Developing Tools for the Management of Freshwater Crayfish

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The candidate confirms that the work submitted is her own, except where work which has formed part of jointly-authored publications has been included. The contribution of the candidate and the other authors to this work has been explicitly indicated below. The candidate confirms that appropriate credit has been given within the thesis where reference has been made to the work of others.

Chapter 2 is from the following jointly-authored paper:

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Contributions of each author:

Stephanie Peay carried out a literature review and the risk assessments for *Pacifastacus leniusculus* and *Procambarus clarkii* and wrote most of the paper.

David M. Holdich carried out risk assessment for *Orconectes limosus* contributed to the introduction and reviewed and edited.

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Stephanie Peay designed the study, carried out the crayfish survey, helped with some of the fisheries survey, analysed the data and wrote the paper.

Neil Guthrie was the project manager for the Environment Agency, which funded part of the work and he helped to edit the paper.

Jack Spees carried out the fisheries survey work in 2008 and provided comparative data on the substrates and helped to edit the paper.

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http://www.bcsf.org.uk/sites/default/files/files/Rees%20et%20al_%202011%20(2010%20crayfish%20conference,%20high%20resolution%20version%20available%20on%20request_).pdf

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Abstract

The introduction of non-indigenous crayfish into Europe is causing the loss of indigenous crayfish, due to transmission of crayfish plague and competition. Other factors are reductions of habitat quality and in some areas harvesting. This study deals with issues facing environmental agencies and other resource managers about how manage crayfish; from prevention of further introductions, to eradication where feasible, or control if it is cost-effective, or where it is not, then applying measures to mitigate the effects of invasion by finding or establishing isolated areas for indigenous crayfish, i.e. ark sites. It provides a range of decision-making tools for management. The study includes a literature-based risk assessment for non-indigenous crayfish in Great Britain. It presents the first evidence of the negative impact of signal crayfish *Pacifastacus leniusculus* on salmonid fish in a headwater stream. The study shows how the technique of biocide treatment against signal crayfish has developed, the outcomes and the lessons learned from the projects and factors that will contribute to other successful eradication treatments in future. Another new potential method for eradication or control is electric shock treatment, which was field-tested in this study. As an aid to assessing the feasibility of eradication or control, a simple cost-model was developed using the potential impact on salmonid fish and a re-stocking cost as a surrogate for environmental impact of crayfish invading a catchment. This was used to compare the costs of eradication or control and showed the benefit of early eradication and the unsustainably high cost of control by trapping. As signal crayfish are already widespread in England and Wales, risk-based selection criteria were developed to help identify potential ark sites for white-clawed crayfish. In addition, a decision-making tool has been prepared to help conservation managers understand the issue and develop conservation action plans at catchment scale.

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White-clawed crayfish Austropotamobius pallipes (left), an indigenous European species and signal crayfish Pacifastacus leniusculus (right), a non-indigenous North American species. The signal crayfish has been introduced into many waterbodies in Great Britain and is replacing white-clawed crayfish in much of its range.

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Graphic by Nathan Ward

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Part 1

Invasion ecology: risks and impacts of non-indigenous crayfish

Chapter 1
Introduction: biological invasion and management tools

1.1 Biological invasions

Human pressure on ecosystems is leading to global decline in biodiversity (Butchart et al., 2010). Biological invasions are among the major global threats to biodiversity (e.g. Vermeij, 1996, Wilcove et al., 1998, Mack et al., 2000a), together with habitat loss or reduction of quality and over-exploitation. Biological invasions, over-exploitation of natural resources, pollution and habitat degradation or loss all lead to losses of indigenous species and there is interaction between the factors.

Cascade effects can occur in which the impacts of one invasive species then facilitates invasion by others, described by some authors as “invasional meltdown” (Moyle and Light, 1996, Simberloff and Von Holle, 1999, Light and Marchetti, 2007). Disturbance in ecosystems often opens up opportunities for non-indigenous species, e.g. the development of ‘weed’ communities. An example of this is the invasion by a series of invaders into multiple structural layers of forest (Asner et al., 2008). The greatest impacts of invasive plants are often the changes that can occur in soil nutrient processes (Ehrenfeld, 2003, Vila et al., 2011). Gonzalez et al. (2008) developed models to show how an invasive species could act as an ecosystem engineer, modifying habitat to an extent that it facilitated its success at the expense of indigenous species. An example of an ecosystem in invasional meltdown would be the East African Lake Naivasha, where red swamp crayfish *Procambarus clarkii* was introduced, damaged indigenous macrophytes and reached very large populations in the extensive mats of the invasive water hyacinth *Eichhornia crassipes.* Red swamp crayfish became an important component of the diet of another introduced invasive species, American bigmouth bass *Micropterus salmoides*. When the water hyacinth became a severe problem, limiting access to the shore and clogging dams, bio-control was implemented by the introduction of the non-indigenous beetle *Cyrtobagus eichhorniae*. The enormous rafts of water hyacinth disappeared from the edges of the *Cyperus papyrus* reedswamp and the population of *Procambarus* subsequently declined in this unstable, invasives-dominated system (Foster and Harper, 2004).

