THE UNIVERSITY OF HULL

The impact of cormorants (*Phalacrocorax carbo carbo* and *Phalacrocorax carbo sinensis*) on inland fisheries in the UK

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ABSTRACT

Cormorants are piscivorous birds with a daily food intake (DFI) of approximately 500 g. They are a protected species under the Wildlife and Countryside Act 1981. In the UK, the number of over-wintering, inland cormorants increased steadily between 1970 and 1987, at a rate of between 5 and 10 % per annum. An increase of 74 % occurred between winter 1987/88 and 1990/91, and the population is still believed to be rising. The population growth was observed in all regions of the UK, on all habitat types. As cormorants exploited new habitats, ornithologists welcomed their increased presence. This contrasts with the views of angling bodies, who assert that the presence of cormorants, feeding daily on their fisheries, has a damaging impact on fish stocks with inevitable financial losses. Due to a lack of effective non-lethal control methods, the angling bodies wish to see the cormorants removed from the protected species list so their inland numbers can be controlled. Ornithologists insist that there is no scientific evidence proving cormorants are damaging to inland fisheries and so are opposed to any culling.

A review of previous cormorant studies was undertaken to evaluate information on their ecology, feeding behaviour and predation impact. The general conclusion was no study had been able to prove cormorant predation damages fish populations, because few studies had moved beyond determining the mass of fish removed by the birds over the particular study period. No assessment had been made of the impact of that fish removal on the fish population dynamics and the angling performance of the fishery. This highlighted the requirement for research into the impact of cormorant predation on inland fisheries. This study was formulated to estimate cormorant predation impact on fisheries in a more realistic and robust manner than had previously been undertaken.

The principal objective of the study was to integrate fish population and cormorant feeding dynamics data on specific fisheries (study sites) in such a way as to quantify, where possible, the full impacts of the cormorant predation. This required the following criteria at each study site:

- evaluation of the historical status of fish and cormorant populations;
- determination of the population and community dynamics of the fish stocks;
- analysis of the angling effort and angling performance;
- identification of the species, and estimation of the numbers and sizes of fish consumed and wounded by cormorants, and comparison with the numbers and sizes of the fish populations present;
- determination of the occupancy on, and use by, cormorants at the selected sites.

The work programme ran between September 1995 and July 1998, covering three winters of cormorant predation. At each study site, the cormorant feeding dynamics were assessed by detailed feeding observations and cormorant counts. This enabled data to be collected on the species, size and amount of fish being ingested during each foraging bout, and the diurnal and seasonal patterns of cormorant occupancy. As feeding observations were unable to completed at each site everyday, a modelling system was designed, using a Monte Carlo Simulation (MCS), to estimate the number and mass of

fish being removed from the site over the whole winter period. The fisheries data were collected by electric fishing, seine netting, hydro-acoustics and angler catch analysis. The actual methods used at each site were dependent upon the physical conditions present. The data were analysed for fish population dynamics, including length frequency of species, year class strength, natural mortality rate and growth indices; and for angling performance, including catch per unit effort and the relative importance of species. Combining site-specific data for the fish species composition, and the length frequency distribution from fisheries surveys and the cormorants' diet, allowed preliminary predation impact assessment. Reconstruction of life tables from the fisheries data allowed integration of the cormorant feeding data from the Monte Carlo Simulation to assess impact in terms of the numbers of fish consumed on subsequent population densities. This enabled the status of the fish population at each study site to be shown, with and without cormorant predation over the three-year period, resulting in a detailed predation impact assessment.

The fisheries studied were located in two regions of the UK, the Midlands and the North West of England. This enabled the research to be completed in two distinct geographical areas, with known and established over-wintering cormorant roosts. The Midland study sites were Holme Pierrepont Rowing Course, Colwick Park Trout Lake and the River Trent. The North West study sites were the lower River Ribble and Grimsargh number 3 Reservoir. These sites encompassed cyprinid and salmonid fish populations, and covered riverine and lacustrine fisheries.

At Holme Pierrepont Rowing Course, cormorants were first observed to forage on the lake in winter 1993/94. Angling success was high on the lake between 1990 and 1992, but had declined rapidly by 1994. Between 1994 and 1998, angling success improved, although catch rates were still below the levels observed in 1992. During the period of study, large numbers of foraging cormorants, approximately 80 per day, ingested roach, common bream and perch. Prey size was mainly below 100 mm. Foraging bout success was high. The fisheries surveys revealed the species present, including roach (*Rutilus rutilus* (L.)), common bream (*Abramis brama* (L.)) and perch (*Perca fluviatilis* L.), had fast growth rates. Growth analysis of fish from year classes prior to the cormorant foraging on the lake revealed a distinct growth rate shift in 1994, with considerably faster growth in the post-predation period. It was estimated that the fish populations were reduced 59 % by three years of cormorant foraging.

It was concluded that the cormorant predation on the lake had reduced the fish standing crop considerably in the post winter 1993/94 period, and was a major factor in the decline in angling results in the lake in 1994. The cormorant predation, by reducing fish standing crop in the lake, lowered the amount of inter- and intra-specific competition in the fish populations, in aspects such as food supply. This increased the scope for growth and resulted in the faster growth rates. This is a compensation mechanism in the fish population to reduce the damage impact. The faster growth decreased the age of maturity, increased fecundity for age and decreased the time period in which fish were vulnerable to predation, as cormorants mainly ingested fish below 100 mm in the lake. As the growth shift occurred during the period of lowered angling returns, this supports the theory that cormorant predation had reduced the fish abundance in the lake.

In the River Trent, cormorant predation impact was difficult to determine. The numbers of cormorants observed foraging on the river were low during each winter period. Fish population dynamics were stable, with the variation in annual growth increments being dependent on physical factors, such as temperature. Standing crop values revealed annual variation, and this could be accounted for by natural fluctuations in the fish populations, and their migratory behaviour. Angler catch rates were consistent over the study period and were higher than values observed between 1969 and 1990. However, integration of the cormorant feeding data from the Monte Carlo Simulation into the life table analysis, revealed that the fish populations of the study sites were reduced by up to 98 % over the study period. This was explained by the MCS over-estimating the fish losses. This resulted from limitations in the assumptions of the number of observed cormorants flying over the river that actually fed there. It was acknowledged that while cormorants were taking fish from the river, their impact on the fish populations was likely to be low when compared with natural factors, such as temperature and flow.

Cormorants utilised Colwick Park Trout Lake as a day roost site during each winter. This was because there was a low density of fish in the lake, making foraging very inefficient. During March of each year, 2000 rainbow trout were stocked into the lake for anglers to exploit at the beginning of the trout fishing season. This resulted in the cormorants using the lake for feeding before they dispersed from the area to breeding sites. The replacement cost of the predated trout was between £414 and £516 per annum. A further impact of the cormorant predation was non-lethal wounding, where the trout had been captured by the bird, but dropped before swallowing. This resulted in a loss of scales on one side of the fish and an abdominal hole on the other. It was estimated that 11 % of all trout stocked in the lake were wounded during the period of cormorant predation.

The fish community structure at Grimsargh number 3 Reservoir was composed of high densities of roach, common bream and perch below 150 mm. No fish were present between 180 mm and 450 mm. Although cormorant foraging was very efficient on the reservoir, the numbers of cormorants utilising the reservoir as a food patch was low. The maximum number of birds that foraged daily on the lake was 13. Integration of life table analysis and MCS data revealed cormorants had a minimal impact on the fish populations of the lake because of the high density of small fish present.

The efficiency of fisheries surveys on the lower River Ribble was poor. The river topography made boat-mounted electric fishing particularly ineffective. The river was wide (up to 30 m), with very deep (above 4 m) and very shallow (below 0.5 m) areas found in close proximity. This, combined with the restrictions on the number of electric fishing surveys, resulting from consents only being granted for fishing outside of periods when salmon were migrating up the river, meant that the fisheries data set was limited. Therefore, despite a robust cormorant data set, life table analysis was unable to be carried out and an accurate predation impact assessment was not possible. The biomass of fish removed during each winter by cormorants was compared with the biomass of fish available in the river - based on electric fishing results - but this was not considered to be a satisfactory method of impact assessment.

The cormorant impact assessment on each study site revealed it was difficult to make a general statement on the impact of cormorant predation on inland fisheries. Actual cormorant predation impact on a specific fishery is dependent on a combination of factors, such as the foraging habitat for the cormorants, the fish community structure and density, and the number of alternative foraging sites in the locality.

The cormorant control methods for fishery managers were reviewed, with a cormorant control strategy elucidated for a general inland fishery in the UK, and for each specific study site. Despite a number of non-lethal control methods being available, the majority of methods were either ineffective or could not be applied in a realistic strategy. For example, scaring methods only force the cormorants to feed on alternative areas of the fishery, or on neighbouring fisheries. Their increased activity results in an increased DFI. Licensed shooting has been shown to be effective in reducing cormorant occupancy on fisheries in the short term. However, the licence application procedure is, at present, difficult and the legal constraints on the use of firearms in public areas means the method is not viable at the study sites. An exception was the lower River Ribble, where cormorant predation impact was difficult to elucidate.

At Holme Pierrepont Rowing Course, the construction of 'cormorant-proof' fish refuges was the most favourable option in protecting the vulnerable fish species from cormorant predation. In the River Trent, off-stream refuges should be constructed to increase the recruitment potential of the cyprinid populations. This will increase the potential for compensation of any fish losses to cormorants in subsequent years. Manipulation of the rainbow trout stocking policy during the period of cormorant occupancy would reduce losses to the birds at Colwick Park Trout Lake. As cormorant predation impact was minimal at Grimsargh number 3 Reservoir, management procedures should aim to reduce the high density of slow growing cyprinid fish below 150 mm. This will aim to reduce the amount of intra- and inter-specific competition in the lake, increase the growth of the fish and increase the density of fish in the angler exploitable cohort. Similar to the River Trent, recruitment potential should be maximised in the lower River Ribble by construction of off-stream refuges which increase the fry and nursery areas.

Cormorant predation was shown to have had a serious impact on Holme Pierrepont Rowing Course and Colwick Park Trout Lake, and there are few options available for fishery owners to reduce cormorant occupancy and predation on specific fisheries by current methods. This highlights a requirement for a thorough review of the present legislation and practices regarding the legal protection of the cormorant. The review should aim to implement cormorant control strategies on a regional and national level, with an aim to protect vulnerable fisheries from inland, over-wintering cormorant predation.

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1. INTRODUCTION

1.1 Rationale of study

1.1.1 *The cormorant*

There are two sub-species of cormorant that inhabit the UK, *Phalacrocorax carbo carbo* (Linnaeus) and *Phalacrocorax carbo sinensis* (Blumenbach). They are relatively large, piscivorous birds with a daily food intake of 340 to 520 g day^{-1} (Kirby *et al.* 1996). They are associated with coastal and estuarine fisheries, but an increasing number have been found over-wintering inland in recent years (Veldkamp 1996). A detailed description of the population dynamics and ecology of the cormorant is given in Chapter 2.

The number of over-wintering inland cormorants in the UK has increased steadily over the last 20 years (Sellers 1979; Feare 1988), with the increase estimated at 5 to 10 % per annum between 1970 and 1987 (Kirby *et al.* 1995). Between 1987/88 and 1990/91, the rate accelerated by a total of 74 %, equivalent to a linear growth rate of 24.8 % per annum, and it is still thought to be increasing (Kirby *et al.* 1995). The growth was observed in all regions of the UK, on all habitat types, but was especially noticeable at reservoirs and gravel pits (BTO 1997).

It is believed that a minimum of 6000 over-wintering inland cormorants are present in the UK (Russell *et al.* 1996). These birds form colonies in September and October which are the focus of night roosting and the base from which daily activities, such as feeding, are undertaken. Colonies of over 400 wintering inland cormorants are known to be present at a number of UK sites, for example, Abberton Reservoir, Essex (Russell *et al.* 1996) and Attenborough Gravel Pits, Nottingham (T. Holden pers. comm.; Section 4.5.1). The birds remain inland until March or April when they disperse to coastal breeding sites (Section 2.2). Therefore, between October and March these cormorants tend to forage on inland waters.

Inland cormorant colonies require an abundant food source to meet the birds' daily nutritional requirements (Section 2.6.2). The increase in stocked recreational inland fisheries over the last 10 to 15 years, covering both salmonid and cyprinid species, together with the existing fish stocks available in rivers, canals and lakes, have presented the over-wintering cormorants with an attractive and abundant food source. Eutrophication of many inland fisheries has also increased inland fish production in recent years (Russell *et al.* 1996). As cormorants are accomplished piscivores and are evolved to be capable of exploiting coastal fisheries, efficient foraging on inland fisheries is possible.

1.1.2 *The ornithologists' view*

Where cormorants have set up large inland wintering roosts, they are a habitual and highly visual visitor to surrounding fisheries where they forage at dawn and the early daylight hours. The increased winter presence of cormorants inland is welcomed by ornithologists who are pleased at their expanded habitat range (BTO 1997). The birds have been described as:

"Dramatic, especially if they have been far afield and return at a great height, gliding downwind at speed, wings curved back, feet sticking out either side to control the helter-skelter descent." (Wynde and Hume 1997).

1.1.3 *The anglers' view*

Angling has been reported as one of the fastest growing leisure activities in the UK. Large amounts of money are generated from licences and revenues from over three million anglers in the country (NRA 1994a). Since 1993, the sales of rod licences have increased by 40 %, with a total of 1,196,417 angling licenses sold in England and Wales in the 1997/98 financial year, raising £13,386,885 (EA 1999). Of these, 97 % were for non-migratory trout and coarse fish and 3 % were for migratory salmon (*Salmo salar* L.) and sea trout (*Salmo trutta* L.) (EA 1999). The total national annual expenditure on angling is estimated at £2.4 billion (NRA 1994a). Thus, it may be considered important that the fisheries and fish stocks on which this industry is dependent are managed on a sustainable basis and are exploited rationally.

A number of UK freshwater and diadromous fish species are widely exploited by recreational anglers, with all coarse fisheries operating catch and release policies and stillwater salmonid fisheries usually operating put-and-take systems. The majority of riverine salmonids are also taken, subject to new bylaws introduced in April 1999 (A. Starkie pers. comm.). Recreational angler targeted species include: Atlantic salmon, brown trout (*Salmo trutta* L.), rainbow trout (*Oncorhynchus mykiss* (Walbaum)), barbel (*Barbus barbus* (L.)), bleak (*Alburnus alburnus* (L.)), common bream (*Abramis brama* (L.)), carp (*Cyprinus carpio* L.), chub (*Leuciscus cephalus* (L.)), dace (*Leuciscus leuciscus* (L.)), gudgeon (*Gobio gobio* (L.)), perch (*Perca fluviatilis* L.), pike (*Esox lucius* L.), roach (*Rutilus rutilus* (L.)) and grayling (*Thymallus thymallus* (L.)).

Angling associations and societies view the cormorant as a problem, capable of causing large damage to inland recreational fisheries due to their daily food intake of 340 to 520 g day^{-1} (Section 2.6.2), and their over-wintering presence for 5 to 6 months a year. Over 80 cormorants have been observed feeding daily on some inland fisheries between October and March (Fitzpatrick 1997; T. Holden pers. comm.). Angling media calculated the total weight of fish consumed by inland wintering cormorants during 1996 was 1,433,455 kg, at a replacement cost of £5,294,701 (Fitzpatrick 1997), with the impact of this level of predation questioned. A variety of nicknames have been used to describe the cormorants, for example, "the black plague" (Fitzpatrick 1997; Figure 1.1).

1.1.4 *Cormorant control methods*

When anglers and angling associations complain of large scale cormorant predation on their fisheries, they are unable to control the cormorants by lethal methods, such as shooting. In the UK, cormorants are fully protected under the EU Birds Directive (EEC/79/409) through the Wildlife and Countryside Act 1981 (Figure 2.1). Legal protection of *sinensis* was deregulated by the EU in 1997 (Fitzpatrick 1997), but this has not yet been incorporated into UK legislation. Also, *Phalacrocorax carbo sinensis* is less abundant in the UK than *Phalacrocorax carbo carbo* (Section 2.1). Licensed shooting of cormorants is allowed in the UK, "*for the purposes of preventing serious damage to..........fisheries*" ((Wildlife and Countryside Act 1981; Figure 2.1). However, the legislation does not give the criteria for 'serious damage'.

The number of licences for shooting cormorants granted annually from 1983 to 1992 was between 26 and 51. In the winter of 1996/1997, 47 licences were granted to shoot cormorants in England with 180 birds shot, and 32 licences in Scotland with 244 birds shot (Kirby *et al.* 1996).

A number of non-lethal methods have been used by fishery owners to protect their fisheries and livelihoods from cormorant predation, including visual and acoustic scarers (Section 10.2). However, these control methods often fail due to habituation by the cormorants which then return to the fishery to continue foraging (Section 10.2). Furthermore, when cormorants are driven from one fishery, it is likely they will settle on a neighbouring one until scaring tactics are implemented there. Therefore, a long term solution is apparently not available using non-lethal control methods.

1.1.5 *The inland cormorant issue*

The increased presence of cormorants on inland fisheries in winter, coupled with the limited success of non-lethal methods of cormorant control, including fisheries where such control methods are not practical (for example, unmanaged rivers, navigable rivers, canals and drains), have resulted in anglers and angling bodies increasingly voicing their opinion that the legal protection for cormorants should be revoked. They claim that only shooting cormorants will protect the fish populations from serious predation (Figure 1.1).

However, conservationists and ornithologists insist that there is insufficient scientific evidence in the majority of cases to support the existence of large scale damage to inland fisheries by cormorants (Feltham *et al.* 1999), and the law cannot be changed on the basis of anecdotal evidence alone. It is argued that when decreases in angling catches occur, cormorants, being large, visible and very obvious, are easy to blame with little regard given to other factors controlling fish production, such as water quality, environmental factors and habitat management (Wynde and Hume 1997).

In reviewing the current legislation surrounding the cormorant, and to fill the void of scientific data in cormorant predation impact studies, the Ministry of Agriculture, Food and Fisheries set up a £1 million, three-year cormorant evaluation study which began in September 1995. Four projects were implemented investigating inland cormorant ecology, control methods, feeding behaviour and predation impact. This study forms part of the project, "Case studies of the impact of fish-eating birds on inland fisheries in England and Wales" (Feltham *et al.* 1999). This was a collaborative project, involving the University of Hull and Liverpool John Moores University. On the selected UK fisheries, the University of Hull studied the fish populations and Liverpool John Moores University studied the cormorant populations, in order to assess the cormorant predation impact on the fish populations (Chapter 3). Hence, in this study, the cormorant data used to support the fisheries interpretation of cormorant predation impact were collected by colleagues at Liverpool John Moores University (Chapter 3).

1.2 Project overview

1.2.1 *Project aims*

The aim of this study was to assess and quantify the impacts of UK over-wintering inland cormorants (*Phalacrocorax carbo carbo* and *Phalacrocorax carbo sinensis*) on inland freshwater fish populations.

For this study, the term 'impact' must be defined at the outset. Although the removal of fish by over-wintering cormorants from the standing crop of a fishery is a predation impact, it only indicates the level of cormorant predation that the fishery has incurred. A more sensitive assessment of cormorant predation impact on an affected fishery is the subsequent effects of that predation on the fish populations and their population dynamics. Such impacts may include:

- shifts in fish population dynamics, for example, changes in growth, fecundity and recruitment;
- changes in the community structure and standing crop of the fish populations in the affected fishery;
- changes in fish habitat utilisation, such as increased use of vegetated areas, and changes in shoaling behaviour, for example, increased shoaling during the period of cormorant predation.

The principal aim of this study was to integrate fish population and cormorant feeding dynamics data on specific fisheries in such a way as to quantify, where possible, the full impacts of the cormorant predation. Although a great deal of effort has been put into estimating the 'impact' of cormorants at freshwater fisheries, few previous studies have moved beyond estimating the number or mass of fish removed from a particular fishery by cormorants, with little regard given to the effect of that predation on the fish populations (Section 2.7). This study was formulated to estimate the impact in a more realistic and robust manner than has previously been undertaken.

1.2.2 *Specific objectives*

The specific fisheries objectives of the study were:

- to evaluate, using existing data sources, the historical status of fish and cormorant populations at a number of key sites to provide baseline data;

- to determine the population and community dynamics of the fish stocks at these key sites;

- to quantify the angling effort and angling performance of the sites using match records, log book returns and liaison with angling clubs, and relate these data to stock status and cormorant feeding observations;

- to identify the species and estimate the numbers and sizes of fish consumed and wounded by cormorants at the sites from observations of their feeding behaviour and the analysis of cormorant stomach contents, and compare with the numbers and sizes of the fish populations present (from fisheries surveys and angling catch data);

- to identify 'hotspots' on the sites where the cormorants were present in greatest numbers and/or forage most often;

- to develop models to assess the impact of known cormorant predation losses on the resident fish stocks of the sites.

The fisheries and cormorant predation impact data were supported by cormorant data collected by Liverpool John Moores University. The cormorant objectives were:

- to estimate the annual and seasonal variation in the numbers and distributions of flying, feeding and loafing cormorants on river sites, by co-ordinated monthly counts;

- to record the occupancy on, and use by, cormorants of the selected sites, including aspects of cormorant feeding ecology, through detailed field observations.

BRUM WANT TO GO BANG!

Britain's biggest club apply for a licence to shoot river cormorants

Anglers Mail, February 1996

'TRENT WRECKED' BY BLACK PLAGUE

Angling Times, January 1997

REMOVE THOSE BIRDS

Angling Times, November 1997

KILLER BIRD BLITZ

Angling Times, February 1996

CORMIES THREAT TO STOCK LIFELINE

Don't give these birds a free lunch, says agency

Angling Times, November 1996

Figure 1.1 Headlines relating to cormorant articles from angling media.

2. CORMORANT BIOLOGY, ECOLOGY AND IMPACT STUDIES

2.1 Characteristics and taxonomy

The cormorant is a heavy-bodied bird with a long sinuous neck, long wings and a long wedge-shaped tail (Plate 2.1) (Veldkamp 1996). They are predominantly black in colour, often with a metallic sheen, and the legs are set towards the tail of the bird (Veldkamp 1996). They are aquatic and piscivorous.

Cormorants belong to the family Phalacrocoracidae. They are a polytypic species with five subspecies distributed throughout Europe, Asia, Africa, Australasia and North America (Russell *et al.* 1996). In Europe two of the subspecies breed and over-winter: *Phalacrocorax carbo carbo* (Linnaeus) and *Phalacrocorax carbo sinensis* (Blumenbach).

Phalacrocorax carbo carbo is the North Atlantic race of the cormorant. Britain is a main stronghold supporting 19% of the west European population, with only Norway contributing higher abundance (Kirby *et al.* 1996). Present population figures show 7200 breeding pairs in the UK (Kirby *et al.* 1996), with overall numbers of more than 19 000 birds (Kirby *et al.* 1995).

Phalacrocorax carbo sinensis is the European species of cormorant found from the Western Baltic and Central Europe eastwards and sporadically through Russia to India, China and Japan (Cramp and Simmons 1977). They are considered winter migrants into UK waters, comprising 5 to 10 % of the British wintering population (Kirby *et al.* 1996).

Recent research in south-east England has shown that the subspecies live sympatrically and that hybridisation may be occurring (Goostrey *et al.* 1998).

Comparison of the main features of the two sub-species reveals variation in breeding plumage and body size (Table 2.1) (Stokoe 1958; Alstrom 1985; Sellers 1993). Cramp and Simmons (1977) noted that the features were individually and transitory variable, and must be considered in combination. Alstrom (1985) considered the method too unreliable and concluded that the shape of the gular pouch was a more reliable feature for identification. Marion (1995) indicated that even in full nuptial plumage exact field identification of the two subspecies was virtually impossible.

Cormorant body weight has been determined by a number of workers (Piggins 1958; Cramp and Simmons 1977; Koffiberg and van Eerden 1995). Variation was found between sex, breeding and wintering birds, and different stages of maturity (Table 2.2).

2.2 Reproduction

Phalacrocorax carbo carbo generally occupies breeding colonies between mid-March and mid-September, with egg laying occurring in late April or early May (Cramp and Simmons 1977). The colonies are coastal, situated on rocky cliffs, skerries, stacks and offshore islands (Russell *et al.* 1996), and consist of between 10 and 200 pairs (Lloyd *et al.* 1991). However, a small number of tree-nesting, breeding colonies are apparent at inland sites, these having mainly been established within the past decade (Sellers 1991; Debout *et al.* 1995).

Table 2.1 **Distinguishing features between the European subspecies of cormorant (Cramp and Simmons 1977).**

Identification		Feature on *carbo*	Feature on *sinensis*
• Breeding plumage	Gloss on the body plumage	Blue/ bluish purple	Blue/green
	Extent of the white filoplume feathering on head		Much greater in *sinensis*
	Size of white thigh patches		Larger in *sinensis*
• Body size	Wing length	Larger than *sinensis*	
	Bill dimensions	Larger than *sinensis*	
	Weight	Heavier than *sinensis*	

Table 2.2 **The body weight of the two European subspecies of cormorant (Cramp and Simmons 1977).**

Subspecies	Season	Mean weight of male (g)	Mean weight of female (g)
• *sinensis*	Winter	2605 +/- 418	2145 +/- 398
	Breeding (April)	2423	2085
	Breeding (June)	2283	1936
• *carbo*	Winter	3104 +/- 337	2486 +/- 284
	Breeding (May to October)	3559	2820

Both parents incubate the eggs and care for the young after hatching (Pickering *et al.* 1992). The fledgling period is approximately 50 days, although young may return to the nest site to be fed for a further 40 to 50 days before becoming independent (Pickering *et al.* 1992).

After the breeding season the birds disperse widely, with birds from different breeding areas moving in specific directions. Some inland movement also occurs (Coulson and Brazendale 1968; Summers and Laing 1990). Cormorants rarely fly over open sea as indicated by North Sea cormorants being exclusively coastal (Tasker *et al.* 1987). The majority of breeding cormorants in the UK are considered residents (Pickering *et al.* 1992).

The age at first maturity has been determined in *carbo* populations in Wales, with breeding being first observed in birds in their second summer (Sellers 1991). The majority of cormorants breed at three years of age (Russell *et al.* 1996).

Phalacrocorax carbo sinensis breeds almost exclusively inland. However, the literature suggests they do not breed in the UK and are immigrants (Russell *et al.* 1996). The age at first maturity in Europe varies (Russell *et al.* 1996). In Denmark, breeding is common at two years of age (Bregneballe and Gregersen 1995), while a small proportion of birds have been found to breed at one-year old (Bregneballe *et al.* 1996). Cormorants in the

Netherlands mature at four or five years old, and occasionally at three years old (Kortlandt 1942).

The reproductive success of *carbo*, measured as the number of young fledged per pair, varies considerably between year and colony. It is related to food availability, weather conditions and social factors in the colony (Russell *et al.* 1996). Reproductive success of colonies on British coasts ranges between 1.8 and 2.3 young per laying pair (Carrier and Baker 1991; Sellers 1991). Norwegian cormorant reproductive success varied from 2.3 to 3.1 young per pair over a three-year period (Rov 1984).

The reproductive success of *sinensis* increases with age and stabilises at five years old (Russell *et al.* 1996). In Denmark, the mean numbers of young fledged per pair were 1.31 at two years, 1.65 at three years, 1.95 at four years, and 2.18 to 2.41 at five years and older (Bregneballe and Gregersen 1995).

2.3 Mortality

The estimation of cormorant mortality rates utilises data from ringing recovery experiments. Mortality rates for first year birds in Britain were 70 % and between 28 and 46 % for older birds (Stuart 1948). Age specific mortality rates for *sinensis*, calculated at a time when the populations were increasing at an annual rate of 10%, were 36 % for first year birds, 22 % for second year, 16 % for third year and 7 to 14 % for adult birds (Kortlandt 1942). Cramp and Simmons (1977) considered these values rather high due to ring wear and loss. However, the adult survival rates were comparable with the observed annual return rates of other individually-colour-coded seabirds to their nest sites (Russell *et al.* 1996).

2.4 Behaviour

The two cormorant species are pursuit diving birds capable of diving for prey to depths over 9 m, but surface between dives to breathe (Welsh Water Authority 1980). Diving behaviour is related to the depth and foraging habitat being fished (Wilson and Wilson 1988). Birds diving in shallow water, or feeding on surface shoaling fish, tend to have relatively short dive periods (Russell *et al.* 1996). The feet and hooked bill are used to chase and grasp fish, which are usually seized just behind the gills (van Dobben 1952). Cormorants return to the surface to swallow their prey (Cramp and Simmons 1977).

It has been postulated that because the shag (*Phalacrocorax aristotelis* Linnaeus), a piscivorous bird related to the cormorant, swallows fish underwater when feeding on sandeels (Wanless *et al.* 1993), cormorants may also swallow fish underwater. This was supported by B. Hughes (pers. comm.) who considered the head lifting behaviour of a cormorant on surfacing after a dive was indicative of the bird moving a fish (swallowed underwater) from the throat to the gizzard. However, Feltham and Davies (1995, 1996) showed feeding cormorants in north west England would return to the surface to ingest even small cyprinid fry.

Feeding occurs exclusively during daylight hours. Schafer (1982) and Feare (1988) found no clear daily peak of feeding activity in winter whilst Kennedy and Greer (1988) found feeding in spring was mainly in the first few hours after dawn.

2.5 The population of cormorants in the UK and Europe

2.5.1 *Historical populations*

Cormorants and man inhabit the coastal and inland waters of Europe, and both exploit similar fish resources. The cormorant, an accomplished piscivore, has traditionally attracted a hostile attitude with persecution controlling the historical European population numbers (Veldkamp 1996). With the development of more effective measures of population control in the mid Twentieth Century, the breeding populations of cormorants became small and localised throughout Europe at that time (Van Eerden *et al.* 1995). Complete cormorant elimination was seen in some European countries (Veldkamp 1996).

A factor suggested for this decline in the European cormorant population was the use of persistent pesticides in agriculture after 1945. Veldkamp (1996) considered these were responsible for reduced breeding success in at least some of the European colonies. Boudewijn and Dirksen (1995) concluded organochloride contaminants were the most probable cause of reduced breeding success of cormorants through measured reductions in egg shell thickness and high mortalities of embryos and small nestlings. Van den Berg *et al.* (1995) reported cormorant egg contamination above the 'no observed effect level' of polychlorinated biphenols, dibenzo-*p*-dioxins and dibenzoflurans (PCB, PCDD, PCDF). However, in Poland, contamination of breeding birds by organochloride pesticides, such as DDT, and heavy metals, such as lead and mercury, were found not to have any adverse effect on breeding success or population growth (Mellin *et al.* 1991; Lindell *et al.* 1995).

Historically, the cormorant populations of Britain were viewed as such a threat to inland fish stocks that in some regions bounty schemes were employed to encourage their control. The Dee and Clyde River Authority and the Gwynedd and Wye Water Divisions are three organisations from Wales which deployed control schemes in the 1960s, offering from 25 pence to one pound per cormorant head (Welsh Water Authority 1980). These schemes were considered ineffective and were abandoned as recorded annual claim figures were low. Similar schemes were operated in Scottish regions but were disbanded due to misuse by claimants (Mills 1965). MacDonald (1987a) reported 3527 cormorants were shot during the 1973 to 1976 bounty schemes in Ireland.

There are extensive historical records of inland tree-nesting colonies in the UK, especially in Norfolk (Seago 1967), Suffolk (Payn 1962), Pembrokeshire (Lockley *et al.* 1947) and Scotland (Baxter and Rintoul 1953). Colonies also existed on several parts of the British coast (Pickering *et al.* 1992). These colonies are all thought to have become extinct due to persecution (Russell *et al.* 1996). However, the overall population decline was difficult to ascertain as the species shifted breeding sites regularly. Inland colonies were reported common in Ireland earlier this century, but also declined due to persecution (Russell *et al.* 1996).

Thus, the cormorant population of Europe was held below the carrying capacity of the available habitats during the mid twentieth century (Cramp and Simmons 1977; Kirby *et al.* 1996; Veldkamp 1996).

A shift in attitude towards conservation in Europe during the late 1960s and early 1970s saw an emerging lobby of conservationists safeguarding the last remaining European breeding colonies in nature reserves. This halted the population decline (Van Eerden *et al.* 1995). The cessation of persecution in this period is well documented for both the Netherlands (Zulstra and Van Eerden 1991) and Denmark (Gregersen 1991). A ban in a number of persistent pesticides also occurred at that time (Veldkamp 1996). Hence, these combined factors enabled cormorant population numbers to stabilise.

2.5.2 *Cormorant legal protection*

A significant factor in the reversal of the cormorant population decline was the EC Directive on the conservation of wild birds (EEC/79/409) of 1979. The Directive covers protection, management, control and rules for exploitation applicable to birds, their eggs and their habitats. The cormorant is included in Annex I. The directive was implemented into UK law under the Wildlife and Countryside Act 1981 (Figure 2.1).

Prior to implementation in the UK, all wild birds were protected by law under the Protection of Birds Act 1954 - 1976. Within the Act were four schedules under which protection could be withdrawn (Welsh Water Authority 1980). The second of these included the cormorant. It stated that pest species of wild birds may be killed or taken by authorised people without prosecution for the protection of property if it can be proven to court that it was preventing damage to crops, fruit, fisheries etc. (Welsh Water Authority 1980).

In the Act of 1981, Section 16 (1) (k) meant killing of cormorants became unlawful unless a license was granted (Figure 2.1). A license for shooting will only be granted if it is proven that it is solely for the, "purposes of preventing serious damage to .... fisheries" (Wildlife and Countryside Act 1981; Figure 2.1). Licenses are issued where:

- cormorants are / likely to be causing serious damage to a fishery;
- non-lethal techniques have been attempted and shown to have failed, or are impractical;
- there are no other evident causes of the damage;
- it is considered that shooting may help reduce the problem;
- there is no other satisfactory solution.

Kirby *et al.* (1996) reported that between 26 and 51 licenses were issued annually between 1983 and 1992. In the winter of 1996/97, 47 licenses were granted to shoot cormorants in England with 180 birds shot, and 32 licenses in Scotland with 244 birds shot.

2.5.3 *Population patterns after legal protection*

After the implementation of the EU directive throughout Member States, an initial small increase in the population occurred (Veldkamp 1996). During the mid 1980s the population of the breeding strongholds of *Phalacrocorax carbo sinensis* in the Netherlands and Denmark began to increase exponentially (Veldkamp 1996). The eutrophication of many European inland waters, which resulted in increased fish production, supported the population increase. Although Central European cormorant population growth started later than the West, the wave of increase has now reached the Baltic States and Russia. Finland and the Central and Northern territories of Sweden are on the brink of colonisation (Veldkamp 1996). Rose and Scott (1994) reported that the Northern and Central Europe population is 200 000 birds and the Baltic Sea and Mediterranean 100 000 birds.

Shifts in *Phalacrocorax carbo carbo* populations have been less dramatic (Veldkamp 1996). In Britain, the breeding population increased from 6400 breeding pairs in 1969 to 7200 breeding pairs in 1985/87 (Kirby *et al.* 1996). Lloyd *et al.* (1991) reported 8000 pairs in 1969/70 and 10 400 pairs in 1985/87, with an overall increase rate of 3 % per annum observed (Cramp *et al.* 1974; Lloyd *et al.* 1991; Debout *et al.* 1995). Regional differences have occurred, with large increases in Northern England, Eastern Scotland and Ireland, and declines in Western and Northern Scotland, the latter being linked to intense local persecution at fish farms (Lloyd *et al.* 1991).

The winter distribution of *carbo* populations in the UK was surveyed initially for the 1980 to 1982 Winter Atlas of the British Trust for Ornithology (Lack 1986). Inland colonies comprising of at least 25 birds were widespread, with overall numbers around 20 000 to 25 000 birds (Russell *et al.* 1996). From 1986, cormorants were included in national monthly counts of water bird species organised by the BTO (Russell *et al.* 1996). The surveys highlighted the pattern of cormorant distribution in the UK between 1986 and 1991 being similar to that observed in 1980, but numbers had risen markedly (Starling *et al.* 1992). In 1990/1991 the estimated wintering population was 18,700 birds. The winter population exceeded 19 000 in 1993 (Kirby *et al.* 1995).

However, it is not the *Phalacrocorax carbo carbo* wintering population figures which are raising concern amongst angling groups. It is the increasing number of over-wintering *inland* birds. Inland over-wintering cormorant numbers have been increasing steadily over the last 20 years (Sellers 1979; Feare 1988), with inland annual population growth estimated at 5 to 10 % between 1970 and 1987 (Kirby *et al.* 1995). Between 1987/1988 and 1990/1991 the rate increased by a total of 74 %, equivalent to a linear annual growth rate of 24.8 % (Kirby *et al.* 1995), and it is still thought to be increasing. The growth was observed in all regions of the UK on all habitat types, but was especially noticeable at reservoirs and gravel pits (BTO 1997).

There are a number of theories why this increase in the inland over-wintering population numbers of *Phalacrocorax carbo carbo* has occurred:

- they reflect the increase in absolute numbers of breeding birds in Britain (Kirby *et al.* 1995), aided by the protection of the Wildlife and Countryside Act of 1981;

- the numbers shot under licence since legal protection has been relatively small and thought unlikely to have affected the population as a whole (Debout *et al.* 1995);

- an influx of *Phalacrocorax carbo sinensis* from Europe. At present they comprise between 5 and 10 % of the total wintering population (Kirby *et al.* 1995). However, no evidence exists that this is an overall increase in their number in Britain (BTO 1997);

- an apparent shift in winter quarters of some birds, as shown by cormorants in Pembrokeshire, Wales. An increasingly eastward movement of cormorants in the last 30 years has been seen, with a tendency for the movement distance to be shorter (Kirby *et al.* 1995). Marion (1995) suggested the overall decrease in movement distance was due to increased competition between British bred *carbo* birds and continental *sinensis* birds over-wintering in France;

- a general shift from coastal to inland habitats. It is estimated between 50 and 54 % of birds are inland by February with evidence that birds vacate coastal waters sooner than ever before, resulting in maximum inland abundance being reached quicker (Kirby *et al.* 1995);

- as a consequence of deteriorating conditions in coastal waters due to decreased food supplies, perhaps due to competition with other species or commercial fishing. However, no evidence exists to support this (Kirby *et al.* 1995);

- reduction in human persecution inland resulting in a more suitable habitat (Kirby *et al.* 1995);

- increased availability of suitable inland habitats with heavily stocked reservoirs, lakes and flooded gravel pits (Kirby *et al.* 1996), which also create a more favourable habitat and food source.

Inland over-wintering cormorants, similar to coastal cormorants, will form large colonies. These are used for overnight roosting and the base from which they fly for daily foraging activities. Russell *et al.* (1996) documented the major winter roosts in England from September to March between 1985 and 1993 (Table 2.3).

Table 2.3 Major inland over-wintering cormorant roosts of England 1985 to 1993 (Russell *et al.* 1996).

Location	Number of cormorants
Abberton Reservoir, Essex	405
Rutland Water, Leicestershire	371
Ranworth/Cockshoot Broad	324
Queen Mary Reservoir	314
Grafham Water	294
Ouse Waste	262
Harringfield Reservoir	222

The actual number of *sinensis* birds wintering in the UK is not known. It is thought the inland numbers range from 1000 to 3000 birds (van Eerden and Munsterman 1995), equal to between 5 and 10 % of the total winter population. *Sinensis* birds are known to winter at Abberton Reservoir, Essex, with estimates of 150 birds (Ekins 1990).

Kirby *et al.* (1995) discussed cormorant population numbers in relation to available habitat. They concluded that carrying capacity of the habitats had yet to be fully utilised, so further increases in inland population numbers could be expected.

2.6 Cormorant feeding ecology

2.6.1 *Cormorant diet composition*

Cormorants are a top predator in freshwater ecosystems (Van Eerden *et al.* 1995). They are opportunistic feeders, taking advantage of feeding in fresh water in winter and coastal habitats in the summer breeding period (Van Eerden *et al.* 1995). Quantitative evaluation of cormorant diet is difficult due to problems in the methods available, resulting in biased and incomplete records of the numbers and sizes of the fish consumed (Russell *et al.* 1996). Qualitative evaluation of diet is easier to determine by three main methods: field observations, stomach contents analysis (shot birds and regurgitates) and pellet contents.

Due to the wide range of habitats exploited, cormorants catch a large variety of fish species (Veldkamp 1996). There have been many studies on the food of both cormorant subspecies, and over 115 fish species have been recorded in the diet in Europe (Veldkamp 1996). These include fish from the families Cyprinidae, Esocidae, Salmonidae and Percidae. All are well represented in the UK freshwater fish species.

Twenty species of fish have been recorded in the diet composition of *carbo* from British freshwater habitats (Russell *et al.* 1996) (Table 2.4).

The main prey species of cormorants are usually the locally dominant species (West *et al.* 1975; Welsh Water Authority 1980), with diet predictable and variable according to the availability of the principal prey fish (Kirby *et al.* 1996; Marquiss and Carss 1997).

Table 2.4 Species of fish found in cormorant diet at UK river and stillwater sites (* = present in cormorant diet at river / stillwater sites).

Species	River	Stillwater
Lamprey (*Lamptera fluviatilis* (L.))	*	
Eel (*Anguilla anguilla* (L.))	*	*
Salmon	*	
Brown/Sea trout	*	*
Rainbow trout		*
Grayling	*	
Pike	*	*
Perch	*	*
Roach	*	*
Common bream	*	*
Dace	*	
Gudgeon	*	*
Tench		*
Carp		*
Minnow (*Phoxinus phoxinus* (L.))	*	
Stickleback (*Gasterosteus aculeatus* L.)	*	

(Hartley 1948; Mills 1965; McIntosh 1978; Ransom and Beveridge 1983; MacDonald 1987b; Kennedy and Greer 1988; McCarthy *et al.* 1993; Feltham and Davies 1995; Warke and Day 1995; Russell *et al.* 1996)

This pattern has been highlighted in UK cormorant studies. Cormorants shot in spring on rivers and lakes of Scotland and Ireland during the smolt run had consumed brown trout and salmon parr and smolts (Mills 1965; McIntosh 1978; Kennedy and Greer 1988; Warke and Day 1995). Rainbow trout were a common dietary component of birds feeding in Scottish lochs where rainbow trout had escaped from fish farms (Ransom and Beveridge 1983). Cyprinids, especially roach, were an important dietary component in

British rivers (McIntosh 1978; MacDonald 1987; Feltham and Davies 1995), and perch a major prey in lakes (Hartley 1948; Mills 1965; Ransom and Beveridge 1983; MacDonald 1987; McCarthy *et al.* 1993).

A study on stillwater and riverine fisheries in England and Wales (MAFF unpublished, in Russell *et al.* 1996), found only four species made up 82 % of total diet composition, these being roach (51 %), perch (14%), bream (10%) and common carp (7%).

The size of fish taken ranges from 30 to 650 mm, with the majority in the range 100 to 300 mm (Marquiss and Carss 1997). A summary of the range of sizes by weight of different fish species is shown in Table 2.5.

Table 2.5 The size by weight of individual fish ingested by cormorants in Europe.

Family and species		Mean weight (g)	Maximum weight (g)	Source
• Salmonidae	Brown Trout	300	700	Keller 1992
	Rainbow trout		825	Ransom and Beveridge 1983
• Esocidae	Pike	383	688	Keller 1992
• Anguillidae	Eel	322	900	Keller 1992
• Percidae	Perch	35	367	Keller 1992
	Ruffe	10	15	van Dobben 1952
	Zander	108	557	Keller 1995
• Cyprinidae	Roach	202	632	Keller 1995
	Bream	144	532	Keller 1995
	Tench	286	416	Keller 1995
	Chub		795	Suter 1991

Salmon are usually consumed as smolts or large parr (Russell *et al.* 1996). There are records of adult salmon being taken, with a grilse of 1800 g consumed in Scotland (St John 1882, cited in Mills 1965), and Carss and Marquiss (1996) reported two kelts of 1000 g and 1500 g in cormorant stomachs from the same area. Brown and rainbow trout are taken in the range 100 to 400 mm, cyprinids and perch in the range 50 to 300 mm, and eels between 100 and 300 mm (Russell *et al.* 1996) (Table 2.5).

Specific data were not available for *sinensis* feeding in the UK. However, studies in Europe highlight a similar situation to *carbo* in the UK, with diet covering a wide and varied range of fish species composed of locally dominant species (van Eerden *et al.* 1991; Platteeuw *et al.* 1992; Dirksen *et al.* 1995; Veldkamp 1995; Suter 1997).

2.6.2 *Daily food intake (DFI)*

Estimates of the daily mass of food consumed by cormorants are useful to calculate the potential impact of cormorant predation on fish stocks. Extensive research has been completed in the area since Collinge (1924) reported the cormorant is capable of consuming 6800 g of fish per day. Estimates have been made based on the following methods:

- analysis of fish meals regurgitated by breeding birds;
- analysis of the fish remains in regurgitated pellets;
- feeding experiments with captive fully grown birds;
- feeding experiments with large nestlings;
- stomach analysis of dead birds;
- stomach temperature data;
- energy demand calculations.

The summary of published estimates shows wide variation (Table 2.6). This has led to contention between fisheries, conservation groups and biologists (Russell *et al.* 1996).

Kirby *et al.* (1996) agreed a DFI for the *carbo* species of between 340 and 520 g. A symposium in Bologna in 1995 concluded future estimates should be based upon the energy requirements of wild birds (Russell *et al.* 1996), but no indication of how this would be estimated was forwarded.

Table 2.6 Cormorant daily food intake estimates.

Subspecies	Sample size	DFI (g)	Source
• *sinensis*	109	130 - 411	Martijn & Dirksen 1991
	2061	243 - 540	Veldkamp 1994
		739	Feltham & Davies 1995
	1758	273	Keller 1992
	1738	365 - 516	Worthmann & Spratte 1990
	4	516	Keller 1995
• *carbo*	9	356	Hartley 1948
	74	425 - 700	Mills 1965
		470	Rae 1969
	4	650 - 700	McIntosh 1978
		661	Barret *et al.* 1990
		881	Feltham & Davies 1995

2.7 Impact of cormorants on inland fisheries

Early cormorant feeding impact studies were based on stomach analysis with impact evaluated from the proportion of commercially-important species in the diet. Studies by McAtee and Beal (1912), Lumley (1935), Cottam and Uhler (1937) and Vladykov (1943) assumed no damage was being done to the fish species populations if the proportion of that species in the diet was below 5 % of total cormorant diet composition. Another method used was based on the premise that, when a particular commercial fish species was found in the diet, then damage was being done to stocks merely by their consumption (MacIntyre 1934; Schiemenz 1936; Leonard and Shetter 1937). This resulted in many workers suggesting cormorants were harmful to the fish stocks (Gladstone 1920; Pycraft 1934; Ikade 1952). Furthermore, dramatic declines in fish numbers, caused by human interference or other factors, in certain fisheries were wrongly blamed on cormorants (Hall 1925; Pycraft 1934). A number of authors found the diet of cormorants to be very dependent upon local availability of fish (Forbush 1921;

Lewis 1929; Mendall 1936), with conclusions erring caution on general statements of cormorant impact.

Subsequent work on impact assessment of cormorants has involved research on river and stillwater fisheries and at aquaculture sites (McIntosh 1978; Osieck 1991a; Platteuw *et al.* 1992; Callaghan *et al.* 1994; Dieperink 1995; Veldkamp 1995; Modde and Wasowicz 1996; Pilcher and Feltham 1997).

2.7.1 *Cormorant predation at aquaculture sites*

Due to the close confinement of large numbers of fish, usually of the optimum cormorant foraging size, fish farms are very attractive feeding sites for cormorants.

Dieperink (1995) studied predation of hatchery-reared rainbow trout in a commercial pound net in a Danish fjord adjacent to a colony of 5000 breeding cormorants. A control was used to establish the background mortality in a pound net by preventing cormorant access to the fish. The background mortality was 15 % per day. With the protection removed, mortality increased to 98 % per day. Observations showed the cormorants were able to virtually empty the net in 30 minutes, consuming 110 fish, equivalent to approximately 50 kg total mass.

In America, Bunker (1998) estimated that double crested cormorants caused an annual loss of $20 million in the aquaculture programmes of 13 States, an industry worth $714 million. Non-lethal methods proved ineffective in controlling the cormorant predation. The US Fish and Wildlife Service published a predation order allowing lethal methods to be implemented at sites when non-lethal methods have been tried and failed.

The effects of Florida double crested cormorant (*Phalacrocorax auritus floridanus*) predation at an American channel catfish (*Ictalurus punctalus* (Rafinesque)) culture site were studied by Schramm *et al.* (1984). Two culture ponds were stocked with fingerlings of 5 to 20 cm for grow-out. Prior to stocking, there were few piscivorous birds observed near the ponds, but twelve cormorants were recorded foraging two weeks post stocking. A resident population of 13 cormorants became established within 3 months, with an estimated 246 catfish consumed per day, an average of 13 per bird. As an average catfish weighed 16 g, each bird consumed 340 g per day. It was concluded that the high consumption rate by double crested cormorants, their ability to nest inland and quickly establish a large breeding colony, imposed a significant economic risk to a culture site.

Results of other work on cormorant damage at culture sites are shown in Table 2.7 (Russell *et al.* 1996).

The literature highlights that where high concentrations of both fish and cormorants occur, the potential exists for a high level of predation. As culture farms are run for economic profit, it can be assumed the impacts are largely economic in nature, although loss of valuable broodstock may occur.

Table 2.7 Cormorant predation levels and fish consumption in aquaculture ponds.

Location	Species	Estimated loss	Cormorant population	Outcome
Netherlands	Carp	35 - 75 %	16000 bird visits per day	Farm abandoned
Germany	Carp	300 tonnes		Unprofitable to stock larger sizes
Germany	Carp	50 - 80 %		
France	Carp, perch, roach, mullet (*Mugilidae* sp.)	43 %		Some farms closed
Israel	Carp, mullet.	90 tonnes	10000	Industry continues

(Osieck 1982; Im and Hafner 1984; EIFAC 1988; Vaadia *et al.* 1989)

2.7.2 *Cormorants at stillwater fisheries*

A total of 344 returns were received in a national questionnaire of angling clubs on cormorant predation, from which it was concluded that there is an increasing problem on stillwaters, especially in SE and SW England (Carss and Marquiss 1995). Salmonid and cyprinid stillwaters were affected increasingly throughout the year. The main findings were:

- 50 to 60 % of the fish at two fisheries were claimed to have been taken;
- economic losses between £100 and £100000 were claimed, attributed to extra restocking costs and revenue loss from falling permit and subscription fees;
- two clubs experienced a fall in revenue of 50 % in 2 years and another by 20 % in 4 years;
- an unquantifiable financial loss in local economies was thought to have arisen due to falling catches attributable to cormorants.

Salmonid stillwater fisheries

Cormorant occupancy and impact were studied at 167 stillwater salmonid fisheries in England and Wales by Callaghan *et al.* (1994, 1998) between 1988/89 and 1992/93. Cormorants were widespread at inland stillwater sites throughout the year, with higher densities between October and March. Over the period, peak cormorant numbers increased up to 20 %, regardless of season. Fishery managers perceived the cormorants to be responsible for: economic losses due to fish consumption; wounding of stocked trout; affecting trout behaviour and catchability; and causing a perceived impact where anglers choose to fish waters where cormorants were not present.

To determine the impact of the cormorants on the fisheries, multiple regression analysis was used. The relationships between the numbers of trout caught at the sites and the number and mean mass of fish stocked, the angling effort, and the cormorant density were determined. Bird density was found to account for very little of the catch variation. Angling success was dependent upon stocking density at smaller sites (below 10 hectares) and dependent upon stocking density, angler density and the mean weight of fish at larger sites (above 10 hectares). However, no mention was made that where cormorants consume stocked trout, the stock density of the fishery is reduced. This lowers angler catch rate as it is directly related to the stock density (Crisp and Mann 1977; O'Grady 1980; Pawson 1982; North 1983). Additionally, each fish consumed represents a financial loss incurred by the fishery, for each fish was purchased and stocked by the fishery owner.

In a South Utah, USA, reservoir double crested cormorants and western grebes *(Aechmophorus occidentalis* Lawrence*)* ate 31.8 % and 8.8 % of stocked fingerlings respectively in 2 weeks following stocking (Modde and Wasowicz 1996). It was found the birds consumed more hatchery trout than the more abundant Utah chub *(Gila attraria* L.).

Trout were also more vulnerable to cormorant predation than other fish species in another 13 reservoirs and lakes in South Utah, USA (Ottenbacher *et al.* 1994). As the stocked trout were hatchery reared, known to have characteristics making them vulnerable to bird predation (Vincent 1960; Flick and Webster 1964; Fraser 1974; Ayles *et al.* 1976), they were more likely to be consumed than the indigenous fish. The indigenous fish were not acting as buffer populations for the trout.

The USA stocked trout fisheries were protected from avian predation by moving stocking dates to late spring or early summer to avoid the migrating avian predators (Modde and Wasowicz 1996). However, the consumption of stocked fish by resident birds was difficult to ameliorate (Modde and Wasowicz 1996).

Cormorant predation impact assessment on other recreational trout fisheries in USA have found conflicting results. Cormorant predation has impacted on some important recreational fisheries by consumption of considerable numbers of game fish (Ayles *et al.* 1976; Christie *et al.* 1987; Campo *et al.* 1988). However, little evidence of any predation impact by cormorants was found on other fisheries (Baille 1947; Carrol 1988; Findholt 1988).

Non-salmonid stillwater fisheries

Cormorant predation has been suggested to have positive impacts on non-salmonid stillwater fisheries where: removal of benthic cyprinids (e.g. common bream) has reduced water turbidity; diseased and weak fish were consumed (Feare 1988); and cormorant predation on perch has resulted in reduced predation pressure on brown trout populations, for example, Lake Windermere, UK (Macan and Worthington 1951).

In biomanipulation programmes in the Netherlands, benthic cyprinids are removed from eutrophicated, lacustrine fisheries where nutrient input has been reduced. This aims to reverse the eutrophication process by increasing zooplankton survival. This increases the level of the phytoplankton grazing and results in a decrease in water turbidity. This allows the establishment of macrophytes and populations of pike, tench and rudd (Veldkamp 1996). Cormorants have been utilised as a method of removing the benthic cyprinids, and, hence, have a positive impact on the biomanipulation programmes (Hosper *et al.* 1992).

Impact assessments in other cormorant research on non-salmonid stillwater fisheries have been limited due to restricted work on the fish population dynamics (Osieck 1991a; Platteeuw *et al.* 1992; Veldkamp 1995; Pilcher and Feltham 1997).

The impact of feeding cormorants was assessed at Rye Meads, UK, by comparing fish mortality on a control lagoon, where cormorants were excluded, with a test lagoon where cormorants were able to forage (Pilcher and Feltham 1997). Similar numbers of measured roach, carp and bream of known size were stocked into the lagoons. The experiment was carried out over the winter of 1995/96 and repeated in the period April to October 1996. A total of 303 hours of cormorant observations were made, comprising 29 foraging bouts and 339 dives. Only 12 fish were observed to be consumed.

After completion of the monitoring, the lagoons were drained and the mortality rates of the fish calculated. It was found in the test lagoon, after accounting for post-stocking mortality (calculated in the control lagoon), 52.5 % (winter) and 57.3 % (summer) of the fish stocked were lost. However, only 17 % and 6 % of these mortalities were directly attributable to cormorant predation. Thus, an alternative mortality source occurred in the test lagoon, such as nocturnal feeding by other piscivorous birds.

An observation made in the summer experiment in the test lagoon was the surviving fish demonstrated minimal growth, whilst normal growth of fish was observed in the control lagoon. This indirect cormorant impact may have implications for summer cormorant exploited fisheries, such as those in close proximity to breeding colonies.

Coastal reclamation in the Netherlands has seen the creation of many stillwaters - polder lakes - supporting major multi-species fisheries, such as IJsselmeer and adjoining lakes. Large breeding and winter populations of cormorants (*sinensis*) are present allowing study of the interaction of fish and bird populations (Table 2.8). Large numbers of cormorants were present on the lakes with high numbers of fish consumed by cormorants. Estimates of between 2 and 27 % of the fish stock were taken by cormorants on the lakes (Table 2.8).

Table 2.8 Cormorant predation figures from data on the IJsselmeer and surrounding Dutch Polder lakes.

Lake	Species	Cormorant consumption	Number of birds	Source
IJsselmeer (1938 - 40)		840 t	11600 breeding	van Dobben (1952)
IJsselmeer (1989)	Cyprinids, percids	2.7 % fish stock	>24000 breeding	Osieck (1991 a)
IJsselmeer (1987)	Cyprinids, percids	100 % commercial fishery catch		EIFAC (1988)
IJsselmeer (1991 - 92)	Cyprinids, percids	27 % fish stock		Dirksen *et al.* (1995)
Wolderwijd and Veluwemeer (1989 - 90)	All species	2 % fish stock	900 in winter	Marteijn and Dirksen (1991)
Ketelmeer (1989 - 91)	Cyprinids and perch	2 - 10 % fish stock	> 1500 in winter	Platteeuw *et al.* (1992)
Wannerpervenn (1991)	Roach, bream	5 - 16 % fish stock	> 2000 breeding	Veldkamp (1995)

IJsselmeer supports a commercially-important fishery. Hence, cormorant predation may affect commercial catches by lowering the biomass of the commercially important species. Eel, pike, perch, roach, bream, zander (*Stizostedion lucioperca* (L.)) and smelt (*Osmerus eperlanus* (L.)) are all exploited by commercial fishing and cormorants. Osieck (1991a) published biomass data allowing a comparison to be made between catches (Table 2.9).

Table 2.9 Comparison of commercial catch and cormorant predation with fish biomass on IJsselmeer in 1982.

Species	Fish biomass (t)	Commercial catch (t)	Cormorant consumption (t)
Roach	19,000	170	190
Bream	12,000	68	1
Perch	14,000	683	470
Pike	1000	70	< 24
Eel	1500	842	< 6
All species	70,000	4300	1200

It was not determined if the level of cormorant predation was compensated for by the fish population. However, the commercial catch of fish was higher than the cormorant consumption figures, so it may be assumed that such predation had a minimal long term effect when compared with commercial fishing.

Comparative cormorant predation studies have also been completed on Lake Erie, USA and Lake Malawi, Africa. On Lake Erie, the annual piscivorous bird consumption, which included cormorant predation, was estimated at 13,368 tonnes. This was equivalent to 15.2 % of the prey fish biomass required to support the walleye (*Stizostedion vitreum* Rafinesque) population during a single growing season (Maderijian and Gabrey 1995). On Lake Malawi, the annual cormorant consumption was 95 tonnes, with the average annual commercial catch (1979 to 1987) being 10,417 tonnes. The cormorants consumed 70 tonnes of species found in the commercial catches (2.7 % of the human harvest). The research concluded that cormorant predation had no significant impact on the commercial fisheries (Linn and Campbell 1992).

The relationship of prey availability and cormorant ecology has found cormorant productivity is constrained fundamentally by fluctuating fish populations (Warke *et al.* 1994). In periods where fish were in limited supply to cormorants in Lough Neagh, the birds responded with short term regulatory mechanisms, such as deferred breeding (Warke *et al.* 1994), and increased use of marine feeding sites (Warke and Day 1995).

The biomass of roach and perch in lakes has been shown to influence choice of feeding site in cormorants in Switzerland, where fish stocks govern cormorant numbers over time (Suter 1995). When the European population of cormorants was stable, the number of cormorants wintering at Lake Neuchatel increased in parallel with the increase in roach commercial catch (Suter 1995). Cormorant numbers have declined on the lake in recent years, apparently in response to declining roach stocks due to reduced nutrient input (Müller 1990, Suter 1995).

2.7.3 *Cormorants on river fisheries*

Salmonid river fisheries

Scottish salmonid rivers are constituted by salmon and trout, and species such as bullhead (*Cottus gobio* L.), minnow and stone loach (*Barbatula barbatulus* (L.)) (McIntosh 1978; MacDonald 1987; Russell *et al.* 1996). It is generally considered that cormorants are detrimental to such fisheries if only the salmonids are taken (Russell *et. al.* 1996). However, the other species may play an important dietary role for salmonid species and any predation is likely to reduce their numbers.

Cormorants were monitored on a 22-km stretch of the River Tweed, Scotland in winter 1972/73 (McIntosh 1978). Numbers peaked in December (99 birds) and declined in March. Cormorant stomach analysis found roach, salmonids and flounders (*Platichthys flesus* (L.)) were the main prey species (size range 36 to 650 mm). Salmonids, mainly parr averaging 106 mm, comprised less than 30 % of diet. The smolt run was identified as a vulnerable life stage to cormorant predation due to heavy shoaling. However, since the smolts migrated to sea between March and June, after cormorant dispersal, then losses were expected to be minimal and not affect adult returns due to compensation mechanisms (McIntosh 1978).

In reviewing the work by McIntosh (1978), Russell *et al.* (1996) discussed that any losses due to cormorants on the smolt run could not to be compensated for. Over-winter mortality, including cormorant predation, would have weakened the pre-smolt cohorts in relation to the carrying capacity of the river leaving little scope for compensatory growth and survival. Thus, any mortality associated with the smolt run would be additive mortality.

The smolt run has been found to be vulnerable to cormorant predation in salmonid river fisheries in Co. Mayo, Republic of ~~Northern~~ Ireland, where between 5 and 13.1 % of salmon and sea trout smolts were consumed annually by cormorants (MacDonald 1987). The majority of these were hatchery stock with poor predator defence responses, and 40 % were taken in the first 3 miles after release. As a positive relationship existed between the numbers of smolts migrating and the number of returning adults, any cormorant predation on the smolt run reduces the number of returning adult salmon. However, in comparison to some mortality sources, such as drift netting off the Irish coast which caught up to 80 % of returning adults, losses were probably low.

Blackwell *et al.* (1997b) found smolts figured highly in cormorant diet as they migrated to sea from the Penobscot River, Maine, USA. This was perceived as a potentially limiting factor to successful recovery of the salmon population of the river. Blackwell *et al.* (1997a) discussed that in New England and in maritime Canada, efforts to restore Atlantic Salmon by stocking with fry and smolts have contributed to the increased use of inland, riverine habitat by cormorants.

Piscivorous bird predation on hatchery-reared pink salmon fry (*Orncorhynchus gorbuscha* (Walbaum)) and chum salmon fry (*Onchorynchus keta*) was studied in North America (Scheel and Hough 1997). The losses attributable to bird predation, including the double crested cormorant, were 1.1 to 2.4 % of 2.7 to 5.9 million fry released.

The diet and impact of double crested cormorants were studied on stocked rainbow and cutthroat trout (*Onchorynchus clarki*) on an American river in 1994 (Derby and Lovvorn 1997). Prior to stocking, the bird diet was composed of 17 % trout and 61 % *Catostomus* fish species. Post-stocking, trout consumption rate increased to 60 %, with a large decrease in *Catostomus* fish species diet composition. It was estimated the birds consumed over 80 % of the trout stocked in 1994.

Thus, salmonid rivers are utilised by cormorants for feeding. The smolt run and post stocking periods of hatchery-reared fish are highly vulnerable to predation due to heavy shoaling characteristics and poor predator defence responses.

Non-salmonid river fisheries

There are few studies relating to the impact of cormorants on non-salmonid river fisheries. Staub *et al.* (1992) reported declines of 70 to 90 % in angler catches of barbel and grayling in response to cormorant predation on a number of large rivers in Germany, including the Rhine and Aare. On the River Ribble and its tributary the River Darwen, in north west England, up to 138 cormorants wintered along 36 km of river. An estimated 2 to 38 % of the stock of chub, roach and dace were removed (Feltham and Davies 1995). Compositions of fish caught by anglers and cormorants showed considerable overlap in the sizes of fish taken. Cormorants were therefore assumed to compete with anglers. Poor fish stock data prevented the determination of the extent to which the predation might reduce angler catches (Feltham and Davies 1995).

2.7.4 Summary of the cormorant predation impact research

Although a large number of cormorant feeding studies have been completed in recent years, the research has been unable to link the level of cormorant predation with the impact on the fish populations in terms of fish population dynamics, growth rates, year class strength, productivity, angler catch rates and fisheries economics. Such impact assessments are necessary in order to develop management strategies.

2.8 Non-lethal effects of cormorant predation

2.8.1 *Wounding*

Cormorant wounding occurs when a fish is attacked by the bird but it is unable to be swallowed and subsequently escapes. This may be due to a defence mechanism of the fish or that the fish was too large for the bird to swallow. Cormorant wounds on a fish can be recognised by marks consistent with being hooked and grasped by the sharp beak. A characteristic deep triangular wound or abdominal hole is evident on one side of the fish with an area on the other side where the scales have been scraped off by the lower mandible (van Dobben 1952; Ransom and Beveridge 1983; Carss 1990a; Davies *et al.* 1995). Some injured fish may die from the damage inflicted or from secondary infection (Russell *et al.* 1996). However, capture of damaged fish with healed wounds does occur, so a proportion of fish survive (Suter 1995). Wounding research has revealed variations in wounding frequency (Table 2.10). An example is the greater proportion of chub than dace wounded by cormorants in Boyce's Beck, north west England, a tributary of the River Ribble (Davies *et al.* 1995) (Table 2.10). The wounded individuals were of a greater average size than the resident chub and dace populations, showing size played a role in determining cormorant wounding vulnerability.

Table 2.10 Wounding rates of wild fish populations due to cormorants.

Location		Species	Size	Wounding frequency (%)	Source
Boyce's Beck, NW England		Dace	20-26 cm	2-7	Davies *et al.*(1995)
		Chub	26-29 cm	14-18	
		Roach		6-7	
Chew Valley Lake, UK		Trout		7-37	Russell *et al.* (1996)
Irish Loughs		Sea trout		'numbers'	Piggins (1958)
R.Rhine, Linth Canal, Switzerland		Grayling		10-16	Suter (1995)
		Pike		25-30	Staub *et al.*(1992)
Germany	River Mattig	Grayling		10-15	Kainz (1993)
	River Ebrach	Trout		50	Wissmath *et al.* (1991)
		Chub		60	
	gravel pit	White fish	800 g	'large numbers'	
	R.Rhine/Aare	Grayling		10-15	Staub *et al.* (1992)

The avoidance of fish from avian predators suggest intrinsic escape mechanisms and body defence structures are important in capture and swallow prevention (Recher and Recher 1968). Fish species whose post-capture escape mechanisms were no more elaborate than violent flexions of the body escaped swallowing by the heron species *Ardea occidentalis* and *Ardea herodias* on 0.5 % of capture occasions. Fish species possessing elaborate escape behaviour or physical structures interfering with swallowing (e.g. perch and zander) escaped on 7.8 % of capture occasions. Eels were noted to wrap and contract in escape with violent contortions of the body and escaped on 5.7 % of 88 capture occasions

2.8.2 *Population structure*

Following cormorant predation on a fishery, the fish population structure may be altered. Brönmark *et al.* (1995) found population size structures of crucian carp (*Carassius carassius* (L.)) and tench were clearly related to the presence or absence of piscivores. Where piscivores were absent the populations were dominated by small bodied individuals in high densities, but the populations consisted almost exclusively of large individuals in piscivore presence.

2.8.3 *Habitat use and feeding*

Research into shifts of habitat use and feeding of fish in cormorant presence has not been carried out. However, experiments have been completed using other predators and the results may be applicable to fish in cormorant presence.

Simple laboratory experiments on salmon parr have shown in predator presence the habitat use of parr is altered, with increased use of hiding places and reduced foraging range (Metcalfe *et al.* 1987; Huntingford *et al.* 1988). This shows predator avoidance for the parr is incompatible with active feeding, a trend shown in a number of other predator-prey studies (Caraco *et al.* 1980; Dill and Fraser 1984; Lendrem 1984; Magurran *et al.* 1985; Fraser and Huntingford 1986).

The growth and recruitment performance of fish populations may be affected by habitat choice in the presence and absence of predators (Fraser and Cerri 1982; Werner *et al.* 1983; Holopainen *et al.* 1991; Tonn *et al.* 1992; Fraser and Gilliam 1992). In the presence of a fish predator, juvenile bluegill sunfish (*Lepomis macrochirus* (Rafinesque)) obtained a greater percentage of their diet from the vegetation habitat, where foraging return rates were reduced (Werner *et al.* 1983). Declines in growth of up to 27 % resulted in these age groups. The growth rate of the larger fish, unaffected by the predation, increased due to lowered foraging competition in the open habitats. Juvenile crucian carp were shown to aggregate among macrophytes in the littoral zone of lakes in predator presence (Tonn *et al.* 1992).

Talbot *et al.* (1984) indicated that fish will feed more intensely than normal after recovering from the effects of predator stimuli. Salmon parr which were subjected to a period of starvation subsequently consumed larger meals to offset any deficiency incurred. Broekhuizen *et al.* (1994) found in this recovery phase growth rates were higher than those where continuous periods of high food availability occurred. This compensatory growth was found to be so intense that a higher net growth rate was sometimes achieved than in fish which feed continuously.

2.8.4 *Behaviour changes*

Fish behaviour may be changed during prolonged periods of cormorant foraging. Although work investigating fish behaviour changes in predator presence has not been carried out using cormorants, the results from work using other predators may be applicable.

Prey aggregation is a behavioural refuge against predator attack and has been shown to occur by a number of workers (Neill and Cullen 1974; Poole and Dunstone 1975; Milinski 1979; Magurran and Pitcher 1987), with studies indicating aggregated prey have greater individual survival than solitary prey (Radakov 1973; Neill and Cullen 1974; Morgan and Godin 1985; Pitcher 1986). Johannes (1993) showed golden shiner (*Notemigonus crysoleucas*) aggregated positively in relation to predator density (Johannes 1993). Young fish aggregated more than older fish at low predator densities whilst older fish responded increasingly to increased predator densities

2.9 Fish population dynamics

The determination of the impact of cormorant predation on inland fish populations is inconclusive, with impact often made on disputable assumptions (Draulans 1988). Little regard is given to the population dynamics of the fish species involved.

Fish population dynamics are important to consider as they affect the performance of the fishery before cormorant predation has occurred and will affect how the fish populations respond. Density dependent and independent mortality, the carrying capacity of the fishery, the scope for compensation for losses, and the density dependent and independent factors limiting performance are important considerations.

2.9.1 *Mortality rate and compensation*

The mortality rate of a population and of individual cohorts (the members of a particular year class) determine the numbers of surviving individuals with time. Factors which cause mortality at a rate related to the population size are density dependent. Factors which result in the proportional mortality of a cohort or population, irrespective of their initial density, are density independent. If the relative importance of the mortality processes can be understood within the lifetime of a cohort then estimating predation impact will be easier.

Compensation and regulatory processes are the response of the fish populations in limiting the effects of mortality losses. An example of compensation after depletion of fish stocks is the increased growth rate of the surviving individual fish resulting from decreased competition. This often increases survival, lowers the age of sexual maturity and increases fecundity. The following are examples where regulation acts on different life stages in salmonid populations (Russell *et al.* 1996) and where cyprinid compensation has occurred, although not as a result of cormorant predation.

Salmonid population regulation

- Where moderate predation by piscivorous birds on salmon and trout fry occurs, compensation for the losses is possible. Full compensation for the losses is more likely in favourable environments where there is an adequate number of spawners.

- Predation losses of salmonid parr and smolts are likely to consist of dispensatory mortality with little scope for compensation. This is because survival and growth of parr and smolts are mainly density independent.

- Inter-cohort interactions may compensate for predation losses. This is dependent on the physical structure of the habitat and the growth rates of the fish. Where habitat is favourable compensation for the losses is more likely as growth rates are greater resulting in increased survival.

Cyprinid compensation

- Where large numbers of fish eating birds, including the cormorant, concentrated, taking advantage of areas with prey species in high densities, their foraging resulted in, "the thinning populations of fish to a point where those escaping are able to find adequate food to make rapid growth and thereby attain large sizes," (Barret 1971). Thus, remaining individuals were able to compensate for the predation by increased growth, survival and fecundity).

- Commercially exploited vendace (*Coregonus albula* L.) populations in Lake Pyhajarvi, Finland, compensated for an exploitation rate of 90 % removal of total cohort production by increased growth rates, increased fecundity and a lowered age of maturity. Fishing mortalities of other species in the lake were low with no change in their population dynamics (Sarvala *et al.* 1994).

- Botsford (1981) reported on several aquatic populations where commercial fishing resulted in population declines to low levels which remained low even after the exploitation was removed. The growth rate of the fish had increased with the population losses, which remained after exploitation. The continued increased growth rate had contributed to the continued depressed fish abundance.

2.9.2 *Carrying capacity*

The carrying capacity of a fishery is the maximum potential size of the fish populations as limited in time by constraining factors. Examples of potential constraints are food availability, territorial and aggressive fish behaviour, and shelter availability. If the population reaches a size where these factors begin to constrain it, then the carrying capacity of the system has been reached. Carrying capacity restraints may be broadly categorised into density independent and density dependent factors (Section 2.10.3, 2.10.4). These factors limit the potential size of fish populations before any outside mortality factors, such as bird predation, exert their influence on the population.

2.9.3 *Density independent factors*

Temperature

Temperature is probably the most widely studied variable which influences recruitment to coarse fish populations (Cowx *et al.* 1995). It affects the growth of fry in their first summer directly through its effect on metabolism and food consumption, and indirectly through food abundance (Cowx *et al.* 1995). The effects of temperature may be summarised as affecting spawning time, growth and year class strength.

• Spawning time:
Spawning time has fundamental consequences on the amount of growth achieved by fish in their first growing season. Subsequently, this is important in year class strength determination. The water temperature at which spawning occurs varies from species to species. The majority of cyprinids, for example, roach, common bream, barbel and bleak, spawn at water temperatures between 12 and 16 °C (Cowx *et al.* 1995). This usually occurs in May or June (Cowx *et al.* 1995). Exceptions include dace (6 to 10°C) (Kennedy 1969, Mann 1974, Alabaster and Lloyd 1982), chub (18 to 20°C) (Alabaster and Lloyd 1982), and perch (8.5 to 13.5°C) (Dalimier *et al.* 1982). Thus, dace have a longer growing season than chub.

• Growth:
The correlation of water temperature with growth is relatively high, suggesting temperature has a considerable influence on the amount of growth achievable in the first growing year (Cowx *et al.* 1995). Broughton and Jones (1978) found that the growth of 0-group roach in the River Hull was correlated most strongly to the number of degree days over 14 °C. Cowx *et al.* (1995) observed this relationship on the Rivers Ure and Ouse, but found on the River Swale that 12 °C was the critical temperature. Similar relationships were shown for species such as dace, chub and gudgeon. The greater the fry length at the end of the first growing year, the higher the expected over winter survival, and the stronger the resulting recruiting year class.

• Year class strength:
Cyprinid populations are dominated by a small number of dominant year classes (Mills and Mann 1985), which translates into angler catches as catches are dominated by these year classes. As angler catches are a function of stock abundance (Crisp and Mann

1977; O'Grady 1980; Pawson 1982; North 1983), where strong year classes exist angler catches may be expected to be good. Year class strengths have been correlated to water temperature by a number of workers (Le Cren 1958; Craig *et al.* 1979; Pivnicka 1982; Mills and Mann 1985; Craig 1987; Cowx *et al.* 1995). Cowx *et al.* (1995) found that the correlation between year class strength and temperature was not as strong as that between growth and temperature and suggested other biotic and abiotic factors may be equally important in regulating year class strength.

Flow rates

Temperature and food supply explain much of the variance in year class strengths. However, poor year classes do occasionally occur where conditions appear favourable, so other factors are exerting an influence (Cowx *et al.* 1995). Evidence has been shown that flow rate can be important in recruitment dynamics as fish larvae and 0-group juveniles can be lost through downstream drift where flows are strong (Copp and Cellot 1988). In years of high river flows this loss will be greater, especially if that river has embanked channels (Cowx *et al.* 1995). Swimming speed is a function of body length (Cowx *et al.* 1995), so years of higher spring temperatures will reduce this loss due to early spawning allowing an increased growth period.

Food supply

After temperature, food supply is probably the most important variable influencing the growth and survival of fry, and subsequent year class strength (Cowx *et al.* 1995). As the two factors are intrinsically linked, it is difficult to discriminate between them (Cowx *et al.* 1995). There have been few studies on the precise diets of coarse fish fry. Those carried out (Starkie 1976; CGRT 1980; Cowx 1980; Hodgson 1993) highlight a high similarity between species.

As all year class strength variance cannot be explained by temperature and flow fluctuations, other key factors may have a regulating effect, including species diet similarity. This was shown by year class strength production in UK rivers in 1989, 1990 and 1991. If temperature was the key regulating factor, each year would have produced a strong year class (Cowx 1998). However, it was generally found 1989 produced a strong year class, 1990 was average and 1991 slightly above average (Cowx *et al.* 1995). As it appears production of two or three successive year classes are rare, the high density of juvenile fish in the 1989 cohort may be successfully competing for food resources with the 1990 cohort, resulting in reduced food resources and increased mortality in the 1990 cohort. Hence, food supply can act as a regulating factor in year class strength production through diet similarity as it may result in inter-cohort competition.

Food supply in the initial period of life is also extremely important in juvenile fish survival. Adequate food resources must be available for larvae within a short period of hatching so the fish can move from endogenous to exogenous food quickly and avoid starvation (Cowx 1998). Thus, hatching must coincide with explosions of plankton or epibenthic food resources. In this period, not only should adequate food be available, but it should also be of optimum size and be preferred to alternative, less abundant sources, if starvation in the 0-group fish is to be avoided and survival maximised.

2.9.4 *Density dependent factors*

Density independent factors often obscure the influence of density dependent factors (Mills and Mann 1985). This makes their impact on fish populations difficult to assess (Cowx *et al.* 1995). However, Townsend (1989) showed that where a single year class is responsible for the production of eggs, strong density dependent factors are in evidence in the population dynamics. These may act through fecundity, egg and fry survival, and the interactions of competition and predation.

Where resources of food and space are limiting, intra-specific competition may occur within the same year class, which may result in a depressed growth rate (Cragg-Hine and Jones 1969; Persson 1983). An overabundance of piscivorous species may result in cannibalism which has been suggested as a density dependent control mechanism limiting their numbers (Kipling and Frost 1970; Grimm 1981; Mann 1982; Craig and Kipling 1983), as it increases the shortage of other suitable prey (Cowx *et al.* 1995).

A juvenile competitive bottleneck where inter-specific competition exerted a profound effect on the population dynamics was discussed by Johansson and Persson (1986) and Persson (1983, 1988). An overlap in roach and perch food supply caused an early diet shift in the perch. This increased intra-specific competition in the perch fry, which lowered their growth rate. When the roach population numbers were reduced, the perch delayed this diet change, decreasing the intra-specific competition resulting in increased growth rates.

2.9.5 *Put and take trout fisheries*

A different situation occurs on put and take trout fisheries as the fish are stocked on an annual basis and removed by anglers. In recent years, there has been a rapid increase in the number of these specialist fisheries, which cormorants have been able to exploit (Anglian Water 1997).

In general, the fisheries are stocked with batches of rainbow trout at the beginning of the trout angling season and at intervals throughout the season to maintain the stock at a level capable of providing anglers with acceptable returns (Pawson 1982). Thus, population dynamics are irrelevant here and monitoring of cormorant impact cannot be measured in the same way as on cyprinid fisheries.

However, cormorant predation mortality analysis can be carried out in a different way. The loss of stocked fish in a put and take fishery occurs through fish being taken by anglers (F) and through natural mortality (M). Natural mortality includes the combined effects of disease, starvation, predation and ageing. Where a fishery is operated in a consistent manner over a number of years, for example, in the numbers and sizes of fish stocked, the fishing mortality will tend to remain constant over the period (Pawson 1982). Thus, any significant increase in natural mortality due to cormorant predation will manifest itself quickly in the analysis of fisheries data, for the stock will be decreasing at a greater rate than usual. This will be reflected in reduced angler returns (Pawson 1982).

Overall, fish population dynamics and constraining environmental effects must be taken into consideration before cormorant predation impact is assessed. It has to be stressed that the factors that govern fish population dynamics with time are very complex and inter-related (Figure 2.2), with only a simplistic over-view presented here. Cormorant predation is only one potential constraint of fish production. Its significance on fish production will be governed by the relationships of the other constraining factors (Figure 2.2).

Cormorant predation impact assessment and interpretation is very subjective. For some, very small losses to cormorants are unacceptable. Others argue that birds do not pose any threat overall to stocks and, if left alone, a prey/predator equilibrium balance between fish and bird numbers will be reached, and compensation mechanisms in the fish populations will limit any long term damage to stocks. However, it is indisputable that once a fish has been eaten by a cormorant it no longer has a value and cannot be caught again (Russell *et al.* 1996).

Plate 2.1 Cormorant - *Phalacrocorax carbo carbo* (Laguna 1998).

Wildlife and Countryside Act 1981

CHAPTER 69

4.—(1) Nothing in section 1 or in any order made under section 3 shall make unlawful—

(a) anything done in pursuance of a requirement by the Minister of Agriculture, Fisheries and Food or the Secretary of State under section 98 of the Agriculture Act 1947, or by the Secretary of State under section 39 of the Agriculture (Scotland) Act 1948;

(b) anything done under, or in pursuance of an order made under, section 21 or 22 of the Animal Health Act 1981; or

(c) except in the case of a wild bird included in Schedule 1 or the nest or egg of such a bird, anything done under, or in pursuance of an order made under, any other provision of the said Act of 1981.

(2) Notwithstanding anything in the provisions of section 1 or any order made under section 3, a person shall not be guilty of an offence by reason of—

(a) the taking of any wild bird if he shows that the bird had been disabled otherwise than by his unlawful act and was taken solely for the purpose of tending it and releasing it when no longer disabled;

(b) the killing of any wild bird if he shows that the bird had been so seriously disabled otherwise than by his unlawful act that there was no reasonable chance of its recovering; or

(c) any act made unlawful by those provisions if he shows that the act was the incidental result of a lawful operation and could not reasonably have been avoided.

(3) Notwithstanding anything in the provisions of section 1 or any order made under section 3, an authorised person shall not be guilty of an offence by reason of the killing or injuring of any wild bird, other than a bird included in Schedule 1, if he shows that his action was necessary for the purpose of—

(a) preserving public health or public or air safety;

(b) preventing the spread of disease; or

(c) preventing serious damage to livestock, foodstuffs for livestock, crops, vegetables, fruit, growing timber, or fisheries.

3.—(1) The Secretary of State may by order make provision with respect to any area specified in the order providing for all or any of the following matters, that is to say—

(a) that any person who, within that area or any part of it specified in the order, at any time or during any period so specified, intentionally—

(i) kills, injures or takes any wild bird or any wild bird so specified;

(ii) takes, damages or destroys the nest of such a bird while that nest is in use or being built;

(iii) takes or destroys an egg of such a bird;

(iv) disturbs such a bird while it is building a nest or is in, on or near a nest containing eggs or young; or

(v) disturbs dependent young of such a bird,

shall be guilty of an offence under this section;

(b) that any person who, except as may be provided in the order, enters into that area or any part of it specified in the order at any time or during any period so specified shall be guilty of an offence under this section;

(c) that where any offence under this Part, or any such offence under this Part as may be specified in the order, is committed within that area, the offender shall be liable to a special penalty.

(2) An authorised person shall not by virtue of any such order be guilty of an offence by reason of—

(a) the killing or taking of a bird included in Part II of Schedule 2, or the injuring of such a bird in the course of an attempt to kill it;

(b) the taking, damaging or destruction of the nest of such a bird;

(c) the taking or destruction of an egg of such a bird; or

(d) the disturbance of such a bird or dependent young of such a bird.

(3) The making of any order under this section with respect to any area shall not affect the exercise by any person of any right vested in him, whether as owner, lessee or occupier of any land in that area or by virtue of a licence or agreement.

(4) Before making any order under this section the Secretary of State shall give particulars of the intended order either by notice in writing to every owner and every occupier of any land included in the area with respect to which the order is to be made or, where the giving of such a notice is in his opinion impracticable, by advertisement in a newspaper circulating in the district in which that area is situated.

EXPLANATORY NOTE

(This note is not part of the Regulations)

Part I of the Wildlife and Countryside Act 1981 ("the Act") contains provisions for, amongst other matters, implementing Council Directive 79/409 EEC on the conservation of wild birds.

These Regulations make amendments to Part I of the Act for the purposes of further implementing that Directive.

Subsection 4(3) of the Act provides certain defences to the killing or injuring by an authorised person (as defined in section 27(1) of the Act) of any wild bird, other than a bird included in Schedule 1 to the Act, contrary to section 1 or to any order made under section 3 of the Act. Under section 4(3)(c) those defences include action necessary for preventing serious damage to livestock, foodstuffs for livestock, crops, vegetables, fruit, growing timber, or fisheries.

Regulation 2—

(a) adds "inland waters" (as defined in regulation 4) to the list of matters in section 4(3)(c) in respect of which action to prevent serious damage may be taken;

(b) adds sections 4(4)-(6) to the Act to limit the defence available under section 4(3)(c) by providing—

(i) that an authorised person must show that there was no other satisfactory solution to the killing or injuring;

(ii) that if the killing or injuring was foreseeable, a licence under section 16 of the Act must be applied for as soon as reasonably practical and not determined;

(iii) that the authorised person must notify the agriculture Minister (as defined in section 27(1) of the Act) as soon as reasonably practical after the killing or injuring.

Section 16(1) of the Act provides for licences to enable action to be taken which would otherwise be unlawful under sections 1, 5, 6(3), 7 and 8 and orders under section 3 of the Act. Regulation 3(3) adds section 16(1A) to the Act to provide that an appropriate authority shall not issue a licence for any purpose in section 16(1) unless it is satisfied that there is no other satisfactory solution. Regulation 3 also makes various minor amendments to section 16.

Figure 2.1 Articles from the Wildlife and Countryside Act 1981 relating to the protection of cormorants.

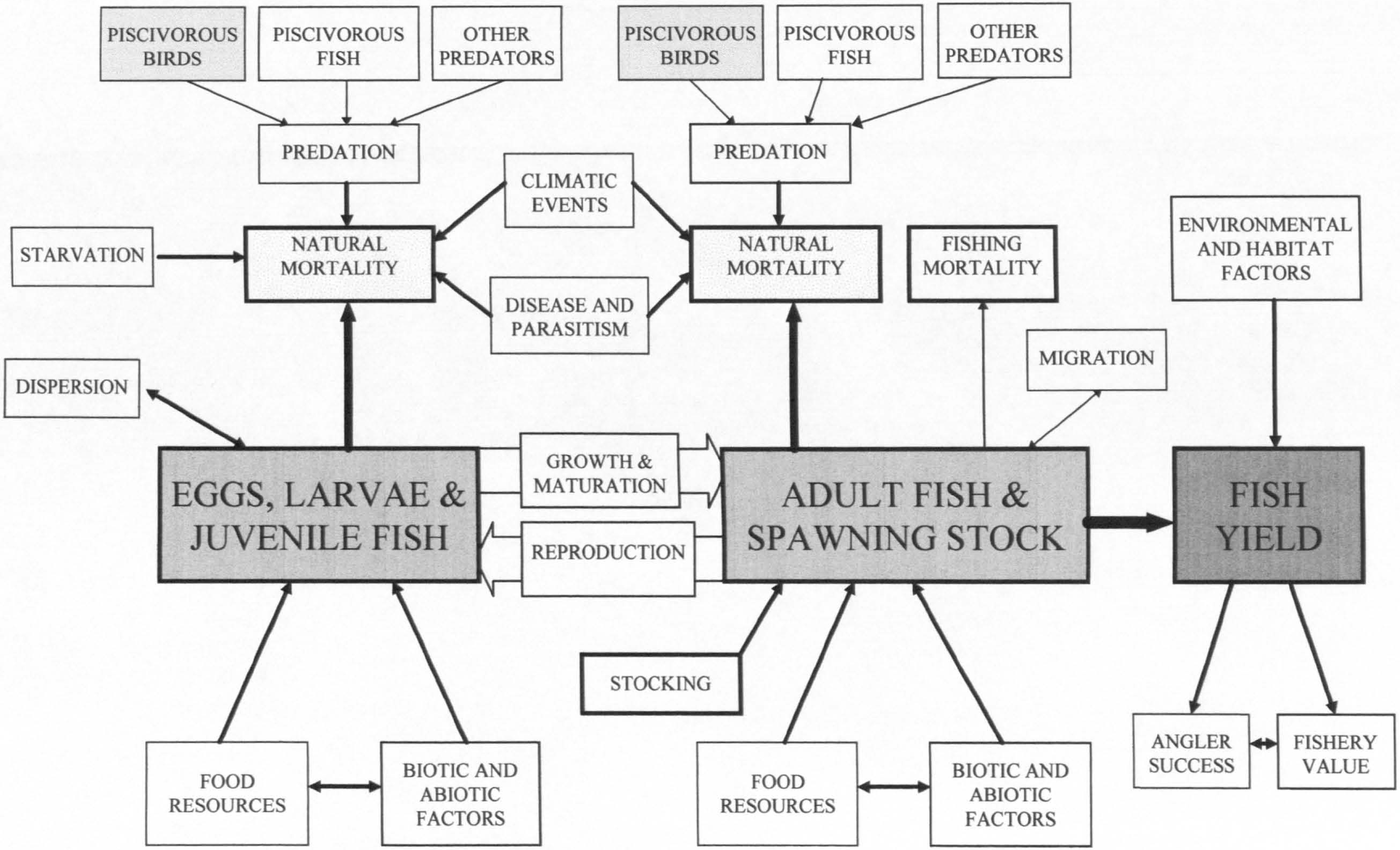


Figure 2.2 Model illustrating the complexity of factors that govern fish abundance in a cyprinid fishery (Cowx 1998).

3. GENERAL METHODS AND DATA ANALYSIS

3.1 Sites

The study sites were located in two regions of the UK where over-wintering cormorant populations were known to roost, the Midlands and the North West of England (Figure 3.1). This enabled the research to be completed in areas with known and established over-wintering cormorant roosts. River and stillwater fisheries, with populations of cyprinid and salmonid species, were utilised for the cormorant observations and fisheries surveys. This allowed comparison to be made of cormorant predation impact in respect of fisheries type and the dominant fish species. The study sites utilised are shown below (Figure 3.2, 3.3), with full site descriptions given in the appropriate chapter.

Midlands

- Holme Pierrepont Rowing Course (SK 620395) - a cyprinid stillwater fishery.
- Middle River Trent (SK490311-SK648424) - a cyprinid lowland river fishery.
- Colwick Park Trout Lake (SK 610395) - a put-and-take rainbow trout stillwater fishery.

North West

- Grimsargh Number 3 Reservoir (SD 590347) - a cyprinid reservoir fishery.
- Lower River Ribble (SD 462280-SD 711382) - a cyprinid and salmonid river fishery.

3.2 Cormorant monitoring methodology

The cormorant monitoring data were collected by colleagues at Liverpool John Moores University, as part of the MAFF study, "Case studies of the impact of fish-eating birds on inland fisheries in England and Wales," (Feltham *et al.* 1999; Section 1.1.5). Monitoring of cormorant occupancy and feeding ecology provided data relating to feeding success rates and the annual numbers and biomass of fish consumed at each specific fishery. The monitoring comprised of the following activities:

- cormorant counts (including monthly co-ordinated counts, roost counts and dawn flight counts);
- cormorant feeding observations;
- shooting.

The contribution of Liverpool John Moores University in collecting the data is acknowledged and its use in the study is to support the fisheries data interpretation in elucidating cormorant predation impact.

3.2.1 *Cormorant counts*

The abundance and distribution of cormorants at the sites was monitored by counting the number of birds at each site once each month. Counts began at first light for all sites, with additional counts at still waters completed during feeding observations. On the River Trent, the river was divided into four sections, with each section counted by separate observers walking upstream until count completion. The methodology for the River Ribble was similar, except it was only divided into two sections. The stillwater site counts were carried out from a parked vehicle, due to good access round each of the sites (Feltham *et al.* 1999).

During counts, the approximate locations of cormorants were plotted on count maps together with the time of sighting. Bird activity was assigned one of the following categories: feeding, flying or loafing/roosting (Feltham *et al.* 1999).

Additionally, a monthly count was undertaken at the Midlands main cormorant night roost (Attenborough, Section 4.5.1) and the North West main cormorant night roosts (Jacksons Bank and Stubbins Wood, Section 8.6.1) to establish the seasonal fluctuations in cormorant numbers in each region (Feltham *et al.* 1999).

3.2.2 *Cormorant feeding observations*

Routine observations of foraging cormorants were made using Opticron 8x41 magnification binoculars and a Geoma x25 telescope. Observations were made from a fixed point where vision was easy and habitat provided good camouflage. The Holme Pierrepont Rowing Course, Colwick Park Trout Lake and Grimsargh Reservoir observations were generally carried out from a parked car. The River Trent and River Ribble observations were carried out from bankside strategic points. Cormorant feeding observations commenced at dawn and continued while birds were active at the venue or until weather conditions made further foraging bouts unlikely, for example, low light intensity and high wind conditions (Feltham *et al.* 1999). During the feeding observations, the following cormorant data were collected:

- behavioural category (swimming, diving, loafing, preening, wing drying, flying etc.);
- the general location of bird, plotted on site maps, including time of day;
- the duration of feeding bout;
- the distance covered during feeding bout;
- the number of dives and the proportion of successful dives;
- the proportion of successful feeding bouts;
- the prey caught and their approximate size in relation to cormorant bill-length (Feltham *et al.* 1999).

The advantage of using feeding observations rather than alternative methodologies, such as stomach analysis of birds that have been shot, to assess cormorant diet was that large amounts of dietary data could be collected at known times, with minimal disturbance to the cormorants. However, identification of cyprinid species was difficult at most observational distances and was only possible to species level at Holme Pierrepont, where observations were possible at short range. Fish below 50 mm were also difficult to identify and were collectively termed 'fry' (Feltham *et al.* 1999).

Collection of feeding data to species/species group level, with their sizes estimated to the nearest 50 mm, allowed accurate comparison with the fish species represented in fisheries surveys and angler catches. The size of each ingested fish was estimated by comparison of fish size with cormorant bill length, which was taken as 115 mm. This followed bill measurements taken on shot cormorants from the River Ribble (B. Wilson pers. comm.). Use of length-weight equations enabled fish weight to be determined (Section 3.5.6), allowing an estimation of the biomass of fish removed from the fishery to be determined during each feeding observation. The feeding observations and bird count data were then used in a Monte Carlo Simulation model which estimated the biomass of fish consumed over the whole winter period at each site (Feltham *et al.* 1999, Section 3.4.2).

3.2.3 *Shooting*

Shooting was used to validate the feeding observations of the fish species predated on by cormorants, particularly of those fish below 100 mm. The results were compared to the observer's field notes taken during that bird's last feeding bout, so verifying the feeding observations. The method utilised analysis of the birds' stomach contents, which provided more detailed assessment of the diet than simple field observations. However, sample sizes were usually very small, due to the difficulty in obtaining a license to shoot cormorants (Section 2.5.2), and provided little data on spatial and temporal diet variation.

Extraction and examination of cormorant stomach contents was undertaken using a standard method (Carss *et al.* 1997). An incision was made in the body cavity of the cormorant, starting at the cloaca and passing upwards to the base of the skull. The stomach was separated from the other internal organs, removed and opened from top (anterior) to bottom. Any whole fish were removed, identified, measured and weighed. All partially digested material was washed out of the stomach into water-tight containers, with a biological agent added to remove any remaining flesh, and stored in an oven at 35°C for 2 to 3 days. The remains were then washed through a sieve, before being air-dried prior to analysis.

The dried samples were examined for the identification of 'key bones' (Feltham and Marquiss 1989). These bones included pharyngeal teeth, first vertebra and jaw bones. A reference collection and published descriptions of these bones allowed determination of the prey species (Feltham and Marquiss 1989, Veldkamp 1995). Indicator key bone measurements allowed prey size to be determined from published regression equations (Feltham 1990, Veldkamp 1994). Examples included pharyngeal bone gape, first vertebra width and length of the pre-opercular bone.

3.3 Fisheries monitoring methodology

The fisheries survey programme involved sampling the fish populations at four specific times of the year between October 1995 and July 1998. The rationale for this is explained below. This programme was not carried out in full on the River Ribble because permission to electric fish was denied at certain times of the year due to concerns about damage to salmon stocks (Section 8.2), and on Grimsargh number 3 Reservoir, because of essential maintenance work (Section 7.1).

- September/October

This related to the peak abundance of the fish stocks prior to any over-winter mortality. It allowed the assessment of the potential juvenile recruitment and gave an evaluation of the status of the fish stocks prior to the influx of wintering cormorants on the study sites. The period also marks the cessation of summer fish growth.

- January

This related to the period when cormorant numbers were typically at, or near, their peak. Hence, it was when cormorant predation pressure on the fish populations was likely to be greatest.

- March/April

These data gave information on the pre-recruitment phase from spawning and gave an assessment of the over-winter survival of fish. Cormorants also began to leave the study sites to breed in coastal areas in this period.

- June/July

These data corresponded with the post spawning phase of cyprinid populations and the period when fish growth rates were at their maximum and cormorants were generally absent from the sites.

3.3.1 *Electric fishing*

The quantitative assessment of fish stocks in large fresh water bodies is notoriously difficult, because no methods are available which adequately sample the fish populations to allow such analysis to be undertaken (Cowx 1996). Consequently, the methods adopted to assess the status of the fish populations in the study sites were largely semi-quantitative and based on gear calibrations (Bayley 1985; Cowx 1996). This approach was deemed acceptable, because it provided an estimate of stock size and structure on which the impact of cormorant predation could be measured.

The electric fishing gear calibration method enabled an estimate of population size to be determined for each site on each sampling occasion. The approach involved determination of gear efficiency on suitable fisheries prior to its application to the survey site by depletion sampling (hand-held electric fishing) or by determining efficiency from the proportion of the first catch against the estimated population size (boat mounted electric fishing). The calculated gear efficiency was then related to each survey site, allowing the calculation of the total population size by its application on the actual sample size.

In electric fishing, fish capture relies on the application of an electric current to the water body having an effect on the nervous system and muscle of the fish. The nervous response of the fish ultimately leads to it being immobilised, allowing removal from the water by a hand net. The fish is unharmed and can be returned to the water once appropriate biological measurements have been taken (Section 3.3.6). Two types of electric fishing gear were used.

Hand-held electric fishing

The hand-held electric fishing gear consisted of twin hand-held insulated poles each supporting a ring-shaped electrode (Plate 3.1). Each hand-held pole was connected to a control box powered by a 2 kVA generator (Plate 3.1). The output from the control box was 100 Hz PDC with an operating current of 1-2 A. An electric field was created around each electrode and fish in the effective range of the electric field were attracted towards the ring electrodes. This gear was used in the marginal areas of lakes in the study.

The gear efficiency was assessed by calibration experiments undertaken on a number of small rivers (Cowx 1994, 1996, 1998). This involved quantitative assessment of the fish population size using a three-catch depletion method (Zippin 1959, Cowx 1983). The efficiency of the gear was then determined as the proportion of fish caught in the first catch as a measure of the total population size. The efficiency of the gear (P) was found to vary between 0.32 and 0.50, with a mean of 0.41. These values were used on the actual sample size of fish at each appropriate site to give the estimated total population size.

Boat mounted electric fishing

On larger rivers and lakes, hand-held electric fishing efficiency is low, as a disproportionate amount of effort is required for small catches of fish (Hickley & Starkie 1985). Hence, for surveys on large rivers and lakes, boat-mounted electric fishing gear, specifically constructed for surveying these water bodies (Harvey & Cowx 1995a, b), was used (Plate 3.2).

The basic design consisted of two electrode arrays, each consisting of three concentric rings of decreasing radii: 0.80 m, 0.53 m, 0.26 m. Each ring was electrically isolated and supported a number of pendant electrodes constructed from stainless steel wire rope. Each ring array was supported on a boom structure fixed to the front of a stable, cathedral-hulled boat (Plate 3.2). Electrical power was supplied from a 7.5 kVA Allam generator via a Millstream electric fishing control box. The output from the control box was a precise ¼ sine wave at 100 Hz at all power settings. The fishing control box fired the electrode rings sequentially and fish were attracted to the ring arrays.

A different gear calibration methodology had to be applied to the large water bodies, as depletion sampling was not possible due to survey areas being impossible to isolate. The efficiency of the boat-mounted gear could not be calibrated for individual water bodies as no measure of the absolute population size was possible. However, the gear had been calibrated for large water bodies in general (Harvey 1996), and the efficiency was found to range between 0.10 and 0.36 (mean 0.23). The efficiency varied due to the size of water body being sampled, depth, temperature, water conductivity, and size and species of target fish.

The effective electric field width was determined from experimental work as 8 m (Harvey 1996). Hence, the area fished on each survey was: *distance fished* x *electric field width*. Fish density was calculated as: *(catch/area fished)/efficiency*.

3.3.2 *Seine netting*

Seine netting is a technique that is generally used on large stillwaters or canals. It relies on encircling the fish with a wall of net which is hauled to the bank for the removal of fish. The method was not applied to the river sites because the water current would have made the hauling of the net difficult, and it could not be applied to Holme Pierrepont and Colwick Park Trout Lake due to underwater obstructions. Hence, its use was limited to Grimsargh number 3 Reservoir.

The seine net used was 75 m long and 3 m deep with a mesh size of 20 mm, and was set from a boat. Efficiency of capture of fish was taken as 50 % (Coles *et al.* 1985), with the efficiency applied as for electric fishing catches to obtain an estimate of total population size. All fish captured in the net were transferred to water filled containers for data collection (Section 3.3.6).

3.3.3 *Micro-mesh seine netting*

Electric fishing is known to be selective against juvenile fish (Zalewski and Cowx 1990), although previous trials showed the boat-mounted electric fishing gear caught fish as small as 20 mm (Harvey 1996). Consequently, it was deemed necessary to use a supplementary method to capture juvenile fish at some study sites. Because juvenile fish are generally associated with the shallow, vegetated marginal areas of rivers and lakes, especially in the summer, micro-mesh seine netting was appropriate to provide information on this component of the fish stock.

The micro-mesh seine net used was 25 m long, 3 m deep and had a 6 mm mesh size. It was operated by wading into the water body and encircling a certain area of water. All fish contained within the net were transferred to water filled containers for data collection (Section 3.3.6).

3.3.4 *Hydro-acoustic surveys*

Unfortunately, electric fishing and seine netting did not provide an absolute estimate of fish population size or biomass, parameters which are relevant in assessing fish predation by cormorants. In recent years there have been considerable developments in the use of hydro-acoustic gear in shallow lakes and rivers (Kubecka 1996).

Hydro-acoustic techniques provided information on the fish distribution, biomass and length frequency. However, fish community structure and population characteristics (for example, species composition and growth rates) are not assessed. Thus, the methodology was only used to complement data obtained from the other survey methods. Hydro-acoustic estimates were considered as minimum estimates only, for they were only effective in the open water due to echo interference ('noise') from the bottom and bank in shallow areas. Also, resolution of dense shoals of fish into multiple fish targets, rather than a single target for each fish in the shoal, was a problem in the data analysis and meant single fish targets were only used in data analysis, with multiple targets not used. Thus, only a minimum estimate of fish density was able to be achieved.

Hydro-acoustic surveys were only possible on the River Trent and Holme Pierrepont Rowing Course. The surveys were undertaken by the Environment Agency (Midlands Region). A Simrad EY500 portable echo-sounder controlled by a Toshiba T4900CT laptop computer was employed using a 4x10 degree, 120 KHz split beam transducer from a 5 m dory hulled boat. The sampling method applied to each water body is described in the appropriate study section.

3.3.5 *Angler catch data*

Angler catch data have been used by a number of authors as an indicator of the performance of a fishery in both riverine and lake systems (Cooper and Wheatley 1981; Cowx 1990, 1991; Axford 1991). In cyprinid fisheries, information is provided on angler exploitation rates and changes in fish population community structure (Cowx 1990; O'Hara and Williams 1991). However, it cannot provide a quantitative assessment of the fish stock (Hickley and Starkie 1985). The method relies on the recording of various parameters from angling matches and pleasure anglers, including: total weight caught at each peg, time spent fishing, species composition and number of anglers fishing. Collection of angler catch data was undertaken at a number of study sites, including the Stoke Bardolph section of the River Trent, where data were collected by the Nottingham and District Federation of Anglers Society.

On put-and-take trout fisheries, angler catch per unit effort has been found to be dependent on the stocking policy of the fishery, with the relationship between trout stocking and catch per unit effort well documented (Crisp and Mann 1977; O'Grady 1980; Pawson 1982; North 1983). Catches have been shown to be a function of the current stock density and previous stock/catch relationships (North 1983), with higher angler catch rates generally attributable to increased stock abundance. This relationship is less evident on cyprinid fisheries due to the influence of weather and water conditions on angler success, switching of angler target species in response to these changes, the perceived abundance of alternative species and the varied ability of anglers (Jacklin 1995). However, as angler catches have been shown to reflect changes in fish community structure (Cowx 1990; O'Hara and Williams 1991), angler catch per unit effort is likely to reflect the relative fish stock abundance changes in the fishery, subject to the factors outlined by Jacklin (1995).

Angler catch data was collected on angler return forms and included approximate estimates of sizes and importance of each fish species caught, length of time fished, and in the case of matches, the number of anglers fishing. These data were used to estimate various angler activity parameters (Section 3.5.12). In some cases, attendance at fishing matches allowed a sub-sample of fish to be collected for comparison with data collected by other survey methods.

3.3.6 *Fish data collection*

All fish captured by the sampling methods were transferred to water filled containers for data collection. Individual fish were identified, measured (fork length (mm)), and several scales removed for later analysis in the laboratory. The individual weights of fish from a representative proportion of the catch were measured to generate site- and species-specific length-weight relationships. The total weight of all individuals for each fish species was also measured. The general condition of each fish was noted, paying particular attention to any wounding attributable to cormorants. Following collection of data all fish were returned live to the water.

3.4 Cormorant data analysis

3.4.1 *Analysis of cormorant diet and occupancy*

The data obtained from the feeding observations (Section 3.3.2) allowed the calculation of the catch composition and length frequency of the species predated upon at each site (Section 3.5.1 and 3.5.2). This allowed comparison of the data from the fisheries surveys and angler catches. The feeding success of the cormorants at each site was calculated as the percentage of the number of dives successful in a feeding bout and as the percentage of feeding bouts where at least one fish was captured.

The roost count data were analysed by month to show the seasonal variation in cormorant abundance in the regions. Cormorant occupancy at each site by month allowed comparison with the changes in cormorant abundance at the main roost, and with changes apparent at the site, such as stocking.

3.4.2 *Estimation of biomass removed by cormorants over a winter period - the Monte Carlo Simulation*

The detailed site-specific cormorant feeding observations and cormorant occupancy allowed the calculation of biomass of fish removed by the birds during each winter period at each site. Due to the number of sites covered by each observer, cormorant observations were not completed at each site on a daily basis. Hence, observations at each site were limited. Thus, a model had to be developed which would be able to utilise the temporal and spatial data collected by the field observer at each site, to estimate the biomass of fish consumed over the whole winter period. The model was developed by Liverpool John Moores University, utilising a Monte Carlo Simulation methodology (MCS).

To estimate the biomass removed, a starting point was required. This was an estimate of the amount of fish removed by the cormorants during observations. Simple models have been used, utilising such data, to provide 'knife-edge' estimates. However, these have been criticised for their limitations (Marquiss and Carss 1994, Feltham 1995), as they generate estimates which are isolated in time with no estimate of maximum and minimum values. Consequently, the estimations are of limited accuracy. The development of the MCS model sought to address the limitations of the simple models by developing a more realistic model (Feltham *et al.* 1999).

The starting point of the model is:

$$yi = N.c.pi \qquad \text{(equation 1)}$$

where: yi = mass of species i removed from the fishery by the birds in the period;
N = the number of bird days at a fishery (number of birds present multiplied by days);
c = cormorant daily food intake (g);
pi = proportion of species i in the diet (Feltham *et al.* 1999).

The output is the mass of the species removed from a defined area, for a given time (Marquiss and Carrs 1994, Feltham 1995). However, equation 1 is limited in use because:

- N, c, pi are usually assumed to be annual constants;
- no confidence limits have been set on the data;
- it suggests only one estimate of biomass removal for a fishery.

The constants of N, c and pi cannot be assumed to be annual constants, for the number of cormorants utilising the fishery, their diet, feeding behaviour and daily food requirements vary with time and the different parts of a fishery utilised for feeding. Also, fish populations are known to be highly mobile, the mass of fish removed from different areas of a fishery are unlikely to be the same, and temporal variation in fish predation must be considered. Thus, multiplying the single constant values, without incorporating variation about them, will produce a 'knife-edge' and potentially inaccurate estimate of biomass removal. The actual level of fish biomass removed will probably lie between two extremes, a lower and an upper limit, which can only be set using variance around the constants of *N, c* and *pi* (Feltham *et al.* 1999).

Thus, equation 1 can be used as a building block for the model and is only used to estimate the likely mass of fish removed from smaller sections of the fishery during discrete time periods. Summing the resulting values would enable the biomass removed over the whole winter period to be calculated. Hence, values of *N, c* and *pi* were estimated for different areas of the specific sites during the course of the winter, resulting in the equation:

$$yi = yi1 + yi2 + yi3 + yi4 + \ldots\ldots yin \qquad \text{(equation 2)}$$

where: $yi1 + yi2 + yi3 + yi4 + \ldots\ldots yin$ represent the individual estimates of *yi* at each study site (Feltham *et al.* 1999).

For the river study sites, n was the number of sections of river sampled multiplied by the number of times they were sampled in the winter. For the stillwater sites, n was the number of times the site was sampled each winter (Feltham *et al.* 1999).

Derivation of estimates of variation in *N, c* and *pi* enabled the resulting distributions to be substituted for the constant values in equation 1 and 2. This allowed the confidence limits to be set around the biomass removal estimate. The general methodology for the generation of these distributions is detailed below. Site specific methodologies were sometimes used to derive *N, c* or *pi* values, with their rationale explained in the relevant sections.

The distribution in *N* - bird days at the fishery

Despite being expressed as a single parameter, the value of *N* encompassed three factors:

- the number of birds recorded during counts;
- the proportion of these birds that met their DFI at the fishery; and
- the number of days cormorants are present at the fishery.

Hence, these factors were incorporated into equation 1, to derive equation 3 (Feltham *et al.* 1999):

$$yi = (N \times d \times f) \times c \times pi \qquad \text{(equation 3)}$$

where: *N* = number of cormorants counted at a fishery;
d = number of days cormorants present;
f = proportion of cormorants that actually fed there on a given day.

Assuming the cormorant counts reflected the average number of cormorants present on each day of the month, estimating variance in bird days required only the determination of *N* and *f* variation. *N* was derived by recalculating the monthly count data as three point moving averages. This provided more realistic estimates of how the numbers of cormorants recorded in each area changed with time. Variation was measured around these values by calculating the standard deviation for each monthly count, the preceding count and the count immediately after. Thus, estimation of standard deviation in November would utilise the three point moving averages for October, November and December. This allowed a normal distribution with a quasi-empirically-derived mean and standard deviation to be substituted for the constant value of N (Feltham *et al.* 1999).

The variation in f was derived utilising certain assumptions. As individual marking of cormorants at the study sites was not possible, variation around f could not be estimated quasi-empirically. Field observations allowed each cormorant to be designated a behavioural category - feeding, loafing or flying. Thus, f could be estimated as the proportion of cormorants observed feeding during counts. This would be regarded as a minimum estimate (f_{min}), as it cannot be assumed that all roosting and flying birds would not feed at the study site during the day. The maximum value of f, (f_{max}), would be all the birds recorded roosting and flying at the site during counts (Feltham *et al.* 1999). As the counts were recorded during the first three hours of daylight, the model assumes:

- most loafing/roosting birds observed have already fed at the site, or were about to do so;

- the roosting birds showed a high degree of site fidelity and were unlikely to have fed elsewhere.

However, as cormorants were known to fly to other study sites to feed in and outside of the study area, f_{min} and f_{max} were quantified as (Feltham *et al.* 1999):

f_{min} = the number of birds seen feeding during counts;

f_{max} = (1.0 x feeding birds) + (1.0 x roosting birds) + (0.5 x flying birds) (equation 4)

<u>The distribution in *c* - daily food intake</u>

The contentious issue of DFI estimation in cormorant ecological studies has been discussed in Section 2.6.2. Because of limited data on cormorant DFI, it was difficult to provide a variance around one DFI figure. The estimates utilised were based in the energy requirements of cormorants, using c_{min} as 0.4 kg day^{-1} and c_{max} 0.8 kg day^{-1} (Feltham 1995; Grémillet 1995; Feltham and Davies 1996; Feltham *et al.* 1999).

<u>The distribution in *pi* - the proportion of species *i* in the diet</u>

Species *i* refers either to the precise species (at Holme Pierrepont) or the species group identified, for example, cyprinids (at all the other study sites) in cormorant diet. Values were derived from field observations and cormorant stomach analysis from shot birds. The proportion by mass of each prey type taken during individual foraging bouts in a given area and during a particular month was calculated with the median and inter-quartile range of the data recorded. These data were used to generate appropriate distributions to substitute in to the model for the constant *pi* (Feltham *et al.* 1999).

Thus, the distributions of *N*, *c* and *pi* were calculated to provide the lower and upper limit of the biomass of fish removed. The distributions were then applied to the MCS model, to derive the required calculated data, by use of the Minitab computer program. The model generated known distributions around each of the input parameters, prior to randomly sampling them to provide a single biomass estimate of *yi* for the site in question during the particular time period. This value was calculated by month for each study site and for each species and size group. The method was repeated 100 times, the values averaged with the range of the potential amount of biomass removed from the site representing the confidence limits on the data (Feltham *et al.* 1999).

The generated predation losses from the model were integrated with the fisheries data to estimate the fish losses attributable to cormorants over the study period (Section 3.5.10). The MCS-generated fish biomass losses were broken down into numerical data (using site specific or empirically derived length-weight equations, Section 3.5.5) and into age class data (Section 3.5.3), to allow comparison with the status of the fish stocks present at each study site (Section 3.5.9).

3.5 Fisheries data analysis

3.5.1 *Species composition*

Species composition was expressed as the percentage of each species caught in the fisheries survey by site. The data were evaluated on an annual basis, and displayed as pie charts. This allowed direct comparison of the fish species caught in fisheries surveys with those in angling catch surveys and those predated on by cormorants.

3.5.2 *Length frequency distribution*

Length frequency distributions of each fish species were derived from the catches in fisheries surveys. The methodology involved assigning each fish length of a particular species into a 10-mm length class and determining the total number of fish in each size class. The length frequency distribution was evaluated on an annual basis and was used to discriminate the age group modes.

Observations of size of fish predated on by birds allowed classification into 50-mm size classes only. Hence, for comparison, the length frequency distribution of fish derived from fisheries surveys and angler catches were reassigned to 50-mm size classes. This elucidated the sizes of fish vulnerable to cormorant predation in relation to the resident fish population as revealed by fisheries surveys and angler catches.

3.5.3 *Age and growth determination*

The determination of the age and growth of fish is an important tool in the assessment of fish population dynamics (Bagenal 1978). The age and growth of fish was determined by the interpretation and counting of annual growth checks (annuli) which appear on the scales of the fish (Bagenal & Tesch 1978). These are formed during the periods of faster (summer) and slower (winter) growth found in fish species of temperate regions.

All individual fish were measured for fork length (mm) and approximately four scales were removed from each fish, apart from when large numbers of a particular species were caught, when a representative sample was obtained. The representative sample involved removing a sample of scales from a maximum of 20 individuals in each 10 mm size class. The scales were removed from the shoulder region of the fish, above the lateral line and below the insertion of the dorsal fin. This is where the scales were first laid down and provided a full growth history of the fish (Bagenal and Tesch 1978).

The age and growth of each fish was determined using standard methodology (Bagenal and Tesch 1978). Scales from each individual fish were examined under a microfiche projector. The fish were aged by counting the number of annuli on the scales taken from each fish. More than one scale was examined to ensure correct interpretation of the annuli. The scale radius was measured from the nucleus to the scale edge, along the dorsal-ventral axis.

For each fish species the regression relationship between fish length (mm) and total scale radius (arbitrary units) was determined as:

$$\text{Length} = a + b \,(\textit{Scale radius}) \qquad \text{(equation 5)}$$

where a and b are constants.

For each individual fish, the length when each annulus was laid down was calculated from the scale radius to each annuli using equation 5. This calculation was repeated for each fish in a particular species and the mean length for each age from all the fish in each species was calculated.

The resulting length for age data was applied to growth models to provide more detailed information on the performance of the fishery (Section 3.5.4). The growth models used were the Von Bertalanffy growth model (1938), the growth index (Hickley and Dexter 1979) and the relative growth index (Kempe 1962; Mann 1973).

3.5.4 *Growth models*

Von Bertalanffy model

Fitting of the Von Bertalanffy model to the growth data allowed two important parameters to be determined, L_∞ (the mathematical asymptote of the growth curve, often referred to as the final or maximum theoretical size of the fish species); and K (the catabolic coefficient, or the rate at which the fish grows towards L_∞) for each fish species.

The parameters L_∞ and K were calculated from a Ford-Walford plot of L_{t+1} (length at time $_{t+1}$) against L_t (length at time $_t$). A Ford-Walford plot is formulated on the basis that the growth rate of fish decreases with age. A regressed line fitted to the plot of L_{t+1} against L_t will approach the 45° diagonal of the graph and at the point of the interception, $L_{t+1} = L_t$, and length has reached its asymptotic value, L_∞.

The slope of the regressed line in the relationship L_{t+1} against L_t is equal to e^{-k}, and K (the catabolic coefficient) was calculated as:

$$K = e^{-k} \qquad \text{(equation 6)}$$

where: K = catabolic coefficient; k = the slope of the graph of L_{t+1} against L_t

The values were compared to growth parameters from other water bodies.

Growth indices

Two growth indices were determined. The first, the relative growth index, involved calculation of relative growth in different years to identify periods of good and poor growth of a particular species (Kempe 1962; Mann 1973). The second method, the growth index, compared growth in different years to a standard derived for a particular species (Hickey and Dexter 1979).

The relative growth index

The average length increment at each age was calculated and used as a standard. The growth of each year class in each growth year was then calculated as a percentage of this standard. The mean growth rate for each year was calculated as a mean of these percentages for each age group (Mann 1973). An average value below 100 % indicated a poor growth, a value above 100 % indicated a good growth year.

The value of the method is in a fishery where growth is below the national standard, good and poor growth years may still be identified.

The growth index

This method was identical to that described for the relative growth index except that instead of using the average length increment from the age and growth data, standard growth rate data were used as the reference (Hickley and Dexter 1979).
Additionally, instantaneous growth rates (G) were calculated:

$$G = (\log L_t - L_{t-1}) / t \qquad \text{(equation 7)}$$

3.5.5 Length-weight relationships

Length-weight relationships of fish were represented by:

$$\textit{weight} = a(\textit{length})^b \qquad \text{(equation 8)}$$

where a and b were constants.

The logarithmic transformation gave the straight line relationship:

$$\log \textit{weight} = \log a + b\ (\log \textit{length}) \qquad \text{(equation 9)}$$

The length-weight relationship for each fish species was derived using a subsample of fish measured for individual length and corresponding weight. The length-weight relationship was then used for derivation of biomass estimates (Section 3.5.11).

3.5.6 *Mortality rate*

Mortality rates are an important parameter in the analysis of fish population dynamics. The change in the number of fish with time is a basic consideration in determining the relative level of production within a fish population. A high mortality rate highlights a sharp decline in population numbers with age, with most individuals dying early in life.

A low mortality rate usually indicates a population with more individuals surviving to an old age.

The standard mortality model for fisheries is:

$$Z = M + F \qquad \text{(equation 10)}$$

where Z = total mortality; M = natural mortality; F = fishing mortality.
The natural mortality (M) consisted of mortality (M_n), due to starvation, disease and predation from non piscivorous bird sources, and piscivorous bird predation mortality (M_{bp}), included if cormorants were feeding on the fishery. In UK catch and release fisheries, fishing mortality (F) is considered zero, hence the relationship becomes:

$$Z = M = M_n + M_{bp} \qquad \text{(equation 11)}$$

To calculate natural mortality (M), the age class was plotted against the natural logarithm of the numbers in the age classes. The inverse slope of the regression line represents natural mortality (M).

The mortality rates derived for each specific fishery were compared to data from other UK fisheries to assess whether mortality was high or low. It should be noted that natural mortality is the rate of decline of the population with time and is linked to the survival rate of the population (Section 3.5.7). It is not a percentage, so a mortality rate of 0.50 does not imply 50 % of fish die annually.

3.5.7 *Survival rate*

The survival rate represents the ratio of the number of fish alive at the end of a time period relative to the number of the same group which were alive at the beginning of that time period. It is assumed the group is closed except for total mortality, i.e. numbers are only influenced by mortality.

Two main methods were used to calculate the survival rate. Equation 12 uses the total mortality calculated from equation 11 to derive survival rate:

$$S = e^{-Z} \qquad \text{(equation 12)}$$

where S = survival rate
Z = total mortality

Alternatively, the numbers of fish at times $_t$ and $_{t+1}$ are used to derive survival rate as:

$$S = N_{t+1} / N_t \qquad \text{(equation 13)}$$

where S = survival rate
N_t = numbers at time t
N_{t+1} = numbers at time t_{+1}
The calculation of survival rate from the mortality rate (equation 12) is critical to the cohort reconstruction (section 3.5.9) and allows mortality schedules to be produced for

the data. This analysis is important as mortality and survival directly influence stock abundance. Note, angler catches are a function of stock abundance and where external pressures influence stock abundance, angler catches will be affected.

3.5.8 *Year class strength*

Year class strengths (YCS) are an important parameter in the understanding of the dynamics of fish populations. Year class strength is used to show the dominance of certain cohorts in the population structure and the influence they can have on fisheries, both at the present time and in the future. The year class strength was determined by back-calculating the number of fish (N_0) that would have been recruited to the population at time t_0 (Cowx in press), assuming constant mortality throughout life (equation 14). Although mortality is known to be higher in the juvenile life stages, this was irrelevant for this procedure because a comparative index of YCS was generated based on mortality in >1 year old fish.

$$N_0 = N_t \exp Z_t \qquad \text{(equation 14)}$$

where Z = Total mortality rate
N_0 = Numbers in starting population
N_t = Numbers at time t
t = time

Year class strength was calculated as the number of fish recruited divided by the mean number recruited from all year classes, multiplied by 100. A value above 100 is considered a strong year class, a value below 100 is considered a weak year class.

3.5.9 *Cohort reconstruction analysis (life table analysis)*

Cohort reconstruction was only undertaken at study sites occupied by cormorants and where fisheries data was adequate. It involved constructing life tables of the fish populations, and the method will be referred to throughout the thesis as cohort reconstruction analysis.

This analysis allowed the status of the main fish species populations to be determined under the influence of cormorant predation and assessment of how the fish populations would possibly respond with the cormorant predation removed. The two main components to the analysis involved reconstructing the cohorts, by use of life tables, from fisheries survey data before adding cormorant predation data derived from the Monte Carlo Simulation (Section 3.4.2).

Reconstructing the cohort was undertaken from data collected for each individual study year. The following methodology was applied to each fish species predated on by cormorants.

- A starting point was required with the first sample of each survey year used. This ensured there were no repeat catches of individual fish which may have occurred in later surveys. The lengths of the fish captured were assigned to an age using the length for age growth data derived from back-calculation. This allowed the assessment of the numbers of fish in each age group.

- Due to the bias in sampling methods against younger age groups, the data rarely showed the highest numbers of fish in the younger age group with an incremental decline in numbers with age, as would be expected. Instead, a representative decline in numbers with age did not occur until the fish were older. In roach, this may have been at age 2, 3 or 4. In common bream it may have occurred at a greater age. Once the representative decline in numbers had been found, the mortality rate for this part of the population was calculated using the methodology outlined in section 3.5.6.

- The mortality rate was then converted to the survival rate using equation 12. This rate was applied to the number of fish in the age group where the representative decline was first observed. This calculation gave an estimate of the number of fish present in the previous age group. Thus, if the exponential decline was first observed at age 4, application of the survival rate on their numbers gave an estimate of the number of individuals that were present at age 3. This calculation was repeated until the numbers at age 1 were found.

- The reconstructed numbers for age were then weighted for year class strength. The number in each year class was multiplied by a YCS multiplying factor = YCS/100, where YCS = Year class strength for fish species derived for a specific year.

- The final stage was to take account of the gear efficiency and area sampled. The generated numbers were multiplied, using the derived survey gear efficiency (Section 3.3.1) and the area sampled, to give the total number of fish in the cohort in the total area of the water body. This figure was then converted into the number of fish per hectare which completed the cohort reconstruction.

Using this methodology the cohort was reconstructed by use of life table analysis and the addition of cormorant predation data undertaken (Section 3.5.10).

3.5.10 *Addition of cormorant predation data*

The Monte Carlo Simulation output (Section 3.4.2) provided the biomass and number of fish taken by cormorants from each age group in each winter period. These data were applied to the data from the cohort reconstruction to illustrate the numbers of fish present with and without bird predation.

The reconstructed cohort shows the number of fish present after a winter of cormorant predation. Note that cohort reconstruction for the October 1995 fisheries surveys do not have predation data applied as that would have occurred in winter 1994/95, before the study began. The cohort reconstruction from October 1996 fisheries survey data utilised MCS data from winter 1995/96. The cohort reconstruction from October 1997 fisheries survey data utilised MCS data from both winter 1996/97 and 1995/96. The cohort reconstruction from June 1998 fisheries survey data used MCS data from the winters of 1997/98, 1997/96 and 1995/96 applied. Natural mortality rates were applied to the historical MCS data to take account of fish that would have died from natural causes, if they had not been eaten by cormorants, before being applied to the model.

The cohort reconstruction data alone details the numbers of fish present with cormorant predation. The cohort reconstruction with the MCS data applied details the numbers of fish that would be present (subject to natural mortality) in the absence of the cormorant predation. These data are displayed graphically and highlight differences in numbers which can be attributed to cormorant predation alone, and represents a direct measure of impact.

3.5.11 *Biomass*

The numbers in each fish species group with and without cormorant predation were converted to biomass using the length-weight relationships (section 3.5.5) to show minimum, mean and maximum values. These data were then compared to estimates derived from hydro-acoustic surveys and biomass removed directly by cormorants.

3.5.12 *Angler catch data*

Angler catch data allowed the assessment of angling effort and performance on specific fisheries. Catch forms were collected from anglers fishing at Holme Pierrepont and the River Trent over the study period. The data were used to determine:

- total fishing effort, calculated as the total number of anglers multiplied by the number of hours fished;
- catch per unit effort (g man hour^{-1}), calculated by total weight (g) of catch against the total effort (man hours) in the fishing period;
- percentage of anglers weighing in, calculated as the percentage of anglers with catch against the total number of anglers fishing;
- species composition of catch. This was calculated by two alternative methods:

  i. where annual data were sparse the total number of fish in each species was represented as a percentage of the total number of fish caught in a particular season;

  ii. where data were adequate, the index of relative importance (RI) was calculated where (Cowx 1990):

  RI = (% relative abundance + % occurrence) * 0.5 (equation 15)

  - Percentage relative abundance was derived by expressing the total points score for each species as a percentage of the total points awarded for all species. The species points scores were calculated from the angler catch forms where the most frequently caught species in a particular angling match were awarded 4 points, the second most frequent 2 points and any other species caught 1 point, with the total scores for each species each year.

  - Percentage occurrence values was determined as the percentage of all matches in which a species was represented in the angler catches.

The data were compared to historical data from similar fisheries in the UK to assess performance of the particular fishery.

Figure 3.1 The location of the study sites in Europe and the UK.

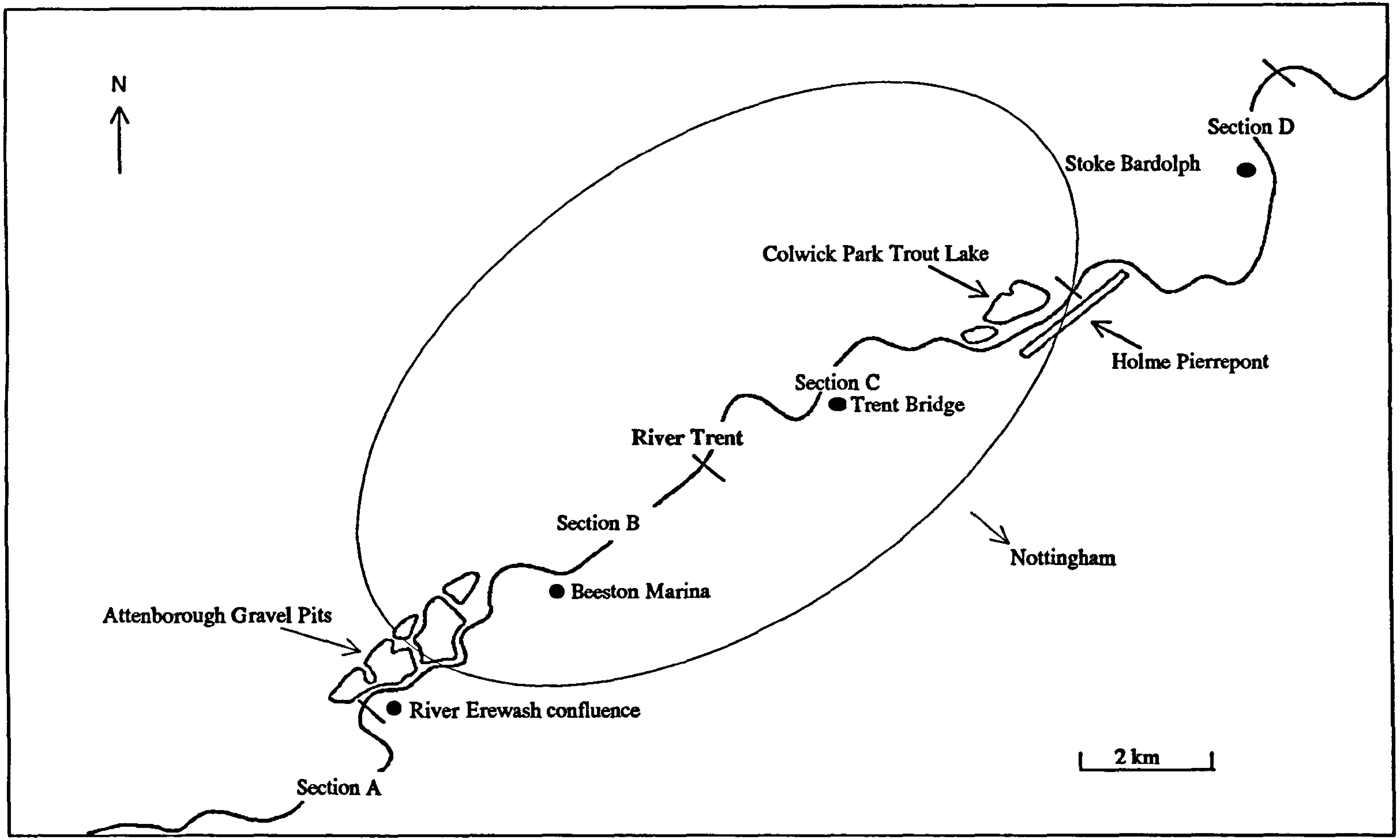


Figure 3.2 The location of the River Trent (with cormorant count sections marked), Holme Pierrepont and Colwick Park Trout Lake in the Midlands study sites.

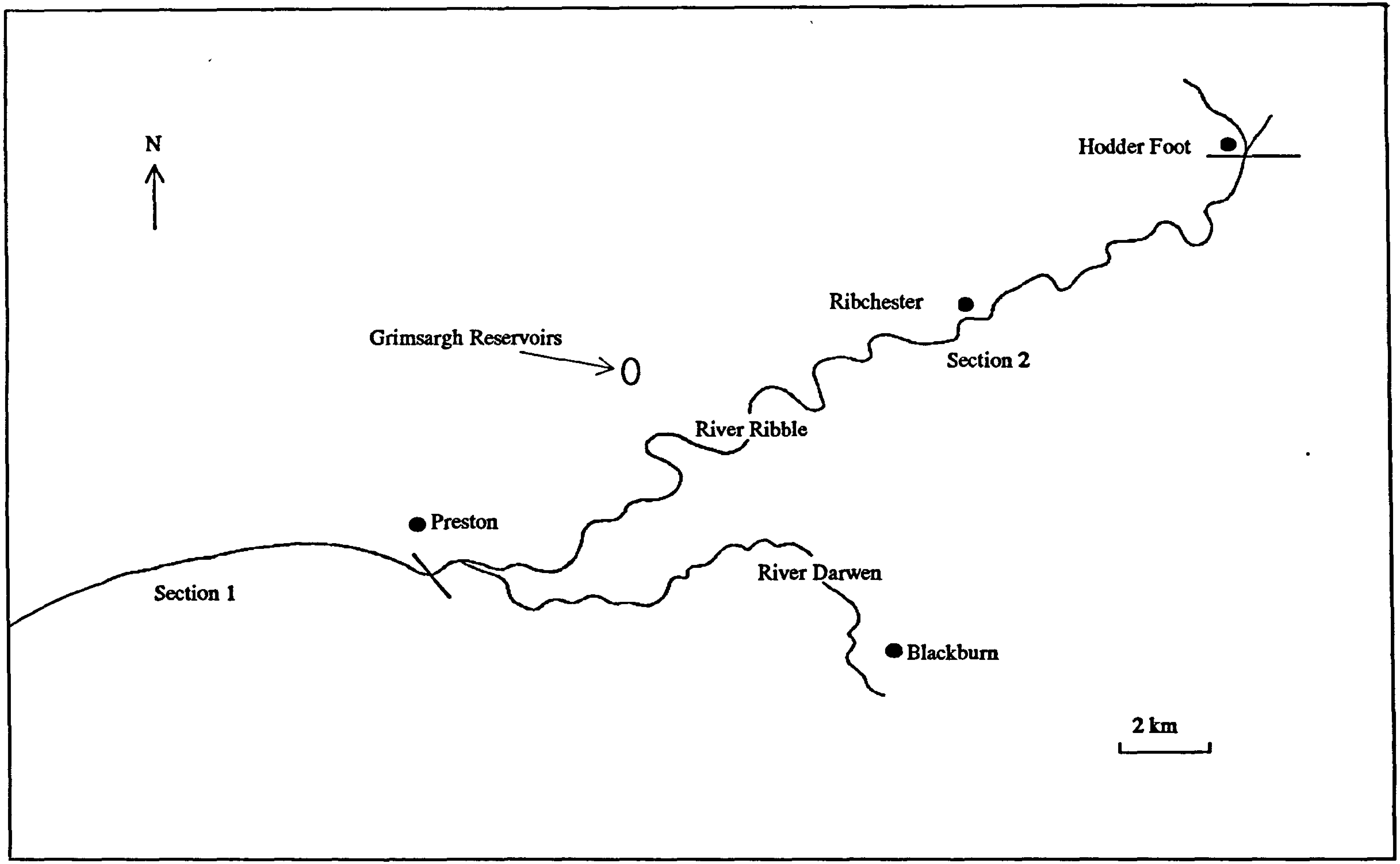


Figure 3.3 The location of the lower River Ribble study sites and Grimsargh Reservoirs in Lancashire, North West England.

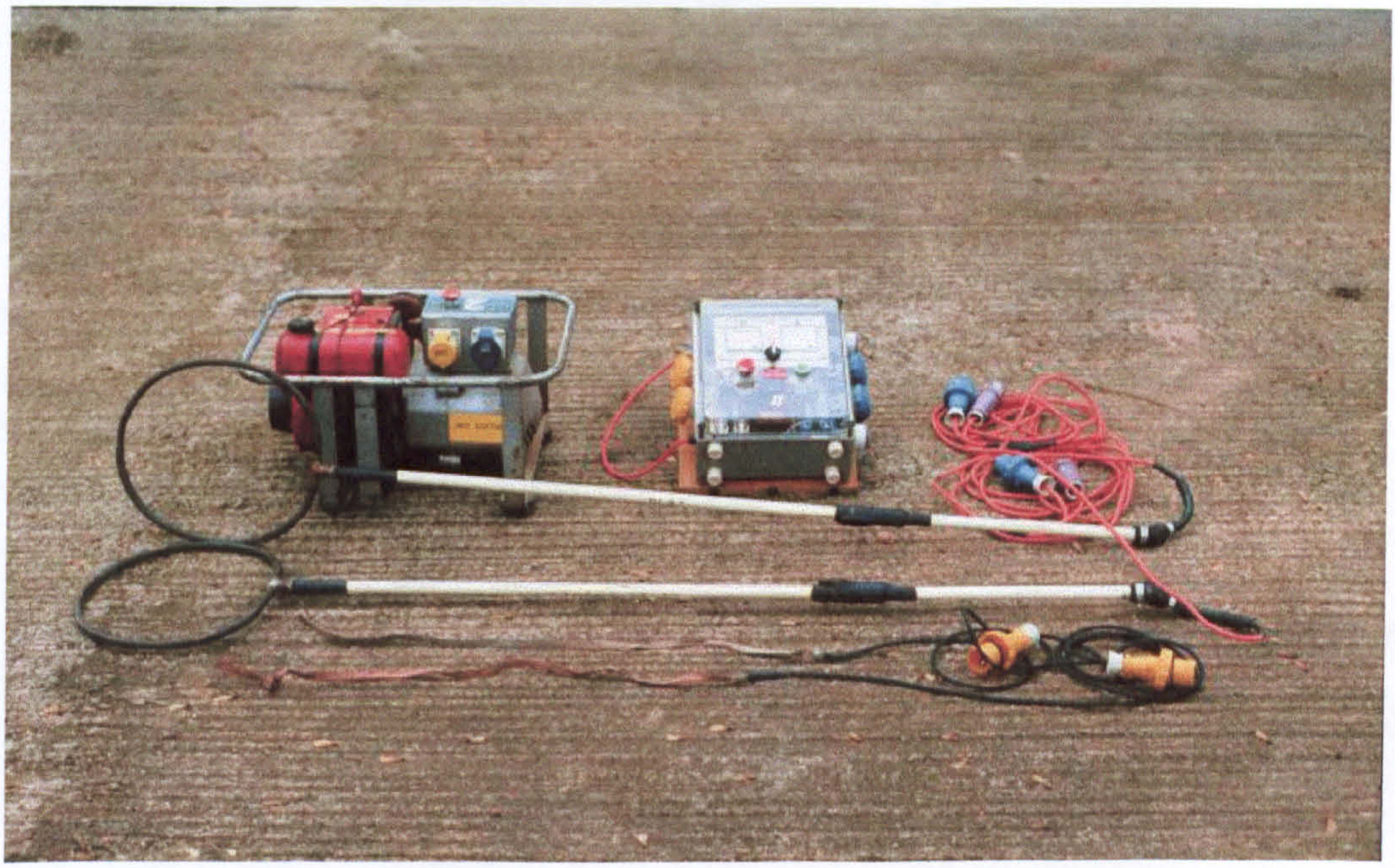


Plate 3.1 Example of hand-held electric fishing equipment, displaying generator, control box and electrodes (anode and cathode).

Plate 3.2 Boat mounted electric fishing equipment, displaying the two electrode arrays, each consisting of three concentric metal rings.

4. HOLME PIERREPONT ROWING COURSE, NOTTINGHAM

4.1 Introduction

4.1.1 *General features*

Holme Pierrepont Rowing Course (SK 618393) is situated at the National Water Sports Centre, Nottingham and is managed by the English Sports Council. The rowing course is adjacent to the River Trent, approximately 3 km south-east of Nottingham City Centre (Figure 3.2). The rowing course was constructed from disused gravel workings and flooded from the River Trent to form a lake 2215 m long and 135 m wide, with a total water area of 28.68 ha (Plate 4.1).

4.1.2 *Site management*

Holme Pierrepont Rowing Course is used primarily for water sports such as rowing, sailing, windsurfing and canoeing. The lake is also used for recreational angling and was considered a premier coarse angling venue in Britain and Europe between the early 1980s and 1993. Large catches of cyprinid species were taken regularly from all areas of the lake (S. Sparke pers comm.). Convenient access to pegs and the generally even distribution of fish throughout the lake led to the venue being used for the 1994 World Angling Championships.

The 1994 World Angling Championships saw very poor catches recorded with a general perception of a decline in angling quality on the lake at that time (B. Pluckrose pers. comm.). There are a number of theories purporting to explain this decline, but many parties blamed the large numbers of over-wintering cormorants that foraged on the lake during winter 1993/1994 and have since returned annually. The site management assumes angling visits declined after 1994 due to adverse publicity in the angling press in relation to the poor catches in the World Angling Championships (M. Thompson pers. comm.). However, angler usage of the lake has been in decline since 1992 (Table 4.1). Angling revenue and angler usage of the lake was known to peak at £40 000 in the mid-1980s (B. Pluckrose pers. comm.), with a large decrease observed between 1992 and 1996 (Table 4.1). There has been a marked increase in angler usage of the lake in 1997 and 1998, with a Home International Competition held on the lake on 1 and 2 August 1998. However, angler occupation of the lake in 1998 had been reduced by 88 % in comparison with 1992 (Table 4.1). Thus, historically, the lake attracted large numbers of anglers providing revenues in excess of £20 000 per annum, but at present in the region of £1000 to £3000 per annum (Table 4.1).

Table 4.1 Number of anglers and financial income generated from angling at Holme Pierrepont Rowing Course, 1992 to 1998.

Year	Total number of anglers	Total income from angling (£)
1992	8626	28447.10
1993	3862	20266.60
1994	1314	9312.10
1995	103	304.50
1996	147	1138.50
1997	555	1603.75
1998	1043	2789.11

The management brief for the rowing course is as a water sports lake to be used primarily for aquatic recreational sports training and school activities. Consequently, anglers have little input into the management of the lake, although the management are concerned about the possible impact of over-wintering cormorants and their role in the decline in the number of anglers using the fishery.

4.1.3 *Habitat characteristics*

Holme Pierrepont Rowing Course has a uniform topography of maximum central depth 3.5 to 4.0 m. The marginal areas shelve gently to 2 m approximately 30 m from the bank. The primary water supply is from Bolster Brook, an effluent and storm-water carrier stream from Radcliffe Sewage Treatment Works, which flows into the lake on the south bank, 250 m from the boat house. It is sufficient to maintain normal water levels. Additional water sources are three underground springs located around the west bank which percolate drainage water from surrounding land into the lake, and a direct water supply from the River Trent which enters the lake through a sluice located on the north bank approximately 500 m from the boat house. Water level and water flow through the lake can be controlled by adjusting the sluice and stank board height at the overflow into the River Trent. Inflow rate from the River Trent is periodically increased during periods of poor water quality and blooms of blue-green algae.

Since the beginning of the 1990s, water quality at the site has altered due to improvements in the effluent discharge from Radcliffe Sewage Treatment Works into Bolster Brook. Levels of organic material and suspended solids entering the lake have declined and the water conditions are noticeably less turbid and odiferous (B. Pluckrose pers. comm.). Consequently, the water is frequently very clear, except during high levels of watersport activity or phytoplankton blooms, and the gravel substrate remains free of silt. This present environment contrasts markedly with the turbid water conditions and silty lake bed synonymous with the lake during the 1980s.

The substratum consists of gravel and beds of *Elodea* sp. growing in the margins. Macrophyte growth is minimal throughout the remainder of the water body. Benthic macro-invertebrates are abundant in the lake, with *Gammarus pulex* and *Daphnia magna* being prevalent throughout the year. Bales of barley straw are distributed along the margins to control phytoplankton growth. Bankside vegetation consists of maintained grasslands and there is an absence of trees and shrubs. There are few underwater features of note except for the wires connecting the marker buoys that delineate the rowing lanes.

Although the rowing course is uniform in dimensions and topography, there are a number of features found throughout its length. At the western end of the rowing course, a number of concrete slipways and boat pontoons are present. These extend approximately 20 m into the lake and are used for the launching of boats, canoes and wind-surfing equipment (Plate 4.2).

The surface of the rowing course is divided into eight rowing lanes by buoys connected by underwater wires. The presence of these wires affects the area of water available for fishing by anglers as the first lanes on both banks are situated 20 m from the bank. Anglers are unable to cast beyond these lanes due to the submerged wires, so are restricted to fishing within 20 m of the margin. Hence, the available area for fishing is approximately 94 000 m^2 from a total water body area of 286 875 m^2, i.e. only approximately 30 % of the water area is accessible to anglers. This indicates that anglers are reliant on fish moving into the marginal areas to become available for capture.

The rowing course is connected to an adjacent water ski-lagoon via a shallow culvert at the 1750 m mark on the south bank. The water ski lagoon has a water area of approximately 2 ha. and has an undulating depth profile up to 4 m. Although shallow, the connecting culvert does allow the free movement of fish between the two water bodies and this has been observed by estate staff. The reasons for this movement are unknown but consideration must be given to speedboat activity creating more favourable water conditions for fish than in the rowing course at certain times of the year, for example, increased dissolved oxygen and disturbance of food items.

4.1.4 *Resident fish stocks*

The resident fish stocks of Holme Pierrepont Rowing Course consist of coarse fish, with the dominant species captured by anglers being roach and common bream. However, perch have been an increasing component of catches in recent years. Other species found in angler catches include chub, dace, eel, gudgeon, pike, tench, rudd, carp, bleak and roach/bream hybrids.

The only documented fish stocking occurred in 1994 prior to the World Angling Championships when approximately 1500 kg of roach and bream were introduced. It is thought that the original stock of fish in the lake came from the River Trent when the river flooded into the site during the winter of 1975/76 (B. Pluckrose, pers. comm.). Other supplementation of fish stocks occurs when river water occasionally enters the lake during winter floods and through illegal stocking by anglers transferring catches from the River Trent to the lake.

4.1.5 *Cormorant populations*

No historical data for cormorant numbers occupying Holme Pierrepont Rowing Course were available prior to the study, although their presence was noticed by the site wardens. Their over-wintering presence in large numbers on the lake was believed to have been observed first in winter 1993/94 (M. Thompson pers. comm.).

4.2 Materials and methods

4.2.1 *Cormorant monitoring*

The number of cormorants occupying Holme Pierrepont Rowing Course were recorded by the following methods (Feltham *et al.* 1999):

- monthly co-ordinated counts during the first hour of daylight and co-ordinated with counts at Colwick Trout Lake (Chapter 6) and the River Trent (Chapter 5);
- daily scan counts - a point sample at approximately 7.30 am each morning;
- scan count samples each hour through the day for some days each month;
- complete records of all birds using the site each hour throughout daylight hours for some days each month;
- counts during feeding observations.

The monitoring programme was comprehensive at the site and allowed a detailed picture of the cormorant occupancy to be determined. Hence, in the MCS model, the parameters of the number of birds feeding at the fishery ($N.f$) and cormorant DFI (c) were determined by site specific methodologies (Section 4.3.1).

4.2.2 *Fisheries monitoring*

Five fisheries survey techniques, using the gears described in Section 3.3, were used to assess the status of the fish populations in Holme Pierrepont Rowing Course. The techniques used were electric fishing (boat-mounted and hand held), micro-mesh seine netting, hydro-acoustics and angler catch analysis.

Boat mounted electric fishing was undertaken to sample the complete marginal area of the lake during each survey. The depths encountered in the middle of the lake were too great for efficient electric fishing (pers. obs.). Hand-held electric fishing was carried out underneath the boat pontoons at the western end of the lake, as fish were observed shoaling there in the winter months (Plate 4.2). This area could not be accessed with any other sampling gear due to the shallow nature and physical obstruction of the pontoons.

Large scale seine netting surveys could not be undertaken on the rowing course because of the presence of underwater obstructions, for example, the underwater lane wires. However, micro-mesh seine netting was used to assess the juvenile fish population in the lake as the electric fishing techniques are considered selective against juvenile fish (Zalewski and Cowx 1990). Juvenile fish were generally associated with the shallow, vegetated marginal areas of the lake in the summer, allowing micro-mesh seine netting to be carried out to provide information on this component of the fish stock.

Hydro-acoustic surveys were undertaken by the Environment Agency (Midlands Region). The surveys were conducted using consecutive longitudinal transects, each 2000 m long, to provide sufficient coverage of the lake. The boat was operated at a speed of 5 km h^{-1} and the results were described in single target volume density (the number of fish recorded as single targets by the beam in its measured water volume) (Lyons 1997). This method of data analysis meant that any multiple targets in the hydro-acoustic beam were not included in the fish density estimate. This was due to the possibility of errors in the identification of the actual number of fish in the multiple target. Hence, use of single target volume density results represents the minimum estimate of the number of fish present in the lake, but is an accurate figure since all sources of error have been removed (Section 3.3.4).

Angler catch data were collected on Holme Pierrepont Rowing Course over the study period, with angling match catches and pleasure angler catches both being monitored. Historical angler catch data could only be traced back on an ad hoc basis to 1990.

4.3 Data analysis

4.3.1 *Cormorant data analysis*

The following cormorant data outputs, using the methodologies described in Section 3.4, were generated for Holme Pierrepont Rowing Course from observations and counts:

- seasonal cormorant occupancy;
- diurnal cormorant occupancy;
- cormorant feeding success;
- MCS generated losses of fish attributable to over-wintering cormorants.

The principle of the MCS model utilised at Holme Pierrepont Rowing Course was that described in Section 3.5.2. However, the derivation of the number of birds feeding at the fishery ($N.f$), and the cormorant daily food intake, c, utilised site specific methodologies due to the comprehensive data set that was compiled at the site (Section 4.2.1).

Estimating $N.f$, the number of birds feeding at Holme Pierrepont

The value for N at the site was determined by analysis of the cormorant diurnal occupancy pattern. The greatest occupancy occurred in the first hour after sunrise. Thus, the level of N peaked in this period. All birds present in this peak were observed to feed at least once. Cormorant occupancy then declined at the site, although 'non-peak' birds continued to visit the site at low densities during the daylight hours to feed. These data allowed the values of N_{min} and N_{max} to be determined by the application of two distinct assumptions on the 'non-peak' birds (Feltham *et al.* 1999).

- N_{min}

Assumption : Any birds arriving after the early morning peak were the same birds as some of those observed in the morning peak, which had returned to resume feeding. Thus, N_{min} is represented as only the number of cormorants counted feeding in the early morning peak (Feltham *et al.* 1999).

- N_{max}

Assumption : Cormorants observed arriving at the site after the early morning peak were different birds from those already observed. Thus, N_{max} is represented as the numbers of birds observed in the early morning peak with the addition of the numbers of any cormorants observed in the remainder of the day (Feltham *et al.* 1999).

As the cormorants were not marked during the study, it was impossible to distinguish between individual cormorants during dawn until dusk counts. Thus, the actual number of cormorants that utilised the site during that day will be a uniform distribution of N lying between the values of N_{min} and N_{max} (Feltham *et al.* 1999). This highlights the important use of distributions around the model parameters in the MCS model (Section 3.5.2).

The following factors were used to determine the value for f at Holme Pierrepont:

- all of the cormorants observed at the site were observed to feed there;
- no birds were observed to haul out at the lake;
- no twin peaks of foraging activity were noted at dawn and dusk as observed at other fisheries (Pilcher and Feltham 1997), with only a single peak at dawn.

Using these factors, f, the proportion of observed cormorants that actually fed at the site on a given day, was taken as 1. Therefore, $f_{min} = f_{max}$, and f became a constant (Feltham *et al.* 1999).

Estimating c - the cormorant daily food intake

The general MCS model utilised a uniform distribution of c lying between values of c_{min} and c_{max} from DFI estimates based on energy considerations (Section 3.5.2). This assumes all birds which fed at the fishery met their full DFI there.

At Holme Pierrepont Rowing Course this assumption could not be justified; hence, use of c_{min} and c_{max} were not appropriate (Feltham *et al.* 1999). The approach to estimate DFI at the site utilised the feeding observation data. For each month, the size of prey species was converted to mass using length-weight equations (3.5.6). Where a complete foraging bout was observed, the total mass of fish consumed was known as DFI_{HPP}, where:

$$DFI_{TOTAL} = DFI_{HPP} + DFI_{ELSEWHERE}$$

During observations, a number of foraging bouts were unable to be observed as being complete, due to, for example, losing a specific cormorant amongst a whole flock. Thus, the total mass ingested by complete bouts was compared with the values for incomplete bouts. Where lower estimates were apparent in the incomplete bouts, a correction factor was applied to give the total that would have been achieved by the cormorant for a complete feeding bout (Feltham *et al.* 1999).

Therefore, for each month, the mean mass of fish consumed per foraging bout, with standard deviation, was calculated to represent DFI at Holme Pierrepont Rowing Course, with this distribution of c used in the MCS model rather than the uniform DFI distribution discussed in Section 3.5.2.

4.3.2 *Fisheries data analysis*

The following outputs, using the methodology described in Section 3.5, were generated for Holme Pierrepont Rowing Course from fisheries surveys:

- standing crop;
- species composition;
- length frequency distributions;
- age and growth determination of fish species;
- growth indices and models;
- mortality rates, survival rates and year class strengths;
- cohort reconstruction;
- biomass estimates;
- angler catch performance.

4.3.3 *Assessment of the impact of cormorant predation on Holme Pierrepont Rowing Course*

The information used to assess the impact of the cormorant predation were:

- comparison of species composition of cormorant predation, electric fishing results and angler results. This shows selectivity by cormorants and anglers on the fish species availability represented by electric fishing;

- comparison of length frequency distribution of cormorant predation, electric fishing results and angler results. This shows the size selectivity by species of cormorants and anglers compared with catches by electric fishing;

- comparison of the number and biomass of fish in the lake (represented by electric fishing results) with cormorant predation and without cormorant predation (electric fishing results plus MCS generated cormorant predation losses);

- proportion of the biomass of fish in the lake removed by cormorant predation, revealed by biomass removed by cormorants.

4.4 Status of fish populations at Holme Pierrepont Rowing Course

4.4.1 *Standing crop*

The mean standing crop values of the main fish species were calculated from data derived from the first survey in each year (October 1995, 1996, 1997 and June 1998) and using an electric fishing gear efficiency of 0.23 (Section 3.3.1; Table 4.2). Little variation in total standing crop was observed during the study period, although variations within species did occur.

Table 4.2 Mean standing crop, estimated by electric fishing, of major fish species in Holme Pierrepont Rowing Course during the study period.

	Standing crop (kg ha^{-1})		
Species	**1995/6**	**1996/7**	**1997/8**
Roach	8.25	47.07	17.76
Common Bream	48.30	24.15	48.30
Perch	14.97	14.81	13.22
Total standing crop	71.51	86.03	79.28
Total standing crop g m^{-2}	7.15	8.60	7.92

A hydro-acoustic survey, completed by the Environment Agency (Midlands Region) in August 1997, estimated a mean biomass of 16.96 kg ha^{-1} (range 10.79 to 25.16 kg ha^{-1}). This value is low when compared with the values derived by the electric fishing gear (Table 4.2). These results may be considered low due to only single targets being used as fish identifiers in the beam (Section 4.4.2).

4.4.2 *Angler catch data*

Angler effort and catch per unit effort (CPUE) for the 1990/91 to 1998/99 angling seasons showed variation over time (Figure 4.1). The high CPUE observed in 1990/91 and 1991/92, 1947 g man $hour^{-1}$ and 1185 g man $hour^{-1}$ respectively, fell to 95 g man $hour^{-1}$ in 1994/95, but recovered to 756 g man $hour^{-1}$ between angling seasons 1996/97 and 1998/99. Although annual angler effort also varied over the study period, the angling effort in this period refers only to that recorded as part of the angler catch data monitoring (Table 4.3), and not that recorded by the site management (Table 4.1). Indeed, effort levels were shown to be far higher (Table 4.1), but catch data were unavailable. This was shown by angler visits in 1992 totalling 8626 (Table 4.1). Thus, assuming each angler fished for 5 hours, then the actual angler effort level was 43,130 man hours.

Table 4.3 Annual fishing effort on Holme Pierrepont Rowing Course 1990 to 1998.

Angling season	1990/1	1991/2	1994/5	1996/7	1997/8	1998/9
Angler effort (man hour)	440	1000	774	326	629.5	859
CPUE (g man $hour^{-1}$)	1947	1185	95	567	403	756

Although the catch rate on the lake has declined since 1990, the catch rates (except CPUE in season 1994/95) compare favourably with those from river fisheries, for example, 60 to 231 g man $hour^{-1}$ on the River Trent (Cowx 1991; Jacklin 1996; Section 5.7.1) and 37 to 91 g man $hour^{-1}$ for the Yorkshire Ouse (Axford 1991).

4.4.3 *Species composition of fish populations in Holme Pierrepont Rowing Course*

Roach were the dominant fish species throughout the study period (Figure 4.2). The contribution of common bream varied between years with high catch composition in the 1996/97 surveys only (23.6 %). Perch numbers were seen to increase through the study period and contributed 20.5 % of all electric fishing catches in 1997/98 (Figure 4.2). Other species caught by electric fishing included chub, a fish associated with riverine habitats. These are thought to have arisen from illegal introductions by anglers from stocks caught in the River Trent, which runs alongside the rowing course (T. Holden pers. comm.). Pike were caught in electric fishing surveys in 1995/96 (0.9 %), 1996/97 (14.5 %), and June 1998 (2.8 %), and a small number were observed in surveys in winter 1997/98 but were not captured. The presence of pike is important because they are a piscivorous fish species, representing a further predation influence on the fish stocks.

A number of minor species, including carp, gudgeon and tench were also caught during electric fishing. The contribution of gudgeon to the fish community in 1995/96 was high, at 18.6 % of total catch.

4.4.4 *Length frequency distribution*

The electric fishing surveys suggest low numbers of fish below 100 mm were present (Figures 4.3 to 4.5). This was possibly due to a sampling bias of electric fishing techniques (Zalewski and Cowx 1990).

Roach populations were dominated by fish in the size range 90 to 200 mm, with individuals up to 290 mm caught (Figure 4.3). The size range suggests that roach were recruiting successfully, attaining relative large sizes and establishing a mature stock.

Inter-annual variation in the size structure of common bream populations was observed during the study (Figure 4.4). Good recruiting populations of bream in the size range 110 to 220 mm were found in 1997/98. However, in other surveys the common bream present in the lake were in the size range 430 to 530 mm. These were old fish (10 to 14 years) (Section 4.4.6), and comprised the main mature bream stock of the lake. An absence of fish in the size range 300 to 400 mm was evident in all surveys, probably due to difficulties in sampling common bream by electric fishing in the deeper areas of the lake.

There was evidence to suggest perch were recruiting successfully in the lake (Figure 4.5). In 1997/98, large numbers of perch were captured, especially in the size range 110 to 130 mm. These fish were recruited from successful spawning in March/April 1996. Perch above 280 mm were found in all survey years. Thus, perch were able recruit successfully, attain large sizes and establish a mature stock in the lake.

The roach length frequency distribution from the Home International angling contest of 1998 showed anglers targeted the larger sizes of the roach stock, fish above 150 mm (Figure 4.6). The dominant size classes of roach captured by anglers (150 to 180 mm) were probably the dominant roach size class caught in winter 1997/98 surveys, but exhibiting summer growth (Figure 4.3, 4.6). These roach were from the 1996 year class of roach and highlight the reliance of angler catches on the presence of strong year classes of fish.

4.4.5 *Year class strength*

Strong year class strengths of roach occurred in 1988, 1992, 1995 and 1996 (Figure 4.7). Poor year class strengths occurred in 1989, 1990, 1991, 1993 and 1994. The weak year classes of 1993 and 1994 may have contributed to the poor standing crop of roach (Table 4.2) in 1996, the period when these year classes of roach should have been strongly represented in the roach population of the rowing course.

4.4.6 *Growth rates*

Growth rates for the major fish species, roach, common bream and perch, were derived using data obtained from electric fishing catches. The growth of roach in 1996/96 was above average whilst the growth of roach caught in 1996/97 to 1997/98 was rated as fast compared to standards (Cowx *et al.* 1995) (Figure 4.8). A scale sample from a roach caught in January 1997 aged 2+ from Holme Pierrepont Rowing Course was compared with roach of a similar age from the River Trent (Section 5) and Grimsargh Reservoir (Section 7) in Figure 4.11. It can be seen the roach from Holme Pierrepont attain a greater size than the fish from the other two sites. Overall, a rapid increase in roach growth rate has occurred since the early 1990s (Section 4.6.3; Figure 4.12, 4.19).

The growth rates of common bream from year classes 1992 to 1996 at Holme Pierrepont Rowing Course were fast when compared with standard data (Figure 4.9). However, common bream from the 1982 to 1987 year classes were much slower growing (Section 4.6.3). A bream aged 8 from this early period would achieve the same length as a 5 year old bream from post 1992 year classes. This apparent difference in growth rate is examined in Section 4.6.3.

The growth rate of perch at Holme Pierrepont Rowing Course was also good when compared with standard data (Figure 4.10). In surveys, perch up to lengths of 327 mm were captured. However, scales from these fish were either all replacement (a scale that has replaced a lost scale and grown back without the growth history) or of too poor quality for accurate analysis. Hence, although the maximum age calculated was 6, with a length of 276 mm, larger perch were present in the lake and these were probably over 6 years of age.

4.4.7 *L infinity and K*

The von Bertalanffy growth parameters L_∞ and K for roach, bream and perch at Holme Pierrepont were compared with values derived from other UK fisheries (Table 4.4 to 4.6). The values derived for Holme Pierrepont lie within the ranges found at other UK fisheries, suggesting the populations are typical of fisheries in the region.

Table 4.4 L_∞ and K values for roach in Holme Pierrepont Rowing Course compared with values for roach from other UK fisheries.

Venue	L_∞	K
Holme Pierrepont	302	0.14
Beeston, River Trent	341	0.16
Trent Bridge, River Trent	362	0.16
Stoke Bardolph, River Trent	361	0.15
River Frome (Mann 1973)	400-430	0.13-0.14
River Stour (Mann 1973)	240-370	0.15-0.25
River Thames (Williams 1967)	165	0.30
River Culm (Cowx 1980)	238-287	0.20-0.23
River Nene (Hart and Pitcher 1973)	273-294	0.16-0.30
Slapton Ley (Burroughs *et al.* 1979)	157-258	0.31-0.32
River Severn (Criag Goch Research Team 1980)	328-371	0.18-0.21

Table 4.5 L_∞ and K values for common bream in Holme Pierrepont Rowing Course compared with values for bream from other UK fisheries.

Venue	L_∞	K
Holme Pierrepont	596	0.11
Beeston, River Trent	545	0.15
Trent Bridge, River Trent	530	0.14
Stoke Bardolph, River Trent	515	0.14
River Exe (Cowx 1983)	457-625	0.11-0.14
Tatton Mere (Goldspink 1981)	590	0.11
Ellesmere Mere (Goldspink 1981)	538	0.12
Cole Mere (Goldspink 1981)	510	0.12

Table 4.6 L_∞ and K values for perch in Holme Pierrepont Rowing Course compared with values for perch from other UK fisheries.

Venue	L_∞	K
Holme Pierrepont	368	0.21
Beeston, River Trent	432	0.13
Trent Bridge, River Trent	303	0.19
Stoke Bardolph, River Trent	429	0.14
Tatton Mere (Goldspink 1981)	410-560	0.14-0.25
Rostherne Mere (Goldspink 1981)	320	0.21
River Nene (Hart and Pitcher 1973)	690	0.37
River Thames (Williams 1967)	479	0.05
Slapton Ley (Craig 1974)	291	0.25
River Severn (CGRT 1980)	393	0.18
East England rivers (Hartley 1974a)	177	82.5

4.4.8 *Mortality and survival rates*

The mortality and survival rates of the main species at Holme Pierrepont were compared with values derived for the species from other UK fisheries (Table 4.7) (Cowx *et al.* 1995). The values for species at Holme Pierrepont were high, indicating low annual survival of fish, although mortality for roach and common bream were within the range obtained from other UK fisheries (Table 4.7). Natural mortality rates are very important to consider, for once the fish were out of the vulnerable sizes to cormorant predation, the only factor governing their subsequent survival is the natural mortality rate. Hence, even after growing out of the vulnerable size and age to cormorant predation (Section 4.6.2), fish in Holme Pierrepont were not expected to show an extended life span due to the high natural mortality rate (Section 9.2.5).

Table 4.7 Mortality and survival rates of roach, common bream and perch in Holme Pierrepont Rowing Course 1995 to 1998, compared with other UK fisheries (Cowx *et al.* 1995).

Species	Venue	Z	S	If N_t=1000, N_{t+1}=
Roach	Holme Pierrepont	0.858	0.42	424
	7 UK fisheries	0.20 - 1.40		
C. bream	Holme Pierrepont	0.921	0.40	398
	7 UK fisheries	0.33 - 1.23		
Perch	Holme Pierrepont	0.960	0.39	393
	2 UK fisheries	0.46 - 0.53		

4.5 Cormorant observations

4.5.1 *Temporal roost occupancy*

The numbers of cormorants occupying the Attenborough night roost showed a marked seasonal trend, with the greatest numbers between October and March, and reduced numbers between April and September (Figure 4.13). This was related to the spring/summer migration of mature cormorants to coastal breeding grounds (Section 2.2).

4.5.2 *Cormorant seasonal occupancy*

The early morning cormorant counts at Holme Pierrepont Rowing Course revealed peak numbers from December to February of each winter surveyed (Figure 4.14). Numbers dramatically decreased in April. Cormorant numbers at the Attenborough night roost reflected the decline over summer (Figure 4.13). No cormorants were observed at the site between May and September. The peak winter counts decreased over the study period, with 153 birds in February 1996, 83 in December 1996 and 79 in December 1997.

4.5.3 *Cormorant diurnal occupancy*

Cormorant occupancy of the site was generally highest at dawn, with a large decline during the first hours of daylight (Figure 4.15). This coincided with increased numbers of birds arriving at Colwick Park Trout Lake (Figure 4.15; Section 6.4). This suggests that the cormorants fed at Holme Pierrepont at dawn before using Colwick as a day roost area (Section 6.5.3). All of the cormorants using Holme Pierrepont used the site for feeding, with very few birds observed to 'haul out', i.e. leave the water and roost.

4.5.4 *Cormorant feeding success*

A total of 613 foraging bouts were observed during the study of which 398 (64.9%) were successful, i.e. at least one fish was caught (Table 4.8). Variation was observed between years and months, ranging between 51 and 79 %. This was the highest successful foraging bout range observed in the Midlands study sites (Section 5.5.2, 5.6.2, 5.7.2 and 6.5.4).

The proportion of dives that were successful varied annually between 12.7 and 15.2 % (Table 4.8). Of the 6485 dives observed, 876 (13.5%) were successful.

Table 4.8 Feeding success of cormorants at Holme Pierrepont Rowing Course, winter 1995 to 1998.

	1995/96	1996/97	1997/98
Number foraging bouts observed	215	182	216
Number foraging bouts successful	160	105	133
% foraging bouts successful	74.4	57.7	61.6
No. dives observed	2647	1675	1863
No. dives successful	402	212	262
% dives successful	15.2	12.7	14.1

4.6 Impact assessment of cormorant predation at Holme Pierrepont Rowing Course

4.6.1 *Species composition of electric fishing surveys, angler catches and cormorant predation*

Comparison of the species composition of angler catches and cormorant diet suggests their exploitation patterns change between years, were selective and not always representative of the fish stocks available (Figures 4.16).

Although perch were a major component of cormorant diet and angler catches over the period, their representation decreased over time. Cormorant diet selection for perch fell from 63 % in 1995/96 to 19 % in 1997/98, whilst the composition of perch in angler catch fell from 71 % to 3 % over the same period. By contrast, electric fishing catch composition demonstrated a reverse of this pattern, with a shift from 6 % to 21 % over the period. Perch standing crop was stable over this period at around 14 kg ha^{-1} (Table 4.2).

