Conservation of the Freshwater Pearl Mussel

2. Relationship with Salmonids

Conserving Natura 2000 Rivers

Conservation Techniques Series No. 3

Conservation of the Freshwater Pearl Mussel

2. Relationship with Salmonids

Conserving Natura 2000 Rivers Conservation Techniques Series No. 2

Lee C Hastie and Mark R Young

University of Aberdeen
Culterty Field Station

For more information on this document, contact:
English Nature
Northminster House
Peterborough
PE1 1UA
Email: enquiries@english-nature.org.uk
Tel: +44 (0) 1733 455100
Fax: +44 (0) 1733 455103

This document was produced with the support of the European Commission's LIFE Nature Programme. It was published by **Life in UK Rivers**, a joint venture involving English Nature (EN), the Countryside Council for Wales (CCW), the Environment Agency (EA), the Scottish Environment Protection Agency (SEPA), Scottish Natural Heritage (SNH), and the Scotland and Northern Ireland Forum for Environmental Research (SNIFFER).

ISBN 1 85716 778 3

A full range of **Life in UK Rivers** publications can be ordered from:
The Enquiry Service
English Nature
Northminster House
Peterborough
PE1 1UA
Email: enquiries@english-nature.org.uk
Tel: +44 (0) 1733 455100
Fax: +44 (0) 1733 455103

This document should be cited as: Hastie LC & Young MR (2003). *Conservation of the Freshwater Pearl Mussel 2. Relationship with Salmonids.* Conserving Natura 2000 Rivers Conservation Techniques Series No. 3. English Nature, Peterborough.

Technical Editor: Lynn Parr
Series Ecological Coordinator: Ann Skinner

Cover design: Coral Design Management, Peterborough.
Printed by Astron Document Services, Norwich, on Revive, 75% recycled post-consumer waste paper, Elemental Chlorine Free. 1M.

Cover photo: Sue Scott

Conserving Natura 2000 Rivers

This report on the relationship between the freshwater pearl mussel (*Margaritifera margaritifera*) and salmonids has been produced as part of **Life in UK Rivers** – a project to develop methods for conserving the wildlife and habitats of rivers within the Natura 2000 network of protected European sites. The project's focus has been the conservation of rivers identified as Special Areas of Conservation (SACs) and of relevant habitats and species listed in annexes I and II of the European Union Directive on the Conservation of Natural Habitats and of Wild Fauna and Flora (92/43/EEC) (the Habitats Directive).

One of the main products is a set of reports collating the best available information on the ecological requirements of each species and habitat, while a complementary series contains advice on monitoring and assessment techniques. Each report has been compiled by ecologists who are studying these species and habitats in the UK, and has been subject to peer review, including scrutiny by a Technical Advisory Group established by the project partners. In the case of the monitoring techniques, further refinement has been accomplished by field-testing and by workshops involving experts and conservation practitioners.

Life in UK Rivers is very much a demonstration project, and although the reports have no official status in the implementation of the directive, they are intended as a helpful source of information for organisations trying to set 'conservation objectives' and to monitor for 'favourable conservation status' for these habitats and species. They can also be used to help assess plans and projects affecting Natura 2000 sites, as required by Article 6.3 of the directive.

As part of the project, conservation strategies have been produced for seven different SAC rivers in the UK. In these, you can see how the statutory conservation and environment agencies have developed objectives for the conservation of the habitats and species, and drawn up action plans with their local partners for achieving 'favourable conservation status'.

The project has also developed new conservation techniques for practical management of key species.

For each of the 13 riverine species and for the *Ranunculus* habitat, the project has also published tables setting out what can be considered as 'favourable condition' for attributes such as water quality and nutrient levels, flow conditions, river channel and riparian habitat, substrate, access for migratory fish, and level of disturbance. 'Favourable condition' is taken to be the status required of Annex I habitats and Annex II species on each Natura 2000 site to contribute adequately to 'favourable conservation status' across their natural range.

Titles in the Conserving Natura 2000 Rivers ecology and monitoring series are listed inside the back cover of this report, and copies of these, together with other project publications, are available via the project website: www.riverlife.org.uk.

Contents

1 Background

1.1 General introduction

The general decline of the endangered freshwater pearl mussel (*Margaritifera margaritifera* L.) throughout its holarctic range is well documented (Kerney 1975; Young & Williams 1983a; Bauer 1986 1988; Ziuganov *et al.* 1994). Several reasons have been suggested, including the effects of overfishing, industrial and agricultural pollution, and habitat reduction due to river engineering (Young 1991). Scotland is considered to be a stronghold of *M. margaritifera*, containing approximately half of the world's known remaining viable populations (Young *et al.* 2001).

However, even here the majority of populations have declined and many have disappeared completely. According to Cosgrove *et al.* (2000a), populations of pearl mussels are now either extinct or no longer viable (reproducing) in almost 70% of historical sites that were occupied 100 years ago. Although remaining populations are now provided with better protection by a recent ban on pearl fishing, stronger pollution control measures and restrictions on river engineering activity (Cosgrove & Hastie 2001), the fate of the pearl mussel in Scotland and elsewhere is by no means secure. There is concern that recent changes in native salmonid populations may pose a serious threat to the long-term survival of *M. margaritifera*. (Hastie & Cosgrove 2001).

Freshwater mussels have a short parasitic larval phase on the gills of suitable host fish. The larvae (glochidia) of *M. margaritifera* are very host-specific and, as far as is known, can only complete their development on Atlantic salmon (*Salmo salar* L.) or brown trout (*Salmo trutta* L.), usually 0+ fish (fry in their first year after hatching) (Young & Williams 1984a, Hastie & Young 2001).

Changes in salmonid host populations are not considered to have been a significant factor in the general decline of *M. margaritifera* over the past 50–100 years (Young & Williams 1983, Young 1991). However, although very little is known about the mussel-host relationship, long-term survival clearly depends on host availability, and there is concern that recent significant changes in wild salmonid stocks may threaten mussel populations (Bauer1988; Chesney & Oliver 1998; Cosgrove *et al.* 2000b; Hastie & Cosgrove 2001).

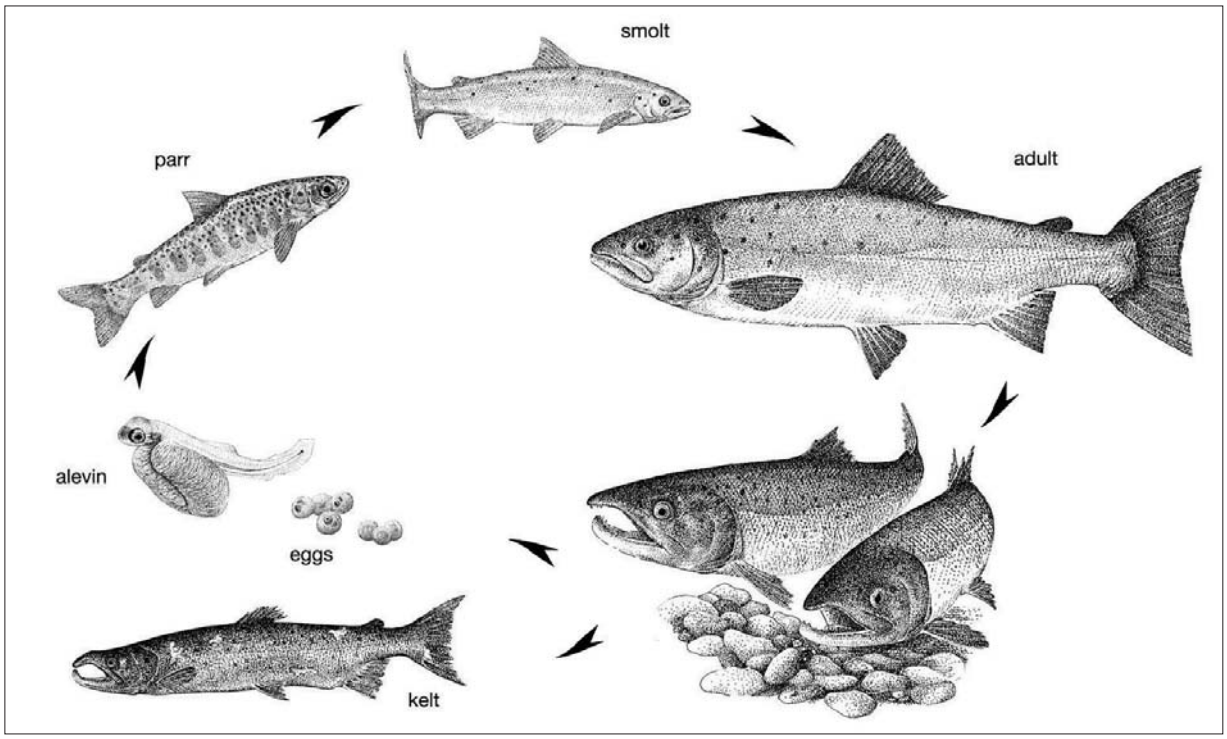
According to Ziuganov *et al.* (1994), a low density of fish hosts can be a limiting factor in some mussel populations. Trout densities lowered by acidification have been implicated in the decline of *M. margaritifera* in Sweden (Bauer 1988). In the north-west of Scotland, several migratory trout stocks have collapsed recently and some salmon stocks are declining (Walker 1993). Since more than 90% of surviving Scottish *M. margaritifera* populations are found in this area (Cosgrove *et al.* 2000a), it is important that appropriate research is carried out in order to determine the significance of these changes in terms of pearl mussel conservation (Hastie & Cosgrove 2001).

1.2 The salmonid life-cycle

The life-cycles of *S. salar* and *S. trutta* are well known but not fully understood. Both species are migratory fish; adults return to spawn in parent rivers, juveniles undergo their early development in freshwater and pre-adults grow to reproductive size at sea (Walker 1993). A proportion of *S. trutta* populations (that have access to the sea) migrate to feed in coastal waters (sea trout), while others (mainly males) mature and remain wholly in fresh water (brown trout). Above natural and artificial obstacles (for example, waterfalls, dams, polluted reaches), a sex ratio of about 1:1 would be expected for this species (J Watt, pers. comm.). Some male salmon also mature and spend most of their lives in fresh water (precocious male parr).

In Scotland, trout and salmon usually spawn in the autumn and winter. Eggs are buried in gravel nests (redds) and hatch the following spring. The newly hatched alevins remain in the gravel for several weeks until their yolk-sacs are absorbed, when they emerge as fry (0+ fish). The migratory fish remain in fresh water for another 1–5 years as parr. The parr eventually become silver-coloured smolts that migrate

downstream during spring and spend one or more years feeding in the sea. Fish (*S. salar*) that return to breed in fresh water after only one winter are known as grilse, whereas those that spend more than two years at sea are known as salmon. Most of the spent fish (kelts) die after spawning, but a few females survive and repeat the process and spawn again. In contrast, adult trout typically spawn annually for many years. More complete descriptions of the life-cycles of *S. salar* and *S. trutta* are provided in reviews by Gibson (1993) and Elliot (1994).



Sarah Wroot

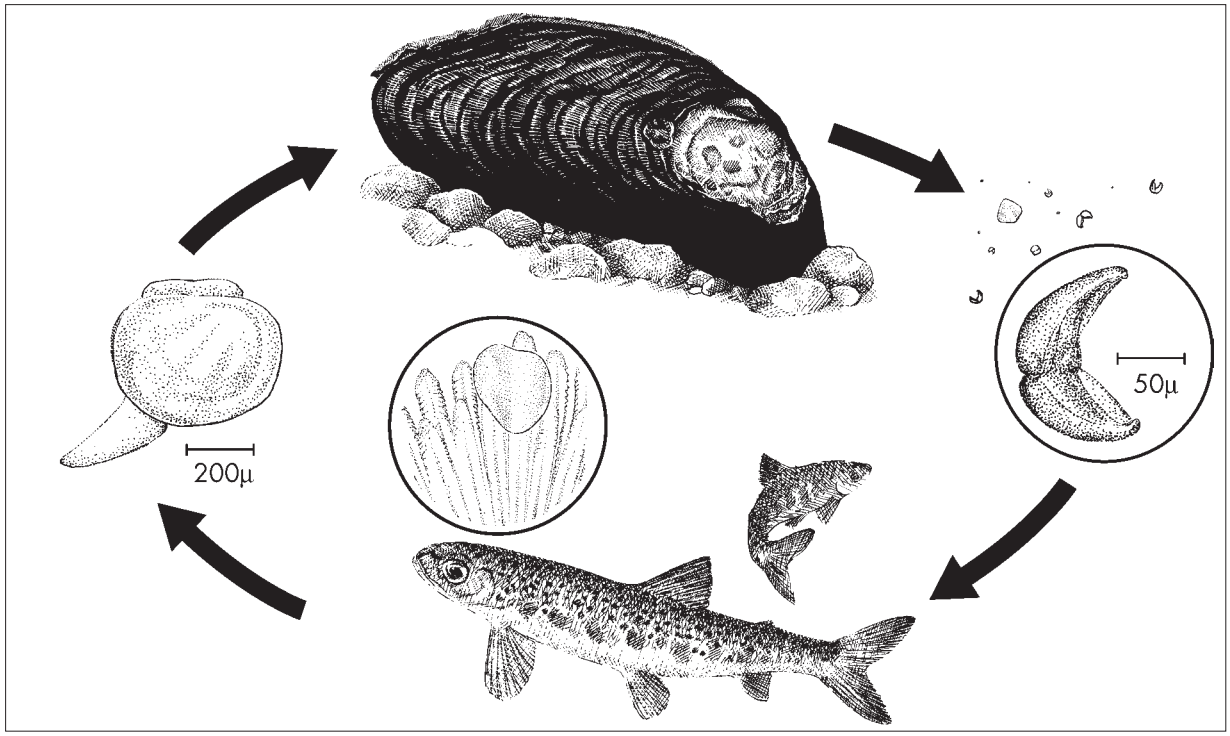
The Atlantic salmon has several life stages (beginning bottom right). The adult female lays eggs, which are fertilized by the male. The spent adults are then known as kelts, and while a few return to sea to spawn the following year, most die. The eggs hatch into alevins, dependent on their yolk sacs; then grow into fry, parr and smolt, when they first migrate to sea. Up to four years later, they return to their natal river to spawn.

1.3 The freshwater pearl mussel life-cycle

The life-cycle of the freshwater pearl mussel is less well known. The slow-growing *M. margaritifera* is one of the longest-lived invertebrates known, capable of reaching ages greater than 100 years (Bauer 1992). In common with other freshwater bivalves, the sexes are separate. Both sexes mature at age 12–20 years (Young & Williams 1984a).

An annual cycle of gametogenesis is apparent (Ross 1992). Up to 3 million unfertilised eggs pass out of the ovary into the mantle cavity and collect in brood pouches in the modified gills (marsupium), where they are fertilised in early summer. The female mussels inhale sperm by normal filtering action, in which a stream of water (containing food particles) enters the mantle cavity via the inhalant siphon. In mid- to late summer, following an incubation period of several weeks, the females discharge their glochidia into the river (Hastie 1999).

Glochidia resemble miniature mussels, measuring 0.06–0.08 mm across (Buddenseik 1991). They are obligate parasites of fish and are found encysted on the gills of their hosts. Of the many glochidia released by pearl mussels, only a few that are ingested or inhaled by host fish become attached to and encyst on their gills. The parasitic phase of *M. margaritifera*, which does not appear to harm wild fish, lasts for several months, before the glochidia metamorphose into tiny mussel 'seed' (by then approximately 0.4 mm across), excyst from the host gills, drop off and settle onto the riverbed (Young & Williams 1984a). Those that settle in clean, stable sand may survive to adulthood.



Sarah Wroot

The freshwater pearl mussel has a unique association with juvenile salmonids. Mussel larvae, known as glochidia, are released in summer and attach themselves to the gill filaments of host fish. Here they encyst until the following spring, when they drop off and begin to mature.

1.4 Salmonid stocks in Scotland

Fishery managers and biologists have been concerned about the plight of sea trout in north-west Scotland for some time (Anon. 1993). Catches in this region have declined since the 1950s and are now at the lowest levels ever recorded (Figure 1). Historical catch data, which are influenced by fishing effort and the types of gear used, are of limited use for assessing the status of wild fish stocks (Walker 1993). Nevertheless, the downward trend (supported by a small number of independent surveys) has been so dramatic that the general consensus is that sea trout are disappearing in north-west Scotland (Anon. 1993). A number of stocks have collapsed completely.

The causes for this general decline have been attributed to numerous factors, including climatic/oceanographic changes, over-fishing, increased predation, infestations by sea lice, pollution (acidification) and physical habitat degradation (Marshall 1998). Whatever the reasons, the implications of this decline for the future of local sea trout fisheries are grave (Butler 1998).

Another concern is that during the past decade, average weights of sea trout have started to fall in some populations (Walker 1993, Butler 1998). Most sea trout caught in north-west river systems are female, and these exhibit a significant relationship between body size and fecundity – smaller females produce fewer eggs (Walker 1993). Therefore, the decline in both numbers and sizes of individual fish may significantly reduce the potential fecundity of local sea trout populations, and consequently affect their ability to recover in the long term (Walker 1993). In contrast, non-migratory brown trout stocks in Scotland have largely remained stable, and in some areas they may even be increasing as sea trout stocks collapse (Butler 1998).

