0.014
Post liming	68	0.006	0.002	0.021	0.005
T2					
Pre-liming	3	0.016	0.010	0.020	0.018
Post-liming	48	0.008	0.002	0.023	0.007
GF					
Pre-liming	4	0.007	0.004	0.011	0.007
Post-liming	15	0.003	0.002	0.005	0.003
T3					
Pre-liming	21	0.014	0.006	0.019	0.014
Post-liming	72	0.008	0.002	0.033	0.007
T5					
Pre-liming	No data				
Post-liming	59	0.044	0.011	0.091	0.042
GW					
Pre-liming	4	0.005	0.005	0.006	0.005
Post-liming	15	0.004	0.002	0.007	0.004
T6					
Pre-liming	20	0.044	0.015	0.077	0.040
Post-liming	62	0.038	0.010	0.100	0.036

Table A5.2.10

Zinc -total (mgl^{-1})				
No.	Mean	Min	Max	Median
22	0.019	0.012	0.035	0.018
91	0.012	0.002	0.083	0.010
10	0.013	0.006	0.023	0.014
93	0.008	0.002	0.054	0.006
6	0.022	0.010	0.031	0.022
74	0.009	0.002	0.054	0.007
4	0.009	0.007	0.011	0.008
19	0.003	0.002	0.006	0.003
23	0.014	0.008	0.022	0.014
92	0.010	0.002	0.092	0.008
No data				
69	0.047	0.013	0.173	0.042
4	0.008	0.005	0.011	0.008
19	0.004	0.002	0.007	0.004
20	0.047	0.030	0.072	0.042
73	0.042	0.020	0.242	0.037

Appendix 5.2. continued. Results of spot samples for selected determinands.

Table A5.2.11

Site	Magnesium -filterable (mgl^{-1})				
	No.	Mean	Min	Max	Median
T1					
Pre-liming	28	1.005	0.707	1.388	1.004
Post-liming	89	0.927	0.630	2.010	0.910
D1					
Pre-liming	12	1.346	0.110	3.140	1.215
Post liming	93	0.935	0.610	1.440	0.910
T2					
Pre-liming	6	1.188	1.010	1.450	1.135
Pre-liming	68	0.949	0.680	1.200	0.970
GF					
Pre-liming	6	0.972	0.190	1.330	1.130
Post-liming	19	0.992	0.730	1.480	1.000
T3					
Pre-liming	29	1.101	0.691	1.758	1.100
Post-liming	91	0.994	0.099	4.460	0.940
T5					
Pre-liming	No data				
Post-liming	69	1.045	0.780	1.570	1.040
GW					
Pre-liming	6	1.455	0.640	2.110	1.520
Post-liming	19	1.524	1.100	2.310	1.460
T6					
Pre-liming	23	1.250	0.877	2.040	1.204
Post-liming	72	1.162	0.640	3.330	1.115

Table A5.2.12

Site	DOC (mgl^{-1})				
	No.	Mean	Min	Max	Median
T1					
Pre-liming	26	1.790	0.900	5.100	1.400
Post-liming	90	1.940	1.200	4.100	1.900
D1					
Pre-liming	10	1.260	0.930	1.800	1.150
Post liming	76	2.550	1.100	6.100	2.125
T2					
Pre-liming	6	1.250	1.100	1.400	1.250
Pre-liming	74	2.038	1.000	3.100	1.900
GF					
Pre-liming	4	4.875	0.900	16.30	1.150
Post-liming	17	1.516	0.560	2.700	1.540
T3					
Pre-liming	27	1.950	0.630	5.400	1.300
Post-liming	91	2.018	1.200	7.000	1.900
T5					
Pre-liming	No data				
Post-liming	66	1.969	0.082	3.500	1.900
GW					
Pre-liming	4	0.875	0.650	1.000	0.925
Post-liming	19	1.448	0.600	2.800	1.300
T6					
Pre-liming	23	2.118	0.800	7.800	1.300
Post-liming	73	1.984	1.000	5.300	1.800

Appendix 5.2. continued. Results of spot samples for selected determinands.

Table A5.2.13

Site	Iron -filterable (mgl^{-1})				
	No.	Mean	Min	Max	Median
T1					
Pre-liming	22	0.075	0.005	0.168	0.079
Post-liming	90	0.118	0.023	0.338	0.115
D1					
Pre-liming	8	0.044	0.034	0.059	0.042
Post liming	75	0.100	0.016	0.393	0.076
T2					
Pre-liming	5	0.063	0.036	0.086	0.058
Post-liming	69	0.114	0.012	0.970	0.101
GF					
Pre-liming	4	0.023	0.015	0.036	0.021
Post-liming	17	0.023	0.003	0.056	0.021
T3					
Pre-liming	26	0.056	0.008	0.145	0.053
Post-liming	91	0.096	0.012	0.215	0.091
T5					
Pre-liming	No data				
Post-liming	68	0.091	0.034	0.190	0.088
GW					
Pre-liming	4	0.016	0.008	0.022	0.016
Post-liming	15	0.040	0.005	0.205	0.018
T6					
Pre-liming	21	0.051	0.007	0.143	0.047
Post-liming	71	0.082	0.019	0.170	0.082

Table A5.2.14

Iron -total (mgl^{-1})				
No.	Mean	Min	Max	Median
22	0.156	0.010	0.539	0.143
91	0.180	0.039	0.384	0.176
10	0.159	0.018	1.130	0.051
77	0.126	0.011	0.508	0.100
6	0.113	0.079	0.163	0.104
69	0.163	0.045	0.387	0.155
4	0.166	0.024	0.571	0.034
19	0.036	0.010	0.091	0.025
23	0.111	0.030	0.571	0.081
92	0.148	0.017	0.612	0.131
No data				
69	0.157	0.060	0.439	0.144
4	0.019	0.010	0.024	0.020
19	0.081	0.003	0.550	0.019
19	0.098	0.020	0.314	0.081
73	0.138	0.054	0.482	0.120

Appendix 5.2. continued. Results of spot samples for selected determinands.

Table A5.2.15

Site	Conductivity (Us cm^{-1})				
	No.	Mean	Min	Max	Median
T1					
Pre-liming	25	54.88	52.00	68.00	54.00
Post-liming	90	54.79	34.60	66.00	56.20
D1					
Pre-liming	10	53.00	49.00	62.00	52.00
Post liming	78	54.79	34.60	66.00	56.20
T2					
Pre-liming	6	52.43	49.60	54.00	53.00
Post-liming	68	50.86	37.00	63.20	52.00
GF					
Pre-liming	4	60.50	56.00	66.00	60.00
Post-liming	19	46.96	33.00	55.00	47.10
T3					
Pre-liming	26	54.08	47.10	65.00	54.00
Post-liming	90	50.53	38.00	62.20	51.00
T5					
Pre-liming	No data				
Post-liming	69	53.33	39.00	62.40	54.78
GW					
Pre-liming	4	70.75	54.00	82.00	73.50
Post-liming	19	70.18	54.00	105.3	66.60
T6					
Pre-liming	22	59.91	52.00	69.00	60.50
Post-liming	72	56.62	43.00	67.60	57.00

Table A5.2.16

Site	Silicate- filterable (mgl^{-1})				
	No.	Mean	Min	Max	Median
T1					
Pre-liming	23	2.243	2.011	2.503	2.225
Post-liming	69	2.129	1.177	2.524	2.139
D1					
Pre-liming	6	1.213	0.950	1.530	1.225
Post liming	70	1.787	0.747	3.337	1.818
T2					
Pre-liming	6	1.083	0.960	1.160	1.085
Post-liming	63	1.736	0.085	2.824	2.075
GF					
Pre-liming	No data				
Post-liming	No data				
T3					
Pre-liming	23	2.446	1.797	2.886	2.439
Post-liming	69	2.304	0.519	2.952	2.289
T5					
Pre-liming	No data				
Post-liming	65	1.876	0.800	3.187	2.053
GW					
Pre-liming	No data				
Post-liming	No data				
T6					
Pre-liming	23	2.627	1.602	1.540	2.652
Post-liming	69	2.500	1.562	3.337	2.545

Appendix 5.3 The results of mixing experiments carried out with water samples from the River Doethie and the River Tywi (T1).

Ten litres of water from each site were taken on two different occassions and mixed at the laboratory. Samples were mixed with a ratio ranging from 1:1 to 7:1 dilution of Doethie to Tywi water.

Table A5.3.1 Results of samples taken on the 31 October 1991. (f)-filtered, (t)-total. All determinands to standard units.

Determinand	Tywi	Doethie	Ratio Doethie : Tywi 1:1	2:1	3:1	4:1	5:1	6:1	7:1
pH	6.92	4.98	6.41	6.01	5.72	5.62	5.42	5.46	5.78
Conductivity	60.9	45.9	48.7	44.9	43.2	42.0	41.4	41.4	41.5
Hardness	17.40	6.62	12.30	10.10	9.28	8.81	7.37	7.42	6.94
Alkalinity	9.18	1.12	4.80	3.22	2.34	2.14	1.70	1.78	2.38
Calcium (f)	5.31	1.37	3.48	2.74	2.43	2.21	1.75	1.72	1.58
Aluminium (f)	.086	.186	.140	.155	.164	.160	.166	.182	.180
Aluminium (t)	.126	.488	.298	.388	.522	.469	.544	.491	.593

Table A5.3.2 Results of samples taken on the 7 January 1992. (f)-filtered, (t)-total. All determinands to standard units.

Determinand	Tywi	Doethie	Ratio Doethie : Tywi 1:1	2:1	3:1	4:1	5:1	6:1	7:1
pH	6.70	5.30	6.20	6.20	6.00	6.10	6.00	6.00	5.90
Conductivity	48.2	41.9	45.3	44.1	43.6	43.7	43.3	43.0	43.0
Hardness	15.10	8.73	11.60	10.70	10.40	10.30	9.84	9.69	9.67
Alkalinity	6.00	1.24	3.56	3.76	2.54	2.64	2.28	2.32	1.98
Calcium (f)	4.51	1.90	3.10	2.73	2.57	2.51	2.33	2.30	2.26
Aluminium (f)	.060	.050	.060	.050	.050	.050	.060	.050	.040
Aluminium (t)	.060	.060	.100	.080	.080	.080	.080	.070	.060

CHAPTER 6

THE EFFECT OF LIMING ON BENTHIC MACRO-INVERTEBRATE POPULATIONS

6.1 Introduction

Upland rivers with their characteristic substratum of boulders, cobbles and pebbles normally support a rich insect fauna, with Ephemeroptera, Plecoptera, Diptera and Trichoptera in prominence. Oligochaeta, Coleoptera and Mollusca are also often present, representing a diverse benthic invertebrate community (Hynes, 1960).

Acidification has been shown to have detrimental effects upon benthic invertebrate taxon richness (Sutcliffe and Carrick, 1973; Otto and Svensson, 1983; Fjellheim and Raddum, 1990; Simpson *et al.*, 1985), with taxa such as Ephemeroptera, Molluscs and some Trichoptera being reduced or totally absent.

Sutcliffe and Hildrew (1989) proposed three hypotheses for this. First that the chemical conditions in acid waters are intolerable to some taxa, or have sublethal physiological effects, and might thus cause active avoidance. The detrimental mechanisms to invertebrates correspond with those found for fish, i.e. impaired salt balance, reduced gas exchange, impaired calcium metabolism, element deficiencies and toxic effect of metals such as aluminium (Vangenechten *et al.*, 1989). Second, their food supply is affected by acid waters, viz. reduced primary production and inhibition of microbial degradation of litter. Third, the absence of fish in acidified water may affect communities by removing predation pressure (Bendell and McNicol, 1987).

While it is probable that all three hypotheses contribute to an explanation of invertebrate community structure in acidic streams, the first is regarded of primary importance as indicated by the large amount of literature linking invertebrate diversity with chemical water quality (Stoner *et al.*, 1984; Burton and Allan, 1986; Ormerod *et al.*, 1987; Weatherley *et al.*, 1990; Rutt *et al.*, 1990; Fjellheim & Raddum, 1990).

The aim of this chapter is to evaluate the changes in invertebrate faunal assemblages in the main river following liming. In the context of this study the possible compounding effects from the regulating reservoir and its hypolimnial discharge are also discussed.

6.2 Materials and Methods

6.2.1 Sampling strategy

To establish the effects of liming, the main river was sampled at a number of sites below the impoundment before and after liming. Tributaries within the upper river were sampled as reference streams and also to establish sources of possible recolonising taxa.

Autumn sampling was carried out since this was the first season available to this study and small instars present at this time may be particularly affected by acidification (Allard & Moreau, 1986). Spring sampling was also undertaken since the effects of extreme winter water quality may be most apparent at this time (Wade *et al.*, 1989).

6.2.2 Sampling sites

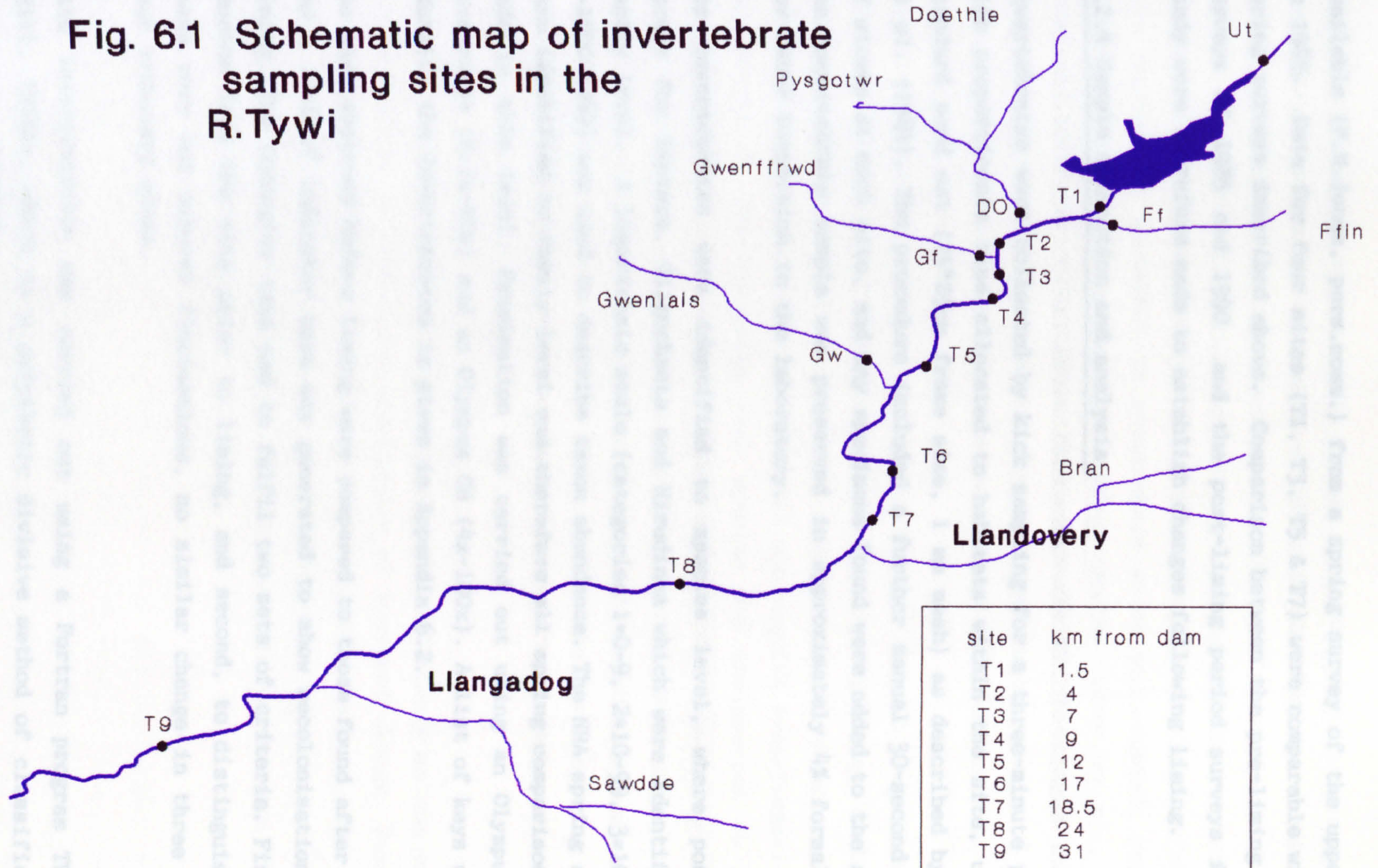
7 main river sites (T1-T7; Fig. 6.1) were sampled within the anticipated impacted area, established from personal communications with NRA personnel, with a further 2 sites (T8 & T9) in lower reaches of the main river. Reference sites were sampled on the tributaries Gwenlais (GW), Gwenffrwd (GF) and Ffin (FF). The River Doethie (DO) was chosen as an acidic reference tributary, with a further site above the reservoir (UT) providing an additional acidic reference for spring samples (see below).

6.2.3 Sampling season

The first set of samples collected during the study period were in the autumn of 1990, prior to the first liming exercise in February 1991. Subsequent surveys were carried out after liming in the autumn of 1991 and 1992. Exact dates of sampling and National Grid References for site location are given in Appendix 6.1.

Spring samples were collected in 1991 and 1992. Since the liming of Llyn Brianne took place in February 1991 both of the above surveys were deemed to be after liming. The spring sampling of 1991 was undertaken some 40-50 days after liming, which could have allowed some recolonisation by some taxa. The NRA's river quality survey in spring 1990, however, included some of the sites in this study (T1, T3, T5, T7, DO, GF, GW) and an additional site UT (River Tywi upstream of Llyn Brianne). Data were also

Fig. 6.1 Schematic map of invertebrate sampling sites in the R.Tywi



site	km from dam
T1	1.5
T2	4
T3	7
T4	9
T5	12
T6	17
T7	18.5
T8	24
T9	31

available (F.H.Jones, pers.comm.) from a spring survey of the upper Tywi in 1985. Data for four sites (T1, T3, T5 & T7) were comparable with the spring surveys described above. Comparison between the pre-liming period surveys of 1985 and 1990 and the post-liming period surveys in this study were therefore made to establish changes following liming.

6.2.4 Sample collection and analysis

Invertebrates were collected by kick sampling for a three-minute period, with proportionate time allocated to habitats within the site, using a standard pond net (25*25cm frame size, 1 mm mesh) as described by Furse *et al.* (1981). The procedure included a further manual 30-second search of stones at each site, and any specimens found were added to the sample. The invertebrate sample was preserved in approximately 4% formaldehyde for later inspection in the laboratory.

The invertebrates were identified to species level, where possible, except for Diptera, Oligochaeta and Hirudinea which were identified to family level. A logarithmic scale (categories 1=0-9, 2=10-99, 3=100-999, 4=1000-9999) was used to describe taxon abundance. The NRA spring samples were identified to family level and therefore all spring comparisons were made to this level. Examination was carried out using an Olympus SZ40 microscope (6.7x-40x) and an Olympus CH (4x-100x). A list of keys used to identify the invertebrates is given in Appendix 6.2.

The taxa observed before liming were compared to those found after liming and a list of indicator taxa was generated to show recolonisation after liming. The indicator taxa had to fulfil two sets of criteria. First, an absence from the site prior to liming, and second, to distinguish that these were not natural fluctuations, no similar change in three of the four tributary sites.

Data interpretation was carried out using a Fortran program TWINSPAN (Hill, 1979b), which is a polythetic divisive method of classification. Essentially TWINSPAN is a program that produces an ordered two-way table by constructing a classification of the samples, and then using this to construct species classification on the basis of presence in site groups. A species indicator key is produced based on some species with high affinity for particular groups.

TWINSPAN was used to classify sites from the autumn surveys into hierarchical groups on the basis of their fauna. Species abundance was incorporated using the abundance categories as 'pseudospecies' (Hill, 1979b). Since all the sites were not surveyed by the NRA in spring 1990 the spring dataset available for TWINSPAN analysis was smaller and so a more appropriate procedure to group these sites was adopted, using the TWINSPAN acid indicator key for spring samples (Rutt *et al.*, 1990); developed from 323 upland sites throughout the UK. This acid indicator key places sites into one of four groups with groups 3 and 4 corresponding to sites predicted to be affected by acidification.

Data interpretation was also carried out using a Fortran program called DECORANA (Hill, 1979a) utilising detrended correspondence analysis to ordinate sites according to the fauna composition. Abundance categories were logarithmic, therefore the data did not require transformation, but the option of down-weighting rare species was implemented to prevent distortion of the analysis. DECORANA reduces the complexity of fauna variation between sites, by compiling sites into an order with respect to their fauna composition. Sites with similar invertebrate assemblages occur closely together. Site axis scores are produced to enable ordination and axis scores can also be used to determine relationships with environmental variables.

For both TWINSPAN and DECORANA, analysis was undertaken on the combined autumn samples to enable the examination of temporal change. Similarly, the spring samples from each year were analysed together. The aim of this ordination analysis was to distinguish between sites that demonstrated a marked difference in fauna composition between the pre- and post-liming periods.

6.3 Results

6.3.1 Autumn macro-invertebrates

The list of individual taxa found during the study period is shown in Appendix 6.3. Prior to liming the main river sites showed a paucity of taxa while the lower sites (T8 & T9) and the tributaries displayed a rich insect fauna. After liming the main river sites displayed a more diverse invertebrate composition, and this is shown by the list of taxa

indicating recolonisation of the main river due to liming (Table 6.1).

The most marked changes were observed with the Ephemeroptera *Baetis rhodani*, *Rhithrogena semicolorata* and *Ecdyonurus* spp. Prior to liming *B.rhodani* was absent from T3 and T5 and found in very small numbers at T1, T2, T4, T6 and T7. *R.semicolorata* and *Ecdyonurus* spp. were not detected at any of these sites in 1990. After liming the autumn samples of 1991 and 1992 were dominated by these species which typically represented over 50% of the benthic samples numerically.

Table 6.1. Taxa recolonising the main river after liming. (autumn collection)

Species	Site T1	T2	T3	T4	T5	T6	T7	T8	T9	Trib.
Glossosoma spp.			C	C		C	C	C	C	
B.scambus	C	C				C	C			C(GW)
B.rhodani	*	*	C	*	C	*	*			
R.semicolorata		C	C	C	C	C	C			
Ecdyonurus spp.		C	C	C	C	C	C	C		
A.fluviatilis		C	C			C	C			
P.microcephala	C	C	C	C	C		C			C(DO)
S.personatum		C		C			C	C	C	C(DO)
P.meyeri	C	C	C	C	C	C		C		C(FF)

C denotes recolonisation using the following criteria for each taxon:
(i) absence at that site pre-liming.
and (ii) no similar change in three of the four tributary sites.

* denotes an increase in abundance in post-liming samples from scarce (1-9 individuals) to abundant (>100).

Also seen to recolonise the main-river were the Plecoptera *Perlodes microcephala* and *Protonemura meyeri* and the Trichoptera *Glossosoma* spp. and *Seriscotoma personatum*. Other taxa such as *Isoperla grammatica*, *Hydropsyche siltalai*, *Hydropsyche pellicudula* and *Lepidostoma hirtum* were detected in the main river for the first time after liming, but showed a similar trend in the tributaries, suggesting that these changes may have been natural fluctuations and not due to liming.

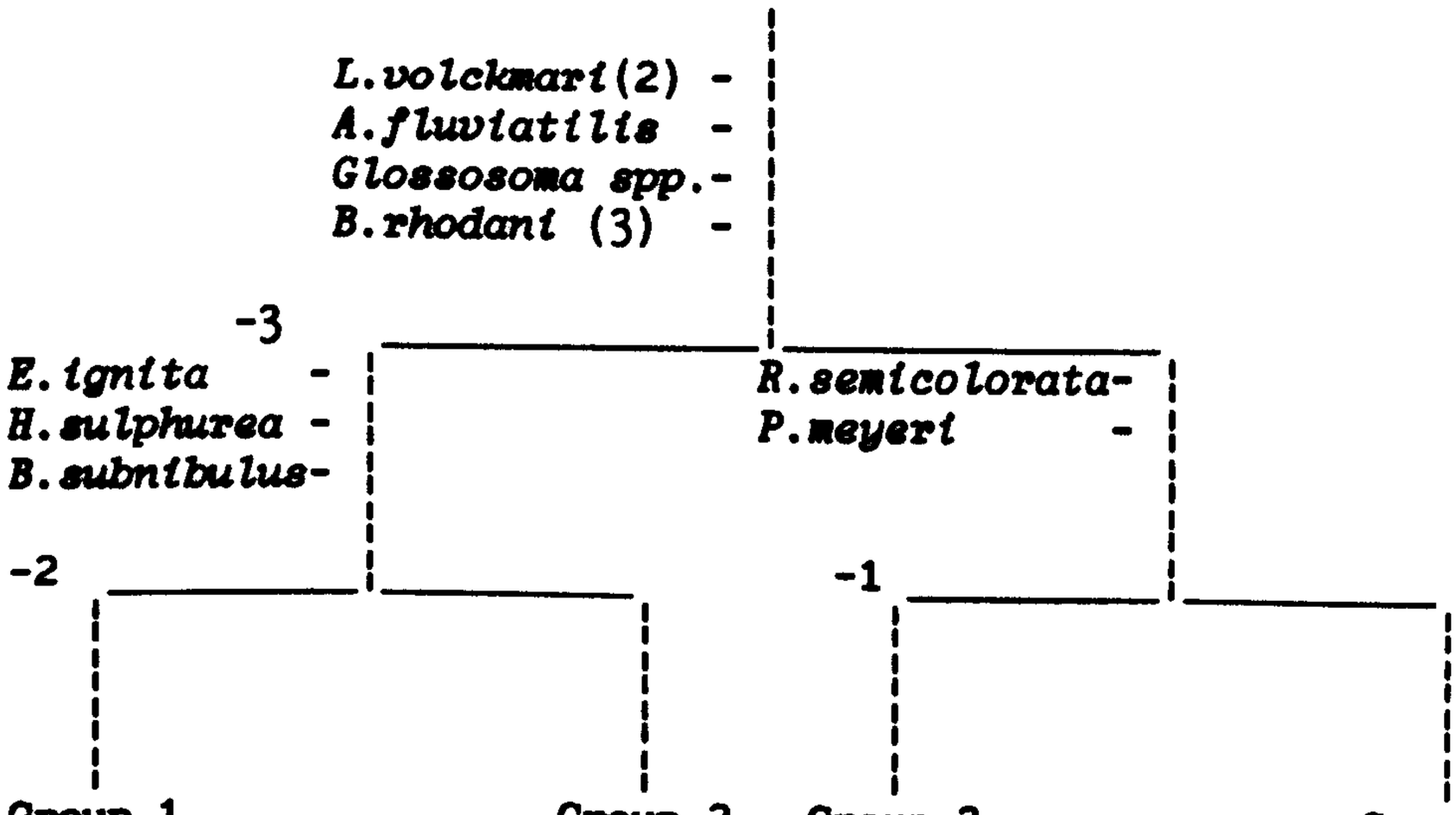
The mollusc *Ancylus fluviatilis* was absent from sites T1-T7 prior to liming but was present in the tributaries FF, GF, and GW. After liming

the mollusc was found at four of the the main river sites, during the autumn collection.

The only taxon to show a decrease after liming was the Chironomidae which were found at lower abundance.

The autumn TWINSPAN analysis was stopped at the second level to give four hierarchical groups (Fig. 6.2).

Fig. 6.2. TWINSPAN groupings - autumn results. Indicator key with species and pseudospecies indicating divisions. The scores given at each branch are the maximum allowed for continuation.



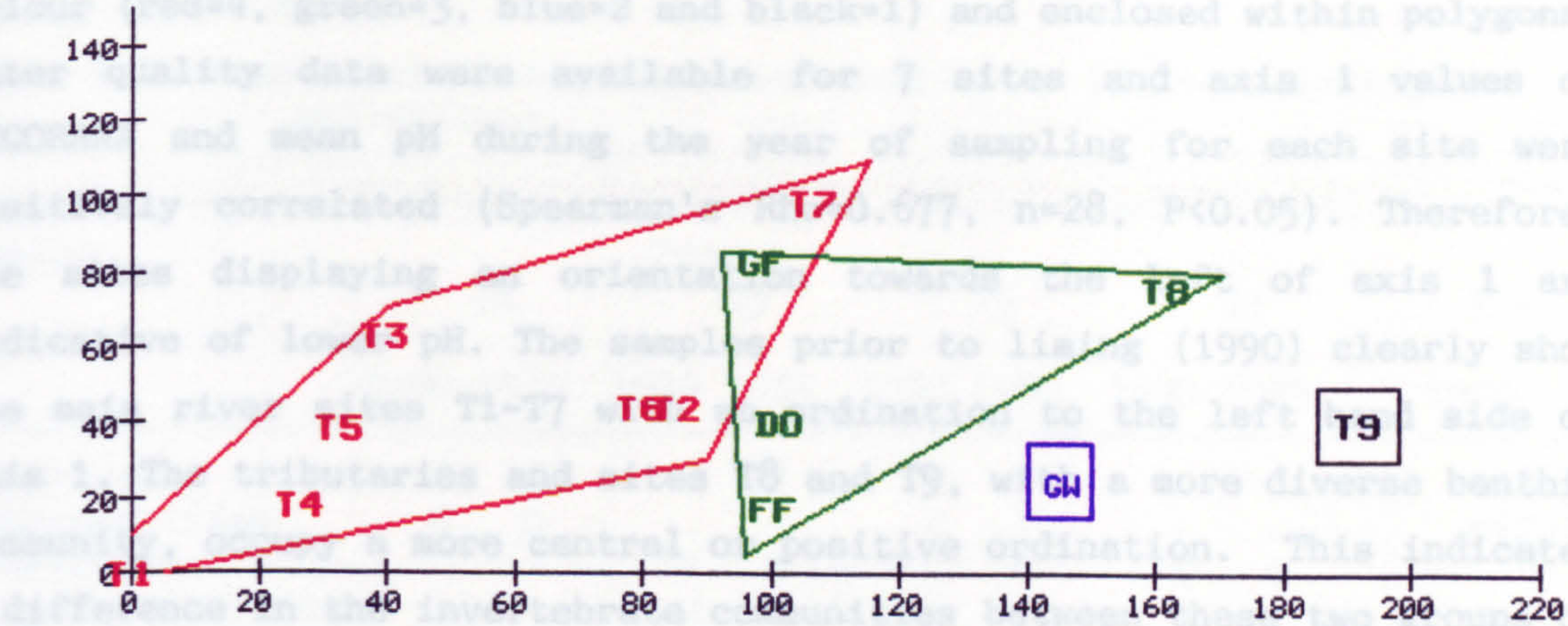
Prior to liming			
1990- T9	GW	GF,DO,FF,T8	T1,T2,T3,T4,T5 T6,T7
After liming			
1991- T9,T8	GW,T6,T7	GF,DO,FF,T1 T2,T3,T4,T5	
1992- T9	GW,T6,T7,FF T3,T8,	GF,DO,T1,T2 T4,T5	

In 1990 (pre-liming), sites T1-T7 were in group 4 while the tributaries and T8 were in the higher groups of 3 or 2. T9 was the only site in group 1.