Roach dominated the fish community structure in electric fishing surveys in all years of the study (Figure 4.16). The species were also an important dietary component of cormorants, contributing between 28 and 39 % of the fish intake by number. Therefore, roach were a consistent food source for the cormorants. The contribution of roach to angler catches increased dramatically, from 13 to 94 %, over the study period. This was at the expense of perch which showed a marked drop in contribution to angler catches over the same period.

Common bream did not play an important role in the diet of cormorants at Holme Pierrepont in winter 1995/96 (Figure 4.16). However, the proportion of common bream in cormorant diets increased from 1996 to 1998, and by winter 1997/98, 42 % of all predated fish were common bream. Electric fishing catches of common bream were low in all years, although a small increase was observed in 1996/97.

4.6.2 *Length frequency distribution*

Observations of cormorant feeding behaviour allowed the classification of prey size into 50-mm increments. Accordingly, all data collected from electric fishing surveys were reassigned to 50-mm size classes to allow comparison.

The length frequency distribution of roach predated by cormorants revealed they mainly selected fish below 100 mm for consumption (Figure 4.17). These sizes were poorly represented in electric fishing due to sampling bias, with roach electric fishing catches dominated by fish between 150 to 200 mm. Hence, cormorants appear to select roach below 100 mm for consumption against the larger individuals present. As the roach reach 100 mm in under 2 years (Figure 4.8), it is evident these fish were mainly vulnerable to predation only in their first winter of life.

The length frequency distribution of cormorant predated common bream revealed they selected fish up to 250 mm for consumption (Figure 4.18). However, cormorant predation concentrated on fish below 100 mm in winter 1995/96 (66 %) and 1996/97 (70 %). In winter 1997/98, 45 % of common bream consumed were below 100 mm. Cormorants only predated upon larger common bream above 200 mm in 1995/96 (8 %). As with roach, the growth rate of bream was very rapid, achieving 200 mm by the end of the third year (Figure 4.11). Consequently, bream were mainly vulnerable to cormorant predation in their first and second winters of life.

Cormorants mainly selected perch below 100 mm in length for consumption, fish which comprised 76 to 94 % of total predated perch in all years (Figure 4.19). Perch growth rate was fast in Holme Pierrepont Rowing Course (Figure 4.12), suggesting perch were only vulnerable to predation in their first winter.

4.6.3 *Growth indices*

Interpretation of the growth of roach and common bream from Holme Pierrepont Rowing Course indicated a marked change in growth rates around 1993 (Section 4.4.6). The relative annual growth of roach was poor in the pre-1993 era when compared with recent growth from the lake (Figure 4.12, 4.20) and the national standard (Figure 4.21). Between 1992 and 1995 the growth rate increased by 90 % (Figure 4.20) and the shift was first detected in the 1994 growth year. Although the good rate of growth has continued, the years when the highest growth rate was observed produced the lowest angler catch per unit effort (4.4.2).

The historical growth rate of common bream was also poor when compared with recent data (Figure 4.22). Between 1983 and 1990, growth was below the national standard for bream (Figure 4.23) (Hickley and Dexter 1979). A marked increase in growth first occurred in 1993 and has continued to increase with some annual variation. At present, the growth rate exceeds the national standard for bream by 100 % (Figure 4.23) (Hickley and Dexter 1979).

In summary, a dramatic growth rate shift occurred in the roach and bream populations during the period 1993/94. However, there was no indication of such a change in the perch population (Section 4.4.6). Cormorants were first observed feeding on the lake in winter 1993/4 (B. Pluckrose pers. comm.), around the time of the growth rate shift.

4.6.4 *Cohort reconstruction with and without cormorant predation*

Evidence from the cormorant feeding observations suggest they eat a considerable quantity of roach, common bream and perch, both in terms of number and biomass (Table 4.9 to 4.11; Figure 4.24 to 4.41). The difference in the values with and without cormorant predation can **only be attributable to cormorant predation** and not any other source of mortality.

The model suggests that roach abundance in the lake was been reduced by 62 % in number and 72 % in biomass after three winters of cormorant predation (Table 4.9). When cormorants are absent, this equates to an increased roach abundance of 258 % by number and 354 % by biomass. Common bream abundance was reduced by 51 % in number and 67 % in biomass after the three winters of cormorant predation (Table 4.10), which equates to an increased abundance of 205 % by numbers and 305 % by biomass when no cormorants were present. Perch abundance was reduced by 65 % by numbers and 75 % by biomass by the three winters of cormorant predation (Table 4.11), which equates to an increased abundance of 287 % and 402 % by numbers and biomass respectively in the absence of cormorants.

These data must be considered as mean cormorant predation impact figures because they are based on electric fishing surveys using a gear calibration efficiency of 0.23 (Section 3.3.1). As the efficiency is known to vary between 0.10 and 0.36, the range of impact values shift accordingly (Table 4.12). This is particularly true for bream which have a lower susceptibility to electric fishing than roach and perch (Zalewski and Cowx 1990). Thus, the mean estimates for predation impact on bream may be over estimated.

Notwithstanding this, the electric fishing assessment was considered to be a reasonably accurate estimate of the fish stock abundance. This was confirmed by the hydro-acoustic data which provided a minimum estimate of abundance in the rowing course in August 1997 of 16.96 kg ha^{-1} (range 10.79 to 25.16 kg ha^{-1}) in the presence of cormorant predation (estimated at 13 to 34 kg ha^{-1}). When compared with cormorant predation rates from the MCS output, the hydro-acoustic assessment, which provided lower fish abundance estimates than electric fishing gear, shows that cormorant predation accounted for a high proportion of fish in the lake (Table 4.9-4.11, Figures 4.24-4.41).

Indeed, the cormorant predation rates lead to some anomalous results, for example, winter 1997/98. Cormorants were estimated to remove 34 kg ha^{-1} of fish from the rowing course when the hydro-acoustic assessment of August 1997 showed a maximum 25 kg ha^{-1} of fish were available. The anomaly is also certainly due to the limitations of hydro-acoustic assessment in shallow waters and suggests the electric fishing studies provided a more accurate reflection of the fish stocks.

Table 4.9 Numbers and biomass of roach with and without cormorant predation in Holme Pierrepont Rowing Course.

		Number ($n\ ha^{-1}$)	% reduction	Biomass ($g\ ha^{-1}$)	% reduction
1996	With cormorants	225.1		8246.9	
	Without cormorants	416.1	45.9	21962.3	63.4
1997	With cormorants	1173.6		47065.3	
	Without cormorants	1560.5	24.8	70723.2	33.5
1998	With cormorants	322.2		17762.7	
	Without cormorants	856.6	62.4	62882.6	71.8

Table 4.10 Numbers and biomass of common bream with and without cormorant predation in Holme Pierrepont Rowing Course.

		Number ($n\ ha^{-1}$)	% reduction	Biomass ($g\ ha^{-1}$)	% reduction
1996	With cormorants	419.7		48300.1	
	Without cormorants	456.9	8.1	55253.3	12.6
1997	With cormorants	209.8		24150.1	
	Without cormorants	481.0	56.4	59501.1	59.4
1998	With cormorants	419.7		48300.1	
	Without cormorants	861.7	51.3	147305.7	67.2

Table 4.11 Numbers and biomass of perch with and without cormorant predation in Holme Pierrepont Rowing Course.

		Number ($n\ ha^{-1}$)	%	Biomass ($g\ ha^{-1}$)	%
1996	With cormorants	230.8		14967.4	
	Without cormorants	1299.1	82.2	48834.4	69.3
1997	With cormorants	253.6		14809.5	
	Without cormorants	928.2	72.3	57588.4	74.3
1998	With cormorants	249.6		13200.2	
	Without cormorants	715.6	65.1	53146.3	75.2

Table 4.12 Impact of cormorant predation on the populations of roach, common bream and perch ($n\ ha^{-1}$) in Holme Pierrepont Rowing Course over the range of electric fishing gear efficiency (CP = cormorant predation, -CP = without cormorant predation).

		1996		1997		1998	
Species	Efficiency	CP	-CP	CP	-CP	CP	-CP
Roach	0.10	396	632	2699	3085	788	1284
	0.23	225	461	1174	1560	343	838
	0.36	110	346	750	1136	219	714
Common bream	0.10	965	1001	483	753	1448	1888
	0.23	420	455	210	480	420	860
	0.36	148	183	162	432	402	843
Perch	0.10	531	1596	583	1256	574	7039
	0.23	231	1296	256	928	250	716
	0.36	148	1212	134	806	160	625

The cohort analysis not only indicated the proportion of fish that would have been available should cormorants not have been present, but also an adjustment of the population structure of the target species. If cormorants were not present, it is probable that the roach, perch and bream would have lived longer. Consequently, the fishery may have supported a greater number of large, older specimen fish, which may have been attractive to anglers.

4.7 Discussion

4.7.1 *Cormorant occupancy*

The seasonal cormorant occupancy at Holme Pierrepont Rowing Course showed peak abundance in the winter months of December to February, with a large decline in numbers in March and April. No cormorants were observed on the lake between May and September. This was shown to be related to the numbers of cormorants at the main Attenborough night roost. Hence, the seasonal abundance of the cormorants on Holme Pierrepont was related to their breeding behaviour and not temporal changes in fish distribution in the lake.

Cormorant occupancy of the site was generally greatest at dawn, with a large decline during the first hours of daylight. These birds were utilising the site for feeding, before leaving to day-roost elsewhere, for example, Colwick Park, Nottingham (Chapter 6).

4.7.2 *Cormorant feeding success*

All cormorants visiting the site were observed to feed there. Foraging bout success was relatively high, and the highest among the sites studied in the Midlands Region. Thus, in comparison to the River Trent (Section 5.5.2, 5.6.2, 5.7.2) .and Colwick Park Trout Lake (Section 6.5.4), Holme Pierrepont Rowing Course provided a profitable feeding patch. This was probably due to the high number of densely shoaled fish in the area around the boat pontoons, which allowed good diving access, resulting in the high feeding bout success rate (Section 4.7.6; Plate 4.2).

4.7.3 *Cormorant impact assessment*

The main conclusion in assessing cormorant predation impact is that a high proportion of the fish standing crop were removed each year. The majority of those fish were individuals of below 100 mm. Although anglers generally target fish of above this size (Figure 4.6), this assessment cannot be interpreted as negligible impact, for a proportion of the small fish would have grown on to a larger size (number determined by the natural mortality/survival rate) and been available for angler exploitation. This is an important consideration as angler success is a function of the stock abundance of catchable-sized fish (Pawson 1982). Cormorant predation was shown to reduce fish availability through the cohort analysis and this will have had a negative affect on the fishery performance.

However, a number of other factors must be taken into consideration when assessing the scale of the impact of cormorant predation on the fishery.

4.7.4 *Response of fish populations to predation*

The growth rates of roach, bream and perch, the species heavily predated upon by cormorants, were extremely fast. Fast growth rates of roach have also been recorded at Hornsea Mere, East Yorkshire, which is a water body of similar topography to Holme Pierrepont Rowing Course and is also subject to cormorant predation (unpublished data, J. Harvey pers. comm.). The fast growth rate may have been the result of lowered abundance/standing crop of fish, i.e. the effect of reduced levels of inter- and intra-specific competition, due to the cormorant predation, combined with high food availability. The growth shift occurred during the period of lowered angling returns, hence, supports the theory that cormorant predation reduced fish abundance in the lake and resulted in reduced competition and increased fish growth.

This growth response of the fish populations had considerable advantages because it enabled the fish to grow out of the catchable cohort of cormorants quickly (Section 4.6.2) and appear in angler catches earlier. This was reflected in the angler returns in the Home International angling match in August 1998, where the dominant roach size classes captured were 150 to 180 mm, fish of 2+ years old (Section 4.4.6), resulting from the strong 1996 year class (Section 4.4.5).

In 1998, recruitment from the 1995 and 1996 year classes in the fishery was strong, as would have been predicted for the climatic conditions in the first year of life of these fish (Section 4.4.5). Thus, the strong recruitment appeared to have been adequate to override the effect of the observed cormorant predation. This has resulted in sufficient fish being able to grow beyond the optimal size for cormorant predation in a short time, generally after 1 year, and ensured a satisfactory head of angler exploitable fish in the lake. These gave the acceptable angling returns when compared with river fisheries (Section 4.4.2). However, it is likely the angling returns during the study period would have been increased in cormorant absence due to the increased abundance of fish available in the lake for exploitation.

The summer of 1998 was generally cold and wet, and such conditions tend to result in poor juvenile recruitment (Cowx *et al.* 1995). As it is likely cormorant predation on these fish will be high in winter 1998/99, due to no cormorant management measures being implemented at the site, it has to be asked whether sufficient fish will escape the predation to be able to grow to a reasonable size for anglers to subsequently exploit? If the cormorant food resources are limited due to the poor fry survival, will the cormorants target the larger fish as their food base and begin to compete directly with anglers? These questions can only be answered when the fish population response to cormorant predation had been measured over a number of years with variable fish recruitment patterns. However, the compensation processes limiting the predation impact may only be able to occur when good fry recruitment has occurred. An increased negative impact of cormorant predation may occur during years of poor fry survival.

It should be recognised that coarse fish populations undergo considerable natural fluctuations in abundance (Townsend 1989; Perrow *et al.* 1990; Townsend *et al.* 1990). These result in fluctuations in fishery performance. In many fisheries, a series of poor recruitment years will be reflected in poor angling results two or three years later, especially if the larger, older fish die out and are not replaced (Cowx 1991). This has been observed in many rivers, for example, River Trent (Cowx 1991), Yorkshire Ouse (Axford 1991) and Warwickshire Avon (A. Starkie pers. comm.), and results in complaints by anglers of poor angling. It is likely the angling would improve after a period of years if strong year classes were able to develop. At Holme Pierrepont Rowing Course, the situation is more difficult to assess because the heavy cormorant predation pressure may not allow even an average year class to be represented in angler catches in subsequent years.

4.7.5 *Water quality*

Water quality in Holme Pierrepont Rowing Course has undoubtedly increased in recent years due to improvements in Bolster Brook and the adjacent River Trent which maintain the level of the lake. Historically, the lake was very turbid with recent improvements resulting in the lake often being very clear, especially in the marginal areas. This may lead to fish moving away from the marginal areas to deeper central areas during the times of clear water, as they exhibit a natural avoidance behaviour to factors such as predation. This phenomenon was observed during electric fishing surveys, when catches in the margins were low at times of clear water, but were much higher when water was turbid. The possible movement of fish away from the marginal areas to deeper areas may have serious implications on angler catches. Anglers on the rowing course are restricted to fishing within 20 m of the bank because of the presence of the underwater wires which support lane buoys. This means that anglers are fishing the marginal areas of the lake and in clear water conditions fish may avoid this zone, reducing catches. This problem was highlighted in recent press reports on the River Trent, with complaints that the river is too clean and as a result fish are easily 'spooked' into the deeper areas and angling efficiency is decreased (Fitzpatrick 1997).

4.7.6 *Boat pontoons*

An important consideration in the cormorant impact assessment was the role of the boat pontoons in the cormorant predation levels (Plate 4.2). The pontoons, used as launching facilities for water users, are situated at the western end of the lake. During winter there is a general fish migration to this area of the lake (B. Pluckrose pers. comm.). The hand-held electric fishing gear revealed the fish were shoaled tightly in the cover provided by the boat pontoons (pers. obs.). Fish were only caught when the electrode was placed next to, or under, the pontoons, whereupon large fish samples were obtained. These dense shoals of fish were observed in water below 50 cm depth, with some fish even located in small concrete cracks. Thus, the fish were utilising the boat pontoons as a winter refuge area.

Cormorant feeding observations revealed their ability to be able to dive underneath the boat pontoons to predate on the fish, with a high degree of success (Section 4.5.2). Thus, the pontoons were not acting as a safe refuge for the fish, whose dense shoaling behaviour apparently provided easy feeding for the birds, resulting in the high fish losses (Section 4.6.4). Consequently, any fish refuge constructed to deter cormorant predation would have to prevent cormorant diving access (Chapter 10).

4.7.7 *Water-ski lagoon*

The adjacent water-ski lagoon may play a role in the fluctuations in catches both by electric fishing and anglers. Movements of fish are known to occur between the water-ski lagoon and the rowing course through the small culvert. A hydro-acoustic survey in 1996 found low numbers of fish in the rowing course coupled with high densities of fish in the water-ski lagoon, which could cause some difficulties in interpreting the impact assessment.

Consequently, the lagoon and rowing course must be considered as one system when assessing the status of the populations. It was not felt that this issue obscured the interpretation of the findings of the study, except that standing crop of fish in the system may be slightly greater than that estimated from electric fishing and hydro-acoustic surveys. Notwithstanding, the biomass of fish eaten by cormorants still remains high relative to the potential standing crop of a temperate lake of this nature, i.e. 30 to 200 kg ha^{-1} (Grimm and Backx 1994).

Plate 4.1 Holme Pierrepont Rowing Course, viewed from the boathouse end. A cormorant maybe viewed in the foreground, foraging around the boat pontoons.

Plate 4.2 The boat pontoons at Holme Pierrepont Rowing Course.

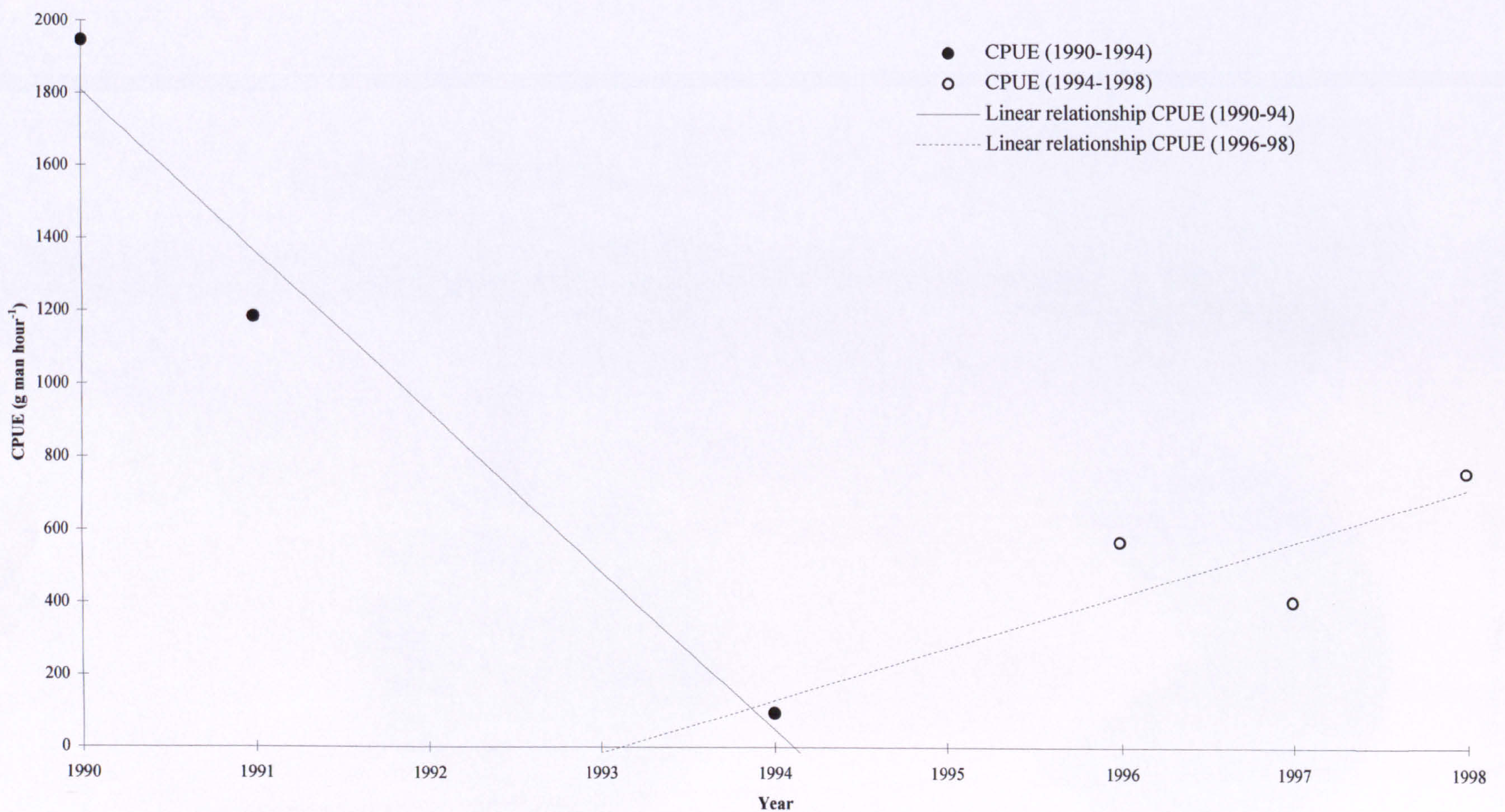


Figure 4.1 Angling performance, measured as Catch per unit effort, at Holme Pierrepont Rowing Course, 1990 to 1998.

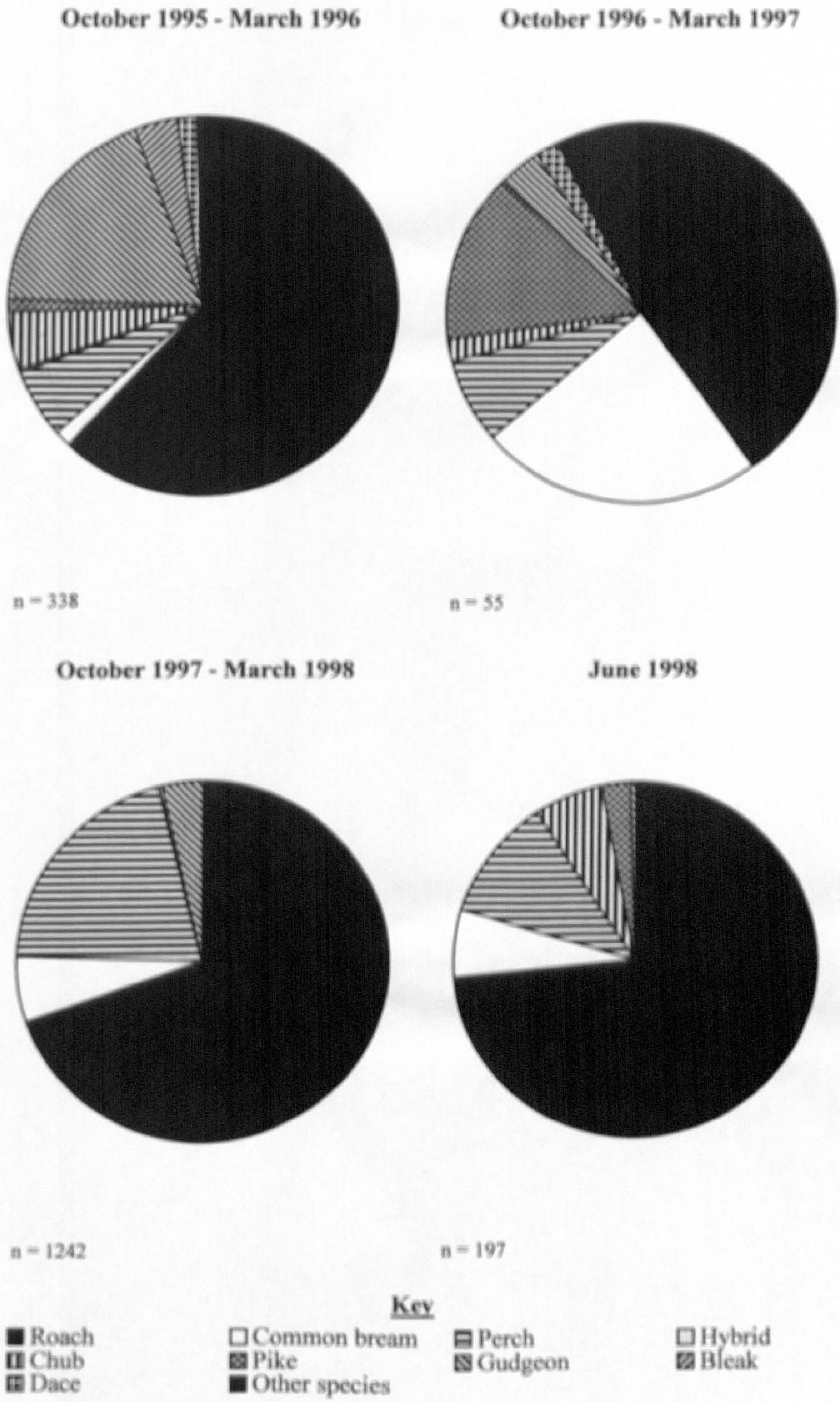


Figure 4.2 Species composition of electric fishing surveys at Holme Pierrepont Rowing Course.

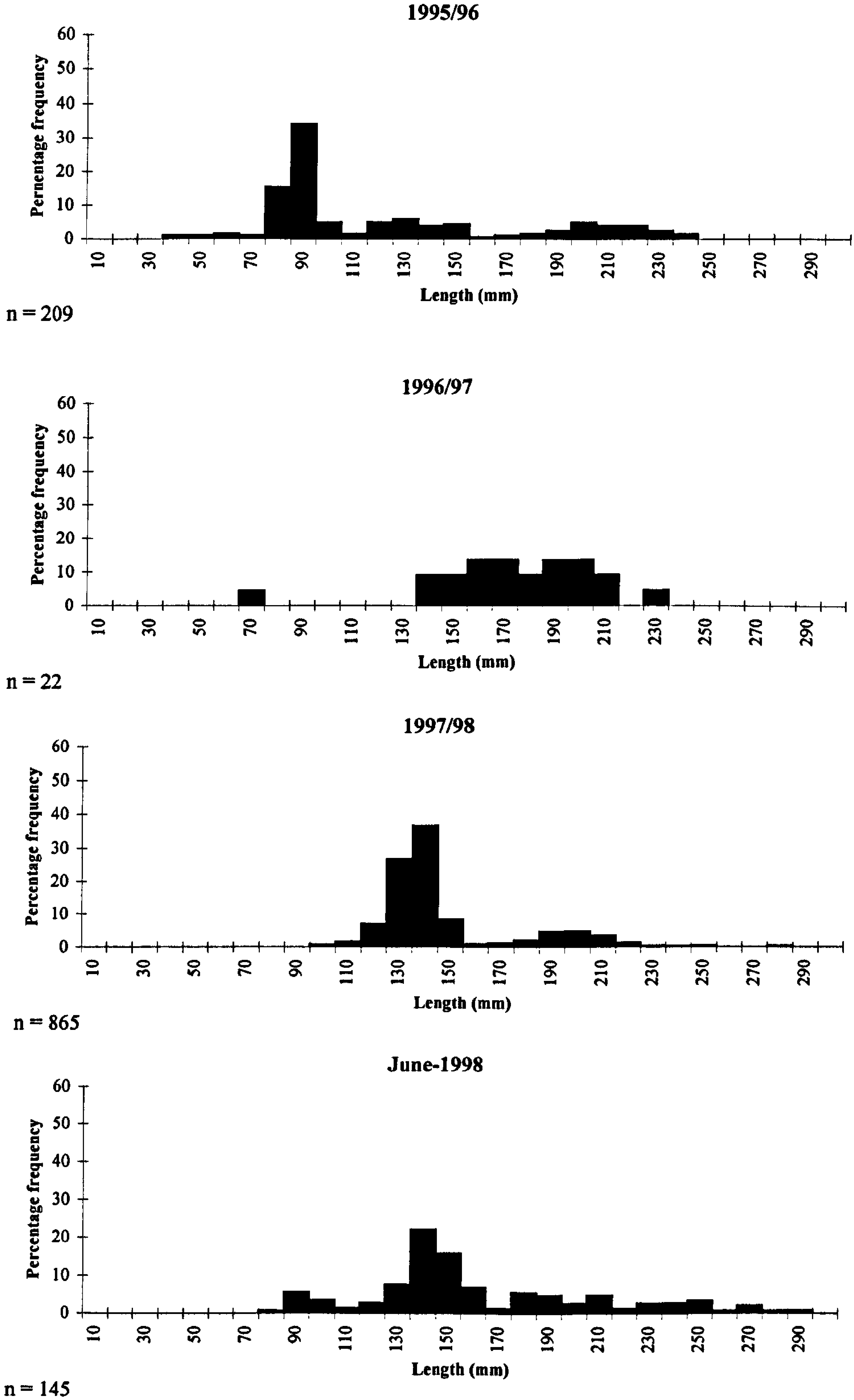


Figure 4.3 Length frequency distribution of roach at Holme Pierrepont Rowing Course.

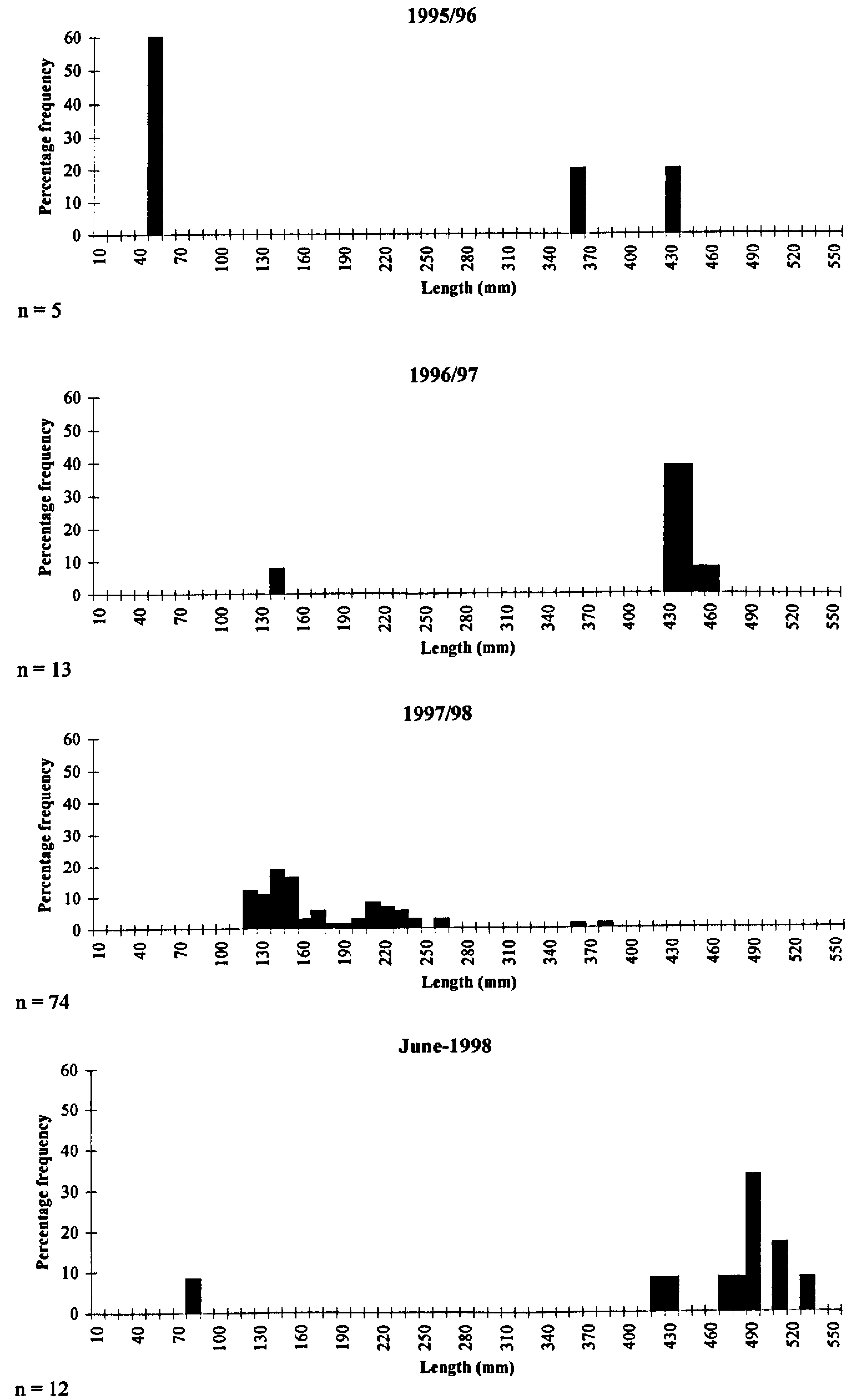


Figure 4.4 Length frequency distribution of common bream at Holme Pierrepont Rowing Course.

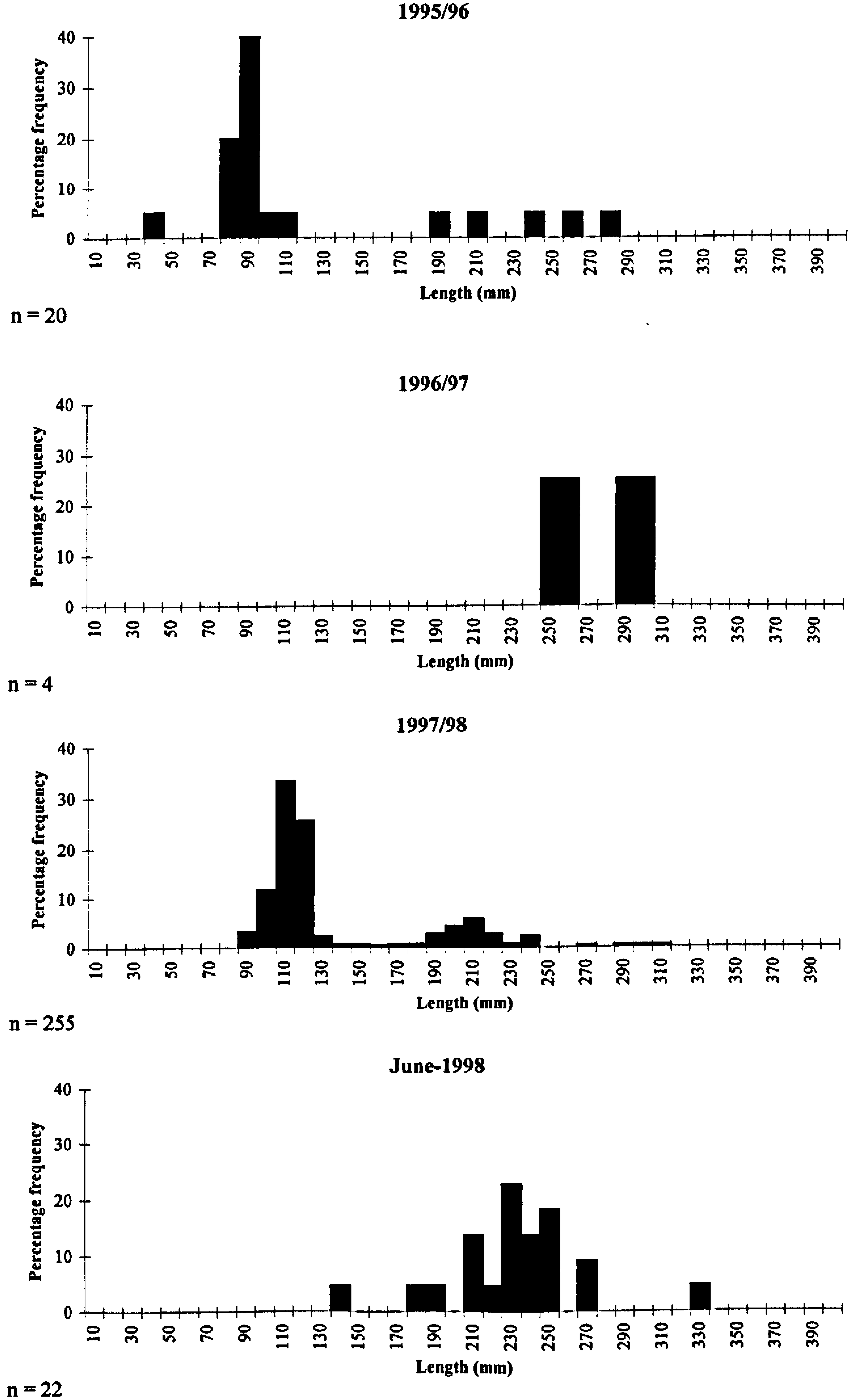


Figure 4.5 Length frequency distribution of perch at Holme Pierrepont Rowing Course.

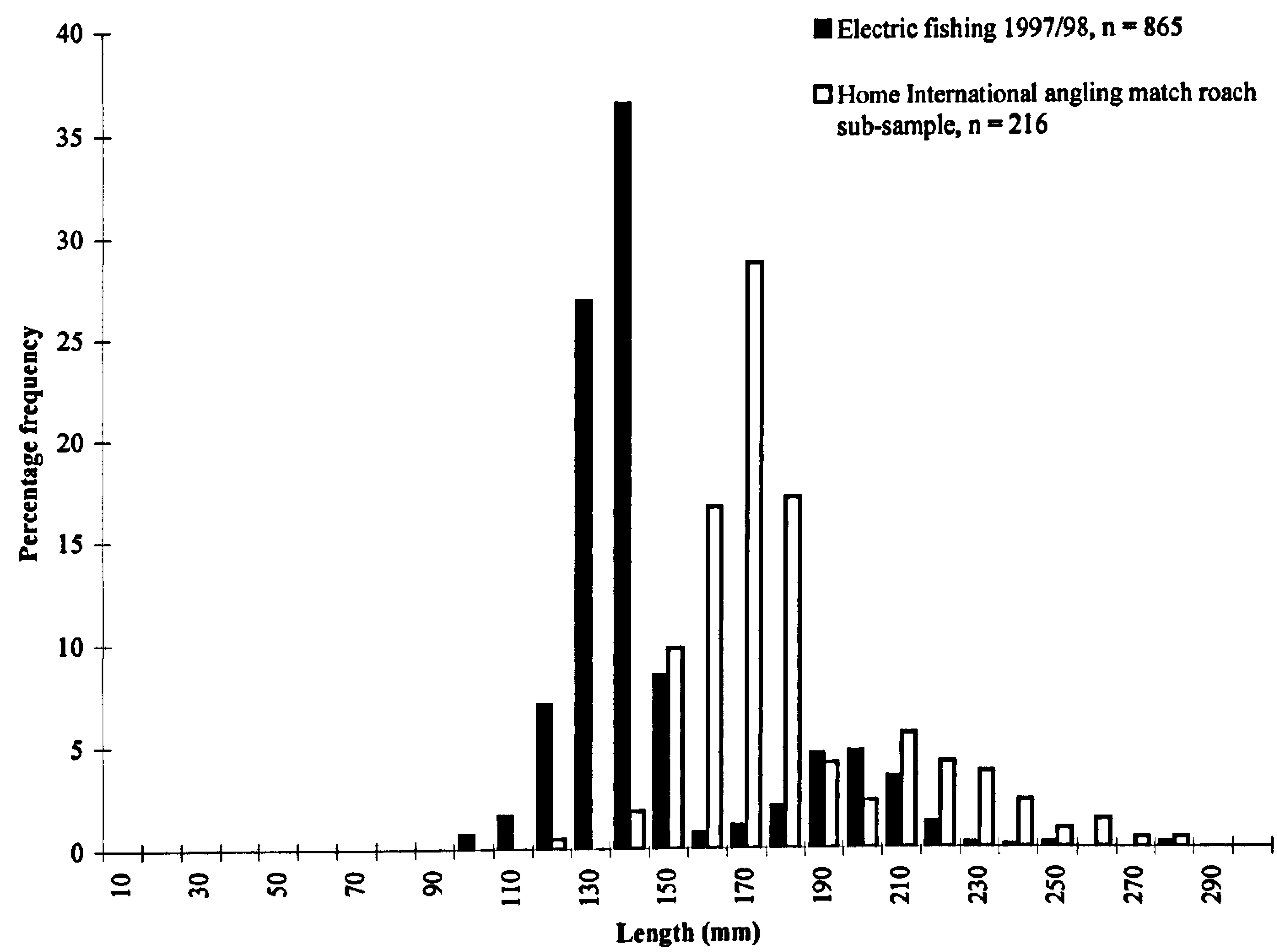


Figure 4.6 The percentage length frequency of roach caught by electric fishing, winter 1997/98, compared with the measured sub-sample of roach from the 1998 Home International angling competition.

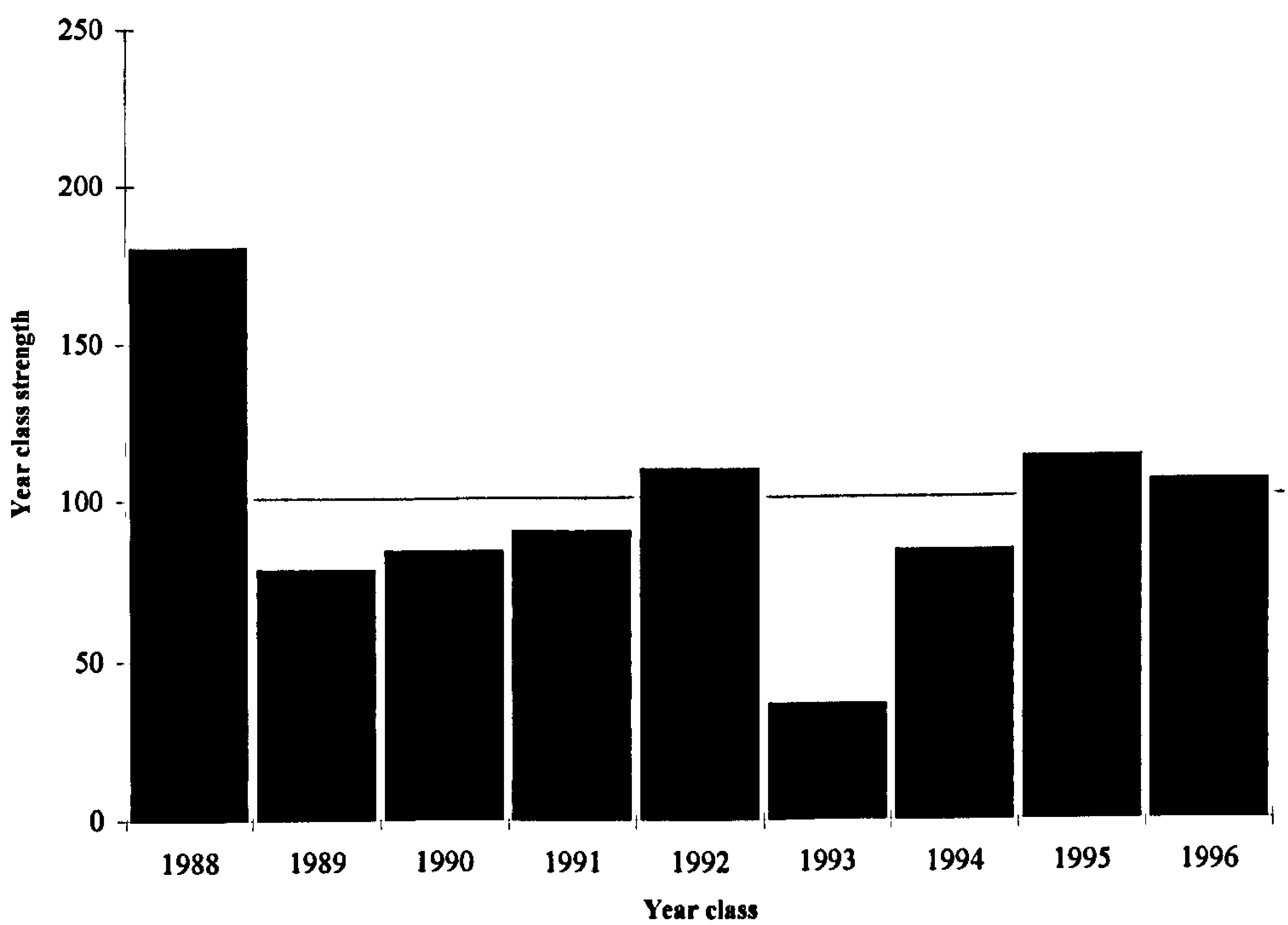


Figure 4.7 Year class strength of roach at Holme Pierrepont Rowing Course.

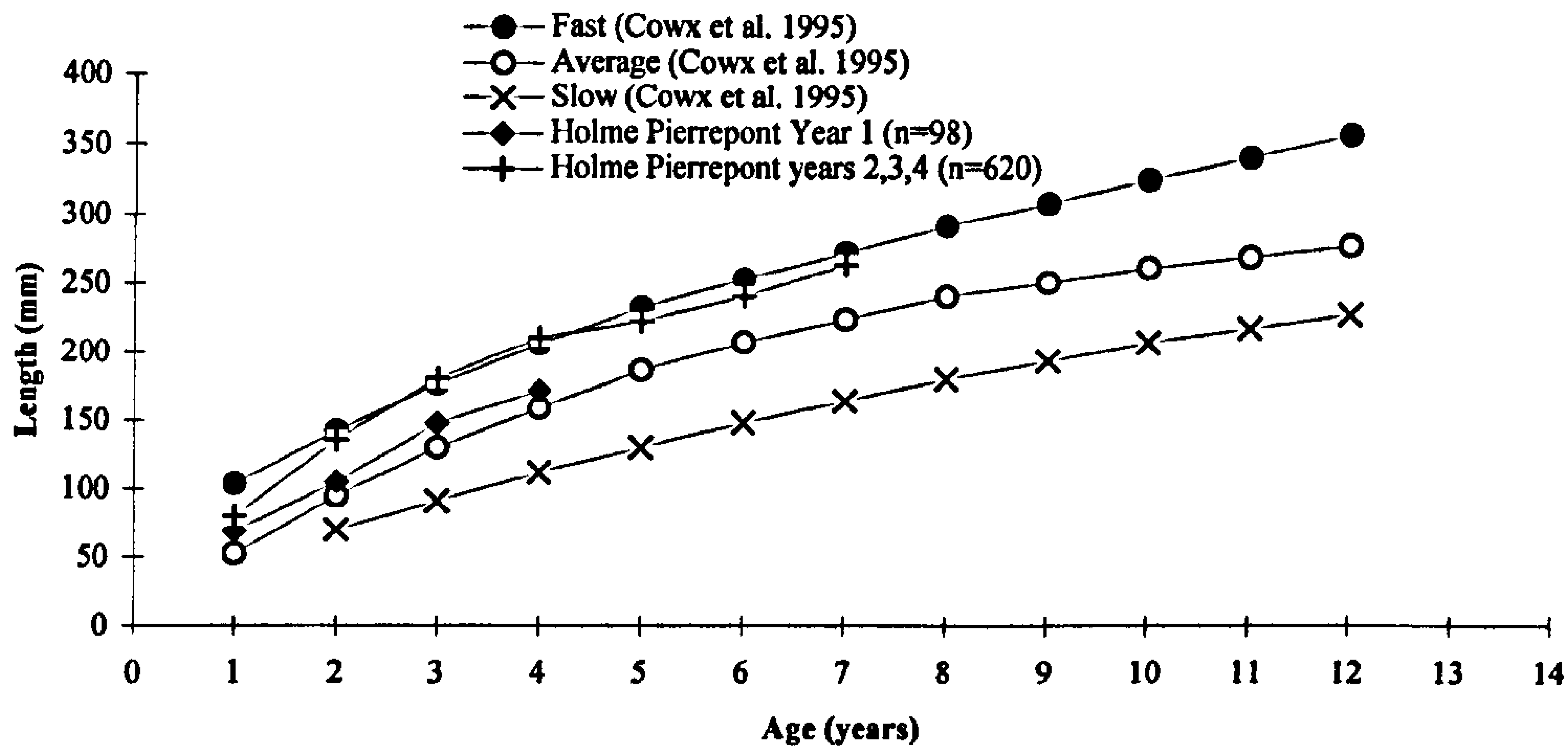


Figure 4.8 Growth of roach at Holme Pierrepont compared with standard growth curves.

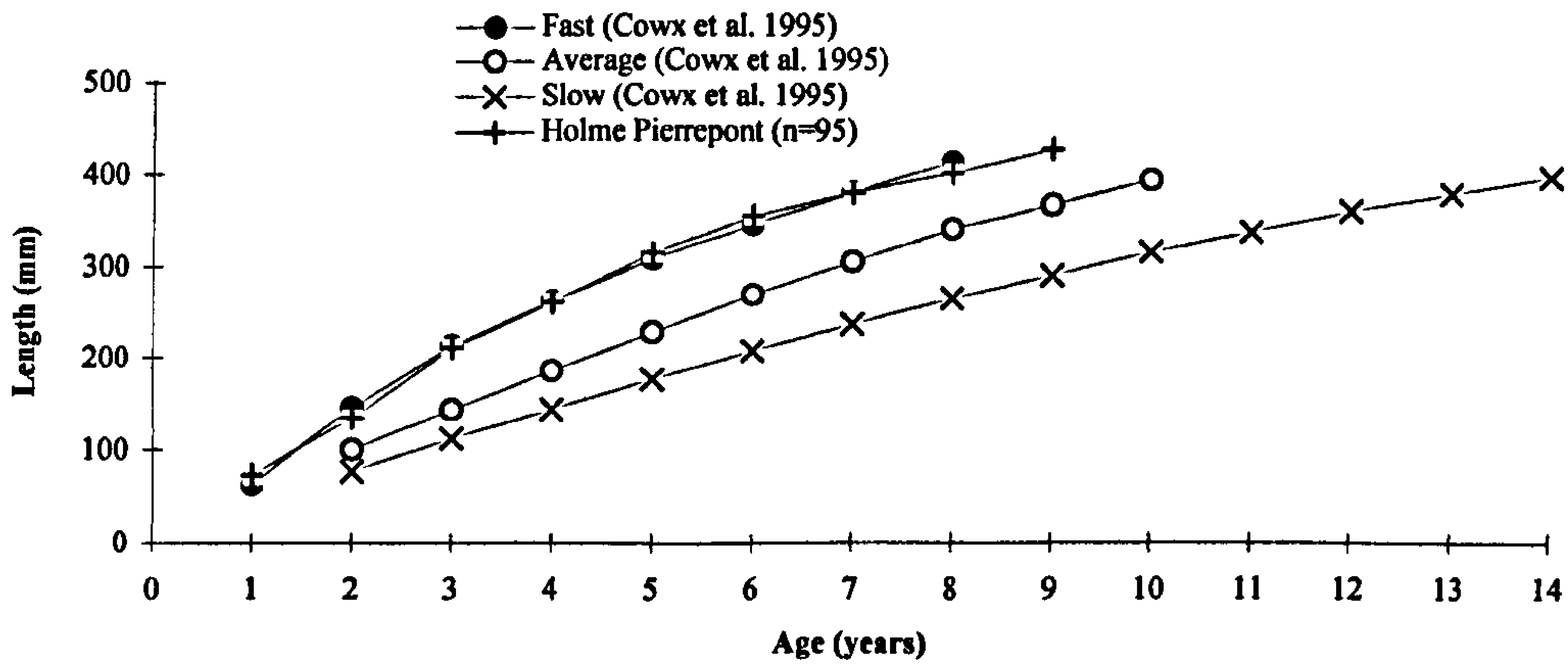


Figure 4.9 Growth of common bream at Holme Pierrepont compared with standard growth curves.

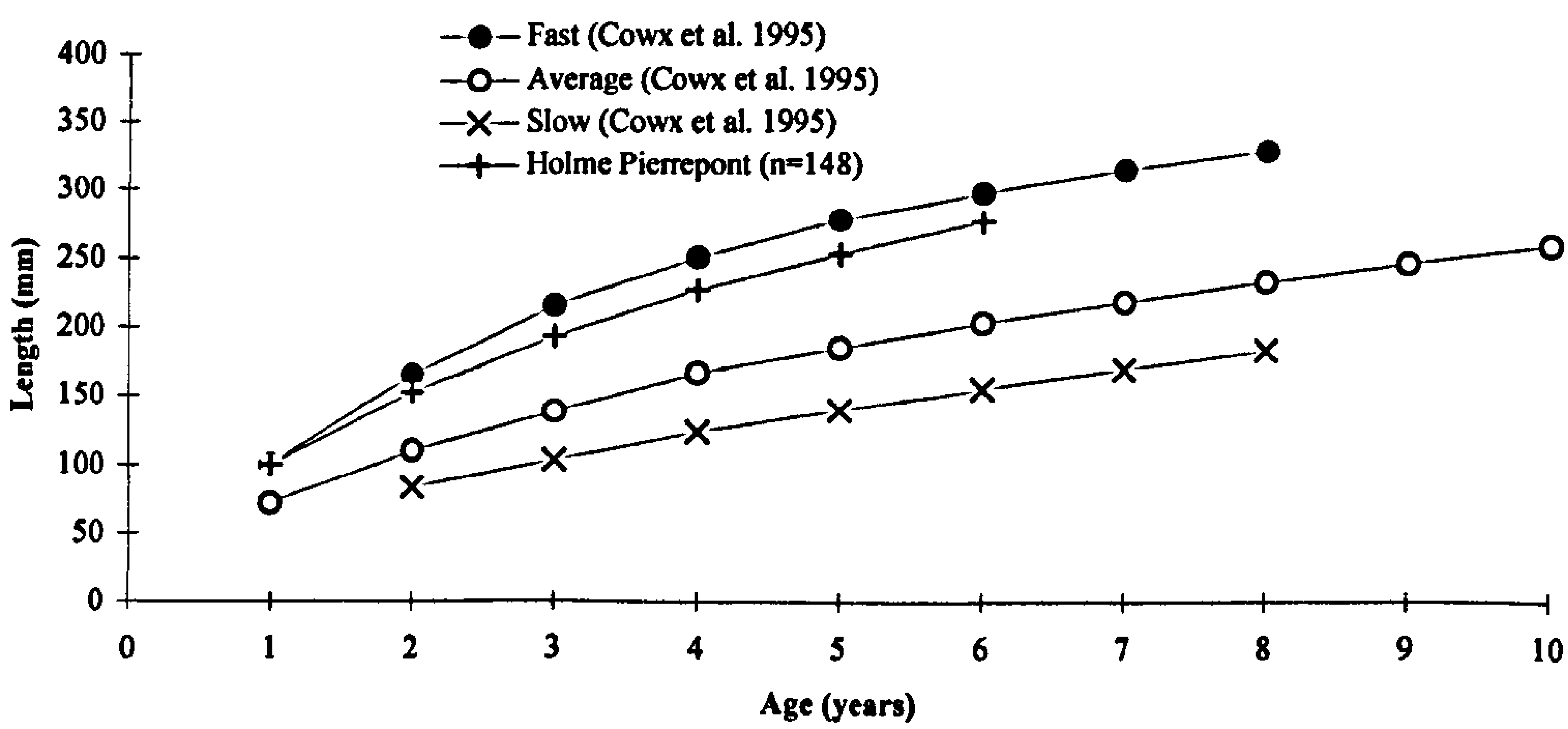


Figure 4.10 Growth of perch at Holme Pierrepont compared with standard growth curves.

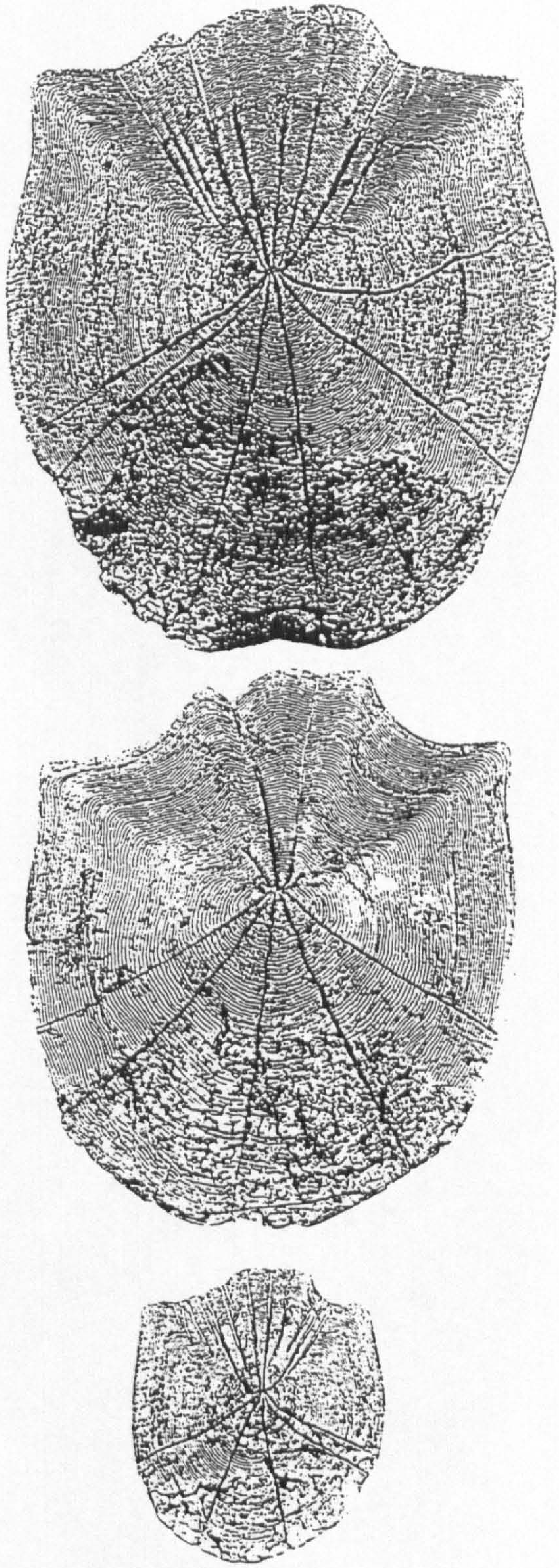


Figure 4.11 Comparison of scales from roach of age 2+:
Top: Holme Pierrepont, caught Janaury 1997, 210 mm.
Middle: River Trent, caught January 1998, 125 mm.
Bottom: Grimsargh number 3 Reservoir, caught October 1995, 99 mm.

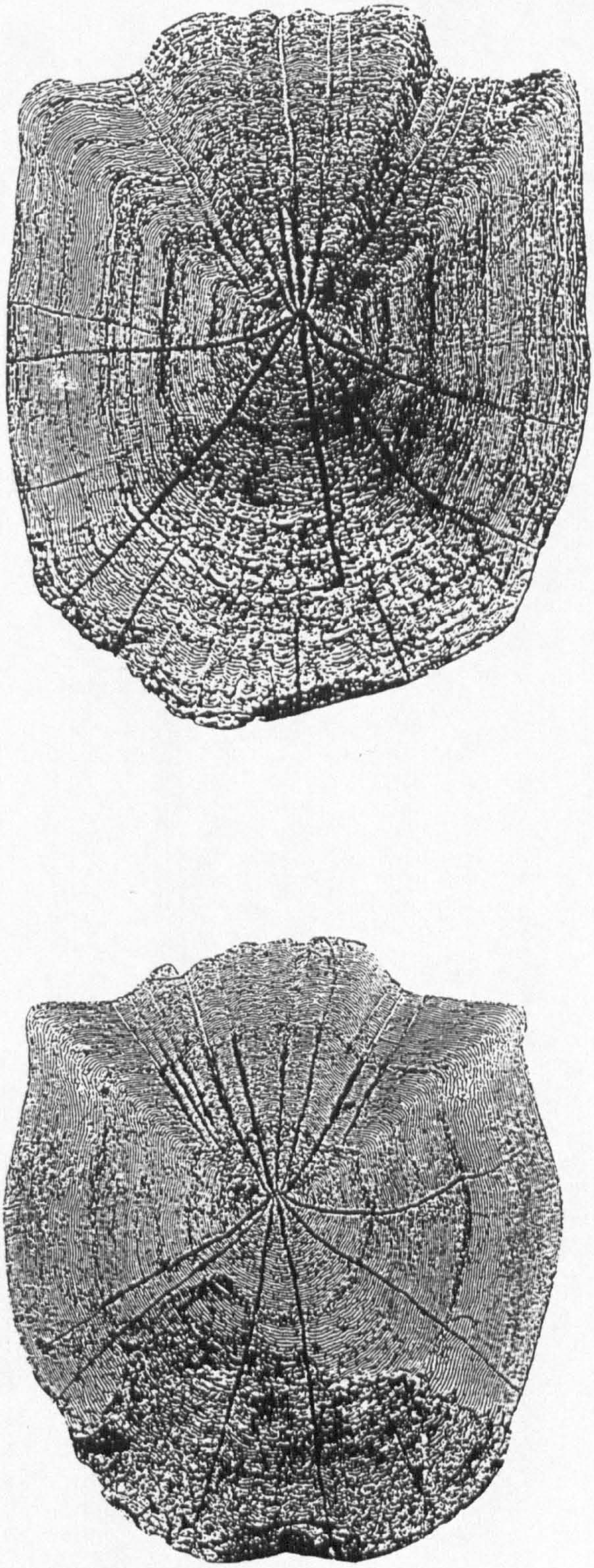


Figure 4.12 Scales from roach at Holme Pierrepont:
Top: caught October 1995, 206 mm, age 7+.
Bottom: caught January 1997, 210 mm, age 2+.

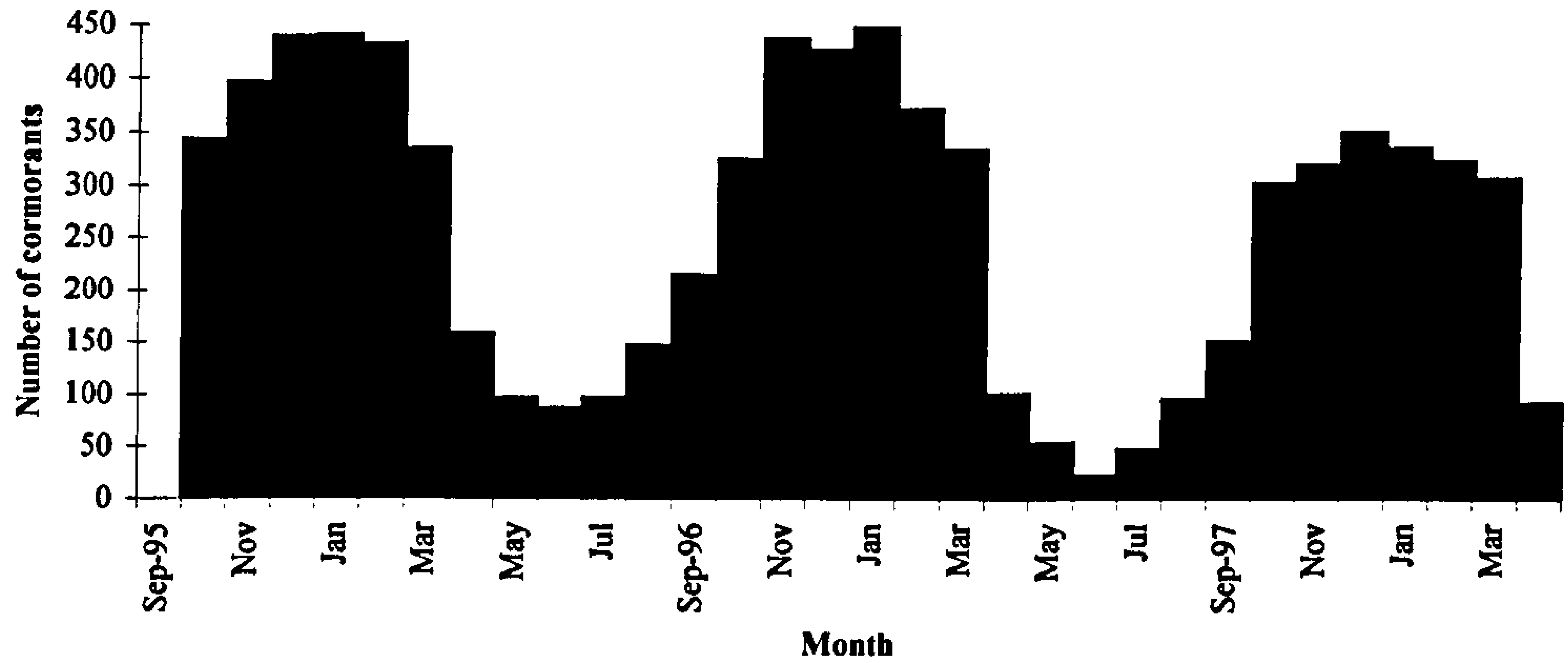


Figure 4.13 Attenborough cormorant roost numbers, September 1995 to April 1998.

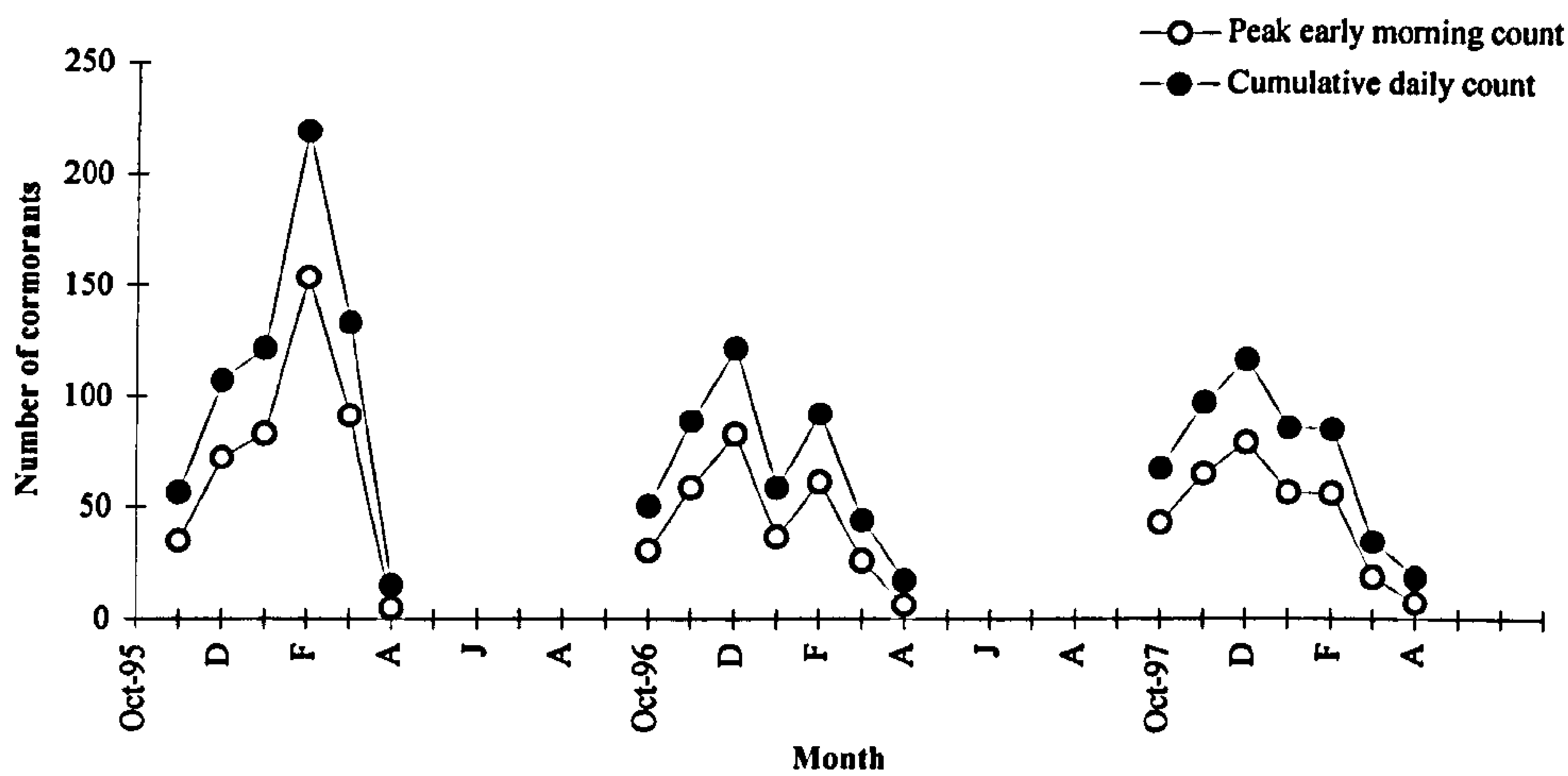


Figure 4.14 Peak early morning and estimated daily counts of cormorants feeding at Holme Pierrepont Rowing Course, 1995 to 1998.

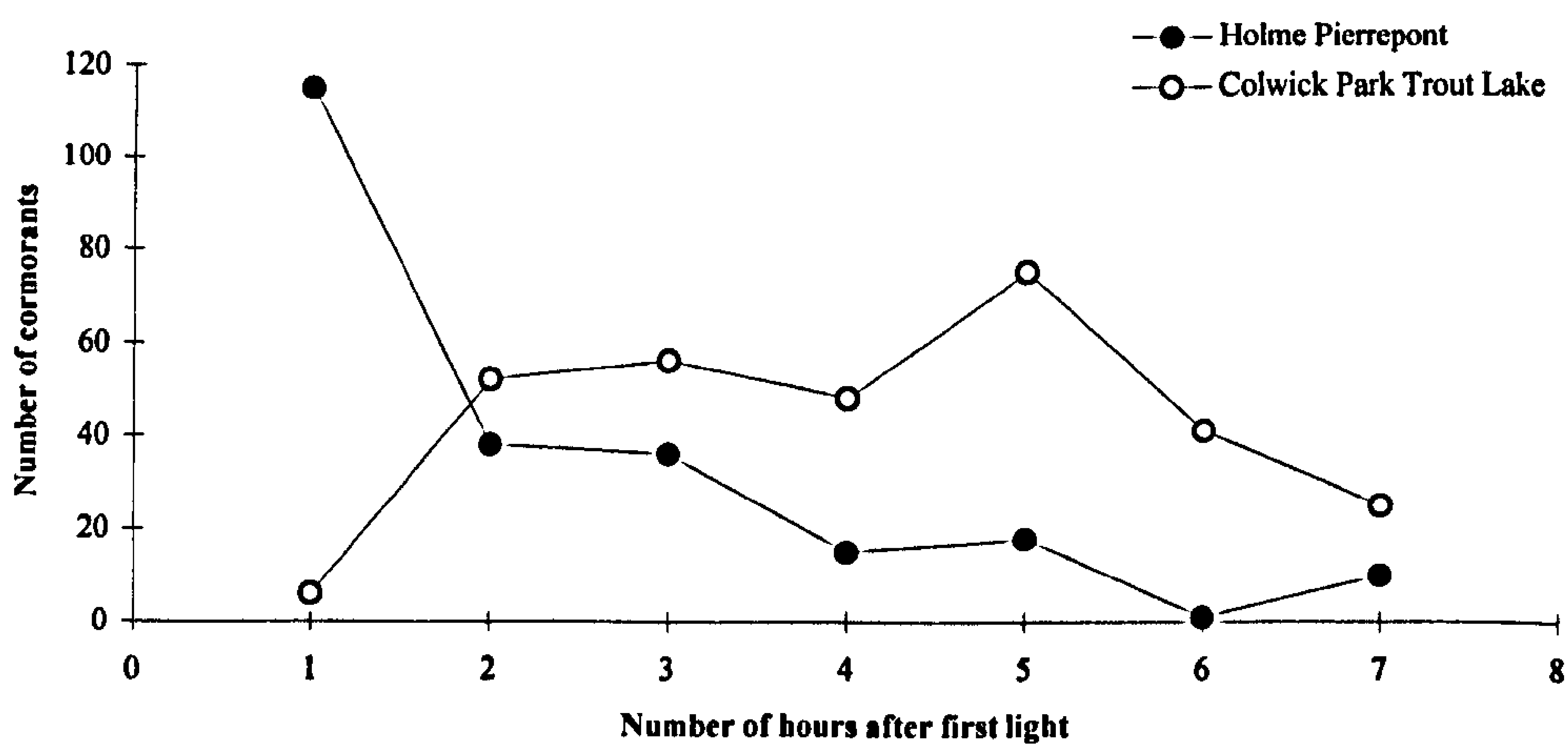


Figure 4.15 Typical pattern of the cormorant numbers recorded at Holme Pierrepont and Colwick Park Trout Lake in relation to the number of hours after first light (data from February 1996).

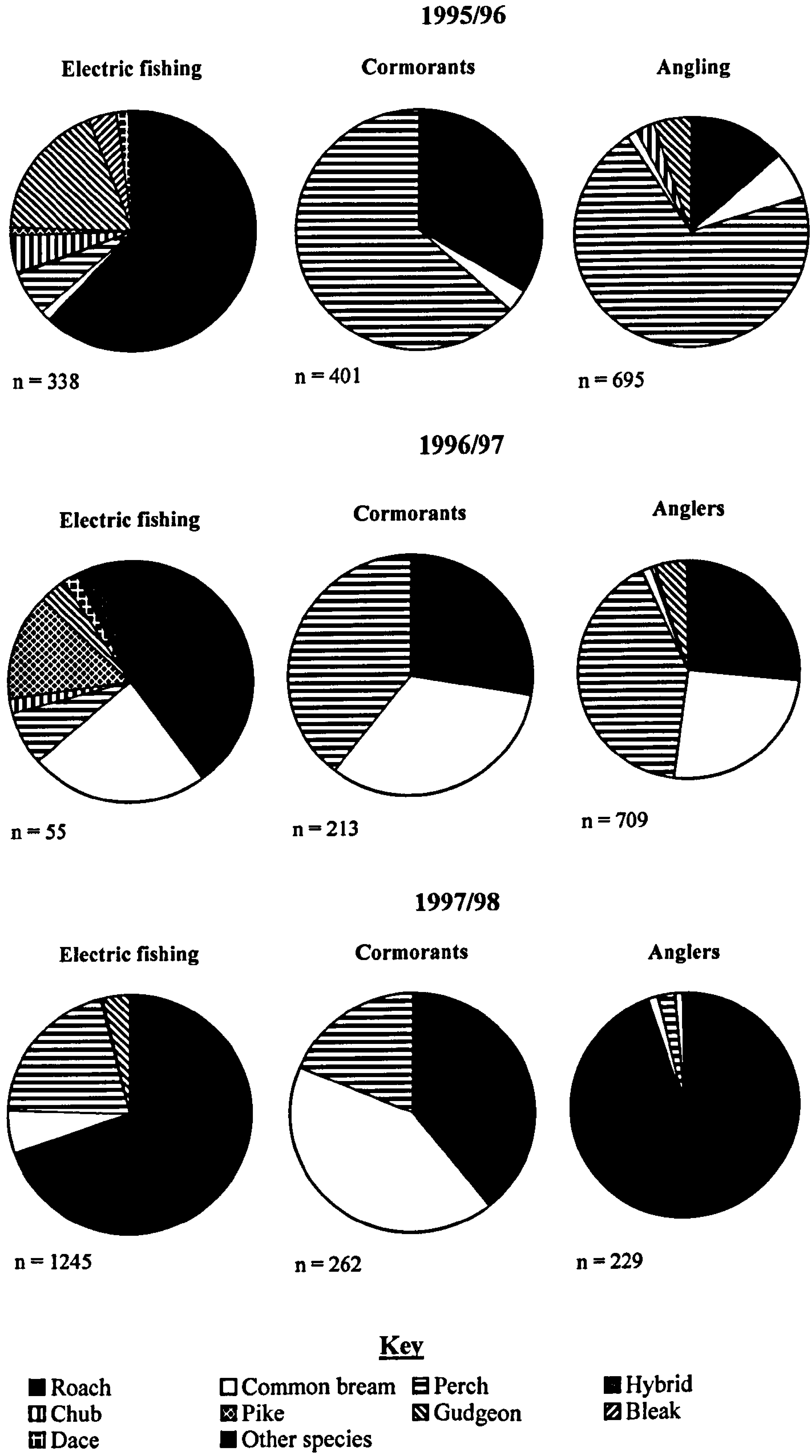


Figure 4.16 Species composition of cormorant predation, electric fishing and angler catches at Holme Pierrepont.

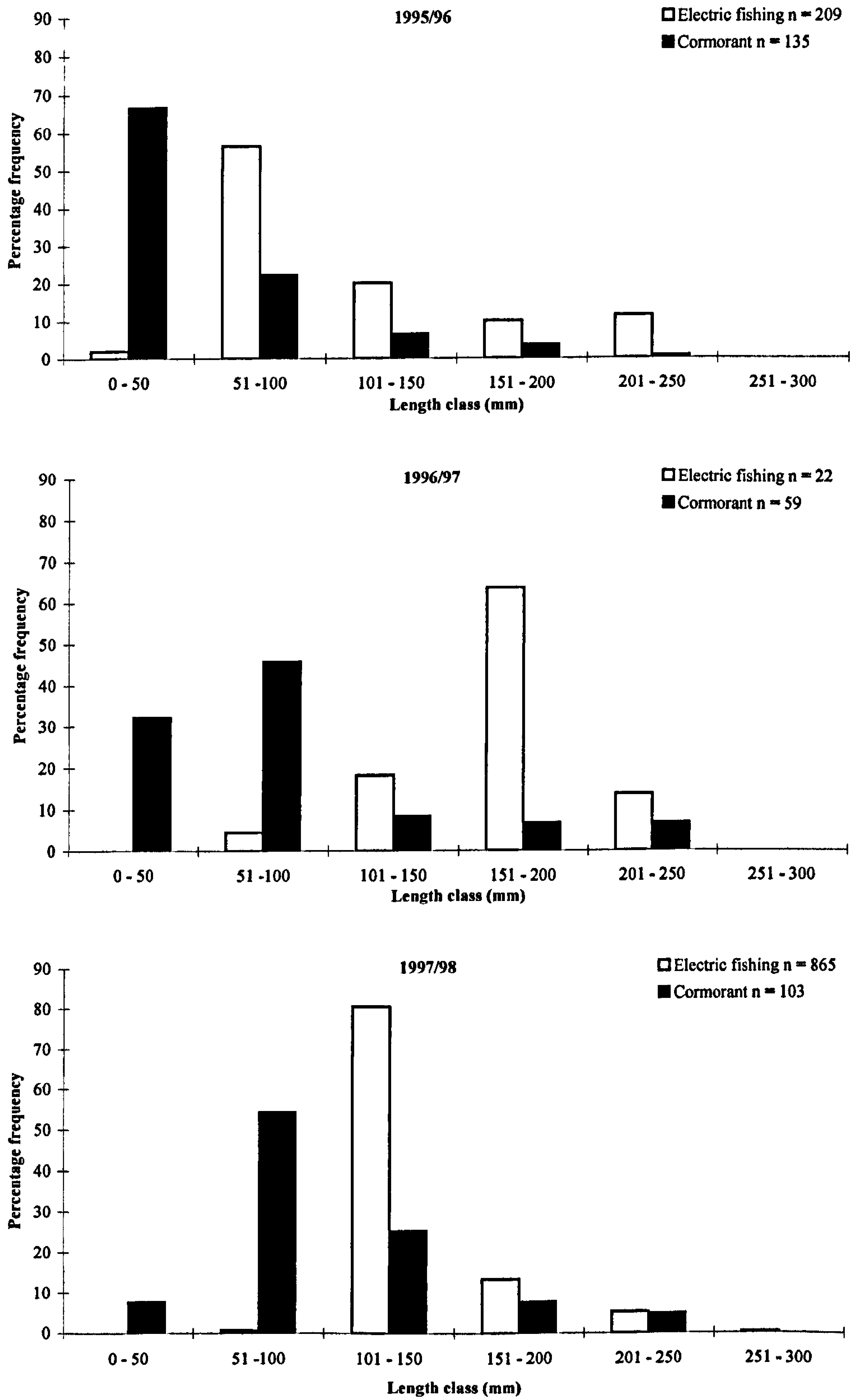


Figure 4.17 Length frequency distribution of roach ingested by cormorants and caught in electric fishing surveys at Holme Pierrepont Rowing Course.

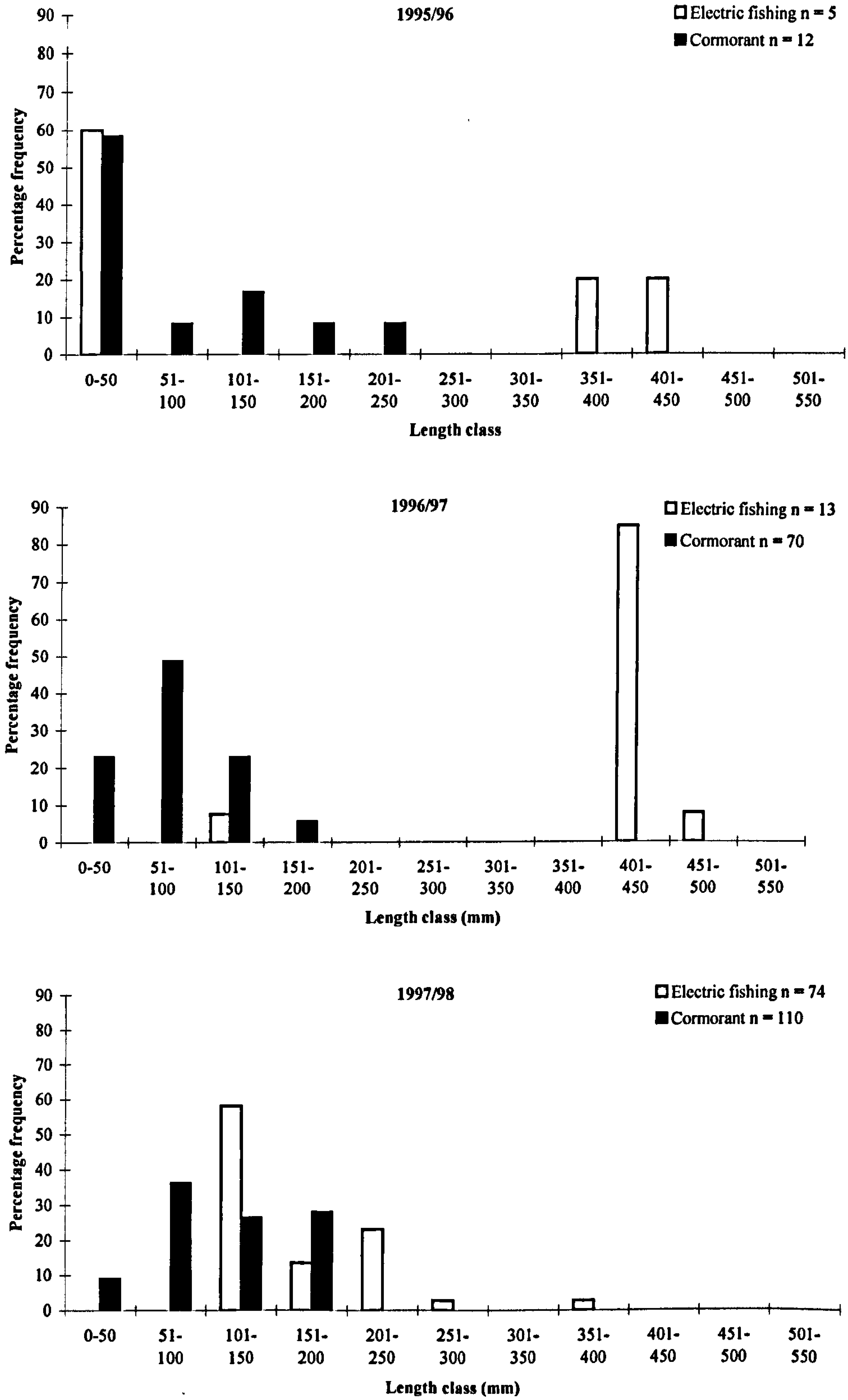


Figure 4.18 Length frequency distribution of common bream ingested by cormorants and caught in electric fishing surveys at Holme Pierrepont Rowing Course.

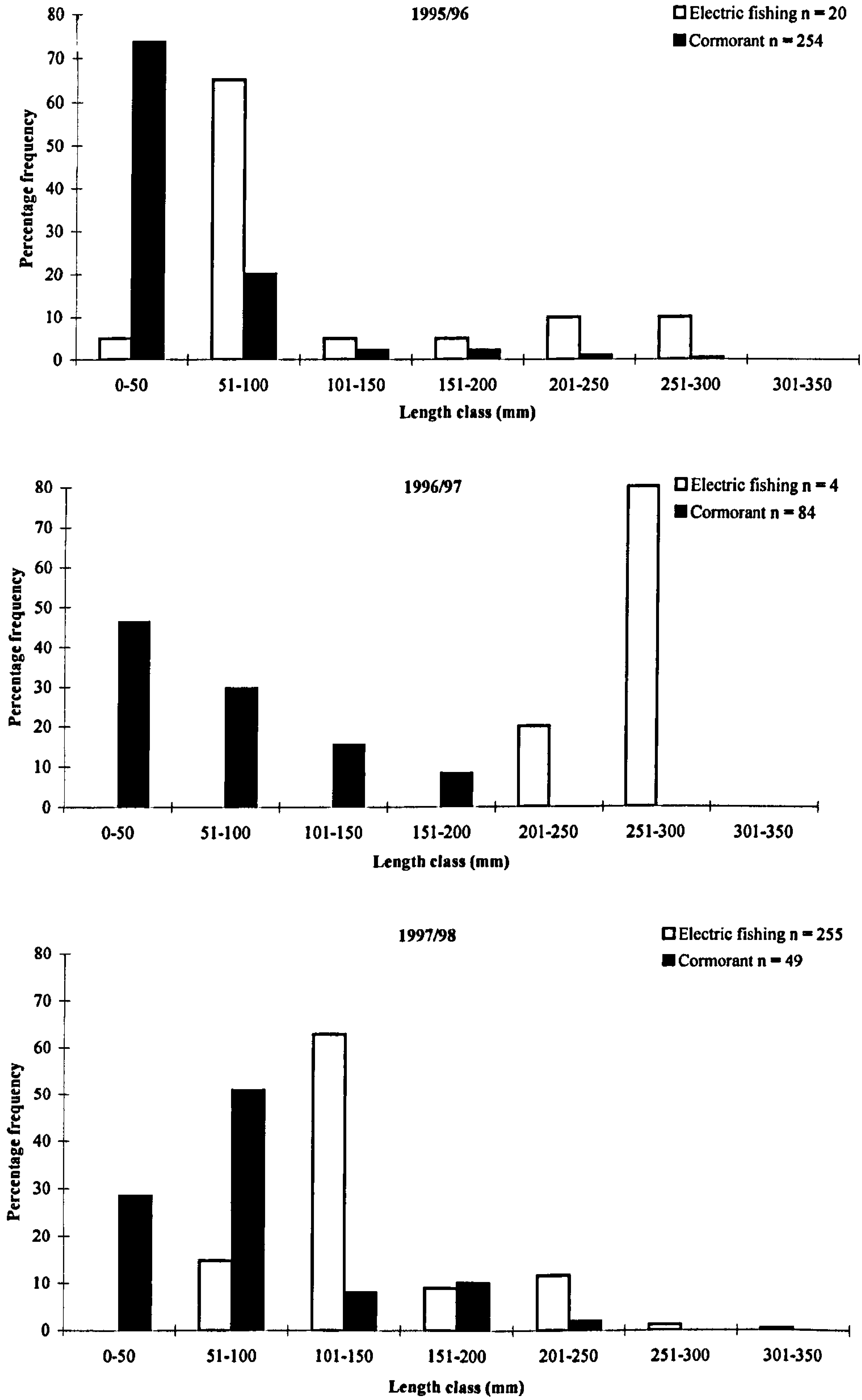


Figure 4.19 Length frequency distribution of perch ingested by cormorants and caught in electric fishing surveys at Holme Pierrepont Rowing Course.

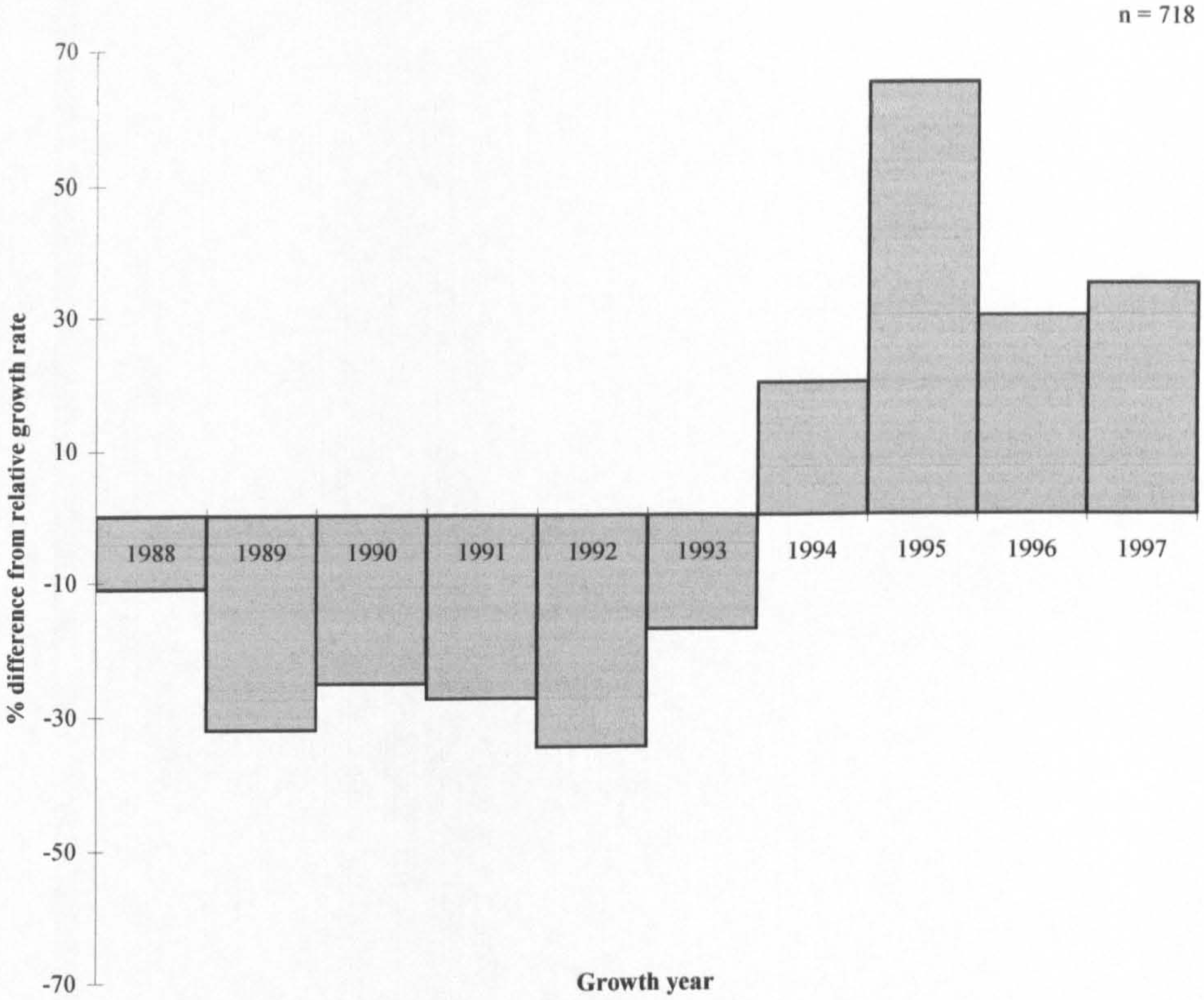


Figure 4.20 Relative annual growth rate of Holme Pierrepont roach 1988 to 1997.

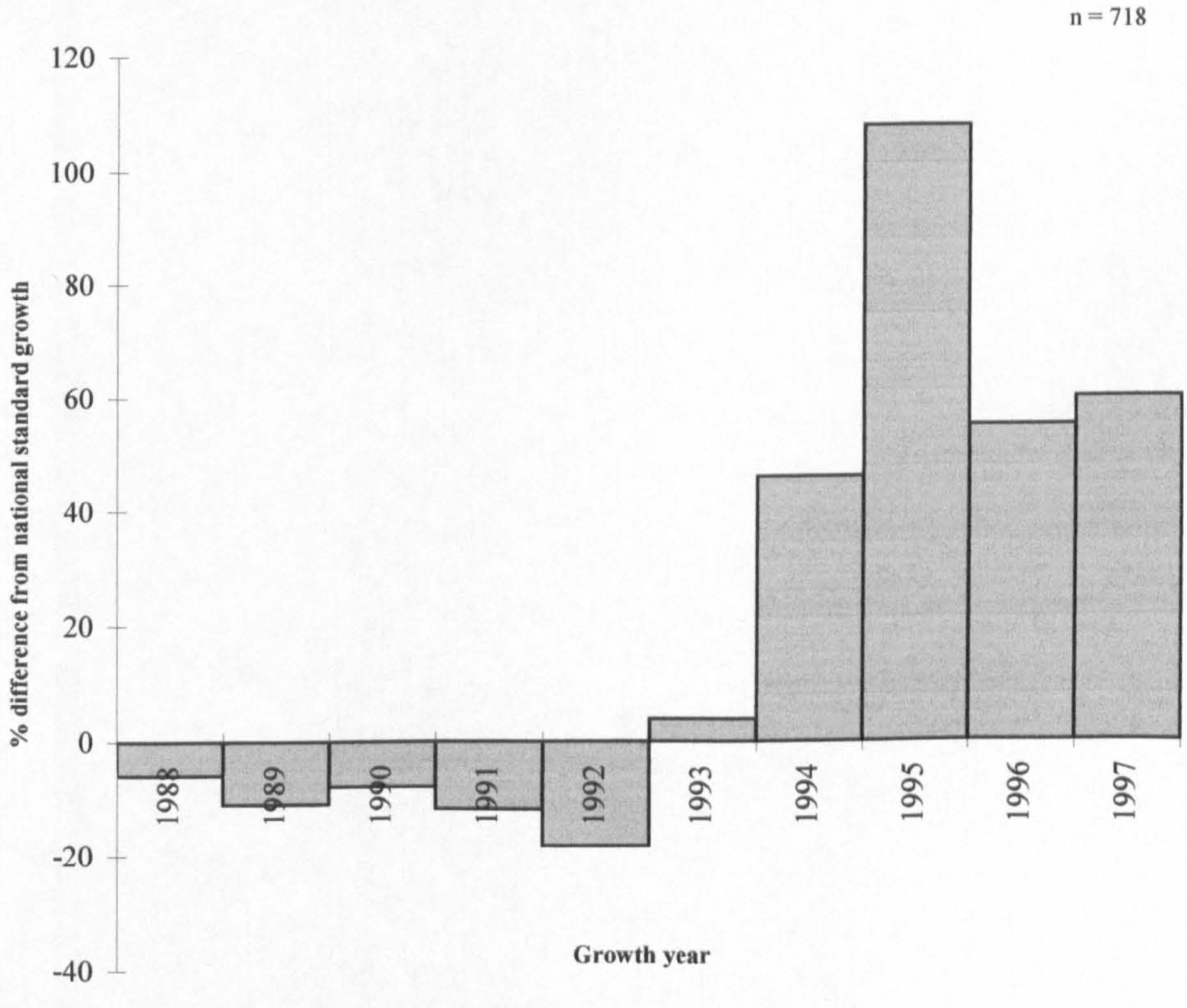


Figure 4.21 Annual growth rate of Holme Pierrepont roach compared to a standard roach growth rate (Hickley and Dexter 1979).

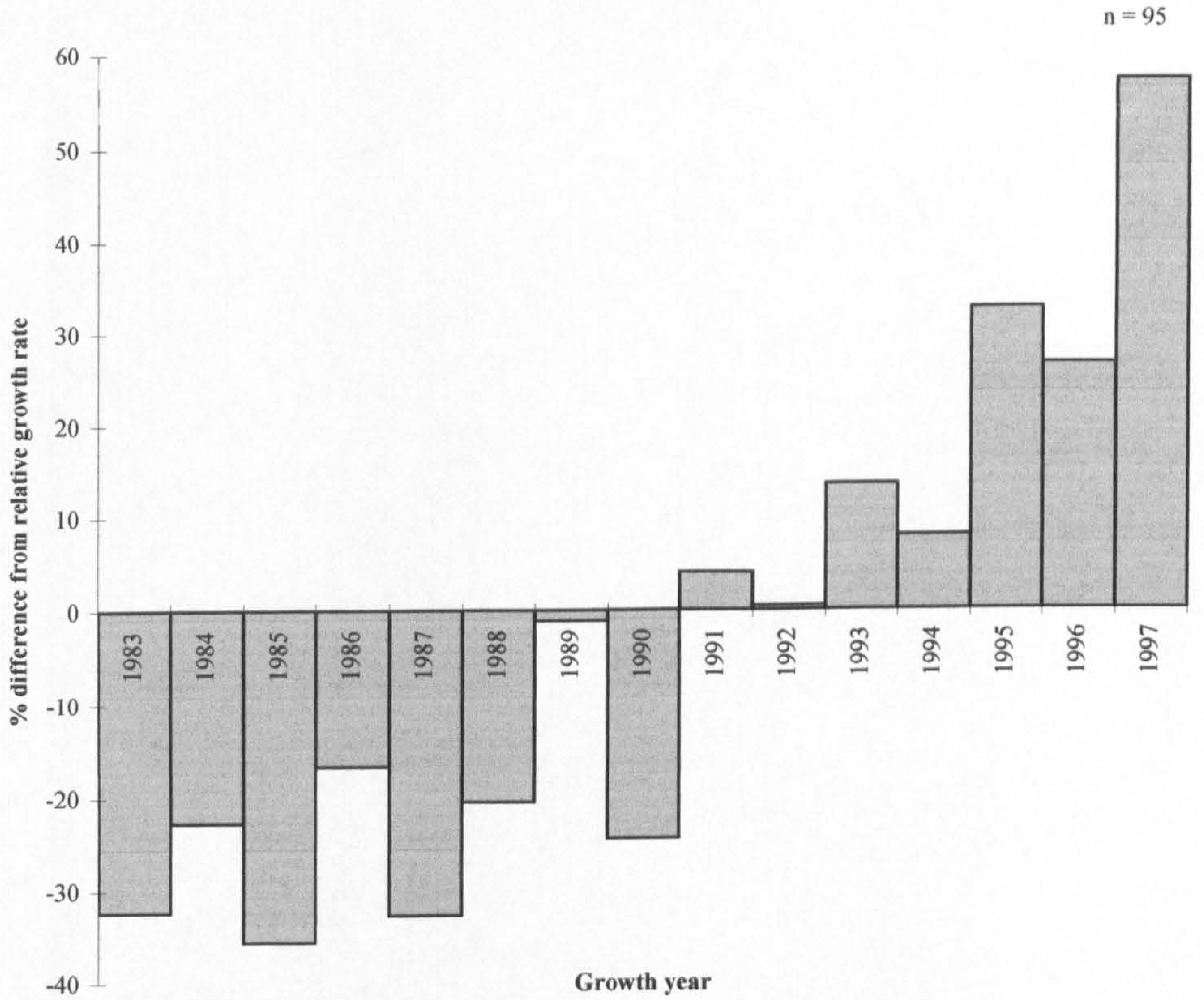


Figure 4.22 Relative annual growth of Holme Pierrepont common bream 1983 to 1997.

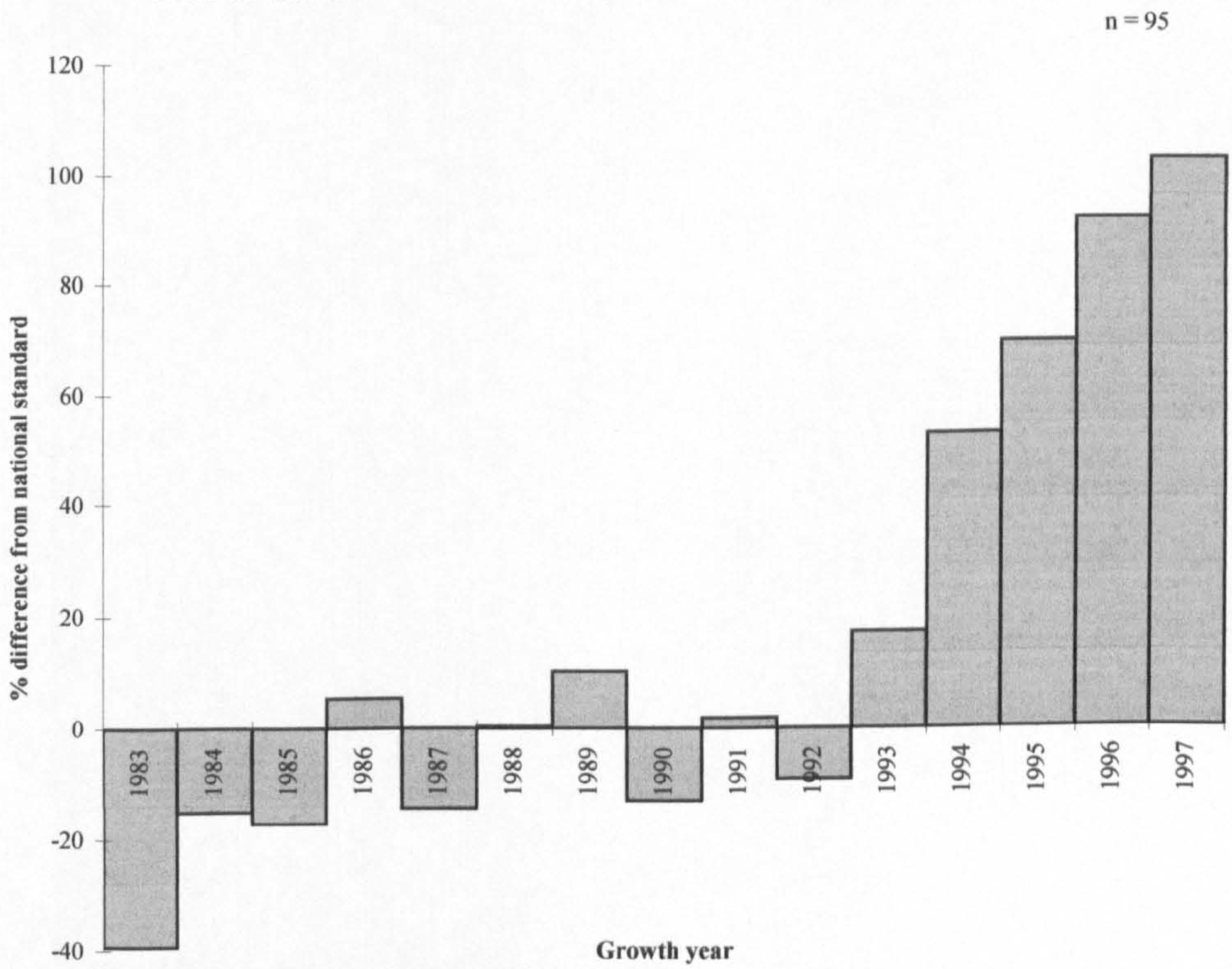


Figure 4.23 Annual growth of Holme Pierrepont common bream compared to a standard bream growth rate (Hickley and Dexter 1979).

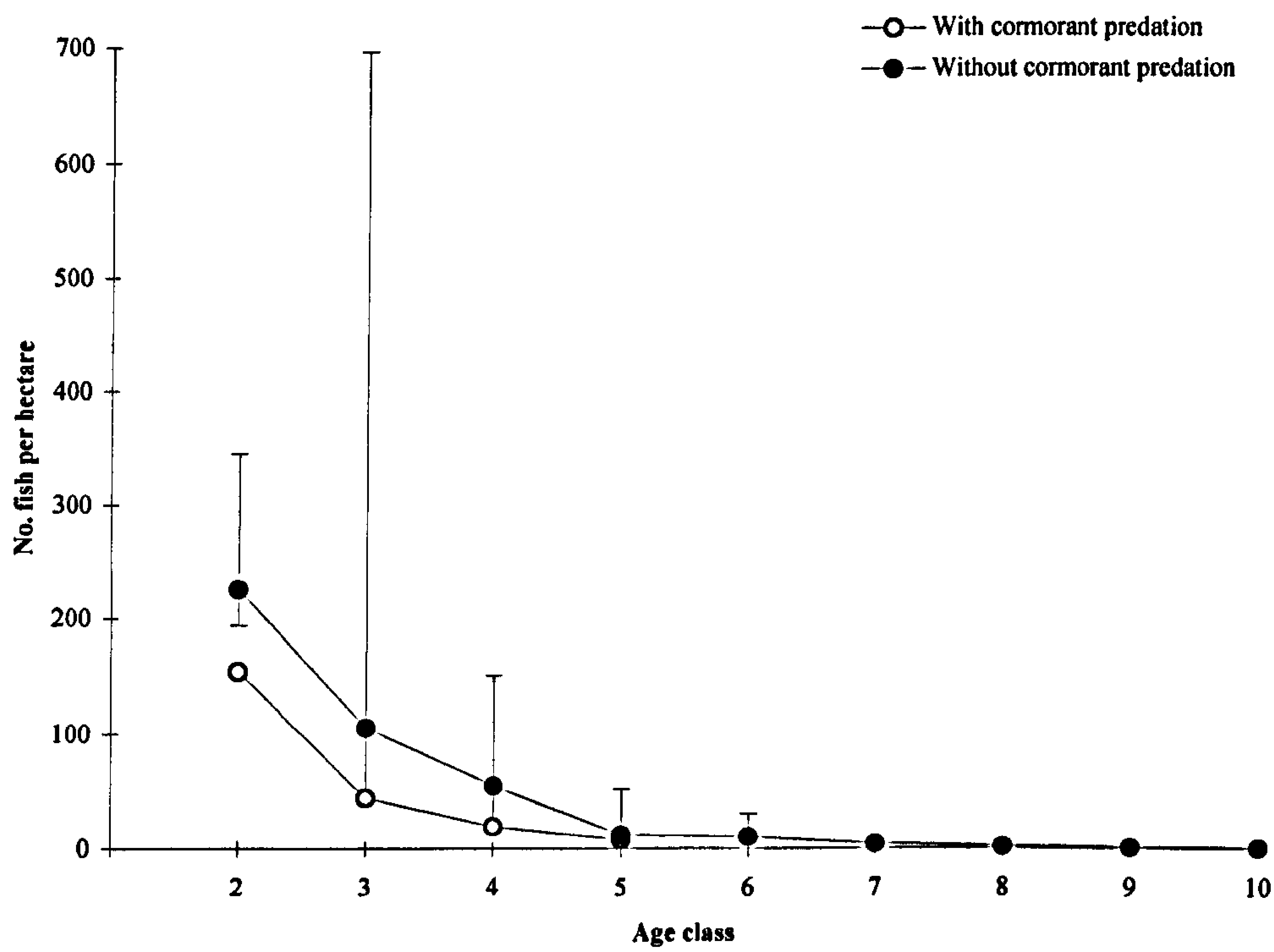


Figure 4.24 Impact of cormorant predation on the numbers of roach in 1996 at Holme Pierrepont Rowing Course.

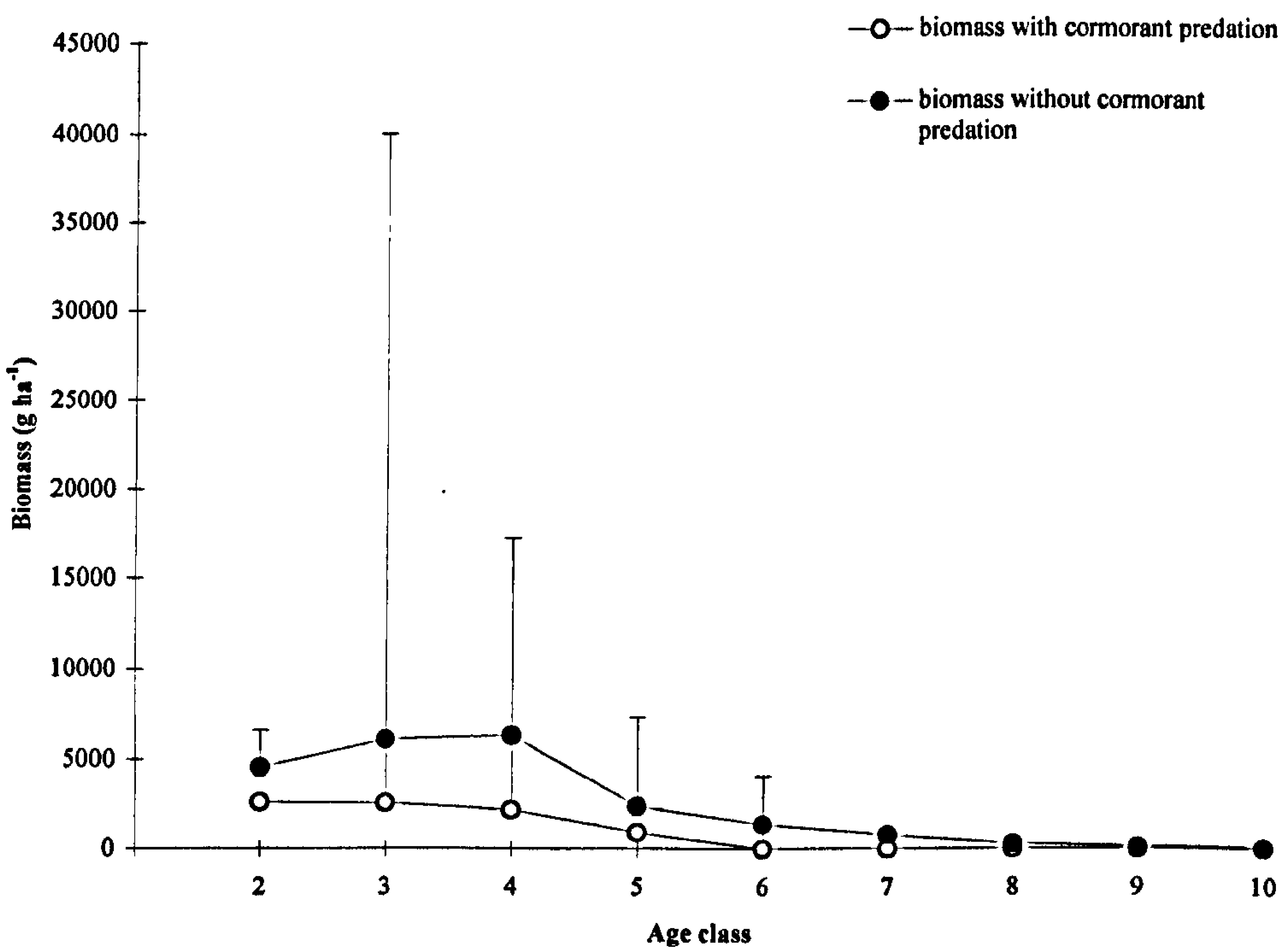


Figure 4.25 Impact of cormorant predation on the biomass of roach in 1996 at Holme Pierrepont Rowing Course.

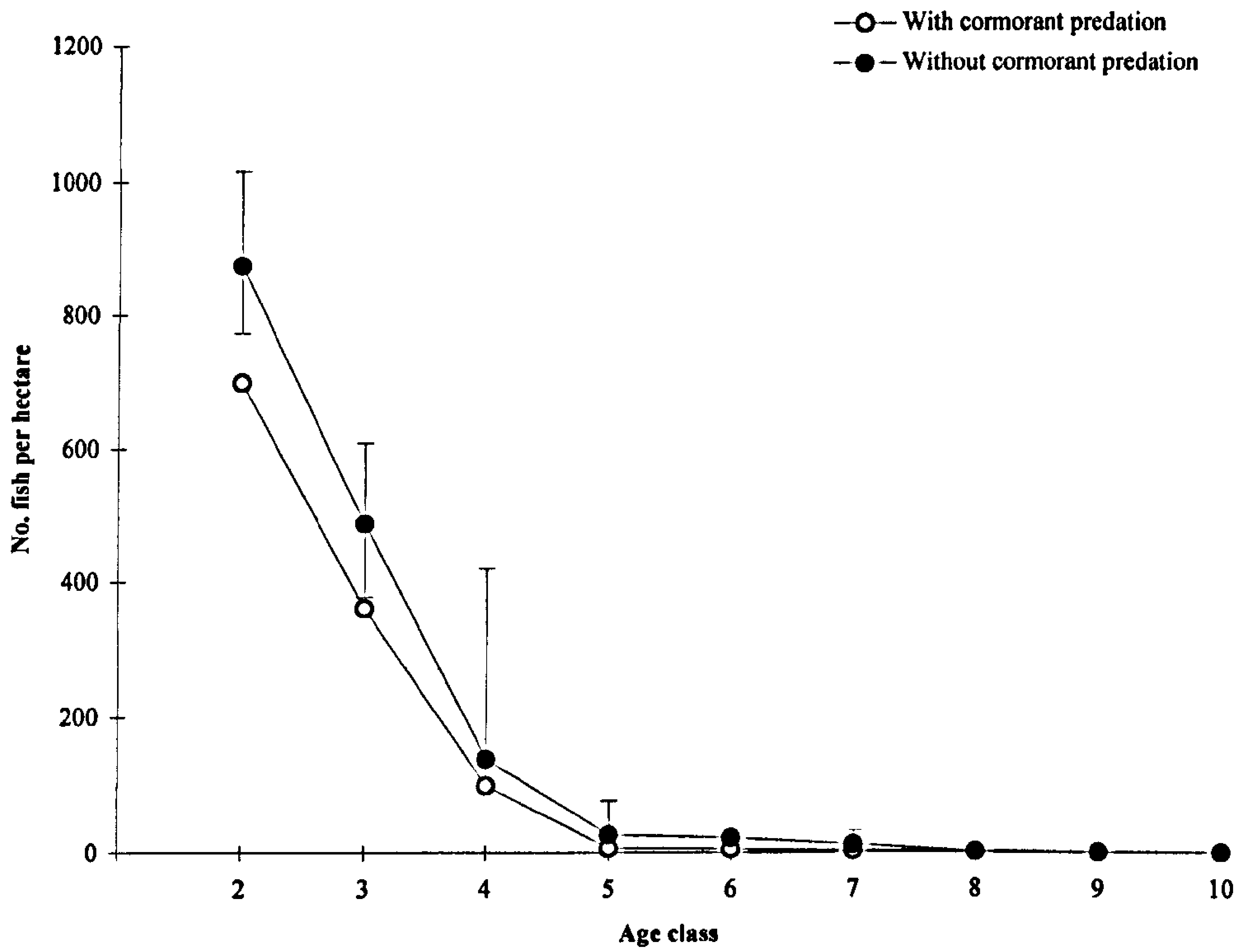


Figure 4.26 Impact of cormorant predation on the numbers of roach in 1997 at Holme Pierrepont Rowing Course.

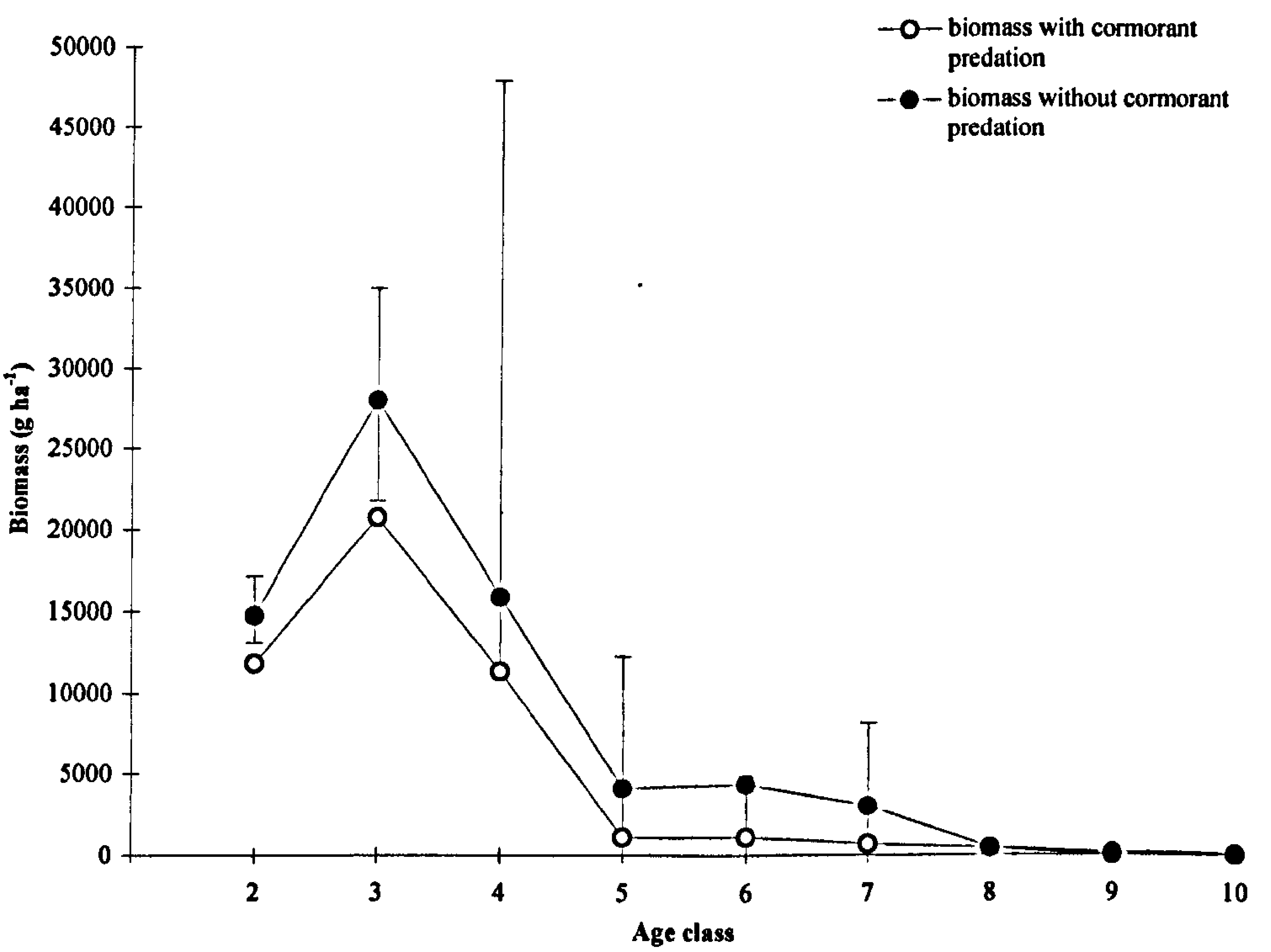


Figure 4.27 Impact of cormorant predation on the biomass of roach in 1997 at Holme Pierrepont Rowing Course.

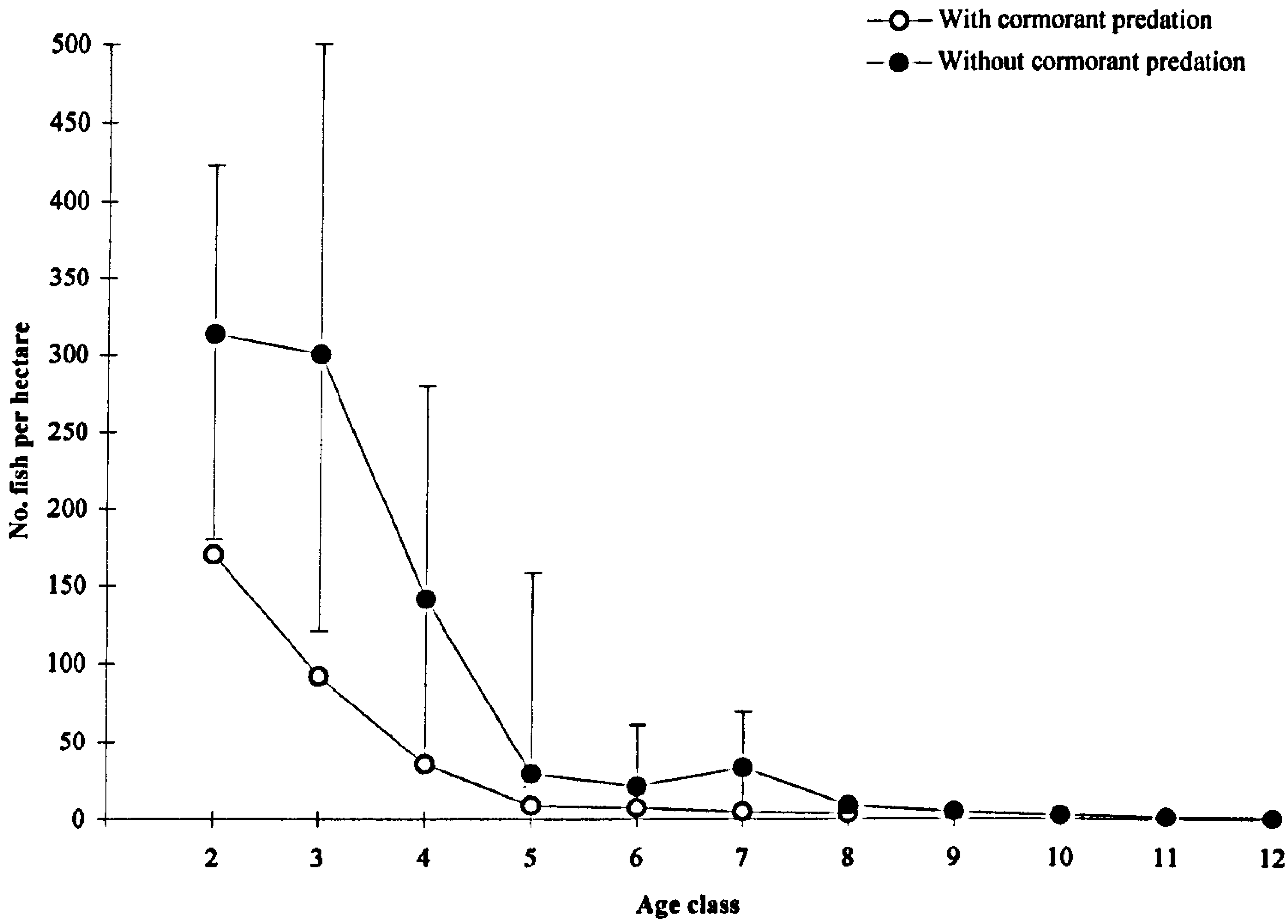


Figure 4.28 Impact of cormorant predation on the numbers of roach in 1998 at Holme Pierrepont Rowing Course.

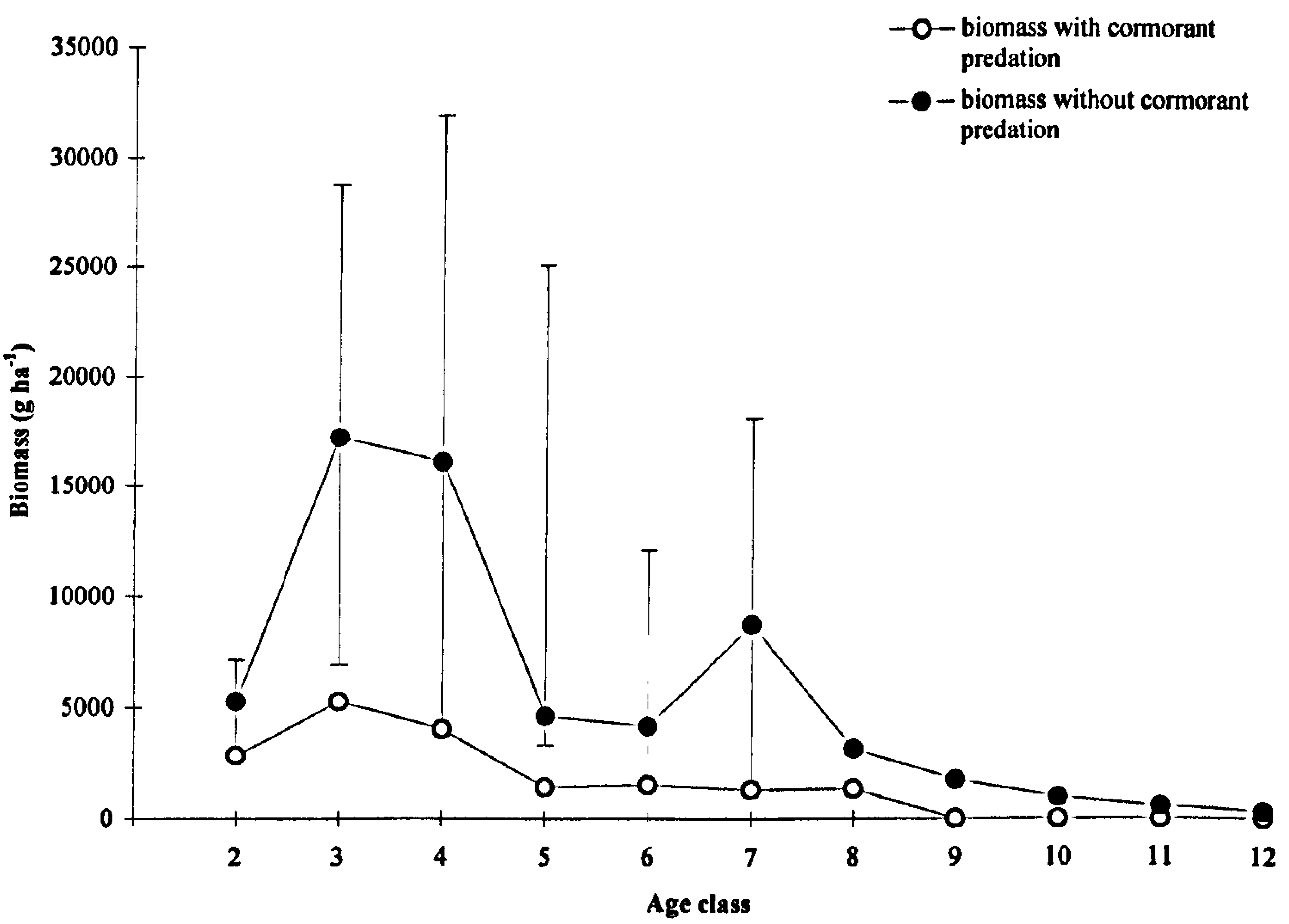


Figure 4.29 Impact of cormorant predation on the biomass of roach in 1998 at Holme Pierrepont Rowing Course.

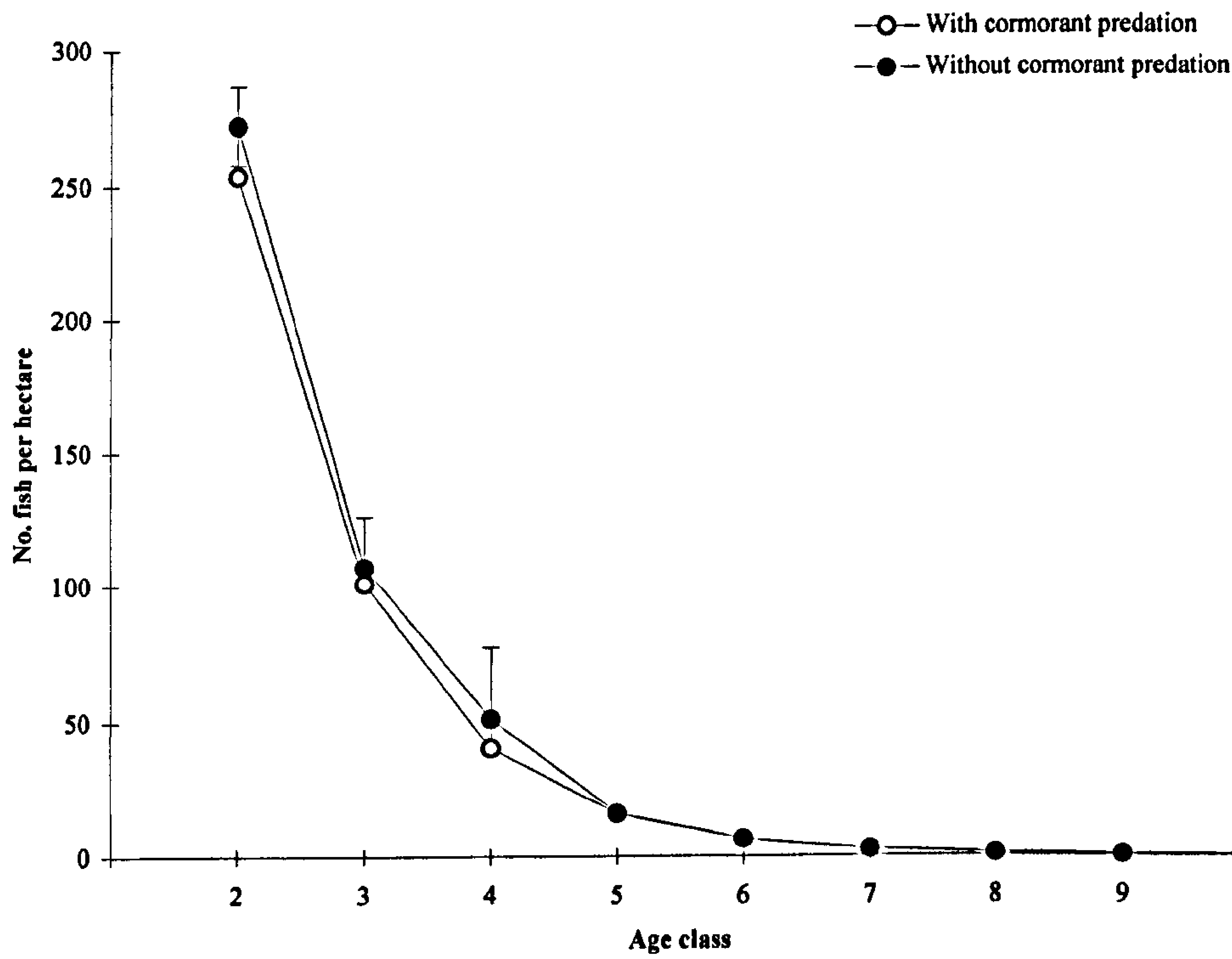


Figure 4.30 Impact of cormorant predation on the numbers of common bream in 1996 at Holme Pierrepont Rowing Course.

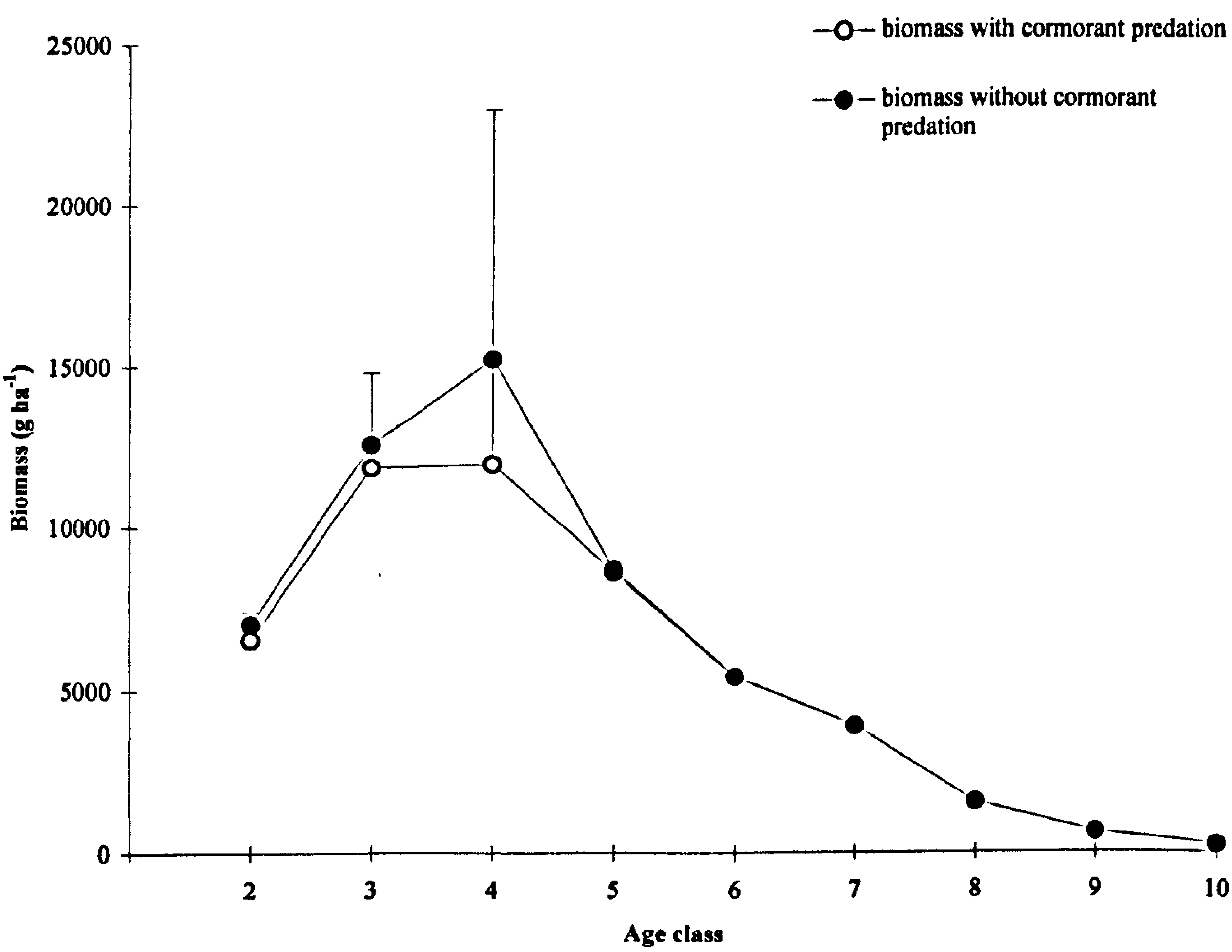


Figure 4.31 Impact of cormorant predation on the biomass of common bream in 1996 at Holme Pierrepont Rowing Course.

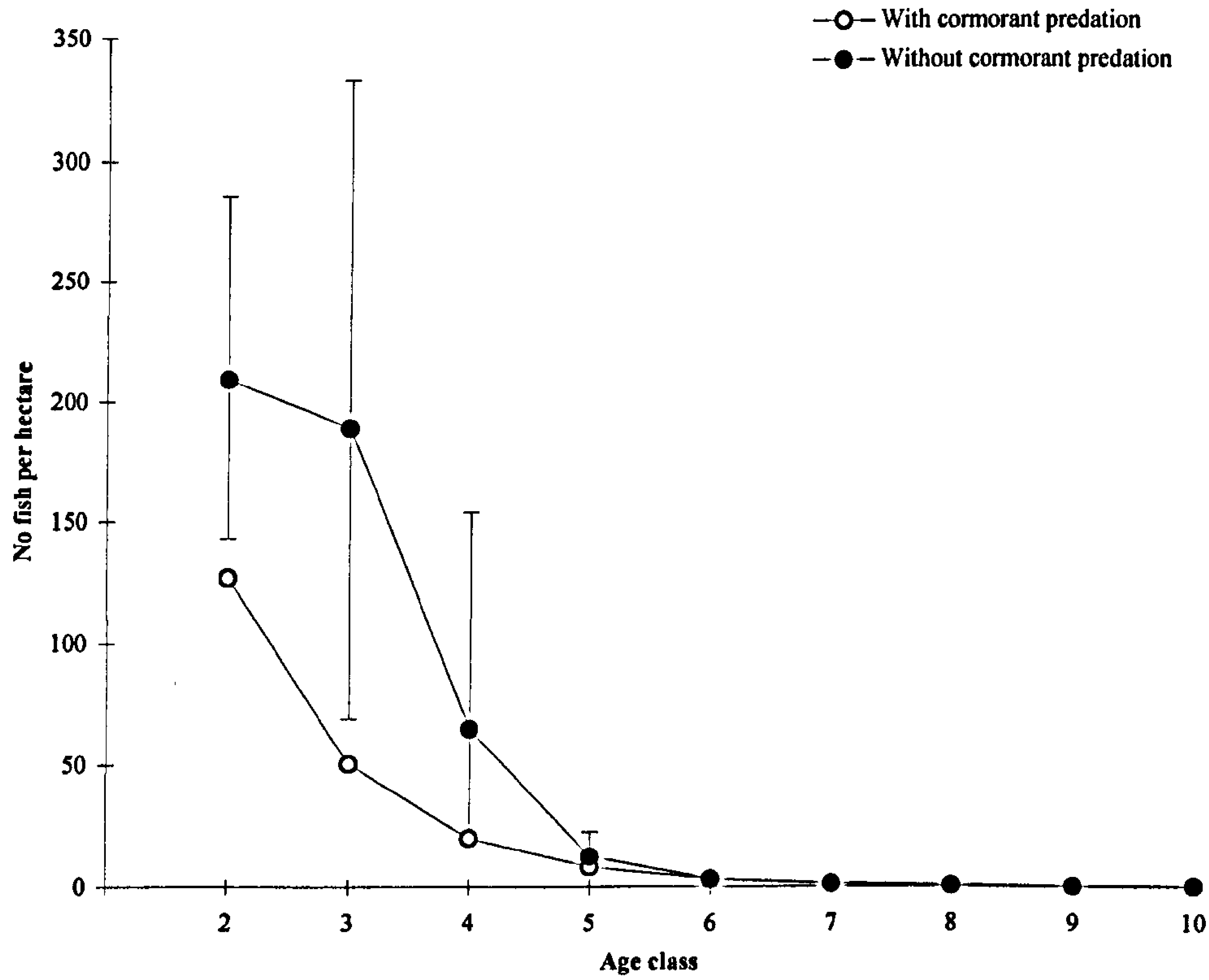


Figure 4.32 Impact of cormorant predation on the numbers of common bream in 1997 at Holme Pierrepont Rowing Course.

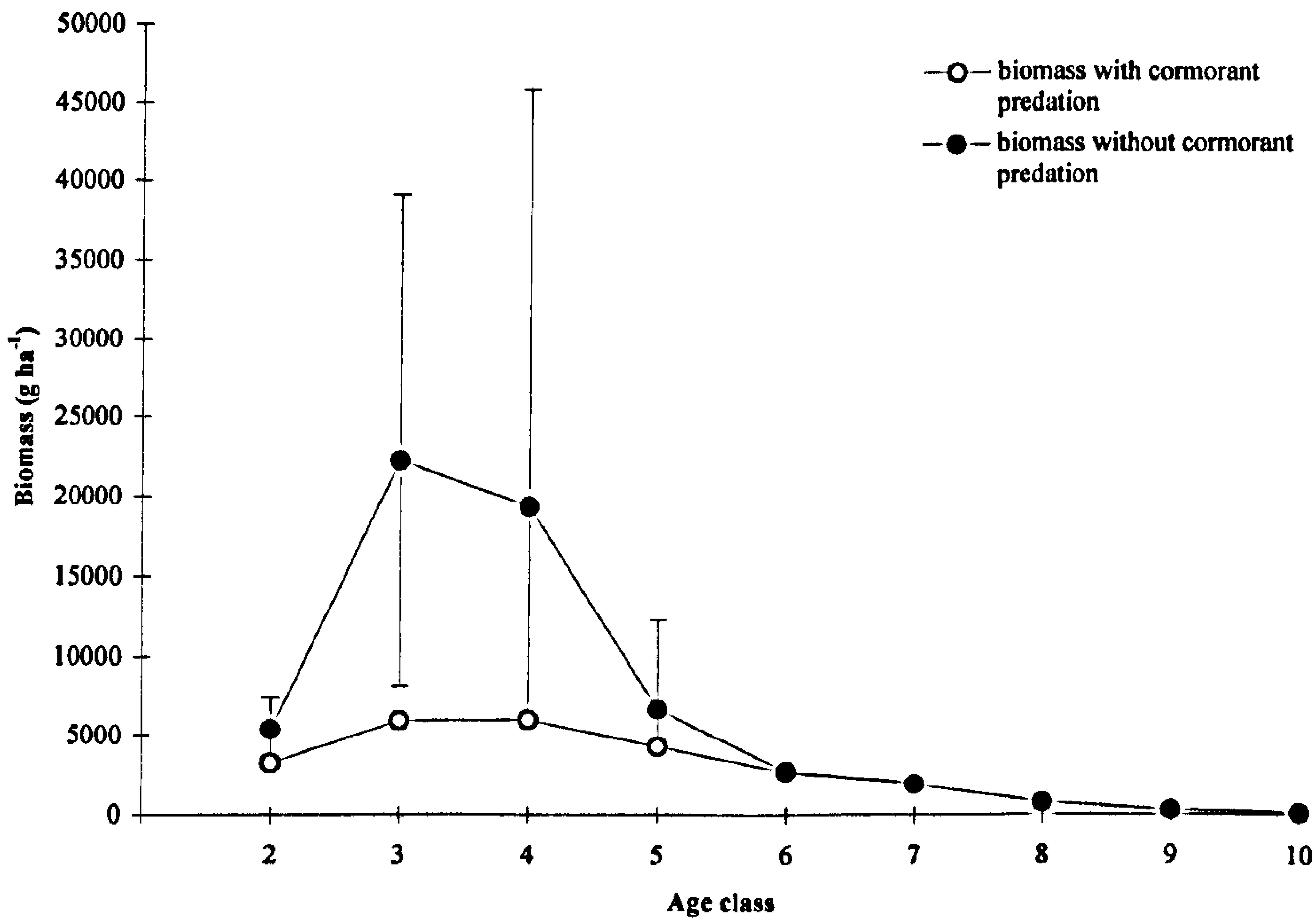


Figure 4.33 Impact of cormorant predation on the biomass of common bream in 1997 at Holme Pierrepont Rowing Course.

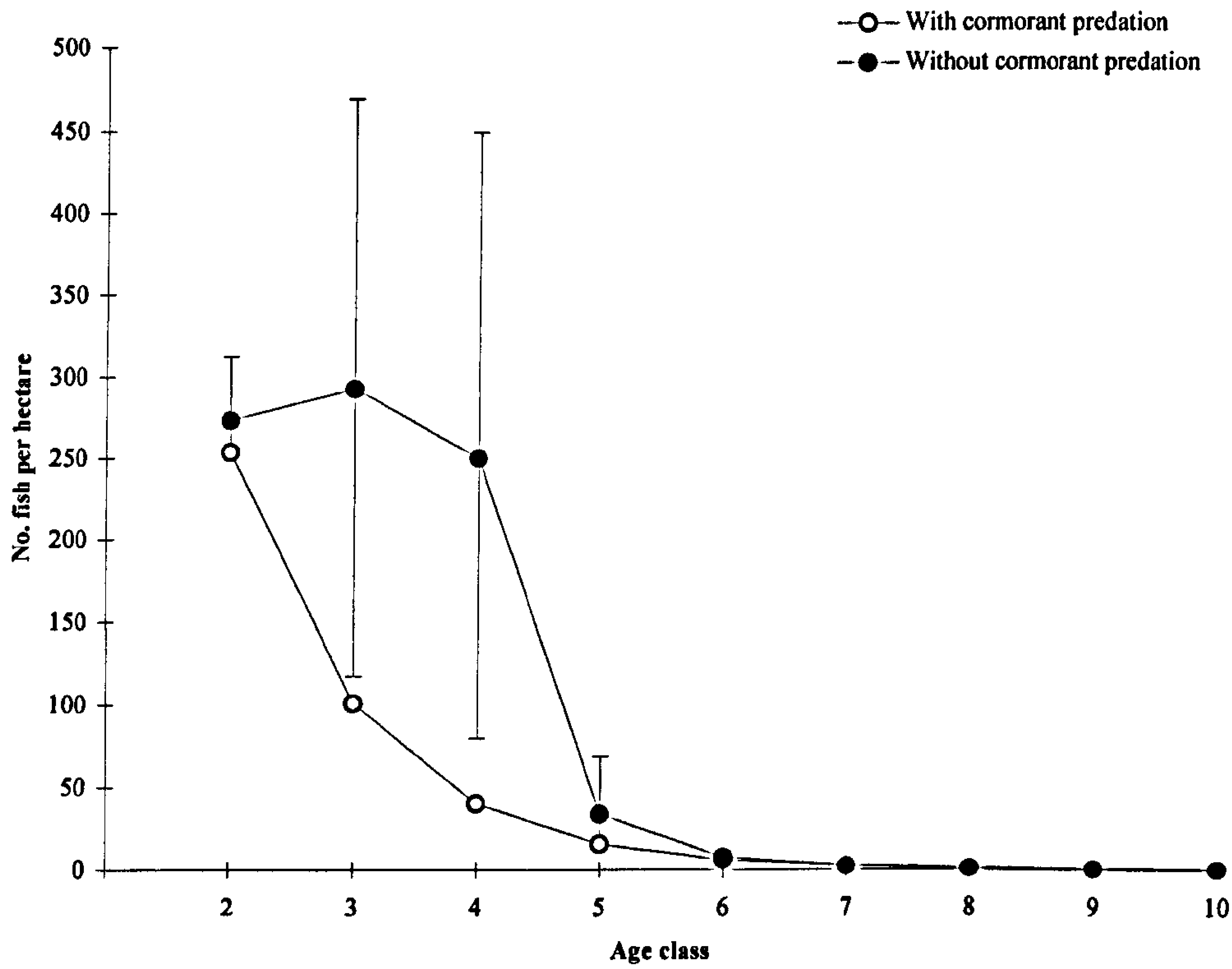


Figure 4.34 Impact of cormorant predation on the numbers of common bream in 1998 at Holme Pierrepont Rowing Course.

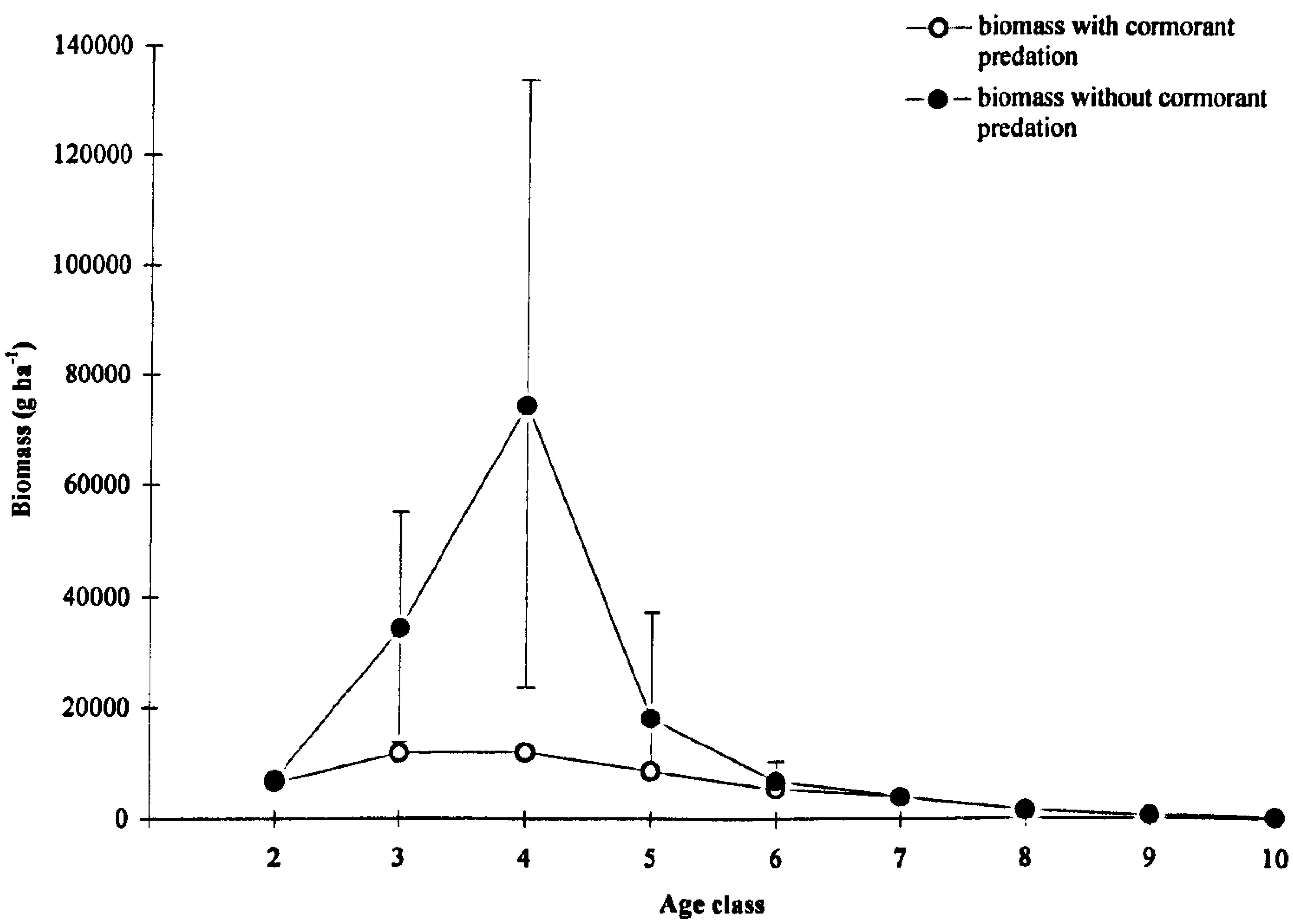


Figure 4.35 Impact of cormorant predation on the biomass of common bream in 1998 at Holme Pierrepont Rowing Course.

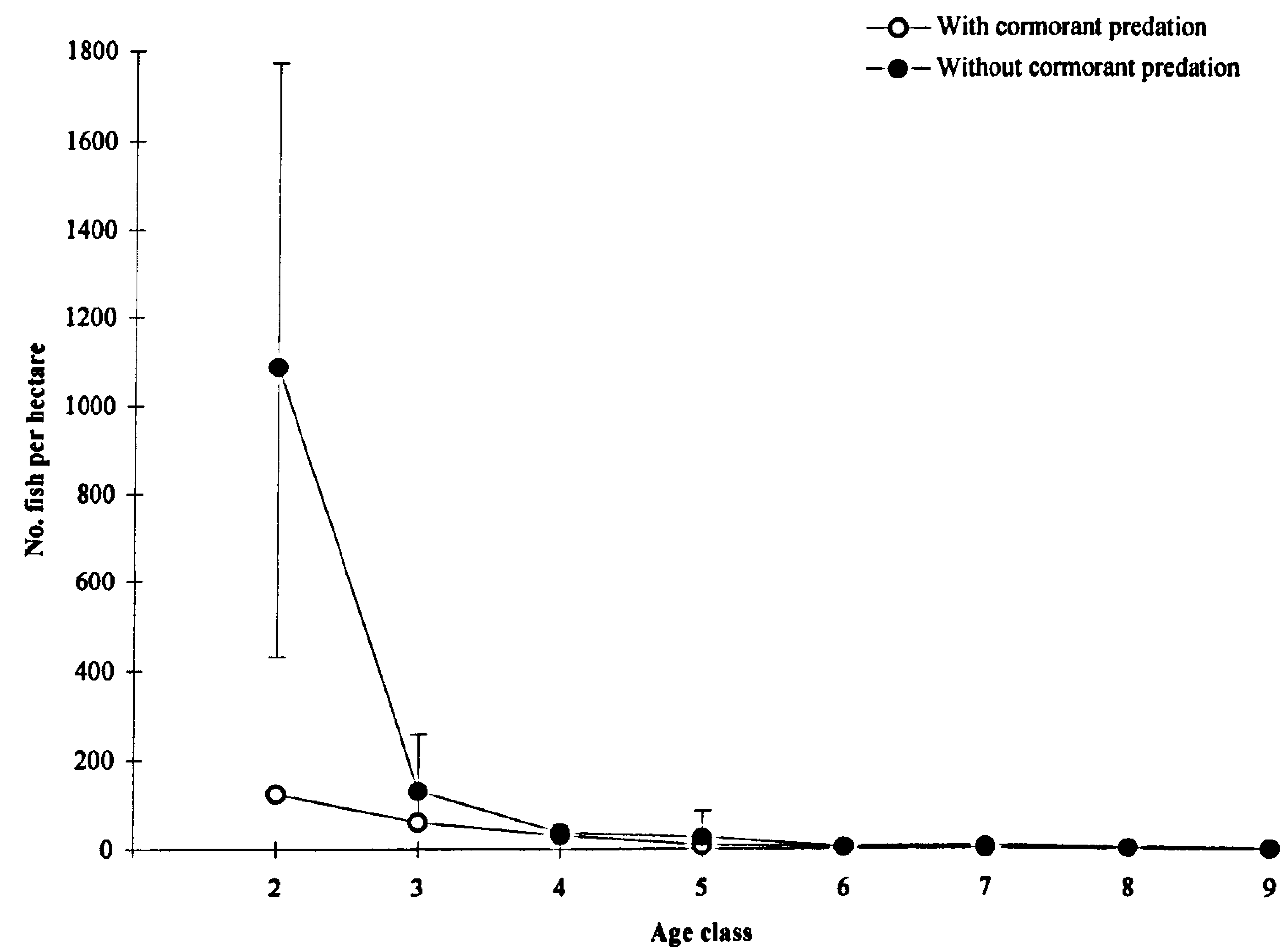


Figure 4.36 Impact of cormorant predation on the numbers of perch in 1996 at Holme Pierrepont Rowing Course.

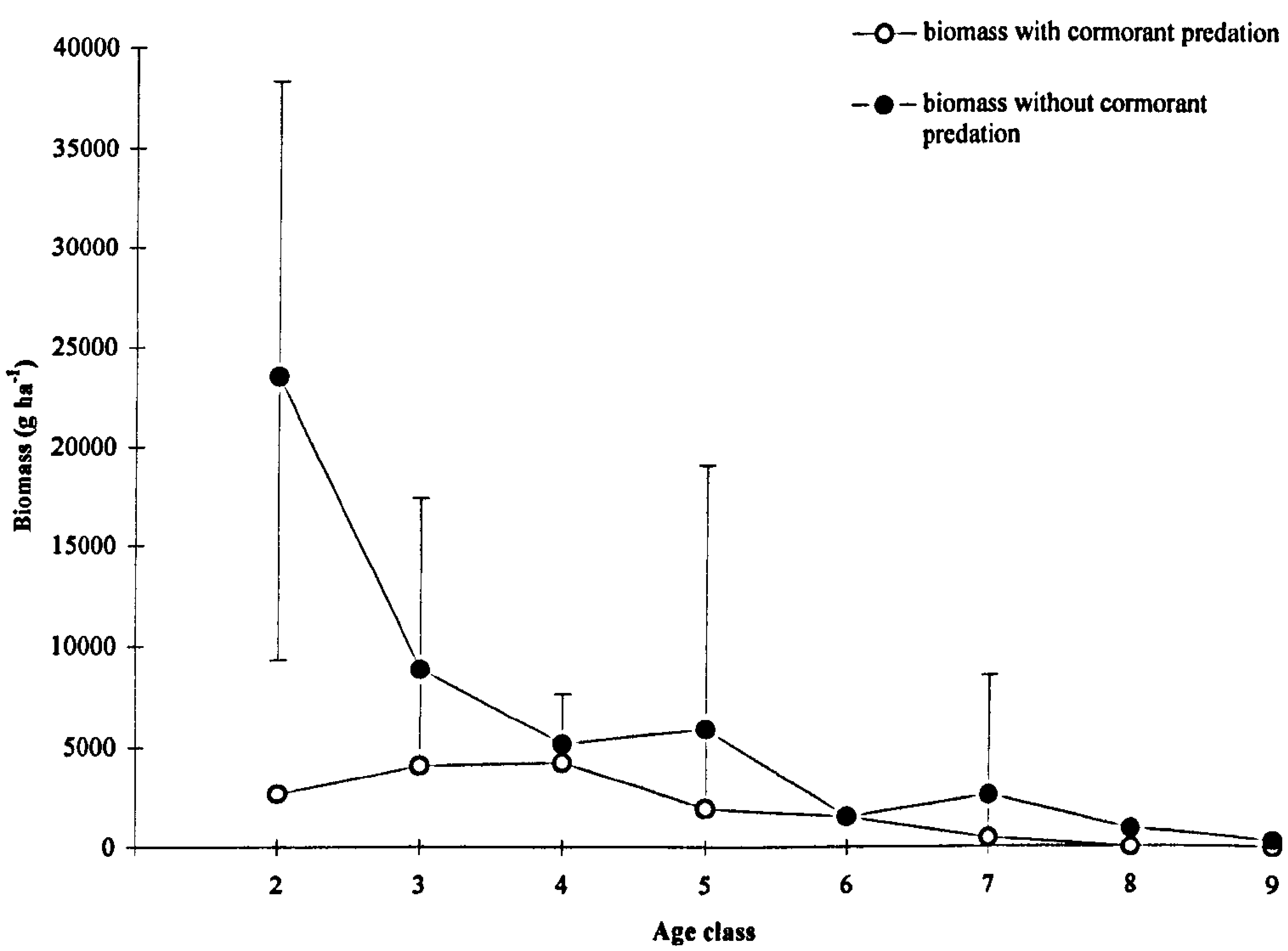


Figure 4.37 Impact of cormorant predation on the biomass of perch in 1996 at Holme Pierrepont Rowing Course.

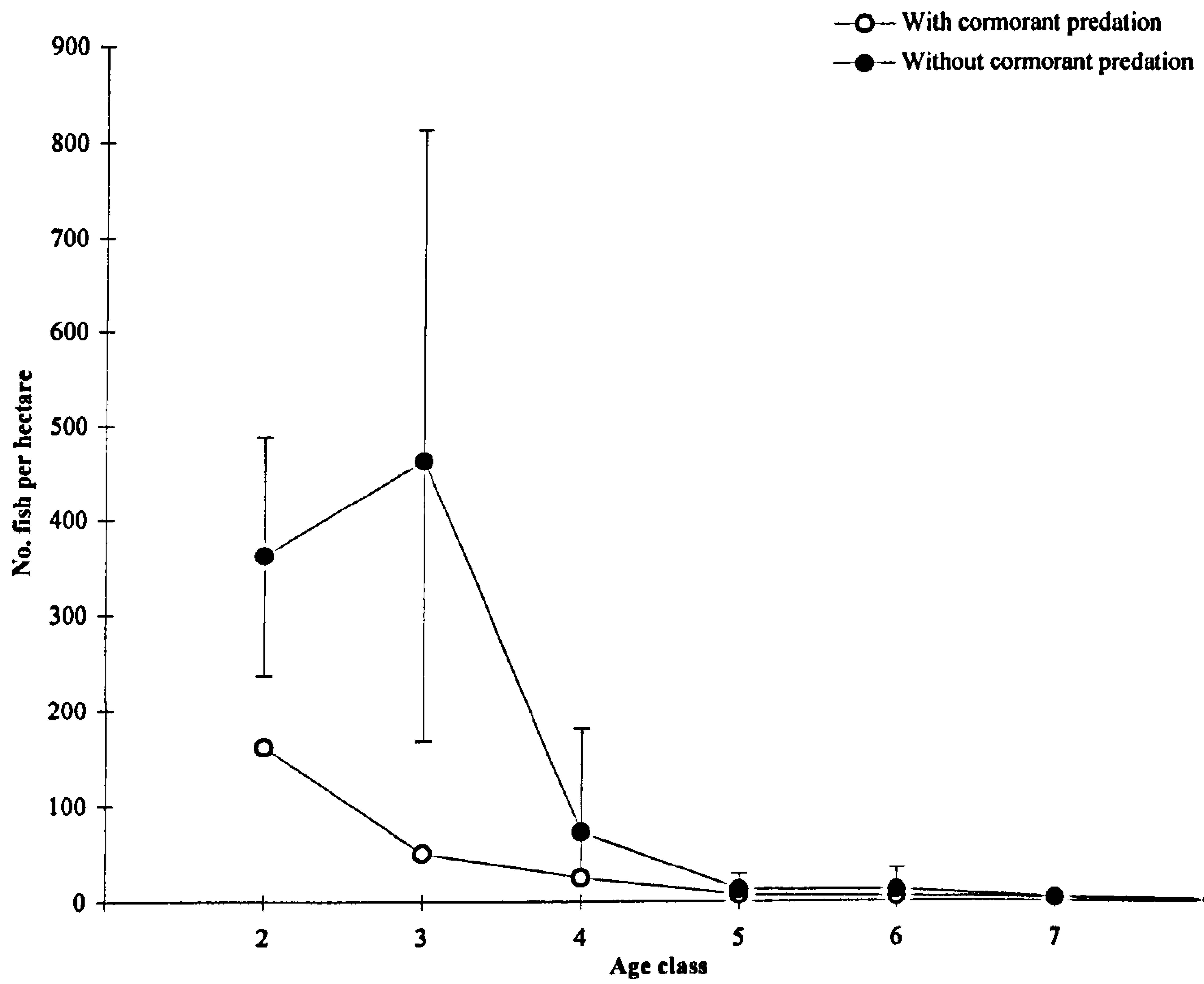


Figure 4.38 Impact of cormorant predation on the numbers of perch in 1997 at Holme Pierrepont Rowing Course.

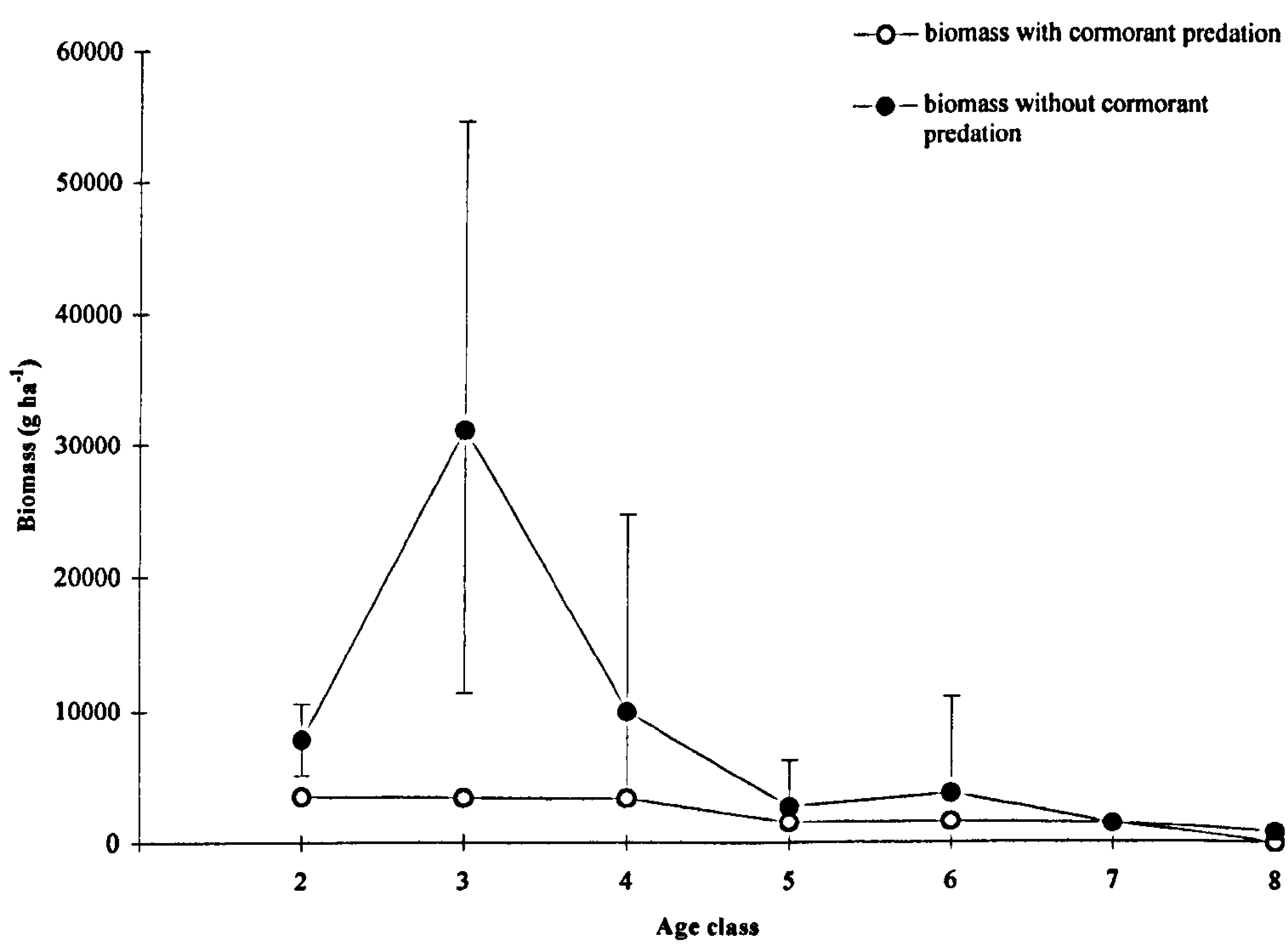


Figure 4.39 Impact of cormorant predation on the biomass of perch in 1997 at Holme Pierrepont Rowing Course.

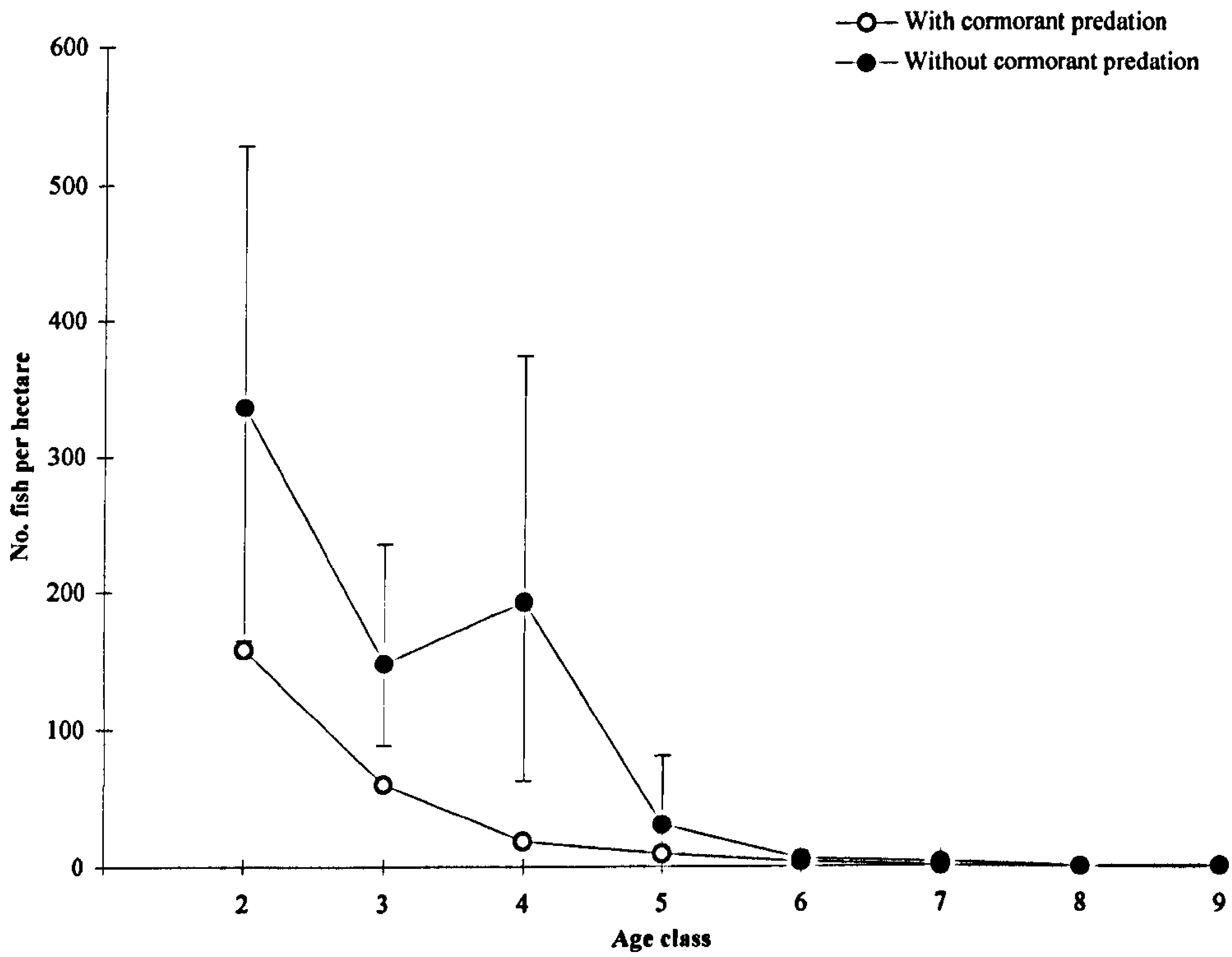


Figure 4.40 Impact of cormorant predation on the numbers of perch in 1998 at Holme Pierrepont Rowing Course.