Salmon catches in north-west Scotland fluctuated considerably during the period 1952–1990, but overall the numbers appear to have been relatively stable for at least four decades. However, during the 1990s a marked downturn occurred and catches are now at historically low levels. Salmon catches actually increased in a few local fisheries, but this is thought to be due to the significant numbers of fish

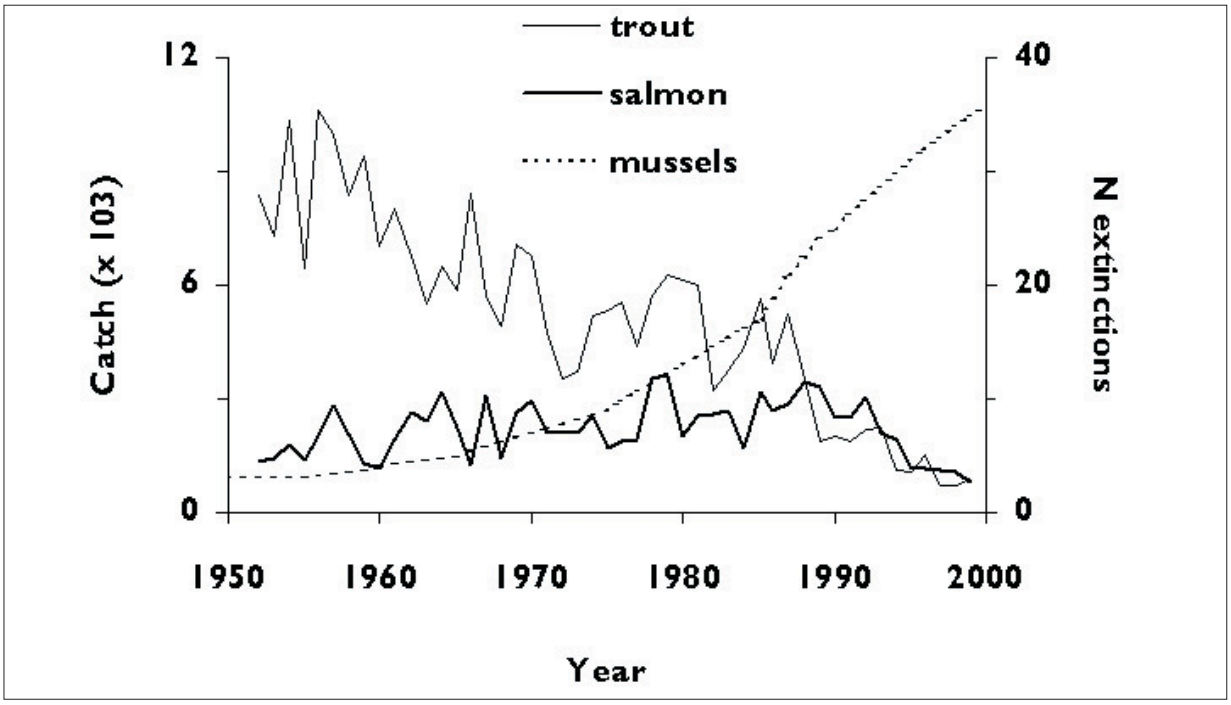


Figure 1. The coincidental declines in catches of sea trout and salmon, and increasing loss of freshwater pearl mussel populations in north-west Scotland during the period 1952-2000 (Hastie & Cosgrove 2001).

farm escapees that invaded some rivers (Walker 1993; Webb 1993; Butler 1998), and possibly some intensive stock enhancement programmes.

Despite the dramatic decline of sea trout catches over the past 50 years, very few electric-fishing surveys of juvenile salmonids in north-west rivers were carried out before 1990. As a result, it is still unclear whether or not local densities of fry and parr have been significantly affected by the reduced spawning stock biomass and lowered egg production now evident in many rivers. Walker (1993) compared densities of salmon and trout in north-west rivers over periods of 5–10 years (1984–1993) and found no evidence of a major change in overall juvenile abundance. However, in some small streams, very low densities – believed to be sub-critical for smolt production – have been recorded more recently (Butler 1999).

There is some evidence that salmonid juvenile densities fluctuate widely between years, but little is known about the mechanisms involved (Butler 1998). Unfortunately, there is a lack of long-term data-series based on regular juvenile surveys in north-west Scotland (Walker 1993). In the 1990s, an attempt was made to rectify this by establishing local fisheries trusts committed to long-term monitoring of target rivers (Anon. 1997). A number of baseline surveys have already been carried out by the fisheries trust biologists (for example, Butler 1998; Marshall 1998; Watt 1999), although it will take many years of monitoring before any long-term trends in juvenile salmonid abundance are apparent.

1.5 Implications for mussel conservation

Since non-migratory brown trout are suitable hosts for *M. margaritifera* glochidia, it is possible that some (trout-dependent) pearl mussel populations will remain viable. In central Europe, brown trout are the main hosts of *M. margaritifera*. However, there is concern that the non-migratory trout that grow in very small oligotrophic streams in north-west Scotland do not produce enough fry to sustain mussels in the long term (Hastie & Cosgrove 2001). Adult sea trout only return to rivers to spawn and therefore are not limited by stream resources. Since brown trout (and smaller sea trout) produce far fewer eggs, the decline in abundance and size of sea trout has resulted in substantial reductions in the fecundity of local trout populations, particularly in small streams (Walker 1993). Several *M. margaritifera* populations in small streams are already showing signs of reduced recruitment (Hastie *et al.* 2000a).

At present, populations of *M. margaritifera* that are able to utilise salmon hosts may be less vulnerable, but this is unconfirmed. Salmon stocks naturally fluctuate between years (Butler 1998). As pearl mussels can have a reproductive life-span of up to 80 years (Bauer 1992), they are probably not greatly affected by annual fluctuations in numbers of host fish. However, the fact that north-west salmon stocks have also declined recently, and currently are at historically low levels, is cause for considerable concern. Unless salmon and sea trout stocks recover, then the long-term survival of all remaining *M. margaritifera* populations in north-west Scotland eventually may be threatened.

Although a number of baseline studies of the relationship between pearl mussels and their hosts have been carried out (Young & Williams 1984a; Bauer 1987; Cunjak & McGladdery 1991; Ziuganov *et al.* 1994; Hastie 1999; Hastie & Young 2001), more research is required. For example, there is a distinct lack of field data from individual rivers, and little is known about the relationship between host stock sizes and the reproductive success of mussels (Chesney & Oliver 1998).

Some workers (for example, Ziuganov *et al.* 1994, Cosgove *et al.* 2000b) have argued that the margaritiferid-salmonid relationship may be symbiotic, in that salmon and trout may benefit from the presence of mussels in some rivers. The rationale is as follows:

- Mussels may be important for the maintenance of water quality in salmonid redds (spawning beds) and nursery areas because they reduce suspended organic material by filter-feeding and secrete 'pseudofaeces' that are rapidly degraded to harmless products. A single *M. margaritifera* can filter up top 50 litres of river water each day (Ziuganov *et al.* 1994).
- Mussel beds may, either as water-flow refugia or as a source of calcium (leached from shells), provide critical micro-habitats for aquatic invertebrates upon which juvenile salmonids feed. Large numbers of small fish and invertebrates are often found in mussel beds (pers. obs.).

There is a need for the different fisheries interests and conservation organisations to work together. Given the close relationship between salmonids and pearl mussels, and with their coincidental decline in a number of rivers in north-west Scotland, perhaps an integrated approach to their conservation management is the best way forward. For example, in rivers where mussels may be threatened by a lack of juvenile salmonid hosts, regular stocking of native fish (perhaps jointly financed by fishery and conservation organisations) may be a very useful short-term remedial action. Methods used to manage the physical environment of a salmon or trout river may have a number of implications for the survival of local mussel beds and juvenile fish nurseries. However, since more understanding of the mussel-host relationship is urgently required, a considerable amount of research should be undertaken before any measures are implemented.

In this report, the results of an investigation of the relationship between pearl mussels and salmonids in north-west Scotland are described. The implications of these findings for salmonid fishery operations and the general management of rivers containing mussels are also discussed.

2 Riverbed surveys: estimation of mussel densities

2.1 Summary

From 1996 to 2002, riverbed surveys of 12 100 m lengths of river for freshwater pearl mussels were carried out in each of the River Kerry candidate Special Area of Conservation (cSAC) and River Moidart cSAC in north-west Scotland. Estimated total numbers of visible mussels ranged from 2,340–4,1900 mussels per 100 m in the Kerry and 1,370–1,7760 mussels per 100 m in the Moidart. Observed mussel densities ranged from 1.81–32.23 mussels per m^2 (Kerry) and 0.96–16.60 mussels per m^2 (Moidart). Overall numbers of mussels at selected sites ranged from 3,225–6,4582 mussels per 100 m (Kerry) and 1,679–3,7864 mussels per 100 m (Moidart). Absolute numbers of juvenile mussels (<65 mm shell length) ranged from 500–1,5112 juveniles per 100 m in the Kerry and 190–2,499 juveniles per 100 m in the Moidart. Relative numbers of juvenile mussels ranged from 13.8–23.4% (Kerry) and 6.6–11.3% (Moidart).

2.2 Introduction

This represents the first phase of the investigation of the relationship between freshwater pearl mussels and their salmonid hosts. For each site, an estimate of the number of mussels in a standard length of river was required. The general techniques used were based on those successfully developed during previous surveys of Scottish *M. margaritifera* populations (Cosgrove *et al.* 2000a; Hastie *et al.* 2000a; Young *et al.* 2001). Most of the quantitative information about mussel beds described here also contributed toward the determination of annual mussel (surplus) production rates at selected sites in north-west Scotland (Section 7).

2.3 Methodology

2.3.1 Site selection

The work was carried out in the River Kerry cSAC and River Moidart cSAC in north-west Scotland. These two small rivers (overall length <20 km) support large, functioning *M. margaritifera* populations of international importance. Recent estimates of population size are >500,000 mussels in the River Kerry and >100,000 mussels in the River Moidart (Hastie *et al.* 2000). In both rivers, the lower reaches where mussels are found are <5 km in length. A number of sites on the lower Kerry and the lower Moidart (covering a range of observed mussel densities) were selected and surveyed during the period 1996–2001. For logistical reasons, only 1–4 sites per river were sampled annually, but other studies have indicated that mussel beds are usually very stable from year to year (Young & Williams 1983b).

2.3.2 Riverbed surveys

At each site, a 100 m length of riverbed was divided into 20 x 5 m sections. Equally spaced cross-river transects (1 m wide) were positioned at the start (downstream end) of each section. The transects were surveyed by wading across the river channel and counting the numbers of 'visible' mussels in a 1 m wide strip of riverbed using a metal 1 m^2 quadrat frame and a glass-bottomed viewing bucket. The term 'visible' refers to mussels seen without disturbing the riverbed, whereas 'hidden' refers to those completely buried in the riverbed sediments (Hastie *et al.* 2000a). This resulted in a total length of 20 m completely surveyed per 100 m length of river at each site. For each 100 m length of river the corresponding area of riverbed (m^2) was also estimated by multiplying by observed mean river widths.

2.3.3 Estimation of mussel density

For each site, the overall number of mussels per 100 m length of river (N) was estimated by multiplying the number of mussels found in the fully surveyed 20 m length (n) by 5. The approach was first tested and validated by comparing an estimate based on a 10 m section with the number of mussels counted during a complete survey of 50 m river length (estimate = 4,150 mussels per 50 m; actual count = 4,126 mussels per 50 m). Overall mussel density (D mussels per m^2) was estimated by:

D = N/A, where A = area of riverbed (m^2) in corresponding 100 m length of river.

For a sub-sample of the 20 m length sites (n = 7), estimates of mussel density and overall abundance of juvenile mussels (immature specimens < 65 mm shell length (L) – measured by callipers: Hastie 1999) were made using unpublished data collected during a number of previous *M. margaritifera* surveys on the River Kerry and River Moidart (Cosgrove 1998a, 1998b; Cosgrove & Young 1998; Hastie 1999; Cosgrove *et al.* 2000a; Hastie *et al.* 2000a, b; Hastie & Cosgrove 2002).

Total numbers of mussels (t mussels) found in the exhaustively searched 1 m^2 quadrats are obtained by adding together the visible (v) and hidden (h) mussel counts (t = v + h). Using the ratio of hidden to visible mussels from the quadrat counts (h/v), it is possible to estimate the overall total number of mussels (T) for each 100 m section. This total equals the visible number (v) plus an estimated number of hidden mussels based on h/v (T = v + (vh/v)). The mussels found in the quadrats were also measured and juveniles counted, where juveniles <65 mm L. This also allows an index of juvenile

abundance (% juveniles) to be calculated. Based on detailed knowledge of the mussel beds and riverbed habitats in the River Kerry and Moidart, an index of juvenile abundance was used for all sites (Table 1).

Table 1. Index of predicted juvenile abundance for River Kerry and River Moidart *M. margaritifera* sites, based on detailed local knowledge (1996–2002).

Juvenile abundance code	Predicted proportion of juvenile mussels (%)	General description
A	>20	Probably viable, sufficient recruitment.
B	15–19.9	Possibly viable, sufficient recruitment.
C	10–14.9	Possibly non-viable, insufficient recruitment.
D	<10	Probably non-viable, insufficient recruitment.

2.4 Results

During 1996–2002, riverbed surveys for mussels were carried out at 12 sites each on the River Kerry and the River Moidart. The areas of mussel bed surveyed (per 100 m length of river) ranged from 1280–1460 m^2 (Kerry) and 1070–1420 m^2 (Moidart). Estimated overall 'visble' mussel densities ranged from 1.81 to 32.23 mussels per m^2 (Kerry) and 0.96–16.60 mussels per m^2 (Moidart). The results of the Kerry and Moidart surveys are summarised in tables 2 and 3 respectively.

Table 2. Visible mussel counts and density estimates for 1 sites on the River Kerry (1996–2001).

Site	Location (NGR)	Year	River width (m)	Overall area (m^2)	Visible mussels (/20 m)	Estimated mussels (/100m)	Mussel density (/m^2)
K1	NG8187772869	1996	13.0	1300	8380	41900	32.23
K2	NG8156173951	1996	14.4	1440	2400	12000	8.33
K3	NG8156173951	1997	13.7	1370	2385	11925	8.70
K4	NG8156573937	1998	14.6	1460	1450	7250	4.97
K5	NG8156173951	1998	13.7	1370	7566	37830	27.62
K6	NG8156373928	1999	13.6	1360	1680	8400	6.18
K7	NG8157973914	2000	12.8	1280	1598	7990	6.24
K8	NG8189272884	2000	13.2	1320	7618	38090	28.86
K9	NG8185972844	2001	13.1	1310	6750	33800	25.80
K10	NG8158373906	2001	13.1	1310	1609	8045	6.14
K11	NG8185972844	2002	12.9	1290	468	2340	1.81
K12	NG8157863885	2002	12.8	1280	6704	33520	26.19

Table 3. Visible mussel counts and density estimates for 12 sites on the River Moidart (1997–2001).

Site	Location (NGR)	Year	River width (m)	Overall area (m^2)	Visible mussels (/20 m)	Estimated mussels (/100m)	Mussel density (/m^2)
M1	NM7395672157	1997	12.8	1280	3400	17000	13.28
M2	NM7294671840	1997	10.8	1080	840	4200	3.89
M3	NM7294671840	1998	10.8	1080	610	3050	2.82
M4	NM7300072115	1998	13.5	1350	1990	9950	7.37
M5	NM7296371855	2000	13.0	1300	766	3830	2.95
M6	NM7397672175	2000	11.1	1110	3246	16230	14.62
M7	NM7249171839	2001	14.2	1420	274	1370	0.96
M8	NM7319571989	2001	11.9	1190	1012	5060	4.25
M9	NM7389772138	2001	11.2	1120	2942	14710	13.10
M10	NM7393772131	2001	11.0	1100	3134	15670	14.25
M11	NM7394773140	2002	10.9	1090	2586	12930	1.77
M12	NM7390572144	2002	10.7	1070	3552	17760	16.60

Table 4 provides estimates of total (visible + hidden) mussel number and juvenile abundance at four Kerry sites (K3, K9, K10, K11) and three Moidart sites (M1, M7, M10). The quadrat data used to estimate these are also shown. Overall numbers of mussels at these sites ranged from 1679 mussels per 100 m (M7) to 64582 mussels per 100 m (K9). Absolute numbers of juvenile mussels (<65 mm shell length) ranged from 190 juveniles per 100 m (M7) to 15112 juveniles per 100 m (K9). Relative numbers of juvenile mussels ranged from 6.6% (M1) to 23.4% (K9). At all seven sites, very small mussels (<30 mm) were found.

Table 4. Observed (1 m^2) quadrat counts of mussels carried out at selected sites on the River Kerry and River Moidart. Predicted juvenile abundance codes (all sites) and estimates of the total (visible + hidden) numbers of mussels are also provided.

				Overall estimates		
Site	**N quadrats**	**Mussels visible / total**	**No. juveniles (%)**	**Juvenile abundance code***	**Mussels (per 100 m)**	**Juveniles (per 100 m)**
K1				A		
K2				B		
K3	17	187 / 245	41 (16.7)	B	15624	2609
K4				C		
K5				A		
K6				B		
K7				C		
K8				A		
K9	4	168 / 321	76 (23.4)	A	64582	15112
K10	16	233 / 320	44 (13.8)	C	11049	1525
K11	13	230 / 317	49 (15.5)	B	3225	500
K12				A		
M1	13	88 / 196	13 (6.6)	D	37864	2499
M2				C		
M3				C		
M4				D		
M5				C		
M6				C		
M7	14	253 / 310	35 (11.3)	C	1679	190
M8				D		
M9				C		
M10	19	212 / 340	32 (9.3)	D	25131	2337
M11				D		
M12			D			

***Definitions of juvenile abundance codes are provided in Table 1.**

Sites K3, K9, K10, K11, M1, M7 and M10 were targeted for more detailed information because of their importance to the investigation of the relationship between mussels and salmonid hosts – both fish density data and samples of wild fish (for determining the rate of glochidial infection) were obtained for these sites. Other data from previous studies (Cosgrove 1998a, b; Cosgrove & Young 1998; Hastie 1999; Cosgrove *et al.* 2000a; Hastie *et al.* 2000a, b; Hastie & Cosgrove 2002) and detailed knowledge of the distributions of mussel beds and the physical habitat conditions in the River Kerry and Moidart were used to make crude estimates of relative juvenile abundance at all twelve sites. Based on these, five Kerry sites were coded A (>20%) and four Kerry sites were coded B (15–19.9%). Three Kerry sites and six Moidart sites were coded C (10–14.9%), and six Moidart sites were coded D (<10%) (Table 4).

2.5. Discussion

These results indicate that large numbers of *M. margaritifera* were present at each site. In the field, it was usually difficult to assess the sizes of mussel beds and practically impossible to pre-determine the approximate numbers of mussels on the riverbed. However, the range of observed mussel numbers (approximately 1,500–6,5000 mussels per 100 m) was considered to be appropriate for testing relationships between mussel density and salmonid density and quantifying mussel seed production rates and survival rates required to maintain mussel numbers in the long term (Section 7).

