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**MONITORING OF THE SCHELLY OF HAWESWATER,
APRIL 2010 TO MARCH 2011**

FINAL REPORT

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EXECUTIVE SUMMARY

1. Following previous projects which had established that the status of the rare schelly (*Coregonus lavaretus*) in Haweswater of the English Lake District in north-west England was poor, the present project was commissioned to monitor the status of this population from April 2010 to March 2011 by examining entrapment records and specimens and by hydroacoustics, to undertake a series of cormorant (*Phalacrocorax carbo*) observations and roost counts over the same period, and to assess cormorant management undertaken at the lake in recent years. A further initial objective to develop modelling of schelly population dynamics was not addressed because resources were fully used by the above higher priority objectives.

2. 1 schelly (length 335 mm, weight 540 g, age 10 years, male), 84 Arctic charr (ranging in length and weight from 105 to 300 mm and 14 to 273 g, respectively) and 1 perch (*Perca fluviatilis*) (length 206 mm, weight 89 g) were entrapped from April 2010 to March 2011. The entrapment record thus demonstrated that the schelly was extant in Haweswater in early 2011 and, taking a general view of the data from recent years, provides some evidence for a limited population recovery, although the rate of increase in total entrapped schelly has noticeably slowed in recent years.

3. A hydroacoustic survey was carried out in July 2010. The population density of all fish had a geometric mean of 10.1 fish ha^{-1} (lower and upper 95% confidence limits of 4.7 and 21.6 fish ha^{-1}), while that of large fish of length greater than 250 mm was 2.4 fish ha^{-1} (lower and upper 95% confidence limits of 1.0 and 5.4 fish ha^{-1}). The latter group of fish is probably

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dominated by adult schelly and their population density converted to an estimated population size of 327 individuals with lower and upper 95% confidence limits of 144 and 741 individuals, respectively. In addition, an indeterminable number of smaller adult and young schelly are probably also present, although these cannot be differentiated from similar sizes of Arctic charr, brown trout (*Salmo trutta*) and perch (*Perca fluviatilis*). Overall, these hydroacoustic data indicate that schelly recruitment is has taken place in recent years and has led to some increase in the adult component of the population.

4. The number of roosting cormorants between April 2010 and March 2011 varied from 0 to 12 birds, with the maximum of 12 birds recorded in August 2010. No cormorant shooting or any other form of management was undertaken, but no nesting activities were observed. All roosting was confined to the island of Wood Howe. Although these cormorant counts were lower than those observed before shooting was begun in 2004, overall they tended to be higher than those recorded in 2007 when shooting was last carried out. However, there was no clear peak associated with the potential nesting season, which may represent a shift back to the pattern of cormorant residence at Haweswater observed before the first recorded nesting of 1992.

5. During 2010, local cormorant feeding behaviour was reduced to 45% of the unmanaged 1997 level. This is considerably higher than the corresponding figures of 25% and 27% for 2008 and 2009, respectively. Modelling of the schelly population suggested that this level of predation is incompatible with a population recovery and, if it is maintained or exceeded in subsequent years, the schelly can be expected to resume its drift towards local extinction.

6. Assuming that the population of adult schelly has remained above the minimum viable level and if impacts from cormorants can be successfully controlled to sustainable levels, a general improvement in spawning conditions in terms of water levels over those existing in the 1980 and early 1990s should lead to a significant recruitment to the adult spawning stock. On a more negative note, it is notable that the recent possible faltering in the recovery of the schelly population, as suggested by the entrapment record and hydroacoustic surveys, has been accompanied by a substantial increase in the amount of cormorant feeding activity at Haweswater.

7. A continued relatively low population abundance means that the current status of the schelly population of Haweswater is considered to be poor.

8. It is recommended (a) that the current monitoring of the Haweswater schelly population by a combination of examination of entrapped specimens and hydroacoustics is continued, with the frequency of the latter maintained at a single annual survey in July, (b) that communications are maintained with all stakeholders concerning the potential future control of cormorants at Haweswater through non-shooting scaring and shooting methods, (c) that the operation of the reservoir continues to minimise lake level variations particularly through the critical period of February to April, (d) that if further cormorant management is undertaken then its effects are monitored not only in terms of the numbers of cormorants shot, but also in terms of non-lethal effects on local cormorant abundance, distribution and behaviour, (e) that whether or not shooting is undertaken, the numbers of cormorants at Haweswater are monitored by a programme of at least monthly roost counts, (f) further development is undertaken of the modelling of schelly population dynamics including best

assessment of the impact of local foraging by cormorants, and (g) some consideration is given to undertaking gill-net surveys of schelly in the refuge sites of Blea Water and Small Water.

CHAPTER 1 INTRODUCTION

1.1 Background

The present investigation continues a series of projects undertaken by the Centre for Ecology & Hydrology (CEH, formerly the Institute of Freshwater Ecology) under commission to United Utilities (UU, formerly North West Water Limited (NWW)) and the Environment Agency (EA, formerly the National Rivers Authority) on the ecology and conservation of the schelly (*Coregonus lavaretus*, also known as gwyniad in Wales and powan in Scotland) in Haweswater, Cumbria. Among the freshwater fish of the U.K., the national rarity of schelly is second only to that of the closely related vendace (*Coregonus albula*), with both species being protected by the Wildlife and Countryside Act of 1981 and on the U.K. List of Priority Species and Habitats of the U.K. Biodiversity Action Plan (www.ukbap.org.uk). These studies, the first field work of which was carried out in 1991, are reported in Winfield *et al.* (1994a), Winfield *et al.* (1994b), Winfield *et al.* (1995), Winfield *et al.* (1999), Winfield *et al.* (2001a) and a series of annual monitoring reports the three most recent of which are Winfield *et al.* (2008), Winfield *et al.* (2009) and Winfield *et al.* (2010). Findings have also been published in Winfield *et al.* (1996), Winfield *et al.* (1998a), Winfield *et al.* (2002a), Winfield *et al.* (2003a), Winfield *et al.* (2004a) and Winfield *et al.* (2007a).

Prior to the 1990s, the only publications concerned with this schelly population were those of Swynnerton & Worthington (1940), Bagenal (1970) and Maitland (1985), which were concerned with diet in the 1930s and population biology in the 1960s and 1980s. A limited amount of further information on the schelly from the 1970s and 1980s is available in the

unpublished theses of Broughton (1972) and Mubamba (1989). Reviews of schelly ecology and Haweswater as a habitat for this species are given in Winfield *et al.* (1994b) and Winfield *et al.* (1995), while Talling (1999) summarises other aspects of this oligotrophic reservoir.

The above investigations revealed that the schelly population of Haweswater in the 1990s was of very poor status, both when compared with the contemporary status of other schelly populations in England and Wales (Winfield *et al.*, 1994a; Winfield *et al.*, 1996) and when compared with its own earlier status in the 1960s (Winfield *et al.*, 1994b; Winfield *et al.*, 1995; Winfield *et al.*, 2002a). This deterioration has been attributed to increases in lake level variability from the 1960s to the mid 1990s, particularly during the critical approximately February to April spawning season and egg incubation period, arising due to its operation as a drinking water reservoir (Winfield *et al.*, 1998a; Winfield *et al.*, 2004a).

As a result of this concern, several conservation actions have been instigated at Haweswater including the development of an artificial spawning substratum system, a more sympathetic lake level management regime (both summarised in Winfield *et al.*, 2003a), and attempts to establish two new refuge populations of Haweswater schelly in nearby Blea Water and Small Water by the introduction of schelly eggs in early 1997 (Winfield *et al.*, 1997). Subsequent hydroacoustic and fyke-netting surveys in 2000 found no evidence of successful establishment in Blea Water, but possible evidence of success in Small Water through the detection of echoes that may have originated from schelly (Winfield *et al.,* 2001a). Confirmation of the establishment of a self-recruiting population of schelly in Small Water was produced in October 2002 by the sampling by gill net of eight schelly of 3 year classes, none of which was that of the original introduction as eggs (Winfield *et al.*, 2003b). In

addition, it now appears that the second introduction of schelly to Blea Water (Winfield *et al.*, 1997) has been successful following the capture, and return alive, of a schelly of length approximately 150 mm by an angler on 29 May 2005 (P. Corkhill, *pers. comm.*).

In addition to the above studies, the potential impact on the schelly population of Haweswater of a recently established local breeding colony of cormorants (*Phalacrocorax carbo*) has also been assessed through extensive field observations in 1996 and 1997 and found to be a cause for concern (Winfield *et al.*, 1998b). Adopting the precautionary principle, scaring measures to prevent further nesting have subsequently been undertaken by UU (as NWW) in 1999 and 2000 (Winfield *et al.*, 2003a), and from 2002 to 2004. Scaring activities were prevented in 2001 by local control measures against foot-and-mouth disease. However, even in years when scaring was implemented, its effect was limited to preventing successful nesting and it did not reduce the numbers of adult cormorants at the lake (Winfield *et al.*, 2003a). Consequently, it was recommended that shooting of adult cormorants be undertaken as the only means of further reducing the impact of foraging by cormorants on the schelly population (Winfield *et al.*, 2003a). Limited shooting was carried out for the first time in 2004 and continued to 2006, resulting in a total of 29 cormorants being shot over the 3 years (Winfield *et al.*, 2007b). A small amount of shooting was performed in 2007, although no birds were actually shot, and in subsequent years no further shooting has been attempted because of the much lower numbers of cormorants present (Winfield *et al.*, 2008; Winfield *et al.*, 2008; Winfield *et al.*, 2008). This management action was not undertaken lightly, but done so on the basis of an overall consideration of conservation issues at Haweswater. A considered discussion of this sensitive issue, including its wider conservation implications in the English Lake District, is given by Winfield *et al.* (2003a).

Given the above gravity of the situation relating to the conservation of the schelly in Haweswater, a monitoring programme for this nationally and internationally important population is clearly highly desirable. Such a programme has now been in place since 1997, with previous annual reports given in Winfield *et al.* (1999), Winfield *et al.* (2000), Winfield *et al.* (2001b), Winfield *et al.* (2002b), Winfield *et al.* (2003b), Winfield *et al.* (2004b), Winfield *et al.* (2005), Winfield *et al.* (2006), Winfield *et al.* (2007b), Winfield *et al.* (2008), Winfield *et al.* (2009) and Winfield *et al.* (2010). This programme forms the core investigation of the present project.

1.2 Objectives

The objectives of the present project were to monitor the status of the schelly in Haweswater from 1 April 2009 to 31 March 2010 by examining entrapment records and specimens and by hydroacoustics, to undertake a series of cormorant observations and roost counts, and to assess cormorant management undertaken at the lake in recent years in terms of its long-term effect in reducing the predation impact of cormorants on the schelly population. A further initial objective to develop modelling of schelly population dynamics including best assessment of the impact of local foraging by cormorants was not addressed because resources were fully used by the above higher priority objectives.

CHAPTER 2 ENTRAPMENT

2.1 Introduction

The entrapment of fish at Haweswater was described in detail in Winfield *et al.* (1994b), in which long-term records of such individuals collected by UU (as NWW) staff between 1 April 1972 and 31 March 1994 were used to investigate changes in schelly population abundance between the time of that study and the time of a comparable investigation in the mid 1960s by Bagenal (1970). In addition, the collection and examination of entrapped specimens during the study of Winfield *et al.* (1994b) allowed a comparison of biological features of the 1990s schelly population with those recorded in the 1960s by Bagenal (1970). Further data were collected for the period of 1 April 1994 to 31 March 2010 within the Haweswater schelly monitoring programme reported most recently by Winfield *et al.* (2010).

The studies of Winfield *et al.* (1994b), Winfield *et al.* (1995), Winfield *et al.* (1999) and the most recent full report of the present monitoring programme (Winfield *et al.*, 2010) revealed that the numbers of schelly and Arctic charr (*Salvelinus alpinus*) entrapped between 1973 and early 2010, which accounted for over 90% of all fish entrapped during this period, had declined. Furthermore, detailed analysis by Winfield *et al.* (1994b) to incorporate the influence of changing sampling effort arising from variations in the volume of water abstracted indicated that this decline reflected a real decrease in the population abundances of schelly and Arctic charr in Haweswater.

Examination of biological specimens of schelly by Winfield *et al.* (1994b) showed that this decline had been accompanied by a reduction in the equitability of age classes and a reduction in growth rate when fish entrapped during the early 1990s were compared with those examined during the mid 1960s by Bagenal (1970). Subsequent entrapment records and entrapped specimens remained consistent with the interpretation of a declining schelly population with generally little recruitment since 1990 (Winfield *et al.*, 1995; Winfield *et al.*, 1999, although with some indication of improvement in recent years (Winfield *et al.*, 2010).