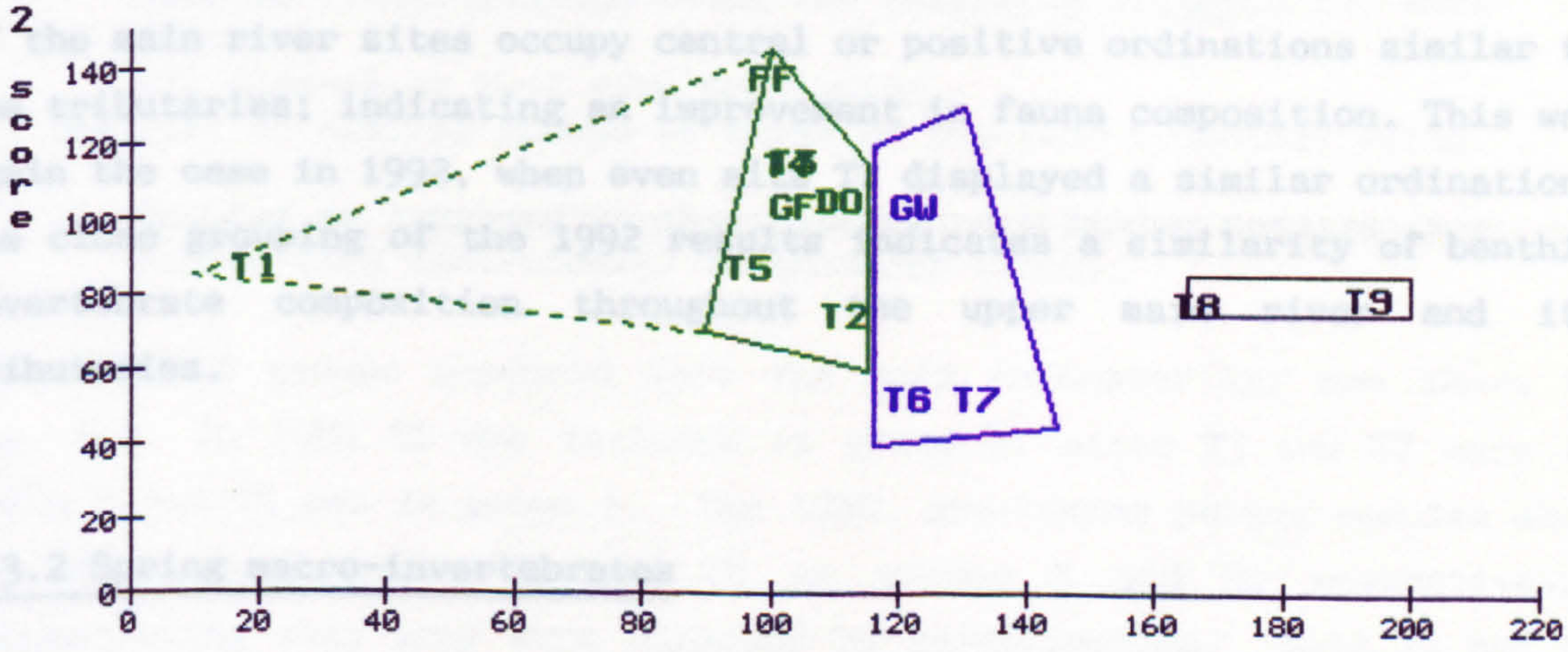
In 1991 (post-liming) all sites previously in group 4 had moved to either group 3 or 2, though site T1 was still borderline group 4. All tributaries remained in the same group as in 1990. Site T9 was again in group 1 but was joined by T8. The groupings in 1992 were similar to 1991, but site T3 moved to group 2, FF moved from group 3 to 2 and site

Fig. 6.3 Autumn DECORANA & TWINSPAN results
TWINSPAN groups enclosed in polygons

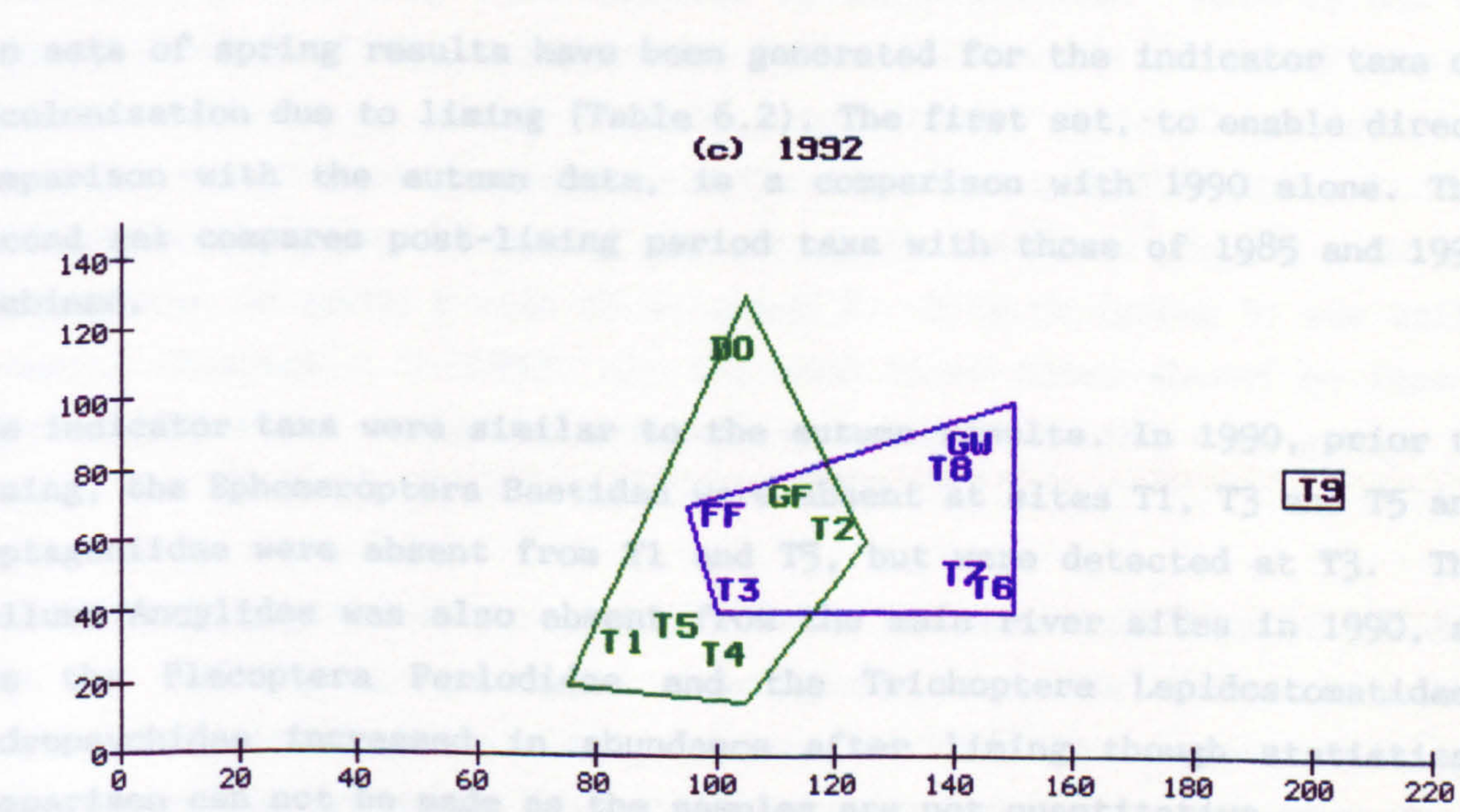
(a) 1990



(b) 1991



(c) 1992



Axis 1 score

T8 moved from group 1 to 2.

The DECORANA plots of axes 1 and 2 for the three autumn sampling periods are shown in Figs. 6.3a-c, with the TWINSPAN groupings superimposed in colour (red=4, green=3, blue=2 and black=1) and enclosed within polygons. Water quality data were available for 7 sites and axis 1 values of DECORANA and mean pH during the year of sampling for each site were positively correlated (Spearman's Rho=0.677, n=28, $P<0.05$). Therefore, the sites displaying an orientation towards the left of axis 1 are indicative of lower pH. The samples prior to liming (1990) clearly show the main river sites T1-T7 with an ordination to the left hand side of axis 1. The tributaries and sites T8 and T9, with a more diverse benthic community, occupy a more central or positive ordination. This indicates a difference in the invertebrate communities between these two groups of sites attributed to pH.

After liming the results of 1991 show that, with the exception of T1, all of the main river sites occupy central or positive ordinations similar to the tributaries; indicating an improvement in fauna composition. This was again the case in 1992, when even site T1 displayed a similar ordination. The close grouping of the 1992 results indicates a similarity of benthic invertebrate composition throughout the upper main river and its tributaries.

6.3.2 Spring macro-invertebrates

Two sets of spring results have been generated for the indicator taxa of recolonisation due to liming (Table 6.2). The first set, to enable direct comparison with the autumn data, is a comparison with 1990 alone. The second set compares post-liming period taxa with those of 1985 and 1990 combined.

The indicator taxa were similar to the autumn results. In 1990, prior to liming, the Ephemeroptera Baetidae were absent at sites T1, T3 and T5 and Heptageniidae were absent from T1 and T5, but were detected at T3. The mollusc Ancylidae was also absent from the main river sites in 1990, as was the Plecoptera Perlodidae and the Trichoptera Lepidostomatidae. Hydropsychidae increased in abundance after liming though statistical comparison can not be made as the samples are not quantitative.

The comparison with both 1985 and 1990 combined displayed colonisation for similar families, though Baetidae were found at some sites (T3, T5) in 1985 but not in 1990. Heptageniidae were found at T5 in 1985 but not 1990. Both these families demonstrated marked increase in abundance after liming (Table 6.2).

Table 6.2 Taxa recolonising the main river after liming. (spring collection).

family	Site									
	T1		T3		T5		T7		Trib	
	1990	1985	1990	1985	1990	1985	1990	1985	1990	
Baetidae	C	C	C	*	C	*			C(DO)	
Heptageniidae	C	C			C	*			C(DO)	
Ancylidae			C	C	C	C	C	C		
Lepidostomatidae			C		C	C	C		C(GW)	
Perlodidae	C									
Gyrinidae			C		C	C	C		C(DO)	

C denotes recolonisation using the following criteria for each taxon:
(i) absence at that site pre-liming.
and (ii) no similar change in three of the four tributary sites.

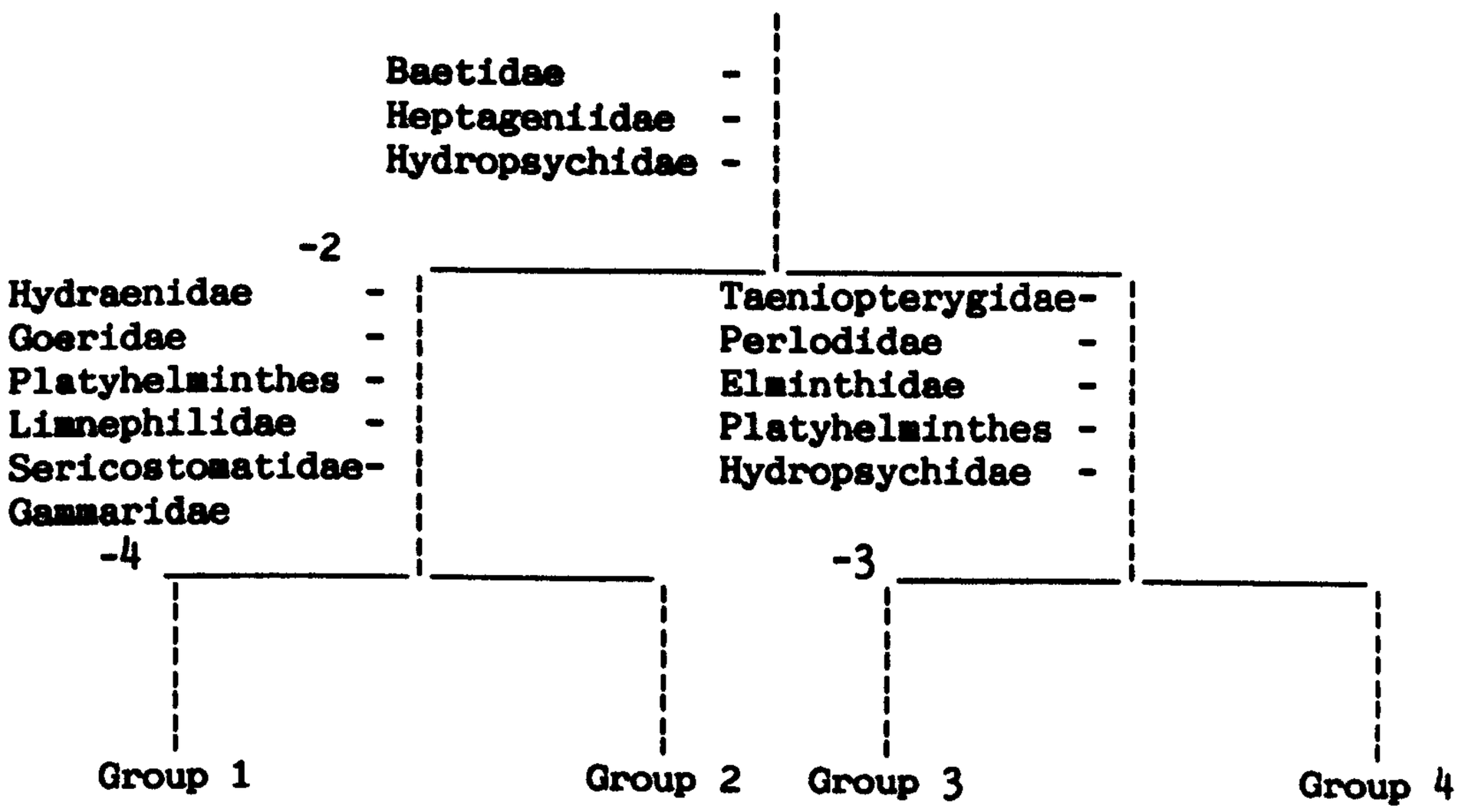
* denotes an increase in abundance in post-liming samples from scarce (1-9 individuals) to abundant (>100).

The TWINSPAN groups produced from the acid indicator key are shown in Fig. 6.4. In 1985 T1 was included in group 4, sites T3 and T7 were in group 2 and T5 was in group 1. The 1990, pre-liming period results show the main river sites T1 and T5 in groups 4 and 3, respectively, demonstrating that they were impacted by acidification. Both T3 and T7 were in group 2 of the indicator key. The tributaries GW and GF showed no impact by being in groups 1 and 2, respectively, while DO was impacted (group 3) and UT showed severe acidification (group 4). In 1991, T1 was still impacted (group 4) but T5 now joined T3 and T7 in group 2. GW and GF were both in group 1 with DO in group 2. Site UT (group 4) was still severely impacted. In 1992, all the main river sites showed no impact from acidification while the tributaries remained in the same groups as in 1991 and UT remained in group 4.

The DECORANA plots for the spring results are shown in Figs. 6.5(a-d), with the TWINSPAN groups again superimposed with the same colours. A positive correlation between axis 1 scores of DECORANA and mean pH in the year of sampling was again evident (Spearman's Rho=0.422, n=21, $P<0.05$).

The DECORANA plots of sites before liming indicate positions to the left or middle of the graph. After liming the ordinations of sites displayed a similar result to the autumn analysis, in that the main river sites progressively moved to converge towards the ordinations displayed by the tributaries.

Fig. 6.4. Acid indicator key developed by Rutt *et al.* (1990) for spring macroinvertebrates- family level. The scores given at each branch are the maximum allowed for continuation. Groups 3 & 4 show impact by acidification.



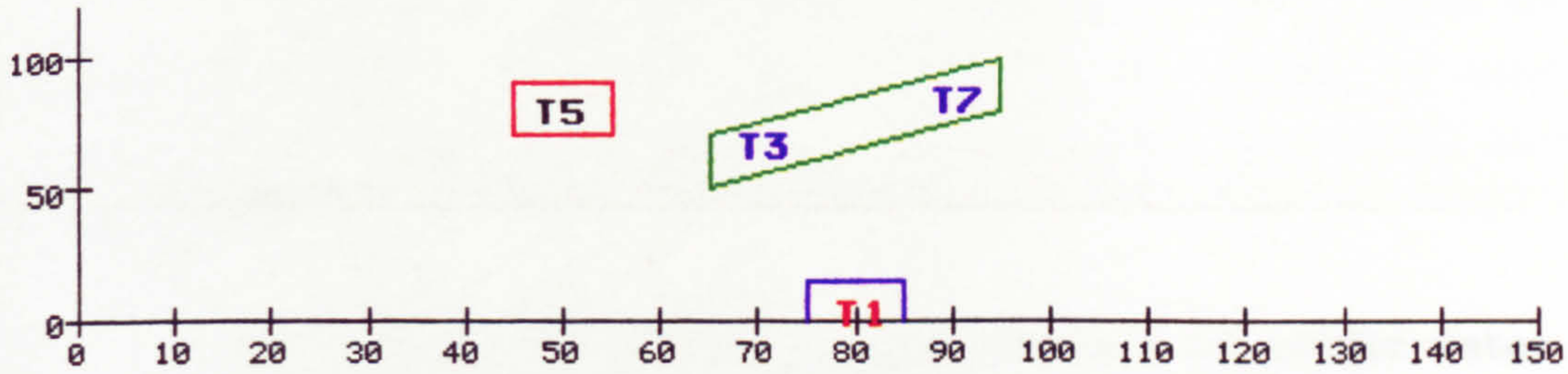
Prior to liming			
1985- T5	T3,T7		T1
1990- GW	GF,T3,T7	DO,T5	T1,UT
After liming			
1991- GW,GF	T3,T7,T5,DO		T1,UT
1992- GW,GF	T3,T7,T5,DO T1		UT

The tributaries GW and GF showed very little movement between years indicating a similar fauna from year to year. The reference tributary DO displayed a change in ordination from 1990 to 1991 and 1992 by moving in a positive direction. Site UT displayed similarity between 1990 and 1991 but did show an ordination slightly to the right in 1992. (This was due to the inclusion of one extra taxon in 1992, Rhyacophilidae, which had a high taxon score and since there were a small number of taxa at the site, the site average was increased giving the positive change in ordination).

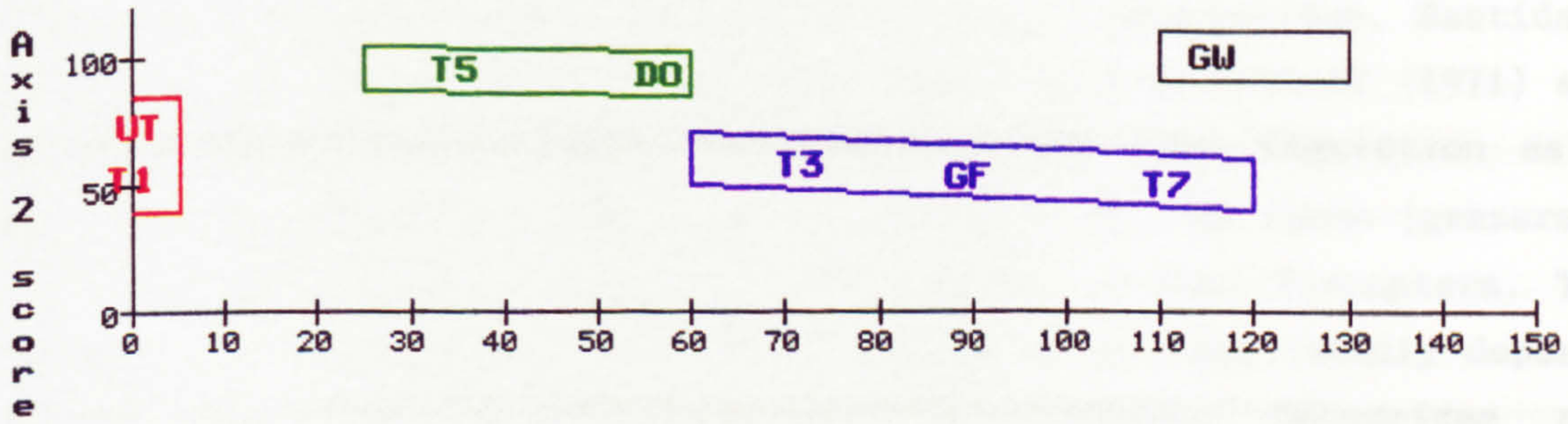
The autumn and spring analyses combined suggest improvements from the pre- to post-liming period in the invertebrate communities of the main

Fig. 6.5 Spring DECORANA & TWINSPAN results
TWINSPAN groups enclosed in polygons

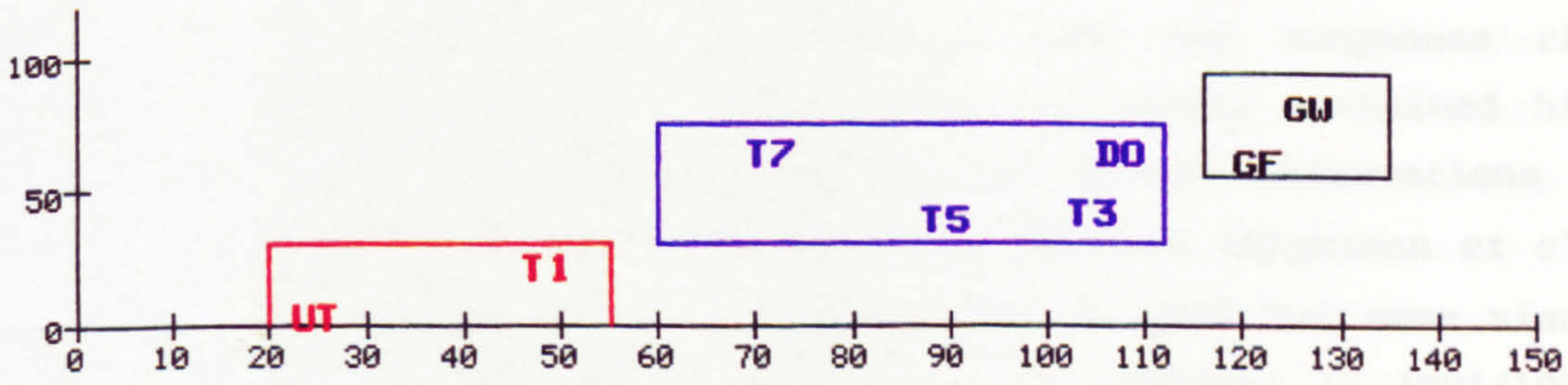
(a) 1985



(b) 1990



(c) 1991



(d) 1992

Axis 1 score

river sites, since the paucity of taxa displayed prior to liming is replaced with an increased diversity, similar to the tributary references.

6.4 Discussion

6.4.1 Regulation

Prior to discussing the effects of the amelioration of acidic water on the invertebrate community it is necessary to consider a further complicating factor to this study, i.e. regulation. The effect of impoundment was reviewed by Brooker (1981), and Armitage *et al.* (1987) demonstrated that fauna assemblages can be modified by regulation with a reduction in the abundance of some taxa e.g. Heptageniidae, Baetidae, Perlodidae, Chloroperlidae and Rhyacophilidae. Both Hilsenhoff (1971) and Armitage *et al.* (1987) give increased siltation from regulation as a major explanation for the reduction in abundance of some fauna (grazers), with a constant flow regime being detrimental to some Plecoptera. The reduced flows of regulation therefore favour other taxa (mainly deposit feeders) such as Polycentropodidae, Chironomidae, Sphaeridae and Oligochaeta.

Inverarity *et al.* (1983) described the effect of sedimentation with a decrease in fauna due to the presence of iron and manganese rich deposits. The water from Llyn Brianne prior to liming contained high levels of both these metals (Chapter 5) and visual observations of sedimentation were noted for c.3Km below Llyn Brianne (Wightman *et al.*, 1990). These were consistent with observations in 1990 but some visual evidence of 'cleaner' gravels in this area were apparent in 1991/1992. This was probably as a result of lower concentrations of these metals after liming (Chapter 5).

Temperature effects from the hypolimnetic draw off might also have an affect on some aquatic insects (Lehmkuhl, 1972; Elliott, 1980) in disrupting egg development, hatching and subsequent emergence.

The influence of regulation will diminish with increasing distance from the reservoir (Crisp, 1977) and with the addition of flow from unregulated tributaries.

No change occurred in the operating regime from the reservoir during the study period and therefore for the purpose of the discussion the effects of regulation, apart from possible reduction in sedimentation in the first few kilometres, are assumed constant in all years. Many of the sites that showed improvement were downstream of major tributaries which would have depressed the effects of regulation. The changes in benthic communities observed after liming were therefore attributed to the liming operation.

6.4.2 Water quality

Early work describing the invertebrate fauna of the River Tywi was reported by Jones (1951). He surveyed the whole of the main river during the summer of 1949, with uppermost sites corresponding to T1 and T6 in this study. He noted that at these sites invertebrates such as the Ephemeroptera *Ecdyonurus venosus*, *Baetis rhodani* and *Ephemerella spp.* were common, as were the mollusc *Ancylus fluviatilis* and the Trichoptera, *Seriscotoma personatum* and *Glossosoma boltoni*. The surveys in 1985 and 1990 clearly show reduced invertebrate diversity compared with the above study, but the building of the reservoir between these surveys further complicates issues.

The influence of pH on faunal composition has been described elsewhere (Rutt *et al.*, 1990; Weatherley *et al.* ,1990; Kullberg, 1992) and this was shown here by the relationship between the axis 1 of DECORANA and pH. The low benthic diversity in the main river sites T1-T7 before liming is consistent with the low pH (5-6) and high aluminium concentrations ($0.1\text{-}0.3\ mgl^{-1}$), experienced at the time (Chapter 5). Improvement of water quality downstream from T1 to T7 before liming, as a result of the ameliorating effect of tributary inputs (Chapter 5), was reflected in some increase in taxa with progressive downstream distance. However, large acid episodes or repeated events would prevent permanent colonisation by extremely acid-sensitive species at the lower sites (McCahon *et al.*, 1989).

The distinct absence or infrequency of Ephemeroptera is consistent with their known acid sensitivity (e.g. Fjellheim and Raddum, 1990; Weatherley *et al.*, 1990). McCahon *et al.* (1989) noted *R.semicolorata* and *B.rhodani* as being most sensitive with the latter being especially sensitive to

aluminium at low pH (0.27 mgl^{-1} filt., pH 4.87 : McCahon & Pascoe, 1989). These species were common in the tributaries during the pre-liming period survey but absent from the main river.

6.4.3 TWINSPAN and DECORANA

The autumn analysis displayed a clear trend of improvement within the main river for both the DECORANA and TWINSPAN analysis. Both analyses were observed to demonstrate acidification, due to the relationship of the DECORANA scores with pH and the known acid sensitivity of the indicator invertebrates chosen by TWINSPAN (Fjellheim & Raddum, 1990; Rutt *et al.* 1990).

The spring DECORANA analysis also displayed an improvement from the pre- to post-liming period but some inconsistencies were observed with the TWINSPAN results. In spring 1985 sites T3, T5 and T7 were not considered to be impacted by acidification, as were T3 and T7 in 1990. This arose because the presence of two of the three indicator taxa (Baetidae, Heptageniidae, Hydropsychidae) allowed allocation to the 'non-impacted' side. However, only small numbers (1-9) of these taxa were observed and were probably explained by the nature of the upper catchment prior to liming. These acid sensitive families were common in the un-acidified tributaries (FF, GF, GW) and thus due to the drifting of invertebrates in fresh water (e.g. Waters, 1972) these tributaries would constantly introduce the invertebrates into the main-river. The acidic water in the main river would then, however, prevent permanent colonisation (McCahon *et al.* 1989). Ephemeroptera display particular tendency to drift (e.g. Elliott, 1967) and Hemsworth (1979) demonstrated the seasonal change in drift with peaks in April and July. The spring surveys therefore coincide with a period of high drift, which increased the chance of intercepting temporary colonisers (particularly the 1985 survey, undertaken in early May). This hypothesis of 'patchy' temporary colonisation is supported by the inconsistency of observing taxa at all the main river sites.

The Heptageniidae at T3 and T7 present in the spring of 1990 were not observed in the 1990 autumn samples. Some taxa would have been of younger instars in the autumn and possibly more difficult to detect. Allard and Moreau (1986) stated that early instars, present in the autumn, were very sensitive to pH. However, Hall (1990) could show no evidence for a more

detrimental effect of pH between spring and autumn. This difference in observations is again probably explained by the temporary colonisation during the spring.

Other acid sensitive species such as *I.grammatica*, *H.siltalai*, *H.pellucidula* and *L.hirtum*, which were absent from the upper main river sites before liming but present after liming, did not satisfy the criteria for recolonisation since they showed a similar pattern in more than one of the tributaries. Although invertebrate assemblages in streams show remarkable annual constancy (Weatherley and Ormerod, 1990) there may be variability with some individual species. 1990 was a particularly dry year and followed a drought year in 1989, which has been shown to reduce species abundance (Cowx *et al.*, 1984; McElravy *et al.*, 1989). Even though the recolonisation criteria were not fulfilled, the hypothesis that liming may have enabled the above four acid sensitive species to inhabit the main river cannot be rejected, since the apparent natural increase in abundance in the tributaries may not have been reflected in the main river with un-limed conditions.

Constancy in annual faunal assemblages was also shown by most of the tributaries. The exception to this was the reference acidic tributary the River Doethie. The paralleled improvement with the main river sites from the spring DECORANA and TWINSPAN results invokes a question as to whether the main river improvements were due to liming or were a natural event. The Doethie is prone to storm-induced episodic acidification (Chapter 5) resulting in pH c.4.5 and filterable aluminium of >0.2 mgl^{-1}. Invertebrates are known to tolerate short episodes (Sutcliffe and Hildrew, 1989; McCahon *et al.*, 1989) and Weatherley and Ormerod (1991) discussed the dynamic nature of faunal assemblages in being able to maintain diversity, though large or prolonged events may be detrimental to acid sensitive invertebrates in the River Doethie. 1990 may therefore have been a particularly poor year and the River Doethie should be looked upon as having semi-acidic conditions. It should also be noted that the improvement in the main river in 1991 would have led to increased aerial adults in the vicinity of the River Doethie. This could possibly have led to increased ovideposition in the River Doethie and therefore the faunal assemblage in this river in 1992. This could possibly affect the contribution of the River Doethie as a reference river to this study in 1992.

Even if any improvements common to both the River Doethie and the main river were ignored, the main river still showed further recolonisation due to liming. *Glossosoma spp.*, *B.scambus* and *A.fluviatilis* recolonised the main river but were not found in the River Doethie. Also the spring 1992 survey showed Baetidae to be absent from the River Doethie whereas they were abundant in the main river.

Further evidence to support the hypothesis that liming has enabled recolonisation was provided by the quality of the fauna in the major afferent stream to the reservoir (site UT). Evidence of severe acidification was observed at this site (TWINSPAN group 4) in each spring, thus the improvement at the site immediately below the reservoir (T1) from group 4 to 2 can be attributed to liming.

Chironomid larvae were different to other taxa in that they showed a decrease in abundance after liming. Olem (1991), however, describes how acid-tolerant chironomids may be replaced by non acid-tolerant species over 1 to 2 years, and so further changes in the Chironomid numbers may take place in the future.

So far most of the changes in faunal assemblages after liming have been discussed in relation to the changes in water quality. However, changes relating to the other two hypothesis of Sutcliffe & Hildrew (1989), food and predation, also need to be noted. Olem (1991) discussed how the increases in detrivores such as Ephemeroptera could be due to the increased microbial activity after liming. The increases in fish observed in this study after liming (Chapter 7) could also have possibly affected invertebrate numbers or composition (Bendell & Nicol, 1987; Shortelle & Colburn, 1987). Weatherley (1988) also indicated the possible effects of interspecific interactions between different invertebrate taxa on the change in faunal assemblages, though little is currently known concerning this.

6.4.1 Recolonisation mechanisms

Following liming the recolonisation by many of the acid-sensitive species was rapid and drift from the tributaries is suggested as the predominant mechanism for this, although upstream movements and aerial recolonisation (Williams and Hynes, 1976) could all have contributed. Drift was considered as the main reason for the rapid recolonisation of the River

Audna after liming (Fjellheim and Raddum, 1992). Sutcliffe and Carrick (1973) suggested that ovidepositing female *B.rhodani* avoided acidic streams and that a pH of around 6 units was needed at egg laying (Bell, 1971). Since liming took place in February, water quality would have been adequate for aerial recolonisation during the following spring/summer.

The slower recolonisation at site T1 might be explained by the greater impact of regulation in the reach immediately downstream of the dam, and the absence of a tributary above this site would not have allowed the level of recolonisation afforded to the other main river sites.