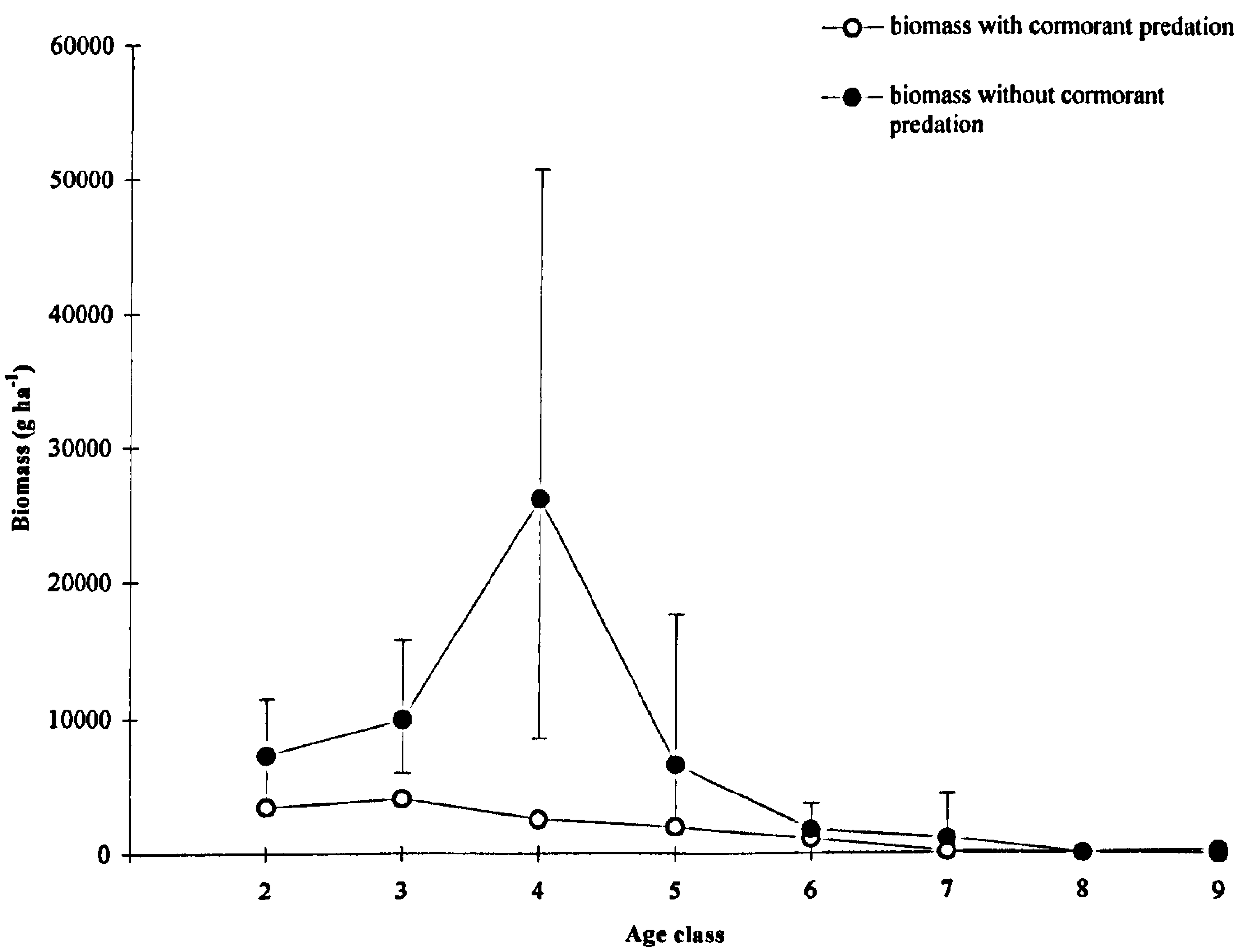


Figure 4.41 Impact of cormorant predation on the biomass of perch in 1998 at Holme Pierrepont Rowing Course.

5. RIVER TRENT, NOTTINGHAM

5.1 Introduction

5.1.1 *General features*

The River Trent rises north of Stoke-on-Trent and flows south-east through Rugeley before adopting a north-easterly direction. The river then flows through the urban conurbations of Burton-on-Trent, Nottingham and Newark before its confluence with the Humber estuary at Trent Falls. The River Trent is approximately 280 km long and has a total catchment area of 10 500 km^2 (Cowx and Broughton 1986).

The River Trent is a popular venue for coarse anglers from the large catchment area of the East Midlands and South Yorkshire. However, concern has been expressed by anglers in recent years with respect to a perceived decline in catches. A number of factors have been cited for the apparent decline, including cormorant predation, water quality improvements and lower winter water temperatures due to the decommissioning of power stations in the Trent valley.

5.1.2 *Water quality*

Water quality in the upper reaches of the Trent is affected by the industrial and domestic effluents of Stoke-on-Trent. Effluent loading in the River Tame, a major tributary of the Trent, from Birmingham and the Black Country causes further pollution downstream of its confluence. Downstream of Burton-on-Trent the River Soar, draining Leicester, and the River Erewash, draining several urban areas of Nottinghamshire and Derbyshire, dilute the River Tame inputs in to the Trent. The River Trent then flows through the city of Nottingham with only two more major towns, Newark and Scunthorpe, affecting water quality before reaching the Humber estuary.

Before the Industrial Revolution, the river supported diverse and prolific fish stocks with salmon and eel fisheries operating. There are numerous reports from the early eighteenth century which highlight the good quality of the river fishery (Jacklin 1996). River water quality declined in the late eighteenth century with the onset of the Industrial Revolution and by 1920 the River Tame was devoid of fish. The decline in water quality peaked in the 1950s with long stretches of the River Trent and its tributaries experiencing low dissolved oxygen and poor aquatic life.

Water quality improvements were initiated by the Rivers (Prevention of Pollution) Act of 1951. In the 1960s improvements were initiated by the diversion of industrial effluents to sewers, better sewage treatment, the introduction of biodegradable detergents and the cessation of coal gasification (Jacklin 1996). In 1980 the River Tame was diverted through a series of purification lakes at Lea Marston, which improved water quality further.

The biological quality of the river, measured as the Biological Monitoring Working Party score (BMWP), has increased significantly since 1980 (Figure 5.1). Measures of organic pollution, such as biochemical oxygen demand (BOD), dissolved oxygen and ammonia, have also improved, although not to the same degree (Figures 5.2 to 5.8). However, this is to be expected due to the nitrification of the River Tame organic pollutants by the time the effluent reaches Nottingham. Lea Marston Lakes have considerably reduced the loading of suspended solids and heavy metal pollution in the Tame (Jacklin 1996). The reduction in suspended solids has also decreased the turbidity of the water.

One further major source of pollution is the discharge from Stoke Bardolph sewage outfall (Section 5.3.4; Plate 5.3), which continues to cause water quality problems. Stoke Bardolph sewage treatment works serve the city of Nottingham. In a 1990 survey, the water quality of the river at Stoke Bardolph was reduced from class 2 to 3, due to increased effluent concentration of ammonia (Jacklin 1996). The ammonia concentration has since been reduced to levels only slightly higher than those found upstream at Nottingham (Figure 5.8). The biological breakdown of this effluent in the river has also resulted in an increased BOD compared with the river at Nottingham (Figure 5.6).

A summary of the water quality improvements are:

- the reduction in organic and toxic pollution since 1960, aided by the construction of Lea Marston Purification Lakes in 1980;
- improvements in biological quality and dissolved oxygen levels with decreases in ammonia, suspended solids and biological oxygen demand;
- improvements in water quality from Stoke Bardolph since 1990.

These long-term water quality improvements have resulted in a shift in the fish community structure from a roach and dace fishery to a chub and bream fishery, to the dissatisfaction of anglers (Cowx 1991).

5.1.3 *Water temperature*

An important change in the river ecology in recent years has resulted from the decommissioning of a number of direct water cooled power stations along the river valley. When operational, the power stations used the river water for cooling during electricity generation and returned the water to the river at an elevated temperature. With decommissioning, the removal of this warm water input has reduced the water temperature in the river. This is most noticeable in winter with river temperature regimes now governed only by climatic conditions. This shift in river temperature has had an impact on angling in general, but winter fishing in particular (Section 5.7.1).

5.2 <u>The study area</u>

The length of the River Trent under study extended from Trent Lock (SK 488312) downstream to Burton Joyce (SK 646340), a distance of approximately 25 km (Figure 3.2). The river flows primarily through pastureland, apart from urban areas centred around Beeston and Nottingham City Centre. For the purposes of the cormorant monitoring the length of river was divided into four separate sections:

- Trent Lock (SK488312) to Attenborough South (SK570330) Section A 6 km;
- Attenborough South (SK570330) to Clifton Bridge(SK562367) Section B 6 km;
- Clifton Bridge (SK562367) to Holme Sluice (SK612392) Section C 7 km;
- Holme Sluice (SK612392) to Burton Joyce (SK646340) Section D 6 km.

The fish populations were investigated in three areas of the River Trent, to integrate with the cormorant population studies. The areas surveyed were influenced by the ease of access for the electric fishing boat, and permission to electric fish by angling clubs and landowners. The three sections assessed for fish populations overlapped the cormorant monitoring sites. For analysis of fish populations the following identifiers were applied to the sections:

- Trent Lock to Clifton Bridge (Cormorant sections A and B) Beeston
- Clifton Bridge to Holme Sluice (Cormorant section C) Trent Bridge
- Holme Sluice to Burton Joyce (Cormorant section D) Stoke Bardolph

5.3 Site details

5.3.1 *Trent Lock (SK 488312) to Attenborough South (SK 570330) (Section A)*

The River Trent from Trent Lock to Attenborough South has a mean width of 60 m and a depth range of 2 to 4 m. The river is slow flowing and navigable to a variety of commercial and pleasure craft. A public footpath, which runs along the left hand bank of the river, is regularly used by walkers, birdwatchers, cyclists and anglers. Angling is possible from both banks of the river with a number of associations controlling the angling rights on the river.

Within the study area a number of habitat features occur. These include Trent Lock Marina, Thrumpton Weir, Cranfleet Lock and Attenborough Gravel Pits. At the eastern end of the study area, on the right hand bank, there are a number of moored house boats and summer houses. The study area finishes at Attenborough South where the River Erewash flows into the River Trent on the left hand side of the river through a regulating sluice gate. The River Erewash flows through several industrial towns in Nottinghamshire and Derbyshire and carries effluent from these areas into the River Trent. During the study, flood defence work was undertaken on the banks surrounding the sluice gate at the confluence of the Erewash.

Within the section the river meanders with exposed stone/gravel banks and shallow bays on both banks of the river. Aquatic vegetation is limited to occasional water lilies and reeds. Bankside vegetation is dominated by grasses, small shrubs and overhanging trees. Grass cutting at angling pegs is undertaken prior to the start of the angling season to improve access. Adjacent land-use is predominantly pastureland with some private houses. Attenborough Gravel pits are located at the eastern end of the study length.

5.3.2 *Attenborough South (SK 570330) to Clifton Bridge (SK 562367) (Section B)*

The River Trent from Attenborough South to Clifton Bridge has a mean width of 60 m and depth range of 1 to 4 m. The river is slow flowing and turbid in nature. The river is non-navigable to large craft from Clifton Bridge upstream to Beeston Weir. Access to the navigable area above Beeston Weir is possible via the Nottingham Beeston Canal link which runs from Trent Bridge, Nottingham to Beeston marina. A public footpath runs along the left hand bank of the river for the whole length of this study reach. On the right hand bank a path runs from Clifton Bridge to Beeston Weir. Angling is possible from both banks of the river, with Nottingham Angling Association and Nottingham and District Federation of Anglers controlling most of the angling rights.

A number of features of note occur in the study length. These include Barton Island (Plate 5.1), Attenborough Nature Reserve, Beeston Marina, Beeston Weir and Clifton Bridge. Barton Island is situated approximately 2 km upstream of Beeston Weir. The right hand channel formed by the island is shallow and not navigable to boat traffic. Attenborough Gravel Pits are situated immediately behind the left hand bank of the river in this section with a regulated outflow from the reserve situated approximately 600 m downstream of Barton Island. The gravel pits are home to Attenborough Nature Reserve where the main cormorant roost for the area is located.

Beeston Marina is located downstream of Attenborough Gravel Pits. A number of mooring stages and boats are situated along the left hand bank of the river for approximately 500 m. The next feature is Beeston Weir. Boat traffic has to navigate the Nottingham Beeston Canal to avoid the shallow nature of the river below the weir. The remainder of the study length of the river is generally uniform with few features of note. At the eastern extremity of the stretch the A52 crosses the river at Clifton Bridge.

In the section, the river meanders with exposed stone/gravel banks and shallow bays on both banks. Aquatic vegetation is limited to occasional water lilies and reeds. Boat moorings are prevalent on the left hand bank near to Beeston Marina with a number of summer houses present on the right hand bank of the river. Bankside vegetation is dominated by grasses, small shrubs and occasional overhanging trees. Adjacent land use on both banks is pastureland, except for Attenborough Gravel Pits and Beeston village where urban and industrial development dominate.

5.3.3 *Clifton Bridge (SK 562367) to Holme Sluice (SK 612392) (Section C)*

The river from Clifton Bridge to Holme Sluice has a mean width of 70 m and depth range of 2 to 5 m. The river is slow flowing, turbid in nature, and navigable from Trent Bridge to Holme Sluices by commercial and pleasure craft. The section flows immediately south of Nottingham City Centre through the urban area of West Bridgford. A public footpath runs along the right hand bank of the river for the whole of the study reach. Angling is possible from both banks of the river and a number of angling associations control the fishing rights on the river. There is an area of free fishing in the centre of Nottingham on the Victoria Embankment.

A number of habitat features are present in this area including the Toll Bridge, Suspension Bridge, Victoria Embankment, Trent Bridge (Plate 5.2), Ladybay Bridge and Colwick Marina. The land use and habitat characteristics vary along the study length. From Clifton Bridge to the Toll Bridge bankside vegetation consists of grasses, small shrubs and the occasional tree. On the left hand bank of the river there is a retaining wall supporting the inner ring road around Nottingham.

Downstream of the Toll Bridge the habitat characteristics of the river corridor alter completely as the river passes through the urban and industrial area surrounding Nottingham. On the right hand bank of the river, from the toll bridge to the suspension bridge, bankside vegetation consists of grasses and occasional trees while downstream of the suspension bridge the bank consists of concrete steps all along to Ladybay Bridge.

On the left hand bank below the toll bridge is the Victoria Embankment. This consists of concrete steps which reinforce the bank. Downstream of the Embankment there is an industrial area from Trent Bridge to Ladybay Bridge. Adjacent to the Trent Bridge area the river is shallow, with depths below 2 m, and the river is non-navigable to large boat traffic. Downstream of Ladybay Bridge the habitat characteristics alter again as the river leaves the urbanised area of the city centre. The right hand bank consists of grasses and occasional trees and land use is pastureland. Holme Pierrepont Water Sports Centre is situated on this bank at Holme Sluices. The left hand bank is used for industrial purposes, including a disused warehouse and dockyard for approximately 500 m , and a section of boat moorings 500 m further downstream. The remainder of the left hand bank of the study length consists of land associated with Colwick Country Park with Colwick Marina situated on the left hand bank of the river, approximately 400 m upstream of Holme Sluices.

The substratum comprises predominantly of silt and mud. Little aquatic vegetation is present apart from occasional areas of reed and water lily. The vegetation is limited at the eastern end of the study area due to the limited shallow margins with depths of 3 to 5 m encountered within three metres of the bank.

5.3.4 *Holme Sluice (SK 612392) to Burton Joyce (SK 646340) (Section D)*

The River Trent from Holme Sluices to Burton Joyce has a mean width of 70 m and depth range of 2 to 4 m. The river is slow flowing, turbid in nature and is navigable from Holme Sluices to Burton Joyce by commercial and pleasure craft. A public footpath, which runs along the left hand bank of the river, is used regularly by walkers, birdwatchers, cyclists and anglers. Angling is possible from both banks of the river with a number of angling clubs controlling fishing rights.

A number of habitat features are present in this section including Holme Sluices, Radcliffe Viaduct, Stoke Weir and Stoke Bardolph Sewage Treatment Works outfall. Holme Sluices are situated at the upstream end of the section and are used to regulate the flow of the River Trent for navigational purposes. Adjacent to Holme Sluices is a navigable lock which is operated by British Waterways and a canoe slalom operated by Holme Pierrepont Water Sports Centre. Radcliffe Viaduct crosses the river approximately 1 km downstream of Holme Sluices supporting a railway line into Nottingham City Centre. The next notable features are Stoke Weir and Stoke Lock, the latter allowing passage of boat traffic past the weir. The final feature of note is the sewage effluent outfall from Stoke Bardolph Sewage Treatment Works, which treats industrial and domestic outputs from the Nottingham area (Plate 5.3, 5.4). The effluent is discharged at the outfall approximately 1 km below Stoke Weir but the effluent does not fully disperse until 1 km further downstream.

Exposed stone/gravel banks and shallow bays are present on both banks of the river. Aquatic vegetation is limited to occasional water lilies and reeds. Bankside vegetation is dominated by grasses, small shrubs with the occasional overhanging tree and land use is predominately pasture.

5.3.5 *Resident fish stocks*

Monitoring of fish stocks has been undertaken on the River Trent for a number of years through angler catch (Cooper and Wheatley 1981; Cowx and Broughton 1986; Cowx 1991) and hydro-acoustic surveys (Lyons 1995, 1996, 1997). The principal fish species are roach, common bream, chub, gudgeon and perch. Other species include pike, barbel, dace, bleak, eels, carp, tench, rudd, minnow, stoneloach, brown trout and salmon. In recent years a population of hybrid fish has emerged in the river and it is unclear as to the parentage of these fish.

5.3.6 *Cormorant populations*

Little historical data are available for the numbers of cormorants occupying the study area. Anecdotal evidence from birdwatchers, anglers and Environment Agency bailiffs suggests cormorants have been present in the area since the early 1990s.

5.3.7 *Angling interests*

The River Trent in Nottingham is a popular venue with coarse anglers and a large number of angling clubs control the angling interests in the study area. The majority of anglers fish for roach, bream, hybrids, chub, dace and gudgeon. A number of specialist anglers target the less abundant species of specimen size, such as carp and barbel. Angling matches are regularly held on the river in the summer and autumn by the controlling angling organisations and by visiting clubs. Recent major angling events include the UK Team Championships, held annually on Victoria Embankment, and the National Championships. Popular areas for angling include the Victoria Embankment, Stoke Bardolph sewage effluent outfall, between Trent Bridge and Ladybay Bridge, and in the vicinity of river features such as weirpools, bridges and boat moorings. Historically, large numbers of winter angling matches were held on the river, although these have not been organised in recent years due to poor winter angling results.

There is a large amount of angling activity on the river with a number of associations financially dependent on its angling performance. If cormorant predation reduces the number of fish available for anglers, an unquantifiable financial loss may be incurred by these associations as anglers visit alternative venues.

5.4 Materials and methods

5.4.1 *Cormorant monitoring*

Cormorant monitoring techniques comprised the fieldwork activities outlined in Section 3.2.

5.4.2 *Fisheries monitoring*

Four fisheries survey techniques, using the gears described in Section 3.3, were used to assess the fish population dynamics in the River Trent, Nottingham. The techniques used were electric fishing (boat-mounted), micro-mesh seine netting, hydro-acoustics and angler catch analysis. Large scale seine netting could not be carried out on the River Trent because of the flow velocity encountered.

Boat mounted electric fishing was undertaken on the three stretches of the River Trent (Section 5.2) at Beeston, Trent Bridge and Stoke Bardolph, which corresponded to the four sections monitored for cormorant occupancy and feeding. Surveys were undertaken in the marginal areas of the river, as fish catches were poor in the middle of the river due to the depths encountered. During each survey the electric fishing boat was operated to cover all of the available marginal habitat.

Micro-mesh seine netting was used to assess the juvenile fish populations in the River Trent, with surveys being undertaken in the shallow, marginal areas of the river.

Hydro-acoustic surveys were undertaken annually on the River Trent by the Environment Agency Midlands Region between 1994 and 1997. The method involved undertaking upstream and downstream transects along three sections of the river at night. The three sections assessed were:

- Thrumpton to Beeston;
- Clifton Bridge to Holme Sluices;
- Stoke Bardolph to Gunthorpe Bridge.

These sections overlapped the sites assessed for cormorant occupancy and fish population assessment. The hydro-acoustic surveys provided additional data on the fisheries of the River Trent in the study area and gave a minimum estimate of the density of fish present, as the results were described in single target volume density (Section 4.2.2).

Angler catch data have been collected by the Environment Agency (Midlands Region), and its predecessors the National Rivers Authority (NRA) and Severn Trent Water Authority (STWA), since 1969, as part of ongoing monitoring of angling results on the River Trent. These data were collected from the River Trent at Stoke Bardolph and have formed the basis of studies by Cowx and Broughton (1986) and Cowx (1991). The data were collected on angler catch return forms compiled by angling clubs which held angling contests on the stretch of river operated by Nottingham Federation and District Angling Society (NFDAS). These data were supplemented by angler catch returns forms from other sections of the River Trent.

5.4.3 *Cormorant data analysis*

The following cormorant data outputs were generated for the River Trent study sections from the cormorant monitoring fieldwork:

- cormorant site occupancy;
- cormorant feeding success;
- MCS generated losses of fish attributable to over-wintering cormorants.

The MCS model for derivation of over-wintering cormorant predation losses was that described in Section 3.4.2, with the exception of the parameters *N.f* (the number of birds feeding at the fishery) and *c* (the cormorant daily food intake at the site).

<u>Estimating *N.f*, the number of birds feeding at River Trent sites</u>

The general MCS model, described in section 3.4.2, used the number of birds observed feeding during counts as N_{min} and used roosting and flying birds to estimate N_{max}. This method was impractical on the River Trent study sections due to the mobility of the observed cormorants and their perceived use of the river as a 'fly-way' to locate adjacent stillwaters (Feltham *et al.* 1999).

Thus, determining the appropriate number of flying and roosting cormorants observed during a count that would have fed on a particular river section became complicated and river site specific. By use of the detailed cormorant observations, it was possible to establish the approximate cormorant use at the four cormorant monitoring sections.

Sections B and C were noted for their use primarily as 'fly-ways' for cormorants. The observed flying cormorants mainly used the river to locate the adjacent stillwaters, for example, Holme Pierrepont Rowing Course (Chapter 4; Feltham *et al.* 1999). The assumption used on these sections was 90 % of all flying and roosting cormorants were feeding / had fed at other sites (Feltham *et al.* 1999).

For cormorants observed flying and roosting in Sections A and D, the pattern was different. If those cormorants had not fed at the site, then they would either fly to other sections of the river outside of the study area or to Sections B and C for feeding (Feltham *et al.* 1999). Due to the lower number of observed feeding cormorants on the river compared with the adjacent stillwaters, the assumption used for Sections A and D was 50 % of flying and roosting cormorants fed, or were likely to feed, in those count sections, with 50 % likely to feed elsewhere (Feltham *et al.* 1999). Therefore, the values for N_{min} and N_{max} were derived as:

N_{min} = the number of birds counted actually observed feeding = $N.f_{min}$;

$N_{max} = N.f_{max}$ = (1.0 x feeding birds) + (a x roosting birds) + (b x flying birds);

where a and b are the section specific constants derived for the proportions of flying and roosting birds utilising the sites for feeding (Feltham *et al.* 1999).

Estimating c - the cormorant daily food intake

The general MCS model utilised a uniform distribution of c lying between values of c_{min} and c_{max} from DFI estimates based on energy considerations (Section 3.4.2). This assumes all birds which fed at the fishery met their full DFI there.

On the River Trent study sections this assumption could not be justified due to the likelihood of cormorants feeding on alternative and adjacent sites. Hence, the described methodology to determine c_{min} and c_{max} in Section 3.4.2 was not appropriate (Feltham *et al.* 1999). As with Holme Pierrepont (Section 4), the proportion of DFI at the site is not a constant and will often be below 1.0 (Feltham *et al.* 1999).

The value for c_{max} had to be determined from the literature (Section 3.4.2), as there was no method of generating an independent estimate for c_{max} (Feltham *et al.* 1999). Thus, to determine c_{max}, a uniform distribution was sampled between the lower (400 g d^{-1}) and upper (800 g d^{-1}) theoretical estimates (Feltham *et al.* 1999) (Section 3.4.2).

The value for c_{min} was determined from an empirically-derived distribution of observed fish intake. For each month, each prey item captured was identified to cyprinid / perch / eel level, and the size estimated by comparing bill length to fish length (Section 3.2.2). The mass of each fish consumed was then determined by converting the estimated fish length to weight by use of length-weight equations (Section 3.5.6), before totalling the mass for each species group. Cyprinid fish mass was then converted to cyprinid species level using their representative value in the fisheries surveys. Any incomplete bouts were recorded and treated identically to those at Holme Pierrepont (Section 4.3.1).

As the numbers of birds feeding at the fishery each month (*N.f*) and their DFI (*c*), with respect to species and size (and, hence, age group) were now determined by the site specific methodologies, the values were substituted into the MCS model as described in Section 3.4.2.

5.4.4 *Fisheries data analysis*

The following outputs, using the methodologies described in Section 3.5, were generated for the River Trent from fisheries surveys:

- standing crop;
- species composition;
- length frequency distributions;
- age and growth determination of fish species;
- growth indices and models;
- mortality rates, survival rates and year class strengths;
- cohort reconstruction;
- angler catch performance.

5.4.5 *Assessment of the impact of cormorant predation on the River Trent*

The following data were used to assess cormorant predation impact:

- comparison of species composition from cormorant predation observations, electric fishing and angler catches. This shows species selectivity by cormorants and anglers relative to electric fishing;

- comparison of length frequency distribution of fish predated on by cormorants, exploited by anglers and caught by electric fishing. This will show the size selectivity by species of cormorants and anglers compared with population structure;

- comparison of the number and biomass of fish in the sections with (electric fishing results) and without (electric fishing results plus MCS generated cormorant predation losses) cormorant predation.

Plate 5.1 River Trent at Beeston, viewed from downstream of Barton Island, looking upstream.

Plate 5.2 River Trent at Trent Bridge, located within the city of Nottingham.

Plate 5.3 Stoke Bardolph sewage treatment works outfall, River Trent, viewed from upstream.

Plate 5.4 Stoke Bardolph sewage treatment works outfall, River Trent, viewed from further upstream.

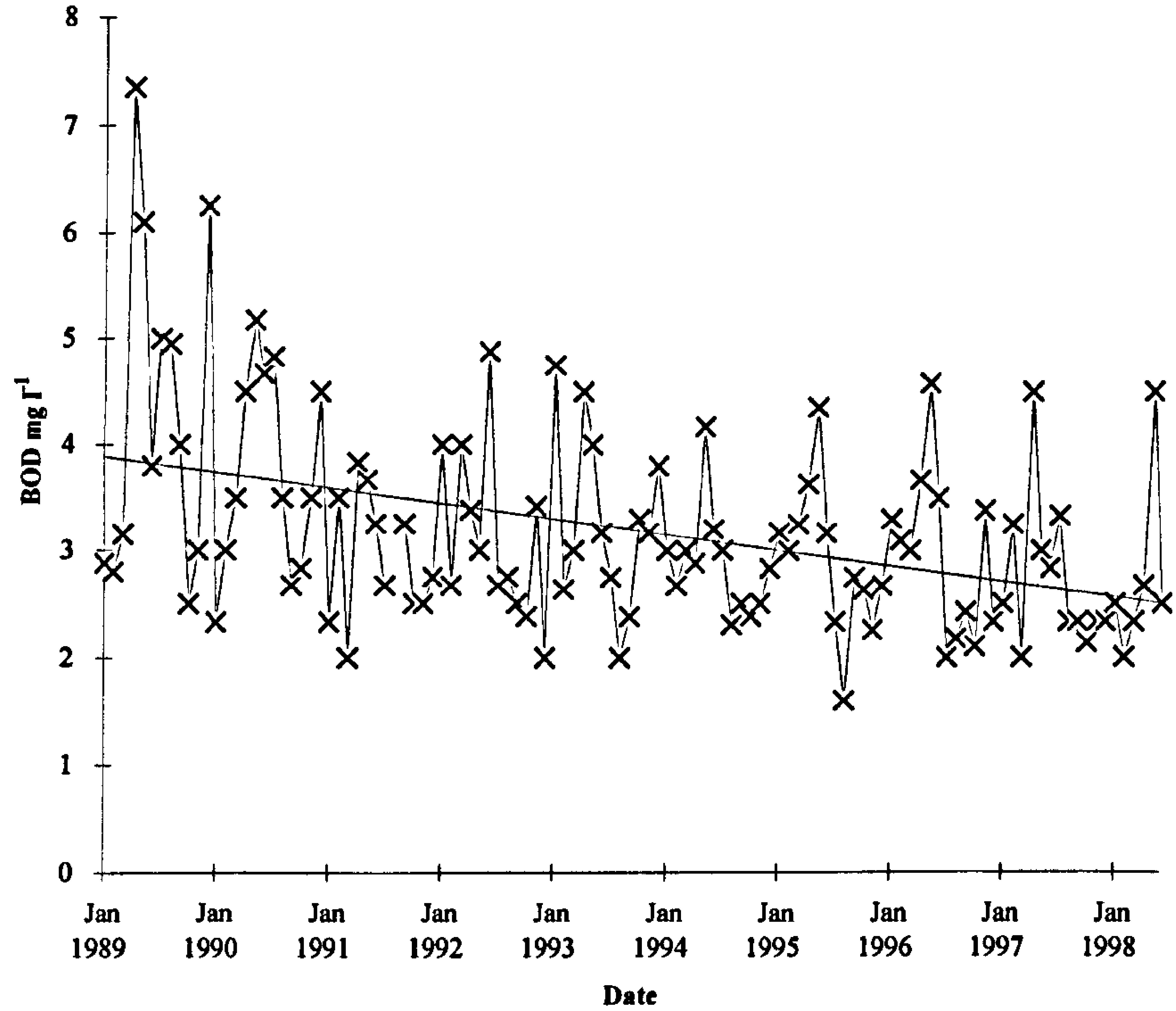


Figure 5.1 Mean biochemical oxygen demand in the River Trent at Trent Bridge, 1989 to 1998.

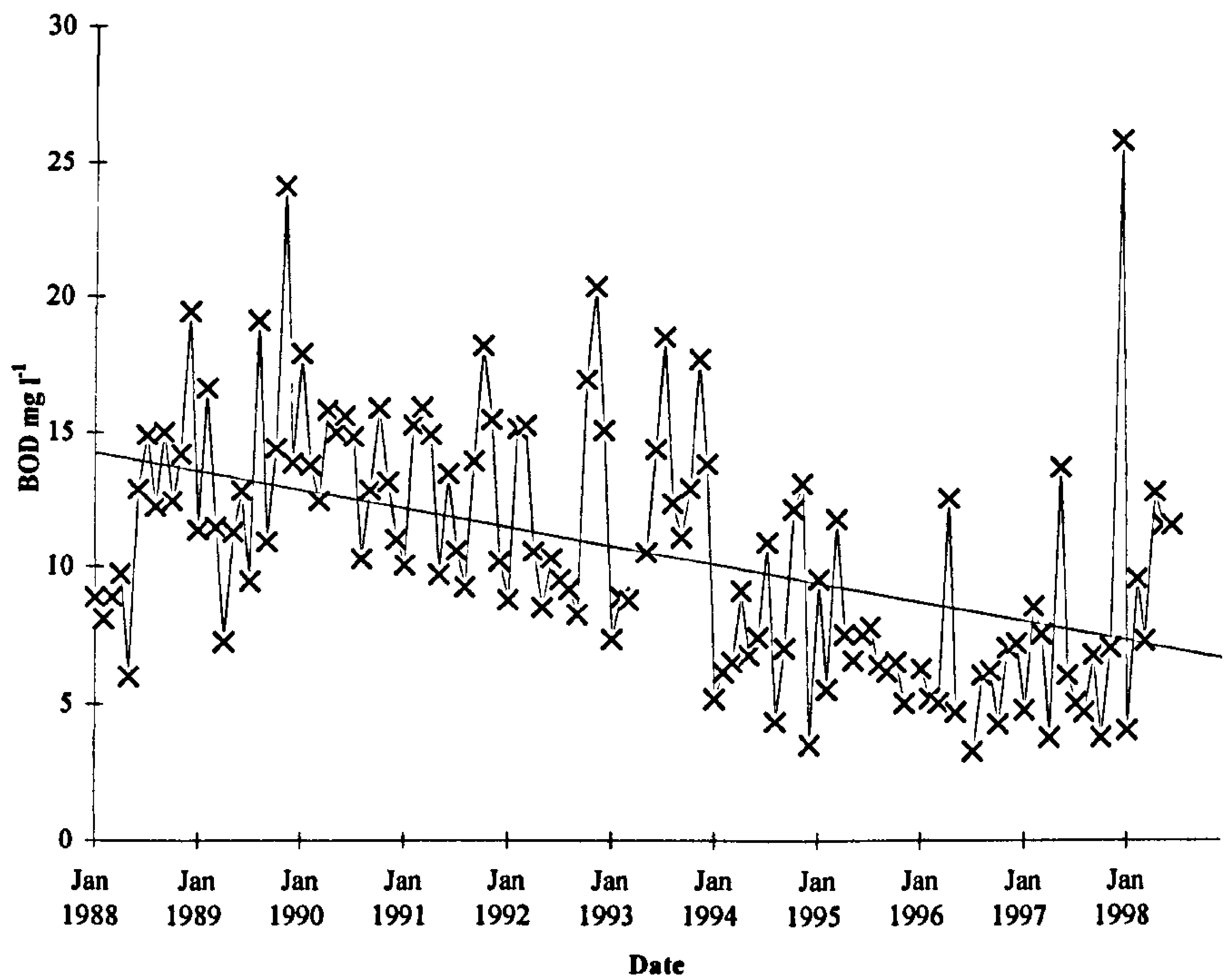


Figure 5.2 Mean biochemical oxygen demand in the River Trent at Stoke Bardolph, 1988 to 1998.

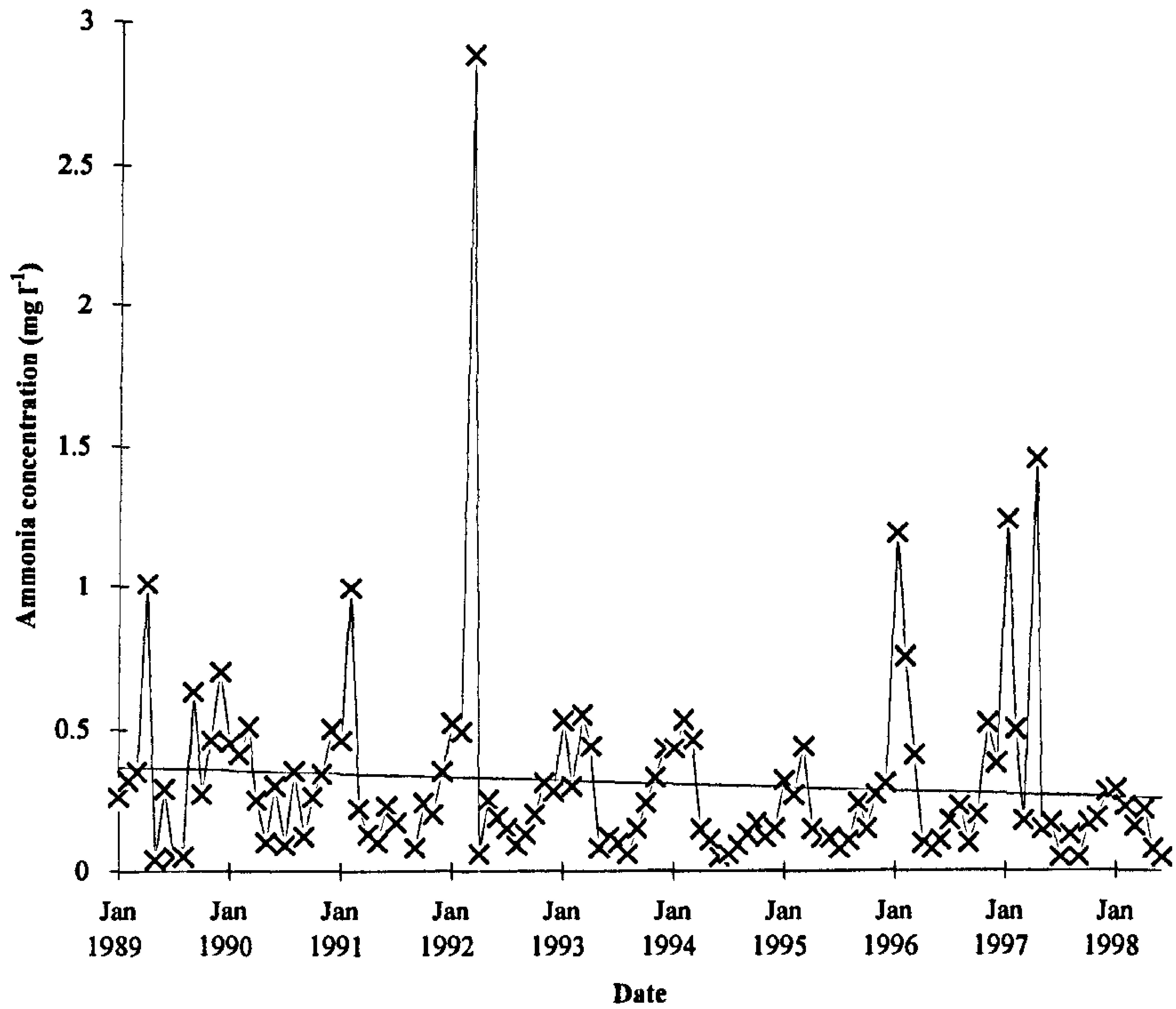


Figure 5.3 Mean ammonia concentration in the River Trent at Trent Bridge, 1989 to 1998.

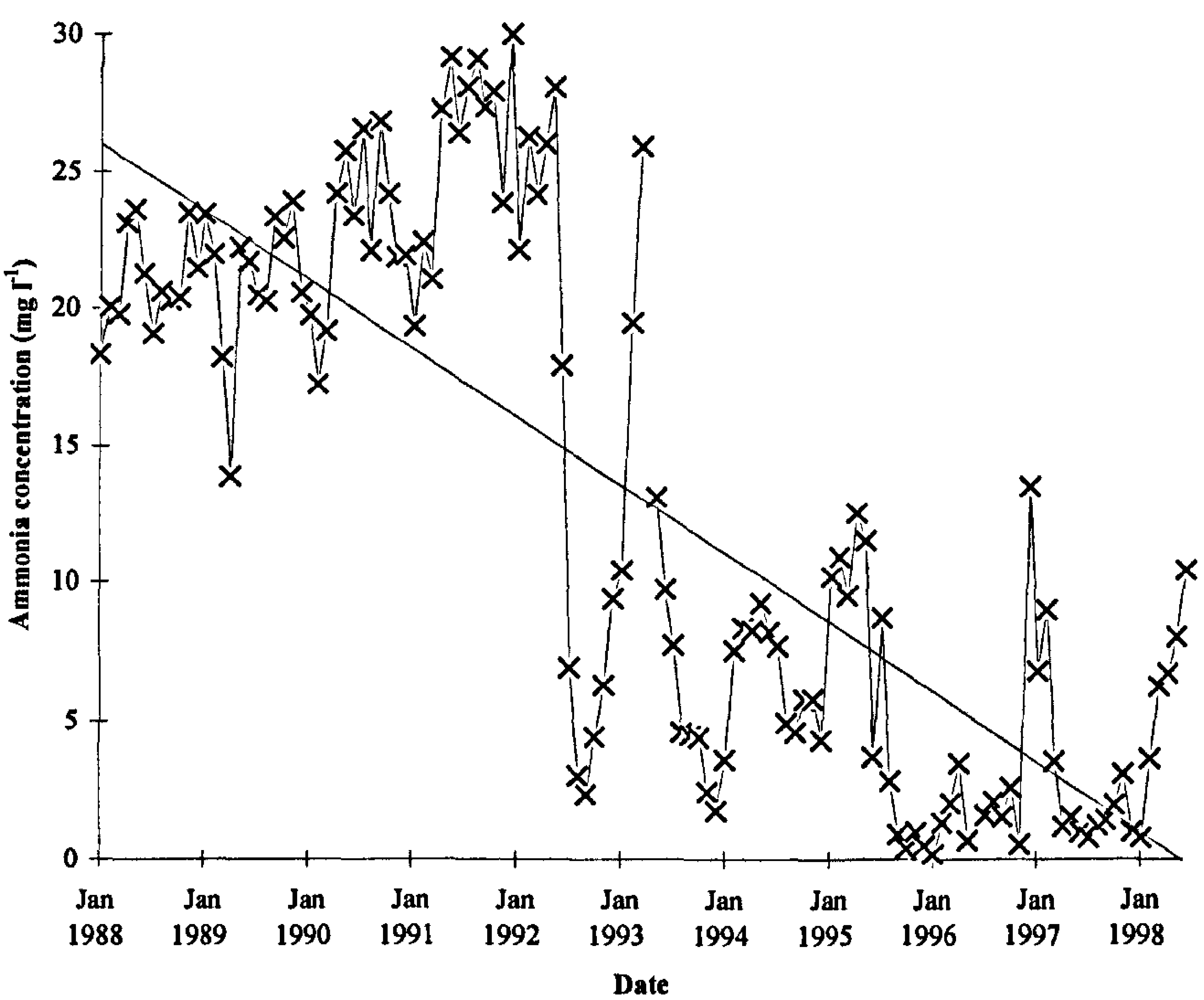


Figure 5.4 Mean ammonia concentration in the River Trent at Stoke Bardolph, 1988 to 1998.

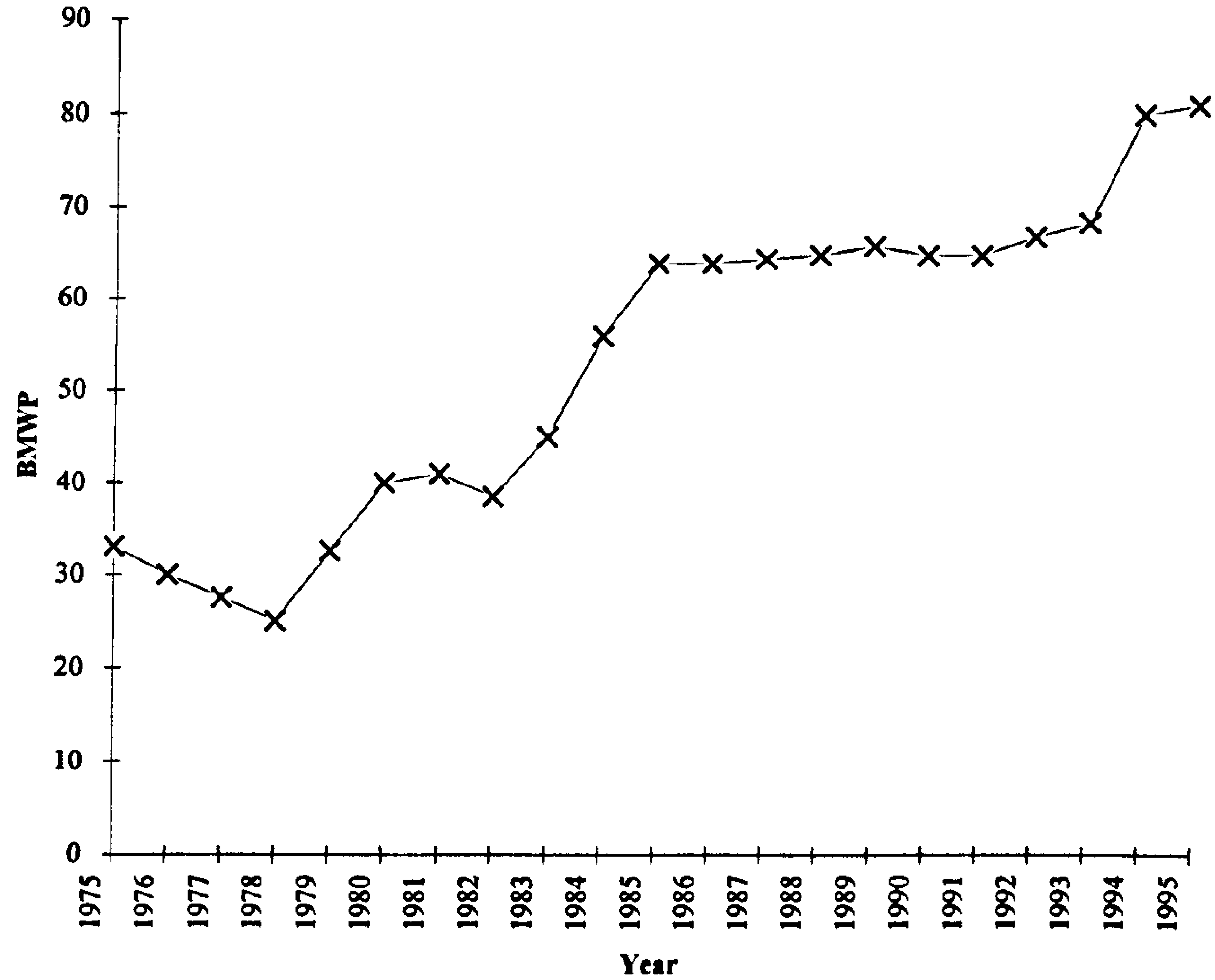


Figure 5.5 Biological water quality (BMWP score) in the River Trent at Nottingham, 1975 to 1995 (Jacklin 1996).

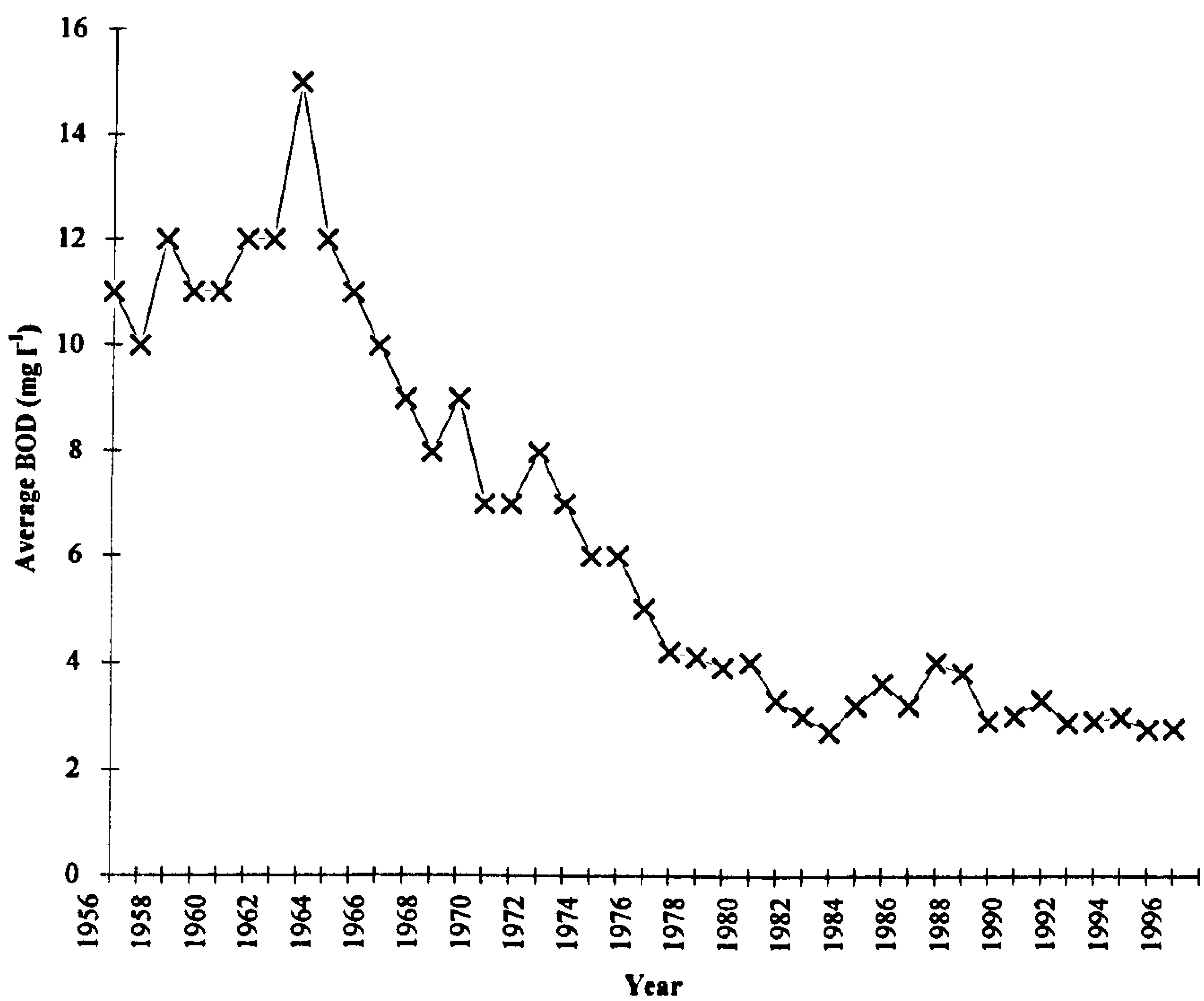


Figure 5.6 Biochemical oxygen demand in the River Trent at Nottingham, 1957 to 1998 (Jacklin 1996).

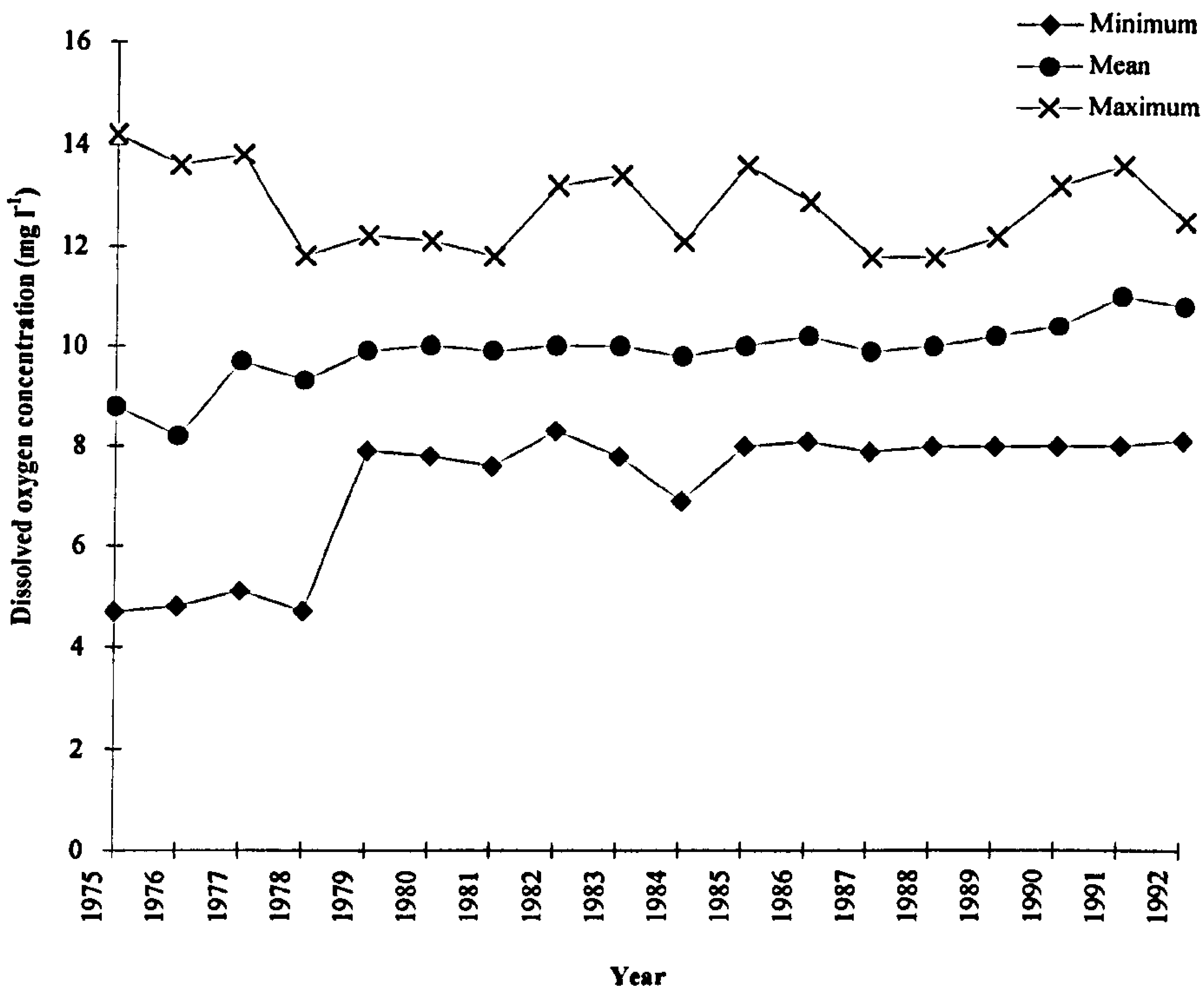


Figure 5.7 Dissolved oxygen levels in the River Trent at Nottingham, 1975 to 1992 (Jacklin 1996).

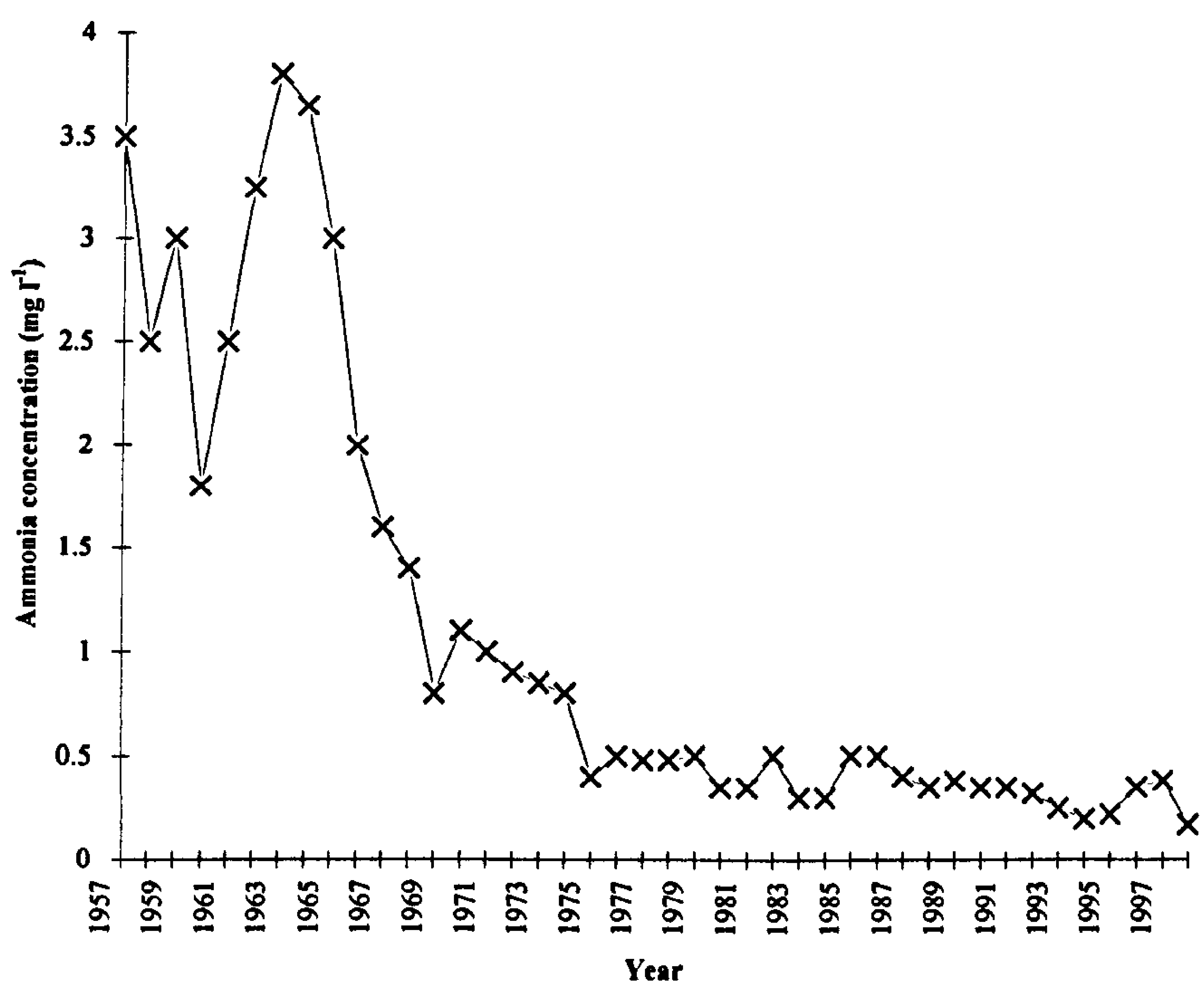


Figure 5.8 Ammonia concentration in the River Trent at Nottingham, 1957 to 1993 (Jacklin 1996).

5.5 Status of fish populations, cormorant populations and the impact of cormorants on the River Trent at Beeston

5.5.1 *Status of fish populations*

Standing crop

The standing crop of the main fish species in the Beeston study area varied from 100.4 g m^{-2} to 8.5 g m^{-2} (October 1995 to June 1998) (Table 5.1).

Table 5.1 Variation in standing crop of the main fish species, estimated by electric fishing, over the study period at Beeston, River Trent.

	Standing crop (kg ha^{-1})		
Species	**1995/96**	**1996/97**	**1997/98**
Roach	161.72	88.67	68.98
Common bream	753.08	5.67	0
Chub	14.70	90.58	0
Hybrid	22.25	3.07	3.07
Perch	52.75	79.12	13.22
Total standing crop	1004.49	267.10	85.27
Total standing crop g m^{-2}	100.4	26.7	8.5

Environment Agency hydro-acoustic data allowed an estimate of fish density in the section (Lyons 1997). Although the hydro-acoustic data provided only a minimum estimate of fish abundance in the section (Section 4.2.2), the data also showed fish abundance changed with time (Table 5.2).

Table 5.2 Density of fish, estimated by hydro-acoustics, present in the Thrumpton to Beeston Lock section of the River Trent.

Year	Fish per hectare (n ha^{-1})
1994	175
1995 (June)	762.5
1995 (Sept)	172.2
1996	69.5
1997	132.2

Angler catch data

Angler catch data for the fishing season 1996/97 were available for analysis from matches held at Thrumpton, a stretch approximately 1 km upstream of Beeston. A catch per unit effort of 327 g man $hour^{-1}$ (Total effort = 3619 man hours) was recorded, which compared favourably with Stoke Bardolph further downstream (Jacklin 1996; range 60 to 231 g man $hour^{-1}$) and the River Ouse, Yorkshire (Axford 1991; range 37 to 91g man $hour^{-1}$).

The percentage of anglers with catch at Thrumpton was 94 %. This is comparable with data from Jacklin (1996), where 45 to 80 % of anglers caught fish, and Axford (1991), where 56 to 81 % of anglers caught fish. Species composition of angler catches was dominated by roach (69 %).

Species composition of fish populations

Electric fishing catches in 1995/96 were dominated by perch (Figure 5.9), but roach dominated in 1996/97 and 1997/98 (65 % and 63 % respectively). It is likely the low numbers of roach in 1995/96 and June 1998 were probably due to poor electric fishing efficiency at the time of sampling because of very clear water. This is likely to cause movement of roach into deeper and darker areas of the river, away from the effective range of the electric fishing gear. This phenomenon occurred on other sections of the River Trent and at Holme Pierrepont where catches improved in turbid water (pers. obs.).

Piscivorous fish comprised a major component of catches between October 1995 and March 1996, perch comprising 42 % of catches and pike 14 % (Figure 5.9). However, this dominance was not observed in later surveys, possibly indicating the movement of predators into the marginal area during the 1995/96 survey.

The other major angling cyprinids, common bream, chub and hybrids, were all present throughout the study period. However, their contribution to catches was generally low, less than 10 % for each species, in all study years (Figure 5.9). Other minor fish species caught by electric fishing were dace, tench and eel.

The minor angling cyprinid species, gudgeon and bleak, comprised a significant proportion of catches (combined total of 80 %) in June 1998 (Figure 5.9). This may be partly due to the poor catches of other cyprinids in the survey. Species composition for both species varied between 3 and 10 % in other surveys.

Length frequency distribution

The length frequency distribution of roach revealed a population dominated by fish in the size range 90 to 190 mm, with the largest roach caught being 290 mm (Figure 5.10). A significant proportion of roach below 100 mm were present in all years showing good recruitment.

The length frequency distribution of common bream revealed few individuals in the size range 100 to 350 mm (Figure 5.11). This may be due to the difficulty of catching bream as they generally inhabit deep and slow flowing areas of the river where electric fishing is inefficient. Catches were either dominated by bream below 100 mm or above 350 mm. The number of fish below 100 mm indicated a successful spawning population of bream and good fry survival. The individuals above 350 mm indicated good survival, capacity for growth and a mature spawning stock.

Only low numbers of chub in the size range 60 to 450 mm were found (Figure 5.12). The results suggests chub recruit annually with a mature spawning stock present with the potential to attain large individual sizes. The decline of chub in the River Trent in recent years has been well documented (Jacklin 1996), and has been related to dominant year classes dying out through natural mortality and not being replaced by strong year classes in recent years. This has also been observed on the Warwickshire Avon, where the dominant 1969 year class of chub resulted in good angler catches into the 1980s (Starkie 1993). Their decline in numbers through natural mortality has yet to be fully replaced by recent, strong chub year classes in the river.

The hybrid population was dominated by individuals in the size class 100 to 190 mm, with the largest fish caught being 220 mm (Figure 5.13). Few fish were observed below 100 mm, possibly due to electric fishing sampling bias. The results showed a recruiting population of hybrids was present in the stretch.

Most perch caught were in the size range 100 to 200 mm (Figure 5.14) although individuals up to 400 mm were captured. The results show successful juvenile recruitment and a spawning stock of older fish present.

The pike population was represented mainly by individuals in the size range 330 to 850 mm (Figure 5.15). The data highlighted a mature population of pike in the study area which were able to reach relative large sizes. The lack of pike below 330 mm may be due the areas sampled by electric fishing not being used by juvenile pike or considerable cannibalism by the larger pike regulating the numbers of small pike.

Year class strength

Only the year class strength for roach was calculated as insufficient data were available for other species. Strong year class strengths of roach occurred in 1990, 1991, 1992 and 1995 (Figure 5.16), which contribute to the good angler catches at the present time. Weak year classes were observed in 1993 and 1994 (Figure 5.16). Analysis of the temperature profile of the River Trent over the period, expressed as annual number of degree days above 14°C, show strong correlation between year class strength and temperature ($r = 0.91$).

Growth

The growth rate of roach was average (Figure 5.17) when compared with national standards (Hickley and Dexter 1979) (Cowx *et al.* 1995), with the maximum age attained 8+. The roach growth rate was below that at Holme Pierrepont (Section 4.4.6), but above that at Grimsargh number 3 Reservoir (Section 7.5.7). Roach scales from the sites are compared on Figure 4.11. Common bream growth rates were above the national standard (Hickley and Dexter 1979) and comparable with waters where bream growth is considered good (Cowx *et al.* 1995) (Figure 5.18). The maximum age attained by common bream was 12 years, indicating a long life span.

Perch growth rates were also above average when compared with other waters (Figure 5.19). Perch up to 6 years were caught during electric fishing surveys. A small number of older perch were present in catches but scales were either replacement or too unclear to enable accurate reading. The growth rate of chub was average when compared with other fisheries (Figure 5.20), with fish up to 13 years present.

Standard growth rates of hybrids, bleak and gudgeon were not available, so they are compared with other river systems where growth rates have been determined (Table 5.3).

Growth rates of bleak and gudgeon were similar to those from other rivers (Table 5.3). However, the longevity of bleak and gudgeon at Beeston, River Trent, were lower than those found in other UK water bodies. The hybrid species captured at Beeston was similar in appearance to the hybrids captured at other River Trent sites, although some confusion exists over their exact parentage. Hence, comparison with growth rates of other hybridised populations was not possible. Overall, the hybrid population achieved slow growth (Table 5.3).

Table 5.3 Growth rate of bleak, gudgeon and hybrids at Beeston, River Trent, compared with other UK rivers.

		Length at age (mm)							
Species	**Location**	**I**	**II**	**III**	**IV**	**V**	**VI**	**VII**	**VIII**
Bleak	Beeston, R. Trent	46	81	118					
	Standard growth	42	76	107	120	125	147		
	River Severn	41	75	104	120	127			
	River Thames	39	70	92	111	121	130	138	140
Gudgeon	Beeston, R. Trent	55	85						
	River Thames	55	86	104	117	123	129		
	River Frome	42	96	120	137	152			
Hybrid	Beeston, R. Trent	62	95	114	132	153	172	195	

(Williams 1967; Mann 1973; Cowx *et al.* 1995).

L infinity and K values

Values obtained from the Von Bertalanffy growth model for roach, common bream, perch and chub (Table 5.4) were similar to values derived for the species in other UK fisheries (Tables 4.4 to 4.6).

Table 5.4 L_∞, and K values derived for roach, common bream, perch and chub at Beeston, River Trent.

Species	L_∞	K
Roach	341	0.16
Common bream	545	0.15
Perch	432	0.13
Chub	660	0.10

Mortality and survival rates

The mortality and survival rates derived for various species at Beeston, River Trent, were similar to other UK fisheries (Table 5.5), but lower than those for Holme Pierrepont (Section 4.4.8).

5.5.2 *Status of cormorant populations at Beeston*

The status of the cormorant populations at Beeston was assessed by monitoring two sections, A and B (Section 5.2), which covered the areas used in the Beeston fisheries surveys.

Temporal night roost occupancy

The night roost occupied by the cormorants utilising the Beeston section was Attenborough (Section 4.5.1, Figure 4.13).

Table 5.5 Mortality and survival rates of species at Beeston, River Trent, compared to those derived from other UK fisheries.

Species	Location	Z	S	If N_t=1000, N_{t+1}=
Roach	Beeston, River Trent	0.54	0.583	583
	7 UK fisheries	0.20-1.40		
Common bream	Beeston, River Trent	0.24	0.787	787
	7 UK fisheries	0.33-1.23		
Chub	Beeston, River Trent	0.21	0.811	811
	3 UK fisheries	0.15-0.44		
Perch	Beeston, River Trent	0.78	0.458	458
	2 UK fisheries	0.46-0.53		
Bleak	Beeston, River Trent	1.08	0.340	340
	3 UK fisheries	0.82-1.49		
Gudgeon	Beeston, River Trent	1.42	0.242	242
	2 UK fisheries	1.02-1.12		
Hybrid	Beeston, River Trent	0.43	0.651	651

(Cowx *et al.* 1995).

Cormorant site occupancy

Section A

The numbers of cormorants observed on the section showed a high degree of variation within and between winters (Figure 5.21). In winter 1995/96, the numbers of observed cormorants varied between 20 and 37 birds. With the exception of February, less than 10 of the cormorants were observed feeding on the section in all months. The majority of cormorants were observed flying. Thus, determining diurnal occupancy was difficult, for the majority of observed cormorants were not occupying the site, but appeared to be using it as a 'fly-way' to locate adjacent river sections (Feltham *et al.* 1999). Similar occupancy patterns were observed in 1996/97 and 1997/98, with flying the main cormorant activity in the section. The numbers of cormorants increased to between 15 and 55 birds in 1996/97 and between 18 and 65 birds in 1997/98.

Thus, the majority of cormorants observed at Section A at Beeston were utilising the site as a 'fly-way' to locate adjacent feeding sites. The seasonal use of the section revealed very low numbers of cormorants utilising the section between May and September, due to breeding dispersal from the Attenborough roost (Section 4.5.1, Figure 4.13).

Section B

A larger number of cormorants were observed at Section B compared with Section A, with numbers varying by month between 10 and 180 birds (Figure 5.22). However, an even greater proportion of cormorants was observed flying over the section, utilising it as a 'fly-way' to locate adjacent stillwaters (Feltham *et al.* 1999).

Cormorant feeding success

The foraging success of cormorants on Section A was between 29.5 % and 45.2 % of all bouts resulting in ingested fish (Table 5.6). These are below values obtained for Holme Pierrepont (Section 4.5.4), but greater than those in the pre-trout stocking period at

Colwick Park Trout Lake (Section 6.5.4). Foraging success was slightly increased in section B with between 39.1 and 52.0 % of all bouts resulting in ingested fish.

The proportion of dives that were successful was low in all years for Sections A and B, with between 5.7 and 11.5 % of dives being successful (Table 5.6). This can be compared with Holme Pierrepont (12.7 to 15.2 %, Section 4.5.4) and Colwick Park Trout Lake (1.8 to 13.9 %, Section 6.5.4).

Table 5.6 Feeding success of cormorants on count sections A and B, River Trent, 1995 to 1998.

	Section A			Section B		
	1995/96	1996/97	1997/98	1995/96	1996/97	1997/98
Observed foraging bouts	42	39	17	25	69	61
Successful foraging bouts	19	14	5	13	27	26
% foraging bouts successful	45.2	35.9	29.5	52.0	39.1	42.6
Dives observed	439	226	141	261	598	488
Successful dives	32	26	8	16	42	45
% dives successful	7.3	11.5	5.7	6.1	7.0	9.2

5.5.3 *Impact of cormorants at Beeston*

The impact of cormorant predation on the fish populations at Beeston was assessed by combining results from cormorant count sections A and B.

Species composition of fisheries surveys, angler catches and cormorant predation

Observation of cormorant feeding behaviour did not allow the identification of fish predated on to species level and consequently fish were classified as cyprinids and perch. Accordingly, all fisheries data collected from electric fishing surveys and angler catch returns were reclassified as cyprinids, perch, pike and eels to allow direct comparison.

Cyprinids were the main dietary component selected by cormorants, comprising between 70 and 83 % of all predated fish in the study period (Figure 5.23). Cyprinids were also the dominant fish species in angler catches, representing 93 to 97 % of all fish caught (Figure 5.23). These observations compare favourably with the community structure (Figure 5.23), indicating cormorants and anglers were selecting the most abundant fish species for exploitation.

Perch were the only other species observed in the diet of cormorants and in angler catches, with only a small proportion represented in the latter (3 to 7 %). Cormorants generally had a constant proportion of perch in their diet (Figure 5.23), reflecting their presence in the community.

Pike and eels were caught in electric fishing surveys, but did not appear in the cormorant diet or in angler catches. It is known that specialist anglers do target these species on the River Trent (T. Holden pers. comm.). Thus, cormorants selected against this component of the fish population during foraging bouts in preference for cyprinids and perch.

Length frequency distribution

Observation of cormorant feeding behaviour allowed the classification of prey size into 50-mm increments. Accordingly, all data collected from electric fishing surveys were reassigned to 50-mm size classes for comparison.

In 1995/96 and 1996/97 cormorants predominantly took cyprinids below 100 mm. However, in 1997/98 this size selection increased to fish in the size range 151 to 200-mm, although the numbers caught were low (6) (Figure 5.24). Cyprinids above 250 mm were not taken by cormorants, although bream and chub of this size were present in the population.

Thus, cormorants selected cyprinids below 100 mm for consumption, except in 1997/98 when fish of 151 to 200 mm were favoured. Although electric fishing revealed cyprinids in these size ranges, a high proportion of fish outside of these ranges were present which the cormorants selected against.

Interpretation of growth rate data for the main cyprinid species allowed the approximate age that fish were susceptible to cormorant predation to be predicted. Roach and hybrids were generally vulnerable to predation throughout their life, while bream were vulnerable up to the age of four and chub up to the age of six (Figure 5.17, 5.18, 5.19, 5.20; Table 5.3). Bleak and gudgeon were vulnerable to predation throughout their life (Table 5.3). This was found to contrast with cyprinids at Holme Pierrepont which were vulnerable to cormorant predation up to a maximum age of two, due to their extremely fast growth rates (Figures 4.7 and 4.8).

Perch in the size range 51 to 250 mm were predated on by cormorants over the study period (Figure 5.25). Variation in the dominant size classes predated on by cormorants was observed between study years, with perch in the size classes 51 to 100 mm and 101 to 150 mm being dominant in cormorant diet in 1995/96 (Figure 4.22). Perch in the size class 51 to 100 mm were the dominant size taken by cormorants in 1996/97. In 1997/98 cormorants selected larger perch in the size ranges 101 to 150 mm and 151 to 200 mm (Figure 5.25). Thus, cormorants selected different size classes of perch during feeding bouts over the study period. Perch above 250 mm were captured in electric fishing surveys, indicating a proportion of the population not susceptible to cormorant predation.

Comparison of cormorant predated perch lengths and perch growth rate data indicated they were susceptible to predation all their life (Figure 5.19, 5.25). However, this was not strictly true because older perch up to 400 mm were caught but their age determination was not possible.

Growth indices

The annual growth increments of roach at Beeston revealed fluctuations in the increments (Figure 5.26, 5.27), with good growth years often being followed by poor growth years (for example, 1995 and 1996; Figure 5.26). Comparison of the growth indices and year class strength results of roach at Beeston (Figure 5.16, 5.26) show a general trend of good growth years corresponding to years of strong year classes. Hence, 1990, 1991, 1992 and 1995 produced strong year classes of roach and good relative growth rate. Year class strength of roach was related to temperature (Figure 5.16). Thus, the growth of roach at Beeston can be tentatively related to annual temperature. It was independent of the influence of the observed cormorant predation.

Reconstruction of cohort with and without cormorant predation

Data on the impact of cormorant predation on the population structure of roach, common bream, chub, hybrid and perch revealed that cormorants removed a considerable quantity by number and biomass of the species present in the Beeston section of the river (Table 5.7 to 5.11; Figures 5.28 to 5.51). The difference in the values with and without cormorant predation can **only be attributable to cormorant predation** and not any other source of mortality.

After three winters of feeding, cormorants removed large numbers and biomass of fish from the Beeston section of the River Trent, equivalent to a reduction in numbers of 34.1 % roach, 98.1 % common bream, 34.5 % perch, 21.8 % hybrids and 55.3 % chub (Table 5.7 to 5.11). Figures 5.27 to 5.50 showed how the cormorant predation affected the population structure of each species by age in terms of numbers and biomass.

Table 5.7 Numbers and biomass of roach with and without cormorant predation in the River Trent at Beeston.

		Number ($n\ ha^{-1}$)	% reduction	Biomass ($g\ ha^{-1}$)	% reduction
1996	With cormorants	5068.1		161720.3	
	Without cormorants	5679.1	10.7	169528.9	4.6
1997	With cormorants	2815.6		88671.1	
	Without cormorants	4452.5	36.8	113300.9	21.7
1998	With cormorants	2249.7		68984.2	
	Without cormorants	3411.8	34.1	113579.7	39.3

Table 5.8 Numbers and biomass of common bream with and without cormorant predation in the River Trent at Beeston.

		Number ($n\ ha^{-1}$)	% reduction	Biomass ($g\ ha^{-1}$)	% reduction
1996	With cormorants	1303.7		753084.3	
	Without cormorants	1835.7	29.0	766985.2	1.8
1997	With cormorants	32.3		5671	
	Without cormorants	1677.8	98.1	81473.2	93.0

Table 5.9 Numbers and biomass of hybrids with and without cormorant predation in the River Trent at Beeston.

		Number ($n\ ha^{-1}$)	% reduction	Biomass ($g\ ha^{-1}$)	% reduction
1996	With cormorants	773.7		22247.4	
	Without cormorants	782.7	1.1	22633.3	1.7
1997	With cormorants	127.4		3067.6	
	Without cormorants	147.6	13.7	3861.8	20.6
1998	With cormorants	129.4		3067.6	
	Without cormorants	165.4	21.8	5325.7	42.4

Table 5.10 The numbers and biomass of chub with and without cormorant predation in the River Trent at Beeston.

		Number ($n\ ha^{-1}$)	% reduction	Biomass ($g\ ha^{-1}$)	% reduction
1996	With cormorants	77.9		14694.9	
	Without cormorants	181.9	57.2	16871.4	12.9
1997	With cormorants	265.8		89975.3	
	Without cormorants	594.1	55.3	99826.5	9.7

Table 5.11 The numbers and biomass of perch with and without cormorant predation in the River Trent at Beeston.

		Number ($n\ ha^{-1}$)	% reduction	Biomass ($g\ ha^{-1}$)	% reduction
1996	With cormorants	982.3		52744.7	
	Without cormorants	1478.3	33.6	75243.5	29.9
1997	With cormorants	1473.4		79117.0	
	Without cormorants	2250.8	34.5	128291.0	38.3

A large difference was observed between fish biomass and numerical abundance in the fishery, as revealed by electric fishing, and the cormorant predated fish biomass and numerical abundance, as calculated by MCS data (Table 5.7 to 5.11; Figures 5.28 to 5.51). The fisheries values were considered to be reasonably accurate because hydro-acoustic assessments were of similar dimensions, with annual values between 69.5 to 762.5 fish per hectare between 1994 and 1997 (Table 5.2). Cormorant predation over 1995-1998 removed 611 to 1638 of roach per hectare annually, with other species also showing heavy losses.

When the fish population minimum and maximum densities are considered (Table 5.12), using the electric fishing gear efficiency of P = 0.10 and P = 0.36 (Section 3.3.1), it can be seen cormorant predation had a major effect on the fish populations even when maximum fish abundance values were used (P = 0.10).

It should be noted that the high cormorant predation rates and wide confidence limits are the result of the assumptions made to support the MCS (Section 5.4.3). This model assumes that in cormorant section A, 50 % of the birds observed flying over the river feed on the river and take their daily food intake (DFI) (Section 5.4.3), compared with 10 % in Section B. This is probably high for Section A as many birds probably feed elsewhere in the region and only use the river infrequently or in low numbers for feeding (Section 5.8).

Table 5.12 Impact of cormorant predation (CP) on the populations of roach, common bream, chub, hybrids and perch (n ha^{-1}) over the range of electric fishing gear efficiency at Beeston, River Trent.

		1996		1997		1998	
Species	P	CP	-CP	CP	-CP	CP	-CP
Roach	0.10	11656	12268	6476	8140	5174	6654
	0.23	5068	5679	2816	4480	2580	3730
	0.36	3238	3849	1799	3463	1437	2917
Common bream	0.10	2998	3530	74	1720		
	0.23	1304	1836	32	1678		
	0.36	833	1365	21	1666		
Chub	0.10	179	283	611	940		
	0.23	78	182	266	594		
	0.36	50	154	170	498		
Hybrid	0.10	1795	1804	284	304	283	321
	0.23	780	789	124	143	123	160
	0.36	499	508	79	99	79	116
Perch	0.10	2259	2755	3389	4166		
	0.23	982	1478	1473	2251		
	0.36	628	1124	941	1719		

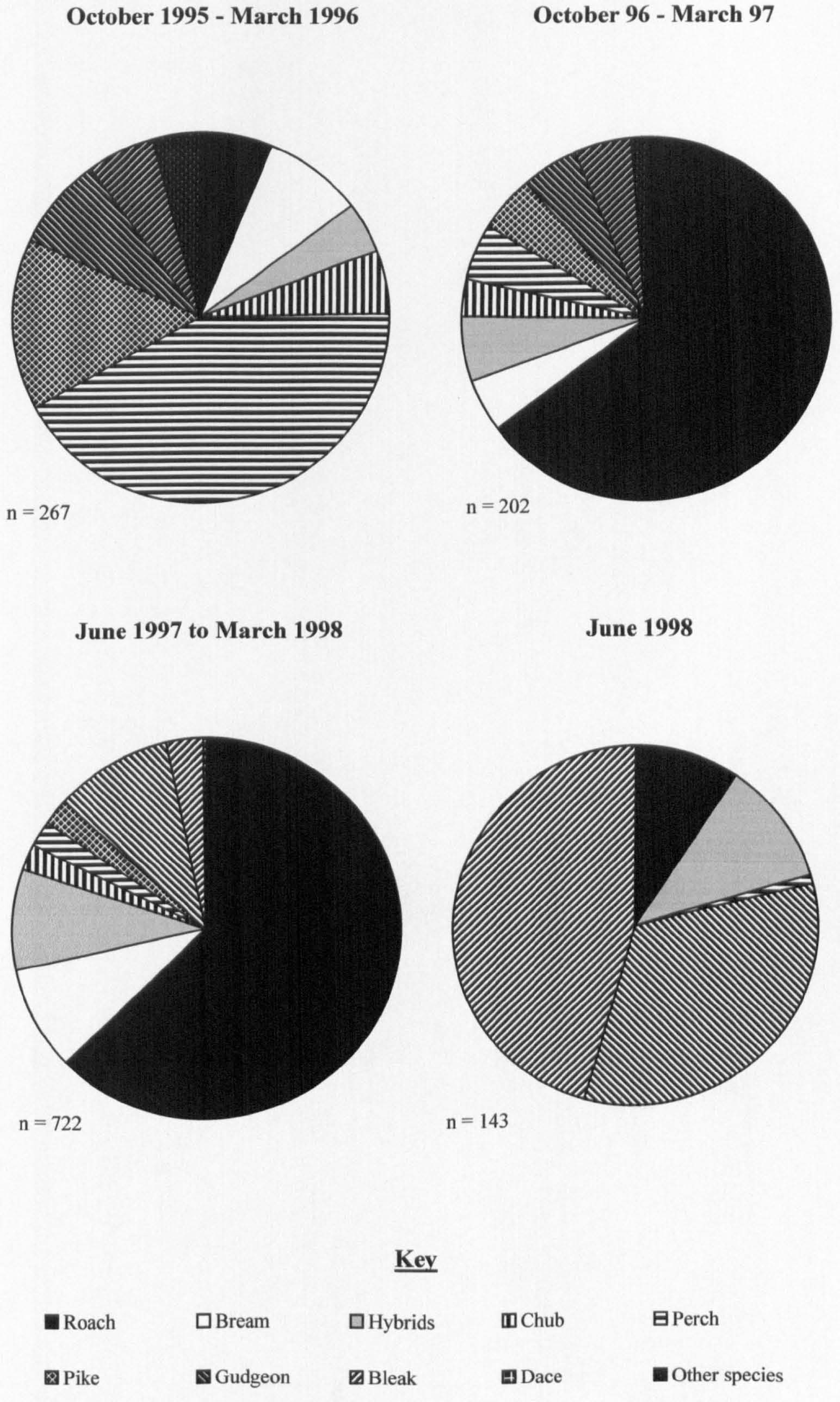
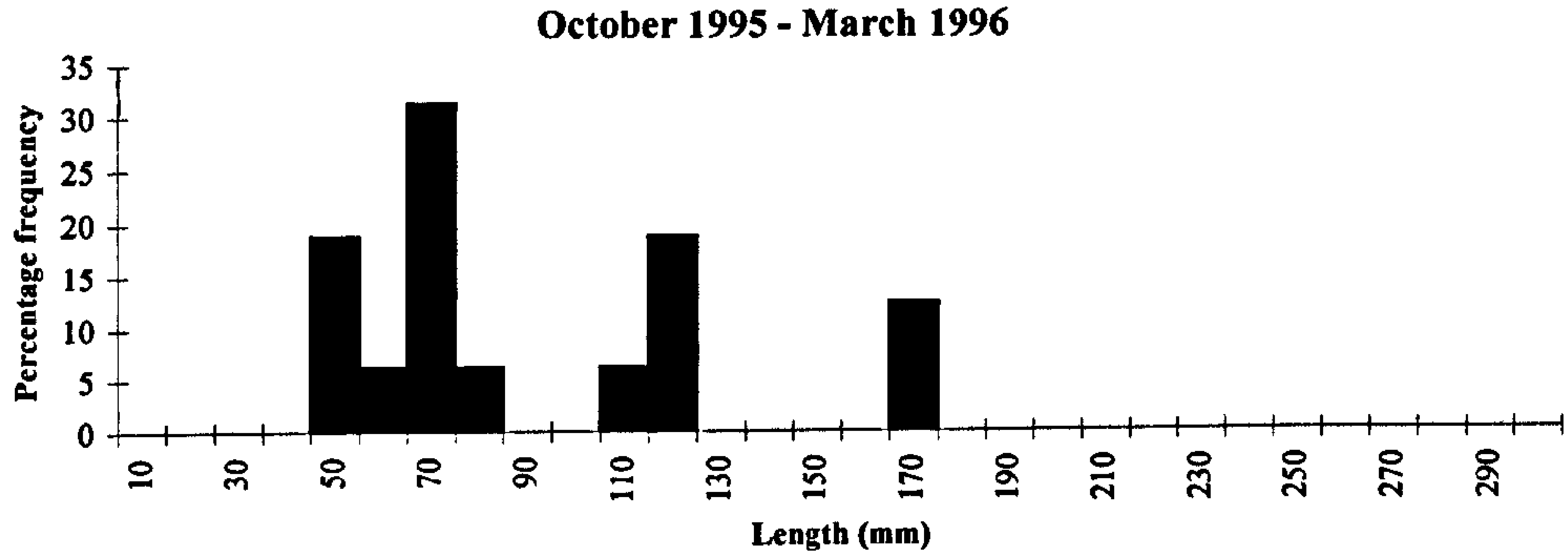
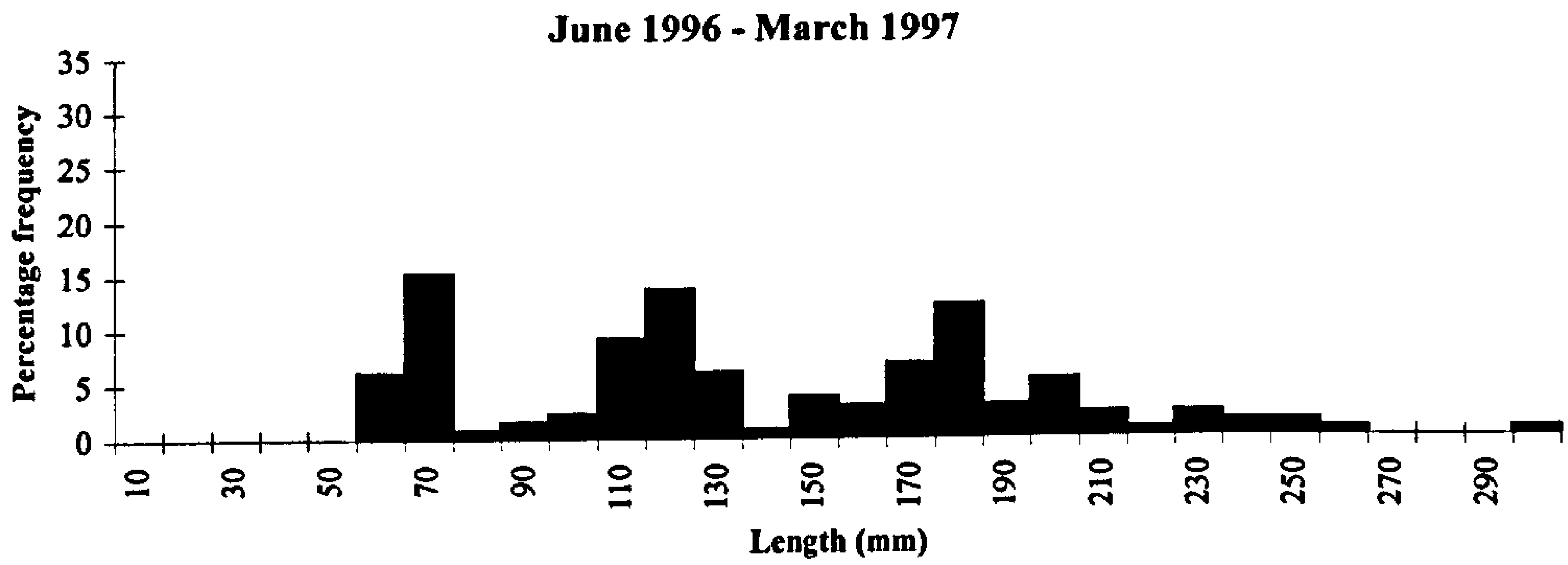


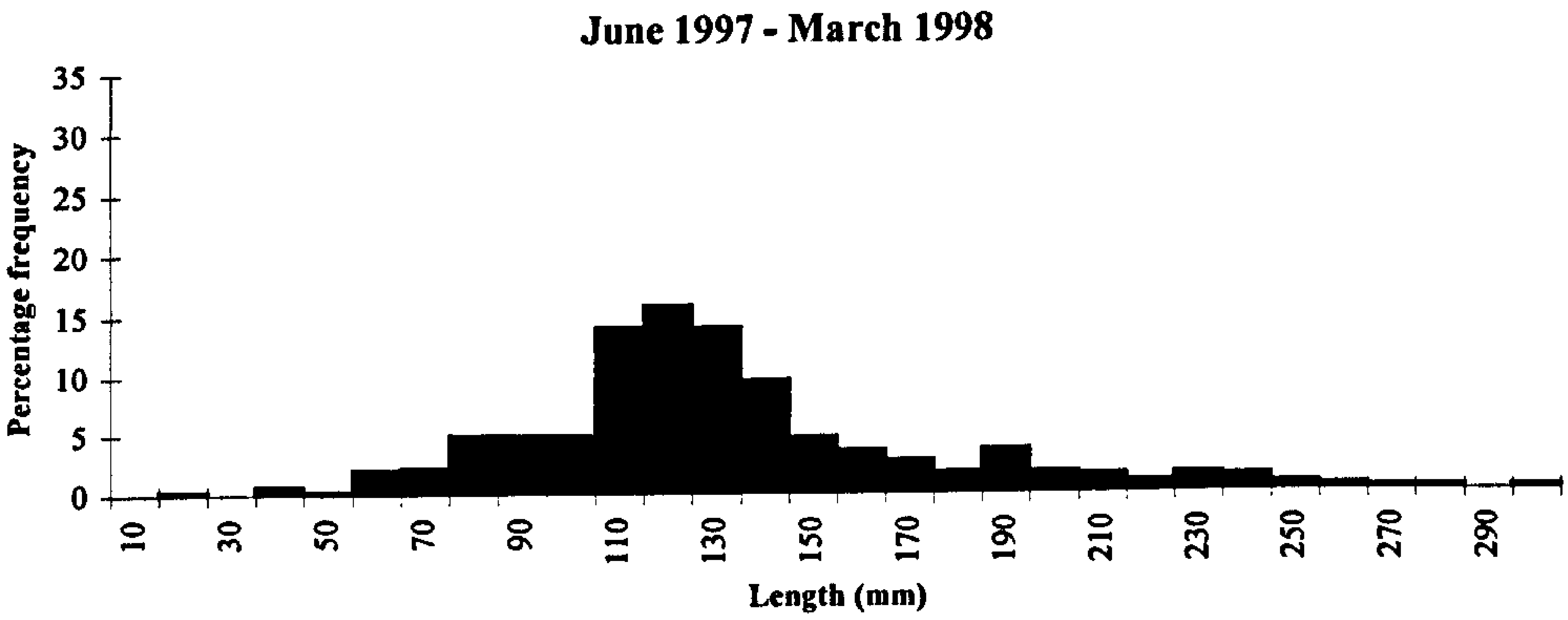
Figure 5.9 Species composition of electric fishing surveys at Beeston, River Trent.



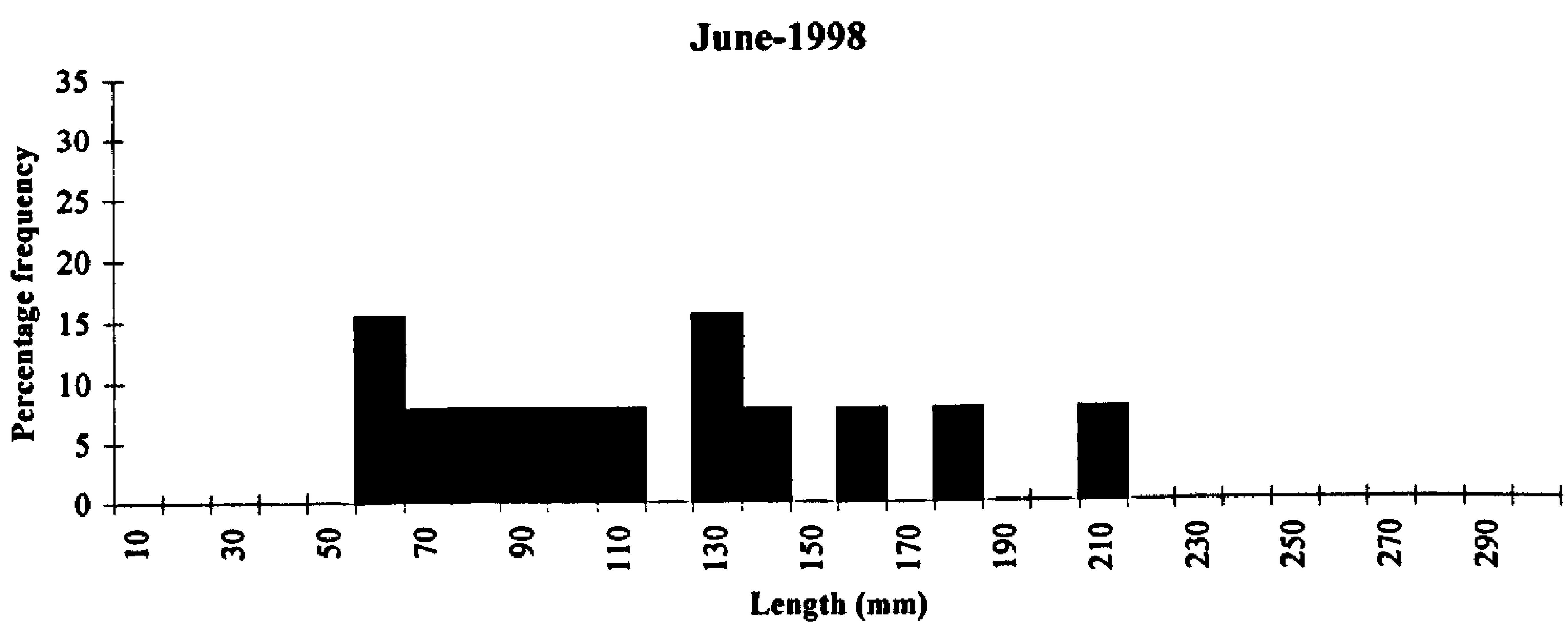
n = 16



n = 131



n = 454



n = 13

Figure 5.10 Length frequency distribution of roach caught by electric fishing at Beeston, River Trent.

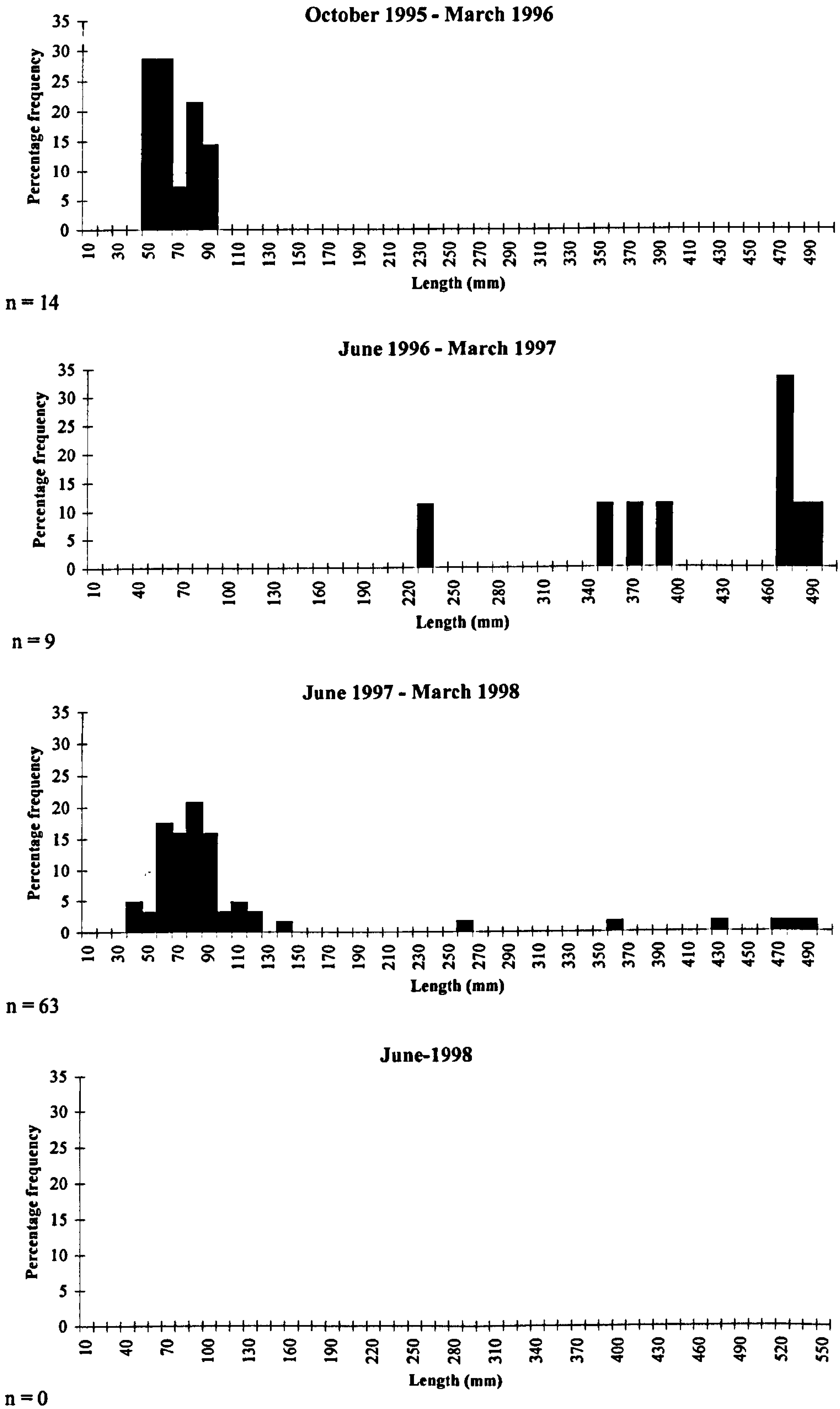


Figure 5.11 Length frequency distribution of common bream caught by electric fishing at Beeston, River Trent.

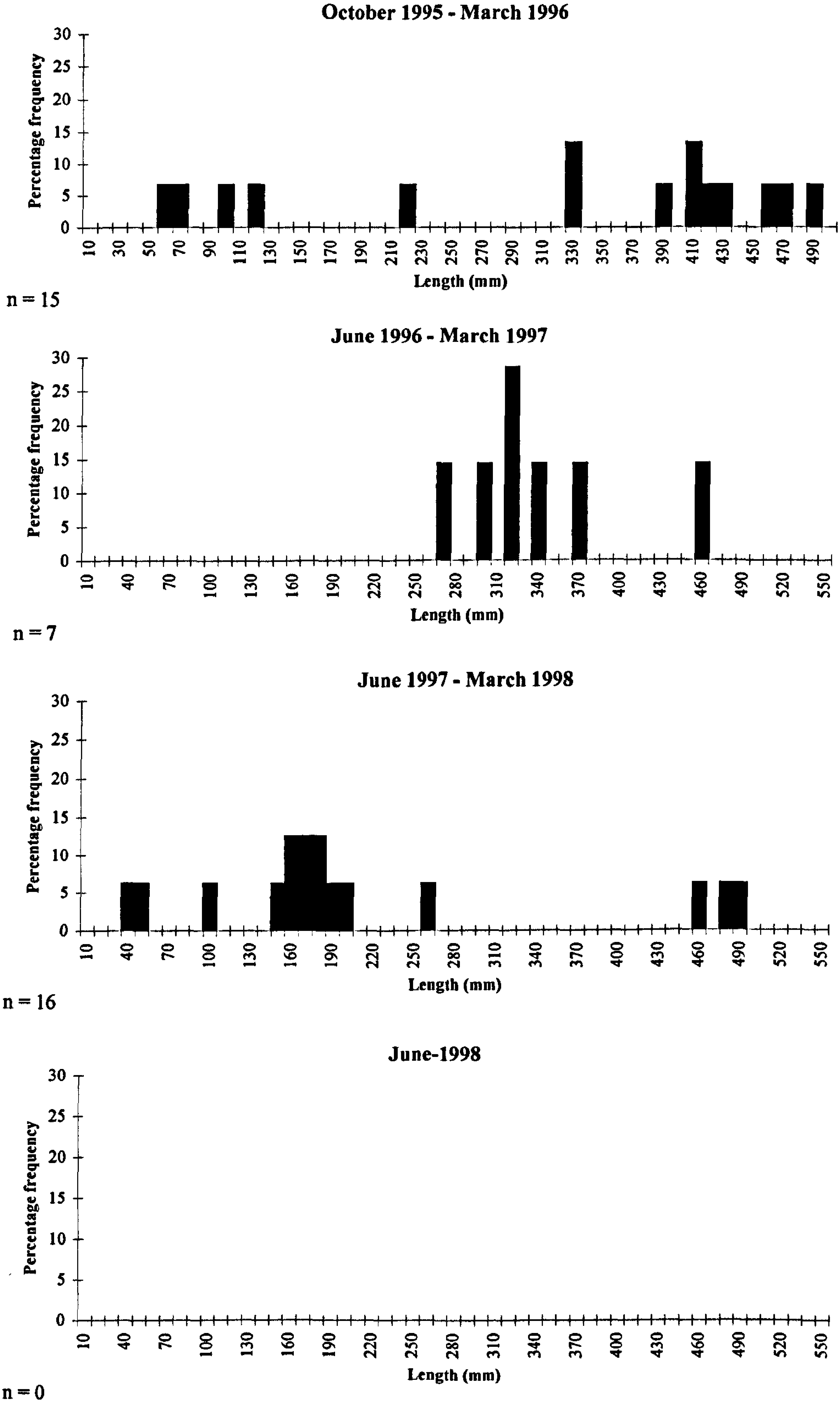


Figure 5.12 Length frequency distribution of chub caught by electric fishing at Beeston, River Trent.

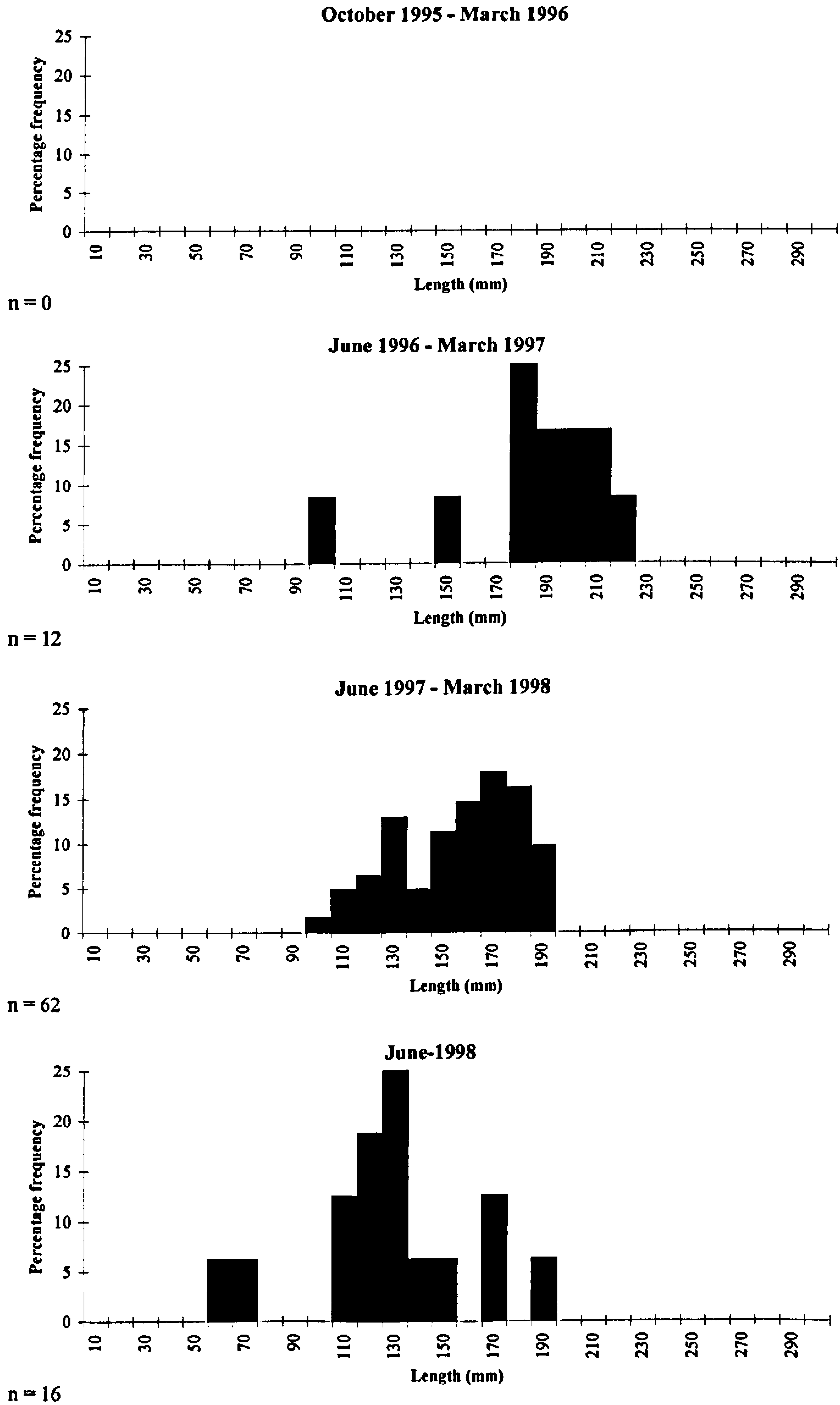


Figure 5.13 Length frequency distribution of hybrids caught by electric fishing at Beeston, River Trent.

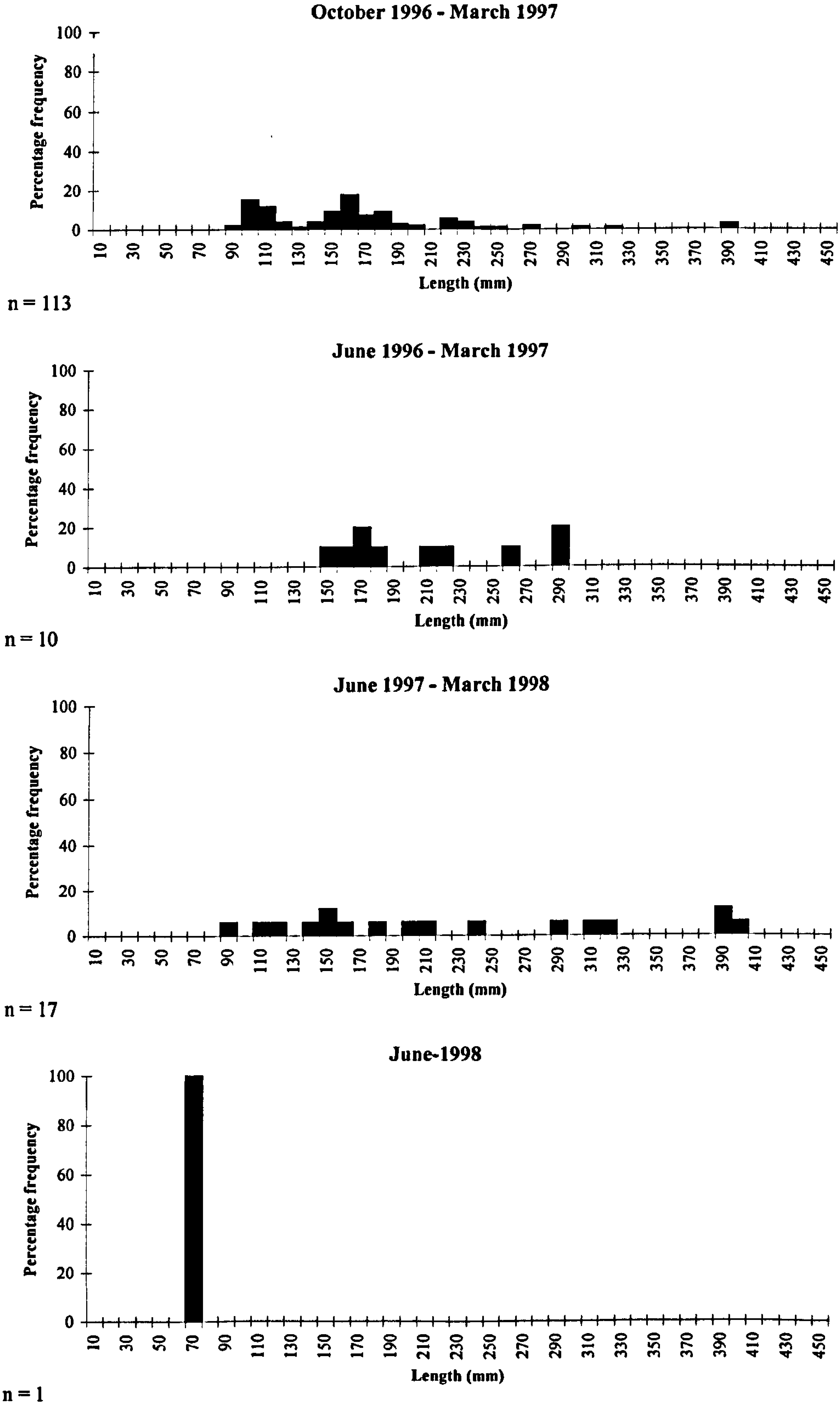


Figure 5.14 Length frequency distribution of perch caught by electric fishing at Beeston, River Trent.

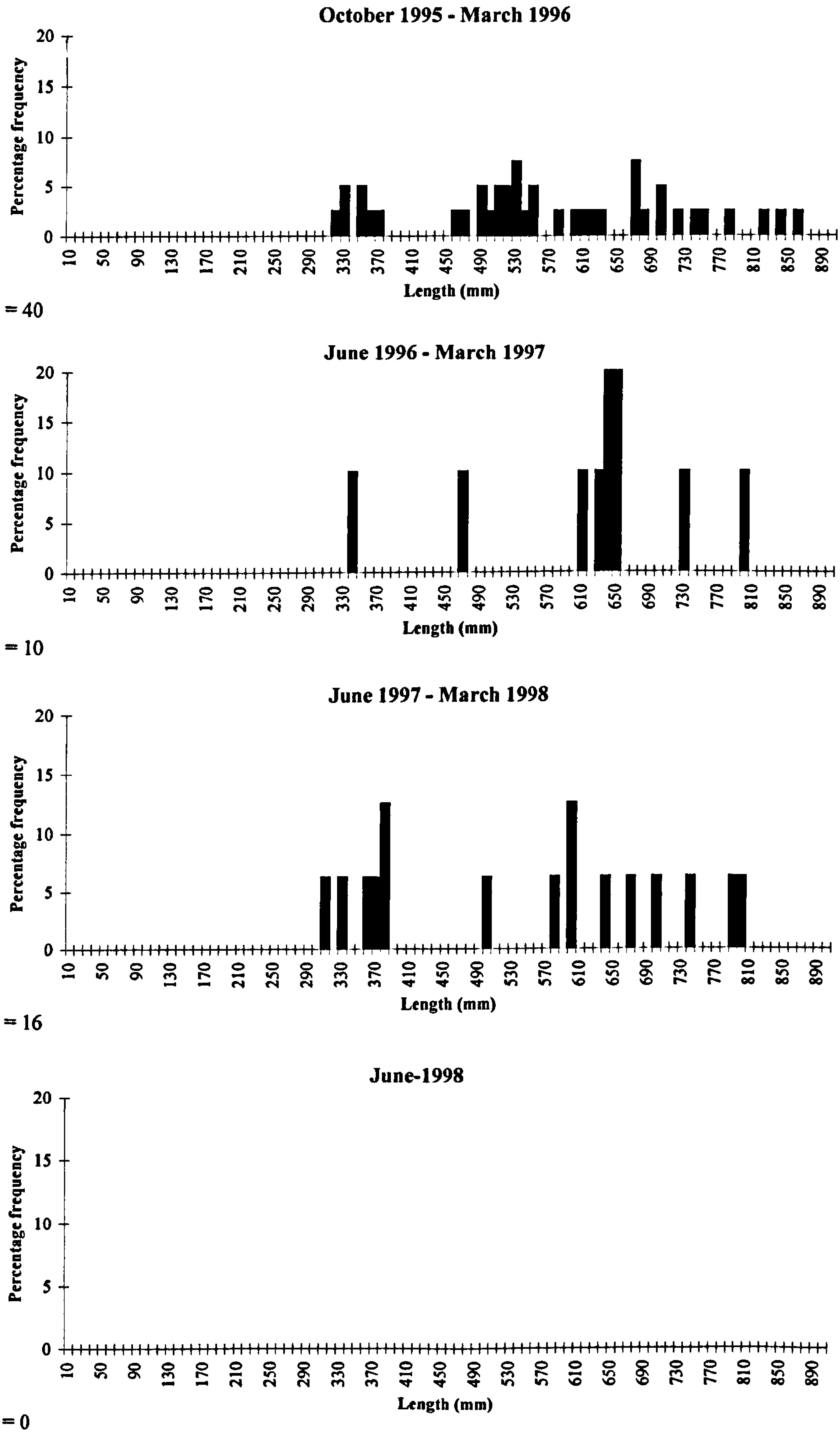
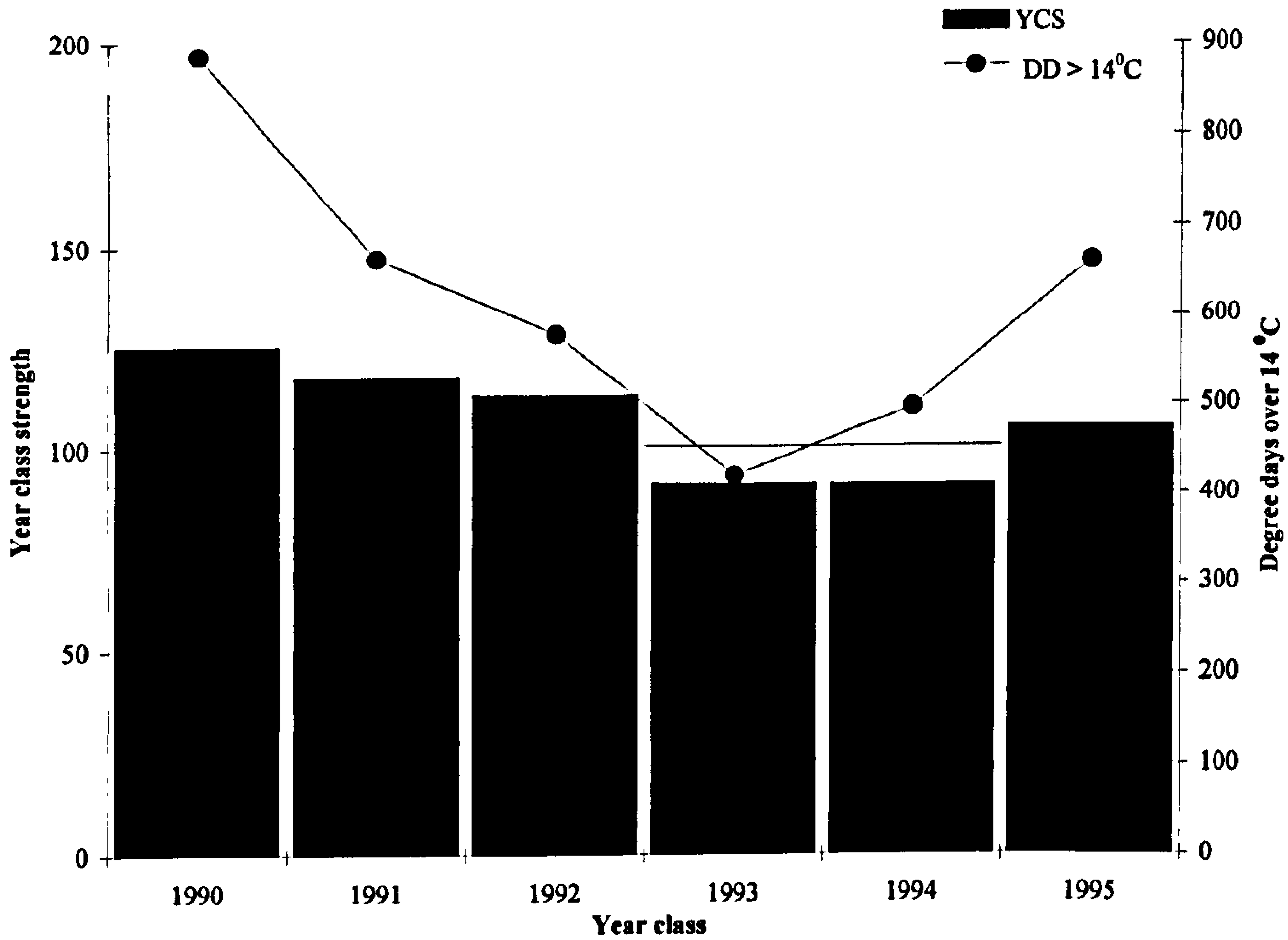


Figure 5.15 Length frequency distribution of pike caught by electric fishing at Beeston, River Trent.



Correlation coefficient = 0.91

Figure 5.16 Year class strength of roach at Beeston, River Trent, and the relationship with degree days above 14 °C.

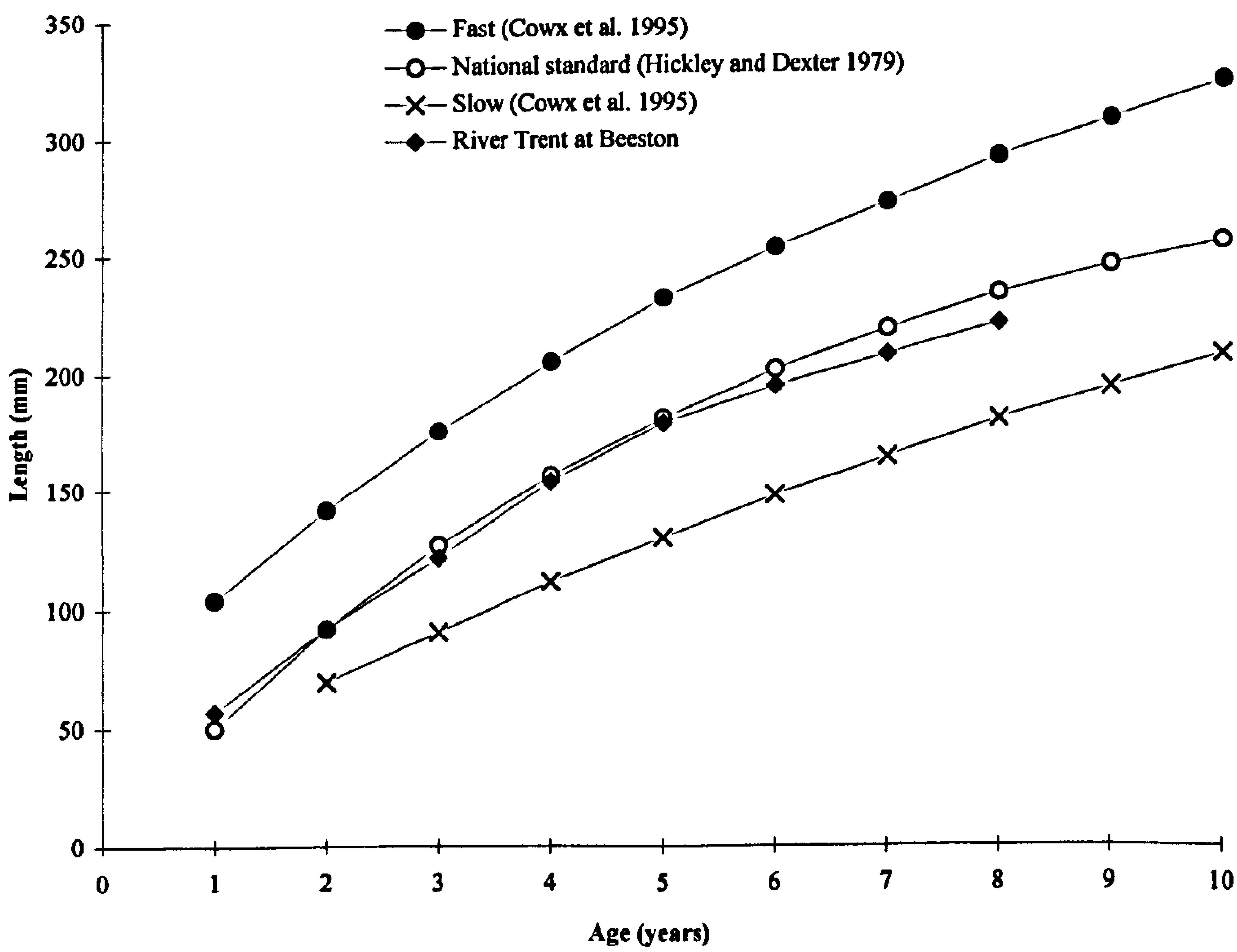


Figure 5.17 Growth of roach at Beeston, River Trent, compared with standard growth curves.

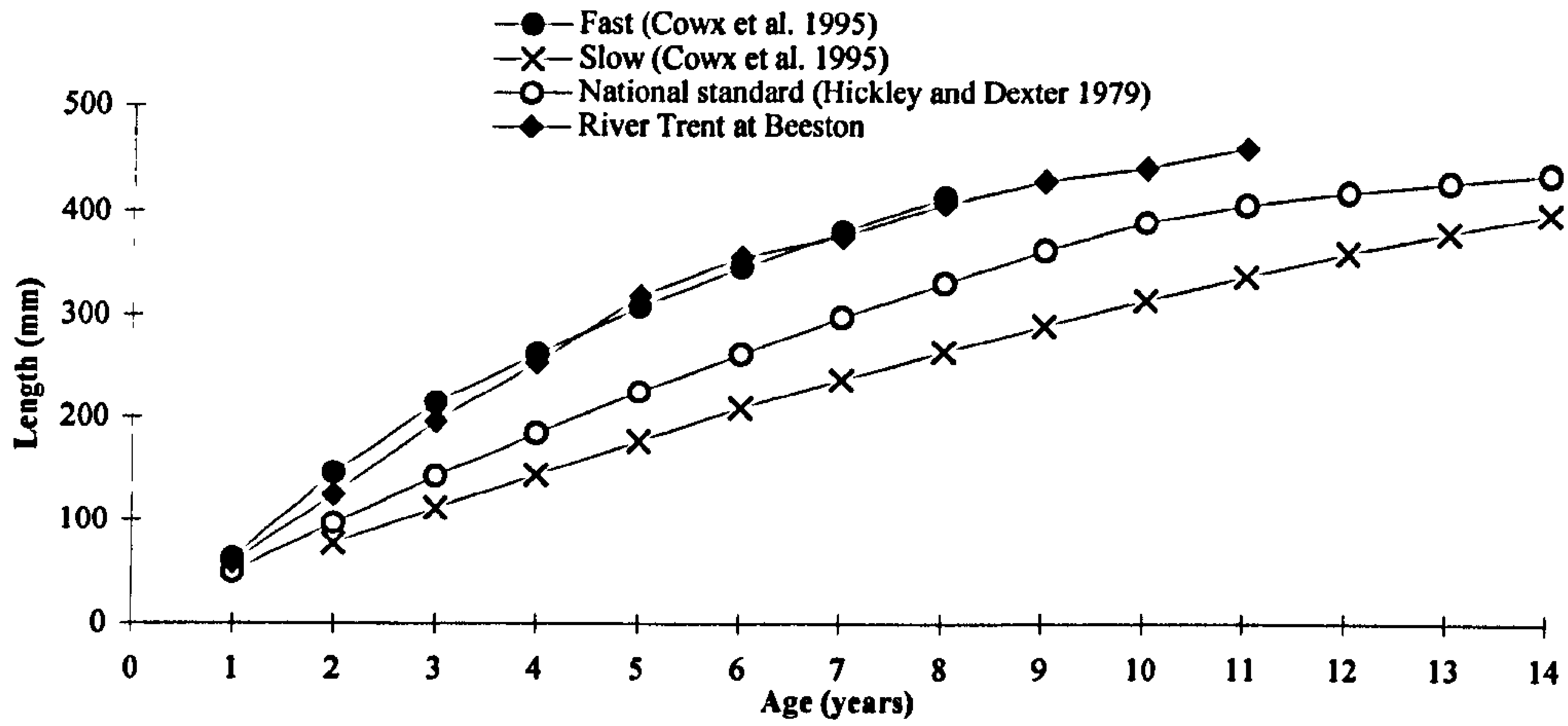


Figure 5.18 Growth of bream at Beeston, River Trent, compared with standard growth curves.

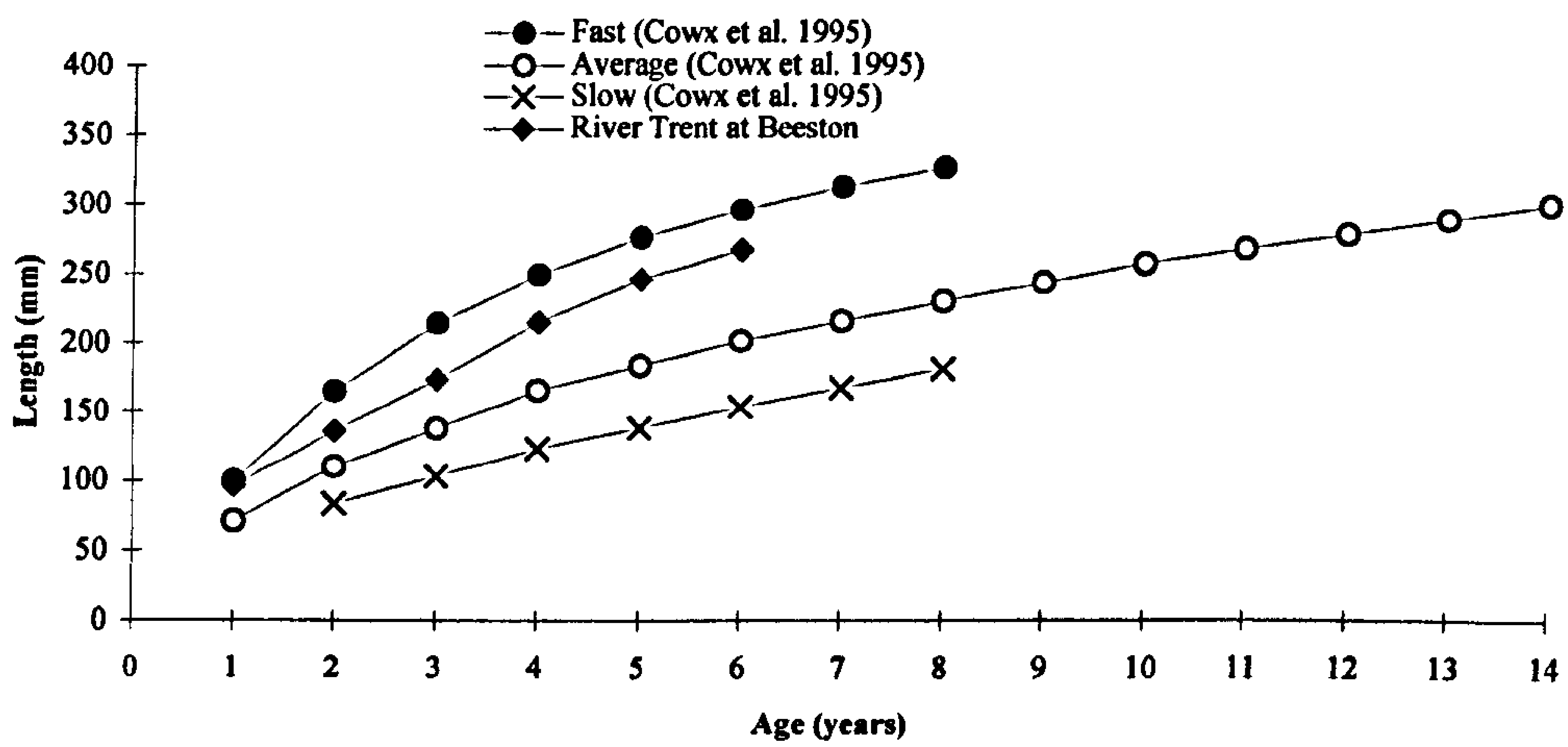


Figure 5.19 Growth of perch at Beeston, River Trent, compared with standard growth curves.

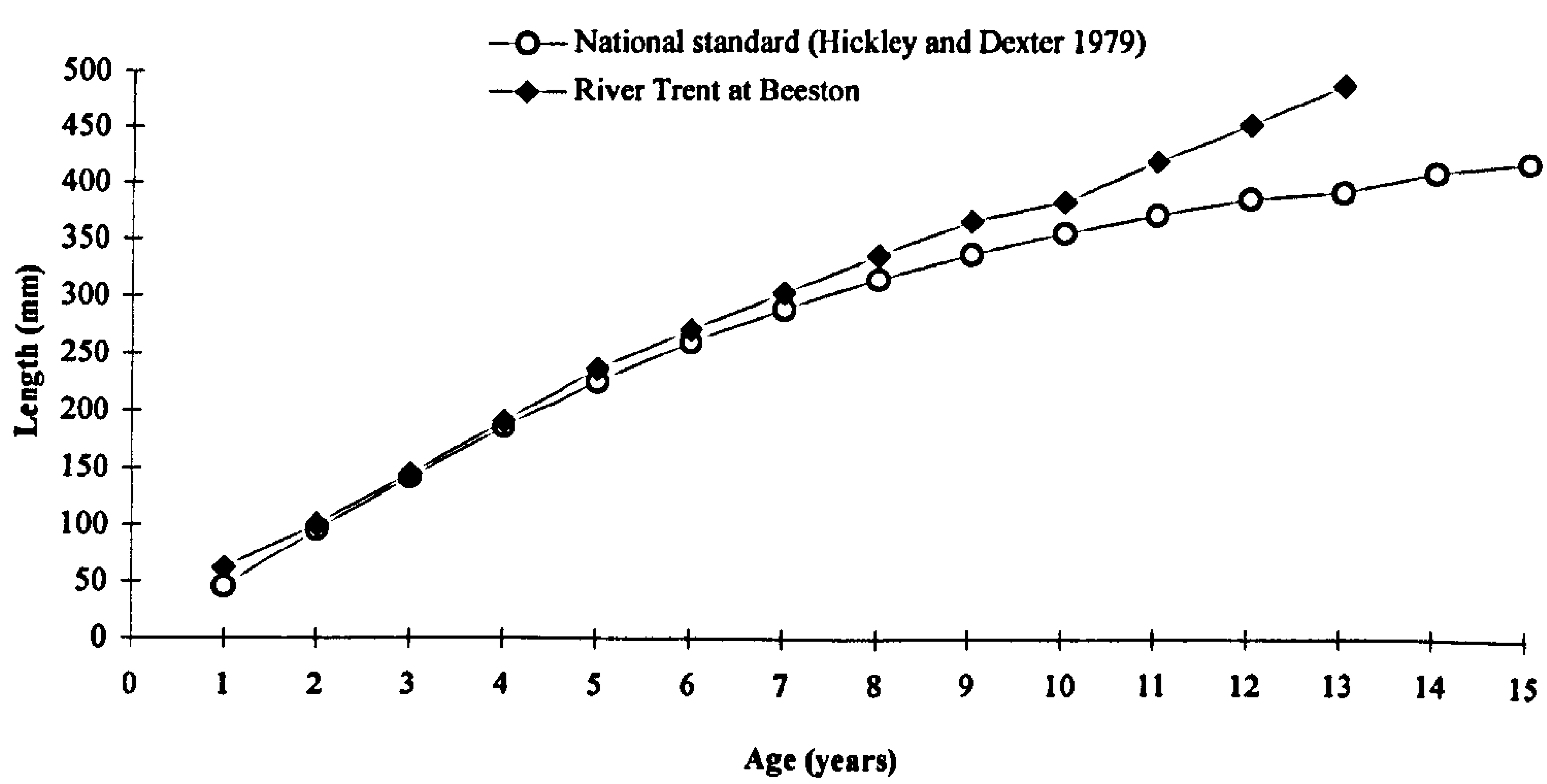


Figure 5.20 Growth of chub at Beeston, River Trent, compared with standard growth curves.

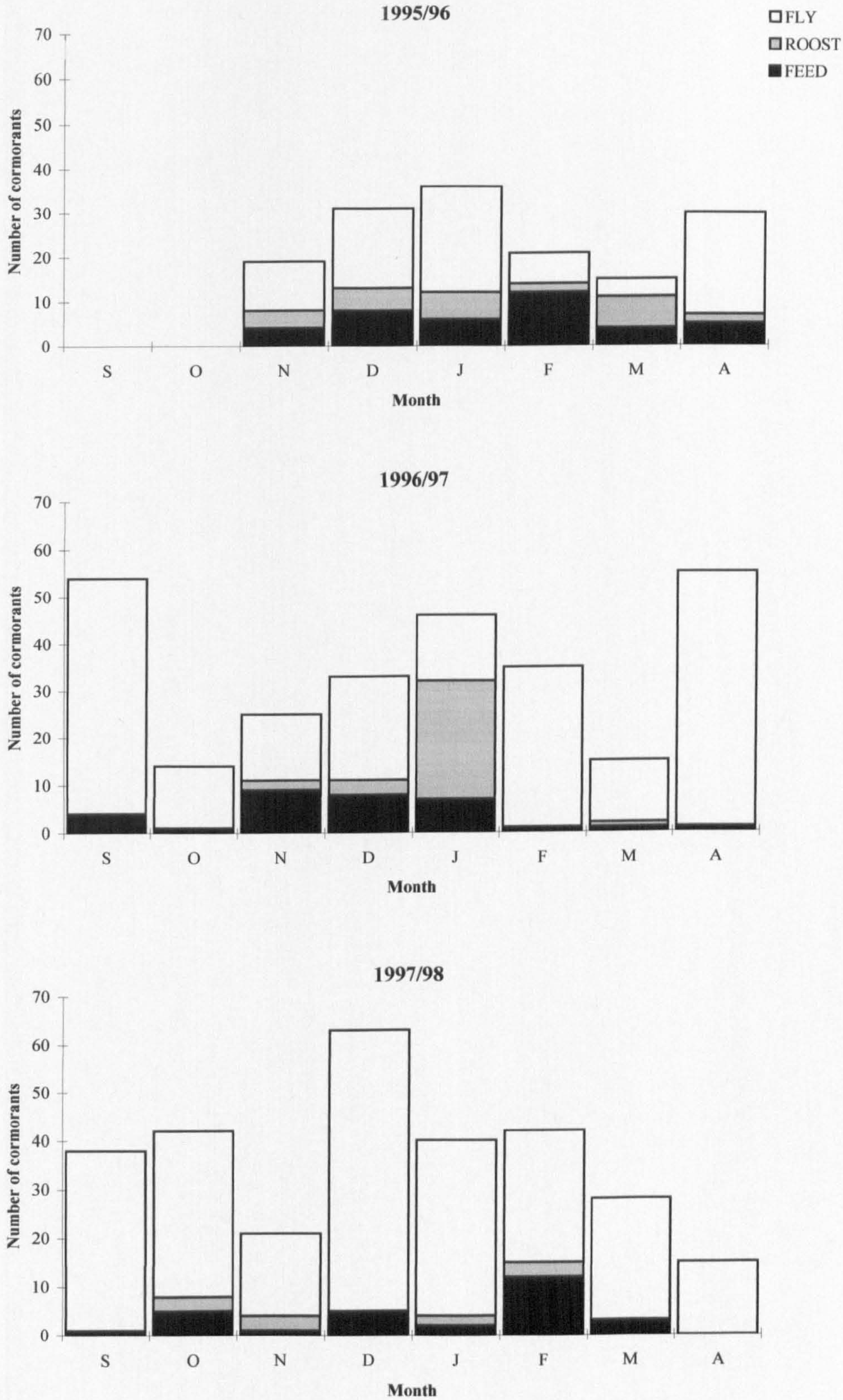


Figure 5.21 The number of cormorants utilising Section A (Beeston) of the River Trent over the study period.

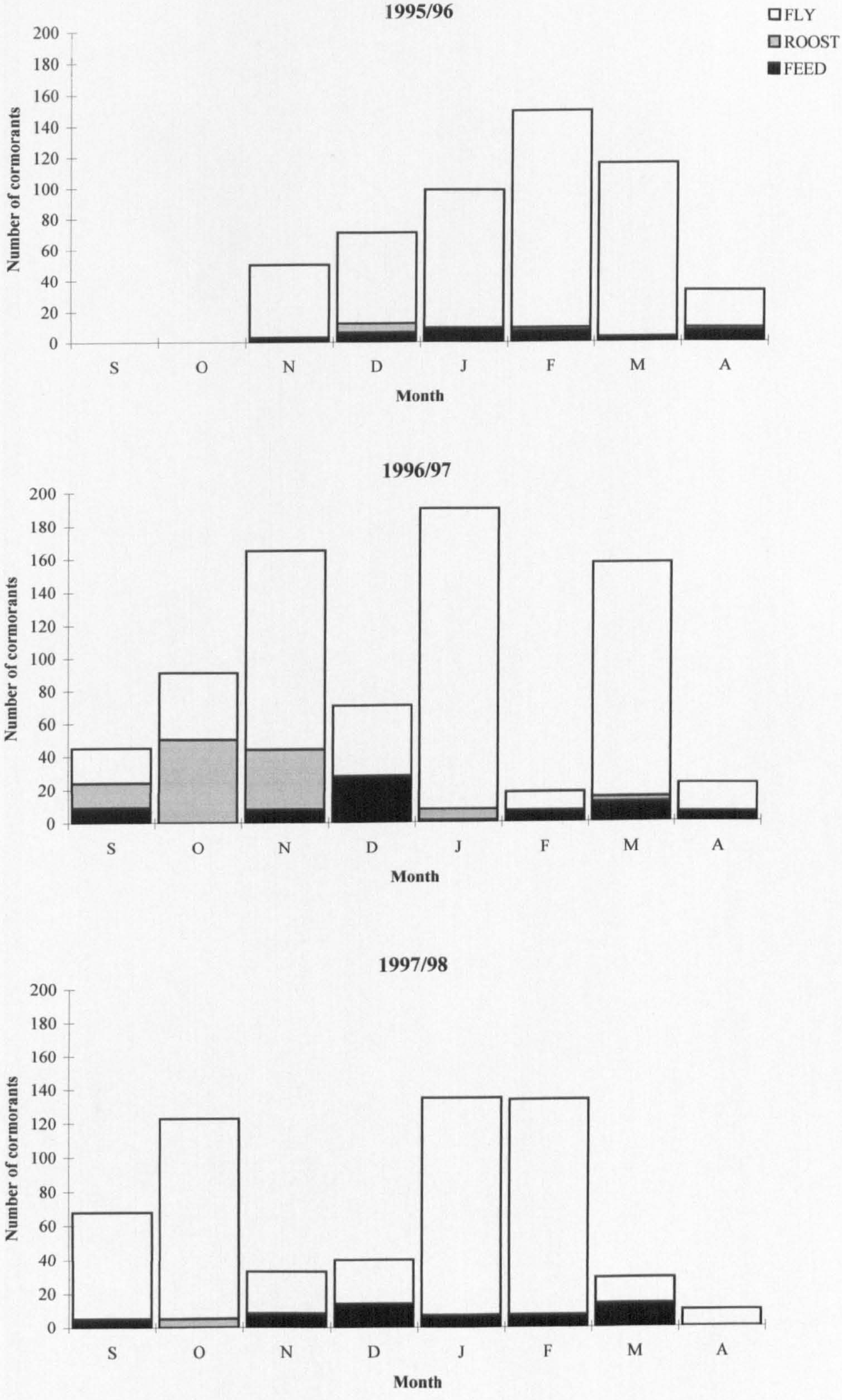


Figure 5.22 The number of cormorants utilising Section B (Beeston) of the River Trent over the study period.

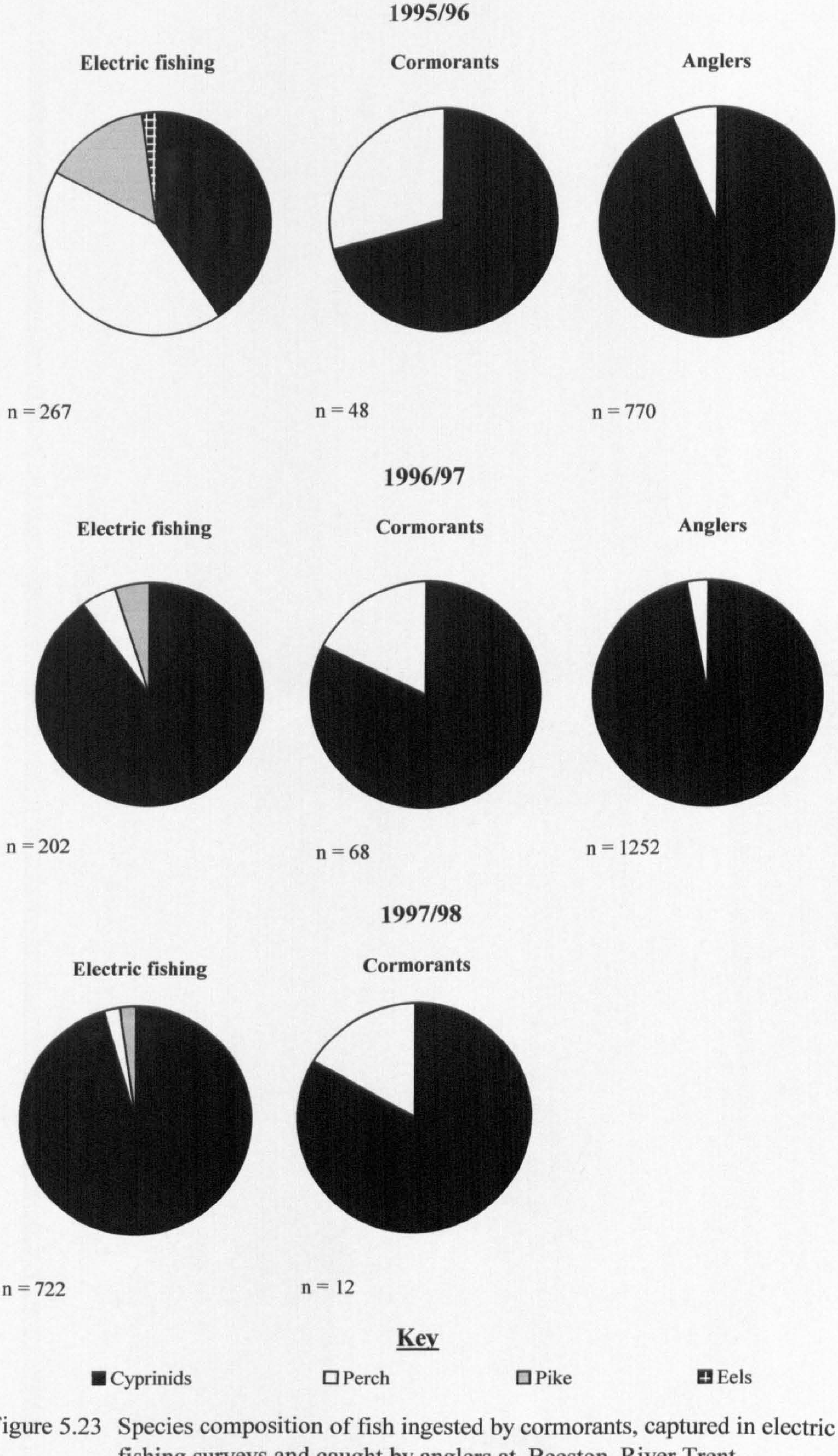


Figure 5.23 Species composition of fish ingested by cormorants, captured in electric fishing surveys and caught by anglers at Beeston, River Trent.

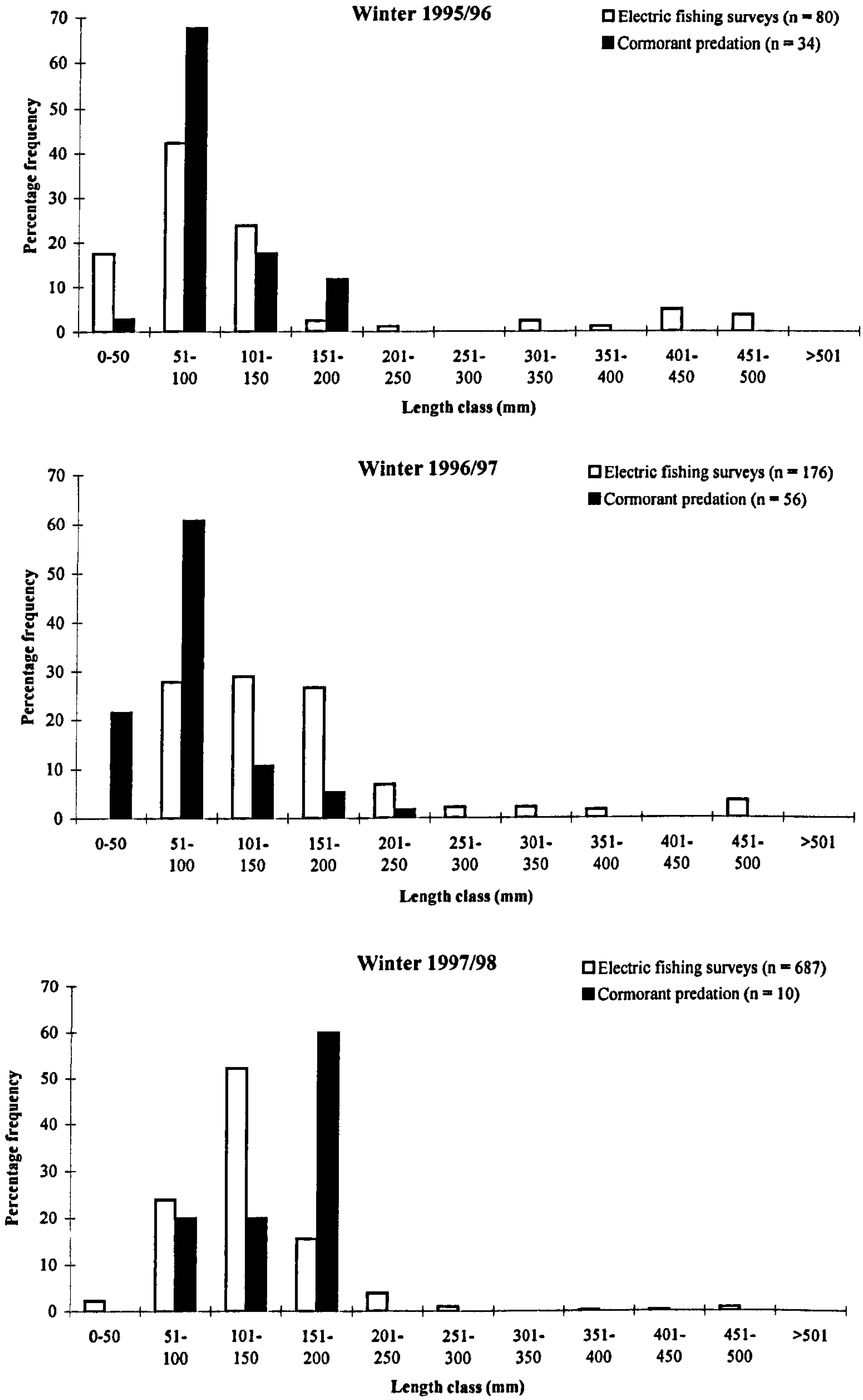


Figure 5.24 Length frequency distribution of cyprinids ingested by cormorants and represented in electric fishing surveys at Beeston, River Trent.

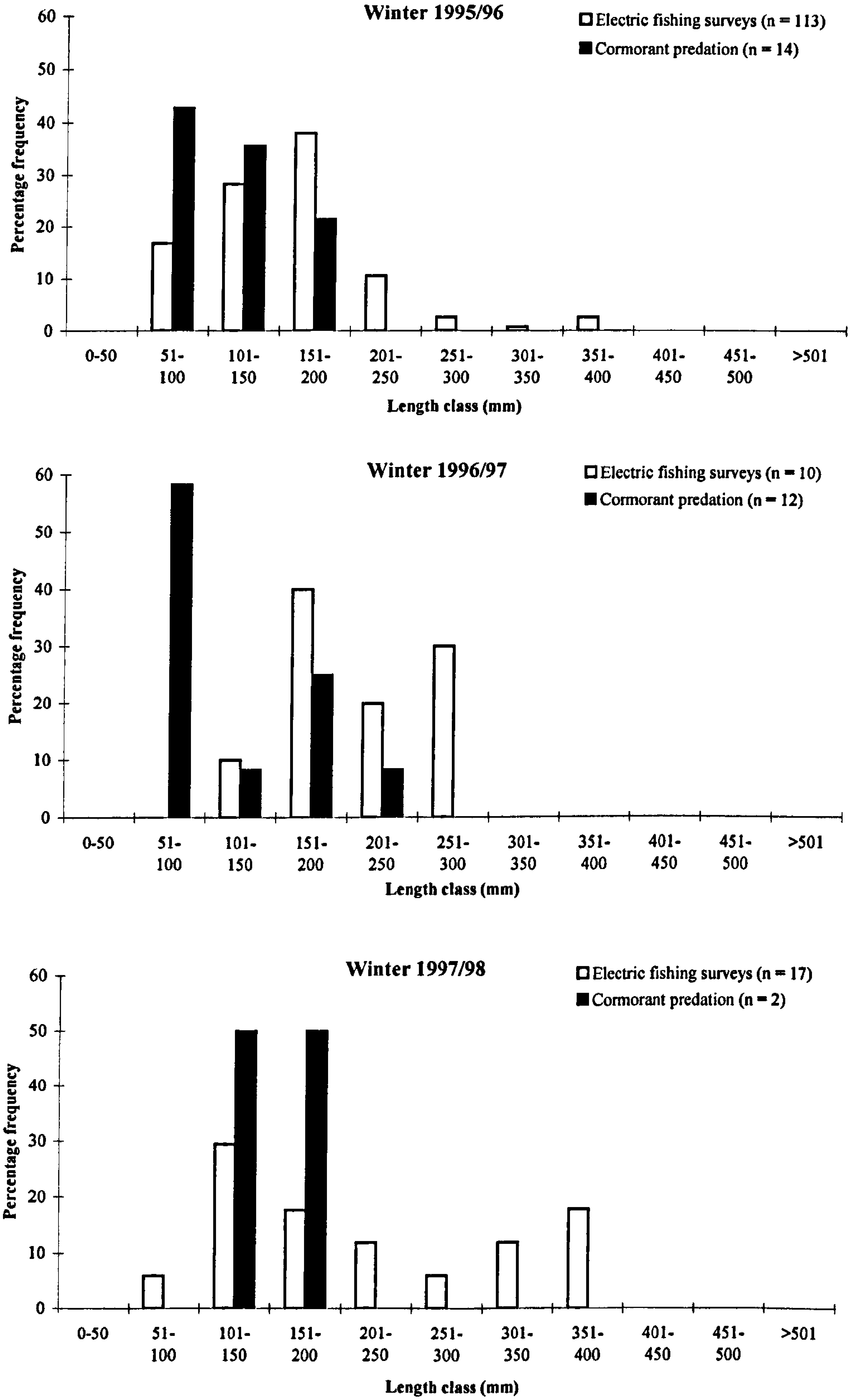


Figure 5.25 Length frequency distributions of perch ingested by cormorants and represented in electric fishing surveys at Beeston, River Trent.

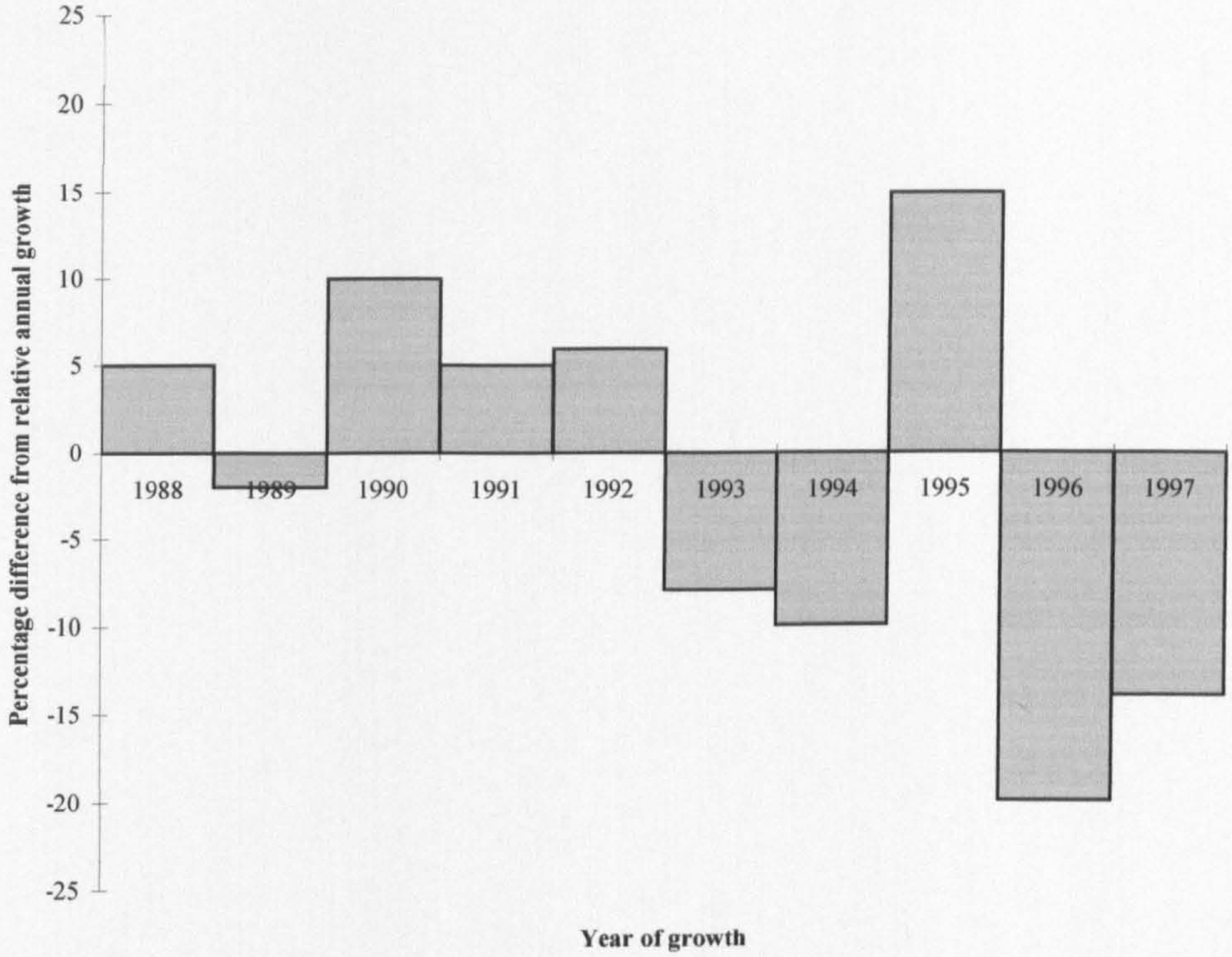


Figure 5.26 Relative annual growth of roach at Beeston, River Trent.

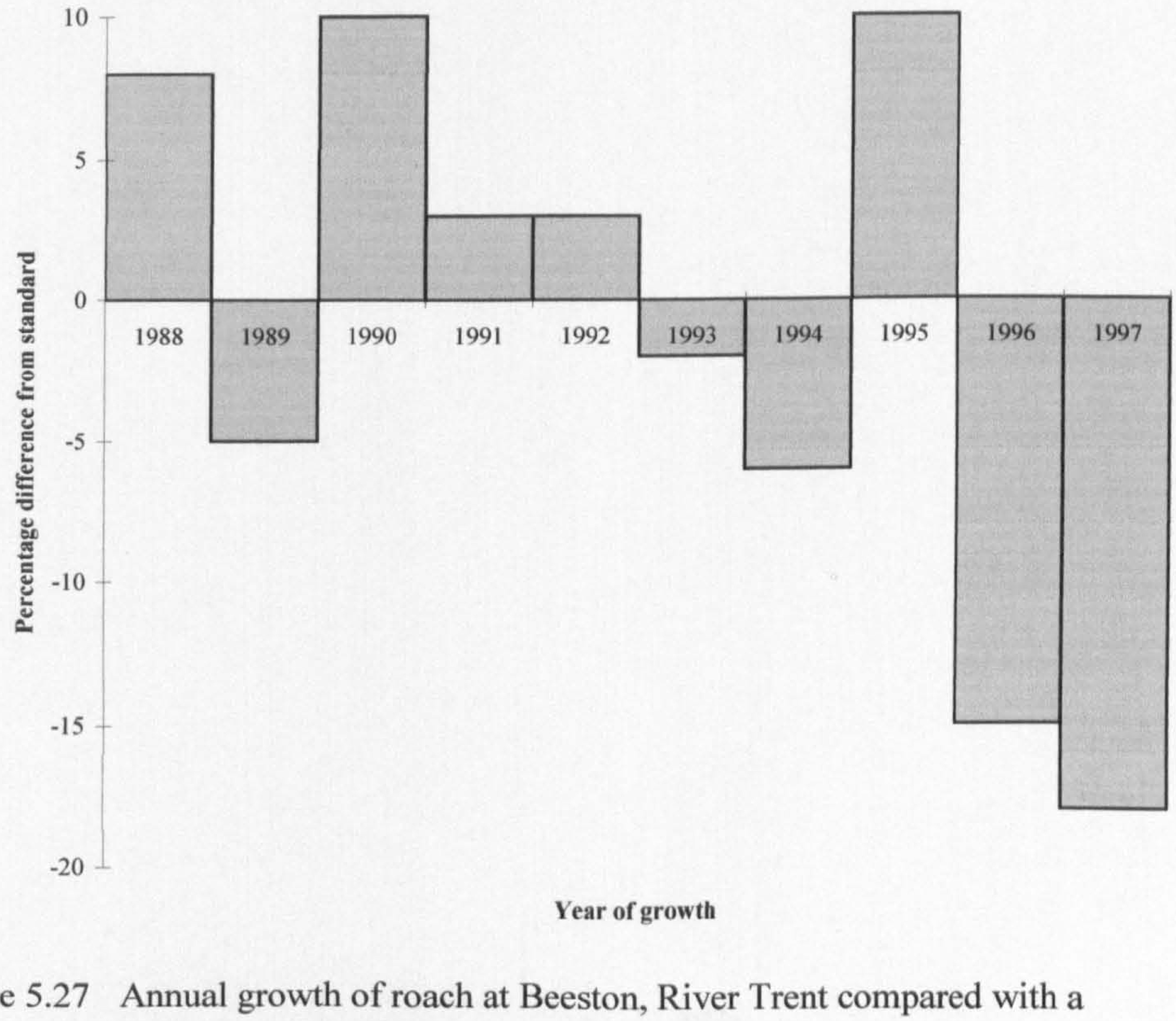


Figure 5.27 Annual growth of roach at Beeston, River Trent compared with a standard roach growth rate (Hickley and Dexter 1979).

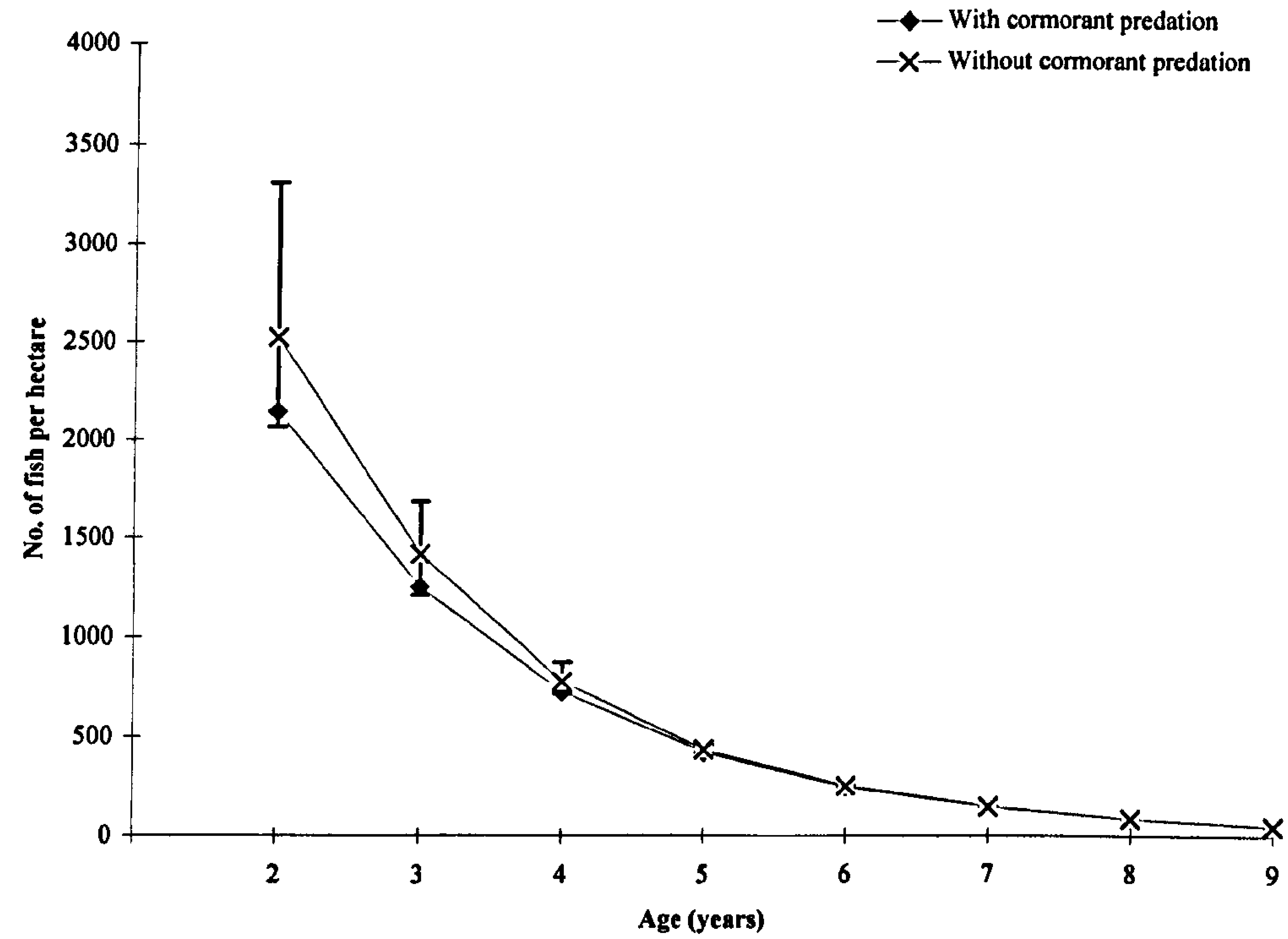


Figure 5.28 Impact of cormorant predation on the numbers of roach at Beeston, River Trent in 1996.

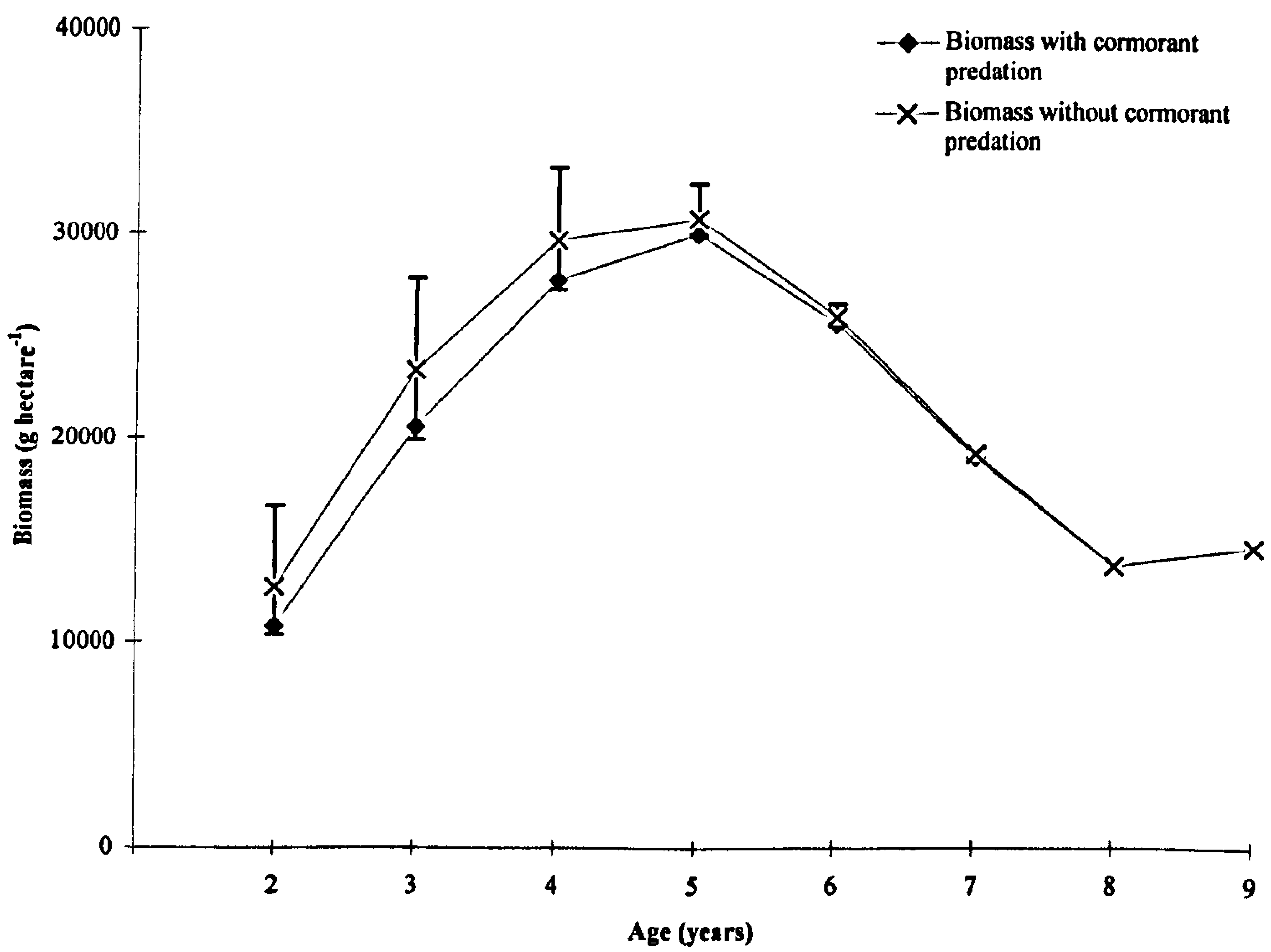


Figure 5.29 Impact of cormorant predation on the biomass of roach at Beeston, River Trent in 1996.