More specifically with respect to this more encouraging trend, Winfield *et al.* (2005) noted that 13 schelly ranging in length from 223 to 332 mm were entrapped during January to March 2005, which followed a report by Winfield *et al.* (2004b) that 24 schelly ranging in length from 228 to 336 mm had been entrapped during the same period in 2004. Given the extreme scarcity of entrapped schelly in previous years, this was an encouraging development because although the absolute numbers involved were still relatively low, the observed size range was greater than that recorded during the early 1990s by Winfield *et al.* (1994b). However, it remained smaller than that observed during the 1960s by Bagenal (1970) and so did not in itself represent cause for any alteration to the status of the population in early 2005, which was considered to be extremely poor (Winfield *et al.*, 2005). Although no schelly were subsequently entrapped from April 2005 to March 2006, because no other fish species were entrapped during this period Winfield *et al.* (2006) attributed this failure to entrapment collection problems rather than to a sudden and further decline in the lake population. More recently, Winfield *et al.* (2007b) recorded the entrapment of 15 schelly ranging in length from 216 to 350 mm during January to March 2007, Winfield *et al.* (2008) reported a further 11 individuals ranging in length from 294 to 346 mm during the corresponding period of 2008,

Winfield *et al.* (2009) reported a further 4 individuals ranging in length from 310 to 361 mm during the corresponding period of 2009, and finally Winfield *et al.* (2010) reported a further 4 individuals ranging in length from 216 to 347 mm during the corresponding period of 2010.

One further source of data relevant to the present study was discovered by the authors during 2001 in the form of a brief unpublished study of entrapped Haweswater schelly from the early 1970s. Broughton (1972) found entrapped schelly from a 1972 sample to include a wide range of individual lengths, including young individuals, but a depressed growth pattern similar to that subsequently recorded in the 1990s. In the 1980s, specimens entrapped in 1983 and examined but not aged by Maitland (1985) included only large individuals. Furthermore, the numbers of schelly entrapped since 1973 showed a marked decrease in the early 1980s between the sampling periods of Broughton (1972) and Maitland (1985). This information is useful in the context of the long-term interpretation of population dynamics at Haweswater because it fixes the start of the deterioration in the status of schelly to the early 1970s, which is consistent with the interpretation given earlier of an overriding influence of increasing lake level fluctuations (Winfield *et al.*, 1998a).

The objective of this part of the present project was to monitor the entrapment of schelly and other fish species from 1 April 2010 to 31 March 2011.

2.2 Methods

Fish entrapped by the abstraction system at Haweswater have for many years been transported by aquaduct to the Garnett Bridge filtration plant near Watchgate Water

Treatment Works, or the latter installation itself, near Kendal, where they are removed by meshes or, as is the case in recent years, by hand netting. Near-daily records of these fish have been kept by NWW, and subsequently by UU, since 1 April 1972 and data for the period up to 31 March 2010 were obtained and analysed in a series of studies including those of Winfield *et al.* (1994b), Winfield *et al.* (1995), Winfield *et al.* (1999), Winfield *et al.* (2006), Winfield *et al.* (2007b), Winfield *et al.* (2008), Winfield *et al.* (2009) and Winfield *et al.* (2010). Arrangements were made to obtain equivalent data from 1 April 2010 to 31 March 2011, i.e. through the period of the present project.

In addition, and again with the co-operation of UU as previously undertaken in the above studies, arrangements were made for the collection of all fish not in a state of decay (which was itself rare given the prevailing low water temperatures) and their storage in dated plastic bags in a freezer at -20 °C to await collection by CEH staff for standard processing as described below.

Entrapped fish were subsequently returned to the laboratory where at a later date they were thawed, identified, measured (fork length to nearest mm), weighed (wet weight to nearest g), sexed and their reproductive state classified as immature, mature, mature and ripe (hereafter abbreviated to ripe), mature and running (hereafter abbreviated to running), or mature and spent (hereafter abbreviated to spent). Opercular bones and otoliths were also removed from any schelly and Arctic charr, respectively, for subsequent ageing as described by Mubamba (1989), although the remit of the present project only included processing of material from the former species.

2.3 Results

1 schelly (length 335 mm, weight 540 g, age 10 years, male), 84 Arctic charr (ranging in length and weight from 105 to 300 mm and 14 to 273 g, respectively) and 1 perch (*Perca fluviatilis*) (length 206 mm, weight 89 g) were entrapped from April 2010 to March 2011.

Long-term trends in the numbers of schelly and Arctic charr, and perch and brown trout (*Salmo trutta*) entrapped from January 1973 to March 2011 (incorporating data from previous monitoring reports and Winfield *et al.* (2000)) are given in Fig. 1 and Fig. 2, respectively.

Length and age frequency distributions of schelly entrapped from April 2010 to March 2011 are given in Fig. 3.

2.4 Discussion

Following completion of the extensive engineering works at Watchgate Water Treatment Works over the period 2001 to 2003, and thus the ending of its associated disruption to entrapment monitoring, Winfield *et al.* (2005) noted the encouragement subsequently offered by the recording of 24, 24 and 13 schelly in 2003, 2004 and early 2005, respectively. These biological observations confirmed the assumption of the hydroacoustic analyses and interpretation of recent years that the schelly was still extant in Haweswater. A failure to record any entrapped schelly in the first part of 2006 was subsequently disappointing, although not necessarily alarming because of a suspicion that it was due to technical rather than biological reasons (Winfield *et al.*, 2006). This optimism was subsequently justified by

the entrapment of 3 schelly in late 2006 (Winfield *et al.*, 2007b), 18 schelly in 2007 (Winfield *et al.*, 2008), 11 schelly in 2008 (Winfield *et al.*, 2009), 5 schelly in 2009 (Winfield *et al.*, 2009) and 4 schelly in the first three months of 2010 (Winfield *et al.*, 2010).

In this context, taken together the numbers of entrapped schelly and Arctic charr recorded since April 2010 continue to be somewhat encouraging, although the rate of increase in total entrapped schelly has noticeably slowed in recent years. For schelly, the very low number of individuals entrapped so far in 2011 is comparable with the low levels of entrapment observed in the late 1990s. The situation for entrapped Arctic charr is more positive, with a general and substantial increase being recorded since the completion of the disruptive engineering works in 2003.

For schelly, the biological features of the single individual entrapped since April 2010 also offered little encouragement. Specifically, the observed length of 335 mm was clearly much narrower than that of 216 to 347 mm recorded in the previous reporting period (Winfield *et al.*, 2010). However, small entrapped schelly have thus been few in number in recent years and the current length range remains far from spanning that of 50 mm to in excess of 350 mm reported for entrapped schelly in the 1960s (Bagenal, 1970). Similarly, the single age observed since April 2010 of 10 years was clearly much narrower than that of 2 to 10 years recorded over the previous 12 months (Winfield *et al.*, 2010) and in the 1960s (Bagenal, 1970).

The entrapment study has thus invaluably demonstrated that the schelly is still extant in Haweswater in early 2011 and, taking a general view of the data from recent years, provides

some evidence for a limited population recovery. However, it also indicates that population abundance is still some way below that recorded in the 1960s by Bagenal (1970) prior to the marked reduction observed during the early 1980s. The issue of the present abundance of the schelly population of Haweswater is returned to and considered in a more quantitative way in the hydroacoustics study of Chapter 3.

CHAPTER 3 HYDROACOUSTICS

3.1 Introduction

In all previous recent population studies of the Haweswater schelly (Winfield *et al.*, 1994a; Winfield *et al.*, 1994b; Winfield *et al.*, 1995; Winfield *et al.*, 1999; Winfield *et al.*, 2004a; Winfield *et al.*, 2006; Winfield *et al.*, 2007b; Winfield *et al.*, 2008; Winfield *et al.*, 2009; Winfield *et al.*, 2010), the status of the population has been found to be poor in terms of its recruitment record and abundance as indicated by low catch-per-unit-effort or low abundance during hydroacoustic surveys when compared with sampling elsewhere. For example, a netting survey in May 1993 involving the deployment of nine survey gill nets resulted in the capture of just two individuals (Winfield *et al.*, 1994b), while one in May 1996 involving five survey gill nets also produced just two schelly (Winfield *et al.*, 1999).

Even entrapment, which produced significant schelly samples for biological analysis of 67 and 63 individuals as recently as 1993 and 1994, respectively (Winfield *et al.*, 1995), subsequently produced a total of only 37 individuals in the following eight years up to and including 2002, albeit with significant disruption to sampling in the later years (Winfield *et al.*, 2003b). Somewhat encouragingly, post-disruption entrapment reported in Chapter 2 has now amounted to 103 individuals, although its rate of increase has slowed in recent years.

Given the above situation, hydroacoustics has become increasingly important as a method of monitoring the abundance of schelly in Haweswater. This technique was first used in an extensive survey of the schelly and other fish populations of this lake in May 1992 by

Winfield *et al.* (1994b). Although this technique is not without its own problems of interpretation, it gives an estimate of population abundance with confidence limits and has the advantage of being completely non-destructive.

Following the conclusion by Winfield *et al.* (2007b) that the three (May, July and September) surveys conducted each year from 1997 to 2006 could now be justifiably reduced to a single annual survey, this approach was subsequently adopted in 2007, 2008 and 2009 during which a single survey was performed in July of each year (Winfield *et al.*, 2008; Winfield *et al.*, 2009; Winfield *et al.*, 2010). The objective of this part of the present project was to survey the schelly and other fish populations by a hydroacoustic survey in July 2010.

3.2 Methods

3.2.1 Field work

A hydroacoustic survey was carried out on 23 July 2010 using a BioSonics DT-X echo sounder with a 200 kHz split-beam vertical transducer of beam angle 6.5° operating under the controlling software Visual Acquisition Version 6.0.1.4318 (BioSonics Inc, Seattle, U.S.A.). Throughout the surveys, data threshold was set at -130 dB, pulse rate at 5 pulses s^{-1}, pulse width at 0.4 ms, and data recorded from a range of 2 m from the transducer. In addition to the real-time production of an echogram through a colour display on a laptop computer, data were also recorded to hard disc. The system was deployed from a 4.8 m inflatable dinghy powered by a 25 horse power petrol outboard engine and moving at a speed of approximately 2 m s^{-1}, depending on wind conditions. The transducer was positioned approximately 0.5 m

below the surface of the water. Navigation was accomplished using a Garmin GPSMAP 60CSx GPS (Global Positioning System) (www.garmin.com) with accuracy to less than 10 m, while a JRC Model DGPS212 GPS (www.jrc.co.jp) with accuracy to less than 5 m inputted location data directly to the hydroacoustic system where they were incorporated into the recorded hydroacoustic data files. Prior to the surveys, the hydroacoustic system had been calibrated using a tungsten carbide sphere of target strength (TS) -39.5 dB at a sound velocity of 1470 m s^{-1}.

Following Winfield *et al.* (1994b), the hydroacoustic survey was undertaken in full daylight by a zig-zag survey incorporating a total of 15 transects across the entire lake, of which 10 were over the original lake (Table 1, Fig. 4). The survey was run from the south to the north, was of approximately 75 minutes duration, and for the area of the lake greater than 5 m in depth gave a ratio of coverage (length of surveys : square root of research area) of 7:1.

3.2.2 Laboratory examination and analysis

Data analysis in the laboratory was performed by trace formation, which is also known as fish tracking, using SonarData Echoview Version 3.40.47.1551 (Myriax, Hobart, Australia, www.echoview.com) with a target threshold of -70 dB applied individually to each transect of the surveys. Default software settings were used for all single target and track detections. In this context, the term 'trace' is synonymous with 'fish', each being composed of a number of echoes.

Mean target strength of each trace produced by Echoview was converted to fish length using the relationship described by Love (1971),

$$TS = (19.1 \log L) - (0.9 \log F) - 62.0$$

where TS is target strength in dB, L is fish length in cm, and F is frequency in kHz.

Mean target strength of each trace was then categorised into 'small' (i.e. -52 to -45 dB, length 40 to 99 mm), 'medium' (-44 to -37 dB, length 100 to 249 mm) or 'large' (greater than -37 dB, length greater than 250 mm) length classes. The large size class was probably mainly adult schelly, while the small and medium size classes were probably composed of a combination of schelly, Arctic charr, brown trout and perch. A detailed justification for these assumptions is given by Winfield *et al.* (1994b) following previous extensive hydroacoustic surveys and netting at Haweswater. Traces of each transect were also categorised into 1 m deep strata from a depth of 2 m below the water surface down to the lake bottom. Such counts were then converted to fish densities for each transect expressed as individuals per hectare of lake surface area by the use of a spreadsheet incorporating the insonification volume for each depth stratum.