The rate of biological invasions has been massively accelerated by deliberate and accidental introductions (Millennium Ecosystem Assessment, 2005) and by the accelerated pace of global transport, especially shipping (Gollasch, 2007), which has brought numerous invaders between oceans, e.g. most of more than 660 non-indigenous marine species in the Mediterranean Sea (Galil, 2012). As the number of species being moved between regions and continents increases, so does the number that become successful invaders. Global trade in crops, together with horticulture is the primary source of invasive plants (Bradley et al., 2012). Williamson and Fitter (1996) described a three-fold “rule of tens”, suggesting that about 10% of species overcome each successive barrier to invasion: arrival and escape, establishment, invasion with impacts. Their work was based on studies of introduced plants, although it had wider application. The authors found exceptions, however, including birds in Hawaii, crop plants and some mammal populations on islands, and Holdich (1999) noted that introduced freshwater crayfish in Europe are a further exception.

Freshwater systems appear to be particularly vulnerable to biological invasions. Approximately 15% of known animal species live in freshwater systems (more than 70,000 species (Brönmark and Hansson, 2002), but a large proportion of these is under threat. The extinction rate of freshwater species was estimated by Ricciardi and Rasmussen (1999) to be comparable with that for tropical rain forests. Whilst 14-18% of terrestrial vertebrates in the United States were classed as vulnerable, imperilled or extinct, among amphibians and fish the proportion was 35-37% and among freshwater crayfish and unionid mussels the proportions were 65% and 67% respectively (Richter et al., 1997). For the imperilled aquatic fauna of the United States the greatest threats are altered sediment loads, altered flow due to impoundments and non-indigenous species (Richter et al., 1997). Increasing future threats for freshwater fauna, in developed countries and in developing countries, were considered by Brönmark and Hansson (2002) to be global warming and associated changes in climate, and above all, the invasion by non-indigenous species. Strayer (2006) estimated around 10,000 species of freshwater invertebrates were already threatened.

The susceptibility of freshwater systems to invasion appears to arise from five main factors: 1. the high dispersal ability of freshwater species; 2. the human assisted movement of species via ships hulls and ballast water and the linking of catchments and seas by the construction of canals; 3. the widespread introduction of fish, crayfish and some molluscs for aquaculture or recreational fisheries; 4. the ease of transmission of pathogens and parasites, and 5. the sensitivity to variations in water temperature, which due to global warming, facilitates invasion of species to higher latitudes.

Examples of pathways for introductions in freshwater systems are shown in Figure 1.1. Globally, movement of invertebrates between continents in shipping is a major issue, for example the introduction of amphipods (Berezina, 2007) and the zebra mussel *Dreissena polymorpha* (Johnson and Padilla, 1996, Ricciardi and MacIsaac, 2000) from the Ponto-Caspian region to the Baltic and North America. The zebra mussel is one of the aquatic invaders readily moved by shipping and boating and it has had major impacts on indigenous unionid species. Other pathways include aquaculture, food for consumption, angling bait and aquarium discards, all of which are pathways for the introduction of invasive crayfish.