Ancylus fluviatilis colonised the main river within some 12 months after liming. It was found at all sites downstream from the confluence of the uppermost tributary containing the mollusc (FFin), indicating drift contribution. However, it was not found upstream of this confluence as this would necessitate upstream movements, which is presumably a slow process. This slow upstream movement was supported by Elliott (1971) who did not collect any *A.fluviatilis* in upstream traps in 72 hours sampling, even though they were common in the benthos. Molluscs are hindered in shell formation by acidic water and Sutcliffe and Carrick (1973) observed an absence of *A.fluviatilis* below pH 5.7. Maitland (1965) indicated that *A.fluviatilis* required over 2mgl of calcium. Consequently the use of this species as a useful indicator of sustained improvements in the Tywi should be considered.

The imminence of liming to the start of the study only allowed one years pre-liming data to describe the fauna of the main river prior to liming. In assessing the changes following liming it was therefore necessary to assume that these were representative of pre-liming conditions. One sample was taken at each site during each season, therefore assessing the variation due to sampling, to improve statistical confidence in data interpretation, was not possible.

To summarise, water quality from the reservoir prior to liming did not allow permanent colonisation by acid-sensitive invertebrates within the main river for c.17 km below the impoundment. The neutralisation of acidity by liming permitted recolonisation of many taxa, and drift from the unacidified tributaries is proposed as the primary mechanism of colonisation. Further changes in the invertebrate fauna may yet occur before a stable faunal assemblage is finally established.

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Appendix 6.1 Details of location and dates for the sampling surveys.

Site	NGR	Date sampled						
		1985	1990		1991		1992	
T1	SN785472	7/5	5/4	26/9	20/4	19/10	21/4	14/10
T2	SN772460	ns	ns	26/9	20/4	19/10	21/4	14/10
T3	SN768447	7/5	5/4	26/9	20/4	20/10	21/4	14/10
T4	SN779434	ns	ns	10/10	20/4	20/10	21/4	14/10
T5	SN765399	7/5	4/4	27/9	20/4	20/10	21/4	14/10
T6	SN762357	ns	ns	10/10	28/4	24/10	21/4	20/10
T7	SN762348	7/5	4/4	10/10	28/4	24/10	21/4	20/10
T8	SN737319	ns	ns	10/10	28/4	24/10	21/4	20/10
T9	SN687265	ns	ns	27/9	28/4	24/10	23/4	20/10
FF	SN787472	ns	ns	26/9	20/4	19/10	21/4	14/10
DO	SN777471	ns	3/5	26/9	20/4	19/10	21/4	14/10
GF	SN763452	ns	5/4	26/9	20/4	20/10	21/4	14/10
GW	SN759389	ns	4/4	27/9	20/4	20/10	21/4	14/10

Appendix 6.2

A list of invertebrate keys used in this study.

Disney, R.H.L. (1975). A key to British Dixidae. FBA key 31. 78 pp.

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Friday, L.E. (1988). A key to the adults of British Water Beetles. Field Studies, 7, 1-151.

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Macan, T.T. (1977). A key to the British fresh- and brackish-water Gastropods. FBA key 13. 44 pp.

Reynoldson, T.B. (1978). A key to British species of freshwater Triclads. FBA key 23. 31 pp.

Unwin, D.M. (1984). A key to the families of British Coleoptera (and strepsiptera). Field Studies, 6, 149-197.

Wallace, I.D., Wallace, B. & Philipson, G.N. (1990). Case-bearing caddis larvae of Britain and Ireland. FBA key 51. 237 pp.

Appendix 6.3. Taxa observed during the study period. Abundance in the configuration Aut 90.Spr 91.Aut 91.Spr 92.Aut 92. with categories 1=1-9, 2=10-99, 3=100-999, 4=1000+

	T1	T2	T3	T4	T5	T6	T7
Ephemeroptera							
Baetis buceratus Eaton							
Baetis muticus (L.)	-.-.1.-.-			-.-.1.-.-	-.-.-.-.1	-.-.-.1.-	-.-.-.1.-
Baetis rhodani (Pictet)	1.1.2.2.3	1.1.3.2.3	-.1.2.2.3	1.1.2.3.3	-.1.3.2.3	1.2.3.2.3	1.2.3.2.3
Baetis scambus Eaton	-.-.-.-.1	-.-.-.-.1				-.-.2.-.2	-.-.2.-.2
Baetis vernus Curtis				-.-.-.-.1	-.-.-.-.1	-.-.-.-.1	
Baetis niger (L.)	-.-.-.1.-				-.-.-.2.-	-.-.-.1.-	-.-.-.1.-
Centroptilum luteolum (muller)							-.-.-.1.-
Rhithrogena semicolorata (Curtis)	-.-.-.1.-	-.1.2.2.3	-.2.2.2.3	-.1.2.3.3	-.1.3.3.3	-.2.3.3.3	-.-.3.3.3
Heptagenia sulphurea (Muller)	-.-.-.-.1				-.1.-.-.-	-.1.-.-.-	-.1.-.-.-
Heptagenia lateralis (Curtis)			-.1.-.-.-			-.1.-.-.-	-.1.-.1.-
Ecdyonurus spp.		-.-.2.1.2	-.-.1.2.2	-.1.2.2.2	-.1.2.1.2	-.1.2.2.2	-.1.3.1.2
Caenis spp.							
Paraleptophlebia spp.						-.-.1.-.-	
Siphlonurus spp.							-.-.-.-.1
Ephemera danica Muller						-.-.-.-.1	-.-.-.1.1
Ephemerella ignita (Poda)	-.-.-.-.1			-.-.-.-.1			
Plecoptera							
Brachyptera risi (Morton)		-.1.-.-.-		-.1.-.-.-	-.1.-.1.-	-.-.-.1.-	-.-.-.1.-
Protonemura meyeri (Pictet)	-.-.1.-.1	-.-.2.-.2	-.-.2.-.1	-.-.2.-.2	-.-.3.-.2	-.-.2.1.1	-.-.2.-.-
Protonemura praecox (Morton)		-.-.1.-.-	-.-.1.-.-				-.-.1.-.-
Amphinemura spp.	-.1.1.2.1	-.2.-.2.-	-.1.1.2.-	-.2.2.2.-	-.2.2.2.-	-.1.-.2.-	-.1.-.1.-
Nemoura avicularis Morton				-.-.-.-.1	-.-.1.-.1	-.-.1.-.2	-.-.1.-.1
Nemoura cambrica (Stephens)	-.-.1.-.-					-.-.1.1.-	
Nemoura cinerea (Retzius)	-.1.1.-.-						
Nemoura erratica Claassen	-.1.-.-.-					-.-.1.-.-	-.-.1.-.-
Leutra fusca (L.)	-.-.1.-.2	1.-.1.-.2	1.-.1.-.2	2.-.1.-.2	1.-.1.-.2	1.-.1.-.2	1.-.1.-.2
Leutra hippopus (Kempny)	-.1.-.-.-	-.1.-.-.-	-.-.-.1.-	-.1.1.1.-	-.1.1.1.-	-.1.-.-.-	-.1.-.1.-
Leutra inermis Kempny	-.-.-.2.1	-.2.-.2.-	-.2.-.2.-	-.2.-.2.-	-.2.-.3.-	-.2.-.2.-	-.2.-.2.-
Leutra moselyi Morton	-.-.-.-.1	-.-.-.-.1		1.-.-.-.1	-.-.-.-.1	1.-.-.-.-	
Leutra nigra (Olivier)	-.1.-.-.1		-.-.-.1.-		-.-.-.1.-		1.1.-.1.-
Perlodes microcephala (Pictet)	-.-.-.-.1	-.-.1.-.1	-.-.1.-.1	-.-.1.-.2	-.-.1.-.2	1.-.2.-.2	-.-.2.1.2
Diura bicaudata (L.)				-.-.-.-.1			-.-.1.-.-

Isoperla grammatica (Poda)	-.-.-.2.-	-.1.1.2.1	-.1.1.2.1	-.2.1.2.1	-.1.1.2.-	-.2.1.2.-	1.1.-.1.-
Perla bipunctata Pictet							
Chloroperla torrentium (Pictet)	-.1.-.1.-	-.1.-.1.-	-.2.1.1.-	-.2.-.1.-	1.1.-.1.-	1.1.-.1.-	1.2.-.2.-
Chloroperla tripunctata (Scopoli)	-.1.1.1.-	-.2.1.1.2	1.2.1.2.1	2.2.1.2.1	1.1.1.2.1	1.1.1.2.1	2.2.1.2.2
Trichoptera							
Rhyacophila dorsalis (Curtis)	-.-.1.1.2	1.1.1.1.1	-.-.1.1.2	-.1.1.1.2	1.1.2.1.2	-.1.2.1.2	1.-.2.1.2
Rhyacophila septentrionis McLachlan	-.-.1.-.-			-.-.-.-.1	1.-.1.-.1	-.-.1.-.-	-.-.-.-.1
Glossosoma spp.			-.-.1.1.1	-.1.-.-.1	-.1.-.-.-	-.1.-.1.1	-.-.-.1.1
Agapetus spp.							
Philopotamus montanus (Donovan)							
Plectrocnemia geniculata McLachlan	-.1.-.-.-	-.-.-.1.-			-.-.-.1.-	-.-.-.1.-	-.-.-.1.-
Polycentropus flavomaculatus (Pictet)			-.1.-.-.	-.-.-.1.1	-.-.1.-.1	-.-.-.-.1	
Polycentropus kingi McLachlan	1.-.-.-.-		-.-.-.-.1	1.-.-.-.1			-.-.1.-.-
Holocentropus spp.						-.-.-.-.1	
Hydropsyche angustipennis (Curtis)							-.-.1.-.-
Hydropsyche fulvipes (Curtis)		-.-.1.-.-					
Hydropsyche pellucidula (Curtis)		-.-.-.-.1		-.1.-.-.-	-.-.-.-.1	-.-.-.1.1	1.-.1.-.1
Hydropsyche siltalai Dohler		-.1.1.2.2	-.1.-.1.2	-.2.1.1.-	-.2.1.2.-	-.2.1.2.2	-.-.-.1.1
Brachycentrus subnubilus Curtis							
Crunoecia irrorata (Curtis)							
Lepidostoma hirtum (Fabricius)		-.-.-.1.1	-.1.-.1.-	-.1.1.1.-	-.-.1.1.-	-.-.1.1.1	-.-.1.1.-
Drusus annulatus Stephens		-.1.-.-.-			-.1.-.1.1	-.2.-.2.-	-.1.1.1.-
Ecclisopteryx guttula (Pictet)				-.1.-.-.-		-.-.-.-.1	
Allogamus auricollis (Pictet)							
Halesus spp.	-.1.-.-.-	-.-.-.1.-	-.1.1.1.-	-.-.-.1.-	-.1.-.1.-	-.-.-.1.-	-.1.-.1.1
Micropterna lateralis (Stephens)							
Potamophylax spp.	1.1.1.-.1	-.1.1.-.-	-.1.-.-.2	1.1.-.1.-	1.-.1.1.1	1.-.1.1.2	-.-.1.1.2
Stenophylax spp.	-.-.1.-.-						
Anabolia nervosa (Curtis)						-.-.-.1.-	-.-.-.2.-
Limnephilus spp.							-.-.-.-.1
Goera pilosa (Fabricius)						-.-.1.-.-	
Silo pallipes (Fabricius)		-.-.-.1.1	-.-.1.-.-	-.-.-.-.1	-.-.-.-.1	-.1.-.2.1	
Sericostoma personatum (Spence)		-.-.-.-.1	-.-.-.1.-	-.-.1.1.1	1.-.1.2.2	1.1.1.2.1	-.1.1.1.1
Odontocerum albicorne (Scopoli)							
Athripsodes spp.							
Oecetis spp.	-.-.-.-.1	-.-.1.-.-				1.-.-.1.-	-.-.-.1.-

Coleoptera							
Noterus spp.							
Oreodytes septentrionalis			-.-.-.1.-			-.-.1.1.1	-.1.-.2.-
Oreodytes sanmarkii	-.-.-.-.1	-.-.-.-.1	-.-.-.1.1	1.-.-.1.1	1.1.-.1.1	1.1.1.1.1	-.1.-.2.-
Gyrinidae		-.-.-.-.1	-.1.1.1.-	-.-.1.1.-	-.1.1.1.-	1.-.-.1.1	1.-.1.1.1
Hydraena gracilis						-.-.-.-.1	
Elmis aenea	-.1.-.-.	1.1.1.-.1	-.-.-.1.2	-.-.1.1.1	-.-.1.1.1	1.-.2.2.2	1.-.2.1.1
Esolus parallelepipedus			-.-.-.1.-			-.-.-.1.-	-.-.-.1.-
Limnius volckmari	-.1.-.-.	1.1.1.1.1	1.-.1.1.1	1.1.2.2.1	1.1.2.2.2	1.2.2.2.2	2.2.2.2.2
Oulimnius spp.							
Molluscs							
Ancylus fluviatilis Mull.		-.-.-.1.1	-.-.-.1.1	-.-.-.1.-	-.-.-.1.-	-.-.1.1.1	-.-.-.1.1
Potamopyrgus jenkinsi (Smith)					-.1.-.-.-		-.-.1.-.-
Lymnaea spp.						-.-.-.1.1	-.-.-.1.1
Planorbis spp.							
Sphaerium spp.						-.-.1.-.1	-.-.-.-.1
Other taxa							
Oligochaeta	2.2.2.2.2	2.2.2.1.2	2.2.1.2.2	2.1.2.1.1	1.2.2.2.2	1.2.2.2.2	2.2.2.2.2
Planariidae	-.1.1.2.1	-.1.1.1.1	-.1.-.-.1	1.1.1.-.1	1.1.1.2.1	-.1.2.1.1	1.1.1.1.1
Glossiphoniidae						-.1.-.-.	
Erpobdellidae							
Tipulidae	-.1.2.1.2	1.-.2.-.2	-.-.2.1.2	1.1.2.1.2	1.1.2.1.2	1.1.2.2.2	2.1.2.2.2
Simuliidae	-.-.2.-.1	-.-.1.-.-	-.-.-.1.1	1.-.1.-.1	-.-.1.1.1	1.1.-.-.-	
Chironomidae	2.1.1.1.1	2.2.1.-.2	2.1.1.1.1	2.1.1.1.1	2.2.2.2.1	1.1.1.2.1	-.2.1.2.1
Psychodidae	-.1.-.-.-						-.-.-.1.-
Tabanidae							1.1.-.-.-
Empididae		-.-.-.1.-	-.-.-.1.-	-.1.-.1.-	-.1.-.-.-	-.1.-.1.-	-.-.-.1.-
Cordulegaster spp.			1.-.-.-.-				
Atherix spp.		-.-.1.-.1		-.-.1.1.-		-.1.1.1.1	1.-.1.-.1
Sialis Fuliginosa Pictet					-.-.-.-.1		
Gammarus pulex (L.)						-.-.1.-.-	
Dixa maculata					-.-.-.1.-		

Appendix 6.3 continued.

	T8	T9	FF	DO	GF	GW
Ephemeroptera						
Baetis buceratus Eaton				-.-.-.-.1		
Baetis muticus (L.)	-.-.-.2.-	-.1.-.1.-				-.-.-.1.-
Baetis rhodani (Pictet)	2.2.3.3.3	3.3.3.3.3	2.2.2.2.3	2.1.2.-.2	2.2.2.2.2	2.3.2.3.3
Baetis scambus Eaton	1.-.1.-.-	1.-.1.1.2				-.-.-.-.1
Baetis vernus Curtis				-.-.-.-.1		
Baetis niger (L.)	-.-.-.1.-	-.1.-.1.-				
Centroptilum luteolum (Muller)						
Rhithrogena semicolorata (Curtis)	2.2.4.4.4	2.3.3.3.3	2.2.3.2.3	2.2.2.2.2	2.2.3.2.3	2.3.3.3.3
Heptagenia sulphurea (Muller)	1.1.2.-.-	1.2.2.2.-				
Heptagenia lateralis (Curtis)	-.-.-.1.-	-.1.-.1.-	-.1.-.1.-		-.1.-.2.-	-.1.-.2.-
Ecdyonurus spp.	-.2.3.3.2	2.1.2.2.2	2.2.2.1.2	1.-.1.-.1	2.1.2.2.2	2.1.2.2.2
Caenis spp.		-.1.-.-.-				
Paraleptophlebia spp.						
Siphlonurus spp.	-.-.1.-.-					
Ephemera danica Muller		-.1.-.-.1				1.-.-.-.-
Ephemerella ignita (Poda)	-.-.1.1.-	2.-.2.1.1	2.-.-.-.-	1.-.-.-.-		1.-.-.- -
Plecoptera						
Brachyptera risi (Morton)	-.-.-.1.-	-.-.-.1.-	-.2.-.2.-	-.2.-.1.-	-.2.-.1.-	-.-.-.1.-
Protonemura meyeri (Pictet)	-.-.2.1.1	-.-.2.-.2	-.-.2.1.2	1.-.2.1.2	2.-.2.-.2	1.-.2.-.2
Protonemura praecox (Morton)	-.-.-.-.1		-.-.1.-.-	-.-.1.-.-	-.-.-.-.1	-.-.-.-.1
Amphinemura spp.	-.1.1.2.-	-.2.-.2.-	-.2.1.-.-	-.1.1.2.1	-.1.-.1.-	-.1.-.2.-
Nemoura avicularis Morton	-.-.1.-.-		1.-.-.-.2		-.-.1.-.-	-.-.1.-.-
Nemoura cambrica (Stephens)						
Nemoura cinerea (Retzius)						
Nemoura erratica Claassen			-.1.-.-.-	1.-.-.-.-		
Leutra fusca (L.)	1.-.2.-.2	2.-.1.-.2	2.-.1.-.1	1.-.1.-.-	2.-.1.-.2	2.-.1.-.2
Leutra hippopus (Kempny)	-.1.1.1.-		-.1.1.1.-	-.-.-.-.1	-.-.1.-.-	-.1.-.-.-
Leutra inermis Kempny	-.2.-.3.-	-.2.-.2.-	-.2.1.2.-	-.2.-.2.-	-.2.1.3.-	-.2.-.2.-
Leutra moselyi Morton	-.-.-.-.1		1.-.1.-.1	1.-.1.-.-	-.-.1.-.1	
Leutra nigra (Olivier)	-.-.-.1.-	-.-.-.1.-	-.-.1.-.-	-.-.-.1.-	-.-.1.-.-	
Perlodes microcephala (Pictet)	1.-.2.1.2	1.-.2.-.1	2.-.2.-.2	-.-.1.-.-	1.-.1.-.1	2.1.1.1.-
Diura bicaudata (L.)	-.-.1.-.-					-.1.-.-.-

Isoperla grammatica (Poda)	1.2.1.2.-	-.2.1.2.-	-.2.2.2.2	1.1.2.2.2	-.2.-.2.2	-.2.1.2.-
Perla bipunctata Pictet	-.1.1.1.-	1.1.2.2.2	-.-.1.1.1		1.-.1.-.-	-.1.1.1.1
Chloroperla torrentium (Pictet)	-.2.-.2.-	-.2.-.1.-	1.2.1.1.-	-.1.1.2.-	1.1.1.1.1	-.2.1.2.-
Chloroperla tripunctata (Scopoli)	1.2.1.2.1	1.1.1.1.-	1.2.2.2.2	1.1.1.2.2	1.2.2.2.2	1.2.2.2.2
Trichoptera						
Rhyacophila dorsalis (Curtis)	1.-.1.1.1	1.-.1.1.1	2.1.-.1.2	-.-.-.1.2	1.2.1.-.1	2.-.1.1.1
Rhyacophila septentrionis McLachlan	-.-.-.-.1				-.-.-.-.1	
Glossosoma spp.	-.-.1.1.2	-.1.2.-.2	-.1.-.-.-		-.1.-.2.-	1.1.1.1.2
Agapetus spp.	-.2.-.2.-	-.1.-.2.-				1.-.-.1.-
Philopotamus montanus (Donovan)			-.-.1.-.-		-.-.1.-.-	
Plectrocnemia geniculata McLachlan	-.-.-.1.-			-.1.-.-.-	-.-.1.1.-	-.1.-.-.-
Polycentropus flavomaculatus (Pictet)	-.1.1.-.-	1.1.1.-.-		1.-.1.-.-		1.-.-.1.-
Polycentropus kingi McLachlan						
Holocentropus spp.						-.-.-.1.-
Hydropsyche angustipennis (Curtis)						
Hydropsyche fulvipes (Curtis)				1.-.-.-.-	1.-.1.-.-	
Hydropsyche pellucidula (Curtis)	1.1.1.1.2	1.1.1.-.2		-.-.1.-.-	-.1.-.-.-	-.-.1.1.1
Hydropsyche siltalai Dohler	1.2.2.2.2	-.2.2.3.2	1.2.2.2.-	1.1.1.2.2	1.2.1.2.1	-.2.2.2.2
Brachycentrus subnubilus Curtis	1.-.-.-.-	1.-.2.-.2				
Crunoecia irrorata (Curtis)			-.-.1.-.-		-.-.1.-.-	
Lepidostoma hirtum (Fabricius)	1.2.2.2.-	-.1.2.2.-	-.-.1.-.-	-.-.1.2.-	-.1.1.1.-	-.2.1.2.-
Drusus annulatus Stephens	-.2.-.-.-	-.1.-.-.-	1.2.-.2.1		-.2.-.1.1	-.2.-.2.1
Ecclisopteryx guttula (Pictet)			-.1.-.-.-		-.1.-.-.-	1.1.-.1.-
Allogamus auricollis (Pictet)			-.-.2.-.-			
Halesus spp.	-.-.-.1.-	-.-.-.1.-	-.2.1.1.1	-.-.-.1.-	-.-.-.1.-	-.1.1.1.-
Micropterna lateralis (Stephens)						1.-.-.-.-
Potamophylax spp.	-.-.1.1.-		1.-.-.1.1	1.-.-.-.-	-.-.-.-.1	1.-.1.1.-
Stenophylax spp.			-.-.1.-.-			-.-.1.-.-
Anabolia nervosa (Curtis)	-.-.-.1.-	-.1.-.-.-				
Chaetopteryx villosa (Fabricius)				-.-.-.1.-		
Limnephilus spp.						
Goera pilosa (Fabricius)		-.-.-.-.1				
Silo pallipes (Fabricius)		-.-.1.-.-	1.1.2.2.1	-.1.-.-.-	1.1.2.1.-	-.1.2.2.-
Sericostoma personatum (Spence)	-.-.2.2.1	-.1.2.2.2	1.1.2.1.2	-.-.-.1.1	2.1.1.2.1	2.1.2.2.2
Odontocerum albicorne (Scopoli)			-.1.1.-.1			
Athripsodes spp.		-.-.-.1.1				
Oecetis spp.	-.1.1.1.-	-.1.1.2.-	-.1.-.-.-			-.1.-.-.-

Coleoptera						
Noterus spp.		-.-.-.-.1				
Oreodytes septentrionalis	-.1.-.1.-		-.2.-.-.-	-.1.-.1.-		
Oreodytes sanmarkii			2.2.-.1.1	1.1.-.1.-		1.-.-.1.-
Gyrinidae	-.1.-.1.1	2.-.2.-.1	-.-.-.-.1	1.1.1.1.1	-.1.1.1.1	1.-.1.2.1
Hydraena gracilis	-.1.1.1.-	-.1.1.1.-	-.1.1.1.1		-.-.1.1.1	-.1.1.1.1
Elmis aenea	-.-.2.2.1	-.-.2.2.1	1.2.2.2.2	1.-.2.2.1	1.1.1.1.1	1.2.2.2.2
Esolus parallelepipedus	-.-.1.2.-	-.-.-.1.-	-.-.-.1.1	-.-.-.1.-		-.-.-.1.1
Limnius volckmari	1.2.2.2.2	2.2.2.2.2	2.2.2.2.2	2.2.2.2.2	1.1.1.2.1	2.2.2.2.2
Oulimnius spp.			-.-.-.-.1			
Molluscs						
Ancylus fluviatilis Mull.	1.1.2.2.1	2.2.2.2.2	1.1.1.1.1		1.-.1.1.1	2.1.2.2.2
Potamopyrgus jenkinsi (Smith)	-.-.1.-.-	-.1.2.2.1				
Lymnaea spp.	-.-.1.-.-	1.-.1.1.-				1.-.-.-.-
Planorbis spp.	-.-.1.-.-	-.-.1.1.-				
Sphaerium spp.			-.-.1.-.-		-.-.1.-.-	
Other taxa						
Oligochaeta	1.2.2.2.2	1.2.2.2.2	2.2.2.2.2	2.2.2.3.2	2.2.2.2.2	1.2.2.2.2
Planariidae	-.1.2.2.2	-.-.-.1.1	-.1.1.1.2	1.1.2.2.2	-.1.1.1.1	-.2.2.2.2
Glossiphoniidae	-.-.1.1.1	-.1.-.-.-				
Erpobdellidae	-.-.1.-.-					
Tipulidae	1.2.2.3.2	1.2.2.2.2	2.1.2.1.2	1.1.2.1.1	-.1.2.2.2	2.1.2.2.2
Simuliidae	1.-.-.-.1	1.1.1.1.-	1.1.1.1.2	2.1.1.-.1	-.-.1.2.1	1.1.1.1.2
Chironomidae	1.2.2.2.2	1.2.1.2.2	2.2.2.1.2	2.1.1.2.-	2.2.1.2.1	2.2.2.2.2
Psychodidae						
Tabanidae	-.-.-.1.-	1.1.-.-.1	-.1.-.1.-	1.-.-.-.-	-.-.1.-.-	
Empididae	-.-.-.1.-	-.-.-.1.-	-.-.-.1.-	-.-.-.1.-	-.1.-.1.-	-.-.-.1.-
Cordulegaster spp.						
Atherix spp.	1.1.1.1.2	2.1.2.1.2		1.1.1.1.1	-.1.-.-.-	-.1.-.-.-
Sialis Fuliginosa Pictet						
Gammarus pulex (L.)	-.-.1.-.-	-.1.1.-.1				
Dixa maculata						

CHAPTER 7

THE EFFECT OF LIMING ON JUVENILE SALMONID POPULATIONS

7.1 Introduction

In recent decades there has been a growing awareness of the detrimental effects of acidification on fisheries in parts of Northern Europe and Northern America (Beamish & Harvey, 1972; Jensen & Snekvik, 1972; Wright & Snekvic, 1978; Haines, 1981; Sevaldrud & Skogheim, 1986).

Acidification, however, was inflicting losses in fish populations in Norway in the 1960s (Muniz and Leivestad, 1980) and according to the Directorate for Nature Management, Norway (1990) probably as far back as 1900. Unfortunately the lack of adequate rigorous historical data (Ormerod & Gee, 1990) has made it difficult to determine what could have been gradual changes, and delayed attention being focused on the adverse effect of acidification. This gradual change is highlighted by Watt *et al.* (1983) who describe detriment to Atlantic salmon (*Salmo salar* L.) in Nova Scotia rivers from an apparent 0.017 pH units per year decrease over 27 years.

The impact of acidification on fisheries in the UK was demonstrated in the early 1980s (Harriman & Morrison 1981; Stoner *et al.*, 1984; Stoner and Gee, 1985). Within Wales, the headwaters of the River Tywi have been extensively studied (Edwards *et al.*, 1990). Poor juvenile production from the fishery protection scheme for Llyn Brianne caused concern and in 1981 highlighted a water quality problem above the reservoir (Gee, 1990).

Among the most studied fish are the salmonid family, salmon, brown trout (*Salmo trutta* L.) and the rainbow trout (*Oncorhynchus mykiss*) and the brook trout (*Salvelinus fontinalis* L.). This is partly due to their high commercial value, but also because their habitat or nursery areas in upland streams may be within the areas of slow weathering rocks and low neutralising capacity soils typically affected by acidification.

This chapter describes the situation in the upper Tywi prior to liming, conditions that can be considered to reflect the status of the fisheries as a result of the acidification process. These baseline data are compared with the status after liming and possible causes for the changes are examined.

7.1.1 Literature review

There is a difficulty in describing a generic effect of acidification upon fish since, as described below, many complexing factors exist which modify the apparent impact.

Effects of low pH on fish

While low pH (<4.0) *per se* is damaging to fish with the incursion of H^+ ions causing acid base imbalance, blood acidosis, impeding O_2 uptake and transport (Fromm, 1980; Wood & McDonald, 1982), Wood (1989) argues that such low pH will rarely occur in the field and that in soft water the failure of ion regulation is of primary concern. In fresh-water fish the ionic gradient increases from freshwater to blood plasma and so the losses to diffusion and urine need to be balanced by uptake from the water. The gills are the main site of ion uptake, mainly facilitated by their large surface area and the separation of the blood and external water by a thin respiratory epithelium. Blood plasma contains c. 150 mequiv $Na^+ l^{-1}$ (Leivestad *et al.*, 1980; Potts & McWilliams, 1989) and monitoring of this parameter has been a target for many workers to describe the physiological effects of acid water (e.g. Fivelstad & Leivestad, 1984; Booth *et al.*, 1988).

Low pH (<5.5) typical of natural conditions disrupts ion regulation by increasing the efflux of both Na^+ and Cl^- and inhibiting influx (e.g. Potts & McWilliams, 1989) resulting in a net loss. Potts & McWilliams (1989) stated that the inhibition of Na^+ uptake at low pH is to be expected from a system exchanging Na^+ and H^+ ions and give the competition between Na^+ and H^+ at binding sites, the augmented effort from the pump to expel H^+ against the large gradient, and the impairment of the ATPase at depressed pH as the possible processes involved. The increased Na^+ efflux is attributed to the change in transepithelial potential from the high permeability to H^+ ions and an increase in permeability of the fish gill to the Na^+ ion which is probably due to the displacement of Ca^+ ions from the surface of the gill. Wood (1989) suggested that when the blood plasma electrolytes show a rapid loss of more than 30% then death occurs within hours as a result of cardiac failure due to haematological and fluid volume disorder (Wood & McDonald, 1982). The detrimental effect of rapid ion loss is also advocated by Booth *et al.* (1988) who showed that a fish losing more than 4% of its

total body Na^+ in 24 hours had a greater than 90% chance of mortality within 11 days.

Effect of low pH and Aluminium

Fresh water of low pH is often synonymous with elevated heavy metals (Chapter 5), especially aluminium, and this has been an area of extensive investigations (see references in Howells *et al.*, 1990). The complex nature of aluminium in fresh water was also stressed in Chapter 5 and has led to great debate as to the causal mechanisms of aluminium that have been demonstrated at low pH (e.g. Stoner *et al.*, 1984).

A pH of 5.2-5.4 is considered the most toxic to fish in the presence of aluminium (e.g. Fivelstad & Leivestad, 1984) and the inorganic monomeric (labile) fraction of aluminium is the most toxic to fish (Wood, 1989; Howells *et al.*, 1990).

At a pH <4.0 aluminium is almost completely in the trivalent state of Al^{3+} and it has been suggested that this may have a beneficial effect on fish (Schofield & Trojnar, 1980) by competing for branchial binding sites with H^+, acting in a similar role to calcium as discussed later (Wood, 1989). As pH increases the cationic hydroxides $Al(OH)^{2+}$ and $Al(OH)_2^+$ predominate at pH's of around 5.0 and 5.5 respectively. The hydroxide $Al(OH)_3$ occurs around pH 6 and as pH increases above pH 6 the dominant form is $Al(OH)_4^-$ (Lydersen, 1990). Lydersen (1990) stresses the effect of temperature on the relative concentrations of the hydrolysis complexes with a shift of the toxic hydroxide species to a higher pH as temperature decreases.

The hydroxide species $Al(OH)^{2+}$ and $Al(OH)_2^+$ have therefore been suggested as the most toxic (Fivelstad & Leivestad, 1984; Sadler & Lynam, 1987) though Lydersen (1990) questions whether the high aluminium toxicity observed at pH 5.2-5.4 may be an effect from the pure aluminium hydroxide polymerization. A further complication is that conditions in the gill micro-environment are probably less acidic (Neville, 1985) than the surrounding water due to the NH_3 excretion and buffering from mucus and gill anions (McDonald *et al.*, 1989), with an increase of some 0.4 pH units. Thus conditions for aluminium precipitation are possible at the gill-water interface, which could deposit on the mucous on the gill epithelium. This would necessitate the fish to 'cough' to remove it, so

that ventilation is not impaired.