Following Jurvelius (1991) and Baroudy & Elliott (1993), the average density of each length class of fish during the survey was calculated as the geometric mean with 95% confidence limits of the component transects. Such calculations were performed for both the 15 transects (where available) covering the entire lake, and for the 10 transects (where available) covering the original lake. However, only the latter calculations are given in the present report

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because adult schelly have never been netted outside the southern limit of the original lake. The absolute population size of adult schelly was calculated by multiplying the geometric mean (with 95% confidence limits) density of large fish in individuals per hectare by the surface area of the original lake, i.e. 138 ha (Ramsbottom, 1976).

As 2010 was the ninth year in which a BioSonics split-beam echo sounder (DT6000 for 2002 to 2004, DT-X for 2005 to 2010) had been used in the present monitoring as a replacement for an older and less sophisticated Simrad EY 200P portable echo sounder, all fish population densities produced earlier using the older system were converted to values that would have been recorded by the BioSonics machines using a series of inter-calibration relationships determined during 2003 (CEH, unpublished data). Only these converted values are presented and considered in this report.

3.3 Results

The population densities of small, medium, large and all fish recorded at Haweswater during July 2010 are given in Table 2, together with corresponding data from July 2008 and July 2009 from Winfield *et al.* (2010). In 2010, the population density of all fish had a geometric mean of 10.1 fish ha^{-1} (lower and upper 95% confidence limits of 4.7 and 21.6 fish ha^{-1}), while that of large fish, which were assumed to be dominated by schelly, was 2.4 fish ha^{-1} (lower and upper 95% confidence limits of 1.0 and 5.4 fish ha^{-1}).

The estimated geometric mean of the population size of small fish was 704 individuals with lower and upper 95% confidence limits of 323 and 1537 individuals, respectively. For

medium fish, the corresponding figure was 315 individuals with lower and upper 95% confidence limits of 143 and 696 individuals, respectively, while for large fish it was 327 individuals with lower and upper 95% confidence limits of 144 and 741 individuals, respectively. The estimated geometric mean of the population size of all fish was 1397 individuals with lower and upper 95% confidence limits of 654 and 2982 individuals, respectively.

Finally, the above estimated population sizes for July 2010 can be put into a longer-term context using corresponding data for this month of 1997 to 2009 sourced from Winfield *et al.* (2010) and earlier reports. Such time series for small, medium, large and all fish are presented in Figs 5, 6, 7 and 8, respectively.

3.4 Discussion

In the present analysis, large fish as identified by hydroacoustics are assumed to be dominated by adult schelly, while medium and small fish echoes are likely to comprise varying proportions of schelly, Arctic charr, brown trout and perch. This interpretation was first made by Winfield *et al.* (1994b) for reasons explained in detail therein and there have since been no changes in the Haweswater fish community which would invalidate this argument. In essence, this situation means that only adult schelly in excess of 250 mm in length can be reliably identified to species and counted as such by hydroacoustics, with younger and smaller individuals being irretrievably mixed with Arctic charr, brown trout and perch. Thus, all population densities and population sizes of adult schelly presented in this report relate only to those individuals in excess of 250 mm in length. However, unless

recruitment fails completely for a number of years, there will thus always be additional numbers of younger and smaller schelly present within the less tractable small and medium length classes of fish. Without extensive netting operations, which are not recommended, it is unfortunately impossible to produce a current quantification of this younger component of the schelly population. Notwithstanding the issue of species identification, it is clear that there has been considerable variation in the estimated population sizes of small, medium and large fish since the first hydroacoustic survey of Haweswater in May 1992 by Winfield *et al.* (1994b).

Small fish, which are likely to comprise not only young schelly but also significant numbers of young Arctic charr, brown trout and perch, were generally more abundant in the period 1992 to 2000 than they have been from 2002 to recent years (Winfield *et al.*, 2009) and their abundance in 2010 conformed to this pattern. However, it is encouraging that they showed an appreciable increase in abundance within the courses of the four summers of 2003 to 2006, although the single survey strategy adopted for 2007 onwards means that such dynamics can no longer be recorded (Winfield *et al.*, 2008). The dynamics of this length class of fish are a function not only of recruitment, which tends to increase their abundance, but also of their individual growth which ultimately takes them out of this length class and so tends to decrease their abundance. Consequently, the fall in small fish abundance observed between the end of one summer and the beginning of the next in 2003, 2004 and 2005 reflects in substantial part the progression of individual fish from the small to medium length classes (Winfield *et al.*, 2008).

The consequences of the above patterns in small fish abundance are apparent in the abundance of medium fish, which again are likely to comprise not only schelly but also significant numbers of Arctic charr, brown trout and perch. Following a low in 2002, the abundance of this length class was consistently higher in 2003, 2004 and 2005, although somewhat similar in 2006 and 2007. Abundance again increased in 2008 and this was maintained in 2009, although it was followed in 2010 by a slight decrease and the considerably higher abundance of 1992 reported by Winfield *et al.* (2008) has not yet been attained. Even though this length class of fish does not include the largest adult schelly, at least some of its schelly members are likely to be reproductively active (see Winfield *et al.*, 1994a) and so can potentially contribute to the recovery of the population.

The abundance of large fish, assumed to be dominated by adult schelly, has now clearly stopped its medium-term decline observed from 1992 to 2002. Furthermore, recent years have shown an increasing, albeit erratically so, overall trend in mean estimated abundance and this pattern was maintained in 2010. However, large individuals do remain extremely scarce and, like medium fish, are still considerably less abundant than they were in 1992 as reported by Winfield *et al.* (2008). Overall, these hydroacoustic data indicate that schelly recruitment has taken place in recent years and has led to some increase in the adult component of the population.

In a wider U.K. context, the population densities of small, medium and large fish in Haweswater as revealed by the hydroacoustic surveys continue to be very low when compared with figures for fish communities including schelly elsewhere in England and Wales (Winfield *et al.*, 1994a) and Scotland (Winfield *et al.*, 2006b). For large fish, taken

here to be adult schelly, the observed range of mean densities in Haweswater between July 1997 and July 2010 of from 1 to 4 individuals ha^{-1} may be compared with a figure of 12 individuals ha^{-1} for adult schelly in the essentially unsuitable eutrophic Wahnbach Reservoir, Germany (Brenner *et al.*, 1987), a range of 25 to 70 adult schelly ha^{-1} for the fished Lake Constance, Germany (Eckmann, 1995), and a range of 10 to 70 adult schelly ha^{-1} found in Lake Osensjoeen, Norway (Linloekken, 1995). Clearly, the population densities of adult schelly in Haweswater in the late 1990s, the 2000s and 2010 are very low when compared with populations elsewhere in Europe.

An interpretation of these changes in relation to the effects of recent trends in lake levels, impacts by cormorants (see Winfield *et al.*, 1998b; Winfield *et al.*, 2003a; Winfield *et al.*, 2004c; Winfield *et al.*, 2007a) and the absolute size of the schelly spawning stock (see Winfield *et al.*, 1999) is made in the final chapter.

CHAPTER 4 CORMORANT OBSERVATIONS AND ROOST COUNTS

4.1 Introduction

A continuous and at least monthly count of cormorants roosting at Haweswater, which was started in February 2002 and for which earlier comparable data had been collected between November 1996 and December 1997 by Winfield *et al.* (1998a), has been maintained up to March 2010 (Winfield *et al*., 2010). In addition to their contribution to the study of cormorant ecology at Haweswater, such counts are invaluable in the assessment of management measures as considered further in Chapter 5.

The objective of this part of the present project was to continue the above data series by undertaking cormorant roost counts at Haweswater from April 2010 to March 2011.

4.2 Methods

The numbers of cormorants roosting at Haweswater, primarily on or near the island of Wood Howe (54°, 29.987' North, 2°, 48.567' West), were counted from April 2010 to March 2011 mainly by a volunteer (Nick Lloyd) using x10 binoculars and occasionally by the senior author using x8 binoculars and a x20 telescope. Regular counts were made at last light at approximately 2 week intervals between April and September 2010, then at approximately 4 week intervals between October 2010 and January 2011, and then again at approximately 2 week intervals between February and March 2011.

All observations followed the detailed methodology of Winfield *et al.* (1998a).

4.3 Results

The number of roosting cormorants varied from 0 to 12 birds, with the maximum of 12 birds recorded in August 2010 (Fig. 9). No nesting activities were observed.

In contrast to observations made occasionally during some previous years, no cormorants were seen to roost high in tall trees on the west shore of the lake (54°, 30.116' North, 2°, 48.917' West), on the shore of the west shore of the lake a considerable distance north of Wood Howe (54°, 31.908' North, 2°, 47.530' West), nor high in tall trees of The Rigg on the south-west shore of the lake (54°, 29.854' North, 2°, 48.643' West).

The above counts are put into a longer-term context in Fig. 10 which shows all such available data between November 1996 and March 2011.

4.4 Discussion

The results of this component of the study are primarily of value in the context of their contribution to the longer-term monitoring of roosting cormorants at Haweswater which facilitates assessment of local management measures, as considered further in Chapter 5. Consequently, their discussion here will be brief.

The observations of 2010 and early 2011 were notable in three respects. Firstly, although these cormorant counts were lower than those observed before shooting was begun in 2004, overall they tended to be higher than those recorded in 2007 when shooting was last carried out. Secondly, the monthly pattern of abundance contrasted with the pattern typical of pre-2007 years when it displayed a substantial peak in the late spring or early summer followed by a general decline. Instead, numbers were relatively steady and low throughout the year. Thirdly, no nesting activities were observed at any time during 2010. It appears that in the last four years of 2007 to 2010, cormorants have been using Haweswater not as a nesting site but rather as a post-nesting dispersion site or as an over-wintering site, which may represent a shift back to the pattern of cormorant residence at Haweswater observed before the first recorded nesting of 1992.

Cormorant spatial distributions in 2010 were similar to those of 2005 to 2009 in that no birds roosted high in tall trees of the west shore of the lake near the island of Wood Howe as first recorded during non-shooting scaring in early April 2003 (Winfield *et al.*, 2003b), and subsequently shown in a more marked manner during April and June 2004 when shooting was being carried out (Winfield *et al.*, 2005). Furthermore, in 2010 as in 2007 to 2009 they also did not roost on the shore of the west shore of the lake a considerable distance north of Wood Howe as they were observed to do in August and September 2005 (Winfield *et al.*, 2006), nor high in tall trees of The Rigg on the south-west shore of the lake as they did occasionally after the shooting period in November 2005 and February 2006 (Winfield *et al.*, 2006) and September 2006 (Winfield *et al.*, 2007b).

Notwithstanding the above encouraging observations, as first suggested by Winfield *et al.* (2004b) it remains a possibility that any non-shooting scaring or inefficient shooting may lead to nesting in such diverse locations in future years, where any further management of the breeding colony would be extremely difficult. As a result, it is again recommended that if any future shooting is carried out it should be undertaken as efficiently as possible, i.e. birds are shot rather than simply scared.

CHAPTER 5 ASSESSMENT OF CORMORANT MANAGEMENT

5.1 Introduction

As described in more detail in Chapter 1, investigations at Haweswater have indicated that the feeding activities of a local breeding colony of cormorants founded in 1992 have had a significant negative impact on its population of schelly.   Such concerns led to the introduction by UU of scaring procedures to stop the production of young cormorants at the colony from 1999 onwards, with the exception of 2001 when scaring activities were prevented by local control measures against foot-and-mouth disease, but the presence of adult cormorants continued to pose a threat to the schelly population (Winfield *et al.*, 2003a). Consequently, UU applied to Natural England (then English Nature) for a licence to undertake controlled shooting of adult cormorants during the spring of 2004.   The licence which was ultimately obtained allowed for the shooting of substantially fewer birds, i.e.15 individuals, than that requested in the initial application.

The cormorant management by shooting subsequently undertaken at Haweswater in 2004 had some success in reducing the level of impact on the schelly population (Winfield *et al.*, 2005), but the results of population modelling undertaken over the same period by Winfield *et al.* (2004c) and Winfield *et al.* (2007a) indicated that it was still insufficient to allow a population recovery. Consequently, in 2005 UU applied to shoot another 50 cormorants and were granted permission to shoot 35 individuals, although only 12 individuals were actually shot (Winfield *et al.*, 2006).   In 2006, UU applied to shoot all cormorants encountered at Haweswater and were again granted permission to shoot 35 individuals although this time

only 2 individuals were actually shot (Winfield *et al.*, 2007b), while in 2007 although a license was again obtained to shoot 35 cormorants no individuals were actually shot (Winfield *et al.*, 2008).   In 2008, UU again applied to Natural England to shoot 35 cormorants and were licensed to do so although, given the consistently low numbers of cormorants observed at Haweswater throughout the potential nesting season and the absence of nesting activities, no shooting attempts were actually made (Winfield *et al.*, 2009).   In 2009 (Winfield *et al.*, 2010) and 2010, UU did not submit an application to shoot.