Nevertheless, even though the sampling protocol was validated by comparing sample cross-sectional mussel counts with those in a completely surveyed 50 m section of river, it is stressed that considerable degrees of sampling error (not quantified) are expected in the data presented here. This is because it is very difficult to sample freshwater pearl mussels accurately in the field (Hastie & Cosgrove 2002) and they typically exhibit highly aggregated (clumped) distribution patterns on the riverbed (Hastie *et al.* 2000b). These features introduce error to estimates made by raising observed numbers of mussels in samples. The potential error is too great to determine sensible confidence limits. However, based on the findings of previous studies (Hastie *et al.* 2000a, b), the mussel density estimates presented here are considered to be of the correct order of magnitude.

There were insufficient data to determine the total numbers of mussels at all sites. Furthermore, it was apparent from the quadrat searches at selected sites that only 50–60% of the total numbers of mussels are usually visible on the riverbed at any given time. However, in a recruiting mussel bed, most of the hidden mussels are usually small, immature specimens (Hastie *et al.* 2000a). In other words, under-estimations of adult mussel numbers by counting only visible mussels are likely to be smaller, possibly 5–10%, in the River Kerry and River Moidart.

The quadrat data also indicated that the Kerry *M. margaritifera* population is producing relatively greater numbers of juveniles than the Moidart population at present. For cSAC monitoring purposes, the following criteria have been proposed to determine whether or not a *M. margaritifera* population is in favourable condition:

- At least 20% of the population are juvenile (<65 mm L), indicating adequate recruitment levels to maintain mussel numbers.
- The presence of small mussels (<30 mm L), indicating recent recruitment (within previous five years) (Young *et al.* 2003).

Based on these, only five Kerry sites (K1, K5, K8, K9 and K12) and no Moidart sites would be considered to be in favourable condition at present. Of the seven selected sites surveyed more thoroughly, only site K9 had >20% mussels <65 mm L, although two sites (K3 and K11) had 15–20% mussels <65 mm L. Only one Moidart site (K7) had >10% mussels <65 mm L. However, small numbers of mussels <30 mm L were found at all seven sites, indicating that measureable levels of recruitment are still occurring in both rivers.

In summary, the results provide mussel density data for appropriate observed ranges of mussel abundance (1679–64582 mussels per 100 m) and recruitment success (6.6–23.4% juveniles) for quantifying relationships between *M. margaritifera* and their salmonid hosts in Section 7. Unfortunately, the sample and sub-sample sizes (n = 24 and n = 7, respectively) are small.

3 Monitoring glochidial release events

3.1 Summary

During 1996–2002, annual releases of freshwater pearl mussel glochidia were monitored at several sites on the River Kerry cSAC and River Moidart cSAC in north-west Scotland. Estimated daily peak releases

ranged from 10.8–441.1 million glochidia per day (Kerry) and 0.3–31.3 million glochidia per day (Moidart) (whole river estimates).

3.2 Introduction

The following section represents the second phase of the investigation of the relationship between freshwater pearl mussels and their salmonid hosts. For each site, estimates of the numbers of glochidia released by the mussels during annual spat events were required. The general techniques used were based on those successfully developed during previous studies of Scottish *M. margaritifera* populations (Hastie 1999). Most of the quantitative information about glochidial releases described here contributed toward the determination of annual mussel production rates at selected sites in north-west Scotland (Section 7).

3.3 Methodology

3.3.1 Site selection

The work was carried out in the River Kerry cSAC and the River Moidart cSAC in north-west Scotland. Full details of the sites used are provided in Section 2.2.1.

3.3.2 Examining mussels

In order to determine the timing of spawning and predict the onset of glochidial release, small samples of live adult mussels (shell length L >70 mm, n = 50) were taken from the rivers Kerry and Moidart on various dates during May–July each year. These were examined non-destructively for gravidity by carefully opening the shell valves with special opening tongs, and checking for the presence of glochidia in the modified gill structures (marsupia) of the female mussels. The developing glochidia can be easily seen as a cream-coloured mass within the translucent brown gills (Young & Williams 1983a). Since sex could not be determined in the field, the fertility rate was recorded as an overall proportion of examined mussels containing glochidia. Prior to and immediately after examination, the mussels were kept in plastic buckets of river water before being returned to the riverbed.

3.3.3 River sampling

At each site, a 5 m x 5 m area immediately downstream of each of the 100 m long surveyed section of mussel bed (see Section 2.2.2) was searched and any visible mussels were removed. This was carried out in order to reduce sampling error associated with large, undiluted spats from individual female mussels situated nearby. Small plankton nets (25 cm diameter, 53 mm mesh), facing upstream were then stationed at the lower end of the cleared area (5 m below the surveyed mussel bed. An attempt was made to position each net in a typical (intermediate) flow within the channel section. The nets were sampled and re-set daily (usually between 0800 and 1000 hours soak (immersion) times were recorded). Samples were transferred in total and maintained in 500 ml plastic bottles containing river water for 1–4 days at 4°C prior to processing in the laboratory.

River widths and mean cross-sectional depths (adjacent to the net openings) were also determined, in order to obtain cross-sectional area estimates (m^2) for each site.

3.3.4 Estimating glochidial numbers

In the laboratory, background glochidial numbers were estimated volumetrically using a small counting dish and a low-power stereo microscope (x 35 magnification). The counting dish was a clear, plastic petri dish (9 cm diameter) with a grid of 0.25 cm^2 squares ink-marked below the base. The samples were usually diluted 2–3 times (occasionally more, when large amounts of suspended material were collected in the nets during spate conditions) and 5 x 15 ml volumes of diluted water were sub-sampled, with replacement using a graduated pipette. For each 15 ml volume placed in the petri dish, 10

grid squares were examined and the numbers of glochidia counted, giving a total count for 50 x 0.25 cm^2 squares per sample. The sub-sample size (number of grid square counts) required was previously determined by calculating the moving average of up to 200 counts at 10 count intervals for five different samples (Hastie 1999). The glochidia counts were converted to total number estimates as follows:

Calculation of counting dish area:

Petri dish radius (r) = 4.5 cm

Petri dish area = πr^2 = 63.585 cm^2

Grid square area = 0.25 cm^2 (area of 50 squares = 12.5 cm^2)

Addition of 15 ml sub-sample to dish:

63.585 cm^2 area contains 15 ml (diluted) river water

50 squares (12.5 cm^2) contains $\frac{12.5 \times 15}{63.585}$ = 2.949 ml (diluted) river water

Constant for converting to glochidia per ml (c) = $\frac{1}{2.949}$ = 0.339

Total estimate:

No. of glochidia/sample = NcVD

where N is the number of glochidia counted in 50 grid squares
V is the volume of diluted sample (ml)
D is the dilution of original sample
c is the conversion constant

Finally, converting to a daily estimate:

No. of glochidia/net/day (g) = $\frac{24}{t}$.NcVD

where t is the net soak time (h)

An attempt was also made to estimate the total number of glochidia that passed the entire river cross-section where each net was positioned. Based on the assumption that mean river current passed through each net, and that the number of glochidia trapped in the net was proportional to the overall number of glochidia that passed that point on the river, then:

$$\text{No. of glochidia/section/day (G)} = \frac{g.A}{a}$$

Where A = cross-sectional area of river (m^2)
a = area of net opening (= 0.049 m^2)

3.4 Results

During 1996–2001, annual releases of glochidia (spats) were monitored at 12 sites on the River Kerry and 12 sites on the River Moidart. Represented cross-sectional areas of river sampled ranged from 3.01–9.32 m^2 (Kerry) and 1.24–5.80 m^2 (Moidart). Estimated daily peak releases in the River Kerry ranged from 10.8 million glochidia/section/day (site K10, 1999) to 441.1 million glochidia/section/day (site K12, 2002). Estimated daily peak releases in the River Moidart ranged from 0.3 million glochidia/section/day (site M5, 2001) to 31.3 million glochidia/section/day (site M8, 2001). The results of the Kerry and Moidart glochidia monitoring programs are summarised in tables 5 and 6, respectively.

Table 5. Estimated annual peak glochidia releases for 12 sites on the River Kerry (1996–2001).

Site	Date	River section area (m^2)	Estimated glochidia per day (net) x 10^6	Estimated glochidia per day (section) x10^6
K1	15.07.96	3.01	1.400	86.000
K2	15.07.96	6.79	0.273	37.830
K3	16.07.97	5.81	0.193	22.884
K4	13.07.98	9.32	0.221	42.035
K5	13.07.98	5.33	0.510	55.476
K6	15.07.99	7.21	0.100	14.714
K7	04.07.00	4.46	1.257	114.413
K8	04.07.00	3.36	1.383	94.834
K9	09.07.01	3.21	0.660	43.237
K10	09.07.01	6.00	0.088	10.776
K11	28.06.02	5.81	0.403	47.784
K12	28.06.02	5.36	4.032	441.051

Site locations provided in Section 3.4.

Table 6. Estimated annual peak glochidia releases for 12 sites on the River Moidart (1997–2001).

Site	Date	River section area (m^2)	Estimated glochidia per day (net) x 10^6	Estimated glochidia per day (section) x10^6
M1	14.07.97	4.30	0.137	12.022
M2	14.07.97	1.60	0.137	4.473
M3	12.07.98	1.35	0.045	1.240
M4	12.07.98	5.00	0.061	6.224
M5	04.07.00	1.85	0.008	0.302
M6	04.07.00	1.24	0.203	5.137
M7	08.07.01	4.66	0.005	0.476
M8	08.07.01	5.00	0.040	4.082
M9	08.07.01	4.26	0.070	6.086
M10	08.07.01	5.08	0.302	31.309
M11	01.08.02	4.59	0.154	14.426
M12	01.08.02	5.80	0.077	9.114

3.5 Discussion

From previous studies (Young & Williams 1984a, Hastie 1999), it is known that *M. margaritifera* spats are highly synchronised events, with most of the glochdia released within one to two days. In the laboratory, female pearl mussels can be induced to release all their glochidia in a few hours (Young & Williams 1984b). In the field, it is estimated that approximately 50% and 90% of the total glochidia produced by all mussels in the population may be released within 24 hours and 48 hours, respectively (unpublished data). Therefore, it is possible to estimate the total number of glochidia released at each site, based on the numbers observed during the period of peak release.

However, the numbers of glochidia sampled were consistently much smaller than expected, based on the estimated numbers of gravid female mussels at each site and the known number of glochidia that develop in 'average' sized mussels (see Section 7). This may be due in part to sampling error (for example, under-estimation of numbers of glochidia collected in plankton net raised to cross-section of river).

The pattern of glochidial dispersal is not fully understood. In small streams with large mussel populations, glochidia may be dispersed everywhere by the currents (Young & Williams 1984a), but this is unconfirmed. There is likely to be a steep decline in numbers with distance from the mussel beds due

to dilution effects and the glochidia becoming trapped or damaged as they drift downstream. An assessment of the distances glochidia can be transported and remain viable and the rate of decline in numbers would be useful in determining overall host availability. Based on the results of preliminary trials involving artificial releases of glochidia and plastic tubes and trout hosts as collectors in two German streams, Jansen *et al.* (2001) reported that some *M. margaritifera* glochidia could drift and remain infective over distances of 200–500 m. In Scotland, distances of 100–500 m have been reported in some rivers (Hastie & Young 2001).

It is also likely that a significant proportion of the glochidia produced do not enter the water column. Perhaps glochidia are not all released by the female mussels (some may be re-absorbed). Spat glochidia are usually released together in small sticky masses and short strings – some of these may either become trapped in the riverbed sediments or directly ingested by fish, rather than being transported downstream in the water column. Consequently, it is clear that the number of glochidia caught in plankton nets in the water column are much lower than the total developing in the mussels. Nevertheless, the estimates of glochidia numbers collected in the nets are considered to be of the correct order of magnitude. It would also be appropriate to consider their use as an index of the total numbers of glochidia released at each site and so a useful guide to the numbers that will be encountered by potential fish hosts.

4 Does electric-fishing harm pearl mussels?

4.1 Summary

Two experiments were carried out in the River South Esk cSAC, north-east Scotland, to test the effects of electric-fishing on endangered freshwater pearl mussels. In the first experiment, two areas of mussel bed were marked out as treatment and control sites, and the former was electric-fished using standard equipment. The mussels were examined 10 minutes, 24 hours and 35 days after treatment, and their shell valve closure responses, burrowing capabilities and gravidities were recorded as signs of normal functioning. In the second experiment, individual mussels were marked as treatments and controls, and the former were electric-fished. These mussels were examined as before.

No mortalities occurred, and no significant differences in functioning between treatments and controls were observed in either experiment. Since no measurable treatment effect could be demonstrated, it appears that electric-fishing did not adversely affect the short-term survival of *M. margaritifera*.

4.2 Introduction

A principal aim of this project was to promote the conservation of both mussels and salmonids. Since the project would have involves a considerable amount of electric-fishing near mussel beds, it was considered worthwhile to conduct a short study of potential detrimental effects of electric-fishing on *M. margaritifera* before any fieldwork was carried out. The findings of this work have already been published (Hastie & Boon 2001) and are reproduced here with the kind permission of John Wiley & Sons Ltd.

Electric-fishing – the application of an electrical current to water in order to capture fish – is a standard cost-effective sampling technique that has been widely used by fishery biologists for almost 50 years (Snyder 1995). It was developed primarily for stock assessments and ecological studies of freshwater fish, particularly salmonids (Bohlin *et al.* 1989), but it has now also been used successfully to sample populations of freshwater shrimp (Penczak & Rodriguez 1990), crayfish (Rabeni *et al.*1997) and other aquatic invertebrates (Nagel 1993).

Although electric-fishing has long been considered to be the ideal non-destructive fish-sampling tool,

there is now growing concern about the injurious effects of electric-fishing, particularly on endangered fish populations, and a number of studies have been undertaken recently (Barrett & Grossman 1988; Snyder 1995; McMichael *et al.* 1998; Nielsen 1998; Habera *et al.* 1999). By contrast, very little is known about the effects of electric-fishing on aquatic invertebrates, other than the fact that certain groups (for example, arthropods) can be successfully sampled by this method (Penczak & Rodriguez 1990; Nagel 1993; Rabeni *et al.* 1997). Since many aquatic invertebrate species around the world are now threatened, it is important that studies of the effects of electric-fishing on different invertebrate groups (for example, arthropods and molluscs) are also carried out.

The freshwater mussels (Unionacea) comprise a highly threatened group of aquatic invertebrates (Bauer & Wachtler 2001). In Scotland, the endangered *M. margaritifera* is found in several important salmon and trout rivers that have been extensively electrofished for many years (Hastie 1999).

Freshwater mussels use different fish species as larval hosts. For example, *M. margaritifera* larvae (glochidia) can only complete their development on the fry and parr of commercially important Atlantic salmon (*Salmo salar*) or brown trout (*Salmo trutta*), and large numbers of these young fish are often found within or near mussel beds (pers. obs.). Given this type of association, it is likely that a large number of threatened mussel beds in different rivers around the world have been exposed to the effects of electric-fishing. Therefore, in the interests of conservation, studies to determine whether electric-fishing has any damaging effects on freshwater mussels would be worthwhile.

The following investigation was carried out to assess the effects of electri-fishing on a population of *M. margaritifera* in a river in northern Scotland.

4.3 Methodology

The study was carried out during June–August 2000 at two sites (A and B) on a river in northern Scotland, each supporting a large, viable *M. margaritifera* population. It is necessary to keep specific site details confidential because of the present threat of illegal pearl fishing (Hastie *et al.* 2000a). Site A is a natural mussel bed in the river (approximately 50–100 m^2 of riverbed). Site B, located about 2 km further upstream, is an old disused mill lade (100 m long) with running water, which usually contains several hundred live mussels washed in during floods.

Site A

Two 10 m^2 areas of mussel bed (10 m apart) were marked out as treatment and control areas. The control areas were not located between the electrodes. Standard electric-fishing equipment (250 V, 100 Hz DC, 25% duty cycle) was used and the entire treatment area was 'swept' three times. A number of fish in the treatment area were temporarily immobilized, indicating that the equipment was functioning properly, but no immobilized fish were seen in the control area during treatment. The control area was also 'swept' (with no electric-fishing) by wading over the mussel bed three times in order to produce a similar level of physical disturbance in both areas. The mussels in the treatment and control areas were then checked and compared 10 minutes, 24 hours and 35 days after treatment.

The shell valve closure responses of the mussels to gentle prodding (10 minutes, 24 hours and 35 days), burying capabilities (24 hours, 35 day) and gravidities (35 day only, during annual spawning event) were checked as signs of normal functioning. Mussels were checked for gravidity by carefully opening the shell valves with special tongs, and checking for the presence of glochidia in the modified gill structures (marsupia) of the female mussels (Young & Williams 1983a). Since the sex of *M. margaritifera* cannot be determined in the field, gravidity was recorded as an overall proportion of examined mussels containing glochidia (Hastie 1999).

Site B

A random sample of 100 adult mussels (shell length L range 70–120 mm) was taken from the upper lade where mussels collect during floods. The samples were divided into two, and the mussels marked by scoring either 'E' (treatment) or 'X' (control) on their shells (n = 2 x 50). The mussels were

transported in an insulated plastic container of water to the lower lade (approximately 100 m downstream, no mussels) where the treatment mussels were placed in the water and electric-fished as described previously (the control mussels were simply disturbed and then returned to the container). A number of fish in the lower lade were immobilized during treatment. The treated mussels were put in the container with the control mussels and were then returned to the upper lade. These were also checked 10 minutes, 24 hours and 35 days after treatment.

4.4 Results

Site A

Most of the mussels in both the treatment and control areas closed their shell valves during the sweeps, probably as a result of physical disturbance. Even when the power was on and the fish were affected, the mussels in the treatment area appeared to react (by withdrawing their siphons and closing their shell valves) only when they were physically disturbed. Within 10 minutes after electric-fishing, several mussels in both areas began to open their shell valves again and filter-feed normally (indicated by protruding siphons). Mussels of all sizes (10–120 mm L) appeared to be unaffected by the treatment.

By 24 hours, all visible mussels in both areas appeared to be filter feeding normally and they exhibited a normal shell valve closing response to prodding. After 35 days, the mussels continued to react normally. The mussels had already commenced spawning by this time, but there were still a number of gravid mussels in both areas. The proportions of gravid mussels observed were 6/50 (12%) and 9/50 (18%) in the treatment and control areas, respectively.

Site B

The treated mussels again reacted only to physical disturbance, even when the power was on. Within 10 minutes after they were returned to the lade, some of both the treated and control mussels began to open up and filter feed normally again. By 24 hours, most of the mussels had re-buried themselves in the sandy bottom of the lade so that only a third of their shells remained visible above the surface. As before, all mussels appeared to react normally to prodding.