The actual mechanism of aluminium toxicity in the pH range 5-6 (often encountered in the field) seems somewhat confused between disruption of ion regulation and impairment of respiratory exchange.

Aluminium exacerbates the effects of low pH (Wood, 1989) by acting synergistically with H^+ to increase membrane permeability (Turnpenny, 1990) and disrupting ion regulation. Dalziel *et al.* (1985) showed inhibition of Na^+ influx occurred in brown trout in the presence of aluminium at pH 4.0-4.5, with an increase in efflux at pH 5.4. Potts and McWilliams (1989) thus postulated that Al^{3+} and $Al(OH)^{2+}$ are responsible for influx inhibition and $Al(OH)_2^+$ is responsible for efflux. Staurnes *et al.* (1984) observed a decrease in the activity of the enzyme responsible for Na^+ uptake (Na-K-ATPase) with 0.2 mgl-1 Al at pH 5. Dietrich & Schlatter (1989) found that rainbow trout exposed to relatively low aluminium concentrations (0.1-0.2 mg/l) at pH 5.2 exhibited electrolyte loss but also showed cell necrosis and fusions of the secondary gill lamellae, impairing gas exchange, which was attributed to inorganic monomeric aluminium. Impaired gas exchange would presumably increase electrolyte loss due to hyperventilation and therefore increased blood flow to the gills (Fivelstad & Leivestad, 1984) which are already damaged. At higher pH (>5.4) and aluminium levels (>0.2 mg/l), toxicity was attributed to impairment of gas exchange from clogging of the gills with mucus, with the physical irritation of the gills by accumulating polymeric aluminium. Neville (1985), also using rainbow trout, showed that electrolyte loss was responsible for death at pH 4.0-4.5 in the presence of inorganic aluminium but respiratory failure (hypoxia) was accountable at pH 6.1, at the same aluminium concentration. The intermediate pH range of 5.0-5.5 displayed transitionary affects between the ionoregulation and respiratory disruption mechanisms.

Low pH and trace metals

While many of the investigations into metal toxicity in acid water have centred around aluminium there is also an awareness of the effects of other trace metals such as iron, manganese, zinc, copper, lead, nickel and cadmium (Andersson & Nyberg, 1984; McDonald *et al.*, 1989; Reader *et al.*, 1989; Sayer *et al.*, 1989). Turnpenny *et al.* (1987) showed that poor fish biomass in some Welsh streams was attributed to heavy metal

concentrations, with poor biomass in the English Pennine streams being aluminium related.

Andersson & Nyberg (1984) quoted iron (0.55-1.2 mgl^{-1}) as being the primary cause of death to brown trout at pH c.5.5, though in the presence of both manganese and aluminium. Reader *et al.* (1989) exposed brown trout fry to eight trace metals (Cu, Ni, Pb, Zn, Al, Cd, Fe and Mn) at concentrations typical of soft, acid waters at pH 4.5 and 6.5 . All eight metals increased mortality at pH 4.5 with negligible effect at pH 6.5, though only five (Al, Cd, Cu, Pb and Zn) gave < 60% survival. In contrast, Cusimano *et al.* (1986) (quoted in McDonald *et al.* (1989)) stated that the toxicity to rainbow trout should decrease at pH 4.7 for Cu, Cd and Zn compared to pH 5.7 and 7.0, from increased competition for binding sites with H^+. McDonald *et al.* (1989) discussed the sublethal effects and mechanisms of trace metals upon branchial ion regulation and summised that Cu and Al were disruptive to Na^+/Cl^- balance and Cd, Zn and Mn mainly interfered with Ca^{2+} balance.

In assessing the effects of acid waters on fish it is clear that a number of trace metals should be considered and not aluminium in isolation.

Sublethal effects of acidified waters

While many investigations have focused on lethal levels of low pH and trace metals to fish there is also the important consideration of sublethal levels and effects. Among those effects most noted are retarded growth and development of fry (Baker & Schofield, 1982; Cleveland *et al.*, 1986) at pH>5.2 with aluminium exposure, and behavioural impairments (Hunn *et al.*, 1987) which could lead to less viable fry survival. Disruption in skeletal development from reduced mineral uptake (calcium, sodium and potassium) has also been noted (Reader *et al.*, 1988; Turnpenny, 1990).

Variability in response of different life stages

Some of the difficulty with interpreting the general effects of acidified waters is attributable to the relative effects on different life stages. The toxicity to aluminium has been suggested to increase with age (Brown & Sadler, 1989; Ingersoll *et al.*, 1990) but certain life stages may be more susceptible, e.g. young fry and migrating smolts of salmonids are

considered particularly vulnerable (Howells *et al.*, 1990). Toxicity to pH alone has been stated to decrease with age (Brown & Sadler, 1989) but in contrast Daye & Garside (1979) showed embryos of Atlantic salmon to be more resistant to acidity than alevins, with LL_{50} of pH 3.9 for embryos (median hatch 94-97 days) and pH 4.3 for alevins (over 40 days).

Carrick (1979) found that a pH of 3.5 was lethal to salmon, trout and sea trout eggs but at pH 4.5 and above there was no demonstrable effects from acidity. Brown & Sadler (1989) also showed toxicity increased with lower pH but stressed the role of calcium in ameliorating this (see below). Eggs are particularly vulnerable to the H^+ ion when at the freshly fertilised stage (Ingersoll *et al.*, 1990; Turnpenny, 1990) but show less sensitivity 'post eyeing'. The presence of aluminium has been shown to reduce toxicity to eggs (Baker & Schofield, 1982; Ingersoll *et al.*, 1990) and is probably as a result of polyvalent aluminium reducing H^+ membrane permeability similar to calcium (Schofield & Trojnar, 1980).

While Ingersoll *et al.* (1990) showed a pH of <5.2 affected brook trout eggs a pH of <4.4 was required to reduce survival of fish beyond the yolk-sac stage; the addition of aluminium (>0.333 mgl^{-1}), however, was toxic to the latter age group. Brown (1983) showed how brown trout 10 to 30 days post hatch survived some 12 days in water of pH 5.4, 1 mgl^{-1} calcium and 0.5 mgl^{-1} aluminium. Survivorship in this water decreased rapidly post swim-up to less than 1 day when the fish were 70 days post hatch and to around 8 hours when 450 days post hatch.

The pre-smolt and smolt stages of anadromous fish such as the Atlantic salmon have been described as particularly sensitive (Henriksen *et al.*, 1984; Howells *et al.*, 1990) with two-year-old fish more sensitive than one-year olds (Rosseland & Skogheim, 1984).

The literature on large adult fish is sparse, especially anadromous fish, probably as a result of difficulty in size, handling and availability. Potts *et al.* (1989) showed that migrating adult salmon transferred from sea water to freshwater underwent ion regulation difficulties and mortality when exposed to pH 5, >0.5 mgl^{-1} aluminium over 2 days. Skogheim *et al.* (1984) described the death of migrating adult Atlantic salmon below a mixing zone of pH 4.7, aluminium rich water from a hydroelectric power station and pH 6.0 water in the river Ogna.

Susceptibility of different fish species

Amongst the most studied species have been the brown trout and the brook trout. The salmonid group is the most sensitive to acid and aluminium (Howells *et al.*, 1990) and from the literature an increasing sensitivity is observed for the following fish species :-
Oncorhynchus mykiss > *Salmo salar* L.> *Salmo trutta* L.> *Salvelinus fontinalis* L.

Rosseland & Skogheim (1984) noted that for *Salmo trutta* the anadromous form was more sensitive than the resident form.

Factors ameliorating the effects of low pH and trace metals

The important role of calcium for fish in acidified waters was shown by Wright & Snekvik (1978) who found that lake fishery status was better described by calcium than pH. Many workers have since described the mitigative effect of calcium on low pH and trace metals (Muniz & Leivestad, 1980; Nelson, 1982; Brown , 1983; Stoner *et al.*, 1984; Witters, 1986; Hunn *et al.*, 1987; Booth *et al.*, 1988; Thomsen *et al.*, 1988; Wood *et al.*, 1990).

Potts & McWilliams (1989) described the beneficiary mechanism of calcium as reducing the permeability of the fish gill to other ions, inhibiting H^+ incursion and Na^+ efflux, therefore reducing toxicity to fish. Brown & Sadler (1989) described the effect of pH on brown trout egg survival and the influence of calcium levels. At pH 4.2 with 0.25 mgl^{-1} calcium mortality was 100% but decreased to 0% at pH 5.1 at the same calcium level. However, increasing the calcium level to 8 mgl^{-1} at pH 4.2 also prevented mortality.

Stoner *et al.* (1984) stated that calcium <2 mgl^{-1} was detrimetal to salmonids with pH< 5.5 and 0.18 mgl^{-1} filterable aluminium. Turnpenny (1990) described the effect of calcium on hatching success of salmon, trout and sea trout eggs and quoted the ratio of $Ca^{2+}:H^+ = 100$ as being critical for proper development and skeletal calcification. Salmon were distinct in having a threshold pH of 6.0 for surviving at this ratio (Ca. 100 $uequivl^{-1}$ or 2mgl^{-1}). Trout eggs displayed less stringent criteria with low survivorship (<50%) below $Ca^{2+}:H^+ = 100$ but increasing asymptotically above this. Turnpenny (1990) also stated that the above

ratio should prevent toxicity from labile aluminium levels up to 0.1 mgl^{-1}. Since calcium is a major fraction of total hardness it follows that an increase in total hardness offers protection against trace metal toxicity; the degree of protection varying logarithmically with hardness concentration (Brown, 1968). Magnesium also contributes to total hardness and Zitco & Carson (1976) described its importance by reducing toxicity to juvenile salmon by increasing the molar concentration of magnesium to zinc to 5.6:1.

Other inorganic and organic components of water may also abate aluminium toxicity, notably fluorides, silicates and organic, humic substances. Fivelstad & Leivestad (1984) showed that AlF^{2+} was relatively unimportant in the toxicity to fish compared to the hydroxides and Birchall (1989) described the elimination of aluminium toxicity at a 13:1 ratio of Si:Al from silicic acid. The addition of citrate has been shown to complex aluminium and reduce toxicity to fish (Driscoll *et al.*, 1980; Weatherley *et al.*, 1990). As stated earlier, it is the inorganic rather than organic aluminium that is regarded as being most toxic to fish. Therefore, waters with high DOC (>10 mgl^{-1}) tend not to be as toxic for the concentration of aluminium. Lacroix & Townsend (1985) stated that pH was more important than aluminium toxicity in Nova Scotia rivers where DOC was 10-20 mgl^{-1} and less than 10% of dissolved aluminium was in the labile form.

Water quality criteria

From the above account it can be seen that there are many variables influencing the toxicity to fish in acid waters, detrimental and ameliorative. Much of the difficulty in describing exact cause and effects of acidified waters on fish is in drawing together results from investigations on different species (even strains within species) and observations from a myriad of different combinations of water quality. Even if the exact mechanisms have not as yet been fully established, experiments and empirical data have given fishery managers some targets to aim for to minimise any detrimental effect on fish.

Dalziel *et al.* (1990) suggested that a water quality of pH> 6, calcium >2 mgl^{-1} and total aluminium <0.1 mgl^{-1} should protect all life stages of brown trout.

Howells *et al.* (1990) gave more detailed recommendations with regard to maximum aluminium levels, based on fry or one-year-old salmonids. For waters with DOC <10 mgl^{-1} (As in the River Tywi):-

Ca <2 mgl^{-1}; pH >4.5- <5.0; Al <0.030 mgl^{-1}.
Ca <2 mgl^{-1}; pH >5.0- <6.0; Al <0.015 mgl^{-1}.
Ca <2 mgl^{-1}; pH >6.0 ; Al <0.075 mgl^{-1}.
Ca >5 mgl^{-1}; pH >4.5- <5.0; Al <0.060 mgl^{-1}.

Aluminium level referred to is that of the soluble inorganic fraction.

7.2 Materials and methods

7.2.1 Sampling strategy

The strategy to examine the effects of liming on the juvenile salmonids in the upper main river was split into two components.

(i) An annual electrofishing survey was carried out in August/September of each year to examine any change in population size, sampling along the length of the main river to investigate the extent of the impacted area and recovery. Annual surveys of tributaries within the catchment served as control reference sites.

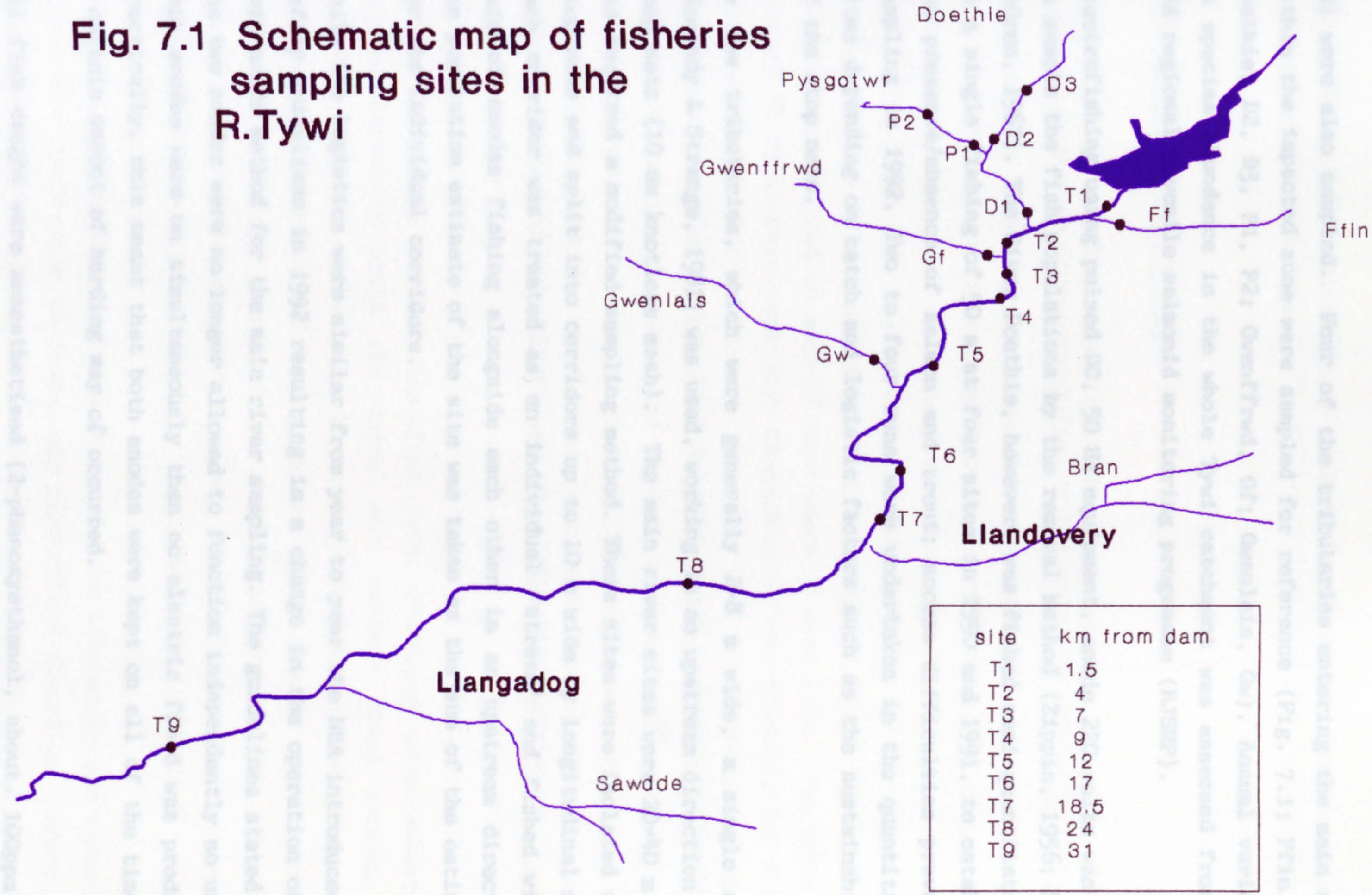
(ii) A complementary pre and post-liming period physiological experiment with caged brown trout was undertaken to investigate the detrimental mechanisms and extent of impact and recovery.

7.2.2 Electrofishing surveys

To quantitatively assess changes in the salmonid populations following liming main-river electrofishing surveys were carried out in 1990, prior to liming, and compared with similar surveys in 1991 and 1992, after liming.

Seven main river sites (T1 to T7; Fig. 7.1) within the suspected impacted area were surveyed. These were identified by personal communications with NRA personnel and where possible were consistent with locations used in previous surveys (1985/1986). Two sites downstream of this zone (T8 &

Fig. 7.1 Schematic map of fisheries sampling sites in the R.Tywi



site	km from dam
T1	1.5
T2	4
T3	7
T4	9
T5	12
T6	17
T7	18.5
T8	24
T9	31

T9) were also sampled. Four of the tributaries entering the main river within the impacted zone were sampled for reference (Fig. 7.1; Ffin, FF; Doethie, D2, D3, P1, P2; Gwenffrwd, Gf; Gwenlais, Gw). Annual variation in species abundance in the whole Tywi catchment was assessed from the NRA regional juvenile salmonid monitoring programme (RJSMP).

Electrofishing using pulsed DC, 50 Hz equipment, anode 220 volts was used to sample the fish populations by the removal method (Zippin, 1956; Seber LeCren, 1967). The River Doethie, however, was fished semi-quantitatively by a single fishing of 50 m at four sites in 1990 and 1991, to establish the presence/absence of salmon and trout; access difficulties prevented sampling in 1992. Two to four runs were undertaken in the quantitative sites depending on catch and logistic factors such as the sustainability of the stop nets.

In the tributaries, which were generally 2-8 m wide, a single anode (Kennedy & Strange, 1981) was used, working in an upstream direction with stop nets (10 mm knotless mesh). The main river sites were 20-40 m wide and required a modified sampling method. These sites were isolated using stop nets and split into corridors up to 10 m wide by longitudinal nets. Each corridor was treated as an individual 'stream' and fished with a pair of anodes fishing alongside each other in an upstream direction. The population estimate of the site was taken as the sum of the estimates for the individual corridors.

While the logistics were similar from year to year the NRA introduced new safety guidelines in 1992 resulting in a change in the operation of the twin anode method for the main river sampling. The guidelines stated that the two anodes were no longer allowed to function independently so unless both anodes were on simultaneously then no electric field was produced. Practically, this meant that both anodes were kept on all of the time and a certain amount of herding may of occurred.

All fish caught were anaesthetised (2-phenoxyethanol, about. 100ppm) and measured (fork length) by 'pricking' their length onto 'aquascribe' paper. Scales were taken from a sub-sample of fish to aid age class determination. Individual fish length was measured to the nearest 1mm and age classes of fry (0+) and parr (>0+) were determined by length frequency and scale readings. Quantitative population estimates were obtained from a computer package utilising the Zippin (1956), Seber

LeCren (Seber & LeCren, 1967) or minimum (total catch) method, according to adequacy of catch data.

7.2.3 Physiological surveys

As part of the pre-liming period fisheries survey the response of brown trout to acid conditions was assessed using blood plasma analysis during an eleven day exposure period in February 1991 (7th-18th). Due to the lack of native brown trout large enough for this work (>15 cm), hatchery reared 1+ brown trout were used (hatchery conditions pH 6.2, 22 mg $CaCO_3 l^{-1}$). Five main river sites were used, T1-T4 and T6 (Fig. 7.1), with a further reference site in the unacidified River Gwenlais (Gw). Five fish were placed in each of four cages at each site. The cages were made of 15 mm plastic mesh with dimensions of approximately 0.5 x 0.3 x 0.3 m and weighted with bricks. The cages were placed in positions with shading and low water velocity to minimise stress. At least ten fish were left in the transporting tank and kept at the laboratory for ten days to assess any loss due to transportation. The sites were inspected daily where possible and one cage from each site was removed at two or three-day intervals for caudal vein blood sampling and gill tissue metal analysis.

To obtain plasma samples the fish were overdosed with anaesthetic (2-phenoxyethanol) while the cages were opened. Blood samples were withdrawn from the caudal vein, between the anal and caudal fins, with a heparinised 25G needle and 1 ml syringe. Blood samples were centrifuged in the field and the plasma stored at -20 °C until analysis. The plasma was analysed for sodium at the Regional NRA laboratory (South), Llanelli using the methodology described for water samples in Chapter 5.

The complete gill arches were dissected in the laboratory and rinsed with double distilled water to remove any particulate aluminium, before being freeze dried for a minimum of 14 hours. The sample was ground and a 0.5 g sample taken and microdigested in AristaR HNO_3 before analysing for Al, Mn, Zn and Fe by ICPOES or ICPMS (see Chapter 5). Due to analytical requirements the gills from all five fish from one cage were pooled for each sampling session.

To maintain similar temporal conditions this experiment was repeated after liming in February 1992 (18th-29th), with brown trout from the same hatchery. An additional site (D1) was included on the River Doethie due to its known episodic acidity (Chapter 5).

7.3 Results

7.3.1 Electrofishing surveys

It is important to note that the main comparisons should be intra-specific comparison of the same site from year to year. Inter-specific comparisons of different sites needs care due to the variation in habitat available at each site (Appendix 7.1). The upper sites (T1-T6) were dominated by boulders/cobbles and good parr habitat whereas the lower sites tended to be more featureless with pebbles/gravel. Variation in the amount of overhanging trees providing bankside cover also creates different habitat potential between sites. As a consequence, inter-site comparisons may display gross trends or presence and absence limits, but not absolute density.

In order to compensate for these cross-site habitat differences, the use of a habitat model to estimate salmonid density potential (HABSCORE) was considered. However, the model was developed from streams generally <10 m and may not therefore be appropriate for the river widths (20-40 m) in this study (N.Milner pers. comm.). As a consequence this approach was not adopted.

The main river survey of 1990 showed a dearth of salmonids in the upper sites (Figs 7.2a-d; Table 7.1). No salmon fry were found at sites T1 to T5, with small numbers at T6, increasing downstream to T9. Salmon parr showed a similar trend with no fish observed upstream from site T5. Trout fry were detected from the uppermost site (T1) downstream, as were trout parr, but both were in low densities (<5/100 m^2).

In 1991, after liming, salmon fry were found from site T2 downstream, with densities at site T5 and T6 particularly high (25 - 40 100 m^2). While sites T2-T6 showed obvious increase in density from the absence in 1990 the main river reference sites (T8 & T9) differed, with the density decreasing at T8 but increasing at T9. Salmon parr were also observed

Fig. 7.2 Mainriver electrofishing results (1990-1992)
NS - not sampled * estimate from 1 run