The objective of this part of the present project was to assess the cormorant management undertaken at the lake in recent years in terms of its long-term effect in reducing the predation impact of cormorants on the schelly population.

5.2 Methods

An assessment of the persisting effects of the shooting of earlier years in reducing the predation impact of cormorants on the schelly population of Haweswater was made by comparing measures of cormorant roosting and derived indices of feeding behaviour from January to December of 2010 (no management undertaken), 2009 (no management undertaken), 2008 (no management undertaken), 2007 (shooting and non-shooting scaring undertaken), 2006 (shooting and non-shooting scaring undertaken), 2005 (shooting and non-shooting scaring undertaken), 2004 (shooting and non-shooting scaring undertaken), 2003 (non-shooting scaring undertaken), 2002 (non-shooting scaring undertaken) and 1997 (no scaring undertaken). This procedure used the methodology described in detail by Winfield *et al.* (2004c), but using the strengthened relationship between roost counts and feeding activity

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subsequently derived using additional field data by Winfield *et al.* (2006), i.e. y = 0.2126Ln(x) - 0.0670 (r^2 = 0.3981, df = 26, p < 0.001).

5.3 Results

For 1997, 2002, 2003, 2004, 2005, 2006, 2007, 2008, 2009 and 2010, the January to December measures of cormorant roosting were 7860 cormorant-days, 4309 cormorant-days, 4640 cormorant-days, 3075 cormorant-days, 2489 cormorant-days, 1774 cormorant-days, 796 cormorant-days, 1181 cormorant-days, 1078 cormorant-days and 1618 cormorant-days, respectively (Fig. 11).

For 1997, 2002, 2003, 2004 2005, 2006, 2007, 2008, 2009 and 2010, the January to December indices of feeding behaviour were 179, 127, 145, 103, 99, 83, 34, 44, 48 and 80, respectively (Fig. 12). Thus, the indices of feeding behaviour of the managed populations in 2002, 2003, 2004, 2005, 2006, 2007, 2008, 2009 and 2010 were 71%, 81%, 58%, 55%, 46%, 19%, 25%, 27% and 45%, respectively, of that of the unmanaged population in 1997 (Fig. 13).

5.4 Discussion

The absence of any cormorant shooting activity at Haweswater in 2010 matched that of 2009 and 2008 and, when combined with the unsuccessful shooting attempts of 2007, means that no birds have actually been shot since the two individuals of 2006. This pattern follows the downward trend of 15, 12 and 2 birds shot in 2004, 2005 and 2006, respectively. Prior to

2010, the recent overall situation with respect to cormorants and schelly at Haweswater was considered to be very encouraging. However, the 2010 observations revealed a substantial increase in local cormorant numbers and, by implication, in feeding behaviour. These observations raise a number of issues.

Firstly, it is clear that the reduction in recent years in the numbers of cormorants attempting to nest at Haweswater has been maintained in 2010, with the small number of birds present making no apparent attempts to construct nests. However, the number of individuals present after the nesting season has shown a substantial increase. The absence of any cormorant management activities of any kind during the nesting seasons of 2008 to 2010 has thus now been followed by an increase in local cormorant abundance. Although the non-nesting pattern of recent years may persist into 2011, it is highly desirable to continue to count cormorants throughout the coming potential nesting season and to review the situation in late 2011 with a view to deciding upon any cormorant management to be undertaken in 2012.

Secondly, the extensive non-shooting scaring procedures used in 2002 and 2003 and the extensive shooting and non-shooting scaring procedures used in 2004, 2005, 2006 and 2007, while successfully preventing the production of young (Winfield *et al.*, 2004b; Winfield *et al.*, 2005; Winfield *et al.*, 2006), only resulted in reductions in cormorant feeding behaviour to 71%, 81%, 58% 55%, 46% and 19%, respectively, of the unmanaged 1997 level. Using a schelly population model incorporating impacts from water level fluctuations and cormorant predation, Winfield *et al.* (2004b) and Winfield *et al.* (2007a) found that levels of predation above 40% resulted in predicted extinction even with no recruitment losses due to water level fluctuations. Furthermore, the model indicated that in order to achieve a significant recovery

of the schelly population in the medium-term future, predation impact from cormorants must be reduced to 10% or less of the 1997 level. While it was encouraging that the relative levels of predation observed in 2008 and 2009 were 25% and 27%, the level of 45% recorded in 2010 gives considerable cause for concern. It is notable that this 2010 substantial increase in predation was accompanied by a possible faltering in the recovery of the schelly population, as suggested by the entrapment record and hydroacoustic surveys. If this level of predation by cormorants is maintained or exceeded in subsequent years, the schelly population can be expected to resume its drift towards local extinction. Further schelly population modelling work incorporating cormorant data from recent years would be informative.

Thirdly, it is a welcome development that temporary nesting-season moves of cormorants to roosting high in tall trees of the west shore of Haweswater observed during the non-shooting scaring of 2003 (Winfield *et al.*, 2004b) and the shooting scaring of 2004 (Winfield *et al.*, 2005) have not been repeated in any of the subsequent years of 2005 to 2010. It had been feared that such a response may lead to nesting in such locations in future years, where any further management of the breeding colony would be extremely difficult.

CHAPTER 6 GENERAL DISCUSSION AND RECOMMENDATIONS

6.1 General discussion

The findings of the various components of this project have already been discussed within their specific chapters. However, a brief and more general discussion is warranted here with respect to the present situation at Haweswater and the conservation of its schelly population. Inevitably in a long-term monitoring project, much of the following repeats the general discussion of the previous annual report given by Winfield *et al.* (2010).

The U.K. *Coregonus* populations are of great national and international conservation value, in part because their gene pools have not been subjected to the effects of translocations, stockings or intense fisheries as is common elsewhere in Europe (see Beaumont *et al.*, 1995). Nevertheless, Haweswater is a strategically-important supply of potable water for north-west England and the conservation management of its schelly population must operate within this context. The most damaging aspects of lake level variations are now minimised as far as possible, with an additional amelioration system in reserve in the form of an artificial spawning substratum system. In addition, at least one refuge population has been successfully established at Small Water (Winfield *et al.*, 1997) and it appears that a second introduction to Blea Water (Winfield *et al.*, 1997) has been successful following the capture, and return alive, of a schelly of length approximately 150 mm by an angler on 29 May 2005 (P. Corkhill, *pers. comm.*).

In addition to the above improvements in the situation at Haweswater, it is also encouraging that the medium-term decline in adult schelly has now definitely stopped and this component of the population is in contrast showing some qualified signs of recovery. Further limited encouragement is provided by the entrapment study because schelly recorded in recent years have comprised a range of lengths and particularly ages, even though only one individual was entrapped in 2010. These observations show that the schelly remained extant in Haweswater in early 2010, and thus that at least some recruitment is continuing. Nevertheless, both the hydroacoustic and entrapment studies suggest that any population recovery is limited and indicate that the present population abundance of schelly in Haweswater is still some way below that recorded in the early 1990s and previous decades.

Schelly population modelling presented by Winfield *et al.* (2004a) demonstrated that the decline in population abundance observed during the 1980s can be explained by the changing pattern of water level fluctuations between the early 1960s and the early 2000s. It also indicated that, in the absence of other significant impacts, the population could recover relatively quickly under conditions of water level fluctuations compatible with the operation of Haweswater as a strategically important reservoir. However, it was notable that the descriptive model also showed a slight, although not statistically significant, increase in population abundance during the 1990s which was not observed in the entrapment data, even before it was disrupted by the engineering works. This discrepancy was subsequently accounted for by the addition to the model of an impact from cormorant foraging, based on actual observations at Haweswater, by Winfield *et al.* (2004c) and Winfield *et al.* (2007a).

While it is virtually impossible to obtain conclusive proof for the hypothesis that the local foraging activities of cormorants became a significant negative factor during the 1990s for the continued survival of schelly in Haweswater, the above findings provide very strong supporting evidence. Adopting the precautionary principle, this information was used by UU and others to justify attempts to prevent nesting by cormorants from 1999 to 2008, although such activities planned for 2001 were prevented by local access restrictions due to foot-and-mouth disease. Furthermore, as described in detail in Chapter 5, from 2004 to 2007 such activities were augmented by the shooting of a limited number of cormorants, with similar plans for 2008 being abandoned given the low numbers of birds present and no such activities even contemplated for 2009. This multi-year intervention brought about a reduction in the impact from cormorants close to that required for a significant recovery of the schelly population as indicated by the modelling by Winfield *et al.* (2004c) and Winfield *et al.* (2007a), although the recent post-shooting period has seen some increase in cormorant feeding at the lake. This resilience of cormorants to management measures supports the view presented in Winfield *et al.* (2004b) that among the range of options suggested for the control of cormorants in a guidance leaflet issued by the then Ministry of Agriculture, Fisheries and Food (Ministry of Agriculture, Fisheries and Food, 2000), considerations or trials at Haweswater have shown that noise generating scarers, stocking control, 'buffer' species and fish refuges are inappropriate, visual scarers are ineffective, and roost management and human disturbance have had only limited success. This left shooting of the adult birds as the only realistic management option that was likely to reduce the predation impact on the schelly to levels that may allow the population to recover.

Assuming that the population of adult schelly in Haweswater has remained above the minimum viable level and if impacts from cormorants can be successfully controlled to sustainable levels, a general improvement in spawning conditions in terms of water levels over those existing in the 1980 and early 1990s (UU, unpublished data) should lead to a significant recruitment to the adult spawning stock. On a more negative note, it is notable that the recent possible faltering in the recovery of the schelly population, as suggested by the entrapment record and hydroacoustic surveys, has been accompanied by a substantial increase in the amount of cormorant feeding activity at Haweswater. Further monitoring of this complex system by entrapment analysis, hydroacoustics and cormorant counts continues to be essential.

In the meantime, a continued relatively low population abundance means that the current status of the schelly population of Haweswater is considered to be poor.

6.2 Recommendations

It is recommended that the current monitoring of the Haweswater schelly population by a combination of examination of entrapped specimens and hydroacoustics is continued, with the frequency of the latter maintained at a single annual survey in July. Although the current programme has some inherent data interpretation problems, it does provide information on both the biological condition and absolute abundance of the schelly population. In this context it is strongly recommended that the collection of entrapped fish by UU staff at Watchgate Water Treatment Works is fully maintained and supported by UU.

It is also recommended that communications are maintained with all stakeholders, including licensing authorities, concerning the potential future control of cormorants at Haweswater through non-shooting scaring and shooting methods. Although no applications for licensed shooting were made for 2009, 2010 or 2011, it is strongly recommended that the situation is reviewed in late 2011 with a view to agreeing any cormorant management to be undertaken in 2012. Such action is not recommended lightly, but is done so on the basis of an overall consideration of conservation issues at Haweswater. A considered discussion of this sensitive issue, including its wider conservation implications in the English Lake District, is given by Winfield *et al.* (2003a).

With respect to reservoir operational issues at Haweswater, it is recommended that efforts are continued to minimise lake level variations through the critical period of February to April when schelly eggs incubate on shallow, inshore spawning grounds. Where possible, relatively stable levels in the spring and early summer would also be beneficial to the schelly population as they may appreciably improve feeding conditions for underyearlings in inshore nursery areas.

If further management of cormorants is undertaken at Haweswater, then it is strongly recommended that its effects are monitored not only in terms of the numbers of cormorants shot, but also in terms of non-lethal effects on local cormorant abundance, distribution and behaviour on shooting and non-shooting days. It is also recommended that whether or not shooting is undertaken, the numbers of cormorants at Haweswater are monitored by a programme of at least monthly roost counts.

It is also recommended that further development is undertaken of the modelling of schelly population dynamics including best assessment of the impact of local foraging by cormorants, the latter of which can now be substantially improved by use of the 2002 and subsequent cormorant field observations.

Finally, following the capture of a single schelly by an angler at Blea Water on 29 May 2005 (P. Corkhill, *pers. comm.*) it is recommended that some consideration be given to undertaking a gill-net survey to provide a more robust assessment of the establishment or otherwise of a second refuge population of Haweswater schelly in this water body. A further assessment of the successfully established refuge population in Small Water last surveyed in 2002 by Winfield *et al.* (2003b) would also be informative.

ACKNOWLEDGEMENTS

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Table 1. GPS locations for 15 hydroacoustic transects used at Haweswater in 2010. Locations are given in degrees and decimal minutes.