After 35 days, only 47 treated and 48 control mussels had recovered, but these continued to react normally. The mussels had already begun spawning by this time, but a number of treated and control mussels were still gravid. The proportions of gravid mussels observed were 14/47 (30%) treated and 16/48 (33%) control mussels.

4.5 Discussion

Electric-fishing had no observed effect on the mussels, and the conclusion from this work is that it does not adversely affect the short-term survival of *M. margaritifera*. The only significant difference observed was between the proportions of gravid mussels in the lade and those in the river. It appears that the mussels in the river had begun spawning 1–2 days earlier. Since the lade is very shaded and has a much slower flow of water, this may be due to slight differences in water temperature and/or dissolved oxygen content (which can influence the timing of spawning in this species: Hastie & Young 2003). Whatever the reasons, it is clear that electric-fishing was not a factor. However, these results do not rule out entirely the possibility that the mussels might have suffered some undetected injury.

Since *M. margaritifera* is highly endangered (Young *et al.* 2001), it would certainly be worthwhile to carry out further research to confirm or refute the findings of this study. For example, the control and treated mussels could have been examined during the next annual spawning episode in 2001 to check for any longer-term effects. Various types of electric-fishing gear (and different gear settings) are widely used by fisheries biologists (Tillma 1996) and the possible adverse effects of these on the mussels should be tested. Sampling protocol may be important (P. Maitland, pers. comm.). A small number of detailed histological examinations and comparisons of the tissues of treated and control mussels would also be worthwhile.

The possible indirect effects of electric-fishing on mussels should also be investigated. For example, it has been demonstrated that electric-fishing can harm host fish (including significant egg mortality and spinal injuries), although how these translate into population effects has not been adequately studied (Nielsen 1998). According to Ziuganov *et al.* (1994), a low host fish density may be a limiting factor in some *M. margaritifera* populations. In Scotland, stocks of wild salmonids are declining (Walker 1993) and there is concern now that host fish densities may be sub-critical in some mussel rivers (Hastie & Cosgrove 2001). A number of small streams have mussel populations that do not appear to be recruiting adequately, shown by a lack of juveniles (Hastie *et al.* 2000a). Perhaps, as a precaution, repeated electric-fishing over mussel beds in these streams should be avoided, if possible, until further research is carried out.

5 Glochidia present on samples of wild fish

5.1 Summary

During 1997–2002, samples of wild juvenile salmonids were taken from the River Kerry cSAC and River Moidart cSAC in north-west Scotland, and examined for freshwater pearl mussel glochidia. The observed incidence of infected 0+ salmon ranged from 70–95% (Kerry) and 33–83% (Moidart). Individual loads on 0+ salmon ranged from 0–1260 glochidia per fish (Kerry) and 0–253 glochidia per fish (Moidart). The observed incidence of infected 1++ salmon was 29% (Kerry) and ranged from 20–50% (Moidart). Individual loads on 1++ salmon ranged from 0–750 glochidia per fish (Kerry) and 0–16 glochidia per fish (Moidart). Overall, fewer glochidia were found on juvenile trout (0–46% incidence), although one large specimen (aged 3+) caught in the River Kerry had 3,920 glochidia.

5.2 Introduction

The following section represents the fourth phase in the investigation of the relationship between freshwater pearl mussels and their salmonid hosts. For each site, estimates of the numbers of encysted *M. margaritifera* glochidia on the gills of juvenile salmon and trout were required. The work was carried out in collaboration with local salmon fishery trust biologists. Techniques used to estimate glochidial loads on fish samples were based on those successfully developed during previous studies of *M. margaritifera* glochidiosis in Scottish wild and farmed salmonid stocks (Hastie & Young 2001). Most of the quantitative information about glochidial infection levels described here contributed toward the determination of annual mussel (surplus) production rates at selected sites in north-west Scotland (Section 7).

5.3 Methodology

5.3.1 Site selection

The work was carried out in the River Kerry cSAC and the River Moidart cSAC in north-west Scotland. Full details of the sites used are provided in Section 2.2.1.

5.3.2 Collection of wild fish

Fish were captured by electric-fishing (100 m^2 swept area at each site) and were killed (overdosed) in 30 ppm benzocain anaesthetic solution. In the field, specimens were identified (to species) and measured (fork length FL to nearest mm) and then fixed in 10% buffered formalin (excess $CaCO_3$) solution for >48 h.

5.3.3 Examination of wild fish

In the laboratory, the fish were rinsed in tap water. A small sample of scales (approximately 20) was removed from each specimen by scraping below the base of the dorsal fin with a scalpel. Scales were examined under a stereo microscope (x 10) and ages were determined by counting the number of winter growth rings. The fish were then examined for glochidia. The opercula were carefully removed using a scalpel and scissors, and the excised gills were placed between two glass slides. The numbers of glochidia were then counted using a compound microscope (x 50) and a tally counter.

5.4 Results

During 1997–2002, seven samples of Kerry fish and five samples of Moidart fish were examined for pearl mussel glochidia. The observed incidence of infected salmon fry (aged 0+) ranged from 70% (site K7, 2000) to 95% (site K3, 1997) in the River Kerry, and from 33% (site M7, 2001) to 83% (site M1, 1997) in the River Moidart. Individual loads on 0+ fish ranged from 0–1260 glochidia/fish (site K10, 2001) in the River Kerry, and from 0–253 glochidia per fish (site M5, 2000) in the River Moidart. Fewer glochidia were found on older fish. The observed incidence of infected salmon parr (aged 1++) was 29% (site K3, 1997) in the River Kerry, and ranged from 20% (site M1, 1997) to 50% (sites M7+10, 2001) in the River Moidart. Individual loads on 1++ fish ranged from 0-750 glochdia per fish (site K4, 1997) in the River Kerry, and from 0–16 glochidia per fish (site M10, 2001) in the River Moidart. The results of the Kerry and Moidart juvenile salmon examinations are summarised in tables 7 and 8, respectively.

Table 7. Observed glochidial infection loads on juvenile salmon caught in the River Kerry (1997–2002).

				Infection loads (glochidia per fish)					
Site	Date	Age	N	Mean	SE	Median	SD	Range	Incidence(%)
K3	15-10-97	0+	21	452	72	548.0	328	0–1200	95
K3	15-10-97	1++	13	76	41	0	201	0–750	29
K7	30-10-00	0+	10	21	14	2.5	44	0–132	70
K8	30-10-00	0+	20	76	30	9.0	134	0–483	80
K9	10-10-01	0+	21	269	61	206.0	280	0–774	81
K10	10-10-01	0+	25	524	82	464.0	407	0–1260	84
K11	08-10-02	0+	29	33	9	10.0	46	0–176	72

Table 8. Observed glochidial infection loads on juvenile salmon caught in the River Moidart (1997–2001).

				Infection loads (glochidia per fish)					
Site	Date	Age	N	Mean	SE	Median	SD	Range	Incidence(%)
M1	28-10-97	0+	12	34	14	8.0	48	0–130	83
M1	28-10-97	1++	15	1	1	0	3	0–12	20
M5	07-11-00	0+	14	60	23	28.5	84	0–253	79
M5	07-11-00	1++	4	1	1	0.5	0	0–2	50
M6	07-11-00	0+	20	45	11	30.0	51	0–180	80
M7	19-10-01	0+	15	10	6	0	21	0–76	33
M10	19-10-01	0+	15	43	17	20.0	66	0–212	87
M7+10	19.10.01	1++	4	4	4	0.5	8	0 - 16	50

Very small samples of trout were also obtained from the River Kerry (n = 14, 1997) and River Moidart (n = 2, 2000) and examined for glochidia. Overall, the incidence of trout fry infected with glochidia (46%) appeared to be lower than that observed for salmon fry (95%) (site K3, 1997). However, one large trout (aged 3+) that was captured and included in the same sample had 3,920 glochidia. The results of the Kerry and Moidart trout examinations are summarised in Table 9.

Table 9. Observed glochidial infection loads on trout caught in the River Kerry and River Moidart (1997–2000).

Site	Date	Age	N	Infection loads (glochidia per fish)					
				Mean	SE	Median	SD	Range	Incidence(%)
K3	15-10-97	0+	13	87	69	0	245	0–890	46
K3	15-10-97	3+	1	3920	--	3920	---	------	100
M5	07-11-00	0+	1	0	--	0	---	------	0
M5	07-11-00	1++	1	0	--	0	---	------	0

5.5 Discussion

The prevalence of natural glochidial infections of *M. margaritifera* in the Kerry and Moidart juvenile salmonid stocks are comparable to those reported elsewhere (Young & Williams 1984a; Bauer 1987a; Cunjak & McGladdery 1991; Beasley 1996). For example, Bauer (1987b) recorded a 100% incidence of glochidia in a sample of 0+ trout from a Bavarian stream. Incidences as low as 3% (salmon) and 7% (trout) in some Irish rivers by Beasley (1996). In the Stac Burn in north-west Scotland, 70% of captured 0+ trout were found to have glochidia (Young & Williams 1984a). In Nova Scotia, 58–92% prevalence in 0+ salmon was reported by Cunjack & McGladdery (1991). Mean infection loads of 400 glochidia per fish and 12–153 glochidi per fish have been observed in 0+ trout and 0+ salmon samples, respectively (Young & Williams 1984a, Cunjak & McGladdery 1991). Naturally large loads of >1000 glochidia per fish have occasionally been observed in salmon and trout stocks (Young & Williams 1984; Bauer 1987b; Beasley1996).

The results presented here indicate that in both rivers, 0+ salmon carry far more glochidia than 1++ salmon. In a number of rivers, older salmonids are clearly less important as hosts for *M. margaritifera* than 0+ fish (Hastie & Young 2001). Greater infection loads on young wild salmonids have been frequently reported (Awakura 1968; Bauer 1979, 1987b, c; Bauer & Vogel 1987; Young & Williams 1984a), and this seems to be a general feature of the margaritiferid–host relationship (Hastie & Young 2001). Although, exceptionally, a few older fish carry glochidia (for example, 3+ trout (site K4)), in the wild it is mainly 0+ fish that appear to be infected (Young & Williams 1984a; Bauer 1987b; Hastie & Young 2001). A number of possible reasons why older fish are less likely to carry glochidia have been suggested, including reduced exposure, increased resistance and acquired immunity (Hastie & Young 2001).

These results also demonstrate that young salmon are the main natural hosts of *M. margaritifera* in both rivers. Salmon appear to be the main hosts in Nova Scotia (Cunjak & McGladdery1991), and Russia. (Ziuganov *et al.* 1994). Bauer (1987a,b) suggests that salmon become increasingly important as hosts at higher latitudes. Farther south, trout are apparently their main hosts – for example in Ireland (Beasley 1996) and Germany (Bauer 1987a, b). In Scotland, there appears to be an overlap in host utilisation. In many rivers that support *M. margaritifera* populations and both host species, juvenile salmon are far more abundant than trout (Hastie & Young 2001). In such circumstances, one would expect salmon to be the most important host. However, there are several small streams in northern Scotland (and many in Germany) with *M. margaritifera* populations that have no salmon (Young & Williams 1984a) and mussels in these must be trout-dependent.

An important final point to make is that all the samples were taken in the autumn (October–November), at least three months after the fish were exposed to glochidia. In the River Kerry and River Moidart, *M. margaritifera* usually spat late June-early July annually (Hastie & Young 2003). The numbers of *M. margaritifera* glochidia on individual host fish tend to decline rapidly after initial attachment (Fustish & Millemann 1978; Young & Williams 1984a, b; Bauer 1987a; Young *et al.* 1987; Ziuganov *et al.* 1994). This is thought to be due to a humoral response initiated by the host (Young & Williams 1984a, Bauer, 1987a). For the present study, the (October-November) sampling period was chosen in order to allow the numbers of glochidia to have partly stabilised (avoiding the greatest fall in numbers following exposure). Based on previous work in north-west Scotland (Young & Williams 1984b; Hastie & Young 2001), approximately. 50% of *M. margaritifera* glochidia initially attached to the

host gills remain encysted three months after exposure. Therefore, the estimates presented here may be doubled to correspond to the numbers of glochidia initially attached to the host fish in July.

6 Salmonid host density data

6.1 Summary

Juvenile salmonid density estimates for the River Kerry cSAC (1997–2002) and River Moidart cSAC (1997–2001) were obtained from Wester Ross Fishery Trust (WRFT) and Lochaber & District Fishery Trust (LDFT), respectively. Observed densities of 0+ salmon ranged from 2.0–102.2 fish per 100 m^2 (Kerry) and 10.6–50.8 fish per 100 m^2 (Moidart). Observed densities of 1++ salmon ranged from 11.5–25.1 fish per 100 m^2 (Kerry) and 0.0–19.6 fish per 100 m^2 (Moidart). Observed densities of 0+ trout ranged from 0.0–0.4 per 100 m^2 (Kerry) and 0.0–3.3 per 100 m^2 (Moidart). Observed densities of 1++ trout ranged from 0.0–8.4 per 100 m^2 (Kerry) and 0.0–1.6 per 100 m^2 (Moidart).

6.2 Introduction

For each site, overall estimates of juvenile salmonid numbers present in a standard length of river were required. The work was carried out on two rivers in collaboration with salmon fishery trust biologists, who supplied fish density data obtained during annual and biannual electric-fishing surveys. Attempts were also made to obtain juvenile salmonid data for a list of electric-fishing sites in Scotland known to support *M. margaritifera* populations. Most of the quantitative information about salmonid densities derived here contributed toward the determination of annual mussel (surplus) production rates at selected sites in north-west Scotland (Section 7).

It is axiomatic that successful mussel recruitment depends upon the presence of sufficient host fish, yet there have been no substantial studies of the relationship between the numbers of hosts and mussel recruitment success. The vague suggestions in the literature as to the densities of juvenile salmonids necessary for optimal mussel recruitment, (such as those of Zuiganov *et al.*1994) may not apply in Scottish rivers. Consequently, the data collected here are an essential starting point for a proper understanding of this vital topic.

6.3 River Kerry cSAC

Juvenile salmonid density data for two main stem sites on the River Kerry cSAC (1997–2002) were provided by Wester Ross Fisheries Trust (WRFT). Where possible, these were matched with adjacent mussel sites (surveyed during the same year, within 100 m). Observed densities of juvenile salmon in the Kerry ranged from 2.0–102.2 fry per 100 m^2 and 11.5–25.1 parr per 100 m^2. Observed densities of juvenile trout ranged from 0.0– 0.4 fry per 100 m^2 and 0.0–8.4 parr per 100 m^2. Overall densities of juvenile salmonids ranged from 25.9–118.3 fish per 100 m^2. These are summarised in tables 10 and 11.

Table 10. Densities of juvenile salmon and trout recorded during WRFT electric-fishing surveys of the River Kerry (1997-2002).

Year	Site*	NGR	Density (fish per 100 m^2)				
			Salmon 0+	Salmon 1++	Trout 0+	Trout 1++	Total density
1997	KRY5 (K3)	NG815740	9.2	15.7	2.8	0.0	27.7
1997	KRY3	NG820729	2.0	23.5	2.5	0.7	28.7
1999	KRY5 (K6)	NG815740	38.2	17.8	0.0	0.0	56.0
1999	KRY3	NG820729	35.7	13.0	4.6	0.0	53.3
2001	KRY5 (K10)	NG815740	8.8	17.1	0.0	0.0	25.9
2001	KRY3 (K9)	NG820729	102.2	11.5	4.6	0.0	118.3
2002	KRY5 (K12)	NG815740	64.9	20.5	0.0	8.4	93.8
2002	KRY3 (K11)	NG820729	65.3	25.1	0.0	8.0	98.4

*** Nearest mussel survey site in parentheses and only loose association is possible.**

Table 11. Summary statistics of juvenile salmonid densities (fish per100 m^2, pooled samples) recorded in the River Kerry (1997–2002).

Site(KRY)	Species	Age	N	Mean	Median	SD	Range
3	salmon	0+	4	51.3	50.5	42.7	2.0–102.2
3	salmon	1++	4	18.3	18.3	7.0	11.5–25.1
3	trout	0+	4	2.9	3.6	2.2	0.0–4.6
3	trout	1++	4	2.2	0.4	3.9	0.0–8.0
5	salmon	0+	4	30.3	23.7	26.9	8.8–64.9
5	salmon	1++	4	17.8	17.5	2.0	15.7–20.5
5	trout	0+	4	0.7	0.0	1.4	0.0–2.8
5	trout	1++	4	2.1	0.0	4.2	0.0–8.4
3+5	salmon	0+	8	40.8	37.0	34.9	2.0–102.2
3+5	salmon	1++	8	18.0	17.5	4.8	11.5–25.1
3+5	trout	0+	8	1.8	1.3	2.1	0.0–4.6
3+5	trout	1++	8	2.1	0.0	3.8	0.0 - 8.4

6.4 River Moidart cSAC

Juvenile salmonid density data for three main stem sites on the River Moidart cSAC (1997–2001) were provided by Lochaber & District Fishery Trust (LDFT). As for the Kerry, these were matched with mussel survey sites. Observed densities of juvenile salmon in the Moidart ranged from 10.6–50.8 fry per 100 m^2 and 0.0–19.6 parr per 100 m^2. Observed densities of juvenile trout ranged from 0.0–3.3 fry per 100 m^2 and 0.0–1.6 parr per 100 m^2. Overall juvenile salmonid densities ranged from 17.8–65.1 fish per 100 m^2. These are summarised in tables 12 and 13.

Table 12. Densities of juvenile salmon and trout recorded during LDFT electric-fishing surveys of the River Moidart (1997–2001).

Year	Site*	NGR	Density (fish per 100 m^2)				
			Salmon 0+	Salmon 1++	Trout 0+	Trout 1++	Total density
1997	MOI2 (M2)	NM727718	12.2	5.6	0.0	0.0	17.8
1997	MOI4 (M1)	NM735721	15.8	5.3	0.3	0.3	21.7
1998	MOI2 (M3)	NM727718	33.5	6.6	0.0	0.0	40.1
1998	MOI4 (M4)	NM735721	31.8	19.6	3.3	1.6	56.3
1999	MOI2	NM727718	17.9	0.5	0.1	0.0	18.5
1999	MOI4	NM735721	50.8	12.2	2.1	0.0	65.1
2001	MOI2 (M7)	NM727718	15.8	0.0	0.4	0.4	16.6
2001	MOI2a (M8)	NM726718	10.6	6.9	1.6	0.0	19.1
2001	MOI4 (M9+10)	NM735721	18.0	0.8	0.8	0.0	19.6

* Nearest mussel survey site in parentheses and only loose association is possible.