Fig. 7.2a salmon fry O+

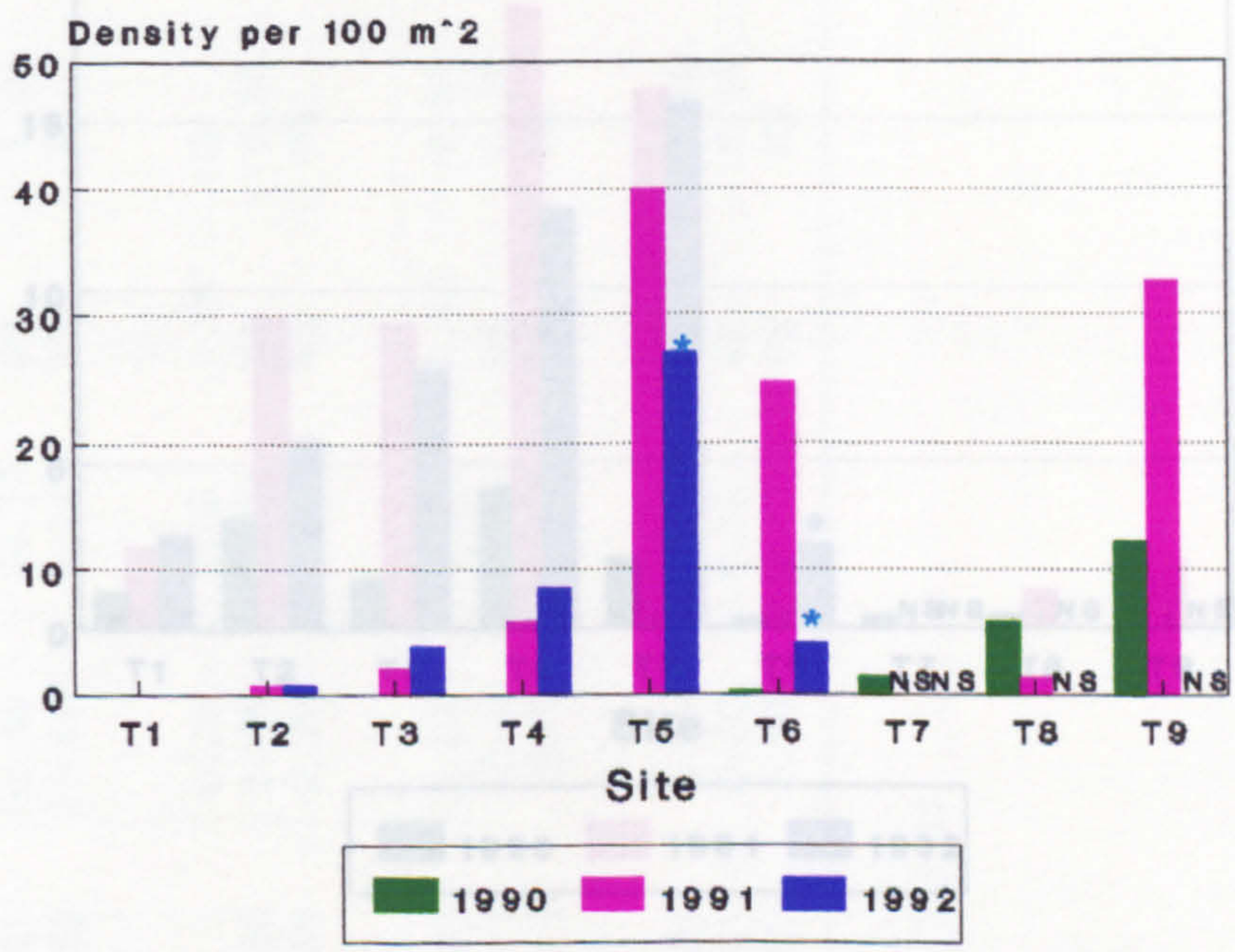


Fig. 7.2b salmon parr >O+

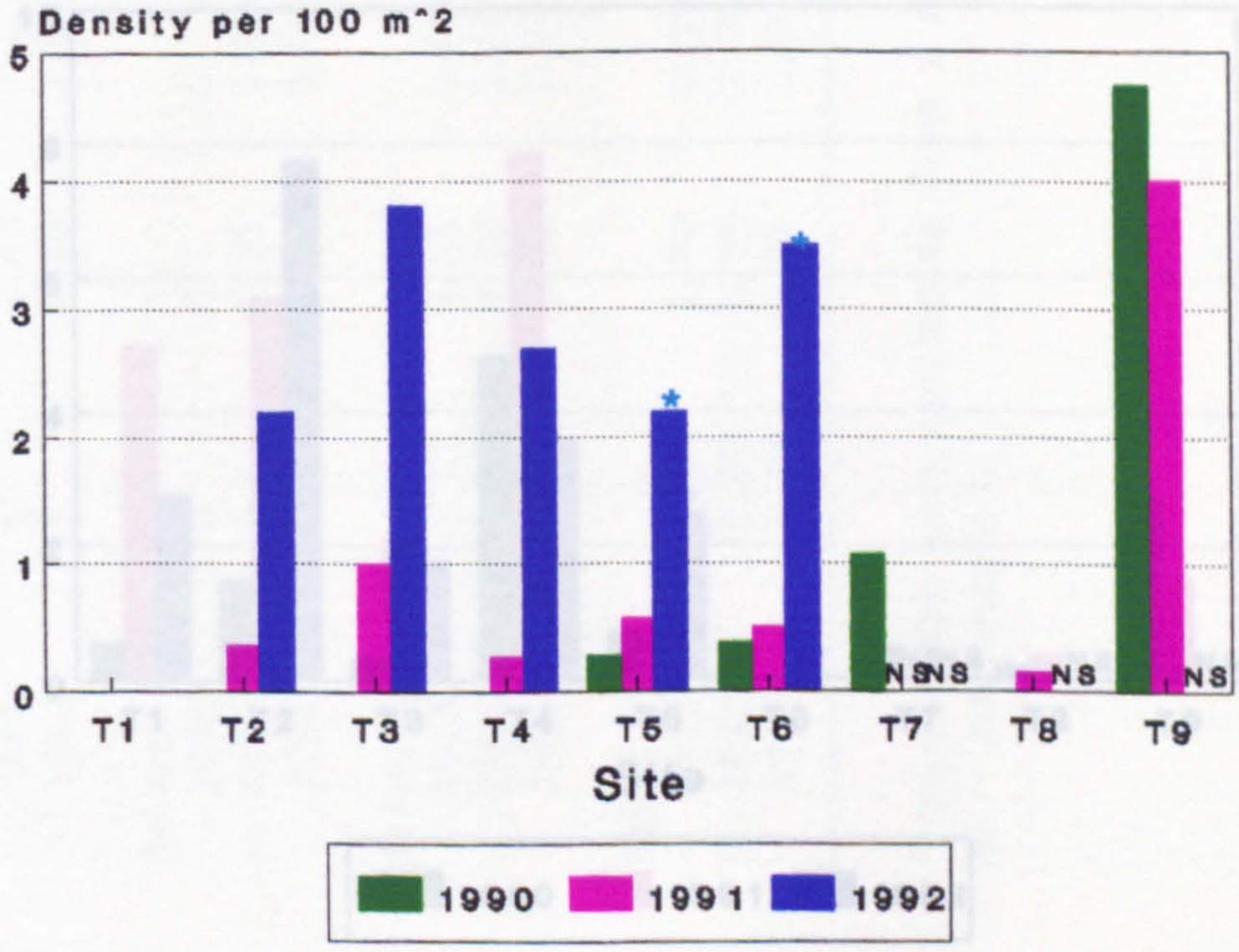


Fig. 7.2 continued

Fig. 7.2c trout fry 0+

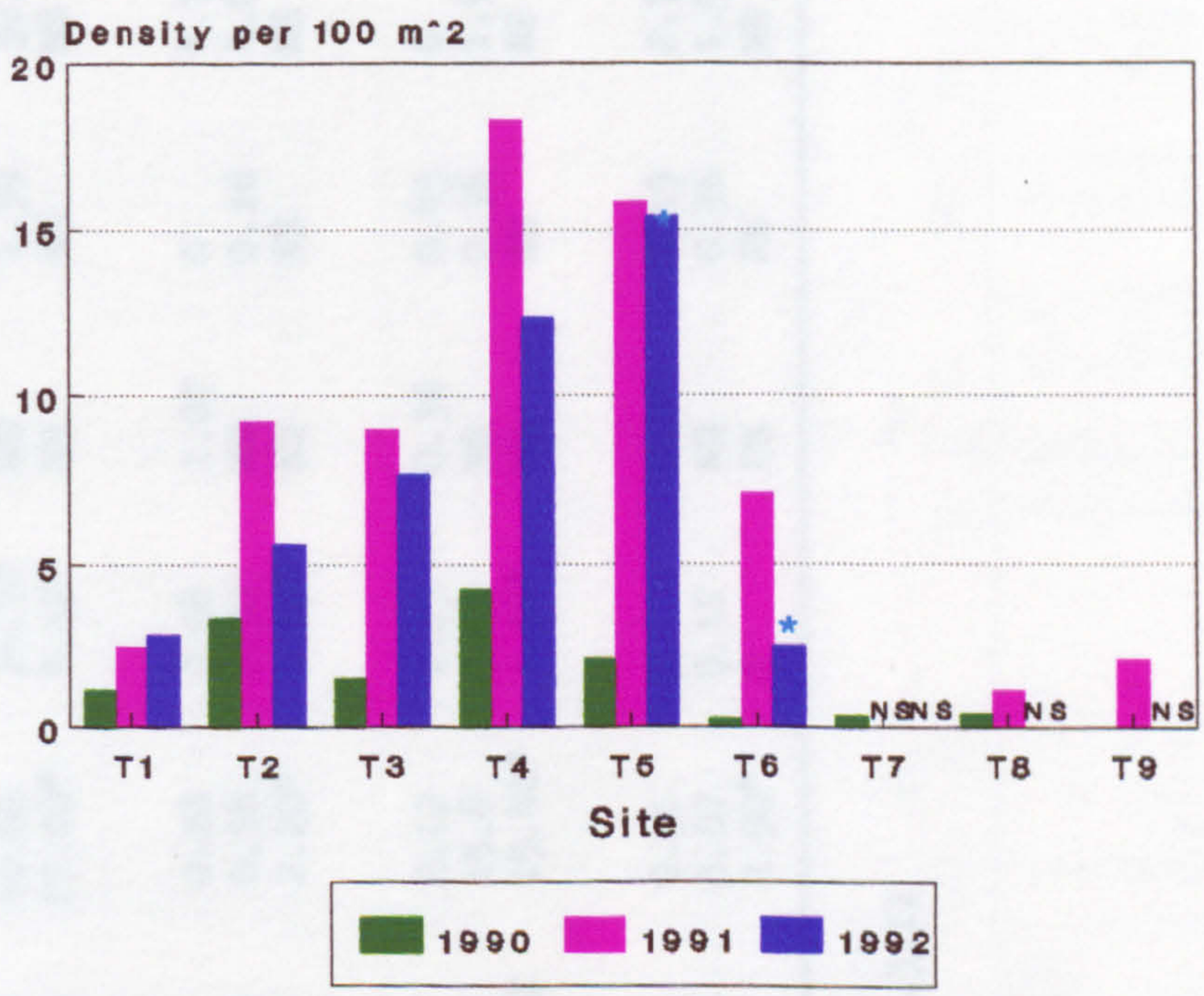


Fig. 7.2d trout parr >0+

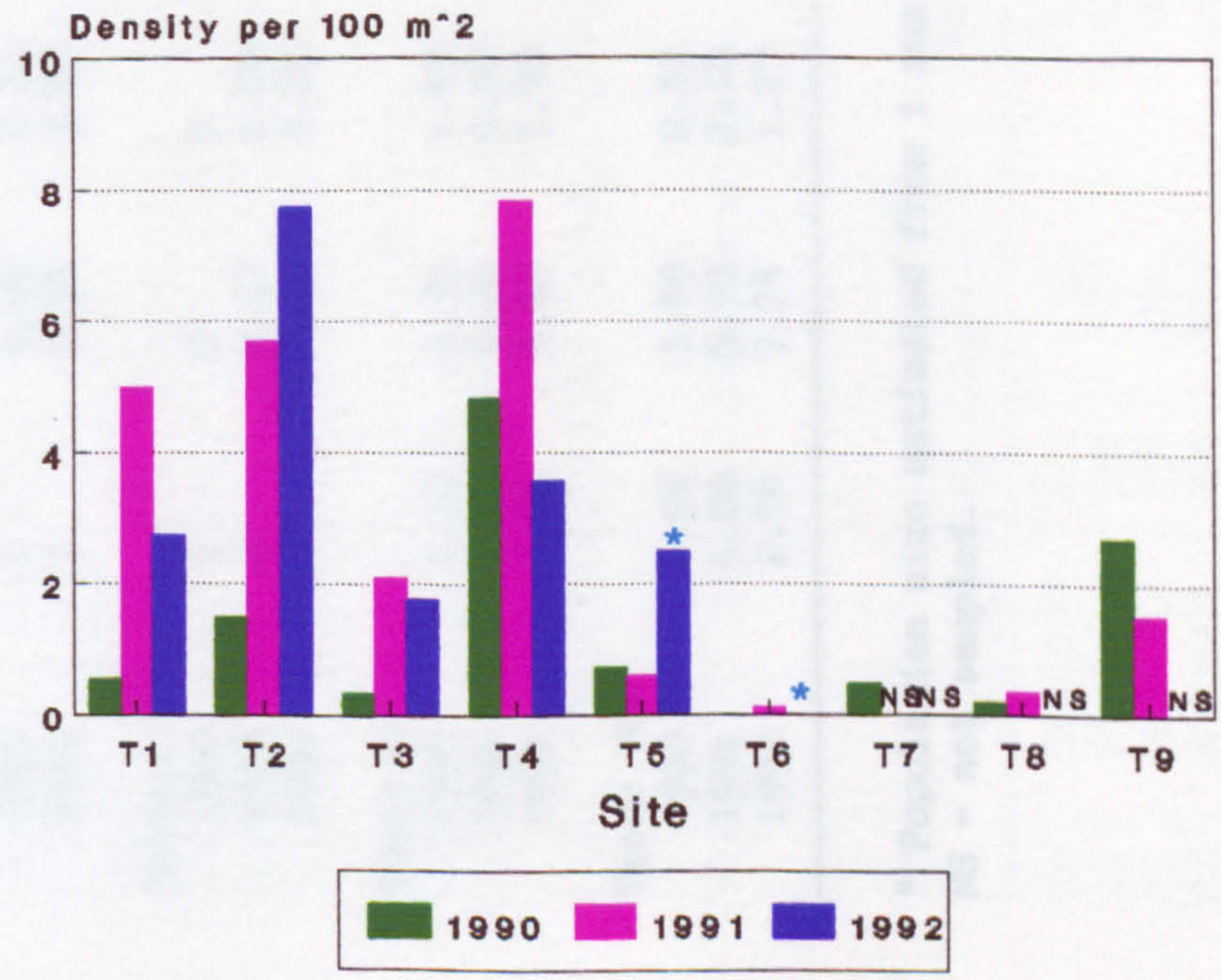


Table 7.1 Juvenile salmonid densities (N/100m^2) from the main river electrofishing surveys.

	Site T1	T2	T3	T4	T5	T6	T7	T8	T9
Salmon 0+									
1990	0	0	0	0	0	0.25	1.32	5.80	12.30
1991	0	0.80	2.10	5.87	40.00	24.70	NS	1.39	32.70
1992	0	0.78	3.85	8.59	27.00*	4.12*	NS	NS	NS
Salmon >0+									
1990	0	0	0	0	0.28	0.38	1.07	0	4.74
1991	0	0.37	1.00	0.28	0.58	0.50	NS	0.16	4.01
1992	0	2.20	3.80	2.69	2.20*	3.50*	NS	NS	NS
Trout 0+									
1990	1.12	3.35	1.49	4.23	2.12	0.25	0.36	0.43	0
1991	2.50	9.20	9.00	18.3	15.8	7.10	NS	0.36	1.54
1992	2.86	5.62	7.70	12.32	15.40*	2.54*	NS	NS	NS
Trout >0+									
1990	0.56	1.49	0.33	4.83	0.71	0	0.48	0.19	2.71
1991	5.00	5.70	2.10	7.83	0.60	0.12	NS	0.36	1.54
1992	2.74	7.74	1.77	3.57	2.50*	0	NS	NS	NS

* Population size estimated from 1 run (see 7.3.1)
NS - not sampled.

Table 7.2 Quantitative electrofishing results (N/100m^2) from the NRA RJSMP survey 1990-92.

River	Salmon						Trout					
	0+			>0+			0+			>0+		
	90	91	92	90	91	92	90	91	92	90	91	92
Blotweth	0.5	0.0	4.9	6.3	2.5	3.0	16.7	10.5	12.7	35.6	21.7	77.0
Melindwr	0.0	1.9	0.7	1.7	0.0	7.3	28.1	20.1	28.0	15.3	11.4	20.7
Cennen	8.8	5.7	13.5	1.1	3.9	9.8	70.2	118.4	168.4	9.8	7.7	33.9
Sawdde	1.2	14.0	15.9	12.4	1.3	12.8	29.9	3.6	41.4	8.1	2.1	34.7
Gwenlais	65.4	36.4	20.7	11.7	4.5	17.5	68.7	36.5	27.2	12.8	16.0	30.3
Gwenffrwd	25.5	40.1	2.7	2.7	6.8	3.9	25.6	19.3	22.6	6.4	7.7	13.4
Ffin	0	0	0	0	0	0	369.4	236.4	173.0	13.1	19.4	28.6
Mean	19.1	16.4	9.7	6.0	3.2	9.1	86.9	63.5	67.6	14.4	12.3	39.5

Note. Ffin not included in mean calculations for salmon.

Table 7.3 Electrofishing results (minimum N/100m^2) from the River Doethie catchment (1 run).

Site	NGR	Area fished m^2 1990 / 1991		salmon 0+		salmon >0+		trout 0+		trout >0+	
				90	91	90	91	90	91	90	91
D2	SN765484	277	55	1.1	1.82	0	3.6	4.7	9.1	0	9.1
D3	SN767490	304	NF	1.0	-	0	-	3.3	-	0.3	-
P1	SN764483	335	296	0.3	0	0	0.3	5.7	12.2	0.6	0.3
P2	SN759485	295	333	0	0	0.3	0	4.7	13.8	0.7	1.8

from T2 downstream though at low densities. Again sites T2-T4 showed an increase from an absence in 1990 but sites T5-T9 showed a similarity with pre-liming period densities. Trout distribution was similar to 1990. Trout fry densities, however, showed a marked increase at sites T1-T6 with up to a seven-fold increase in density. Site T8 showed a small increase while T9 had low numbers of trout fry compared to 1990, when they were not observed. Trout parr showed up to a ten-fold increase in the upper sites (T1-T4) whilst at T5 and T8 the densities were similar to 1990, with T9 actually decreasing slightly.

In 1992, the survey programme was hampered by high rainfall during August and early September reducing possible sampling days. Sites T1-T4 were sampled quantitatively as before, but T5 and T6 were surveyed by one run through the site to give a semi-quantitative estimate (n). The 1992 densities for T5 and T6 were thus estimated by assuming the efficiency of capture (p) on the single run was the same as the mean efficiency of the first runs at the other sites (trout 0+ p=0.50, trout >0+ p=0.49, salmon 0+ p=0.48, salmon >0+ p=0.49). The density (N) was estimated as N=n/p (Cowx, 1983).

Salmon fry were again found from site T2 downstream, with densities comparable to 1991 at T2-T4. The estimated densities for T5 and T6 were lower than 1991. Salmon parr showed an increase in density at sites T2-T6 with typical increases of 3-5 fold (Table 7.1). Trout fry showed a very similar pattern to 1991 but with densities slightly lower at most sites. The increased trout parr densities in 1991 were again evident in 1992, although some sites showed increases on 1991 densities and some showed decreases.

A total of eight tributaries were sampled consecutively in 1990, 1991 and 1992 for the RJSMP by quantitative means. These included the FF, GF and GW which are within the impacted area. The mean densities for each site (Table 7.2) showed no significant increase for trout or salmon fry over the three years (paired t-test P>0.05). Moreover, a downward trend was exhibited (Table 7.2). Trout and salmon parr densities at each site showed no increase from 1990 to 1991, but there was a significant increase in parr densities in 1992 (paired t-test, P<0.05).

The semi-quantitative fishing of the River Doethie demonstrated the presence of both salmon and trout in 1990 and 1991 but both salmonid

species were in low numbers (Table 7.3).

7.3.2 Physiological surveys

Conditions during the pre-liming period survey were generally stable with respect to pH, but low water temperatures (0-4 ^{o}C) were encountered (Figs. 7.3 a-b). Snow prevented returning to the site until four days post-stocking and some extreme results were found. All the brown trout at sites T1-T4 showed 100% mortality and post-stocking blood samples were therefore unobtainable for these sites. Examination of the dead fish showed mucous production on the gills and distended opercula. The fish had been dead some time and anecdotal evidence (G.Copp pers. comm.) allowed an estimate of death at around 2 days after stocking.

Site T6, where mean pH>6, and the reference site in the River Gwenlais showed no mortality over the 11-day exposure period. However, trout at site T6 displayed signs of stress indicated by a decrease in mean blood plasma sodium levels from 126 meql^{-1} at introduction to 90 meql^{-1} after 11 days (Fig.7.4a). Analysis of variance showed this to be significant with time ($p=0.001, F=7.5$) and the samples from the 8th and 11th days after stocking were significantly lower (5%) than at introduction. Analysis of the samples from site GW (Fig. 7.4b) showed no significant change with time ($p=0.051, F=2.91$) and no group of samples were significantly different at the 5% level.

Analysis of the gills of the brown trout (Fig.7.5a-d) indicated a low level of metals from the fish farm environment (Al,10.9 mgkg^{-1} dry wt.; Mn,4.5 mgkg^{-1} dry wt.; Zn, 384 mgkg^{-1} dry wt.; Fe, 40.1 mgkg^{-1} dry wt.). The samples taken from the fish at sites T1-T4 demonstrated a rapid accumulation of aluminium, manganese and iron. Zinc showed no increase from background levels. The largest increases were generally at site T1, where concentrations of aluminium, manganese and iron were 135, 18 and 103.6 mgkg^{-1} dry weight, respectively. These represented a twelve-fold increase in aluminium, a four-fold increase in manganese and over doubling of the iron concentrations, compared to fish farm levels. Gill tissue concentrations from site T6 did not rise to the levels obtained at T1, even after 11 days and the reference tributary showed none of the increases found in the main river (Figs. 7.5a-d).

Fig. 7.3a pH and temperature from T1 - February 1991

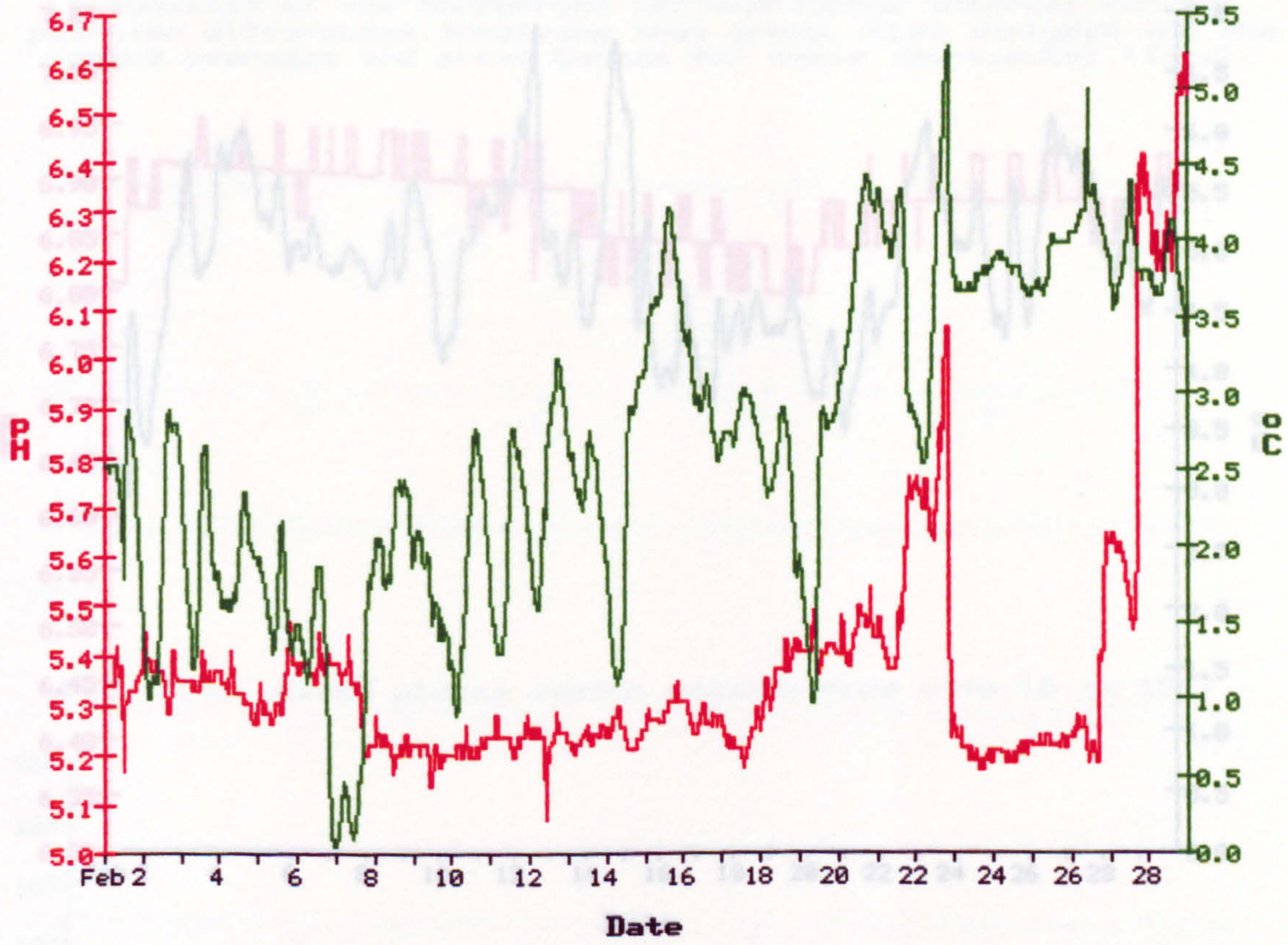


Fig. 7.3b pH and temperature from T6 - February 1991

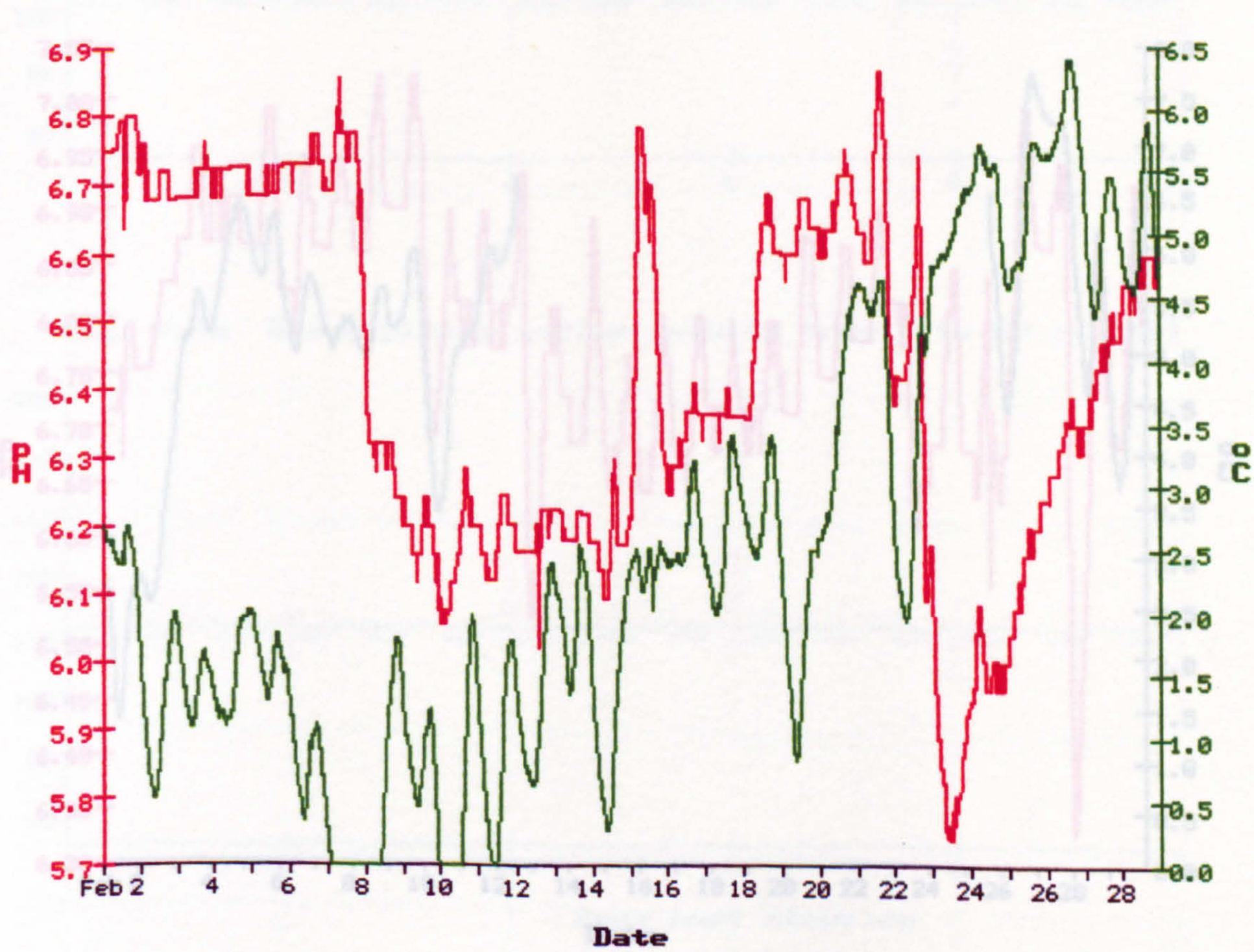


Fig. 7.3c pH and temperature from T1 - February 1992

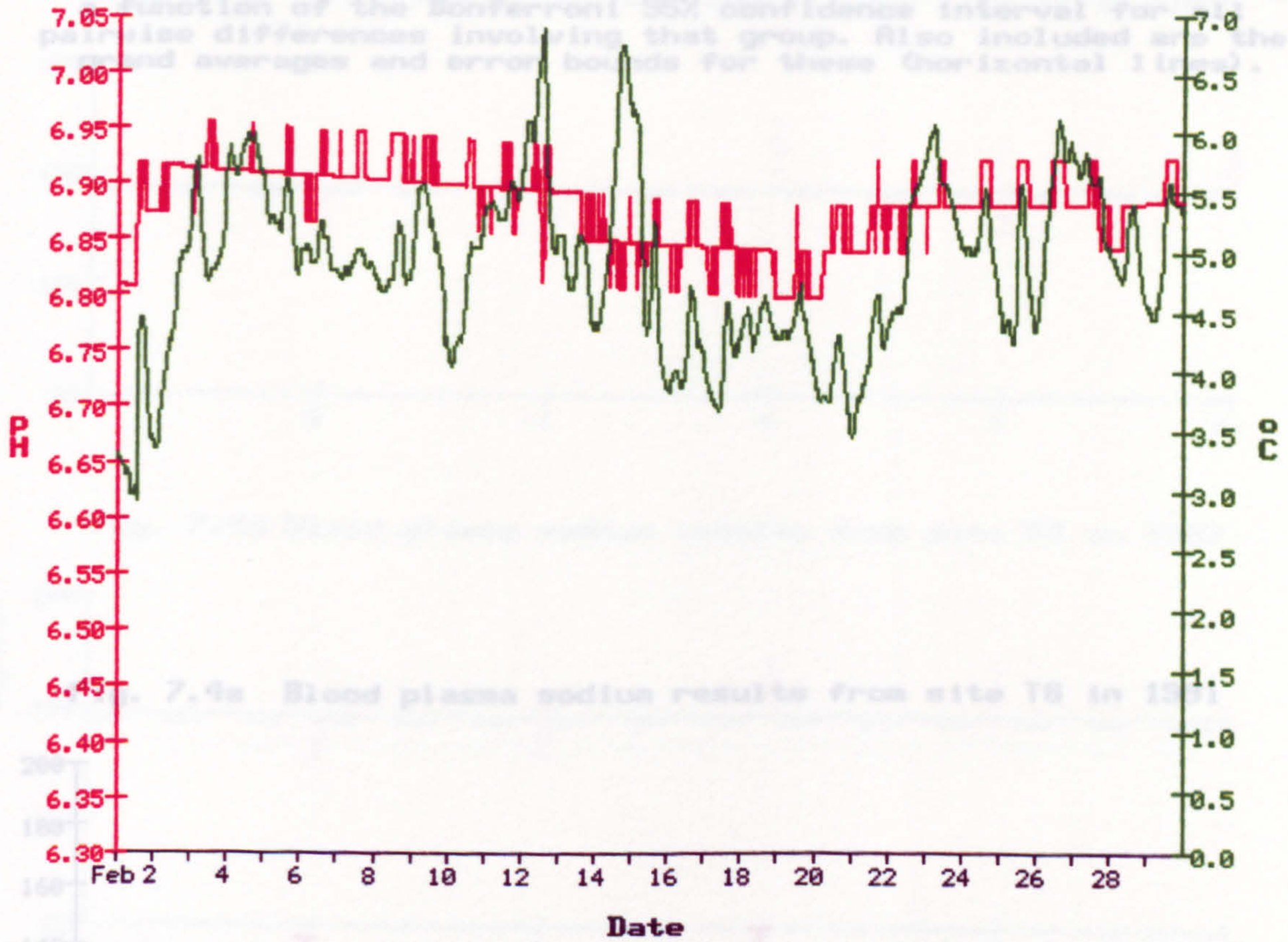


Fig. 7.3d pH and temperature from T6 - February 1992

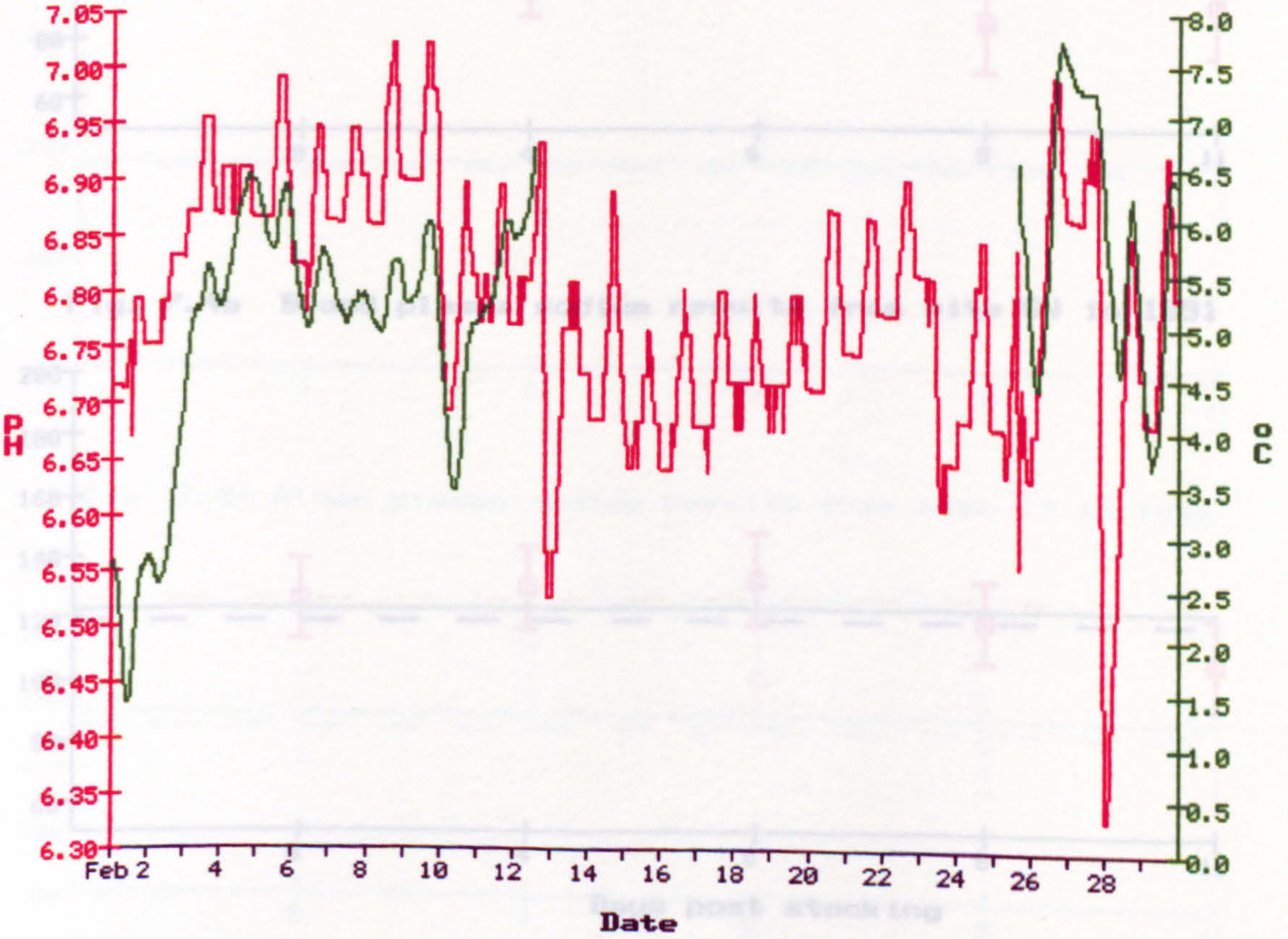


Fig. 7.4 Blood plasma sodium results from the physiological experiments. The graphs show simultaneous comparison between groups with the length of the confidence bar for each group mean being a function of the Bonferroni 95% confidence interval for all pairwise differences involving that group. Also included are the grand averages and error bounds for these (horizontal lines).