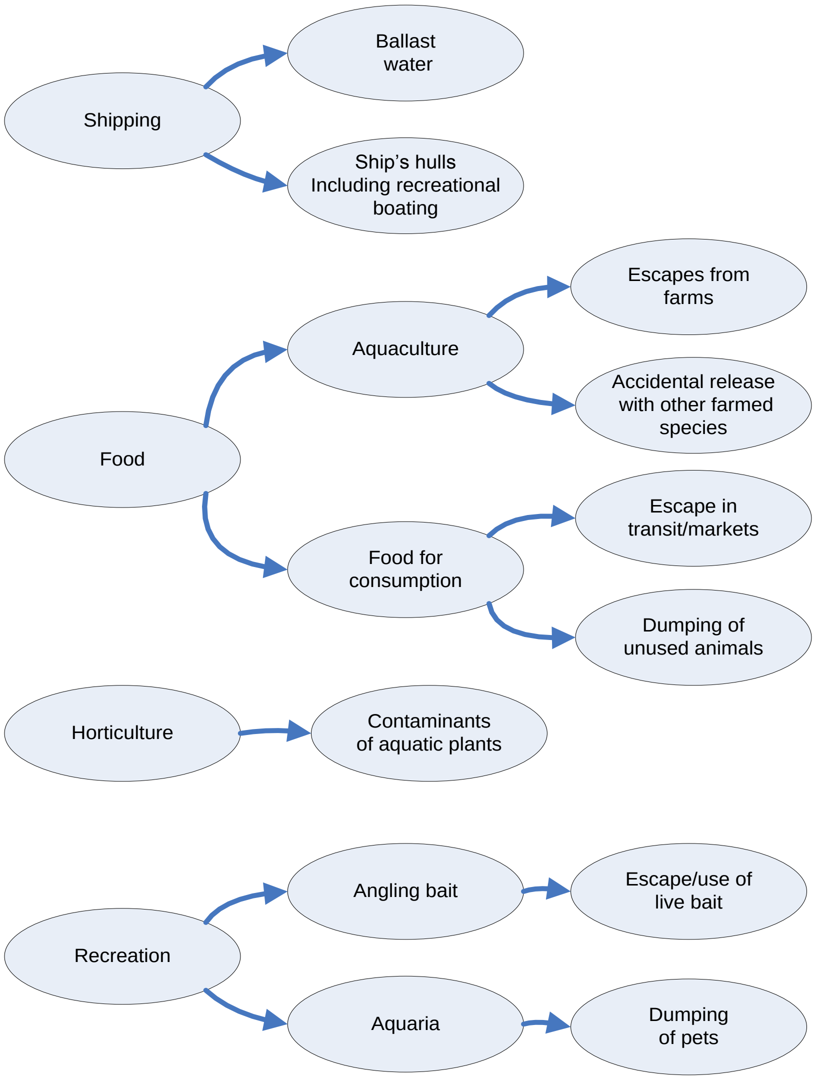

Figure 1.1 Examples of pathways for introductions into freshwater ecosystems

Many of the invasive species in freshwater share similar biological characteristics. Statzner et al. (2008) reviewed the biological traits of freshwater macroinvertebrates and identified the traits that were most likely to be found in species that became invasive when introduced outside their natural range: more frequent reproduction and high abundance; significantly more ovovivipary; large size; flexibility in utilisation of food resources and more efficient exploitation of food, and dominance in their communities. These traits facilitate the rate of invasion and the competitiveness of invasive species. Species that act as ecosystem engineers are often particularly effective as invaders and have the greatest impacts on the aquatic ecosystems they colonize because they alter habitats to conditions in which they can thrive.

One example of an ecosystem engineer is the zebra mussel, which as well as carpeting and outcompeting other benthic species, provides habitat for other invasive species (Ricciardi et al., 1998). It has become a food source for non-indigenous fish, round goby *Neogobius malanostomus* and white bass *Morone chrysops* (French, 1993). Interactions of invaders in ecosystems can be complex, with impacts cascading to species at different trophic levels. For example, filamentous algae can modify the effects of invasive zebra mussel (Ward and Ricciardi, 2010) and abundance of algae *Cladophora* spp. is modified by nutrient status, especially eutrophication and by the turbidity of water. Turbidity can be increased by the bioturbation of bottom-feeding fish, such as goldfish *Carassius auratus*, which also graze on macrophytes (Richardson et al., 1995) and both grazing and turbidity affects the relative abundance of *Cladophora* and other algae.

Invasive species may also bring with them novel parasites, which go on to affect indigenous species in the new community. An example of the role of invasive species as carriers of new parasites is, again, the zebra mussel, which is the first intermediate host for a parasitic trematode *Bucephalus polymorphus* (Minguez et al., 2012), whose final host is the pike perch *Stizosledion lucioperca*, a species widely introduced for sport fishing. With the presence of non-indigenous zebra mussel, the parasite spreads to indigenous and non-indigenous cyprinid fish (Kvach and Mierzejewska, 2011) leading to increased mortality in the fish.