Table 13. Summary statistics of juvenile salmonid densities (fish per 100 m^2, pooled samples) recorded in the River Moidart (1997–2001).

Site(MOI)	Species	Age	N	Mean	Median	SD	Range
2	salmon	0+	4	19.9	16.9	9.4	12.2–33.5
2	salmon	1++	4	3.2	3.2	3.4	0.0–6.6
2	trout	0+	4	0.1	0.1	0.2	0.0–0.4
2	trout	1++	4	0.1	0.0	0.2	0.0–0.4
4	salmon	0+	4	29.1	24.9	16.1	15.8–50.8
4	salmon	1++	4	9.5	8.8	8.2	0.8–19.6
4	trout	0+	4	1.6	1.5	1.4	0.3–3.3
4	trout	1++	4	0.5	0.2	0.8	0.0–1.6
2+2a+4	salmon	0+	9	22.9	17.9	13.2	10.6–50.8
2+2a+4	salmon	1++	9	6.4	5.6	6.3	0.0–19.6
2+2a+4	trout	0+	9	1.0	0.4	1.1	0.0–3.3
2+2a+4	trout	1++	9	0.3	0.0	0.5	0.0–1.6

6.5 Discussion

It is difficult to determine accurately juvenile salmonid densities by electric-fishing, and it is stressed that considerable degrees of error (not quantified) are expected in the data presented here. Nevertheless, in general the electric-fishing survey data provided by WRFT and LDFT indicate the following:

- The River Kerry supports greater numbers of juvenile salmonids than the River Moidart.
- Salmon are more abundant than trout in both rivers.
- Fry are more abundant than parr in both rivers.

Ziuganov *et al.* (1994) suggested that a critical minimum salmonid host density of 0.2 fish per m^2 is required for maintaining *M. margaritifera* populations in the long term. All samples from the River Kerry produced overall juvenile salmonid density estimates of >0.2 fish per m^2 (>20 fish per100 m^2), whereas five samples from the River Moidart produced estimates of <0.2 fish per m^2 (this is discussed further in Section 7). It was also planned to compare salmonid density estimates for a number of other *M. margaritifera* sites in Scotland, but unfortunately, data held by the Scottish Salmon Fisheries Database were not accessible.

7 The relationship between mussels and host fish

7.1 Summary

The relationship between the freshwater pearl mussel and its salmonid hosts was investigated. The study was based on quantitative information on mussel abundance, glochidial production/infestation and juvenile salmonid abundance observed in the River Kerry cSAC and River Moidart cSAC in north-west Scotland. Significant positive relationships were observed between mussel density and glochidial production. However, the numbers of glochidia sampled were much smaller than were expected, based on the estimated numbers of gravid female mussels at each site and an estimate of the average number of glochidia present in each female mussel. In general, mussel abundance appeared to be positively correlated with juvenile salmon abundance.

At all sites, the majority of encysted glochidia (75–99%) appeared to be carried by 0+ salmon hosts. Relatively small numbers of encysted glochidia (0–4%) appeared to carried by trout hosts. Positive relationships were observed between the number of encysted glochidia and the number of unattached glochidia in the rivers. Estimates of the probability of a glochidium successfully encountering and

encysting on a suitable host gill were very low (<0.04%). Estimates of the total number of mussel seed produced at each site within a river ranged from 113–18,505 mussels per year. Estimates of post-settlement survival rates required in order to maintain present mussel numbers at five sites ranged from 4–30%. However, at two sites on the River Moidart, the numbers of mussel seed produced appeared to be less than required to maintain present mussel numbers. An apparent trend between the surplus of mussel seed produced and the relative abundance of juvenile mussels at each site was observed.

7.2 Introduction

Although a number of baseline studies have been carried out (usually of the general ecology of the mussel, rather than its specific relationship to fish hosts) (Young & Williams 1984a, b; Bauer 1987a, b, c; Cunjak & McGladdery 1991; Ziuganov *et al.* 1994; Hastie 1999; Hastie & Young 2001; Jansen *et al.* 2001), much more focused research is required. For example, there is a distinct lack of field data from individual rivers, and little is known about the form of any relationship between host stock sizes and the reproductive success of mussels (Chesney & Oliver 1998). According to Ziuganov *et al.* (1994), a critical minimum threshold salmonid host density of 0.2 fish per m^2 is required for maintaining *M. margaritifera* populations. However, this is a vague figure derived from observation rather than from detailed study, and it has never been checked properly in a range of rivers.

In this section quantitative information is provided on the margaritiferid–salmonid relationship in the River Kerry cSAC and River Moidart cSAC. The analyses use data obtained during field studies (1997–2002) as previously described (sections 2, 3, 5 and 6).

7.3. Methodology

This study was based on quantitative information on mussel abundance, glochidial production rates, host fish infestation rates and salmonid host abundance obtained for the River Kerry cSAC and River Moidart cSAC (Table 14). Full details on selected sites and how the data were obtained are provided in sections 2, 3, 5 and 6. Based on these, estimates of the overall juvenile salmonid abundance, glochidial production and (newly-settled) mussel seed production were made for a number of sites.

For each site (100 m section of river), estimates of the total number of unattached glochidia produced were based on the assumption that 40% of the 'visible' mussels became gravid (Hastie 1999) and that the average gravid female mussel produced 1 million viable glochidia (a conservative estimate). It was also assumed that 50% of the glochidia released were shed during the day of maximum release (Hastie 1999).

The glochidial loads on juvenile fish recorded at three months post-exposure were doubled to provide estimates of original loads (following Young & Williams 1984b). Numbers of juvenile salmon and trout (S fish per100 m) were estimated by multiplying observed fish density (d fish per m^2) by the total area of riverbed (A m^2) within the section:

$$S = Ad$$

The total numbers of encysted glochidia (G glochidia per 100 m) on juvenile salmon and trout were then estimated from mean glochidial loads (g glochidia per fish) observed at each site:

$$G = Sg$$

Based on the findings of previous studies (Young & Williams 1984a, Hastie & Young 2001), it was assumed that 5% of the encysted glochidia completed development and excysted as small mussels (number of mussel seed produced = 0.05 G).

It was also assumed that, on average, female *M. margaritifera* had a reproductive lifespan of 50 years (based on work by Bauer 1992).

7.4 Results

7.4.1 Mussel density and glochidial production

Those sites on the Rivers Kerry and Moidart with large mussel beds appeared to produce the most glochidia. Significant positive relationships were observed between mussel abundance and glochidial production (Figure 2), and between mussel density and glochidial production (Figure 3). However, the numbers of glochidia sampled were much smaller than expected, based on the estimated numbers of gravid mussels at each site and the assumption that each female releases 1 million glochidia each year. Comparisons with the expected numbers of glochidia produced indicated that relatively small numbers were actually observed (Table 15). Estimates of the proportions observed based on the net samples ranged from 0.1–2.9% for the River Kerry and 0.1–1.0% for the River Moidart.

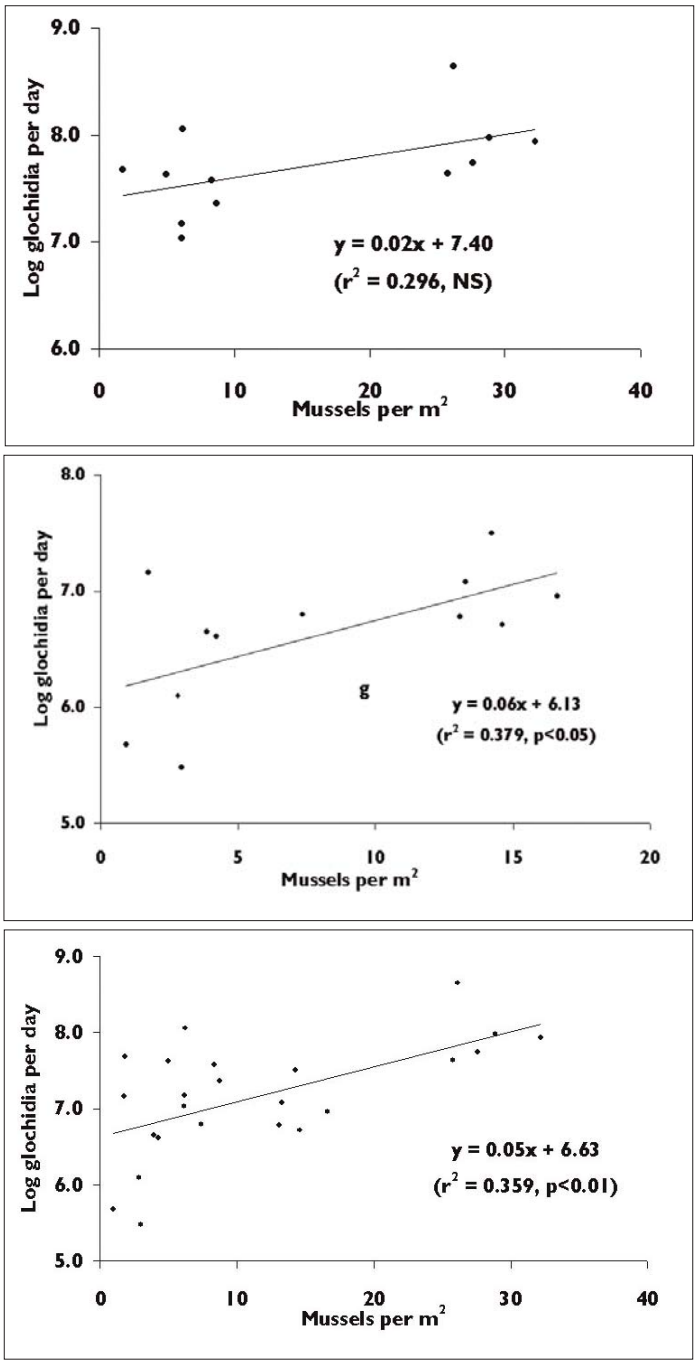


Figure 2. Scatter-plots of glochidial production against mussel density. Regression line equations and correlation coefficients displayed. Top: Kerry, centre: Moidart, bottom: Kerry &

7.4.2 Mussel abundance and juvenile salmonid abundance

In general, mussel abundance appeared to be very weakly positively correlated with juvenile salmon abundance (tables 16–17). The exceptions were negative relationships observed between mussels and 0+ salmon (Moidart, Table 16) and 1++ salmon (Kerry, Moidart, tables 16-17). However, except for a positive relationship observed between mussel density and 0+ salmon density (pooled sample, Table 17), none of the observed relationships were significant, probably due to small sample size (n = 6 Kerry + 6 Moidart sites).

7.4.3 Estimation of encysted glochidia production

From the fish density data and corresponding mean glochidial loads observed, it was possible to estimate the total numbers of encysted glochidia at four Kerry and three Moidart sites. The results are summarised in Table 18. At all sites, the majority of encysted glochidia appeared to be carried by 0+ salmon hosts. Estimates of the proportions of the overall numbers of glochidia on 0+ salmon ranged from 75–97% in the River Kerry and approximately 99% in the River Moidart. Relatively small numbers of encysted glochidia appeared to be carried by trout hosts. Estimates of the proportions of the overall numbers of glochidia on trout ranged from 0-4% in the River Kerry and 0-1% in the River Moidart.

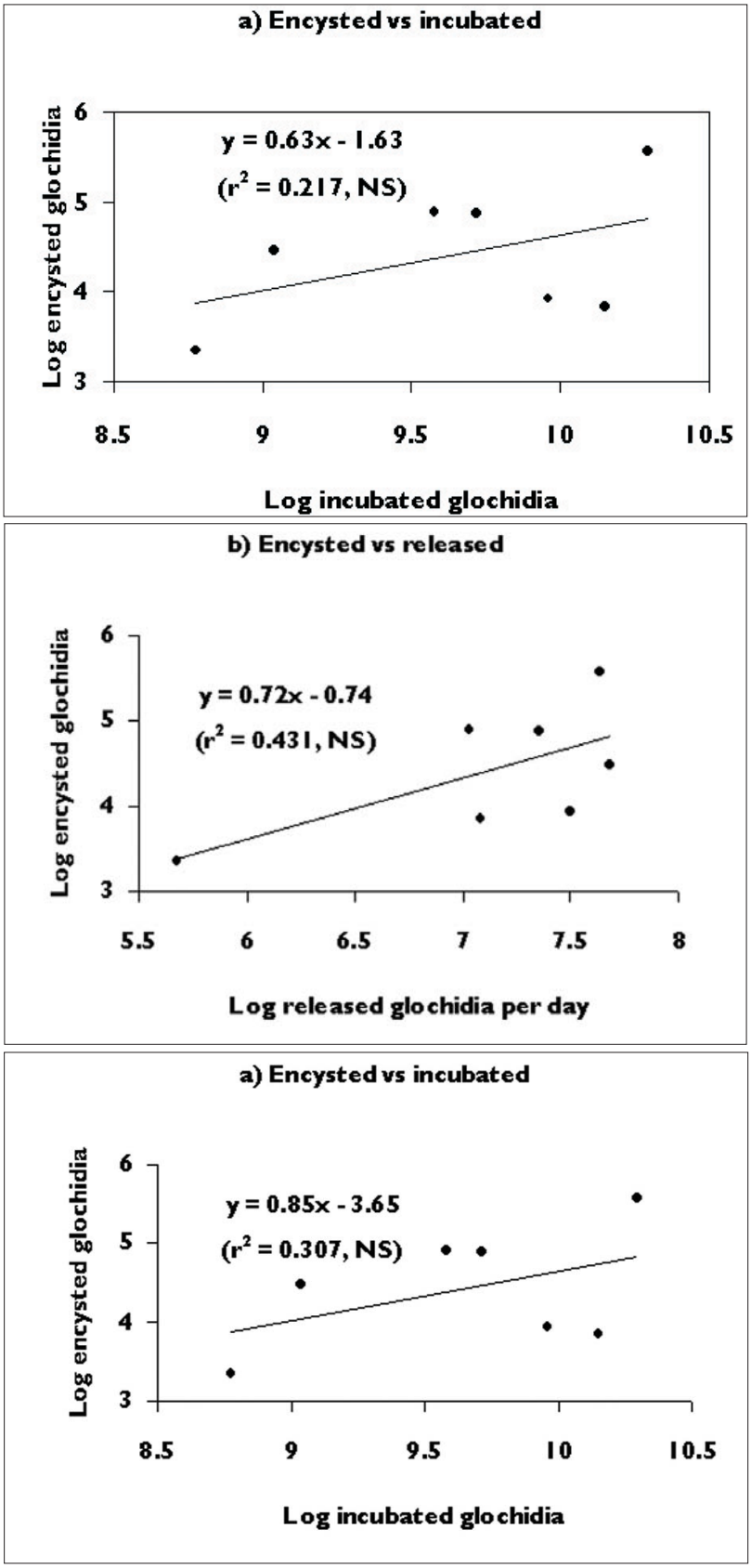


Figure 3. Scatter-plots of number of encysted glochidia against: a) incubated glochidia estimated for gravid females) and b) peak glochidia release. Regression line equations and correlation coefficients displayed. n = 4 Kerry + 3 Moidart sites. NS = not significant at p = 0.05. Top: Kerry, centre: Moidart, bottom: Kerry & Moidart.

Positive relationships were observed between the number of encysted glochidia and the expected number of incubated glochidia based on the numbers of gravid female mussels (Figure 1a), and between the number of encysted glochidia and the number of unattached glochidia collected in the plankton nets during the peak release day (Figure 1b). Table 19 provides survival estimates (to encystment stage) based on the expected numbers of glochidia incubated by gravid female mussels, and the numbers of unattached glochidia collected in the plankton nets during spat events. These indicate enormous losses in glochidia prior to successful encystment. Based on the numbers of glochidia incubated by gravid female mussels, the probability of a glochidium contacting and successfully encysting on a suitable host fish ranged from 0.0001–0.0374%. Based on the observed numbers of unattached glochidia collected in the plankton nets, the probability of successful encystment ranged from 0.03–0.9%.

7.4.4 Estimation of mussel seed production

Based on previously published work (Young & Williams 1984a, Hastie & Young 2001), a survival rate of 5% was expected for encysted *M. margaritifera* glochidia between initial encystment and successful development to mussel 'seed'. This represents the fraction that would probably remain on the host fish over winter, metamorphose and eventually excyst and drop off as viable small mussels (seed) during the following spring. Based on this

Table 14. Mussel abundance, glochidial production rates, host fish infestation rates and salmonid host abundance obtained for the River Kerry cSAC and River Moidart cSAC.

	Mussel density		**Glochidia release**	**Host density**				**Host abundance**				**Glochidial loads**			
Site	Mussels 100 m^{-1}	Mussels m^{-2}	Glochidia day^{-1}	Salmon 0+ m^{-2}	Salmon 1++ m^{-2}	Trout 0+ m^{-2}	Trout 1++ m^{-2}	Salmon 0+ 100 m^{-1}	Salmon 1++ 100 m^{-1}	Trout 0+ 100 m^{-1}	Trout 1++ 100 m^{-1}	Salmon 0+ glo. $fish^{-1}$	Salmon 1++ glo. $fish^{-1}$	Trout 0+ glo. $fish^{-1}$	Trout 1++ glo.$fish^{-1}$
K1	41900	32.23	86000000												
K2	12000	8.33	37830000												
K3	11925	8.7	22884000	0.092	0.157	0.028	0	126	215	38	0	904	152	174	30
K4	7250	4.97	42035000												
K5	37830	27.62	55476000												
K6	8400	6.18	14714000	0.382	0.178	0	0								
K7	7990	6.24	114413000					520	242	0	0	42			
K8	38090	28.86	94834000									152			
K9	33800	25.8	43237000	1.022	0.115	0.046	0	1339	151	60	0	538	90	104	18
K10	8045	6.14	10776000	0.088	0.171	0	0	115	224	0	0	1048	176	202	34
K11	2340	1.81	47784000	0.653	0.251	0	0.08	842	324	0	103	66	12	12	2
K12	33520	26.19	441051000	0.649	0.205	0	0.084	831	262	0	108				
M1	17000	13.28	12022000	0.158	0.053	0.003	0.003	202	68	4	4	68	2	14	0
M2	4200	3.89	4473000	0.122	0.056	0	0	132	60	0	0				
M3	3050	2.82	1240000	0.335	0.066	0	0	362	71	0	0				
M4	9950	7.37	6224000	0.318	0.196	0.033	0.016	429	265	45	22				
M5	3830	2.95	302000									28	8	0	0
M6	16230	14.62	5137000									90			
M7	1370	0.96	476000	0.158	0	0.004	0.004	224	0	6	6	20	8	4	0
M8	5060	4.25	4082000	0.106	0.069	0.016	0	126	82	19	0				
M9	14710	13.1	6086000	0.18	0.008	0.008	0	202	9	9	0				
M10	15670	14.25	31309000	0.18	0.008	0.008	0	198	9	9	0	86	8	16	2
M11	12930	1.77	14426000												
M12	17760	16.6	9114000												

Table 15. Comparison of expected and observed glochidial production at each site. Based on assumptions that 40% of 'visible' mussels were gravid, each gravid mussel produced 1 million glochidia and 50% of total glochdia were released during peak event (day of maximum release).