Fig. 7.4a Blood plasma sodium results from site T6 in 1991

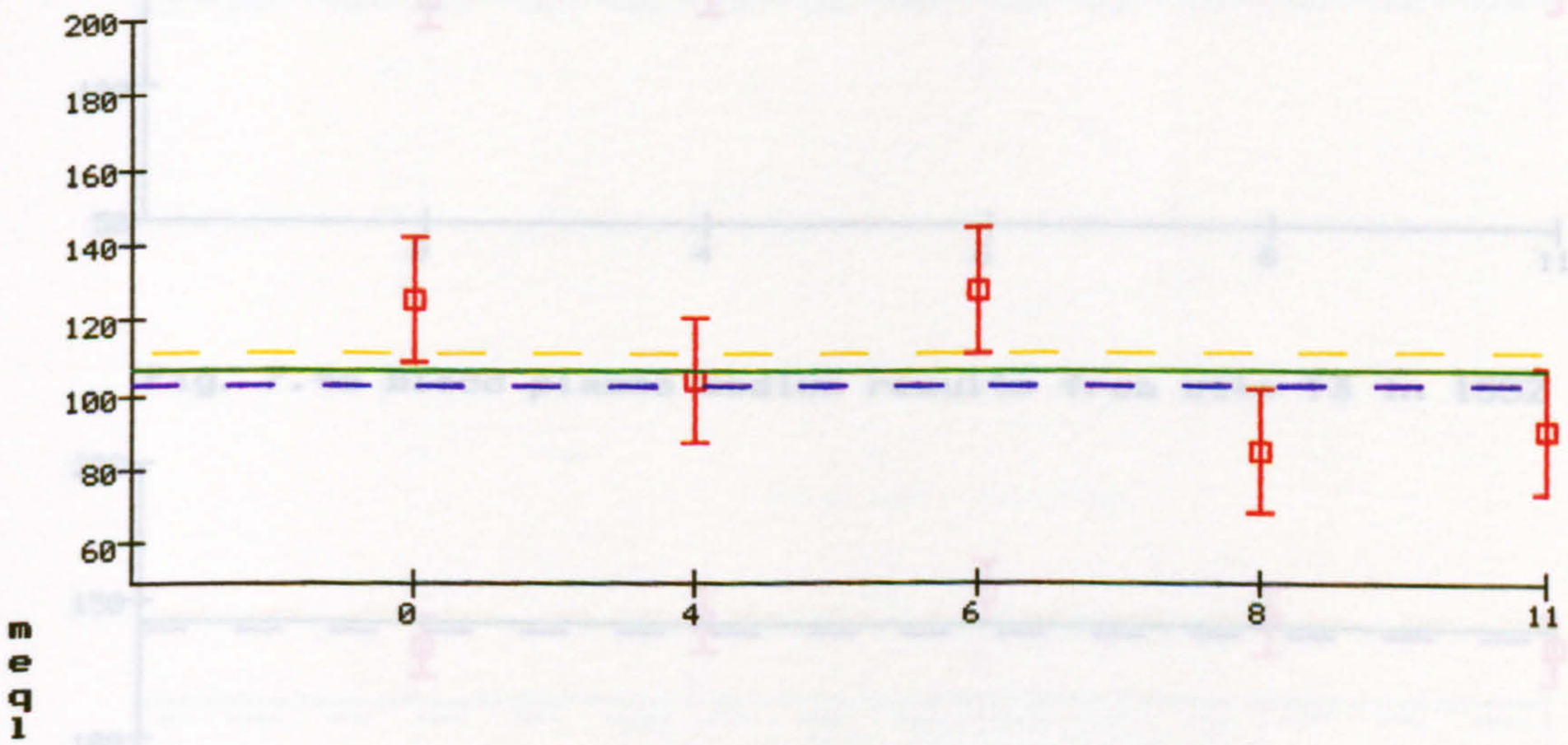


Fig. 7.4b Blood plasma sodium results from site GW in 1991

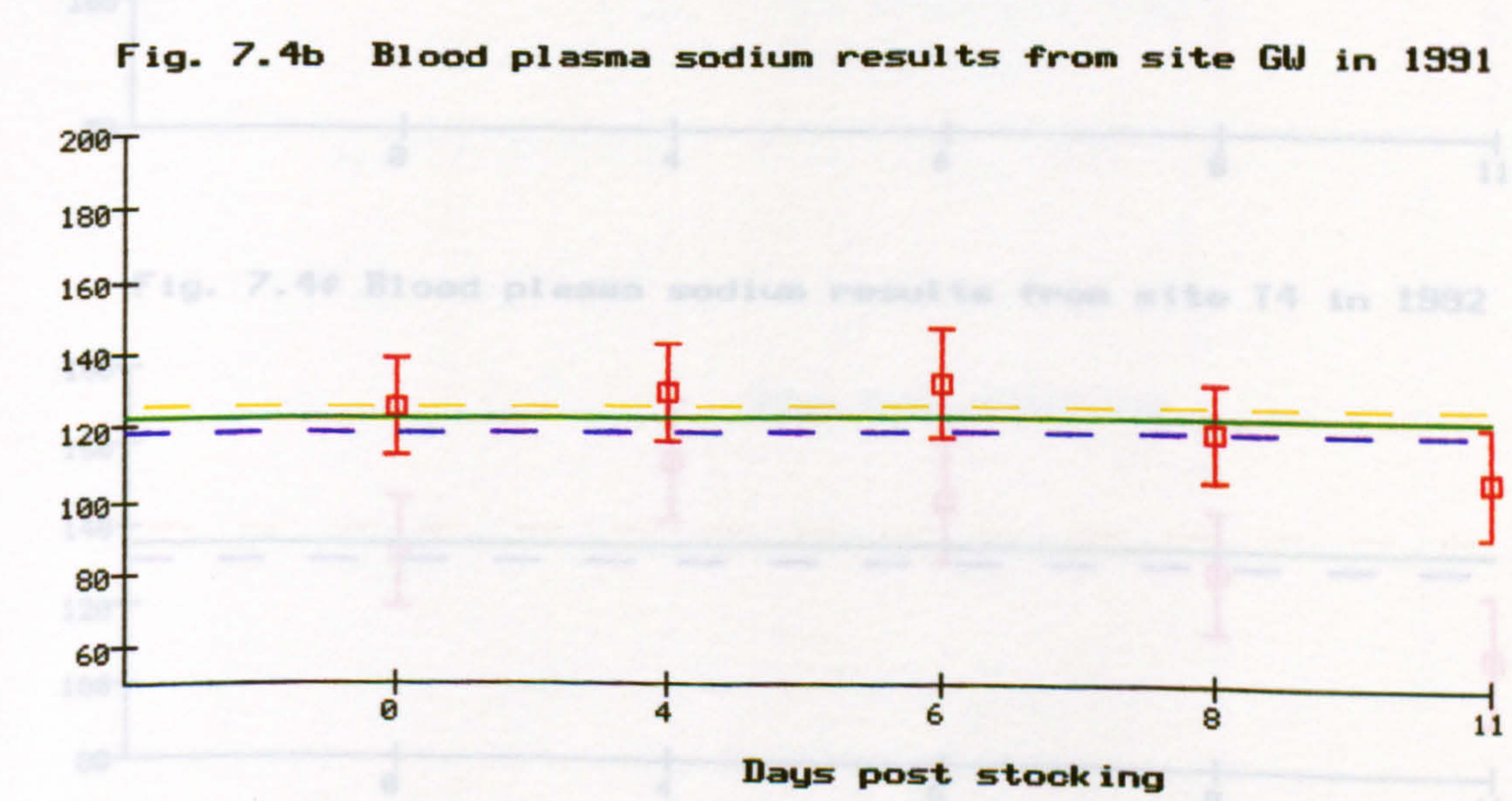


Fig. 7.4c Blood plasma sodium results from site T1 in 1992

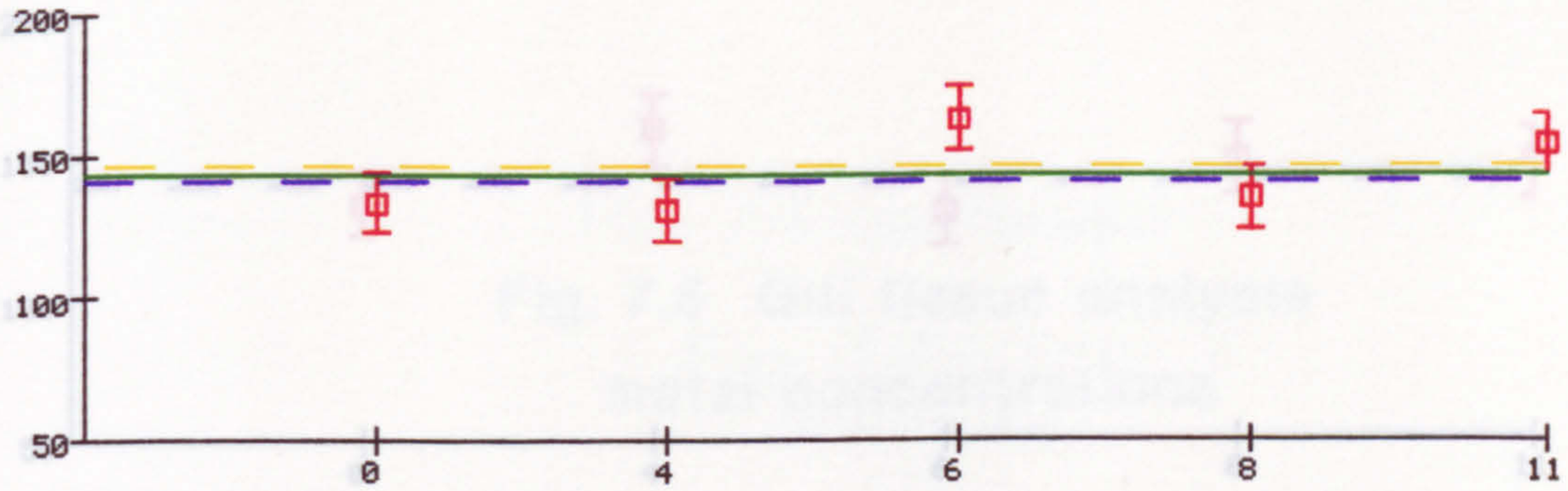


Fig. 7.4d Blood plasma sodium results from site T2 in 1992

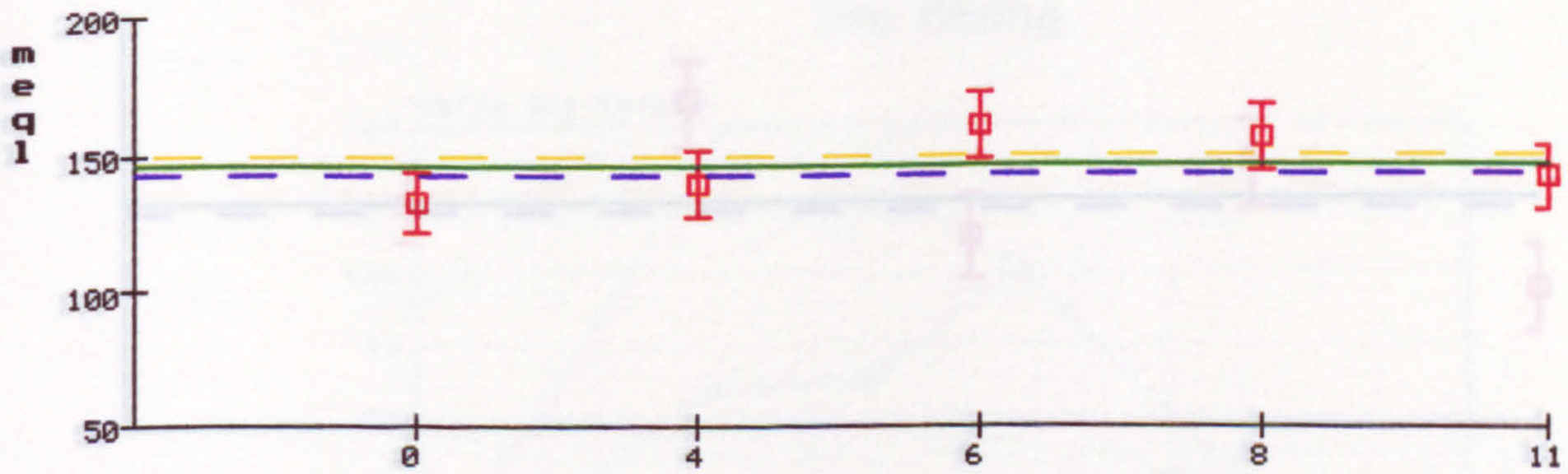


Fig. 7.4e Blood plasma sodium results from site T3 in 1992

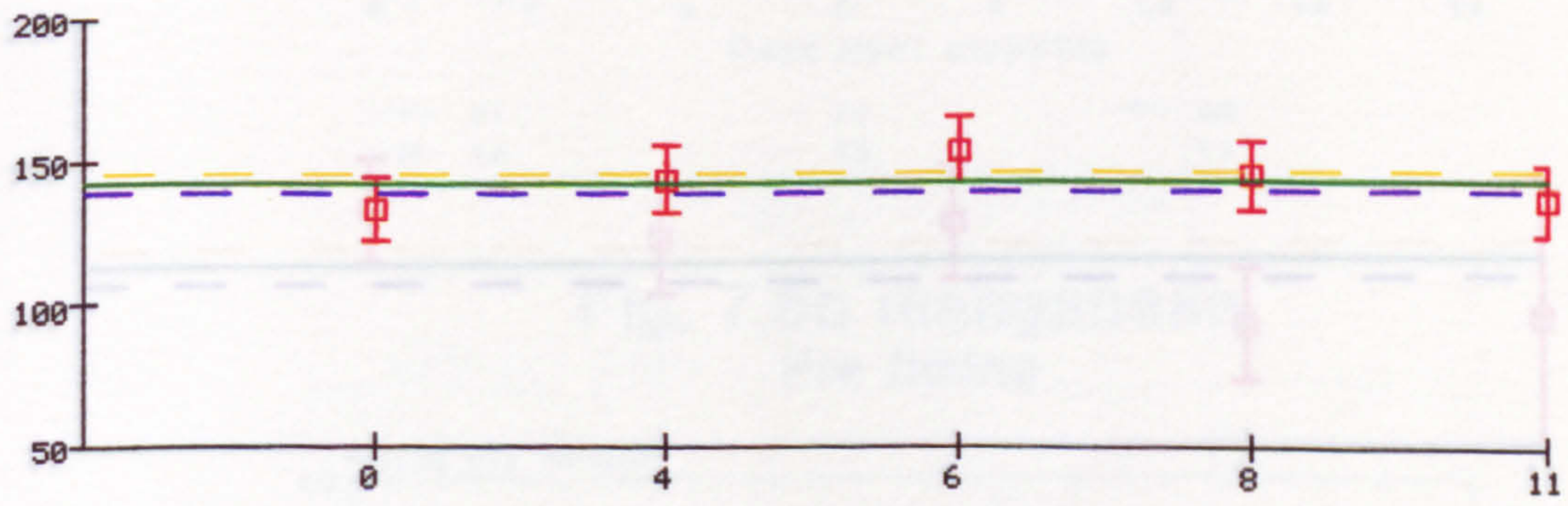


Fig. 7.4f Blood plasma sodium results from site T4 in 1992

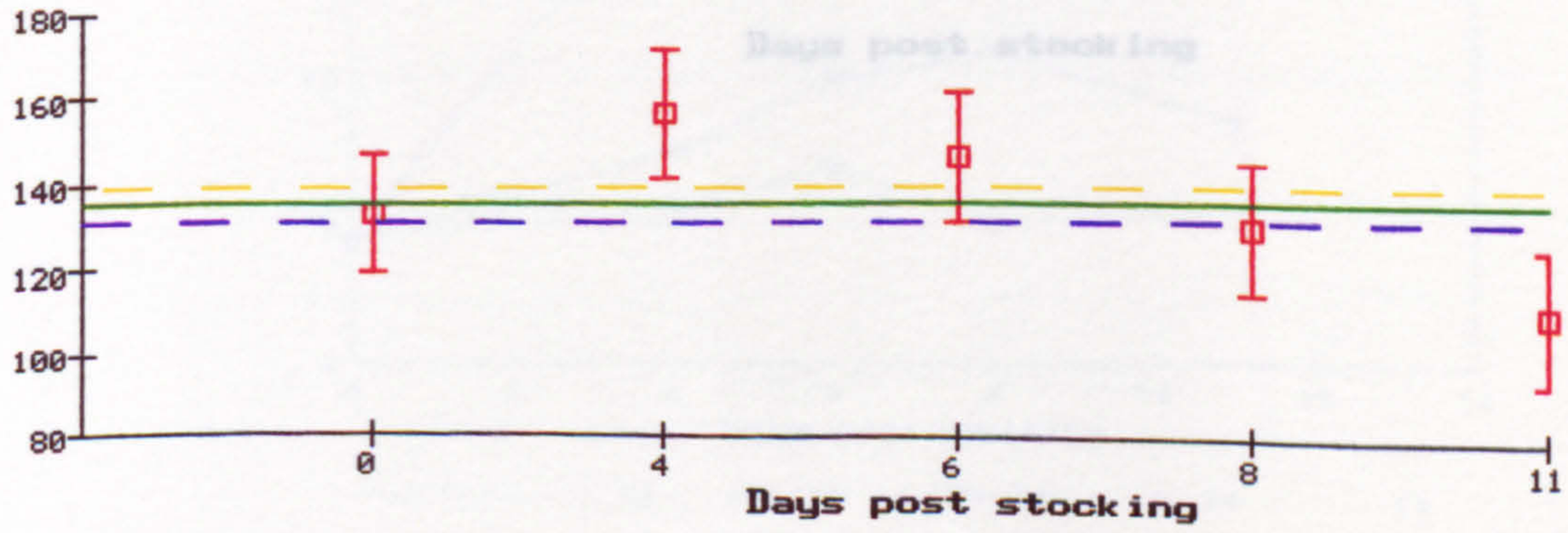


Fig. 7.4g Blood plasma sodium results from site T6 in 1992

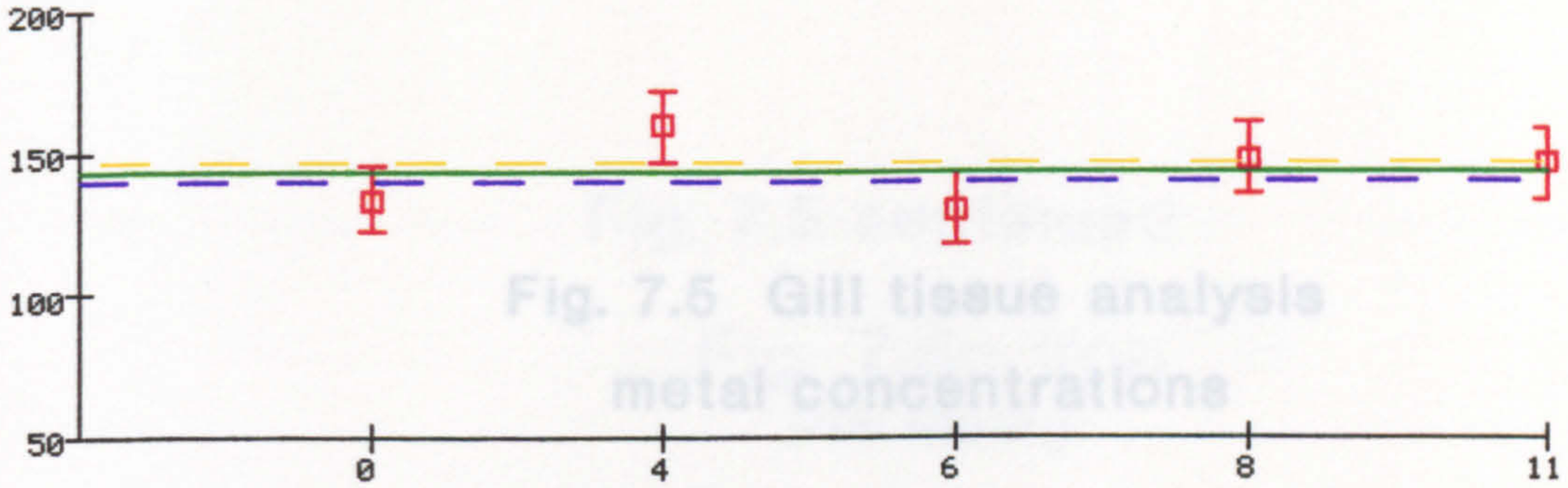


Fig. 7.4h Blood plasma sodium results from site GW in 1992

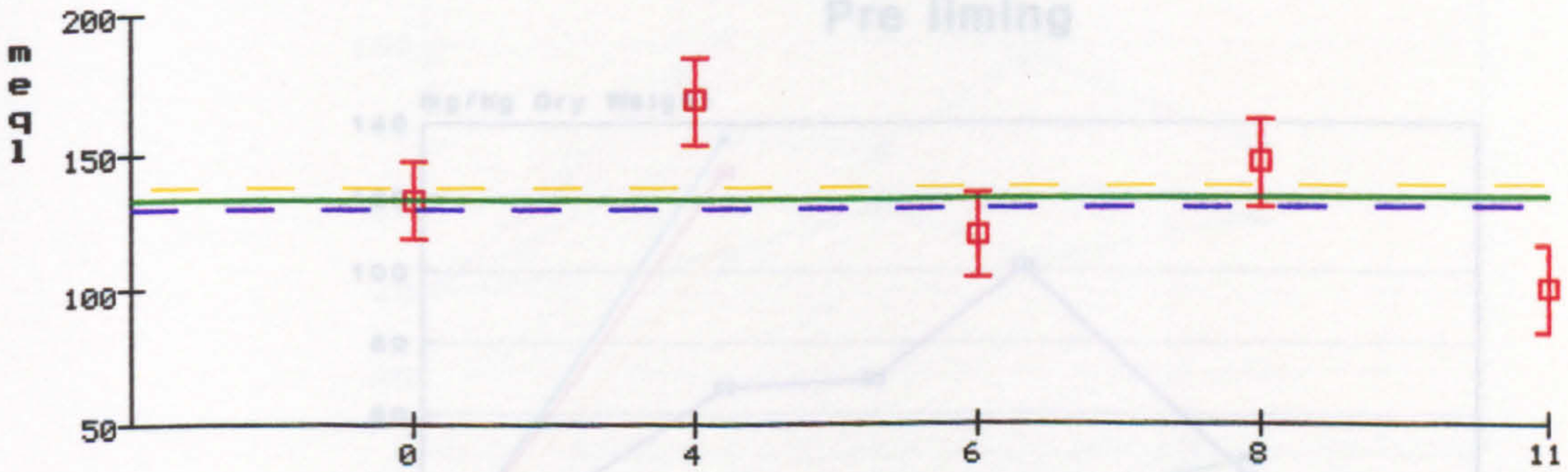


Fig. 7.4i Blood plasma sodium results from site DO in 1992

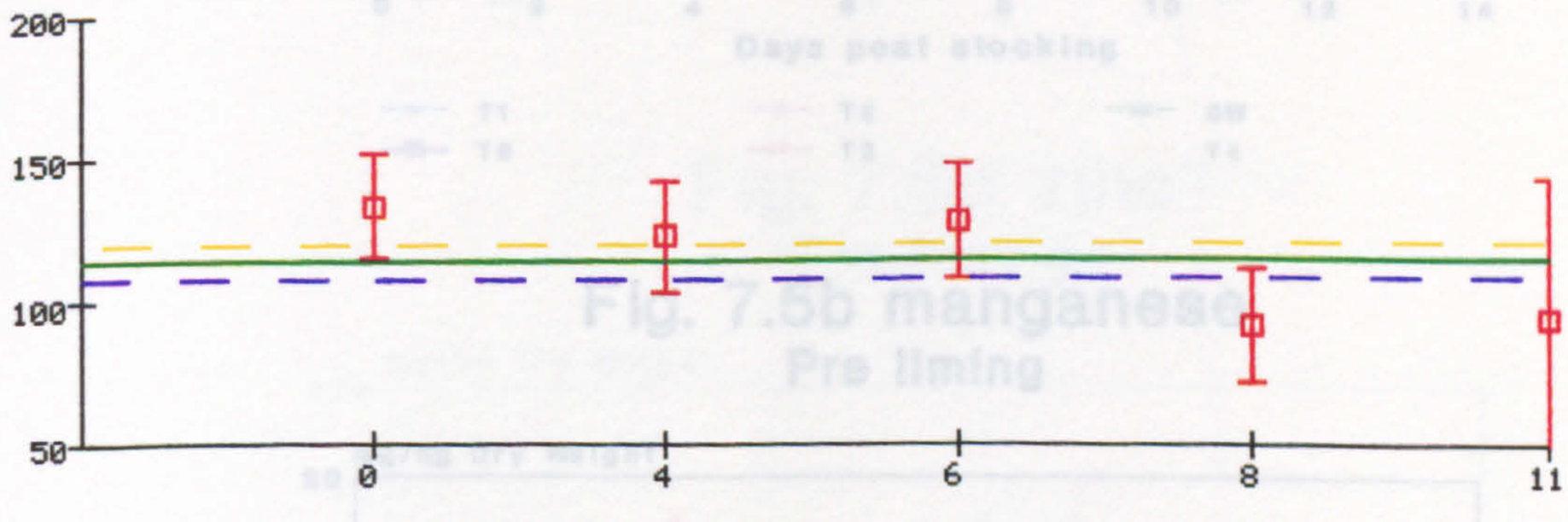


Fig. 7.5 Gill tissue analysis metal concentrations

Fig. 7.5a aluminium
Pre liming

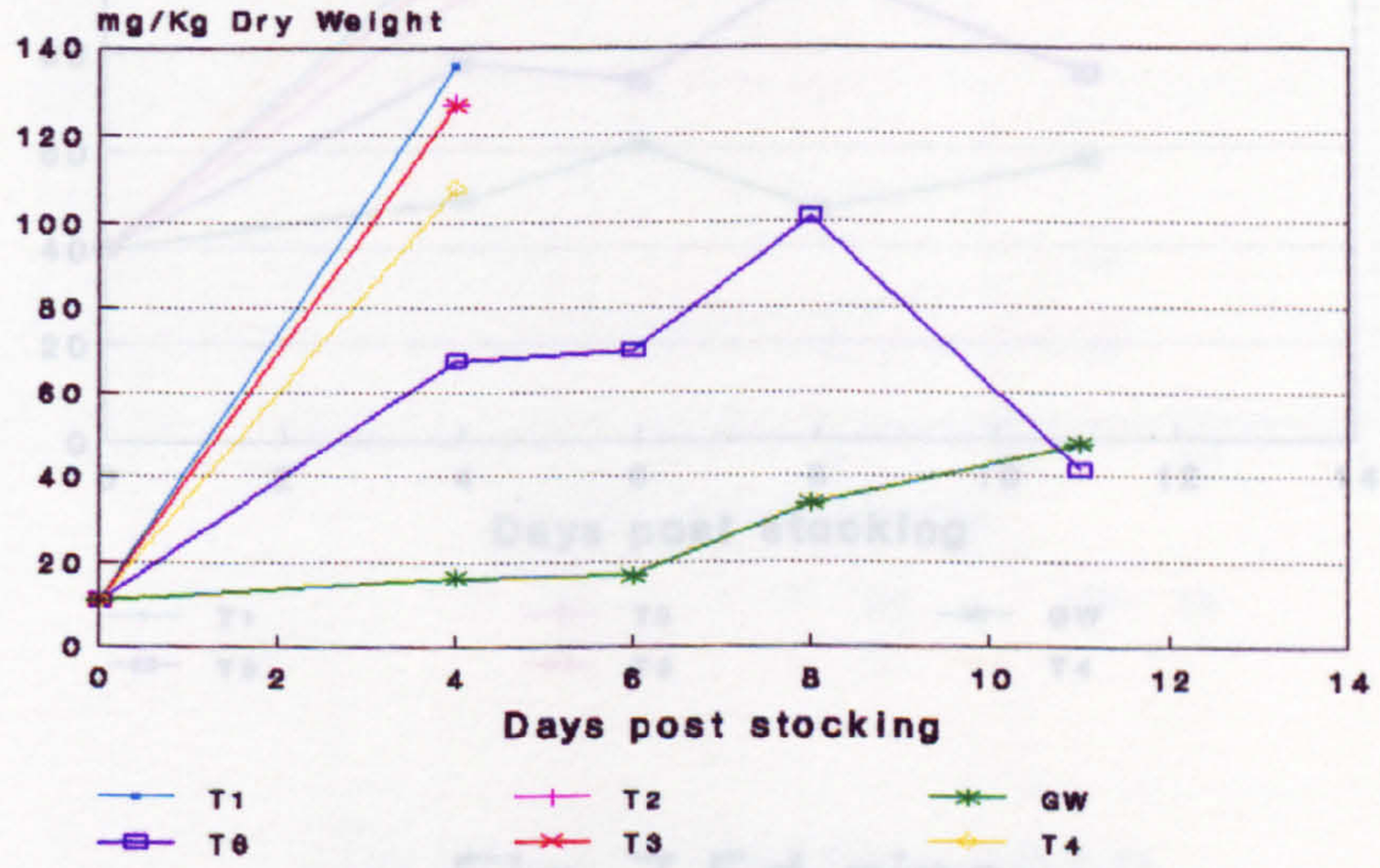


Fig. 7.5b manganese
Pre liming

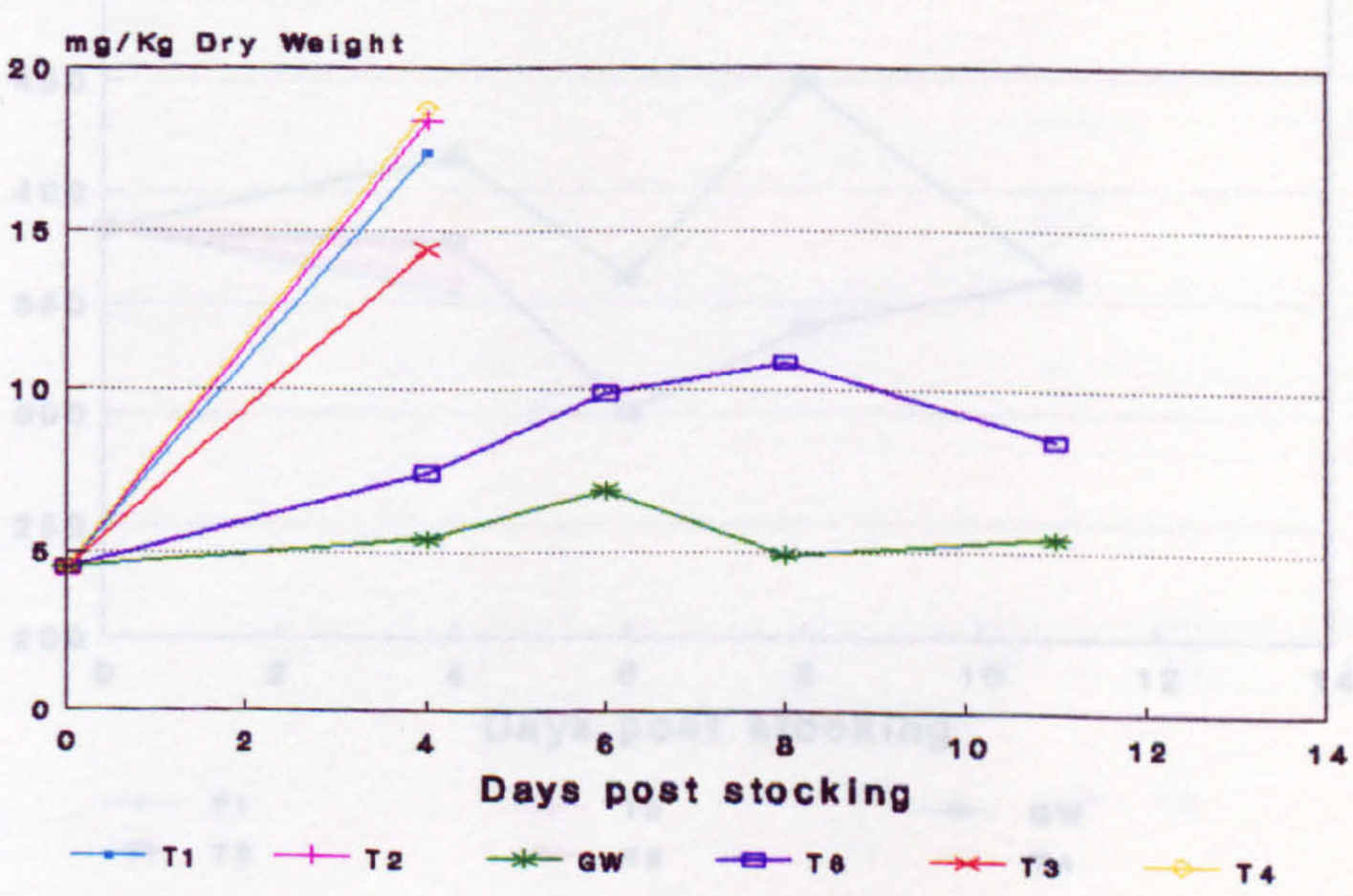


Fig. 7.5 continued

Fig. 7.5c iron
Pre liming

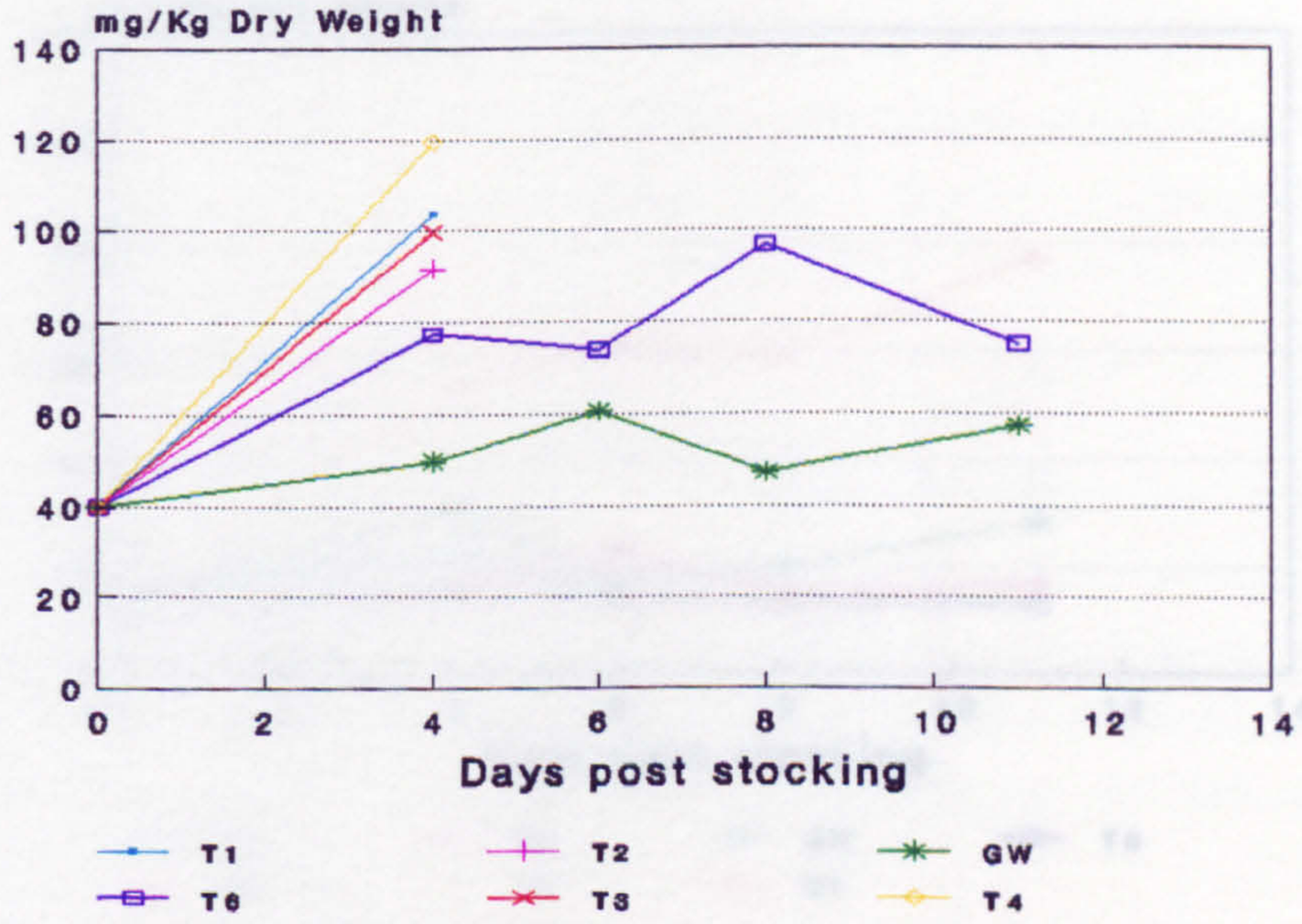


Fig. 7.5d zinc
Pre liming

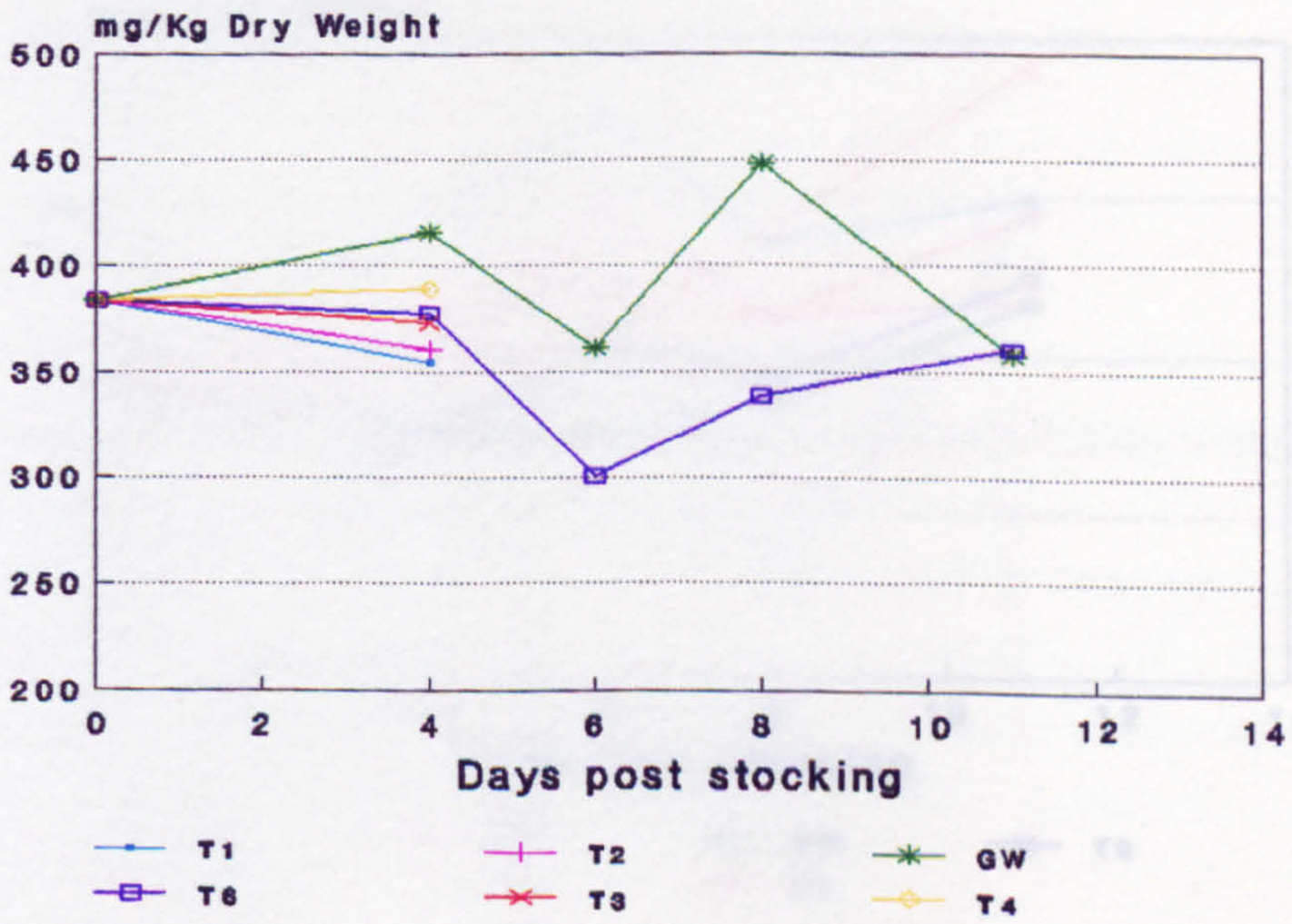


Fig. 7.5 continued

Fig. 7.5e aluminium
Post liming

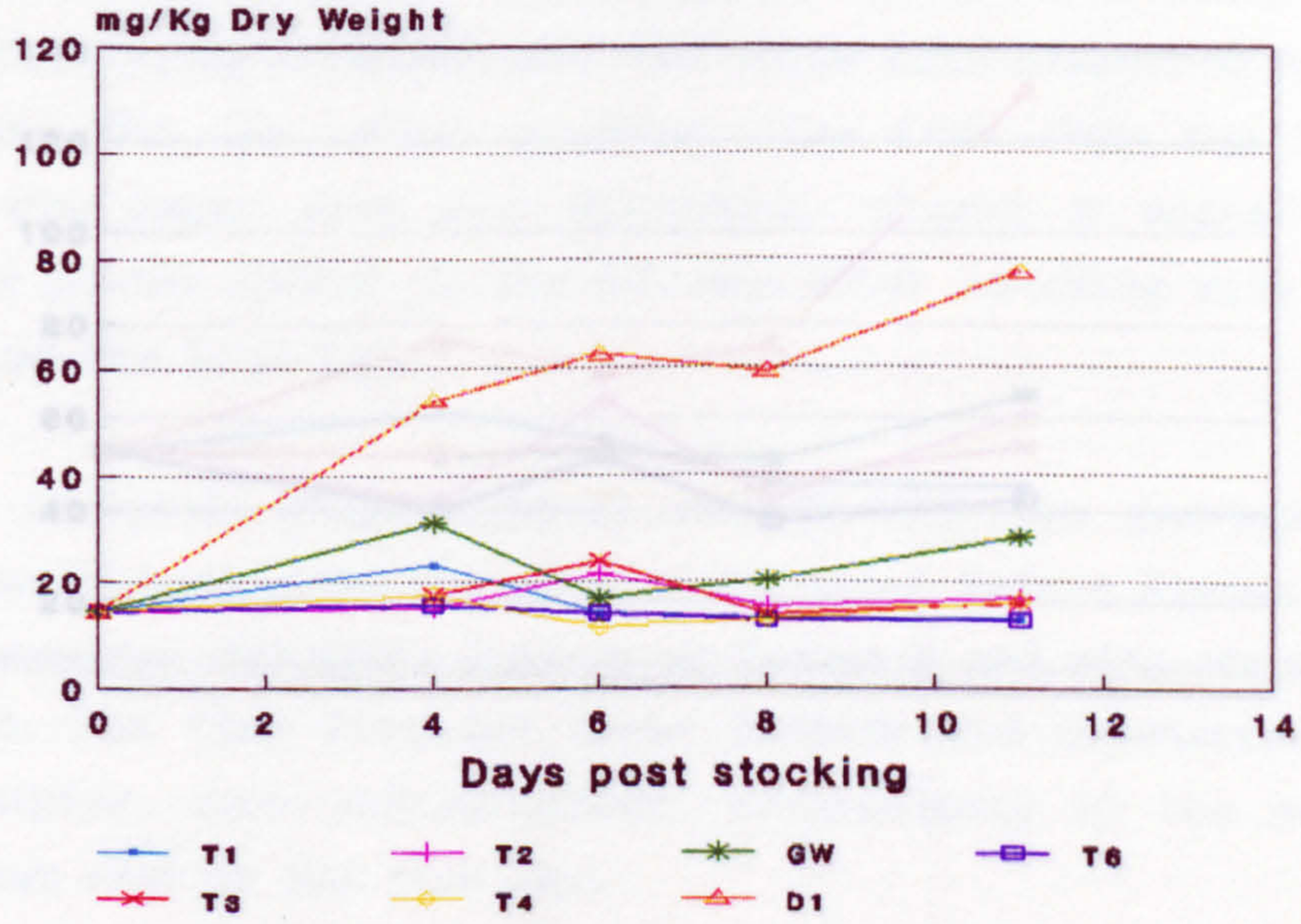


Fig. 7.5f manganese
Post liming

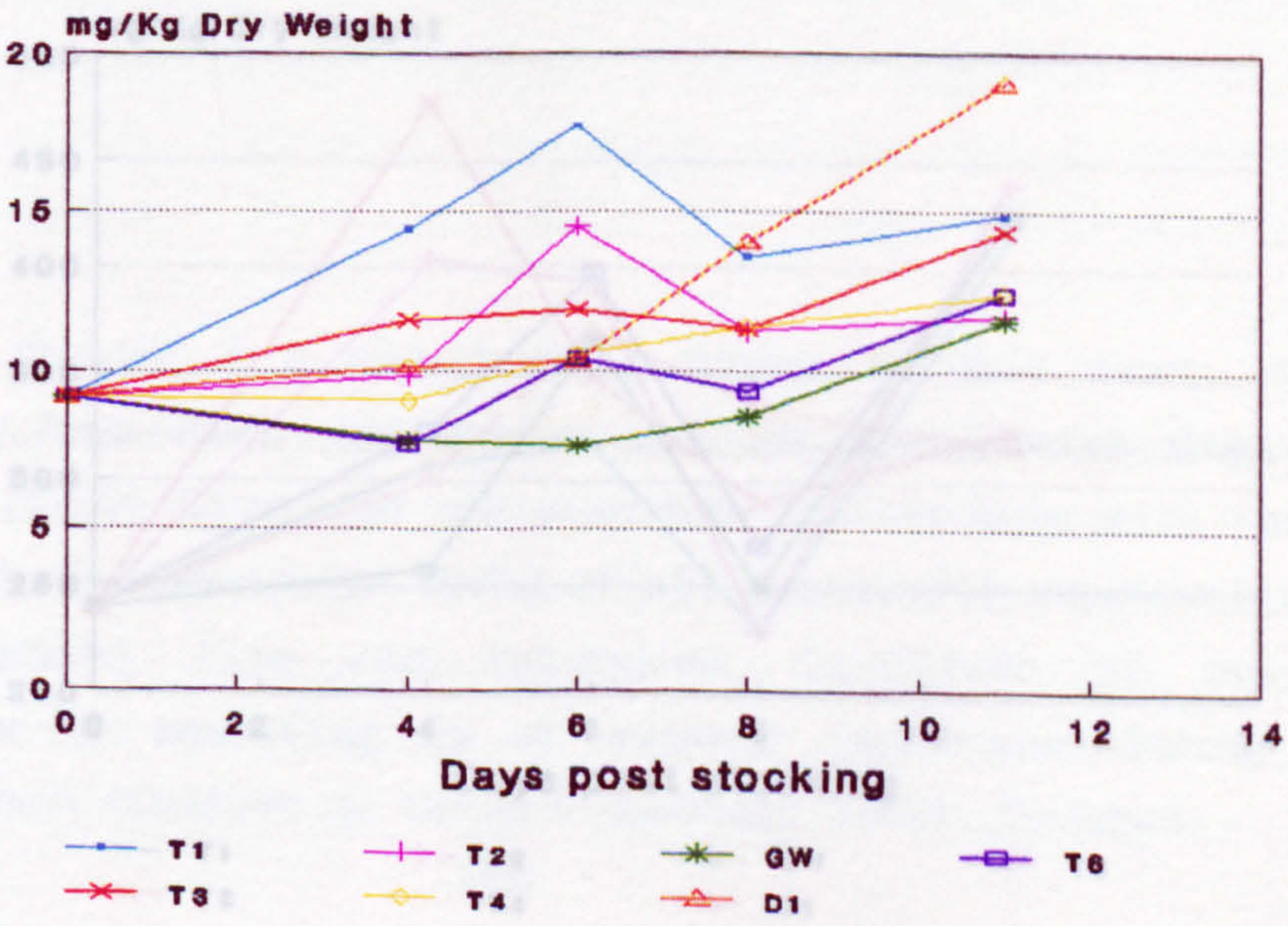


Fig. 7.5 continued

Fig. 7.5g iron
Post liming

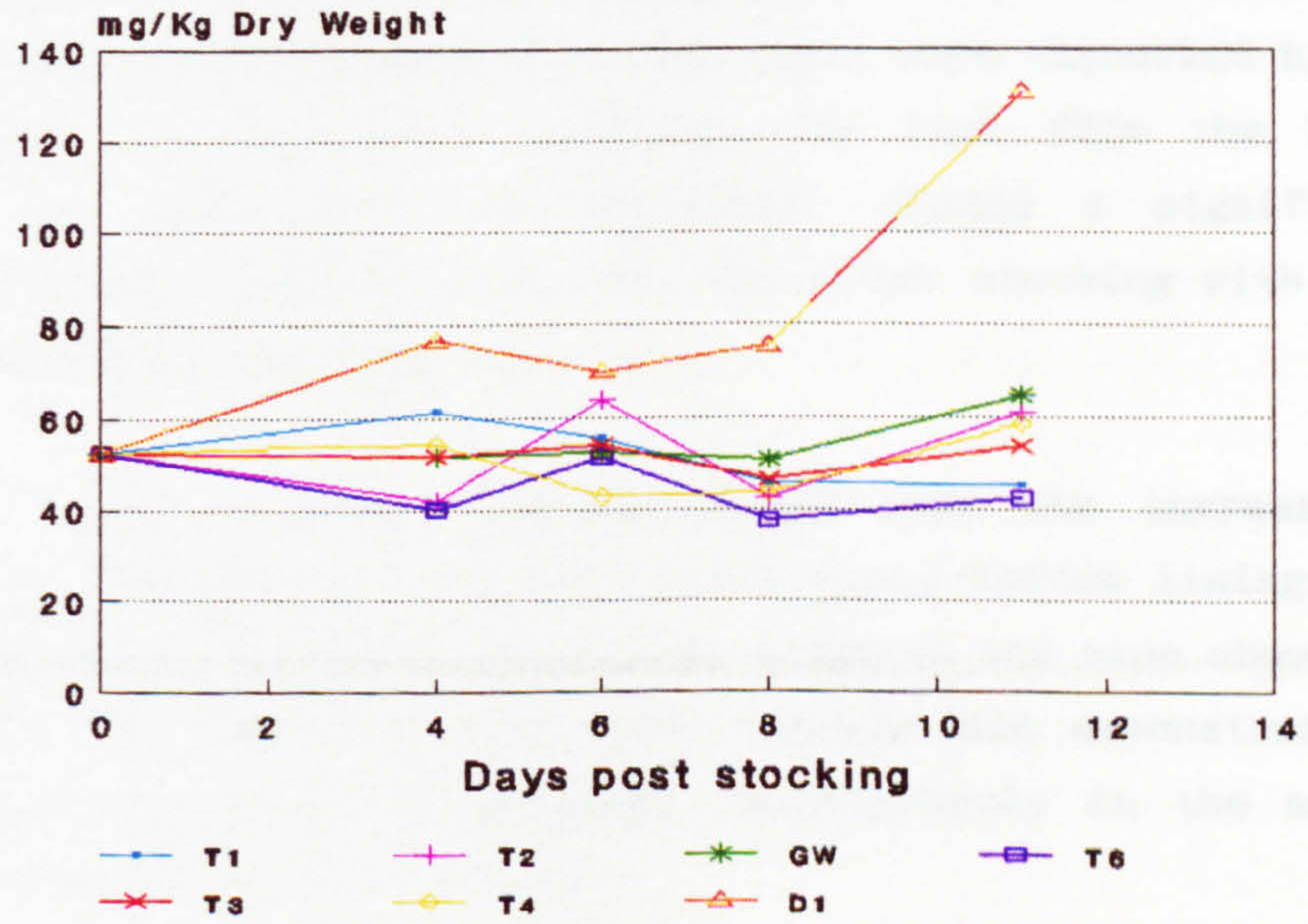
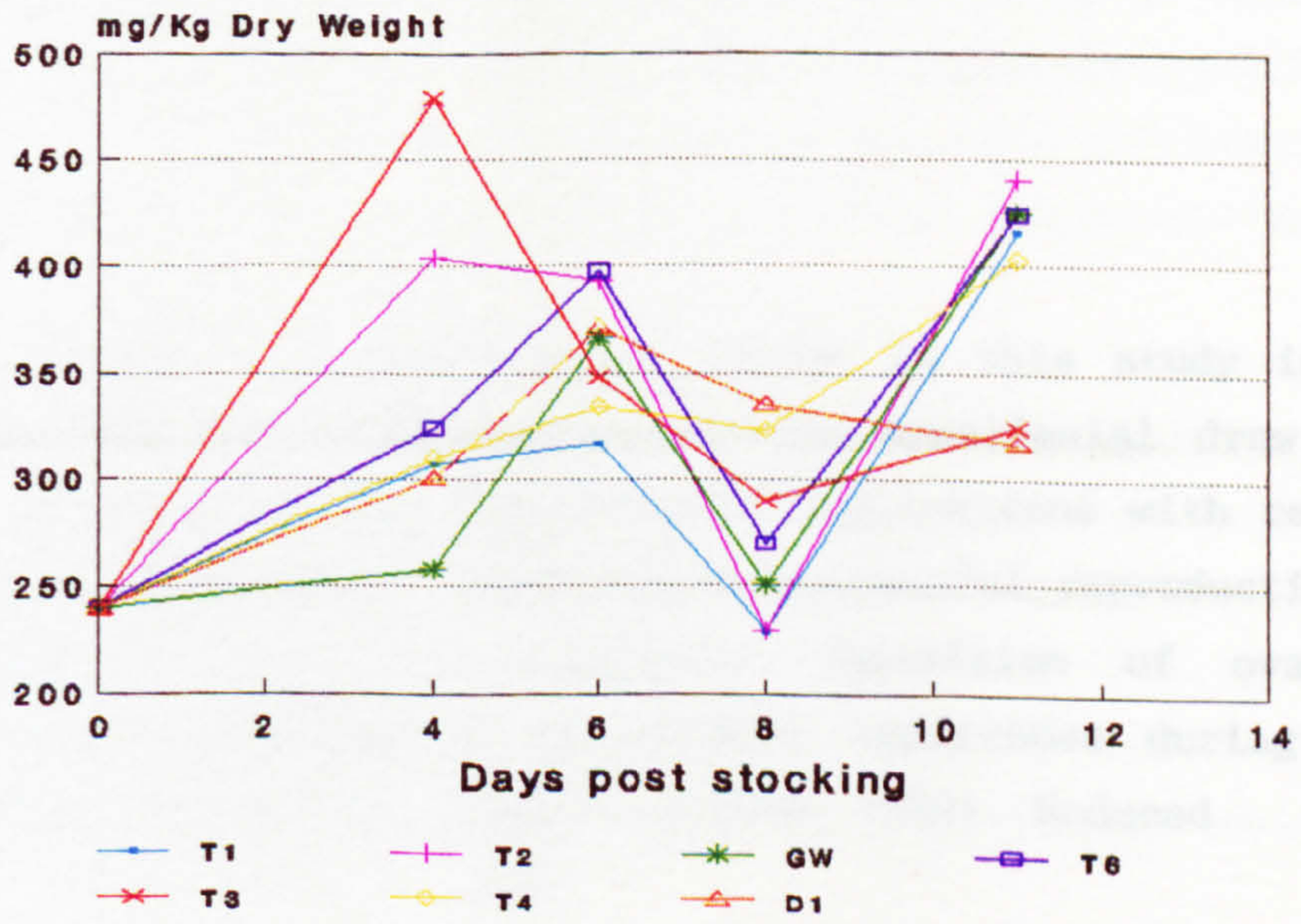


Fig. 7.5h zinc
Post liming



The post-liming period work was hindered by unstable flows during the 11 days, with rainfall on the 5th and 8th days after stocking. The flows on the 8th day were the highest recorded that month. River temperature was again low but some 2-3 degrees above those in 1991 (Fig 7.3c&d). The 11 day exposure resulted in all fish surviving at the sites where mortalities were observed pre-liming. On the last day of sampling 4 of the remaining five fish in the River Doethie were dead on inspection, following a rainfall event over the previous few days. The blood plasma results (Figs. 7.4 c-i) showed that no main river site displayed a significant decrease in blood plasma sodium during the exposure period. The graphs for sites T4 and GW show a decrease on the last day of sampling but these results should be disregarded since they could have been due to the fish being stressed when the cages were disturbed by the high flows on the 8th day after stocking. The fish from the River Doethie, where the cages were not disturbed, showed a significant decrease in blood plasma sodium on the 8th day after stocking with this level maintained on the 11th day.

The gill tissue analysis (Figs. 7.5e-h) showed that the increase in aluminium and iron displayed at the main river sites before liming were not repeated. Manganese did again display an increase and zinc showed no discernable trend. The fish from the River Doethie did demonstrate an increase in aluminium, iron and manganese, particularly in the sample containing the dead fish on the 11th day.

All fish kept in the transporting tank were alive after ten days in both years indicating no transportation induced mortality.

7.4 Discussion

7.4.1 Regulation

As discussed in Chapter 6 a complicating factor in this study is the possible effects from river regulation and the hypolimnial draw off. Wightman *et al.* (1990) discussed the possible implications with respect to the River Tywi. Sedimentation could affect successful reproduction by reducing intra-gravel flow and subsequent deposition of ova, if intra-gravel flow or upwelling is of primary importance during redd siting by the female (Chapter 3, Crisp & Carling, 1989). Reduced

intra-gravel flow could also reduce the viability of any deposited eggs.

The altered thermal regime in the River Tywi (Wightman *et al.*, 1990) was similar to the principal effects described by Cowx *et al.* (1987), viz increased residual flow, a change in the timing and magnitude of the variation of seasonal flow, altered seasonal temperature cycles and the depressing of summer and elevating of winter mean daily water temperatures. A number of implications arise from the altered thermal regime. The increasing of winter temperatures would result in the accelerated development of salmonid eggs since time to hatching is related to temperature (Crisp, 1981). The delaying in onset of temperature increase in the spring would affect subsequent feeding, more so for salmon than trout since salmon show cessation of feeding below 6-7^{o}C (Symons, 1979; Elliott, 1991) while trout feed at 4^{o}C and above (Elliott, 1975). The increased development of ova and the delay in first feeding could therefore lead to starvation of fry.

The growth of juvenile salmonids is also related to temperature (Elliott, 1975, Egglishaw & Shackley, 1985) and the decreased summer temperatures could therefore reduce growth potential and subsequently increase the age of smolting. This would result in decreased production from the impacted area but older smolts may display greater survivorship or different patterns of return (Chapter 2), and might offset some of the production loss.

The initiation of smolt migration by Atlantic salmon has been shown to be suppressed or delayed below 10-12^{o}C (Solomon, 1978). The delay in temperature increases in the spring would therefore affect the smolt migration in the upper Tywi and delay migration. This delay in migration could result in reduced migration success since estuary water quality may be poorer (Owen, 1988), and later migrating smolts have a tendency to migrate during daylight (Solomon, 1978) when bird predation may be higher.

River regulation, however, has been shown to be beneficial by some workers. Cowx *et al.* (1984) described how drought eliminated salmon fry in an un-regulated tributary due to thermal stress but the dampening of summer temperatures in a nearby regulated river prevented this. Crisp *et al.* (1983) described a 40% increase in trout density below Cow Green reservoir following impoundment, though this reservoir operates an

epilimnial release regime. As stated in Chapter 6, the regulating regime of the reservoir did not change from the pre- to post-liming period and therefore the effects from regulation were assumed constant in all years.

7.4.2 Electrofishing surveys

The pre-liming period electrofishing survey demonstrated low densities of juvenile salmonids for some 17 km below the reservoir. The absence of salmon in the first 12 km below the impoundment is consistent with the higher sensitivity of salmon to acidic conditions (Rosseland & Skogheim, 1984; Howells *et al.*, 1990). The highest tributary, the Ffin, was the only one of the four reference tributaries within the impacted area (Ffin, Doethie, Gwenffrwd and Gwenlais) which did not contain juvenile salmon. This may be due to its small size restricting entry to adult salmon (Clarke *et al.* 1987), since the high trout densities suggest no environmental perturbation and adult sea trout are known to utilise it for spawning. The detection of juvenile salmon in the River Doethie suggests that adult salmon therefore migrated through the impacted area at least to the River Doethie junction, and that in normal circumstances juvenile salmon would be expected from this point downstream.

Similar findings were reported by Clarke *et al.* (1987) for electrofishing surveys carried out in 1985 and 1986, with very low densities of juvenile salmonids in the main river below the reservoir (Appendix 7.3). Juvenile salmon were, however, present at low densities within some areas where they were absent in 1990, but water quality conditions have continually deteriorated since the 1985/1986 surveys (Rogers, 1990).