Another example of the transfer of parasites or pathogens by a non-indigenous aquatic species is the crayfish plague *Aphanomyces astaci*, which is transmitted by most of the North America species of crayfish that have been introduced into Europe and which, in all European species of crayfish, has been found to be rapidly lethal, leading to loss of whole populations of indigenous crayfish (Diéguez-Uribeondo et al., 2006, Dieguez-Uribeondo, 2009).

Predatory invasive species are often more successful than indigenous predators, because susceptible indigenous prey species generally have fewer defence mechanisms and greater naïveté toward new predators. Naïveté of prey is a factor of importance in freshwater ecosystems where predatory fish have been introduced (Cox and Lima, 2006). The effect of naïve prey is also evident in the reduced defence response in amphibians to invasive red swamp crayfish *Procambarus clarkii* (Almeida et al., 2011, Gomez-Mestre and Diaz-Paniagua, 2011), as the wetlands used by the amphibians are not usually utilised by indigenous crayfish.

1.2. Biological invasions by crayfish

Non-indigenous crayfish are particularly important as aquatic invaders. The species introduced have been used in aquaculture and they have attributes that predispose them to be effective invaders: high fecundity, relatively rapid growth compared to indigenous crayfish, aggression, able to live in a wide range of conditions, omnivorous and with the ability to switch diet according to availability of food resources. Together, these traits make them ideally suited as aquatic invaders. Whilst indigenous crayfish species have some of these traits in common, the invasive species are those that can readily out-compete the indigenous species.

Crayfish are omnivorous and hence invasive crayfish can affect the invaded ecosystem through predation, herbivory and detritivory (Figure 1.2). The omnivorous diet of crayfish and the effects of crayfish on aquatic communities have been reported by many authors.