	Mussel density		Glochidial production		
Site	Overall 100 m^{-1}	Gravid mussels 100 m^{-1}	Expected glochidia x 10^6	Observed glochidia x 10^6	% expected that were observed
K1	41900	16760	16760	172	1.0
K2	12000	4800	4800	76	1.6
K3	11925	4770 (5206)	4770 (5206)	46	1.0 (0.9)*
K4	7250	2900	2900	84	2.9
K5	37830	15132	15132	111	0.7
K6	8400	3360	3360	29	0.1
K7	7990	3196	3196	229	7.2
K8	38090	15236	15236	190	1.2
K9	33800	13520 (19788)	13520 (19788)	86	0.6 (0.4)
K10	8045	3218 (3810)	3218 (3810)	4	0.1 (0.1)
K11	2340	936 (1090)	936 (1090)	19	2.0 (1.7)
K12	33520	13408	13408	176	1.3
M1	17000	6800 (14146)	6800 (14146)	24	0.9 (0.2)
M2	4200	1680	1680	9	0.5
M3	3050	1220	1220	2	0.2
M4	9950	3980	3980	12	0.3
M5	3830	1532	1532	1	0.1
M6	16230	6492	6492	10	0.2
M7	1370	548 (596)	548 (596)	1	0.2 (0.2)
M8	5060	2024	2024	8	0.4
M9	14710	5884	5884	12	0.2
M10	15670	6268 (9118)	6268 (9118)	63	1.0 (0.7)
M11	12930	5172	5172	30	0.6
M12	17760	7104	7104	18	0.3

*Alternative estimates for selected sites, based on gravid females = 40% total (visible + hidden) adult mussels estimated from 1 m^2 quadrat counts in parentheses.

Table 16. Regression equation parameters and correlation coefficient estimates for a model of relationship between mussel abundance and host fish abundance (Log (n+1) fish per 100m = *a*Log mussels per 100 m + *b*).

River	Species	Age	N	*a*	*b*	r^2	Significance
Kerry	salmon	0+	6	0.242	1.663	0.055	NS
Moidart	salmon	0+	6	-0.001	2.337	0.001	NS
pooled	salmon	0+	12	0.215	1.627	0.065	NS
Kerry	salmon	1++	6	-0.171	3.056	0.467	NS
Moidart	salmon	1++	6	0.491	-0.360	0.080	NS
pooled	salmon	1++	12	0.475	0.018	0.087	NS
Kerry	salmon	all	6	0.136	2.320	0.046	NS
Moidart	salmon	all	6	0.020	2.365	0.002	NS
pooled	salmon	all	12	0.224	1.745	0.087	NS
Kerry	trout	all	6	0.477	-0.694	0.046	NS
Moidart	trout	all	6	0.500	-1.010	0.101	NS
pooled	trout	all	12	0.570	-1.188	0.093	NS
Kerry	both	all	6	0.139	2.330	0.047	NS
Moidart	both	all	6	0.025	2.362	0.002	NS
pooled	both	all	12	0.230	1.743	0.089	NS

NS = not significant at p = 0.05.

Table 17. Regression line equation parameters and correlation coefficient estimates for a model of relationship between mussel density and host fish density (fish per m^2 = *a*mussels per m^2 + *b*).

River	Species	Age	N	*a*	*b*	r^2	Significance
Kerry	salmon	0+	6	0.021	0.214	0.394	NS
Moidart	salmon	0+	6	-0.001	0.205	0.008	NS
pooled	salmon	0+	12	0.020	0.128	0.333	*
Kerry	salmon	1++	6	-0.002	0.204	0.210	NS
Moidart	salmon	1++	6	-0.002	0.070	0.023	NS
pooled	salmon	1++	12	0.001	0.103	0.005	NS
Kerry	salmon	all	6	0.019	0.418	0.326	NS
Moidart	salmon	all	6	-0.003	0.275	0.016	NS
pooled	salmon	all	12	0.020	0.230	0.263	NS
Kerry	trout	all	6	0.001	0.022	0.164	NS
Moidart	trout	all	6	0.001	0.011	0.002	NS
pooled	trout	all	12	0.002	0.009	0.176	NS
Kerry	both	all	6	0.021	0.440	0.326	NS
Moidart	both	all	6	-0.030	0.286	0.013	NS
pooled	both	all	12	0.022	0.240	0.264	NS

* $p<0.05$, NS = not significant at $p = 0.05$.

Table 18. Estimates of initial numbers of encysted glochidia on juvenile salmonids at sites in the River Kerry and River Moidart.

Site	0+ salmon	1++ salmon	0+ trout	1++ trout	Total No.of glochidia
K3	113904 (74.5)	32680 (21.3)	6612 (4.2)	0 (0)*	153196
K9	720382 (97.3)	13590 (1.9)	6240 (0.8)	0 (0)	740212
K10	120520 (75.4)	39424 (24.6)	0 (0)	0 (0)	159944
K11	55572 (93.1)	3888 (6.5)	0 (0)	206 (0.4)	59666
M1	13736 (98.6)	136 (1.0)	56 (0.4)	0 (0)	13928
M7	4480 (99.5)	0 (0)	24 (0.5)	0(0)	4504
M10	17028 (98.7)	72 (0.5)	144 (0.8)	0 (0)	17244

* % overall in parentheses

assumption, the numbers of mussel seed expected to be produced the following year at each site were estimated using the numbers of encysted glochidia in Table 16. If female *M. margaritifera* have a reproductive span of 50 years, then during this period each female must produce at least two viable adult mussels to maintain the population in the long term. In other words, every year 2/50 = 0.04 offspring per female must survive annually.

For the mussel seed produced in the wild, a post-settlement survival rate of 5% (to adulthood) has also been reported (Young & Williams 1984a). Based on these assumptions, expected numbers of mussel seed that would survive to adulthood (and reproduce) and actual numbers of adult offspring required in order to maintain present mussel numbers at the sites studied were estimated.

Comparisons between the actual numbers of mussel seed produced and the numbers required to survive at each site are made in Table 20. These survival estimates can then be compared to the 5% rate previously suggested as typical, to see if they are sufficiently likely to allow the retention of a sustainable mussel population. These estimated required post-settlement survival rates ranged from 3% (site K10), which seems plausible, to 15% (site M7), which seems implausibly high, even at sites where

Table 19. Survival (to encystment stage) estimates, based on expected numbers of incubated glochidia (estimate 1) and recorded numbers of unattached glochidia sampled in plankton nets (estimate 2) at sites in the River Kerry and River Moidart.

Site	Incubated glochidia $x10^9$	Unattached glochidia $x10^6$	Encysted glochidia $x\ 10^3$	Survival estimate (1%)	Survival estimate(2%)
K3	5.206	22.884	153.196	0.0029	0.3348
K9	19.788	43.237	740.212	0.0374	0.8560
K10	3.810	10.776	159.944	0.0042	0.7422
K11	1.090	47.784	59.666	0.0055	0.0624
M1	14.146	12.022	13.928	0.0001	0.0580
M7	0.596	0.476	4.504	0.0008	0.4732
M10	9.118	31.309	17.244	0.0002	0.0276

Assumptions: 40% mussels gravid females, each produces 1 million glochidia (estimate 1); 50% unattached glochidia released during peak day (estimate 2), 50% initially attached glochidia remain encysted after three months (estimates 1+ 2).

sufficient numbers of mussel seed were produced. However, at site M10 (River Moidart), the number of seed produced appeared to be less than the required number of (resulting) adult mussels.

The surplus production ratios (R = number of adults produced per year/number of adults required per year) computed for the River Kerry sites appeared to be at least an order of magnitude greater than those computed for the River Moidart sites. At all Kerry sites, but no Moidart sites, the estimates of adult production were sufficient to maintain mussel numbers in the long term (R >1). If the estimates suggest that a sustainable recruitment rate is being achieved, then there should be observable numbers of juveniles present. In fact, an apparent trend between the surplus of adult mussels produced and the relative abundance of juvenile mussels at each site was observed (Figure 4), although the relationship was not significant ($p>0.05$), possibly due to small sample sizes (n = 7 sites).

Table 21 compares estimates of adult mussel production, juvenile salmonid density and juvenile mussel relative abundance for each site. These indicate that, in terms of long-term viability and *M. margaritifera* conservation status (juvenile abundance, adult production, host availability), the River Kerry cSAC appears to be in good condition at present, whereas the River Moidart cSAC does not. Greater juvenile salmonid densities were observed in the River Kerry (0.259–1.183 fish per m^2) than the River Moidart (0.166–0.217 fish per m^2). Hence, host availability may be one of the factors involved.

Table 20. Annual mussel production estimates and the numbers of newly-settled mussel seed required to survive to adulthood in order to maintain mussel numbers at selected sites on the River Kerry and River Moidart.

Site	No. females per 100 m	Mussel seed per year produced	Adult mussels per year produced	Adult mussels per year required[a]	Post-settlement survival rate (%) required[b]	Surplus production ratio R (produced/ required)
K3	7812	7660	380	312	4.1	1.22
K9	32291	37010	1850	1292	3.5	1.43
K10	5525	7998	400	221	2.8	1.81
K11	1612	2984	150	64	4.0	2.34
M1	18932	696	34	757	Insufficient	0.04
M7	840	226	12	34	15.0	0.35
M10	12565	862	44	503	Insufficient	0.09

[a]**The number of adult mussels required to be produced, if a stable population is to be achieved.**

[b]**The post-settlement survival rate (%) needed to produce sufficient adults to allow a stable population, given the actual number of mussel seed produced.**

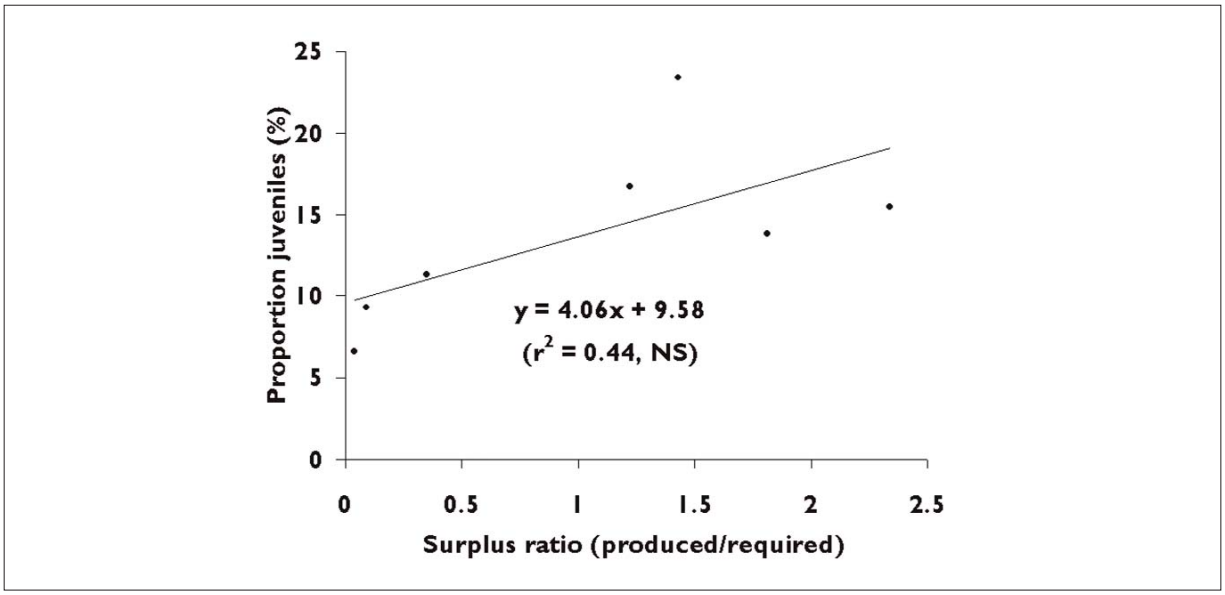


Figure 4. Scatter-plot of proportion of juvenile mussels against predicted surplus ratio of adult mussels produced. Regression line equation and correlation coefficient displayed. n = 4 Kerry + 3 Moidart sites. NS = not significant at p = 0.05.

Table 21. Estimates of observed juvenile salmonid density, predicted number of adult mussels produced and observed abundance of juvenile mussels at selected sites on the River Kerry and River Moidart.

Site	Juvenile salmonid density (fish m^{-2})	Adult production ((mussels $year^{-1}$ (%))[a]		Juvenile mussel relative abundance (% <65 mm L)
		surplus	deficit	
K3	0.277	+68 (32)		16.7
K9	1.183	+558 (43)		23.4
K10	0.259	+179 (81)		13.8
K11	0.984	+86 (134)		15.5
M1	0.217		-723 (96)	6.6
M7	0.166		-22 (65)	11.3
M10	0.196		- 459 (91)	9.3

[a]Figures derived from Table 19.

7.5 Discussion

As discussed briefly in Section 3, the discrepancy between estimates of unattached glochidia collected in the nets and those based on the numbers of gravid female mussels at each site may be due to several factors, including sampling error, failure of females to release all their glochidia into the water column and/or exponential decline of drifting glochidia over very short distances (for example, <10 m). It may be inappropriate to raise numbers of glochidia collected in the nets without proper hydarualic measurements to determine the volumes of water passing though the nets in relation to the entire river cross-section. It would also be worthwhile to station nets across the entire width of the river in order to estimate the total numbers of glochidia at selected sites. Given the present uncertainty associated with obtaining realistic estimates of the numbers of glochidia released per unit of river length, perhaps it would be more appropriate to consider their use as an index rather than as absolute values (in this chapter, absolute values are used for comparative purposes only). In any case, the number of glochidia released was not required for estimating the number of mussels produced at each site. The estimates of survival (glochidia produced to successful encystment) for the River Moidart (0.0001–0.0008%) are comparable to those reported by other workers. Rates of 0.0004% (Young & Williams 1984a) and 0.0002% (Jansen *et al.* 2001) have previously been reported. However, the

estimates for the River Kerry are much greater (0.0029-0.0374%).

There appeared to be a relationship between mussel density and the numbers of glochidia collected in the nets. However, given the enormous numbers of glochidia released in large mussel beds (>10000 mussels per 100 m) and the much smaller numbers of suitable hosts available (for example, 1000 fish per 100 m), it is unlikely that the level of glochidial production is a limiting factor for (future) mussel distribution. It may be different for extremely low densities of mussels (for example, <100 mussels per 100 m) but even less is known about glochidial dispersal in these conditions. In most situations, host availability is likely to be a far more important factor than overall glochidial production.

In general, mussel abundance appeared to be positively correlated with juvenile salmonid abundance. This suggests that host availability may be limiting for *M. margaritifera* distribution at some sites. It is also possible that the margaritiferid:host relationship is mutalistic and that juvenile salmonids may benefit from the presence of mussels (Ziuganov *et al.* 1994). However, it may also simply reflect a shared habitat preference between mussels and salmonids.

It is apparent that, in terms of both overall availability and suitability, 0+ salmon are the most important hosts for *M. margaritifera* in both rivers, although older salmon and trout also appear to be used as hosts when available (Young & Williams 1984a, Hastie & Young 2001). Greater numbers of glochidia have been frequently reported on 0+ salmonids (Awakura 1968; Karna & Millemann 1978; Bauer 1979, 1987a, b, c; Bauer & Vogel 1987) and this seems to be a general feature of the margaritiferid–host relationship. A number of possible reasons why older fish are less likely to carry glochidia include reduced exposure, increased resistance to initial attachment and acquired immunity leading to greater subsequent rejection (Hastie & Young 2001).

The apparent differences in host availability between the River Kerry and River Moidart sites reflect differences in predicted adult mussel production and the observed relative abundance of juvenile mussels (recruitment). According to Ziuganov *et al.* (1994), a critical minimum salmonid host density of 0.2 fish per m^2 is required for maintaining *M. margaritifera* populations in the long term. The salmonid densities recorded in the River Kerry were all >0.2 fish per m^2, whereas only one site in the River Moidart had a host density >0.2 fish per m^2. Thus, it does seem that host density may be a limiting factor in some *M. margaritifera* populations. However, more research is required before this can be properly ascertained.

Although the critical 0.2 fish per m^2 level suggested by Ziuganov *et al.* (1994) appears to be close to the mark in the examples presented here, this may be coincidental. The number of fish required will ultimately depend on the number of mussels required to be replaced and this will vary naturally according to the size of the mussel beds at the reach scale and the overall population at the river scale.

Finally, it is stressed that the results presented here are based on a large number of assumptions and extrapolations, so the potential for error is great. For example, the 5% post-settlement survival estimates (Young & Williams 1984a) were computed for optimal riverbed habitat conditions, and these may vary considerably between sites. Post-settlement survival in extant mussel beds may actually range from 1–10%, depending on the condition of the riverbed, but more research is required to determine this. This is important, because the early post-settlement phase of *M. margaritifera* is particularly sensitive to environmental conditions (Buddensiek *et al.* 1993).
Furthermore, the sample sizes were very small, owing to the difficulty in matching mussel density data with juvenile salmonid density data and obtaining corresponding samples of host fish for examination. Juvenile salmonid stocks are known to fluctuate considerably from year to year (Gibson 1993, Elliott 1994), hence a considerable time-series of fish density data will be required for more thorough analyses. Perhaps the best juvenile salmonid host criteria to use at present for *M. margaritifera* would be to require that native salmonid stocks exhibit long-term stability at natural levels, if the mussels are to survive. This is of particular importance in north-west Scotland, where salmonid stocks in a number of mussel rivers have declined considerably over the past decade (Hastie & Cosgrove 2001).