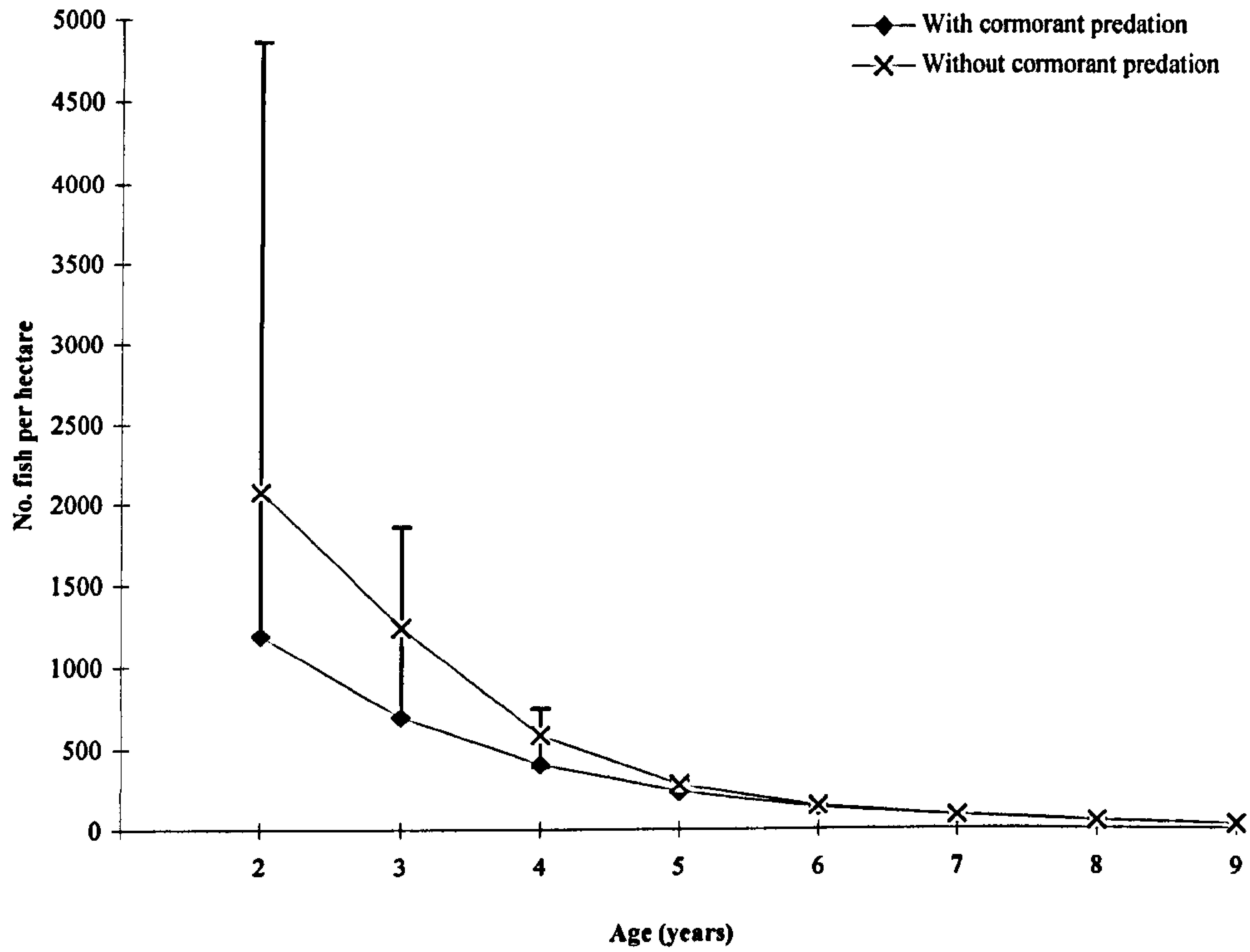


Figure 5.30 Impact of cormorant predation on the numbers of roach at Beeston, River Trent, in 1997.

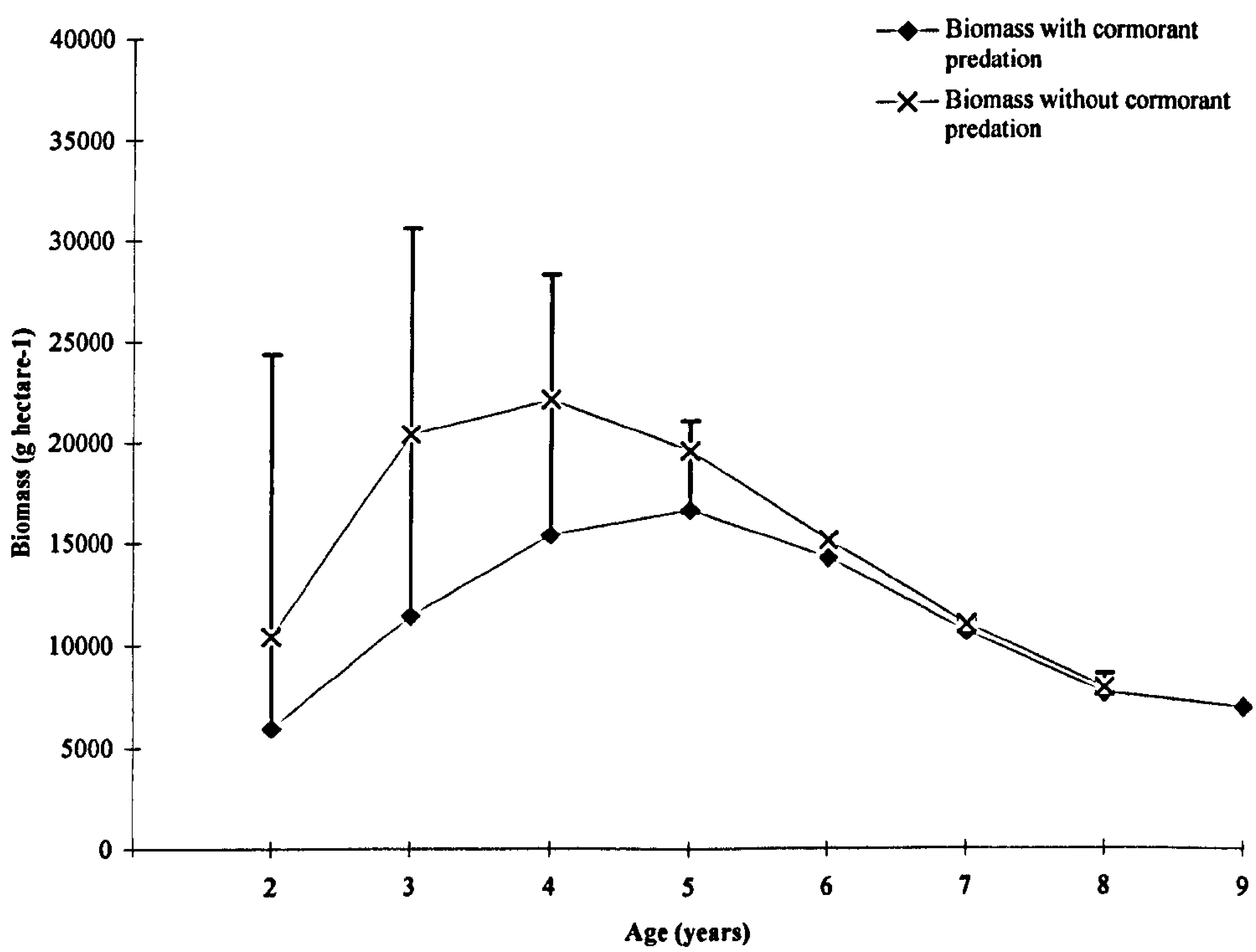


Figure 5.31 Impact of cormorant predation on the biomass of roach at Beeston, River Trent, in 1997.

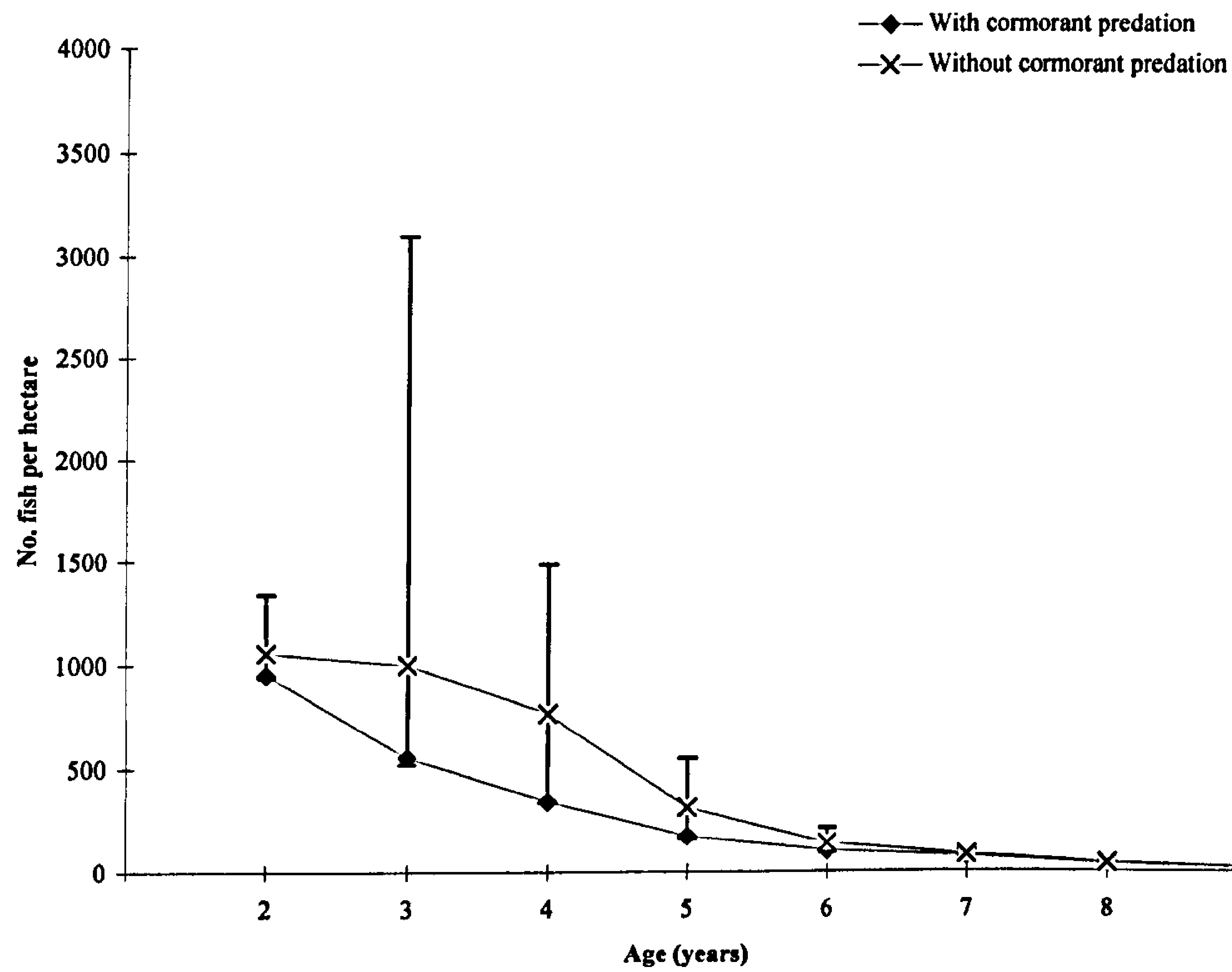


Figure 5.32 Impact of cormorant predation on the numbers of roach at Beeston, River Trent, in 1998.

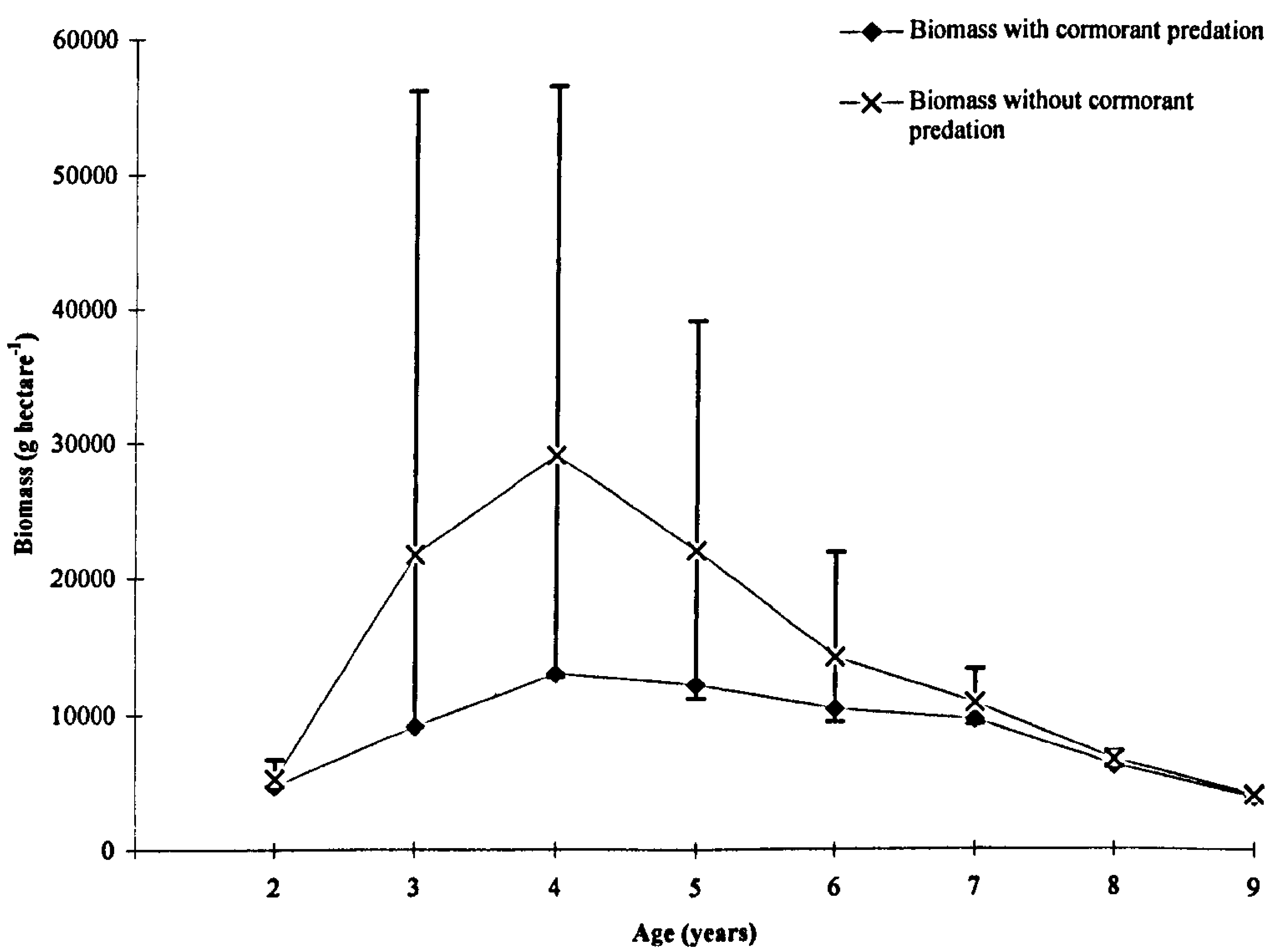


Figure 5.33 Impact of cormorant predation on the biomass of roach at Beeston, River Trent, in 1998.

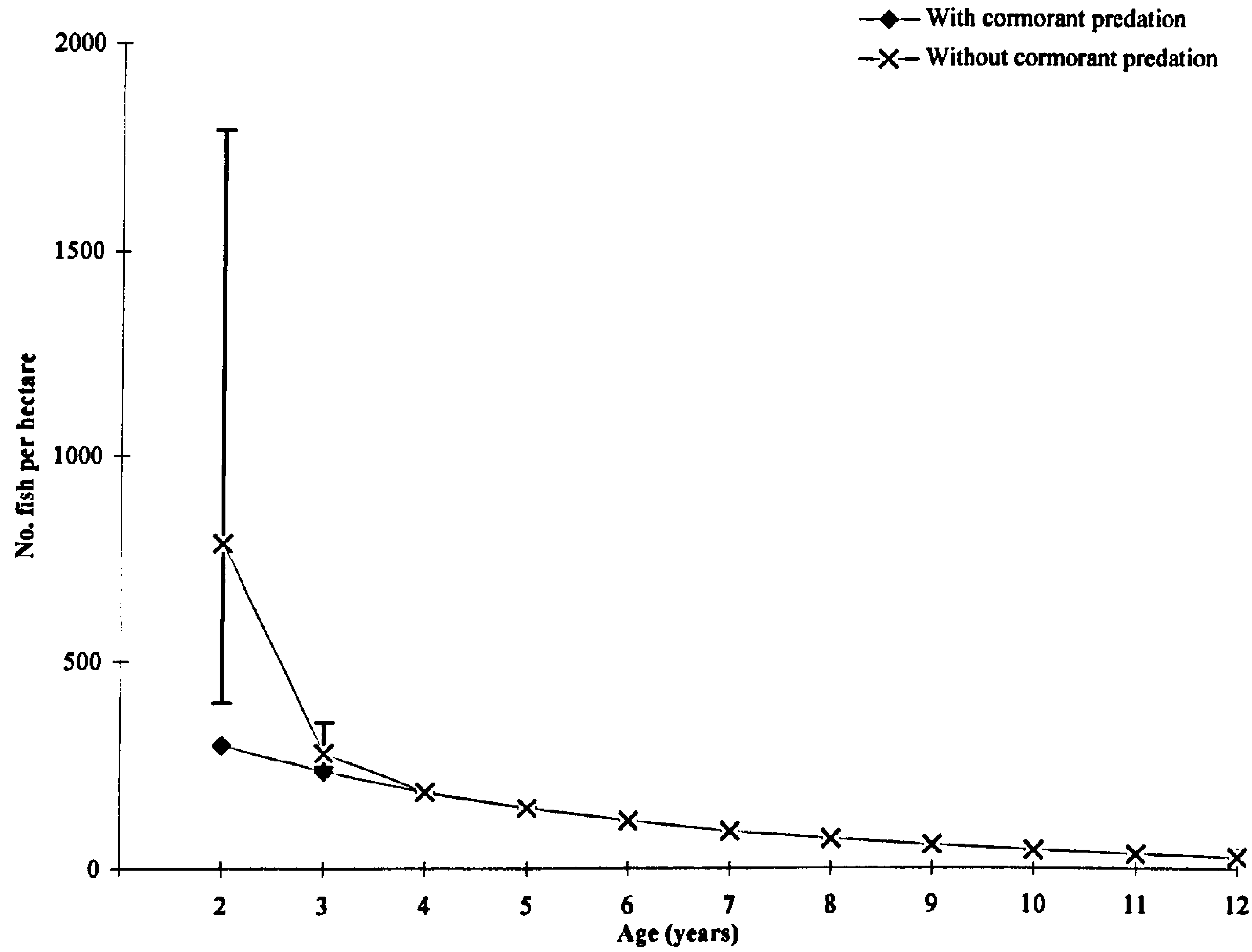


Figure 5.34 Impact of cormorant predation on the numbers of common bream at Beeston, River Trent, in 1996.

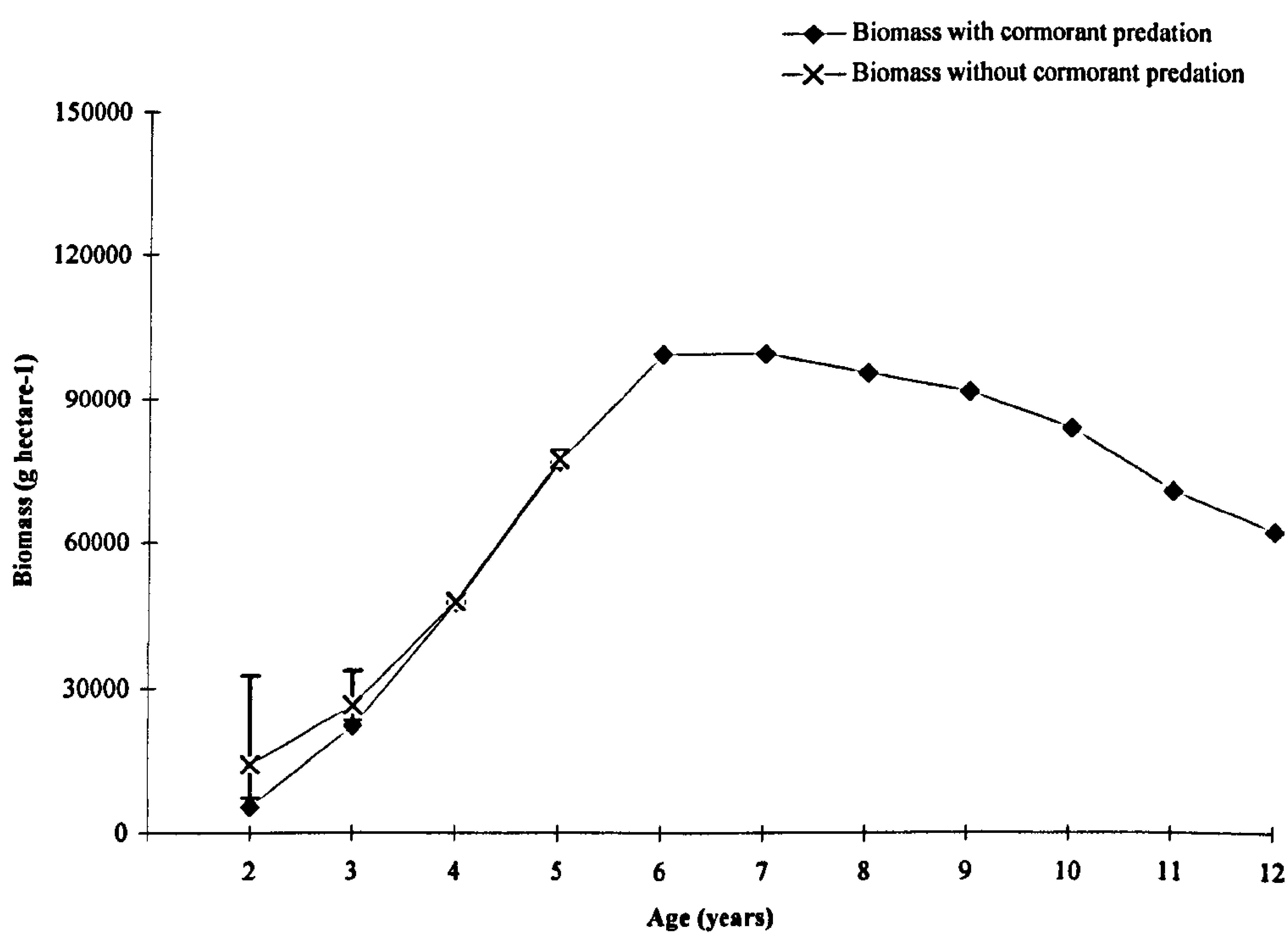


Figure 5.35 Impact of cormorant predation on the biomass of common bream at Beeston, River Trent, in 1996.

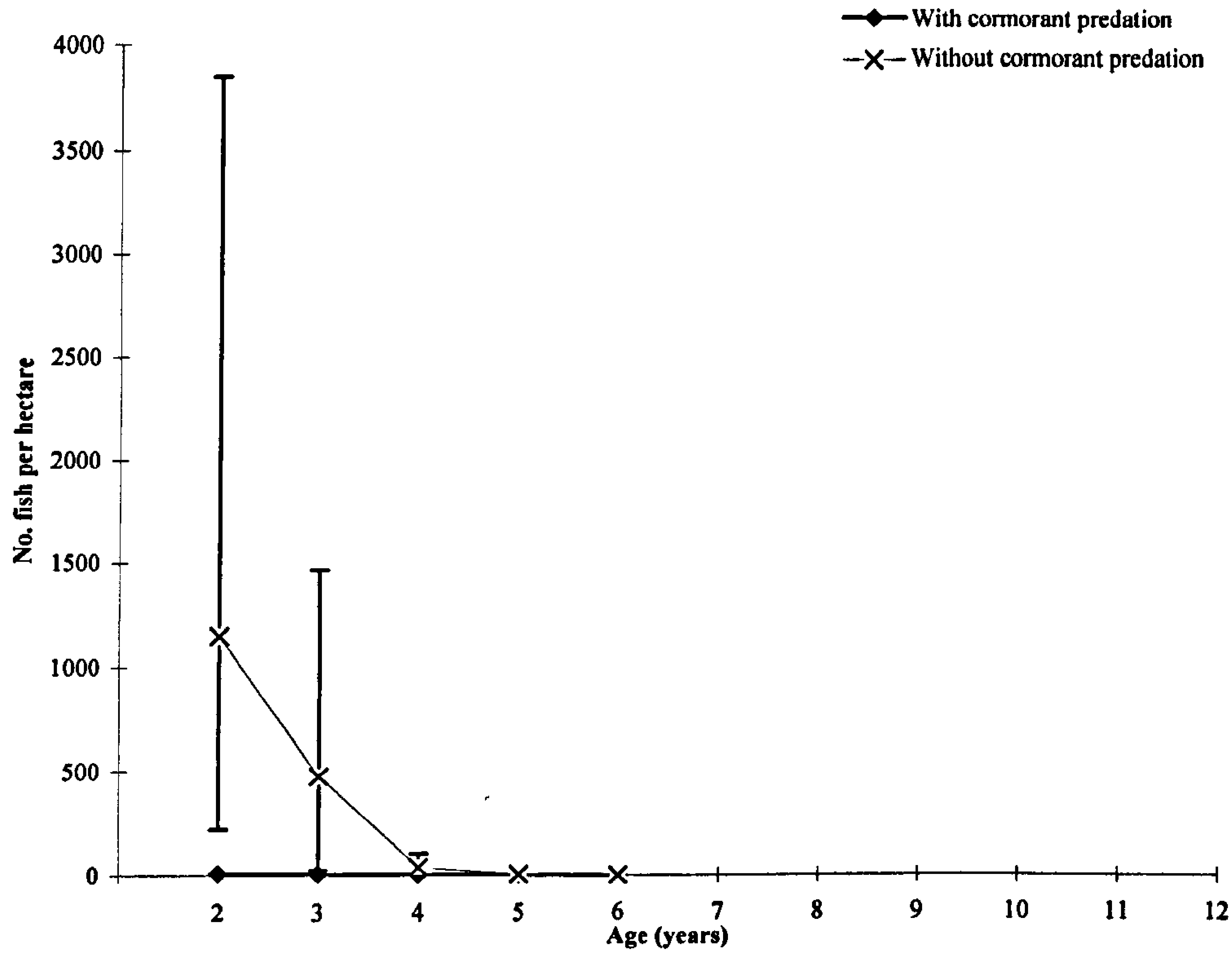


Figure 5.36 Impact of cormorant predation on the numbers of common bream at Beeston, River Trent, in 1997.

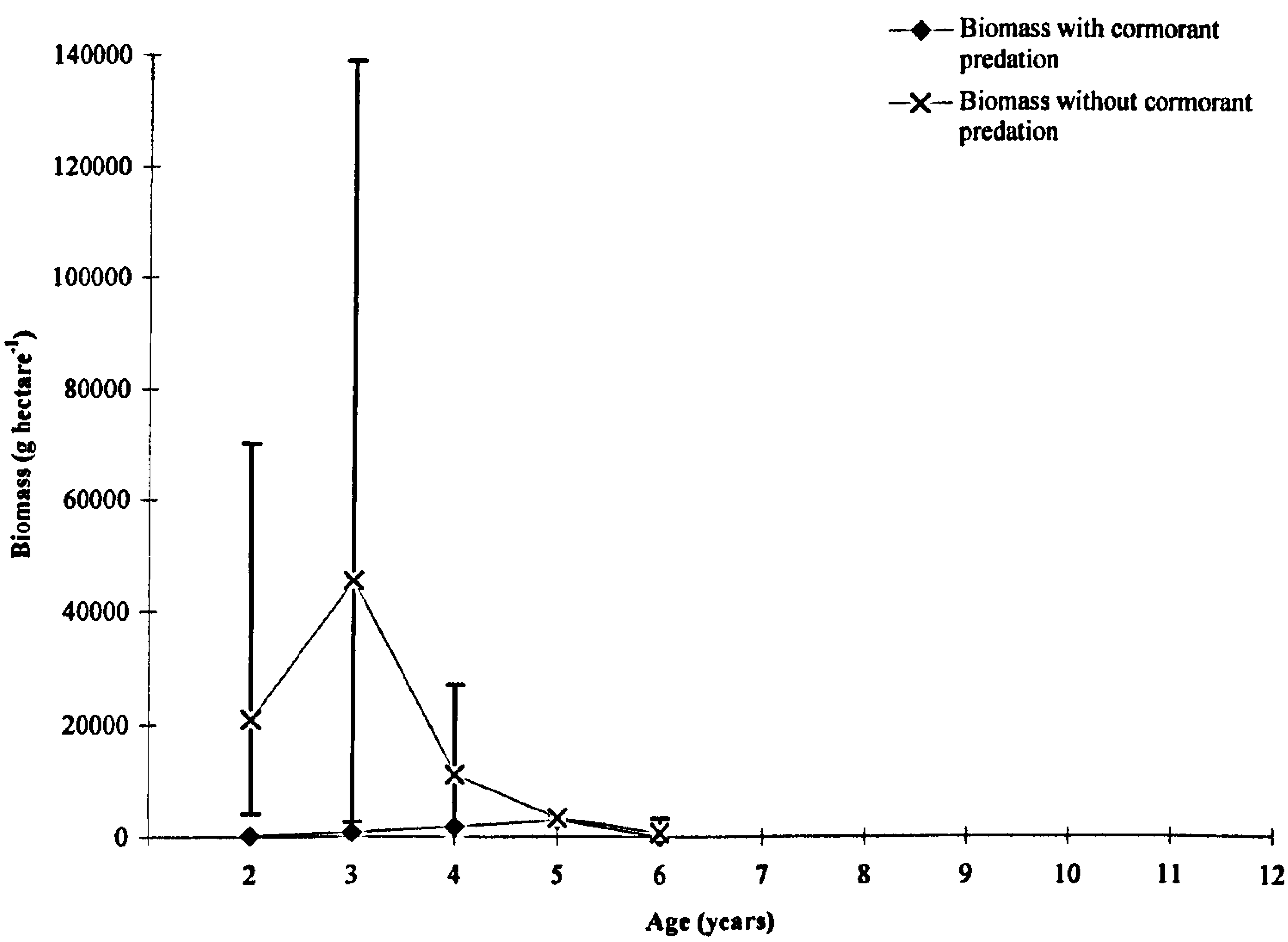


Figure 5.37 Impact of cormorant predation on the biomass of common bream at Beeston, River Trent, in 1997.

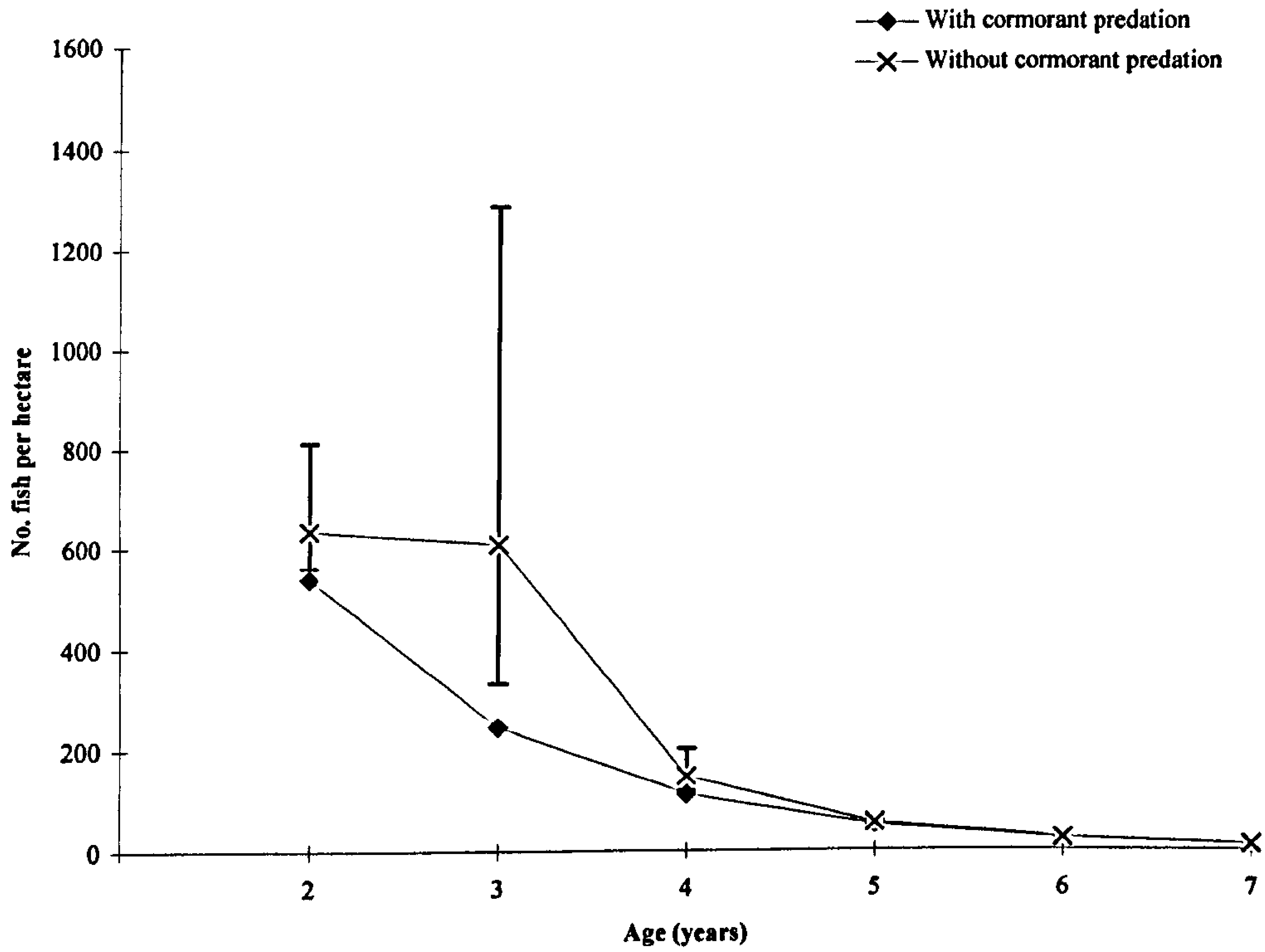


Figure 5.38 Impact of cormorant predation on the numbers of perch at Beeston, River Trent, in 1996.

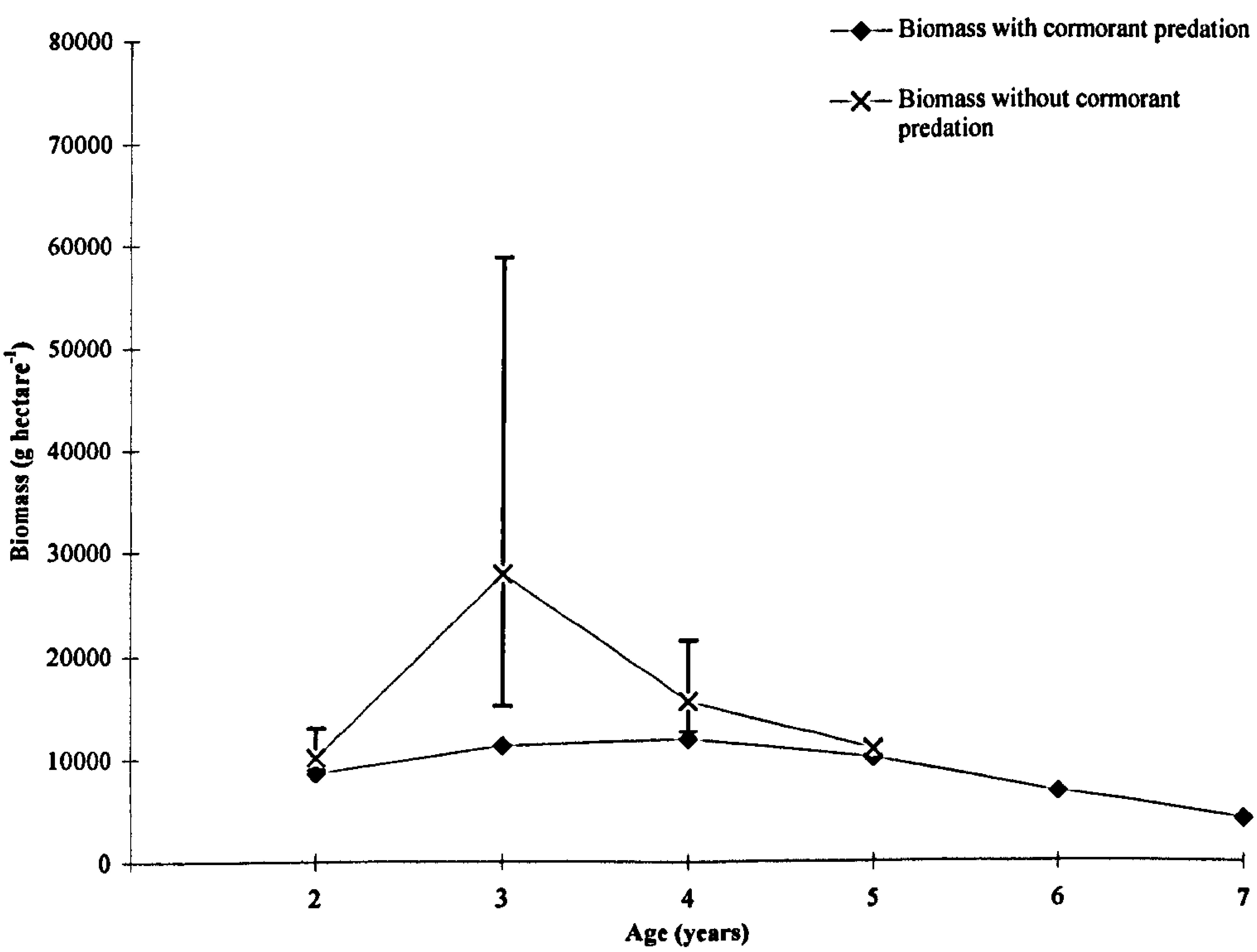


Figure 5.39 Impact of cormorant predation on the biomass of perch at Beeston, River Trent, in 1996.

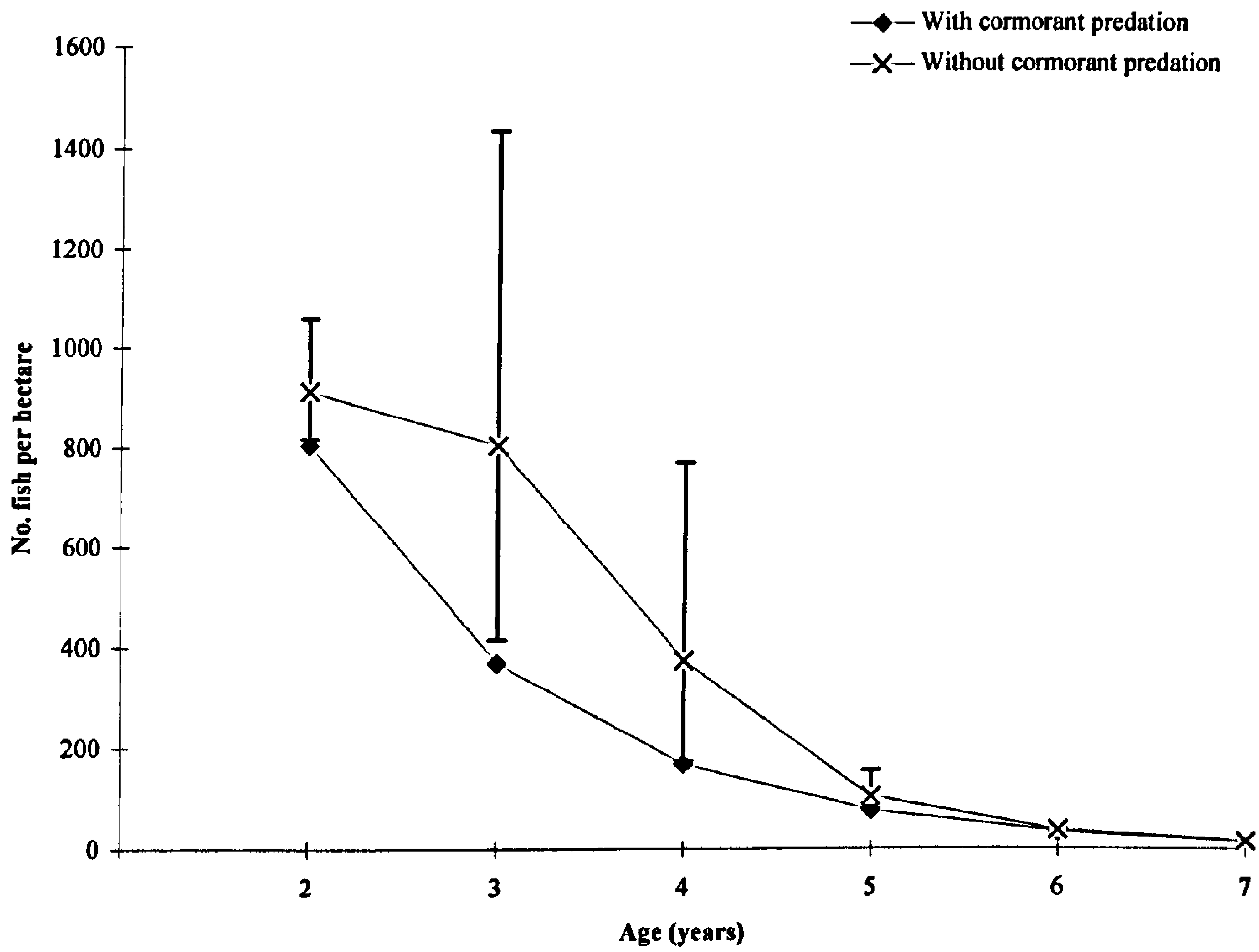


Figure 5.40 Impact of cormorant predation on the numbers of perch at Beeston, River Trent, in 1997.

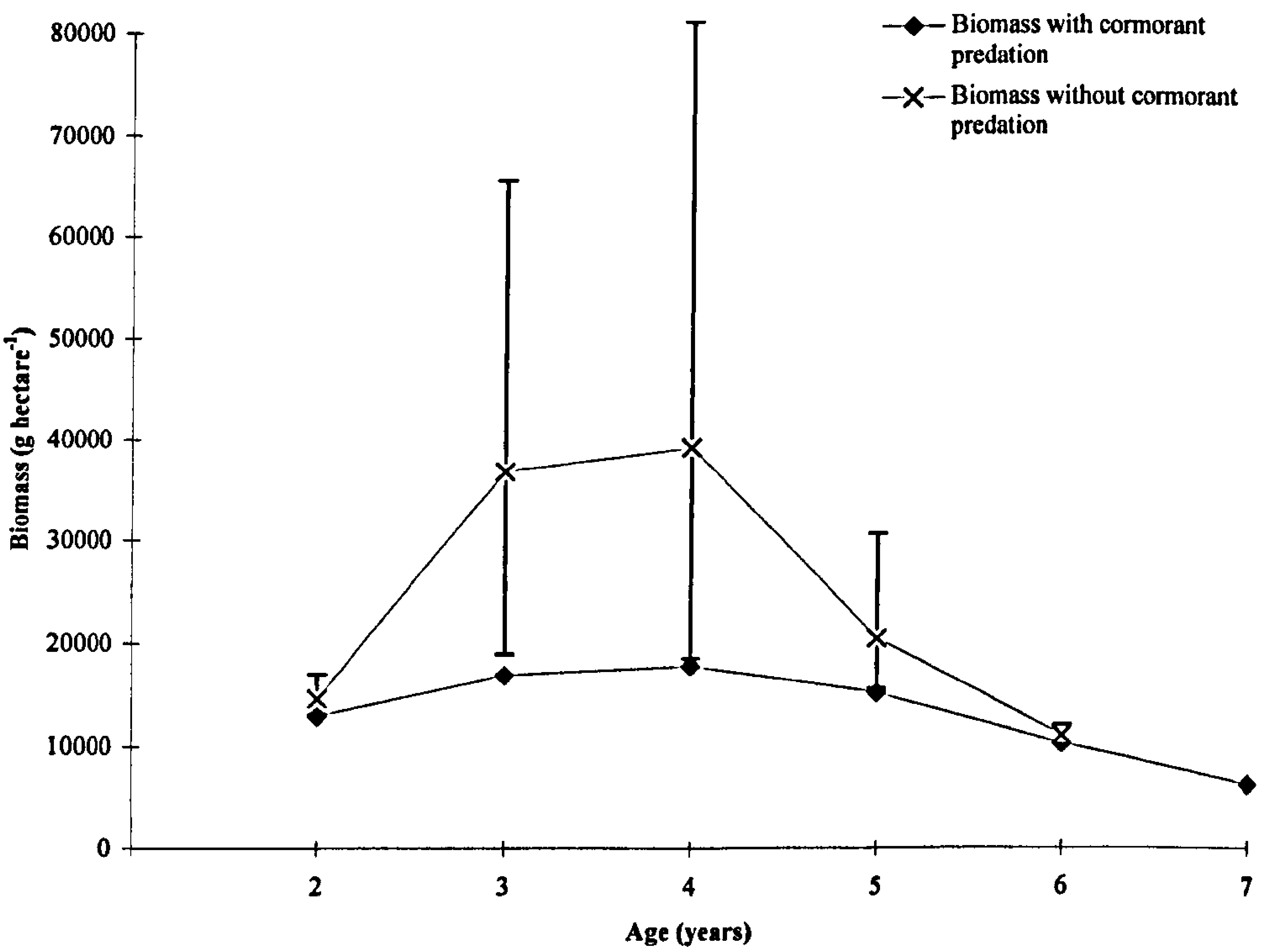


Figure 5.41 Impact of cormorant predation on the biomass of perch at Beeston, River Trent, in 1997.

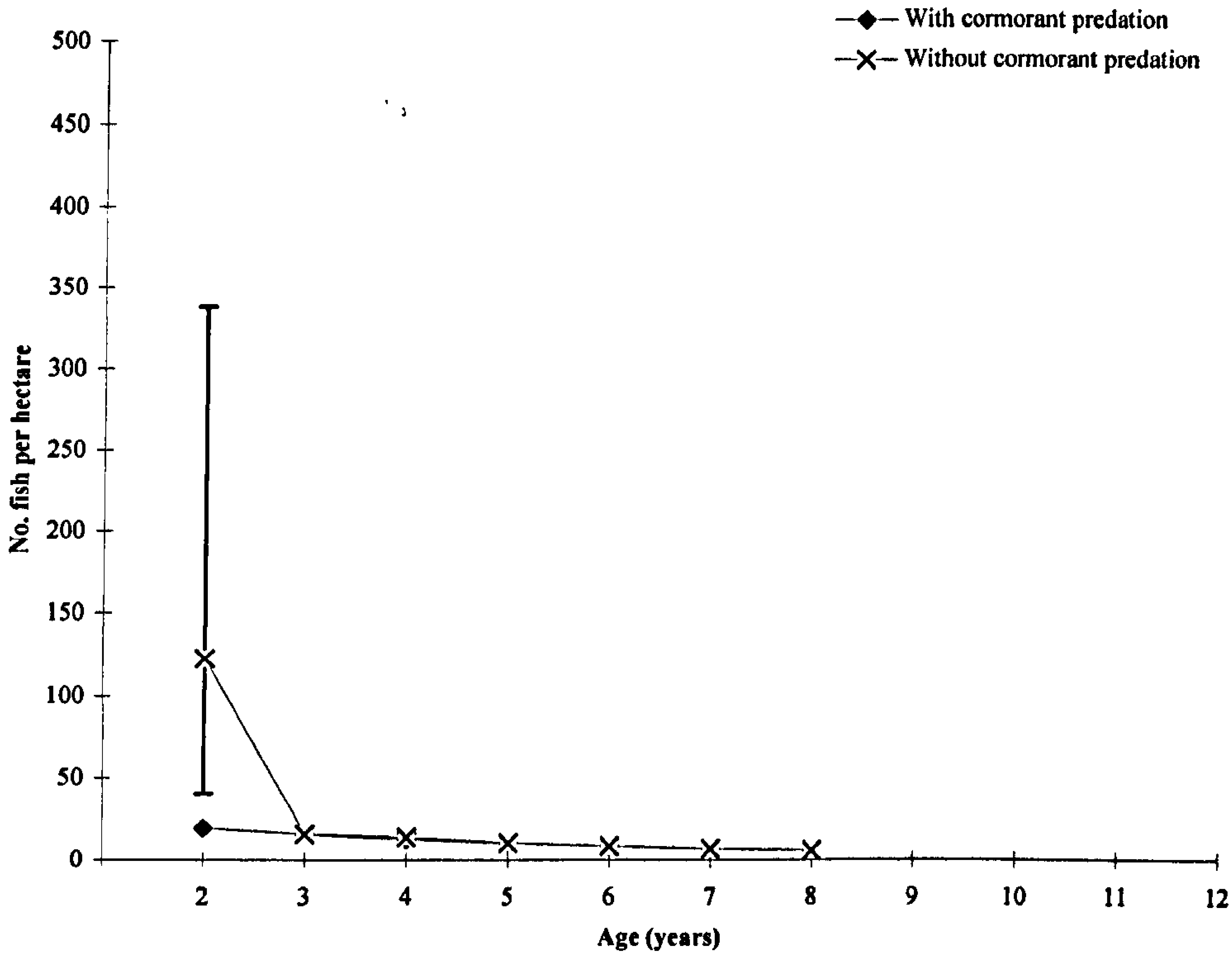


Figure 5.42 Impact of cormorant predation on the numbers of chub at Beeston, River Trent, in 1996.

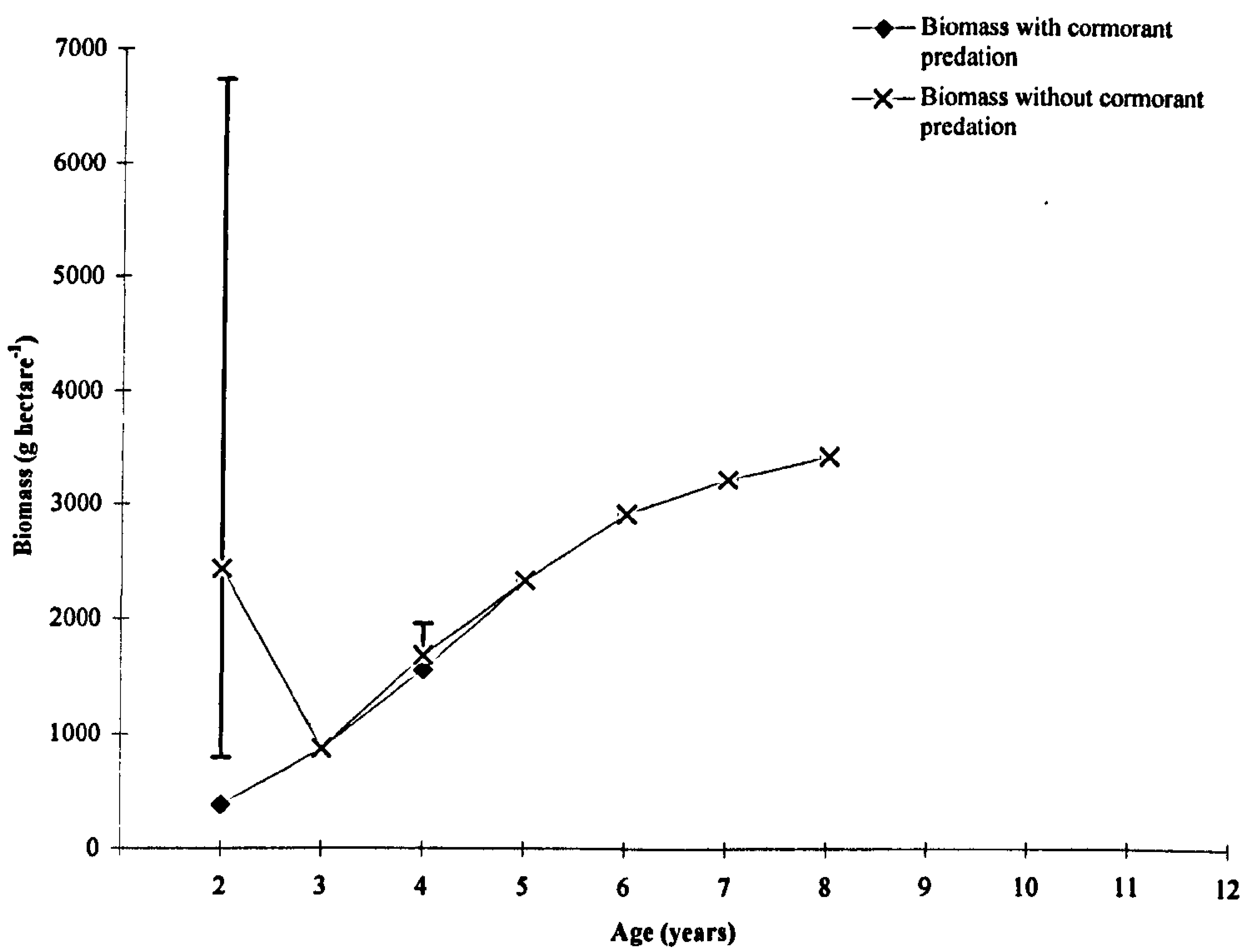


Figure 5.43 Impact of cormorant predation on the biomass of chub at Beeston, River Trent, in 1996.

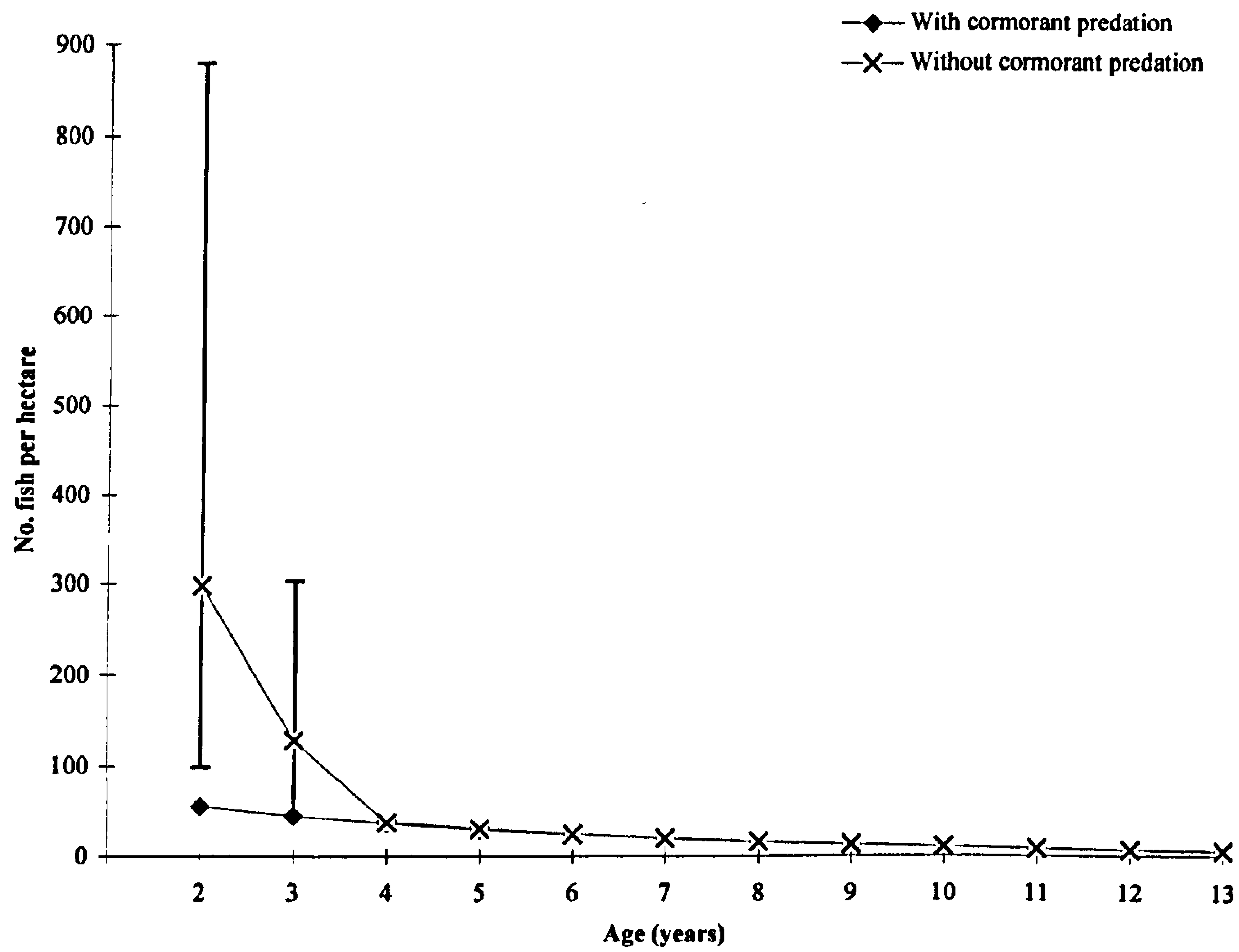


Figure 5.44 Impact of cormorant predation on the numbers of chub at Beeston, River Trent, in 1997.

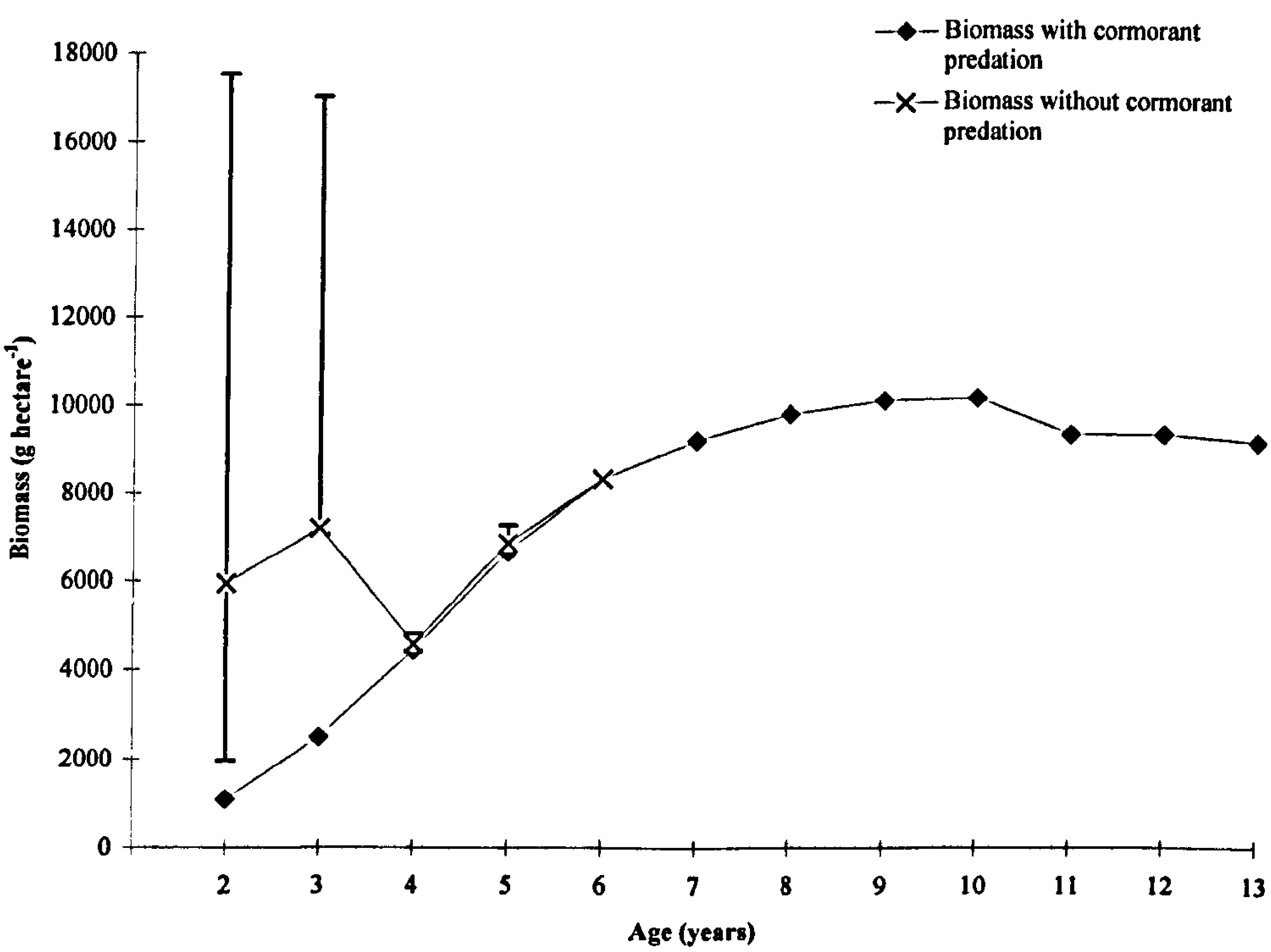


Figure 5.45 Impact of cormorant predation on the biomass of chub at Beeston, River Trent, in 1997.

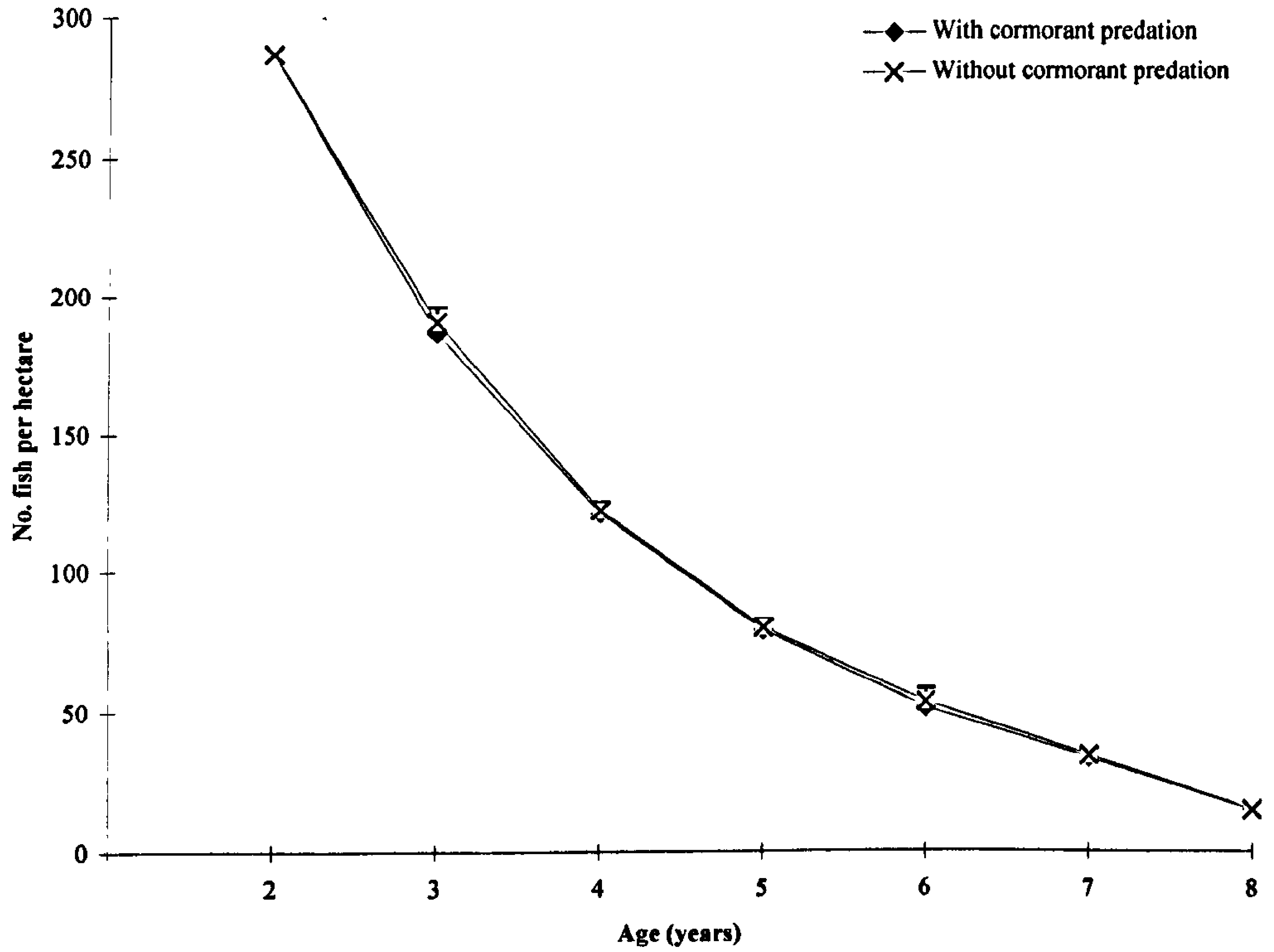


Figure 5.46 Impact of cormorant predation on the numbers of hybrids at Beeston, River Trent, in 1996.

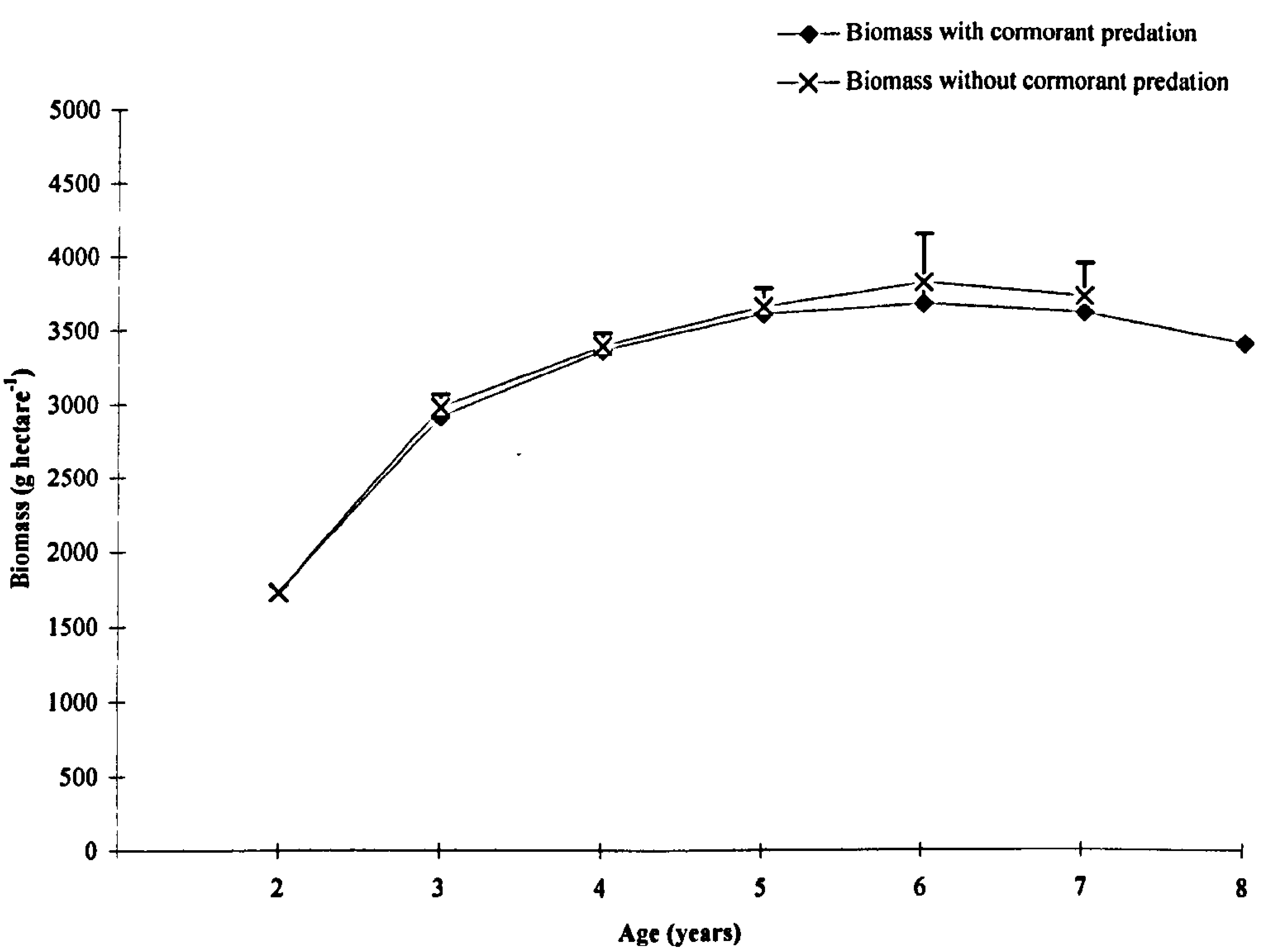


Figure 5.47 Impact of cormorant predation on the biomass of hybrids at Beeston, River Trent, in 1996.

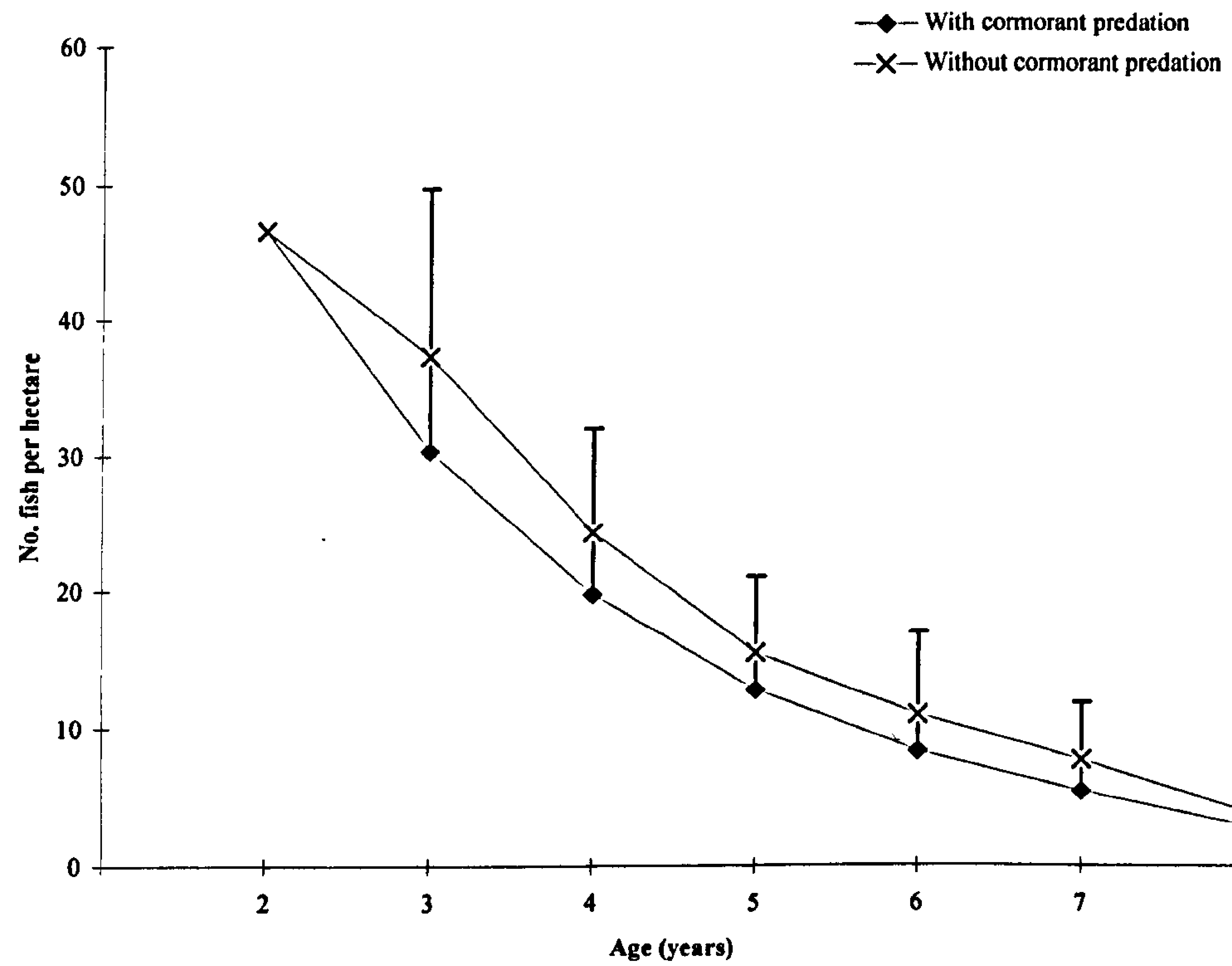


Figure 5.48 Impact of cormorant predation on the numbers of hybrids at Beeston, River Trent, in 1997.

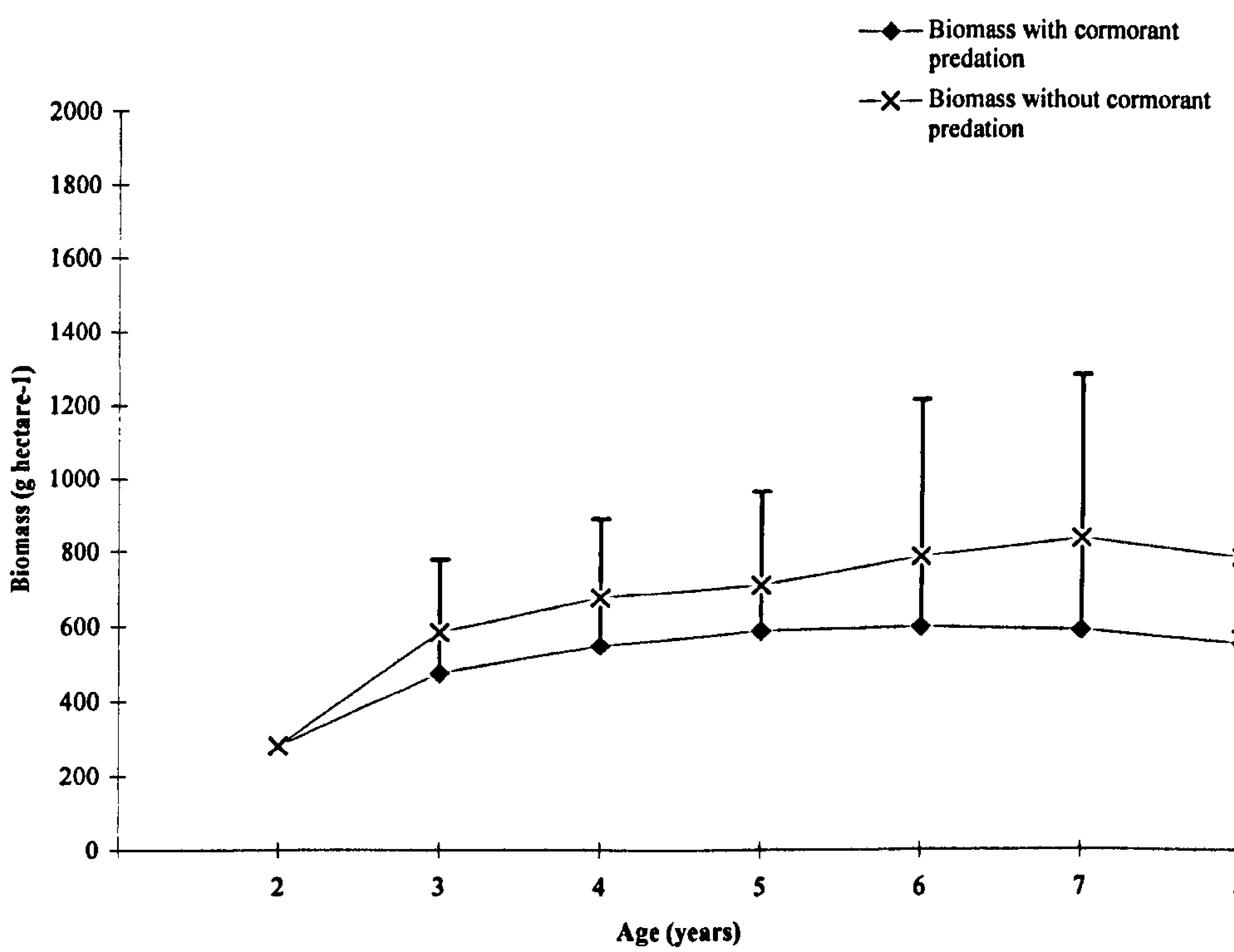


Figure 5.49 Impact of cormorant predation on the biomass of hybrids at Beeston, River Trent, in 1997.

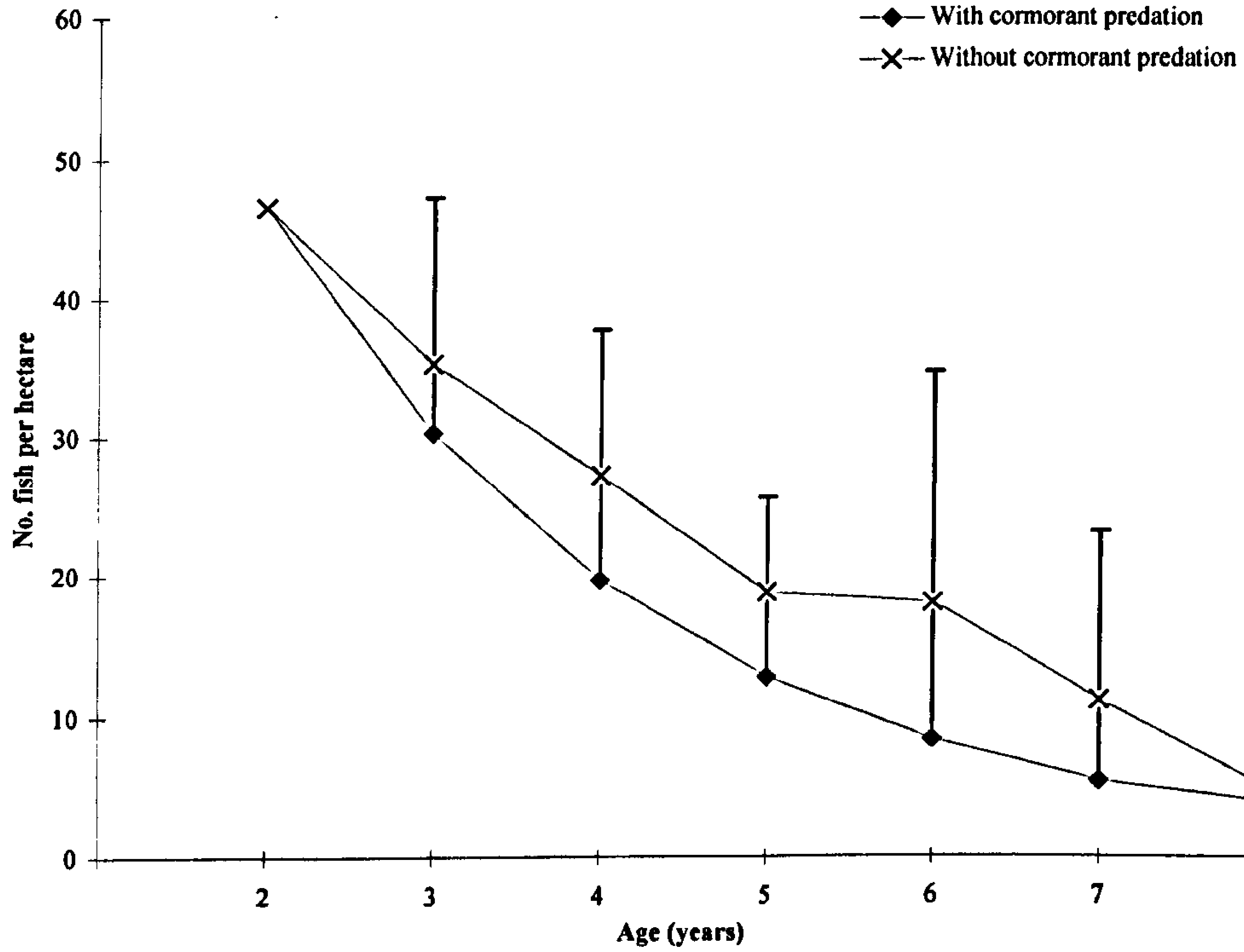


Figure 5.50 Impact of cormorant predation on the numbers of hybrids at Beeston, River Trent, in 1998.

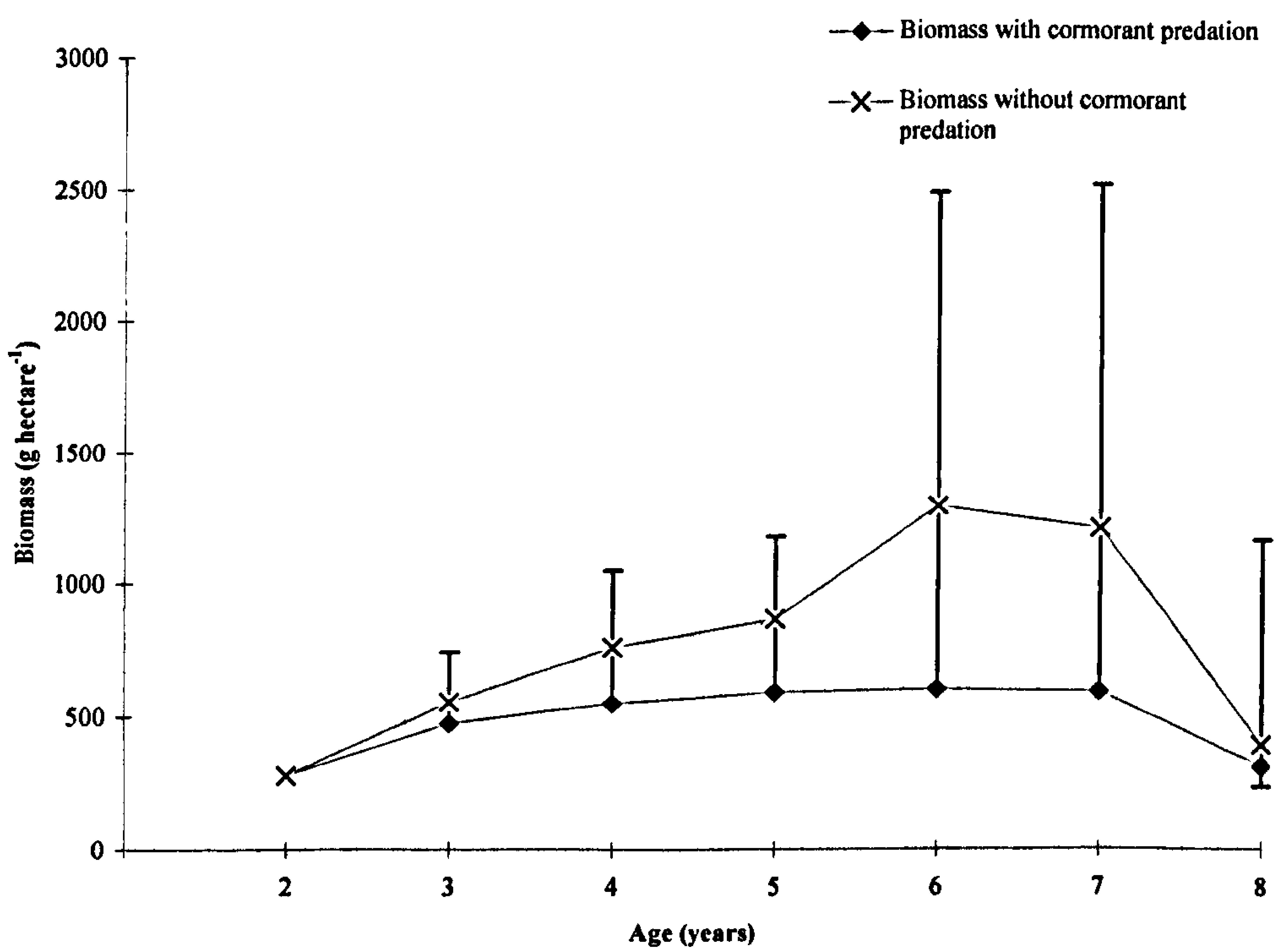


Figure 5.51 Impact of cormorant predation on the biomass of hybrids at Beeston, River Trent, in 1998.

5.6 Status of fish populations, cormorant populations and impact of cormorants on the River Trent at Trent Bridge

5.6.1 *Status of fish populations*

Standing crop

The standing crop of roach, bream and perch in the section (October 1995, 1996, 1997 and June 1998) showed inter-annual variation of 27.5 to 2.9 to 22.3 g m^{-2} (Table 5.13).

Table 5.13 Standing crop of species, estimated by electric fishing, at Trent Bridge, River Trent, over the study period.

	Standing crop (kg ha^{-1})		
Species	1995/96	1996/97	1997/98
Roach	65.21	29.42	49.23
Common bream	199.08	0	103.50
Chub	2.66	0	56.75
Hybrid	6.34	0	0.75
Perch	2.44	0	13.08
Total standing crop	275.73	29.42	223.30
Total standing crop g m^{-2}	27.5	2.9	22.3

Although the hydro-acoustic data provided only a minimum estimate of fish abundance in the section (Section 4.2.2), fish abundance changed with time for the section Clifton Bridge to Colwick Marina (Table 5.14).

Table 5.14 Density of fish present, estimated from hydro-acoustic surveys, in the Clifton Bridge to Colwick Marina section of the River Trent.

Year	Fish per hectare (n ha^{-1})
1994	845
1995 (June)	421
1995 (September)	419
1996	104
1997	83

Angler catch data

Match angler catch data for the angling seasons 1995/96 and 1996/97 were obtained for the stretch from Lady Bay Bridge downstream towards Holme Sluices controlled by Parkside Angling Club (Table 5.15). This area lies within the cormorant observation section and within the area sampled by electric fishing.

Table 5.15 Angler catch data for Parkside Angling Club 1995 to 1997.

	1995/96	1996/97
Total effort (man hours)	4487.75	5381.75
Total catch (kg)	1106.167	1087.612
Catch per unit effort (g man $hour^{-1}$)	246.49	202.10
% anglers with catch	85.30	88.97

The catch per unit effort ranged from 246 g man hour^{-1} to 202 g man hour^{-1} (Table 5.15), which is below the value recorded at Thrumpton in 1995/96 (Section 5.5.1). However, it compares favourably with Stoke Bardolph, River Trent (Jacklin 1996) and River Ouse (Axford 1991) (Section 5.5.1).

Species composition of the electric fishing surveys

Catches of fish were generally good in all winter electric fishing surveys because fish tended to shoal in the Trent Bridge area. It is known that fish migrate from the Holme Sluice and Long Higgin area of the section to Trent Bridge to over-winter, and possibly to spawn in this region during spring (T. Holden pers. comm.).

Roach were the dominant species, comprising between 48 and 81 % of catches (Figure 5.52). Common bream catches were generally very poor in all surveys and chub only exceeded 10 % catch composition in surveys of 1995/96. Poor catch contribution by chub was also observed at Beeston (Section 5.5.1). A population of hybrids, similar in appearance to the hybrids caught at Beeston (Section 5.5.1), was strongly represented in surveys between October 1997 and March 1998 (34 %). The hybrids often contributed a high proportion of angler catches at Trent Bridge (T. Holden pers. comm.).

Piscivorous fish were poorly represented in electric fishing catches, with perch and pike comprising a combined total of below 2 % of all catches between October 1995 and March 1998 (Figure 5.52). Perch comprised 15 % of catches in June 1998, but this may be a reflection of a poor catch of cyprinids. Few pike were observed in all surveys on the section.

The minor species represented in electric fishing samples were gudgeon, bleak, dace, tench and carp. Their combined catch composition did not exceed 8 % in any survey period.

Length frequency distribution of electric fishing surveys

The length frequency distribution of roach consisted of fish in the size range 90 to 290 mm (Figure 5.53). The population in 1995/96 was dominated by fish in the size range 50 to 70 mm and 150 to 190 mm. In 1996/97, the size ranges 80 to 110 mm and 150 to 170 mm were dominant in catches and in 1997/98, roach size classes of 120 to 170 mm roach were dominant (Figure 5.52). Hence, variation in the size range of sampled roach were observed over the study period, and reflects the growth of the strong juvenile cohort from 1995. The large number of individuals below 100 mm indicated a good recruiting population.

A sparse population of common bream was present in the stretch with no dominant size classes (Figure 5.54). However, this may be due to the difficulty of sampling common bream as they inhabit deep and slow flowing areas of the river. The catch of six large common bream was possibly due to their aggregation in shallow water to spawn coinciding with an electric fishing survey.

The population structure of chub was largely composed of fish in the size range 110 to 460 mm (Figure 5.55). Few juvenile chub were captured. The dominance of the larger fish suggests a spawning stock was present, but a potential recruitment bottleneck persists in the system. Fish in the size range 100 to 200 mm should not be subject to sampling bias and all typical chub shoaling areas were sampled in the study period.

The hybrid population was dominated by individuals in the size range 130 to 210 mm, with individuals present up to 270 mm (Figure 5.56). Few fish below 100 mm were observed, possibly due to the sampling bias of the electric fishing gear. The results indicated a recruiting population of hybrids were present which contributed a significant proportion of the cyprinid stock (Figure 4.52).

The length frequency distribution of perch caught by electric fishing revealed a poor population of various sizes up to 370 mm (Figure 5.57). This reveals the presence of a mature spawning stock.

Pike were generally present in low numbers with individuals able to attain sizes up to 770 mm.

Year class strength

Only the year class strength of roach was calculated as the data sets for the other species were inadequate. Good year classes of roach (Figure 5.58) were observed from 1989, 1990, 1991, 1992 and 1995. Poor year classes were observed from 1993 and 1994, a pattern also observed at Beeston (Section 5.5.1).

The temperature profile for the period, expressed as the annual number of degree days above 14°C, was correlated with the year class strength ($r = 0.82$). Hence, the data suggested roach year class strength was dependent on water temperature at Trent Bridge, similar to that at Beeston (Section 5.5.1).

Growth

The growth rates of roach, common bream, perch and chub were compared with national standards (Hickley and Dexter 1979; Cowx *et al.* 1995). Roach growth was average, with the maximum age attained being 8+ (Figure 5.59). Common bream growth rates were considered good, with growth above the national standard (Hickley and Dexter 1979) (Figure 5.60). This was also shown at Beeston (Section 5.5.1). The maximum age of the common bream was 12, showing reasonable longevity of the population (Figure 5.60).

Perch growth rates were also above average, although they were below rates considered fast (Figure 5.61). Perch up to 5 years were found, although older perch were probably present in electric fishing catches which were unable to be aged accurately. Chub growth rates were average, with fish up to 12 years of age (Figure 5.62).

The growth rates for bleak and gudgeon were similar to those from other UK waters with a similar life span (Table 5.16). This is in contrast to bleak and gudgeon at Beeston where no fish were greater than 2+ in age (Section 5.5.1). The hybrids captured at Trent Bridge were similar in appearance to the hybrids captured at the other River Trent sites. Some confusion exists over their parentage and so it is difficult to compare growth rates with other hybridised populations. However, it is clear that they were slow growing and did not attain large sizes.

Table 5.16 Growth rate of bleak, gudgeon and hybrids at Trent Bridge, River Trent, compared with other UK rivers.

		Length at age (mm)								
Species	**Location**	**I**	**II**	**III**	**IV**	**V**	**VI**	**VII**	**VIII**	**IX**
Bleak	Trent Bridge	46	85	100	121	144				
	Standard	42	76	107	120	125	147			
	River Severn	41	75	104	120	127				
	River Thames	39	70	92	111	121	130	138	140	
Gudgeon	Trent Bridge	48	80	99	112	124				
	River Thames	55	86	104	117	123	129			
	River Frome	42	96	120	137	152				
Hybrid	Trent Bridge	65	97	118	136	167	179	190	202	210

(Williams 1967; Mann 1973; Cowx *et al.* 1995)

L infinity and K

Values obtained from the Von Bertalanffy growth model for roach, common bream, perch and chub (Table 5.17) were similar to those derived for the species in other UK fisheries (Tables 4.4 to 4.6).

Table 5.17 L_∞, and K values derived for roach, common bream, perch and chub at Trent Bridge, River Trent.

Species	L_∞	K
Roach	362	0.16
Common bream	530	0.14
Perch	303	0.19
Chub	538	0.11

Mortality and survival rates

The mortality (Z) and survival (S) rates for various species at Trent Bridge (Table 5.18) were relatively low and high, respectively, compared with other UK fisheries (Table 5.5), indicating good annual survival of cohorts.

Table 5.18 Mortality and survival rates of species at Trent Bridge, River Trent.

Species	Z	S	If N_t=1000, N_{t+1}=
Roach	0.48	0.619	619
Common bream	0.23	0.795	795
Chub	0.41	0.664	664
Perch	0.34	0.712	712
Hybrid	0.26	0.771	771

5.6.2 *Status of cormorant populations*

The status of the cormorant populations at Trent Bridge was assessed by monitoring Section C, (Section 5.2), which covered the area used in the fisheries surveys.

Temporal roost occupancy

The cormorants utilising the Trent Bridge section occupied the main Attenborough night roost (Section 4.5.1, Figure 4.13).

Cormorant site occupancy

The number of cormorants observed on the section increased over the study period, from a peak of 42 birds in 1995/96 to a peak of 155 birds in 1997/98 (Figure 5.63). The majority of the cormorants used the section as a 'fly-way', with a smaller proportion using the site for feeding. Very few birds were observed using the site for day roosting and loafing (Figure 5.63).

Feeding success

The foraging bout success rate of cormorants on Section B was 42.9 to 50 % of bouts resulting in consumed fish (Table 5.19). This was similar to that observed for Sections A and B (Table 5.6), below values obtained for Holme Pierrepont (Section 4.5.4) and above those in the pre-trout stocking period at Colwick Park Trout Lake (Section 6.5.4).

The proportion of dives that was successful was low in all years for Sections C with between 7.5 and 9 % of dives being successful (Table 5.19). This was similar to dive success on Sections A and B (Table 5.6), lower than Holme Pierrepont (12.7 to 15.2 %, Section 4.5.4) and overlaps with Colwick Park Trout Lake (1.8 to 13.9 %, Section 6.5.4).

Table 5.19 The feeding success of cormorants at count section C (Trent Bridge), River Trent.

	1995/96	1996/97	1997/98
Observed foraging bouts	14	68	123
Successful foraging bouts	6	34	61
% foraging bouts successful	42.9	50.0	49.6
Dives observed	140	615	1146
Successful dives	12	46	103
% dives successful	8.6	7.5	9.0

5.6.3 *Impact of cormorants at Trent Bridge*

Species composition of fisheries surveys, angler catches and cormorant predation

Observations of cormorant feeding behaviour did not allow the identification of fish predated on to species level. Fish were classified as cyprinids and perch. Accordingly, all fisheries data collected from electric fishing surveys and angler catch returns were reclassified as cyprinids, perch and pike to allow direct comparison.

Cyprinids were the dominant species group present in all electric fishing catches (Figure 5.64) while pike and perch comprised a small proportion of catches. Cyprinids were the main prey items selected by cormorants during feeding bouts, reflecting their dominance in the fish populations. However, cormorants selected a greater proportion of perch in their diet than were represented in the population. In 1995/96, of 12 fish consumed, 50

% were perch and in 1996/97 and 1997/98, where larger numbers of fish were consumed, perch comprised 13 and 11 % of consumed fish. Therefore, cormorants actively selected perch during feeding.

Angler catches in the period were also dominated by cyprinid species (Figure 5.64), with a small proportion of perch captured. However, the proportion of perch in anglers' catches was higher than in the population but below the contribution observed in cormorant feeding observations.

Length frequency distribution

Cormorant predation of cyprinids was dominated by fish in the size range 50 to 200 mm (Figure 5.65). Cyprinids below 50 mm and up to 250 mm were also taken. The size composition of cyprinids caught by electric fishing was dominated by fish up to 200 mm with smaller numbers of cyprinids up to 500 mm present (Figure 5.65). The analysis shows the dominant size classes of the cyprinid populations were selected by cormorants during feeding.

Integration with growth rate data for the main cyprinid species (Figure 5.59, 5.60, 5.62; Table 5.16) allowed the approximate age that fish were susceptible to cormorant predation to be predicted. The roach and hybrid populations were exposed to cormorant predation over their whole life span, while bream and chub were vulnerable up to 5 years of age. This contrasted with cyprinids at Holme Pierrepont, which were vulnerable to cormorant predation up to a maximum age of two, due to extremely fast growth rates (Figures 4.8, 4.11).

Cormorants actively selected perch in their diet at Trent Bridge, River Trent. The sizes of cormorant predated perch were 50 to 200 mm (Figure 5.66). Perch were present in the section up to 370 mm. Hence, a proportion of perch in the section was not susceptible to cormorant predation. Growth analysis (Figure 5.61) shows perch were vulnerable to predation up to the age of five years.

Growth indices

The growth indices at Holme Pierrepont indicated an improvement in growth rate of roach from 1994 (Figure 4.20). This was tentatively linked to the appearance of cormorants on the fishery decreasing stock density, decreasing competition and allowing the increased growth.

The annual growth of roach at Trent Bridge varied between years and showed no overall pattern (Figures 5.67 to 5.68). Comparison of the growth indices and year class strength of roach at Trent Bridge (Figures 5.67, 5.68 and Figure 5.58) showed a general trend of good growth years corresponding to years of strong year class. Hence, 1991, 1992 and 1995 produced strong year classes of roach and good relative growth rate. Year class strength of roach was related to temperature (Figure 5.58). Thus, the growth of roach at Trent Bridge appeared to be related to annual temperature and independent of the influence of cormorant predation.

Cohort reconstruction with and without cormorant predation.

Cormorant predation on the populations of roach, common bream, chub, hybrid and perch revealed that a considerable quantity by number and biomass of the species present were removed (Tables 5.20 to 5.23). The difference in the values with and without cormorant predation can only be attributable to cormorant predation and not any other source of mortality.

Table 5.20 Numbers and biomass of roach with and without cormorant predation on Trent Bridge, River Trent.

		Number ($n\ ha^{-1}$)	% reduction	Biomass ($g\ ha^{-1}$)	% reduction
1996	With cormorants	2016.2		65212.5	
	Without cormorants	2353.2	14.3	71287.5	8.5
1997	With cormorants	803.2		29418.0	
	Without cormorants	4709.8	82.9	68387.9	57.0
1998	With cormorants	1346.1		49232.4	
	Without cormorants	5616.4	76.0	142868.3	65.5

Table 5.21 Numbers and biomass of common bream with and without cormorant predation on Trent Bridge, River Trent.

		Number ($n\ ha^{-1}$)	% reduction	Biomass ($g\ ha^{-1}$)	% reduction
1996	With cormorants	519.5		199076.0	
	Without cormorants	526.5	1.3	199562.2	0.2
1998	With cormorants	445.3		103496.8	
	Without cormorants	628.8	29.2	141046.0	26.6

Table 5.22 Numbers and biomass of hybrids with and without cormorant predation on Trent Bridge, River Trent.

		Number ($n\ ha^{-1}$)	% reduction	Biomass ($g\ ha^{-1}$)	% reduction
1996	With cormorants	181.9		6336.0	
	Without cormorants	190.9	4.7	6566.6	3.5
1998	With cormorants	29.5		747.9	
	Without cormorants	142.89	79.4	8731.3	91.4

Table 5.23 Numbers and biomass of perch with and without cormorant predation on Trent Bridge, River Trent.

		Number ($n\ ha^{-1}$)	% reduction	Biomass ($g\ ha^{-1}$)	% reduction
1996	With cormorants	48.4		2440.2	
	Without cormorants	416.4	88.4	12682.5	80.8
1998	With cormorants	159.4		13075.8	
	Without cormorants	865.8	81.6	51715.9	74.7

After three winters feeding, cormorants removed large numbers and biomass of fish from the Trent Bridge section of the River Trent, equivalent to a reduction in number of 76 % roach, 29.2 % common bream, 81.6 % perch and 79.4 % hybrids. Figures 5.69 to 5.86

show how the cormorant predation affected the age groups of the species between 1996 and 1998 both in terms of numbers and biomass.

A large difference was observed between the fish biomass and numerical abundance in the fishery found by electric fishing, and the cormorant predated fish biomass and numerical abundance estimated by the MCS model (Tables 5.20 to 5.23, Figures 5.69 to 5.86). However, the abundance values caught by electric fishing were considered reasonably accurate, as the minimum estimate calculated from hydro-acoustic data revealed lower fish abundance (83 to 845 fish per hectare, Table 5.14). Cormorants were estimated to remove 338 to 4270 of roach per hectare annually from 1995 to 1998, with other species also experiencing heavy losses. This would appear to be an over-estimation of losses, which is discussed further in Section 5.8.

When the fish population minimum and maximum values were considered, estimated from electric fishing gear calibration data (P = 0.10 and P = 0.36) (Section 3.3.1), it can be seen cormorant predation had a major effect on the fish populations even where maximum fish abundance estimates were calculated (P = 0.10).

Table 5.24 Impact of cormorant predation (CP) on the age classes of roach, common bream, hybrids and perch (n ha^{-1}) over the range of electric fishing gear efficiency at Trent Bridge, River Trent.

		1996		1997		1998	
Species	**P**	**CP**	**-CP**	**CP**	**-CP**	**CP**	**-CP**
Roach	0.10	4431	4768	1847	5754	3096	7366
	0.23	2016	2353	803	4710	1346	5616
	0.36	1231	1568	513	4420	860	5130
Common bream	0.10	1195	1202			1024	1521
	0.23	520	527			445	942
	0.36	332	339			285	781
Hybrid	0.10	414	423			63	179
	0.23	182	191			28	143
	0.36	118	127			18	133
Perch	0.10	111	479			367	1073
	0.23	48	416			159	866
	0.36	31	399			102	808

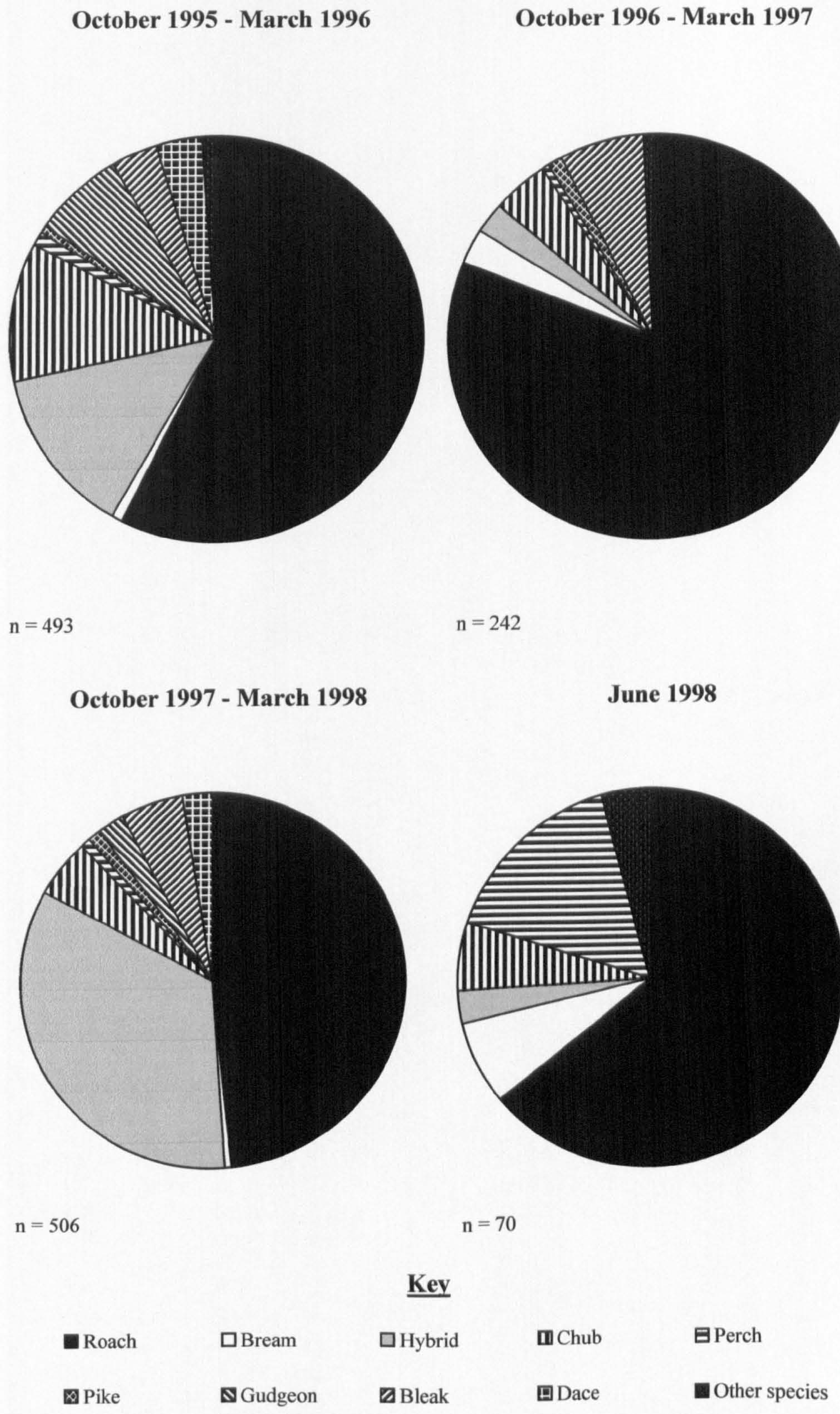


Figure 5.52 Species composition of electric fishing surveys at Trent Bridge, River Trent.

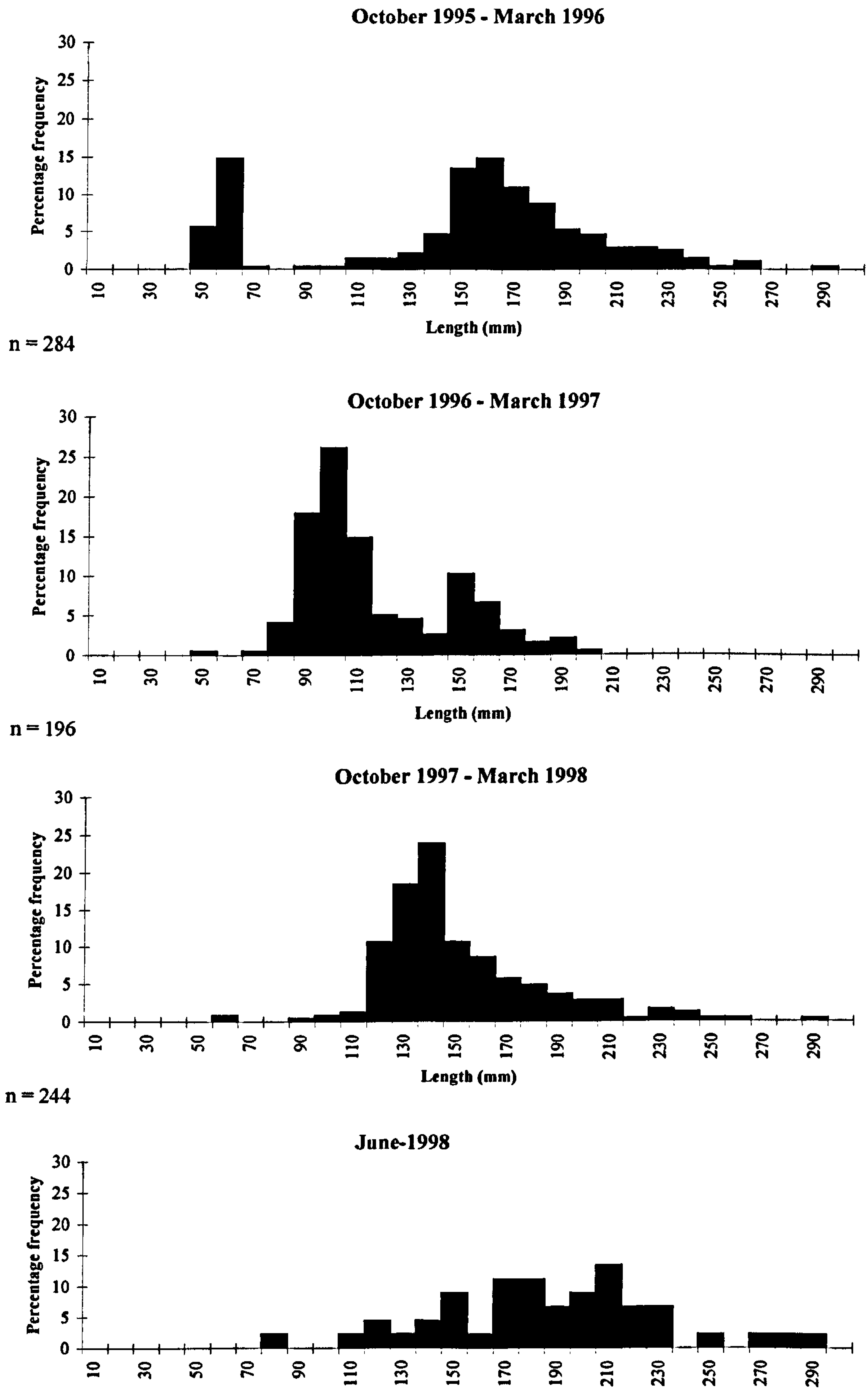


Figure 5.53 Length frequency distribution of roach caught by electric fishing at Trent Bridge, River Trent.

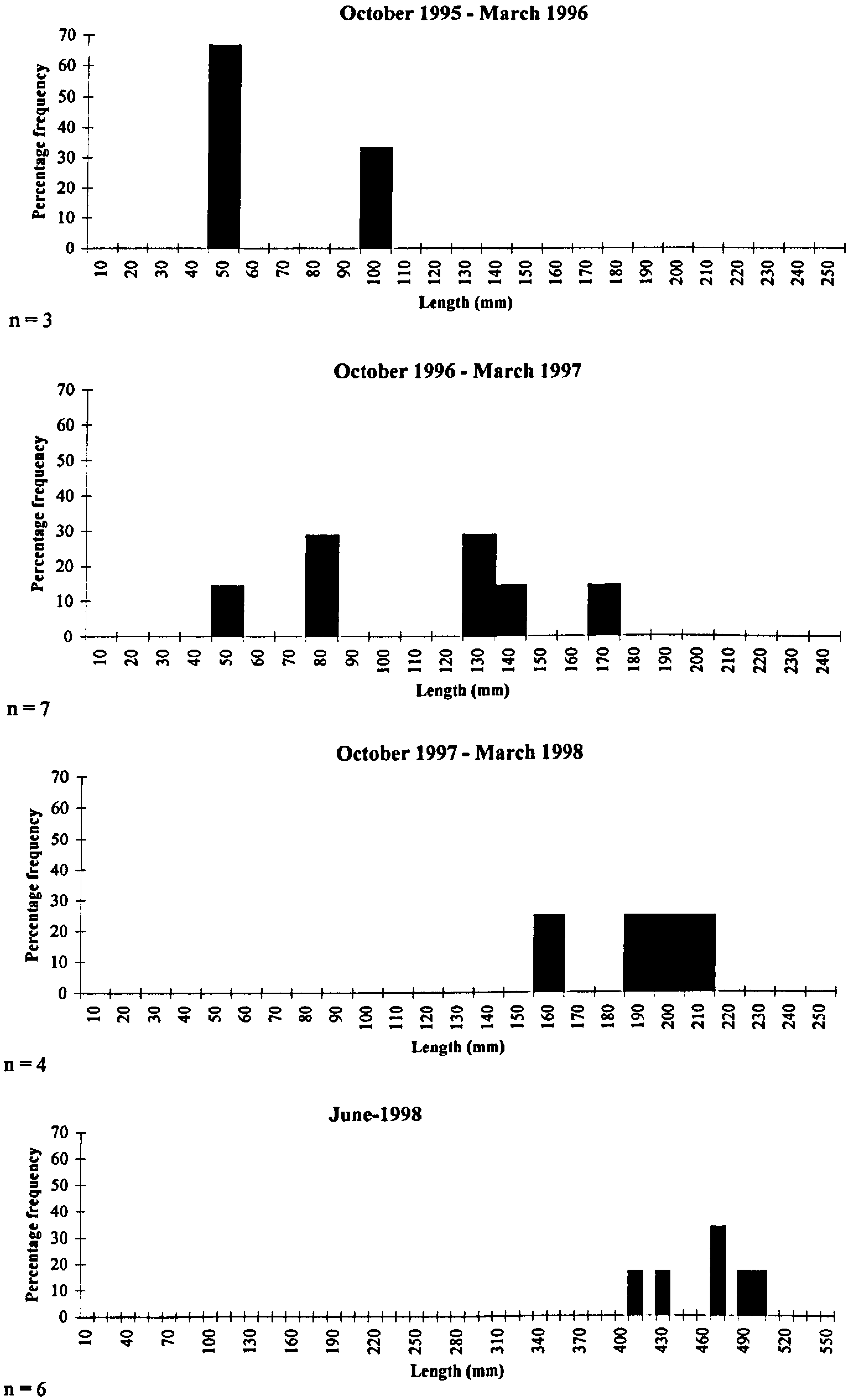


Figure 5.54 Length frequency distribution of common bream caught by electric fishing at Trent Bridge, River Trent.

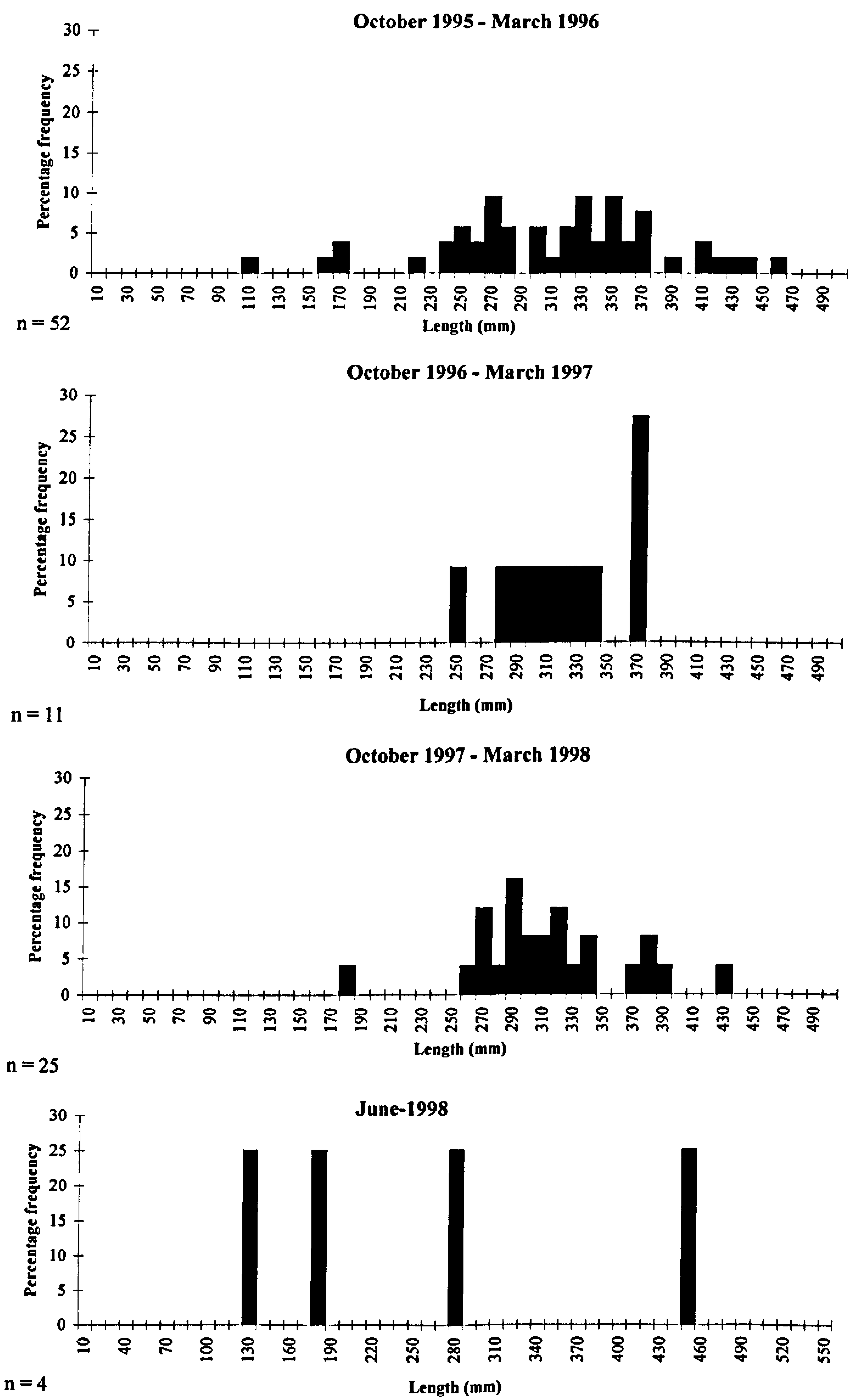


Figure 5.55 Length frequency distribution of chub caught by electric fishing at Trent Bridge, River Trent.

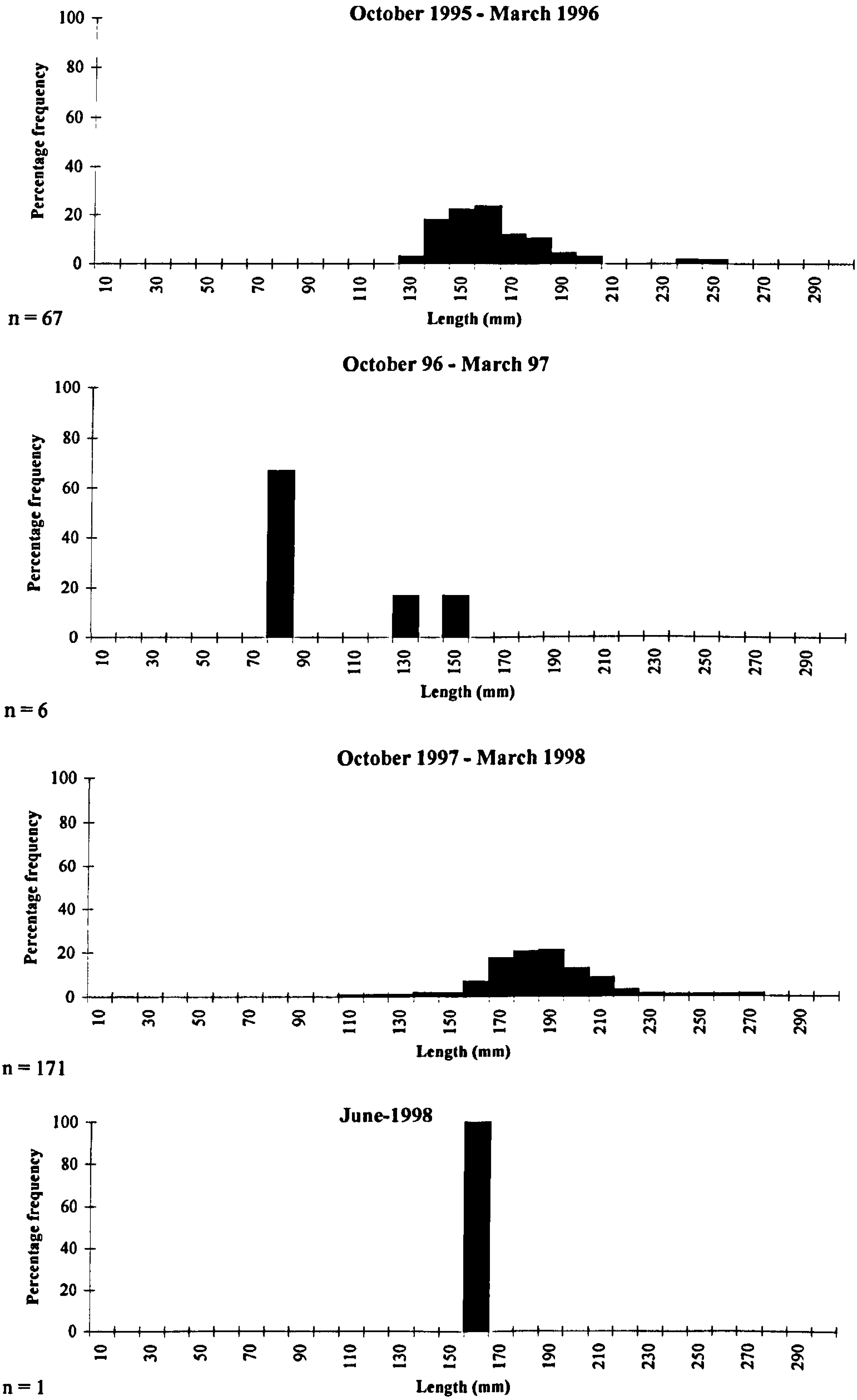
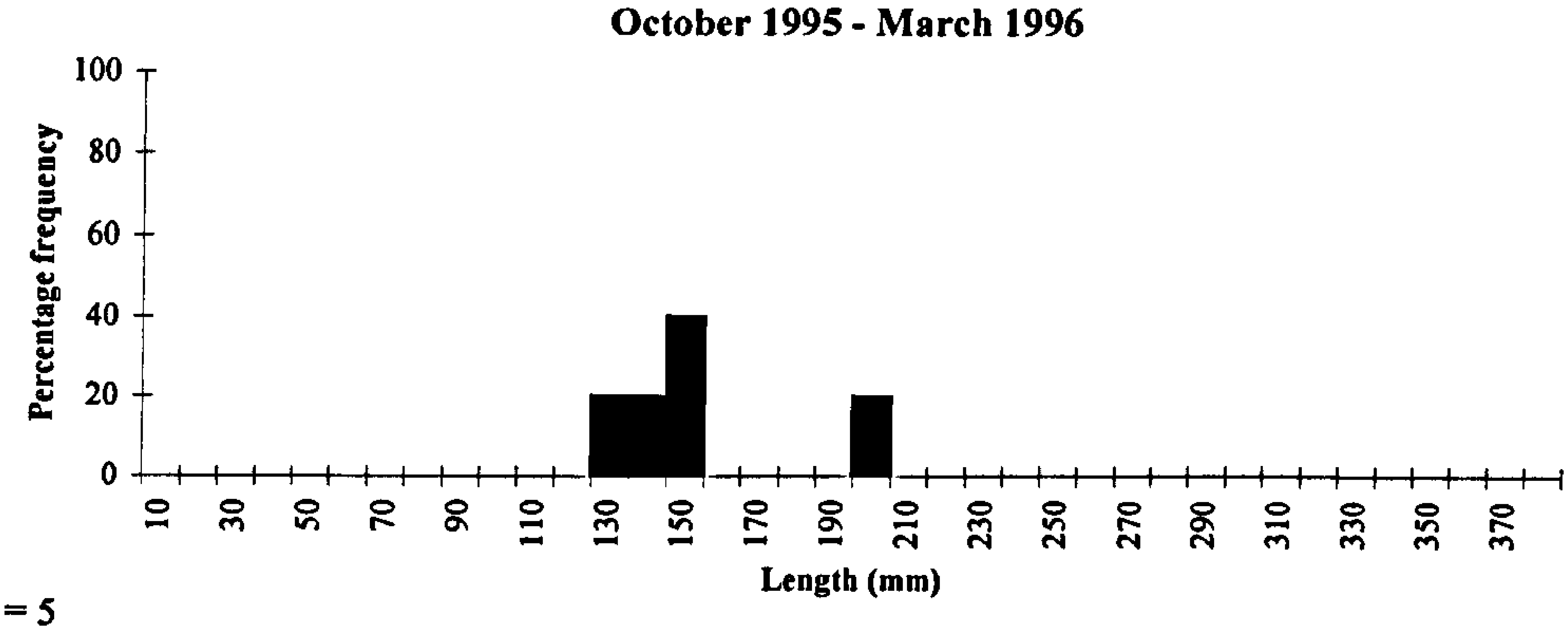
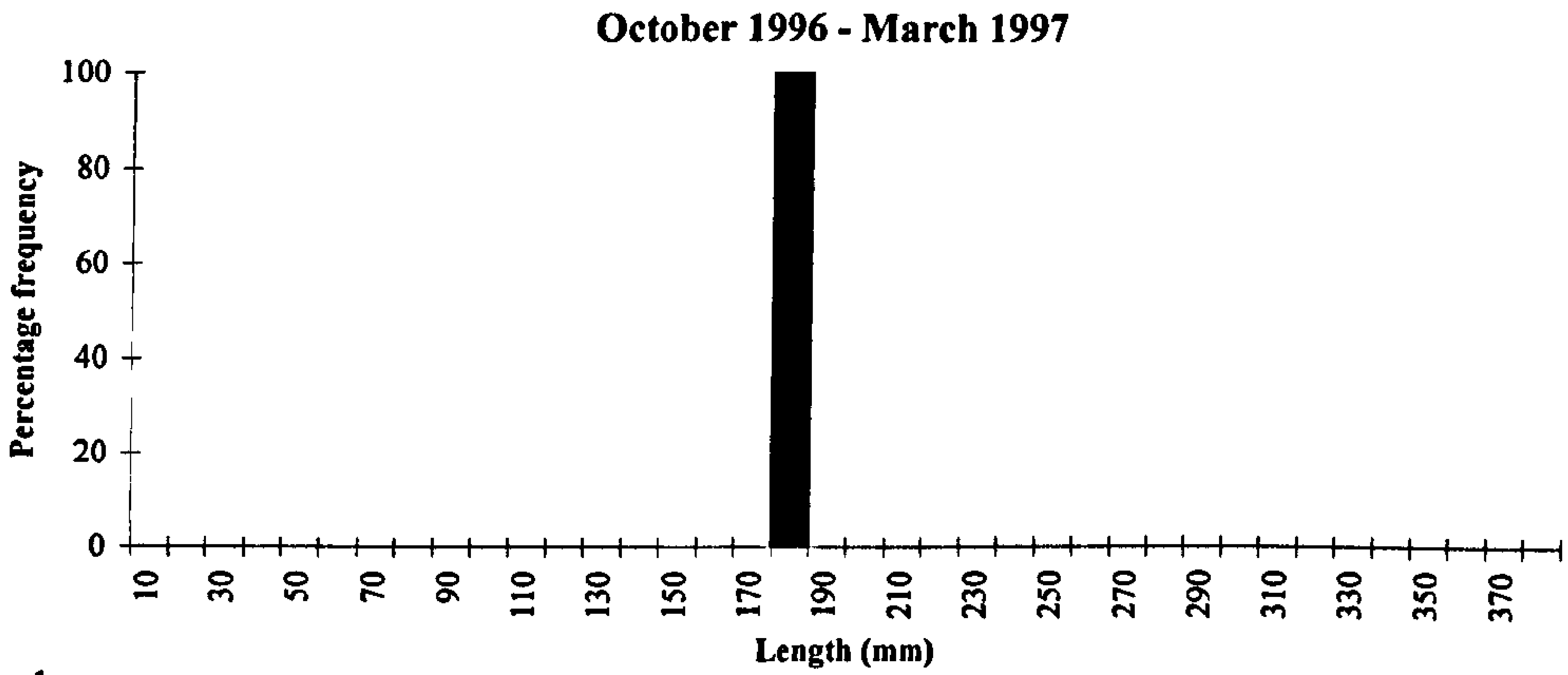


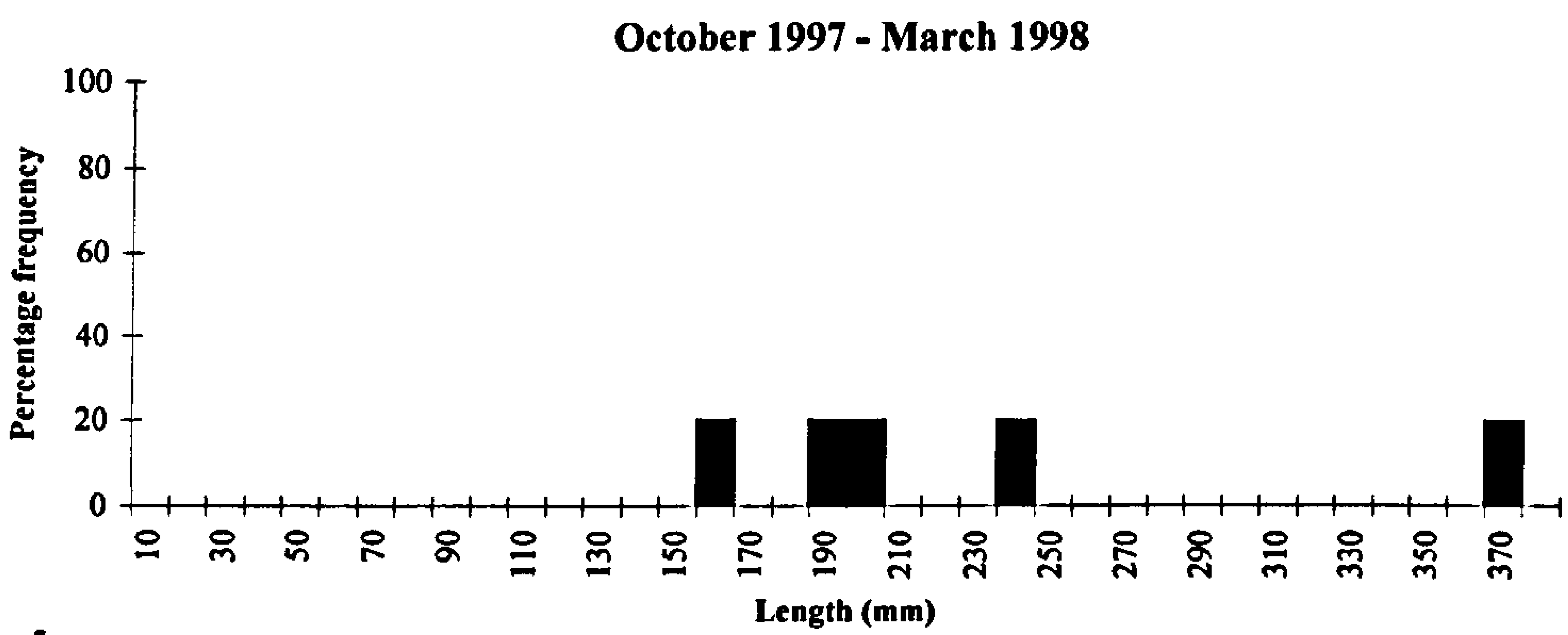
Figure 5.56 Length frequency distribution of hybrids caught by electric fishing at Trent Bridge, River Trent.



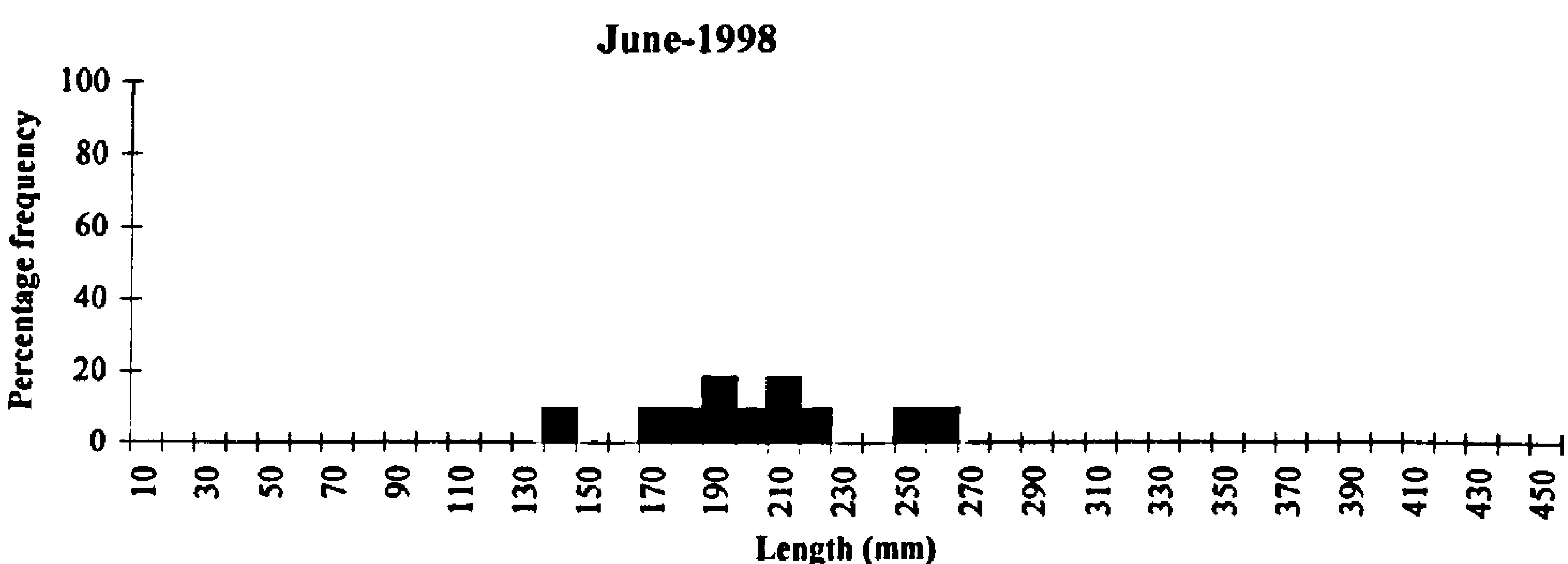
n = 5



n = 1



n = 5



n = 11

Figure 5.57 Length frequency distribution of perch caught by electric fishing at Trent Bridge, River Trent.

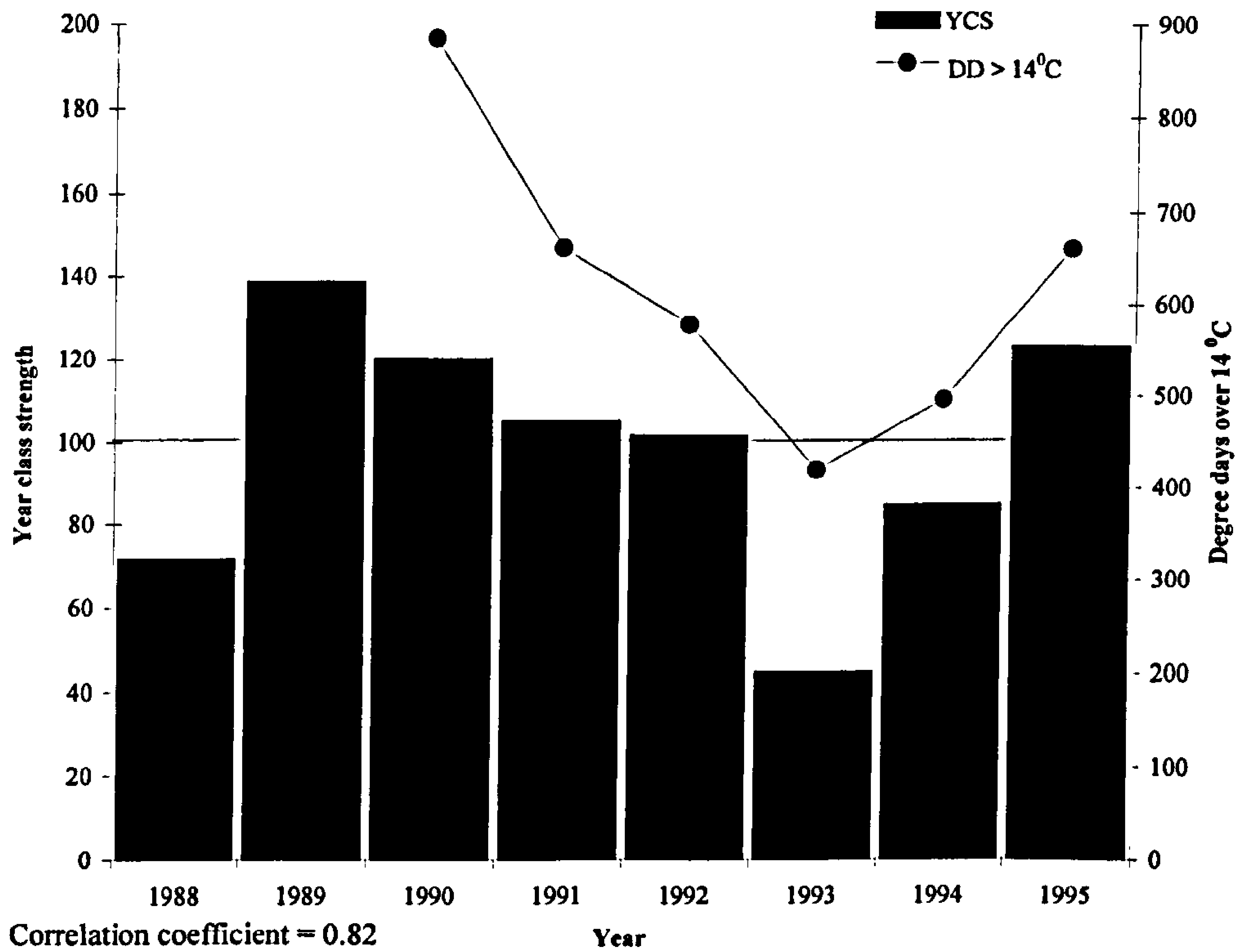


Figure 5.58 Year class strength of roach at Trent Bridge, River Trent, and the relationship with degree days over 14°C.

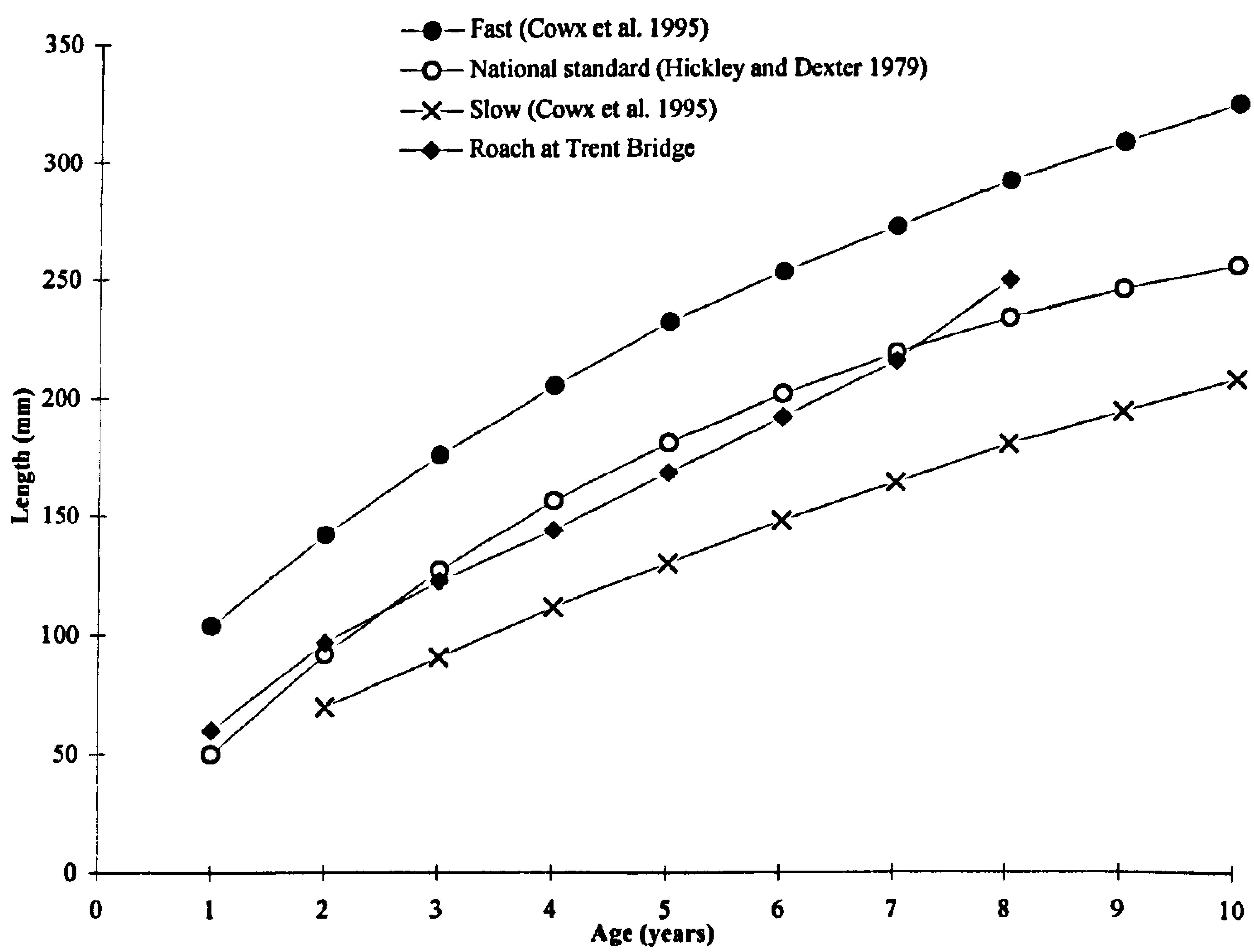


Figure 5.59 Growth of roach at Trent Bridge, River Trent, compared with standard growth curves.

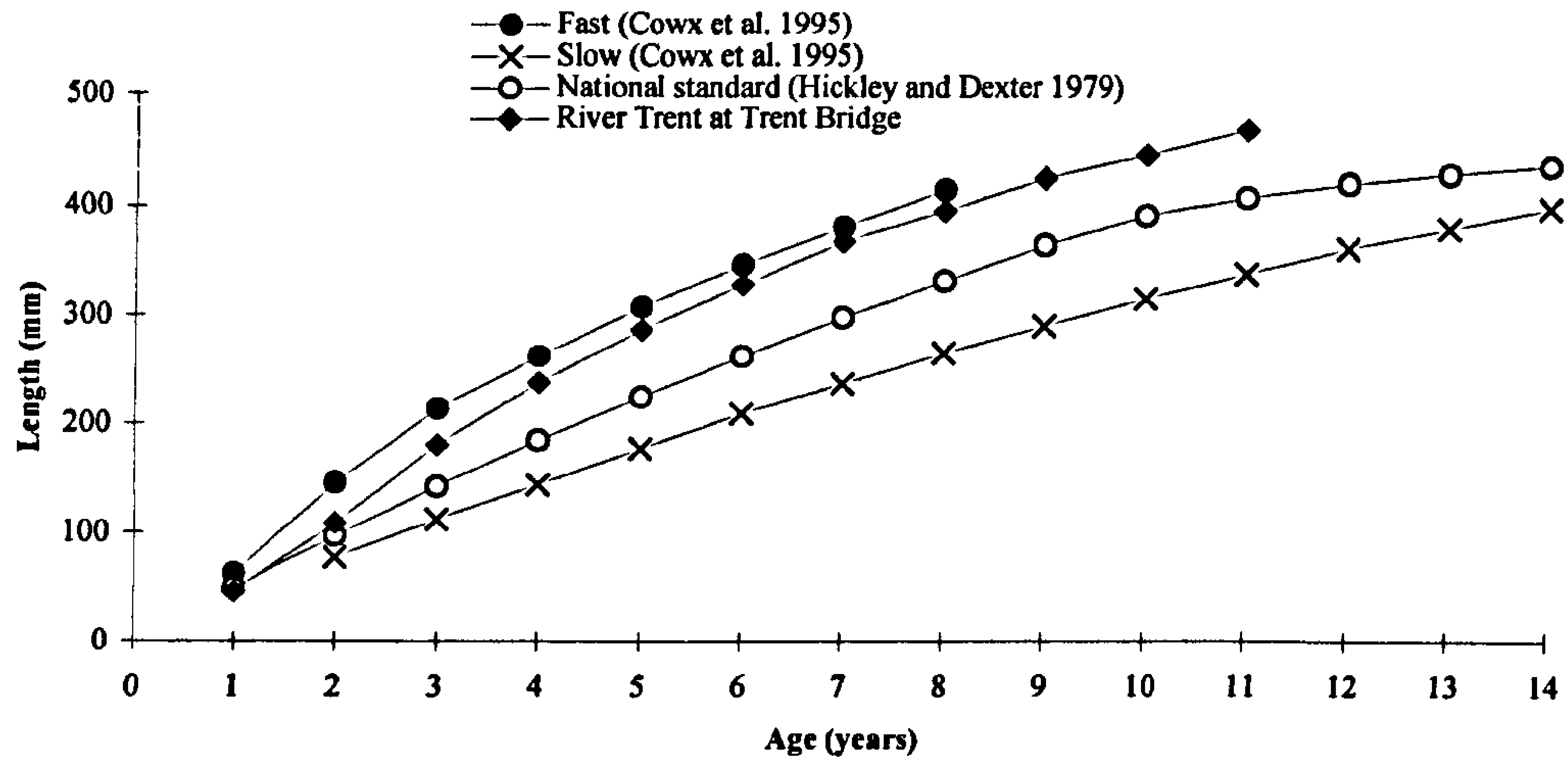


Figure 5.60 Growth of common bream at Trent Bridge, River Trent, compared with standard growth curves.

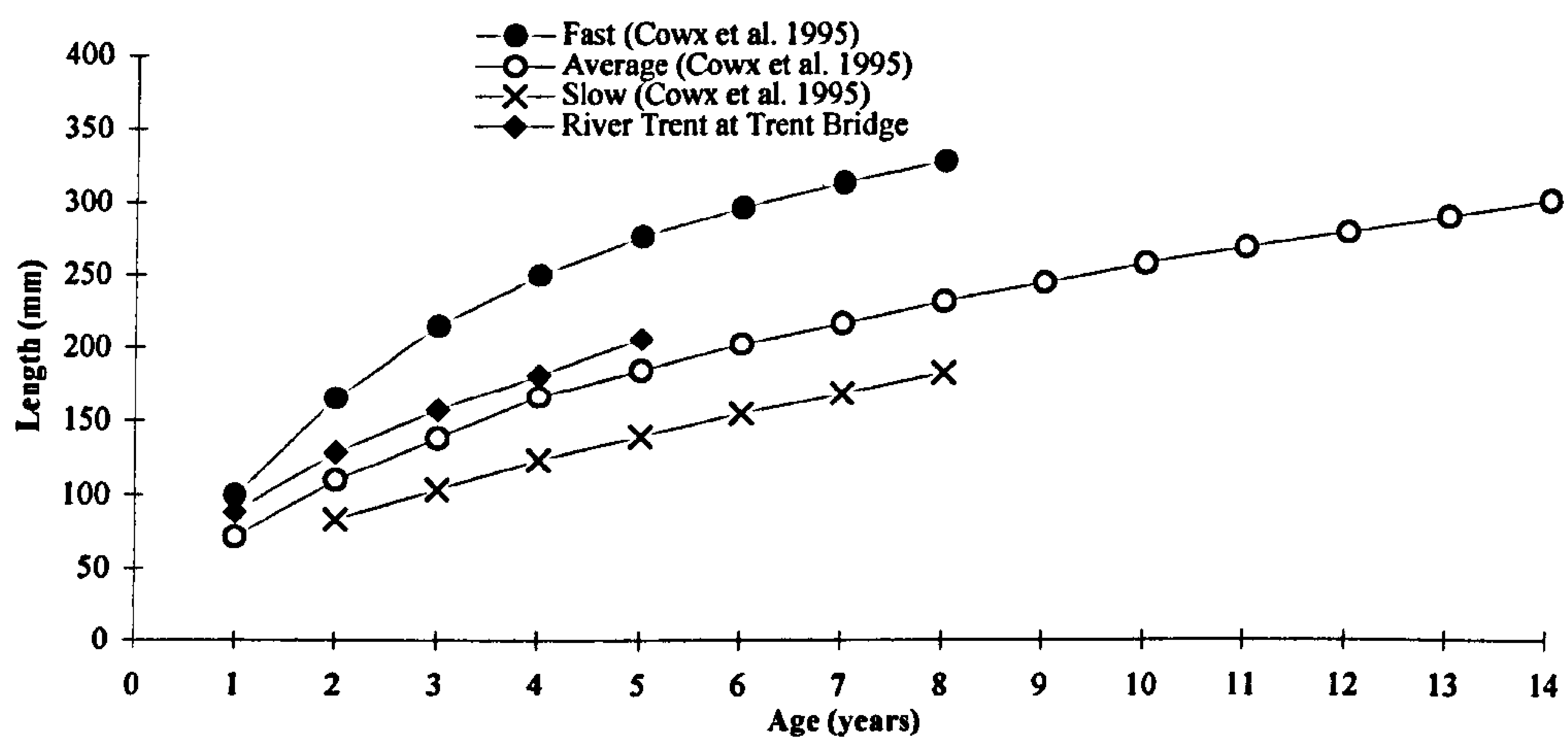


Figure 5.61 Growth of perch at Trent Bridge, River Trent, compared with standard growth curves.

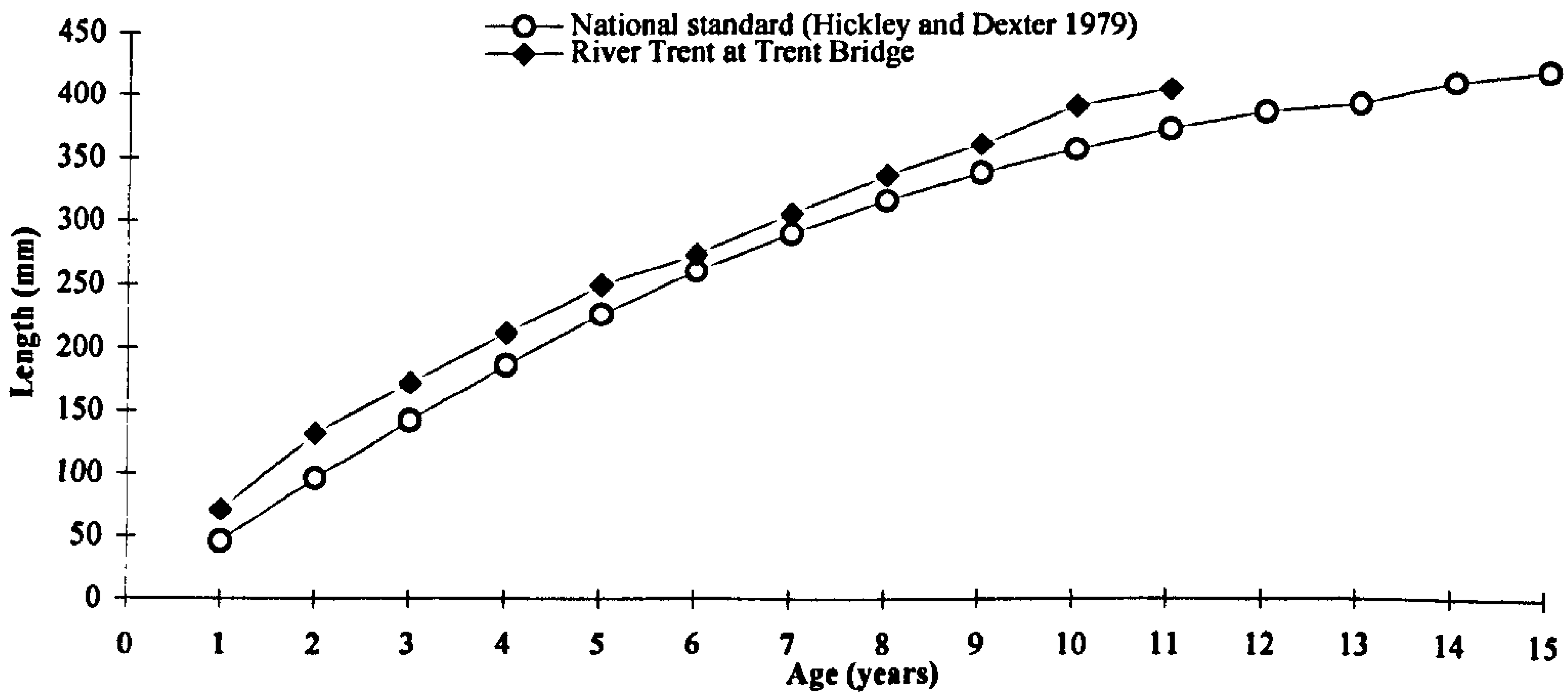


Figure 5.62 Growth of chub at Trent Bridge, River Trent, compared with standard growth curves.

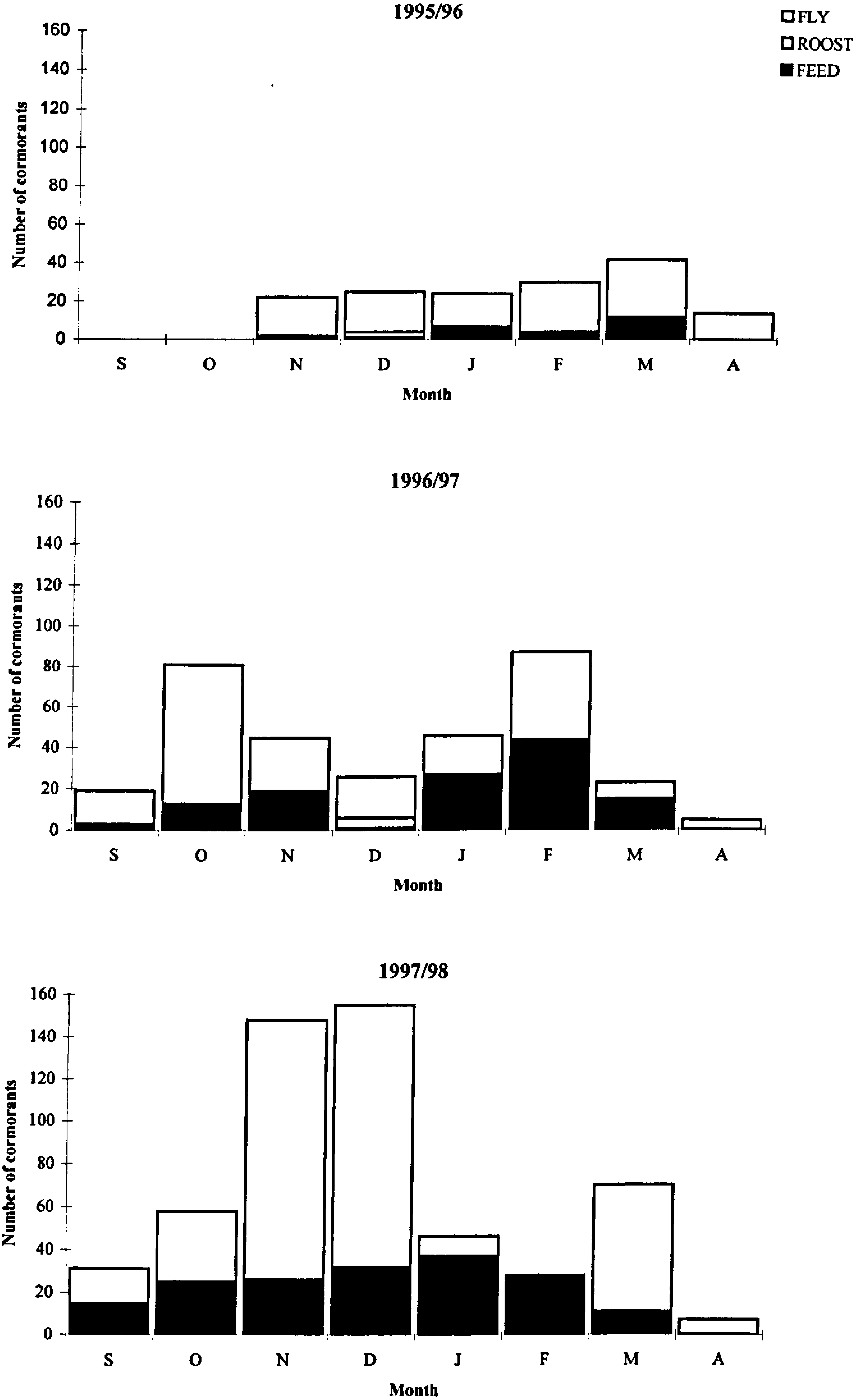


Figure 5.63 The number of cormorants utilising Section C (Trent Bridge) of the River Trent over the study period.

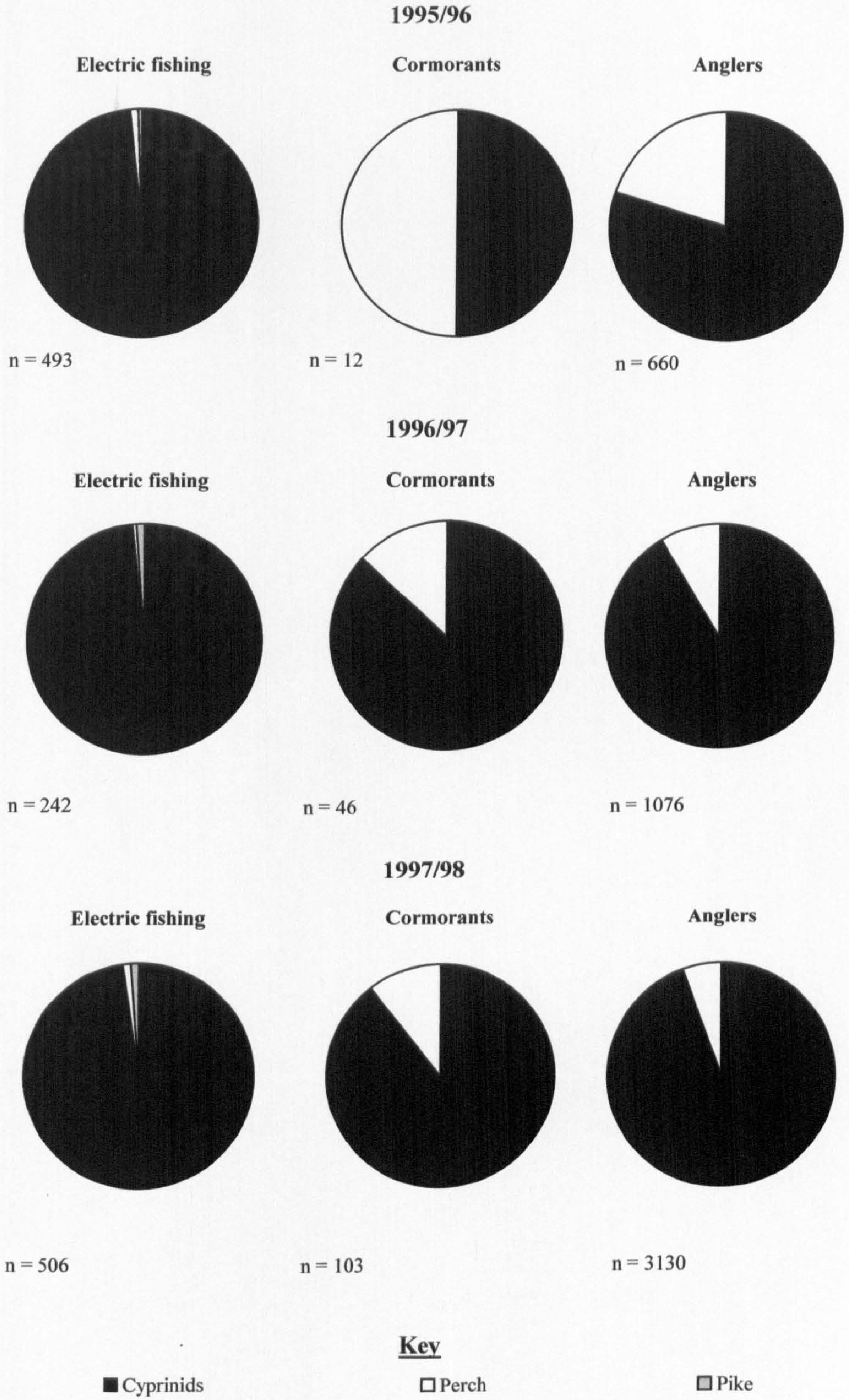


Figure 5.64 Species composition of cormorant diet, electric fishing surveys and angler catches from Trent Bridge, River Trent.

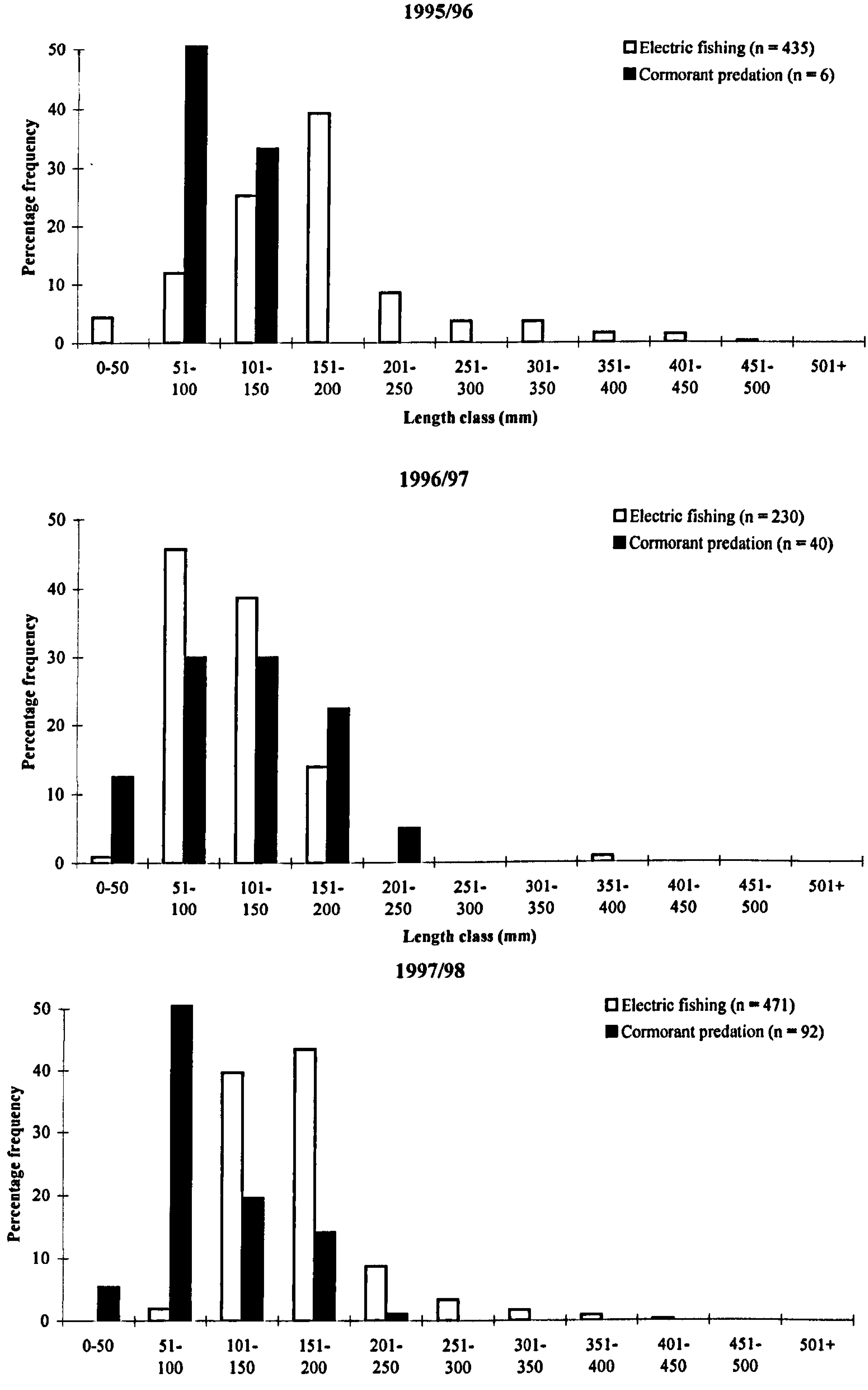


Figure 5.65 Length frequency distribution of cyprinids ingested by cormorants and caught in electric fishing surveys at Trent Bridge, River Trent.

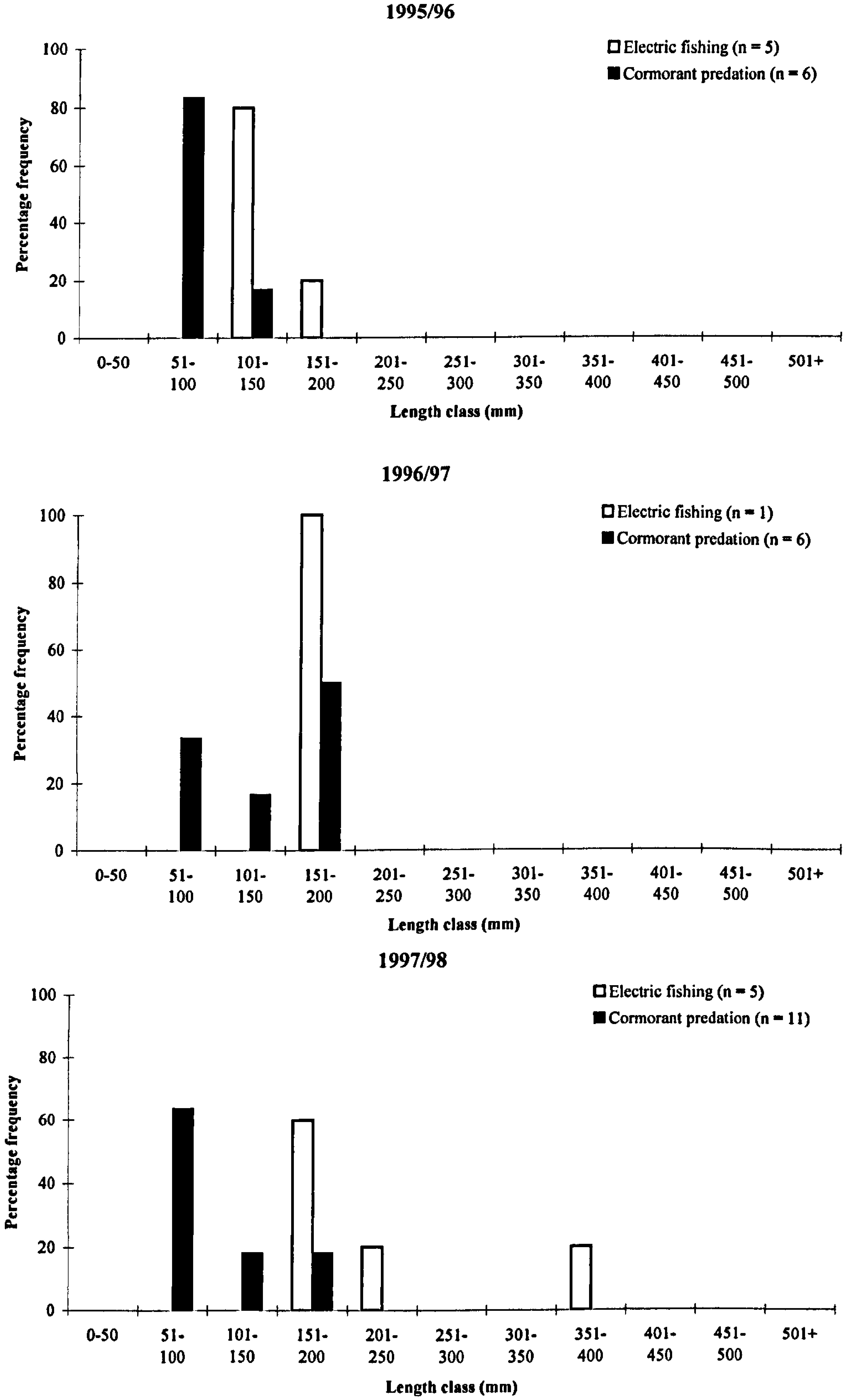


Figure 5.66 Length frequency distribution of perch ingested by cormorants and caught in electric fishing surveys at Trent Bridge, River Trent.