Event	Latitude (North)	Longitude (West)
Transect 1 start	54, 29.425	2, 49.168
Transect 1 end	54, 29.910	2, 48.399
Transect 2 start	54, 29.910	2, 48.399
Transect 2 end	54, 29.915	2, 49.187
Transect 3 start	54, 29.915	2, 49.187
Transect 3 end	54, 30.297	2, 48.481
Transect 4 start	54, 30.297	2, 48.481
Transect 4 end	54, 30.321	2, 49.028
Transect 5 start	54, 30.321	2, 49.028
Transect 5 end	54, 30.638	2, 48.404
Transect 6 start	54, 30.638	2, 48.404
Transect 6 end	54, 30.693	2, 49.045
Transect 7 start	54, 30.693	2, 49.045
Transect 7 end	54, 30.784	2, 48.259
Transect 8 start	54, 30.784	2, 48.259
Transect 8 end	54, 30.899	2, 48.854
Transect 9 start	54, 30.899	2, 48.854
Transect 9 end	54, 31.060	2, 48.079
Transect 10 start	54, 31.060	2, 48.079
Transect 10 end	54, 31.176	2, 48.591
Transect 11 start	54, 31.176	2, 48.591
Transect 11 end	54, 31.408	2, 47.715
Transect 12 start	54, 31.408	2, 47.715
Transect 12 end	54, 31.589	2, 48.080
Transect 13 start	54, 31.589	2, 48.080
Transect 13 end	54, 31.790	2, 47.018
Transect 14 start	54, 31.790	2, 47.018
Transect 14 end	54, 31.999	2, 47.253
Transect 15 start	54, 31.999	2, 47.253
Transect 15 end	54, 32.081	2, 46.254

Table 2. Summary data (given as geometric means with lower and upper 95% confidence limits in parentheses) for densities of small (length 40 to 99 mm), medium (100 to 249 mm), large (250 mm and greater) and all fish recorded during the hydroacoustic survey undertaken on Haweswater in 2010 together with corresponding data from corresponding dates in the previous two years from Winfield *et al.* (2010).

Date	Small fish (fish ha^{-1})	Medium fish (fish ha^{-1})	Large fish (fish ha^{-1})	All fish (fish ha^{-1})
25 July 2008	3.0 (1.3, 7.1)	2.8 (1.3, 6.1)	3.7 (1.6, 9.0)	13.6 (9.9, 18.6)
17 July 2009	6.9 (2.3, 20.9)	3.2 (1.1, 9.9)	2.1 (0.9, 4.9)	10.8 (3.0, 38.9)
23 July 2010	5.1 (2.3, 11.1)	2.3 (1.0, 5.0)	2.4 (1.0, 5.4)	10.1 (4.7, 21.6)

Fig. 1. Numbers of schelly and Arctic charr entrapped from January 1973 to March 2011. Note that the last data point does not yet cover the complete calendar year and that 2001 to 2003 encompassed a period of severe disruption of entrapment monitoring due to extensive engineering works. Pre-April 2006 data are from previous monitoring reports and Winfield *et al.* (2000).

Schelly

Arctic charr
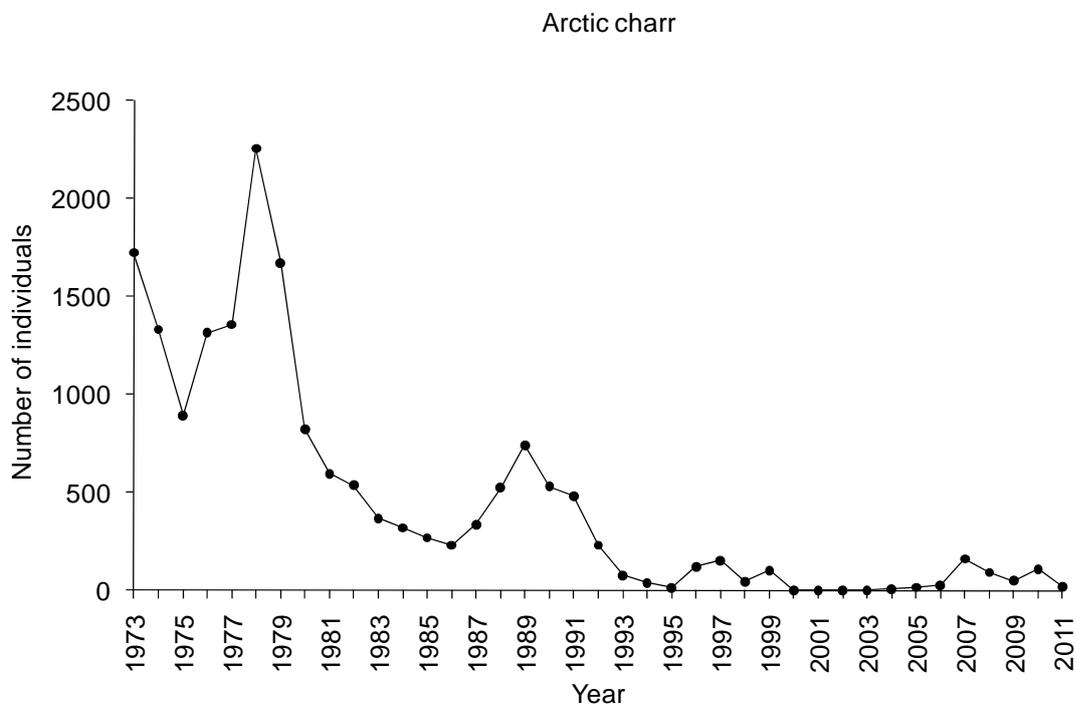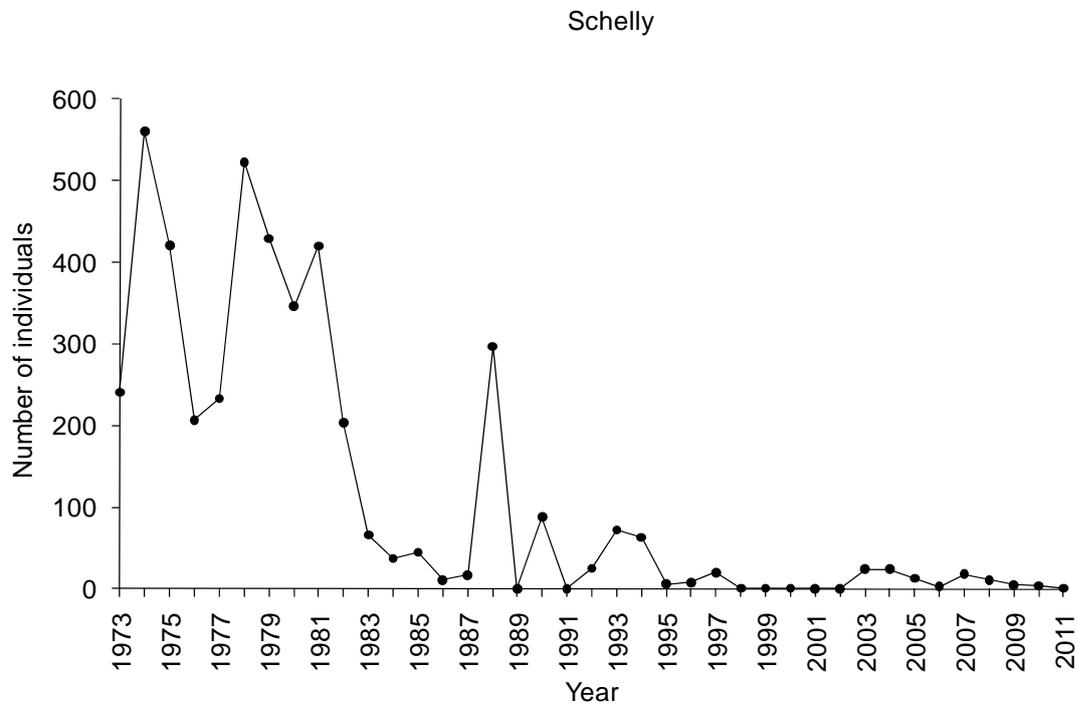
Fig. 2. Numbers of perch and brown trout entrapped from January 1973 to March 2011. Note that the last data point does not yet cover the complete calendar year and that 2001 to 2003 encompassed a period of severe disruption of entrapment monitoring due to extensive engineering works. Pre-April 2006 data are from previous monitoring reports and Winfield *et al.* (2000).
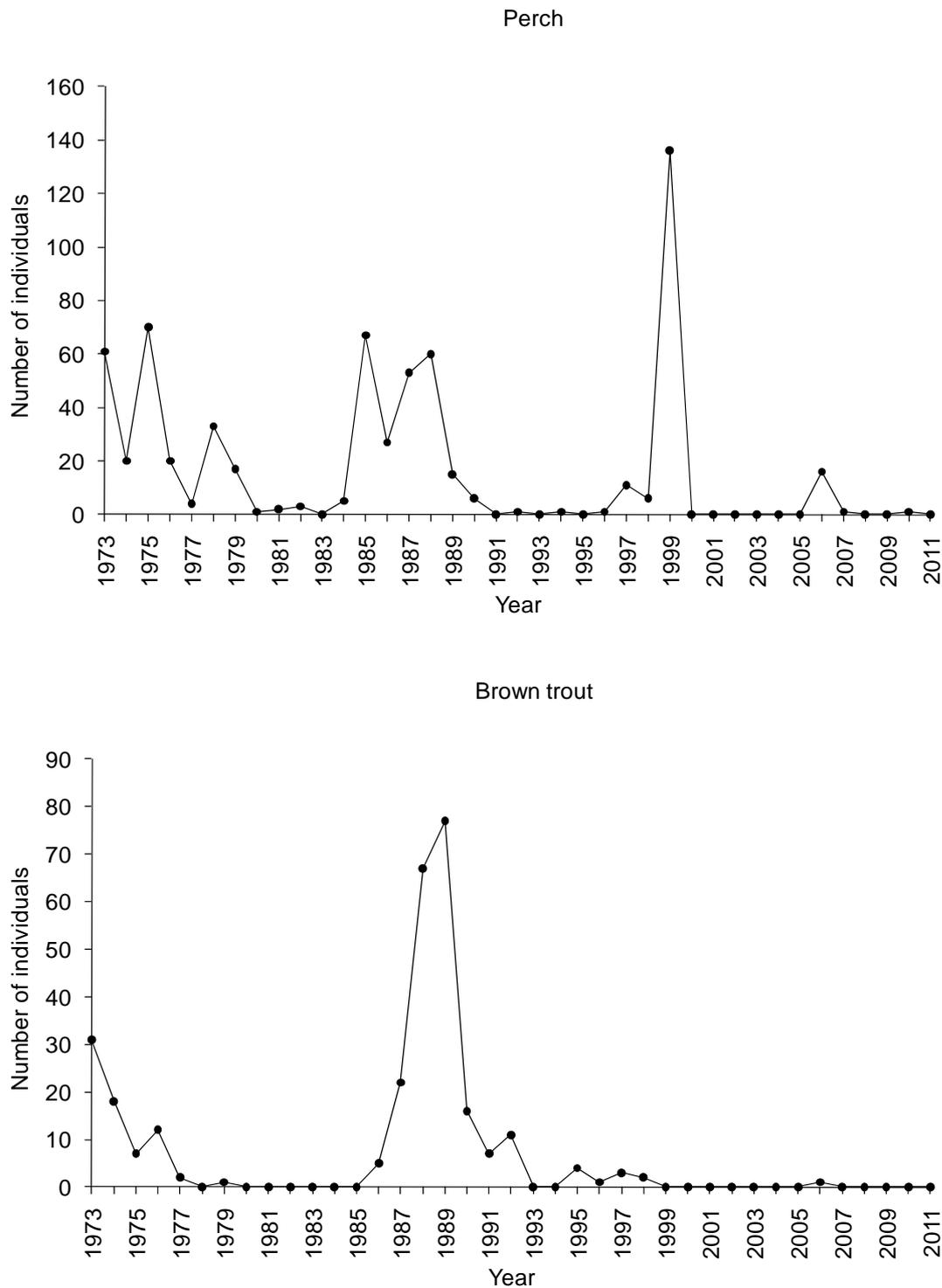
Fig. 3. Length (N = 1) and age (N = 1) frequency distributions of schelly entrapped from April 2010 to March 2011.