In summary, it appears that in certain rivers, host availability may be a limiting factor, although the limitations of the data used are stressed. At present, ensuring the long-term stability of local salmonid stocks should form part of the management strategy to conserve pearl mussels.

8 The effects of fishery-related physical habitat modifications on mussel beds

8.1 Summary

The impacts of river engineering work and physical habitat modification on *M. margaritifera* were reviewed. A number of salmonid fishery-related activities, including construction of bank reinforcements, croys (deflectors), fishing platforms, pool dredging, footbridges and weed control may adversely affect mussel beds to varying degrees. Of these, croys and pool dredging are considered to constitute a major threat to the conservation status of *M. margaritifera*.

8.2 Introduction

River engineering work has historically been responsible for the decline and extinction of a number of *M. margaritifera* populations and, at present it is a potentially significant threat (Cosgrove & Hastie 2001). Several studies have identified that physical habitat disturbances caused by pipe-laying operations, hydro-power generation schemes, channel re-alignments, cattle fords, bridge supports and fishery management schemes can have serious impacts on *M. margaritifera* populations (Jungbluth & Kuehnel 1978; Young & Williams 1983a; Valovirta 1990; Ziuganov *et al.* 1994; Killeen *et al.* 1998; Cosgrove *et al.* 2000; Cosgrove & Hastie 2001). Under the Habitats Directive, several viable pearl mussel rivers are currently being designated as SACs, thus providing some legal protection for the mussels and their habitat. Many of the problems previously associated with river engineering work have been accidental because little, if anything, was known about the presence of mussels beforehand.

However, the situation in western Europe is now different, since most of the remaining *M. margaritifera* populations have been identified as a result of extensive baseline surveys. Nevertheless, a number of important, viable populations are not in specially protected areas and small-scale developments that might affect mussel beds are often overlooked.

In theory, heightened awareness, full legal protection, control over the physical modification of river channels and up-to-date information on the presence of pearl mussels means that accidental damage is preventable. Recent studies have contributed to our knowledge and understanding of the physical habitat requirements of *M. margaritifera* (Buddensiek *et al.* 1993, Hastie *et al.* 2000a, 2003), although more research is urgently required, particularly for the early post-settlement stage of its life cycle.

In practice, however, the presence of *M. margaritifera* is usually one of many factors that have to be considered when managing a river. This is particularly important in the context of salmon fishing in Scotland, where almost half of all the known cases of *M. margaritifera* populations adversely affected by river engineering works involved activities that were directly related to fishery management (Table 22). An added factor is that salmon are also protected, so that the effects of river engineering have to be considered with respect to them, as well.

Table 22. Number of Scottish rivers with viable *M. margaritifera* populations adversely affected by river engineering activities. Based on information from Cosgrove & Young (1998).

Activity	No. rivers affected	Proportion of total (%)
Fishery management	15	47
Dam construction	8	25
Flood defence	8	25
Road maintenance	3	9
Pipe laying	2	6

The scale of observed fishery-related population effects and/or habitat degradation, in terms of area of mussel bed disturbed, ranged from <10 m^2 in small streams to >100 m^2 in large rivers (Cosgrove & Hastie 2001). Small-scale river engineering works and riverbed habitat modifications associated with

fishery management include bank reinforcements, croys (deflectors)/platform constructions, pool dredging, footbridges and weed control. The problems associated with these are discussed in detail below.

8.3 Bank reinforcements

The reinforcement of eroding riverbanks is a widespread artificial feature in many rivers in Scotland and elsewhere. Materials used to reinforce eroding riverbanks include steel piling, wood, brickwork, concrete, building rubble, wire gabions (baskets of pebbles/cobbles), rip-rap (block boulders) and loose, coarse riverbed material (usually pebbles/cobbles/boulders). The main threat to mussel beds occurs during the construction phase, when large amounts of (coarse or fine) sediment from the riverbank may be washed down and deposited on the riverbed. Reinforcement work may be associated with channel dredging operations (where riverbed material is used to reinforce banks), which can have a huge impact on *M. margaritifera* populations (Killeen *et al.* 1998). Based on the number of viable mussel beds found beside very old bank reinforcements, it appears that re-colonisation is often possible provided no further disturbance of the riverbed takes place. However, in some cases, hydrological changes associated with the bank reinforcement (for example, reduced flows) are detrimental, and long-term recovery is threatened by a significant reduction in the survival of juvenile mussels. Therefore, this activity is considered to represent an intermediate threat to the conservation status of *M. margaritifera*.

8.4 Croys (deflectors)

Croys are often used to deflect strong flows away from the riverbank thereby creating deep mid-channels that salmon may utilise and/or reducing the amount of erosion immediately downstream. Materials used to construct croys include concrete, building rubble and coarse riverbed material (cobbles, boulders). Although croys are often small structures that appear insignificant, they can have a huge impact on nearby mussel beds. The reduction of flow downstream and upstream of the structure can degrade *M. margaritifera* habitat by heavy silt/sand deposition on the riverbed. This process may prevent mussel beds from regenerating due to lowered survival of the newly-settled juveniles, which are highly sensitive to siltation (Buddensiek *et al.* 1993). Adult *M. margaritifera* may be able to tolerate these conditions for limited periods of time, depending on the rate of fine sediment deposition (Hastie *et al.* 2000a). However, but no juvenile mussels have ever been found beside croys.

The length of riverbed habitat affected ranges from 10–100 m, depending on the size of the river and the extent of hydrological change (unpublished data). Based on the large numbers of croys and their negative impacts on mussel beds, the construction of croys is considered to represent a major threat to the conservation status of *M. margaritifera*.

8.5 Fishing platforms

A number of small fishing platforms (10–50 m long), usually adjacent to deep water and made of concrete, wood or riverbed (cobble/boulder) materials can be found along some Scottish salmon rivers. At some locations, mussel beds may have been detrimentally affected in the short term during the construction of these structures, and in a small number of cases, associated hydrological changes may have degraded the riverbed habitat by deflection of flow as described for croys.

However, the relatively small size and number of fishing platforms built indicate that any detrimental effects on mussel beds have been limited to a few localities, with probably no significant overall effects on *M. margaritifera* populations. Therefore, fishing platform construction is considered to represent a minor threat to the conservation status of *M. margaritifera*.

8.6 Pool dredging

Dredging of the riverbed channel, in order to maintain fishing pools (or create new ones) to hold salmon is a widespread activity often carried out by fishery managers in Scotland. This activity is highly destructive and can cause considerable damage to nearby mussel beds. Killeen *et al.* (1998) reported the total destruction of one entire *M. margaritifera* population in north-west Wales by a channel dredging operation. Heavy machinery is typically used to remove riverbed material, and the integrity of the riverbed habitat may be completely destroyed. Large deposits of sand/silt associated with dredging operations may also pose a significant threat to mussel beds further downstream. Based on the large number of fishing pools maintained in this way, and the impact of dredging on mussel beds, this activity is considered to constitute a major threat to the conservation status of *M. margaritifera*.

8.7 Footbridges

Small wooden or metal footbridges (< 5 m wide), designed to provide fishing access to opposite banks are often found along Scottish salmon rivers. These are typically single-span (narrow suspension bridges over large rivers) with little impact, if any on the riverbed. A very short length of riverbank may be modified (usually faced with concrete, brick or wood) to support the bridge, and there may be some impact on local mussel beds during construction. Based on the small size and likely impact, the construction of small footbridges for fishing access is considered to constitute a minor threat to the conservation status of *M. margaritifera*.

8.8 Weed control

Fishing pools and river channels sometimes become choked with weeds, making it difficult for anglers to land catches. This has become more of a problem in recent years with the accidental introduction and rapid spread of invasive *Ranunculus* spp. into the River Spey and other rivers in Scotland (Laughton *et al.* 2003). Since the root structure of *Ranunculus* spp. bind fine sediment (sand, silt) particles and degrades the riverbed habitat, it is likely that its spread, if unchecked, could seriously affect the conservation status of mussels and salmonids in certain river systems.

Two methods traditionally used by fishery managers to control *Ranunculus* spp. are physical (hand) removal and chemical (application of herbicide). Until recently, the most effective herbicide available for control of *Ranunculus* spp. was Midstream (Diquat). However, current European Union (EU) legislation prevents the use of Diquat in watercourses. Thus, effective chemical control is no longer available to river and fishery managers in Europe. However, physical control is costly and time-consuming and there are health and safety implications as well as a number of environmental concerns.

Laughton *et al.* (2003) recently investigated the effects of physical removal of *Ranunculus* spp. on pearl mussels and juvenile salmonids in the River Spey. Preliminary results suggested little effect on mussels or fish. An effective hand-pulling method should be developed in order to minimise the impact on mussel beds and the spread of broken plant material downstream. The development of a comprehensive management plan to address the issue is currently under way in a separate study funded by Scottish Natural Heritage (Laughton *et al.* 2003). Based on the preliminary findings of this study, physical weed removal is considered to constitute a minor threat to the conservation status of *M. margaritifera*.

8.9 Discussion

This section illustrates the range of potential fishery-related operations that may impact *M. margaritifera* populations. In particular, the construction of croys (deflectors) and pool dredging is considered to

constitute a major threat to the conservation status of *M. margaritifera*. The effects of many damaging operations could be avoided if fishery managers were aware of the issues associated with the ecological and legal requirements of the species (Cosgrove & Hastie 2001). *M. margaritifera* is now fully protected under law in most countries, and guidance is urgently needed so that river managers can integrate ecological and socio-economic factors when considering the impact of proposed activities on *M. margaritifera* populations. Operations likely to harm mussels and permanently damage their habitat should not proceed.

9 Conclusions and recommendations

The estimates of mussel density (Section 2) were based on established survey techniques and are considered to be reliable. The estimates of numbers of unattached glochidia collected in the plankton nets (Section 3) are considered to be of the correct order of magnitude and may be useful as an index of glochidial release. However, the estimates of absolute numbers of glochidia for the entire river width are much smaller than expected, based on the number of gravid female mussels present at each site. It would be worthwhile to carry out more detailed studies by monitoring the pattern of glochidial dispersal both downstream and across the entire river width.

Electric-fishing is a very useful and cost-effective technique used to sample juvenile salmonid populations. Since it had no measurable adverse short-term effect on mussels (Section 4), it may be used in mussel–salmonid conservation studies. However, as a precaution, repeated electric-fishing over mussel beds in small streams should be avoided until further research is carried out.

The salmonid samples used in this study were very small with much potential for error (sections 5–6). Therefore, any detailed conclusions based on these cannot be considered to be very reliable. Nevertheless, a number of general points can be made.

There appeared to be a general association between mussel and salmonid distribution and abundance in the River Kerry and River Moidart (Section 7). However, this may simply reflect a shared habitat preference. Detailed habitat studies involving mussels and salmonids would help to ascertain this. It is apparent that, in terms of both overall availability and suitability, 0+ salmon are the most important hosts for *M. margaritifera* in both rivers. However, there are a number of other important *M. margaritifera* populations that are entirely trout-dependant.

Although the River Moidart *M. margaritifera* population is recruiting some young mussels, it does not appear to be producing enough at present to maintain present numbers. It is possible that host availability may be one of the factors involved. If this were the case, then enhancement of mussel numbers by the release of hatchery-raised 0+ salmon of local stock infected with mussel glochidia into the River Moidart may be an effective conservation management strategy. However, more research is required before this can be properly ascertained.

Although host availability is likely to be a limiting factor in some *M. margaritifera* populations, the number of fish required will ultimately depend on the number of mussels required to be replaced, and this will vary naturally. Since juvenile salmonid stocks fluctuate considerably from year to year, an appropriate time-series of fish density data (for example, 10–50 years) is required for more thorough investigations of the mussel–host relationship. It is recommended that, in rivers with important mussel populations, native salmonid numbers are monitored, and that long-term stability of the salmonid populations should be considered as one of a number of essential features of favourable condition.

Finally, a number of salmonid fishery-related activities, including the construction of bank reinforcements, croys (deflectors), fishing platforms, pool dredging, footbridges and weed control may adversely affect mussel beds. In particular, the construction of croys and pool dredging are considered to constitute a major threat to the conservation status of *M. margaritifera*. Operations likely to harm mussels and permanently damage their riverbed habitat should not proceed.

10 Summary

During 1996–2002, riverbed surveys of 100 m lengths of river for freshwater pearl mussels were carried out in the River Kerry cSAC and River Moidart cSAC in north-west Scotland. Estimated total numbers of 'visible' mussels ranged from 2340–41900 mussels per 100 m (Kerry) and 1370–17760 mussels per 100 m (Moidart). Observed mussel densities ranged from 1.81–32.23 mussels per m^2 (Kerry) and 0.96–16.60 mussels per m^2 (Moidart).

Overall numbers of mussels at selected sites ranged from 3225–64582 mussels per 100 m (Kerry) and 1679–37864 mussels per 100 m (Moidart). Absolute numbers of juvenile mussels (<65 mm shell length) ranged from 500–15112 juveniles per 100 m (Kerry) and 190–2499 juveniles per100 m (Moidart). Relative numbers of juvenile mussels ranged from 13.8–23.4% (Kerry) and 6.6–11.3% (Moidart).

During 1996–2002, annual releases of *M. margaritifera* glochidia were monitored on the River Kerry and River Moidart cSAC. Estimated daily peak releases ranged from 10.8–441.1 million glochidia. per day (Kerry) and 0.3–31.3 million glochidia per day (Moidart).

Two experiments were carried out in the River South Esk cSAC, north-east Scotland, to test the effects of electric-fishing on *M. margaritifera*. In the first experiment, two areas of mussel bed were marked out as treatment and control sites and the former was electric-fished using standard equipment. The mussels were examined 10 minutes, 24 hours and 35 days after treatment, and their shell valve closure responses, burrowing capabilities and gravidities were recorded as signs of normal functioning. In the second experiment, individual mussels were marked as treatments and controls and the former were electric-fished. These mussels were examined as before. No mortalities occurred and no significant differences in 'functioning' between treatments and controls were observed in either experiment. Since no measureable treatment effect could be demonstrated, it appears that electric-fishing did not adversely affect the short-term survival of *M. margaritifera*.

During 1997–2002, samples of wild juvenile salmonids were taken from the River Kerry cSAC and River Moidart cSAC in north-west Scotland, and examined for *M. margaritifera* glochidia. The observed incidence of infected 0+ salmon ranged from 70–95% (Kerry) and 33–83% (Moidart). Individual loads on 0+ salmon ranged from 0–1260 glochidia per fish (Kerry) and 0–253 glochidia per fish (Moidart). The observed incidence of infected 1++ salmon was 29% (Kerry) and ranged from 20–50% (Moidart). Individual loads on 1++ salmon ranged from 0–750 glochidia per fish (Kerry) and 0–16 glochidia per fish (Moidart). Overall, fewer glochidia were found on juvenile trout (0–46% incidence), although one large specimen (aged 3+) caught in the River Kerry had 3920 glochidia.

Juvenile salmonid density estimates for the River Kerry cSAC (1997–2002) and River Moidart cSAC (1997–2001) were obtained from Wester Ross Fishery Trust (WRFT) and Lochaber & District Fishery Trust (LDFT), respectively. Observed densities of 0+ salmon ranged from 2.0–102.2 fish per 100 m^2 (Kerry) and 10.6–50.8 per 100 m^2 (Moidart). Observed densities of 1++ salmon ranged from 11.5–25.1 per 100 m^2 (Kerry) and 0.0–19.6 per 100 m^2 (Moidart). Observed densities of 0+ trout ranged from 0.0–0.4 per 100 m^2 (Kerry) and 0.0–3.3 per 100 m^2 (Moidart). Observed densities of 1++ trout ranged from 0.0–8.4 per 100 m^2 (Kerry) and 0.0–1.6 per 100 m^2 (Moidart).

The relationship between *M. margaritifera* and its salmonid hosts was investigated. The study was based on quantitative information on mussel abundance, glochidial production/infestation and juvenile salmonid abundance observed in the River Kerry cSAC and River Moidart cSAC in north-west Scotland. Significant positive relationships were observed between mussel density and glochidial production. However, the numbers of glochidia sampled were much smaller than were expected, based on the estimated numbers of gravid female mussels at each site. In general, mussel abundance appeared to be positively correlated with juvenile salmon abundance.

At all sites, the majority of encysted glochidia (75–99%) appeared to be carried by 0+ salmon hosts. Relatively small numbers of encysted glochidia (0–4%) appeared to be carried by trout hosts. Positive relationships were observed between the number of encysted glochidia and the number of unattached

glochidia in the rivers. Estimates of the probability of a glochidium successfully encountering and encysting on a suitable host gill were very low (<0.04%). Estimates of the total number of mussel seed produced at each site ranged from 113–18505 mussels per year. Post-settlement survival rates required in order to maintain present mussel numbers at five sites ranged from 4–30%. However, at two sites on the River Moidart, the numbers of mussel seed produced appeared to be less than required to maintain present mussel numbers. An apparent trend between the surplus of mussel seed produced and the relative abundance of juvenile mussels at each site was observed.

The impacts of river engineering work and physical habitat modification on *M. margaritifera* were reviewed. A number of salmonid fishery-related activities, including construction of bank reinforcements, croys (deflectors), fishing platforms, pool dredging, footbridges and weed control may adversely affect mussel beds to varying degrees. Of these, croys and pool dredging are considered to constitute a major threat to the conservation status of *M. margaritifera*.