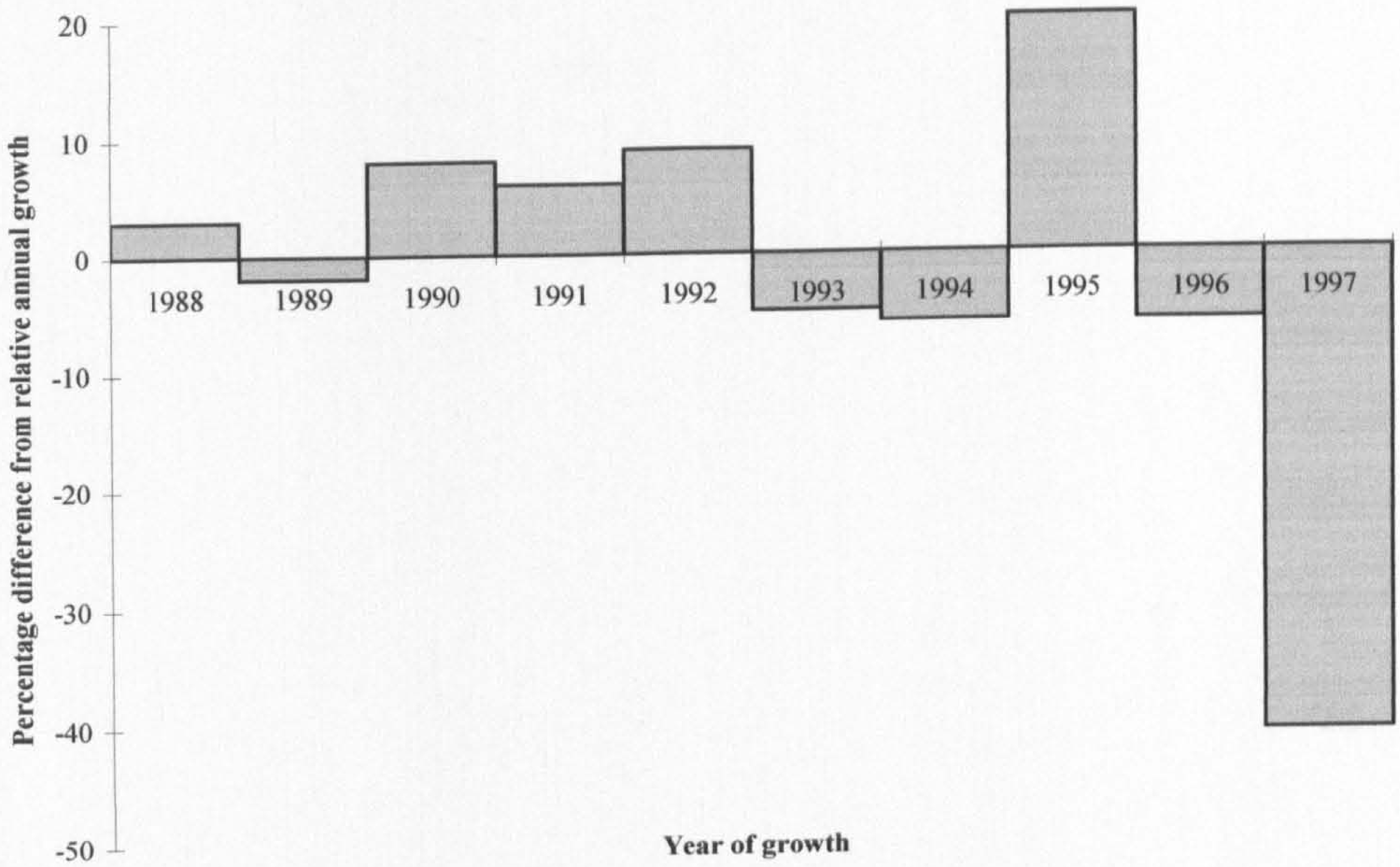


Figure 5.67 Relative annual growth of roach at Trent Bridge, River Trent.

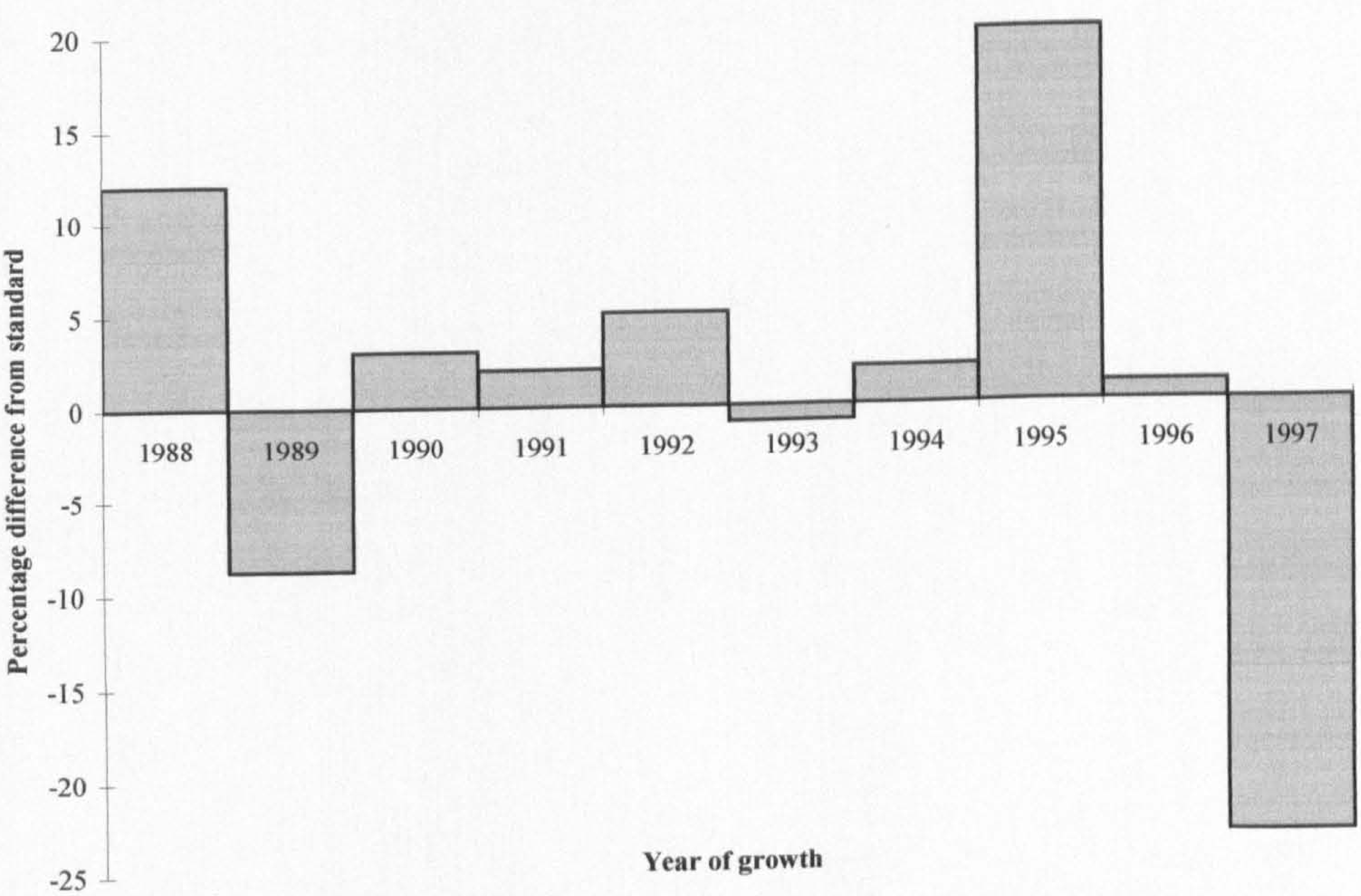


Figure 5.68 Annual growth of roach at Trent Bridge, River Trent compared with a standard roach growth rate (Hickley and Dexter 1979).

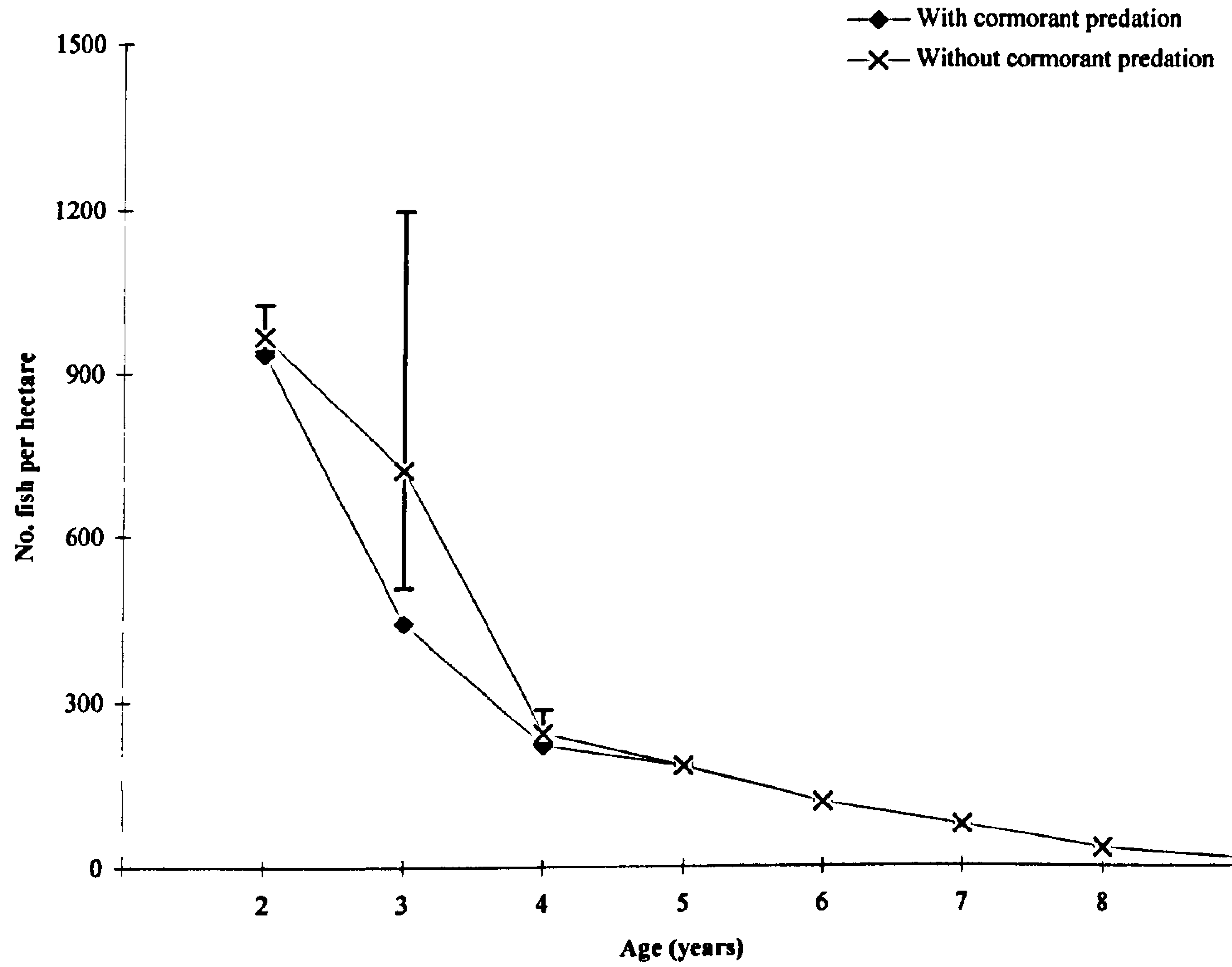


Figure 5.69 Impact of cormorant predation on the numbers of roach at Trent Bridge, River Trent, in 1996.

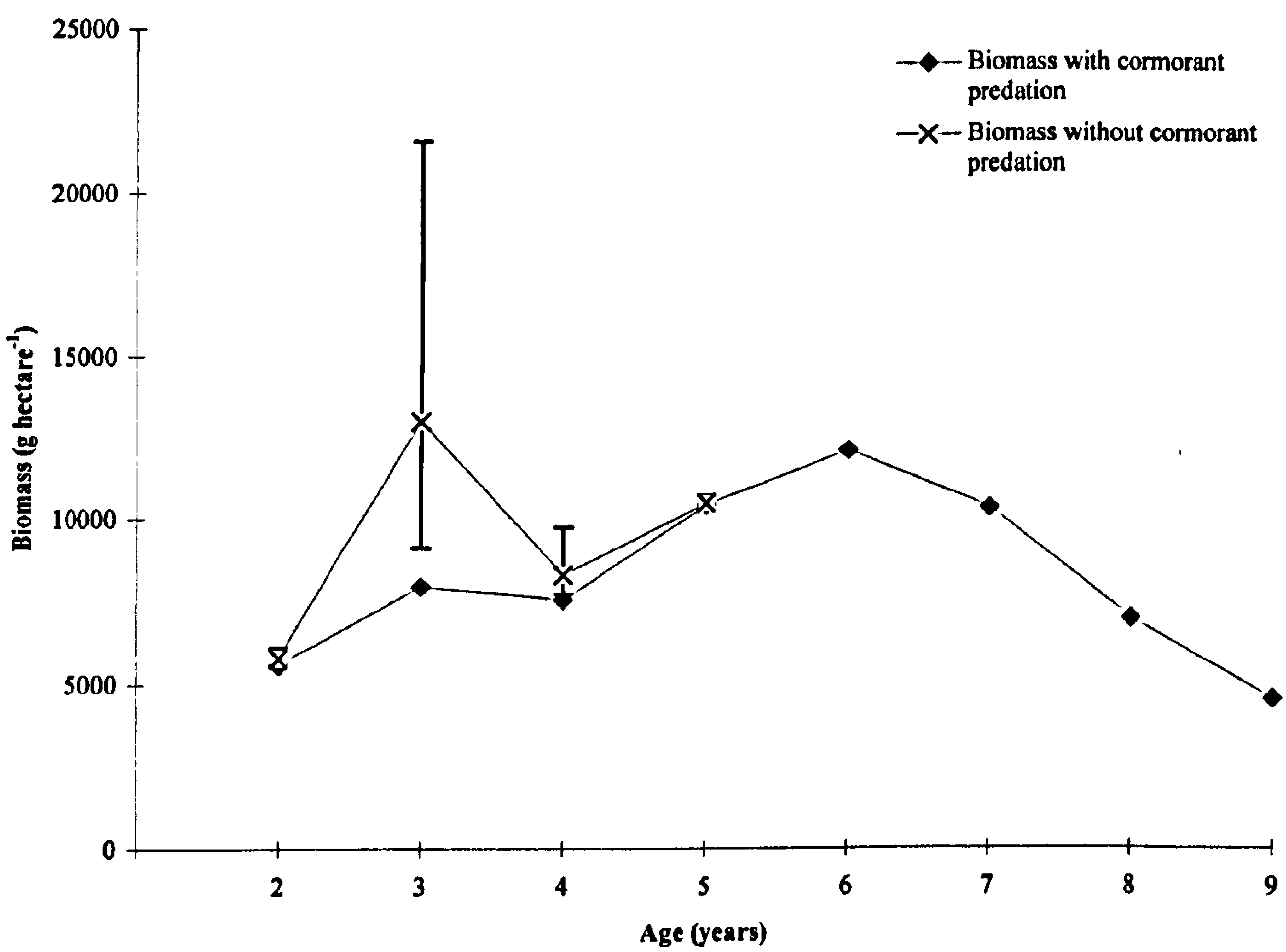


Figure 5.70 Impact of cormorant predation on the biomass of roach at Trent Bridge, River Trent, in 1996.

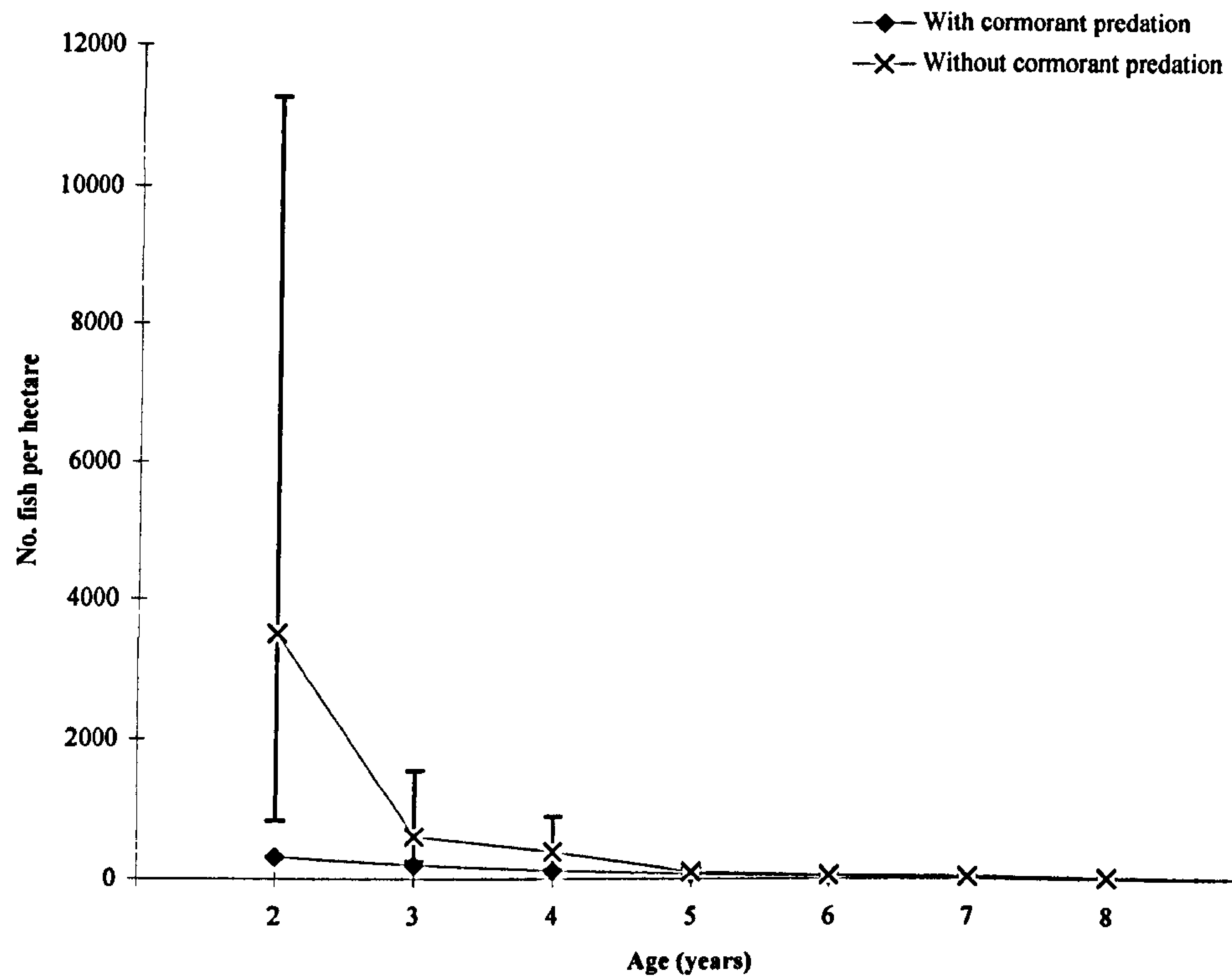


Figure 5.71 Impact of cormorant predation on the numbers of roach at Trent Bridge, River Trent, in 1997.

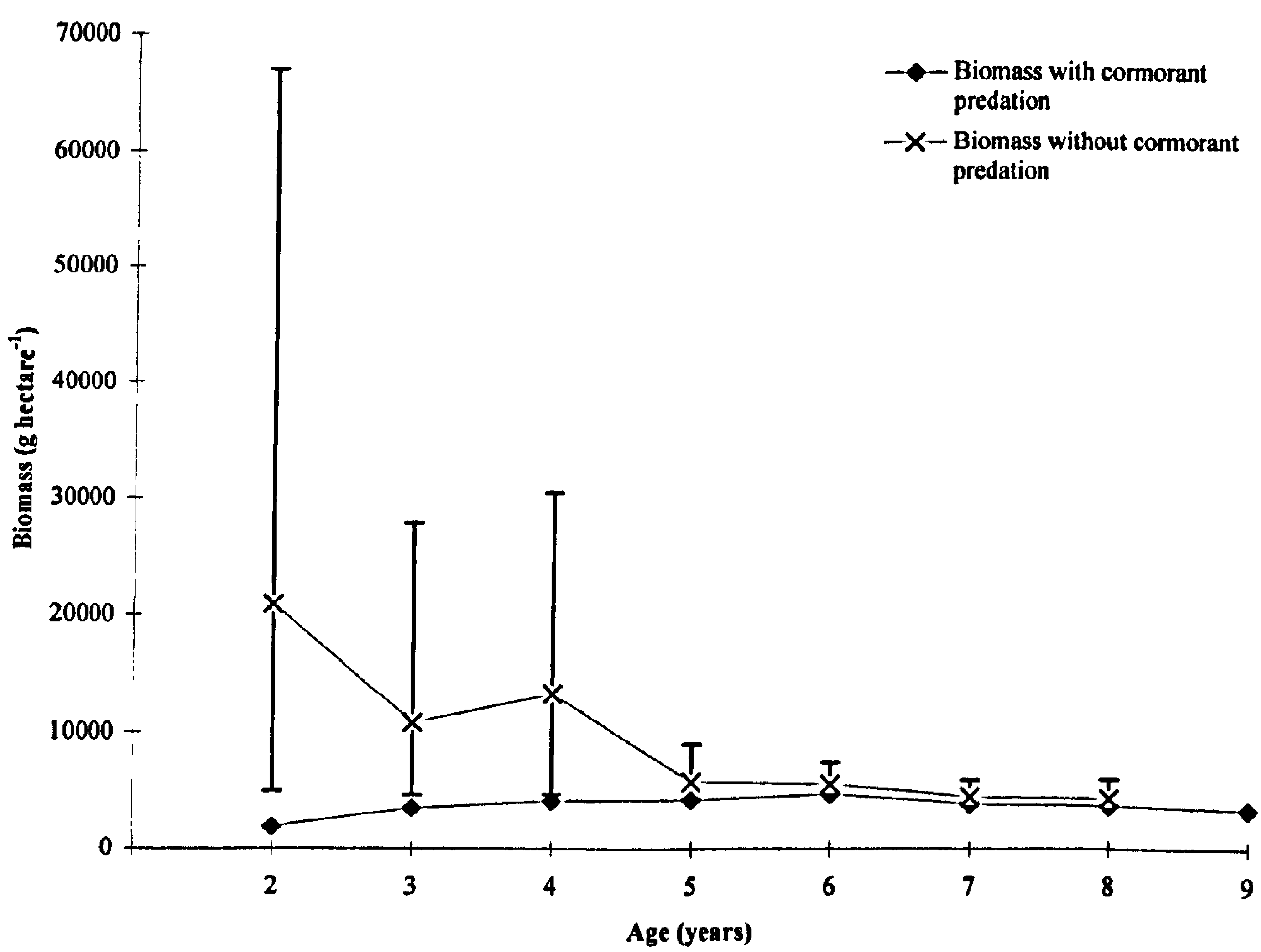


Figure 5.72 Impact of cormorant predation on the biomass of roach at Trent Bridge, River Trent, in 1997.

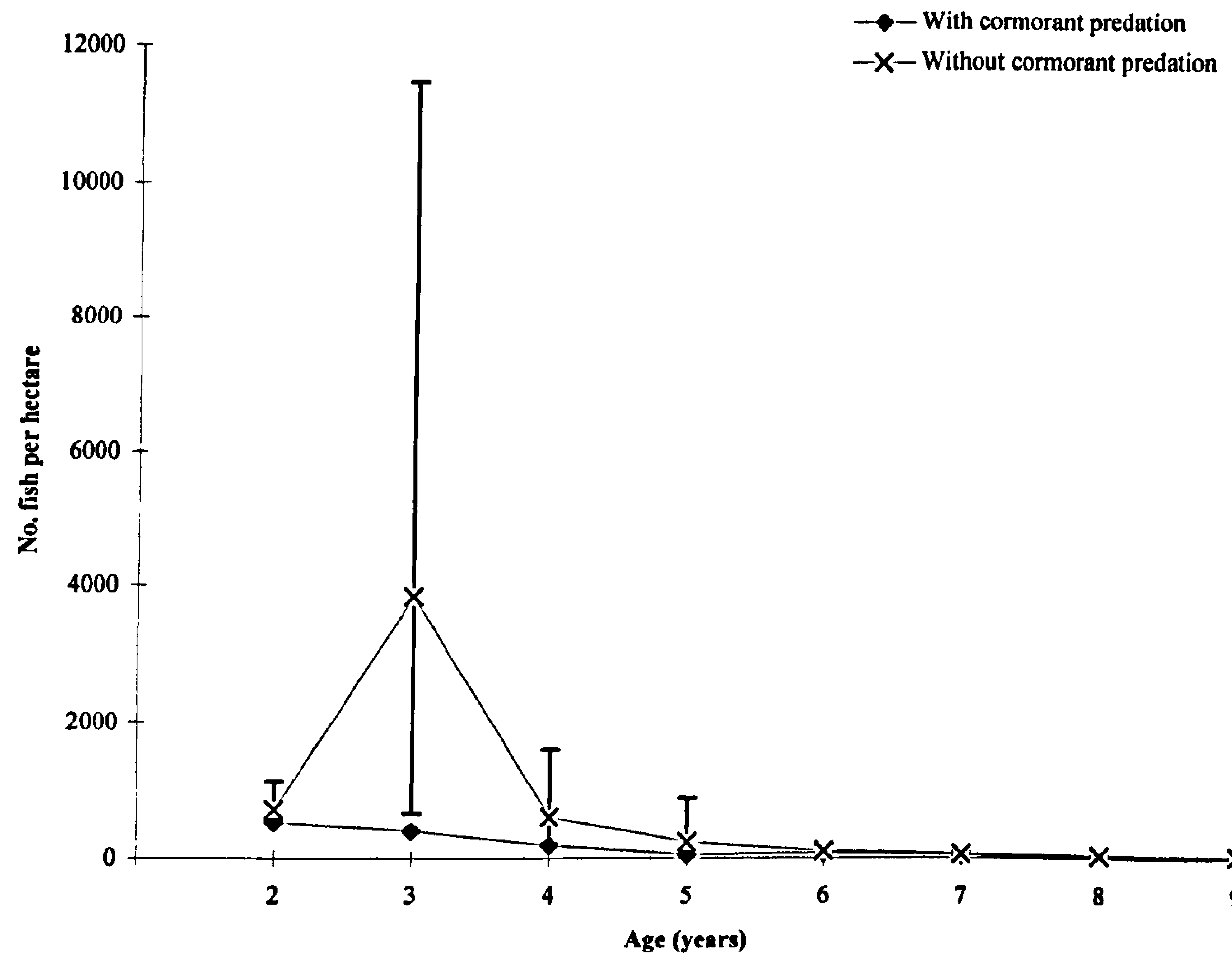


Figure 5.73 Impact of cormorant predation on the numbers of roach at Trent Bridge, River Trent, in 1998.

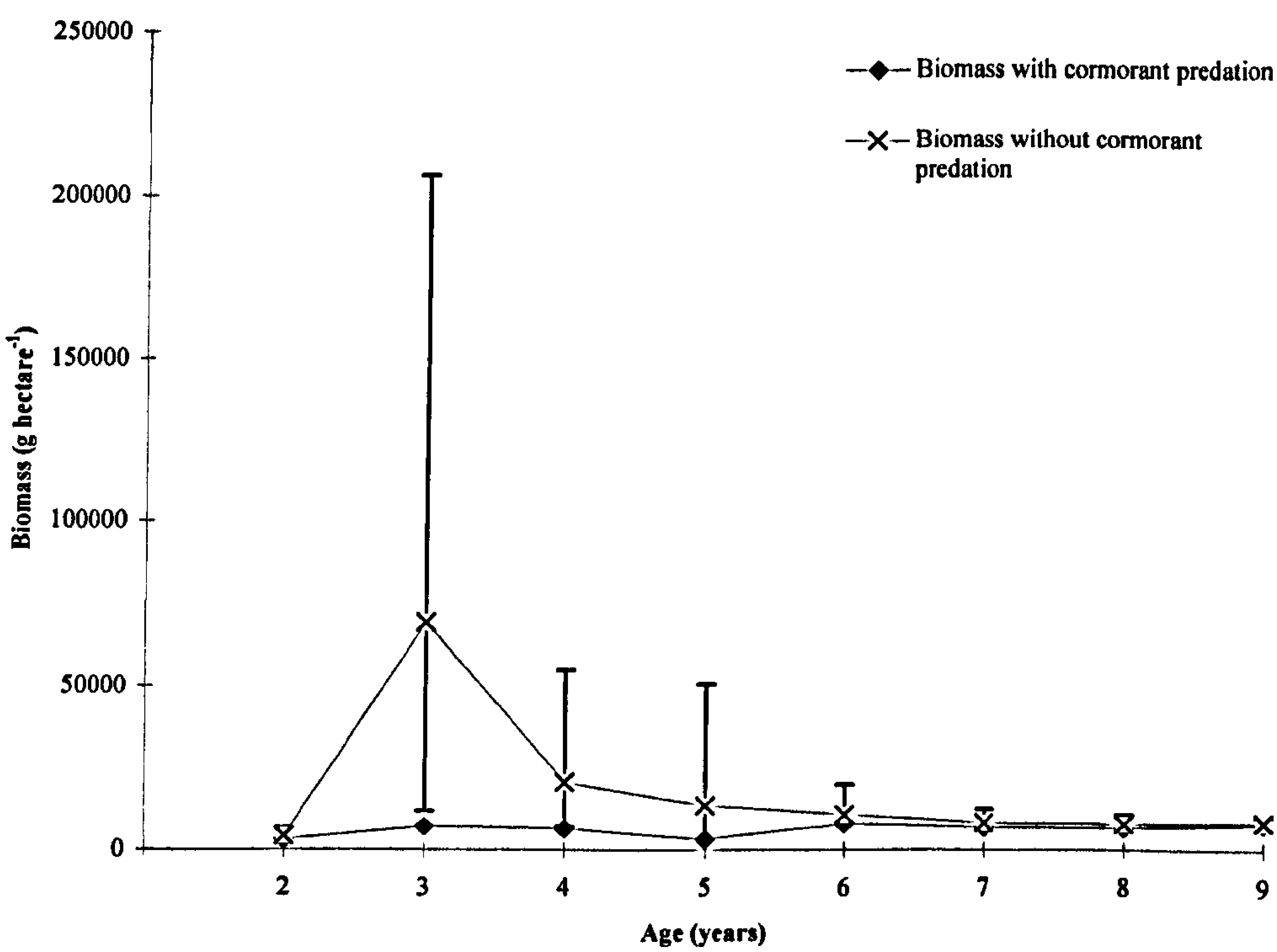


Figure 5.74 Impact of cormorant predation on the biomass of roach at Trent Bridge, River Trent, in 1998.

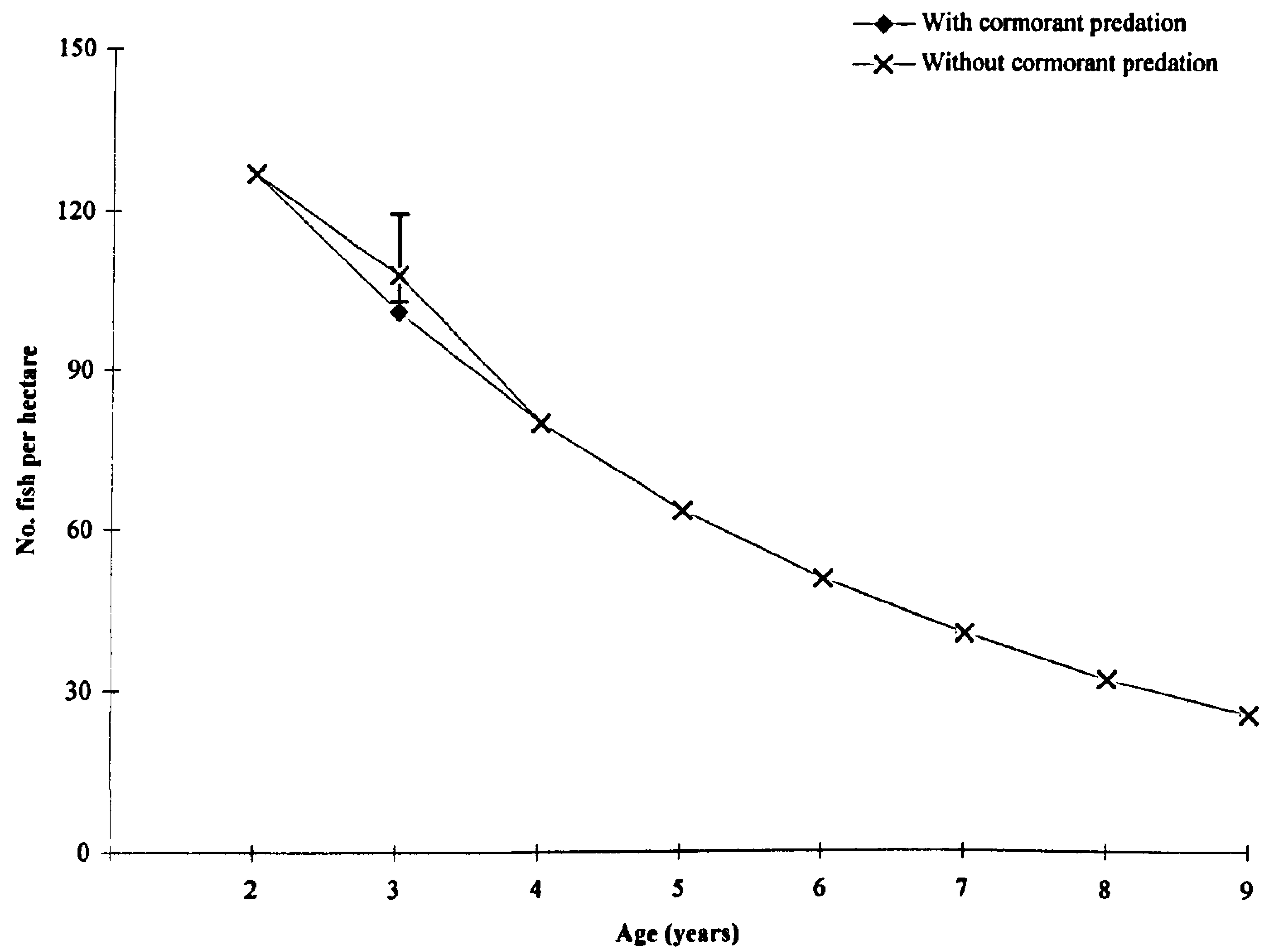


Figure 5.75 Impact of cormorant predation on the numbers of common bream at Trent Bridge, River Trent, in 1996.

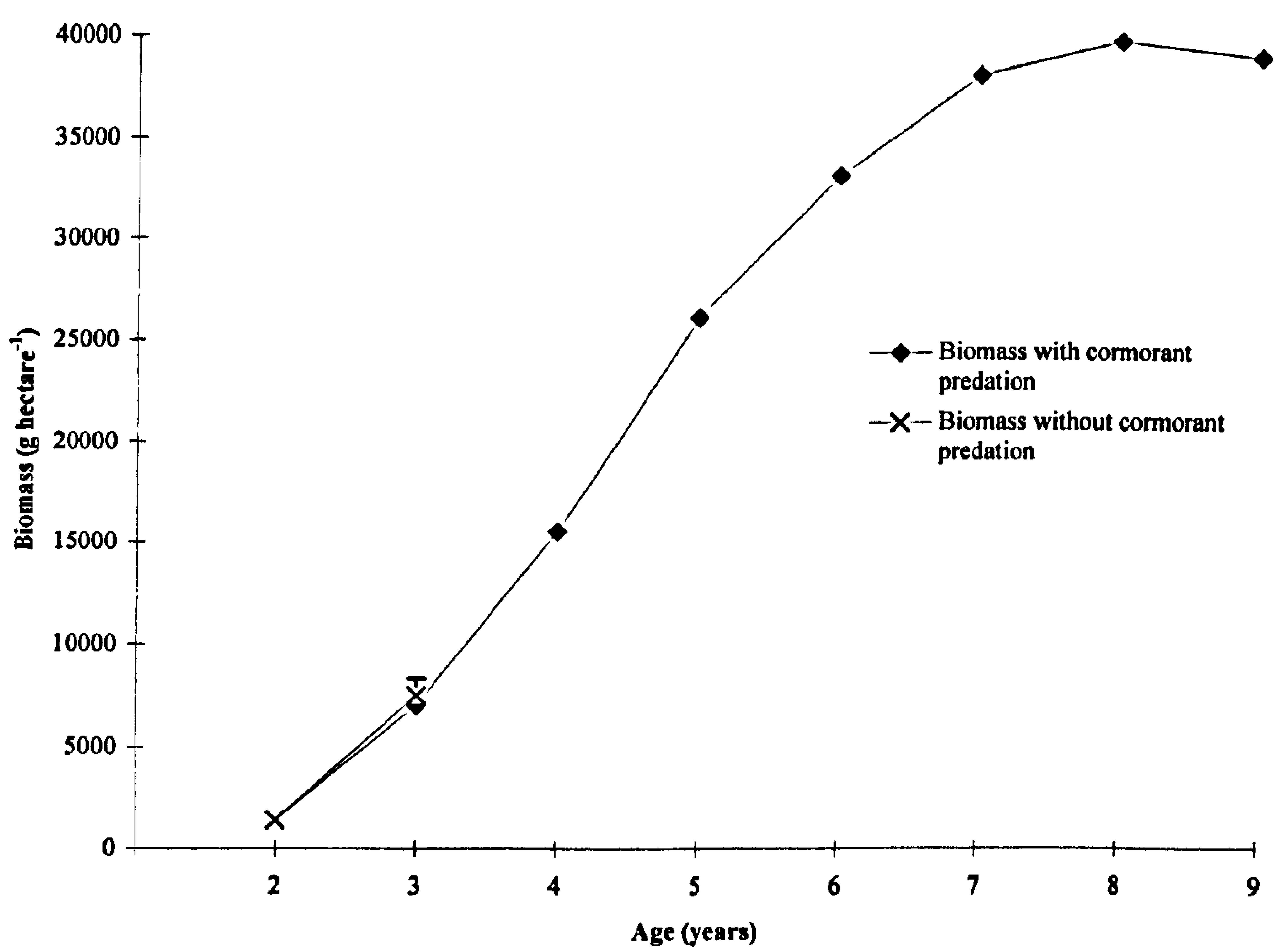


Figure 5.76 Impact of cormorant predation on the biomass of common bream at Trent Bridge, River Trent, in 1996.

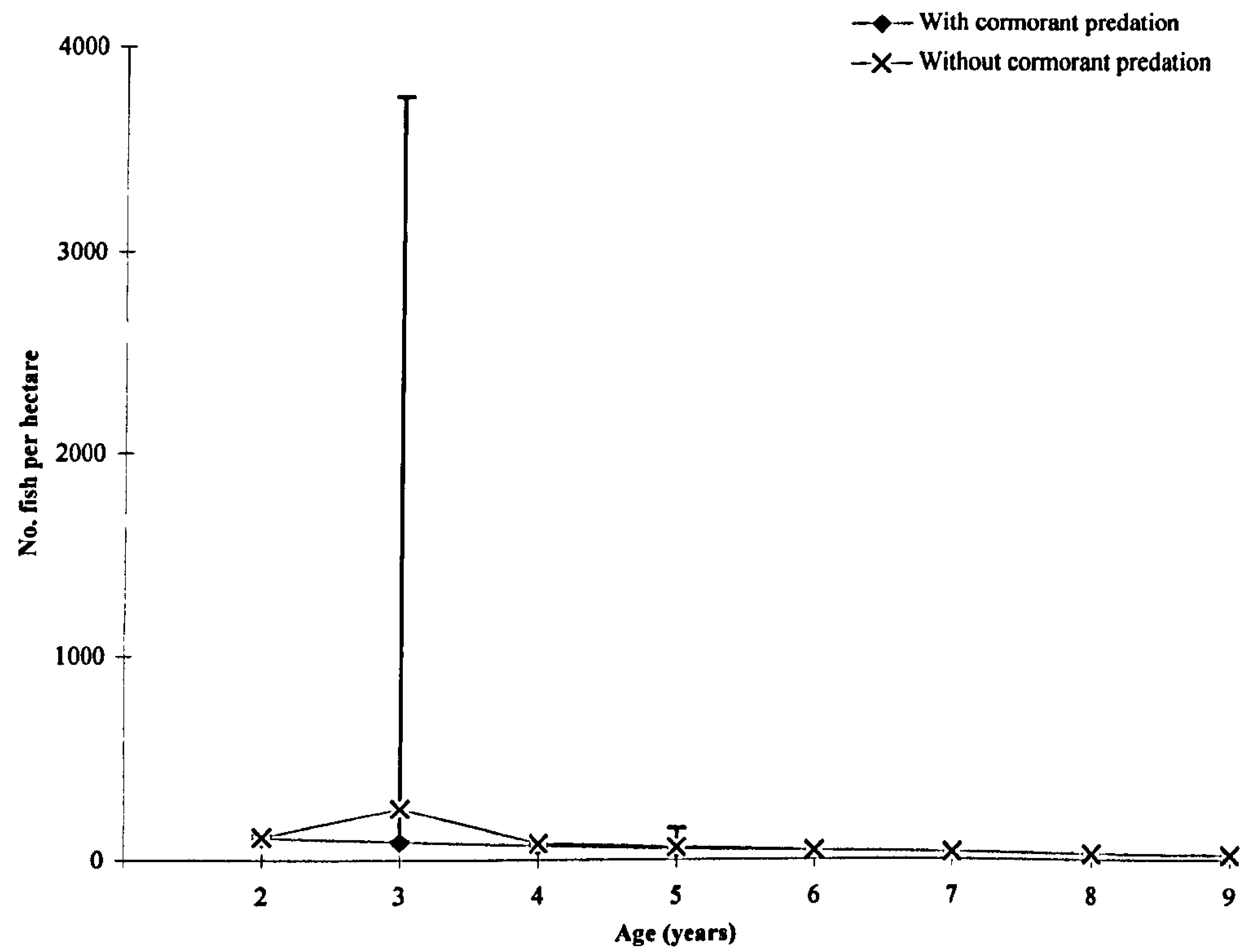


Figure 5.77 Impact of cormorant predation on the numbers of common bream at Trent Bridge, River Trent, in 1998.

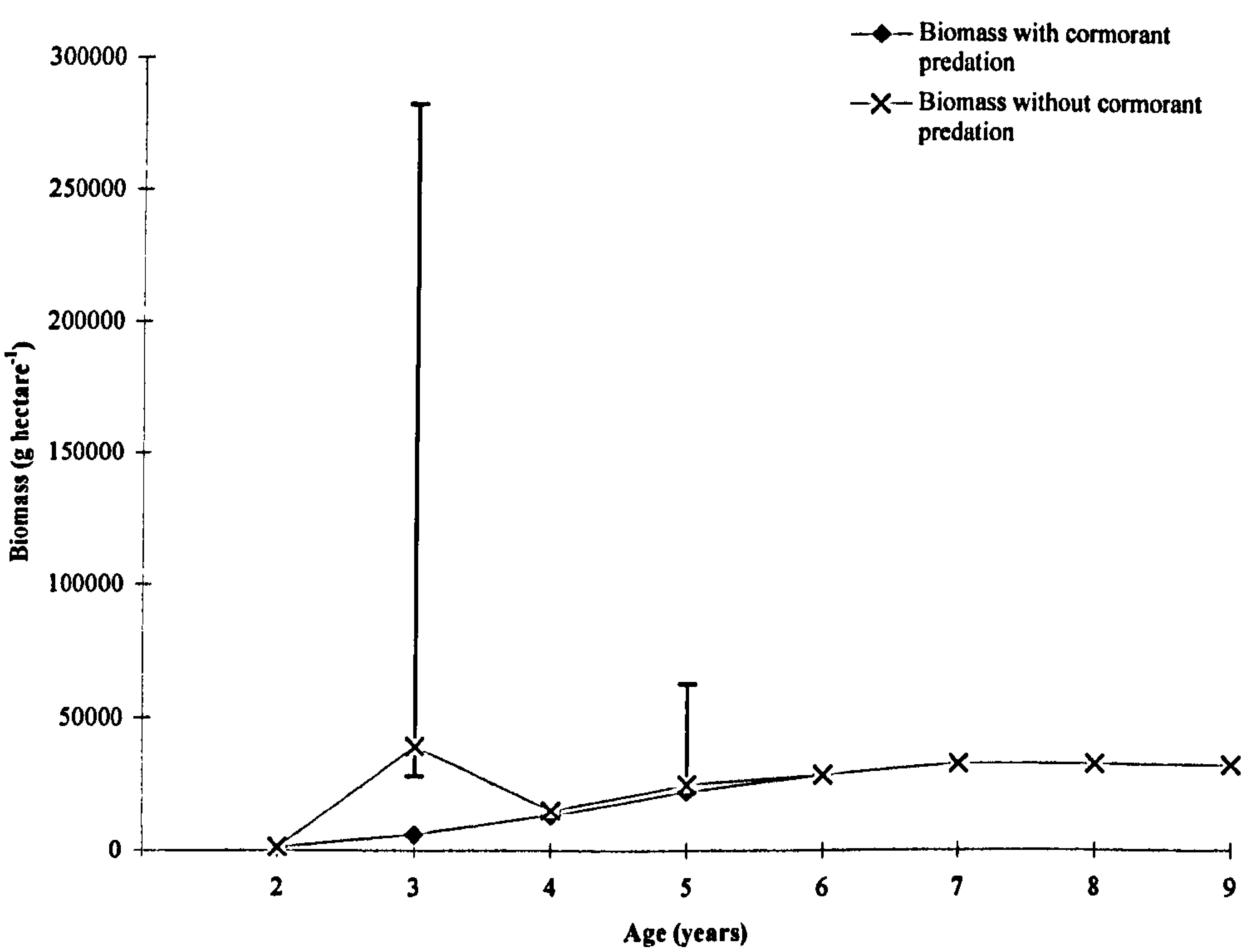


Figure 5.78 Impact of cormorant predation on the biomass of common bream at Trent Bridge, River Trent, in 1998.

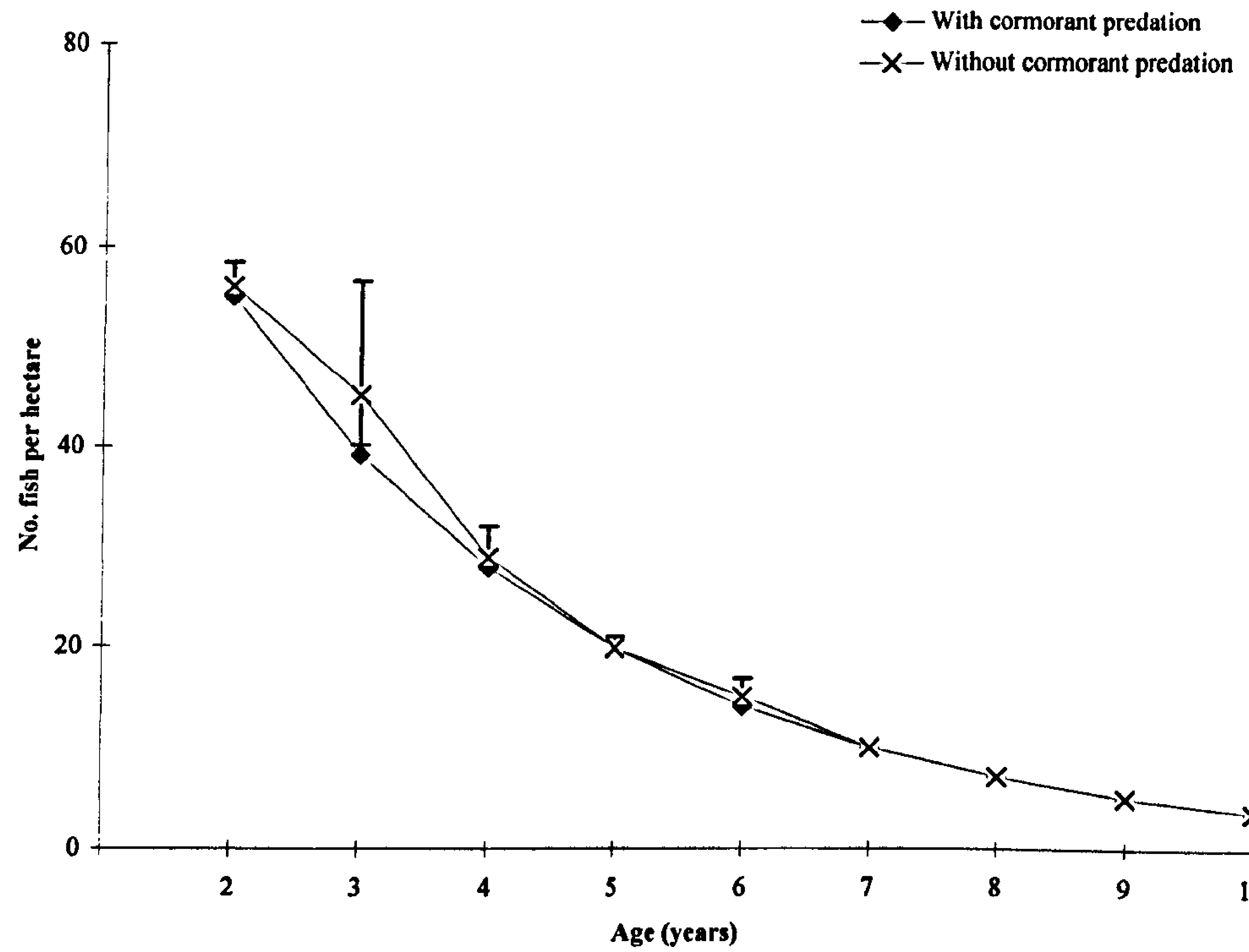


Figure 5.79 Impact of cormorant predation on the numbers of hybrids at Trent Bridge, River Trent, in 1996.

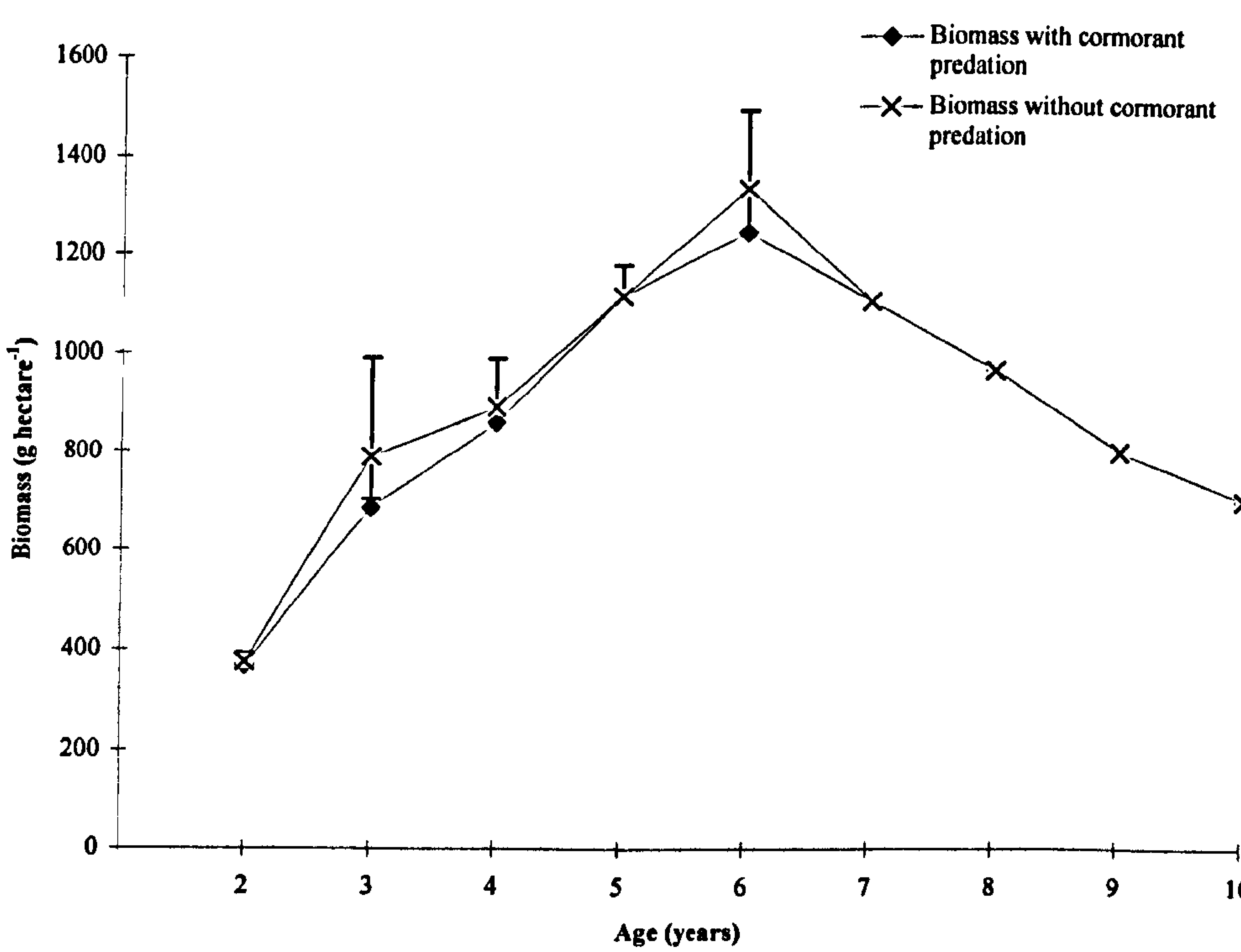


Figure 5.80 Impact of cormorant predation on the biomass of hybrids at Trent Bridge, River Trent, in 1996.

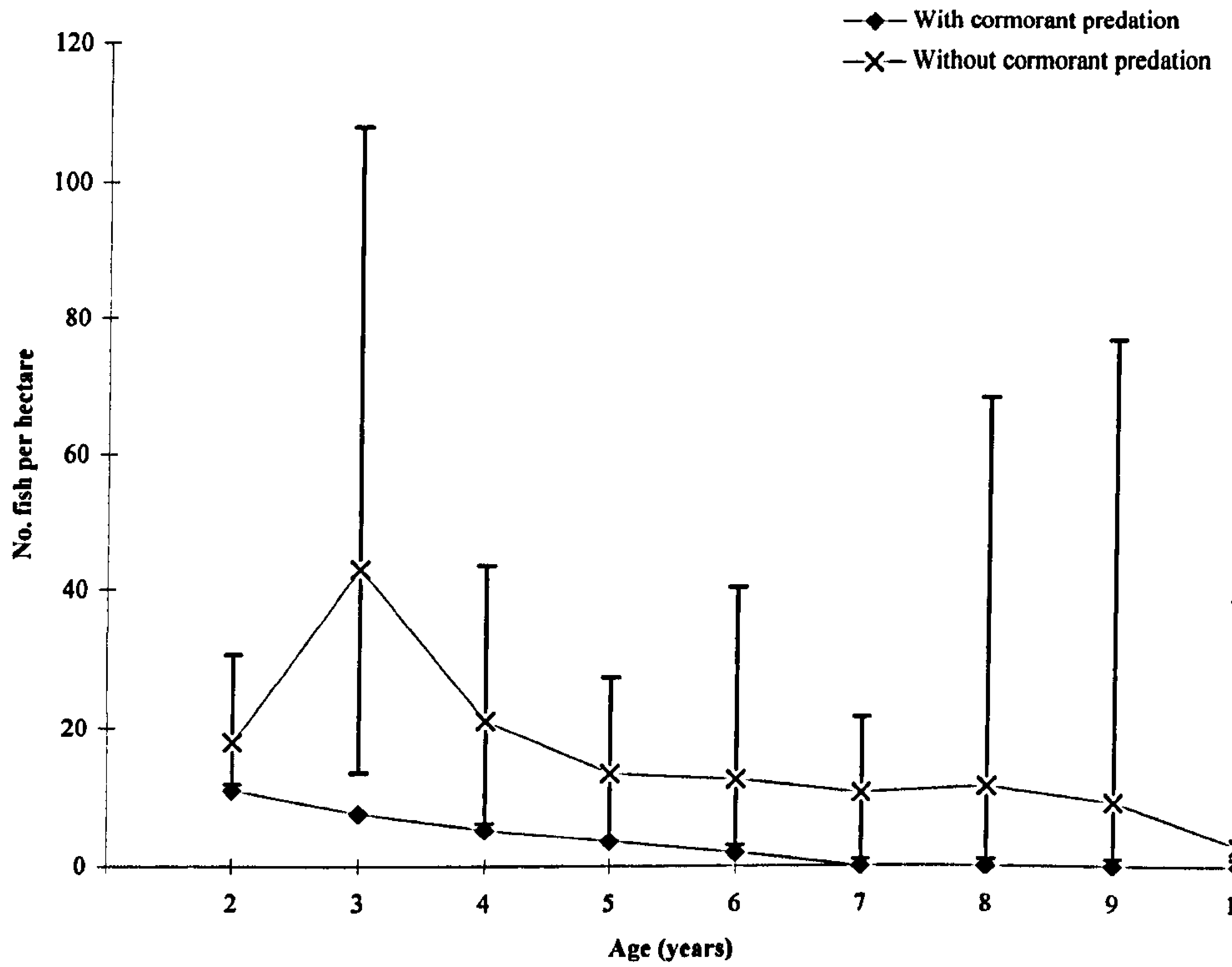


Figure 5.81 Impact of cormorant predation on the numbers of hybrids at Trent Bridge, River Trent, in 1998.

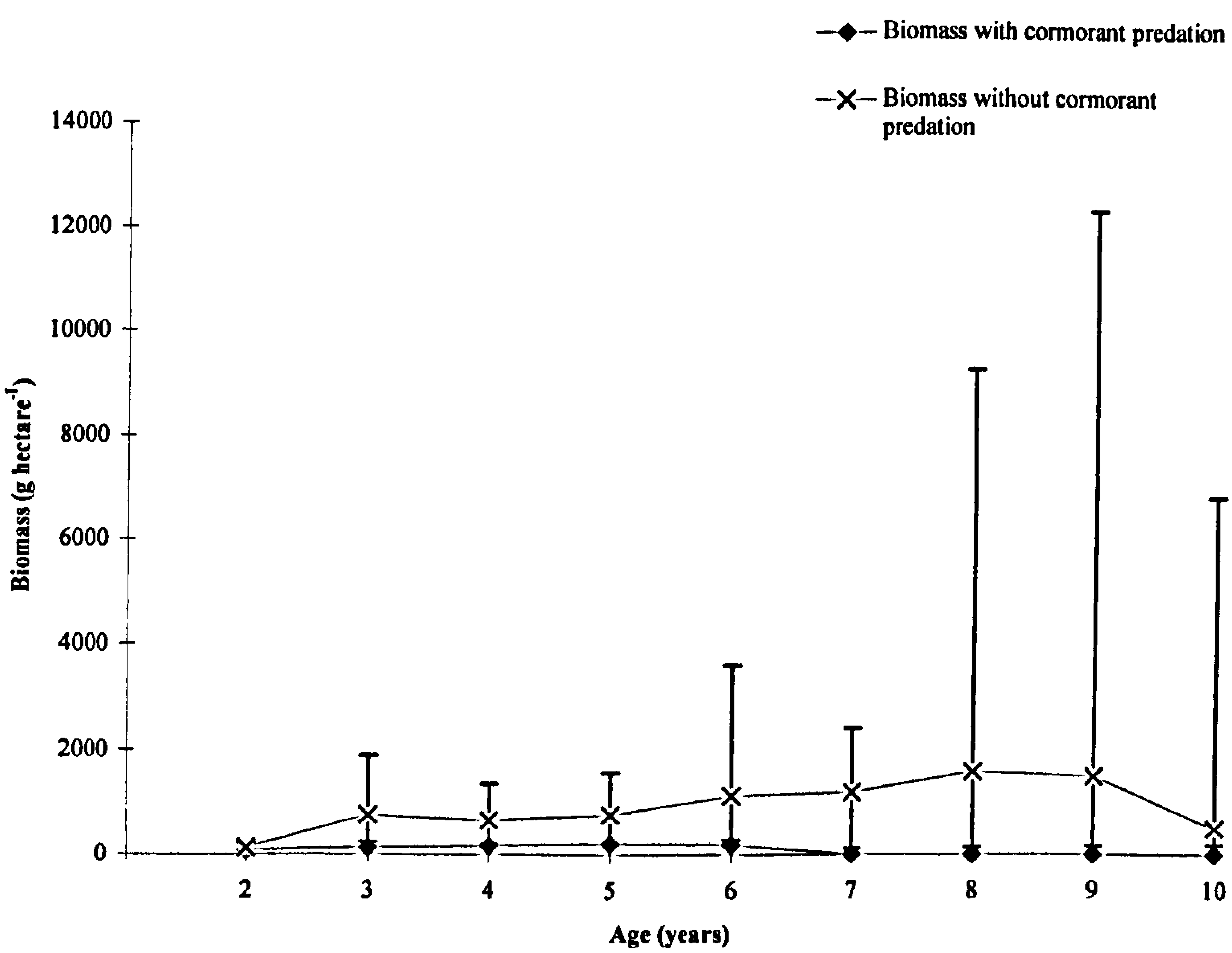


Figure 5.82 Impact of cormorant predation on the biomass of hybrids at Trent Bridge, River Trent, in 1998.

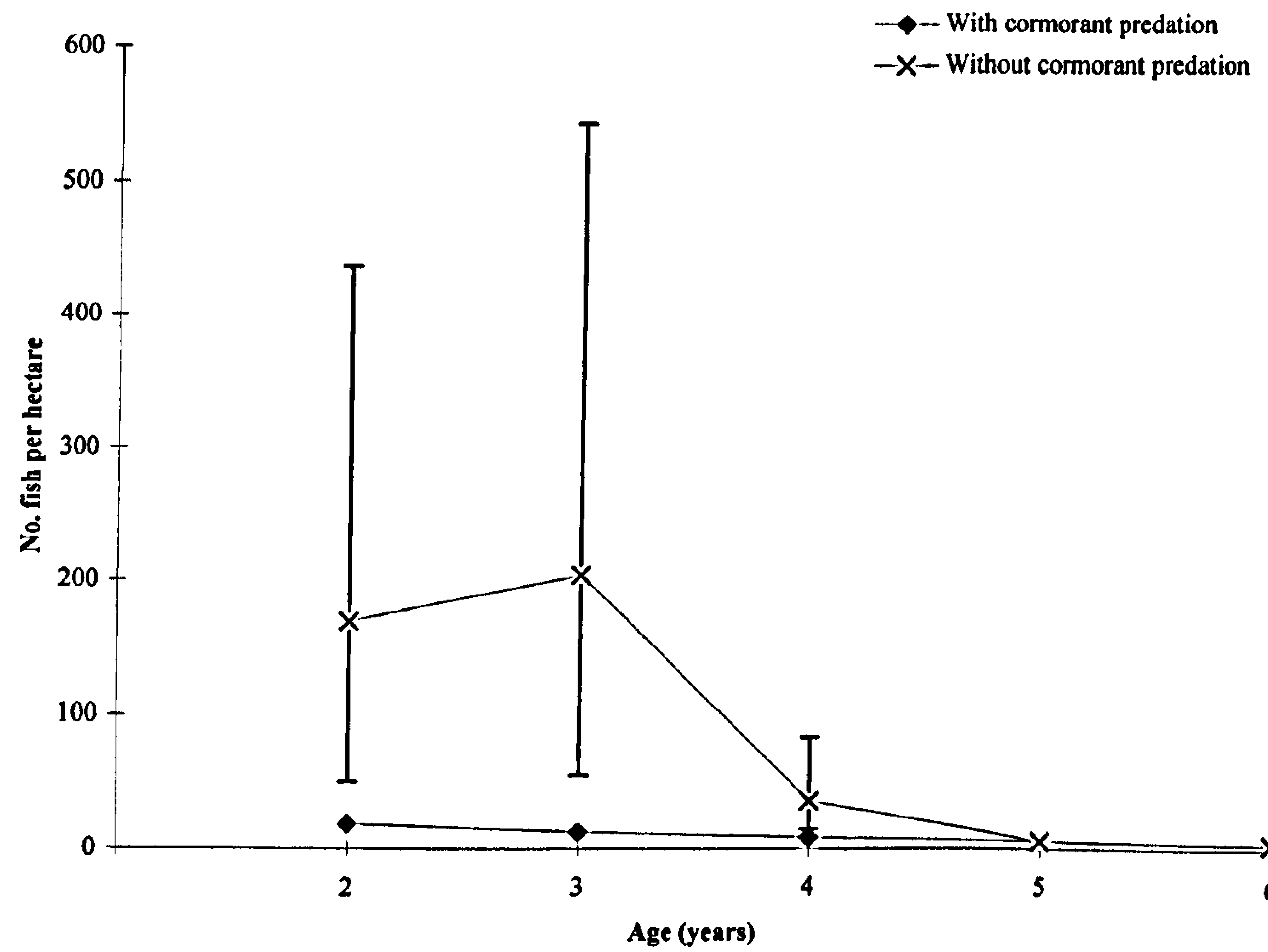


Figure 5.83 Impact of cormorant predation on the numbers of perch at Trent Bridge, River Trent, in 1996.

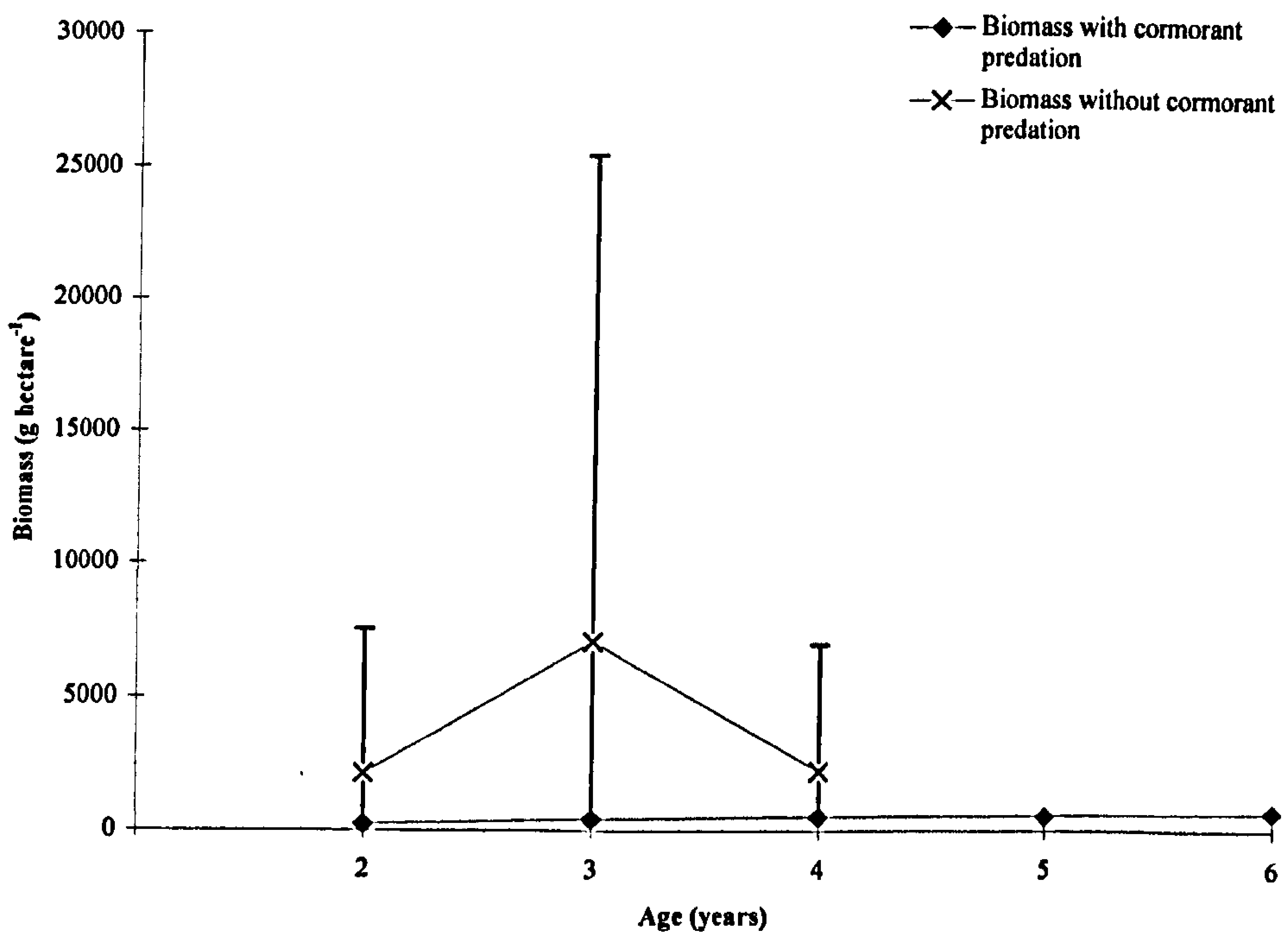


Figure 5.84 Impact of cormorant predation on the biomass of perch at Trent Bridge, River Trent, in 1996.

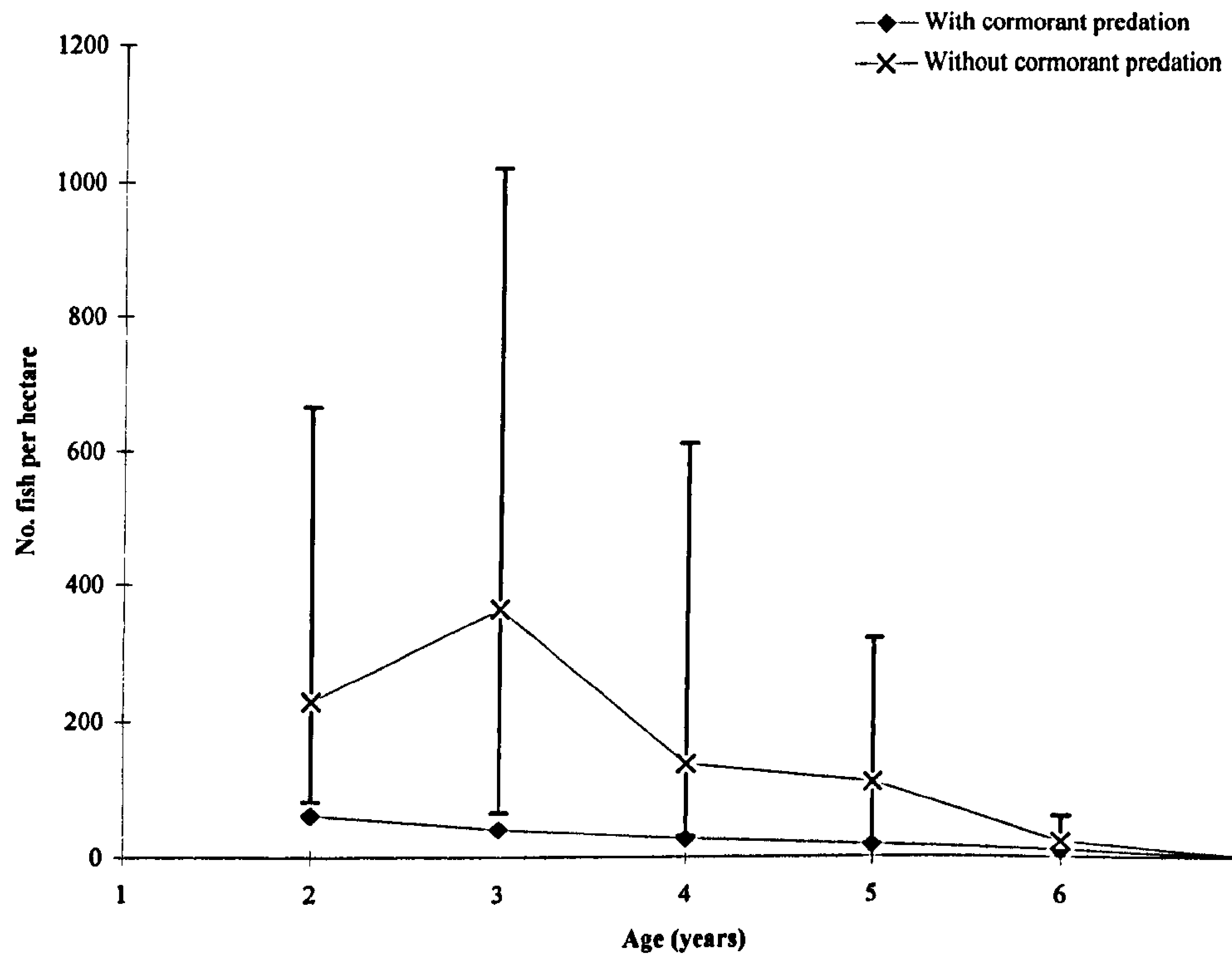


Figure 5.85 Impact of cormorant predation on the numbers of perch at Trent Bridge, River Trent, in 1998.

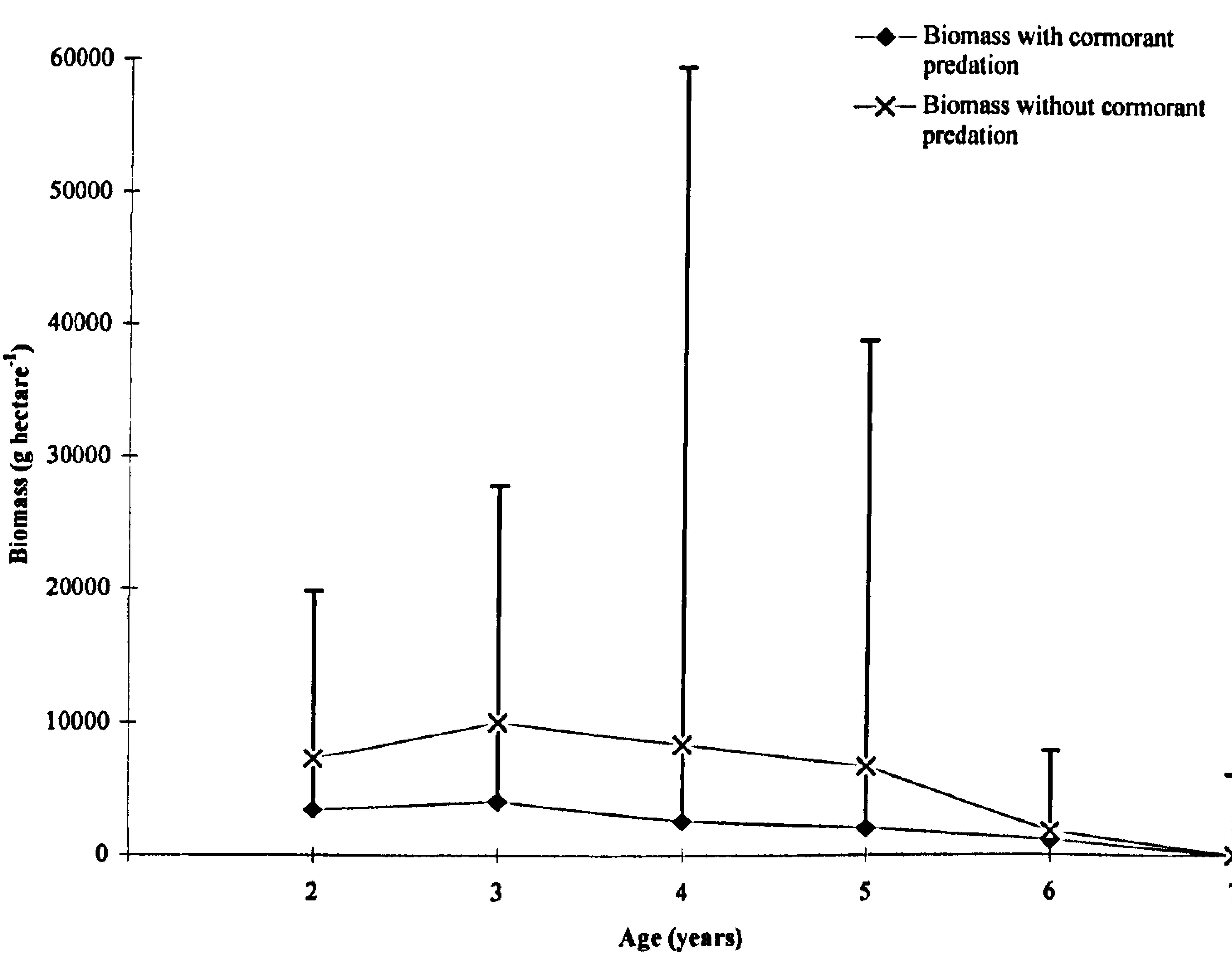


Figure 5.86 Impact of cormorant predation on the biomass of perch at Trent Bridge, River Trent, in 1998.

5.7 Status of fish populations, cormorant populations and impact of cormorants on the River Trent at Stoke Bardolph

5.7.1 *Status of fish populations*

Standing crop

The standing crop of roach, bream and perch stocks in the section varied between survey years (October 1995, June 1996, June 1997 and June 1998), and were similar to the Beeston sections (Table 5.25) (Section 5.6.1).

Table 5.25 Standing crop of species, estimated by electric fishing, at Stoke Bardolph, River Trent, over the study period.

	Standing crop (kg ha^{-1})		
Species	**1995/96**	**1996/97**	**1997/98**
Roach	170.18	67.71	23.65
Common bream	753.08	5.67	0
Chub	133.77	148.23	0
Hybrids	0	0	0
Perch	34.72	1.04	1.16
Total standing crop	1091.75	222.65	24.81
Total standing crop g m^{-2}	109.18	22.27	2.48

Hydro-acoustic data for the section Stoke Bardolph to Gunthorpe, covering the electric fishing area and cormorant observation section, also showed a wide temporal variation in fish abundance in the section, with a decrease in the study period (Table 5.26).

Table 5.26 Density of fish present, estimated by hydro-acoustics, in the Stoke Bardolph to Gunthorpe Bridge section of the River Trent.

Year	No. of fish per hectare (n ha^{-1})
1994	435
1995 (June)	324
1995 (September)	48
1996	67
1997	59

Angler catch data

Angler catch data have been collected on the Stoke Bardolph to Gunthorpe Weir fishery of the River Trent since 1969. Due to the presence of this large data set, a detailed analysis of the status and performance of angler catches was possible.

The catch per unit effort of anglers increased from 60 g man $hour^{-1}$ in the 1969/70 angling season to 231 g man $hour^{-1}$ in 1996/97 (Figure 5.87). Angling catch rates were variable with years of very poor catches often followed by a series of good years. For example, good catches occurred in 1989/90 and 1990/91 which followed poor catches in 1987/88 and 1988/89. This results from the varying year class strengths of the exploited species (Jacklin 1996). Catch per unit effort of a large angling match (410 anglers competed) in July 1998 on the study section was 201 g man $hour^{-1}$. This compared favourably with data collected between 1969 and 1997 (Figure 5.87).

The catch per unit effort for the fishery also compared very favourably with data collected on the Yorkshire River Ouse fishery, 1971 to 1990 (Figure 5.87). A similar overall increase in catch rate was also observed on the River Ouse fishery (Axford 1991).

During the period 1981 to 1997, complaints have been raised by anglers over an apparent decline in catches. Complaints about the quality of the angling in the large angling match of July 1998 on the section featured heavily in the angling press in the following weeks (Fitzpatrick 1998). However, these complaints appear unjustified and probably arise because of the effects of the water quality improvements (Section 5.1.2), and the decommissioning of the power stations along the Trent Valley (Section 5.1.3) on the behaviour of the fish populations in relation to angling susceptibility (Cowx and Broughton 1986, Jacklin 1996).

The percentage of anglers weighing in at Stoke Bardolph changed markedly with time (Figure 5.88). Between the 1989/90 and 1995/96 angling seasons, there was a decrease in the percentage of anglers with catch, 86 to 57 %, although in 1996/97 it increased to 87.5 %. The output in 1996/97 may be high due to a lack of winter angler effort, as more anglers with catch were apparent in summer matches.

The percentage of anglers with catch in the large angling match in July 1998 was 92.7 %. This value compares favourably with the percentage of anglers with catch data from 1969 to 1997 (Figure 5.88). It does not appear to show the decline in angling success reported (Fitzpatrick 1998).

The relationship between the catch rate and percentage of anglers with catch was unclear ($P = 0.50$). Hence, a year of increased catch rate did not necessarily correspond with an increased percentage of anglers with catch.

Data from a fishery on the River Ouse, Yorkshire, 1971 to 1990 (Figure 5.88), revealed a decline in percentage of anglers with catch over the period (Axford 1991), with values below those at the Stoke Bardolph, River Trent. Hence, angling success, measured in terms of catch per unit effort and percentage of anglers with catch, was greater on the River Trent than the River Ouse.

Temperature data from 1990 to 1997 allowed the relationship between monthly catch rate and temperature to be determined (Figures 5.89 to 5.90). In all angling seasons the angler catch rate declined as the water temperature decreased. Hence, angler catch rates were generally higher during summer than winter. It is thought the lower water temperatures adversely affect the feeding rate and behaviour of fish for successful angling, and highlights the effects of the decommissioning of power stations in the Trent Valley on winter angling results. An exception was seen in season 1995/96 when a poor data set may have resulted in ambiguous results.

The winter angling results further decreased as a result of the improvements in water quality (Section 5.1.2) in the river. This generally resulted in increased water clarity during periods of low rainfall. This has been thought to increase the winter shoaling behaviour of the fish, causing large concentrations of fish in the winter migratory areas, leaving large areas of the river with a low fish density (Jacklin 1996). This tight shoaling behaviour by the fish populations in winter in known areas was shown in the hydro-acoustic surveys carried by the Environment Agency (J. Lyons pers. comm.). This winter fish behavioural pattern is likely to adversely affect winter angling results over large areas of the river.

The winter shoaling behaviour of fish at Stoke Bardolph was also reflected in the fisheries surveys. Fish were only captured and observed in the weirpool and sewage outfall at Stoke Bardolph between October and March.

The species composition of angler catches over the period 1969 to 1997 was measured as their relative importance in catches (Section 3.5.12) (Cowx and Broughton 1986; Cowx 1991). Fifteen species were recorded in catches, with seven species, roach, common bream, chub, perch, gudgeon, bleak and dace, being dominant.

Roach were the major species angled over the period (Figure 5.91) and their relative importance declined between 1969 and 1989, as the biotic and abiotic conditions of the river improved. Svardson (1976), Burrough *et al.* (1979) and Persson (1983) all associated roach dominance with nutrient rich, turbid, eutrophic conditions, so the reduction in organic enrichment probably explains the change.

Between 1990 and 1997, the relative importance of roach increased again towards historical levels. This may be due to decreased inter-specific competition from other species, allowing roach to further dominate the cyprinid populations. The changing water quality of the river appears to have affected the growth rate of the roach populations, which now grow slower compared with historical populations (Figure 5.94).

The change in the water quality of the river, which had an adverse effect on the roach populations, favoured the common bream populations between 1969 and 1984. This was seen by an overall increase in their relative importance in angler catches. This increased level of importance was maintained up to 1997 (Figure 5.91). The common bream represented in the angler catches often exceed 1.5 kg in weight, hence, making a significant contribution to angler catch rates. The present growth rate of common bream exceeds that found between 1987 and 1993 (Figure 5.95).

The chub population increased in importance in angler catches between 1969 and 1984, as a number of strong year classes resulted in chub being a major component of the catch (Figure 5.91). From 1984 onwards, chub relative importance declined, especially between 1990 and 1997. This was reflected in the poor chub catches in the electric fishing surveys (Section 5.5.1, 5.6.1, 5.7.1). It is thought this is the result of the natural mortality of a number of strong year classes dating back to 1969 and 1976 which were not replaced due to weak recruitment over the last decade. The present growth rates of chub are similar to rates observed between 1986 and 1994 (Figure 5.96; Jacklin 1996).

The contribution of perch to angler catches (Figure 5.92) has improved since 1969, reflecting the improved water quality and recovery from perch ulcer disease. Gudgeon, bleak and dace have all declined in angler catches over the period (Figure 5.92, 5.93).

The total angler effort on the stretch declined markedly between 1984 to 1997 (Figure 5.94). The lowest effort was recorded in 1995/96 with 2683 man hours recorded. 1996/97 saw an effort of 19150 man hours, which was a reduction of 63 % since 1990/91. Although a reduction in summer angler effort has been seen, this decline in angler effort is due to very low winter angler usage, related to generally poor winter angling success. The total catch has fallen correspondingly (Figure 5.97).

Species composition of fish populations

Roach were the dominant species in electric fishing surveys on the stretch, consisting between 31 and 56 % of catches (Figure 5.98), and this was reflected in angler catches (Figure 5.91). Common bream were poorly represented in catches. However, angler catch data suggest common bream were present in greater numbers in the stretch (Figure 5.91). Common bream inhabit the deep and slow flowing areas of the river. Hence, these fish were likely to be out of the effective range of the electric fishing gear. Chub were present in low proportions in the electric fishing catches, reflecting their contribution to angler catches (Figure 5.91).

Pike and perch were present in low proportions, with perch comprising 1 to 5 % of all catches and pike comprising 7 to 11% (Figure 5.98). In June 1998, no pike were captured although eight were observed immobilised out of hand netting range. The pike and perch were generally captured in the weirpool area.

The minor angling cyprinid species, gudgeon and bleak, contributed high proportions of the catch in some surveys (Figure 5.98).

Thus, the dominant species in the section was roach, a pattern similar to angler catches and the electric fishing surveys at Beeston and Trent Bridge. Bream, chub and hybrids were poorly represented, while perch and pike were present, and gudgeon and bleak were also evident.

Length frequency distribution

The roach population was dominated by fish in the size range 110 to 210 mm, with individuals to 300 mm (Figure 5.99). A significant number of individuals below 10 mm were present in all survey years, especially October 1995 to March 1996, which showed good capacity for recruitment. Hence, in the stretch there was a good recruiting population of roach present, which was able to enter the mature stock, with individuals capable of attaining relative large sizes.

Common bream were present in small numbers due to the inefficiency of the gear. The presence of bream above 400 mm in the third survey year showed a resident spawning stock.

Chub were poorly represented in the electric fishing catches, but were caught in the size range 110 to 480 mm (Figure 5.100). Although the results revealed weak annual recruitment, a mature spawning stock was present with the potential to attain large relative individual sizes.

A small population of hybrids was present in the stretch with fish up to 240 mm present. Perch were also present in small numbers, with fish up to 400 mm being caught. Perch were caught in the size range 90 to 260 mm, indicating a recruiting population was present.

Pike were represented by fish in the size range 470 to 850 mm, with fish in the size range 600 to 860 mm dominant (Figure 5.101).

Year class strength

Only the year class strength for roach was calculated as insufficient data were available to generate accurate year class strengths for the other fish species. A very strong roach year class was obtained in 1988, with strong year class strengths also occurring in 1990, 1991 and 1992 (Figure 5.102). These strong year classes contributed to angler catches during the study period. Weak year classes were observed in 1993 and 1994 (Figure 5.102). Analysis of the temperature profile of the River Trent over the period, expressed as annual number of degree days above 14°C, shows correlation between year class strength and temperature ($r = 0.77$). A similar relationship was also observed at Beeston and Trent Bridge (Section 5.5.1 and 5.6.1).

Growth

The growth of roach, common bream, perch and chub were compared with national standards. Roach growth rates were average and similar to the national standard for the species (Hickley and Dexter 1979) with a maximum age attained of 8+ (Figure 5.103). These relationships were also observed at Beeston and Trent Bridge. Common bream growth rates were above the national standard (Hickley and Dexter 1979) and comparable with waters where bream growth was considered good (Figure 5.104). The oldest common bream aged were 12 years of age.

The growth rate of perch was above average (Figure 5.105). The oldest perch caught and aged was 6 years old. Older perch were present in samples but scales were either replacement or too unclear to enable accurate reading. The growth rate of chub was average, with fish up to 13 years being aged (Figure 5.106).

Bleak and gudgeon growth rates were slightly above average UK rates (Table 5.27). Bleak lived to a similar age to other UK populations while no gudgeon over 2+ years was captured.

Table 5.27 Growth rate of bleak and gudgeon at Stoke Bardolph, River Trent, compared with other UK rivers.

		Length at age (mm)							
Species	Location	I	II	III	IV	V	VI	VII	VIII
Bleak	Beeston, R. Trent	48	89	114	134	154			
	Standard growth	42	76	107	120	125	147		
	River Severn	41	75	104	120	127			
	River Thames	39	70	92	111	121	130	138	140
Gudgeon	Beeston, R. Trent	58	99						
	River Thames	55	86	104	117	123	129		
	River Frome	42	96	120	137	152			

(Williams 1967; Mann 1973; Cowx *et al.* 1995)

L infinity and K

Values obtained from the Von Bertalanffy growth model for roach, common bream, perch and chub (Table 5.28) were similar to those derived for the species in other UK fisheries (Tables 4.4 to 4.6).

Table 5.28 **L_∞ and K values derived for roach, common bream, perch and chub at Stoke Bardolph, River Trent.**

Species	L_∞	K
Roach	361	0.15
Common bream	515	0.14
Perch	429	0.14
Chub	727	0.09

Mortality and survival rates

The mortality (Z) and survival (S) rates at Stoke Bardolph varied between species (Table 5.29) and were comparable with other UK fisheries (Table 5.5). In general, the mortality rates were relatively low when compared with other UK fisheries, indicating good survival in the population.

Table 5.29 **Mortality and survival rates of species at Stoke Bardolph, River Trent.**

Species	Z	S	If N_t=1000, N_{t+1}=
Roach	0.46	0.63	630
Common bream	0.33	0.72	720
Chub	0.37	0.69	690
Perch	0.44	0.64	640

5.7.2 *Status of cormorant populations*

The status of the cormorant populations at Stoke Bardolph was assessed by monitoring of Section D, (Section 5.2), which covered the areas used in the fisheries surveys. These cormorants utilised the Attenborough night roost (Section 4.5.1; Figure 4.13).

Cormorant site occupancy

The number of cormorants utilising Section D, Stoke Bardolph, declined over the study period, from a peak of 96 birds in 1995/96 to the 1997/98 peak of 41 birds (Figure 5.107). In comparison to Sections A to C, very few of these cormorants were observed flying, with the main use of the site being a day roost. Similar to Sections A to C, only a small number of the birds were observed feeding at the site.

Feeding success

The feeding success of cormorants on Section D of the River Trent (Stoke Bardolph) varied between 32.1 and 43.8 % of all foraging bouts resulting in ingested fish (Table 5.30). This success is slightly lower than the feeding success observed on Sections A to C of the River Trent (Table 5.6 and 5.19). In comparison with other studied fisheries, the success rates were below those obtained for Holme Pierrepont (Section 4.5.4) and above those in the pre-trout stocking period at Colwick Park Trout Lake (Section 6.5.4).

The proportion of dives that were successful was low in all years for Sections D, with between 7.2 and 8.2 % of dives being successful (Table 5.30). This was similar to dive success on Sections A to C (Table 5.6, 5.19) and comparable with Holme Pierrepont

(12.7 to 15.2 %, Section 4.5.4) and Colwick Park Trout Lake (1.8 to 13.9 %, Section 6.5.4).

Table 5.30 Feeding success of cormorants at count section D, Stoke Bardolph, River Trent.

	1995/96	1996/97	1997/98
Observed foraging bouts	56	80	62
Successful foraging bouts	18	35	25
% foraging bouts successful	32.1	43.8	40.3
Dives observed	670	754	548
Successful dives	48	62	44
% dives successful	7.2	8.2	8.0

5.7.3 *Impact of cormorants at Stoke Bardolph*

Species composition of fisheries surveys and cormorant predation

Observations on cormorant feeding behaviour did not allow the identification of fish predated on to species level. Fish were classified as cyprinids, perch and pike. Accordingly, all fisheries data collected from electric fishing surveys and angler catch returns were reclassified as cyprinids, perch and pike to allow direct comparison.

In electric fishing surveys, cyprinid species were dominant, comprising 85 to 92 % of fish community (Figure 5.108). Pike and perch were present in smaller numbers with eels comprising less than 1 % of all catches.

Cormorants selected the dominant cyprinids for predation and comprised 52 to 77 % of fish eaten. However, these values were generally lower than in the population (Figure 5.108). The contribution of perch was greater in cormorant diet than in the electric fishing surveys. Eels comprised a significant proportion of cormorant diet in 1997/98, much higher than the proportion found by electric fishing.

Hence, the cormorants selected cyprinids, perch and eels in their diet. The proportions of perch and eels in the diet were above those found in the fish population.

Length frequency distribution

The size of cyprinids eaten by cormorants was dominated by fish below 100 mm (Figure 5.109). However, fish up to 200 mm were taken in all years and fish of 250 mm were taken in 1996/97. The cyprinid populations sampled in electric fishing surveys (Figure 5.109) were dominated by fish in the size range 50 to 250 mm, with individuals present up to 500 mm, the latter mainly comprising common bream and chub. Hence, cormorants selected the dominant size classes of cyprinid for consumption.

The growth rates of cyprinids at Stoke Bardolph (Section 5.7.1) showed roach, bream and chub populations were most vulnerable to cormorant predation for their first two years. However, the risk of cormorant predation remained throughout the life of roach, and up to age 5 for bream and 6 for chub. This relationship is similar to those found at Beeston and Trent Bridge (Section 5.5.3 and 5.6.3), but contrasts with Holme Pierrepont (Section 4).

Cormorants selected perch of below 200 mm for consumption (Figure 5.110). Comparison with growth rates reveal perch attain lengths of 250 mm around 5 years of age (Section 5.7.1).

Growth indices

The growth indices for Holme Pierrepont revealed an increase in growth rate of roach from 1994 (Figure 4.20). This was tentatively linked to the appearance of cormorants on the fishery decreasing stock density, decreasing competition and allowing the increased growth.

The annual growth increments varied between years for roach at Stoke Bardolph and showed no overall pattern (Figures 5.111 and 5.112). Comparison of the growth indices and year class strength of roach at Stoke Bardolph (Figure 5.102 and 5.112) showed a general trend of good growth years corresponding to strong recruitment. Hence, 1988, 1991 and 1992 produced strong year classes of roach and good relative growth rate. Year class strength of roach was related to temperature (Section 5.7.1). Therefore, the growth of roach at Stoke Bardolph appeared to be related to annual temperature and independent of the influence of cormorant predation

Cohort reconstruction with and without cormorant predation

Data on the impact of cormorant predation on the cohorts of roach, common bream, chub, hybrid and perch reveal that the cormorants do remove a considerable quantity by number and biomass of the species present (Table 5.31 and 5.32). The difference in the values with and without cormorant predation can **only be attributable to cormorant predation** and not any other source of mortality.

Table 5.31 Numbers and biomass of roach with and without cormorant predation on the River Trent at Stoke Bardolph.

		Number ($n\ ha^{-1}$)	% reduction	Biomass ($g\ ha^{-1}$)	% reduction
1996	With cormorants	3683.9		151629.3	
	Without cormorants	14017.9	73.7	220615.3	31.3
1997	With cormorants	2118.1		127467.4	
	Without cormorants	11326.7	81.3	205358.1	37.9
1998	With cormorants	114.6		23650.5	
	Without cormorants	10093.9	81.3	263395.0	91.0

Table 5.32 Numbers and biomass of perch with and without cormorant predation on the River Trent at Stoke Bardolph.

		Number ($n\ ha^{-1}$)	% reduction	Biomass ($g\ ha^{-1}$)	% reduction
1996	With cormorants	276.9		34720.8	
	Without cormorants	900.9	69.3	61570.0	43.6
1997	With cormorants	92.0		5380.9	
	Without cormorants	1135.0	91.9	71366.8	92.5
1998	With cormorants	92.0		11557.2	
	Without cormorants	1252.2	92.7	122254.8	90.5

After three winters cormorants removed large numbers and biomass of fish from the Stoke Bardolph section of the River Trent, equivalent to a reduction in numbers of 81.3 % roach and 92.7 % perch. Figures 5.113 to 5.124 show how the cormorant predation affected the different age groups of the species between 1996 and 1998, in terms of numbers and biomass.

A large difference was observed between the fish biomass and fish numerical abundance in the fishery revealed by electric fishing and the cormorant predated fish biomass and numerical abundance as calculated by the MCS model (Tables 5.31 and 5.32 , Figures 5.113 to 5.124). However, the electric fishing data were considered reliable because they were similar to the outputs for hydro-acoustic surveys (Table 5.26). Cormorants were estimated to remove between 8979 and 10334 roach per hectare annually over the study period, which is probably an over-estimate (Section 5.8).

When the fish population minimum and maximum values are considered (Table 5.33), using electric fishing efficiencies of P = 0.10 and P = 0.36 (Section 3.3.1), it can be seen cormorant predation had a major effect on the fish populations even where maximum fish abundance values were used (P = 0.10).

Table 5.33 Impact of cormorant predation (CP) on the populations of roach and perch (n ha^{-1}) over the range of electric fishing gear efficiency at Stoke Bardolph, River Trent.

		1996		1997		1998	
Species	**P**	**CP**	**-CP**	**CP**	**-CP**	**CP**	**-CP**
Roach	0.10	7861	18195	4872	14080	2564	11543
	0.23	3709	14043	2118	11327	1115	10094
	0.36	2184	12518	13530	10562	712	9691
Perch	0.10	635	1259	212	1255	212	1372
	0.23	277	901	92	1135	92	1252
	0.36	176	800	59	1102	59	1219

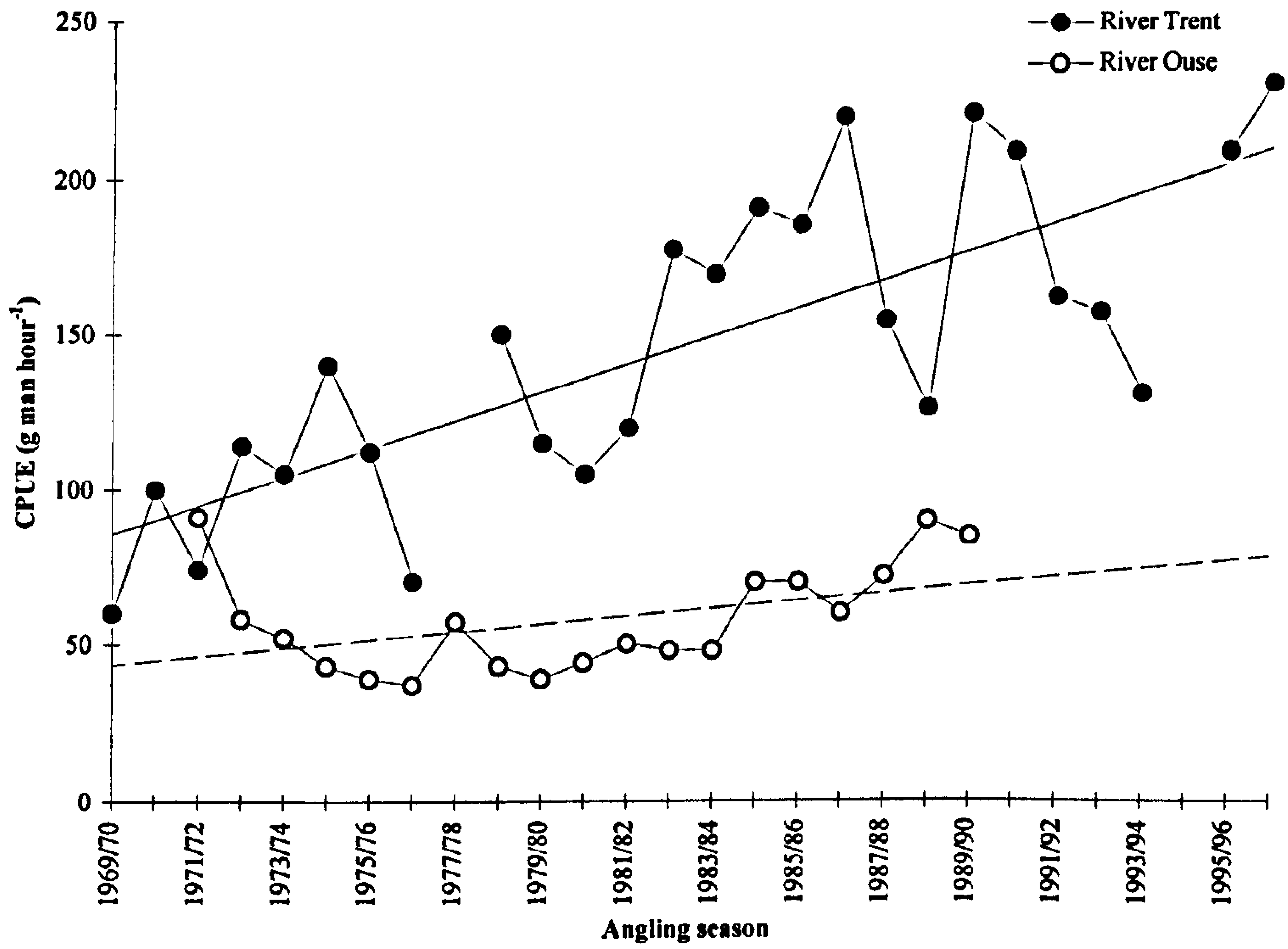


Figure 5.87 Catch per unit effort (CPUE) of anglers at Stoke Bardolph to Gunthorpe, River Trent, 1969 to 1997 compared with the River Ouse, Yorkshire, 1971 to 1990 (Axford 1991).

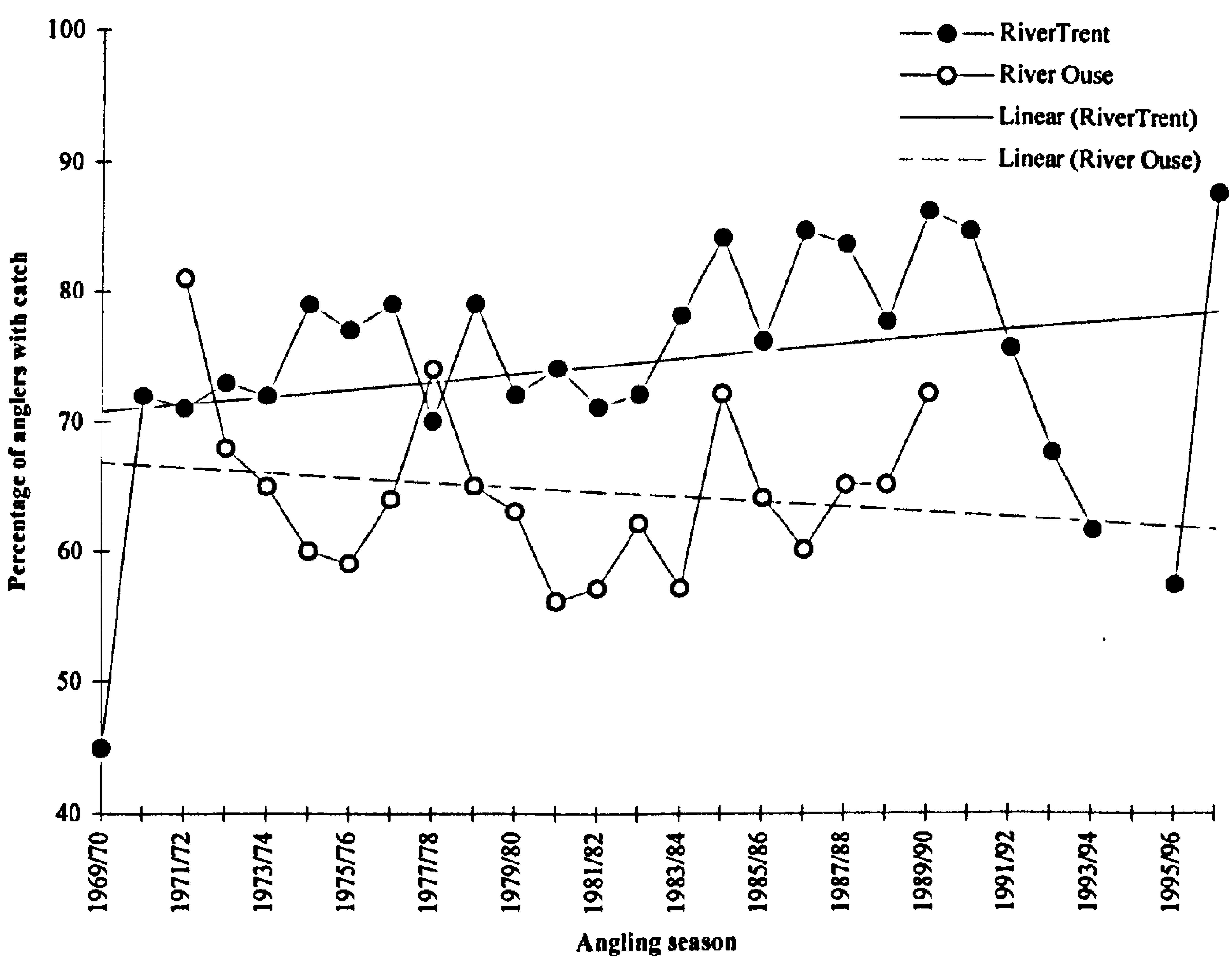


Figure 5.88 Percentage of anglers with catch at Stoke Bardolph to Gunthorpe, River Trent, 1969 to 1997, compared to the River Ouse, Yorkshire, 1971 to 1990 (Axford 1991).

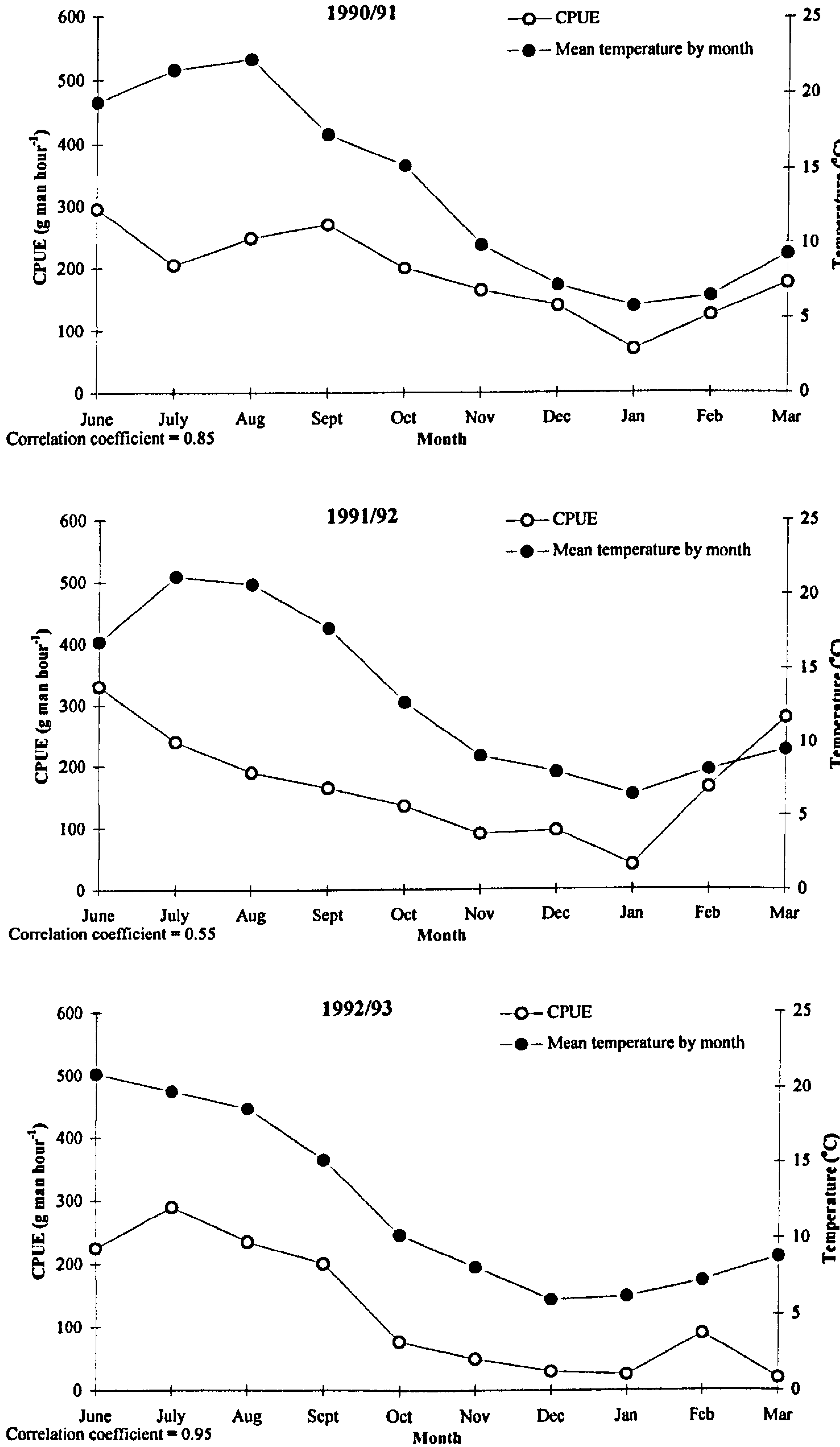


Figure 5.89 Relationship between angler catch per unit effort and water temperature at Stoke Bardolph to Gunthorpe, River Trent, 1990 to 1993.

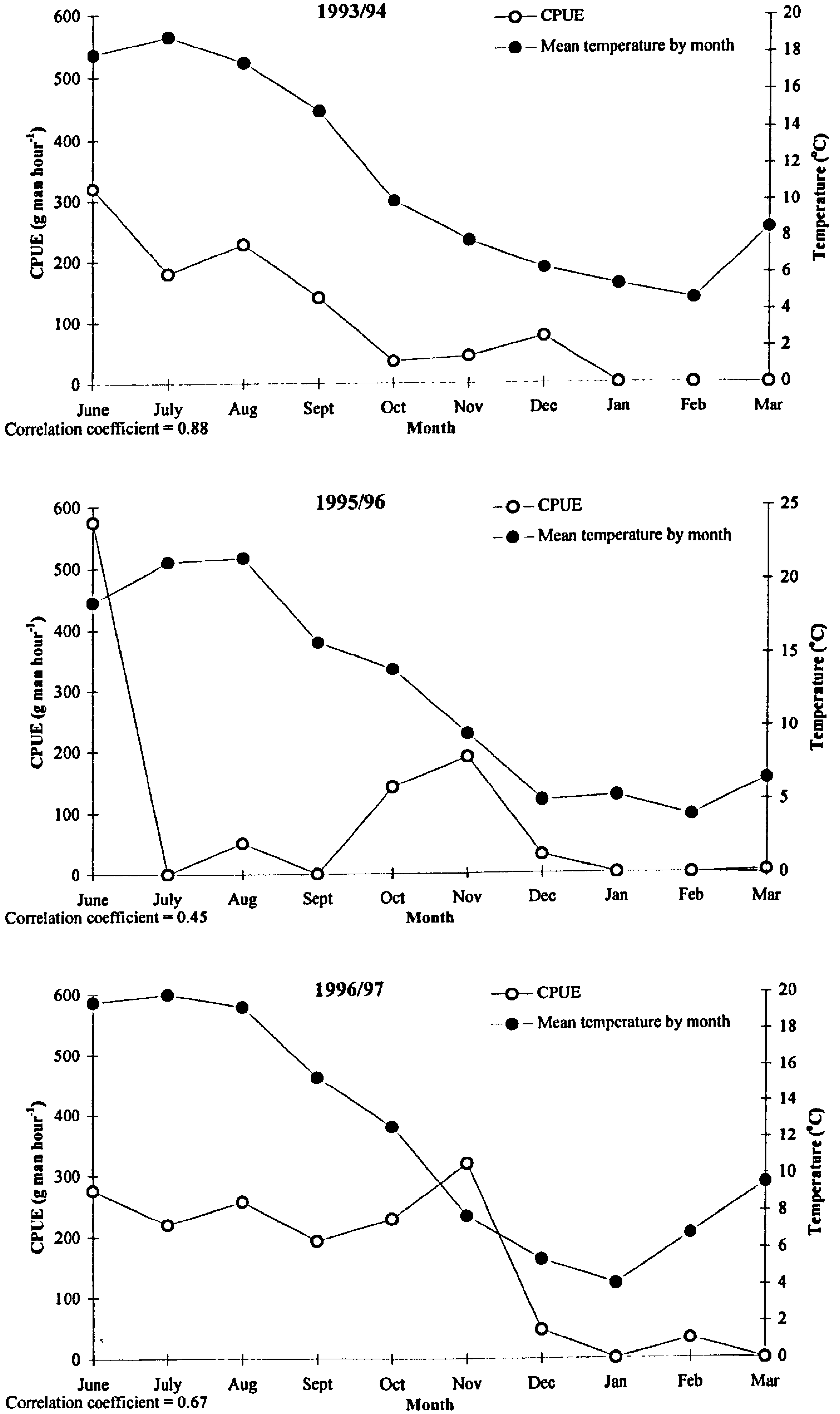


Figure 5.90 Relationship between angler catch per unit effort and water temperature at Stoke Bardolph to Gunthorpe, River Trent, 1993 to 1996.

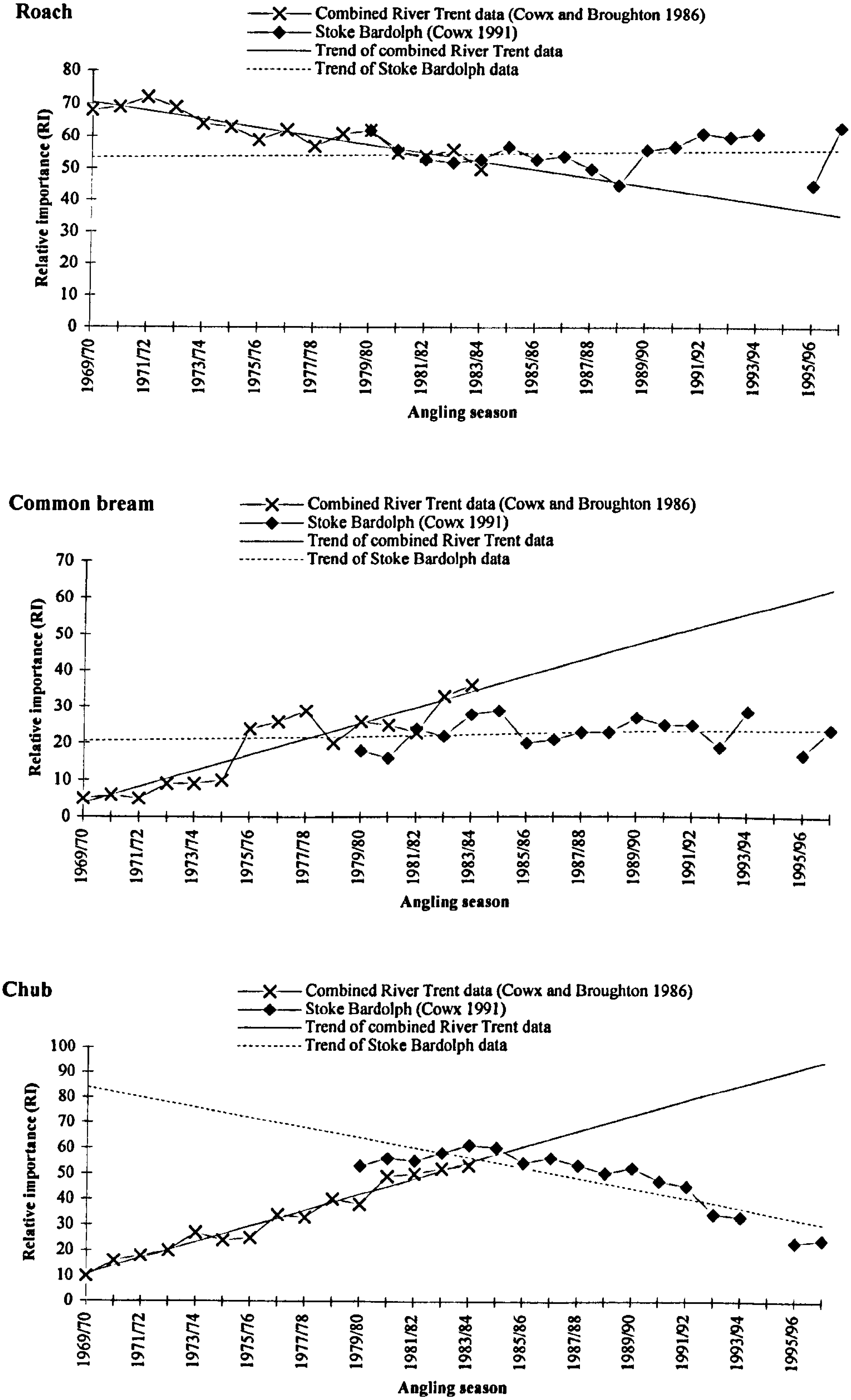


Figure 5.91 Variation with angling season of the relative importance (RI) of species in angler catches from the River Trent, 1969 to 1997.

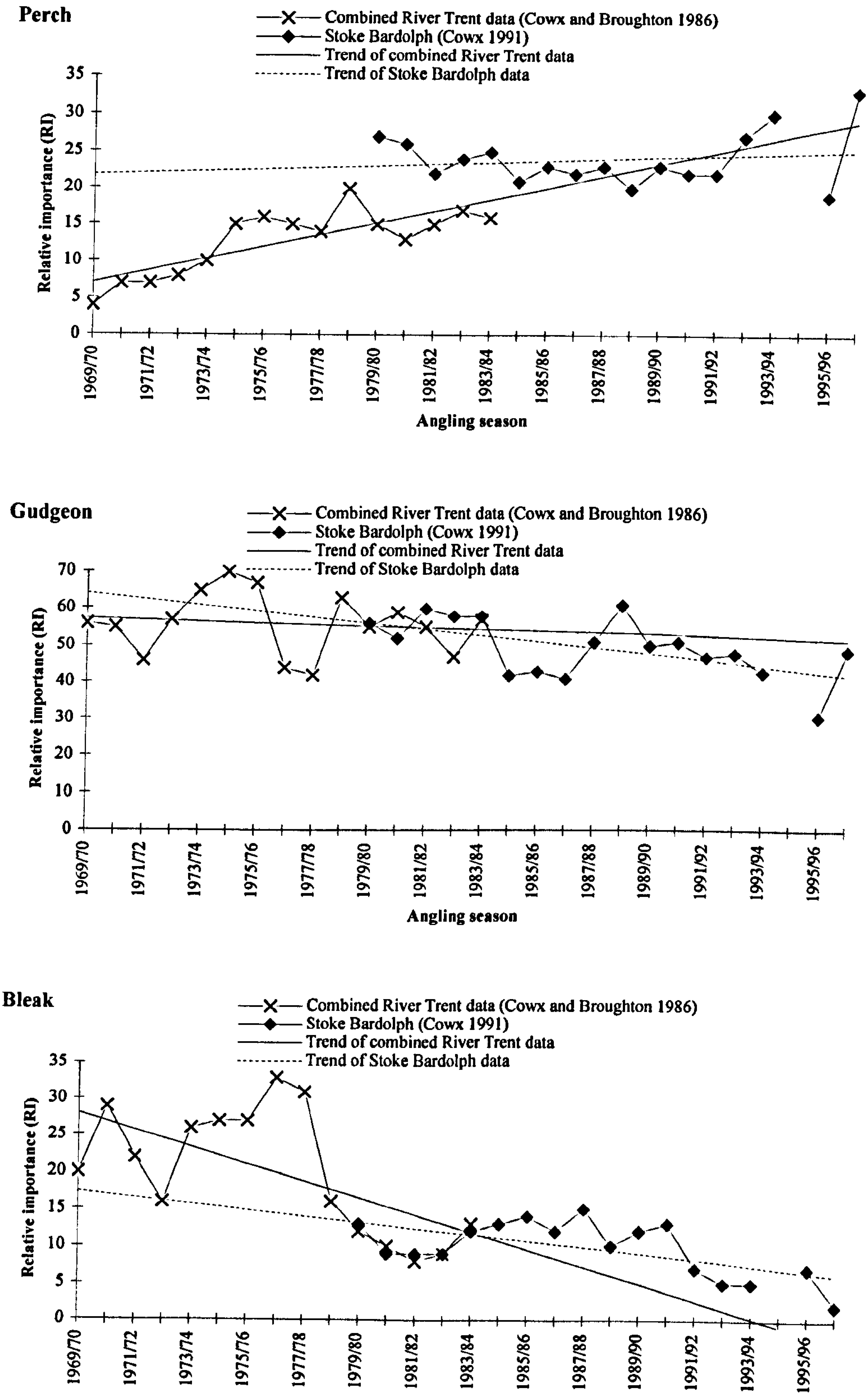


Figure 5.92 Variation with angling season of the relative importance (RI) of species in angler catches from the River Trent, 1969 to 1997.

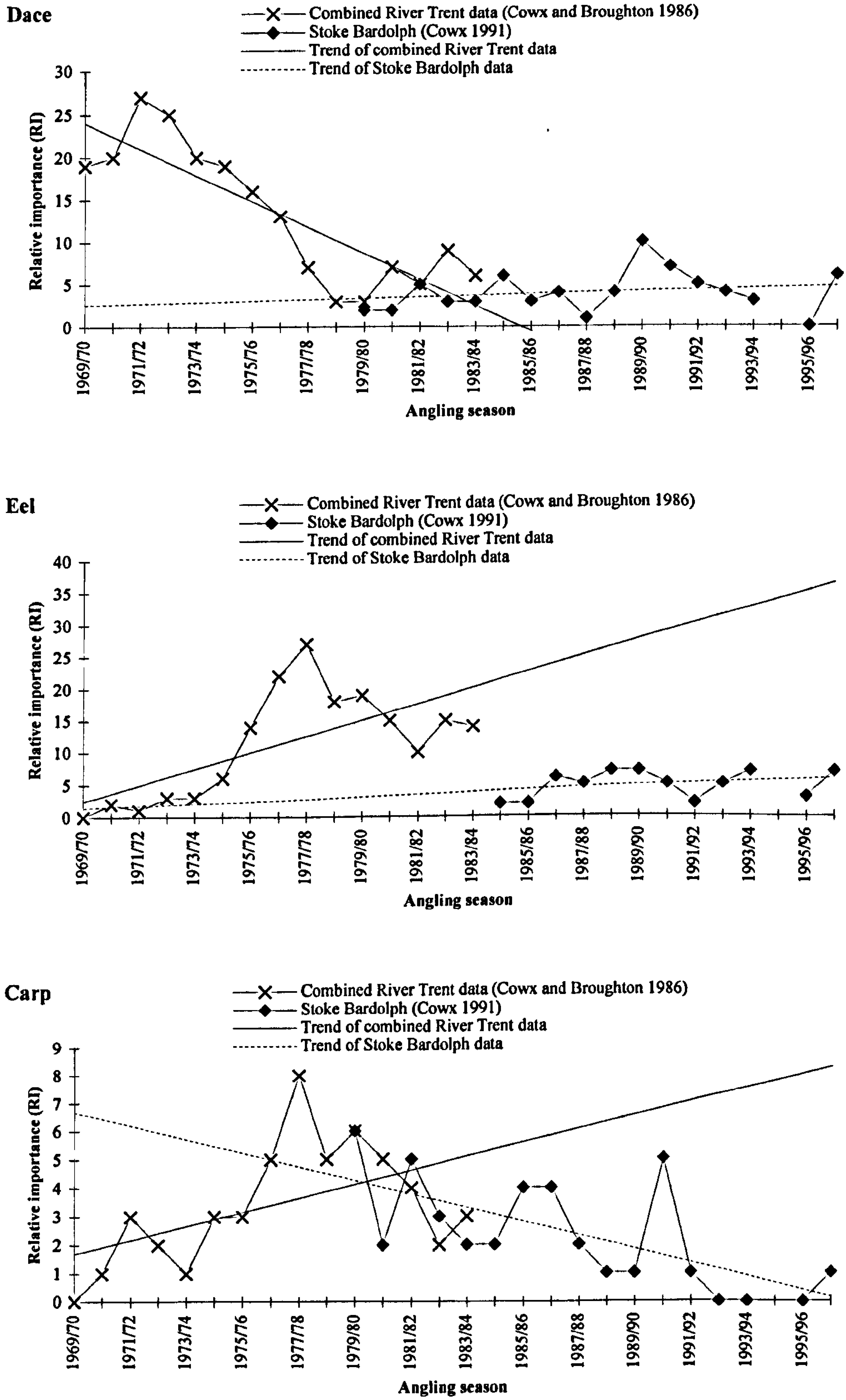


Figure 5.93 Variation with angling season of the relative importance (RI) of species in angler catches from the River Trent, 1969 to 1997.

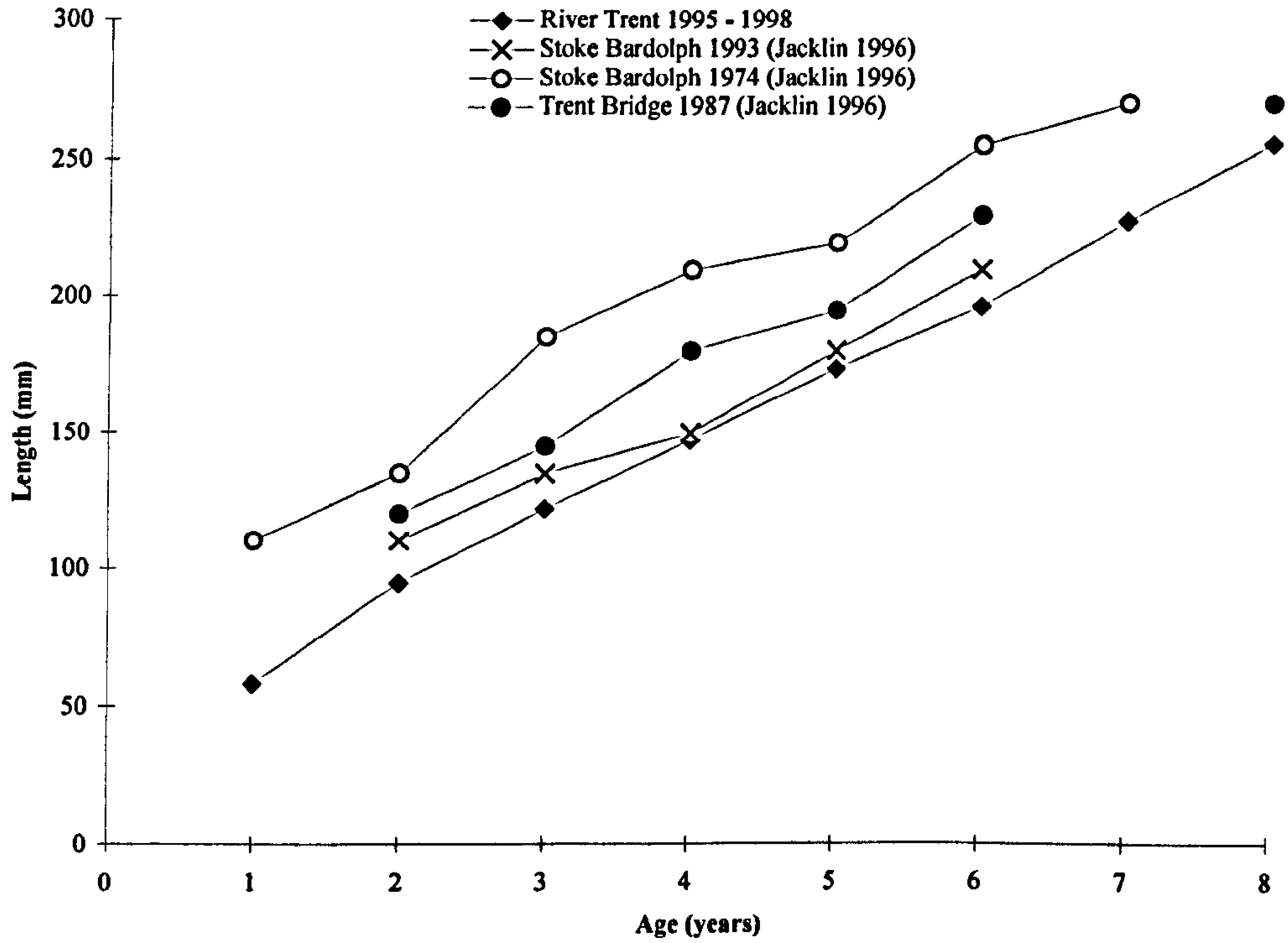


Figure 5.94 Historical and present growth rates of roach from sections of the River Trent (Jacklin 1996).

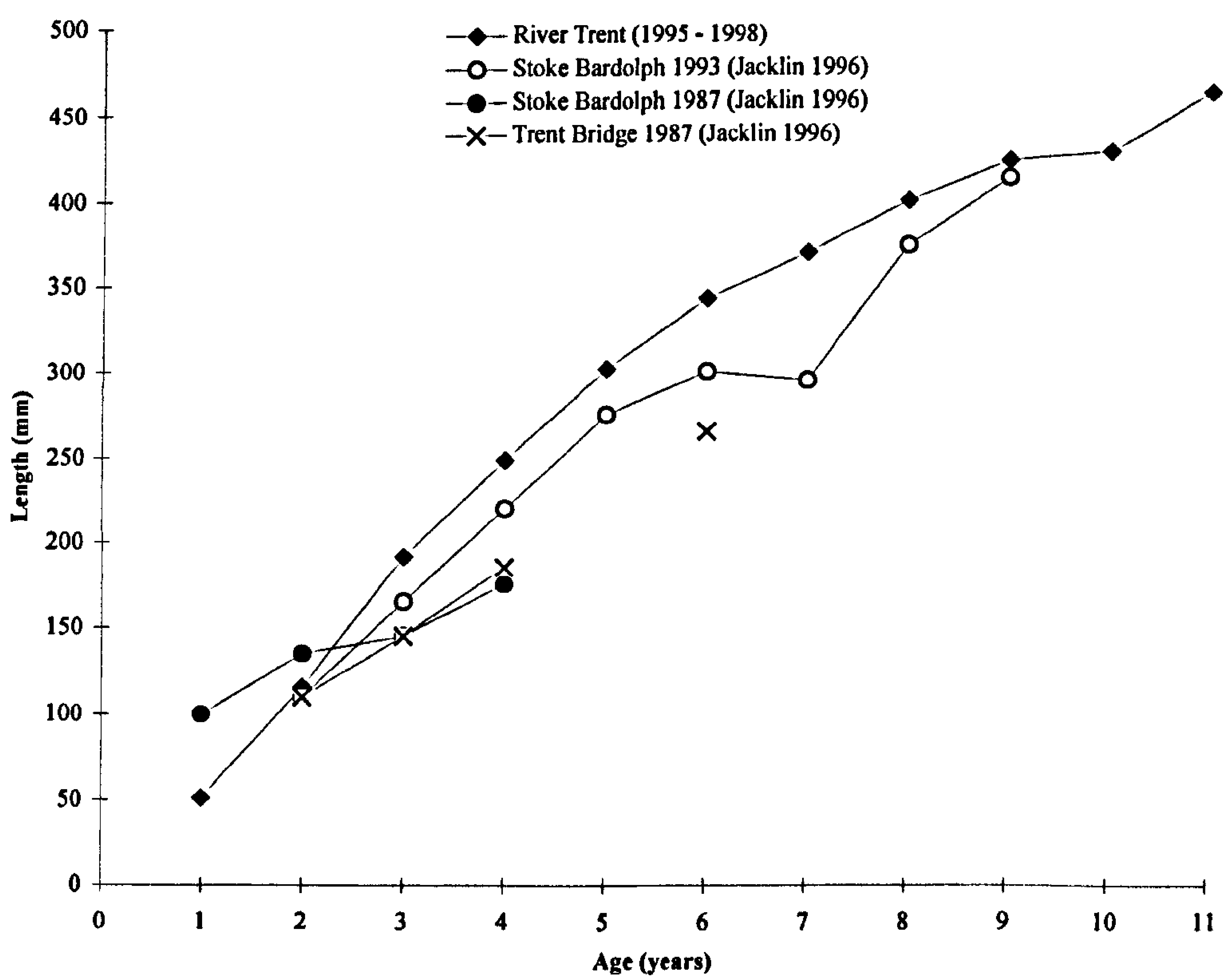


Figure 5.95 Historical and present growth rates of common bream from sections of the River Trent (Jacklin 1996).

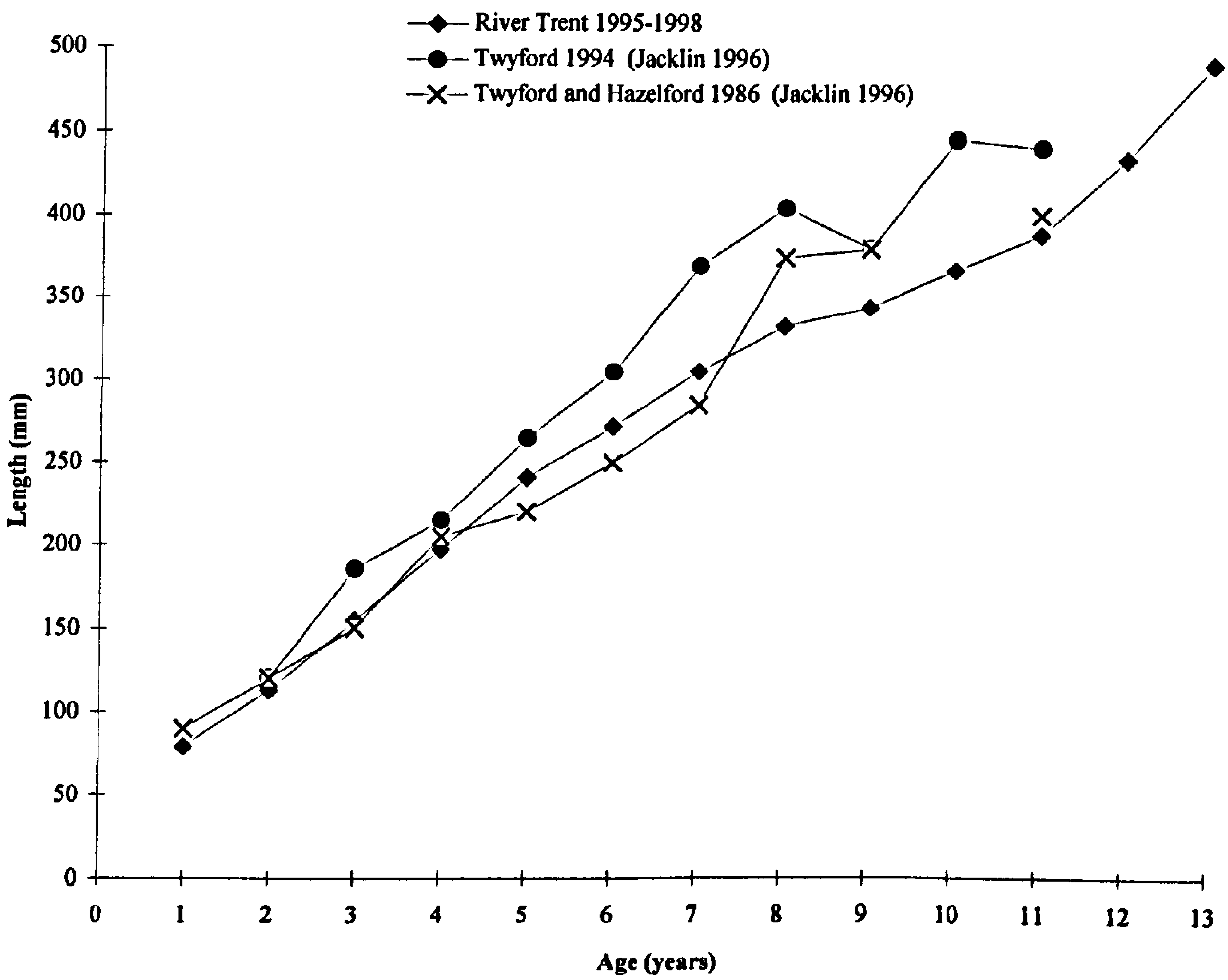


Figure 5.96 Historical and present growth rates of chub from three sections of the River Trent.

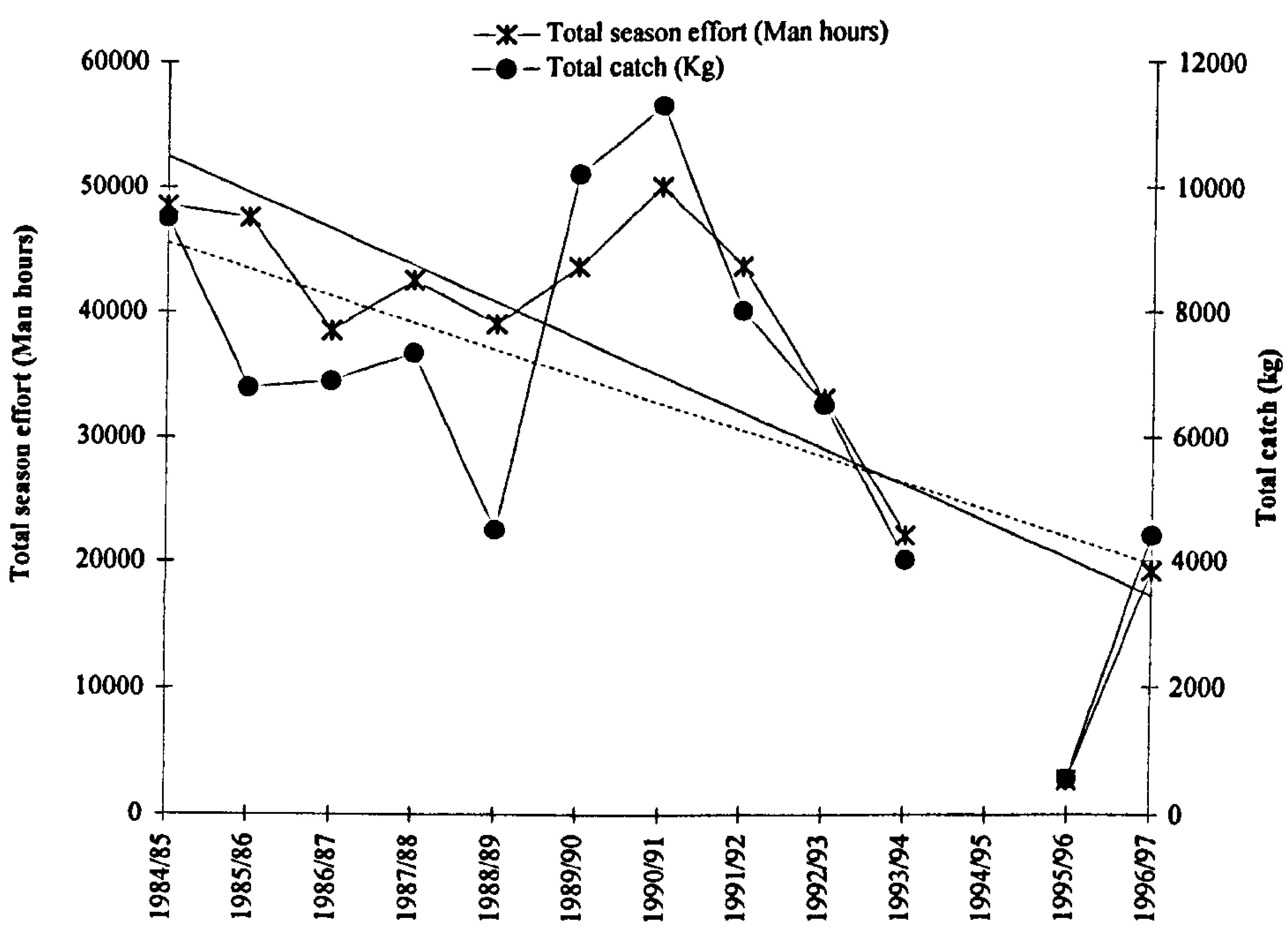


Figure 5.97 Changes in total season effort and total catch of anglers at Stoke Bardolph to Gunthorpe, River Trent, 1984 to 1997 (Jacklin 1996).

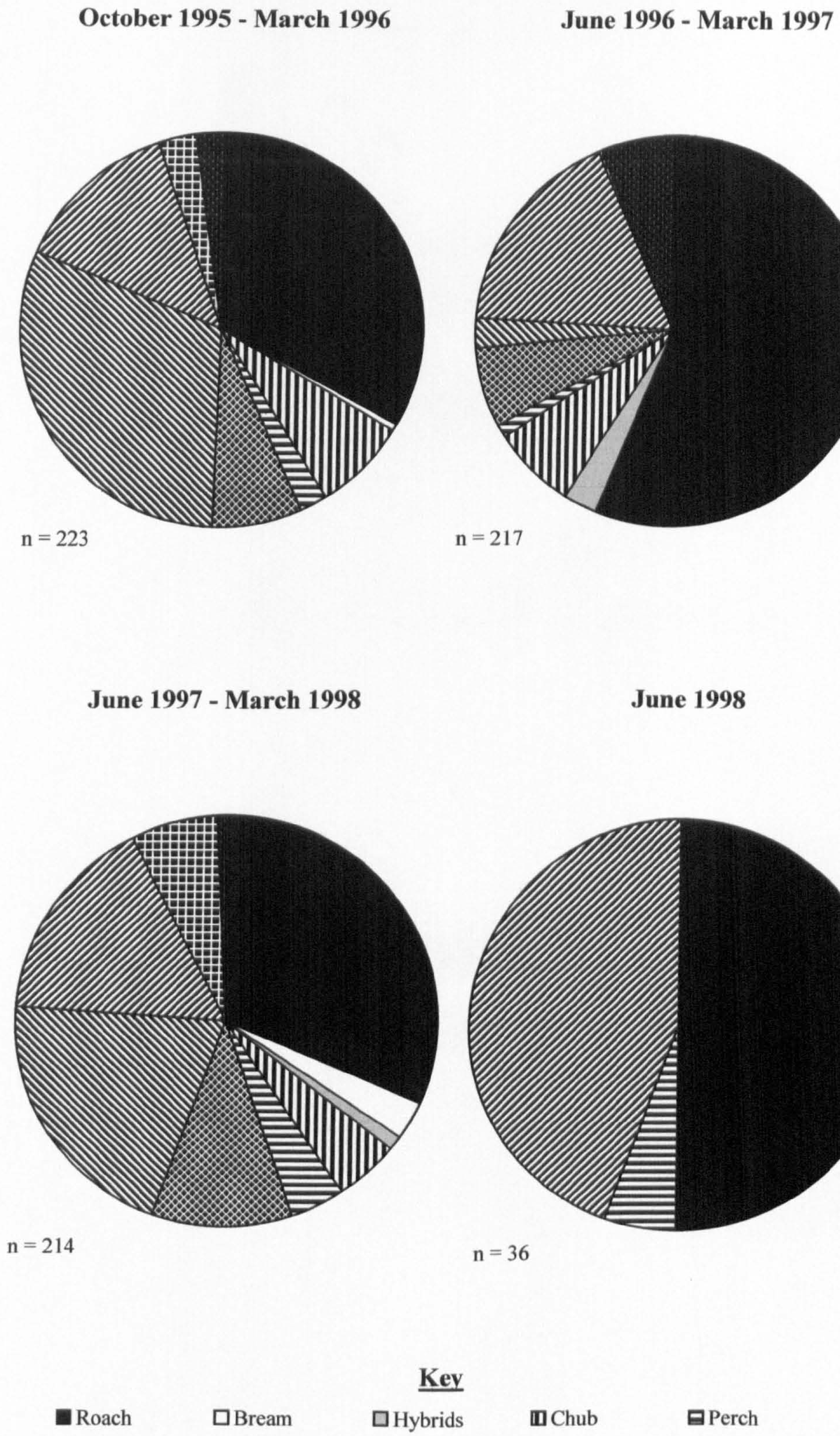


Figure 5.98 Species composition of electric fishing surveys at Stoke Bardolph, River Trent.

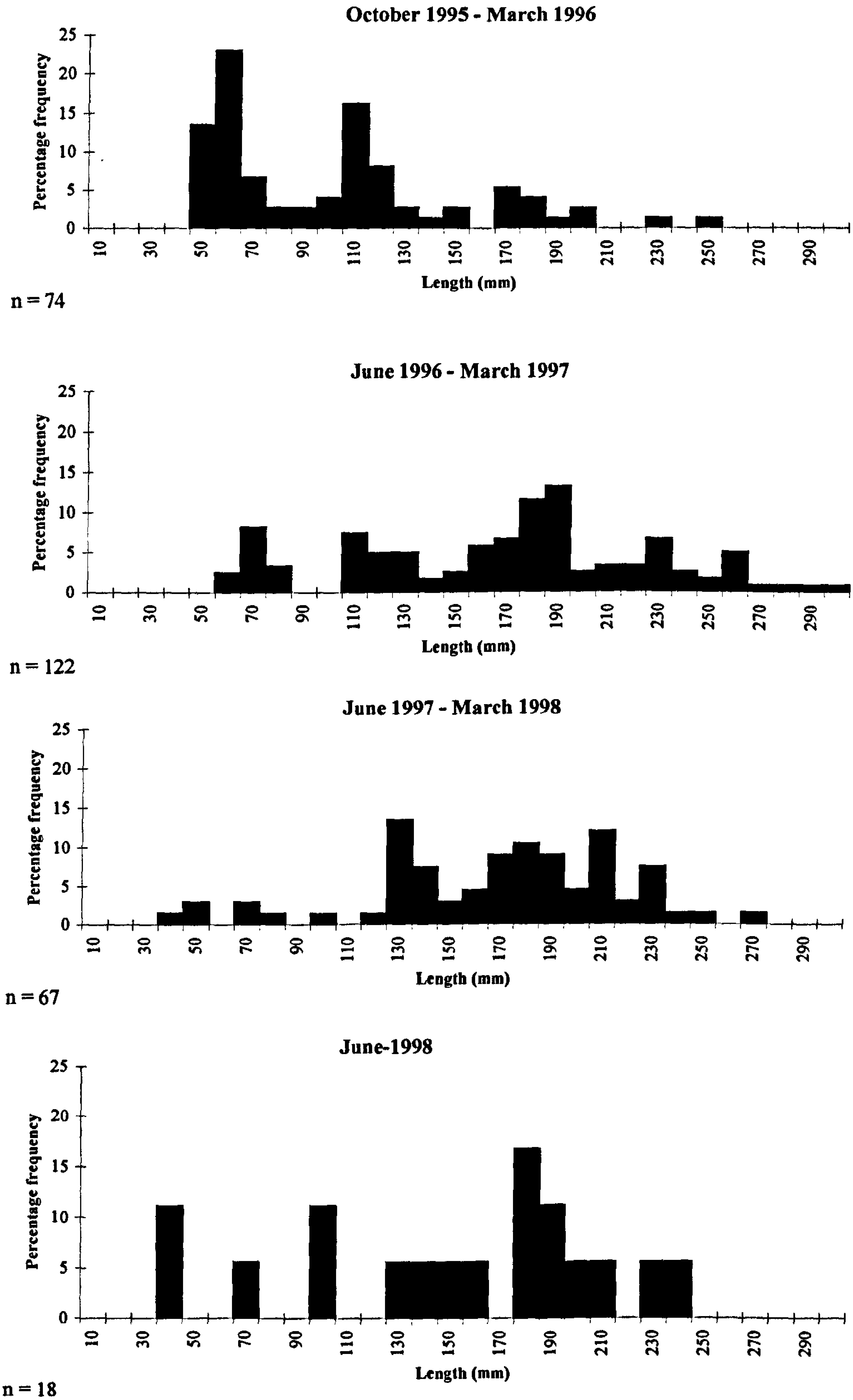


Figure 5.99 Length frequency distributions of roach caught by electric fishing at Stoke Bardolph, River Trent.

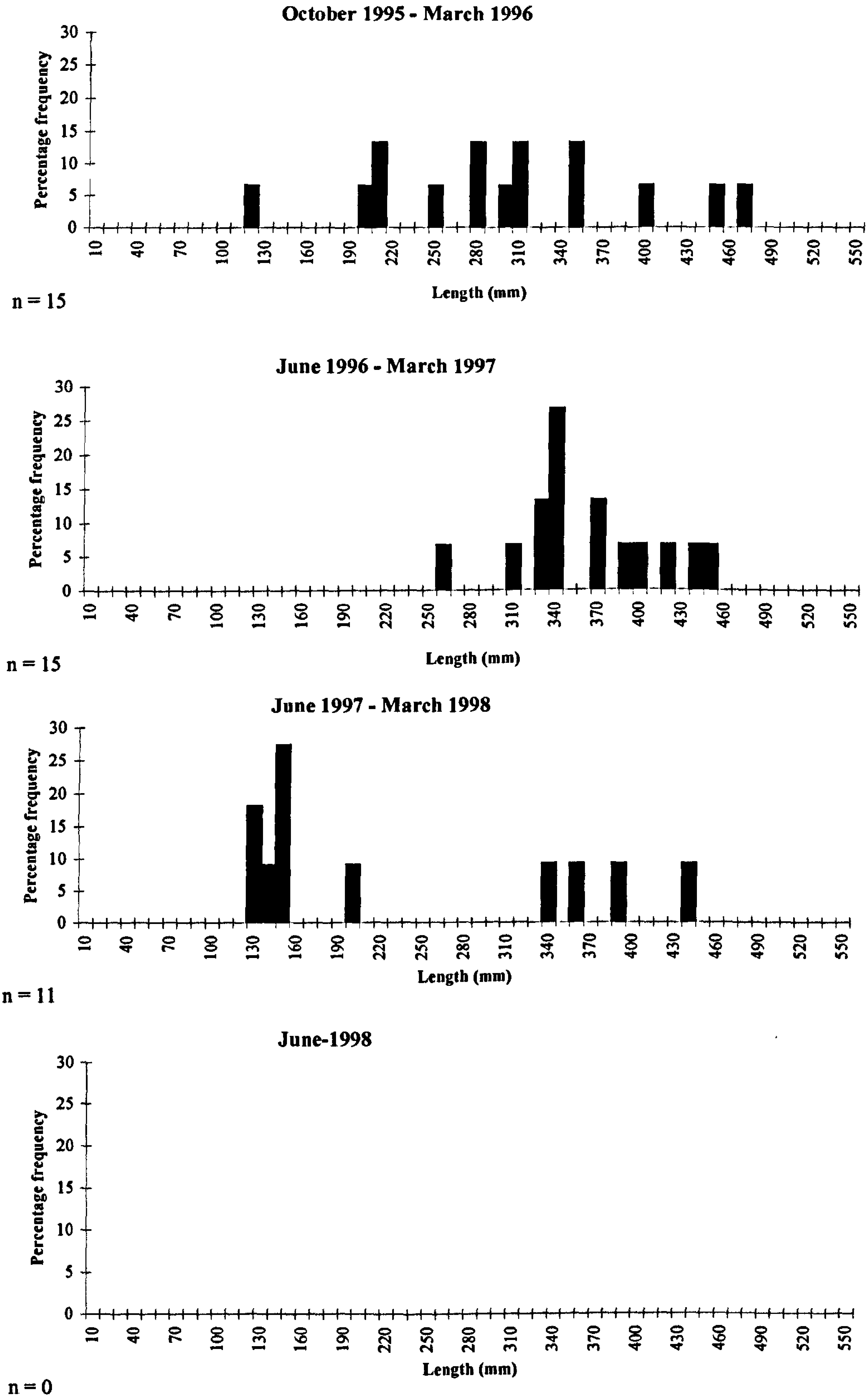


Figure 5.100 Length frequency distributions of chub caught by electric fishing at Stoke Bardolph, River Trent.

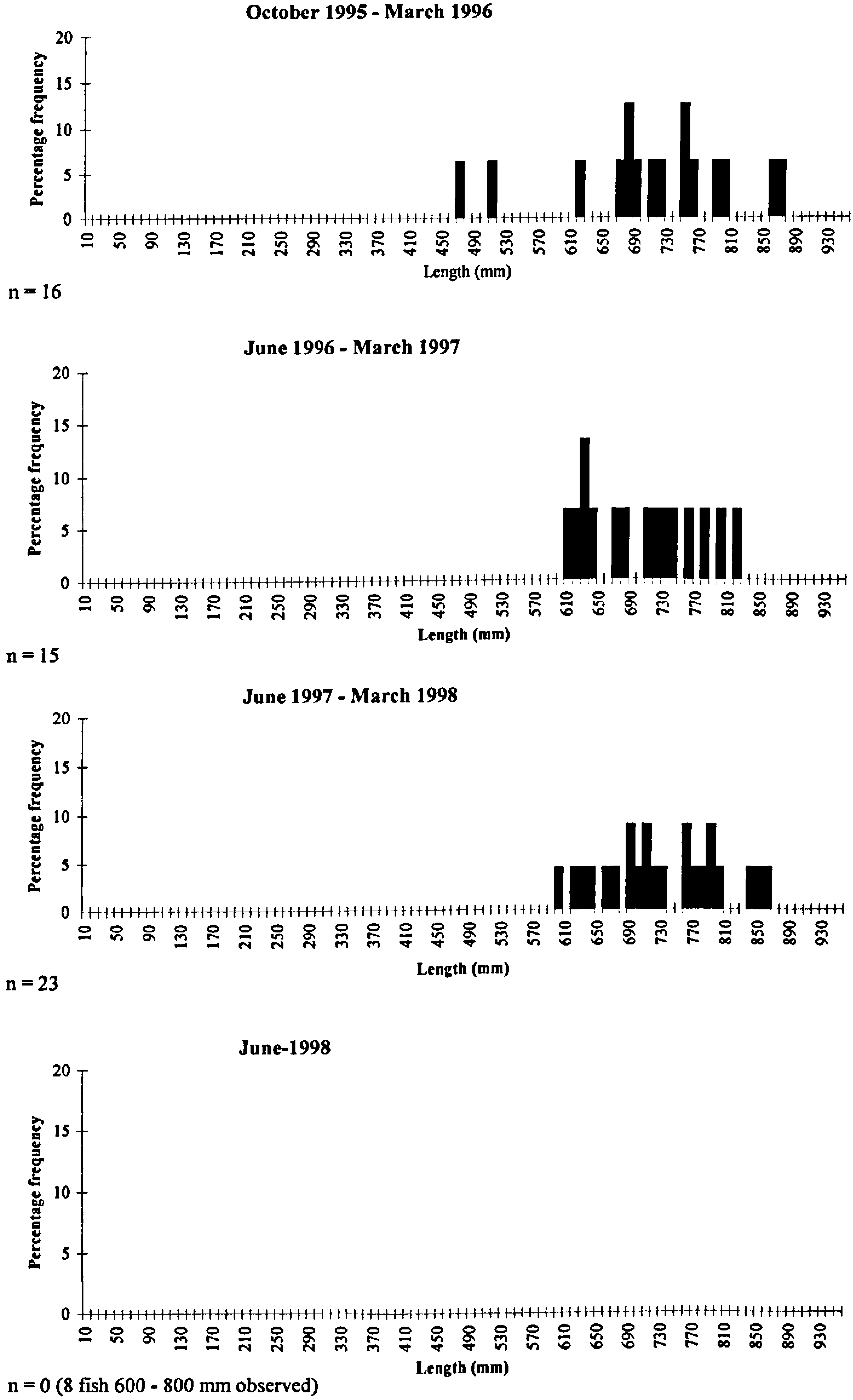


Figure 5.101 Length frequency distribution of pike caught by electric fishing at Stoke Bardolph, River Trent.

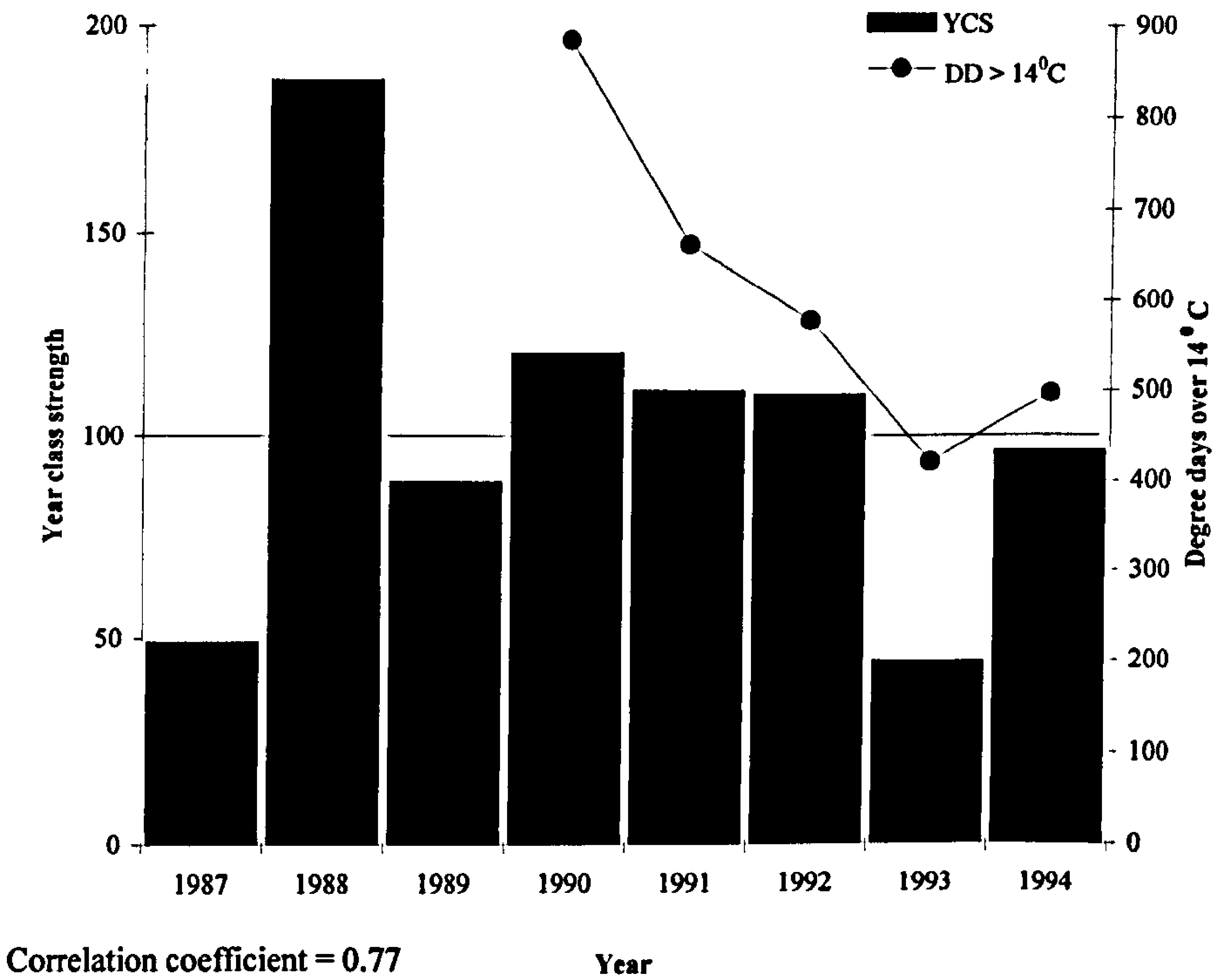


Figure 5.102 Year class strength of roach at Stoke Bardolph, River Trent, and the relationship with degree days over 14°C.

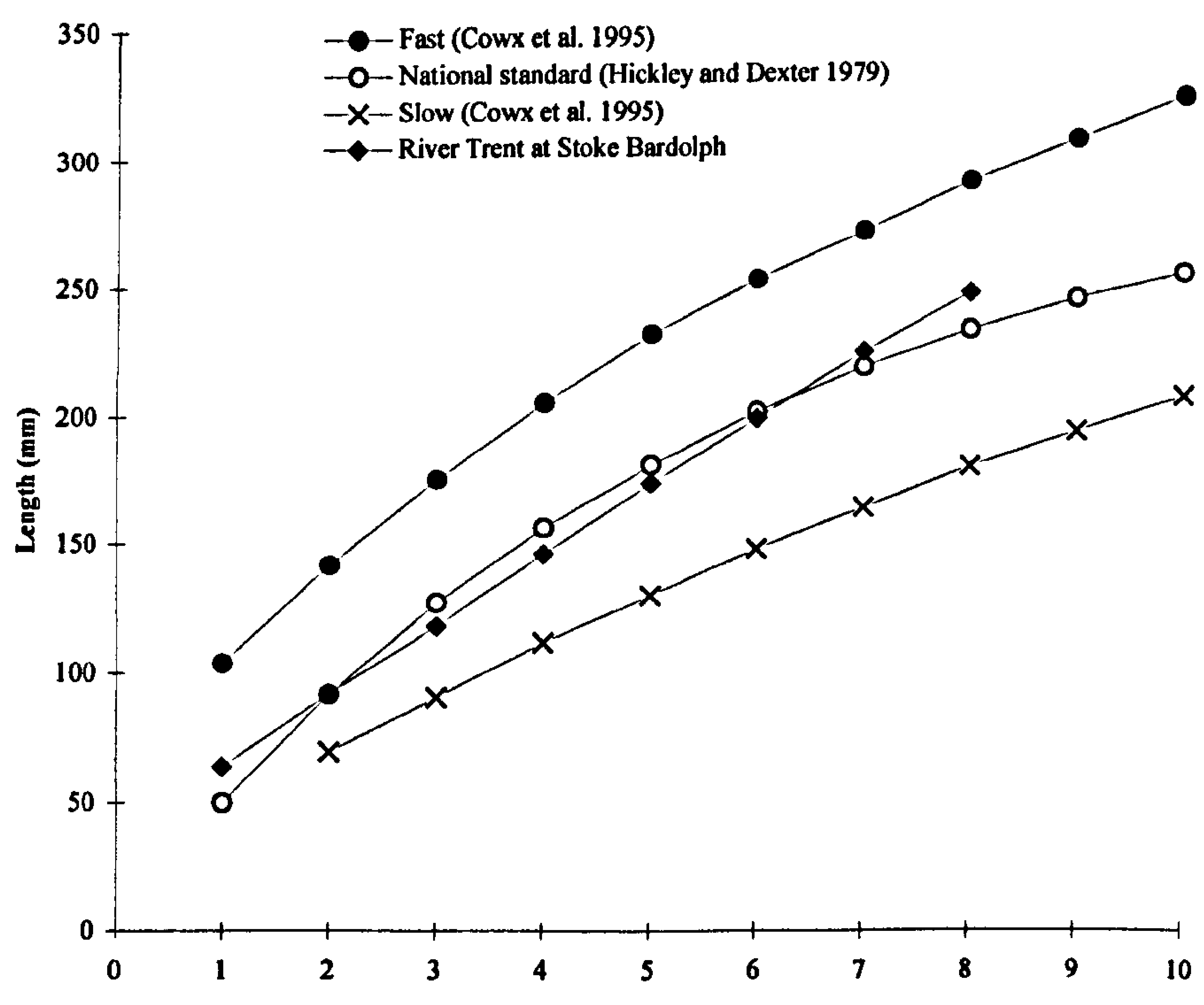


Figure 5.103 Growth of roach at Stoke Bardolph, River Trent compared with standard growth curves.

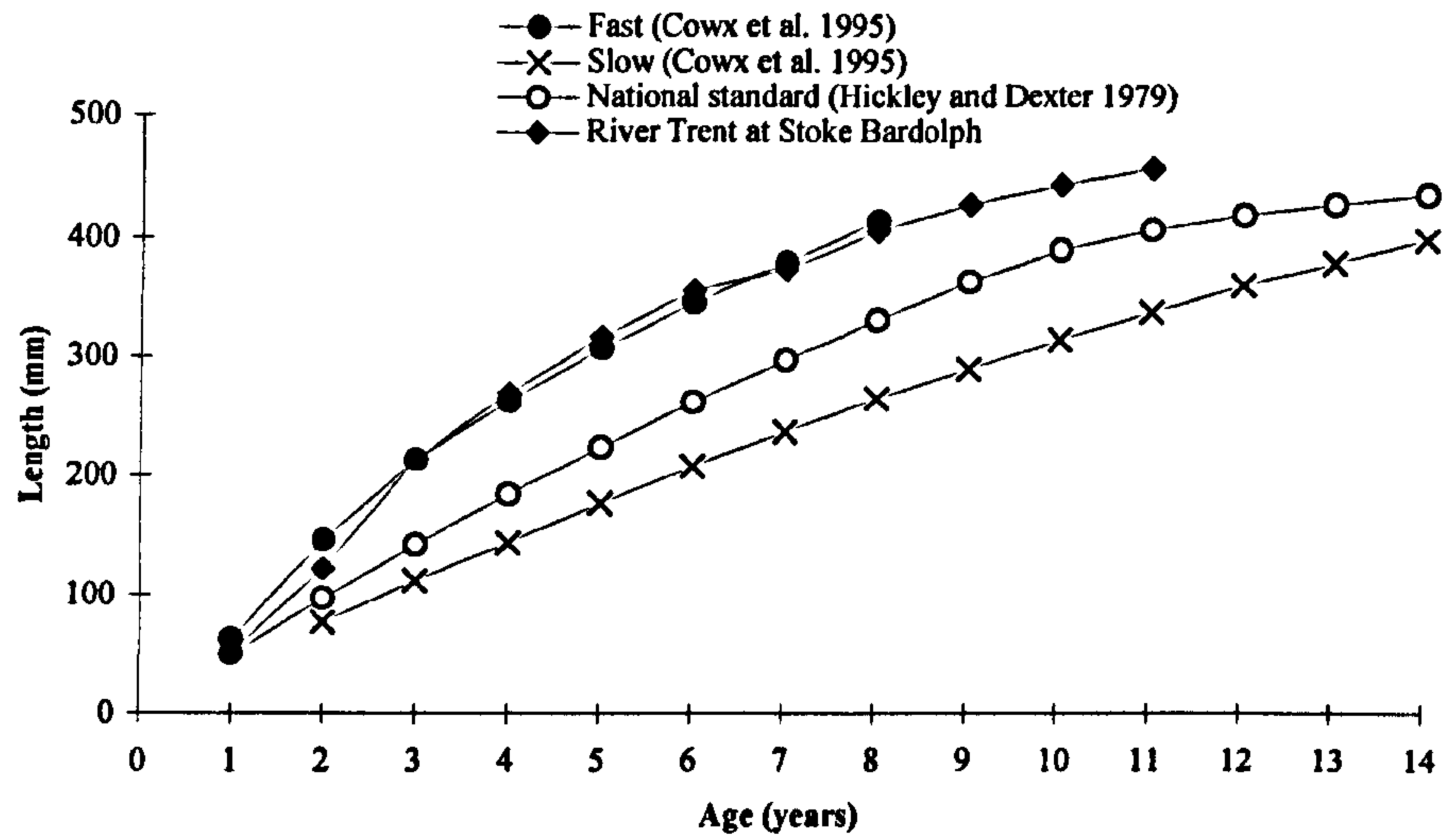


Figure 5.104 Growth rate of common bream at Stoke Bardolph, River Trent, compared with standard growth curves.