Length frequency distribution

Age frequency distribution
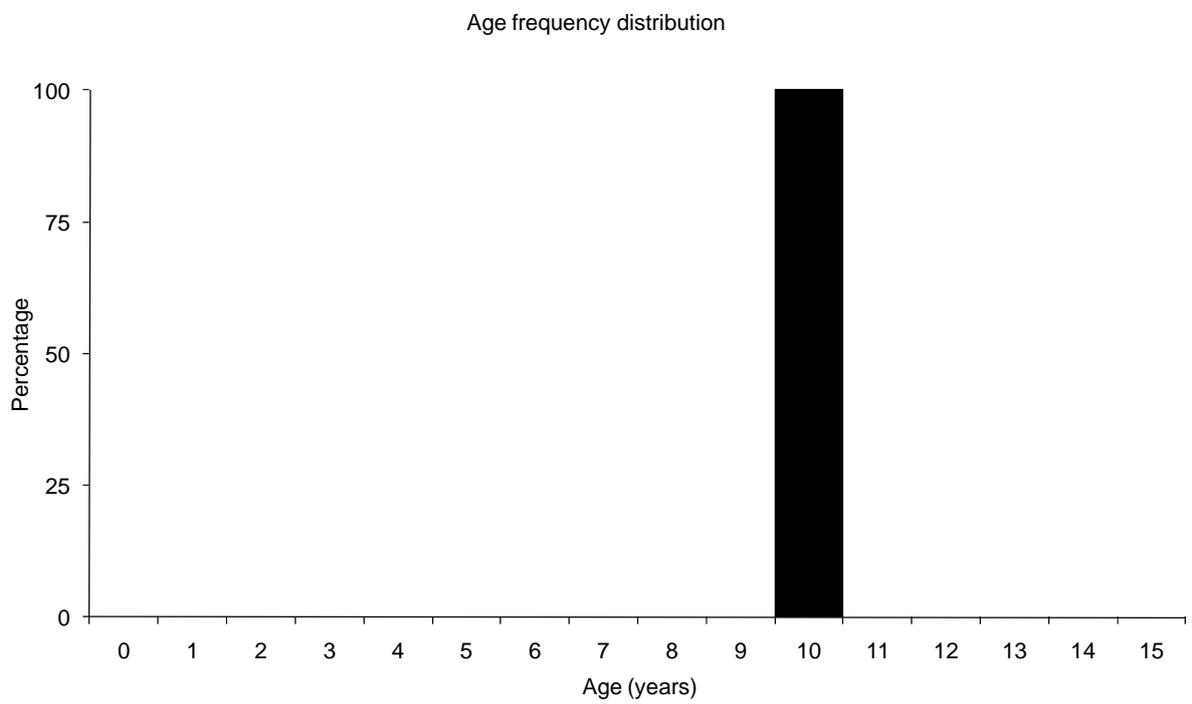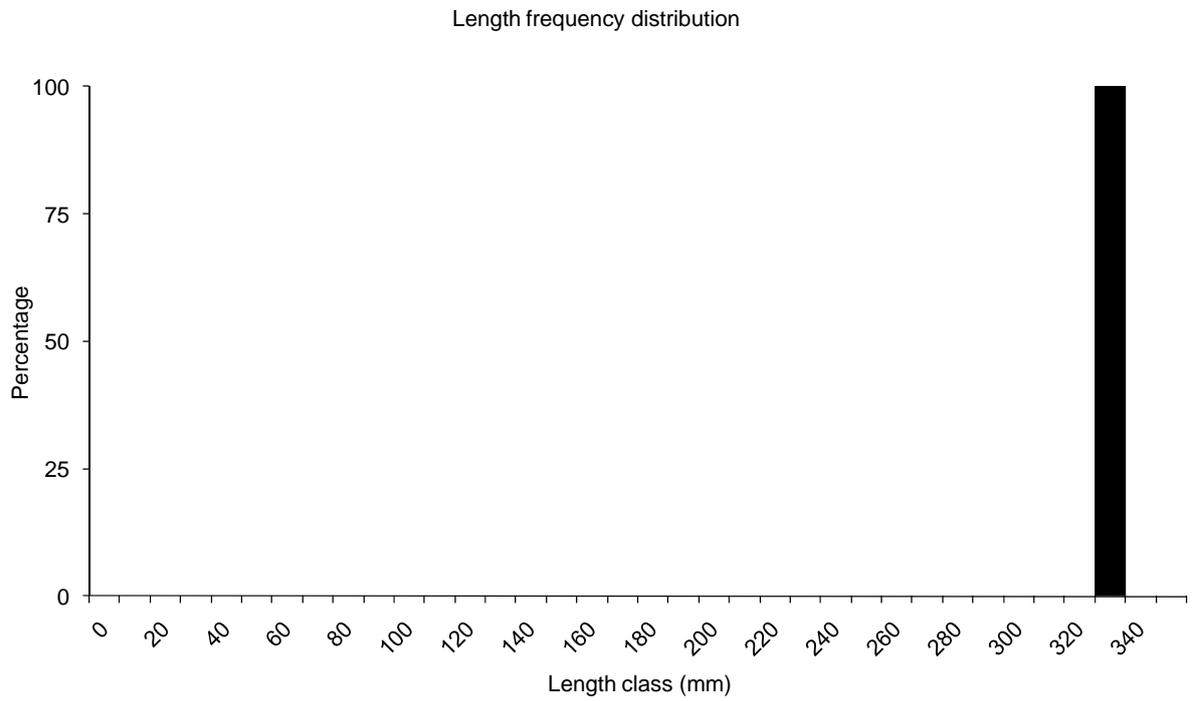
Fig. 4. Hydroacoustic transects (continuous lines numbered in bold italics) used during a survey of Haweswater on 23 July 2010. Depth contours (broken lines) are given in metres while the approximate position of the abstraction point is shown by an asterisk. Redrawn with permission from Ramsbottom (1976).
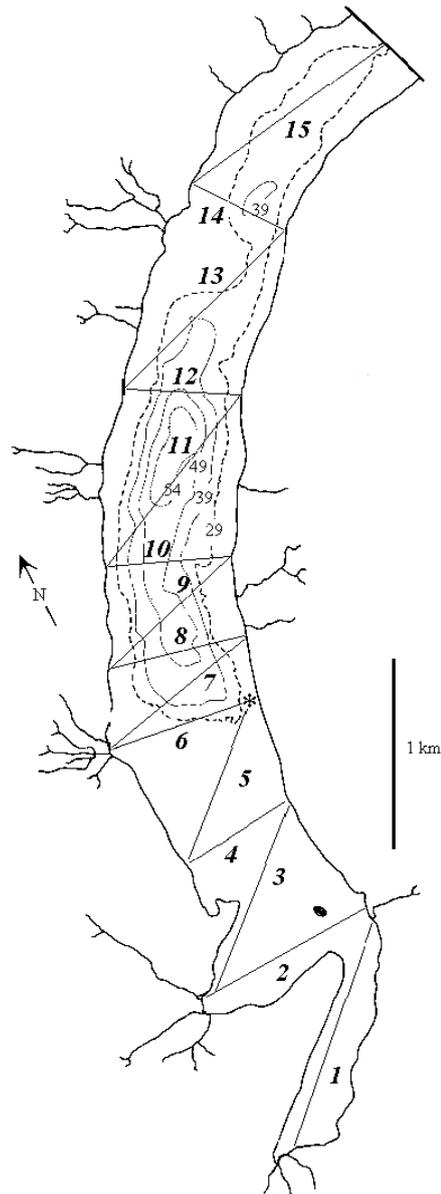
Fig. 5. Estimated population sizes (geometric means with 95% confidence limits, plotted on standard and logarithmic scales) of small fish (40 to 99 mm in length, probably a combination of schelly, Arctic charr, brown trout and perch) in July of 1997 to 2010. Note that a survey could not be undertaken in 2001 because of local access restrictions due to foot-and-mouth disease.
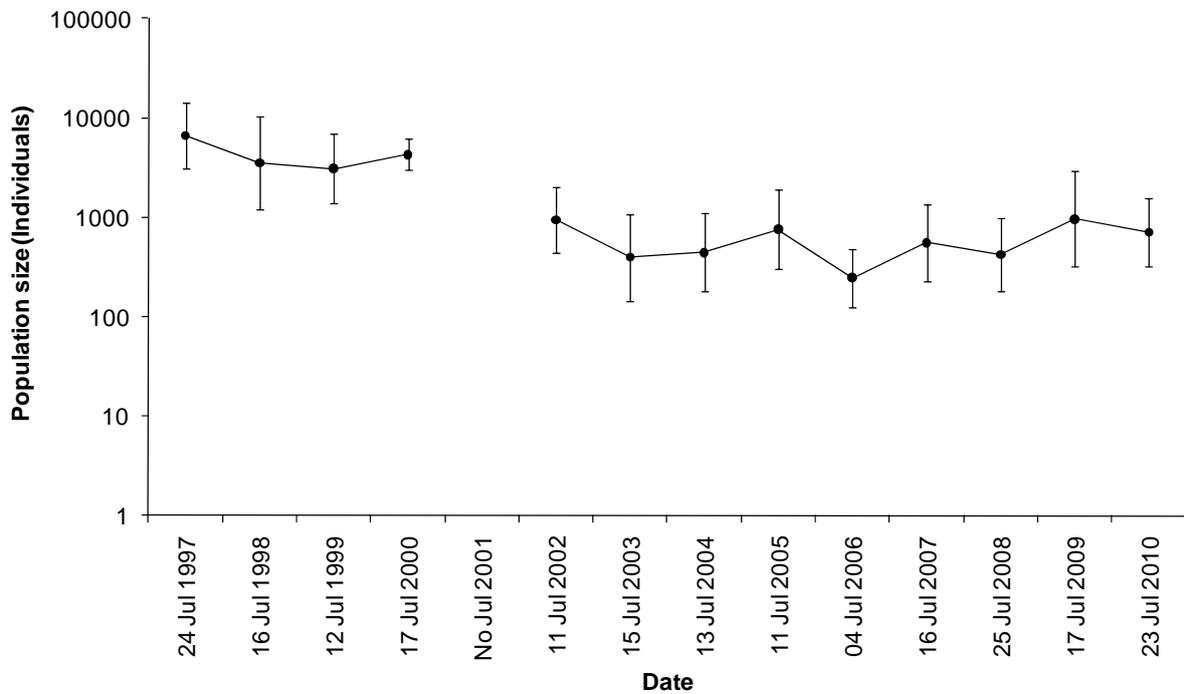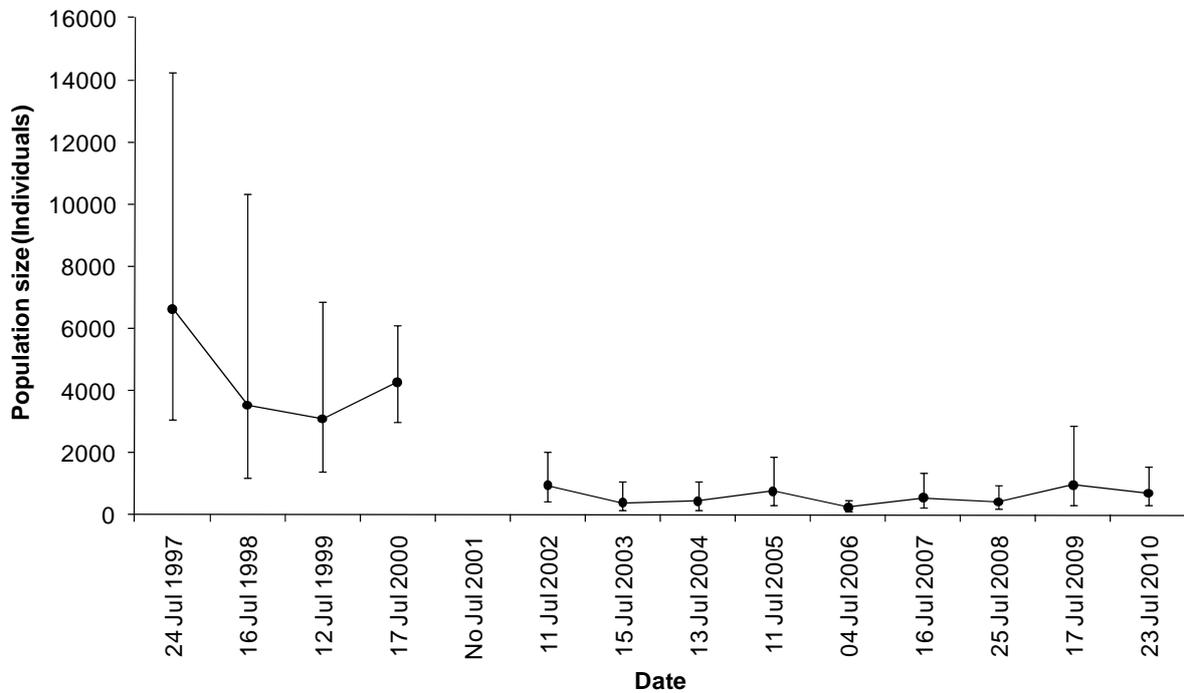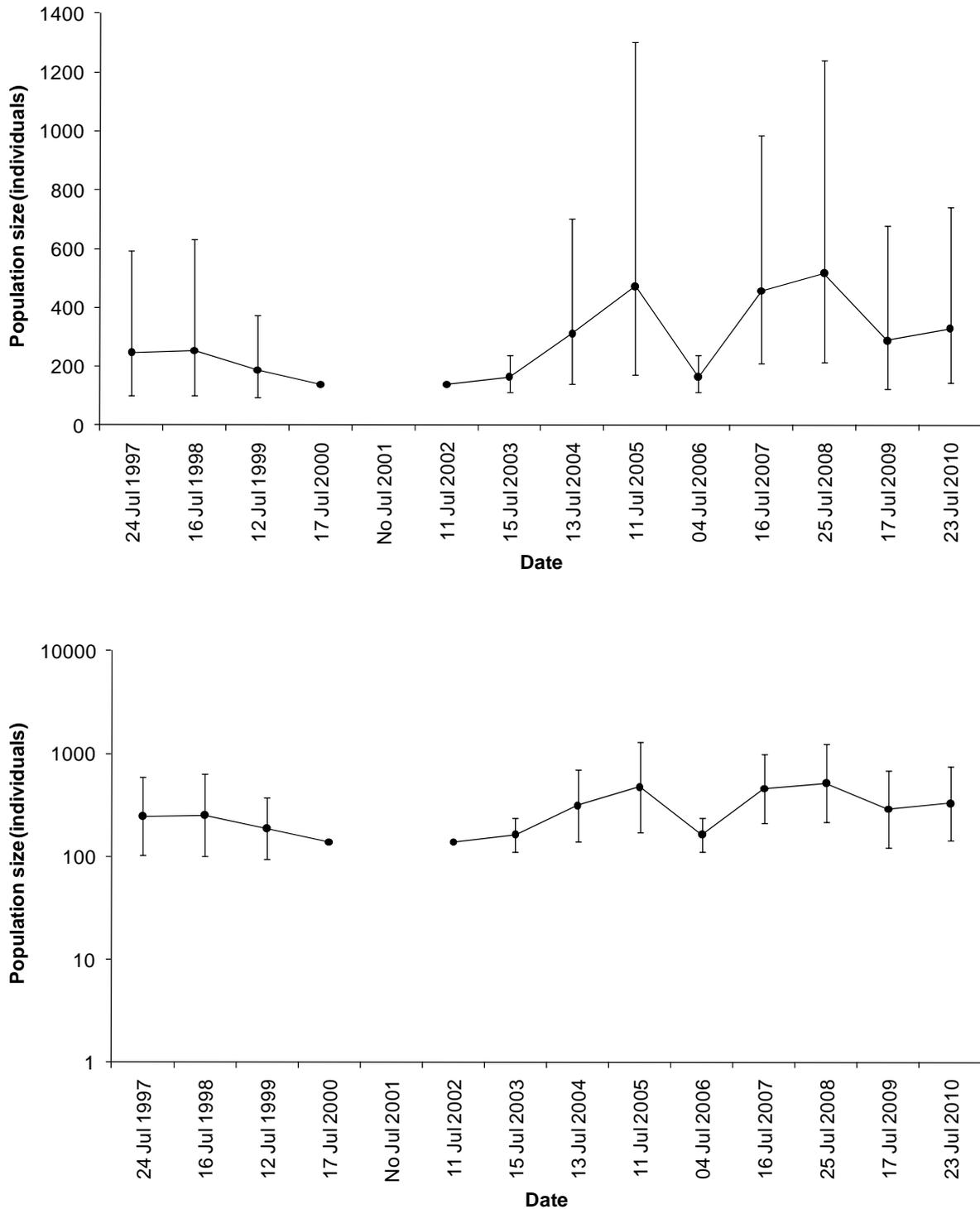
Fig. 6. Estimated population sizes (geometric means with 95% confidence limits, plotted on standard and logarithmic scales) of medium fish (100 to 249 mm in length, probably a combination of schelly, Arctic charr, brown trout and perch) in July of 1997 to 2010. Note that a survey could not be undertaken in 2001 because of local access restrictions due to foot-and-mouth disease.
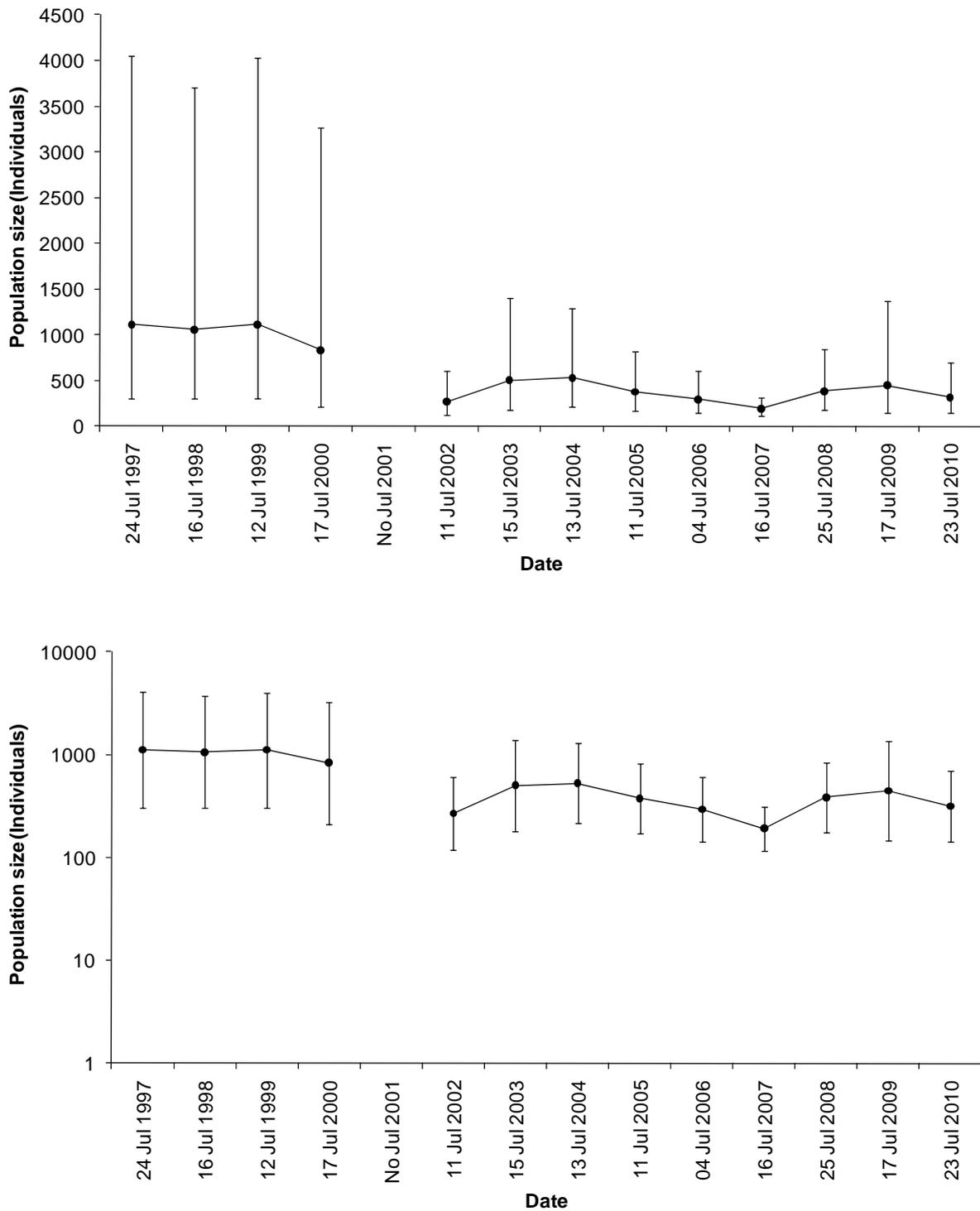
Fig. 7. Estimated population sizes (geometric means with 95% confidence limits, plotted on standard and logarithmic scales) of large fish (250 mm or greater in length, probably mainly adult schelly) in July of 1997 to 2010. Note that a survey could not be undertaken in 2001 because of local access restrictions due to foot-and-mouth disease.