Acknowledgements

This project was funded by Life in UK Rivers, managed by Gareth Morgan and Rob Cathcart (English Nature) with the assistance of Phil Boon (Scottish Natural Heritage). Under the terms of the UK Wildlife and Countryside Act 1981, special licences to work with freshwater pearl mussels in the River Dee, River Kerry, River Moidart and River South Esk cSACs were provided by Scottish Natural Heritage. Permission to reproduce extracts from published material in Section 4 was kindly granted by John Wiley & Sons, Chichester. The following people and organisations provided crucial help and advice during this project: Peter Cosgrove (Cairngorms National Park Authority); Robert Fettes, Adrian Hudson, Mark Patterson (Dee Salmon Fishery Trust); Ann Skinner (Environment Agency); Willie McKenzie, John Shearer (LandCatch); Jon Watt (Lochaber & District Fishery Trust); Ian Cameron, Dougie Hunter, Andy McLean, John Richmond, Jim Treasurer, Claire Treasurer (Marine Harvest); Mary Elliot, Martin Gaywood, Mary Gibson, Isla Martin, John Ralston and Iain Sime (Scottish Natural Heritage); Hillary Anderson (Scottish Fisheries Coordination Centre); James Butler, Bob Laughton (Spey Salmon Fishery Trust); Susan Cooksley, Frances Scougall (University of Aberdeen); Peter Cunningham, Karen Starr (Wester Ross Fishery Trust); Shona Marshall (West Sutherland Fishery Trust). We are extremely grateful to everyone who contributed to this work.

12 References

Anon. (1993). *Problems with Sea Trout and Salmon in the Western Highlands*. Atlantic Salmon Trust, Pitlochry, 79 pp.

Anon. (1997). *Freshwater Fisheries Laboratory Biennial Review*. Scottish Office Agriculture, Environment and Fisheries Department. Crown Copyright, 50 pp.

Awakura T (1968). *The ecology of parasitic glochidia of the freshwaterpearl mussel,* Margaritifera laevis *(Haas)*. Scientific Report of the Hokkaido Fish Hatchery. No. 23.

Barrett JC & Grossman GD (1988). Effects of direct current electrofishing on the mottled sculpin. *North American Journal of Fisheries Management* 8, 112–116.

Bauer G (1979). Untersunchungen zur Fortpflanzungsbiologie der flubperlmuschel im Fichtelgebirge. *Archiv fur Hydrobiologie* 65, 152–165.

Bauer G (1986). The status of the freshwater pearl mussel *Margaritifera margaritifera* L. in the south of its European range. *Margaritifera margaritifera* 38, 1–9.

Bauer G (1987a). The parasitic stage of the freshwater pearl mussel (*Margaritifera margaritifera* L.). III. Host relationships. *Archiv für Hydrobiologie* 76, 413–423.

Bauer G (1987b). The parasitic stage of the freshwater pearl mussel (*Margaritifera margaritifera* L.). II. Susceptibility of brown trout. *Archiv für Hydrobiologie* 76, 403–412.

Bauer G (1987c). Reproductive strategy of the freshwater pearl mussel *Margaritifera margaritifera*. *Journal of Animal Ecology* 56, 691–704.

Bauer G (1988). Threats to the freshwater pearl mussel, *Margaritifera margaritifera* in central Europe. *Biological Conservation* 45, 239–253.

Bauer G (1992). Variation in the life span and size of the freshwater pearl mussel. *Journal of Animal Ecology* 61, 425–436.

Bauer G & Vogel C (1987). The parasitic stage of the freshwater pearl mussel (*Margaritifera margaritifera* L.). I. Host response to glochidiosis. *Archiv für Hydrobiologie* 76, 393–402.

Bauer G & Wachtler K (2001). *Ecology and Evolutionary Biology of the Freshwater Mussels Unionoidea*. Springer Verlag, Berlin.

Beasley CR (1996). *The distribution and ecology of the freshwater pearl mussel,* Margaritifera margaritifera *L. 1758, in County Donegal, Ireland and implications for its conservation*. Unpublished PhD Thesis, Queen's University of Belfast.

Bohlin T, Hamrin S, Heggberget TG, Rasmussen G & Satveit SJ (1989). Electrofishing: theory and practice with special emphasis on salmonids. *Hydrobiologia* 173, 9–43.

Buddensiek V (1991). *Untersuchungen zu den aufwuchsbedingungen flubperlmuschel* Margaritifera margaritifera *L. in iher fruhen postparasitared phas*e. Dissertation an der Tierarztlichen Hochschule Hannover.

Buddensiek V, Engel H, Fleischauer-Rossing S & Wachtler K (1993). Studies on the chemistry of interstitial water taken from defined horizons in the fine sediments of bivalve habitats in several northern German lowland waters II: Microhabitats of *Margaritifera margaritifera* L., *Unio crassus* Philipsson and *Unio tumidus* Philipsson. *Archiv für Hydrobiologie* 127, 151–166.

Butler J (1998). *Wester Ross Fisheries Trust (WRFT): Annual Review 1997*. Wester Ross Fisheries Trust, Inverness, 54 pp.

Butler J (1999). *Wester Ross Fisheries Trust (WRFT): Annual Review 1998–1999*. Wester Ross Fisheries Trust, Inverness, 40 pp.

Chesney HCG & Oliver PG (1998). Conservation issues for Margaritiferidae in the British Isles and western Europe. *Journal of Conchology Special Publication* 2, 231–242.

Cosgrove PJ (1998a). *Survey of the River Kerry for the presence of Freshwater Pearl Mussels (*Margaritifera margaritifera *Linn.).* Confidential Report, Scottish Natural Heritage, Edinburgh, 14 pp.

Cosgrove PJ (1998b). *Survey of the River Moidart for the presence of Freshwater Pearl Mussels (*Margaritifera margaritifera *Linn.).* Confidential Report, Scottish Natural Heritage, Edinburgh, 14 pp.

Cosgrove PJ & Young MR (1998).*The status of the Freshwater Pearl Mussel* Margaritifera margaritifera *in Scotland.* Confidential Report, Scottish Natural Heritage, Edinburgh, 118 pp.

Cosgrove PJ & Hastie LC (2001). River management, translocation and conflict resolution in the conservation of threatened freshwater pearl mussels. *Biological Conservation* 99, 183–190.

Cosgrove PJ, Young MR, Hastie LC, Gaywood M & Boon PJ (2000a). The status of the freshwater pearl mussel *Margaritifera margaritifera* (L.) in Scotland. *Aquatic Conservation: Marine and Freshwater Ecosystems* 10, 197–208.

Cosgrove PJ, Young MR & Hastie LC (2000b). Freshwater pearl mussels in peril. *British Wildlife* 11, 340–347.

Cunjak RA & McGladdery SE (1991). The parasite-host relationship of glochidia (Mollusca: Margaritiferidae) on the gills of young-of-the-year Atlantic salmon (*Salmo salar*). *Canadian Journal of Zoology* 69, 353–358.

Elliot JM (1994). *Quantitative Ecology and the Brown Trout.* Oxford University Press, Oxford, 286 pp.

Fustish CA & Millemann RE (1978). Glochidiosis of salmonid fishes II. Comparison of tissue response of Coho and Chinook salmon to experimental infection with *Margaritifera margaritifera* (L.) (Pelecypoda: Margaritiferidae). *Journal of Parasitology* 64, 155–157.

Gibson RJ (1993). The Atlantic salmon in freshwater: spawning, rearing and production. *Reviews in Fish Biology and Fisheries* 3, 39–73.

Habera JW, Strange RJ & Saxton AM (1999). AC electrofishing injury of large brown trout in low-conductivity streams. *North American Journal of Fisheries Management* 19, 120–126.

Hastie LC (1999). *Conservation and ecology of the freshwater pearl mussel,* Margaritifera margaritifera *(L.).* Unpublished PhD thesis, University of Aberdeen, Aberdeen.

Hastie LC & Young MR (2001). Freshwater pearl mussel (*Margaritifera margaritifera*) glochidiosis in wild and farmed salmonid stocks in Scotland. *Hydrobiologia* 445, 109–119.

Hastie LC & Boon PJ (2001). Does electric-fishing harm freshwater pearl mussels? *Aquatic Conservation: Marine and Freshwater Ecosystems* 11, 149–152.

Hastie LC & Cosgrove PJ (2001). The decline of migratory salmonids: a new threat to pearl mussels in Scotland. *Freshwater Forum* 15, 85–96.

Hastie LC & Cosgrove PJ (2002). Intensive searching for mussels in a fast-flowing river: an estimation of sampling bias. *Journal of Conchology* 37, 309–316.

Hastie LC & Young MR (2003). Timing of spawning and glochidial release in Scottish freshwater pearl mussel (*Margaritifera margaritifera*) populations. *Freshwater Biology* (in press).

Hastie LC, Young MR, Boon PJ, Cosgrove PJ & Henninger B (2000a). Sizes, densities and age structures of Scottish *Margaritifera margaritifera* (L.) populations. *Aquatic Conservation: Marine and Freshwater Ecosystems* 10, 229–247.

Hastie LC, Boon PJ & Young MR (2000b). Physical microhabitat requirements of freshwater pearl mussels, *Margaritifera margaritifera* (L.). *Hydrobiologia* 429, 59–71.

Hastie LC, Cooksley SL, Scougall F, Young MR, Boon PJ & Gaywood MJ (2003). Characterisation of freshwater pearl mussel (*Margaritifera margaritifera*) riverine habitat using River Habitat Survey data. *Aquatic Conservation: Marine and Freshwater Ecosystems* 13, 213–224.

Jansen W, Bauer G & Zahner-Meike E (2001). Glochidial mortality in freshwater mussels. In: Bauer G & Wachtler K (eds). *Ecology and Evolutionary Biology of the Freshwater Mussels Unionoidea*. Springer Verlag, Berlin, 185–211.

Jungbluth JH & Keuhnel U (1978). Wasserguete untersuchungen an perlmuschelbaechen. *Verhandlungen der Gesellschaft fuer Oekologie* 7, 317–322.

Kerney MP (1975). European distribution maps of *Potamias elegans*, *Discus ruderatus*, *Eobania vermiculata* and *Margaritifera margaritifera*. *Archiv fur Molluskenkunde* 106, 243–249.

Killeen IJ, Oliver PG & Fowles AP (1998). The loss of a freshwater pearl mussel *M. margaritifera* population in NW Wales. *Journal of Conchology Special Publication* 2, 245–250.

Laughton R, Cosgrove PJ & Hastie LC (2003). *The effect of* Ranunculus *removal on juvenile salmonids and freshwater pearl mussels in the River Spey*. Confidential report, Scottish Natural Heritage, Edinburgh, 30 pp.

Marshall S (1998). *West Sutherland Fisheries Trust (WSFT): Annual Review 1997*. West Sutherland Fisheries Trust, Inverness, 19 pp.

McMichael GA, Fritts AL & Pearsons TN (1998). Electrofishing injury to stream salmonids; injury assessment at the sample, reach and stream scales. *North American Journal of Fisheries Management* 18: 894–904.

Nagel JW (1993). Estimating relative abundance of stream macroinvertebrates with an electrofishing sampler. *Journal of the Tennessee Academy of Science* 68, 77–78.

Nielsen JL (1998). Electrofishing California's endangered fish populations. *Fisheries* 23, 6–12.

Penczak T & Rodriguez G (1990). The use of electrofishing to estimate population densities of freshwater shrimps (Decapoda, Natantia) in a small tropical river, Venezuela. *Archiv für Hydrobiologie* 118, 501–509.

Rabeni CF, Collier KJ, Parkyn SM & Hicks BJ (1997). Evaluating techniques for sampling stream crayfish (*Paranephrops planifrons*). *New Zealand Journal of Marine and Freshwater Research* 31, 693–700.

Ross HCG (1992). The reproductive biology of the freshwater pearl mussel *Margaritifera margaritifera* (L.) in Co. Donegal. *Irish Naturalist Journal* 24, 43–50.

Skinner A, Young MR & Hastie LC (2003). *Ecology of the Freshwater Pearl Mussel.* Conserving Natura 2000 Rivers Ecology Series No. 2, English Nature, Peterborough, 16 pp.

Snyder DE (1995). Impacts of electrofishing on fish. *Fisheries* 20, 26–39.

Tillma J (1996). Review of the principles of electrofishing. *Fisheries* 21, 35–36.

Valovirta I (1990). *Conservation of* Margaritifera margaritifera *in Finland.* Colloquim of the Bern Convention Invertebrates and their Conservation. Council of Europe, Strasbourg, T-PVS 34, 59–63.

Walker AF (1993). Sea trout and salmon stocks in the Western Highlands. In: Anon. (ed). *Problems with Sea Trout and Salmon in the Western Highlands*. Atlantic Salmon Trust, Pitlochry, 6–18.

Webb JH (1993). Prevalence and behaviour of escaped farmed salmon in coastal waters and rivers: western Scotland. In: Anon. (ed). *Problems with Sea Trout and Salmon in the Western Highlands*. Atlantic Salmon Trust, Pitlochry, 19–23.

Watt J (1999). *Lochaber & District Fisheries Trust (LDFT): 2nd Annual Report 1997–1998*. Lochaber & District Fisheries Trust, Inverness, 49 pp.

Young MR (1991). Conserving the freshwater pearl mussel (*Margaritifera margaritifera* L.) in the British Isles and continental Europe. *Aquatic Conservation: Marine and Freshwater Ecosystems* 1, 73–77.

Young MR & Williams JC (1983a). The status and conservation of the freshwater pearl mussel in Great Britain. *Biological Conservation* 25, 35–52.

Young MR & Williams JC (1983b). Redistribution and local recolonisation by the freshwater pearl mussel *Margaritifera margaritifera* (L.). *Journal of Conchology* 31, 225–234.

Young MR & Williams JC (1984a). The reproductive biology of the freshwater pearl mussel *Margaritifera margaritifera* (Linn.) in Scotland. I. Field studies. *Archiv fur Hydrobiologie* 99, 405–422.

Young MR & Williams JC (1984b). The reproductive biology of the freshwater pearl mussel *Margaritifera margaritifera* (Linn.) in Scotland. II. Laboratory studies. *Archiv fur Hydrobiologie* 100, 29–43.

Young MR, Purser GJ & al-Mousawi B (1987). Infection and successful re-infection of brown trout (*Salmo trutta* (L.) with glochidia of *Margaritifera margaritifera* (L.). *American Malacological Bulletin* 5, 125–128.

Young MR, Cosgrove PJ & Hastie LC (2001). The extent of, and causes for, the decline of a highly threatened naiad: *Margaritifera margaritifera*. In: Bauer G & Wachtler K (eds.), *Ecology and Evolutionary Biology of the Freshwater Mussels Unionoidea*. Springer Verlag, Berlin, 337–357.

Young MR, Hastie LC & Cooksley SL (2003). *Monitoring the Freshwater Pearl Mussel,* Margaritifera margaritifera. Conserving Natura 2000 Monitoring Series No. 2, English Nature, Peterborough, 22 pp.

Ziuganov V, Zotin A, Nezlin L & Tretiakov V (1994). *The Freshwater Pearl Mussels and Their Relationships with Salmonid Fish*. VNIRO, Russian Federal Research Institute of Fisheries and Oceanography, Moscow, 104 pp.

Conserving Natura 2000 Rivers

Ecology Series

1 Ecology of the White-clawed Crayfish, *Austropotamobius pallipes*
2 Ecology of the Freshwater Pearl Mussel, *Margaritifera margaritifera*
3 Ecology of the Allis and Twaite Shad, *Alosa alosa* and *A. fallax*
4 Ecology of the Bullhead, *Cottus gobio*
5 Ecology of the River, Brook and Sea Lamprey, *Lampetra fluviatilis*, *L. planeri* and *Petromyzon marinus*
6 Ecology of Desmoulin's Whorl Snail, *Vertigo moulinsiana*
7 Ecology of the Atlantic Salmon, *Salmo salar*
8 Ecology of the Southern Damselfly, *Coenagrion mercuriale*
9 Ecology of the Floating Water-plantain, *Luronium natans*
10 Ecology of the European Otter, *Lutra lutra*
11 Ecology of Watercourses Characterised by *Ranunculion fluitantis* and *Callitricho-Batrachion* Vegetation

Monitoring Series

1 A Monitoring Protocol for the White-clawed Crayfish, *Austropotamobius pallipes*
2 A Monitoring Protocol for the Freshwater Pearl Mussel, *Margaritifera margaritifera*
3 A Monitoring Protocol for the Allis and Twaite Shad, *Alosa alosa* and *A. fallax*
4 A Monitoring Protocol for the Bullhead, *Cottus gobio*
5 A Monitoring Protocol for the River, Brook and Sea Lamprey, *Lampetra fluviatilis*, *L. planeri* and *Petromyzon marinus*
6 A Monitoring Protocol for Desmoulin's Whorl Snail, *Vertigo moulinsiana*
7 A Monitoring Protocol for the Atlantic Salmon, *Salmo salar*
8 A Monitoring Protocol for the Southern Damselfly, *Coenagrion mercuriale*
9 A Monitoring Protocol for the Floating Water-plantain, *Luronium natans*
10 A Monitoring Protocol for the European Otter, *Lutra lutra*
11 A Monitoring Protocol for Watercourses Characterised by *Ranunculion fluitantis* and *Callitricho-Batrachion* Vegetation

These publications can be obtained from:

The Enquiry Service
English Nature
Northminster House
Peterborough
PE1 1UA
Email: enquiries@english-nature.org.uk
Tel: +44 (0) 1733 455100
Fax: +44 (0) 1733 455103

They can also be downloaded from the project website: www.riverlife.org.uk

The Life in UK Rivers project was established to develop methods for conserving the wildlife and habitats of rivers within the Natura 2000 network of protected European sites.

Set up by the UK statutory conservation bodies and the European Commission's LIFE Nature programme, the project has sought to identify the ecological requirements of key plants and animals supported by river Special Areas of Conservation.

In addition, monitoring techniques and conservation strategies have been developed as practical tools for assessing and maintaining these internationally important species and habitats.

The freshwater pearl mussel (*Margaritifera margaritifera*) is one of the most endangered invertebrates in Europe. In the UK, viable populations are only found in Scotland, and these sites hold almost half the world's remaining populations with active recruitment.

Because of the pearl mussel's unique relationship with juvenile salmonids – its larvae can only complete their development on the fry and parr of salmon or brown trout – any management efforts designed to conserve the mussel must also consider the needs of these host fish.

This publication reports on a project to study the relationship between the freshwater pearl mussel and juvenile salmonids, and the management implications for conservation of these species and their riverine habitat. An accompanying report examines captive breeding techniques for the pearl mussel.

Information on Conserving Natura 2000 Rivers and the Life in UK Rivers project can be found at **www.riverlife.org.uk**

This document was produced with the support of the European Commission's LIFE Nature Programme and published by the Life in UK Rivers project - a joint venture involving English Nature, the Countryside Council for Wales, the Environment Agency, the Scottish Environment Protection Agency, Scottish Natural Heritage and the Scotland and Northern Ireland Forum for Environmental Research.