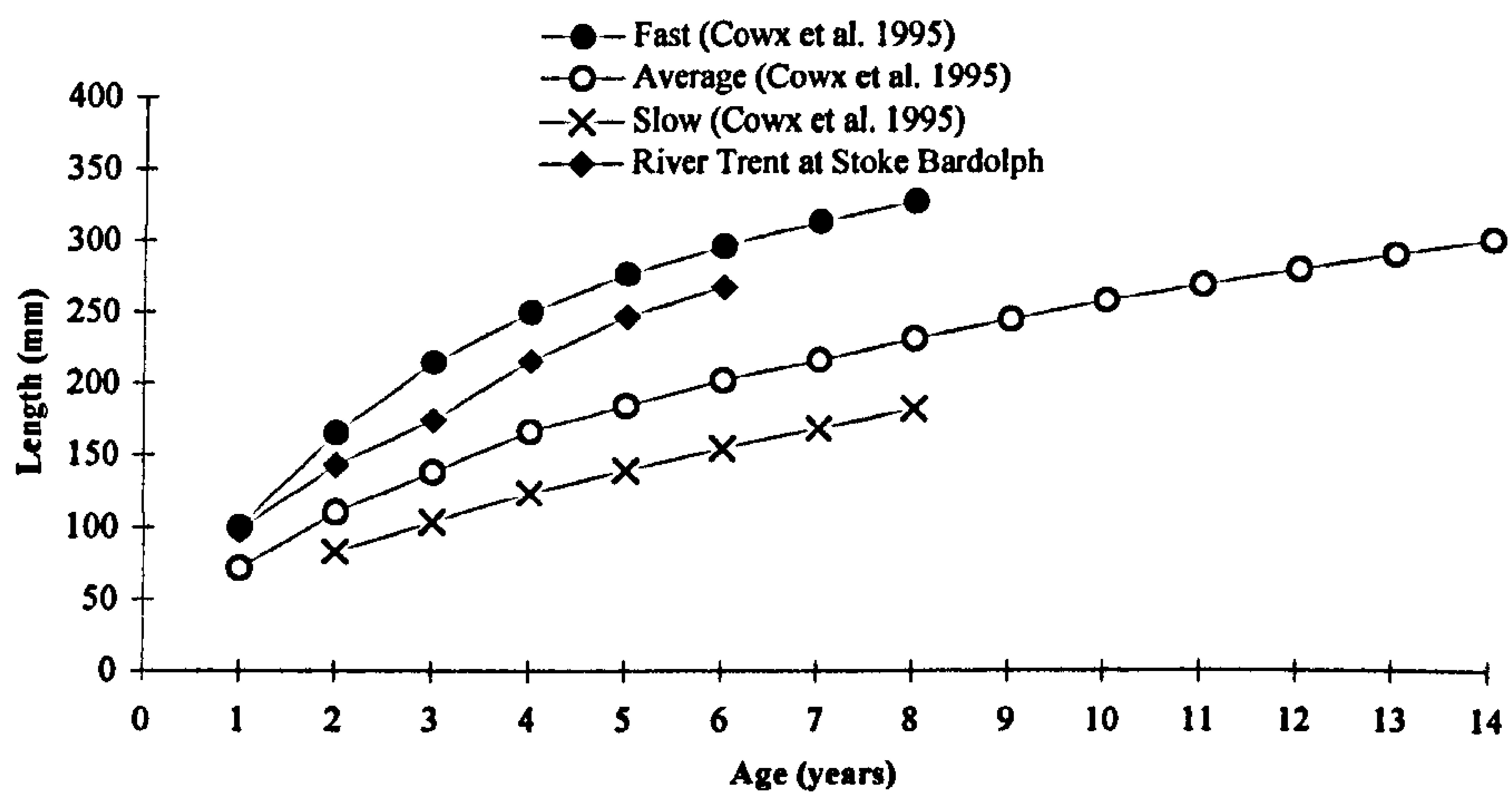


Figure 5.105 Growth rate of perch at Stoke Bardolph, River Trent, compared with standard growth curves.

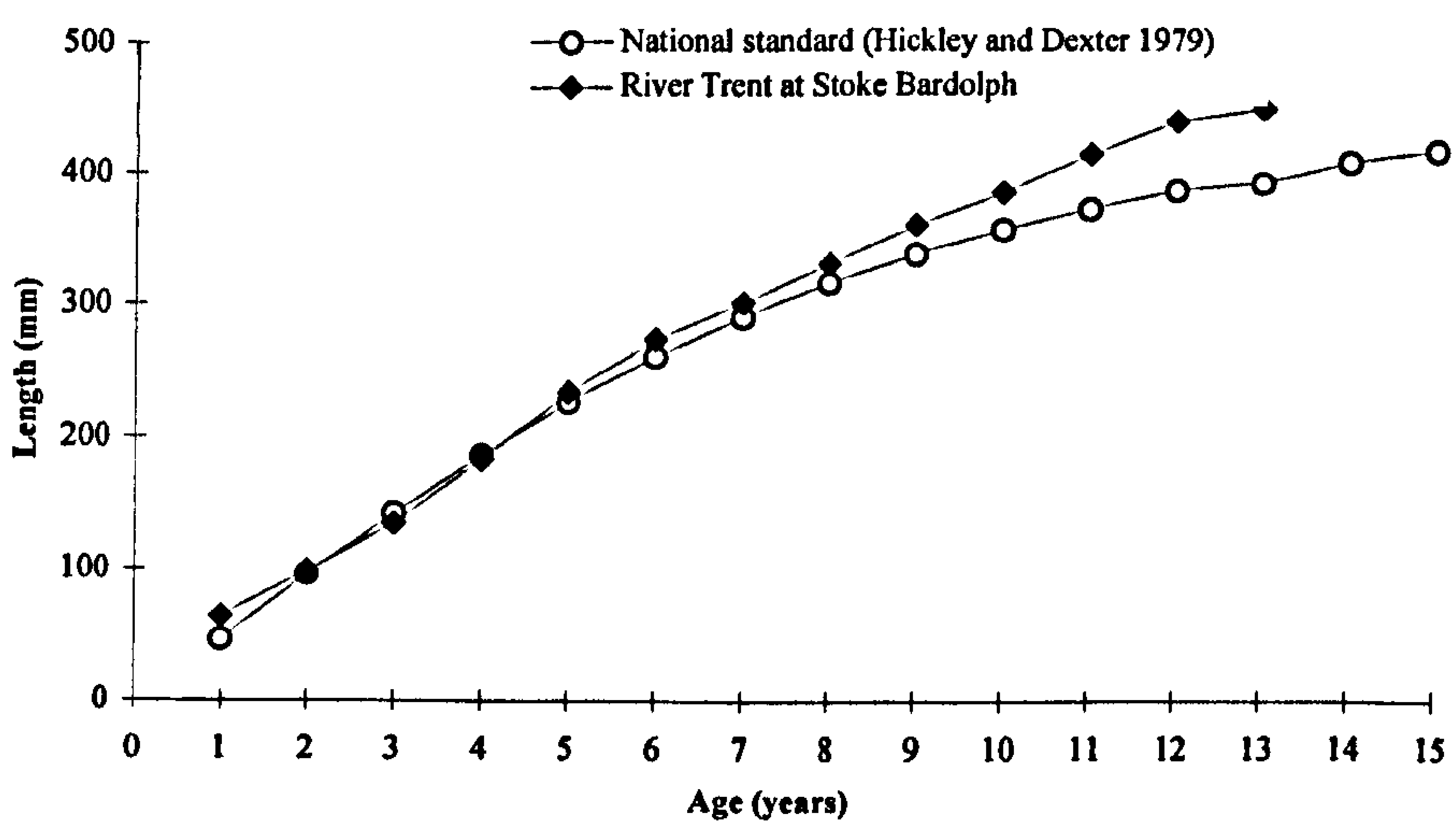


Figure 5.106 Growth rate of chub at Stoke Bardolph, River Trent compared standard growth curves.

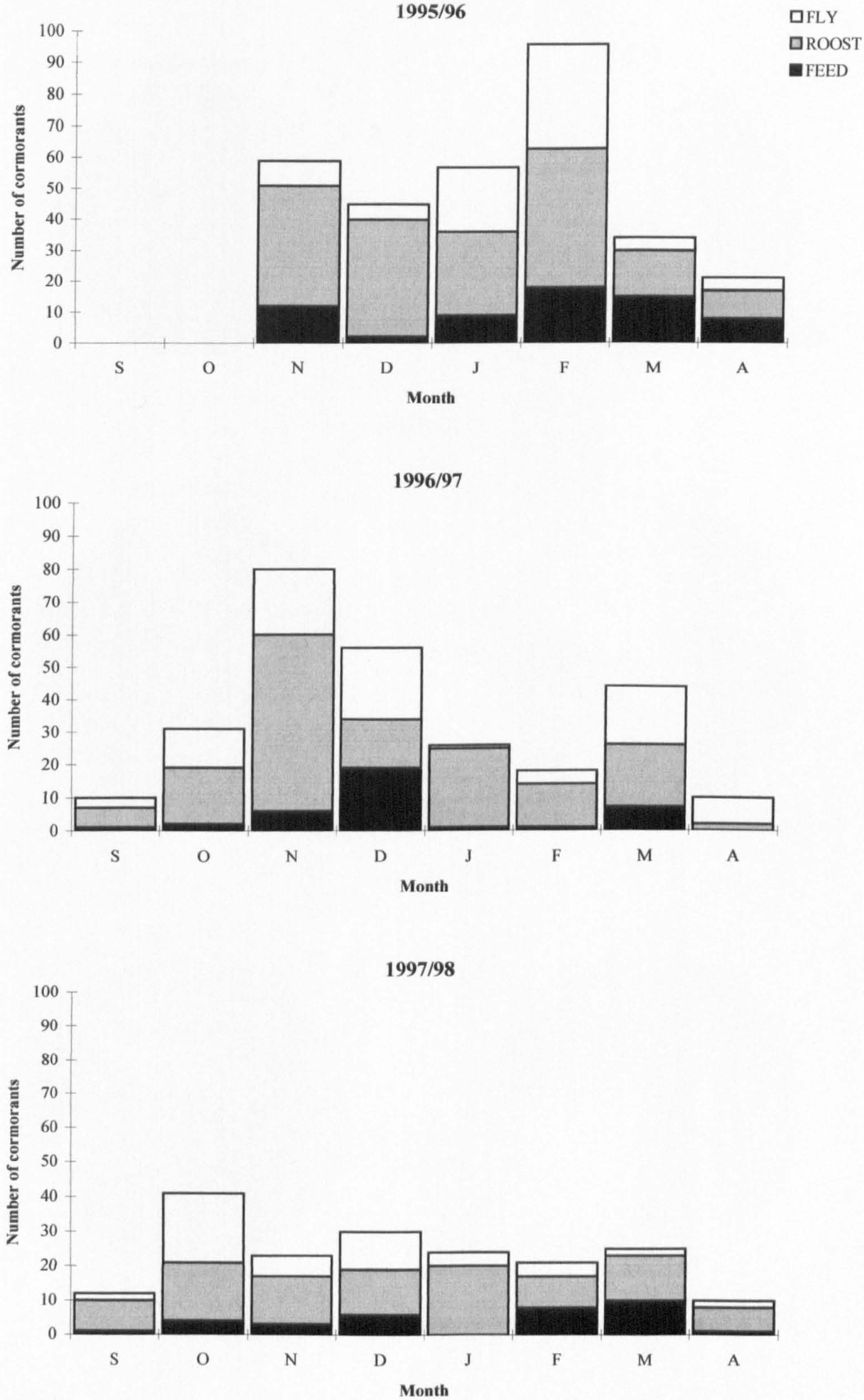


Figure 5.107 The number of cormorants utilising Section D (Stoke Bardolph) of the River Trent over the study period.

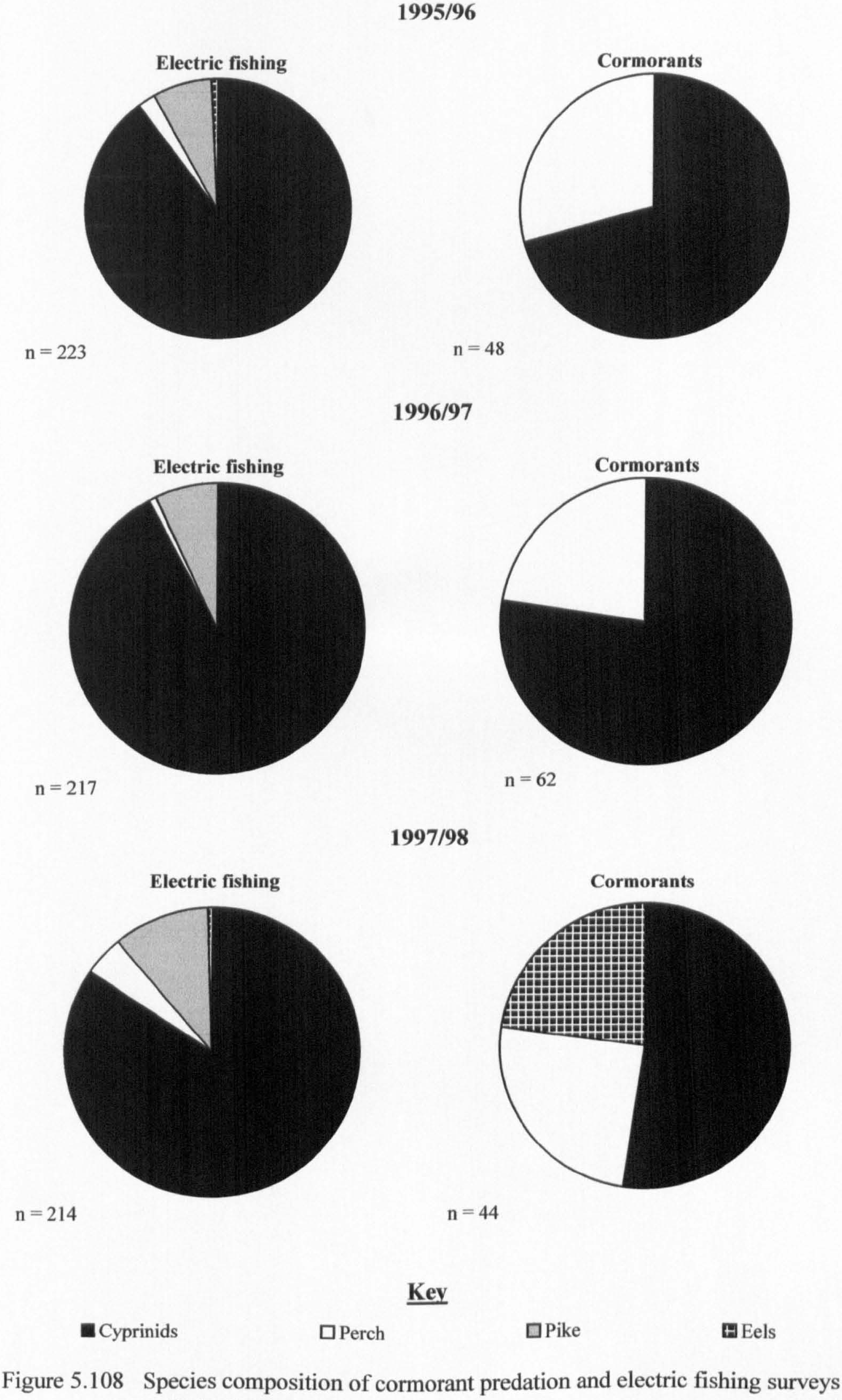
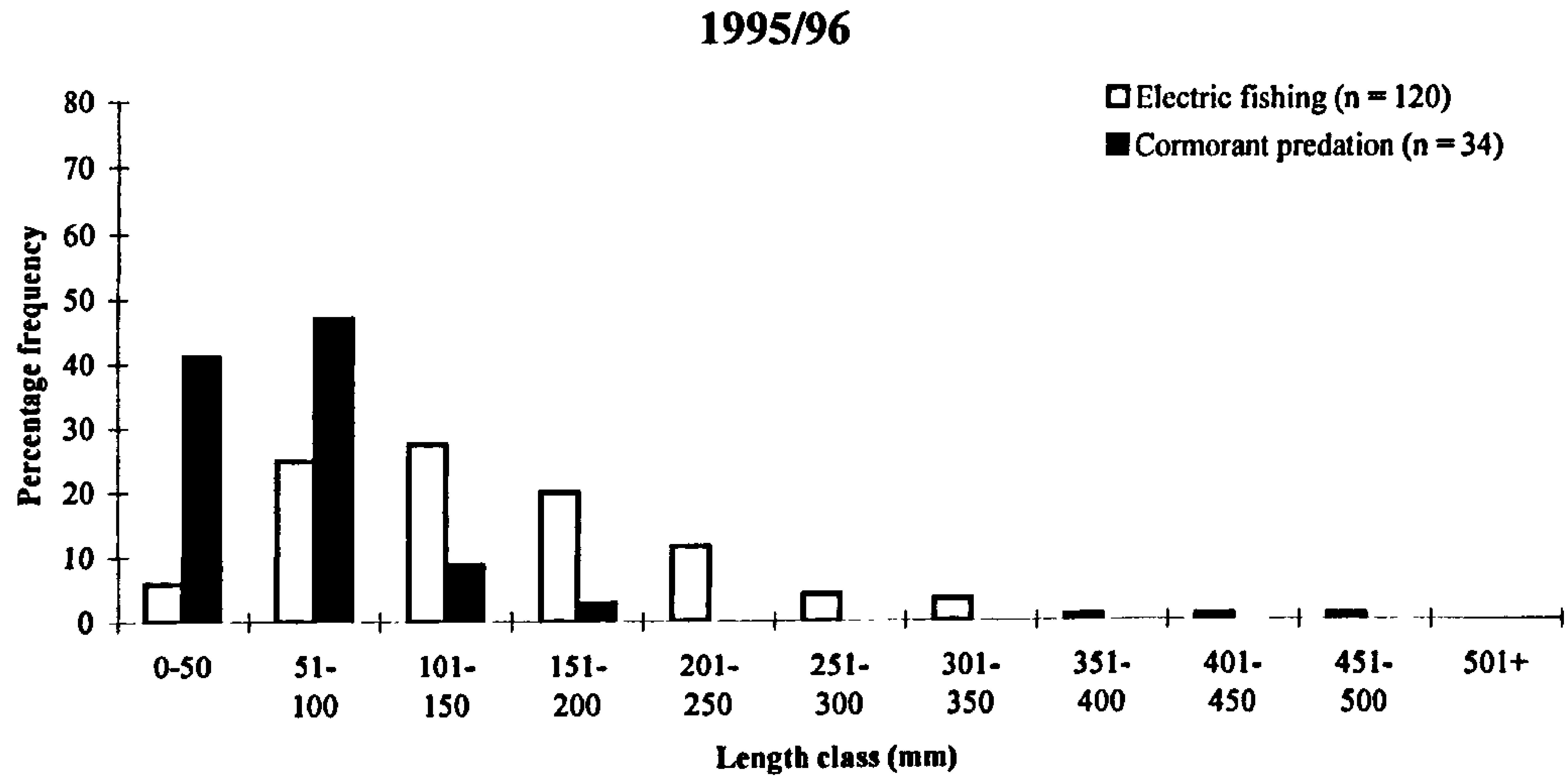
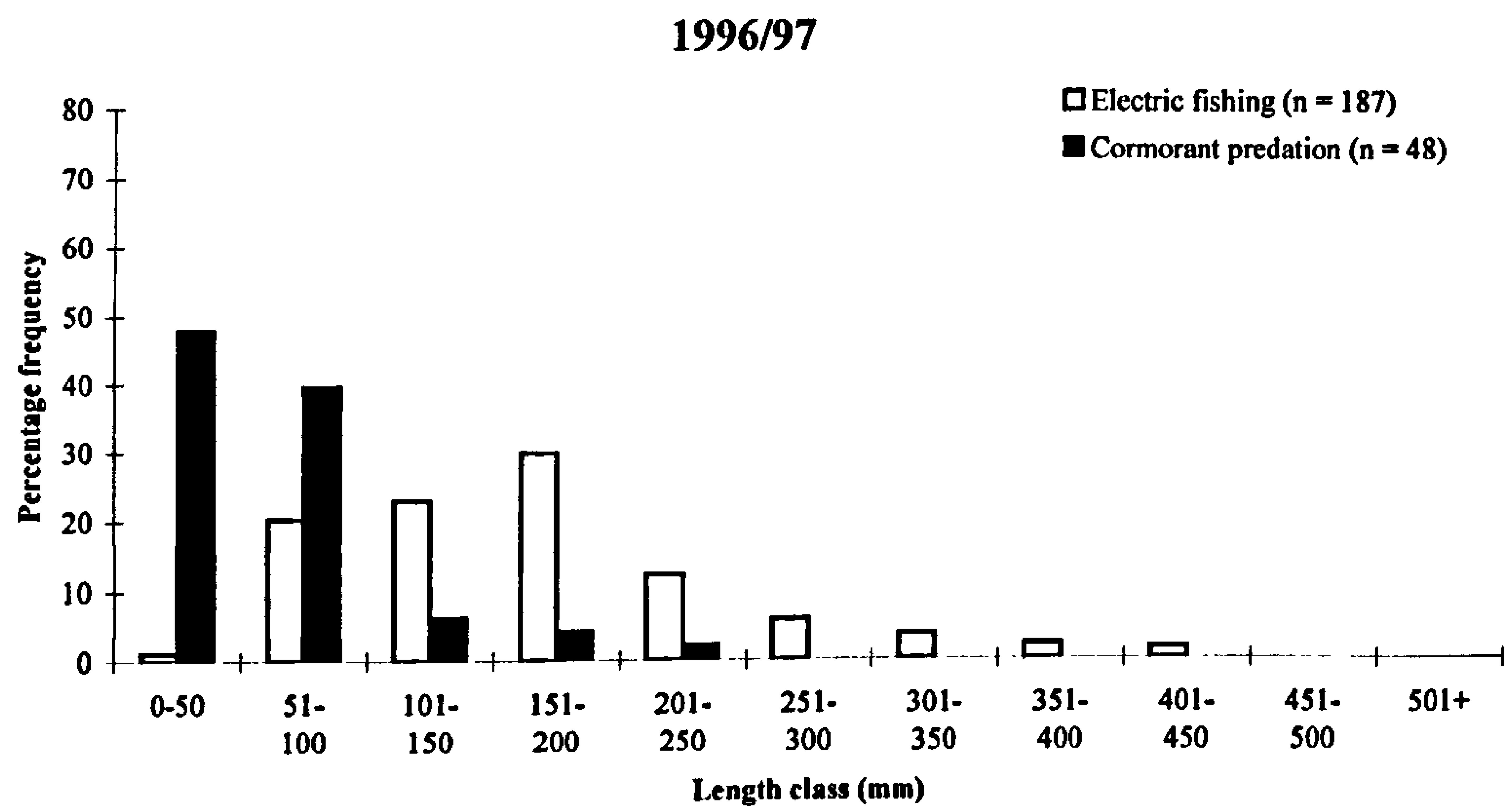


Figure 5.108 Species composition of cormorant predation and electric fishing surveys at Stoke Bardolph, River Trent.





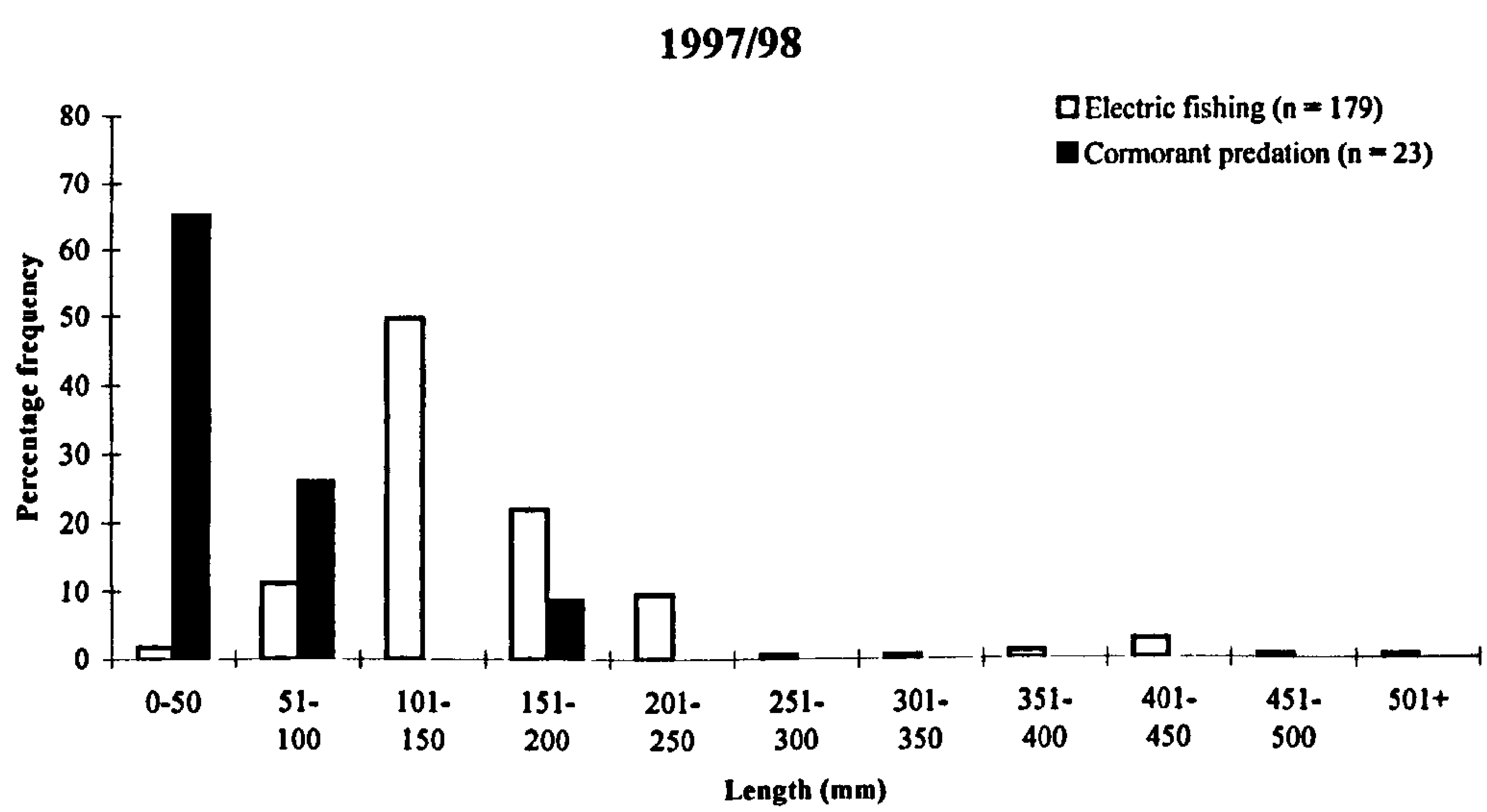
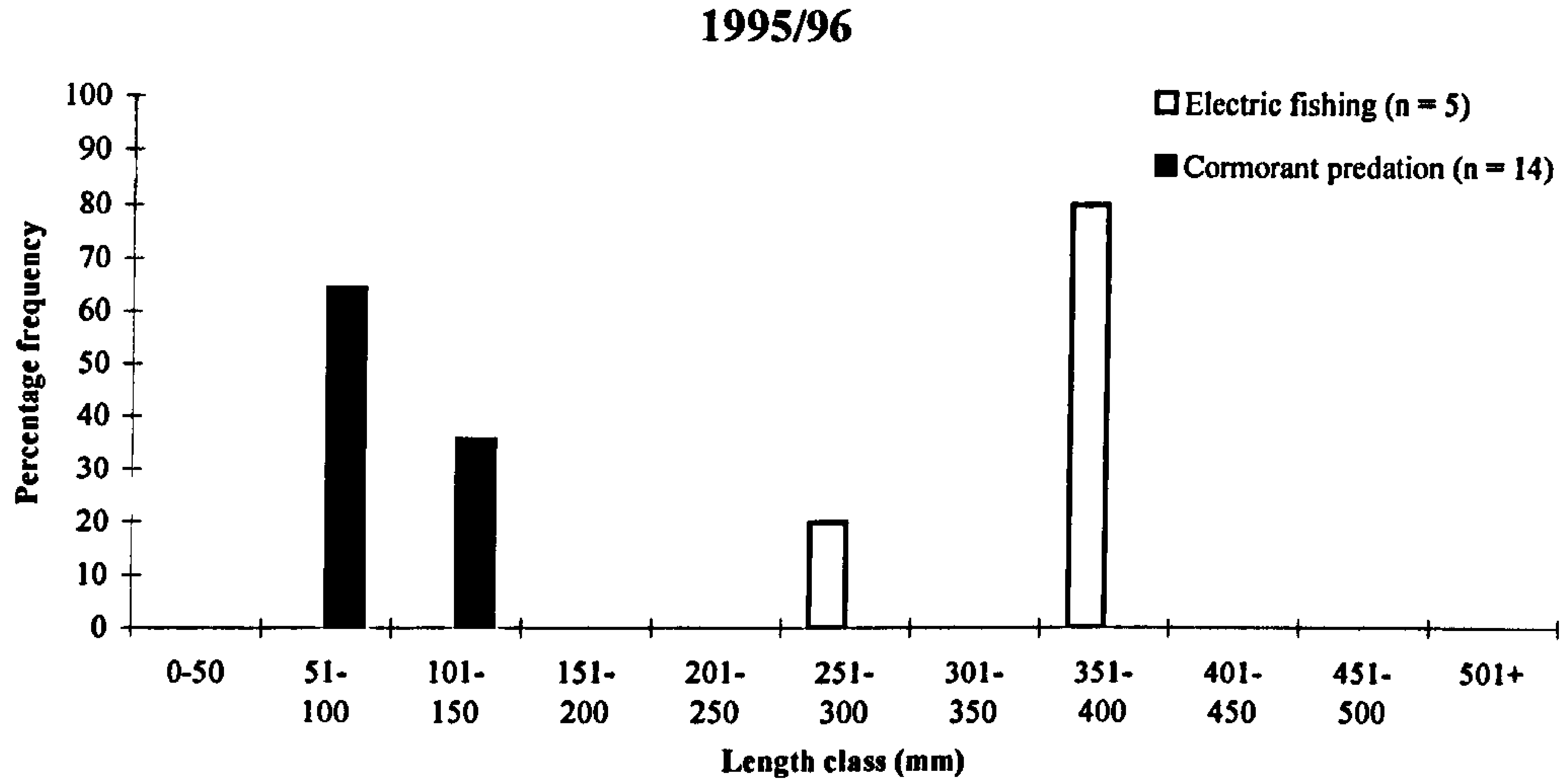
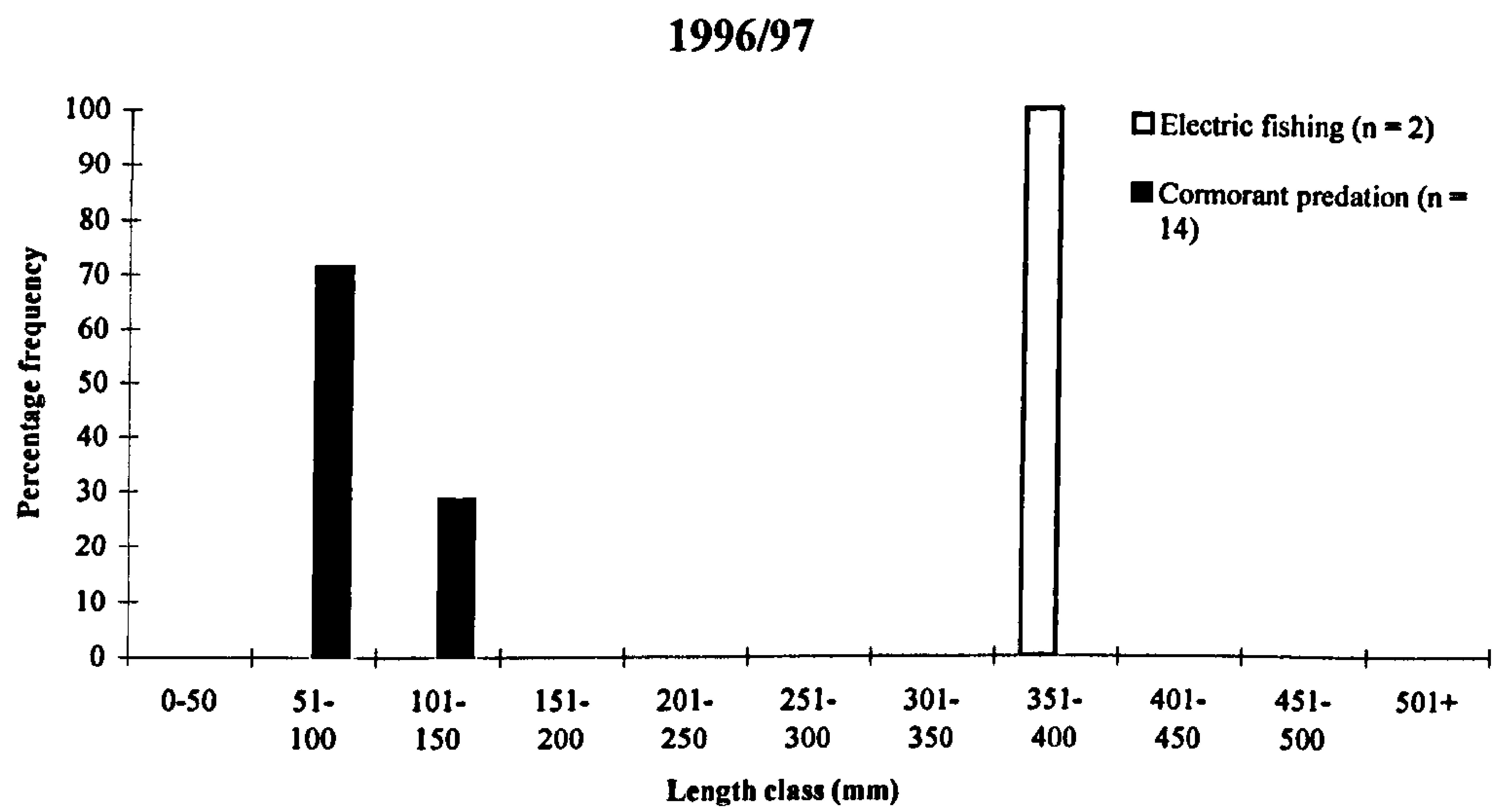


Figure 5.109 Length frequency distribution of cyprinids ingested by cormorants and captured by electric fishing at Stoke Bardolph, River Trent.





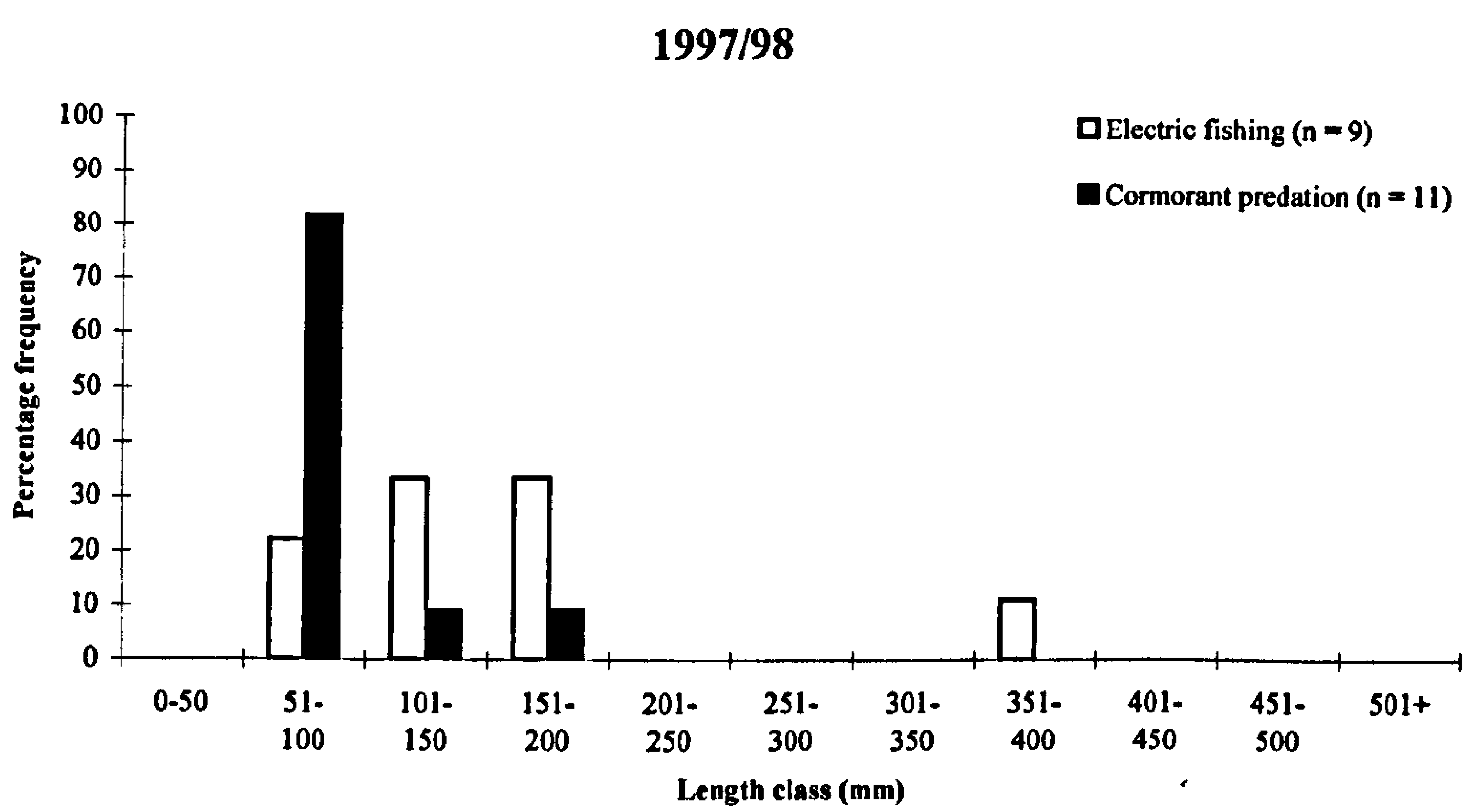


Figure 5.110 Length frequency distribution of perch ingested by cormorants and captured by electric fishing at Stoke Bardolph, River Trent.

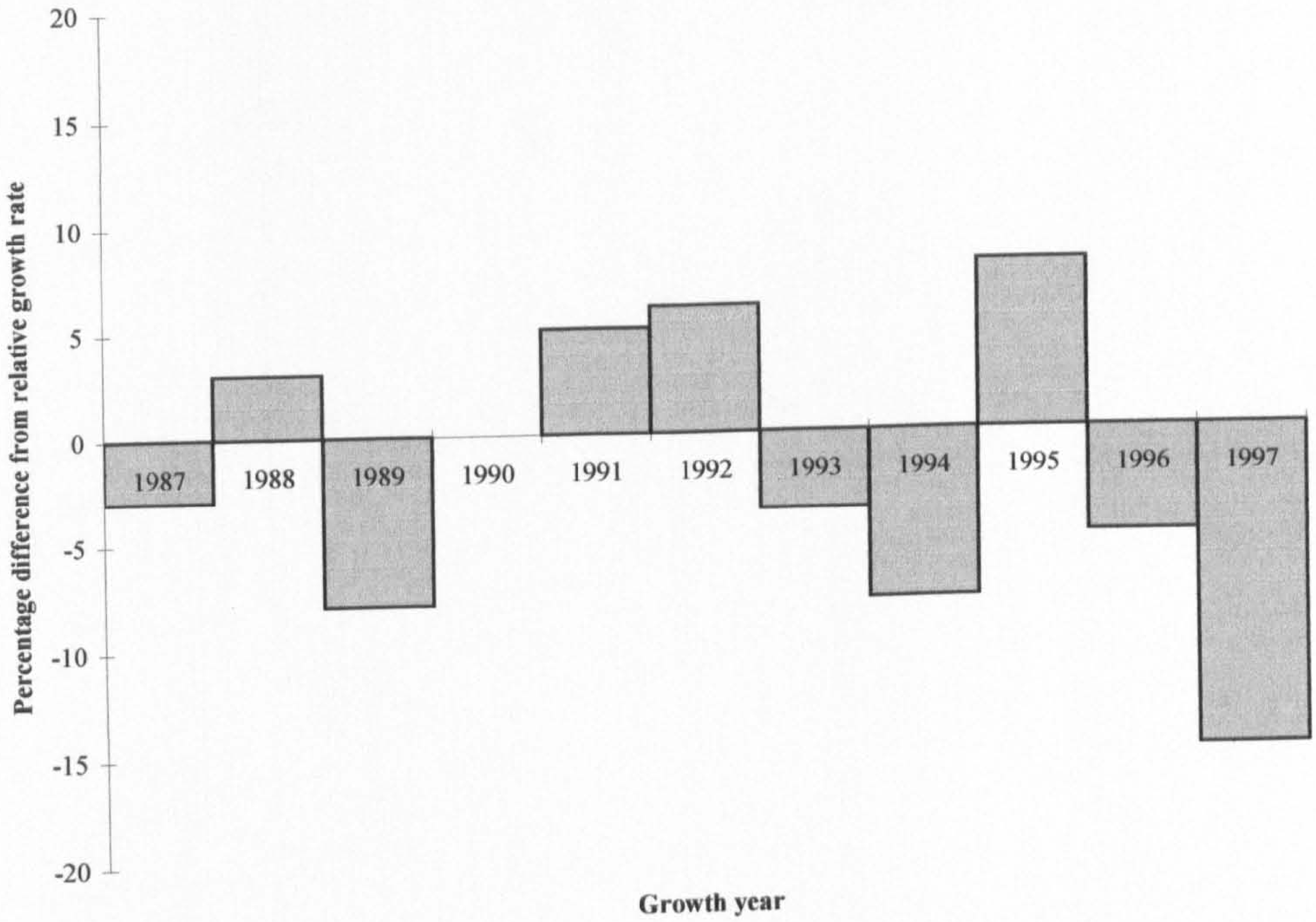


Figure 5.111 Relative annual growth of roach at Stoke Bardolph, River Trent.

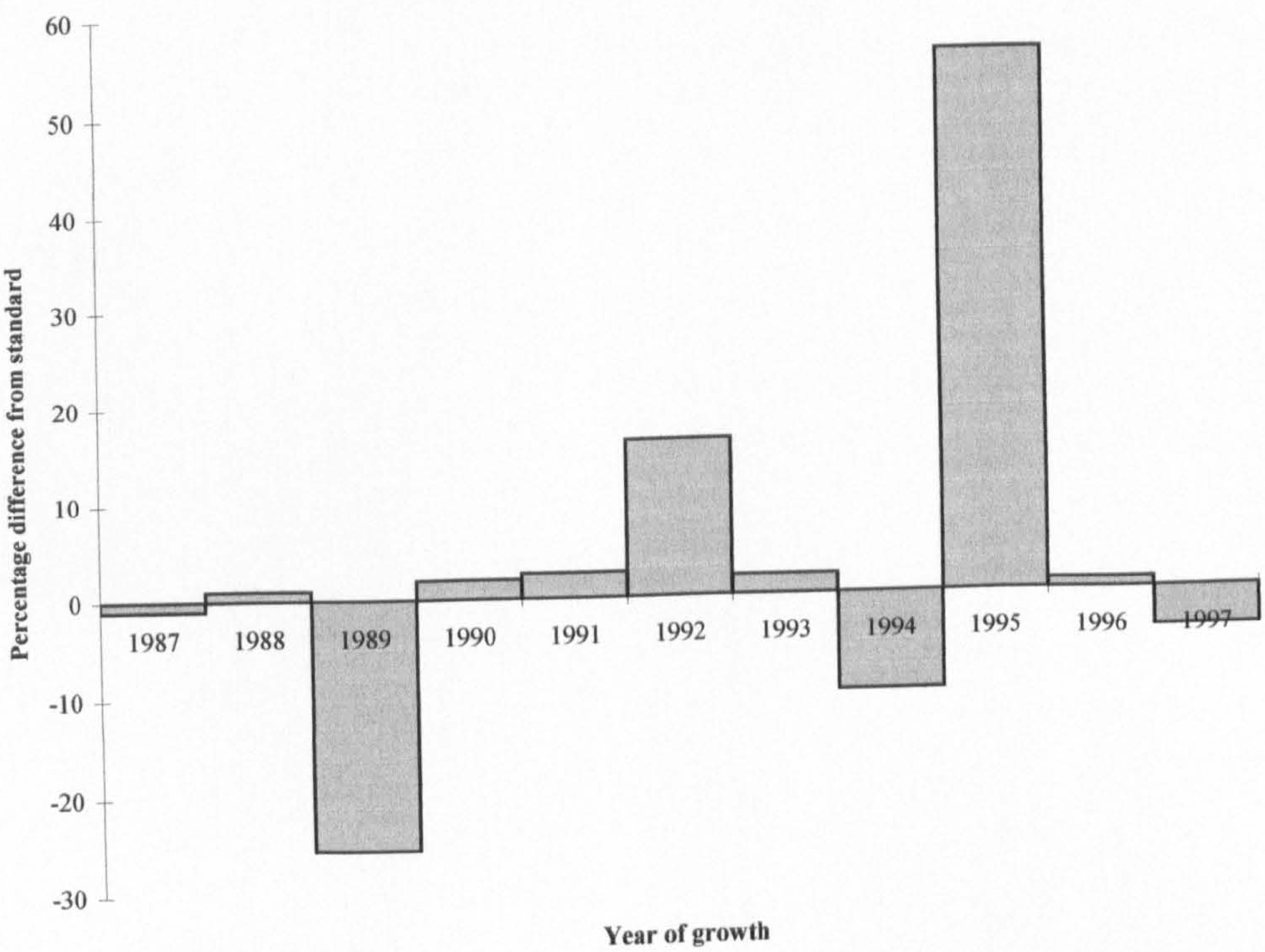


Figure 5.112 Annual growth of roach at Stoke Bardolph, River Trent compared with a standard roach growth rate (Hickley and Dexter 1979).

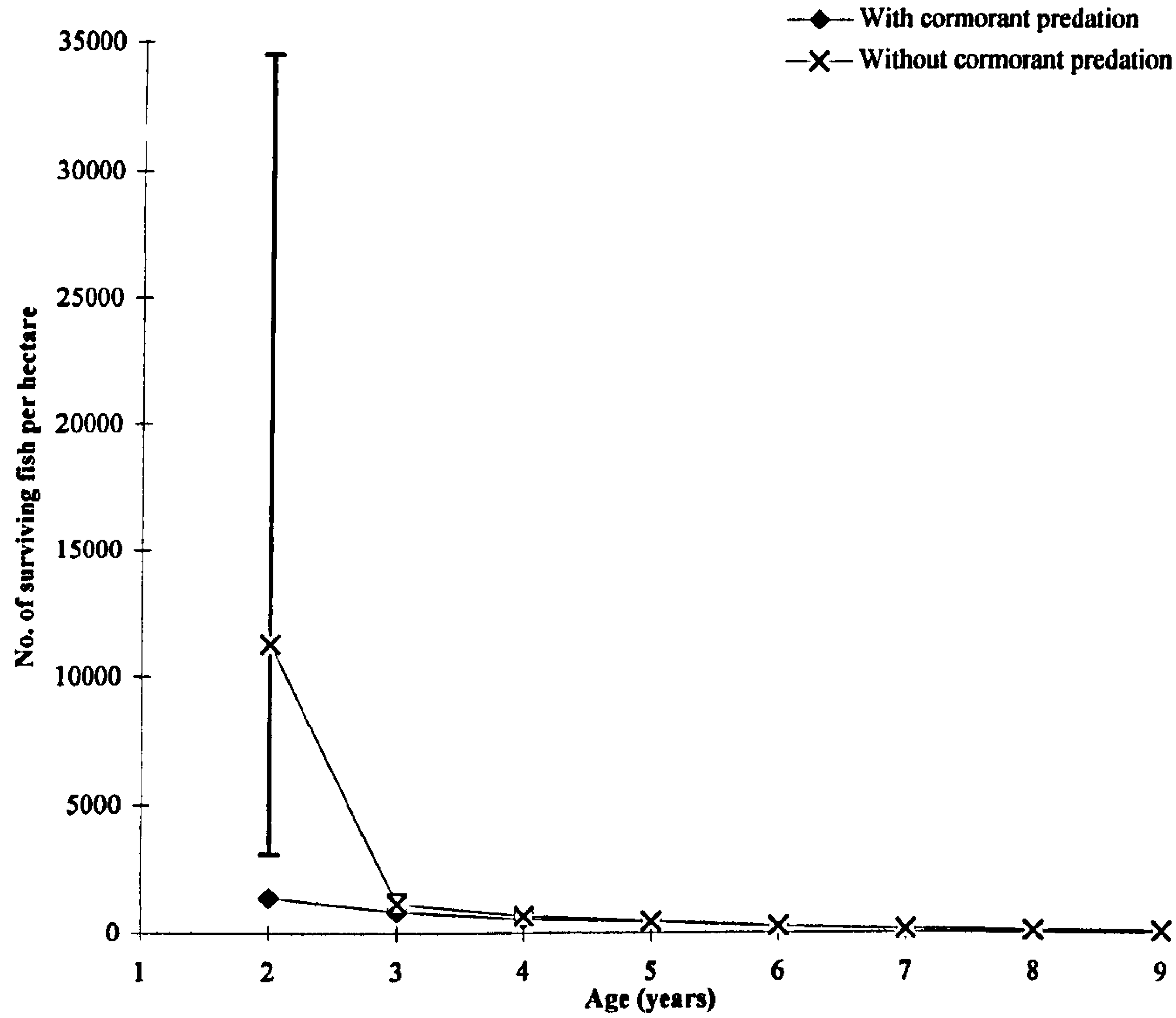


Figure 5.113 Impact of cormorant predation on the numbers of roach at Stoke Bardolph, River Trent, in 1996.

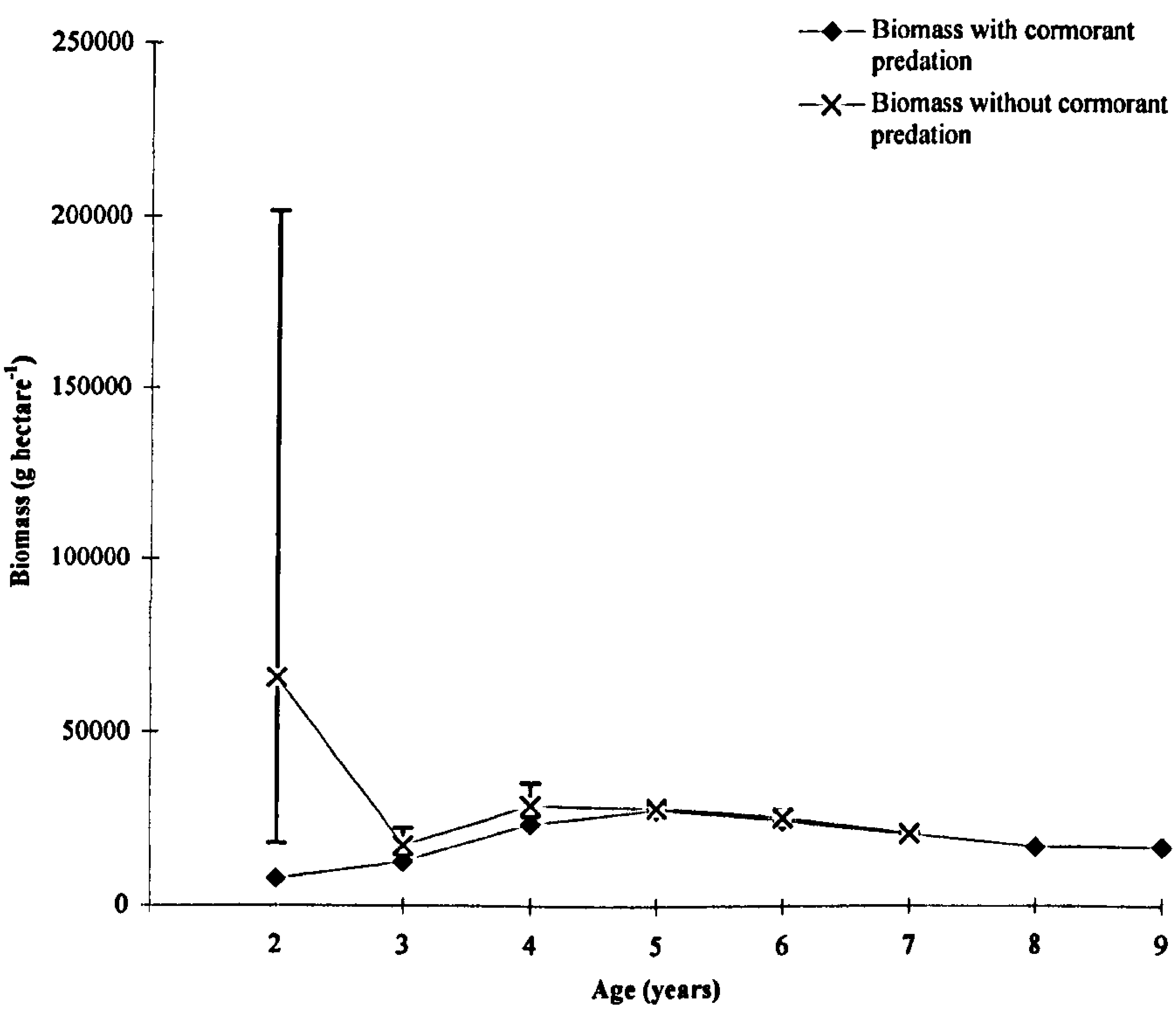


Figure 5.114 Impact of cormorant predation on the biomass of roach at Stoke Bardolph, River Trent, in 1996.

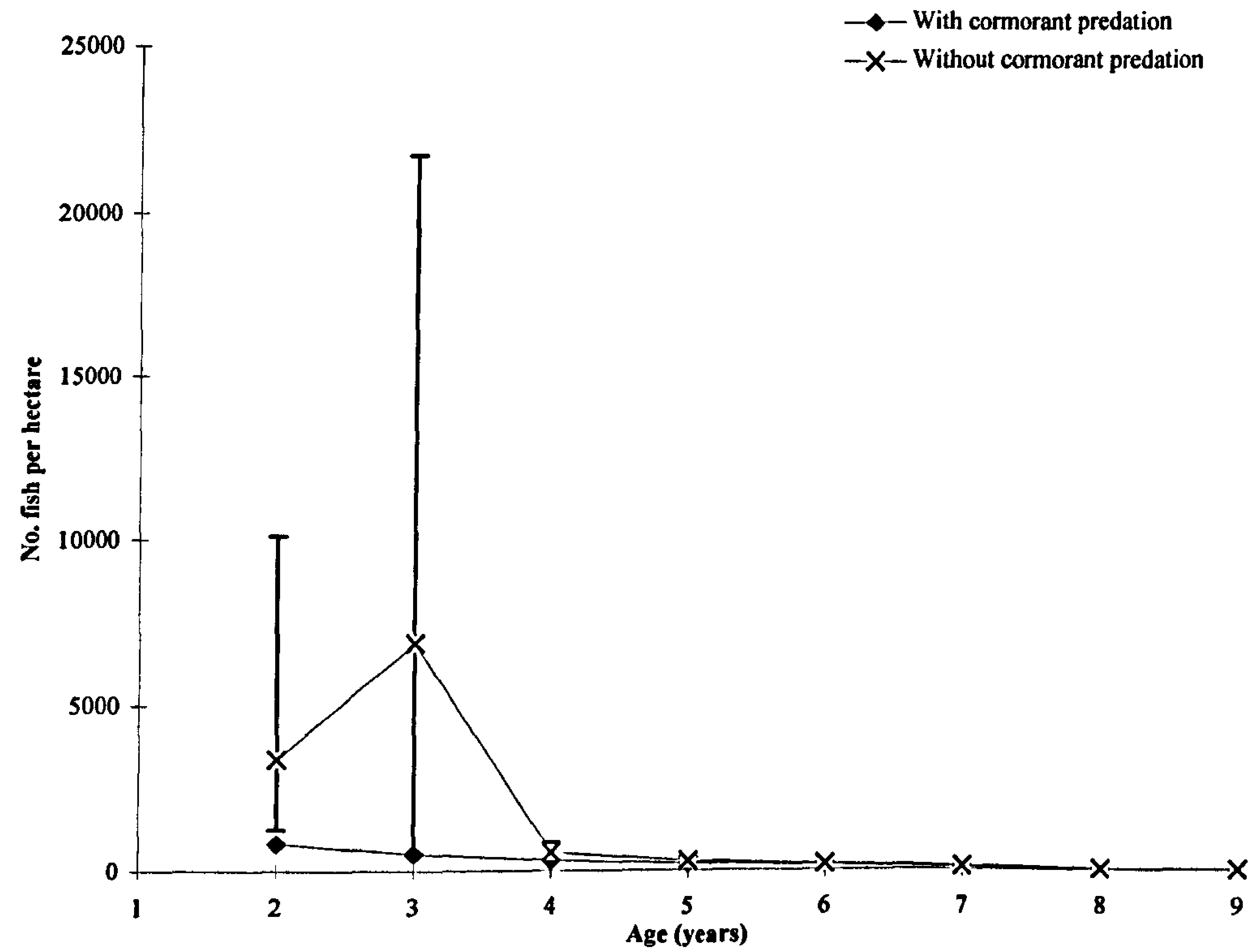


Figure 5.115 Impact of cormorant predation on the numbers of roach at Stoke Bardolph, River Trent, in 1997.

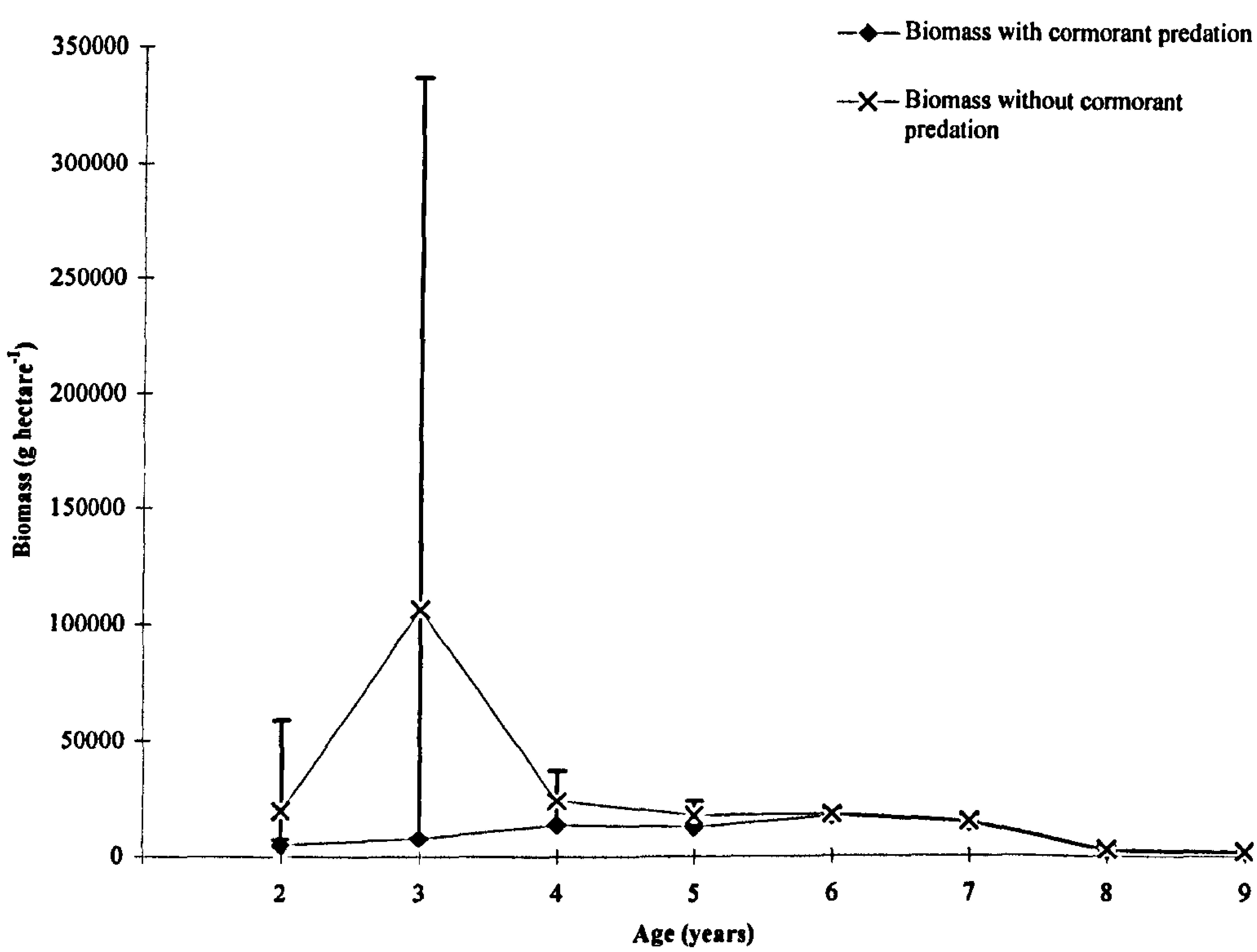


Figure 5.116 Impact of cormorant predation on the biomass of roach at Stoke Bardolph, River Trent, in 1997.

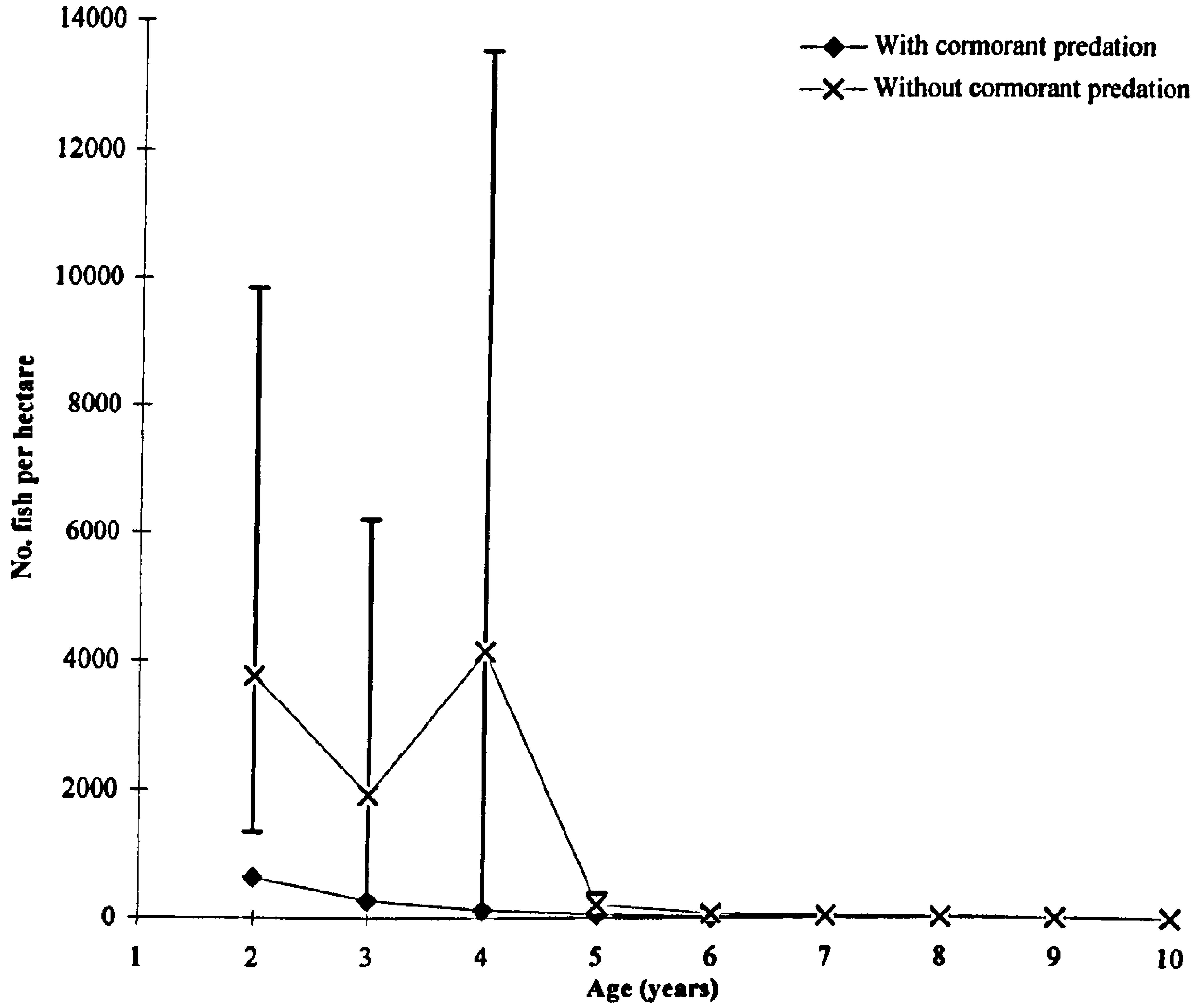


Figure 5.117 Impact of cormorant predation on the numbers of roach at Stoke Bardolph, River Trent, in 1998.

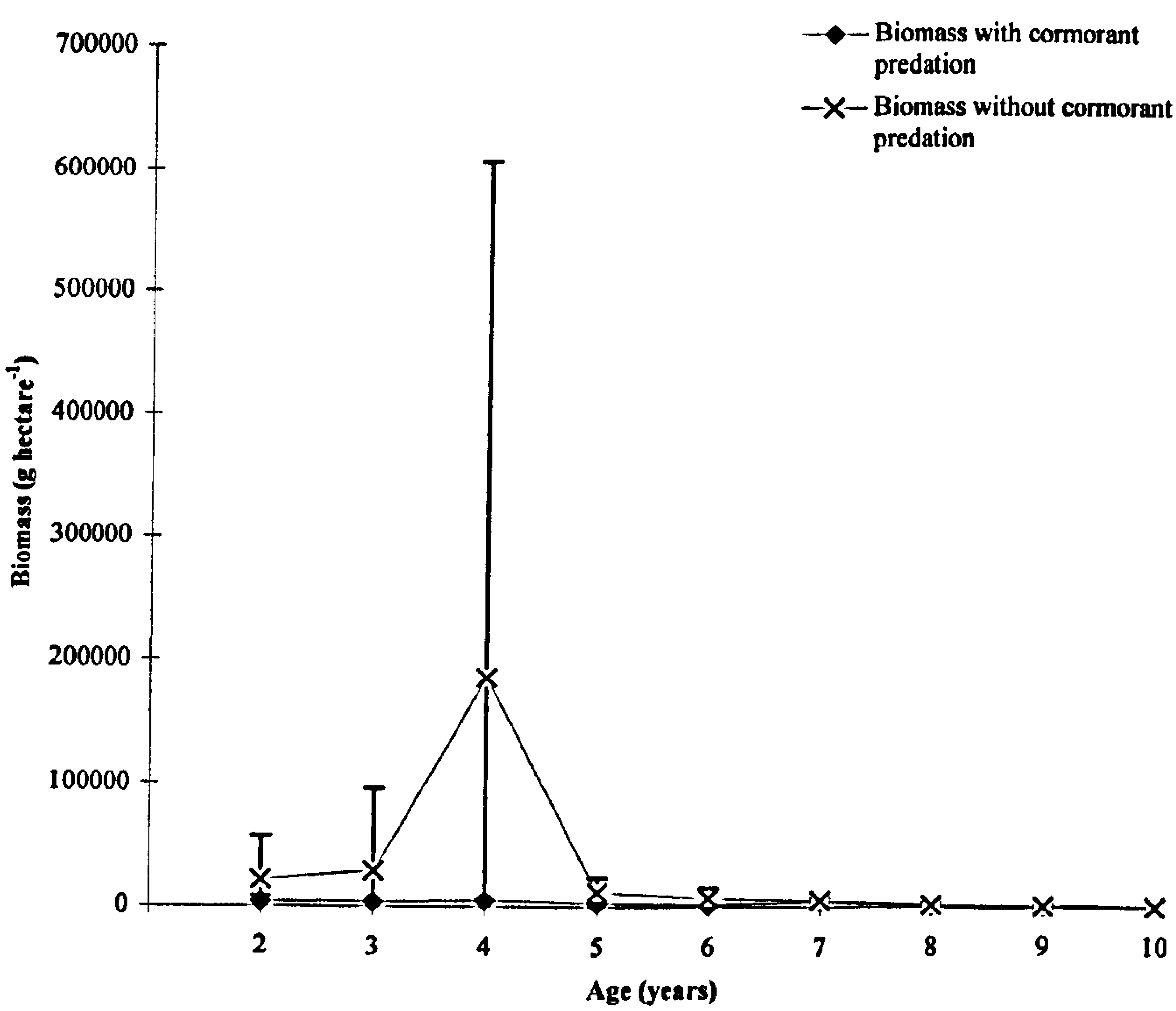


Figure 5.118 Impact of cormorant predation on the biomass of roach at Stoke Bardolph, River Trent, in 1998.

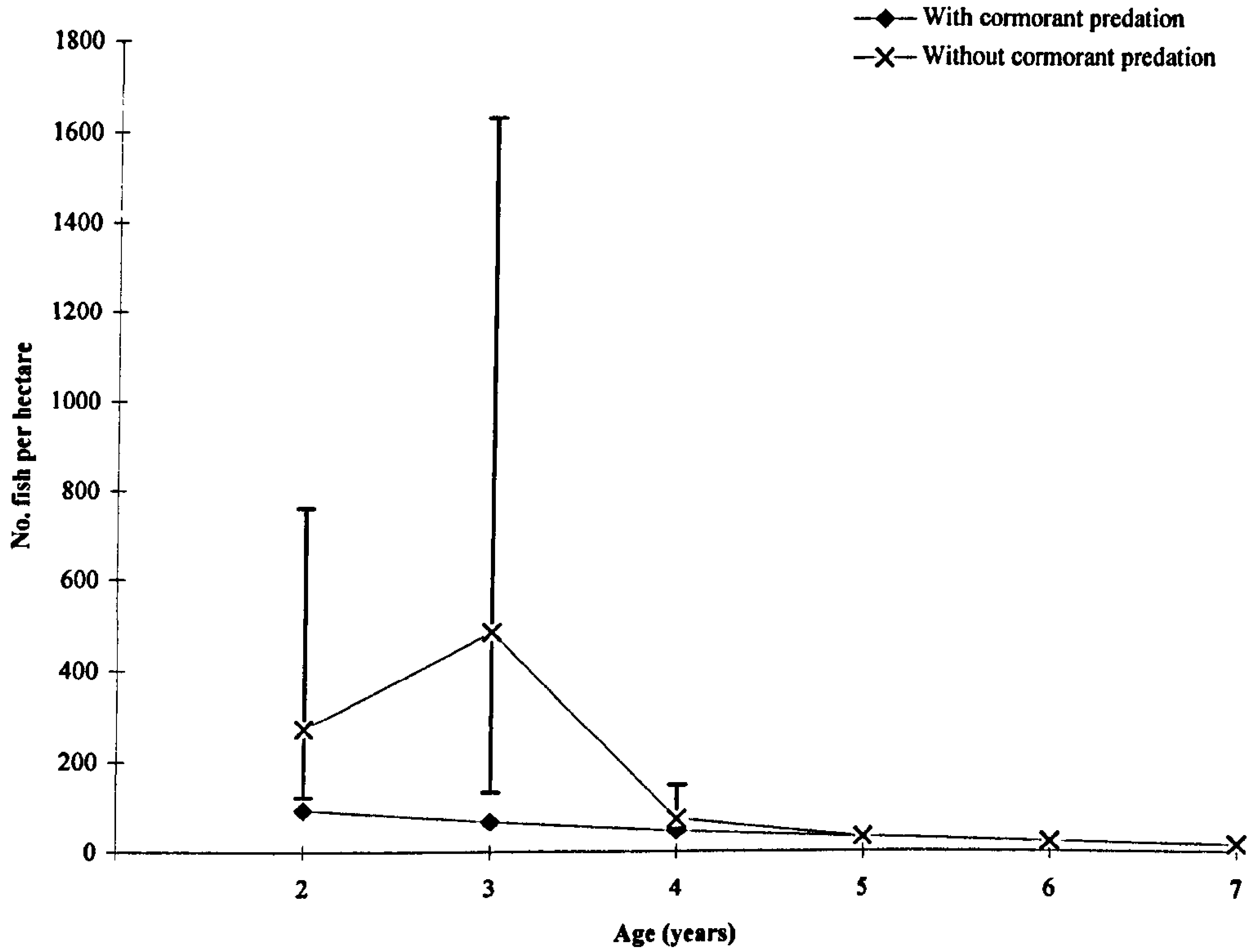


Figure 5.119 Impact of cormorant predation on the numbers of perch at Stoke Bardolph, River Trent, in 1996.

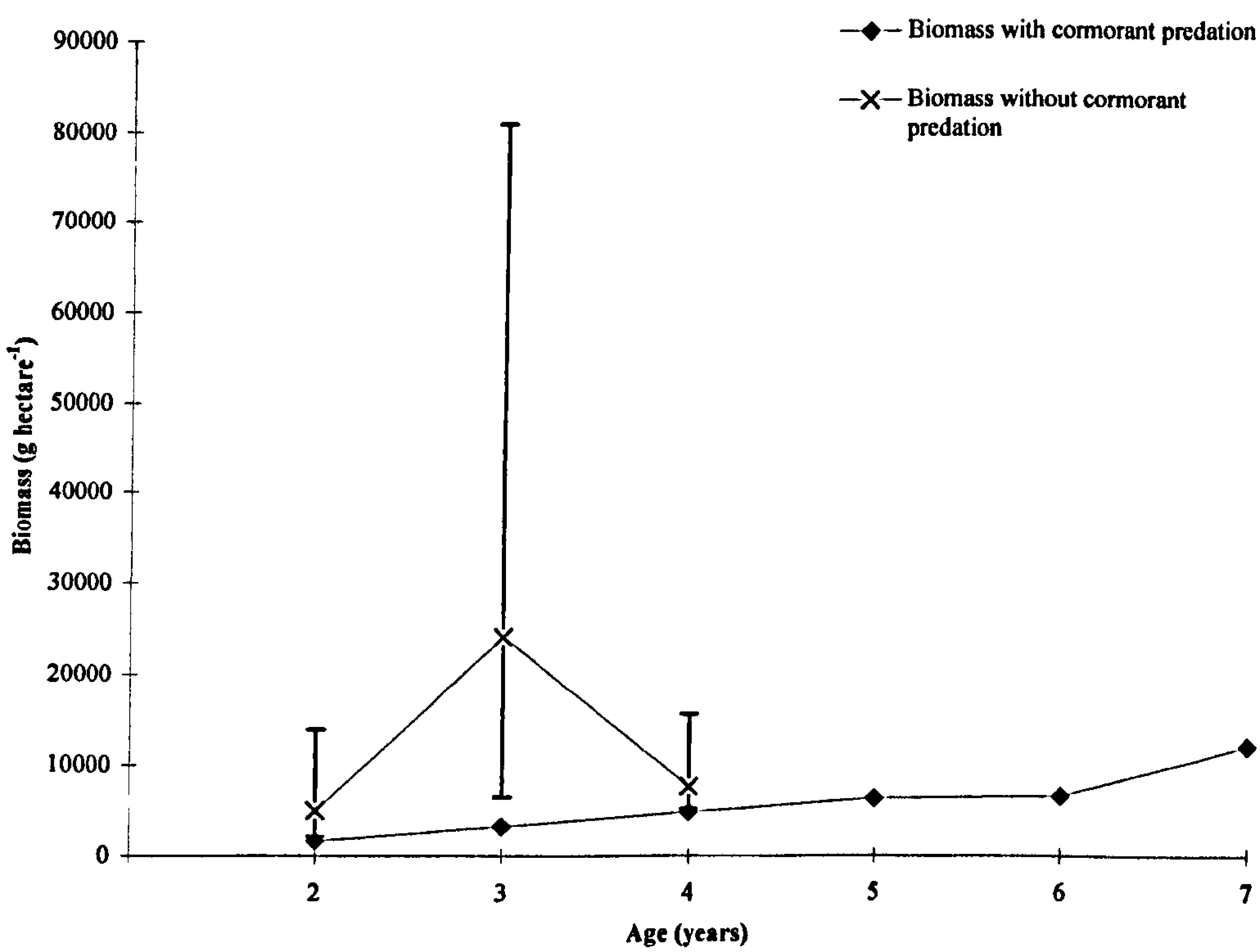


Figure 5.120 Impact of cormorant predation on the biomass of perch at Stoke Bardolph, River Trent, in 1996.

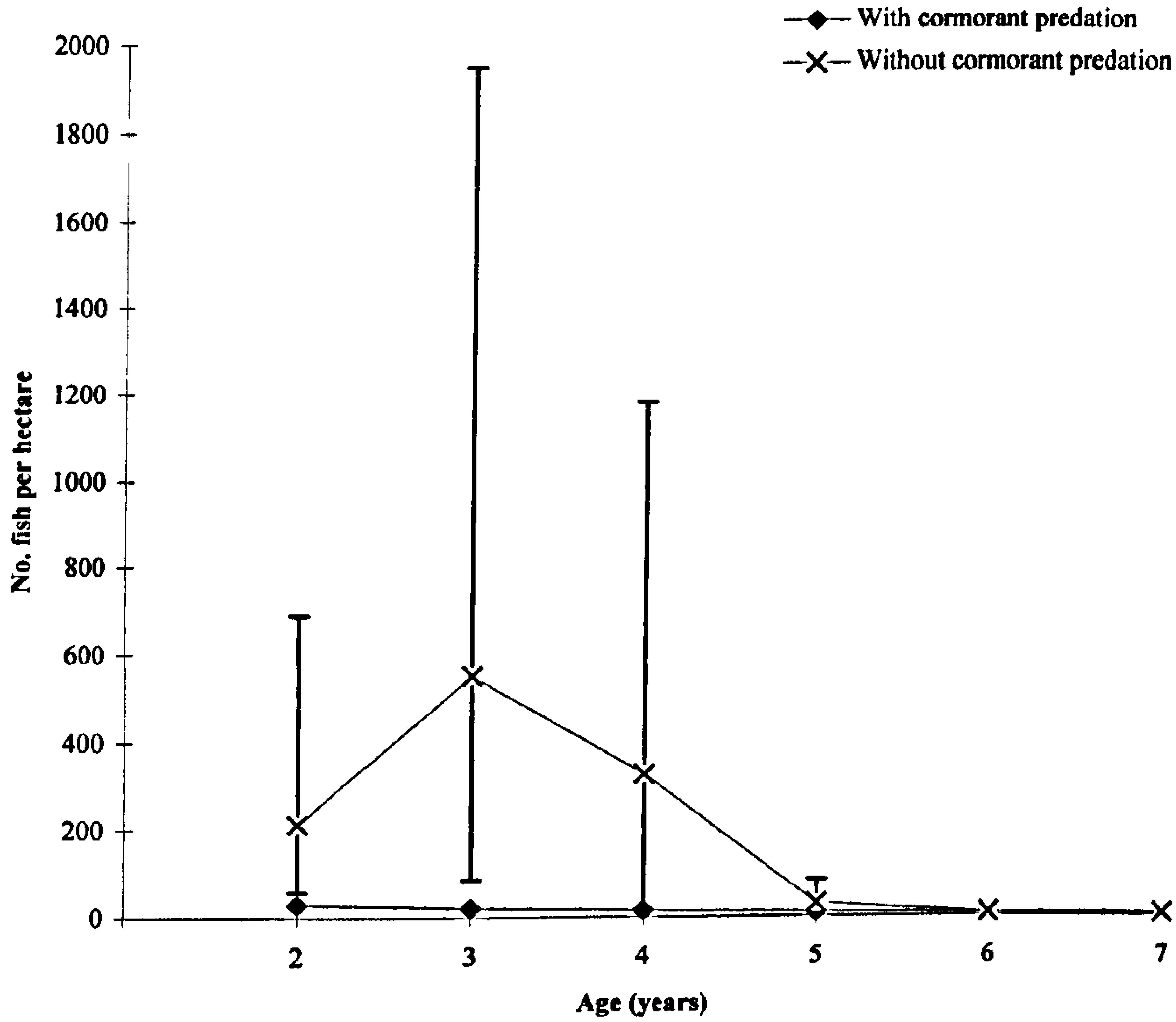


Figure 5.121 Impact of cormorant predation on the numbers of perch at Stoke Bardolph, River Trent, in 1997.

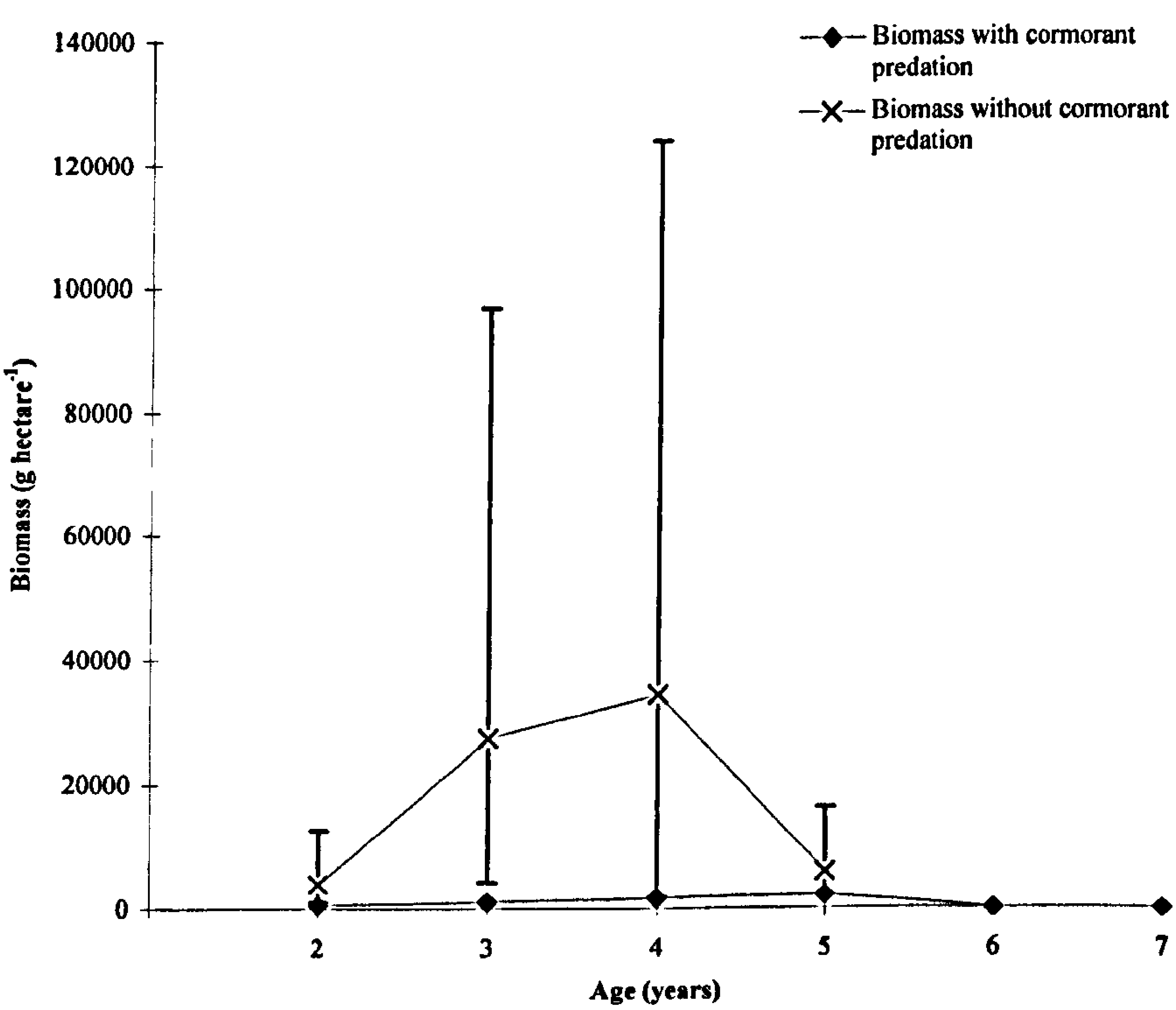


Figure 5.122 Impact of cormorant predation on the biomass of perch at Stoke Bardolph, River Trent, in 1997.

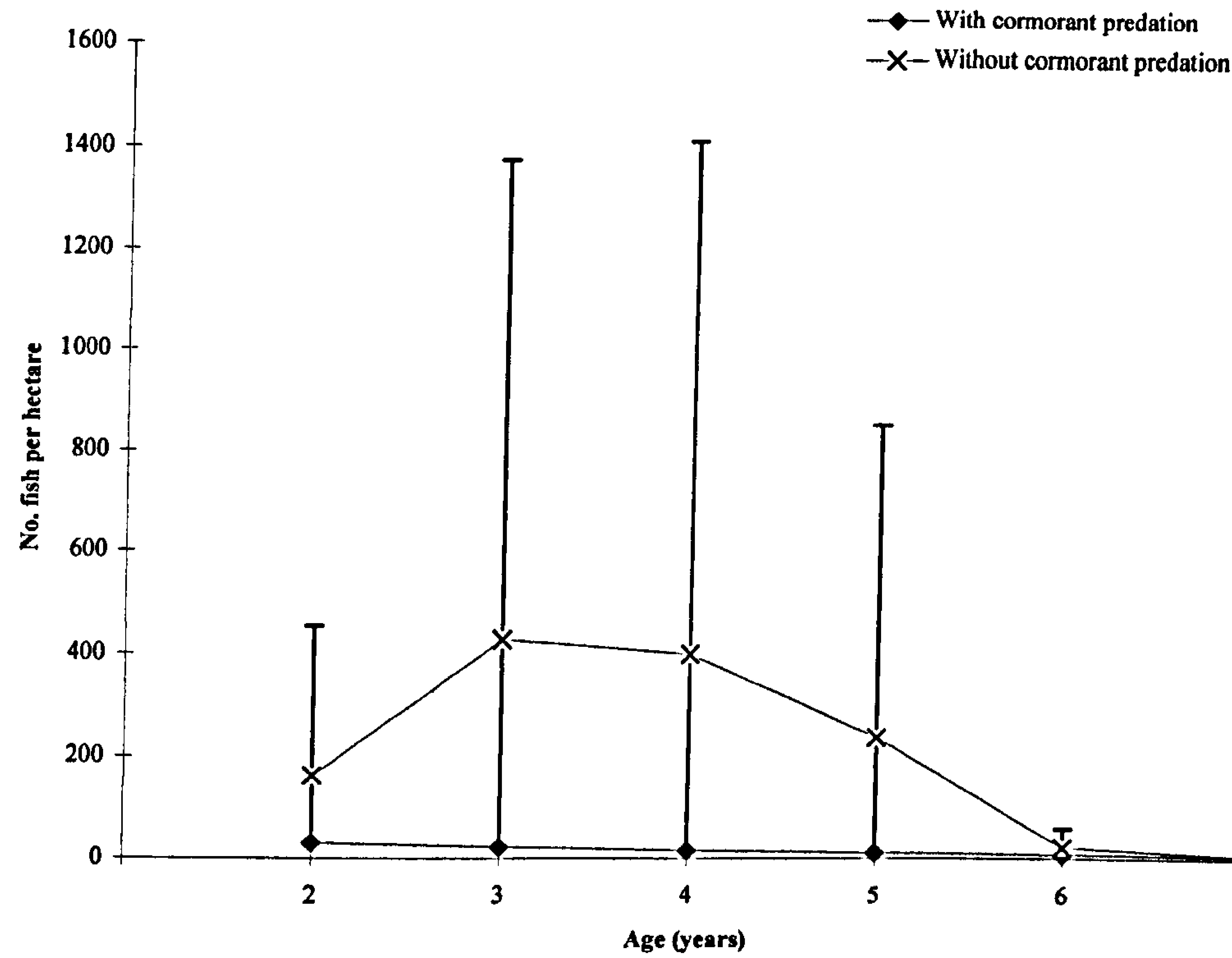


Figure 5.123 Impact of cormorant predation on the numbers of perch at Stoke Bardolph, River Trent, in 1998.

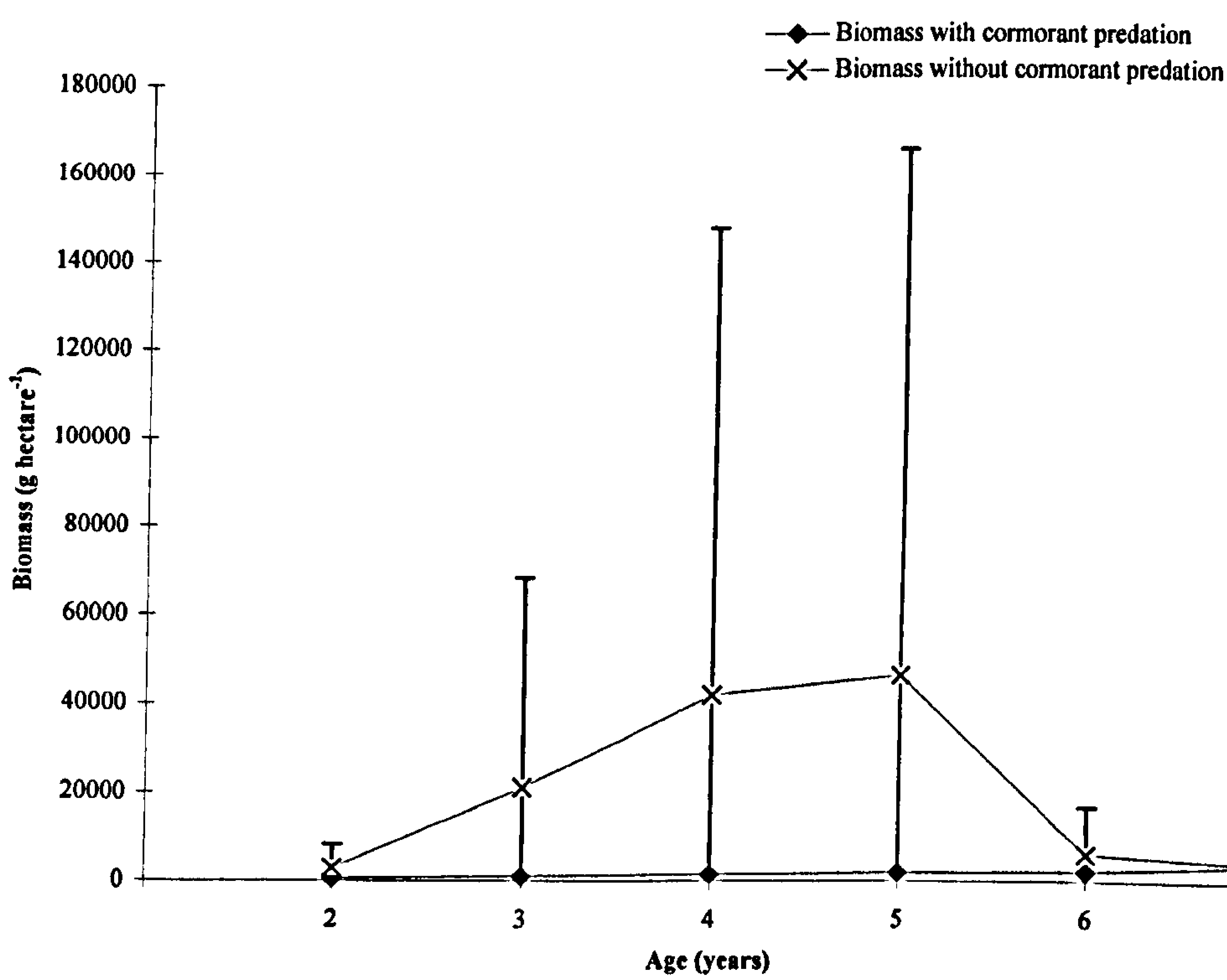


Figure 5.124 Impact of cormorant predation on the biomass of perch at Stoke Bardolph, River Trent, in 1998.

5.8 Discussion

5.8.1 *Fish populations*

The population structure of the fishery as a whole was dominated by cyprinids, typical of a large lowland river. The standing crop and density of fish, assessed by electric fishing and hydro-acoustics, varied each year, indicating the dynamic nature of the fisheries of the River Trent. The growth rates of fish indicated no major changes under the predation pressure of cormorants. The natural mortality rates were generally average when compared to other UK fisheries, even under apparently heavy cormorant predation.

5.8.2 *Cormorant ecology*

Site use

The majority of the cormorants observed on the River Trent were utilising the river as a 'fly-way'. By observing the river contours during flight, the cormorants were able to locate adjacent stillwaters, such as Holme Pierrepont (Chapter 4), for feeding. This was especially apparent at Sections B and C, where the MCS model had to be adapted to account for this behaviour (Feltham *et al.* 1999). Only a small proportion of cormorants utilised the river as a feeding site.

Feeding success

The topography of the River Trent is highly variable, with the presence of deep areas and underwater boulders and snags. The river is also subject to high flows in the winter period. These factors may have made cormorant foraging difficult, with more successful foraging possible on the surrounding stillwaters, such as Holme Pierrepont. This was shown by the increased feeding success at Holme Pierrepont compared with the River Trent sections, although the feeding success at Colwick Trout Park Lake was lower in the period prior to trout stocking, possibly because of poor food resources making foraging inefficient.

The behaviour of fish in the River Trent during the winter period needs consideration when assessing the feeding success of the cormorants. During the fisheries surveys, it was observed that in the summer months the fish were dispersed along the river sections, resulting in good catches of fish. However, during the winter period, dense shoaling of fish in certain areas was observed throughout the River Trent, for example, Trent Bridge during winter. Here, in the built up area of Nottingham, large catches of fish were possible, resulting from winter fish aggregation in the area.

This winter aggregation of fish to specific over-wintering areas, such as Trent Bridge, is known to impact on angler catches along the whole of the River Trent, as fish become concentrated in areas that may only be accessible to a small number of anglers, for example, under bridges, in marinas and in lock cuttings (T. Holden pers. comm.). Predation by cormorants appeared to concentrate in the areas where fish shoal in the winter (T. Holden pers. comm.). This explains some of the foraging success observed, despite otherwise difficult foraging conditions due to the river topography.

5.8.3 *Cormorant impact assessment*

The main conclusion from the cormorant predation impact assessment was that, although predation by cormorants appeared low on the River Trent due to low numbers of feeding cormorants observed, this actually translated into heavy exploitation when applied through the MCS model. This resulted in substantial estimations of fish loss. It is possible that a reasonably high level of fish loss attributable to cormorants was likely to have occurred on the sections. However, due to the low number of fish observed to be consumed by cormorants and the low number of feeding cormorants observed on the river, it is possible the MCS model produced very high loss estimates which were inaccurate (Section 9.4.2).

Evidence for the over-estimation is provided by the fish population data for each section (Sections 5.5.1, 5.6.1, 5.7.1). Little annual fluctuation was observed in angler catch rate, fish growth rates and population size, other than those expected, and could be explained by environmental parameters such as temperature. If fish losses due to cormorants were occurring in the study sections (Sections 5.5.3, 5.6.3, 5.7.3), then angler catches in the sections during the following angling season are likely to have shown a large decline and fish growth rates are likely to have altered as the populations compensated for the losses. These factors were not observed on any of the River Trent study sections. Additionally, the hydro-acoustic and fisheries surveys in the October prior to each winter of cormorant predation revealed that the numbers of fish available for predation were far lower than the losses estimated by the MCS model (Section 5.5.1, 5.5.3, 5.6.1, 5.6.3, 5.7.1, 5.7.3), suggesting the model needs further validation and adjustment (Section 9.4.2).

The main fish species present in the fish population were all vulnerable to cormorant predation, with roach and hybrids vulnerable throughout their life span, and common bream, chub and perch vulnerable for the first four years. This was in direct contrast with Holme Pierrepont where fish were vulnerable only in their first and second year of life (Section 4).

A number of other factors must be taken into consideration when assessing the scale of cormorant predation on a large lowland river such as the Trent.

5.8.4 *Water quality*

Water quality in the River Trent has improved over the last 20 years as a result of the improvement in the sewage treatment works along the river. This has resulted in a decrease in ammonia and suspended solids entering the river, and an increase in dissolved oxygen and biological quality of the river. The decrease in suspended solids is likely to result in movement of fish into areas where the water is more turbid, for example, river confluences, or where shelter is provided, for example, bridges, marinas and underneath boats. This type of activity is a behavioural shift aimed at decreasing the risk of predation. This may make their capture by anglers difficult and may increase cormorant predation in the refuge areas where foraging is possible. The improvement in water quality has had a profound impact on the ecology of the river for the fish populations over a long time period, as revealed by the shifts in the relative importance of species in angler catches (Section 5.7.1).

5.8.5 *Water temperature*

The River Trent has seen a gradual decrease in water temperature since the decommissioning of power stations. During operation, river water was drawn off and used for cooling during the electricity generation and returned to the river at an elevated temperature. This resulted in the River Trent being a major winter angling fishery. However, the decline in winter water temperature has reduced winter angling success due to fish being poikilothermic. At lower temperatures, fish are less active and so feed less, resulting in decreased angler catch rates. Additionally, the historical elevated water temperatures may have induced earlier spawning of fish, which would have resulted in longer growing periods and improved over-winter survival of juvenile fish (Section 2.10.3), possibly resulting in an increased number of recruiting 0-group fish into the population.

The effect of the improved water quality and declines in winter temperatures have had a profound effect on the fish populations of the River Trent. Cormorant predation probably did not have such an impact on the fish populations over the three-year study period as these factors over a fifteen to twenty-year period.

5.8.6 *Angler catches*

Angler catches on the River Trent compare favourably with other lowland fisheries, for example, the River Ouse in Yorkshire (Axford 1991). Anglers have complained of a decline in the fishery in winter, which is due to changes in angling conditions and fish behaviour resulting from the improvements in water quality (Section 5.8.4) and the decrease in winter temperatures (Section 5.8.5), rather than the presence of a reduced stock of fish due to cormorant predation.

In conclusion, cormorants observed on the River Trent mainly used the sections for locating adjacent stillwater feeding sites. Low intensity feeding was observed on all sites, with moderate feeding success in comparison with adjacent feeding sites. The estimated fish losses from the MCS model were considered to be too high due to limitations in the MCS model, although it was acknowledged a substantial amount of fish were likely to have been lost to cormorant predation over the study period.

6. COLWICK PARK TROUT LAKE, NOTTINGHAM

6.1 Site details

6.1.1 *Introduction*

Colwick Trout Lake (SK 610395) (Plate 6.1) is situated within Colwick Park, a nature reserve controlled by Nottingham City Council. Colwick Park is located opposite Holme Pierrepont Watersports Centre (Chapter 4) on the left hand bank of the River Trent (Chapter 5) (Figure 3.2). The Trout Lake has an area of approximately 25 hectares and was constructed from disused gravel workings in 1978. It was filled from the River Trent. The water level is maintained by the removal or addition of stank boards in a channel that links the lake to the River Trent. The channel does not allow the movement of fish into the lake due to the presence of a meshed screen.

Colwick Park is used as a local amenity by a number of user groups for activities including put-and-take rainbow trout fishing, coarse fishing, walking, wind-surfing and boating. The lakes, which also attract large numbers of wildfowl in the winter, are popular with bird watchers.

6.1.2 *Habitat characteristics of Colwick Trout lake*

Colwick Trout Lake has an irregular topography as a result of the gravel abstraction, with depth variations of 2 to 6 m. Three small islands have been constructed on the lake to encourage nesting birds. A number of large boulders are present in the north east corner of the lake which are the remainder of a shallow recreation area. Cormorants have been observed using the boulders as loafing and roosting areas. Some of the boulders were removed in 1997 and 1998 to discourage cormorant occupancy of the site.

Growth of submerged vegetation is extensive in most areas of the lake, especially Canadian pondweed (*Elodea* sp.). This generally hinders angling in the shallow areas of the lake during the summer. Bankside vegetation consists primarily of grassland with an area of tree cover on the north bank.

6.1.3 *Resident fish stocks*

Colwick Trout Lake is stocked annually with approximately 10,000 rainbow trout. Trout are stocked in batches at fourteen-day intervals during the fishing season, from mid-March until late October. The residual stock of trout in the lake from previous seasons is considered to be zero due to poor over-wintering capacity and high natural mortality. Stocking undertaken in March and early April usually involves large fish (350 to 450 mm) in an attempt to reduce the short-term predation from cormorants loafing and roosting on the lake at this time. After emigration of the majority of cormorants to breeding colonies in April and May, smaller trout (250 to 300 mm) are generally stocked.

Other fish species are known to be present in the lake, including roach, bream, pike, perch, tench and eels. These fish are not actively exploited by anglers. The presence of coarse fish is the result of initial flooding from the River Trent and from movement of fish fry into the lake through the connecting channel with the river. Larger coarse fish cannot move into the lake due to a meshed screen.

6.1.4 *Cormorant populations*

No historical data for cormorant numbers occupying Colwick Trout Lake were available prior to the study, although their presence was noticed by the site wardens.

6.2 Materials and methods

6.2.1 *Cormorant monitoring*

The number of cormorants using Colwick Trout Lake was recorded by three methods:

- counts during the first hours of day light. These were part of the monthly co-ordinated counts which coincided with counts at Holme Pierrepont (Chapter 4) and the River Trent (Chapter 5);

- during feeding observations;

- counts throughout the day, on one day each month, allowing cormorant diurnal occupancy patterns to be determined.

The frequency of cormorant counts was increased during the period of trout stocking in mid-March and April. This enabled a detailed picture of cormorant occupancy to be determined in this period. Thus, the estimate of *N*, the number of cormorants present at the fishery, *f*, the proportion of cormorants that fed there on a given day, *c*, cormorant daily food intake, and *Pi*, the proportion of species *i* in the diet, with their distributions in the MCS model (Section 3.4.2), were determined after the trout stocking (Section 6.3.1).

6.2.2 *Fisheries monitoring*

Electric fishing, using the gears described in Section 3.3.1, were used to assess the trout populations in Colwick Park Trout lake. Rainbow trout stocking and angler catch data were also collected.

Boat mounted electric fishing surveys on the Trout Lake were undertaken in all available habitats. Lake bed topography was very uneven with shallow and deep areas found in close proximity. This caused a large variance in catch efficiency.

The number of fish stocked each year during the study was recorded and a sub-sample of these fish were measured and weighed prior to stocking. Seasonal catch data allowed the number of fish removed by anglers to be determined, and angler performance, measured as the number of fish per angler, was calculated. However, daily angler catch rate monitoring in relation to the trout stocking and cormorant predation was not possible.

6.3 Data analysis

6.3.1 *Cormorants*

The following cormorant data outputs, using the methodologies described in Section 3.4, were generated for the Colwick Trout Lake from observations and counts:

- seasonal cormorant occupancy;
- cormorant occupancy in the periods pre- and post-stocking of rainbow trout;
- diurnal cormorant occupancy;
- diurnal cormorant occupancy in the periods pre- and post-stocking of rainbow trout;
- cormorant feeding success pre- and post-stocking of rainbow trout;
- MCS generated losses of stocked rainbow trout attributable to cormorants in the post-stocking period.

The MCS model was only used in the post-stocking period due to the low number of cormorant feeding bouts observed in the pre-stocking period. The model used site specific methodologies on the Trout Lake to determine the values and distributions of *N*, *f*, *c* and *pi* (Feltham *et al.* 1999).

Estimating *(N.f)* - the number of birds feeding at Colwick Trout Lake

To estimate *N*, mean daily count data were plotted against date and a 3rd order polynomial curve was fitted through the data. This generated cormorant number estimates for each day of that month where direct observations were not possible due to a lack of man power resources. Therefore, total bird days per month (*N.d*) was derived by integration of the area beneath the curve (Feltham *et al.* 1999).

The variation around f was generally estimated by setting f_{min} as the proportion of birds actually observed feeding at the site and f_{max} as an upper limit set by assumptions regarding the proportion of flying and roosting birds that would meet some or all of their DFI at the site (Section 3.5.2). However, at Colwick Trout Lake, the data on the changes of cormorant occupancy at the site from daily and hourly counts in the post-stocking period enabled f to be set to 1. Thus, all cormorants that were seen landing, roosting or feeding at the site after stocking were likely to have met some, or all, of their DFI at the site (Feltham *et al.* 1999).

Estimating (c) - the daily food intake or DFI

After the trout were stocked in mid-March, DFI was estimated from the cormorant feeding observations. This was possible as the birds fed exclusively on the stocked trout and were easy to observe. Due to long handling times, the trout were relatively straight forward to identify and size (T. Holden pers. comm.). As the cormorants were observed to only consume one trout per foraging bout, DFI was easy to calculate by use of site specific length-weight regressions which were determined from a sub-sample of the stocked trout (Feltham *et al.* 1999).

Estimating (P_i) - the proportion of prey type *i* in the diet

The estimation of *Pi* was straightforward since all prey consumed after stocking were trout. Thus, *Pi* was constant (*Pi* = 1) ((Feltham *et al.* 1999).

6.3.2 *Fisheries*

The following fisheries data outputs, using the methodology described in Section 3.5, were generated for the Colwick Trout Lake from fisheries surveys:

- species composition;
- length frequency distributions;
- angler catch performance.

6.3.3 *Assessment of the impact of cormorant predation on Colwick Trout Lake*

The impact of the cormorant predation was assessed by use of the following information:

- comparison of species composition from cormorant predation and electric fishing results. This shows selectivity by cormorants on the fish species available, as represented by electric fishing;

- comparison of length frequency of fish predated on by cormorants, trout stocked and fish caught by electric fishing. This shows the size selectivity by species by cormorants compared with sizes of fish stocked and caught by electric fishing;

- number of trout removed by cormorants as a proportion of the original number stocked;

- estimation of the number of cormorant-wounded fish present in the lake.

6.3.4 *Analysis of put-and-take trout fishery data*

The relationship between trout stocking and catch per unit effort is well documented (Crisp and Mann 1977; O'Grady 1980; Pawson 1982; North 1983). Angler catch per unit effort has been found to be dependent on the stocking policy of the fishery whereby catches are a function of the current stock density and previous stock/catch relationships (North 1983). In essence, higher angler catch rates are generally attributable to greater stock abundance.

The catchability of stocked fish diminishes rapidly with time post stocking (North 1983), so frequent restocking with sufficient fish is required to maintain a stock capable of producing a desired CPUE (Pawson 1982). North (1983) found 90 % of trout captured by anglers at Draycote Reservoir, Warwickshire, were caught within 45 days of stocking. The daily mortality rate of the stocked trout was calculated as 1.36 %, composed of natural mortality and undeclared catch by anglers (North 1983). O'Grady (1980) found a positive relationship existed between the number of anglers present in a week with the size of the catch the previous week. Therefore, fishery performance determined angler presence over time.

The important cormorant impact factors on Colwick Trout Lake were the impact of cormorant predation on:

- stock density/abundance (hence, CPUE and angler perception of the attractiveness of the fishery);
- financial cost of replacing lost fish;
- economic impact of reduced CPUE due to cormorant predation;
- the proportion of rainbow trout wounded by unsuccessful cormorant attack.

6.4 Assessment of the fish stocks in Colwick Trout Lake

6.4.1 *Species composition of fish populations in Colwick Park Trout Lake*

The fish community was dominated by rainbow trout (Figure 6.1). This was to be expected due to the annual stocking of 10000 rainbow trout. Cyprinids and pike were also caught in March 1996 and June 1996. In March 1996, 6 pike of between 6700 and 13800 g and a bream of 540 mm were captured. In June 1996, 2 tench of 200 mm were captured. One tench was caught in June 1997.

6.4.2 *Length frequency distribution from electric fishing surveys*

The policy of the manager at Colwick Park was to initially stock only rainbow trout greater than 350 mm to minimise cormorant predation losses. This policy appears to have been carried out reasonably efficiently, as few stocked trout below 350 mm were caught (Figure 6.2).

The trout were able to grow on in the lake, since rainbow trout up to 550 mm were caught, but no trout over 490 mm were stocked. It is impossible to ascertain if these trout had survived over-wintering or were fish stocked during that season.

No age and growth analyses were carried out as the trout were all stocked and had originated from fish farms.

6.4.3 *Angler catch per unit effort*

The number of anglers visiting the fishery declined between 1987 and 1998 (Table 6.1). However, the number of trout stocked per season remained relatively stable during the period. Although a number of fish may survive over-wintering, they are not as exploitable for anglers as recently stocked fish, due to catchability diminishing rapidly post stocking (North 1983). Therefore, it was assumed that only trout stocked in that year were available for exploitation by anglers.

The angler catch rate at the fishery, expressed as the number of trout per angler, showed an upward trend from 1.1 to 1.9 (Table 6.1). As angler effort declined, there were more trout present per angler, which probably resulted in the increased catch per unit effort. This was consistent with other observations that trout angler catch is a function of stock density (Crisp and Mann 1977; O'Grady 1980; Pawson 1982; North 1983) (Section 6.3.5).

Table 6.1 Colwick Park Trout Lake angler catch statistics 1986 to 1998.

Season	No. stocked	No. anglers	Returns (%)	No. fish caught	CPUE (Trout/angler)
1987	9000	6261	87	6113	1.1
1988	10700	5590	87	5444	1.1
1989	9500	6067	83	6703	1.3
1990	9500	5898	87	7120	1.4
1991	9600	5838	87	7007	1.4
1992	10000	4799	85	4691	1.1
1993	10100	3827	87	6288	1.9
1994	9700	4514	87	6898	1.8
1995	9600	4793	91	6647	1.5
1996	10000	4949	89	6230	1.4
1997	9700	4879	88	6130	1.4
1998	9400	3314	87	5521	1.9

6.5 Cormorant observation results

6.5.1 *Seasonal variation of the roost count*

The cormorants which utilised Colwick Trout Lake were birds from the Attenborough night roost. The seasonal variance in the roost count numbers are detailed in Section 4.5.1.

6.5.2 *Seasonal variation*

There was a seasonal change in bird abundance at Colwick Trout Lake over the study period. The periods of peak abundance were between November and March (Figure 6.3). The periods of low abundance were during the summer months (April to August), when the majority of the cormorants roosting at the Attenborough colony had dispersed for breeding in other areas (Section 4.5.1).

6.5.3 *Diurnal Variation*

Pre-stocking diurnal variation

Prior to trout stocking, cormorant occupancy of the site was low at first light, peaked around midday and declined in the remaining light hours (Figure 6.4). This cormorant occupancy pattern was related to reductions in the numbers of feeding birds at other sites in the area, such as Holme Pierrepont (Section 4). Thus, the majority of the cormorants had already fed at other local sites during the initial hours of daylight before utilising Colwick Trout Lake as a day roost area (Section 4.5.1).

Post-stocking diurnal variation

In the period post-stocking, peak cormorant numbers occurred at the site in the first hour after sunrise, with an increased number of cormorants present at that time (Figure 6.4). The number of birds occupying the site then fell before remaining constant throughout the remainder of the day (Figure 6.4). After stocking of the lake with large numbers of hatchery reared rainbow trout, the cormorants utilised the site as a feeding site during the early daylight hours. Thus, the stocking of the trout changed the attractiveness of the site for cormorants from a day roost site to a feeding site.

Despite the behavioural shift observed in the cormorants, the actual number of cormorants using Colwick Trout lake did not increase after stocking with rainbow trout (Figure 6.5). Cormorant occupancy continued to decline similar to the seasonal trend of cormorant abundance at the Attenborough roost (Section 4.5.1). Thus, rainbow trout stocking had little influence on the bird occupancy of the site, except the small 'resident' group of birds that fed daily on the stocked trout in the initial hours of daylight at the site, before breeding dispersal.

6.5.4 *Cormorant feeding behaviour*

In the period of study, a total of 327 cormorant foraging bouts were observed, with 118 (36.1 %) successful (at least one fish caught), although variation in success was seen (18 to 67 %) (Table 6.2). Differences were observed in the percentage of successful foraging bouts pre- and post-stocking, with success increasing post-stocking in all study years (Table 6.2).

The percentage of successful dives by cormorants on the lake was low (1.8 - 13.9 %), with little difference observed between pre- and post-stocking periods (Table 6.2). Thus, the stocking of trout had little influence on the dive success of cormorants, but a large influence on overall foraging bout success.

Table 6.2 Feeding rates and foraging success of cormorants at Colwick Park Trout Lake, 1995 to 1998.

	1995-96		1996-97		1997-98	
	Pre-stocking	Post-stocking	Pre-stocking	Post-stocking	Pre-stocking	Post-stocking
Foraging bouts	38	45	46	51	83	64
Successful foraging bouts	7	15	6	34	23	33
% bouts successful	18	33	13	67	28	52
Dives observed	544	455	529	358	1256	738
Dives successful	10	15	47	34	174	33
% dives successful	1.8	3.3	8.9	9.5	13.9	4.5

6.6 Impact of cormorants at Colwick Park Trout Lake

6.6.1 *Species composition of cormorant predation and electric fishing*

Electric fishing surveys revealed stocked rainbow trout were the main species present in the lake with very small numbers of pike, tench and bream (Figure 6.1). The pike and common bream were considered to be of specimen size.

In 1995/96 the main dietary component of cormorants was rainbow trout (Figure 6.1). Perch and unidentified fry were also taken, but these were not caught in electric fishing surveys. Rainbow trout were again the dominant food eaten in 1996/97 and 1997/98, but unidentified fry were consumed in greater numbers during this period.

In the post-stocking period, all fish predated on by cormorants were rainbow trout, with increased numbers of trout consumed post-stocking when compared with the pre-stocking period (Table 6.3).

6.6.2 *Length frequency distribution*

The sizes of the stocked trout were in the range 300 to 450 mm (Figure 6.2). However, fish up to 500 mm were caught during electric fishing surveys indicating growth of stock (Figure 6.2). Cormorants ingested trout in the size range 200 to 450 mm, with a high proportion of fish taken in the size range 250 to 300 mm (Figure 6.3). They appeared to be selecting the smaller sizes of rainbow trout present in the fishery for consumption. These fish were presumably small sized individuals which were mistakenly stocked with the main batches of fish.

Table 6.3 Cormorant prey in pre- and post- rainbow trout stocking periods at Colwick Park Trout Lake, 1995 to 1998.

	Numbers of cormorant predated species			
	Rainbow trout	Perch	Cyprinids	'Fry'
Pre-stocking 95/96	2	5	1	0
Post-stocking 95/96	15	0	0	0
Pre-stocking 96/97	4	0	0	43
Post-stocking 96/97	34	0	0	0
Pre-stocking 97/98	4	14	0	141
Post-stocking 97/98	33	0	0	0

6.6.3 *Monte Carlo simulated cormorant losses against stocked fish*

The Monte Carlo simulation model was applied to assess the effects of cormorant predation on the stocked rainbow trout populations in Colwick Trout Lake over the study period. In comparison of the MCS data with the stocking data, the stock of trout in the winter was assumed to be zero. The initial stocking of approximately 2000 trout was carried out annually in mid-March with approximately 200 trout then stocked every 14 days.

In the absence of cormorant predation, the numbers of rainbow trout present had an assumed mortality rate of 1.36 % per day applied, consisting of natural mortality and undeclared angler returns (North 1983). This mortality rate was determined from data collected over a number of years at Draycote Reservoir, a put-and-take trout fishery in Warwickshire (North 1983). Angling began on the lake on 15 March. The daily angler catches will have had an impact on the stock abundance by the removal of fish. However, data were not available for daily angler catch and effort so this could not be incorporated into the model.

During the period of study, there was a 16 to 19 % reduction in the abundance of stocked fish to the anglers by the end of April each year, resulting from the cormorant predation (Figure 6.6, Table 6.4). As stock abundance is directly related to angling success (O'Grady 1980; Pawson 1982; North 1983; Crisp & Mann 1997) cormorant predation probably has a negative effect on the fishery performance which may have manifested itself in two ways:

- direct economic loss;
- reduced angler satisfaction which may translate into anglers boycotting the venue.

Over each whole angling season, the trout losses attributable to cormorants equated to 5.9 to 7.8% of all stocked fish. This proportion increased over the study period (Table 6.5). Fortunately for the management of the fishery, cormorant occupancy declined in April until no cormorants were present by May (Section 6.5.2). This was due to annual migration of the mature cormorants from the Attenborough roost to breeding colonies elsewhere (Section 4.5.1).

Table 6.4 Number of stocked trout, ingested by cormorants, in March and April of each study year.

Angling season	Number stocked	Number consumed	%
1996	3640	591	16.2
1997	3623	631	17.4
1998	3781	737	19.4

Table 6.5 Number and value of stocked trout ingested by cormorants at Colwick Park Trout Lake over the angling seasons 1996, 1997 and 1998.

Angling season	Number stocked	Number consumed	%	Value (£)
1996	10,000	591	5.9	413.7
1997	9700	631	6.5	441.7
1998	9400	737	7.8	515.9

6.6.4 *Economic losses*

The loss to the fishery owners was determined from the replacement cost of the consumed trout (Table 6.5). The losses ranged from £413 in 1996 to £515 in 1998. Evaluation of the economic loss, because of fishery avoidance by anglers due to the cormorant predation and its effect on the stock abundance and angler CPUE, was impossible to assess. However, as there are a number of put and take trout fisheries in the Nottinghamshire region, it is likely that loss of revenue due to angler avoidance was significant.

6.6.5 *Wounding rates*

Direct fish and economic losses were not the only impact that cormorants caused at Colwick Park Trout Lake. In electric fishing surveys a number of rainbow trout above 400 mm showed signs of non-lethal attack by cormorants (Table 6.6). Plate 6.2 and 6.3 show a trout of 460 mm with a typical cormorant wound consisting of an abdominal hole caused by the cormorant's sharp beak on one side of the fish and slash marks and scale loss caused by the lower mandible on the other side (Section 2.9.1).

Table 6.6 Percentage and sizes of rainbow trout bearing cormorant damage, sampled by electric fishing, at Colwick Park Trout Lake over the study period.

Sample date	No. sampled	No. wounded	%	Average size sampled trout (mm)	Average size wounded trout (mm)
October 95	41	5	12.2	387	404
January 96	2	1	50.0	445	460
March 96	28	3	10.7	425	428
June 96	3	0	0	485	-
Jan 97-Jun 98	13	1	7.7	439	470

The overall percentage of sampled trout bearing cormorant damage was 11.4 %. This rate was applied to the March/April stocking data to estimate the number of trout which would have suffered non-lethal cormorant attack (Table 6.7).

Table 6.7 The number of stocked trout in March and April over the study period surviving after cormorant predation with the potential number bearing cormorant damage.

Angling season	No. stocked in March/April	No. consumed in March/April	No. surviving predation	No. surviving with cormorant wounds
1996	3640	591	3049	348
1997	3623	631	2992	341
1998	3781	737	3044	347

Thus, a proportion of the surviving rainbow trout are likely to have avoided lethal cormorant attack, but may have incurred physical damage to their body. To many anglers this makes those trout unacceptable if captured. The policy of the management at Colwick Park Trout Lake was to replace any damaged trout presented by an angler with a frozen trout bearing no damage. This ensured wounded trout presence and capture did not diminish the attractiveness of the fishery to visiting anglers. However, it represented an additional financial loss to the fishery. This loss was impossible to evaluate as the role of natural mortality and delayed mortality, caused by secondary infection of the wound, could not be quantified in the stock abundance of wounded fish.

Rainbow trout caught in electric fishing surveys at Colwick Park Trout Lake were the only fish caught over the study period in Midlands study sites which bore obvious cormorant attack wounds.

6.7 Discussion

The other important issues that must also be considered when evaluating the cormorant impact on the fishery are:

- the presence of other fish predators;
- size groups of predated trout;
- timing of the initial rainbow trout stockings;
- stocking mechanism.

Cormorants were not the only piscivores at the site. Several large pike were caught during electric fishing and these could inflict a potential heavy trout predation mortality. Conversely, cormorants were seen feeding on juvenile coarse fish in the trout lake. However, as rainbow trout were the only fish consumed in the period after stocking, these cyprinid populations were only buffering the over-wintering trout population, which was apparently low.

The size groups of trout predated on by the cormorants were rather anomalous. The size of fish stocked were supposed to be in the size range 300 to 450 mm. However, cormorants were mainly consuming fish between 250 and 300 mm. This may be the result of inaccurate assessment of the size of food ingested, for it was a subjective assessment, or undersized fish were being stocked. Irrespective of this, the selection of smaller sized trout by cormorants was a good indicator of the value of stocking large fish (above 400 mm) in cormorant predation prevention. Unfortunately, the effectiveness of the measure is counteracted by the higher level of wounding that was shown to occur in the larger fish. The wounding occurs as the cormorants are unable to handle the larger fish and they escape. There are also increased costs in stocking larger individuals.

The timing of the stocking was important as cormorants only occupied the lake until the end of April. Thus, if trout stocking (and fishery opening) was delayed until after cormorant dispersal from the area, the cormorant predation pressure on the trout would be negligible. This would have to be matched against the loss of revenue during the six weeks when the fishery would still be closed. An assessment of angler activity and income from the fishery against losses due to cormorants would provide a solution to this problem.

The stocking mechanism was an important factor on the rate of cormorant predation. Observations on bird behaviour at the time of stocking revealed predation was heavy when trout were introduced at only one point, until they dispersed (T. Holden pers. comm.). Thus, if the trout were scatter stocked in very low densities, the ease with which cormorants would find the naive, fish farmed fish would be lowered in the period prior to their adaptation to the water body. Alternatively, the cormorants should be continually disturbed at the time of stocking to force them away from the lake until the fish disperse after stocking. However, cormorants that were disturbed at the site during stocking were observed to return immediately and target the newly introduced fish (T. Holden pers. comm.). Other methodologies to prevent cormorant predation on put and take trout fisheries are discussed in Chapter 10.

Plate 6.1 Colwick Park Trout Lake, viewed from the south bank.

Plate 6.2 A puncture wound on a rainbow trout of 460 mm, inflicted by the sharp beak of a cormorant, at Colwick Park Trout Lake.

Plate 6.3 Slash marks and scale loss, resulting from scraping on the lower mandible of a cormorant beak, on the reverse side of the rainbow trout shown in Plate 6.2, at Colwick Park Trout Lake.

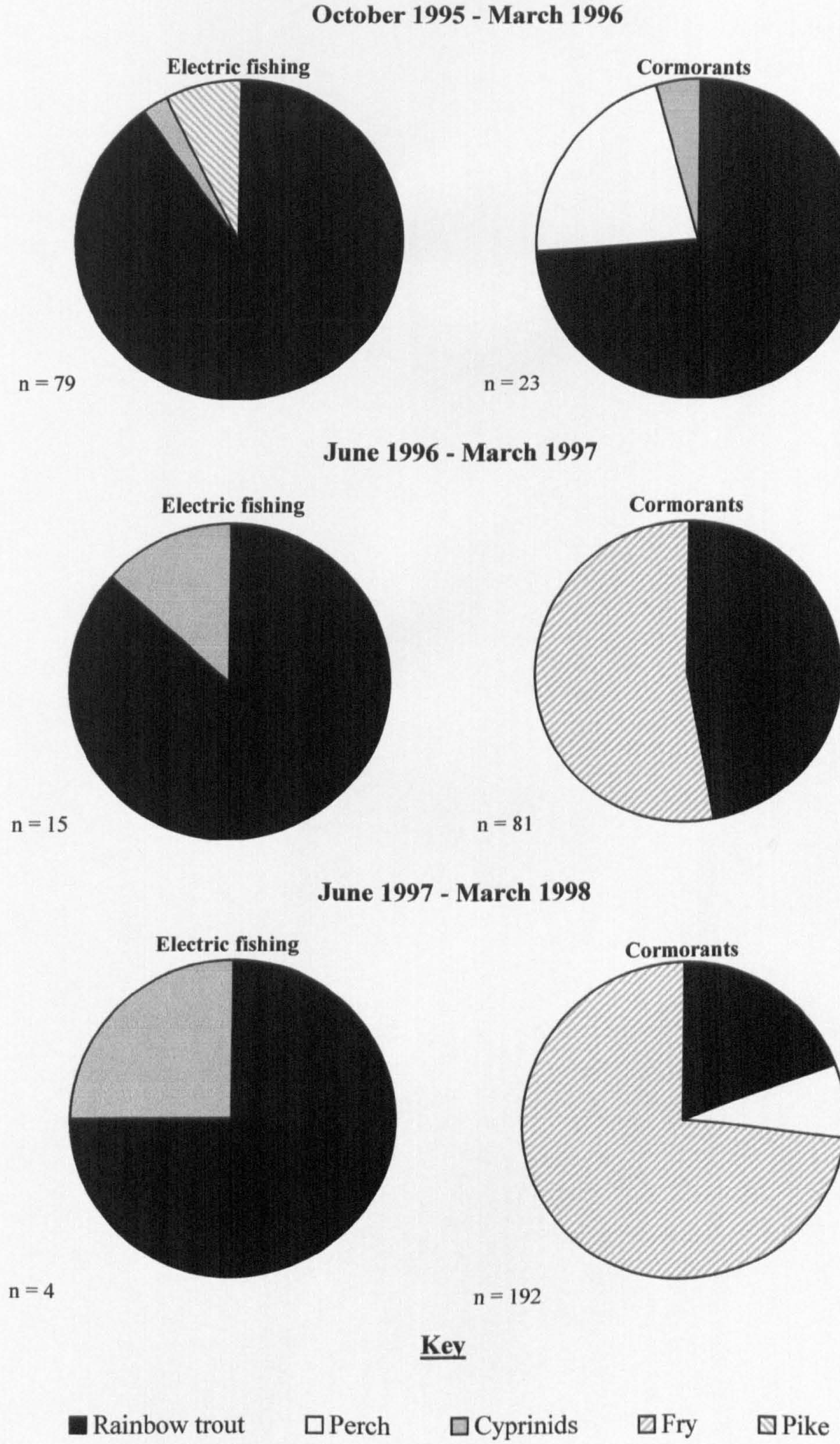


Figure 6.1 Species composition of electric fishing surveys and cormorant predation on Colwick Park Trout Lake.

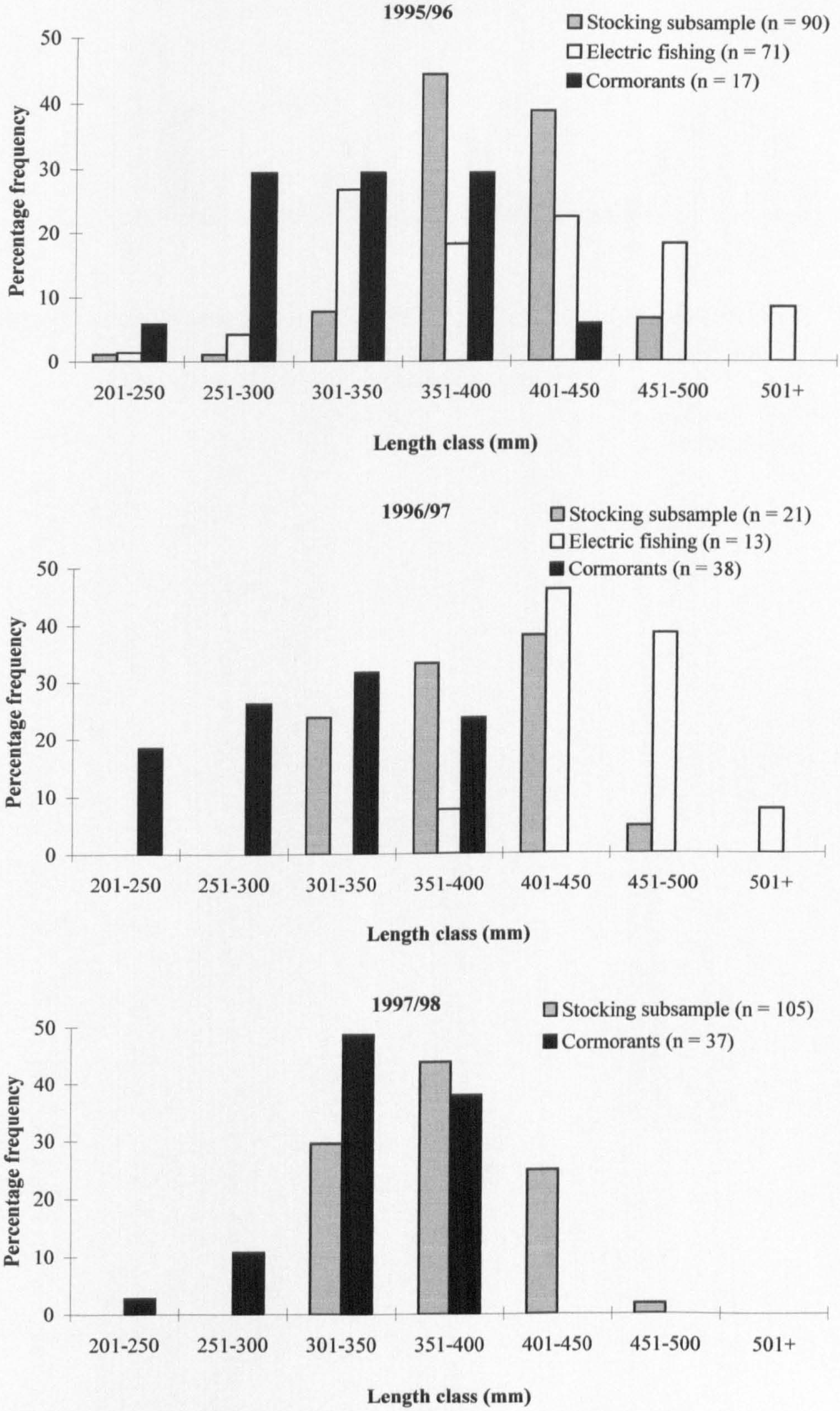


Figure 6.2 Length frequency of rainbow trout caught by electric fishing, stocked and ingested by cormorants at Colwick Park Trout Lake.

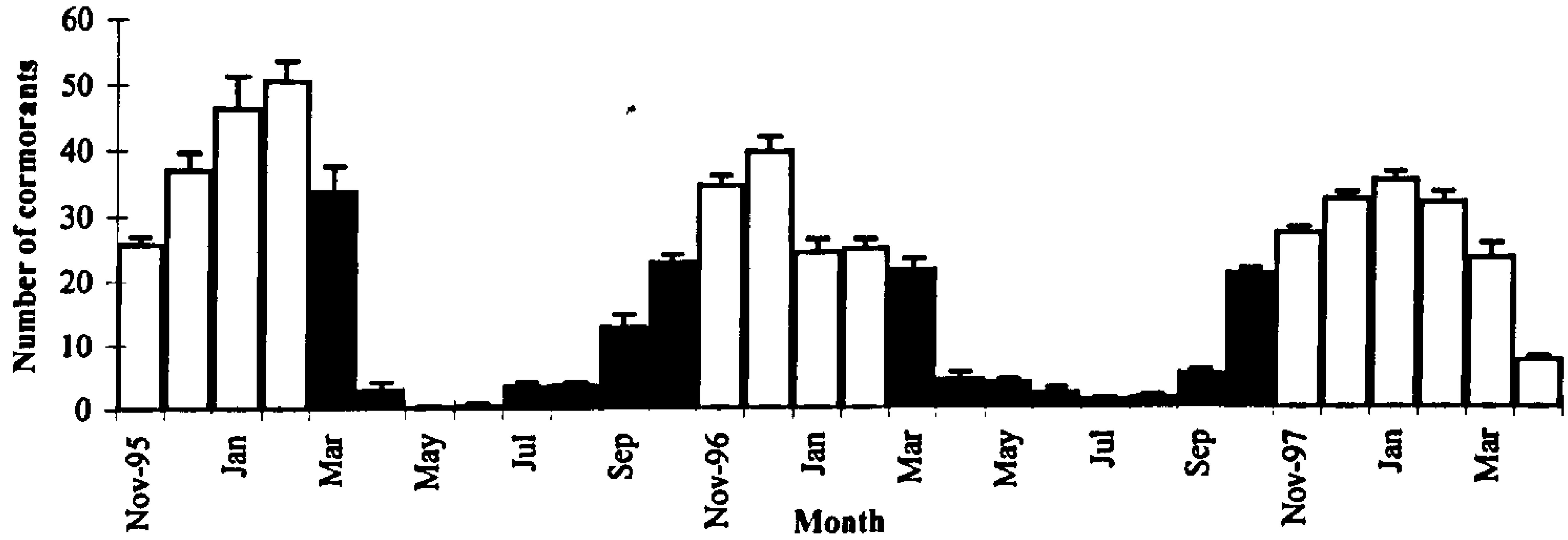


Figure 6.3 Seasonal variation in the number of cormorants observed at Colwick Park Trout Lake over the study period. The solid bars indicate the months when rainbow trout stocking occurred.

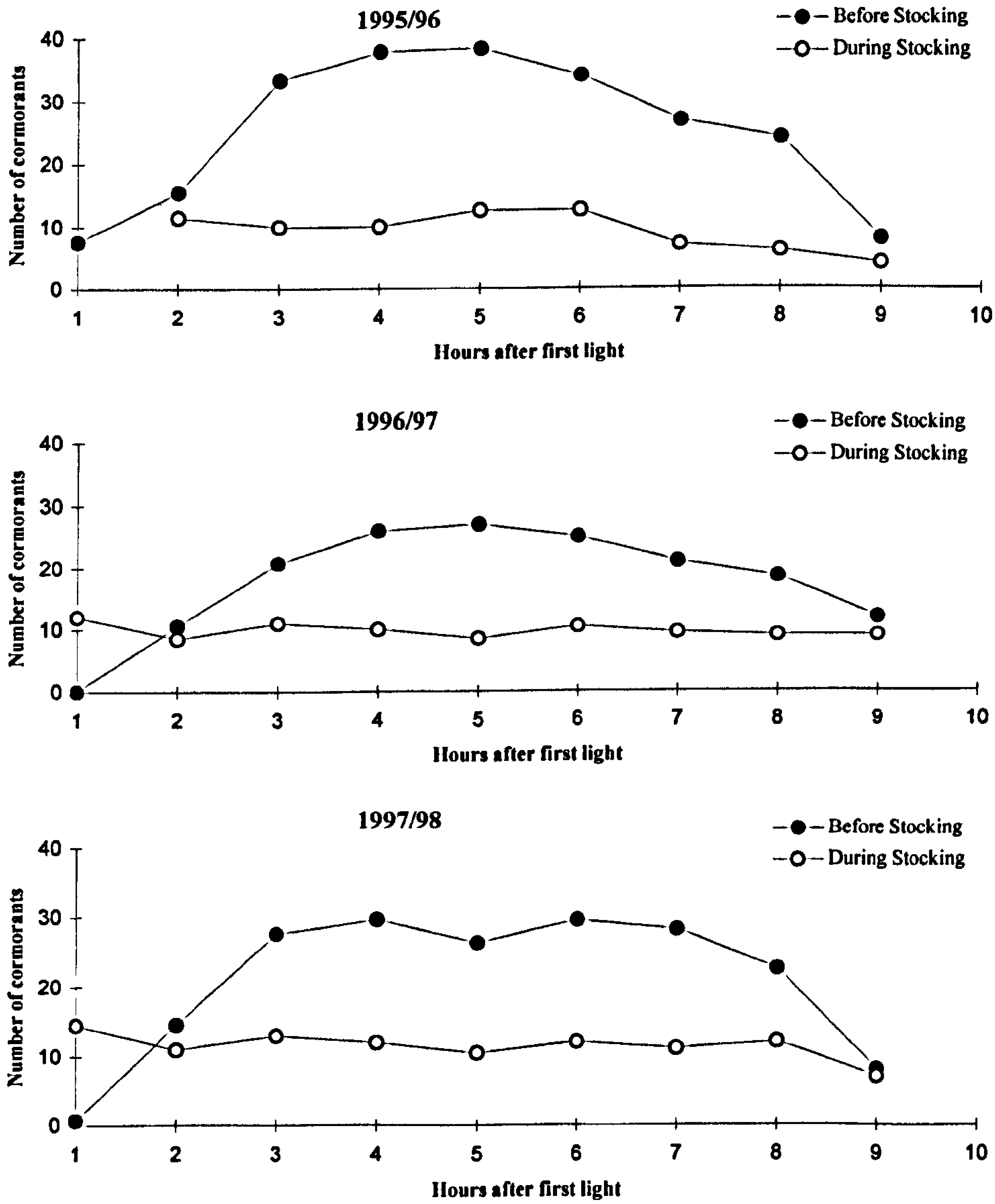


Figure 6.4 Diurnal variation in the mean number of cormorants at Colwick Park Trout Lake during pre- and post- trout stocking periods, 1995 to 1998.

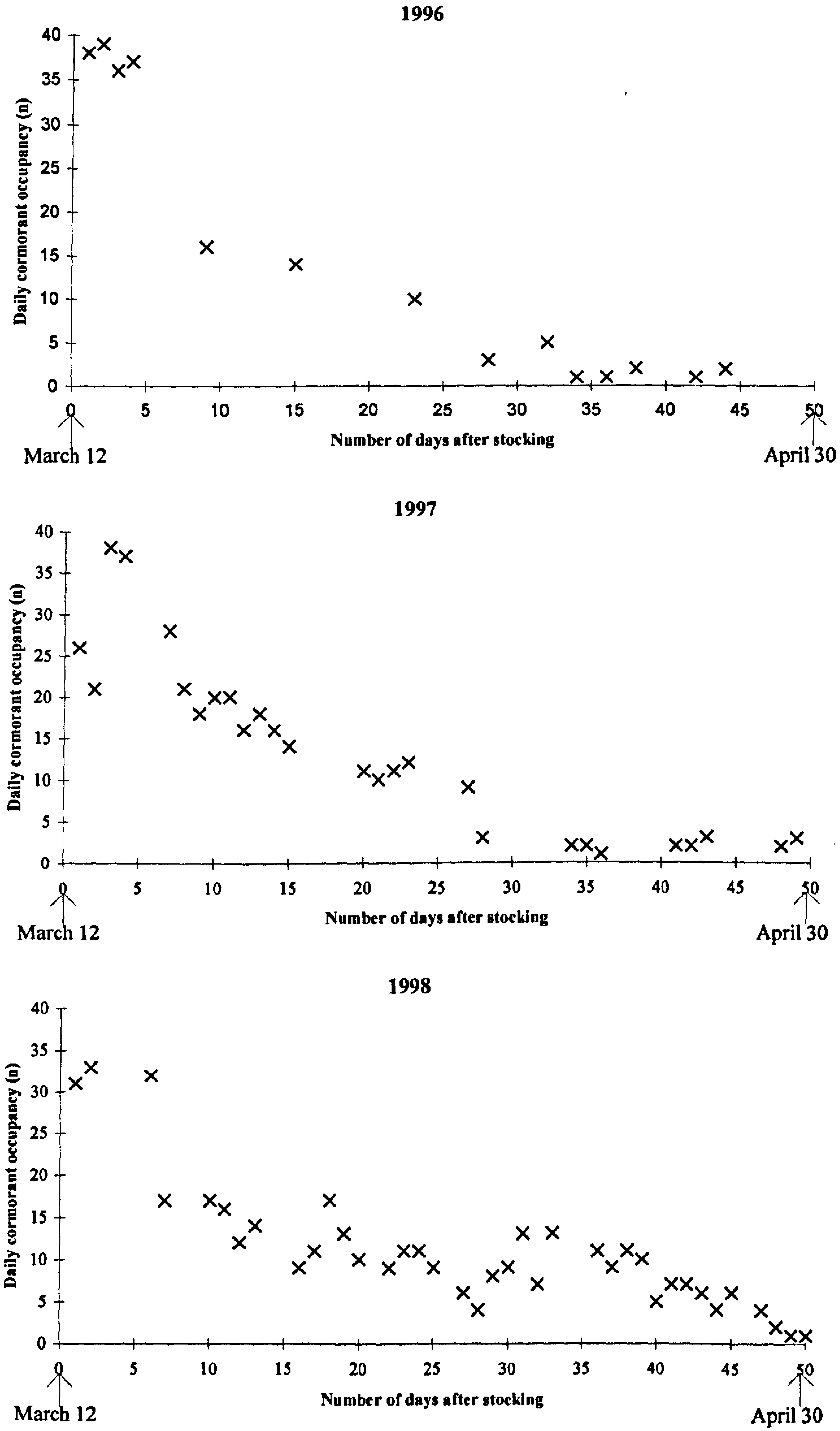


Figure 6.5 The number of cormorants feeding at Colwick Park Trout Lake from initial stocking to April 30th, 1996 to 1998.

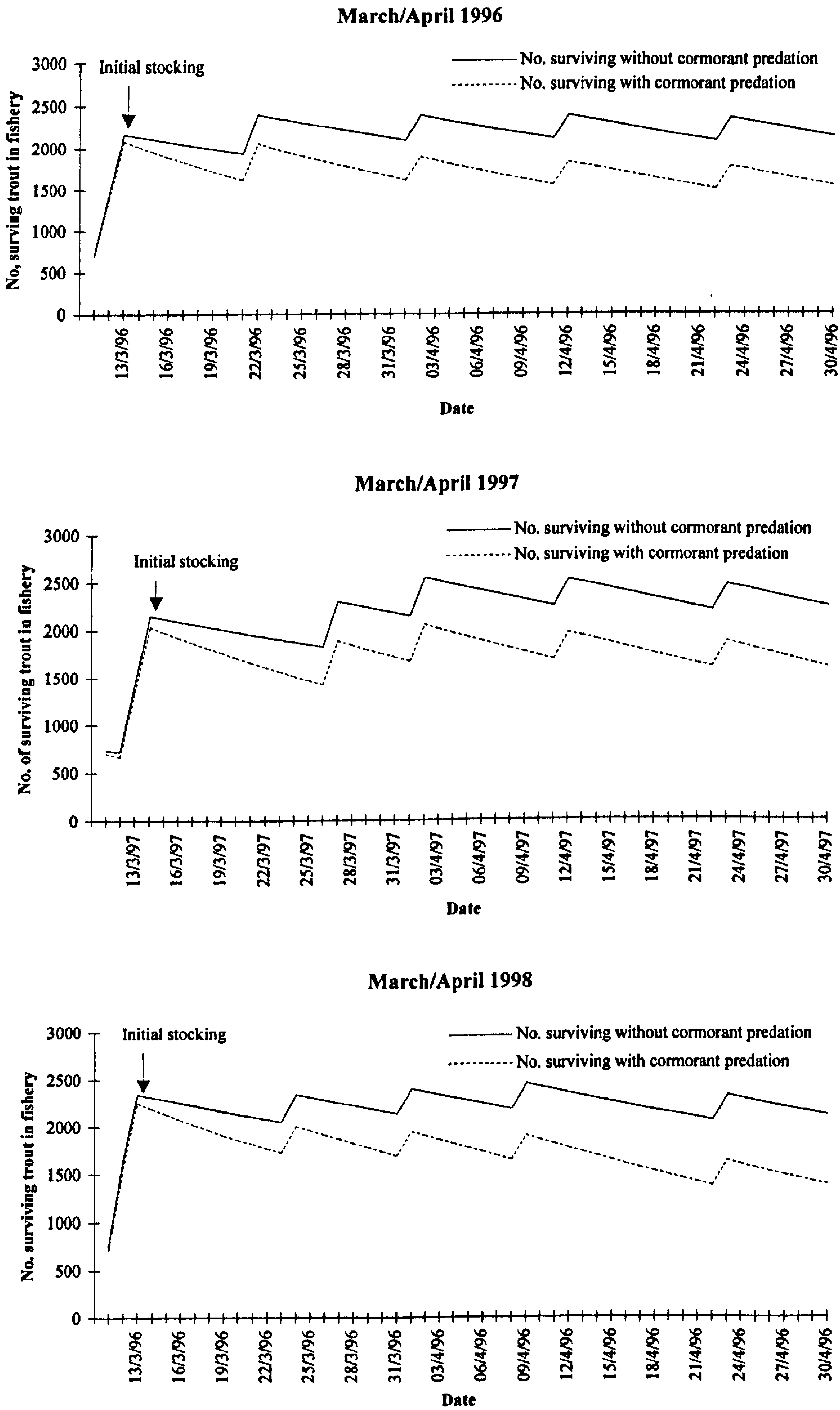


Figure 6.6 Impact of cormorant predation on the numbers of stocked rainbow trout in Colwick Park Trout Lake.

7. GRIMSARGH NUMBER 3 RESERVOIR, LANCASHIRE

7.1 Introduction

Grimsargh reservoirs (SD 590347) are situated approximately 4 km east of Preston, in the village of Grimsargh (Figure 3.3). The Grimsargh reservoir complex consists of three reservoirs, and is used as a water supply by North West Water and as a recreational fishery by Redscar Angling Club. Number 1 reservoir is used exclusively as a put-and-take trout fishery. Number 2 reservoir is a day-ticket coarse fishery, and number 3 reservoir is a members only coarse fishery (Plate 7.1). The water level of the reservoirs is maintained by North West Water and fluctuates during the year. Essential maintenance work on the reservoirs was carried out between March 1997 and March 1998 and involved draining down of each reservoir. Hence, research on the site ended in March 1997.

Anecdotal evidence suggested reservoir number 3 had historically been a productive cyprinid fishery (N. Watson pers. comm.). However, a period of heavy cormorant predation in the early 1990s was suspected to be the cause of a large decline in angler catches by the controlling angling society (N. Watson pers. comm.).

7.2 Site details

Reservoir number 3 was studied to estimate the impact of cormorant predation on the fish populations. The reservoir was 200 m long and 150 m wide (area = 3 hectares), with the water retained by raised flood banks. The average depth was 2.5 m and the substratum consisted of mud and stones. The land on the north and east side of the reservoir was used as pasture. Reservoir number 2 was situated on the west side of the complex and the south side was bordered by residential properties and the main road between Longridge and Preston.

Bankside vegetation consisted of maintained grassland. Little marginal vegetation was present in the reservoir except for localised stands of reed. Two small areas of rooted, emergent vegetation were present on the north and east side of the reservoir in approximately 2 m of water.

7.3 Materials and methods

7.3.1 *Cormorant monitoring*

Cormorant occupancy and feeding behaviour were recorded by dawn until dusk observations (Section 3.2). In winter 1995/96, observations began on 23 January 1996, the day before the reservoir was stocked with 6011 roach (Section 7.3.3). Subsequently, the site was monitored, on average, every 3 to 4 days for 90 days post-stocking. Observations were taken from a vehicle parked on the east bank of the reservoir. In winter 1996/97, dawn until dusk observations were carried out between October and April, the period of cormorant occupancy on the reservoir.

7.3.2 *Fisheries monitoring*

Boat-mounted electric fishing was used to assess the fish populations in the reservoir (Section 3.3.1). Electric fishing surveys were undertaken using a relative assessment strategy which involved sampling the whole of the reservoir area, manoeuvring the boat to provide coverage of all habitats in the reservoir. Electric fishing surveys were conducted in October 1995 and March 1996. However, fish capture efficiency was poor due to the low conductivity of the reservoir. As a result, an alternative survey method was required.

Consequently, fish stock assessments were undertaken in June 1996, October 1996 and March 1997 using a 75 m long and 3 m deep seine net (Section 3.3.2). To provide coverage of the reservoir area, a number of trawls of the seine net were undertaken. All the fish caught were placed in holding nets and, following completion of all the netting, were identified and measured (fork length, mm). The data were processed as described in Section 3.3.6. Seine netting in March 1997 was carried out during draining of the reservoir for maintenance work. All fish were removed and placed in Reservoir 1.

7.3.3 *Stock manipulation*

The electric fishing surveys and angler catches in 1995 indicated few fish were present in the reservoir. Hence, for the study a stock manipulation experiment was performed in the reservoir to assess the impact of cormorant predation on stocked roach of sizes below 200 mm.

Grimsargh reservoir number 3 was stocked with 6011 roach in the size range 45 to 175 mm in January 1996. Each fish was marked by clipping the pectoral fin on fish below 100 mm, or by Panjetting fish of above 100 mm. Cormorant occupancy and feeding on the reservoir was monitored in the period pre- and post stocking to assess any changes associated with the stocking programme.

7.4 Data analysis

7.4.1 *Cormorants*

Cormorant data were analysed to enable the species and size of fish removed by the birds to be compared with the fish species and sizes in the reservoir, as revealed by seine netting, and to determine the effect of the roach stocking in January 1996 on cormorant occupancy.

To estimate the amount of fish removed by the cormorants over the winter periods of 1995/96 and 1996/97, the cormorant feeding observations data were used. As observations covered the entire period of daylight hours on the reservoir, estimating the amount of fish removed by birds was straightforward and did not require the Monte Carlo approach, as N, f, P_i and c were known (Feltham *et al.* 1999). To obtain estimates of the level of cormorant predation on days when observations were not carried out, 3rd order polynomials were fitted to the observation data (Feltham *et al.* 1999). The total number of fish removed on these days was derived by integration of the area beneath the curve (Feltham *et al.* 1999).

7.4.2 *Fisheries*

The following outputs, using the methodology described in Section 3.5, were generated for Grimsargh number 3 reservoir from the fisheries surveys:

- standing crop;
- species composition;
- length frequency distributions;
- age and growth determination of fish species;
- growth indices and models;
- mortality rates, survival rates and year class strengths;
- cohort reconstruction;
- angler catch performance.

7.4.3 *Assessment of the impact of cormorant predation on Grimsargh Reservoir number 3*

The impact of cormorant predation was assessed using the following data.

- Comparison of species composition of fish eaten by cormorants and caught by seine netting and anglers. This will show species selectivity by cormorants and anglers against the baseline netting surveys.

- Comparison of size of fish eaten by cormorants and caught by seine netting and anglers. This will show the size selectivity by species of cormorants and anglers.

- Comparison of the number and biomass of fish in the survey sections (seine netting catch data) and without (seine netting catch data plus MCS generated cormorant predation losses) cormorant predation.

7.5 Status of fish populations in Grimsargh reservoir number 3

7.5.1 *Stock manipulation*

The size class distribution of roach stocked into the reservoir in January 1996 was compared with the seine netting catches in June 1996 (Figure 7.1). Of the 339 fish caught, 89 (26 %) originated from the stocking, the remainder were from natural recruitment.

The catches of non-stocked roach in the survey were approximately three times higher than the stocked roach, indicating a significantly higher natural population of roach in the reservoir than had been thought present on the basis of electric fishing results and angler catches.

The stocked roach showed evidence of some growth between the stocking and netting operations (Figure 7.1).

7.5.2 *Standing crop*

The standing crop of the roach, bream and perch stocks were calculated from data used in cormorant impact assessment using cohort reconstruction (Section 3.5.9). Variation was observed between species with total standing crop relatively stable over the study period (Table 7.1).

Table 7.1 Standing crop of species, estimated by seine netting, at Grimsargh number 3 Reservoir over the study period.

	Standing crop (kg ha^{-1})	
Species	**1995/96**	**1996/97**
Roach	14.00	26.91
Common bream*	35.14	18.23
Perch	11.91	21.67
Total standing crop	61.05	66.81
Total standing crop g m^{-2}	6.11	6.68

*Excluding common bream above 450 mm.

7.5.3 *Angler catch data*

Two angling matches were held in June 1996 when the species and sizes of fish caught were recorded (Table 7.2). Data were not available to calculate catch per unit effort and percentage of anglers with catch.

Table 7.2 Angler catch results from two matches at Grimsargh number 3 Reservoir in June 1996.

	Match 1	Match 2
Number anglers	8	3
Number of fish caught	38	14
% roach	53	50
Median roach length (mm)	121	92
% common bream	36	36
Median bream length (mm)	139	137
% perch	11	14
Median perch length (mm)	96	95

The median length of common bream hides a marked bimodal distribution of small bream (50 to 150 mm) and large (above 450 mm) older bream, with no common bream in the mid-size range, 150 to 450 mm.

7.5.4 *Species composition of fish populations at Grimsargh number 3 Reservoir*

The species composition of fish caught by seine netting was dominated by cyprinid species (Figure 7.2). Cyprinids mainly comprised common bream (62 to 69 %) and roach (20 to 24 %). Perch (6 to 17 %) were the other major species caught, while minor species included pike, carp, eel and tench.

7.5.5 *Length frequency distribution*

The roach population (Figure 7.3) was dominated by individuals in the size range 80 to 140 mm. Few fish were caught over this size range, and no roach above 190 mm were caught. Although roach recruitment into the population appeared good, the roach were unable to attain large sizes. The majority of these roach were resident fish in the reservoir and not part of the stocking in January 1996 (Section 7.5.1).

The common bream population was composed of a similar size structure to roach, consisting of fish in the size range 70 to 110 mm (Figure 7.4). In addition, a small number of large bream over 450 mm was present in the reservoir (Figure 7.4). It is not known if these were historically stocked fish or a natural recruited population. Although common bream recruitment appeared good in the reservoir, no common bream of 190 to 450 mm were present.

The perch population (Figure 7.5) also showed good recruitment with few fish above 120 mm caught.

7.5.6 *Year class strength*

Analysis of the roach, common bream and perch population structure revealed 1995 produced a strong year class (Figure 7.6). 1994 was also a strong year class for roach and common bream. 1992, 1993, and 1996 were weak year classes for all species in the reservoir. Common bream year class strength was strong in 1981, 1985, 1986 and 1987, being related to the large bream present in the fishery (Figure 7.4).

7.5.7 *Growth rates*

The growth rates of roach, common bream and perch were all below average or slow compared with standard growth rates (Cowx *et al.* 1995) (Figure 7.7 to 7.9). Common bream were the only species to have a long life span. However, common bream were only aged between two and five years old and 11 and 16 years old, with no intermediate age classes present.

7.5.8 *L infinity and K*

The values for L_∞ and K obtained for roach, common bream and perch at Grimsargh number 3 Reservoir (Table 7.5) were similar to values obtained from other UK fisheries (Table 4.4, 4.5, 4.6), although the value of L_∞ for roach and perch were low. The presence of the large common bream increased the L_∞ value for this species.

Table 7.3 L_∞ and K values for roach, common bream and perch at Grimsargh number 3 Reservoir.

Species	L_∞	K
Roach	240	0.21
Bream	660	0.08
Perch	174	0.27

7.5.9 *Mortality and survival rates*

The mortality (Z) and survival (S) rates for roach, common bream and perch at Grimsargh No. 3 Reservoir indicated high mortality in these species (Table 7.4) when compared with values from other UK waters (Table 5.5).

Table 7.4 Mortality and survival rates of roach, common bream and perch at Grimsargh number 3 Reservoir.

Species	Z	S	If N_t=1000, N_{t+1}=
Roach	0.72	0.487	487
Common bream	*0.13	0.878	878
Common bream	**1.44	0.237	237
Perch	1.23	0.292	292

* Bream 11 to 16 years of age used in mortality calculation.
** Only bream 2 to 5 years of age used in mortality calculation.

7.6 Cormorant observation results

7.6.1 *Cormorant occupancy*

Stocking experiment - winter 1995/96

Two cormorants were observed on the reservoir on the day prior to stocking of the roach, with a similar number present in the first five days post-stocking (Figure 7.10). The number of cormorants on the reservoir then rose to between 4 and 13 for the next week, before the reservoir froze over. Once thawed, cormorant occupancy returned to the level observed before freezing. Thus, the mean number of cormorants using reservoir No.3 each day peaked in February at 7 ± 1 birds day^{-1} (Figure 7.10). Cormorant occupancy declined during March (Figure 7.10), returning to their pre-stocking numbers. No cormorants were observed at the site during April.

Winter 1996/97

Cormorant occupancy was monitored for all of the three reservoirs in winter 1996/97. Numbers steadily increased with a peak in February of 22 ± 4 birds (Figure 7.11). The low occupancy level in January 1997 was due to the reservoir being frozen for a prolonged period.

7.6.2 *Feeding success*

Stocking experiment - winter 1995/96

Twenty five dawn until dusk visits were completed in the period January to March 1996, with 85 foraging bouts observed, of which 73 (85.9 %) were successful (Table 7.5). This is the highest foraging success rate recorded in the study, with Holme Pierrepont (Section 4.5.4) recording foraging bout success rates of 57.7 to 74.4 %, the River Trent (Section 5.5.2, 5.6.2, 5.7.2) 25.9 to 52.0 %, and Colwick Park Trout Lake (Section 6.5.4) 18 to 67.0 %. However, the percentage of dives that were successful was low (9.5 %), similar to levels observed on other study sites.

Winter 1996/97

Thirty full cormorant site visits were completed on Grimsargh number 3 during winter 1996/97 with 44 foraging bouts observed, of which 32 (74.0 %) were successful (Table 7.5). This level is slightly below the success rate of 1995/96, but was still high compared with the other study sites. Similar to 1995/96, the percentage of dives that were successful was low (11.1 %).

Table 7.5 Feeding success of cormorants in Grimsargh number 3 Reservoir 1995 to 1997.

	1995/96	1996/97
Foraging bouts observed	85	44
Foraging bouts successful	73	32
% foraging bouts successful	85.9	74.0
Dives observed	2453	853
Successful	233	92
% dives successful	9.5	11.1

7.7 Impact assessment of cormorant predation at Grimsargh number 3 Reservoir

7.7.1 *Species composition of fisheries surveys and cormorant predation*

Observation of cormorant feeding behaviour did not allow the identification of fish predated on to species level and fish were classified as cyprinids and perch. Accordingly, all fisheries data collected from seine netting surveys were reclassified as cyprinids, perch, pike and eels to allow direct comparison.

Cormorants actively selected cyprinid species for consumption on Grimsargh No. 3 reservoir in winter 1995/96 and 1996/97. All fish consumed by cormorants in 1995/96 were cyprinids and in 1996/97, 99 % were cyprinids (Figure 7.12). This was reflected in the seine net surveys which were dominated by cyprinids. Perch comprised 6 to 16 % of catches, with pike and eels comprising around 1 % of seine netting catches. These species were not eaten by cormorants.

7.7.2 *Length frequency distribution*

The sizes of cyprinids taken by cormorants were mainly in the size range 50 to 150 mm (Figure 7.13). This was also the main size range of cyprinids found in the seine netting catches. Hence, the cormorants selected prey items from the main size range of cyprinids in the fishery.

Cormorants were also observed to take cyprinids in the size range 150 to 400 mm in 1995/96 and 150 to 250 mm in 1996/97. The seine netting catches (Figure 7.13) revealed few cyprinids over 150 mm in the reservoir and none present between 200 and 400 mm. The last sample from the reservoir was taken when all the fish were removed and all the water drained, so no fish were missed in this process. Thus, an error may have occurred in the estimation of fish size taken by cormorants.

7.7.3 *Cohort reconstruction with and without cormorant predation.*

Cohort reconstruction and MCS calculated data revealed the impact of cormorant predation on the roach, common bream and perch populations (Figure 7.14 to 7.16). Due to the large numbers of roach, common bream and perch present in the reservoir of lengths below 150 mm, the impact of cormorant predated losses was very low.

7.7.4 *Biomass*

The impact of cormorant predation on the biomass of roach, common bream and perch cohorts was also very low, due to the large biomass of these species present compared with that removed by cormorants (Figure 7.17 to 7.19).

7.8 Discussion

7.8.1 *Cormorant occupancy*

Cormorant occupancy was variable on the reservoir over the two winters of study, with generally low numbers of birds observed on the reservoir. Prior to the stocking of roach in 1995/96, occupancy was very low (2), which was observed to increase 5 days post-stocking to a maximum level of 13 birds. However, due to the high numbers of small cyprinid species that were already in the reservoir, it is unlikely that it was the effect of the stocking alone that increased cormorant occupancy of the site. It may have been due to changes in feeding habitat on other feeding sites in the area, such as flooding of river sites forcing the birds to feed elsewhere.

7.8.2 *Cormorant feeding success*

Cormorant feeding success, measured as the percentage of successful feeding bouts, was high on Grimsargh number 3 reservoir, and exceeded levels found on Holme Pierrepont (Chapter 4). This may have been due to the uniform habitat of the reservoir, (for example, the small depth variation), its shallow depth (2.5 m), the dense number of small cyprinids and the lack of suitable refuge habitats for these fish in the reservoir. The high cormorant feeding success on Holme Pierrepont was also related to the fish density, water depth and lack of cormorant refuge areas offering protection (Section 4.7).

7.8.3 *Fish populations*

The fisheries surveys showed the standing crop of fish in the reservoir was 6.11 to 6.68 g m^2 over the period of study, and was shown to be composed of very high numbers of slow growing cyprinids. A number of large common bream (above 450 mm) were present in the reservoir. However, with a lack of common bream between 150 and 450 mm observed in the surveys, which included total fish removal after draining, it would suggest these larger fish were stocked historically.

The growth rates of all of the fish species were slow which indicated high abundance/standing crop of fish and low food availability. The low food availability is likely to be a reflection of the low nutrient loading in the reservoir due to its use as a public water supply.

7.8.4 *Cormorant predation impact*

Cormorants were shown to remove a low proportion of the fish standing crop each year. The majority of fish removed by cormorants were cyprinids, with few perch taken, reflecting the species composition in the reservoir.

Cormorant predation on the reservoir may have been expected to be high as there were abundant stocks of small cyprinid species in the reservoir and cormorant feeding success rates were high. However, the predation was shown to be low and had little impact on the fish cohorts. The low predation levels may have been a reflection of reduced cormorant numbers in the region (Section 8.6.1) and the fact that the reservoir is located in an urbanised area. Cormorants are shy birds (T. Holden pers. comm.). With a major road running close to the reservoir and a number of residential properties close to the reservoir, the birds may have felt insecure whilst feeding and consequently tended to feed elsewhere.

This contrasts with the anecdotal evidence from the controlling angling society, which suggested that cormorant predation had been high on the reservoir in the past and had removed a significant proportion of the fish stock, resulting in decreased angler catches. Although this may have been the case, with the complete absence of fish between 150 and 400 mm, it would appear an additional problem must have occurred in the reservoir. It is unlikely that the cormorants were so efficient in feeding that they were able to remove entire year classes of fish. At present, it would appear there is a production bottleneck in the fishery. Spawning substrate is good and recruitment has been shown to be high in recent years due to the high numbers of small fish up to 150 mm. Food supply may be limited, increasing intra- and inter-specific competition between the fish and resulting in the slow growing populations.

7.8.5 *Management*

Management policies are required to reduce the high abundance of small fish, if the growth rates of the fish species are to be improved. There are a number of measures which could be undertaken.

- Altering the fish community structure by the addition of predators such as pike. These have been historically removed from the reservoir (pers. obs.) and this may have contributed to the high abundance of slow growing fish.

- Cropping fish of below 150 mm every two to four years. This would reduce the amount of inter- and intra-specific competition in the reservoir and allow the remaining fish to attain better growth. Additionally, the cropped fish could be sold at profit as stock fish for other fisheries.

- Habitat manipulation could be carried out to increase the productivity of the reservoir. The present slow growth of the cyprinid populations may be due to a poor food supply in the reservoir. Increasing the nutrient input into the lake, either naturally by adding large amounts of animal manure, or artificially by the addition of chemical fertilisers, would increase the abundance of zooplankton, phytoplankton and macro-invertebrates in the reservoir, increasing fish food supply and decreasing competition.

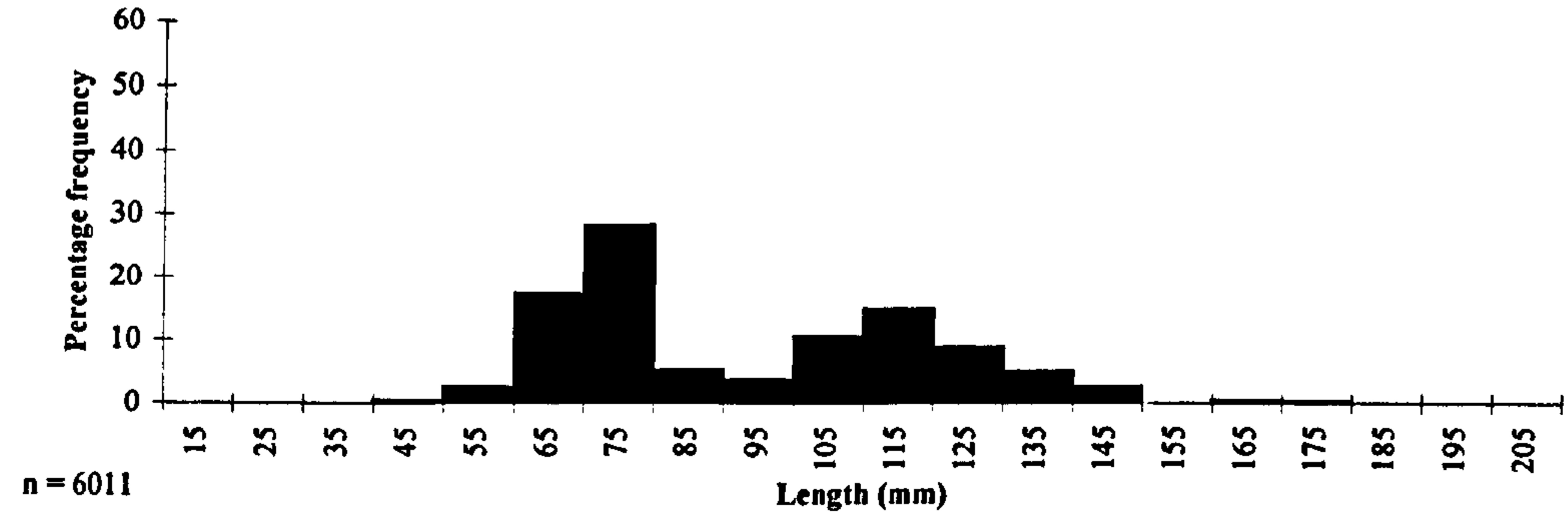
- Stocking of a large number of bigger fish (common bream, tench or carp) would ensure the anglers have an adequate number of exploitative fish in the fishery without having to rely on natural productivity. As this probably would not increase the survival and growth of the small cyprinids, this can be considered a short term measure aiming to improve angler satisfaction.

Any policy which is implemented will have to be consented by North West Water, for the reservoir is used for water supply.

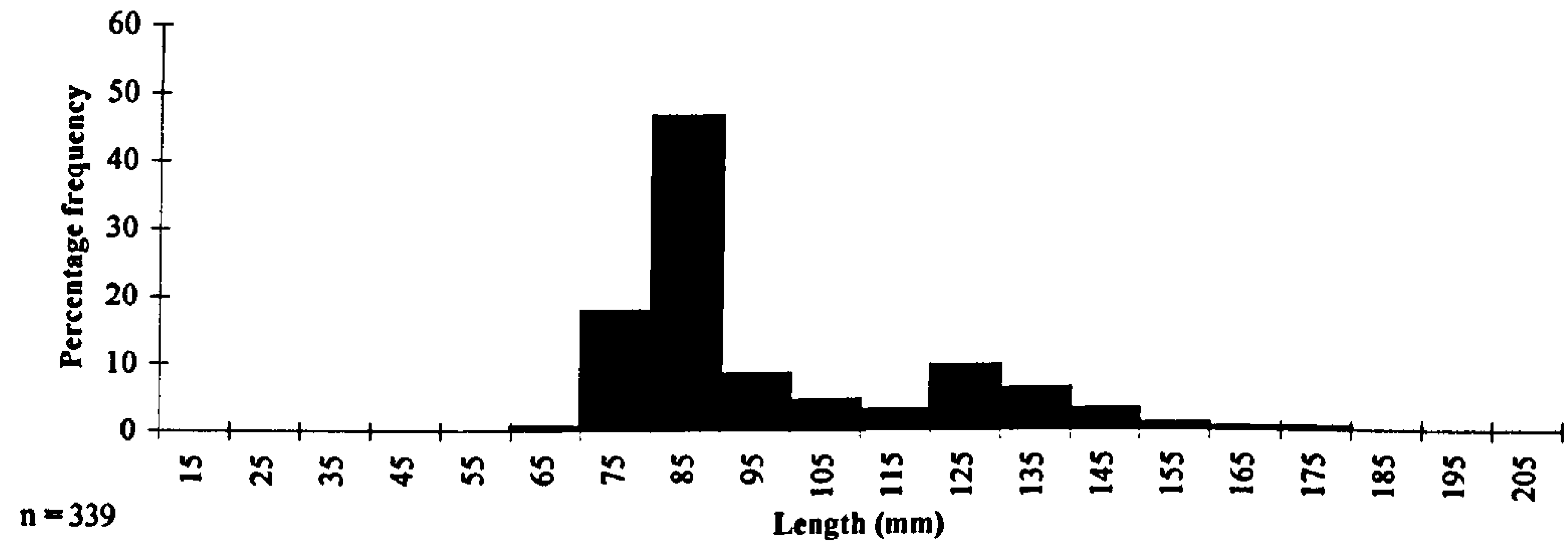
Plate 7.1 Casting the seine net at Grimsargh number 3 Reservoir, as viewed from the east bank.