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Fig. 8. Estimated population sizes (geometric means with 95% confidence limits, plotted on standard and logarithmic scales) of all fish in July of 1997 to 2010. Note that a survey could not be undertaken in 2001 because of local access restrictions due to foot-and-mouth disease.
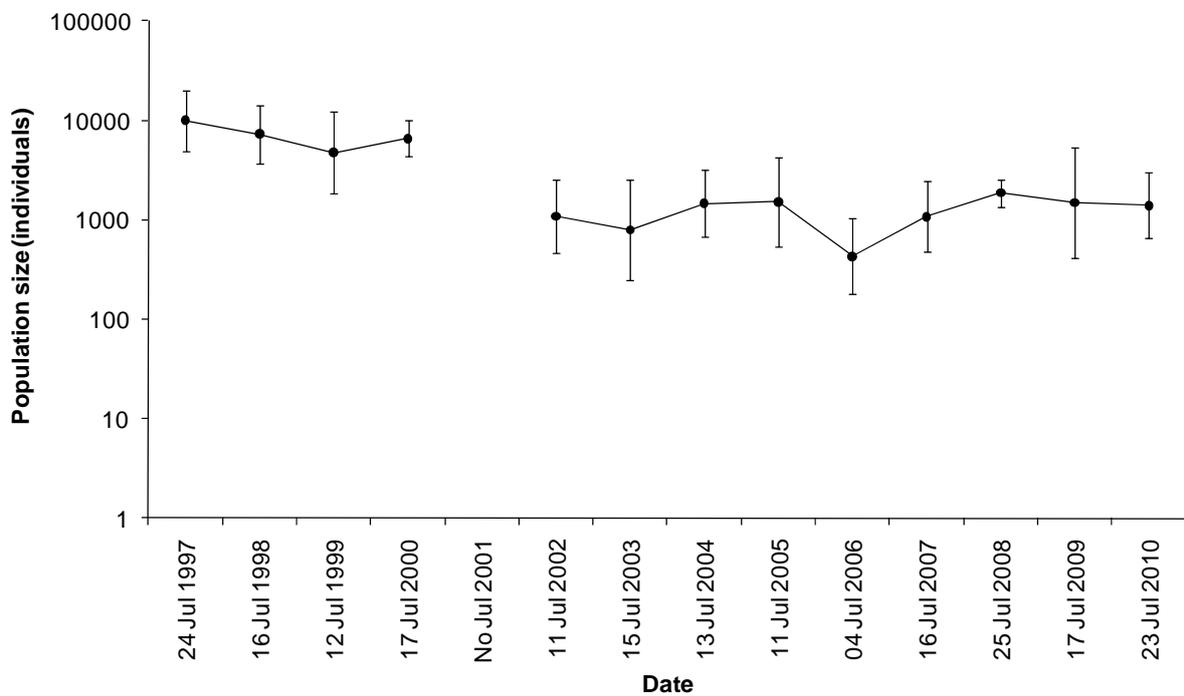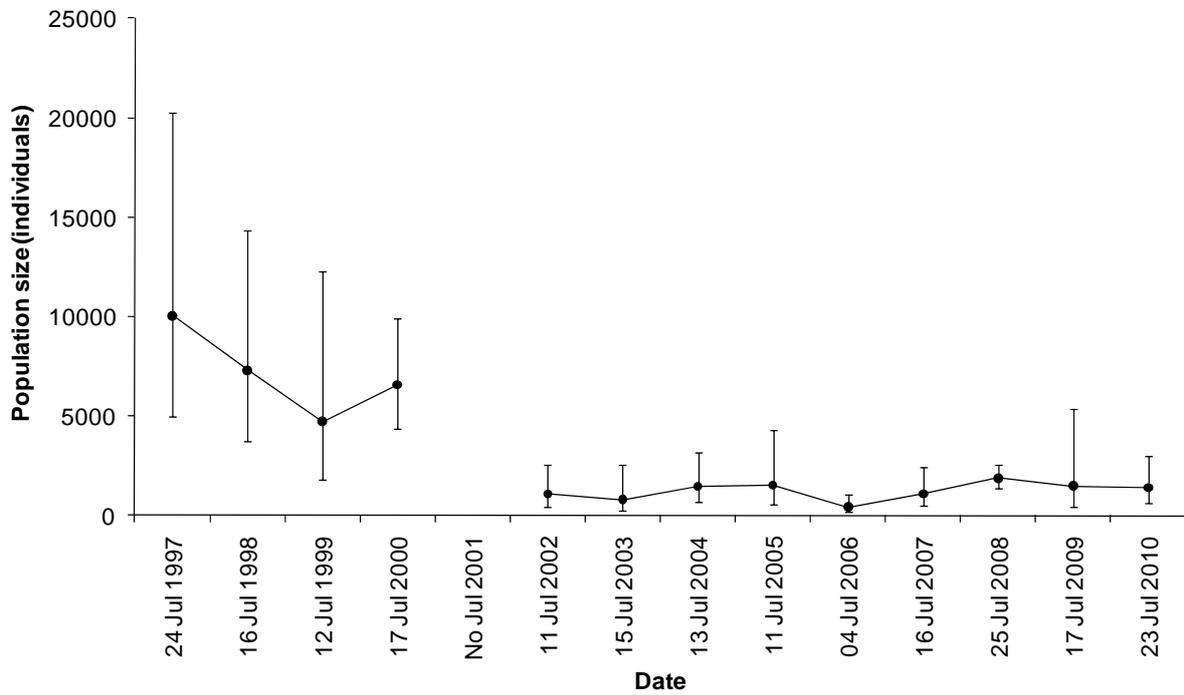
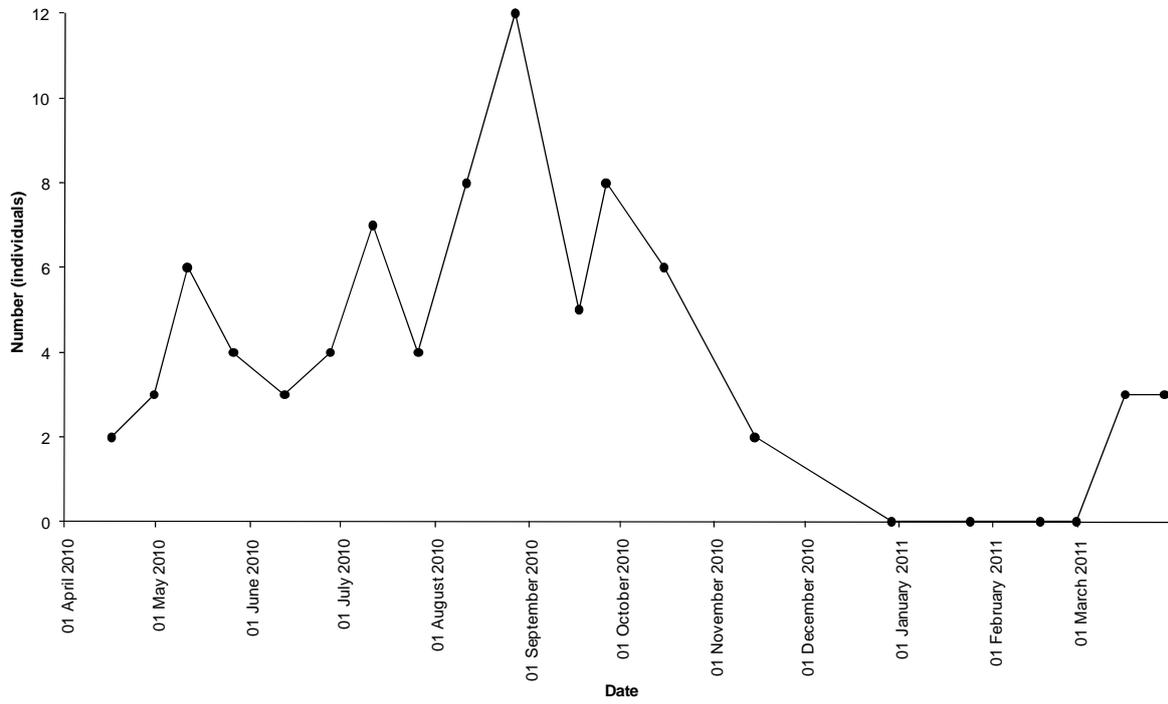Fig. 9. Cormorant roost counts on or near the island of Wood Howe between April 2010 and March 2011.