The first electrofishing survey after liming was carried out in August/ September 1991, six months after the first liming of the reservoir in February. The presence of salmon fry from T2 downstream, and especially the high densities observed at T5 and T6, suggests spawning had taken place in these areas. The adult spawning season is virtually complete by the end of February, indicating that the fry observed in 1991 resulted from ova deposited during the winter before liming. This indicates that successful adult spawning took place in the River Tywi prior to liming. Dispersal of fry from the redds has been noted (Gibson, 1993) and Harding (1986) (quoted in Solomon, 1993) observed maximum downstream dispersion of 743 m. The low numbers of fry observed at T2 and T3 may therefore have

been emigrants from the nearby tributaries (Doethie & Gwenffrwd).

Recruitment failure from reduced hatch or post-hatch mortality is therefore the suggested hypothesis for the absence of juvenile salmon in the year before liming. This is in accordance with the results of Skogheim & Rosseland (1984) and the literature in general (see section 7.1.1). Since liming took place in February 1991, around the expected hatching/post-hatching period, the increase in pH and calcium and the concomitant reduction in aluminium toxicity would have allowed for successful development of the 1991 year class.

Although trout fry were present in small numbers before liming they have been shown to be more resistant than salmon (Turnpenny, 1990) but the increase in density observed in 1991 may be attributed to similar mechanisms as described for salmon.

The presence of salmon parr from T2 downstream in 1991 needs explanation since no fry were discovered in 1990 to develop into parr. Gibson (1993), in his review, described how salmon parr showed both spring and autumn movements and that these movements were primarily in a downstream direction. The low numbers found in the main river in 1991 can probably be attributed to downstream migration from the nearby tributaries (Doethie, Gwenffrwd and Gwenlais) which were able to survive in the post-liming period water quality conditions within the main river. This may also account for the increase in trout parr in 1991, but a second reasoning may be that the main river trout fry in 1990 may have had increased survivorship due to liming in February, since February/March has been shown to be a particularly vulnerable time to aluminium toxicity (Sadler & Lynam, 1988). The large increases in densities observed in 1991 in both juvenile salmon and trout were not duplicated in the tributaries, and since no physical changes in site habitat were observed these increases were attributed to improvements in water quality as a result of liming.

The similar findings in the 1992 electrofishing survey indicate conditions were maintained in the second year. The only notable change from 1991 was the increase in salmon parr. This would be expected from the detection of fry in 1991 surviving to 1992, but a general increase in parr densities in the tributaries indicates that these may have been natural increases, and that 1992 was a good year for parr numbers.

No salmon were found post-liming at T1 but this site is upstream of any tributary containing salmon and thus recolonisation may take longer. T1 is also the closest to the reservoir and would therefore have a greater impact from any effects of impoundment and the hypolimnial release regime. Since trout fry were present at T1 any sedimentation effects do not seem enough to prevent recruitment.

A key management question arising from this work is what level of increased juvenile salmonid production will result from rehabilitation. The historical carrying capacity before acidification is unknown but the surveys in 1985/1986 (Clarke *et al.* 1987; Appendix 7.3), when impact from acidification was evident, showed maximum salmon fry, salmon parr, trout fry and trout parr densities of 35.0, 12.0, 10.7 and 5.6 fish per 100 m^2, respectively. All these maxima, except the salmon parr, have been exceeded in the first two years after liming.

Historically, the main river around sites T3-T4 was renowned for salmon spawning (M.Todd-Tywi head bailiff, pers. comm.) and therefore the current fry densities of <10 fry /100 m^2 (Fig. 7.2, Table 7.1) seem low. The potential carrying capacity of the rehabilitated area will be better estimated after a number of years, when adults from the increased number of parr observed in this study will have returned to spawn (1994-1996), assuming intra-river system homing. A hypothetical estimation of increased production from the rehabilitation is given in Appendix 7.5.

A second assessment of potential production could be made by comparing juvenile densities with those of other rivers, preferably those with similar characteristics as the Tywi. This would require rivers regulated by impoundment, of similar physical characteristics and with salmonid populations of similar traits viz. similar smolt composition and balance of salmon to sea trout. In reality finding similar rivers is difficult and a further complication is the width of the sites sampled.

It was stated earlier that comparison of the main river densities with those of small (<10 m) rivers may not be valid due to variation in physical characteristics such as percentage of the site with bankside cover. Brittain *et al.* (1993) demonstrated that the centre section of a river had lower fish densities than along the bankside. With sites of the width in this study (20-30 m), quantitative sampling requires high resources of manpower and so little comparable data are available.

However, the NRA RJSMP did sample sites of equivalent width to those in this study (Appendix 7.4) between 1990 and 1992, but without nets and with one run only. Minimum estimates were therefore obtained. The unknown probability of capture (p) in the above surveys and the allowance for the densities being minimum estimates allows only a tentative comparison with densities in this study. The RJSMP results (Appendix 7.4) show a general low level of abundance in sites of this width, with trout densities being particularly low. Apart from the results from the Wye (primarily a salmon river) the salmon densities are also low and are probably comparable with those obtained on the River Tywi.

7.4.3 Physiological surveys

The pre-liming period caged fish experiment demonstrated the acute toxicity to the prevailing water quality to hatchery 1+ brown trout, for at least 9 km below the impoundment. Mortality and metal accumulation on the gills has also been found by other authors during caged-fish experiments (Andersson & Nyberg, 1984; Stoner *et al.*, 1984; Lacroix & Townsend, 1987; Weatherley *et al.*, 1990)). Booth *et al.* (1988) showed 79% mortality of brook trout occurred within 2 days when exposed to pH 5.2, 0.333 mgl^{-1} Al and 0.25 mg^{-1} 1- Ca. Aluminium levels in the gills of the dead fish were 837 $mgkg^{-1}$ ww which equates to 167 $mgkg^{-1}$ dw at a conversion factor of 20% (Anderson & Nyberg, 1984). This is similar to the c.130 $mgkg^{-1}$ dw observed in this study. The gill metal concentrations were also similar to those observed by Lacroix & Townsend (1987) but lower than reported by Weatherley *et al.* (1991) or McCahon *et al.* (1987). This may be due to lower exposure periods but also possibly effects from gill preparation. This study used the complete gills whereas the latter two studies may have removed and analysed the lamellae only, which would not include the gill arch cartilage and therefore account for the higher concentrations.