Plate 7.2 Grimsargh number 3 Reservoir, viewed from the south bank. The east bank of Reservoir number 2 can be seen.

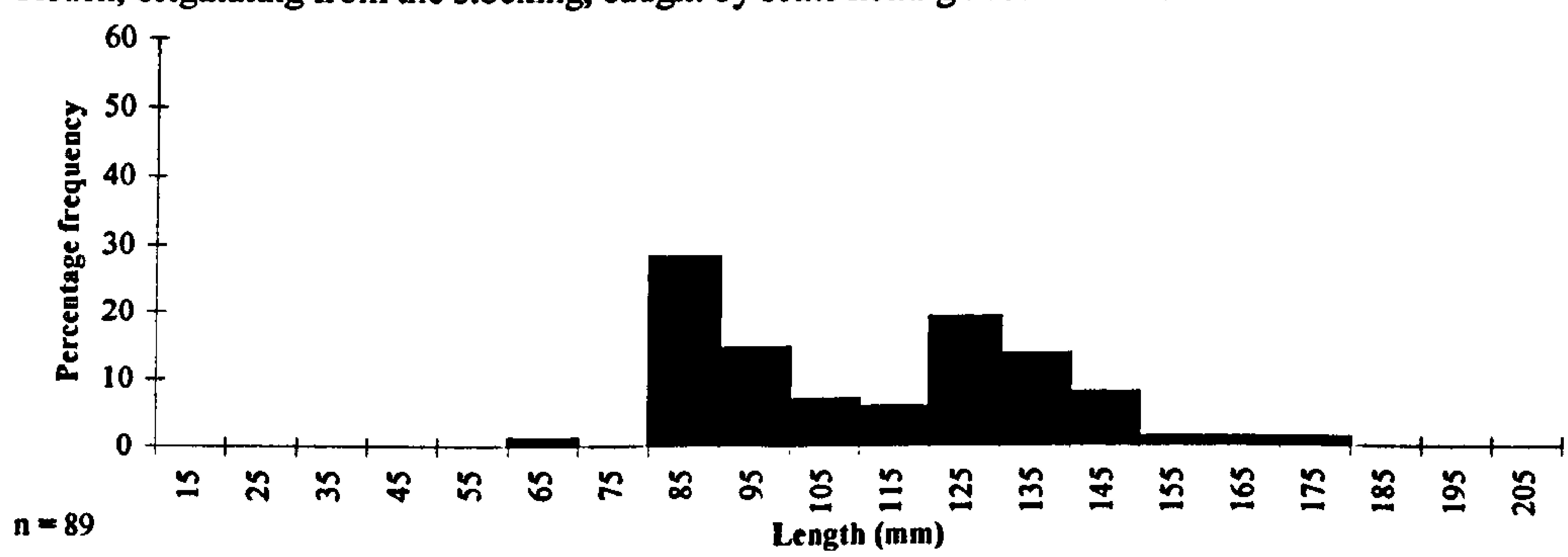
Size distribution of the fish stocked in January 1996.



Total number of roach caught by seine netting in June 1996.



Roach, originating from the stocking, caught by seine netting in June 1996.



Roach, originating from natural recruitment, caught by seine netting in June 1996.

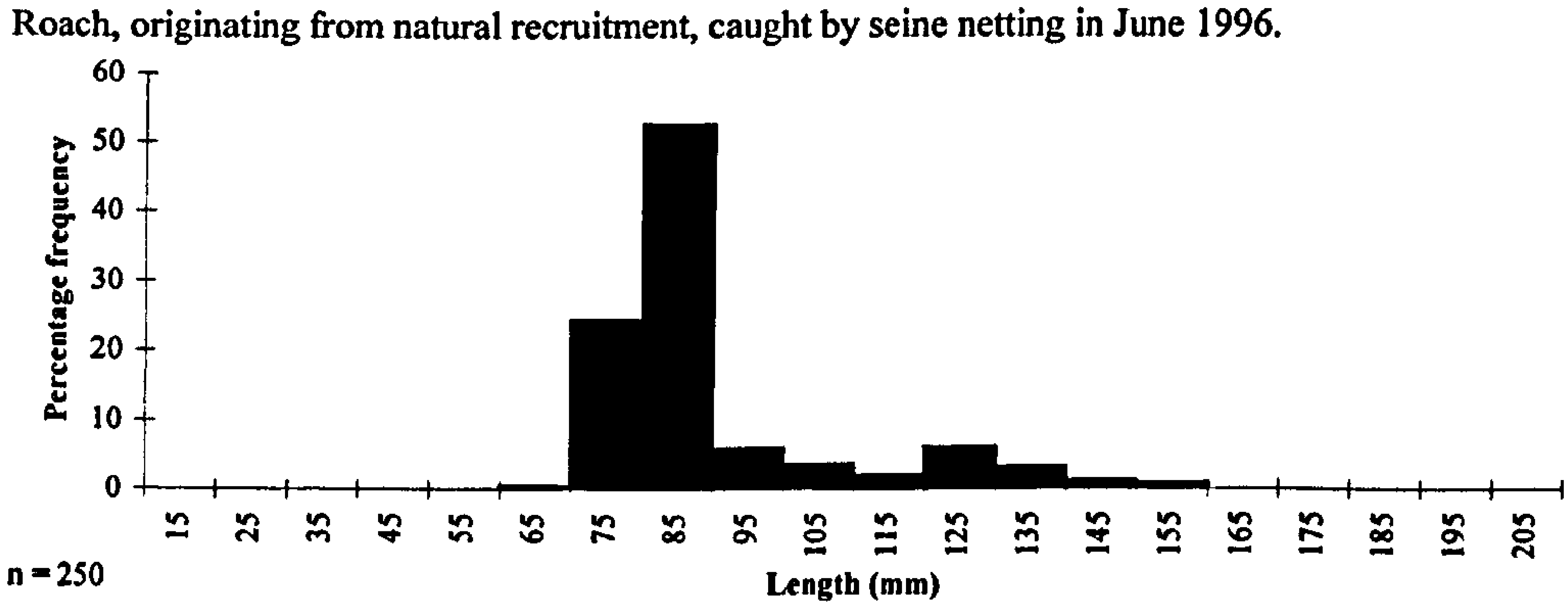


Figure 7.1 Length frequency distribution of roach stocked into Grimsargh number 3 Reservoir in March 1996 and their contribution in the seine net catch in June 1996.

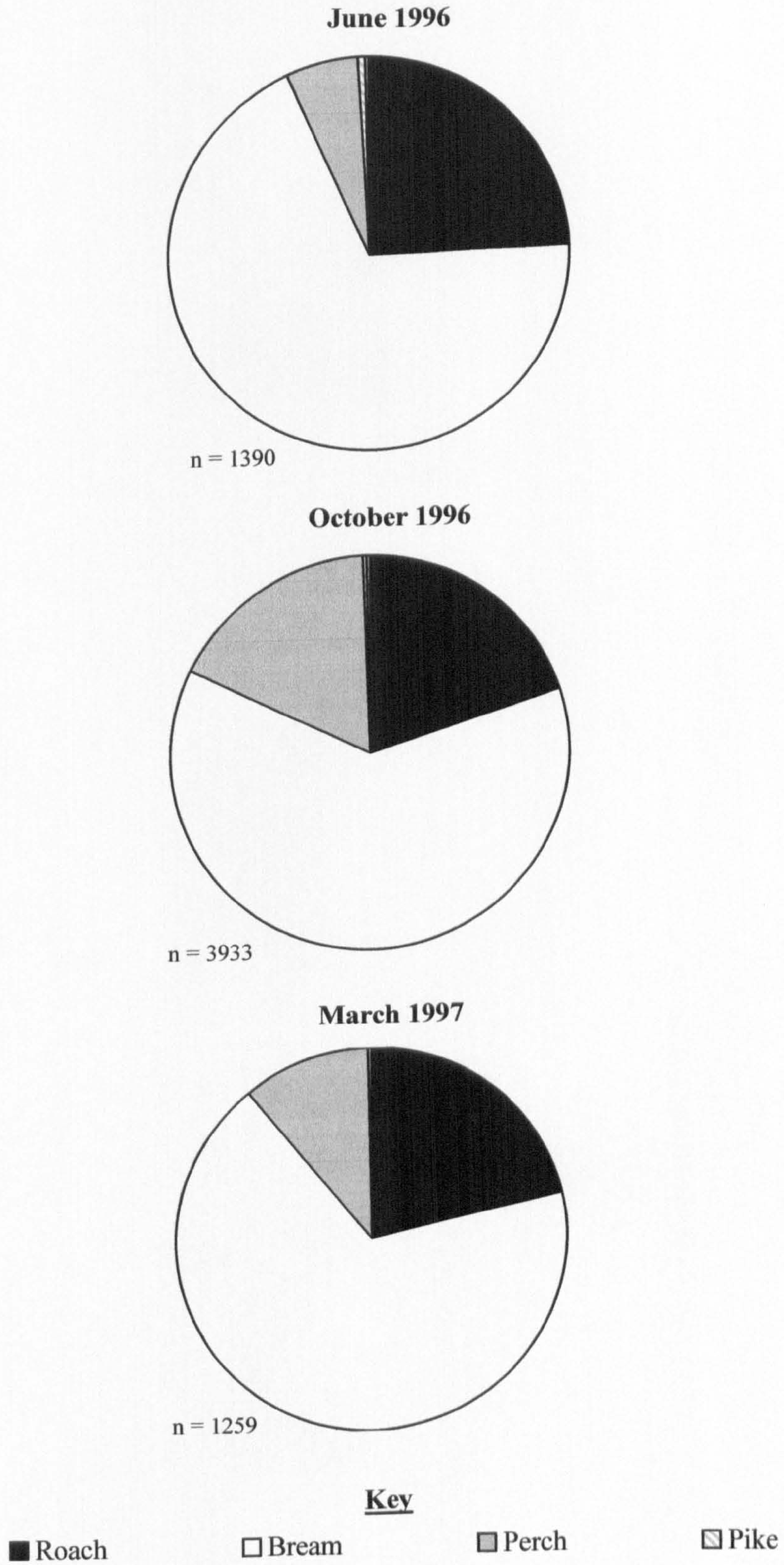


Figure 7.2 Species composition of seine netting surveys at Grimsargh number 3 Reservoir.

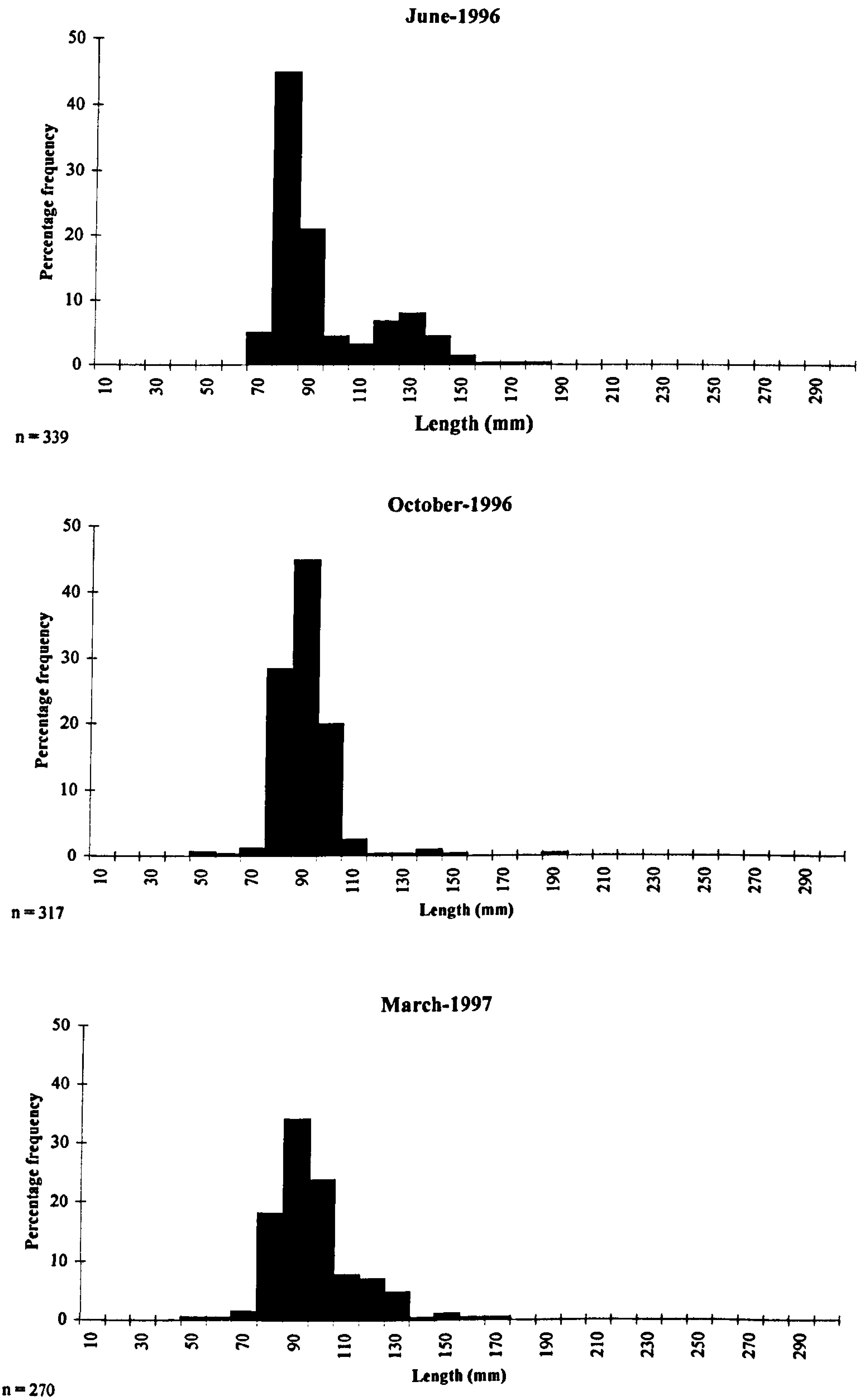


Figure 7.3 Length frequency distribution of roach caught by seine netting at Grimsargh number 3 Reservoir.

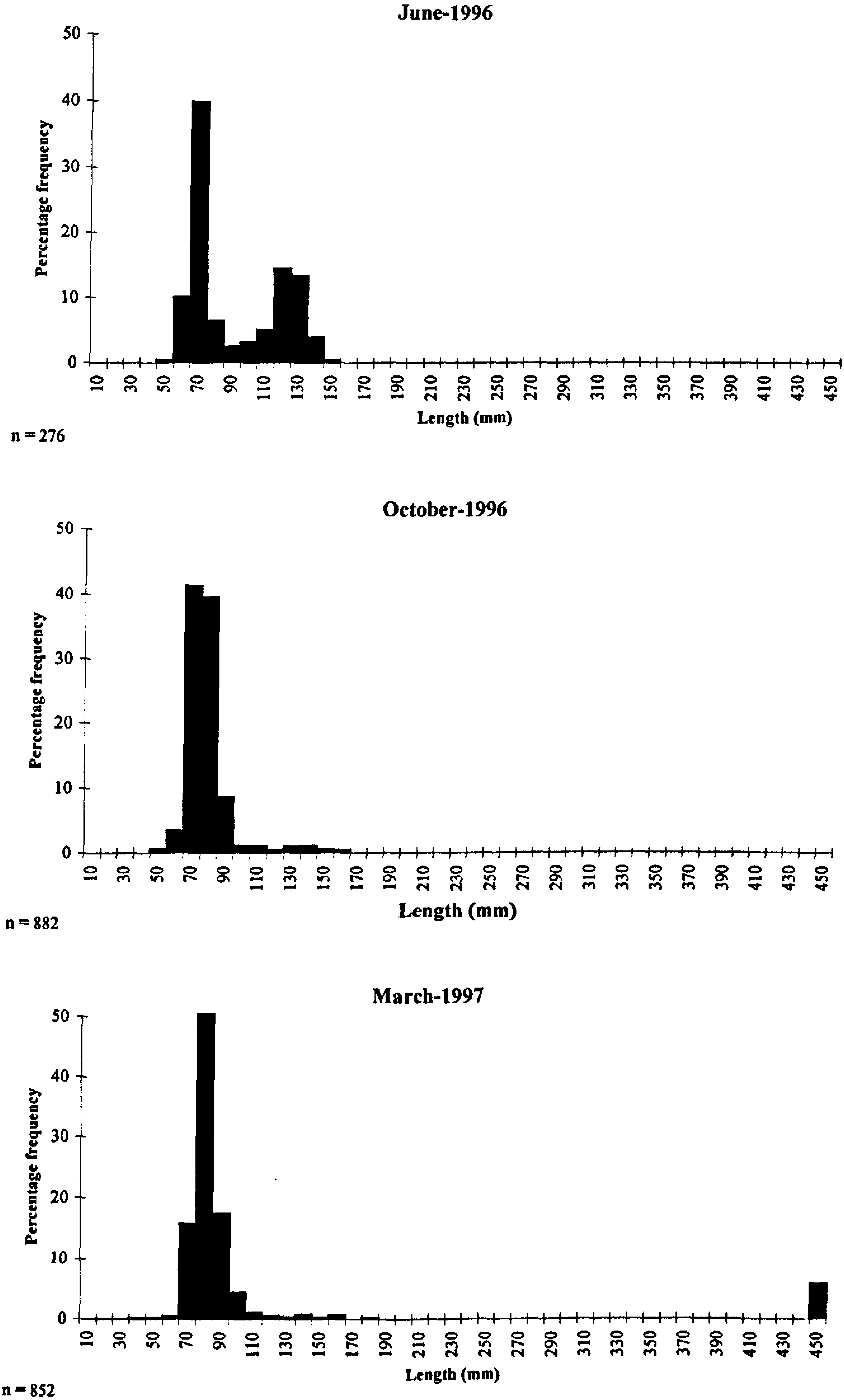


Figure 7.4 Length frequency distribution of common bream caught by seine netting at Grimsargh number 3 Reservoir.

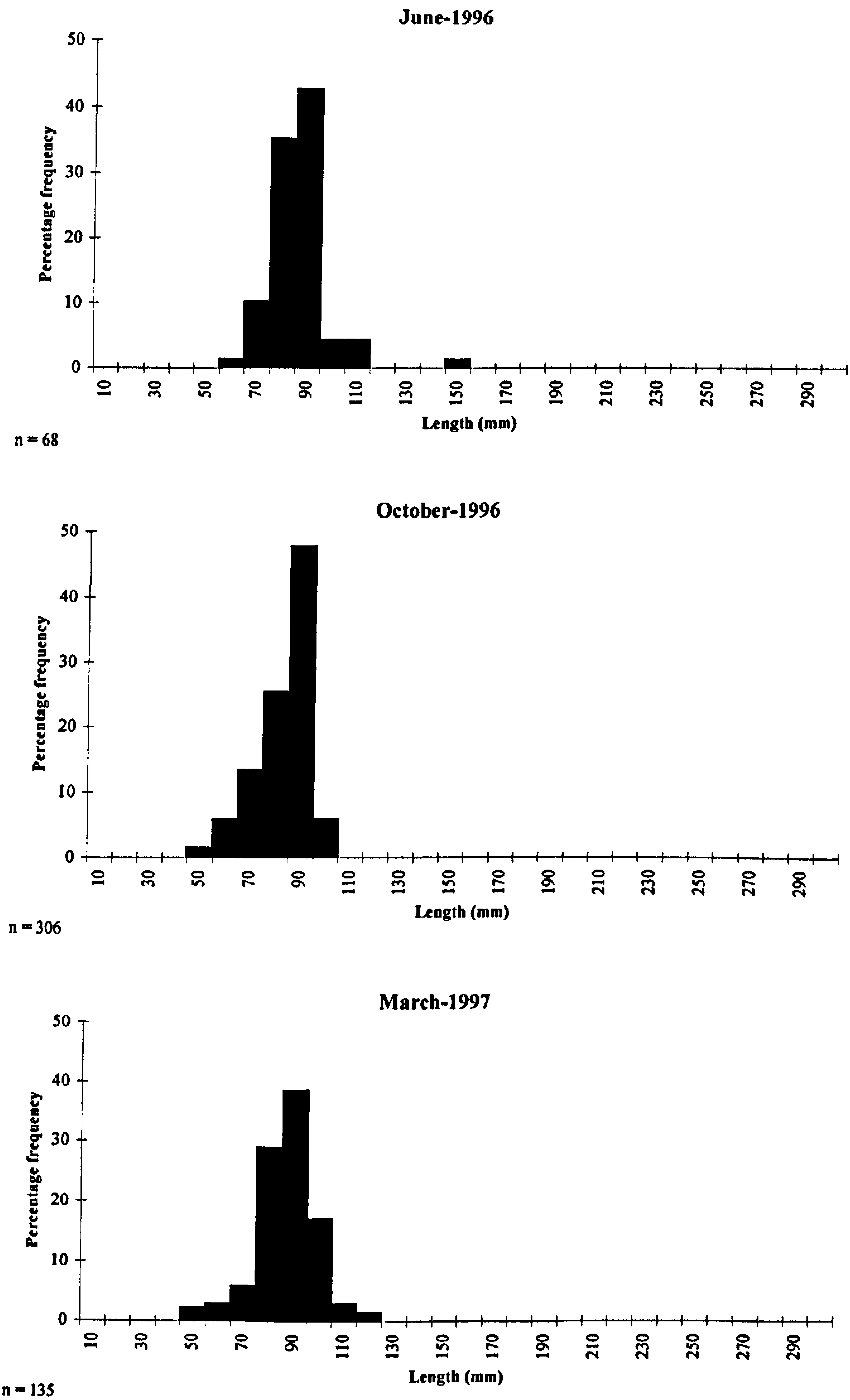


Figure 7.5 Length frequency distribution of perch caught by seine netting at Grimsargh number 3 Reservoir.

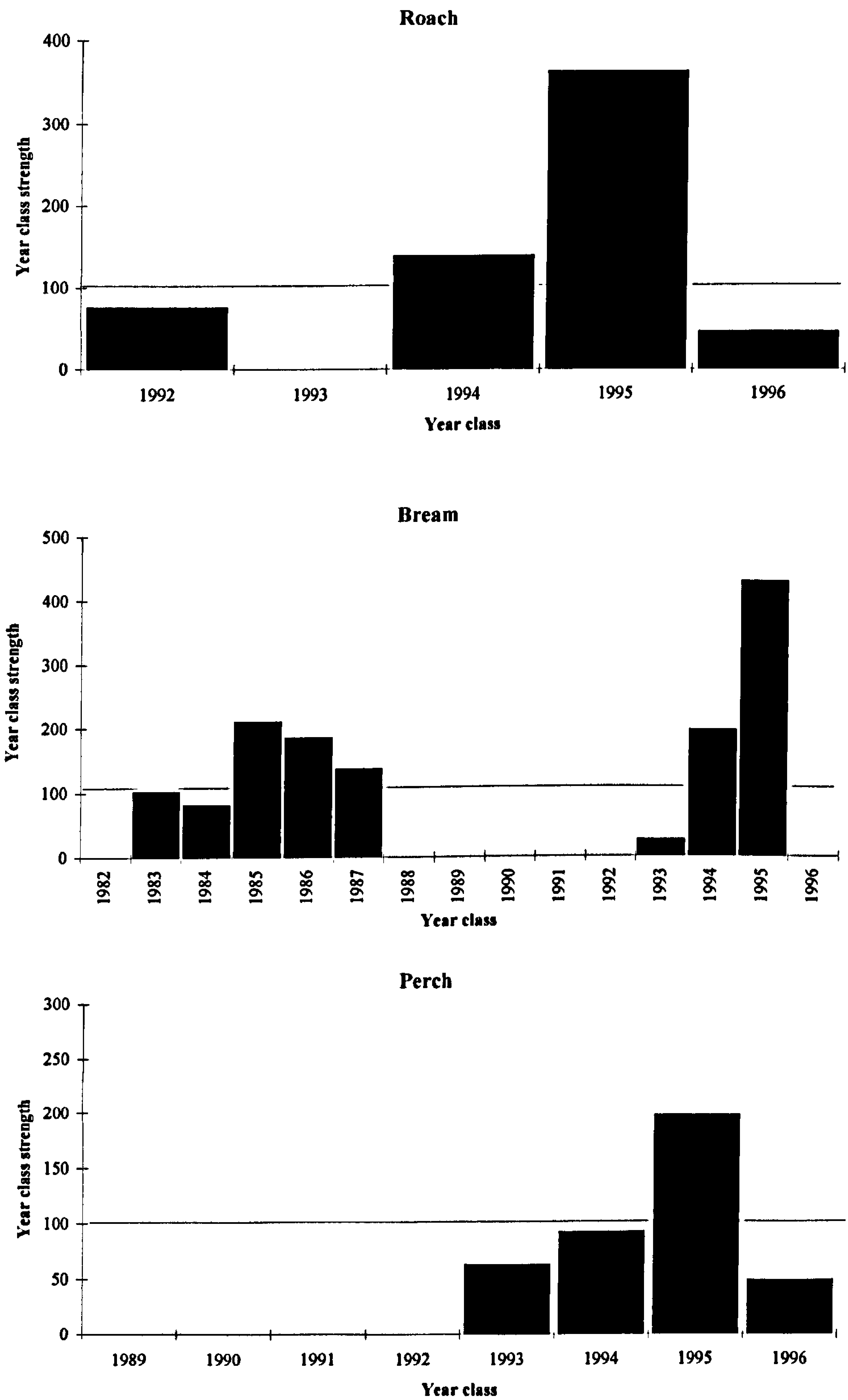


Figure 7.6 Year class strength of roach, common bream and perch at Grimsargh number 3 Reservoir.

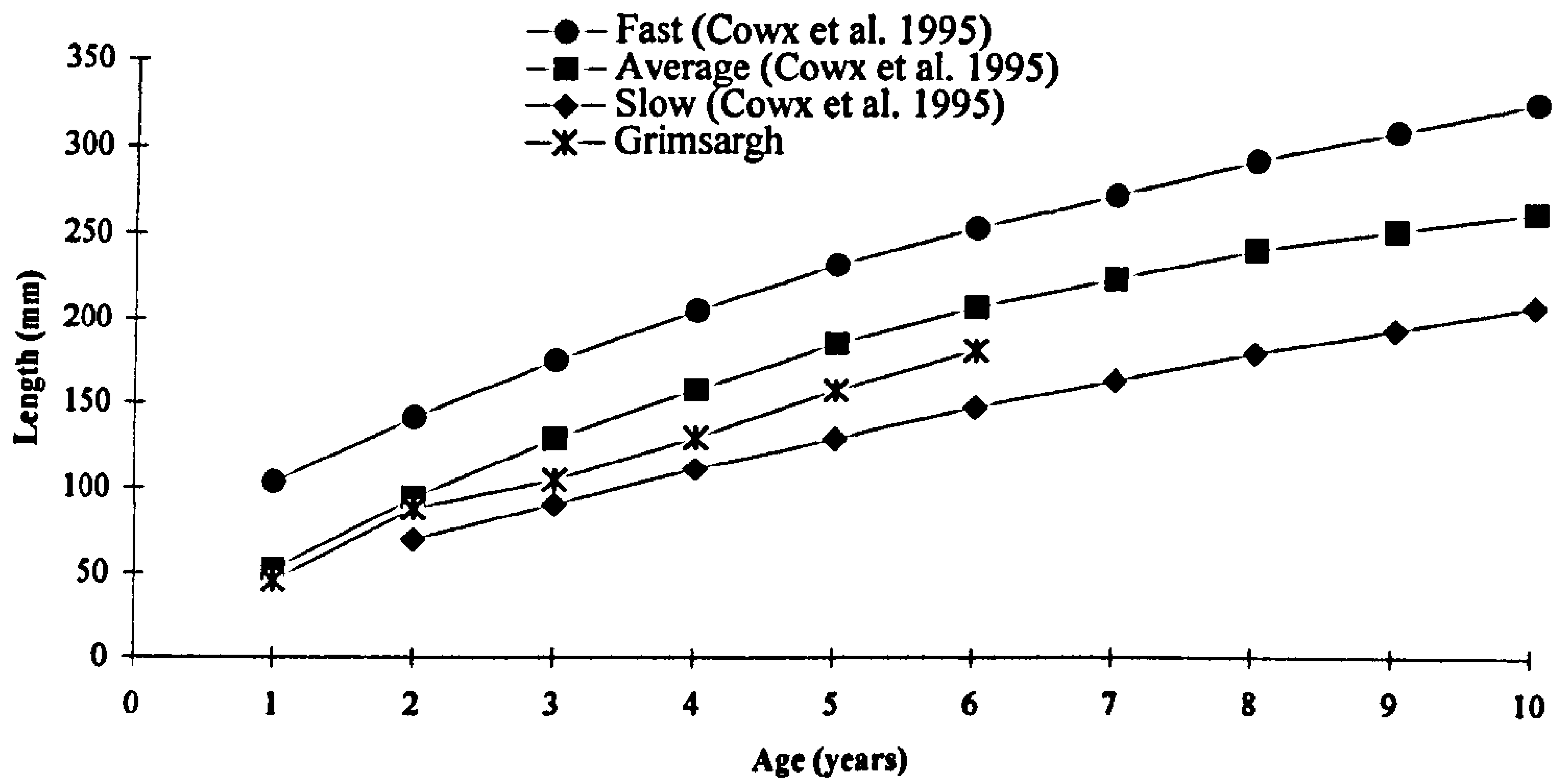


Figure 7.7 Growth of roach in Grimsargh number 3 Reservoir compared with standard growth curves.

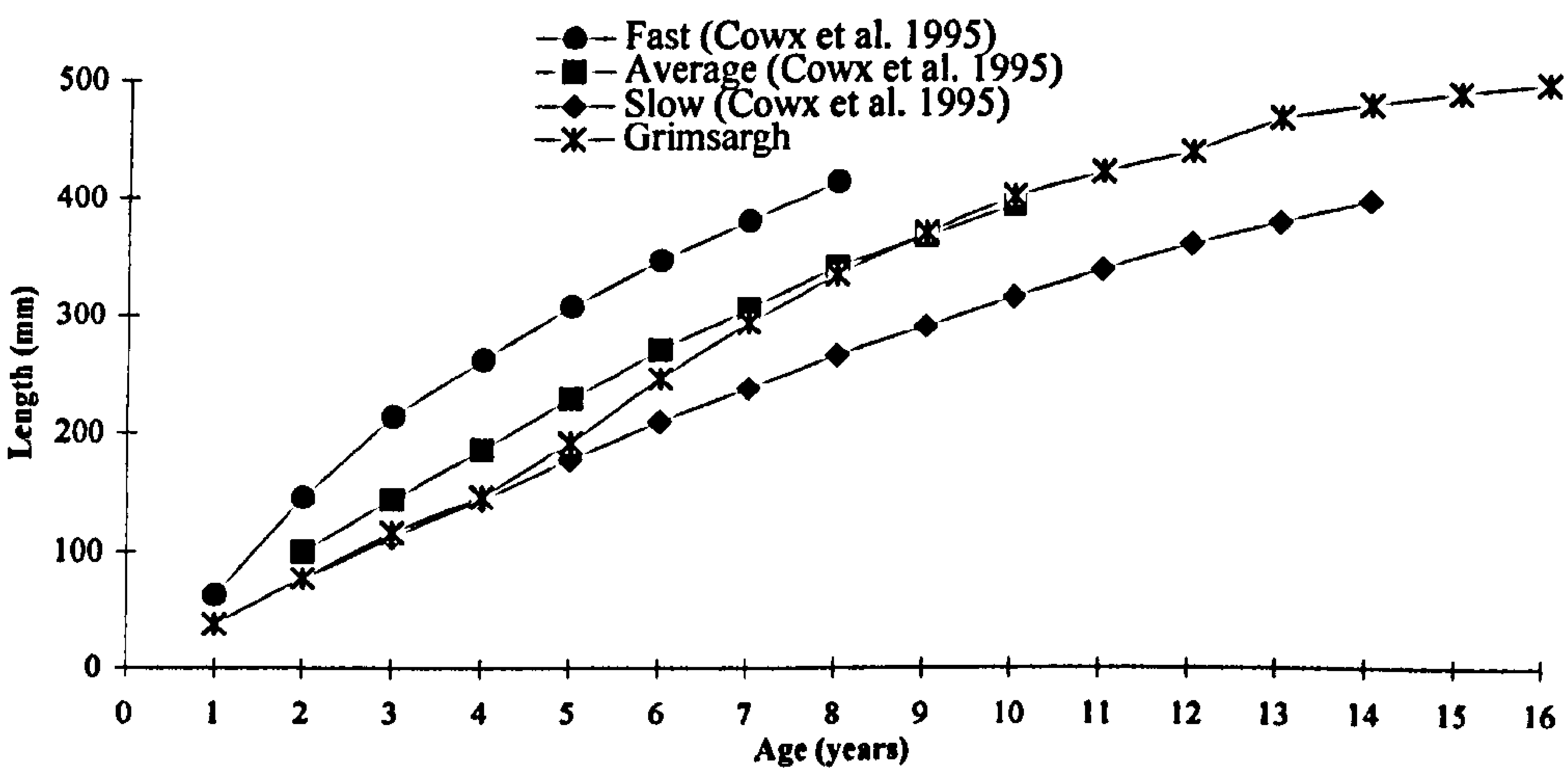


Figure 7.8 Growth of common bream in Grimsargh number 3 Reservoir compared with standard growth curves.

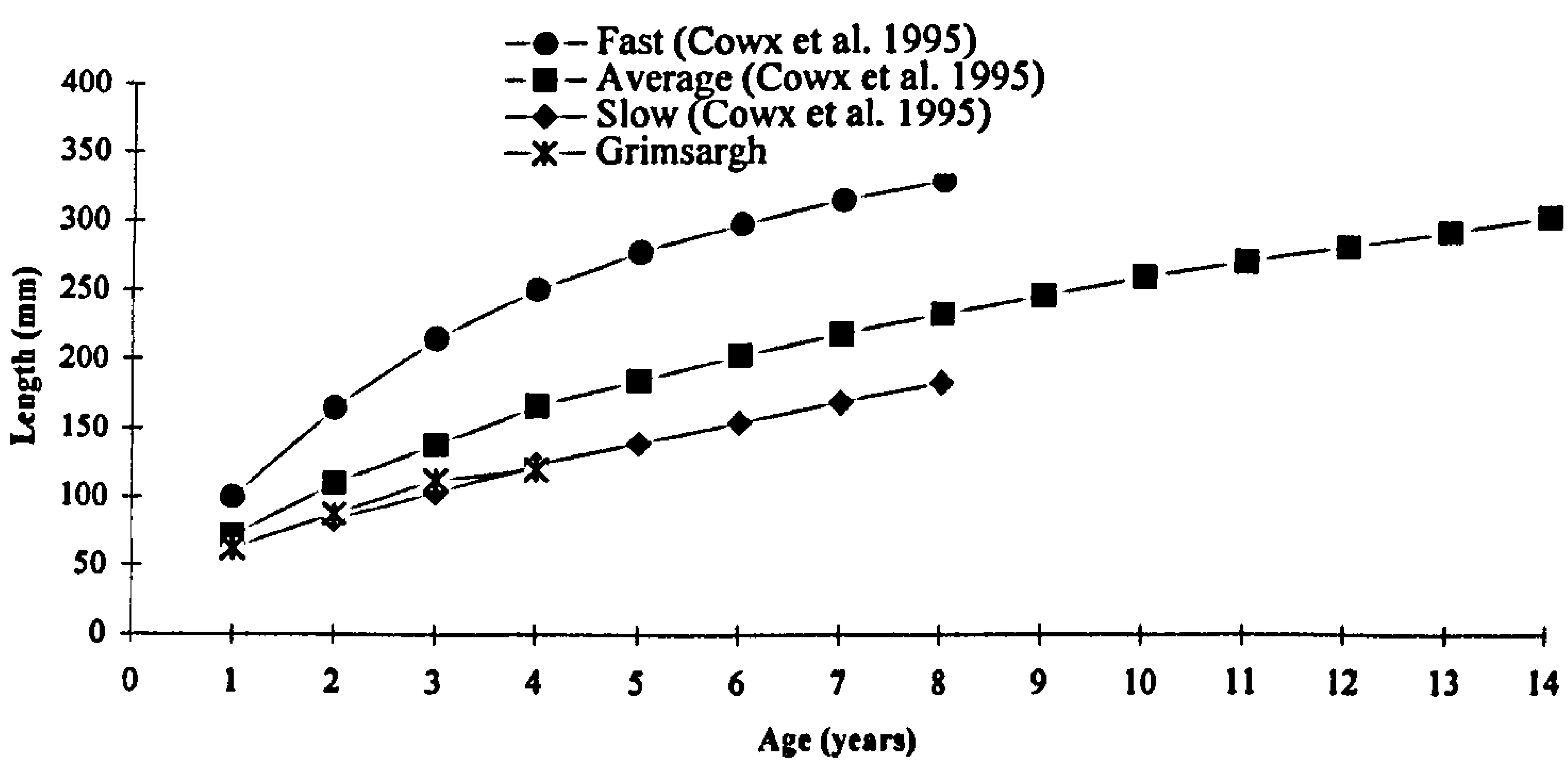
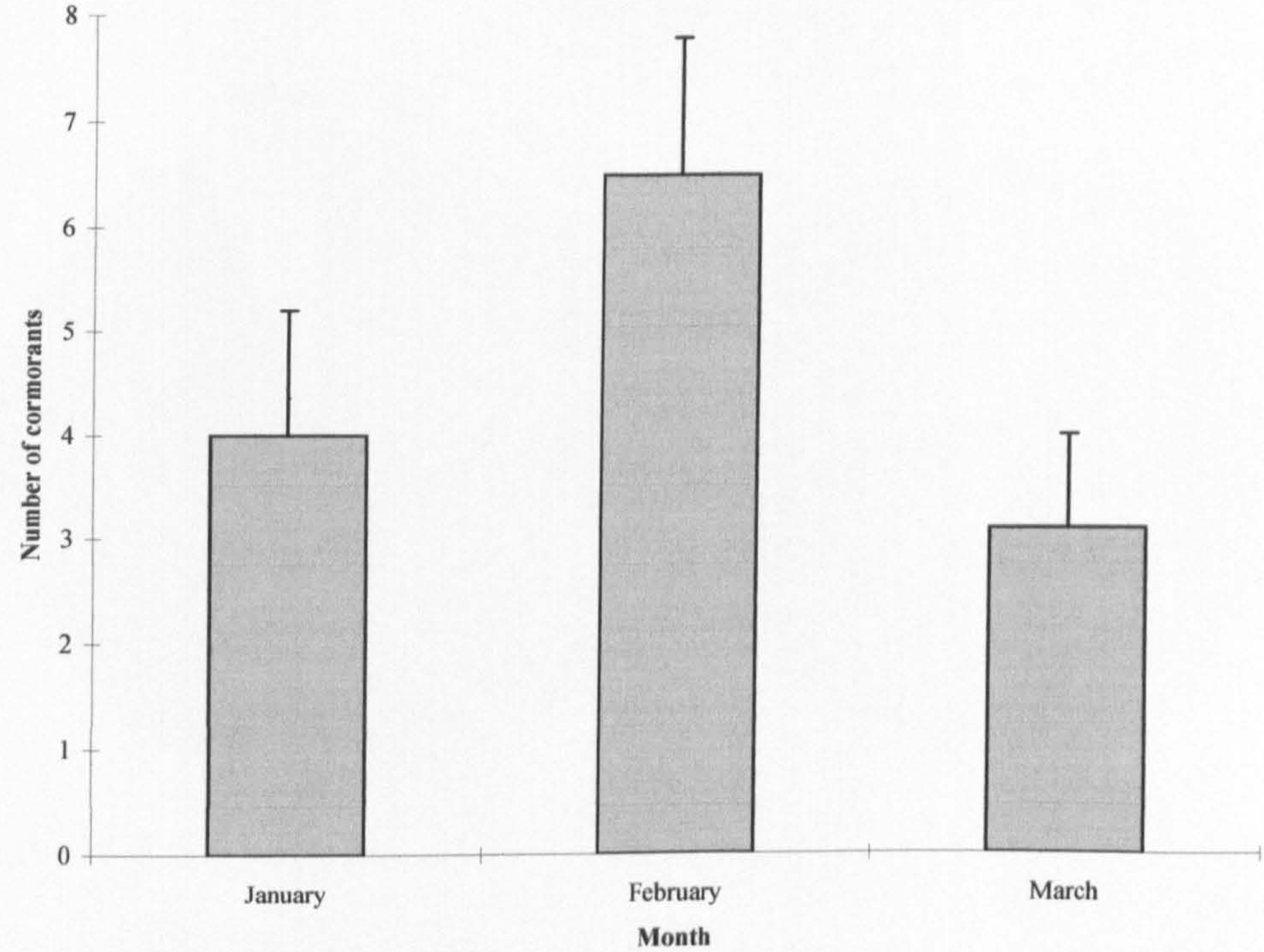


Figure 7.9 Growth of perch in Grimsargh number 3 Reservoir compared with standard growth curves.



Stocking of 6000 roach in January 1996

Figure 7.10 The cormorant occupancy of Grimsargh number 3 reservoir in January to March 1996, in relation to the stocking experiment.

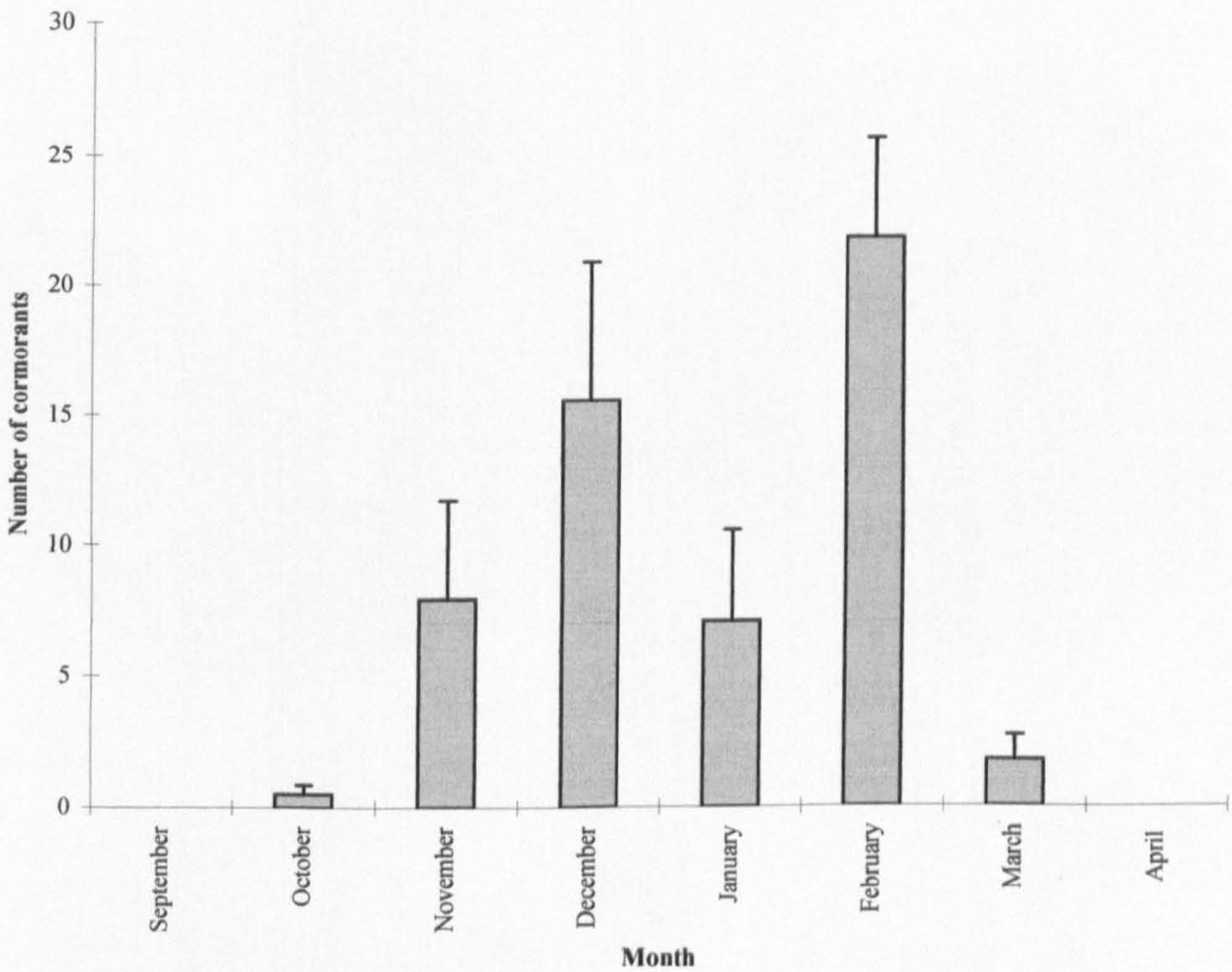


Figure 7.11 Mean daily number of cormorants occupying the Grimsargh reservoirs site between October 1996 and April 1997.

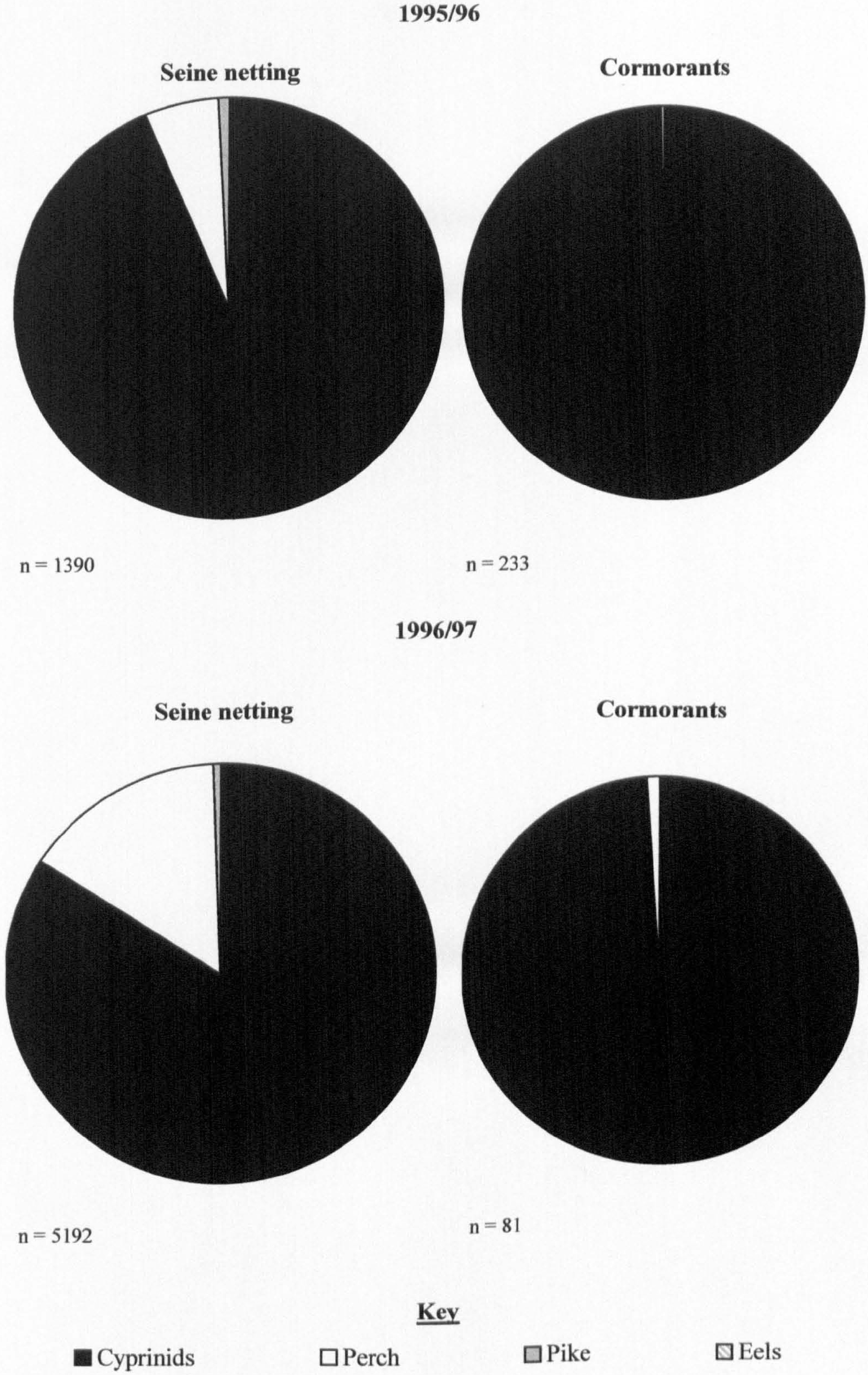
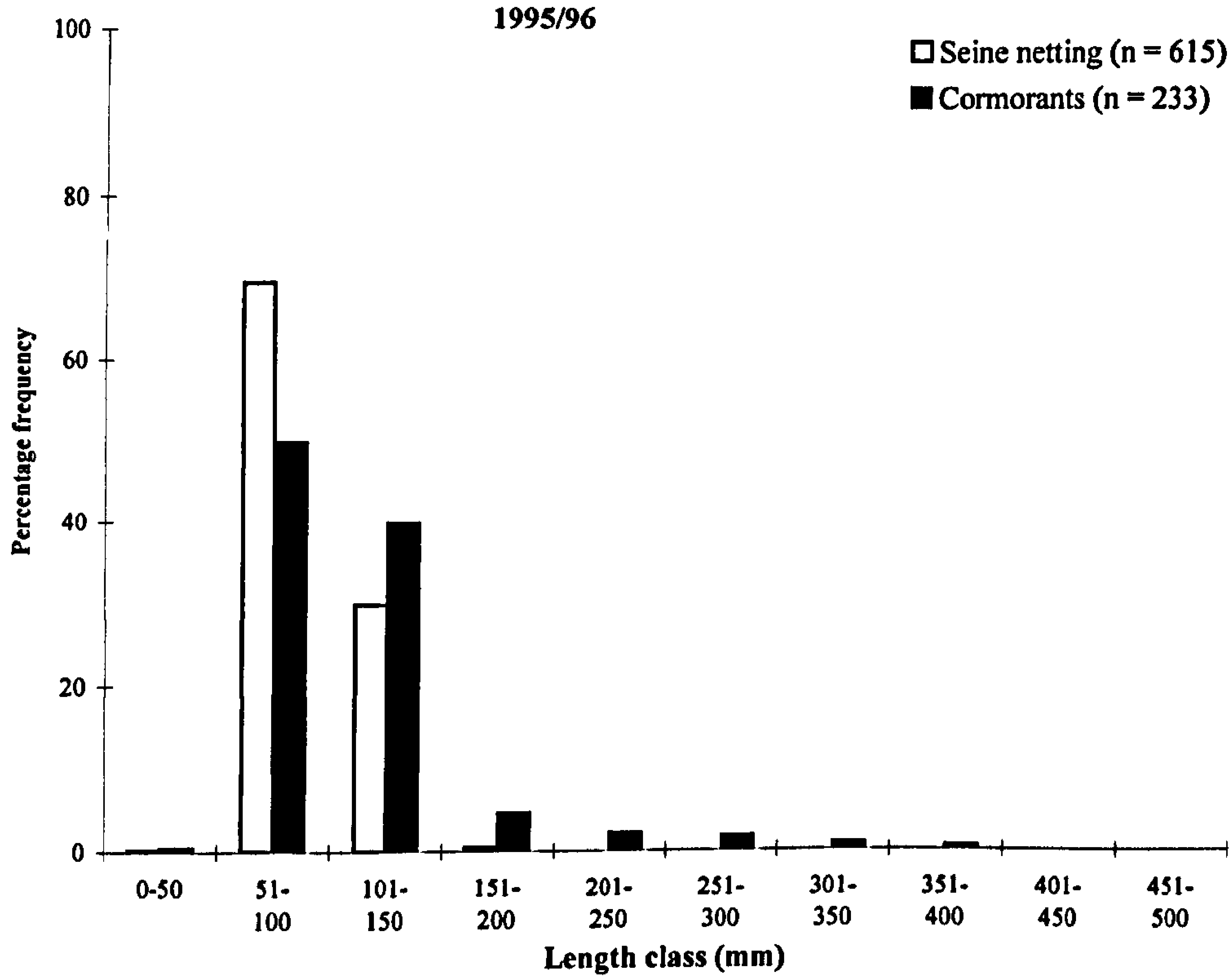


Figure 7.12 Species composition of seine netting surveys and cormorant predation at Grimsargh number 3 Reservoir.



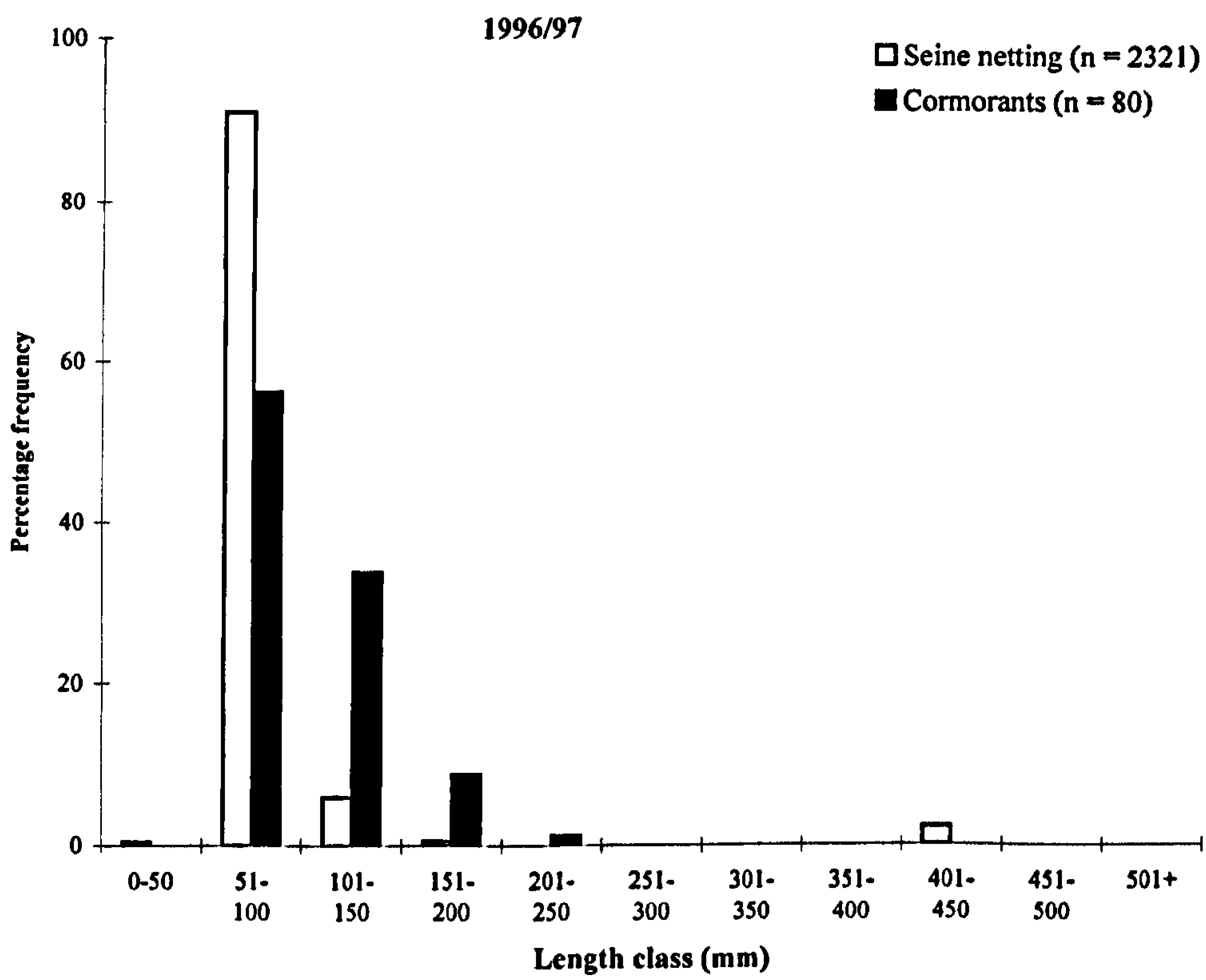


Figure 7.13 Length frequency distribution of cyprinids caught by seine netting and ingested by cormorants at Grimsargh number 3 Reservoir.

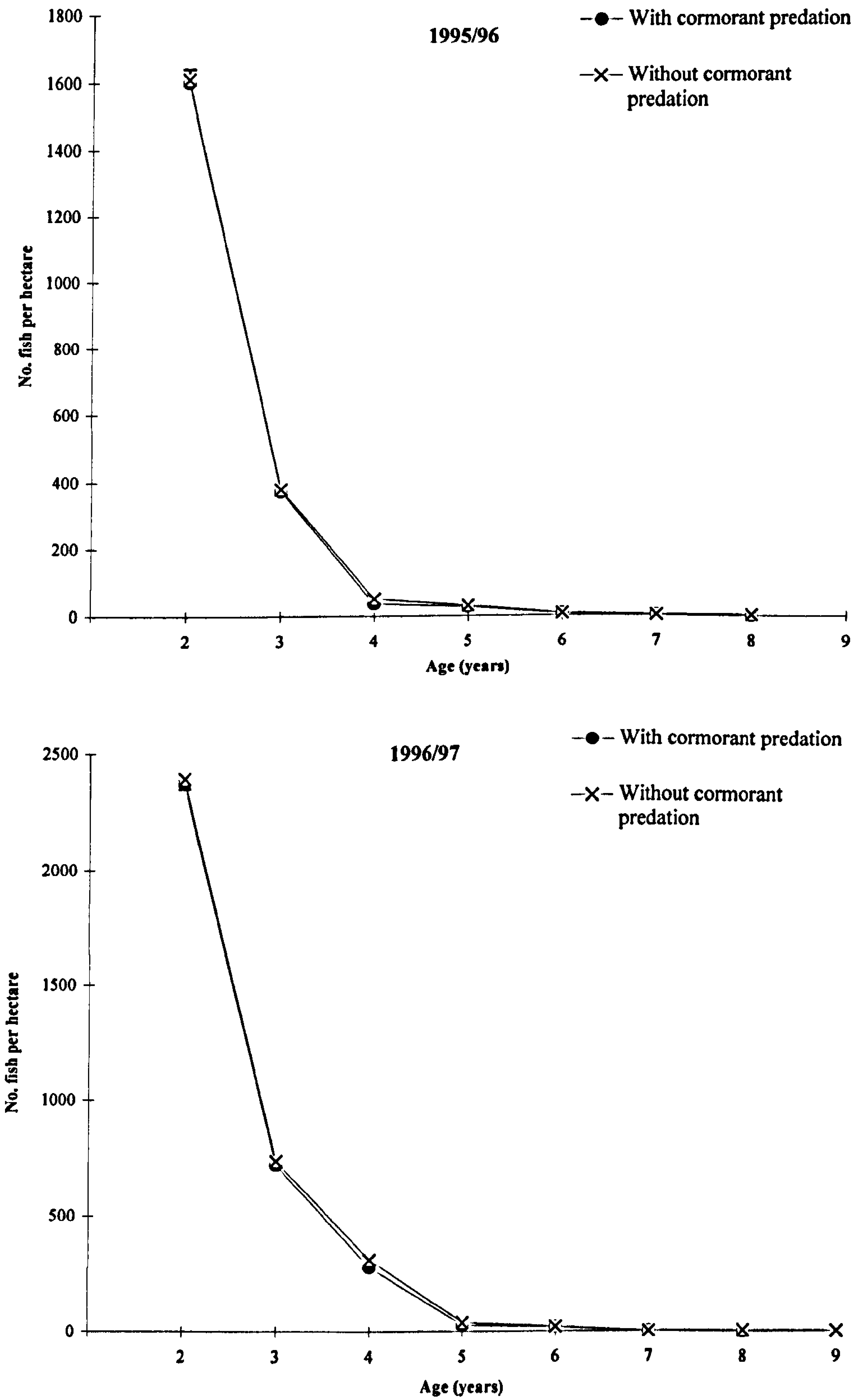


Figure 7.14 Impact of cormorant predation on numbers of roach at Grimsargh number 3 Reservoir.

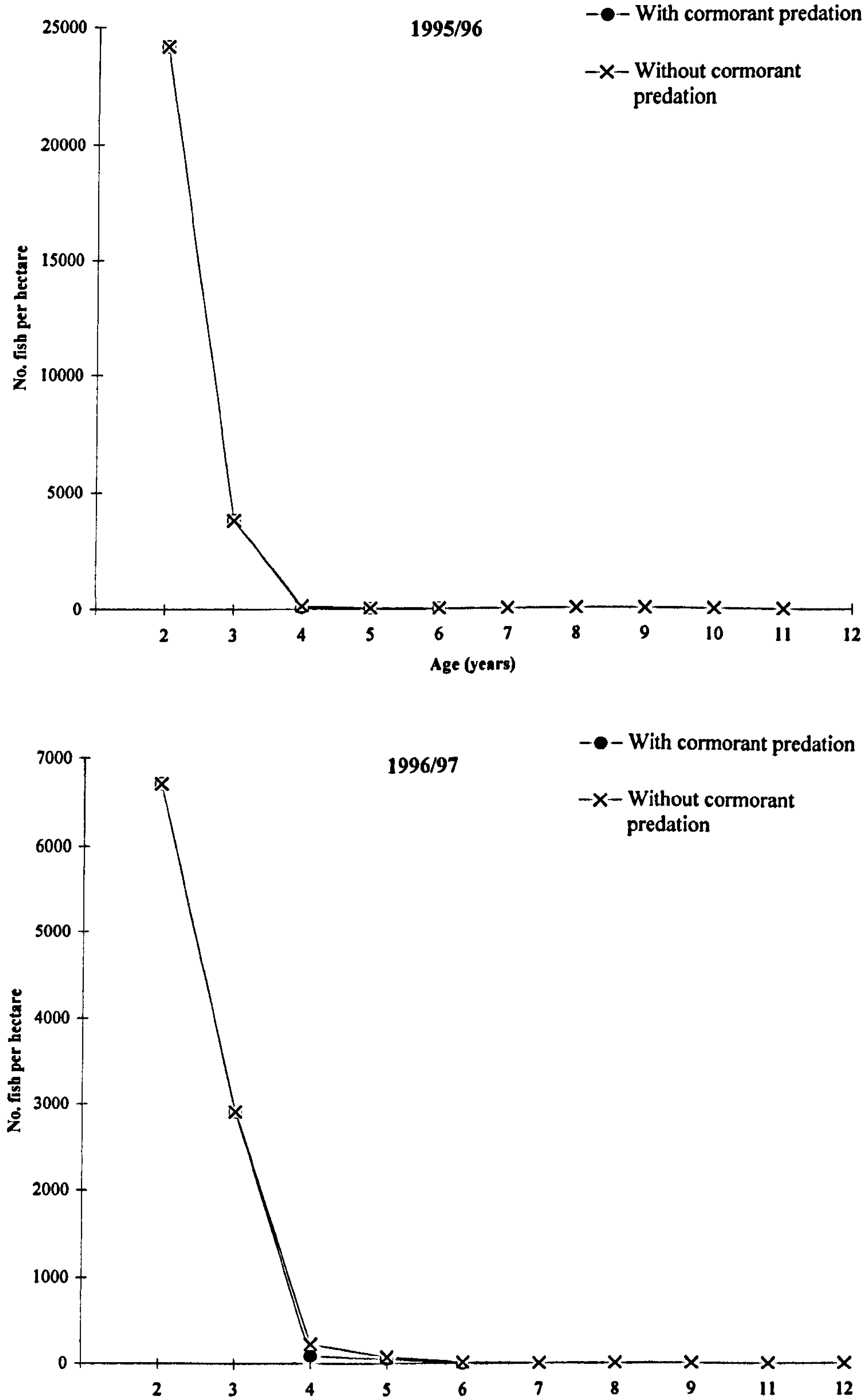
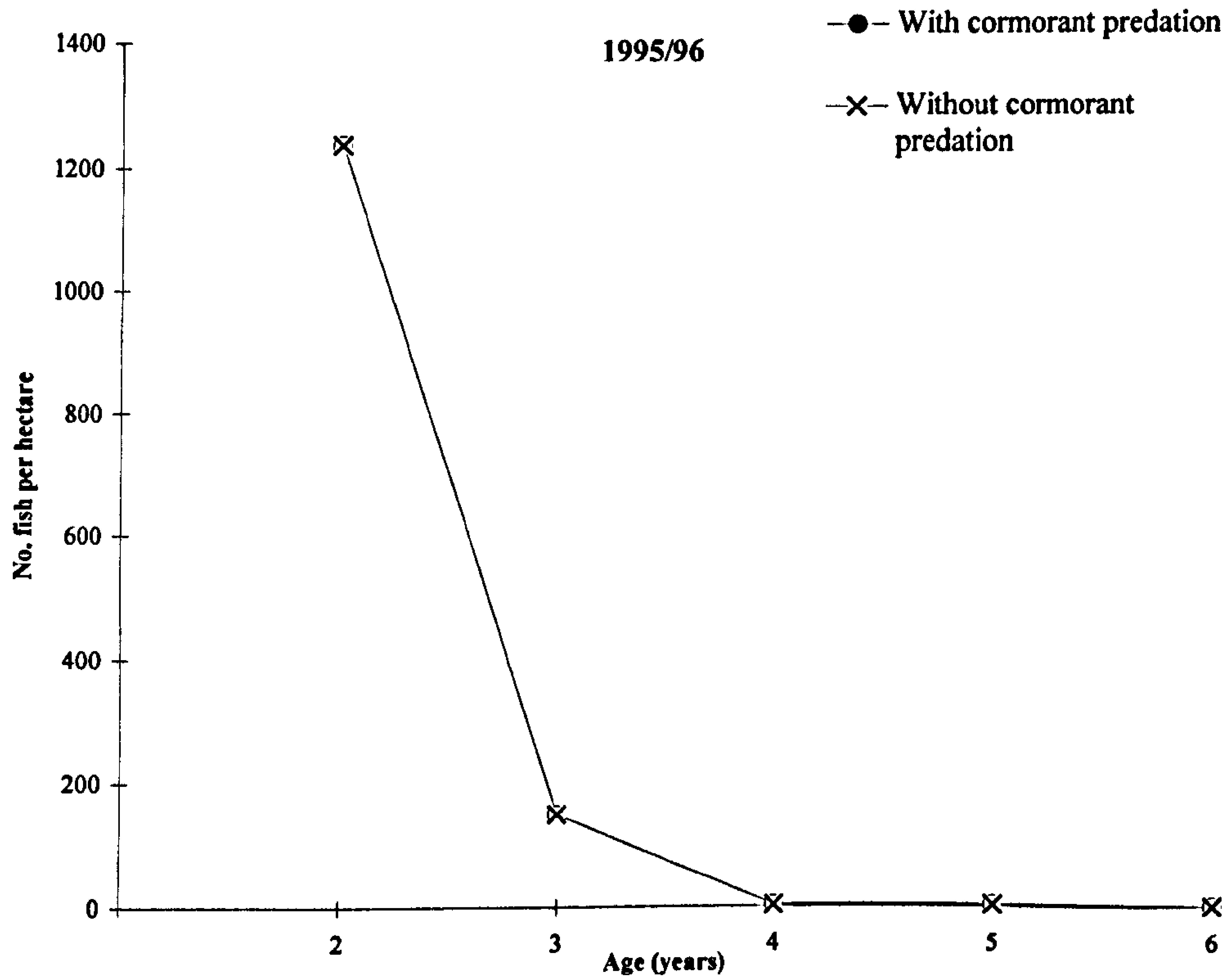


Figure 7.15 Impact of cormorant predation on numbers of common bream at Grimsargh number 3 Reservoir.



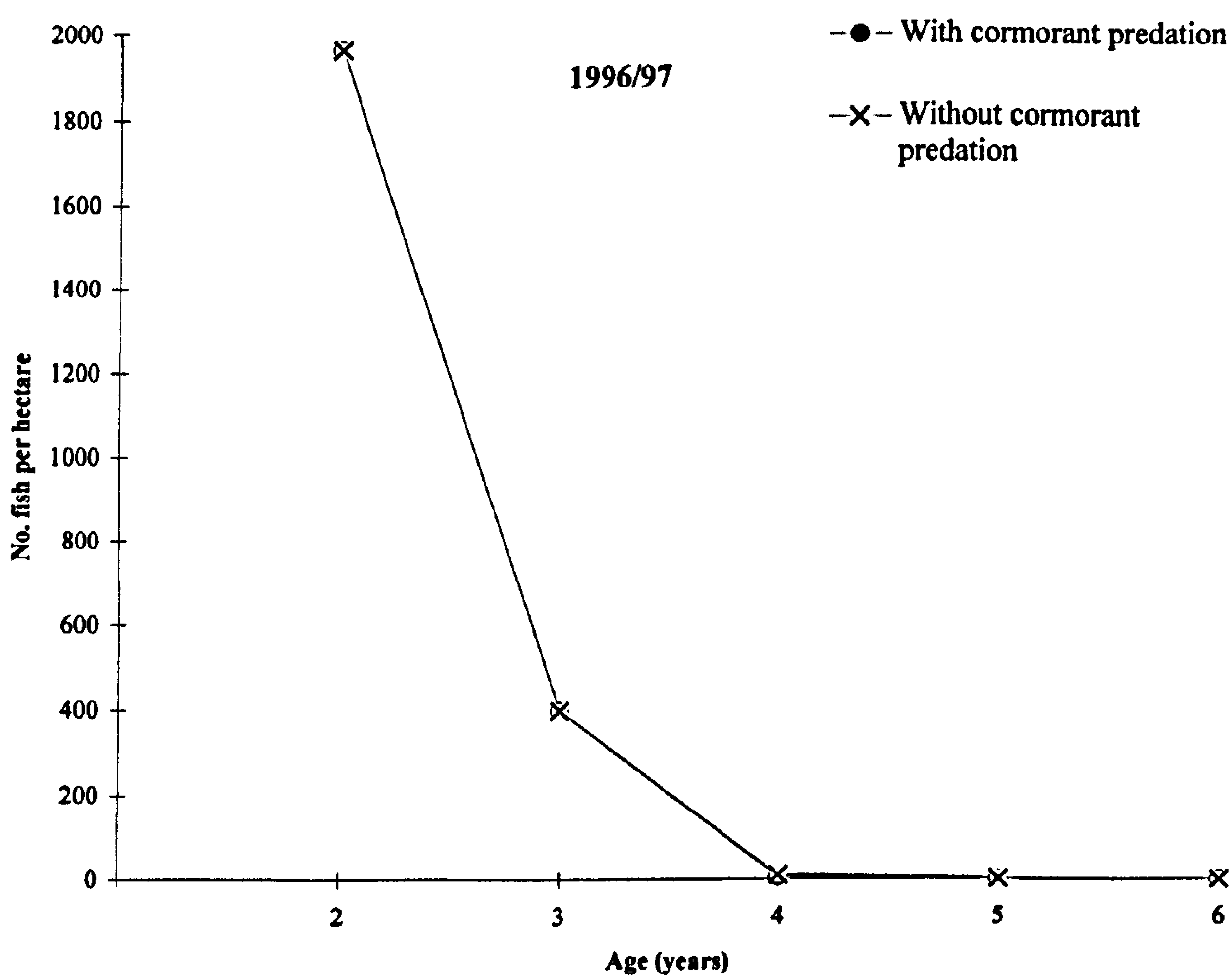


Figure 7.16 Impact of cormorant predation on numbers of perch at Grimsargh number 3 Reservoir.

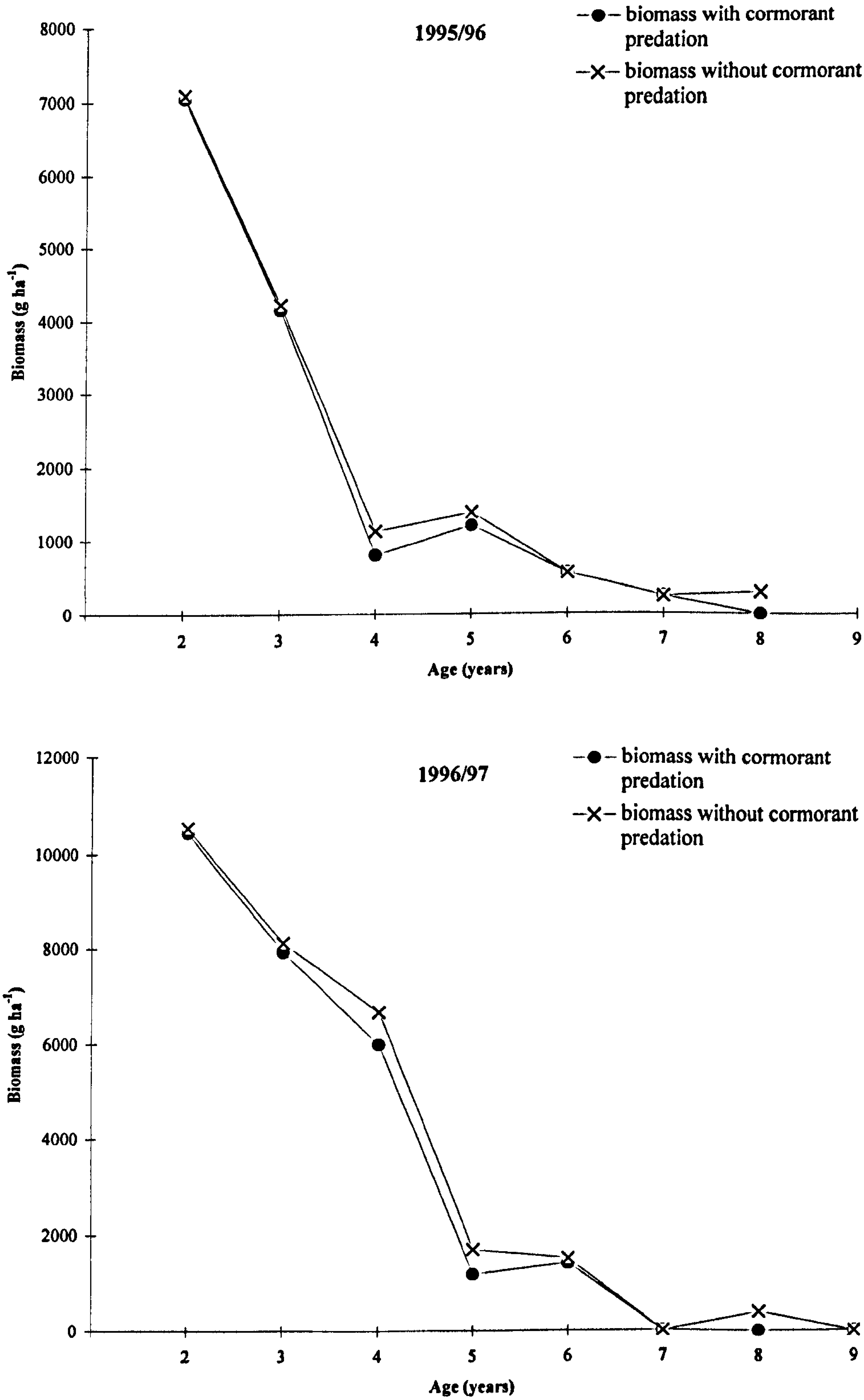


Figure 7.17 Impact of cormorant predation on the biomass of roach at Grimsargh number 3 Reservoir.

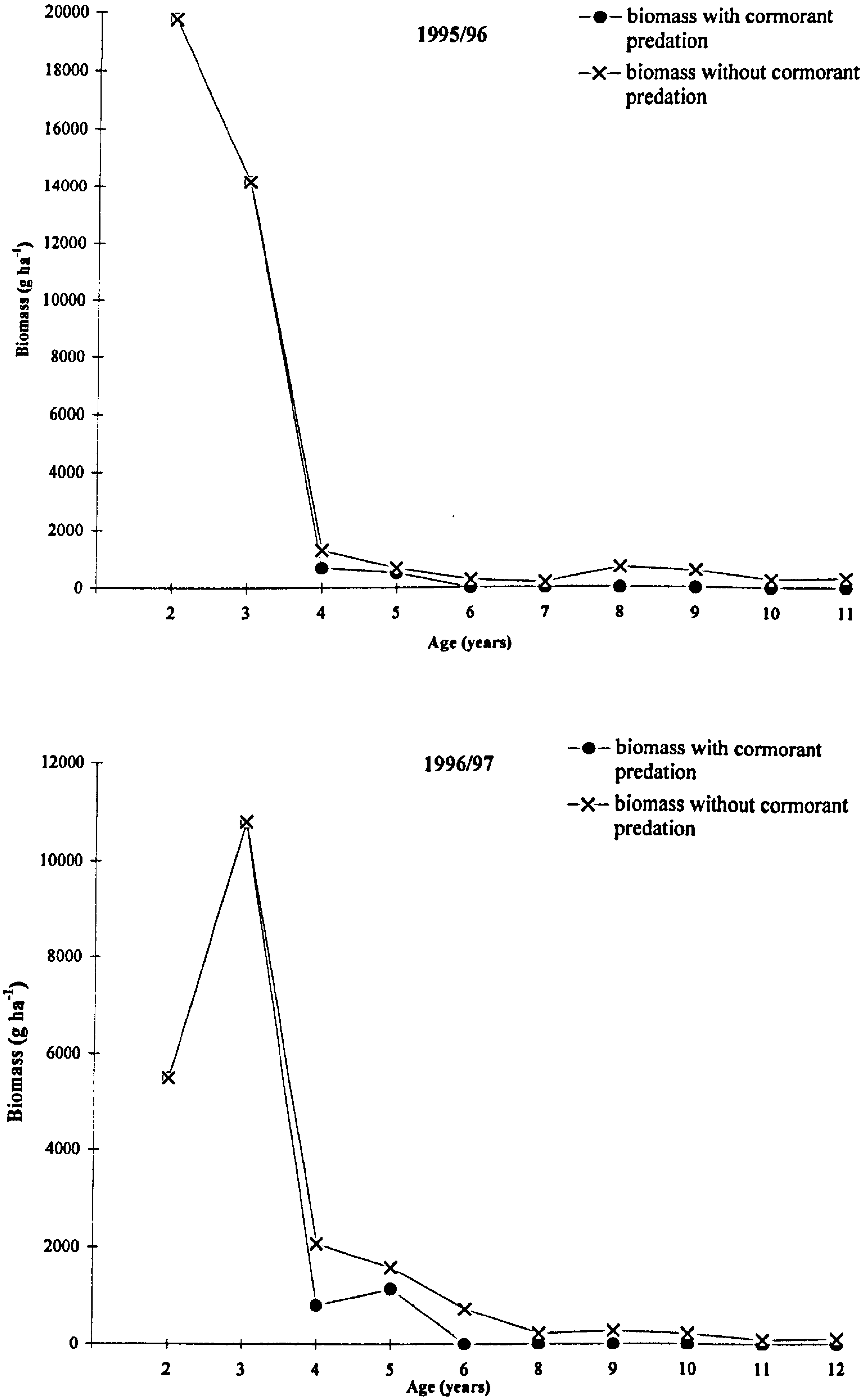


Figure 7.18 Impact of cormorant predation on the biomass of common bream at Grimsargh number 3 Reservoir.

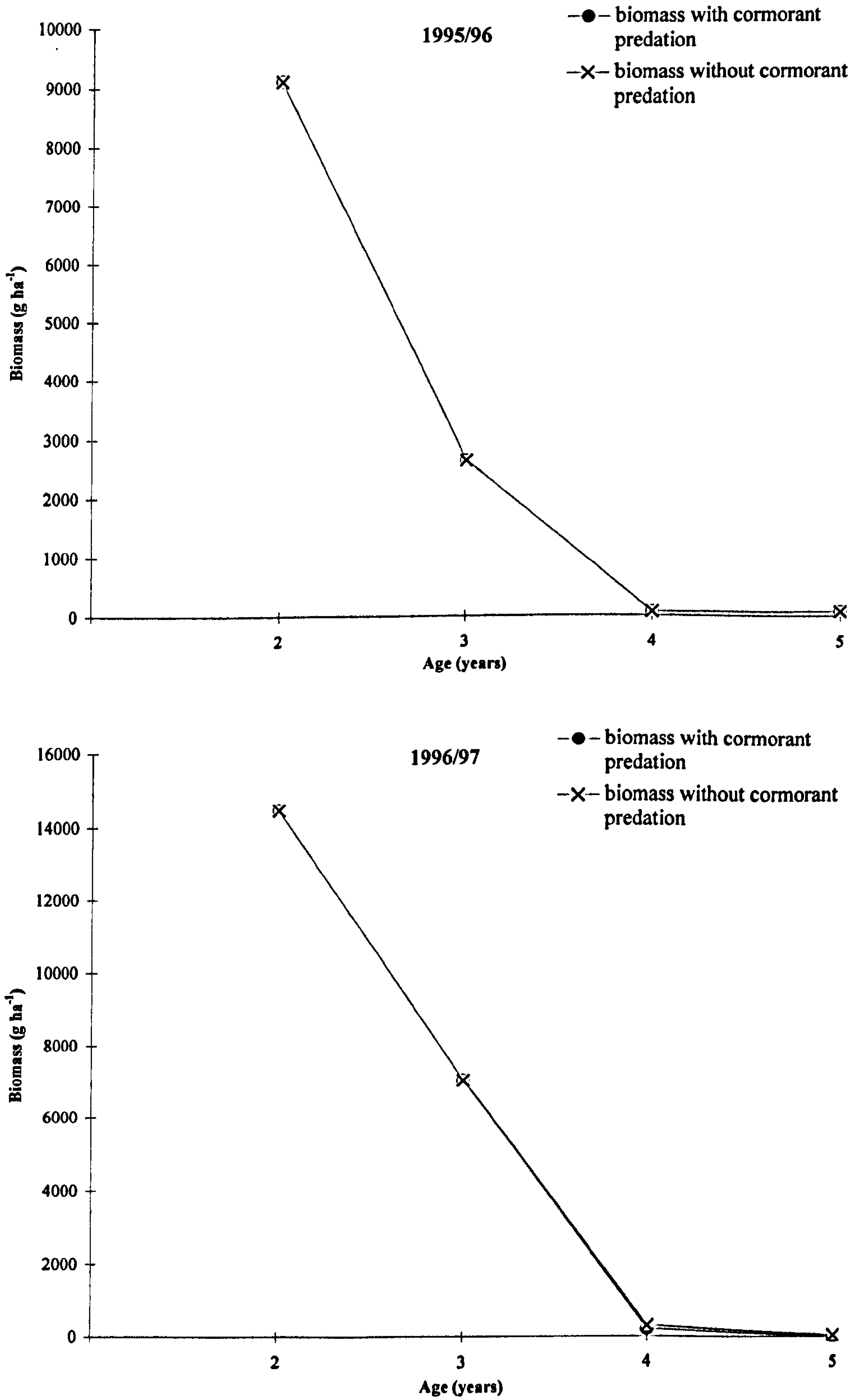


Figure 7.19 Impact of cormorant predation on the biomass of perch at Grimsargh number 3 Reservoir.

8. LOWER RIVER RIBBLE, LANCASHIRE

8.1 Introduction

The River Ribble rises in the Yorkshire Pennines, at Newby Head Moss, 422 m above sea level (SD 795840). The river flows south and then west, joining the Irish Sea at Preston, Lancashire (SD 527827). The river drains a catchment of 2182 km^2, covers a distance of 110 km and has four main tributaries, the rivers Hodder, Calder, Darwen and Douglas. Annual rainfall in the area varies from 1775 mm in the headwaters to below 890 mm in the estuary (Walsingham 1993). The lower River Ribble supports an important coarse fishery, particularly cyprinid species such as chub, roach and dace, with barbel present in some stretches. The lower Ribble is tidally influenced as far upstream as Red Scar weir (SD 587314) (Figure 3.3).

The water quality of the River Ribble catchment is generally good and known to be improving (Walsingham 1993). From the headwaters down to the confluence with Stock Beck the river is class 1A and from this confluence to Calder Foot the river is class 1B. The influence of the River Calder reduces water quality to class 2 down to the tidal limit.

8.2 Site details

For the purpose of cormorant monitoring on the lower River Ribble, the river was divided into two sections on the basis of the tidal influence (Figure 3.3). Section 1 was tidal and extended from Penwortham Bridge (SD 528288) to Red Scar weir, a distance of approximately 8.75 km. Section 2 was non-tidal and extended from Red Scar weir to the confluence with the River Hodder, a distance of approximately 18.75 km.

Fisheries surveys were restricted on the lower River Ribble (SD 462280 to SD 711382) due to limited access across agricultural land, a lack of sites from which to launch the electric fishing boat, angling club concern over damage to fish stocks by electric fishing, the low flow rates during 1996 and 1997, and limited Environment Agency consents due to migrating salmonids between November and April. Only two fisheries survey sites were sampled, which were both located in cormorant section 2.

Fisheries surveys were conducted on the main River Ribble at Ribchester (SD 652345) in July 1996 and June 1998. The length of the site was 600 m with a mean width of 30 m and the topography was riffle/glide. The substratum consisted of gravel, silt and boulders. The depth ranged between 0.5 and 5 m in the main river channel. Both banks were used as pastureland, and the left hand bank had a number of overhanging trees.

Fisheries surveys were also conducted on the main River Ribble at Great Mitton (SD 720388) in May 1996 and June 1998. The length of the site studied was 400 m, with a mean width of 20 m and a mean depth of 1 m. The site had a glide topography with a substratum of gravel, stones and boulders. Adjacent land use was pasture, and on the left hand bank there were a number of overhanging trees.

8.3 Materials and methods

8.3.1 *Cormorant monitoring*

Monthly cormorant counts were conducted from September 1995 during the period of study, with the exception of June and July. This was due to previous studies on the lower River Ribble showing a negligible number of cormorants remaining on the river in this period (Davies 1997). Between September and April of each year, the number of cormorants using the two major roosts on the river at Jackson's Bank (SD 620327) and Stubbins Wood (SD 620342) were also recorded. These roosts were situated at the boundary of the tidal and non-tidal sections of the river. All count and methodologies were as detailed in Section 3.2. Feeding observations were also carried out on the river (Section 3.2).

8.3.2 *Fisheries monitoring*

Boat mounted electric fishing (Section 3.3.1) was used to assess the fish populations on the lower River Ribble. River topography restricted the efficient use of the gear on the sections surveyed, especially in relation to the shallow riffles (below 0.5 m) and deep holes (above 5 m) encountered. The gear was operated across the whole width of the river, ensuring all habitats in the sections were assessed. All fish captured were processed as described in Section 3.4.6.

8.4 Methodology of data analysis

8.4.1 *Cormorants*

Cormorant monitoring data (Section 8.3.1) were analysed to show the size and species of fish consumed by cormorants compared with those shown to be available in the river by electric fishing. To estimate the mass of fish removed by the cormorants each year in the sections, the general MCS model was used (Section 3.4.2).

8.4.2 *Assessment of the impact of cormorant predation on the lower River Ribble*

The following information was used to assess the impact of cormorant predation.

- Comparison of species predated on by cormorants with those determined by electric fishing and from angler catches. This will show species selectivity by cormorants and anglers compared to the electric fishing community structure.

- Comparison of size distribution of fish eaten by cormorants and caught by electric fishing and anglers. This will show the species size selectivity of cormorants and anglers compared with the size of fish vulnerable to electric fishing.

- Comparison of the biomass of fish present (determined from electric fishing) with the biomass consumed by cormorants.

8.5 Status of fish populations at Great Mitton and Ribchester, River Ribble

8.5.1 *Biomass*

The biomass of the fish populations at Great Mitton and Ribchester, River Ribble, based on the gear calibration method using a fishing efficiency of P = 0.23, varied between sites and years (Table 8.1).

Table 8.1 Biomass of the fish populations, estimated by electric fishing, at Great Mitton and Ribchester, River Ribble, over the study period.

Site	Date	Biomass kg ha^{-1} Cyprinids	Salmonids	Other species	Total	Total g m^{-2}
Great Mitton	May 1996	54.3	17.9	1.9	74.1	7.41
	June 1998	23.1	31.9	0	55.0	5.50
Ribchester	May 1996	11.7	65.2	0.7	77.6	7.76
	June 1998	2.1	0	1.4	3.5	0.35

8.5.2 *Angler catch data*

Angler catch data were collected on the lower River Ribble, but as few angling matches were held between 1995 and 1998 catch per unit effort and the percentage of anglers with catch were not calculated. Consequently, only data supporting species composition and length frequency distribution from pleasure anglers' catches were available (Section 8.6.1 and 8.6.2).

8.5.3 *Species composition of fish populations at Great Mitton and Ribchester, River Ribble*

Chub, dace and brown trout were the dominant species in electric fishing catches at Great Mitton (Figure 8.1). A number of migrating sea trout were present in June 1998. Gudgeon, grayling and eels were also present in May 1996, although these species were absent in 1998.

A range of species were caught by electric fishing at Ribchester over the study period (Figure 8.1). Eels dominated in June 1996 with minnows and roach also present; minor species included chub, gudgeon and ruffe. In June 1998 only 5 fish were caught, comprising dace, eels and ruffe.

8.5.4 *Length frequency distribution*

As the electric fishing catches at Ribchester were poor, length frequency histograms were only constructed to compare with cormorant predation (Section 8.7.2).

At Great Mitton, chub up to 500 mm were present although 110 to 320 mm fish were dominant (Figure 8.2). This suggested a recruiting population of chub in the study area with individuals able to attain relative large sizes.

Dace between 140 and 250 mm were present at Great Mitton (Figure 8.2). The absence of small fish suggests poor recruitment and these larger fish may originate from elsewhere in the river system.

Brown trout of between 100 and 340 mm were also present at Great Mitton and the area supported an important trout fishery based on natural recruitment (Figure 8.3).

8.5.5 *Growth rates*

Growth of chub and dace at Great Mitton (Figure 8.4) were above average when compared with the national standards (Hickley and Dexter 1979). Chub were long lived, up to 16 years of age, whilst dace were relatively short lived, up to 4 years age, compared with other UK populations.

8.5.6 *L infinity and K*

The parameters in the Von Bertalanffy growth equation for chub and dace (Table 8.2 and Table 8.3) were within values derived from other UK fisheries (Table 8.2, Table 8.3 and Table 4.4).

Table 8.2 L_∞ and K values derived for chub at Great Mitton, River Ribble, compared with other UK fisheries.

Venue	L_∞	K
Great Mitton, River Ribble	580	0.13
River Severn (Craig Goch Research Team 1980)	649-662	0.09
River Stour (Mann 1976)	438-519	0.15-0.19
River Welland (Leeming 1967)	460-500	0.11
River Lugg (Hellawell 1971a)	387-540	0.11-0.27

Table 8.3 L_∞ and K values derived for dace at River Ribble, Great Mitton, compared with other UK fisheries.

Venue	L_∞	K
Great Mitton, River Ribble	348	0.21
River Severn (Craig Goch Research Team 1980)	294-328	0.22-0.27
River Stour (Mann 1974)	258-265	0.17-0.19
River Frome (Mann 1974)	265-275	0.22-0.28
River Thames (Williams 1967)	210	0.18
Willow Brook (Craig-Hine and Jones 1969)	240	0.32

8.5.7 *Mortality and survival rates*

The mortality (Z = 0.29) and survival (S = 0.75) rates for chub at Great Mitton were low, although within values found in other UK fisheries (Z between 0.15 and 0.44), indicating good annual survival. This ensured high survival rates for chub once individuals had grown out of the sizes vulnerable to cormorant predation (Section 8.7.2), as shown by the chubs' long life span (Section 8.5.5).

8.6 Cormorant results

8.6.1 *Temporal roost numbers*

The cormorant roost counts, taken between October and May in each winter period, showed an annual decline in numbers, with 95 in 1995/96, 85 in 1996/97 and 35 in 1997/98 (Figure 8.5).

8.6.2 *River cormorant numbers*

The number of cormorants recorded during river counts declined during the study period in both river sections (Figure 8.6 and 8.7). This reflects the general decline in the number of night roosting cormorants observed over the study period (Figure 8.5). In Section 1 of the river, the number of birds generally peaked in the early winter period (Sep-Nov) before declining (Figure 8.6). This contrasted to Section 2 where the number of cormorants on the river peaked between December and February (Figure 8.7), a situation most apparent in 1995/96.

The number of cormorants using Section 2 was almost twice as many when compared with the tidal section (peak counts; 32 birds tidal section , 65 birds non-tidal section) in 1995/96. However, in 1996/97 the number of cormorants using Section 1 exceeded those using Section 2. In the final study winter, cormorant numbers were similar in the two river sections (Figure 8.6 and 8.7). These temporal differences in cormorant occupancy could not be accounted for, but may be linked to food availability.

8.6.3 *Feeding success*

Foraging success rates on Section 1 of the Lower River Ribble were 37.7 to 56.1 % of all foraging bouts resulting in ingested fish, with 42.3 to 56.4 % successful in Section 2 (Table 8.4). Thus, little difference was observed between the tidal and non-tidal sections. The percentage of successful foraging bouts for the River Trent (Chapter 5) was between 25.9 and 52.0 % of all bouts resulting in ingested fish. Hence, foraging success rates were broadly similar between the two lowland rivers.

The dive success rate was low for both sections of the river, varying between 3.6 and 8.1 %. Low dive success rates were observed for all study sites.

Table 8.4 Feeding success of cormorants on Sections 1 and 2 of the lower River Ribble.

	Section 1			Section 2		
	1995/6	1996/7	1997/8	1995/6	1996/7	1997/98
Foraging bouts	62	98	53	71	119	39
Successful foraging bouts	27	55	20	30	57	22
% successful	43.5	56.1	37.7	42.3	47.9	56.4
Number of dives	1052	928	948	641	413	638
Number successful	49	157	34	40	68	30
% successful	4.7	8.1	3.6	6.2	4.8	4.7

8.7 Assessment of the impact of cormorant predation on the lower River Ribble

8.7.1 *Species composition*

Observations of cormorant feeding behaviour did not allow the identification of the fish predated on to species level. Fish were classified as cyprinids and perch. Accordingly, all fisheries data collected from electric fishing surveys and angler catch returns were reclassified as cyprinids, perch, pike and eels to allow comparison.

Cyprinids were the dominant species in electric fishing catches at Great Mitton and Ribchester, comprising 48 to 65 % of catches (Figure 8.8). Eels were present and these were selected for consumption by cormorants. Brown trout were present in the section but were not selected by cormorants. The minor species present in the electric fishing catches, but were not taken by cormorants, included perch and grayling. Cyprinids dominated angler catches in 1995/96 (93.2 %) (Figure 8.11). Salmonids and perch only constituted a small proportion of angler catch. Cormorants consumed mainly cyprinids, and to a lesser extent eels and salmon, on section 2 of the River Ribble (Figure 8.8).

8.7.2 *Length frequency distribution*

Cyprinids up to 400 mm were consumed by cormorants on section 2, although most fish were between 100 and 200 mm (Figure 8.9). Electric fishing surveys showed cyprinids up to 500 mm were present, with the dominant size classes 50 to 250 mm (Figure 8.9). Therefore, the dominant size classes of cyprinids were predated upon by cormorants.

In the section, the growth rate of the two dominant cyprinids, chub and dace (Figure 8.4), indicated that chub were vulnerable to cormorant predation up to the age of 9 and dace were vulnerable over their whole life span.

The length frequency distribution of cyprinids in angler catches in Section 2 during 1995/96 revealed cyprinids up to 760 mm were caught, with the majority of angled cyprinids 130 to 230 mm (83 %) (Figure 8.12). Hence, the dominant size classes of cyprinid consumed by cormorants and caught by anglers were similar.

The majority of eels (Figure 8.10) consumed by cormorants were 200 to 350 mm in length and were up to a maximum size of 500 mm. Electric fishing surveys showed eels were present in the section over a wide size range, from 100 to 500 mm.

8.7.3 *Estimated biomass of fish removed from Section 2 of the lower River Ribble*

The fisheries data collated in Section 2 of the River Ribble did not allow accurate assessment of the status of the fish populations. This was due to gear inefficiencies in many of the habitats found in the river, and the limited access and consents that were granted over the study period. Consequently, it was not possible to carry out cohort analysis. Instead, the biomass of fish observed in the fisheries surveys in Section 2 was compared with the biomass of fish removed by cormorants, giving a crude impact assessment of the proportion of biomass removed by the birds (Table 8.5)

Table 8.5 Comparison of the mass of fish removed by cormorants with the biomass of fish available in the fishery, Section 2, lower River Ribble.

	Mass of fish removed by cormorants (kg ha^{-1}).		Standing crop biomass, mean ± se of electric fishing surveys (kg ha^{-1})	Proportion of Standing Crop Biomass removed by cormorants (%)	
	Median	Interquartile Range	Interquartile Range	Median	Interquartile Range
1995/96	24.2	16.8-32.4	64-162	29.8	18.4-54.7
1996/97	7.2	5.3-9.2	40-115	13.9	8.0-23.3
1997/98	11.0	8.4-14.0	5-71	30.1	13.7-59.4

The median biomass values showed cormorants removed between 13.9 and 30.1 % of the standing crop of fish species in Section 2 of the River Ribble between 1995 and 1998.

8.7.4 *Wounding*

During the electric fishing survey at Great Mitton in May 1996 cormorant damage was observed on a number of chub and dace, although no damage was observed on the brown trout (Table 8.6). The wounded individuals were only marginally greater in size than unmarked individuals (Table 8.6). No cormorant wounds were observed on any fish caught by electric fishing at Great Mitton in June 1998 or at Ribchester in July 1996 and June 1998.

Table 8.6 Cormorant wounded fish observed at Great Mitton, River Ribble, in May 1996, compared with fish bearing no cormorant damage.

	Chub	Dace	Brown trout
Number of fish caught	12	6	10
Number caught bearing cormorant damage	3	2	0
% wounded	25.0	33.3	0
Average size of all fish (mm)	299.7	215.2	262.1
Average size of wounded fish (mm)	311.0	218.3	-

8.8 Discussion

8.8.1 *Fish populations*

Assessment of the status of the fish populations in the lower River Ribble was extremely difficult due to problems arising from river topography and fishing access. Hydro-acoustic surveying was not carried out in the study, but was considered to be impractical due to the shallow nature of the river and large number of boulders in the river channel. These factors would create a great deal of interference - 'noise' - in the sonar beam, forming a large number of false single and multiple targets (Section 4.2.2), which would create large inaccuracies during data analysis.

Despite the associated difficulties, a hydro-acoustic survey was undertaken on the river in 1995 (Davies 1997), which revealed a very patchy distribution of fish in the river. This was also demonstrated by the electric fishing survey results, where fish distribution and abundance were shown to be very variable over the study period (Table 8.1). This was also observed for the River Trent study sites (Chapter 5).

8.8.2 *Cormorant numbers*

The number of cormorants present in the River Ribble region over the study period was shown to decline, with night roost peak numbers decreasing by 63 % over the study period. Although cormorant numbers did decline at the Attenborough cormorant night roost in winter 1997/98 (Figure 4.13), the reduction was only 20 % in comparison with the previous winter. The decrease in cormorant numbers at the River Ribble cormorant night roosts may be due to a number of reasons, including inland migration to alternative roosting and feeding sites that provided more efficient energy returns, and illegal shooting of cormorants in the Ribble valley by fishery owners and managers, which was believed to occur due to anecdotal evidence.

8.8.3 *Feeding success*

The feeding success of cormorants on the lower River Ribble was similar to the River Trent (Section 5.8.2), with 37.7 to 56.1 % of all foraging bouts resulting in ingested fish. The success rate was below that of Holme Pierrepont (Section 4.5.4) and Grimsargh number 3 Reservoir (Section 6.5.4), but above that observed at Colwick Park Trout Lake in the period prior to trout stocking (Section 7.6.2). The similarity observed in foraging bout success on the River Ribble and River Trent may be due to the similar habitat observed between the two lowland rivers. It has been shown that both rivers have a patchy and variable fish distribution (Section 8.8.1), and, although specific habitat features will differ between the two rivers, they are broadly similar in topography, turbidity and flow. Hence, cormorants foraging on both rivers will be subject to the same feeding habitat constraints, resulting in a similar foraging bout success rate.

8.8.4 *Cormorant impact assessment*

Cormorant predation impact assessment could only be shown in terms of the biomass removed during each year of study (13.9 to 30.1 % standing crop removed annually) and by the non-lethal wounding rates caused by foraging birds (25 % of all chub and 33 % of all dace showed wounding due to cormorants). This does not allow a full assessment of predation impact after three years of cormorant predation, as has been shown for the other cyprinid survey sites. It highlights the limitations that can occur when a river, which is unsuitable for effective fisheries surveys, is chosen for such a study. Where electric fishing surveys were able to be carried out, catches were very variable (Section 8.8.1) and gear efficiency was likely to be adversely affected by the close proximity of shallow riffle areas (below 0.5 m) and deep pools (above 4 m) (pers. obs.). There was no alternative method that could have been utilised to sample the fish populations, due to the low angler effort and an unsuitable river topography for hydro-acoustic surveys (Section 8.8.1). Thus, despite the river and the cormorant night roosts allowing the collection of a robust cormorant data set, this was very limited in its application without adequate fish population data. However, non-lethal wounding of fish by cormorants was observed on fish in the lower River Ribble, which demonstrated the increased size of fish wounded compared with the resident fish stock (Section 8.7.4). Wounding of cyprinid species by cormorants was not observed on any other study site.

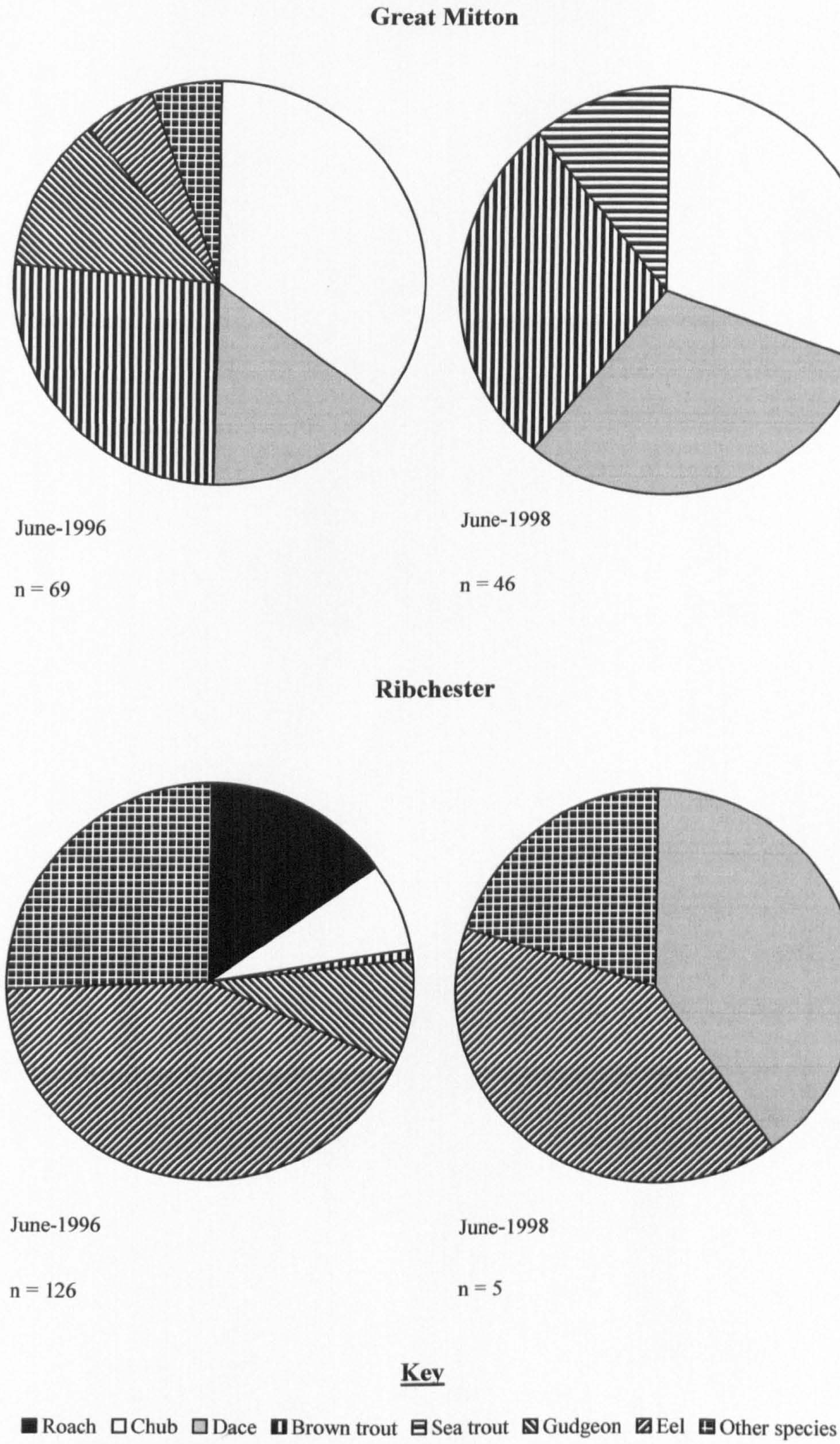


Figure 8.1 Species composition of electric fishing surveys on the lower River Ribble.

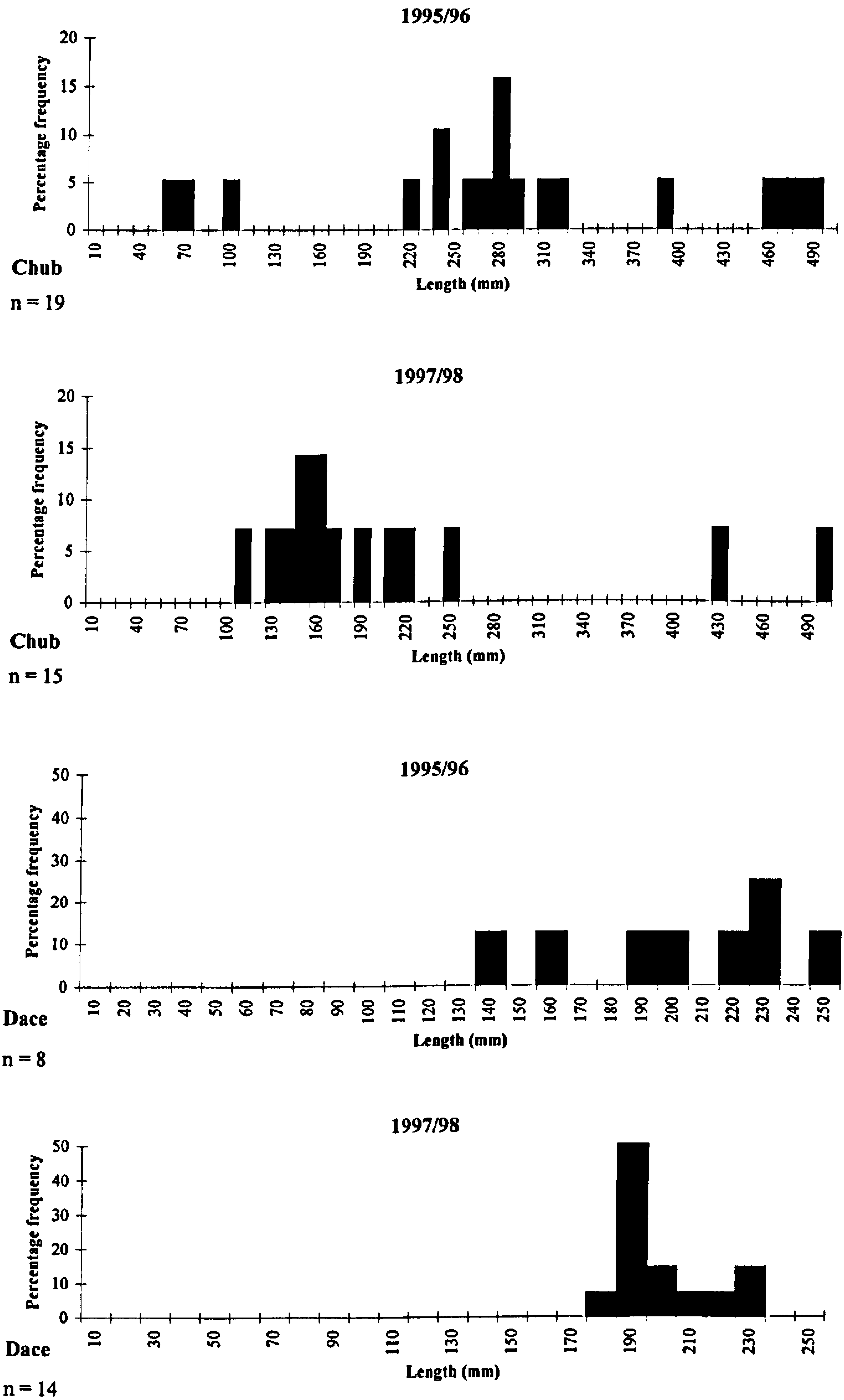


Figure 8.2 Length frequency distribution of chub and dace from the lower River Ribble at Great Mitton.

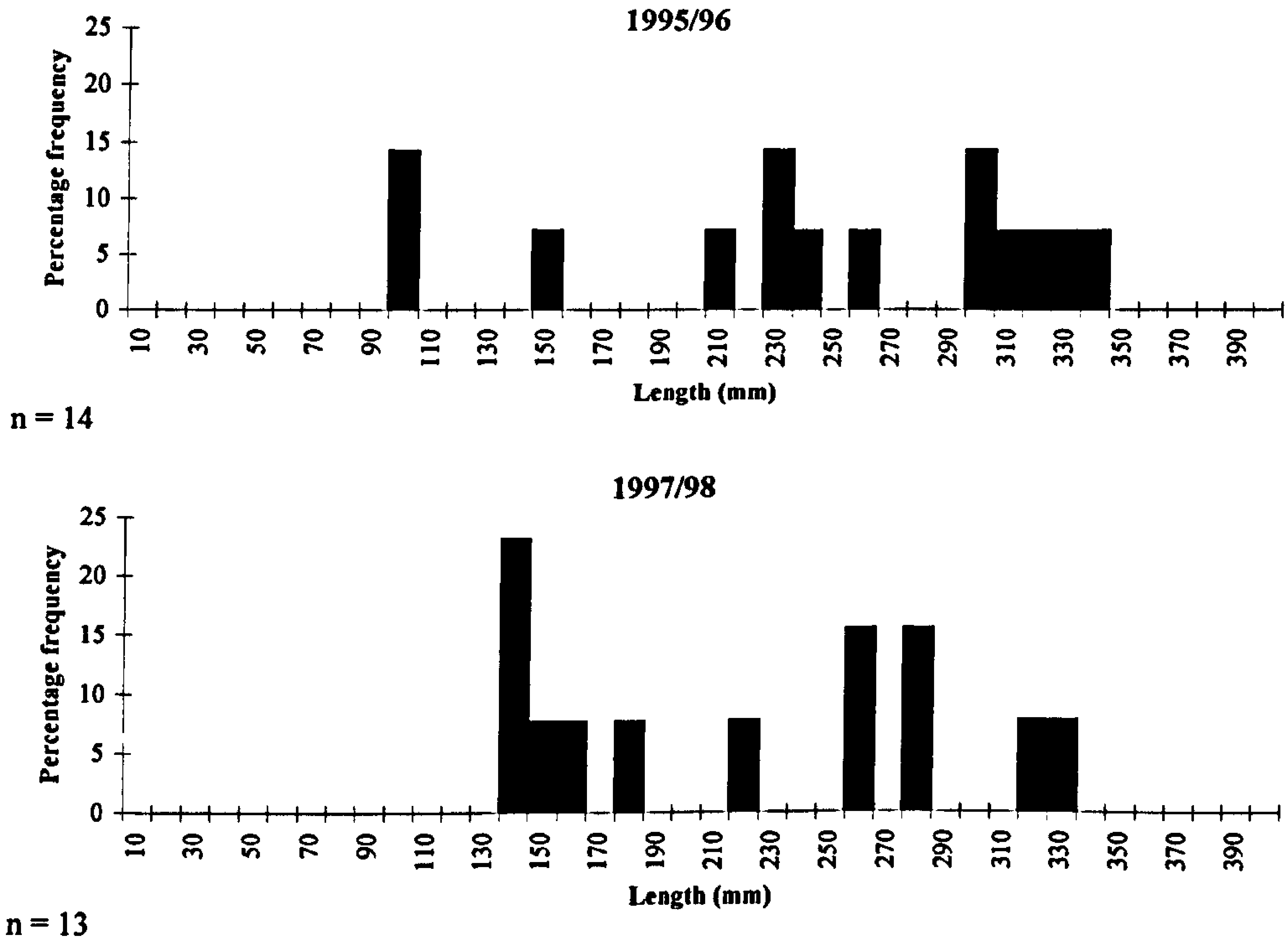


Figure 8.3 Length frequency distribution of brown trout from the lower River Ribble at Great Mitton.

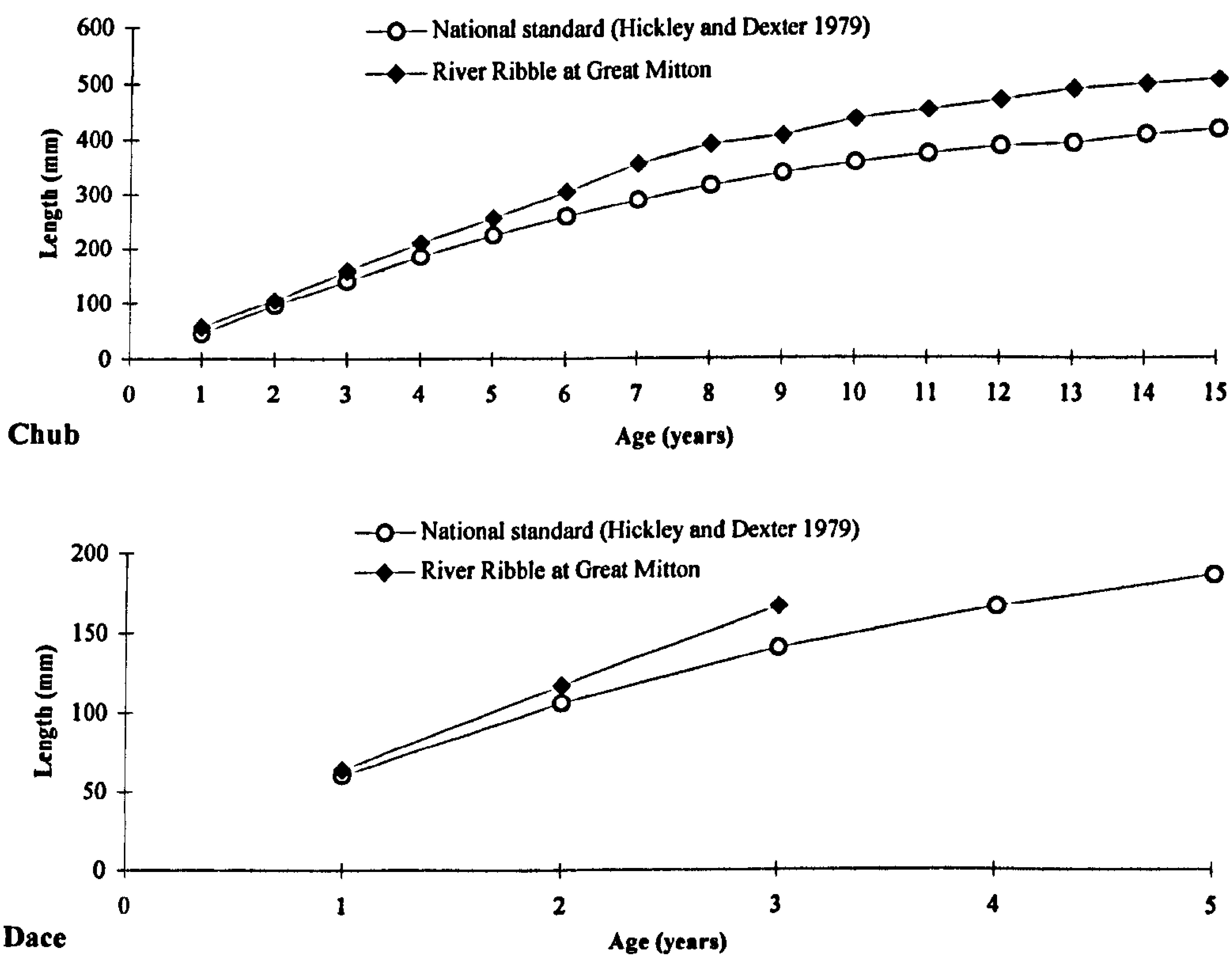


Figure 8.4 Growth of chub and dace from the lower River Ribble at Great Mitton compared with standard growth curves.

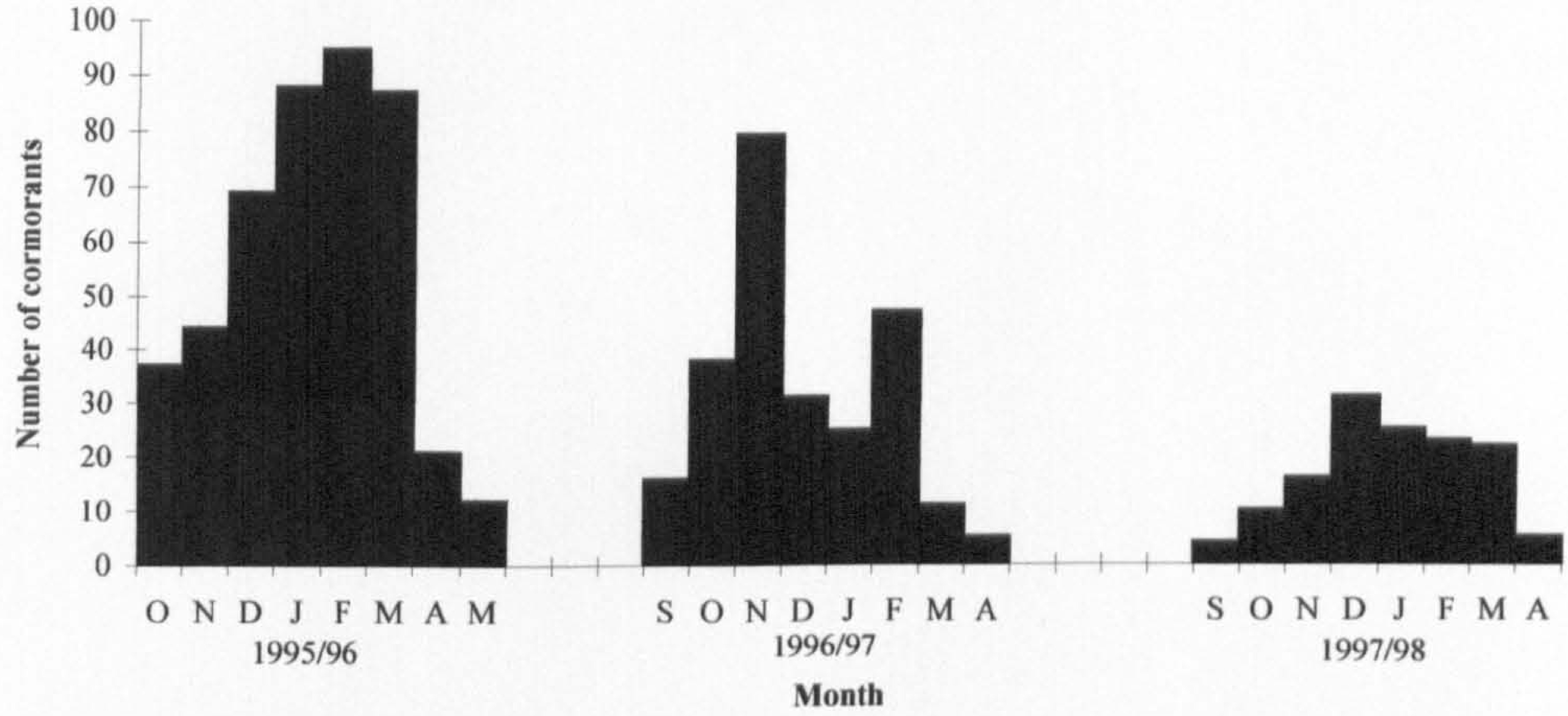


Figure 8.5 The combined number of cormorants recorded during monthly roost count at Jackson's Bank and Stubbins Wood roosts, River Ribble, 1995 to 1998.

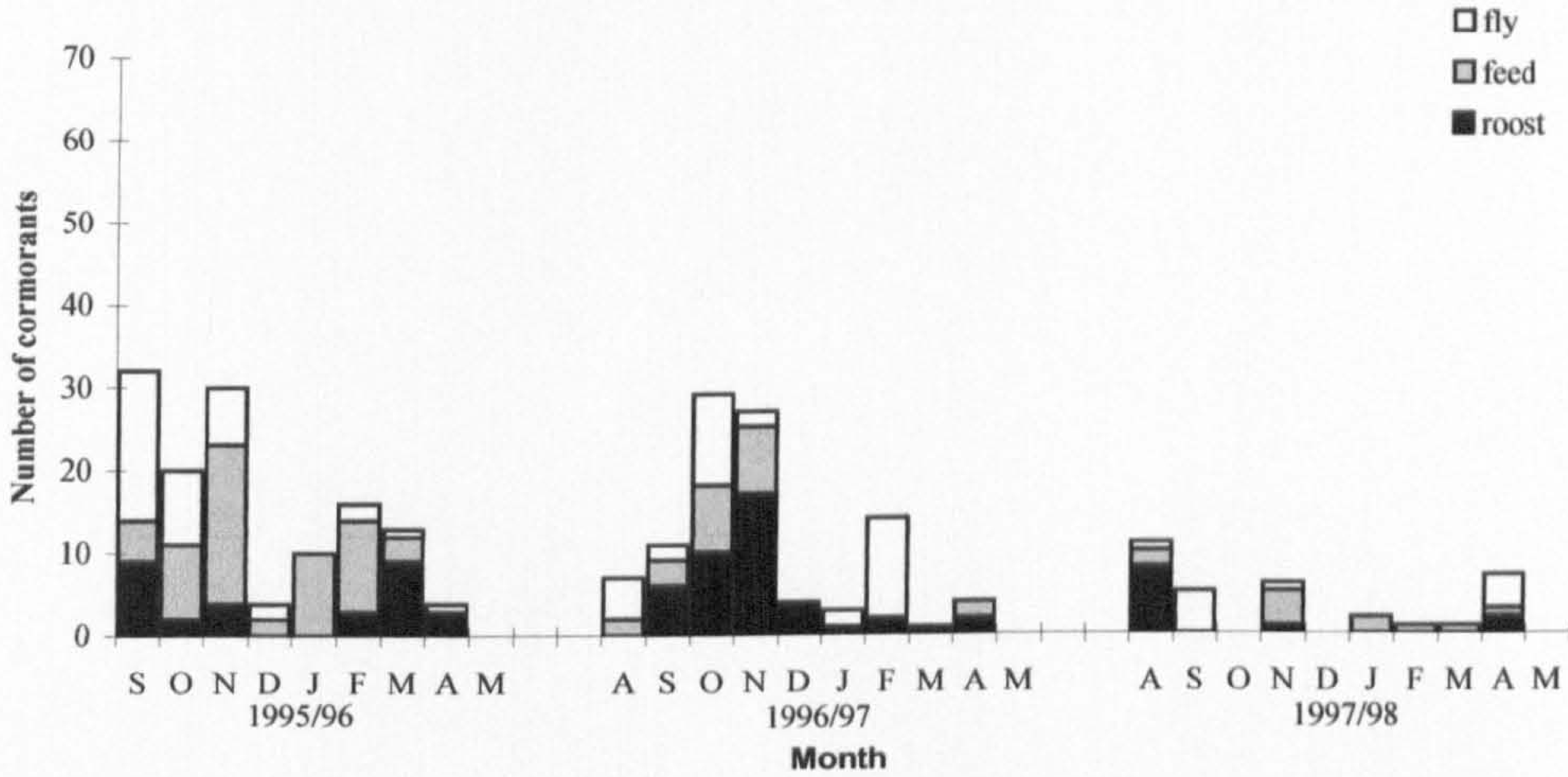


Figure 8.6 The number of flying, roosting and feeding cormorants recorded on Section 1 of the lower River Ribble, 1995 to 1998.

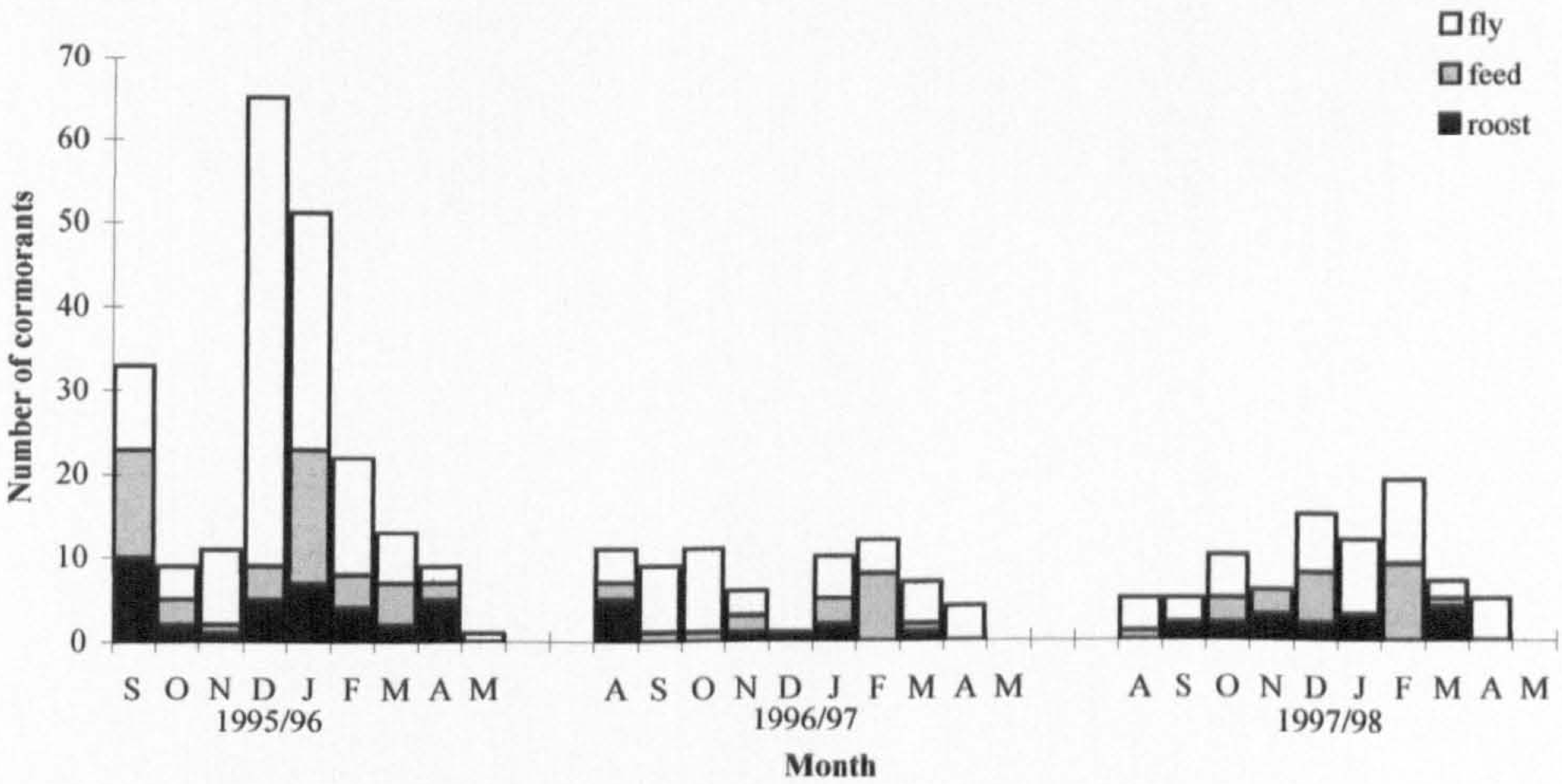


Figure 8.7 The number of flying, roosting and feeding cormorants recorded on Section 2 of the lower River Ribble, 1995 to 1998.

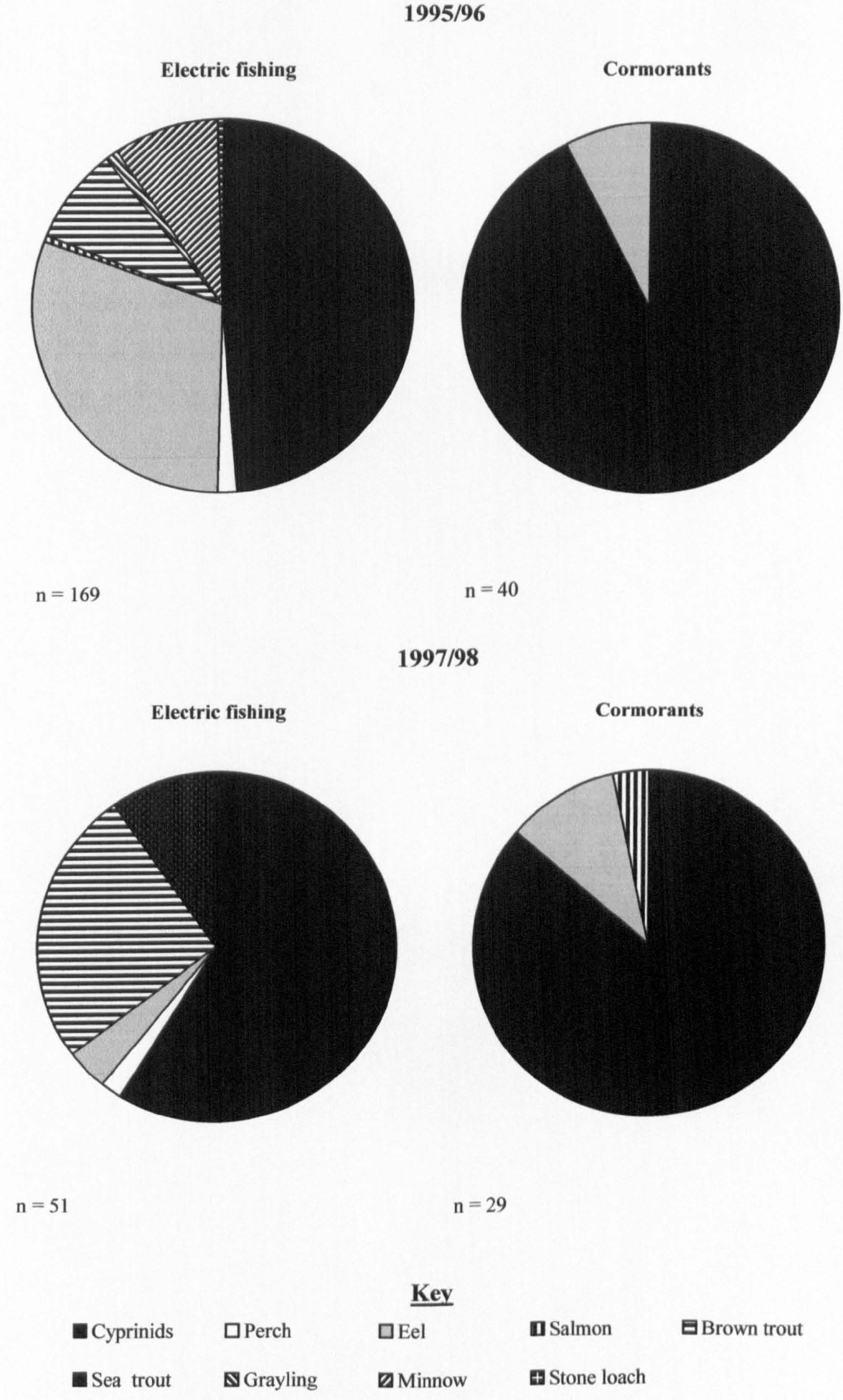


Figure 8.8 Species composition of electric fishing surveys and cormorant predation from section 2 of the lower River Ribble.

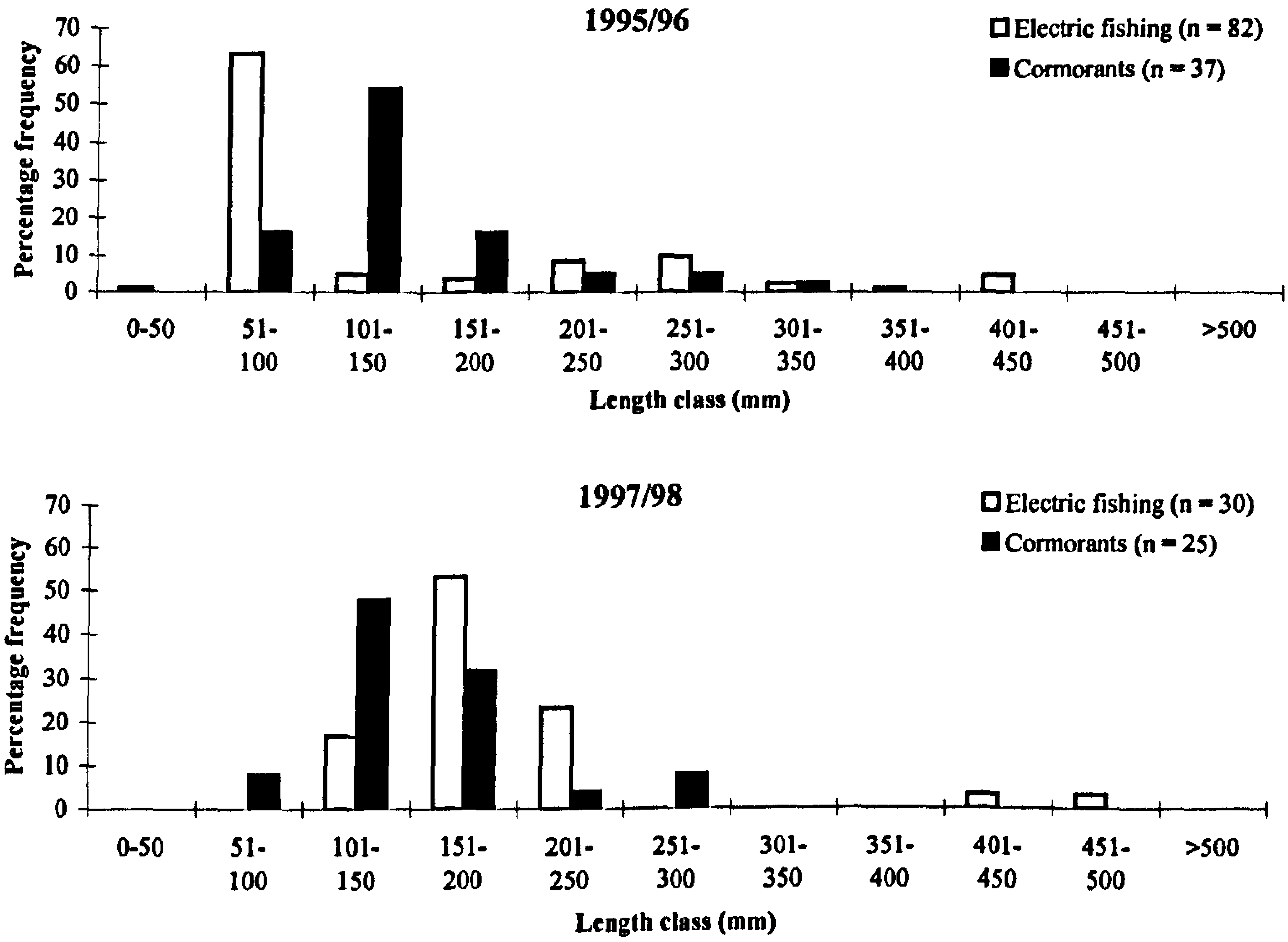


Figure 8.9 Length frequency distribution of cyprinids sampled by electric fishing and ingested by cormorants, lower River Ribble, Section 2, 1995 to 1998.

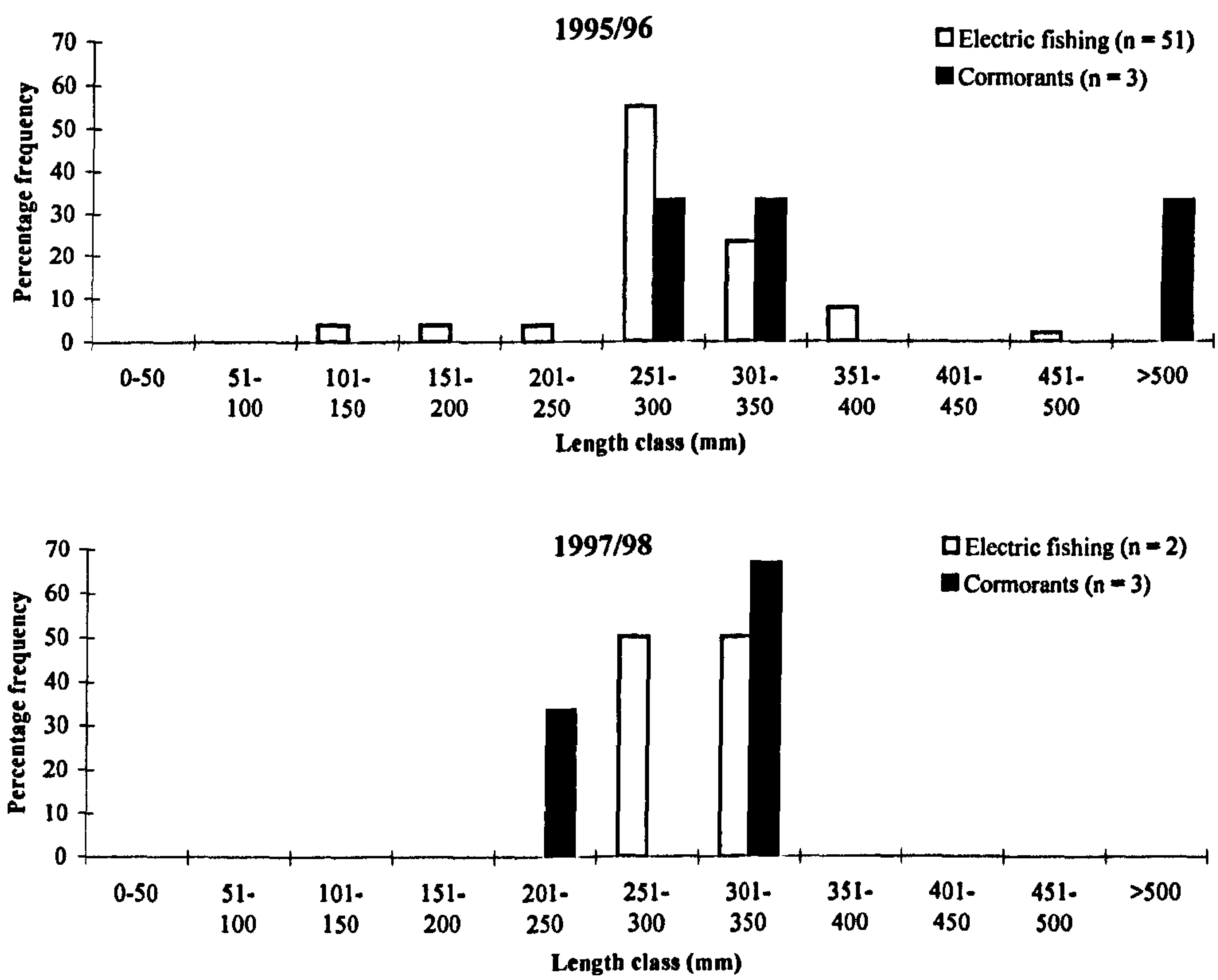


Figure 8.10 Length frequency distribution of eels sampled by electric fishing and ingested by cormorants, lower River Ribble, Section 2, 1995 to 1998.

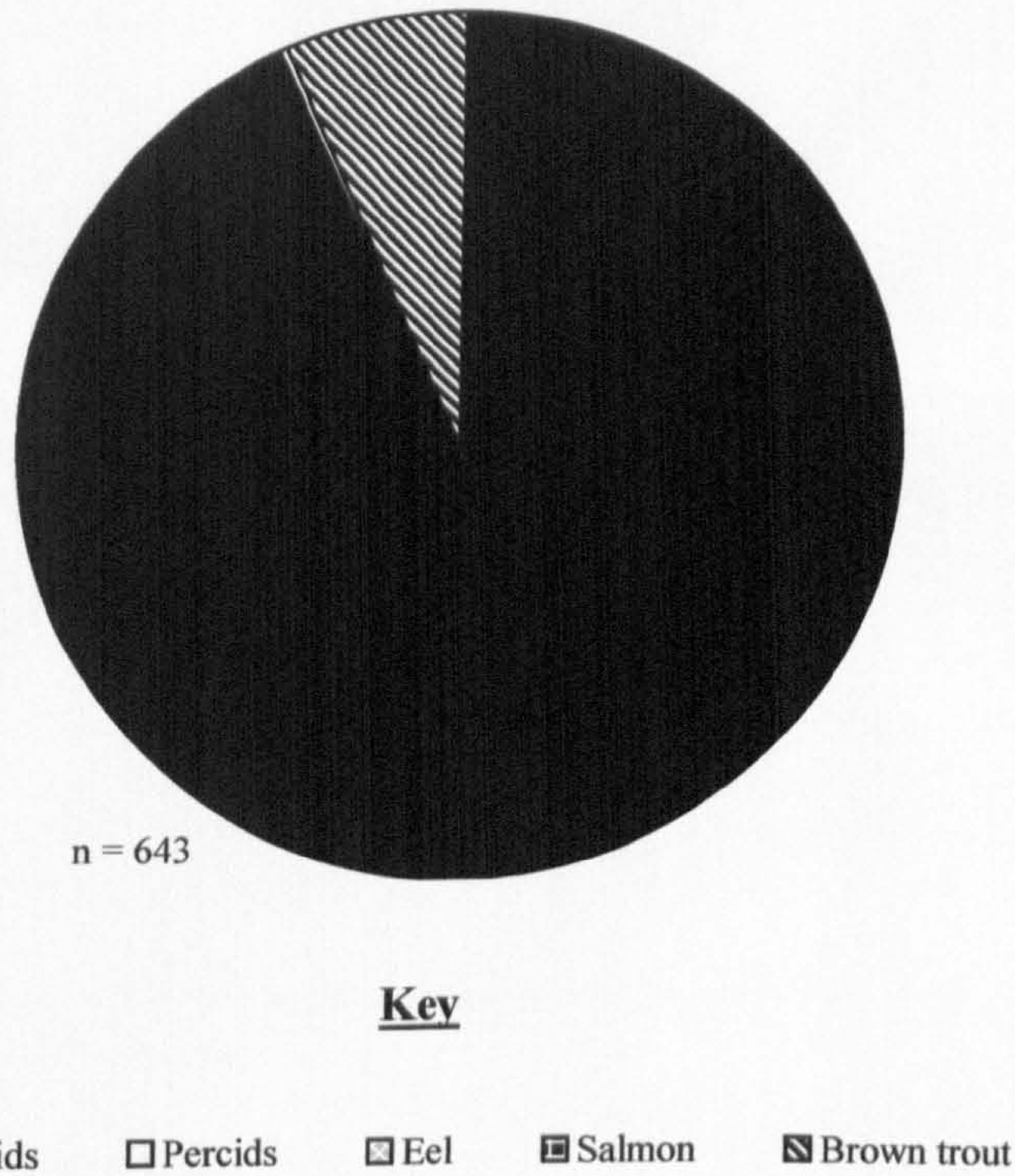


Figure 8.11 Species composition of angler catches from section 2 of the lower River Ribble in 1995/96.

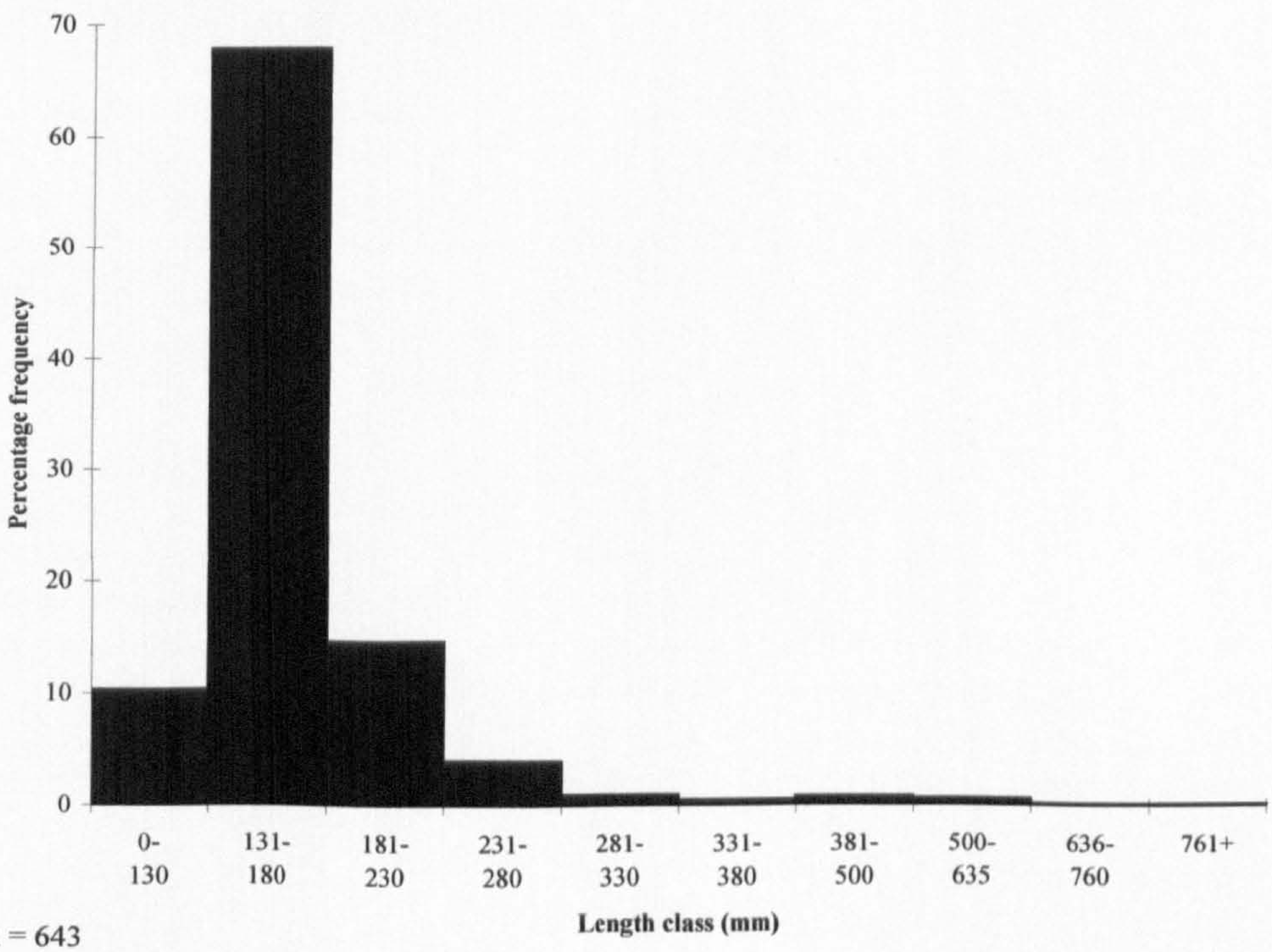


Figure 8.12 Length frequency distribution of cyprinids caught by anglers in Section 2 of the lower River Ribble in 1995/96.

9. DISCUSSION

9.1 Fish population dynamics

9.1.1 *Standing crop*

The standing crop of the dominant species in the study sites was highly variable, with large differences observed between survey years (Table 9.1). The values do not include pike and minor species, such as gudgeon and bleak. The standing crop estimates of 65.9 g m^{-2} and 47.6 g m^{-2} were derived for the River Thames, Reading (Mann 1965; Williams 1967) and Cooper and Wheatley (1981) estimated 44.7 g m^{-2} of the total standing crop was available for angler exploitation in the River Trent at Stoke Bardolph in 1974/75. As this value excluded fish of below 120 mm, total standing crop would have been considerably higher. With the exception of the River Trent sites, the standing crop estimates in the study were comparatively low (Table 9.1). However, the standing crop of 65.9 g m^{-2} in the River Thames, Reading, is considered high, as it was a stunted population in an eutrophic environment (Mann 1965).

Table 9.1 Standing crop range of cormorant exploited species in the study sites, 1995 to 1998.

Study site	Standing crop range (g m^{-2})
Holme Pierrepont	7.15 - 8.60
Beeston, River Trent	8.50 - 100.40
Trent Bridge, River Trent	2.90 - 27.50
Stoke Bardolph, River Trent	2.48 - 109.18
Grimsargh number 3 Reservoir	6.11 - 6.68
Lower River Ribble (biomass all species)	0.35 - 7.76

The largest annual variation in standing crop was observed on the River Trent (Table 9.1) and was supported by the hydro-acoustic surveys carried out by the Environment Agency (Midlands Region) (Sections 5.5.1; 5.6.1; 5.7.1). For example, the hydro-acoustic survey completed between Thrumpton and Beeston in June and September 1995 estimated fish abundance to decline from 762.5 to 172.2 fish per hectare (Table 5.2). Thus, a large difference in fish abundance was observed on the river during a period when cormorants were not present.

Annual differences in fish abundance were probably due to a number of inter-related factors. As cyprinid populations are generally dominated by a small number of strong year classes (Mills and Mann 1985; Cowx *et al.* 1995), the natural mortality of an ageing strong year class, the recruitment of a strong year class, and the recruitment of successive weak year classes will impact on subsequent standing crop. As physical factors, such as annual temperature and flow rates, are important to 0-group fish survival and their subsequent recruitment (Cowx *et al.* 1995; Cowx 1998), these physical factors will also have important implications on subsequent standing crop (Section 9.1.4). Aggregation of fish in winter and spawning periods will also contribute to the fluctuations in standing crop. Fish are known to aggregate in specific areas in the River Trent during winter (Lyons 1995, 1996, 1997; Jacklin 1996; Section 5.8.2), resulting in increased catches of fish in these areas, for example, Trent Bridge, compared with other periods (pers. obs.). In rivers, seasonal fish migrations are associated with reproduction, the opening up of feeding grounds as a result of seasonal changes in water levels, and movement to over-wintering areas (Wootton 1992). For example, Trent Bridge is known to be a spawning area for cyprinids and large numbers of mature fish were caught in this region in March and April of each survey year.

9.1.2 *Angler catches*

Angler catch rates, measured as catch per unit effort (CPUE), varied between sites, being greater on Holme Pierrepont Rowing Course compared with the River Trent sections (Table 9.2). Insufficient data were available for Grimsargh number 3 Reservoir and the lower River Ribble for CPUE and percentage of anglers with catch to be calculated (Section 7.5.3, 8.5.2). Angler effort was low on both fisheries, indicating they were not economically important fisheries for the region. As Colwick Park Trout Lake operates as a put-and-take rainbow trout fishery, CPUE cannot be compared directly with data from Holme Pierrepont and the River Trent.

Comparison with data from Stoke Bardolph, on the River Trent between 1969 and 1994, where CPUE ranged from 60 to 221 g man hour^{-1}, and the Yorkshire Ouse, where CPUE ranged from 71 to 90 g man hour^{-1} between 1971 and 1990 (Axford 1991), suggests the angler catch rates for the study sites were reasonable, with sufficient fish caught by anglers during a visit to be satisfied with the experience (Table 9.2). Thus, cormorant predation on the study sites did not appear to adversely affect angling success in the period 1995 to 1998.

Table 9.2 Catch per unit effort of anglers on Holme Pierrepont and the River Trent, 1995 to 1998.

Site	Catch per unit effort 1995 to 1998 (g man hr^{-1})
Holme Pierrepont	403 - 756
Beeston, River Trent	327
Trent Bridge, River Trent	202 - 247
Stoke Bardolph, River Trent	209 - 231
Stoke Bardolph, 1969 to 1994	60 - 221
Yorkshire Ouse, 1971 to 1990	71 - 90

(Axford 1991; Jacklin 1996).

On the River Trent, angler catches at Stoke Bardolph were strongly correlated to water temperature, with lower water temperatures resulting in decreased angler catch rates (Figure 5.89, 5.90). A relationship between water temperature and angler catch rate was also found on the River Severn (North 1980). This is because the optimum feeding temperature for coarse fish is 18 to 20°C. Thus, decreased angler success is experienced at lower temperatures (North 1980). This trend was also found for the Warwickshire Avon between 1987 and 1998 (Britton 1999).

Furthermore, different fish species have specific thermal optima for feeding, with lower and upper thermal thresholds outside of which feeding behaviour is inhibited (Elliott 1981). Fish with a low thermal optimum, for example, chub and pike (Steel *et al.* 1993), are those most likely to continue feeding during periods of low temperatures. In both the River Trent (Section 5.7.1) and the Warwickshire Avon (Starkie 1993), the chub populations have declined and this may have contributed to the decreased winter angler success.

At Holme Pierrepont, CPUE declined dramatically between 1990 and 1994 (1947 to 95 g man hour^{-1}), but improved thereafter (95 to 756 g man hour^{-1}) (Figure 4.1). However, CPUE in 1998 was still below that observed in the early 1990s (Figure 4.1). As angler catches are a function of the stock size (Crisp and Mann 1977; O'Grady 1980; Pawson 1982; North 1983), it would appear that in the period between 1990 and 1994 a large decline in fish stock abundance occurred.

Comparison of the standing crop range (Table 9.1) with the angler catch rate recorded at each site (Table 9.2) shows that the highest angler catch rates were found at Holme Pierrepont, where the fish standing crop was observed to be low relative to that recorded in the River Trent. Hence, the density of fish required to provide satisfactory angling results, in terms of CPUE, differed between Holme Pierrepont and the River Trent. This indicates that the fish density required to support angler catches is fishery specific, being dependent on the angling conditions of the fishery, and angler catch rates will vary with fish stock abundance in that fishery.

It is possible that the cause of the lower angler catch rate on the River Trent was high angling intensity. Cowx (1991) suggested that the poor catch in relation to the high standing stock in the River Trent was because fish must be caught on several occasions to satisfy the high angling pressure. Fish became less vulnerable to capture as the season progressed due to temporary hook avoidance (Raat 1985; Cowx 1991). At Stoke Bardolph, River Trent, the average recorded annual angler effort (matches only) between 1990 and 1997 was 28 389 man hours (Section 5.7.3), when CPUE was between 131 and 231 g man hr^{-1} (Figure 5.87). At Holme Pierrepont, the average annual angler effort between 1995 and 1998 was 2310 man hours, when CPUE was between 403 and 756 g man hr^{-1} (Table 4.1, 9.2). Hence, the difference in CPUE between the fisheries may have been due to the impact of angling intensity and capture on fish behaviour.

As angler catches have also been shown to be related to the fish community structure (Cowx 1990; O'Hara and Williams 1991; Jacklin 1996), this may be responsible for the increased catch rate observed at Holme Pierrepont in 1998 (Table 9.2). Despite a reduction in total standing crop in comparison with the previous year, CPUE increased (Table 9.3). This was probably due to the 1996 strong year class of roach entering the angler catchable cohort of fish in the lake, as fish below 120 mm are generally not available for angler exploitation (Cooper and Wheatley 1981). This would have increased the number of roach available for anglers, and was reflected in catches during the Home International angling match of August 1998 being dominated by roach of this year class (Figure 4.6).

However, the increased CPUE on Holme Pierrepont in 1998 may have also been due to increased angler efficiency. In the period 1995 to 1997, matches were fished by anglers of mainly average ability (T. Holden pers. comm.). In 1998, many of the anglers competing at Holme Pierrepont were of international standard, because they were practising prior to the Home International angling match held in August (T. Holden pers. comm.). This may have manifested itself as an artificial increase in CPUE in 1998. These results demonstrate the limitations of angler catch data where only a limited data set is available. This limitation was not responsible for the low catch rate recorded in 1994 (95 g man hour^{-1}), as CPUE was based on the period of the World Angling Championships, when only international standard anglers competed. Consequently, a large decline in fish abundance was still thought to have been responsible for the decline in angling success.

Table 9.3 Comparison of standing crop with angler catch per unit effort for Holme Pierrepont Rowing Course, 1995 to 1998.

Date	Standing crop (g m^{-2})	CPUE (g man $hour^{-1}$)
1995/96	7.15	567
1996/97	8.60	403
1997/98	7.92	756

At Colwick Park Trout Lake, annual CPUE increased as angler visits decreased (Table 6.1). This was consistent with other observations that trout angler catch is a function of stock density on put-and take fisheries, with higher angler catch rates generally attributable to greater stock abundance (Crisp and Mann 1977; O'Grady 1980; Pawson 1982; North 1983; Section 6.3.5).

9.1.3 *Growth rates*

The growth rates of the fish species were shown to be site specific, and probably dependent on the ecological and biological conditions present at each site.

The current growth rates of roach and common bream in Holme Pierrepont were fast compared with standard data (Section 4.4.6). Relative growth rates were comparatively slow between 1988 and 1993, but a large increase was observed between 1994 and 1997 (Figure 4.20 to 4.23). This was linked to a decrease in fish abundance in the fishery, a coincidental decline in angler catches and cormorants first being observed to forage on the lake (Section 9.2.2). The decline in fish density is likely to have reduced inter- and intra-specific competition in the fish populations and community, resulting in the improved growth rates (Botsford 1981; Section 4.6.3). The fast growth rates ensured the fish populations were able to grow beyond the cormorant optimum prey size in only one year, so reducing their vulnerability to predation (Section 4.6.2). General patterns observed in fish populations with high growth rates are low asymptotic (L_∞) sizes (Wootton 1992), high mortality rates (Pauly 1980; Wootton 1992) and a reduced age of sexual maturity, with high investment in the early reproduction (Alm 1959, Wootton 1992). This is a potent way of ensuring genetic representation in the succeeding generation (Begon *et al.* 1989). The L_∞ of roach at Holme Pierrepont (302 mm) was lower than for River Trent roach (341 to 361 mm) (Table 4.4), with the latter slower growing. The mortality rate for roach in Holme Pierrepont was also high in comparison to roach in the River Trent (Section 9.1.5).

Decreased fish population sizes have resulted in increased growth rates in other fisheries. A reduction in population size was responsible for increased roach growth rates in Grey Mist Mere (Linfield 1979); a reduction of 40 % in the biomass of the roach population of Lake Söveborg resulted in increased perch biomass by 140 %, due to decreased age-class competition (Persson 1986); and commercially exploited vendace exhibited increased growth when 90 % of the total cohort production was removed (Sarvala *et al.* 1994).

The growth rates of fish in the River Trent were generally similar between sites, with roach and chub growth classed as average when compared with standard growth curves (Sections 5.5.1, 5.6.1, 5.7.1). Common bream and perch growth rates were classed as good. The present growth rate of roach is slower compared with the period 1974 to 1987 (Figure 5.94), while common bream growth rates have improved in recent years.

Chub growth rates have remained constant since the 1970s (Jacklin 1996). These growth patterns are linked to the changes in the ecological conditions of the river which have arisen from the improved water quality of the river. The decrease in organic loading and turbidity now favour common bream and perch rather than roach (Section 5.7.1). Svardson (1976), Burrough *et al.* (1979) and Persson (1983) all associated roach dominance with nutrient rich, turbid, eutrophic conditions, so reduction in organic enrichment possibly explains their decline in growth rates in the River Trent. The decrease in roach growth rates was important because it meant they were vulnerable to cormorant predation throughout their life (Section 5.5.3; 5.6.3; 5.7.3).

Annual growth increments of roach were highly variable in the River Trent (Sections 5.5.3; 5.6.3; 5.7.3). This was tentatively linked to the effect of temperature, with increased growth observed in years of strong roach year class strength (YCS), as YCS was correlated to the increased annual number of degree days over 14°C (Section 9.2.4). Annual fish growth was not governed by falls in fish abundance due to cormorant predation, as was observed at Holme Pierrepont, but was governed primarily by other biotic and abiotic factors, such as water temperature, competition and adequate food supply (Wootton 1990; Cowx *et al.* 1995). Annual variation in roach growth rate has also been observed on a number of other UK rivers, for example, the Willow Brook, Northamptonshire (Cragg-Hine and Jones 1969), the Rivers Stour and Frome, Dorset (Mann 1974) and the River Thames at Reading (Williams 1967). The annual growth variation was also linked to differing annual temperature conditions in these rivers, with increased growth during years of elevated annual temperature (Mann 1974).

The growth rates of fish species in Grimsargh reservoir were slow in comparison with standard data (Figure 7.7 to 7.9), and only common bream demonstrated a long life span. However, as common bream were only caught between 2 and 5 years old and 11 and 16 years old, with no intermediate year classes present in the fishery, it is probable that the older bream were stocked and not from a naturally recruited population. The poor cyprinid growth rates were linked to a potentially overcrowded population of fish below 150 mm in the reservoir, possibly resulting in competition for food resources (Section 7.8.3). This was caused by good recruitment of juvenile fish, indicating good spawning conditions and high fry survival, and culminated in overcrowding. This would increase intra- and inter-specific competition and result in the observed slow growth rates. This pattern was also shown at Grey Mist Mere, Lancashire (Linfield 1979) and the River Thames, Reading (Williams 1965). At Grey Mist Mere, the slow growing roach population was linked to the high survival of successive years of 0-group fish (Linfield 1979). A marked increase in roach growth rate was observed only after three successive years of poor juvenile recruitment, resulting in reduced fish numbers. The reduced fish density probably lowered the inter- and intra-specific competition in aspects such as feeding, resulting in increased growth (Linfield 1979). In the River Thames, the roach, bleak, dace and perch populations were abundant, resulting from eutrophic conditions, with inter- and intra-specific competition resulting in stunted growth (Williams 1965).

The limited growth data for the River Ribble fish populations revealed chub and dace growth rates were above average compared with standard growth curves (Section 8.5.5). The limited data set did not allow further analysis.

9.1.4 *Year class strengths*

Temperature is an important variable in determining year class strength (Le Cren 1958; Craig *et al.* 1979; Pivnicka 1982; Mills and Mann 1985; Craig 1987; Cowx *et al.* 1995; Section 2.10.3). Years of higher temperatures result in stronger year class strength (Mills and Mann 1985; Cowx *et al.* 1995), and periods of lower annual temperatures result in weak year classes (Derback 1947; Smith and Krefting 1953; Walberg 1972). This results in populations often being dominated by individuals spawned in a small number of years (Mann 1979; Linfield 1981; Goldspink 1983; Mills and Mann 1985). As body length and weight are important for juvenile survival, years of increased summer temperatures, resulting in an increased growth period, may reduce fry mortality from factors such as predation and downstream drift (Mann 1979; Cowx *et al.* 1995). It is probable that the fat/lipid content of the fry is also important for survival (Cowx 1998).

However, water temperature does not account for all the variation in year class strengths. In some rivers, years of high temperatures and good growth do not always convert to strong year classes. This was shown by the year class strength fluctuations in many UK rivers in the period 1989 to 1991. If temperature was the key regulating factor, three successive strong year classes should have been produced (Cowx *et al.* 1995). It was found that although 1989 did produce a strong year class, 1990 was average and 1991 slightly above average (Cowx *et al.* 1995). Hence, other factors also affect year class strength, for example, increased flows and biotic factors. Increased flows may have had a flushing effect on juvenile stocks. Biotic factors act through competition. It is apparent that the production of two or three successive strong year classes is rare, possibly due to inter-cohort competition for food resources, especially for 0-group fish, and increasing mortality (Cowx 1998).

Roach year class strengths were highly variable in the study sites between 1988 and 1996, with years of strong and weak year class strength (Table 9.4). In the River Trent and Holme Pierrepont, 1993 and 1994 produced weak year classes of roach. Strong roach year classes were produced in the River Trent from 1990 to 1992. This pattern was correlated to the number of degree days over 14°C (Table 9.5). This has also been observed on a number of other UK rivers (Cowx *et al.* 1995; Table 9.5).

Alternative causes of variation in roach year class strengths in the River Trent were difficult to determine due to the absence of data sets for factors such as flow rates. However, the year class strength of the 1989 cohort roach at Stoke Bardolph was weak when compared with the cohort at Trent Bridge (Table 9.4), and for UK rivers in general (Cowx *et al.* 1995). The year class strength of the 1988 cohort of roach at Stoke Bardolph was very strong (Figure 5.102), so inter-cohort juvenile competition may have acted on the 1989 cohort by reduction of their food resources and resulted in increased fry mortality rates. This feature of year class strength was also observed at Holme Pierrepont (Table 9.4).

The 1996 roach year class was strong in Holme Pierrepont (Table 9.4) and was shown to positively influence angler catches in the 1998 Home International angling match by providing an increased number of angler-exploitable roach in the fish community (Figure 4.6; Section 9.1.2). Thus, under the present conditions, if recruitment of that cohort had been weak, angler catches in 1998 would probably have been reduced in comparison to the levels observed (Section 4.4.2).

Table 9.4 Year class strengths of roach in the study sites, 1990 to 1996. A value above 100 represents a strong year class.

	Year class								
Site	1988	1989	1990	1991	1992	1993	1994	1995	1996
Holme Pierrepont	180	78	84	90	109	36	84	113	101
Beeston			125	117	113	91	91	105	
Trent Bridge	72	139	120	105	102	45	85	123	
Stoke Bardolph	187	89	120	111	110	44	96		
Grimsargh					75	0	137	362	451

Table 9.5 Correlation coefficient of year class strength with relevant water temperature for UK rivers and species (Cowx *et al.* 1995).

River	Species	Temperature (degree days> °C)	Correlation coefficient (r)
Trent at Beeston	Roach	14	0.91
Trent at Trent Bridge	Roach	14	0.82
Trent at Stoke Bardolph	Roach	14	0.77
Ouse	Roach	14	0.79
Ouse	Dace	12	0.13
Swale	Dace	9	0.77
Swale	Chub	12	0.39
Ure	Roach	14	0.77

The year class strength of species in Grimsargh number 3 Reservoir demonstrated a different pattern from the Midlands study sites, with 1994 producing a strong year class and 1992 a weak year class. Although this may have been due to different regional climatic variables compared with the Midlands study area, resulting in a different critical temperature threshold for year class strength determination, it was more likely to have been caused by biotic factors. The high density of slow-growing cyprinids below 150 mm in the reservoir was likely to have caused significant inter-cohort competition for the food resources of 0-group fish, and resulted in increased mortalities and reduced year class strengths. This was also noted at Grey Mist Mere, Lancashire (Linfield 1979).

9.1.5 *Mortality rates*

Natural mortality rate is a key parameter in the assessment of the status of a fish population. Fish die from intrinsic or extrinsic causes (Wootton 1990). Intrinsic causes include genetic deaths from the presence of lethal alleles in the genotype, physiological failures and diseases (Wootton 1990). Extrinsic causes include the lethal effects of abiotic factors such as temperature and salinity, and biotic factors, such as predation, parasitism and malnutrition (Wootton 1990). Hence, the mortality rates of fish species in the study sites were dependent on the intrinsic factors present in the population, and the extrinsic factors at each site, caused by the ecological and biological conditions present.
Comparison with data from other UK populations (Cowx *et al.* 1995), revealed the general mortality patterns at the study sites were:

- high in Holme Pierrepont (Section 4.4.8);
- relatively low in the River Trent (Sections 5.5.1, 5.6.1, 5.7.1);
- high in Grimsargh number 3 Reservoir (Section 7.5.9);
- low for chub in the River Ribble (8.5.7).

The effect of the different mortality rates from the study sites on a cohort of common bream is shown on Figure 9.1. In this hypothetical example, a greater number of common bream were able to survive to spawn in subsequent years and be available for angler exploitation in the River Trent than in Holme Pierrepont and Grimsargh number 3 Reservoir (Figure 9.1).