Fig. 10. Cormorant roost counts on or near the island of Wood Howe for all available data between November 1996 and March 2011.
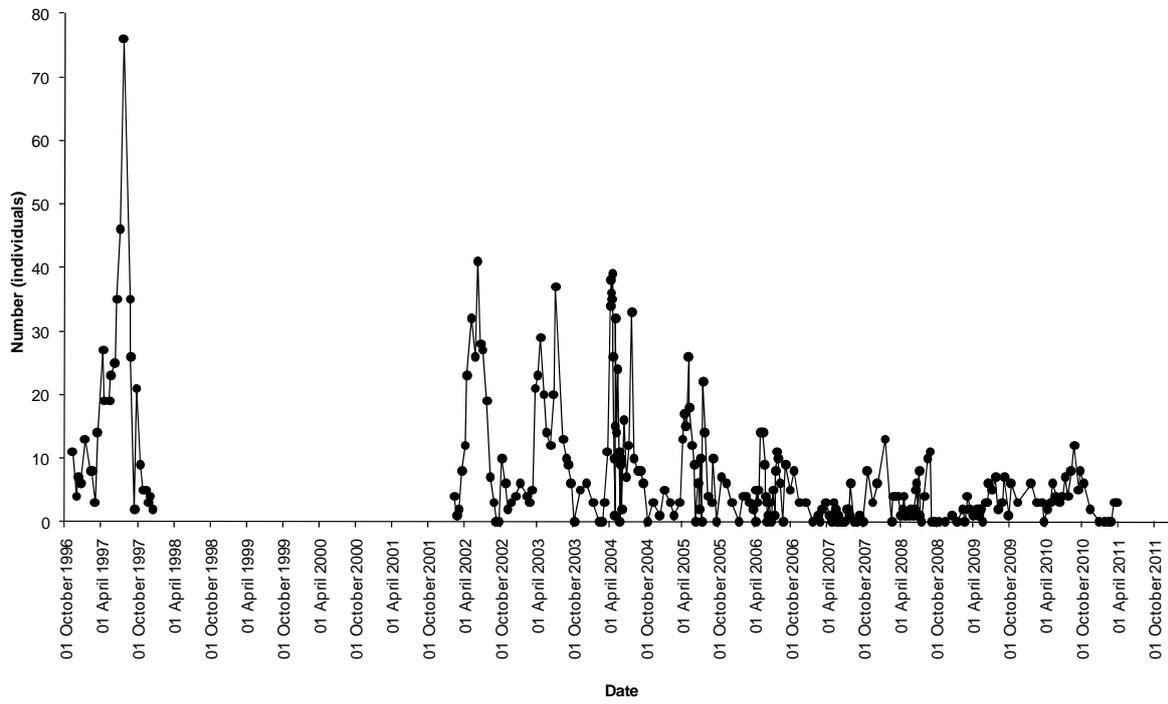
Fig. 11. Cormorant roosting presence (expressed as the number of cormorant-days) on or near the island of Wood Howe, or elsewhere on Haweswater, during 1997 (no scaring), 2002 and 2003 (only non-shooting scaring), 2004 to 2007 (shooting and non-shooting scaring) and 2008 to 2010 (no scaring).
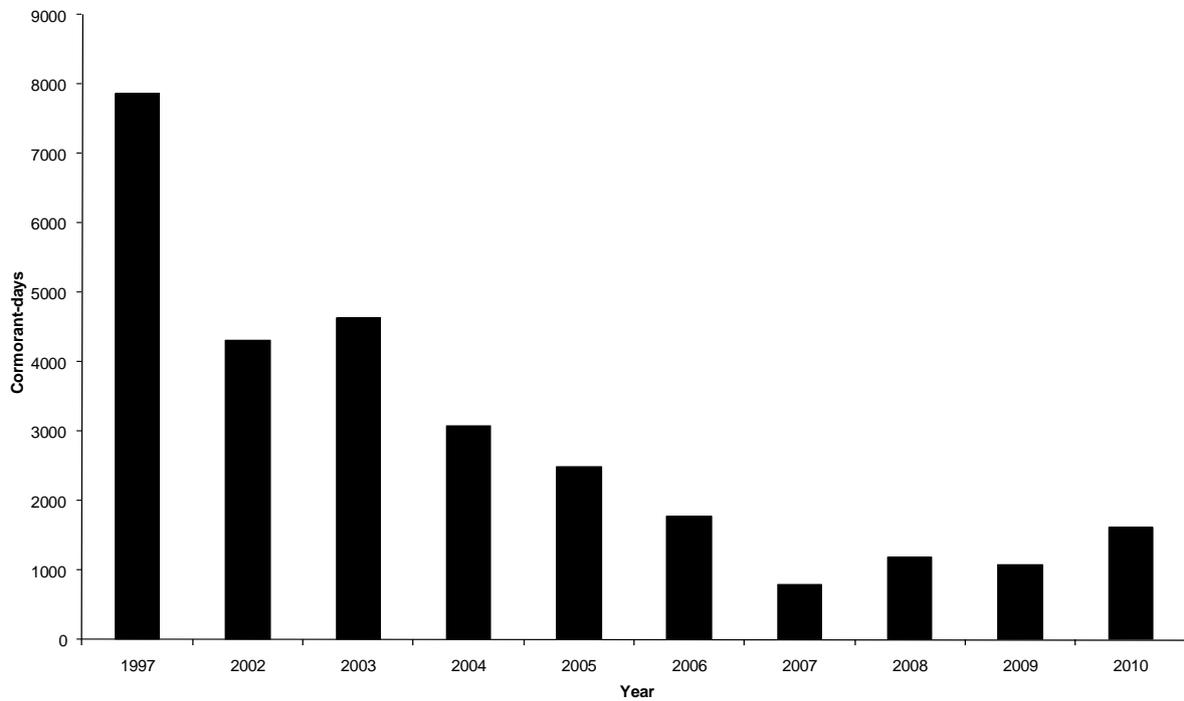
Fig. 12. Cormorant feeding intensity (expressed as an index of feeding behaviour, see text) at Haweswater during 1997 (no scaring), 2002 and 2003 (only non-shooting scaring), 2004 to 2007 (shooting and non-shooting scaring) and 2008 to 2010 (no scaring).
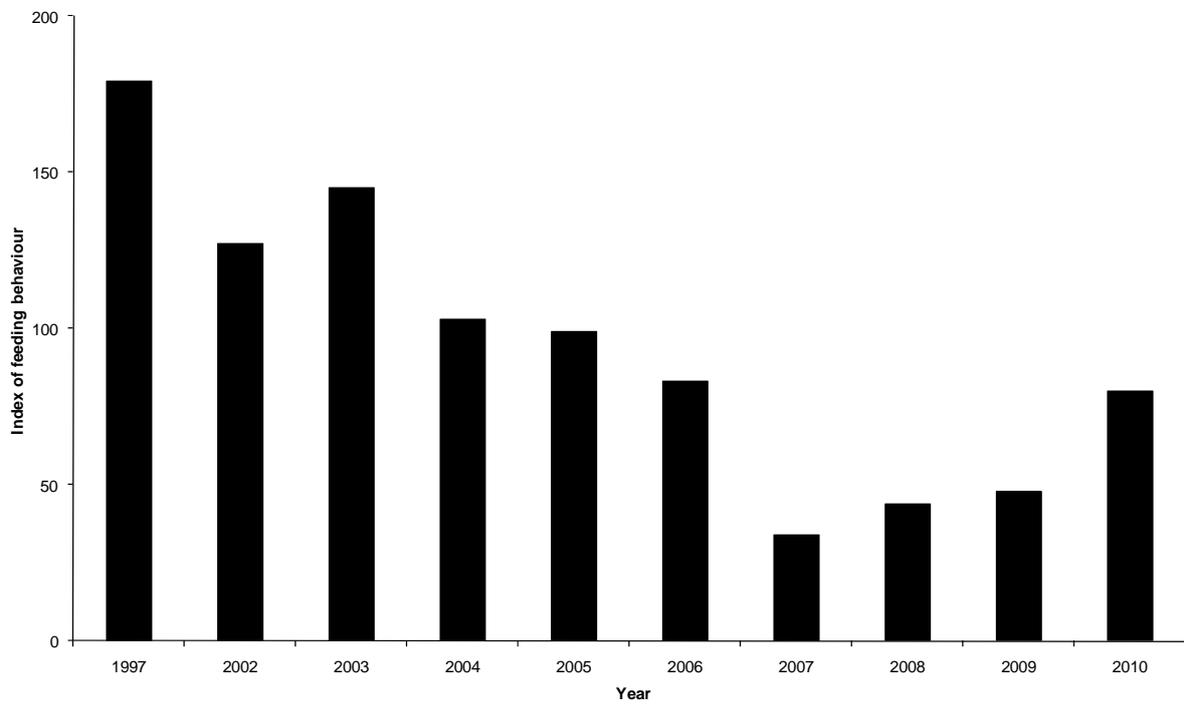
Fig. 13. Cormorant feeding intensity (expressed as an index of feeding behaviour relative to that observed in 1997, see text) at Haweswater during 1997 (no scaring), 2002 and 2003 (only non-shooting scaring), 2004 to 2007 (shooting and non-shooting scaring) and 2008 to 2010 (no scaring).