The distended opercula and mucous production found in the dead fish would suggest death resulted from respiratory failure and hypoxia, which is in agreement with literature results at pH c.5.3 and high aluminium (>0.2 mgl^{-1}), as described earlier (Section 7.1.2). The accumulation of iron and manganese on the gills show that consideration needs to be given to these metals and that these could have acted synergistically with aluminium (Andersson & Nyberg, 1984).

Despite the fish at T6 not showing acute toxicity some effect was evident from the decrease in blood plasma Na^+ concentrations suggesting further exposure may have been detrimental. The pH was >6 at T6 during the experiment and little effect would be anticipated, but the aluminium present (c. 0.15 mgl^{-1}, total) may have been in a more toxic form due to the very low water temperature observed, which alters aluminium speciation (Lydersen, 1990). The hypothesis that T6 may be around the limit of impact of acidification to fisheries during the pre-liming period conditions is tenable from these results. This site represents the upstream limit of salmon fry in the pre-liming period electrofishing survey.

The low numbers of brown trout found downstream of the reservoir prior to liming seems to contradict the complementary physiological experiment, which showed acute toxicity to hatchery brown trout in this area. The caged fish experiments, however, should be viewed as the worse case and there are several possible explanations for this. The hatchery fish could have been more susceptible to acidic conditions than the native fish, a phenomenon reported by Andersson & Nyberg (1984), since wild fish may have slight morphological adaptations or exhibited acclamation (Karlsson-Norrgren *et al.*, 1986a; Wood, 1989). The pre-liming period experiment took place in February which is a period when aluminium may be particularly toxic (Sadler & Lynam, 1988). This is possibly due to seasonal variation in tolerance of fish to factors such as temperature and energy reserves (Gunn, 1986). The water temperature at the time of the experiment (0-4^{o}C) was very low and this has been shown to result in advanced gill lesions with aluminium and pH 5.5 compared to 15^{o}C (Karlsson-Norrgren *et al.*, 1986b). The hatchery fish were transferred directly from their ambient conditions (c. pH 6.2, <0.004 mgl^{-1} total aluminium) to the River Tywi and this rate of change in conditions may be important in the acute response observed (Reader & Dempsey, 1989). There is also the distinct possibility that some of the brown trout sampled in August/September may have migrated from the tributaries and been able to inhabit the main river during summer when conditions were more favourable.

Liming has been shown to reduce toxicity to fish from acid, aluminium conditions by many workers (Reddum *et al.* 1986, Rosseland *et al.*, 1986; Underwood *et al.*, 1987; Rosseland & Hindar, 1988; Weatherley, 1988; McCahon *et al.*, 1989; Degerman & Appelberg, 1992; Lacroix,1992) and allow

the recovery of fish populations (Nyberg, 1984; Turnpenny, 1990).

The post-liming period caged fish experiments complement the electrofishing surveys in that water quality after liming was not toxic to hatchery brown trout, since there were no mortalities and blood plasma sodium and gill metal levels remained stable throughout the exposure period. This is in accordance with Howells *et al.*(1990) who stated that there is little concern for effects from <1 mgl^{-1} of aluminium when pH>6.5, and Reader *et al.* (1989) who found no effect from nominal concentrations of eight trace metals at pH 6.5.

The decrease in blood plasma sodium at T4 and GW was probably due to the cage disturbance from high water, which could alone stress fish. However the increased exertion exhibited during swimming behaviour in increased flows, as would have occurred in this situation, has been shown to duplicate the response of acidic conditions in reducing blood plasma sodium levels (Wood & McDonald, 1982).

The brown trout in the River Doethie (D1) did not display acid stress until the rainfall events preceding sampling on the 8th and 11th days post-stocking. No immediate mortalities were observed from the first event and plasma sodium levels may have stabilised if it were not for the second event three days later. Reader & Dempsey (1989) similarly described how multiple episodic exposure may be detrimental to fish.
The primary concern for episodic acidity from the River Doethie is the possible toxicity to fish in the main river if the limed water cannot buffer the acidic water. The caged fish at T2, which is some 500m below the River Doethie confluence, diplayed no signs of stress that killed the fish at D1 in the post-liming period experiment. The buffering capacity (see Chapter 5) appears adequate in this case despite the reservoir water being at its lowest pH and calcium level due to winter dilution. There could, however, be possible toxicity in the immediate proximity of the mixing zone at the confluence with the limed River Tywi, due to rapid changes in aluminium speciation or precipitation (Weatherley *et al.*, 1991; Rosseland *et al.* (1992). The possibility of toxicity may be reduced if the calcium concentration remains high in the mixing zone (Rosseland *et al.*, 1986). Fish may also reduce any detrimental effects by employing avoidance behaviour to potentially toxic areas (Nakamura, 1986; Gunn, 1986) although this would not be possible for immobile life stages such as eggs.

The input of zinc into the River Tywi in the vicinity of the disused mine water discharges (Chapter 5) is unlikely to be detrimental to fisheries in the main river on a large scale. This is attributed to the magnesium levels which give a molar ratio of 1:13.5 zinc to magnesium ($0.2\ mgl^{-1}$ zinc, $1\ mgl^{-1}$ magnesium) which should protect salmonids (Zitco & Carson, 1976). Magnesium levels were implicated as modifying toxicity in the River Ystwyth (WWA, 1980) where salmonids survived to two years in 95 percentile zinc concentrations of $0.6\ mgl^{-1}$.

Due to the nature of the study period, only one years pre-liming electrofishing data was available from the main-river sites sampled. Therefore, an assessment of annual variation in juvenile densities before and after liming was not possible. However, the physiological experiments incorporated into the study design enabled a direct demonstration of the removal of the effects of the pre-liming water quality conditions. These experiments, together with the numbers of juvenile salmon observed, support the hypothesis that the increased numbers of juveniles observed was due to liming.

To conclude, both the electrofishing and physiological data suggest that the water quality from the reservoir before liming was impacting some 17 km of the river below the impoundment. Recruitment failure is hypothesised for the absence or low juvenile salmonid populations observed. Liming has ameliorated this toxic effect such that numbers of juvenile salmonids, to the parr stage, have shown a dramatic improvement and caged fish now survive. Natural progression to returning adults is therefore expected from these juveniles. The electrofishing results have indicated the importance of the rehabilitated area to the loss of juvenile rearing habitat during pre-liming conditions. A better estimation of the increased juvenile production will require electrofishing surveys in 1995/1996 and beyond.

The NRA will need to assess the timing of re-liming (end of March), with lowest pH and calcium levels after the winter dilution, since this coincides with the egg hatching/post-hatch period and the beginning of the smolt migration. The smolt stage may be particularly susceptible, especially to acidic episodes from the River Doethie, and no facilities or information currently ensure our knowledge of their survival. Certain operational difficulties may be apparent for earlier liming and there are no reports of any effects on smolts in the two years after liming.

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Appendix 7.1 Bank side assessment of the substrate composition (%) at mainriver electrofishing sites (September 1990).

Site	Depth cm	Substrate composition Bedrock	Boulders /Cobbles	Pebbles /Gravel	Sand	Silt
T1	33	10	25	65	0	0
T2	39	0	70	30	0	0
T3	39	0	60	40	0	0
T4	30	0	70	25	4	1
T5	32	0	68	30	2	0
T6	28	0	50	45	5	0
T7	45	10	40	48	2	0
T8	40	0	20	80	0	0
T9	33	0	25	69	5	1

Note. The depth given is the mean depth of 3 measurements taken from a single transect of the river.

Appendix 7.2 Details of location and dates for the mainriver electrofishing surveys.

Site	NGR	Mean W(m)	Mean A(m^2)	Date sampled		
T1	SN785472	19.7	677	30/8/90	22/8/91	19/8/92
T2	SN773460	11.2	535	5/9/90	20/8/91	21/8/92
T3	SN768446	24.3	669	6/9/90	3/9/91	18/9/92
T4	SN778435	23.0	698	7/9/90	21/8/91	11/9/92
T5	SN765398	28.1	684	13/9/90	20/8/91	30/9/92
T6	SN762360	31.2	771	30/9/90	4/9/91	17/9/92
T7	SN762348	18.6	836	4/9/90	NS	NS
T8	SN735319	26.5	1189	3/9/90	13/9/91	NS
T9	SN687266	36.0	1313	10/9/90	20/9/91	NS

Note. Width (W) and area fished (A) are the means from sampling years combined

Appendix 7.3 Juvenile salmonid densities ($N/100m^2$) from the Upper River Tywi main river during 1985/1986. Source - Clarke *et al.* (1987)

	Site Code T1	T2	T3	T4	T5	T6	T7
Salmon 0+							
1985	0	0	0	2.6	NS	NS	NS
1986	0	0	NS	0.8	16.0	6.0	35.0
Salmon >0+							
1985	0	0.9	0	0.7	NS	NS	NS
1986	0	1.0	NS	2.0	12.0	9.0	4.0
Trout 0+							
1985	2.9	10.7	7.1	9.0	NS	NS	NS
1986	0.2	5.0	NS	8.0	10.0	7.0	6.0
Trout >0+							
1985	2.0	5.6	1.6	1.7	NS	NS	NS
1986	5.00	3.0	NS	1.0	5.0	3.0	0.6

Note Although site codes are identical to the codes used in this study they may not represent the exact same sampling locations and hence are not directly comparable for absolute densities.

Appendix 7.4 Range of minimum densities ($N/100m^2$) of juvenile salmonids from the 1990-1992 RJSMP where site width >18 m.

Site (RJSMP no.)	Width (m)	Salmon 0+	Salmon >0+	Trout 0+	Trout >0+
Conwy (24)	35.2	9.3-12.14	0.1-0.2	0.29-1.6	0-0.43
Seiont (15)	18.9	6.3-15.0	1.7-4.8	0.2 -2.0	0.8-1.3
Seiont (13)	30.0	8.5	1.7	1.9	0.9
Wye (27B)	20.1	49.7-83.6	2.6-3.8	0-2.4	0
Elan (65) (regulated)	26.0	4.7-34.6	0.2-1.8	0-0.7	0-1.1
Dwyrid (1)	23.0	8.2	0	3.8	0

Note The above densities are minimum estimates from fishing the sites without stop-nets and with one run only.

Appendix 7.5. A calculation of the possible adult salmonid production from the rehabilitation of the upper Tywi

A range of the benefits to the fishery were estimated from probability calculations using a Monte Carlo type analysis (Weatherley, pers.comm.; Snedecor & Cochran, 1979). The estimate of increased numbers of salmon and sea trout returning to the River Tywi were computed from the potential juvenile production rehabilitated by liming. A modest density of 5 1+ parr/100 m^2 was used for each species in the absence of potential capacity from the main river. The variables affecting the numbers of returning adults were given means and standard deviations from either the literature or personal communications (Tables 1-2). Each variable was assumed to be normally distributed and 50 random values were generated from each distribution. Thus 50 estimates of the increase in adult salmonid numbers were obtained.

Table A2.1. Variables used to calculate increased returns of salmon.

	Mean	s.d.	95% range
Area (00s m^2) (1)	4368	223	3931 - 4805
Parr - smolt survival (2)	0.50	0.076	0.34 - 0.65
Smolt - post smolt (2)	0.63	0.061	0.50 - 0.75
Post smolt - adult (2)	0.50	0.076	0.34 - 0.65
Coastal exploitation (2)	0.25	0.076	0.10 - 0.40
Within season (3)	0.70	0.102	0.50 - 0.90

(1) Assumed 10% error
(2) Kennedy (1988)
(3) Assumed covered.

Table A2.2 Variables used for estimating increased returns of sea trout

	Mean	s.d.	95% range
Area (00s m^2) (1)	4368	223	3931 - 4805
Parr - smolt survival (2)	0.50	0.076	0.34 - 0.65
Smolt - return after coastal exploitation (3)	0.15	0.051	0.05 - 0.25
Previous spawners (4)	1.20	0.051	1.10 - 1.30
Within season (5)	0.70	0.102	0.50 - 0.90

(1) Assumed 10% error
(2) Assumed equivelent to salmon
(3) Variable in the literature from 1.3% to 43.4% though several studies quote between 10 and 20%.(Potter, 1985; Mills & Piggins, 1988; Berg & Jonsson, 1990)
(4) Mee et al. (1990); Chapter 2.
(5) Assumed covered.

The mean estimates of the increase in returning adults were 619 salmon (range 120 - 1188) and 1682 sea trout (range 487 - 3463). It should be noted that there are many assumptions underlying this evaluation open to uncertainty, and the variability in the parameters used to estimate the increased numbers is shown by the order of magnitude difference between minimum and maximum estimates. The sequential variables were calculated independently and therefore any covariance between parameters were ignored.

CHAPTER 8

OVERVIEW and CONCLUSIONS

8.1 Adult sea trout biology

The River Tywi has an early spring run of sea trout in April, May and June comprising large previous spawners and maiden .1+/.2+ fish. From June onwards the catches of previous spawners declines with the arrival of the first whitling in the catch, and the proportion of these whitling increases during the summer. There is a further late run of large fish in October-December but some of these are delayed migrants from the spring.

Four smolt ages were observed (1-4-year olds) and the overall proportion of 1-, 2-, 3- and 4-year-old smolts was 2.9%, 70.6%, 26.1% and 0.4%, respectively. The frequency of 1-year-old smolts increased in 1991/1992 and the decrease in smolt age was found to be influenced by the temperature regime in the catchment from the abnormally mild winters and hot summers of 1988-1990. The implication of this decline in latter years on other studies is discussed. The MSA of a smolt year class was observed to be significantly inversely related to the cumulative mean monthly temperature in the two years prior to smolting.

This study demonstrated the influence of juvenile growth rate on smolt age, with faster growing juveniles smolting earlier. It also demonstrated that there was a difference in smolt age composition between returning sea age groups, such that whitling had a higher proportion of older smolts and no whitling were recorded from 1-year-old smolts. Therefore, it is suggested that juvenile growth rate, and therefore smolt age, of sea trout influences the age at return for adults. This also implies that the relative contribution of each sea age group to the stock is needed before describing the stock MSA. The MSA in this study for the five year period was 2.24 (20% whitling).

The total number of age classes recorded was 44, displaying a plastic life history strategy for the stock, but 60.2% of the stock were maiden .+/.1+ fish representing only 7 age classes. The majority of adults sampled were .1+ maidens but it was recognised that whitling were undersampled due to gear efficiency and thus a quantitative assessment of the relative abundance of each sea age was not possible.

Overall, 63.3% of the 2689 fish where sea age was determined were maidens, and concomitantly 36.7% were previous spawners. There was a variation from year to year in the estimate of age at first maturity and

spawning frequency due to individual year class strength. This highlighted the necessity for sampling over a long time period, preferably five years, to prevent bias from particular year classes. From the overall sample, 62.9% of previous spawners had matured at .1+ and 30.9% as .+; 13.6% of previous spawners had 3 or more spawning marks. This illustrates the longevity of this sea trout population.

Adult sea trout from the Tywi demonstrated a large mean size for age compared to other British stocks, with whitling attaining a mean length of 320 mm and .1+ fish 530 mm. Growth rates of juveniles were observed to affect the length at return such that larger smolts returned as larger adults.

The sex ratio of sea trout was calculated based on gutted fish from the rod fishery and differed between sea age groups. Whitling were observed to have a sex ratio close to unity, with the proportion of females increasing in the .1+/.2+ maidens and previous spawning groups, respectively. Since whitling had a higher proportion of males than older sea age groups, the sex of a smolt may also influence age at return, as well as the influence of smolt age described above. The differential sex ratio between sea age groups again demonstrated the need for information on the relative contribution of each part of the stock component before describing a sex ratio for the stock as a whole.

The different sampling gears used during the study were observed to catch different components of the stock. The trap sampled all age classes but the estuarial net fisheries, using a standard mesh size of 102 mm, did not exploit the whitling component of the stock. A length girth relationship was established which could be used to calculate the size escapement for different mesh sizes. Rods sampled all available lengths of fish but a bias in the relative contribution of each stock component may have occurred.

The majority of comparisons for length at each sea age between each gear was not significantly different, suggesting that each gear could be used for this purpose. However, rods gave lower gradient length/weight relationships, due to the different stock components sampled and the possible weight loss of fish with increasing time spent in-river.

8.2 Spawning behaviour of adult sea trout

The spawning behaviour of adult sea trout was studied using 174 MHz radio transmitters inserted into the stomachs. Sea trout were captured near the spawning beds and tracked prior to, during and after spawning. Ascent into the tributary was shown to be linked to increases in flow for some fish but not others; the latter fish tending to move at night.

Female sea trout were shown to spawn over 1-3 days, with preference for spawning during darkness, before leaving rapidly as kelts. Size was shown to be important in determining dominance among males, with up to 10 males accompanying one female. However, rarely were more than two 'large' males present with each female, the remaining male(s) being whitling or parr. Males were shown to spawn with more than one female and leave and re-enter the tributary on a number of occasions. The greater residence time and active movement within the tributary for males, leads to a greater possibility of male mortality. This would be in accordance with the sex ratio data recorded from the biological characteristics in Chapter 2.

Most of the whitling observed on the spawning beds in the River Llechach were male, but the results from Chapter 2 for the whole river suggest a sex ratio close to unity for this sea age. The low number of female whitling observed spawning either means that a higher percentage of male whitling entering the Tywi mature, or that the spawning population in the River Llechach may not be representative of the whole catchment.

Life history strategies of sea trout may be interpreted from the spawning behaviour and stock structure in this study. These follow the arguments of Parker (1992) as to the evolution of size for reproduction. For females, increased contribution to the spawning stock (egg production) is derived from fish size and therefore there is an advantage in returning for the first time as .1+ fish and not whitling. For males, their contribution will depend upon the number of pairings with females and the dominance over other males. Males returning as whitling will have an advantage in lower natural mortality than that of older males, but due to the size orientated dominance during pairing, male whitling may only be able to 'sneak mate' with females. This will result in a disadvantage in sperm competition with older males and a greater relative expenditure on sperm compared to larger males (Parker, op cit). Age at return for males

can thus be interpreted as a trade between lower natural mortality for whitling and increased dominance and number of pairings for older males.

8.3 The use of angler log book studies

Log books are increasingly being used to improve the quality of catch and effort data from anglers, and thus to obtain a measure of both annual and temporal trends in catches. 297 to 343 log books were distributed annually between 1990 and 1993 to anglers fishing the Tywi catchment. Annual return rates varied between 37% and 51%, and anglers undertook a mean of between 18.6 and 21.8 trips per season.

CPUE (catch per hour) demonstrated that the salmon fishery performed best in 1991 (0.0174), followed by equal performance in 1990 and 1993 (0.0137), and poorest catch rates in 1992 (0.0106). For sea trout, the rod fishery performed best in 1992 (0.1397) followed by 1993 (0.1149), 1991 (0.1029) and 1990 (0.0853), respectively.

Monthly analysis displayed an increase in CPUE and effort through the season for salmon, but in contrast, sea trout CPUE and effort generally increased between March and July and decreased thereafter. Flow was observed to influence salmon CPUE and effort, but such a relationship was less clearly evident for sea trout. Contemporary relationships between flow and sea trout CPUE were confused due to the complicating factors of whitling entering during July-August when flows may be lowest, angling method, and the long-term susceptibility of sea trout to the rods after entering fresh water.

The weight distribution from the log book data suggested that whitling were the most abundant sea age group. The frequency distribution from the sea trout sub-sampled for biological characteristics in Chapter 2 did not show the same level of dominance of whitling. This may be due to the catches of the small number of anglers collecting biological characteristics being different, but it is probably due to the anglers returning whitling alive un-sampled and therefore under-representing their contribution to the catch. This illustrates one advantage of running a log book study in conjunction with a scale collection programme, since any sampling bias in biological sampling from the catch can be recorded.

8.4 The effects of liming Llyn Brianne on the biota of the River Tywi

Liming ameliorated the acidic and high trace metal level of the reservoir water quality observed prior to liming. Immediately below the reservoir, notable changes in determinands included mean pH increasing from 5.30 to 6.77, mean calcium increasing from 1.57 mgl^{-1} to 7.84 mgl^{-1} and filterable aluminium decreasing from 0.182 mgl^{-1} to 0.115 mgl^{-1}. Significant increases in hardness, alkalinity and iron concentrations were also observed. Significant decreases in the metals manganese and zinc were recorded after liming.

The current liming regime has enabled conditions close to neutrality to be maintained since the original liming, with some changes due to dilution associated with winter precipitation and run-off. However, re-acidification of the epilimnion during the summer and the necessity to cope with acid events from the River Doethie may require management action including the possibility of changing the liming method to direct dosing of the major reservoir inputs.

The acidic conditions in the main river prior to liming did not allow permanent colonisation of acid-sensitive invertebrates for c.17 km below the impoundment. The neutralisation of acidity by liming appears to have permitted recolonisation of many taxa, and drift from the unacidified tributaries is proposed as the primary mechanism of colonisation. Among the taxa to have recolonised the main river were the Ephemeroptera *Baetis rhodani*, *B. scambus*, *Rhithrogena semicolorata* and *Ecdyonurus* spp. Also observed to recolonise were the mollusc *Ancylus fluviatilis*, the Trichoptera *Seriscotoma personatum* and Lepidostomatidae and the Plecoptera *Perlodes microcephala* and *Protonemura meyeri* . Further changes to the invertebrate fauna may yet occur before a stable assemblage is established.

The water quality from the reservoir before liming was also impacting upon juvenile salmonids for c.17 km below the impoundment. Recruitment failure is hypothesised for the absence or low density of juvenile salmonids observed. Liming has ameliorated this toxic effect such that numbers of juvenile salmonids, to the parr stage, have shown dramatic improvement. Natural progression to returning adults is therefore expected from these juveniles but estimating the increased adult production is currently difficult because of the lack of a carrying

capacity estimate for the rehabilitated area.

An estimate of an extra 619 salmon and 1682 sea trout returning to the River Tywi has been calculated using a modest prediction of potential juvenile densities (5, 1+ parr /100 m^2). The rehabilitated area comprises some 9.1% of the total rearing habitat available to migratory salmonids in the whole Tywi catchment (D. Mee pers. comm.), stressing the importance of this rehabilitated area to the Tywi. As well as the increased production from the main river, there may be an additional benefit to the tributaries within the rehabilitated area. The acidic conditions prior to liming may have had some detrimental impact on smolts from these tributaries, as they migrated downstream through the main river.

Electrofishing surveys in 1995/1996 and beyond, after some adults have returned from the increased parr production, will furnish better estimates of carrying capacity for the rehabilitated area.

The current timing of the spring reliming (March) should be assessed, with lowest pH and calcium levels after the winter dilution, since this coincides with the most sensitive salmonid stages (egg hatching and smolt migration). Matters of weeks could be crucial in saving year classes and so liming in February, as in the first liming, is preferred. Alternatively, changing the dosing method to direct stream dosing of the major input streams to the reservoir may alleviate the current level of temporal reacidification. The rectification of the damage due to acid water now allows the NRA to assess the possible impact from the hypoliminal release regime (Clarke *et al.*, 1987), since the two factors could not be divorced previously. However, from the findings of this study it would appear that water quality was the dominant factor in reducing production.

Continuation of the angler log book scheme in future years is the best method, currently available, for assessing any changes in returning adults due to liming. The upper main river does not currently perform well for salmon (Chapter 4), and assuming intra-catchment homing, the expected increased adult salmon may therefore improve this situation. The upper reaches of large rivers have been targeted as rearing areas for MSW salmon (Laughton & Smith, 1992), and historically salmon were captured on the Tywi in May/June in greater numbers than in recent years (e.g.

SWWRB, 1965; SWWRA, 1974). Although numbers of early season salmon have declined on many rivers (Gough *et al.*, 1992) part of the decline on the Tywi may have been due to loss in juvenile production in the upper river. Log book studies will again be useful in recording any temporal changes in catches.

8.5 Conclusions for sea trout stock management

Contribution to the spawning stock will depend upon fish size, since fecundity increases with fish length. The spawning behaviour work (Chapter 3) suggested that females of .1+ and older are the more important component for egg deposition, since few mature female whitling were observed. The larger females would also deposit larger eggs which would produce more viable fry (Elliott, 1984), and have a lesser chance of being washed out by floods (Crisp & Carling, 1989). Many of the male whitling were not dominant males but instead formed a pool of mature males able to fertilise females in the absence of larger males, or by darting in during egg deposition. The contribution of whitling to the spawning stock of sea trout in the River Llechach was therefore questionable. This will also be true for the Tywi as a whole if the spawning stock of the River Llechach is representative of the Tywi catchment. Whilst a disregard for whitling is not suggested, the importance of larger fish needs to be emphasised, especially to anglers who practice catch and release.

The importance of the .1+ sea age fish therefore results in the need for management to protect this sea age. Sea trout from the Tywi are fast growing and this enables relatively high egg production per adult female. However, this fast growth results in exploitation by commercial nets of the immature .1+ sea age, where over exploitation could be damaging. Similarly, over-exploitation by the rod fishery would be equally damaging. Current exploitation estimates (D.Mee pers. comm.) do not suggest over-exploitation, a case supported by the longevity of the sea trout population. The current data available, therefore, do not indicate the need for reducing exploitation of the Tywi sea trout stock.

If, however, the situation changed in the future and management required the reduced exploitation of the stock, several hypothetical measures could be introduced. Reducing exploitation by the commercial net

fisheries could be achieved by increasing the mesh size to allow the escapement of some or all of the .1+ sea age. The appropriate mesh size could be determined from the length girth relationship in Chapter 2. To allow the escapement of 535 mm fish (mean length of .1+) for example, a mesh size of 143 mm (246 mm inside perimeter) would be required. Another method of reducing exploitation by the commercial nets would be altering the fishing season. This could be achieved by delaying the beginning of the season, but since peak abundance is typically in May, some two months into the current season, extending the weekly close period would be a preferable approach.

Reducing the exploitation of the commercial fisheries may require an equitable reduction in the rod fishery. Regulating the size of fish taken in the rod fishery is usually achieved by setting a minimum length (currently 19.0 cm). With the conservation of .1+ fish in mind, setting a minimum size to accommodate this age class would also exclude all whitling. An imposition of a protected slot length, e.g. 45 - 55 cm, has been suggested by some workers (O'Farrell *et al.*, 1989), but this may prove difficult to enforce. Delaying the beginning of the rod season would achieve little since peak catches are in July, some five months into the current season, and sea trout remain susceptible to capture through the season. The most practical method of reducing exploitation by the rod fishery, effectively contributing to an increased spawning stock, would be to bring forward the close season date. This may again be difficult to enforce in a mixed fishery such as the Tywi where the salmon rod season would then extend beyond the sea trout season. A method often used to restrict rod exploitation for salmon is to restrict the angling method to fly fishing only. On the Tywi, fly fishing for sea trout may be as effective as any other method (Chapter 4), and therefore restricting the method may not be applicable.

8.6 Conclusions for stock assessment

From this study it was suggested that the best method to quantitatively sample the sea trout stock was a head of tide trap, operated continually or using a temporally stratified approach, throughout the run period, i.e. March to December. A small-meshed seine in the estuary would sample all available sea age classes but the possible oscillatory behaviour of sea trout in the estuary influenced by flow may bias results. Also,

estuary sampling may, in some instances, include fish from other rivers.

The rod fishery sampled all available lengths of fish but may be biased towards particular parts of the stock component. Where possible, biological collection from anglers should be in conjunction with log book studies so that any bias in sampling of anglers catches can be noted. The commercial estuary nets, from regulation, only sample the larger fish in the population. Mean length at sea ages were similar for the different gears, suggesting that any of the gears could be used for this purpose. However, differences did occur in other aspects such as estimates for age at maturation and length weight relationships. Under-sampling of the .+SM+ sea age class by the nets tended to reduce the estimate of maturation as whitling, compared with the rod data. The lower gradient length weight relationships from the rods was partly due to the different sea age classes exploited, but this may also have been due to weight loss with in-river cessation of feeding.

The rod and commercial catches could be used in unison and furnish good qualitative data such as length at age to distinguish stocks of different juvenile and adult growth rates and longevity. The operational resources needed for quantitative trapping is often prohibitive but the qualitative information could be gained cheaply and with apparent ease.

Availability of data over a five-year period enabled annual variation in estimates of biological characteristics to be assessed. It is recommended that a similar time period is used for other studies, since shorter sampling periods may only provide a 'snap shot' and not cover the range of estimates, possibly providing biased information.

The observation that river temperature influences smolt age/juvenile growth rate and that this in turn affects sea age at return has implications for stock assessment of both juveniles and adults. Currently the main method of juvenile assessment is electrofishing surveys during summer months. If smolt age declines, such that parr emigrate as 1-year olds then they will not be present as 1+ parr during the survey 'window', leading to a possible interpretation of lower production in fresh water. For adults, if smolts migrate younger this could lead to a higher production, but this may be offset if smolt survival of these younger/smaller smolts is lower. If whitling display a higher return rate than .1+, due to a shorter sea residence, then younger smolts may lead to

a lower overall number of returning adults. This has repurcussions for sampling methods and data such as angler CPUE.

8.7 The requirement for future research

This investigation examined the short-term effects of ameliorating headwater acidification by liming. Continuation of the surveys will demonstrate the long-term effects and provide a measure of the stable macro-invertebrate assemblage and the carrying capacity for juvenile salmonids.

For the sea trout population, one of the main areas where information is needed is that of the factors governing freshwater production and any links this may have with age at return of adults. The factors given here were determined from scale reading of adults, but to truly investigate this area without the possible bias of varying smolt survival with smolt age, then sampling of the smolt run leaving the river needs to be undertaken. This would entail establishing a long-term study of an experimental catchment or sub-catchment and measuring factors such as water temperature, juvenile densities, juvenile growth rates and relating them to the resulting smolt run.

One of the main questions that must be answered if sea trout stocks are to be managed with any degree of certainty is: 'Why do some smolts return as whitling and some do not ?'. Smolt age and sex may be influencing factors but this question needs to be answered, since any variation in return rates between whitling and .1+ maidens confuses interpretation of production and catches.

The variation in survival rates between smolt ages needs to be established, so that an indication of whether reducing smolt age does increase production or whether it is offset by decreased survival of the younger smolts. This could be determined from trapping returning adults from an experimental catchment, as described above.

Further spawning behaviour work is needed to establish the contribution of whitling to the spawning stock and the proportion of mature whitling entering a river.

The marine phase of the life history is an area where currently knowledge is lacking. To fully understand the life cycle a better understanding of factors such as marine migrations, feeding, survival rates and the effect of environmental variables is required. Information as to the possible differences in the marine phase between sea trout returning as whitling or older fish may be particularly enlightening.

The use of angler log books needs to be developed further, since in many circumstances catch records are the only available method of stock assessment. The catchability of different angling methods needs to be determined since this will enable informed decisions regarding any future restriction of methods. Further gaps in the knowledge include the possible differences in catchability between different sea age groups of fish, viz. whitling and older fish for sea trout and the grilse and MSW component of salmon stocks. This information will aid the interpretation of overall catches between years but a knowledge of the abundance of the stocks is needed, which may be available from studies undertaken with river counters or traps.

There is currently an interest in establishing long-term 'monitored rivers' throughout England and Wales (Solomon, 1993). The Tywi should seriously be considered as a candidate for S.Wales since it is primarily a sea trout river, it has a large run of fish, good catches in the fisheries and there is already a good understanding of the biological characteristics of the stock. The Tywi also has a balance of age at first return between whitling and .1+ maidens, which would enable investigations into this facet of the life-history strategy.

There is, therefore, a major opportunity with the River Tywi to introduce a trapping facility to quantitatively assess fish abundance, taking advantage of the merits noted above. Although installation of a trap would incur capital cost, given the long-term nature of a monitored river, the benefits described above should outway short-term financial considerations.

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