The high mortality rate of fish at Holme Pierrepont was probably linked to the high exploitation of the fish in their first two years of life by cormorants. The removal of up to 62 % of cyprinid species after three winters of cormorant predation (Section 4.6.4), primarily of fish below 100 mm, resulted in reduced survival and short longevity (Section 9.4.1). Natural mortality rates were lower on the River Trent study sites. As cormorant predation was low on the river (Section 9.4.2), the mortality of fish below 100 mm was subject to alternative, less intense, extrinsic and intrinsic factors. The natural mortality rates were very high at Grimsargh number 3 Reservoir. However, cormorant predation was low on the reservoir (Section 9.4.4), so they were due to other factors. As the cyprinid populations were composed of high densities of slow-growing fish under 150 mm, this suggests inter- and intra-specific competition was high and resulted in high mortality due to starvation and disease in stressed individuals (Wootton 1990). However, the plasticity of the growth of fish, their ability to lay down reserves of lipids and their capacity to survive long periods without food has been shown to reduce the importance of starvation as a cause of death, except under unusual circumstances (Wootton 1990).

9.2 Inland cormorant ecology

9.2.1 *Optimum foraging theory*

The cormorant foraging data allow aspects of optimal foraging theory to be applied to the study sites. Optimum foraging theory assumes an animal will maximise the net rate of energy intake during foraging (Ulenaers *et al.* 1992), with a cost-benefit relationship existing between food availability, food intake and growth rate (Cowx 1998). It is able to make predictions about prey size selection in animals (MacArthur and Pianka 1986; Stephens and Krebs 1986; Kamil *et al.* 1987), where information on the time spent searching for and handling prey is available (Ulenaers *et al.* 1992).

As actively searching predators usually hunt for food which is clumped or patchy in distribution, considerable selection in prey and habitat utilisation is necessary to optimise energy returns (Putman and Wratten 1984). This selection is important to the relative fitness of the individual, as fitness will be increased if the net foraging energy intake is maximised (Krebs and Cowie 1976). The net level of energy intake is affected by changes in prey characteristics, such as density and their distribution in the foraging habitat (Werner *et al.* 1983). Design rules for optimal prey selection have been deduced by a number of workers, with similar conclusions (MacArthur 1972; Charnov 1976a):

- predators should prefer 'prey types' (size classes of species), according to their rank order of profitability, defined as net energy yield per unit handling time;

- when the encounter rate with profitable prey is low, the predator will be non-selective and eat all prey types it encounters;

- when the encounter rate with profitable prey is high, the predator should selectively ignore unprofitable prey types and as long as profitable prey are sufficiently abundant it should ignore unprofitable prey regardless of how often they are encountered.

Hence, the optimum foraging habitat for cormorants can be ascertained by simple indices concerning the ease with which they are able to forage successfully at a site, as foraging success may be related to fish abundance in the site. However, it has also been shown that prey availability in a foraging site for piscivorous birds does not depend only on the prey abundance in the habitat, but also on following factors.

- *Availability of cover to fish* (Wood and Hand 1985). Red breasted merganser attack on juvenile salmon decreased with increased salmon cover availability.

- *Water transparency.* The less turbid the water, the easier it is for sight predators to observe their prey (Eriksson 1985). However, in turbid water, the reduced distance over which birds can see fish may be offset by the reduced range over which fish can detect an approaching predator. Fish may be less wary and hide less in turbid conditions and so be easier to catch (Schafer 1982).

- *Prey characteristics.* Fish size, movement, swimming speed and contrast with the background may determine their vulnerability to attack (Wootton 1990; Ulenares *et al.* 1992).

- *Shoaling characteristics.* Small fish may aggregate in shoals and be more conspicuous than single fish, hence, be more vulnerable to attack. Although larger fish may be detected more easily, they may be able to sustain a faster cruising speed than smaller individuals and escape easier (Bond 1979).

Additionally, predatory birds are able to synchronise foraging patterns to coincide with the main activity patterns of their prey (Mikkola 1970; Daan 1981; Rijnsdorp *et al.* 1981), and foraging animals subjected to events repeated at similar times of the day subsequently show daily time-related behaviour patterns (Enright 1975). Modification of the daily foraging pattern allows local variation in temporal prey availability to be exploited (Curio 1976). These aspects of optimum foraging theory will be addressed in relation to the cormorant feeding behaviour at the study sites.

9.2.2 *Cormorant temporal occupancy*

There were three main over-winter night roosts monitored in the study period:

- Attenborough Gravel Pits, Nottingham (SK 520340) (Section 4.5.1);
- Jackson's Bank, Lancashire (SD 620327) (Section 8.6.1);
- Stubbin's Wood, Lancashire (SD 620342) (Section 8.6.1).

The temporal cormorant occupancy at the study sites was related to the numbers present in the night roost (Section 4.5.2; 5.5.2; 5.6.2; 5.7.2; 6.5.2; 7.6.1; 8.6.2). Peak cormorant occupancy at each site occurred between October and March, with low numbers recorded between April and September, with the patterns governed by the breeding behaviour of the birds. Their migration from the study sites in March and April does not appear to be caused by temporal shifts in fish behaviour in the sites, such as decreased shoaling and increased fish activity in warmer temperatures, which may otherwise have forced the cormorants to exploit different food resources because of increased difficulty of capture (Section 9.2.1).

Cormorant breeding colonies are established from approximately mid-March, with egg laying occurring in late April or early May (Cramp and Simmons 1977). These colonies are coastal, situated on rocky cliffs, skerries, stacks and offshore islands (Russell *et al.* 1996) (Section 2.2). Thus, the pattern of low numbers of cormorants in summer at the study roost sites, with decline in numbers beginning in March and numbers increasing from September, was consistent with the breeding migration of mature cormorants to coastal areas (Section 4.5.1, 8.6.1).

9.2.3 *Cormorant diurnal site occupancy*

The main study sites where the diurnal occupancy of cormorants was monitored closely were Holme Pierrepont Rowing Course (Section 4.5.3) and Colwick Park Trout Lake (Section 6.5.3). At Holme Pierrepont, the cormorant numbers peaked in the first hour of daylight, with a subsequent decline in occupancy, before numbers stabilised for the remaining daylight hours. All of the birds on the site were observed to feed there (Section 4.5.3). Therefore, although cormorants did feed over the whole day length at Holme Pierrepont, there was a definite peak in feeding activity in the first hour of daylight. This may have been synchronised with increased fish activity in the lake during the initial period of daylight, making them more susceptible to predation (Mikkola 1970; Rijnsdorp *et al.* 1981). During a 24-hour electric fishing experiment, predatory fish behaviour was also observed to increase in the dawn period, although prey fish activity, for example, roach, was generally greater at dusk (Harvey and Cowx 1995b). It may be assumed cormorants are unable to take advantage of any increased fish activity at dusk because they hunt by sight and are restricted by low light intensities.

A pattern of low cormorant occupancy at first light, a peak around midday and a decline in the remaining light hours was found at Colwick Park Trout Lake in the pre-trout stocking period (October to mid-March) (Figure 6.4). These cormorants used the site primarily as a day roost, with activities such as wing drying and loafing occurring, but little feeding activity. The direction from which the cormorants arrived at the site suggested that they were feeding at adjacent stillwaters, such as Holme Pierrepont, before flying to Colwick Park Trout Lake to day roost (Feltham *et. al.* 1999). Thus, cormorant feeding was probably based on early feeding at an adjacent site, before day roosting at Colwick Park in the remaining light hours. This suggests the existence of a daily behaviour pattern between early-morning foraging to exploit increased fish activity at a profitable site, and day roosting at favourable roosting habitat (Enright 1975). There are a number of large boulders present in the north east corner of Colwick Park Trout Lake which are utilised by cormorants for loafing and day roosting (Section 6.1.2).

Following the initial stocking of the rainbow trout in Colwick Park Trout Lake during mid-March, the diurnal cormorant occupancy of the site changed markedly (Section 9.3.1). Although the period coincided with declining numbers of cormorants using the site due to seasonal migration to breeding colonies (Section 9.3.2), peak cormorant numbers occurred during the first hour after sunrise, rather than around midday, and this was an increase in the number of birds present (Figure 6.4). The number of birds occupying the site then fell in the early afternoon before stabilising throughout the remainder of the day (Figure 6.4). The birds which arrived during the first hour after sunrise all fed on the lake. This showed adaptive behaviour of the daily foraging pattern to exploit local variation in temporal prey availability, in this case the stocking of 2000 rainbow trout at a day roost site (Section 6.1.2, 6.6.3) (Curio 1976).

Therefore, cormorant feeding activity at the sites disagreed with Schafer (1982) and Feare (1988) (Section 2.4), who found no clear daily peak of feeding activity in winter, but showed some agreement with Kennedy and Greer (1988), who found feeding in spring was mainly in the first few hours after dawn. The cormorant foraging patterns may have coincided with the main activity patterns of the fish at the sites (Mikkola 1970; Rijnsdorp *et al.* 1981), resulting in the establishment of a time-related behaviour pattern (Enright 1975), with adaptive foraging behaviour shown to exploit temporal prey availability (Curio 1976).

Peak foraging cormorant numbers were also found in the early hours of day-light on the River Trent, with decreased foraging activity and a switch to day-roosting after this period (T. Holden pers. comm.). Although cormorants were observed to forage and day-roost on the river, its main use was as a fly-way to locate adjacent foraging and roosting areas, such as Holme Pierrepont and Colwick Park Trout Lake (Section 9.4.2).

On the lower River Ribble, peak foraging cormorant numbers occurred in the initial hours of daylight, similar to the Midlands study sites. This was less evident at Grimsargh number 3 Reservoir, perhaps due to the site not being used as a regular food patch by cormorants, as revealed by the low occupancy levels (Section 7.6.1).

9.2.4 *Cormorant feeding success*

There were two main methods utilised to determine cormorant feeding success on the study sites: the percentage of foraging bouts where fish were ingested and the percentage of dives where fish were ingested. However, as the percentage of successful dives was generally low at each study site (<15 %), the percentage of foraging bouts where fish were ingested was a better indicator of feeding success rate (Table 9.6). The proportion of successful dives for roach and rudd by great crested grebes was also shown to be low, where only 3.1 % of dives were successful (Ulenaers *et al.* 1992).

As each site had different habitat characteristics, the different foraging success rates may reflect the ease that cormorants can forage in different habitat types (Section 9.2.1).

Cyprinid, stillwater fisheries

The highest foraging success rates were observed on Holme Pierrepont and Grimsargh number 3 Reservoir (Table 9.6). Both were shallow, stillwater fisheries of low turbidity and limited macrophyte growth. The fish communities and cormorant diet were dominated by cyprinid species. These factors can be related to optimum foraging theory (Section 9.2.1).

Table 9.6 **Percentage of foraging bouts resulting in fish consumption for the study sites, 1995 to 1998.**

Site	Percentage of successful foraging bouts
Holme Pierrepont	58 - 74
River Trent (all sites)	30 - 52
Colwick Park Trout Lake (pre-stocking)	13 - 28
Colwick Park Trout Lake (post-stocking)	33 - 67
Grimsargh no. 3 Reservoir	74 - 86
Lower River Ribble	38 - 56

Although a low standing crop of fish was observed at Holme Pierrepont compared with the River Trent over the study period (Table 9.1), the high density of cyprinids below 150 mm, aggregated under the boat pontoons in winter, provided an attractive feeding patch for cormorants (Section 4.7.6). The turbidity of the water was generally low, allowing the cormorants, sight predators, to easily observe prey fish (Eriksson 1985). Although red-breasted merganser attack on juvenile salmon has been shown to reduce with increased fish cover (Wood and Hand 1985), the fish cover at Holme Pierrepont, provided by the overhanging boat pontoons, did not appear to deter cormorant foraging (Plate 4.1, 4.2). This was because the cormorants were able to dive underneath the pontoons (Section 4.7.6), and the dense fish shoals were probably more vulnerable to cormorant attack than single fish (Bond 1979).

As optimum foraging aims to maximise the net rate of energy intake, cormorant foraging energy expenditure was kept to a minimum at Holme Pierrepont by exploitation of the favourable foraging conditions around the boat pontoons. This increased foraging efficiency and maximised the foraging net energy intake. If fish had shoaled in an alternative area of the lake and were exploited to the same degree, foraging bout success may have been altered, due to the influence of, for example, different depths and available fish refuge. This could affect foraging efficiency and the net energy intake. Indeed, several studies have shown there is a great deal of spatial heterogeneity of fish assemblages within a lake, resulting from variations in habitat structure, which may impact on predatory bird foraging success (Eadie and Keast 1984; Benson and Magnusson 1992; Leslie and Timmins 1992, 1994).

A similar situation was apparent at Grimsargh number 3 Reservoir. Although the standing crop was below that of Holme Pierrepont (Table 9.1), the fish community was composed primarily of slow growing cyprinids of below 150 mm at high density (Section 7.8.3). Water clarity was high at the site and as there was little available cormorant-safe refuge, so any winter shoaling patterns of these fish would have increased their vulnerability to cormorant predation. Therefore, the habitat and fish community structure in the reservoir provided an attractive food patch for cormorants, resulting in high foraging efficiency, which optimised the foraging net energy intake.

Similar to Holme Pierrepont, dense shoaling of fish in confined areas has been observed on other fisheries where large numbers of cormorants forage. At Hornsea Mere, East Yorkshire, where up to 100 over-wintering cormorants foraged daily, dense shoals of cyprinid species were found under boat pontoons in winter 1998/99 (J. Harvey pers. comm.). Seine netting revealed fish density was low in open water habitats (J. Harvey pers. comm.). At Coombe Abbey Lake, Coventry, large shoals of cyprinid fish below 200 mm migrate into the Smite Brook, a small feeder stream of the lake, during winter (pers. obs.). This first occurred in 1996/97, and seine netting of the main lake has also revealed a low winter fish density in open water habitats (A. Starkie pers. comm.). Up to 100 over-wintering cormorants have foraged on the main lake since 1996 (J. Andrews pers. comm.), but few birds forage on the Smite Brook, perhaps because a public footpath runs along its length and the cormorant foraging habitat is poor, resulting from shallow water (< 1m) and narrow channel width (< 6 m) (pers. obs.).

Cyprinid, river fisheries

Cormorant feeding success rates on the River Trent (Section 5.5.2; 5.6.2; 5.7.2) and River Ribble (Section 8.6.3) were similar, with up to 56 % of foraging bouts resulting in fish ingestion. Although the values were below those found on the cyprinid stillwaters, the foraging success rates were reasonably high (Table 9.6).

The fish community of the River Trent comprised primarily cyprinid species of below 200 mm (Section 5.8.1). Their winter shoaling behaviour resulted in a patchy fish distribution along the river, with localised areas of high fish density, for example, Trent Bridge, but with large areas of the river supporting low fish densities (Section 5.8.2). The winter flow rate of the river was high, especially in the period after rainfall, with a general trend of increased turbidity, depth and fish cover compared to Holme Pierrepont over the winter period (pers. obs.).

On the River Trent, 30 to 52 % of foraging bouts resulted in fish ingestion (Table 9.6). Foraging success was expected to be lower, due to the river conditions perhaps not conforming to favourable foraging habitat, particularly in comparison to Holme Pierrepont and Grimsargh number 3 Reservoir. In the river, the patchy distribution of fish and unfavourable turbidity, depth, flow and cover were expected to reduce foraging success and increase energy expenditure during foraging.

Notwithstanding the latter argument, cormorants have been observed to forage successfully in depths to 19.8 m in coastal conditions (assuming benthic foraging occurred), where tidal currents would have been considerable (Cooper 1985). As water depth in the River Trent rarely exceeds 5 m, then depth may not be a limiting factor to foraging success, although an increased energy expenditure would be incurred during deep dives (Ross 1974). Although turbidity will decrease the sight range of a cormorant underwater, this may be offset by the reduced range of predator detection by fish (Schafer 1982). Water flow was likely to cause increased energy expenditure of a dive, for the cormorant has to maintain its movement and speed through the flowing water during fish pursuit. Additionally, the areas of large winter fish shoaling may have increased forging bout success due to the shoals being more conspicuous than single fish (Bond 1979). Therefore, the foraging success was reasonably high, suggesting specific areas of the river provided profitable food patches for the cormorants, especially during winter aggregation of fish.

An important aspect of the River Trent as a foraging site for over-wintering cormorants was its availability as a foraging site during inclement weather when the stillwater fisheries were frozen over and inaccessible to cormorants. Increased numbers of foraging cormorants were observed on the river when Holme Pierrepont was frozen over for extended periods (T. Holden pers. comm.). This is adaptive foraging behaviour due to temporal loss in prey availability in the usual foraging sites (Curio 1976).

Irrespective of this, some reduction in foraging success was found between rivers and stillwaters, presumably because of the factors outlined. However, it should be recognised that the intensity of cormorant foraging was much lower on the river habitats and they were not the primary foraging locations of the birds (Section 9.4.2).

In terms of optimum foraging theory, although the River Trent habitat would have increased cormorant foraging energy expenditure compared with the stillwater sites because of the river's increased depth and flow, its turbidity and cover. Nevertheless, this did not reduce foraging success to very low levels on the river, perhaps due to the winter fish shoaling areas providing adequate foraging patches. Although a similar situation was expected on the River Ribble, field observations were too few to clarify this.

Put-and-take trout fishery

Cormorant feeding success rates at Colwick Park Trout Lake in the pre- and post-trout stocking periods showed a clear difference, with increased foraging bout success observed post-stocking (Section 6.5.4).

In the pre-stocking period, trout stocks were considered to be very low due to over-winter mortality of fish stocked the previous season. As the standing crop of other species in the lake appeared low, and the lake is large (25 ha) with depth variations of 2 to 6 m (Section 6.1), fish location by foraging cormorants may have been very difficult, despite low water turbidity (pers. obs.). As only 13 to 28 % of foraging bouts resulted in fish being ingested, Colwick Park Trout Lake provided an unprofitable food patch to cormorants in this period, with potentially high energy expenditure during foraging.

The introduction of 2000 rainbow trout into the lake in mid-March resulted in cormorants utilising the site for foraging (Section 9.3.2). Although the lake size and depth may have decreased foraging success, the initial behaviour of the trout in the lake, with shoaling in marginal areas and the presence of fish on the surface (pers. obs.), resulted in an increased foraging success rate (33 to 67 %). Thus, the naïve, hatchery-reared trout improved the site for cormorant foraging efficiency, which in turn was likely to increase the foraging net energy intake.

Overall, the cost-benefit relationship between fish availability and fish intake, in terms of foraging bout efficiency, cannot be explained in terms of fish abundance alone. Habitat characteristics and fish community structure were more important than fish abundance in determining foraging efficiency. Habitats favouring the shoaling of small fish, in water of low turbidity and minimal cover, created the most profitable food patches for cormorants. The impact of factors such as turbidity, flow, depth and patchy fish distribution on the foraging efficiency of cormorants was more difficult to determine, with the river sites apparently providing adequate foraging efficiency returns.

9.3 Cormorant predation on the fish populations in the study sites

9.3.1 *Species selection by feeding cormorants*

Cormorants are opportunistic feeders, with the main prey species usually being the locally dominant species (West *et al.* 1975; Welsh Water Authority 1980; Van Eerden *et al.* 1995) (Section 2.6.1). The comparison of cormorant diet with the fisheries surveys on the study sites allowed this relationship to be explored further (Table 9.7). Cormorants did not show selection for some fish species during foraging, i.e. their proportion in the diet was in proportion to their presence in the fish community structure (Table 9.7). For other species, there was positive or negative selection, i.e. their proportion in cormorant diet was over or under their proportion in the fish community (Table 9.7). Cormorant species selection data at Holme Pierrepont enable more detailed analysis, and is discussed later.

Table 9.7 Prey selection of foraging cormorants at specific study sites.

Fishery	Proportionately selected species	Positively selected	Negatively selected
Beeston, River Trent	Cyprinids, perch		
Trent Bridge, River Trent	Cyprinids	Perch	
Stoke Bardolph, River Trent	Cyprinids	Perch, eel	
Grimsargh no.3 Reservoir		Cyprinids	Perch, pike eels
River Ribble	Cyprinids, eels		Trout, perch

- Where cyprinid species were available in high proportions in the fish population, predation by feeding cormorants was high (Table 9.7). Cyprinids were not selected against in preference to any other species at any study site.

- Perch and eel selection by feeding cormorants was site specific and, hence, may be dependent on the availability of other prey species (Table 9.7).

- Cormorants appeared to select against trout when feeding on river fisheries, with cyprinid species providing a buffer against trout losses (Table 9.7).

In terms of optimal prey selection (MacArthur 1972; Charnov 1976a), the 'prey types' preferred by cormorants were cyprinid species in the relevant sites. It may be assumed these were the most profitable prey types, as cyprinids were dominant in the fish communities, which would result in a high encounter rate during foraging (Suter 1997). Their body size, shoaling and defence mechanisms may have also made them more vulnerable to cormorant predation (Bond 1979; Wootton 1990; Ulenaers *et al.* 1992). The unprofitable prey items (those not present in such high densities in the fish community making it unprofitable to always select them, for example, perch) were only likely to have been taken when the encounter rate with the cyprinid species was reduced (MacArthur 1972; Charnov 1976a).

At Holme Pierrepont, the identification of cormorant predated fish was possible to species level, allowing a more accurate assessment to made of the relationship between cormorant feeding selection and the available fish populations. The cormorant exploitation patterns varied between years and showed evidence of selection (Table 9.8).

The proportion of roach predated on by cormorants at Holme Pierrepont was consistent over the study period, and, hence, roach provided a consistent food source for the birds (Table 9.8). However, the proportion of roach was greater in the fish community than ingested by cormorants. Predation on common bream increased in successive survey years, and their proportion in cormorant diet was greater than in the fish community. However, this may be due to gear inefficiency in capturing common bream (Section

4.4.4). The proportion of perch in cormorant diet decreased with time whilst the proportion of perch in the fish community increased.

Table 9.8 Prey selection by cormorants in Holme Pierrepont compared with their proportion in electric fishing surveys (Corm = cormorant selection, Fish = proportion in the electric fishing surveys).

	1995/96		1996/97		1997/98	
Species:	**Corm** (%)	**Fish** (%)	**Corm** (%)	**Fish** (%)	**Corm** (%)	**Fish** (%)
Roach	34	62	28	40	40	70
Common bream	3	2	33	24	42	6
Perch	63	6	39	7	19	21

Although no explanation for the shift in diet was apparent, species selection by cormorants during foraging may be dependent upon the proportion of the fish species in the cormorants' specific feeding patches, rather than in the fishery as a whole. This is determined by the spatial heterogeneity of fish assemblages in the lake, resulting from variations in habitat structure (Eadie and Keast 1984; Benson and Magnusson 1992; Leslie and Timmins 1992, 1994) (Section 9.3.4). It was also likely to be influenced by other factors, including winter migratory patterns, fish shoaling behaviour and water turbidity (Bond 1979; Schafer 1982; Eriksson 1985; Wood and Hand 1985; Wootton 1990; Ulenares *et al.* 1992; Suter 1997).

At Holme Pierrepont, fish were known to migrate between the Rowing Course and the water ski lagoon (Section 4.1.3), and the areas around the inflows of the Bolster Brook and the River Trent (Section 4.1.3) may have provided an alternative, attractive winter habitat for fish. Thus, alternative winter fish shoaling areas to the boat pontoons were available which may have affected the cormorant diet selection. The turbidity of the water was also very variable in winter, being determined by the turbidity of the water entering the lake from the River Trent and Bolster Brook (pers. obs.) and, hence, by local rainfall. This may have affected cormorant diet composition, as it may have impacted on the cormorant foraging efficiency for different fish species, and the habitat utilisation of the prey species. However, with the exception of winter 1995/96, the cyprinid species (64 and 76 % of all species ingested) were more vulnerable to cormorant predation than perch (39 and 19 %).

The dominant fish species ingested by foraging cormorants in Swiss lakes were roach and perch (Suter 1997). The availability of roach, particularly when aggregated in large shoals, was important in determining foraging habitat. This selection in diet was assumed to be due to the numerical dominance and shoaling behaviour of roach, since non-shoaling alternative species were rarely ingested (Suter 1997). This was also shown at Holme Pierrepont, where the dense over-winter shoals of cyprinid species under the boat pontoons were exploited heavily over the study period (Section 9.4.1). On inland fisheries throughout Europe, cormorants have been shown to prefer highly gregarious fish for ingestion, probably because their shoaling allows social foraging by cormorants to be successful (Voslamber 1988; Platteeuw *et al.* 1992; Veldkamp 1994; Dirksen *et al.* 1995; Van Eerden and Voslamber 1995). This suggests cyprinid species were the preferred prey types in cormorant diet in the relevant study sites due to their dominance in the community, coupled with their shoaling tendencies, rather than direct preferential selection (Bond 1979; Suter 1997).

The stock manipulation experiment on Grimsargh number 3 reservoir was shown to be relatively unsuccessful due to the high numbers of cyprinids already present in the fishery. However, the annual stocking of rainbow trout into Colwick Park Trout Lake during March and April revealed how stock manipulation can have a major effect on the prey selectivity of cormorants, as well as in cormorant diurnal site occupancy (Section 9.3.2) and feeding success rates (Section 9.3.3).

Between October and February in each study year, only a small number of over-wintering rainbow trout were ingested, with the main prey species being perch and unidentified fry. However, when the rainbow trout were stocked initially during March, with the change in cormorant diurnal use of the site and the increase in feeding success rate, they became the only species ingested (Section 6.6.1). Cormorants in this instance became opportunistic feeders, switching their diet to take advantage of an easy food source, similar to patterns observed in other studies (Marquiss and Carss 1994; Kirby *et al.* 1996).

9.3.2 *Species size selection by feeding cormorants*

At the study sites, cormorants ingested fish in the size range 50 to 500 mm (Table 9.9). The most vulnerable cyprinid and perch sizes were 50 to 100 mm and 50 to 200 mm respectively, depending on the site. In other studies, the size of fish ingested by cormorants ranged from 30 to 650 mm (Section 2.6.1), with the majority in the range 100 to 300 mm (Marquiss and Carss 1994).

In terms of species and size selection, fish of below 200 mm were the preferred 'prey type' of cormorants (MacArthur 1972; Charnov 1976a) at the study sites (Table 9.9). This suggests that these species and size of fish provided the maximum energy cost-benefit returns for the foraging cormorants. This was probably due to the fish communities in the sites being dominated by these fish, resulting in a high encounter rate (Suter 1997). However, additional factors, such as decreased handling time after capture, their shoaling characteristics making them more vulnerable to cormorant attack than single fish (Bond 1979; Suter 1979), and their slower escape speed compared with larger fish (Bond 1979), may have also contributed to their prey selection.

The main sizes of trout ingested at Colwick Park Trout Lake were 250 to 300 mm (Figure 6.2; Table 9.9). These were the smaller sizes of the trout stocked at Colwick Park Trout Lake, suggesting size played a role in determining their vulnerability to cormorant predation. Measured sub-samples of trout prior to stocking revealed that trout of below 300 mm comprised less than 2 % of the stocked fish (Figure 6.2). Unless an error was made in the trout size estimation taken by cormorants, this suggests that cormorants were feeding selectively on a small proportion of the trout stock in the lake. Predatory animals prefer 'prey types' in terms of species and size, according to their rank order of profitability, defined as net energy yield per unit handling time (MacArthur 1972; Charnov 1976a). This suggests the smaller portion of the trout stock were selected by cormorants due to their decreased handling time making them easier to swallow after capture. Additionally, the smaller trout may possess relatively poor predator response and escape (pre- and post- capture) tactics compared with larger fish, so increasing their vulnerability to predation.

Table 9.9 **Dominant (Dom.) and maximum (Max) sizes (mm) of species ingested by cormorants at the study sites.**

	Cyprinid		Perch		Other species		
	Dom.	Max	Dom.	Max	Species	Dom.	Max
Holme Pierrepont	50-100	250	50-100	300			
Beeston, R. Trent	50-200	250	50-200	250			
T. Bridge, R. Trent	50-200	250	50-200	200			
S. Bardolph, R. Trent	50-100	250	50-150	200			
River Ribble	100-200	400			Eel	200-350	500
Grimsargh no. 3 reservoir	50-150	400					
Colwick Park Trout Lake					Trout	250-300	450

9.3.3 *Non-lethal cormorant predation impacts*

Non-lethal impacts, including wounding, population structure alteration, feeding habitat use and behaviour of fish populations in response to cormorant and/or general piscivorous predator presence, were observed on affected fisheries (Section 2.9).

Wounding

During a period of predation, cormorants will capture a number of fish which then escape, perhaps due to escape mechanisms or being too large to swallow, leaving a characteristic deep triangular wound or abdominal hole on one side of the fish and an area on the other side where scales have been scraped off by the lower mandible (van Dobben 1952; Ransom and Beveridge 1983; Carss 1990a; Davies *et al.* 1995). A number of these fish are likely to die through wound severity or secondary infection of the wound (Russell *et al.* 1996). However, a proportion of wounded fish have been shown to survive (Suter 1995). Size was shown to be a factor in determining wounding frequency, with wounded individuals of greater size than the average size of fish in the populations (Davies *et al.* 1995). The wounding observations noted on the study sites were:

- Cormorant damage was noted on the fish populations present in the River Ribble and Colwick Park Trout Lake. Very little damage attributable to cormorants was noted on fish from other sites.

- At Colwick Park Trout Lake, 11.4 % of stocked trout were wounded in the period of cormorant predation (Section 6.6.5). The wounded fish were of a higher average size than those stocked.

- At Great Mitton, River Ribble, wounding was observed in the chub (25 %) and dace populations (33 %), but the brown trout populations were unaffected (Section 8.7.4). The wounded fish were of a slightly greater average size than healthy individuals. As the section is utilised as a salmonid fishery, then the presence of the cyprinid species probably has a buffering effect on the salmonids from cormorant predation (Section 9.2.3).

When related to habitat variables, the two completely different habitat types in Colwick Park Trout Lake (Section 6.1.2) and the River Ribble (Section 8.2) produced high wounding frequencies in different fish species. There are two possible sources of cormorant wounding: that incurred pre-capture, where wounding occurs due to the cormorant not being able to capture the fish successfully, perhaps resulting from poor water visibility, and that incurred post-capture, where wounding occurs when the cormorant is unable to swallow the fish due to escape tactics of the fish (Recher and Recher 1968), or it is too large to swallow. The severity of wounds observed in trout at Colwick Park suggest wounding occurred post-capture, for abdominal puncture holes were often observed (pers. obs.; Plate 6.2). However, at Great Mitton, River Ribble, results were inconclusive. As the river was relatively turbid and with fast flowing glides, this habitat may have resulted in 'near-misses' by foraging cormorants, causing the wounding observed. However, the increased average size of wounded fish suggest post-capture escape also occurred due to the fish being too large to swallow.

Thus, where cormorant wounded fish were observed in the study sites, a notable percentage of the populations was wounded, and the wounded fish were of greater average size than the resident populations, a trend observed in other studies (Davies *et al.* 1995, Suter 1995).

Fish population structure

Brönmark *et al.* (1995) found in predator absence, fish populations were dominated by small bodied individuals in high densities and in predator presence, the populations consisted almost exclusively of large individuals. This observation was not found on any of the study sites. However, cormorant predation on rainbow trout in Colwick Park Trout Lake was shown to be size selective. The majority of cormorant predated trout were 250 to 300 mm in length, compared with the majority of the stocked fish being of 300 to 450 mm (Section 6.6.2). Hence, if cormorant predation had been prolonged on the fishery, a population consisting only of the larger stocked individuals may have occurred. Additionally, cormorant predation on Holme Pierrepont resulted in an increase in growth rates of the fish species, due to decreased competition (Section 9.1.3). This fundamental, adaptive response of the population to predation has altered the length-age structure of the fish community, with fish attaining greater lengths for age in the period after cormorant predation.

Feeding habitat use

Habitat utilisation by prey populations often changes in predator presence, with associated decreases in foraging range and an increased use of hiding places (Caraco *et al.* 1980; Dill and Fraser 1984; Lendrem 1984; Fraser and Huntingford 1986; Magurran and Pitcher 1987). This may result in reduced fish growth rates due to decreased foraging returns (Fraser and Cerri 1982; Werner *et al.* 1983; Holopainen *et al.* 1991; Tonn *et al.* 1992; Fraser and Gilliam 1992) (Section 2.9.3). It was not possible to show this non-lethal impact caused by cormorant predation on the study sites. However, at over-wintering, cormorant-affected, temperate fisheries, this non-lethal impact may be considered negligible, for fish growth is generally restricted to the summer period only.

This is shown by the formation of annual growth checks (annuli) on temperate fish scales during the period of slow winter growth (Section 3.5.3; Figure 4.10, 4.11). It was also found that on Holme Pierrepont, where cormorant predation impact was high, cormorant predation increased the scope for growth in the affected populations due to decreased competition (Section 9.4.1).

Fish behaviour

Prey aggregation is a behavioural response to predator attack (Neill and Cullen 1974; Poole and Dunstone 1990; Milinski 1979; Magurran and Pitcher 1987), with studies indicating aggregated prey have greater individual survival than solitary prey (Radakov 1973; Neill and Cullen 1974; Morgan and Godin 1985; Pitcher 1986). However, Bond (1979) suggested fish shoals are more vulnerable to cormorant attack. Large aggregations of fish were observed at two study sites where cormorants were observed to feed, Holme Pierrepont and Trent Bridge, River Trent.

The aggregation of fish in the Trent Bridge area of the River Trent during the winter months was thought to be a general over-wintering migration of fish into the area, with fish possibly remaining to spawn in spring prior to dispersal (Section 5.6, 5.8). This migration was observed for a number of years on the river and was not a result of cormorant predation. It appears to be due to the increased water clarity of the river, caused by decreased suspended solids in the water (Section 5.1.2), and decreased winter water temperatures, resulting from power station decommissioning (Section 5.1.3). This has increased the shoaling behaviour of the fish. Cormorants have taken advantage of the situation by feeding on the large shoals of vulnerable-sized fish and may contribute to the fish shoaling behaviour, despite their presence not being the motivating factor.

The aggregation of fish at the boat house end of Holme Pierrepont Rowing Course (Plate 4.2) in the winter months has been occurring for a number of years (B. Pluckrose pers. comm.). However, the shoaling behaviour of the fish in this area of the lake observed in winter 1997/98 was considered to be extreme. Fish, including individuals over 200 mm, were only caught by electric fishing gear when the electrodes were placed underneath the boat pontoons (1 m deep) and into small concrete cracks present in the lake margins (< 0.5 m deep) (Section 4.6.7). Cormorants were observed to forage successfully by diving underneath the pontoons (Section 4.5.2, 4.6.7; Table 9.4). Thus, although the over-winter migration of fish into the boat house area of the lake was due to natural factors, the extreme shoaling behaviour of fish into small refuge areas observed in winter 1997/98 may have been a defence mechanism against cormorant predation. However, with no baseline fish observations on the over-winter fish migrations of Holme Pierrepont in the period prior to cormorant predation, it is not possible to confirm this interpretation.

9.3.4 *Foraging hotspots*

Two cormorant foraging 'hotspots' were identified in the study:

- Holme Pierrepont boat pontoons;
- Colwick Park Trout Lake, after initial stocking in March.

A feature present at both hotspots was the presence of large shoals of accessible fish to cormorants. At Holme Pierrepont, the large number of cyprinids tightly shoaled under the boat pontoons (Plate 4.2) in the winter periods allowed easy feeding for cormorants, resulting in high cormorant foraging rate success (Section 4.5.2, 4.6.7; Table 9.4) and large numbers of fish ingested (Section 4.6.4). The boat pontoons offered the fish cover from above the water, but offered no vertical protection in the water column from diving cormorants.

At Colwick Park Trout Lake, a different situation was observed. It was the initial introductory behaviour of the naïve stocked rainbow trout which made them highly vulnerable to cormorant predation. On introduction to a fishery, hatchery-reared trout are known to exhibit disorientation, shoaling behaviour and a tendency of utilising areas close to the bank (M. Weevers pers. comm.). Thus, when stocked into Colwick Trout Lake, where all the trout were introduced in one area, an artificial cormorant feeding hotspot was created, enabling the cormorants to feed efficiently with minimal effort. This demonstrates that cormorant predation on these rainbow trout could be decreased if stocking practices are improved (Chapter 10).

9.4 Impact of cormorant predation on the fish populations of the study sites

Assessing the impact of cormorant predation is a highly subjective issue (Section 2.7), with assessment often dependent upon the views of those involved. However, in this study, robust assessment may be made by integrating fisheries and cormorant data.

9.4.1 *Holme Pierrepont*

At Holme Pierrepont, the MCS data were considered satisfactory. Although the estimated fish losses due to cormorant predation were high (Section 4.6.4), this was supported by the high numbers of cormorants that fed daily on the fishery (Section 4.5.2). The electric fishing survey data may be considered as robust, with fish abundance estimates supported by hydro-acoustic data from the Environment Agency (Midlands Region) and angler catches.

The integrated MCS and cohort analysis data revealed the total fish abundance in cormorant presence decreased by 59 % after three years of predation. This equates to an increased fish abundance of 246 % in the absence of cormorants, excluding potential changes in density-dependent natural mortality. This can only be considered a damaging impact. These losses would be expected to result in large changes in the fish population dynamics and life history strategy.

Fish life history strategies aim to maximise the survival of their offspring and are controlled by the fluctuations between birth and death rates, and the availability of resources (Pitcher and Hart 1982). These give rise to the two distinct life history strategies of r and K (MacArthur and Wilson 1967). r-strategists rely on their ability to colonise new habitats, make use of short-lived resources and maximise fitness by improving their ability to reproduce rapidly in an uncrowded environment (Pitcher and Hart 1982). K-strategists live in stable environments where it is important for organisms to persist and out-compete rivals by subtle behavioural means, and the main controlling factors are biological (Pitcher and Hart 1982). The characteristics associated with each type of strategy are shown in Table 9.10 (Pianka 1970, 1978).

However, few animals are pure r- or K- strategists, but lie in between (Pitcher and Hart 1982), with fish life history strategies continuous between r and K, depending on environmental conditions. This was shown by bullheads in upland (optimum habitat) and lowland streams (poor habitat) (Fox 1979). In upland streams, a K-strategy was evident in the bullheads, with slow growth, breeding after 2 or 3 years and a maximum age of 6 years. In lowland streams, an r-strategy was evident, with fast growth, breeding after 1 year and a maximum age of 2 years. Hence, considerable phenotypic plasticity in life history traits are seen in fish populations (Wootton 1990). This is an important adaptive trait allowing individuals to respond to environmental and population changes during their lifetime (Wootton 1990).

Table 9.10 Characteristics of r- and K- selection.

Characteristic	r selection	K selection
Habitat	Variable and/or uncertain.	Constant and / or predictable.
Niche	Broad.	Narrow.
Mortality	Density independent, catastrophic.	Density-dependent.
Population size	Variable in time; usually below carrying capacity.	Fairly constant in time; at or near carrying capacity.
Intra- and inter-specific competition	Variable.	Intense.
Selection favours	1. Rapid development. 2. Early reproduction. 3. Short longevity. 4. High fecundity.	1. Slower development. 2. Delayed reproduction. 3. Long longevity. 4. Low fecundity.

The response of fish populations to exploitation can be related to the life history characteristics (Wootton 1990). Exploitation has been shown to artificially impose r-selection on populations, as shown in exploited fish populations by increased growth and fecundity (Alm 1959; Barret 1971), decreased age at maturity (Begon *et al.* 1989; Wootton 1992), and high natural mortality (Pauly 1980). Reduced age of maturity and increased pregnancy rates were observed in exploited whale and seal stocks (Estes 1979).

This suggests the effect of the cormorant predation on the cyprinid populations of Holme Pierrepont was to err towards r-selection in their life history strategy. This had the effect of compensating for the fish losses due to cormorant predation by rapid development and early reproduction, with short longevity (Table 9.10). This was observed in the fish population dynamics as growth rates increased markedly after 1993/94, when cormorant predation was first observed on the fishery (Figure 4.20 to 4.23). The data suggest different population characteristics were followed prior to cormorant predation due to a slower growth rate (Figure 4.20 to 4.23). This would have increased the age of maturity and reduced fecundity at age (Table 9.10). Although both life strategies were r-selected, growth rates increased after the cormorant predation as the fish populations were no longer producing huge numbers of slow growing individuals.

The compensation processes were only thought to manifest into the population when strong year classes of fish were produced, for example, 1995 and 1996 (Section 4.7.3). When cormorants predate on a series of weak year classes, the compensation process may not be seen due to the removal of a large proportion of the weak year class. This leaves few surviving fish to exploit the low competitive environment. However, as fish density declines, a point may exist when the net energy intake of cormorant foraging is reduced to a level where the food patch becomes unprofitable to exploit. This would result in cormorant predation declining on the fishery, so increasing the scope for adequate compensation in the fish populations as losses would be reduced. Further research is required to determine the relationship between fish density and foraging efficiency.

Compared with 1994 to 1997, the angler CPUE was high at Holme Pierrepont in 1998, despite cormorant predation (Section 9.1.4), and was supported by the strong 1996 year class of roach. Future problems may arise when a series of weak year classes are produced in the fishery and are subsequently reduced further in number by cormorant predation. These cohorts may not adequately replace the 1996 year class of roach when they die out through natural mortality and/or cormorant predation. As the year class of 1998 is likely to be weak due to poor climatic conditions in their first growth year (Section 4.7.3, 9.1.4), angler catch rates in 2000 and 2001 may show a large decline compared with 1998.

Thus, the impact of cormorants on the fish populations of Holme Pierrepont revealed:

- a large number (50 to 90) of over-wintering cormorants fed daily on the fishery;

- an overall reduction of 59 % in the fish population in the presence of cormorants;

- compensation mechanisms in the fish population dynamics, resulting from an intense r-life strategy, limited the predation damage, although this was aided by strong year classes in 1995 and 1996;

- angler catches improved from 1994 to 1998, with CPUE 1995 to 1998 in excess of results from the river fisheries studied. The angler catches appeared to be strongly linked to roach year class strength (Figure 4.6).

The shift in growth rate of the cyprinid species was observed to first occur in the period 1993/94, prior to the commencement of the study. Thus, other factors may have also contributed to the changes in the fish population dynamics which must be explored to ascertain whether cormorant predation was the sole cause of the growth shift. These include:

- improvements in effluent discharge in the 1990s from Radcliffe Sewage Treatment Works into Bolster Brook, which feeds in to the lake. This resulted in a decline in organic material and suspended solids entering the lake, causing a decline in water turbidity and nutrient input;

- toxic blue green algae blooms in the early 1990s resulted in the management addressing the problem by placing bales of barley straw, contained within a mesh enclosure, around the lake margins. When the straw rots, it releases chemicals which inhibit algae growth. The effectiveness of this approach has varied, with success in some waters and failures in others (P. Buckland pers. comm.). However, it is a factor that may have caused the water clarity to increase further;

- a high level of angler effort in the period 1990-1994 may have caused high fishing mortality.

The decrease in nutrient input may have adversely affected the fish populations, as was observed in the River Trent (Chapter 5), by altering the abiotic and biotic conditions in the lake, for example, food supply. However, it is unlikely that this would have caused the large decline in the fish populations with a subsequent large increase in growth rate. The decline in turbidity may have increased the feeding efficiency of the cormorants by improving underwater visibility. The control of algal growth is also unlikely to have been the cause, but would also have improved cormorant feeding habitat due to the decreased water turbidity. The large volume of angler effort on the lake, with keepnets being used on a daily basis, with likely repeated capture of fish (Raat 1985, Cowx 1991), may have caused a large number of related fish deaths. The site groundstaff reported large numbers of dead fish following angling matches on the fishery (M. Thompson pers. comm.). However, as this is an immeasurable variable and is only available as anecdotal evidence, it cannot be used as strong evidence to explain the large decline in fish population density. Hence, it would appear that cormorant predation was a major contributor to the decline and resulted in the low population of fish which was then able to demonstrate excellent growth in a low competitive environment.

Cormorant predation was the principal cause of the decline in fish density and angler catches since the 1994/95 angling season. The year prior to cormorants first being observed feeding on the lake, 1993, saw a financial income from angling revenue of £20 266.60 (Table 4.1, Figure 9.1), although this was reduced from £28 447.10 in 1992. Following the poor standard of angling observed in the World Championships of 1994, Holme Pierrepont received a large amount of bad publicity in the angling press (Section 4.1.2). This coincided with the income from angling in 1995 declining to £304.50, a reduction of 98.5 % since 1993 (Figure 9.1). Although the income in 1998 increased to £2789.10, this still represented a 86.2 % reduction in financial returns from the fishery since 1993 (Figure 9.1). Hence, a substantial loss of angler revenue was a further impact attributable to cormorants on the fishery.

9.4.2 *River Trent*

The angler catch data for the River Trent sites in the period of study revealed good CPUE and a high percentage of anglers with catch compared with historical values (Section 5.7.1). However, winter angling results were very poor. This was a result of an improvement in water quality causing increased water clarity, and power station decommissioning causing a more natural temperature regime (Section 5.7.1). This decreased angler success in periods of low water temperature. Temperature has also been shown to be an important factor on angler success on other UK river fisheries (North 1980; Section 9.1.2).

The MCS estimates of fish losses attributable to cormorant predation were inaccurate for the River Trent study sites due to over-estimates of fish losses. This was shown by relatively stable fish population dynamics and angler catch rates over the period of study (Section 5.8.3). The fish populations of the river were estimated to have been reduced by up to 98 % over the three-year period (Table 9.11). Such losses would be expected to impact heavily on angling success due to angler catches being a function of stock size and community structure (Crisp and Mann 1977; O'Grady 1980; Pawson 1982; North

1983; Jacklin 1996). However, this was not observed in the River Trent fishery (Table 9.2).

The over-estimation of fish losses in the MCS model was probably due to limitations in the model. These arise from the assumptions that were determined for estimating the distributions of N_{min} and N_{max} (the number of birds observed flying over the River Trent sites that actually fed there) in the model (Section 5.4.3; Feltham *et al.* 1999). These assumptions were established due to the dynamic nature of cormorant feeding site selection in the Midlands region, with likely daily multi-site feeding and use of the river as a 'fly-way' to locate adjacent stillwaters (Section 5.4.3; Feltham *et al.* 1999). If cormorants were observed utilising only the River Trent for feeding, then the model would have estimated a more realistic distribution of N_{min} and N_{max}. However, cormorants were observed visiting a number of sites over the day-light period, but to assess the actual feeding sites of those birds would require marking birds.

Table 9.11 Reduction in numbers of fish due to cormorant predation by year and site, River Trent, 1995 to 1998.

	Year of survey		
	% reduction in total number of fish by cormorants		
River site	**1995/96**	**1996/97**	**1997/98**
Beeston	18	47	34
Trent Bridge	21	83	73
Stoke Bardolph	74	82	98

The assumptions of the model to estimate N_{max} were that 90 % of all flying and roosting cormorants observed at sites B and C were feeding/had fed at other sites, and 50 % at sites A and D (Section 5.4.3; Feltham *et al.* 1999). These were based on cormorant observations over a three-winter period and were deemed to be as accurate as possible given the nature of the work (Feltham *et al.* 1999). However, they have little scientific basis and it would appear they have caused the model to over-estimate the amount of fish removed from the river. This was shown by the estimated reduction in fish numbers due to cormorant predation in Section D (Stoke Bardolph), where 50 % of observed cormorants were assumed to feed (Section 5.4.3; Feltham *et al.* 1999). Total fish numbers were reduced by 98 % after three winters of predation (Section 5.7.3), yet angler catch rates were relatively stable in the same period (Section 5.7.1). In this instance, the over-estimates are such that the data cannot be applied to the real situation with any degree of confidence.

Accepting that an appreciable fish loss due to cormorant predation did occur on the river, elucidation of their impact is very difficult. A wide range of biotic and abiotic factors influence fish abundance and community structure in a river fishery before cormorant predation can even be considered (Figure 2.2), as is shown by the effects of temperature on year class strength production and growth rates in the River Trent (Section 9.1.3, 9.1.4). An example of the complexity in understanding freshwater fish production is also shown by the decline in chub catches observed in the River Trent (Section 5.7.1). A similar decline was also found in a number of other English rivers, including the Warwickshire Avon, where a series of weak chub year classes in recent years has not replaced the strong chub year classes of 1969 and 1976, which have declined due to natural mortality (Starkie 1993). The reasons for the repeated weak year class production of chub in the 1990s are unknown, but are unlikely to be related directly to water temperatures, due to the large variation in annual water temperatures observed in the period (Section 5.7.1).

Therefore, in the River Trent study sites, the cormorant predated losses were considered to be over-estimated. Evidence was provided by stable fish population dynamics and consistent angler catches in the period of study when large losses of fish to cormorants were being estimated by the MCS model (Table 9.8). It has to be acknowledged that cormorants were removing a proportion of the fish stock, but elucidation of their impact was very difficult due to the large number of inter-related factors governing fish production in river systems (Figure 2.2). It is probable these factors, such as variation in annual temperature and winter flow rates, will have a greater impact than cormorant predation on the annual fish production in the River Trent.

9.4.3 *Colwick Park Trout Lake*

Cormorant impact on Colwick Park Trout Lake was relatively simple to determine when compared with natural fisheries such as the River Trent. The main period of predation was confined to the post-stocking period of March and April, before cormorant breeding dispersal. A total of 16 to 19 % of all stocked trout in this period were removed by the cormorants at a replacement cost of between £414 and £516 per annum. Thus, a financial loss and a reduction in fish stock were the predation impacts.

Evaluation of the impact model highlights the limitations of the MCS model in estimating cormorant predation at Colwick Park Trout Lake. Use of the polynomial curve (3^{rd} order) on the bird count data for the lake to estimate the daily number of birds on the lake (Section 6.3.1) demonstrates how over-estimations occur (Figure 9.3). The number of birds was over-estimated on certain days of the month, resulting in an over-estimation in the number of fish ingested (Table 9.12). Thus, the results of the MCS model must be treated with caution. However, this is an unavoidable limitation of the model, occurring because of data being collected on adjacent study sites in the same period.

A further impact on these stocked trout was non-lethal wounding by cormorants. The average wounding rate was 11.4 % of all stocked fish in the period of cormorant predation (Section 9.3.3). This figure was considered accurate as it was calculated from stocking and fisheries data and did not utilise the MCS model. The wounding will have a further negative effect on the fishery because anglers are unlikely to want to capture trout bearing wounds from a cormorant attack.

9.4.4 *Grimsargh number 3 Reservoir*

Cormorant predation impact was relatively simple to determine in Grimsargh number 3 reservoir. The presence of large numbers of slow-growing roach and common bream in the fishery suggested the observed level of cormorant predation had little impact on fish abundance in the reservoir (Section 7.7). Indeed, increased cormorant predation may have a beneficial effect on the fishery by reducing the fish population numbers, as it may allow the growth rates and longevity of the fish to increase due to decreased competition, providing the food base was sufficient to support the fish population.

Anecdotal evidence suggested that historical cormorant predation was high on the fishery and caused a large decline in the fish stocks and angler results (N. Watson pers. comm.). If this occurred, then the compensation processes limiting damage observed in

Holme Pierrepont were not apparent in the fish populations of Grimsargh number 3 Reservoir. However, the high numbers of slow growing fish, rarely exceeding 3 years of age, may be related to high juvenile survival resulting from initial low competition for food from older fish which may have been removed by the cormorants prior to the study (Section 9.2.3). Additionally, the slow growing, small fish did not provide good angler target species, which resulted in anglers complaining of poor catches, despite a high fish density in the reservoir (Section 7.8.3). This was also observed on the River Thames at Reading, where anglers complained of poor catches, despite a high biomass in the river (Williams 1967). It was shown that the fish community was comprised of a high proportion of stunted fish below 100 mm, providing anglers with a low biomass of target species (Williams 1967).

Table 9.12 Comparison of the observed and estimated number of cormorants foraging at Colwick Park Trout Lake during March 1996 (shading = days of over-estimation of numbers).

Date (March 1996)	Number of cormorants observed in count	Number estimated by polynomial curve (3rd order)
12	38	39
13	39	37
14	36	35
15	37	33
16		31
17		29
18		27
19		25
20	16	23
21		21
22		19
23		17
24		15
25		13
26	14	12
27		11
28		10
29		9
30		9
31		9

9.4.5 *River Ribble*

The fish population data for the River Ribble were poor, due to a number of factors (Section 8.2). As a result, a cohort analysis model could not be constructed. The MCS data were used to compare estimated losses to the biomass of fish observed in the river from the limited fisheries surveys. Thus, rather than demonstrating the number and biomass of fish that had been reduced by cormorant predation over the three-year period, as observed for Holme Pierrepont and the River Trent, this method could only show the fish biomass removed on an annual basis, with no accounting for the fish removed in previous winters.

Cormorants were estimated to remove 29.8, 13.9 and 30.1 % of the standing crop of fish species in Section 2 of the River Ribble in winters 1995/96, 1996/97 and 1997/98 respectively. It was not clear if this had a damaging impact on the fisheries of the River Ribble, with no indication of the potential increased number of fish that would have been present after winter 1997/98 had cormorants not fed on the river. However, the output was similar to other results for cormorant predation levels on the River Ribble, where an estimated 2 to 38 % of the stock of chub, roach and dace were shown to be removed during a winter of cormorant predation (Feltham and Davies 1995).

Similar to the River Trent, there are a number of biotic and abiotic factors that may have a greater impact than cormorant predation on the annual fish production in the River Ribble. These include the decreased river flows, resulting in silting and drying of spawning gravels, and annual temperatures causing variation in year class strength production of important angler species.

In summary, there are two main areas which require serious consideration in future work.

- The MCS model requires further development, perhaps using marked birds, in order to accurately predict the number of foraging cormorants (and hence, the number of fish ingested) on a fishery during periods where observations are not possible. This would reduce the large variance in the MCS output and limit the over-estimates that have been observed on fisheries where cormorants are observed but do not necessarily feed, for example, the River Trent study sites.

- Study sites should concentrate on fisheries where both robust fisheries and cormorant data can be collected. The River Ribble proved to be totally unsuitable for collecting robust fisheries data. Consequently, despite robust cormorant data, precision in determining predation impact was impossible. However, it has to be acknowledged that in general terms, over-wintering cormorants are more likely to forage on the larger rivers and stillwater fisheries in the UK, where collection of fisheries data is difficult, due to aspects such as water depth, resulting in inefficient electric fishing surveys (Hickley and Starkie 1985).

Despite these problems in the study, serious cormorant predation impact was successfully elucidated at Holme Pierrepont Rowing Course, with an overall reduction of 59 % in the fish populations (using P = 0.23 in electric fishing) after three winters of foraging, and at Colwick Park Trout Lake, where economic losses of up to £500 p.a. were noted over only a brief period of predation. The predation impact at Holme Pierrepont resulted in large increases in growth rates and potential changes in reproductive strategies and outputs. Hence, despite cormorant predation impact assessment being very subjective (Russell *et al.* 1996), it is irrefutable that these impacts are damaging in both fisheries and economic terms.

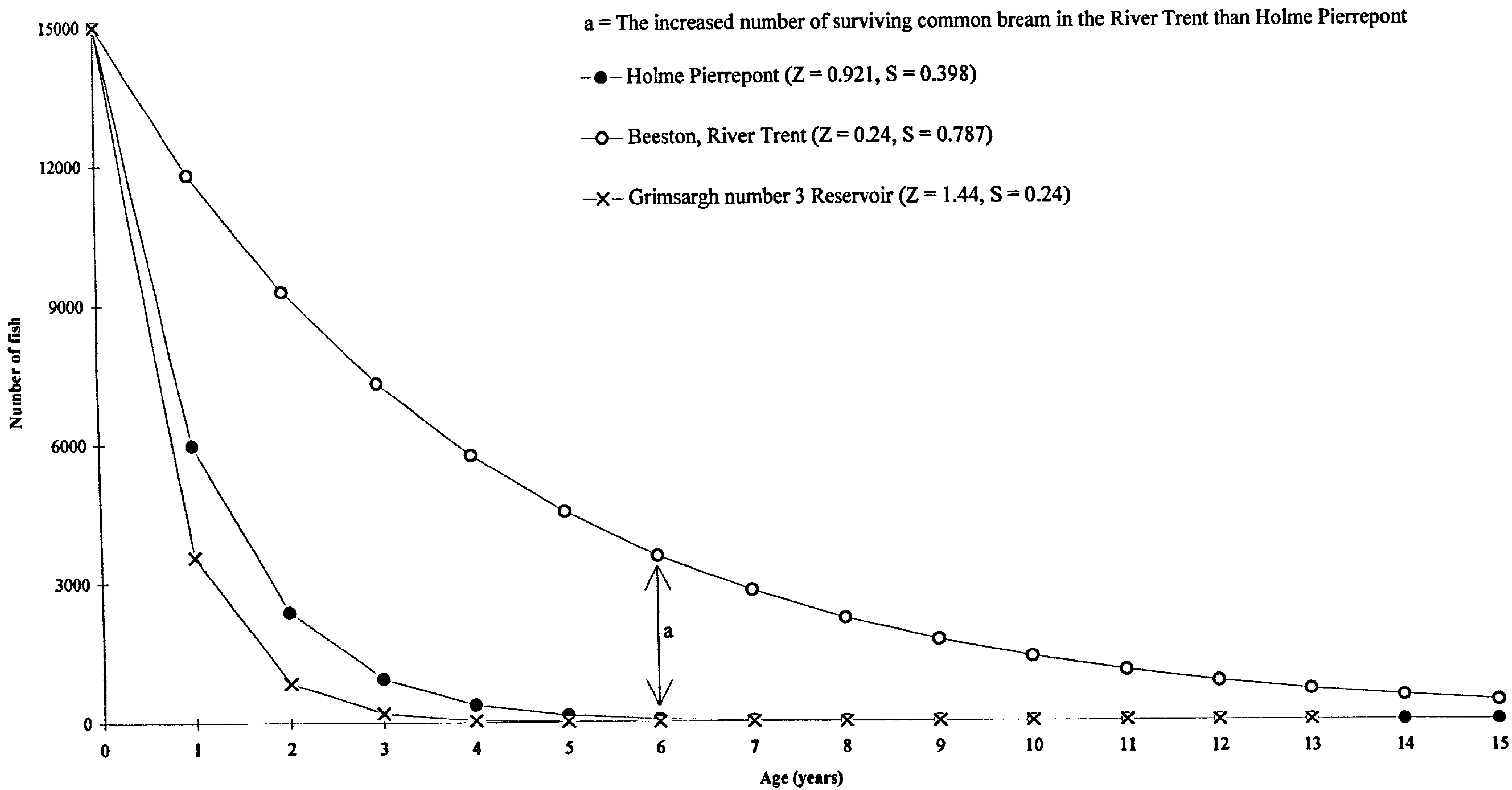


Figure 9.1 Effect of the different natural mortality rates on the number of surviving common bream from a year class producing 15000 recruits in Holme Pierrepont Rowing Course; Beeston, River Trent; and Grimsargh number 3 Reservoir.

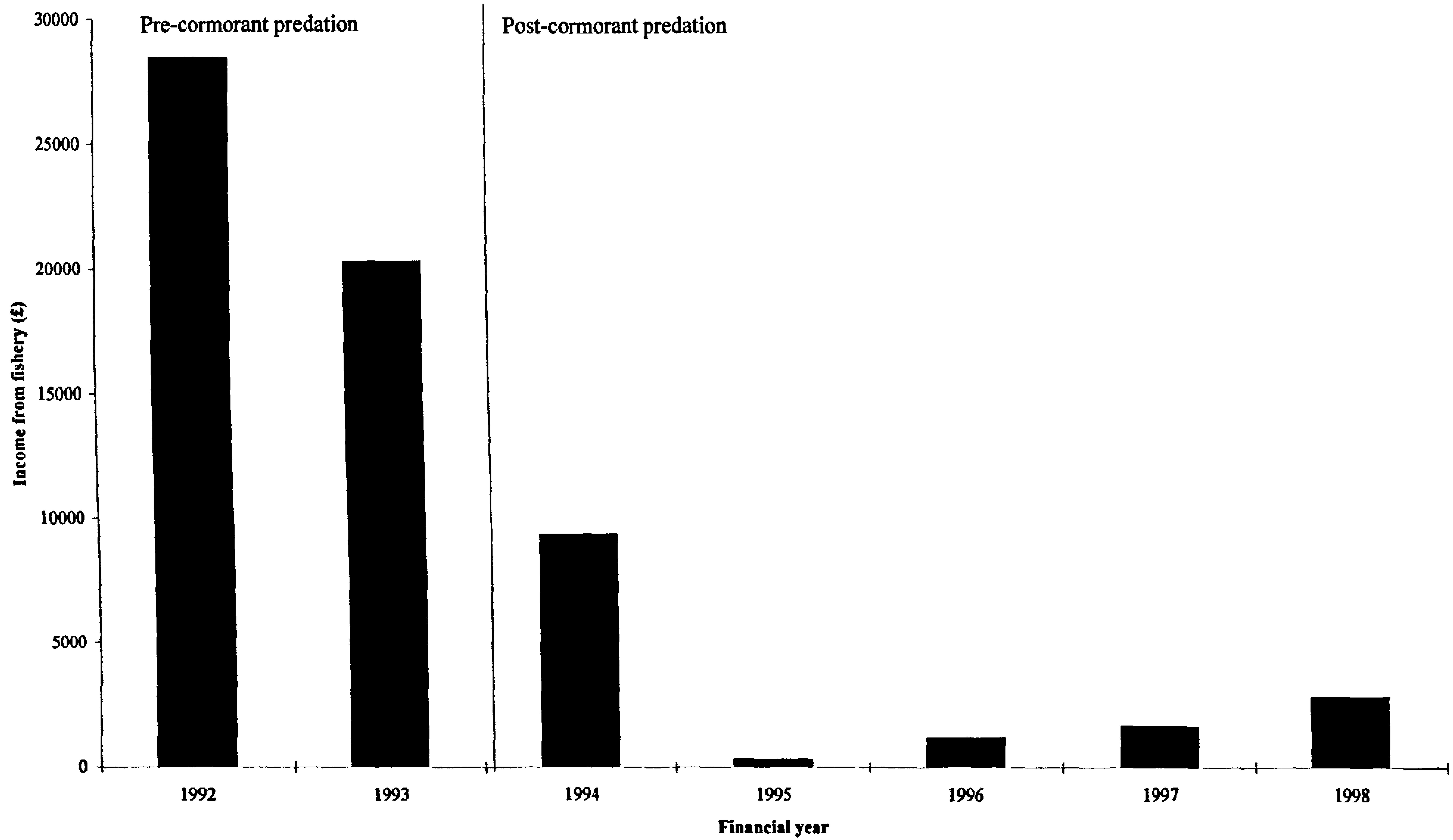
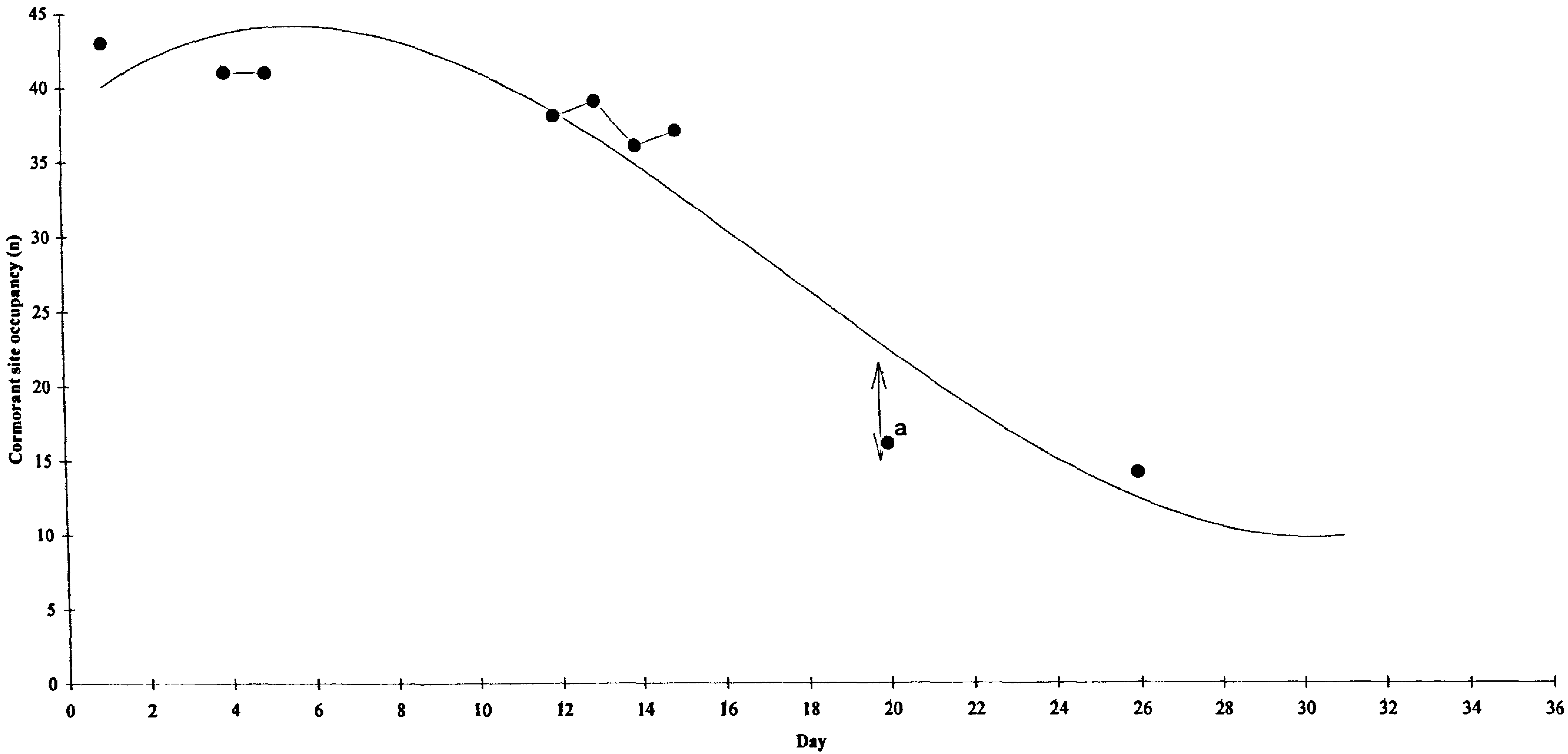


Figure 9.2 Financial returns from angling at Holme Pierrepont Rowing Course in the period prior to, and after, cormorants were first observed foraging on the lake.



a = The difference between the observed number of cormorants utilisng the lake on 20 March 1996 and the estimated number using the method outlined in Section 6.3.1.

Figure 9.3 The polynomial curve (3rd order) used to estimate the daily number of cormorants utilising Colwick Park Trout Lake in March 1996.

10. MANAGEMENT STRATEGIES FOR CORMORANTS AT INLAND FISHERIES

10.1 Introduction

Cormorant management strategies, suitable for UK inland recreational fisheries, can be formulated from the outputs of this study (Chapter 9) and by review of present control techniques (Section 10.2). The following chapter concentrates on fishery specific control techniques considered appropriate to reduce cormorant predation impact on UK inland fisheries, but with particular reference to the sites under study.

Cormorant predation impact was identified at Holme Pierrepont Rowing Course and Colwick Park Trout Lake as damaging in fisheries and financial terms, so it is logical for any fisheries management policy to aim to reduce, or minimise, that impact. Cormorant predation impact was more difficult to quantify on the River Trent and Ribble, but it was acknowledged that cormorants were removing a proportion of the fish standing crop each winter, so any management strategy which can minimise the impact must be considered as a useful tool. At Grimsargh number 3 Reservoir, cormorant predation impact was minimal, so it is unlikely any cormorant management strategy requires implementation. The combination of the diverse nature of the study sites with the different levels of cormorant impact elucidated, show a cormorant management strategy to cover all inland fisheries in the UK would aid management decisions on fisheries affected by cormorant predation.

Any management strategy must be based on the assumption that cormorant occupancy of inland fisheries will be a recurring problem and eradication is not a viable option. This is due to the legal protection of the cormorant (Figure 2.2). Over-wintering cormorant presence will probably have to be accepted by fishery owners, so measures should be implemented to minimise their potential damage to the fishery.

Shooting is a common, and favoured, approach to the control of cormorant populations by fishery managers (Draulans 1987). However, because of firearms control and the protection of the cormorant under the Wildlife and Countryside Act 1981 (Figure 2.2), shooting is not always a viable method, and must be considered in combination with alternative methods (Section 10.2.1). Consequently, a cormorant strategy will aim to integrate the management of the following areas:

- *cormorant population* - to minimise cormorant numbers at the fishery;
- *cormorant habitat* - to minimise cormorant occupancy of the fishery and the region;
- *fish habitat* - to increase the foraging difficulty for cormorants in the fishery;
- *fish population* - to minimise the predation loss to foraging cormorants in the fishery by manipulation of the stocking policy.

To formulate the appropriate management measures, there is a need to examine the existing cormorant control methods in each area (Section 10.2). It is only by doing this that potentially successful methods can be elucidated for use in a management strategy. This information can then be assessed for application in an integrated cormorant control strategy for general, inland, recreational fisheries in the UK (Section 10.3), and the study sites (Section 10.4).

10.2 Existing control methods

10.2.1 *Cormorant population management*

* See page 343

Cormorant population management aims to minimise the cormorant occupancy and foraging on the fishery by reduction of their numbers (Table 10.1; McKay *et al.* 1998). Careful consideration should be given to the methods utilised in scaring techniques, because recent studies have shown that increasing levels of cormorant disturbance may increase their DFI as a result of increased activity (White-Robinson 1982; Belanger and Bedard 1990; Grémillet *et al.* 1995; Riddington *et al.* 1996). If scaring does not completely remove the cormorants from the site, those remaining may consume more fish (McKay *et al.* 1998). Grémillet and Schmid (1993) calculated a disturbance of 30 minutes increased cormorant DFI by 23 g.

Shooting

Shooting as a control method can be used in two ways: as a lethal method to kill cormorants on a fishery and as a non-lethal method where blanks are fired to scare and deter cormorants from the fishery (Table 10.1; Boudewijn and Dirksen 1996). A major problem in fishery-specific shooting programmes is that the attractiveness of the fishery as a cormorant foraging site is not changed post-shooting (Draulans 1987). As cormorants are a highly mobile species concentrating at places with a favourable food supply (Elson 1962; Mills 1967), removed individuals are likely to be replaced quickly by others (Draulans 1987). Additionally, shooting is not always possible in urban regions. There are strict legal controls on firearm use in public areas due to the inherent dangers of their use.

In the UK, the issue of a licence under the Wildlife and Countryside act of 1981 is required to control cormorant occupancy of a fishery by shooting (Section 2.5.2, Figure 2.1). A licence will only be granted when all the following criteria are met:

- cormorants are / likely to be causing serious damage to a fishery;
- non-lethal techniques have been attempted and shown to have failed, or are impractical;
- there is no other evident cause of the damage;
- it is considered that shooting may help reduce the problem;
- there is no other satisfactory solution.

Licensed shooting can be successful in reducing cormorant occupancy on a fishery. A reduction of 11.5 to 43.3 % of pre-shooting cormorant numbers significantly reduced cormorant occupancy in the post-shooting period in a UK study which covered lakes, rivers, and reservoir fisheries, including Foremark Reservoir, Derbyshire; River Trent, Nottinghamshire; and Bewl Water, Sussex (McKay *et al.* 1998). Although killing was not found to enhance the scaring effect of shooting, cormorant numbers took between two and six weeks to return to their pre-shooting numbers. The time was dependent upon site size, the presence of cormorant safe havens, cormorant site-fidelity, the proximity of alternative feeding sites and the timing of shooting in relation to cormorant seasonal movements (McKay *et al.* 1998). Consequently, although success was observed in the short-term, the long-term effectiveness of the method was limited and would have required shooting of birds throughout their period of occupancy. A potential problem was the method's lack of humaneness, for only 49 % of cormorants were killed instantly by a single shot (McKay *et al.* 1998).

Despite the illegality of shooting without an issued licence, it has often been used to control cormorant presence at recreational fisheries (Lagler 1939; Pough 1941; Godin 1979; van Vessen 1981; Cadbury and Fitzherberg-Brockholes 1983; Utschick 1984a; van Vessen *et al.* 1985). Fishery owners consider shooting an effective control method as a dead bird cannot eat any more fish (Draulans 1987). It was shown to be an effective method of controlling cormorant predation on the Upper Tamar Lakes, UK, between 1984 and 1989 (Boudewijn and Dirksen 1996). Three or four birds were shot each year at a roost of 16 birds, with recolonisation of the roost taking up to two weeks. Following implementation of the shooting programme, it was found that the presence of a fishery warden with a gun at the lakes was effective in scaring cormorants away. However, a number of other studies found lethal shooting had no deterrent impact on cormorants (Schiemenz 1936; Creutz 1964, 1981; Utschick 1984a).

Studies investigating the effects of culling programmes on bird abundance and fish production have produced some debatable conclusions (Draulans 1987). White (1939) studied bird culling on the numbers of running salmon smolts. In the year when culling was applied, a higher number of trapped smolts were recorded. However, the increase was within the range of the annual fluctuations of the smolt run, so results were inconclusive. Elson (1962), in a long term programme of merganser control designed to protect salmon production in Canadian rivers, found shooting was efficient in reducing fish losses when applied in a systematic way. However, the validity of the results was questionable due to the smolt counting methods improving over the study period. This resulted in an increased number of smolts counted annually irrespective of any bird control. Krämer (1984) reported the effects of shooting grey herons on the numbers of fish present in streams, and found no increase in fish populations post-shooting. Positive effects of shooting fish-eating birds on fish numbers were not found in a number of other studies (Godin 1979; Osieck 1982; Krämer 1984; Utschick 1984a), probably because of the relatively low number of birds shot (Draulans 1987). For a shooting programme to be successful at a fishery, shooting of a significant number of the cormorants present may be required, as found by McKay *et al.* (1998), where 11.5 to 43.3 % of cormorants had to be shot for a successful, short-term reduction in cormorant occupancy.

Although shooting will possibly reduce cormorant occupancy at the study sites, Holme Pierrepont, River Trent, Colwick Park Trout Lake and Grimsargh number 3 Reservoir all have public access. As shooting is strictly controlled by law in public areas, it is likely the method will be prohibited at these sites.

Auditory stimuli

The use of auditory stimuli aims to drive cormorants from the fishery by producing noises of varying sound levels and frequency (Table 10.1).

The use of gas bangers and cannons to produce a noise to drive cormorants from a fishery has proved ineffective in the majority of cases, despite sounds emitted up to 105 dB (Table 10.1; McKay *et al.* 1998). In the UK, at reservoirs controlled by South West Water, gas cannons failed to reduce cormorant numbers (Boudewijn and Dirksen 1996).

However, cormorant numbers were reduced in the Falsters District of Denmark by use of a gas cannon, although the method was designed primarily to scare seals (Jensen 1984). Experiments in France found gas cannons on reservoirs were ineffective as they only forced the cormorants to move to other areas of the reservoir (Broyer *et al.* 1993), and probably resulted in an energy deficit which had to be compensated for by increased DFI (White-Robinson 1982; Belanger and Bedard 1990; Grémillet *et al.* 1995; Riddington *et al.* 1996).

Pyrotechnics, which consist of exploding and whistling projectiles, usually fired from hand-held pistols or shot guns, were used in south-east USA (Mott and Boyd 1995). However, due to lower cost, greater availability and an increased ability to scare birds at long distances, live ammunition is increasingly being used. Distress calls of double crested cormorants, sirens and electronically generated noises have all also been utilised in attempts to scare cormorants from fisheries in USA, with varying degrees of success (Mott and Boyd 1995). However, the effectiveness of these methods in increasing fish production as a result of reduced cormorant predation has not been assessed.

Alternative acoustic scarers, including the emission of cormorant-specific distress calls of up to 95dB, and the deployment of ultra-sonic emitting sound of frequencies above 20 kHz, have also been unsuccessful in reducing cormorant occupancy on inland fisheries (McKay *et al.* 1998).

Although the true effectiveness of audio scaring has not been determined, their potential use on UK recreational fisheries is likely to be limited. This is because many fisheries are located in urban areas where noise pollution is restricted, for example, the study sites of Holme Pierrepont, Colwick Park Trout Lake and Grimsargh number 3 Reservoir. The majority of audio methods are unlikely to be species specific and may affect other resident and migratory bird populations. The economic use of such methods would also need to be justified before application to a fishery. Additionally, the method scares the cormorants onto alternative foraging sites, so the problem is passed on to the next fishery, with a higher energy cost incurred by the cormorants during scaring. This would result in an increased net amount of fish being consumed from the alternative foraging site. Therefore, it would appear the method can be discounted for effective use in a strategy to minimise cormorant predation.

Visual stimuli

The use of different visual stimuli aims to reduce cormorant occupancy on a fishery by deterring flying cormorants from landing and scaring existing cormorants from the fishery. Although limited short-term success has been found, long-term effects of visual stimuli are limited due to habituation of the cormorants to the stimuli (Inglis 1980; Stickley *et al.* 1995; McKay *et al.* 1998).

The effectiveness of electronically controlled human-effigy-scaring devices in reducing double crested cormorant predation on catfish ponds in USA was determined by Stickley *et al.* (1995). The effigy was programmed to inflate at various time intervals to scare the cormorants and force them to vacate the ponds. In deployed areas, cormorant numbers were reduced by up to 85 % for a period of one week. When harassment patrols were also carried out, the deterrent lasted for two weeks. However, habituation to the stimuli occurred over time, reducing the scaring effectiveness. Inglis (1980) found habituation time decreased with increased exposure to a particular visual stimuli. Stickley *et al.*

(1995) minimised habituation by moving the effigy occasionally, modifying its appearance and placing it where cormorants could only observe it when inflated. The effigy method was judged to be superior to the use of other frightening methods, such as automatic exploders (Stickley *et al.* 1995).

Other visual stimuli that have been used to deter cormorant presence on fisheries include (Table 10.1):

- displaying a cormorant carcass and/or a wooden model of an avian predator to incoming cormorants (Boudewijn and Dirksen 1996);
- the use of angled mirrors to deflect sunlight, which deter flying cormorants from landing (Creutz 1964, 1981; Spanier 1980; Utschick 1984b);
- a model helicopter flying around the fishery to deter cormorants from landing (Boudewijn and Dirksen 1996);
- an ultra-light aircraft flying around the fishery to deter cormorants from landing (Osieck 1982; Moerbeek *et al.* 1987);
- strategically placed scarecrows to deter and scare cormorants from the fishery (Moerbeek 1984);
- sailing radio-controlled boats around the fishery to scare cormorants from the water (McKay *et al.* 1998).

Although short term, limited success has been observed with visual stimuli methods, they have all proved ineffectual in the long term due to factors such as cormorant habituation. Trained raptors have provided positive results (Tusnadi 1959; Festetics 1964), but have proved very expensive and labour intensive (Festetics 1964; Table 10.1). As cormorants are able to ingest DFI in a short period of time at specific food patches with a high density of fish (Osieck 1982, 1983), continuous scaring devices are also required to scare the birds (Boudewijn and Dirksen 1996). As this is likely to result in habituation (Inglis 1980), a high diversity of scaring devices would have to deployed, with irregular changes in their combination and locations (Boudewijn and Dirksen 1996). The scaring activities would have to begin at first light, as cormorants begin foraging at that time (Keller and Vordermeier 1994; Section 9.2.3).

Human presence has been shown to cause cormorants to vacate a fishery for short periods of time (Draulans 1987). Human disturbance caused 90 % of cormorants to fly from a fish pond in Denmark, although only 20 % left the area immediately (Moerbeek 1984; Moerbeek *et al.* 1987). A strategy suggested to reduce cormorant predation at aquaculture facilities is to increase human presence in their vicinity by using adjacent ponds for recreational angling (Moerbeek 1984). Cormorant presence at Holme Pierrepont Rowing Course (Chapter 4) was decreased on days when the lake was being used extensively for rowing and sailing activities (T. Holden pers. comm.), although cormorants were rarely disturbed by human presence during the initial hours of daylight, the peak period of cormorant foraging (Section 9.2.3). At Trent Bridge, River Trent, cormorant foraging was limited in winter 1995/96 (Figure 5.63), perhaps due to the site being located in a busy urban area (Section 5.3.3). However, cormorant foraging increased at Trent Bridge in winter 1996/97 and 1997/98 (Figure 5.63), perhaps due to habituation of the cormorants to the urban environment, which allowed the exploitation of the dense over-wintering shoals of fish in the area (Section 9.3.3).

In conclusion, visual stimuli have produced some successful short-term results in scaring cormorants from fisheries where habituation has been avoided. The method may be successful on a UK recreational fishery where human presence is normally low during the period of cormorant occupancy. The cost of the equipment would have to be balanced by the expected benefits of reduced cormorant occupancy. However, at the study sites which had public access, for example, Holme Pierrepont, River Trent and Colwick Park Trout Lake, vandalism of any structure is a potential problem which would inhibit successful application of any visual stimuli.

Conditioned taste aversion

Conditioned taste aversion (CTA) is a technique involving the use of an aversive chemical to deter predators from eating a particular species of prey. In laboratory trials it was found that CTA could be induced successfully in cormorants (McKay *et al.* 1998). Individual birds that had consumed dead trout treated with the aversive chemical subsequently learned to avoid consuming trout, but continued to consume other species of fish (McKay *et al.* 1998). The avoidance lasted for seven months without reinforcement. However, the laboratory tests only utilised trout and the application of the method for use on multi-species fisheries was not determined. The work concluded that further research, including field testing, was required before the method could be applied on recreational fisheries, especially cyprinid fisheries where many species look similar.

Laser light

Laser light, fired at individual cormorants, was found to displace cormorants from sites with non-target bird species generally not disturbed (McKay *et al.* 1998). However, the method was only effective at low light intensities, was expensive and labour intensive, and could not be used in close proximity to humans (Table 10.1). Thus, it was unlikely to be a cost effective and realistic control method at the majority of inland recreational fisheries.

Methods used in aquaculture

There are other control methods which have been utilised successfully on aquaculture sites to minimise cormorant presence, including anti-predator netting, where stock are placed in pound nets or netting is placed across the pond to prevent cormorant access to the fish (Carss 1989; Cornelisse and Christensen 1993; McCarthy *et al.* 1993; Boudewijn and Dirksen 1996) and wires placed above the water surface to prevent cormorants from landing (Barlow and Bock 1984; Moerbeek *et al.* 1987). However, such aquaculture methods are impractical for use on recreational fisheries as they would prevent effective angling (McKay *et al.* 1998).

10.2.2 *Cormorant habitat management*

Management of cormorant habitat aims to reduce cormorant presence in the area surrounding the fishery (Table 10.2).

Table 10.2 Cormorant habitat management methods.

Tactic	Methodology	Comments
Cormorant night roost disturbance	Disruption/destruction of the cormorant night roost to drive the cormorants from the area	Would require licensing under the Wildlife and Countryside Act 1981. Would only move the cormorants to other fisheries.
Cormorant day roost disturbance	Disruption/destruction of the cormorant day roost to drive cormorants from the area.	Although cormorant occupancy may be reduced, it is unlikely because of probable alternative day roosting sites in the area.

Cormorant night roost disturbance

Cormorant roost disturbance involves removing, or making uninhabitable, the over-wintering cormorant night roost, which usually consists of a number of trees. This method forces the colony to disperse to a different area. It is not a fishery specific control method, but local fisheries would benefit generally as the cormorants are forced to roost elsewhere. It is a non-lethal method requiring a large amount of manpower to destroy the roost. The method has been used throughout Europe to reduce local cormorant predation. However, it is likely they will re-establish a roost elsewhere and the increased energy cost incurred during scaring will result in an increased net amount of fish being consumed from the new foraging sites. Cutting down roost trees is only effective when there are no alternative trees available (McKay *et al.* 1998).

In the UK, a cormorant roost was destroyed on the River Tamar by tree removal using physical force and the wind (Boudewijn and Dirksen 1996), but the roosting cormorants became more dispersed along the river system. However, whether this caused a reduction in the local number of cormorants or reduced predation on surrounding fisheries was not discussed.

There are several examples of roost disturbance in France. In 1991, 12 cormorant roosts, containing a total of 600 to 1000 birds, were disturbed by non-lethal shooting and detonations (Dumeige 1993). However, the method was unsuccessful as it only caused a reduction of 36 cormorants from the roosts after 18 attempts. Broyer *et al.* (1993) also documented the roost disturbance programme and found that methods utilising crackers and shooting in the air had no long lasting effects on the cormorant numbers. At one site all the birds left the roost and returned after a number of days, and on another occasion the roost became only partially deserted. Shooting of some of the birds at the roost over several evenings had a longer lasting effect (Broyer *et. al.* 1993).

Thus, roost disturbance of over-wintering birds has been utilised as a method to control cormorant predation by attempting to drive the birds from local areas. The method has not been shown to reduce cormorant predation on a local scale, and has not been a very successful long-term measure in reducing population numbers. Boudewijn and Dirksen (1996) discussed a proposition that roost disturbance should be used as the first phase of cormorant control, with the next phase utilising deterrent methods.

As roost disturbance aims to reduce local cormorant population numbers, it could be applied to UK situations as a control method in a regional management strategy.

However, location of the roost may cause problems, for example, the Attenborough cormorant night roost, Nottingham, is located in a Site of Special Scientific Interest (SSSI), where it may be difficult to justify destruction of a large number of trees (Section 4.5.1). Additionally, habitat destruction methods would need to be licensed under Section 16(1) of the Wildlife and Countryside Act (1981) (Figure 2.1). It is illegal to destroy the habitat of cormorants. Hence, for a fishery-specific cormorant control strategy, the method can be discounted due to the inherent difficulties.

Cormorant day roost disturbance

Similar to destruction of a night roost, destruction of the day roost areas of cormorants may force them to seek alternative sites. At Colwick Park Trout Lake, there were a number of large boulders protruding from the water in the north east corner of the lake (Section 6.1.2). These were used as areas for day roosting by cormorants, with activities such as loafing and wing drying occurring. Removal of some of these boulders reduced the number of cormorants occupying the lake (M. Weevers pers. comm.). However, these birds were not feeding on the lake and had already fed at alternative sites during the initial hours of daylight (Section 9.2.3). Thus, destruction of this day roost site had no impact on the level of cormorant predation on the lake, because those cormorants were not utilising the lake for foraging. Furthermore, destruction of the day roost in this instance had no impact on the cormorant occupancy on surrounding fisheries.

Removal of the boulders used by cormorants for day-roosting may reduce cormorant predation during the stocking period at Colwick Park Trout Lake. It is the only study site where day-roost disturbance is likely to have any impact on the level of cormorant predation (Section 10.4.3).

10.2.3 ***Fish habitat management***

As cormorant foraging and diving behaviour have been shown to be affected by the habitat of the fishery (Wilson and Wilson 1998), manipulation of the habitat may increase foraging bout duration and decrease success (Table 10.3). Additionally, fish productivity can be increased by habitat manipulation to limit predation losses (Table 10.3).

Fish refuges

Artificial fish refuges, structures which are placed in the fishery to increase the cover for fish from cormorants, are a potentially low cost technique to reduce predation impact (McKay *et al.* 1998). Refuges aim to reduce the encounter rate of foraging cormorants with prey fish by the increased use of cover. There are many different forms of refuge that can be used, including artificial reefs; submerged branches, tubes or pipes; and tyre pyramids (McKay *et al.* 1998). Refuge design and applicability are likely to vary according to species (benthic and pelagic species, cyprinid and salmonid species) and habitat types (river and stillwater) (McKay *et al.* 1998). Fish do utilise cover increasingly with predator presence, as shown by juvenile roach and perch (Eklöev and Persson 1996) and juvenile rainbow trout (Tabor and Wurtsbaugh 1991), although the potential use of refuges to limit losses to cormorants has not been determined fully (McKay *et al.* 1998).

Table 10.3 **Fish habitat management methods.**

Tactic	Methodology	Comments
Artificial fish refuges	Construction of artificial structures which will promote fish shoaling and are safe from foraging cormorants.	Construction is likely to be of low cost. However, the refuges must not make fish unavailable for angler exploitation and they must prevent cormorant foraging.
Increased natural cover for fish	Extra provision for fish of cover from cormorant foraging by provision of extra underwater features, for example, submerged vegetation.	Addition of underwater features may impact adversely on the angling conditions in the fishery and make fish unavailable to exploit.
Construction of in- and off-stream structures	Fish habitat is enhanced to maximise productivity which aims to minimise predation impact.	Habitat modification has been shown to positively impact on juvenile recruitment.
Increased turbidity	Water turbidity is artificially enhanced to reduce the foraging success of cormorants, sight predators.	Dyes, mixing of water with air pumps and introduction of bottom foraging fish (for example, carp) have all been suggested to increase turbidity to minimise cormorant foraging success.

It is critical to ensure that any refuge which promotes shoaling of fish does not impinge on angling success at the fishery (McKay *et al.* 1998), and does not allow cormorants to forage in the refuge. This was shown by Holme Pierrepont boat pontoons (Section 4.7.6), where dense shoals of cyprinid species over-wintered around, and under, the pontoons, which did not provide any vertical protection in the water. Hence, cormorants were able to dive underneath the pontoons and forage very efficiently (Section 4.7.6).

Limited success was observed in protecting fish from cormorants using refuges, because cormorant dive times increased where refuges were present (McKay *et al.* 1998). However, trials in the presence and absence of refuges found that actual fish losses attributable to cormorants were similar, despite the increased foraging bout duration. As artificial refuges are relatively inexpensive compared with alternative cormorant deterrent options, their use may be justified in aiming to protect fish stocks from cormorants. Additionally, reefs may increase the productivity of the fish populations (Prince *et al.* 1978)

Fish refuges can be protected from foraging cormorants by suspending walls of netting in the water around the refuge. These would allow the movement of fish, but prevent cormorants from diving (McKay *et al.* 1998). The principal is similar to the predator nets surrounding pound nets in aquaculture which there has been success in reducing cormorant predation losses (Carss 1989; Cornelisse and Christensen 1993; McCarthy *et al.* 1993; Boudewijn and Dirksen 1996). However, there is a danger that the nets would not be bird species-specific, with non-target, smaller, piscivorous birds getting ensnared in the net, for example, goosanders, mergansers and grebes; and larger fish may get gilled in the net and die.

However, in the attempt to reduce fish losses, fish refuges could be successfully applied to all the study sites where cormorants forage. Refuge structures are likely to differ between stillwater fisheries such as Holme Pierrepont and river fisheries such as the River Trent.

Increased natural cover for fish

Increasing the natural cover for the fish populations, for example, by increasing the amount of aquatic vegetation or terrestrial overhanging vegetation, may increase their protection from cormorant predation (Table 10.3). Extensive floating or emergent vegetation is usually avoided by foraging cormorants (Cramp and Simmons 1977), although Barlow and Bock (1984) report a number of studies where cormorants were able to forage between dense, submerged aquatic weed beds. An additional problem is extensive aquatic vegetation would limit angling on the fishery and cormorant predation in the UK occurs in the winter months, when aquatic vegetation has died back, due to decreased temperatures and in rivers through increased flows. Consequently, the practicality of the method is very limited.

Construction of in- and off-stream structures

In- and off-habitat features are important to fish populations in providing, for example, food, cover from predators and shelter from high flows (McKay *et al.* 1998). Enhancement of these features, or construction of additional features, should aim to optimise the habitat features in the fishery for each different life-stage of the target species. This will aim to avoid inter-cohort competition in the population, which could otherwise lead to over-crowding and slow growth, as observed in Grimsargh number 3 Reservoir (Chapter 7). Optimising fish production in the fishery will ensure that any losses due to cormorants are compensated for. Although construction of off- and in-stream structures have not been designed primarily for minimising cormorant predated losses, it is a method whereby recruitment is maximised, increasing the scope for compensation (Section 10.3.4).

Increased turbidity

As cormorants forage by sight, water turbidity and prey visibility may be important factors in determining foraging success, although cormorants do forage successfully in relatively turbid water (Barlow and Bock 1984; Section 9.2.4). However, where alternative sites of lower turbidity are available, these may be preferred (McKay *et al.* 1998). This was shown by cormorant foraging patterns in the Netherlands (Moerbeek *et al.* 1987; Voslamber 1988). Cormorants moved from favoured sites during periods of high winds and increased turbidity to alternative, less turbid feeding locations. Therefore, artificial enhancement may reduce losses to predators (McKay *et al.* 1998). The use of dyes has been suggested (Mott and Boyd 1995), although the feasibility, cost, environmental impact and the effect on the angling at the fishery have not been assessed (McKay *et al.* 1998). McKay *et al.* (1998) suggested introducing benthic foraging fish, such as carp, although this will have a large impact on the fish community due to the competitive nature of carp and anglers may not wish to catch them. Finally, increased water turbidity could be achieved by using air lifts or water pumps to mix the water.

All the reviewed methods that aim to increase water turbidity appear unrealistic, with little practical application in real situations. Increasing turbidity, either naturally through fish introductions, or artificially through dyes, will impact on the habitat of the fishery and as such may be more damaging to the long-term status of the fishery than cormorant predation.

10.2.4 *Fishery enhancement management*

Fish population control methods apply to fisheries where angling performance is enhanced by stocking of supplementary fish. Therefore, fishery enhancement techniques are particularly relevant for intensively managed fisheries. Put-and-take trout fisheries generally introduce rainbow trout throughout the angling season to provide anglers with a satisfactory catch rate (Pawson 1982). Catch-and-release cyprinid fisheries stock fish on a less regular basis, with species, sizes, frequency and density of the stock dependent on the management objective of the fishery. Thus, alteration of the stocking policy could minimise potential losses to cormorants and limit the encounter rate between cormorants and the stocked fish (Table 10.4).

Size of stocked fish

Alteration of the size of the stocked fish may reduce losses attributable to cormorant predation at both put-and-take trout fisheries and catch-and-release cyprinid fisheries (Table 10.4).

At put-and-take trout fisheries, cormorant predation is size-specific. At Colwick Park Trout Lake, cormorants selectively ingested trout of 250 to 300 mm, which comprised less than 2 % of the stocked fish (Section 9.3.2). McKay *et al.* (1998) reported the majority of trout stocked in UK put-and-take trout fisheries are around 360 to 380 mm fork length. It is believed trout of this length are susceptible to high levels of predation. Increasing the size of the stocked trout, to sizes over 450 mm, was believed to have potential for lowering predation levels (Table 10.4; McKay *et al.* 1998). However, cormorant wounding was observed on 11.4 % of the trout caught in electric fishing surveys at Colwick Trout Lake, with wounded fish of a greater average size than the other trout caught (Section 9.3.3). Consequently, although losses may be reduced by increasing the size of stocked fish, wounding rates may increase.

The economic feasibility of stocking larger trout has to be addressed. McKay *et al.* (1998) consulted a number of fishery managers on the issue, and opinion was divided. A number of fisheries practised stocking with trout of approximately 0.25 to 0.5 kg, as these apparently provided anglers with elevated catch rates, especially in matches. These managers felt stocking of larger fish would not be cost effective due to the increased cost of the fish, as the fish had to be reared for an additional winter, food conversion rates fell as the trout increased in size, and the increased costs could not be recovered by increased angler revenue (McKay *et al.* 1998). However, other managers had experimented stocking with larger trout and found angler return rates were substantially increased, although when cormorants foraged, an increased wounding frequency occurred (McKay *et al.* 1998). Managers expressed concern over the potential effect of high cormorant damage on angler perception of the fishery (McKay *et al.* 1998).

Table 10.4 Manipulation of stocking practice to minimise losses to cormorant predation.

Tactic	Methodology	Comments
Size of stocked fish	Control the size of the fish stocked, because cormorant predation is generally size selective.	Increased wounding frequency is likely to occur when larger fish are stocked.
Frequency of stocking	Regular trickle stocking over an extended time period can be carried out, rather than stocking all the fish in one introduction.	This will reduce the number of naïve fish available for cormorants to exploit at each stocking.
Location at stocking	Stocking the fish at several locations in the fishery, rather than stocking all the fish in one area.	This would split the shoals of stocked fish into lower densities and may reduce the predation pressure.
Timing of stocking	Timing of the stocking can be manipulated to ensure the number of cormorants able to exploit the fish immediately after stocking is minimised.	This can be carried out on a seasonal basis, with stocking only commencing after cormorant dispersal to breeding areas; and diurnally, with stocking of fish in the late evening to avoid the daily peak foraging period.
Density of stocked fish	As fish stock densities can be correlated with cormorant predation levels, reduction in fish density may reduce cormorant foraging on the fishery.	This goes against the principles of fisheries management and is likely to reduce angler success. It would also be dependent upon the management objective of the fishery.
Species of fish	As fish shape can affect species vulnerability to predation (deeper bodied fish are less vulnerable), common bream and selective breeding of carp with high backs may limit predation losses.	This would depend upon the management objective of the fishery, with angler opinion over the species they wish to exploit being an important factor in the decision.
Alternative prey	Stocking of alternative prey may protect the more valuable fish species in the fishery.	The method has not been observed to be successful in multi-species, recreational fisheries, and the introduced fish may adversely affect the resident fish population.

Thus, the size of stocked trout may have a major bearing on cormorant predated losses. Whether or not to increase the average size of trout stocked requires a careful review of fishery policy. Relevant issues for consideration are anglers' perception of the present losses to cormorant predation and the potential future increase in wounding rates caused by cormorant attack. The economic performance of the fishery will have a major bearing on the decision, especially as losses attributable to cormorants in a short time space on Colwick Park Trout Lake were approximately £500 p.a. (Section 9.4.3).

The present study found cormorant predation was size-specific on cyprinid fisheries, with fish under 200 mm most vulnerable to cormorant predation (Section 9.3.2). McKay *et al.* (1998) found larger carp (mean length 270 mm) sustained lower predation rates than smaller carp (mean length 70 mm and 120 mm). However, the larger carp were more likely to sustain cormorant wounds, with 77 % of all large fish displaying wounds attributable to cormorants, compared with 19 % in medium sized fish and 1 % in small fish (McKay *et al.* 1998). Hence, stocking of larger cyprinid fish may be cost effective for fisheries to prevent losses to cormorants, dependent on angler opinion on the increased wounding rate, although it is not known how many of the wounded fish die as a result of the cormorant attack.

At the study sites, stocking of larger trout could be implemented at Colwick Park Trout Lake (Section 10.4.3). The remaining study sites rely only on natural recruitment for subsequent angling success.

Frequency of stocking

Alteration to the frequency of stocking will aim to reduce the chance of large aggregations of recently stocked fish attracting foraging cormorants (Table 10.4; McKay *et al.* 1998). Cormorants are opportunist predators, with cormorant numbers positively correlated to fish stock densities in lakes/specific food patches (Suter 1995; Warke and Day 1995; Section 9.2.4) and increased prey density may result in an increased attractiveness of the site for foraging birds (Elson 1962; Draulans 1987; McKay *et al.* 1998). Increasing the frequency at which fish are stocked, with reduced numbers introduced at each stocking, may reduce the attractiveness of these fish for cormorants. This tactic refers mainly to put-and-take trout fisheries where stocking is carried out on a regular basis, rather than catch-and-release cyprinid fisheries, where supplementary stocking may only occur on an annual basis.

Increasing the frequency of stocking is likely to incur additional transport and handling costs for the fishery and so may be more applicable for fisheries where fish are reared on site, or have on-site fish holding facilities (McKay *et al.* 1998). The trout could be delivered in bulk from the fish farm, held in the on-site facilities, and trickled stocked gradually over the following week. At Colwick Park Trout Lake, it would be difficult to implement the method as the management is reliant on the fish farmer delivering the fish in one journey to reduce costs.

Location of stocking

Alteration of the location of the stocking is an applicable method for both put-and-take trout fisheries and cyprinid fisheries (Table 10.4). When fish are stocked at a specific release point, cormorants are often attracted to the area to forage on the fish which may exhibit naïve behaviour to predation risk (Anglian Water 1997; McKay *et al.* 1998), as observed on trout at Colwick Park Trout Lake (Section 6.7). Stocking fish at a number of locations around the fishery margins, or from a boat, and in areas where there is a high human presence to deter foraging cormorants, may avoid attracting the attention of any cormorants day-roosting at the site. At a number of put-and-take trout fisheries, boats containing dummy fishermen are moored at release sites for 48 hours prior to stocking to act as a predation deterrent on the newly stocked fish (Anglian Water 1997).

This would be a successful technique at Colwick Park Trout Lake as it would disperse the large shoals of trout at the specific release point. These large shoals have been observed to be very vulnerable to cormorant predation in the initial period after stocking (pers. obs.).

Timing of stocking

The timing of stocking fish can be manipulated in two ways: by avoiding peak cormorant numbers at a seasonal level, and on a diel level (Table 10.4). Although the method is mainly applicable to put-and-take trout fisheries, cyprinid fisheries using supplementary stocking to improve angler catches would also benefit from the method.

Seasonal

Ottenbacher *et al.* (1994) found that recreational put-and-take trout fisheries in Utah, USA, were being stocked with hatchery-reared trout in the period prior to, or during, the migrating season for double crested cormorants. Lakes were being stocked with relatively dense populations of vulnerable stocked trout during periods when cormorant numbers were at their peak. To reduce losses where cormorants are primarily migrants, stocking could be carried out to avoid the periods of peak bird abundance (Ottenbacher *et al.* 1994). Osieck (1983), Moerbeek (1984) and Moerbeek *et al.* (1987) recommended delaying stocking of carp into aquaculture ponds in the Netherlands until late June to reduce their availability during peak cormorant numbers. Similar recommendations were made at catfish farms in the USA (Mott and Boyd 1995), and Im and Hafner (1984) recommended removing fish from outside ponds before over-wintering cormorants arrived in the Camargue, France. At Colwick Park Trout Lake (Chapter 6), initial stocking of trout occurred in mid-March, the period when over-wintering cormorants were still present in the region, and resulted in losses of approximately £500 p.a. through predation. Therefore, delaying of stocking until after cormorant dispersal (April) could reduce losses to minimal levels (Section 6.7).

The practical use of the method is dependent on the economic feasibility of its implementation at specific fisheries. As there is an increasing demand for all year round angling at trout fisheries (Pawson 1990), and cormorants are fairly widespread on stillwater fisheries throughout the year (Callaghan *et al.* 1998), stocking trout only in the period outside cormorant occupancy may make it unrealistic for a number of operators. The losses incurred by cormorant predation would have to be weighted against the profits made during that period. On cyprinid catch-and-release fisheries, stocking of fish could be carried out to avoid peak cormorant numbers and to ensure the fish are settled into the fishery to reduce their vulnerability prior to peak predation pressure. However, stocking would have to judged against the different stocking mortalities that are likely to occur at various times of the year.

Diurnal

Diurnal patterns of cormorant predation were observed in this study (9.2.3) and by Kennedy and Greer (1988), where cormorant foraging peaked in the early hours of daylight and reduced with increasing day length. As cormorants feed by sight, foraging is restricted by low light intensities. Consequently, stocking of any fish in the period of cormorant occupancy should be carried out as late in the day as possible to avoid any cormorant foraging in the initial period when fish are most naïve to predation (McKay *et al.* 1998). This is a technique which is likely to be very successful in reducing the initial losses to cormorants at Colwick Park Trout Lake.

Density of fish

As cormorant predation is positively correlated to the fish stock density of the fishery (Suter 1995; Warke and Day 1995), or to the fish density in the specific food patches (Section 9.2.4), decreasing stock density may reduce cormorant predation (Barlow and Bock 1984) (Table 10.4). However, this is opposed to the basic principles of fish management (Draulans 1987). The method could be applied to cyprinid fisheries if the management policy of the lake is to provide anglers with only specimen fish to target. A reduced number of alternative fish will allow the specimen fish to exploit a low competitive environment and the cormorants will not be able to predate on the large fish. Where the fishery is used for competition angling, the method is likely to result in reduced catches, with anglers visiting alternative fisheries. The technique is unlikely to be implemented at any of the study sites.

Fish species

Stocking alternative fish species, which are less vulnerable to cormorant predation, may reduce losses to cormorants and so increase the attractiveness of the fishery to anglers. Matkowski (1989) suggested certain sport fish species, known to be less vulnerable to cormorant predation than others, should be used in recreational fisheries in the USA where cormorant predation is a problem. Deeper bodied fish, for example, common bream, are believed to be less vulnerable to predation than other fish (De Nie 1995). Osieck (1983) suggested a possible means of reducing cormorant vulnerability in fish was to selectively breed carp with higher backs which would deter cormorant predation.

The stocking of alternative fish species which are less vulnerable to cormorant predation is not an option on a put-and-take trout fishery, since anglers visit to catch trout. Cyprinid catch-and-release fisheries can stock alternative fish species, such as common bream and carp, but this is dependent on the management objective of the fishery and if anglers wish to catch these fish.

Alternative (buffer) prey

In a fishery, a buffer population is a species, or a number of species (naturally or stocked), which provides cormorants with an alternative food source to protect species of greater value. An example is the use of crayfish (*Cherax destructor*) in Australian fish farm dams which produce carp (Barlow and Bock 1984). Where crayfish were present, the cultured carp populations were believed to be partially protected from cormorant predation, as in crayfish absence, cormorants removed over 50 % of the carp.

Lagler (1939) first discussed the method of using buffer populations to protect aquaculture stocks. Frogs, toads and forage fish were to be placed in ponds surrounding the culture site. The cormorants would then hopefully forage on these ponds rather than on the fish culture ponds, although Lagler (1939) did not provide any evidence of likely success. Glahn *et al.* (1995) found gizzard shad (*Dorosoma cepedianum* Rafinesque) comprised a large portion of cormorant diet in areas surrounding catfish culture sites, and concluded the shad were an important predation buffer for the catfish.

The use of buffer populations to safeguard more valuable fish stocks in recreational fisheries would be difficult to implement for the following reasons.

- To protect resident valuable fish populations, large numbers of buffer population fish would have to be introduced. This may increase inter-specific competition and cause a decrease in performance of the resident valuable populations, especially where food supply is limited.

- As angler catches are a function of stock size (Crisp and Mann 1977; O'Grady 1980; Pawson 1982; North 1983), it is likely the buffer populations, if stocked in large enough numbers, will feature in angler catches, perhaps against anglers' wishes.

- The increase in total fish density may result in a larger number of cormorants being attracted to the site, so predation levels may increase (Elson 1962; Draulans 1987).

- In natural, multi-species fisheries, such as Holme Pierrepont (Chapter 4) and the River Trent (Chapter 5), the method is not a viable option due to the reasons outlined and the high density of alternative prey that would be required to protect the other fish species. It would also require introductions which have their own inherent problems.

Therefore, manipulation of the present stocking policy of a managed fishery may result in minimising losses to cormorants. Before any method is implemented, consideration will have to be given as to whether the benefits of reduced cormorant predation outweigh the burden of increased costs from introducing a buffer population.

10.3 Potential management strategies for minimising the impact of cormorants in UK fisheries

Prior to implementation of any control strategy, the management objectives of the lake or river must be determined. If angling is a priority, then it is logical to limit the potential damage that cormorant predation can cause. Monitoring of cormorant predation on the fish populations at Holme Pierrepont (Chapter 4) revealed a reduction in the fish stock of over 60 % after three winters of foraging, angler catch rates were adversely affected by predation, the predation coincided with a large decrease in income from angling, and fish behaviour and habitat utilisation were adversely affected. At Colwick Park Trout Lake (Chapter 6), approximately £500 p.a. of stocked trout were taken and considerable non-lethal wounding of fish occurred. Therefore, a do-nothing option will have important implications on the future status and economic viability of these fisheries. However, cormorant impact was more difficult to elucidate in the River Trent and River Ribble, and at Grimsargh number 3 Reservoir, cormorants did not impact adversely on the fish population. A do-nothing option may present a viable alternative to a cormorant management strategy on these fisheries.

10.3.1 *Legal protection*

The cormorant is protected under the Wildlife and Countryside Act 1981 (Figure 2.1). This means a licence must be granted from MAFF before cormorants can be controlled by lethal methods. The purpose of the shooting licence is to reinforce the protection strategy of the fishery and not to necessarily to kill the permitted number of cormorants (McKay *et al.* 1998). Each licence has several conditions, which include a limit on the numbers of cormorants that can be shot, the time of year during which shooting can be conducted and the request that the carcasses are collected for analysis of their stomach contents.

A licence to protect a fishery from cormorants will only be issued, "for the purposes of preventing serious damage to ...fisheries," (Wildlife and Countryside Act 1981; Figure 2.1) and after alternative methods to reduce damage have been tried and proved ineffective (Section 10.2.1). Before a licence is applied for, a number of criteria must be fulfilled by the situation at the fishery. This is a major constraint on the fishery owner/manager, as it requires considerable effort in order to prove a licence should be granted. The number of cormorants that are legally allowed to be shot is an important factor in the licence, because McKay *et al.* (1998) found a minimum of 11.5 % of cormorant numbers in the pre-shooting period had to be killed in order to reduce cormorant numbers after the shooting programme ended (Section 10.2.1).

The justification for the licensing of shooting is a complex issue. In this study, cormorant predation was shown to impact negatively on Holme Pierrepont and Colwick Park Trout Lake; predation impact was more difficult to determine on the River Ribble and River Trent; and cormorants appeared to have negligible impact at Grimsargh number 3 Reservoir. Consequently, it is difficult to generalise the potential impact of cormorant predation on a fishery, making it difficult to determine whether the legal protection afforded to cormorants requires reviewing. However, as removing the requirement of the licence to shoot cormorants is likely to result in a high degree of persecution of cormorants at their roosts, and control of the numbers shot impossible to monitor, the present system should be kept to regulate the shooting of cormorants. Shooting should only be considered as the last alternative option to control cormorant occupancy and foraging at a fishery, but a more structured licence application system should be implemented to aid applicants in meeting the license criteria. This will then allow licensed shooting to become an important part of any management strategy to minimise cormorant impact on a fishery.

As shooting should only be considered as the last viable method to control cormorants, non-lethal methods should be attempted first, with post activity monitoring. A licence would only be applied for once these methods have been shown to be unsuccessful in reducing the predation pressure. Therefore, an appropriate strategy to minimise cormorant predation impact on a managed, recreational fishery must:

- aim to reduce cormorant predation on the fishery by non-lethal methods;

- monitor the success of non-lethal methods, judged by post-implementation cormorant activity;

- collate evidence to justify the claim that cormorants are damaging the fishery before applying for a license to cull. This will be in terms of changes in the fish populations, financial losses attributable to cormorants and the scale of the cormorant predation.

The strategy will consist, where feasible, of (Section 10.2):

- visual control methods, including human presence (non-lethal);
- habitat manipulation, for example, refuge construction (non-lethal);
- stocking policy alteration, for example, fish size, timing of stocking (non-lethal);
- licensed shooting (lethal).

The non-lethal methods would be used in combination, with monitoring of their success on the cormorant activity. If cormorant activity on the fishery is not reduced, then licensed shooting becomes the next viable option.

The non-lethal methods utilising audio scaring, laser lights, roost disturbance, conditioned taste aversion, increased water turbidity, reductions in fish stock density and buffer populations have been discounted (Section 10.2). This is due to factors such as the limited chance of success, the cost and the legal controls (Section 10.2).

A strategy to minimise cormorant predation impact on general UK fisheries is shown on Figure 10.1, with the potential methods in the overall strategy discussed next.

10.3.2 *The do-nothing option*

The do-nothing option is a possible cormorant management strategy (Figure 10.1, Table 10.5). If the fishery is not economically important and is utilised by other user groups, then this option may be preferred. Although the method may appear inexpensive, financial losses may be incurred through a reduction in angling success due to a reduced fish stock, forcing anglers to visit alternative fisheries; and the replacement cost of the predated fish if angling returns are to be maintained to their level prior to cormorant predation. Hence, it may have important implications on the future status of the fishery, in both economic and fish populations terms, as shown by the large economic and fish losses at Holme Pierrepont (Chapter 4) and the financial losses at Colwick Park Trout Lake (Chapter 6) (Section 10.1).

Table 10.5 The do-nothing option in a management strategy to minimise cormorant predation impact.

Problem	Cormorants observed feeding on the fishery.
Aim	Reduce cormorant predation impact on the fish populations of the affected fishery by relying upon fish compensation processes to overcome predation levels.
Feasibility	Dependent on the population dynamics and life history strategies of the fish in the affected fishery.
Effect on fishery	Dependent on level of cormorant predation and the fish population response.
Cost-benefit	If unsuccessful, the costs incurred may include fish stock replacement and the loss of angler revenue due to decreased angler success. Any success will depend on the level of compensation in the fish populations to the losses. The fish population density is likely to be reduced after predation, with an increased output in terms of growth and reproductive effort.

10.3.3 *Visual stimuli*

Visual stimuli could be used to supplement other control methods to reduce the cormorant numbers at a fishery (Table 10.6). They have been successful in reducing cormorant numbers in the short-term, although long-term success was not achieved due to cormorant habituation. For visual stimuli to be successful in reducing cormorant predation at a fishery, a diversity of stimuli would have to be deployed with irregular changes in their location (Section 10.2.1). This would increase the financial and time costs of the method.

If habituation to the stimuli can be avoided, the first measure that could be introduced to a managed fishery, where cormorants are considered a problem, is the application of a visual stimuli, with the chosen method dependent on the cost (Table 10.2). There are a wide variety of methods available (Section 10.2.1). However, vandalism of the equipment would be a concern (Table 10.2). At urban fisheries, any brightly coloured objects may attract unwanted attention and subsequent damage, so decreasing the effectiveness of the method.

Table 10.6 Methodology of the application of visual stimuli to minimise cormorant predation impact.

Problem	Cormorants observed feeding on the fishery
Aim	Reduce cormorant presence on the fishery by scaring cormorants from the water and deterring flying cormorants from landing by application of a visual stimuli.
Feasibility	Dependent on correct application of stimuli, cormorant habituation being avoided, and avoiding vandalism of any structures.
Effect on fishery	Beneficial if the method reduces cormorant presence. However, the method may not be species-specific and may be detrimental to the visual attractiveness of the fishery to visitors.
Cost-benefit	Could be cost effective if the stimuli is not prohibitively expensive, labour costs are limited and valuable fish stocks are successfully protected.

10.3.4 *Habitat modification*

Habitat modification can be applied in two ways to minimise losses to cormorant predation on managed fisheries: by deterring cormorants from foraging on the fishery, and by promoting recruitment and juvenile survival to minimise any losses that may occur.

To deter cormorant predation

Stillwater fisheries provided the most profitable food patches for cormorants in the study, with habitats utilised by shoaling fish in winter providing very attractive foraging sites (Section 9.3). Manipulation of these habitats to discourage cormorant foraging should help protect the fish from predation. This could be done by providing 'cormorant proof' refuges, where fish can gather but cormorants cannot effectively forage, resulting in foraging becoming unprofitable (Table 10.7). Natural (for example, by planting of macrophytes) or artificial (for example, construction of fish refuges which prevent effective pursuit-diving) structures can be utilised.

Fish refuges can only be 'cormorant proof' by ensuring (Section 10.2.3):

- provision of cover from above;
- provision of cover from the side (eliminating cormorants from diving underneath/through);
- location throughout the fishery to promote fish dispersal and reduce the shoal sizes.

Table 10.7 **Application of habitat modification to deter cormorant foraging on a managed fishery.**

Problem	Cormorants successfully foraging on the fishery.
Aim	Decrease cormorant foraging efficiency and make it energetically unprofitable on the fishery. Achieved by construction of cryptic fish habitat and cormorant-proof refuges, and by promoting fish dispersal in the fishery. The method would supplement cormorant scaring tactics by visual stimuli.
Feasibility	Possible, as foraging success was shown to be dependent on habitat types in the study sections (Section 9.3.3).
Effect on fishery	Will be beneficial if cormorant predation can be reduced in the fishery to an appropriate level. May be detrimental if refuges promote fish shoaling, but are unable to prevent cormorant foraging.
Cost-benefit	Dependent on the structures utilised, the labour costs incurred, the value of the protected fish stocks and the success of the structures in protecting the fish.

Two possible structures are brushwood reefs and tyre reefs. Brushwood reefs are constructed by tying bushes together and weighting them to ensure they sink (EA 1998). These would increase fish refuge habitat and prevent cormorants from diving through the shoals of fish. Tyre reefs are constructed from old lorry and car tyres with holes drilled in the upper tread to allow air to escape and weights in the bottom side to ensure sinking. The tyres are lashed together by rope or nylon cord (EA 1998). Although designs would be of personal preference, consideration should have to be given to the aim of the structure, i.e. preventing diving access to foraging cormorants. It is suggested that such reefs occupy a total of 0.25 % of the total area of the fishery and are split into three or four separate units (EA 1998). This would promote three or four separate fish shoaling areas that are cormorant proof.

Additionally, using fencing to protect natural/artificial winter fish refuges may prevent cormorants from foraging effectively in the refuge area. Netting could also be placed vertically in the water to protect the refuge, as long as fish were not gilled as they pass through the net (Section 10.2.3).

As designs for cormorant proof fish refuges have not been fully developed, the Holme Pierrepont boat pontoons (Plate 4.1) would be an ideal location for further research to be carried out. The adjacent pontoons would allow control pontoons and pontoons offering different refuge designs to be constructed and their effectiveness measured under potential heavy predation pressure.

Research on the impact of stock manipulation in relation to cormorant predation on a 0.5 ha stillwater, Gedling Pond, adjacent to Holme Pierrepont, Nottingham, found that despite the natural fish populations being supplemented by cyprinid stocking, cormorants were reluctant to forage there (Feltham *et. al.* 1999). A possible explanation was that the trees surrounding the pond deterred cormorant occupancy on the pond. Cormorants are nervous and shy birds, preferring habitats which allow good peripheral vision (T. Holden pers. comm.). This enables them to notice hazards very quickly. The surrounding trees did not allow good peripheral vision at Gedling Pond; hence, the cormorants did not forage on the pond. Thus, any improvements undertaken at a managed fishery should not involve the large-scale removal of bank-side trees, unless they are likely to provide a cormorant night roosting area.

Lowland river habitats are more difficult to manipulate due to factors such as water flow, navigation and flood defence schemes. Although reefs can be built in large lowland rivers, this is dependent upon the flood defence role of the river and whether the reef will impinge upon this. Natural in-stream cover may be more effective than an artificial refuge in providing cover for fish from cormorant predation. Consequently, any flood defence improvements to a river should not involve large scale destruction of natural fish holding areas, such as sunken and overhanging trees.

As off-stream havens have been observed to provide fish refuge from winter floods on lowland rivers, for example, Colwick Park boat marina in the Trent Bridge study section (T. Holden pers. comm.), cormorants may exploit this fish behaviour by modification of their daily foraging pattern. Clearly, possibilities do exist to construct 'cormorant proof' refuges for fish in off-stream havens.

To promote juvenile survival and recruitment

Increasing juvenile production and survival will maximise their subsequent recruitment into the catchable population (Table 10.8). Construction of in-stream and marginal features and off-stream refuges to increase spawning, nursery and fry habitats may increase the potential recruiting fish numbers, although this is the subject of on-going research (J. Harvey pers. comm.). The structures involved are the same, or similar, to those mentioned in decreasing the foraging efficiency of cormorants (Table 10.7).

Table 10.8 Application of habitat modification to promote juvenile fish recruitment.

Problem	Cormorants forage on fishery with success and impact negatively on the mature spawning stock and recruiting juveniles by reduction of their numbers.
Aim	Maximise juvenile recruitment to ensure adequate compensation can occur in the year classes by removal of recruitment constraints, and improvement of spawning and nursery habitat. This may result in an adequate number of fish being able to grow beyond the optimum size for cormorant predation in the fishery for future spawning and angler exploitation.
Feasibility	Dependent on the physical nature of the fishery, the structures utilised, density independent factors, for example, temperature, water flow and food availability, and density dependent factors, for example, inter- and intra-specific competition.
Effect on fishery	If the scheme is successful, it will have a very beneficial effect on the fishery by increasing stock abundance for angler exploitation in future years. However, an increased fish density may attract increased numbers of foraging cormorants.
Cost-benefit	If successful, the scheme may increase the economic value of the fishery. The short-term costs depend on the structures and manpower utilised.

Suitable structures can be found in Cowx and Welcomme (1997) and examples are shown below. Local Environment Agency fisheries departments will advise on which structures are most suitable for the habitat in the fishery concerned.

Cyprinid stillwater habitat manipulation to improve juvenile recruitment

- Creation of shallow, marginal areas. This allows growth of abundant weed beds, increasing spawning, nursery and feeding areas.
- Creation of shallow, wetland areas. These are generally inaccessible to adult fish, so provide improved fry and nursery habitat.
- Planting reed beds. Increases spawning and nursery habitats.
- Planting overhanging, terrestrial vegetation. Increased marginal cover may reduce predation risk for fish and provide habitat for terrestrial and aquatic invertebrates, increasing the food supply for juveniles.

River habitat manipulation to improve juvenile recruitment

- De-silting of spawning gravels. Improves water circulation through the gravel, maintaining the oxygen supply to the developing eggs.
- Gravel supplementation. Provides improved/increased spawning habitat for salmonid and cyprinid species.
- Current deflector construction. Produces a new sequence of pools and riffles, removes silt and exposes suitable gravel deposits for spawning .
- Creation of off-channel areas. Connected to the main channel, these provide shelter for fry of all species during periods of high flow, preventing downstream drift of juveniles.
- Creation of in-stream cover. Increased in-stream cover can be provided by logs, floating cover boards and macrophytes.

At the present time, study sites on the River Trent (Chapter 5) have few areas of cover for fry, with little marginal macrophytic growth, limited shallow areas out of the main flow and few fry refuge areas during periods of increased flows, for example, boat marinas (pers. obs.). As recruitment may be a constraining factor in the production of strong chub year class strengths in recent years (Section 9.1.4), a number of the activities outlined may increase their fry survival in subsequent years. The creation of off-stream havens is presently being attempted in the area below Beeston Weir, River Trent (J. Lyons pers. comm.).

10.3.5 *Stocking policy alteration*

It was discussed in Section 10.2.4 that the stocking policy of the fishery could be altered to minimise losses to cormorant predation by the following methods (Table 10.4).

- Size of stocked fish. Stock larger fish as cormorant predation is size-selective.
- Frequency. Stock with increased frequency - 'trickle stocking' - with decreased numbers of fish to decrease opportunist cormorant foraging.
- Location. During each stocking occasion, utilise a number of locations in the fishery to decrease the shoal size of vulnerable, naïve fish.
- Timing. Minimise losses to cormorants of stocked fish by attempting, where possible, to avoid peak seasonal and diurnal numbers.
- Density. Minimise losses, where appropriate, by limiting the density of fish in the fishery to avoid attracting foraging cormorants.

- Species. On intensively managed cyprinid fisheries, species less vulnerable to predation should be considered for stock, for example, common bream and carp.
- Alternative prey. Where appropriate, large numbers of cheap, alternative prey for cormorants could be stocked in the fishery to protect more valuable stock.

Implementation of stocking manipulation into a strategy to minimise losses to cormorants may be successful (Figure 10.1). The actual stocking tactics that can be manipulated in the fishery are dependent on the species and current stocking regime of the fishery, the economic benefits and feasibility, and the views of the angler. A policy utilised on a trout put-and-take fishery, where stocking is carried out regularly throughout the year, will differ from a cyprinid catch-and-release fishery, where supplementary stocking may occur only periodically. It is unlikely the density of fish will be reduced on any fishery, due to the negative impact it would have on angler catch rates, and the stocking of alternative prey is unlikely to be a viable option, biologically and economically, for many fisheries (Section 10.2.4). However, a combination of the alteration of the size of stocked fish, the location and timing of stocking, and the species stocked would be relevant for many managed fisheries in the UK in minimising cormorant predation impact (Table 10.4).

10.3.6 *Licensed shooting*

Once non-lethal control techniques have been attempted on a managed fishery and have been assessed as unsuccessful in reducing cormorant predation, application can be made for a licence to control cormorant numbers on the fishery by shooting (Section 10.3.1; Figure 10.1). It has been proposed that the present system for licence application should be retained, as this will allow the regulation and monitoring of shooting to be maintained (Section 10.3.1). However, adjustment must be made in the criteria required for applications, with definition of the term, "serious damage....to fisheries" (Wildlife and Countryside Act 1981; Figure 2.1). Alternatively, the term could be removed from the legislation, with the requirement for a successful licence application being a proven link between decreased fishery performance and the cormorant predation. Fishery performance can be determined in terms of trends in angler catch rates, fish stock assessment and the financial performance of the fishery (Table 10.9). Where decreased performance is elucidated, these data can be allied with cormorant predation data to determine any relationship (Table 10.9).

A step-wise procedure for assessing potential for damage by cormorants at an inland fishery has been developed (Carss and Marquiss 1994, Figure 10.2). However, the flow chart does not demonstrate how any damage can be quantified as 'serious'. A methodology is required for determining serious damage in terms of fisheries monitoring, economic assessment and cormorant ecology monitoring.

Table 10.9 Strategy to determine if cormorant predation is causing 'serious damage' on a managed fishery.

Problem	Cormorant predation has continued on the fishery despite the use of non-lethal control methods, habitat modification and stocking policy alteration.
Aim	To develop the practical application of a realistic licensing programme to cull cormorants as part of a wider strategy to reduce predation at a fishery. The application process will be structured with clear guidance on the requirements for the 'serious damage' criteria of the licence application. Damage will be assessed in terms of fisheries monitoring, cormorant monitoring and financial monitoring.
Feasibility	Dependent on the integration of research, angling, ornithological and government organisations and agencies on agreeing that increased licensed shooting is the way forward in controlling inland cormorant predation. It may reduce illegal shooting practices and allow better understanding of the effect of culling on cormorant occupancy and numbers.
Legality	Requires some revision of the legal conservation status of the cormorant and the application of the Wildlife and Countryside Act 1981 by MAFF regarding the issue of licences to cull cormorants.
Effect on fishery	If the shooting programmes are successful in reducing cormorant predation on the fishery, then the fish stocks will be protected from increased losses attributable to cormorants.
Cost-benefit	Dependent on the effectiveness of the method. Where a licence is granted, the shooting programme reduces cormorant visits to the fishery and the fish stocks are protected, the method is likely to be cost effective, dependent on the labour costs incurred.

Fisheries monitoring

The simplest method of monitoring a fishery is by the continuous assessment of fishery performance through angler catch surveys. If data are available from the period prior to cormorant predation, allowing the baseline fishery performance to be determined, then comparison with performance in the period of cormorant predation will allow a degree of impact assessment. However, natural variation in the fish population must be accounted for, as in this study it has been shown to cause fluctuations in angler catch rates (Section 9.1.2). This will require fish stock assessment (see below).

Angler catch monitoring can be undertaken using pleasure angler cards or match angler returns that are completed at the end of the angling session. The data collected should include:

- angling session duration;
- weight of catch;
- species composition of catch;
- size composition of fish caught;
- health of fish (for example, cormorant wounding marks).

Data analysis allows the catch per unit effort, percentage of anglers with catch, species and size composition of the catches, and the percentage of fish with cormorant wounds to be determined (Section 3.5.12). The angler catch data can be supplemented by the collection of fisheries data from a sub-sample of fish, for example, fork length (mm), individual fish weight (g) and scale samples, which are used to calculate fish population growth characteristics (Section 3.5).

The angler catch and sub-sample data may be sufficient in determining if serious cormorant damage is occurring on the fishery and whether a licence should be applied for. However, these data should be used in conjunction with a fish stock assessment to account for natural variation in the fish population dynamics. This would be carried out by use of electric fishing, seine netting, and/or hydro-acoustics, depending on the nature of the fishery (Section 3.3). This service could be carried out in conjunction with the local Environment Agency fisheries service or by a private contractor, with the cost involved being a major consideration. The analysis of the fisheries survey data would allow assessment of the following:

- fish population and community structure;
- fish density;
- presence of any recruitment bottlenecks;
- population responses to cormorant induced stress, for example, growth rate;
- fish wounding attributable to cormorants;
- the potential for the population to compensate or regulate for predation losses, measured as an index of abundance of juvenile age groups.

These data would allow further determination of the cormorant predation impact and whether the cormorant predation is/has been detrimental to the fishery performance. If data are available for the fishery from the period prior to cormorant predation, this baseline data can be used as a comparison to the data set derived in the period of cormorant predation. This would allow an increased understanding of the effect of natural fluctuations in the fish population dynamics on the fish community and density, and the angling success indices.

Financial assessment

Financial losses are an impact that may be incurred by fishery owners when cormorants feed on their fishery. These may be defined as serious when they affect the profitability of the fishery as a business. Financial losses can be determined from:

- direct fish loss due to predation;
- replacement costs of predated fish and wounded fish;
- indirect financial losses due to angler avoidance, measured by sale of permits and angler visits (difficult to measure);
- loss of angler revenue for the fishery;
- decline in fishery performance attributable to predation and not natural cycles in population abundance.

The daily financial income of the fishery should be monitored in association with the level of cormorant predation and the replacement cost of fish losses. These data would supplement the fisheries data in the licence application.

Cormorant monitoring

To confirm that the changes in the fish populations and angler catches are attributable to cormorant predation requires robust data on the cormorant populations. This would require dawn until dusk observations over a prolonged period of time to obtain the following data:

- temporal numbers of cormorants occupying the fishery;
- activity of the cormorants: roosting, feeding, flying;
- foraging rate success (measured as the percentage of foraging bouts where fish were consumed);
- amount, size and species of the fish consumed.

These data would supplement the fisheries and financial data, and aid the licence application in confirming the damage implied is indeed due to cormorant predation. The number of cormorants occupying the fishery on a daily basis would be used as an indicator of the number of birds that would be culled under licence.

Thus, there must be clarification of the criteria that have to be met to obtain a licence to cull cormorants. This may be done by implementation of a methodology which shows:

- non-lethal control strategies have been attempted and been unsuccessful;
- serious damage in terms of fisheries and financial data, and is shown to be likely to be due to the cormorant predation.

10.4 Specific cormorant management strategies for minimising the impact of cormorants on the study sites

The general cormorant management strategy for UK recreational fisheries (Figure 10.1, Section 10.3) can now be applied to each study site, with consideration given to the results and conclusions of the cormorant impact assessment.

10.4.1 *Holme Pierrepont Rowing Course*

The first stage in structuring a cormorant management strategy for Holme Pierrepont is the determination of the management policy for angling on the lake (Figure 10.3). If angling is considered a priority on the lake, then cormorant predation becomes an issue. The flow chart demonstrates the options available (Figure 10.3).

The basic model demonstrates two aspects in controlling cormorant predation, monitoring and minimisation. If the predation minimising methods fail, a shooting licence could be applied for, as the monitoring data would support the application in fulfilling the 'serious damage' criteria (Figure 10.3).

However, in reality, this model is unlikely ever to be utilised. The lake is used as a general water sports lake, with the complex receiving funding from the Sports Council and Nottingham City Council for daily running costs. The lake is run as a multi-functional water sport complex and, at present, angling receives no management or funding, and is unlikely to in the future (Section 4.1). Consequently, any method which conflicts with the major sports on the lake will infringe on the daily usage of the lake and would not be considered (Table 10.10). Such methods include the extensive planting of macrophytes to increase fish cover and the placement of visual stimuli at strategic locations in the lake (Table 10.10). These would interfere with sports like rowing and wind surfing. Additionally, the legality of shooting cormorants on the complex, due to the public access, would have to be investigated before shooting of the birds becomes a realistic option (Table 10.10).

Table 10.10 Feasibility of methods to minimise cormorant predation impact at Holme Pierrepont Rowing Course.

Method		Feasibility	Explanation
Do-nothing option		Feasible	Although this method has been utilised since cormorants first foraged on the fishery, it has resulted in a damaging impact (Section 9.4.1).
Visual stimuli		Not feasible	The management are unlikely to invest time and money on a method which is unlikely to result in a long-term benefit, will interfere with other water users and is likely to get vandalised if placed on the lakeside.
Habitat modification	Artificial fish refuge(s)	Feasible	As the method is likely to protect the fish populations from predation and can be placed under the boat pontoons away from other water users, it may be practical.
	Increased natural cover	Not feasible	As this would involve extensive planting of macrophytes in marginal areas which would cause conflict with other water users, the method is unlikely to be considered.
Stocking policy alteration		Not applicable	Management do not supplementary stock the lake for angling, because the management does not regard angling as a financial priority.
Licensed shooting		Not feasible	Holme Pierrepont has public access with a perimeter road around the lake. Shooting would not be legal on the complex.

Therefore, the one positive method that could be utilised at Holme Pierrepont is the construction of 'cormorant-safe' fish refuges. As fish already utilise the area immediately underneath, and around, the boat pontoons, these could be fenced off under the water. This would allow fish to retreat through the fences for refuge during cormorant foraging. The fences would be constructed to ensure the cormorants are unable to dive through the structures. This may result in decreased losses. Although netting could be utilised using a similar principle, this may prove more expensive. As there are three main pontoons utilised by the fish for refuge and cormorants for foraging (Plate 4.2), experimentation on the refuge design is possible, with monitoring of the cormorant foraging success rates under each pontoon determining the most successful designs (Section 10.3.4).

Artificial fish refuges could also be distributed at a number of locations throughout the lake to encourage fish refuge and winter shoaling in alternative areas of the lake, preferably in deeper areas than those observed under the boat pontoons, where depths did not exceed 1.5 m (pers. obs.). Brushwood reefs or tyre reefs could be utilised (Section 10.3.4). However, who is responsible for the work and pays for the cost would have to be determined. As mentioned, the owners do not actively spend money on management or promotion of the lake as a fishery.

Additionally, information on the fishery should be disseminated to anglers to explain the status of the fish populations and its effect on angling at present and in the future. An increased understanding of the life-history strategies and population dynamics of the fish in response to the cormorant predation, and the effect of year class strengths on the scope for compensation for losses and angling success, have resulted from this study (Chapter 4 and 9). Growth rates were fast with a short life span (Section 9.1.3), mortality was high (Section 9.1.5), and where recruiting year classes were strong, subsequent angler catch rates were good (Section 9.1.2). Hence, future angler catch rates can be elucidated if monitoring of the cormorant predation and fish populations is continued.

Angler catch rates in 1998 were very dependent on the 1996 year class of roach. This year class was strong, resulting in their domination of the angler catches. However, their mortality will result in decreased numbers available to anglers in 1999, and unless the 1997 roach year class can replace their numbers and mass in the fishery, roach catches are likely to decrease in 1999. The physical conditions in the summer of 1998, with low seasonal temperatures, are likely to have resulted in poor recruitment and weak year classes of all species. As a result, the years when these fish would be expected to contribute positively to angler catches, (2000 and 2001), assuming similar fast growth, will probably see poor catches. Therefore, this information should be disseminated to the local angling fraternity enabling them to understand the dynamic nature of the fishery and the reasons why the angling results are going to fluctuate. The continued over-wintering cormorant predation will continue to reduce the cohorts, decreasing their presence in the fishery, and impacting on future catch rates.

Therefore, in theory, a strategy to minimise cormorant predation impact on Holme Pierrepont can be elucidated (Figure 10.3). However, in reality, due to the management policy of the lake, its funding and its location, there are few viable methods that could be successfully implemented (Table 10.10). Artificial refuges present the only viable and realistic option, and dissemination of information to anglers will enable understanding of any future years of poor angling being the result of the natural fluctuations of the fish populations and the effect of cormorant predation upon them.

10.4.2 *River Trent*

A cormorant management strategy for the River Trent would have to bring together all the angling interests on the affected stretches to decide the policy to be undertaken (Figure 10.4). If a plan of action cannot to be agreed between the parties, then a do-nothing strategy would have to be implemented (Figure 10.4). However, if agreement could be reached, then a strategy can be designed and implemented (Figure 10.4). Potential difficulties arise in where and how the methods are utilised, who finances the methods, and the acceptance of the methods by other users of the river, for example, bird-watchers and boaters. Collaborative parties in any scheme may include the Environment Agency and British Waterways.

Similar to Holme Pierrepont, any strategy would have to comprise of minimisation and monitoring the cormorant predation control techniques (Figure 10.4). If sufficient finance and interest in the management strategy could be generated, non-lethal methods could be applied on the river sections, in conjunction with a monitoring scheme of the fisheries, economics and cormorant populations of the river. If the non-lethal control methods were unable to adequately reduce predation impact in the river, and the monitoring scheme indicated serious damage was occurring, application could be made for a licence to cull the cormorants.

As cormorant predation levels were observed to be low on the river (MCS results were thought to have over-estimated fish losses, Section 5.8), it would be debatable whether a cormorant control strategy is necessary or feasible due to the high economic costs that would be incurred. In reality, the strategy becomes not feasible (Table 10.11). A footpath runs along the length of the River Trent study sites, so public access renders shooting of cormorants impossible. Planting of macrophytes in marginal areas would be a huge task, with much of the marginal area excessively deep for good plant growth, and the navigational usage of the river may cause the newly introduced plants to be washed away. Visual stimuli methods would be difficult to implement, for bankside structures are liable to be vandalised and are of little use if the cormorants are foraging in the middle of the river, up to 30 m away. The study sites extend for a total of approximately 6000 m, with a surface area of over 240000 m^2. Consequently, a large number of stimuli would also be required. The structures could not be placed in-stream as they would obstruct navigation. The biggest potential handicap of the strategy is the fractious nature of the angling fraternity. Amalgamating the opinions of all the angling groups and associations with interests on the river would be extremely difficult.

Table 10.11 Feasibility of methods to minimise cormorant predation impact on the River Trent study sites.

Method		Feasibility	Explanation
Do-nothing option		Feasible	This method has been utilised since cormorants first foraged on the river.
Visual stimuli		Not feasible	The structures are likely to be vandalised due to public access; bankside structures are unlikely to work due to the distance between them and the foraging birds; in-stream structures will interfere with navigation.
Habitat modification	In- and Off-stream refuges	Feasible	These would maximise recruitment success in the river, increasing the year class strength of individual cohorts.
	Increased natural cover	Not feasible	The logistics of the operation would make the method impractical due to the large size of the river and deep margins.
Stocking policy alteration		Not applicable	
Licensed shooting		Not feasible	Due to public access.

Therefore, habitat modification to maximise recruitment success is the only method which may provide positive results in minimising predation impact. This would aim to improve the cyprinid spawning habitat available in the river and increase the habitat available for juveniles (Section 10.3.3). The feasibility of the scheme is dependent on the finance available for refuge construction and the co-operation of riparian owners and Environment Agency departments, such as Flood Defence.

The methods would consist of in- and off-stream refuge areas and de-silting of spawning gravels (Table 10.5). In-stream structures could only be utilised if obstruction of flood defence is avoided. If the planning and implementation of these methods are successful, the potential benefits will include maximisation of the fish production in the river, and minimising over-wintering cormorant predation where refuges are constructed to be cormorant proof (Figure 10.4). Realistically, a river rehabilitation scheme on such a large scale would be very expensive and would require extensive negotiation with interested parties. Hence, success will be dependent on the willingness of outside parties and the costs involved.

10.4.3 *Colwick Park Trout Lake*

The strategy to minimise cormorant predation impact on Colwick Park Trout Lake is based on the same model as Holme Pierrepont and the River Trent (Figure 10.3, 10.4), with differences in the habitat manipulation techniques and the stocking policy review (Figure 10.5). However, the realistic application of the majority of these methods would not be feasible, with stocking practice alteration the only practical technique available to minimise losses to cormorants (Table 10.12). The financial losses of approximately £500 p.a. due to ingested fish alone, without assessment of the loss of angler revenue due to predation and the presence of wounded fish in catches (Section 9.4.3), results in a do-nothing option being unacceptable. The fishery is reliant upon strict funding by Nottingham City Council and any financial losses that can be minimised, with low manpower and costs, should be encouraged.

The use of visual stimuli, for example, inflatable effigies and scare-crows, to decrease cormorant occupancy on the lake would be difficult to implement due to the high risk of vandalism, because Colwick Park is located in an urban area and has 24-hour access (Table 10.12). The construction of 'cormorant proof' trout refuges could be attempted to increase the areas for newly stocked trout to shoal safely and prevent cormorants from being able to predate on them. However, it is possible that the trout will not utilise any refuge structures and, where the refuges are successful, it is likely the trout would be unavailable for angler exploitation (Table 10.12).

A stocking policy review on the fishery would identify the areas where predation can be minimised by alteration of current practices (Section 10.3.4). Cormorant predation in the period of post stocking could be reduced by stocking trout of only above 400 mm and trickle stocking (Section 10.3.4, Figure 10.5). Stocking should also be carried out in the late evening period to reduce the initial losses to cormorants. As such measures have been shown to reduce cormorant predation in other studies, and trout predation in the fishery was shown to be size selective (Section 6.6.2, 10.2.5), implementation of these stocking policies may result in virtual elimination of losses. This would minimise the requirement of other control measures, which have been shown to be impractical.

Table 10.12 Feasibility of methods to minimise cormorant predation impact on Colwick Park Trout Lake.

Method		Feasibility	Explanation
Do-nothing option		Not feasible	Although the method has been utilised since cormorants first foraged on the lake, losses of up to £500 p.a. have been incurred.
Visual stimuli		Not feasible	The structures are likely to be vandalised due to public access; bankside structures are unlikely to work due to the distance between them and the foraging birds; any success in reducing occupancy is likely to only be short-term.
Habitat modification	Trout refuges	Not feasible	Although the refuges may reduce the trout vulnerability to predation, it is likely the trout would also become unavailable for angler exploitation.
	Increased natural cover	Not feasible	The lake already has abundant macrophyte growth and any further planting would impinge on the ease of angling on the lake.
Stocking policy alteration		Feasible	A review of the present stocking policy on the lake would highlight the areas where improvements can be made. These are likely to be in timing, frequency, sizes of fish and location of the stocking. Buffer prey are unlikely to be cost effective or acceptable for the fishery.
Licensed shooting		Not feasible	Due to public access.

Removal of all the boulders in the north-east corner of the lake may also reduce cormorant impact on the newly introduced trout. These boulders are used as a day roost by the cormorants throughout the winter, with daily occupation only after foraging on alternative sites (Section 9.2.3). During the initial stocking, it was the day-roosting cormorants which were first attracted to the trout, that had become disorientated due to their introduction, with many laying on the surface (pers. obs.). Being opportunist predators, the day-roosting cormorants began foraging for these fish immediately, with a number of the trout ingested within the initial 30 minutes of being introduced to the fishery (pers. obs.). Hence, removal of the boulders, reducing day-roosting numbers, and ensuring all the birds are scared off the lake prior to stocking, will minimise these losses. This highlights that stocking in the evening, or during a period of low light intensity, would also be successful in minimising losses, i.e. when the majority of cormorants have returned to the night roost.

Therefore, similar to Holme Pierrepont and the River Trent, an integrated cormorant strategy to minimise losses can be designed for Colwick Park Trout Lake, but in reality it would be impractical, due to the constraints on the fishery of location, likely limited success and costs (Table 10.12). This results in the improvement of stocking practices being the only realistic option to minimise cormorant losses. The actual methods would be dependent upon the management of the fishery, with the possible methods outlined in Section 10.2.4. It must be remembered cormorant predation only impacts on the trout population of the fishery between mid-March (initial stocking) and late April (cormorant dispersal) (Section 9.4.3). As this period attracts a number of regular anglers to the fishery due to the opening of the season, this loyalty and revenue would be lost if stocking was delayed until cormorant dispersal.

10.4.4 *Grimsargh number 3 Reservoir*

Cormorant predation impact was minimal on the fishery due to the high density of slow growing cyprinid fish in the reservoir (Section 9.4.4). Hence, a strategy to minimise losses to cormorants does not appear necessary and fishery management techniques can be concentrated on improving the angling success on the fishery (Section 7.8.5). This should aim to reduce the overcrowded population of the cyprinids under 150 mm in the reservoir (Section 7.8.5). This would decrease competition and increase growth rates. Indeed, any additional predation on the lake, whether by piscivorous fish or birds, may be beneficial in achieving this aim.

10.4.5 *Lower River Ribble*

Cormorant predation impact was difficult to determine on the River Ribble due to poor fisheries data (Section 9.4.5). During the study period, the influence of low flows, loss of spawning habitat, siltation of spawning gravels and poor fry habitat were identified as potential constraints on production in the cyprinid populations of the river (Section 9.4.5). Further investigation of these areas is needed to clarify if these constraints do impact on fish production in the river and whether methods such as off-stream refuge construction could be beneficial in improving recruitment.

The River Ribble is an important salmonid river in the Northwest of England (Chapter 8), with present management of the river aimed at improving salmonid habitat and spawning in the upper catchment areas (pers. obs.). Hence, the cyprinid population of the river has received little management attention. The cyprinid angler effort on the lower River Ribble was limited (Section 8.5.2), with the majority of anglers targeting the specimen sized chub and barbel of the river (pers. obs.). The vulnerable target prey of cormorants receives little attention from anglers at present and cormorants may only impinge on subsequent angling success if they prevent an adequate number of fish attaining specimen size.

10.5 Discussion

Although there are a wide variety of methods that have been utilised to control and minimise cormorant predation, their practical application is limited when applied to real situations on UK recreational fisheries. As the study sites were of diverse nature, covering river and stillwater habitats, and cyprinid and salmonid fisheries, they may be considered representative of the majority of recreational fisheries in the UK. Therefore, it can be extrapolated that there are few feasible and realistic methods for individual UK recreational fisheries to utilise in minimising cormorant predation losses.

Thus, individual fisheries are limited in their options for controlling predation. Non-lethal methods just force the cormorants on to alternative fisheries, with an increased DFI due to energy expenditure. Many fisheries, particularly those located in urban areas, are unable to control cormorant numbers by shooting. Consequently, regional, national, and even European programmes to control inland, over-wintering cormorant populations may be an alternative option. As programmes of this size would require a huge amount of organisation and co-operation between interested and legal parties, their design is beyond the scope of this project. However, it has been shown, through the integration of robust data on cormorant predation and fisheries, that cormorants can have damaging impacts on inland recreational fisheries in terms of fish population dynamics, angling success and economics, and that control techniques are extremely limited in their use on individual fisheries. To conclude, there is clearly a pressing requirement for a thorough review of the present legislation and practices regarding the legal protection of the cormorant.

Table 10.1 Cormorant population management methods.

Tactic	Method	Comments
Shooting	Killing cormorants at the site to enhance scaring techniques.	It is difficult to obtain a license to shoot cormorants in the UK due to protection of cormorants under the Wildlife and Countryside Act 1981 (Figure 2.2) and control on firearms.
Shooting blanks	Firing of blank cartridges to scare cormorants.	Non-lethal shooting has not been proved to scare cormorants from a fishery (Utschick 1983; McKay *et al.* 1998).
Laser light	Aiming a low-powered laser light at individuals to deter occupancy.	Although success was observed using a laser guns to deter cormorants from night-roosts, the equipment is likely to prove prohibitively expensive (McKay *et al.* 1998).
Audio scaring	Deployment of acoustic scarers, bio-acoustics and gas cannons to scare cormorants.	The method has only been successful in the short-term, it is not species-specific, and is restricted by controls on noise pollution in urban areas (Broyer *et al.* 1993; Mott and Boyd 1995; Boudewijn and Dirksen 1996; McKay *et al.* 1998).
Static visuals	Deployment of scare-crows, flags etc. to scare cormorants.	Although limited success has been observed in short time periods, long-term success is very limited due to cormorant habituation to the stimuli (Moerbeek 1984).
Intense lights	Use of search lights, strobes etc. to deter cormorant occupancy at night-roosts.	Although the method may drive cormorants from the roost, they are likely to settle at alternative sites in the region and the disturbance is liable to increase DFI (Boudewijn and Dirksen 1996).
Moving visuals	Use of "pop-up" effigies, kites and helium balloons to deter cormorant presence.	Ineffective in the long-term due to cormorant habituation to the stimuli (Inglio 1980; Stickley et al. 1995).
Raptors	Deployment of live birds of prey, or models, to deter cormorants from the site.	Although the method is effective, it is prohibitively expensive and time consuming to use over a prolonged time period (Tusnadi 1959; Festetics 1964).
Human disturbance	People scaring cormorants from the site.	Despite successes, the method can prove very expensive due to manpower costs (Moerbeek 1984; Draulans 1987; Moerbeek *et al.* 1987).
Model boats	Deployment of a model boat on the fishery to scare cormorants off the fishery.	The method would incur a large time cost and the initial outlay for the boat (McKay *et al.* 1998).
Conditioned taste aversion	Distasteful chemicals are used to condition cormorants away from fish in certain area.	The method is unproven as it has only been tested in the laboratory and is unlikely to work in a multi-species fishery (McKay *et al.* 1998).

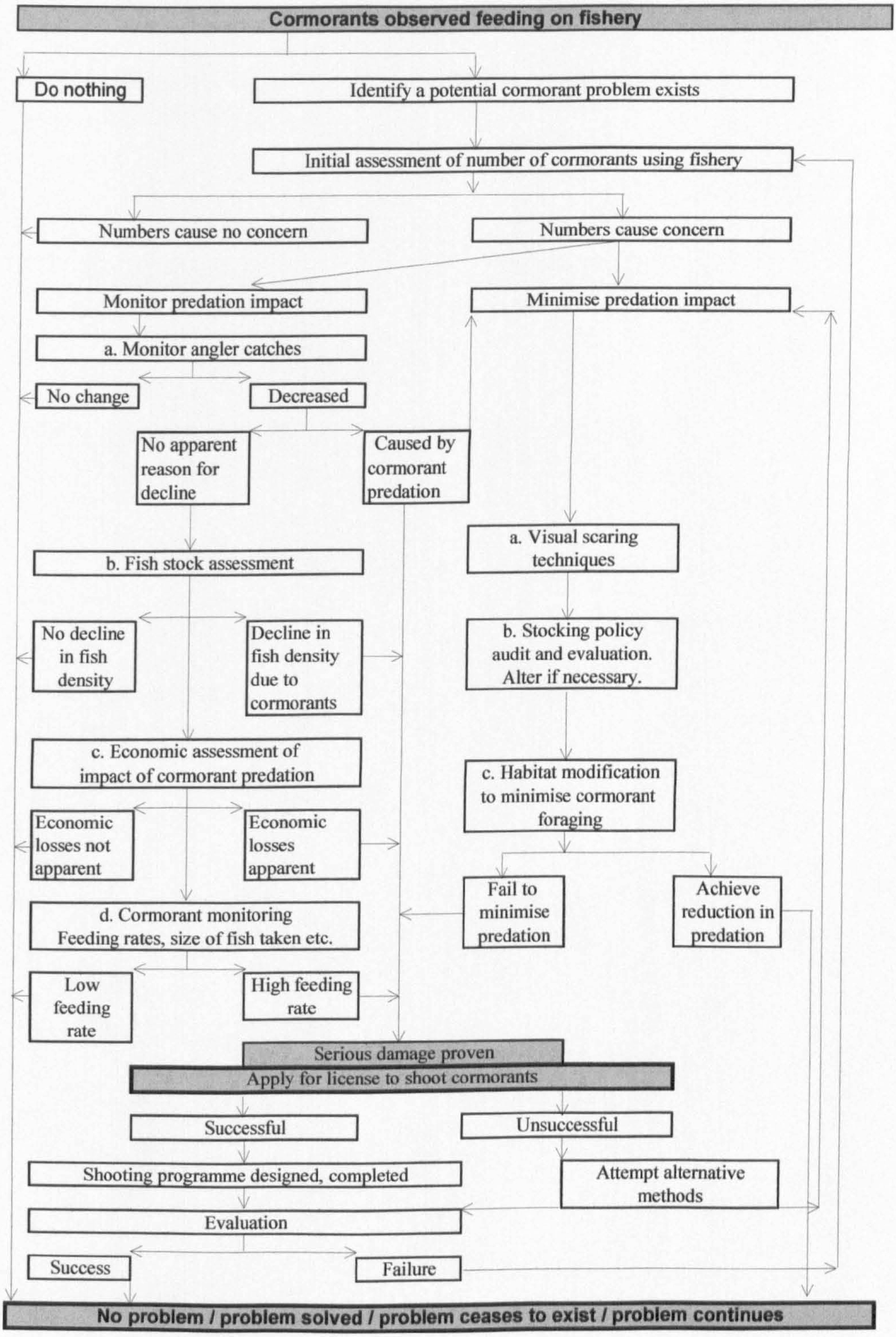


Figure 10.1 Flow chart showing a cormorant management strategy for a general UK fishery.

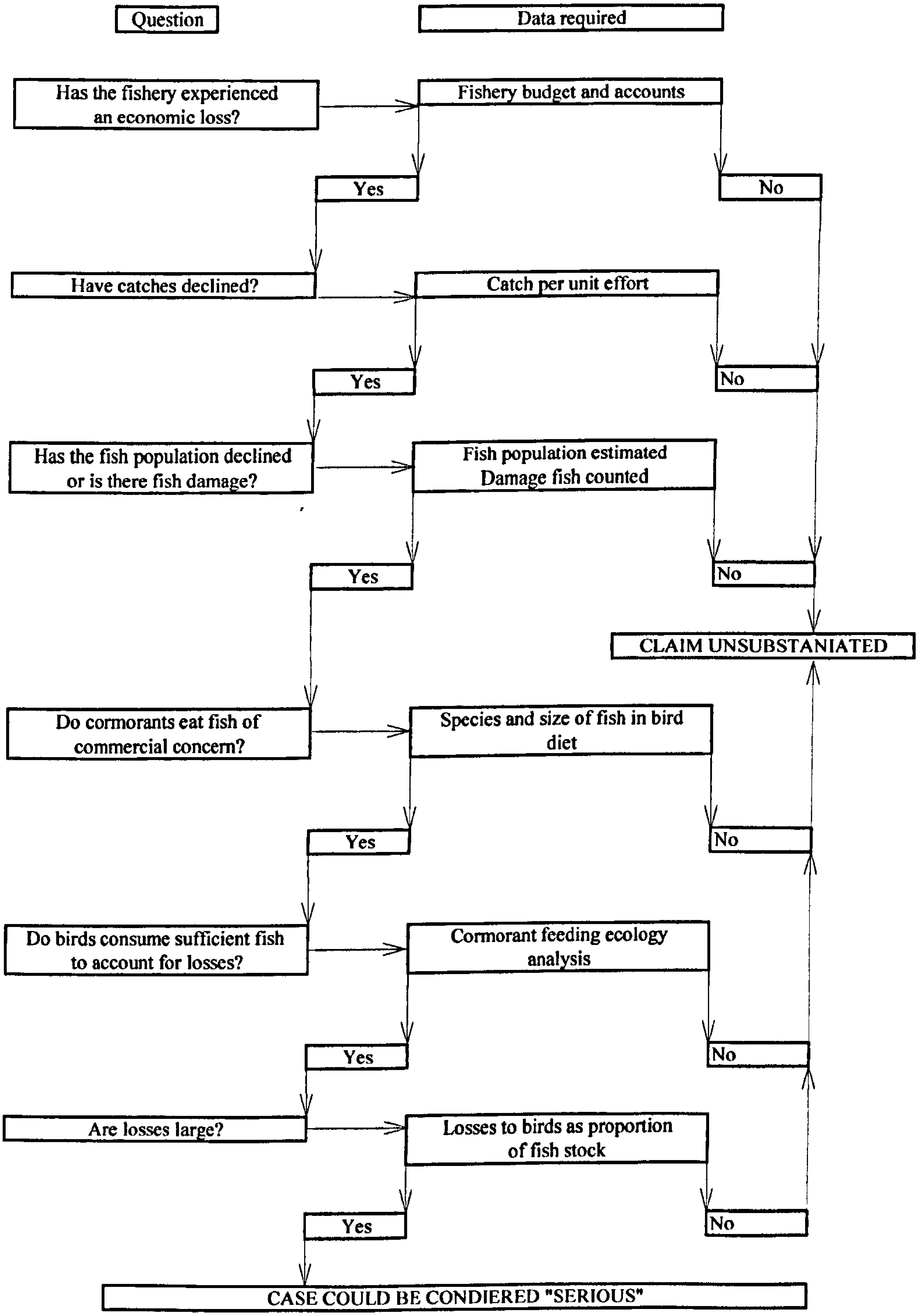


Figure 10.2 Step-wise procedure for potential damage assessment by cormorants (Carss and Marquiss 1994).

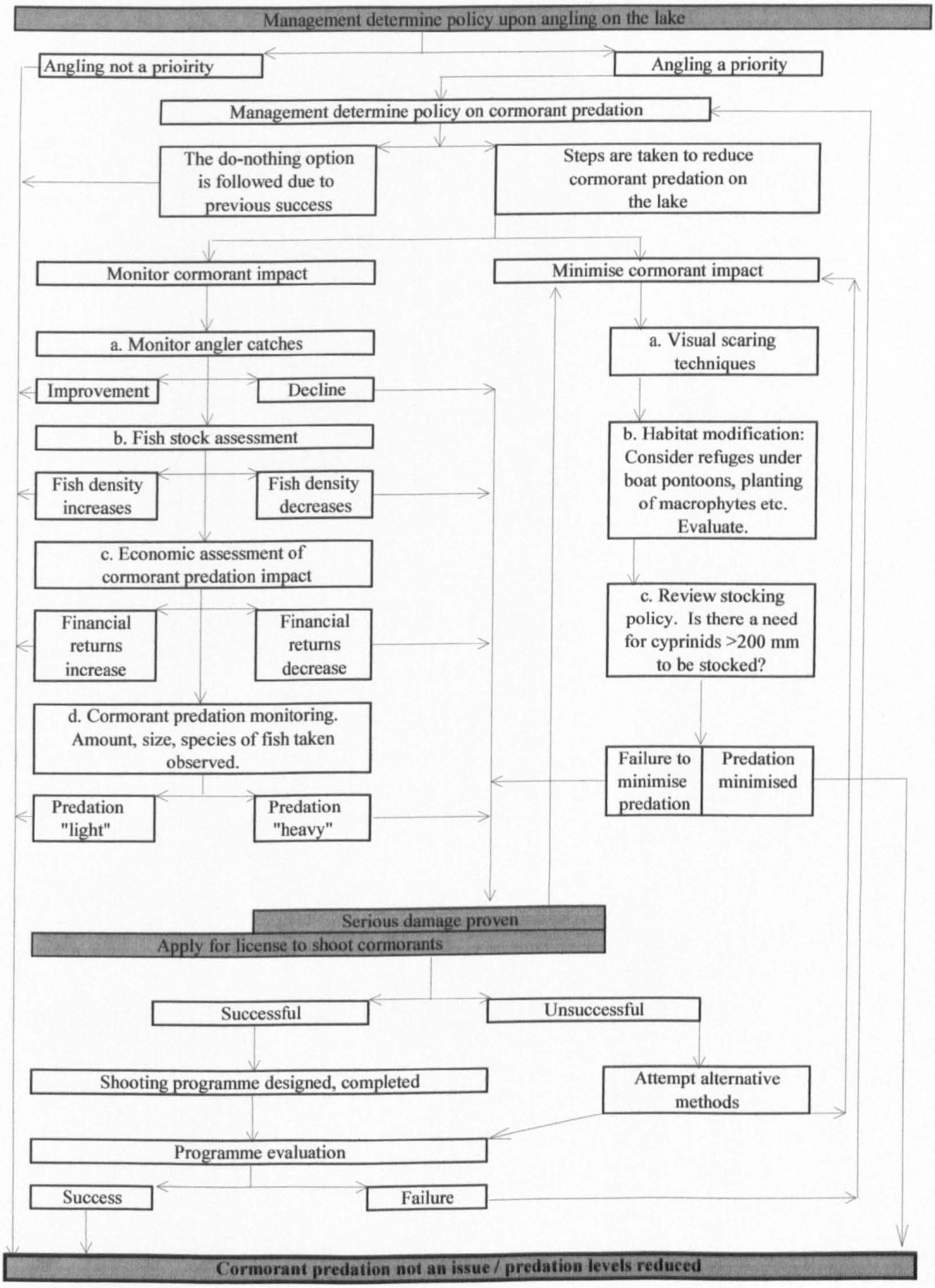


Figure 10.3 Cormorant management strategy for Holme Pierrepont Rowing Course.

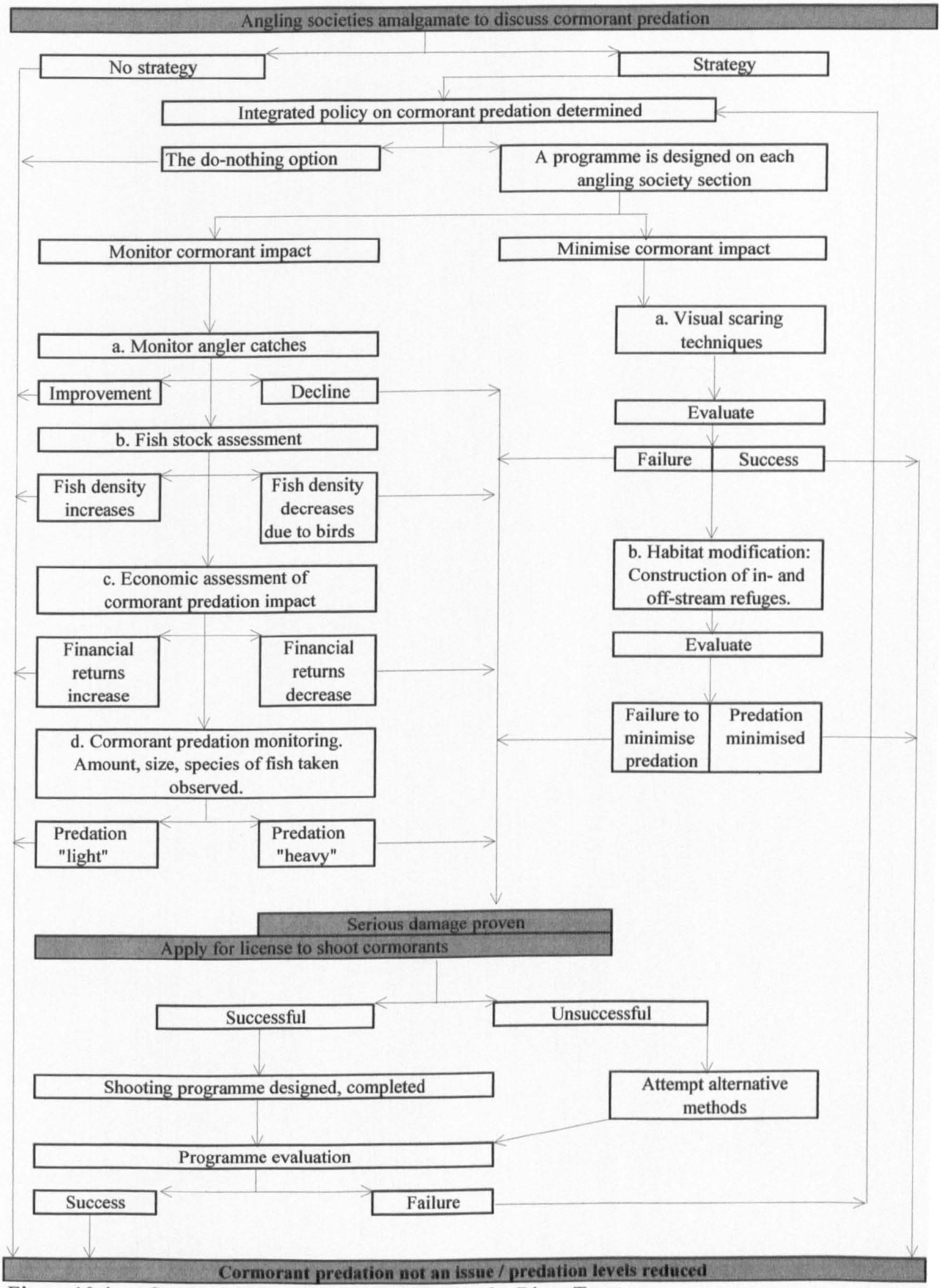


Figure 10.4 Cormorant management strategy for the River Trent.

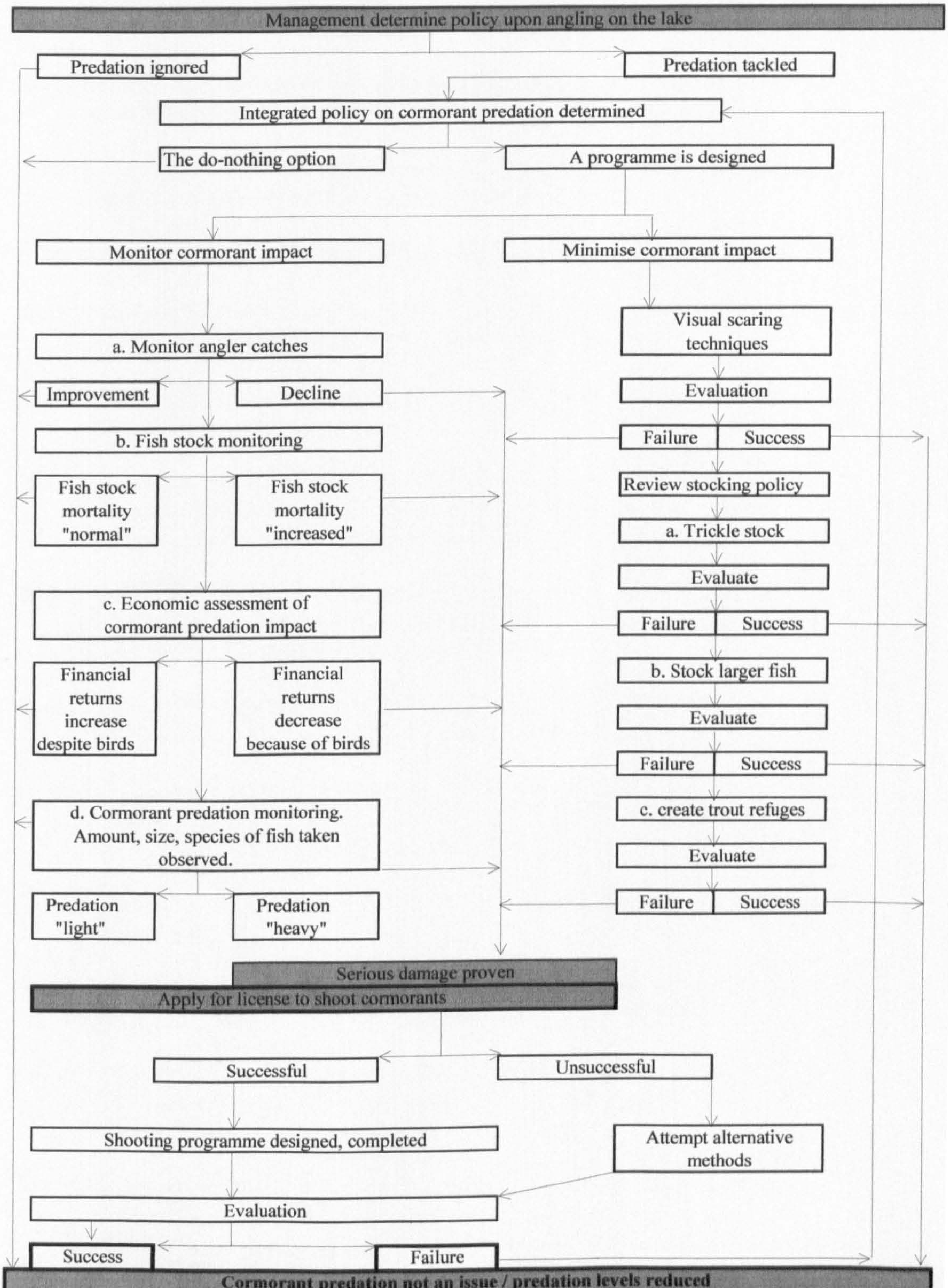


Figure 10.5 Cormorant management strategy for Colwick Park Trout Lake.

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