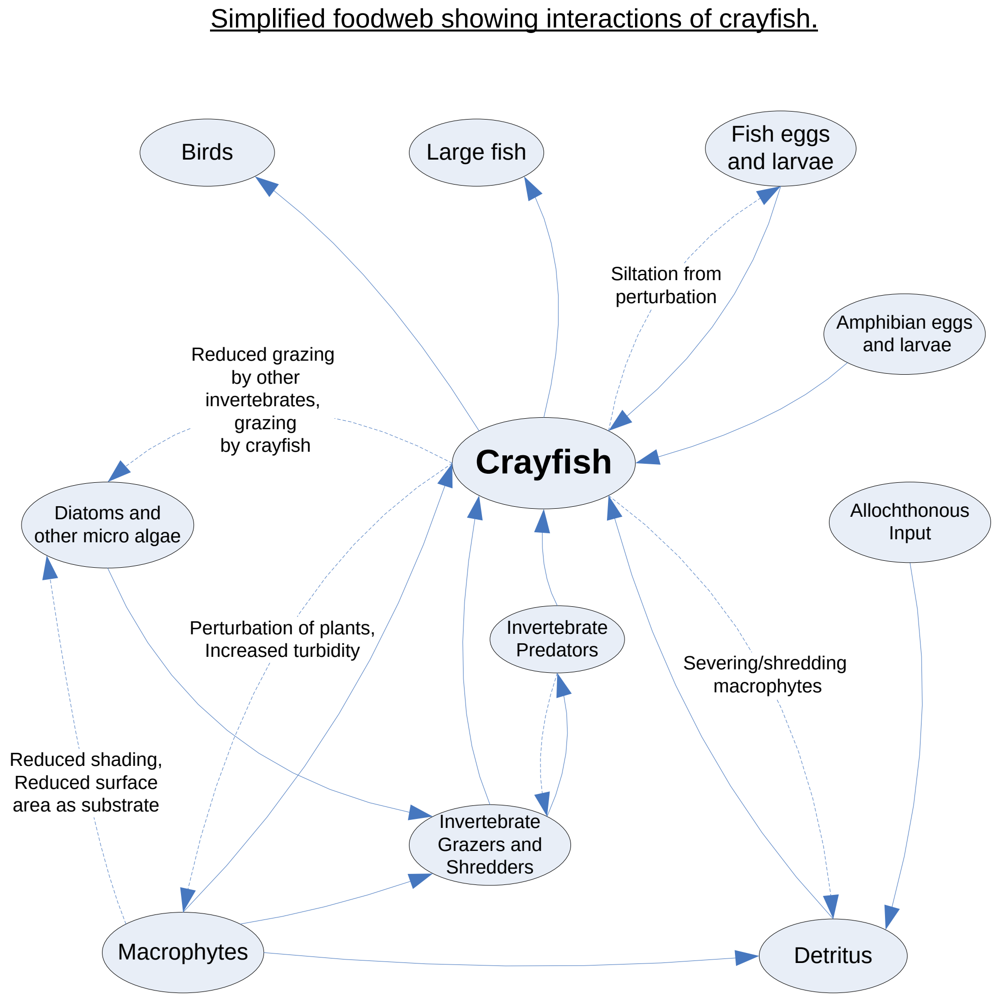

Figure 1.2 A simplified foodweb showing the key role of freshwater crayfish

Crayfish often have a negative effect on both the biomass and species-richness of invertebrates, due to direct predation, or indirect effects mediated via changes in macrophytes. For example, invading rusty crayfish *Orconectes rusticus* consume large quantities of detritus and benthic macroinvertebrates (Rosenthal et al., 2006, Bobeldyk and Lamberti, 2008). Similarly, Stenroth and Nystrom (2003) found that signal crayfish reduced aquatic plants and the diversity and abundance of invertebrates in experimental enclosures. Usio et al. (2009) found similar effects in Japan when signal crayfish invaded wetlands and noted the shredding of plants by signal crayfish, as well as consumption. Nyström and Strand (1996) also recorded the impact of signal crayfish on macrophytes in a Swedish stream. Virile crayfish *Orconectes virilis* also reduce macrophytes (Chambers et al., 1990) and even white-clawed crayfish *Austropotamobius pallipes*, not normally an invasive species, was found to be capable of reducing the abundance of aquatic macrophytes, especially *Chara* species (Matthews et al., 1993). Red swamp crayfish is highly herbivorous (Feminella and Resh, 1989), shredding and consuming macrophytes and pond macroinvertebrates alike (Gherardi and Acquistapace, 2007).

In a river in Scotland, Crawford et al. (2006) found approximately 60% reduction in invertebrate density in areas invaded by signal crayfish. Most of the slower-moving taxa in the river showed reduced numbers of species and/or abundance in the presence of crayfish, including Plecoptera, Chironomidae, Diptera and Hirudinea, Tricladida and Hydracarina. Snails are particularly favoured by crayfish as prey e.g. Nyström and Perez (1998), because the shells are a source of calcium, which is required by crayfish after loss during moulting.

Although the impact on invertebrates is well known, less is known about the impact of non-indigenous crayfish on fish. Some fish predate crayfish (see section 1.5), but crayfish can also predate fish eggs and juvenile fish, as reviewed recently by Reynolds (2011), who summarized the main interactions between fish and crayfish as predation (by both fish and crayfish), competition for food and shelter (by both), modification of habitat (by crayfish), inhibition of foraging behaviour (of crayfish by fish) and effects on community resources and biodiversity (by fish and crayfish). Chapter 3 investigates impact of invasive signal crayfish on fish.

The trophic impact of invaders can cascade through a community. Red swamp crayfish has caused major loss of macrophytes in wetlands in Mediterranean countries, due to grazing, shredding and bioturbation by and consumption of benthic invertebrates (Angeler et al., 2001, Geiger et al., 2005). This can trigger an ecological cascade, leading to a switch from one type of aquatic assemblage to another. Phase change of lakes has occurred, from clear, macrophyte-dominated communities to turbid waters with few macrophytes. Rodríguez et al. (2003) and Rodríguez et al. (2005) described how invasion by red swamp crayfish in Lake Chozas reduced macrophyte cover by 97%, to less than 10% cover within two to three years from introduction. This caused a switch to turbid conditions, dominated by the planktonic blue-green alga *Microcystis*. Crayfish are not very efficient grazers of micro-algae, compared to other invertebrates, but they have large potential for indirect effects by changing macrophyte abundance and by suspension of sediments from burrowing activity. Increases in microalgae can occur as indirect effects due to grazing of macrophytes and predation of herbivorous invertebrates (e.g. Creed, 1994, Charlebois and Lamberti, 1996, Dorn and Wojdak, 2004).

Benthic invertebrates are important in nutrient cycling in freshwater ecosystems, facilitated by burrowing, sediment re-working and processing through faecal production (Covich et al., 1999). In addition to making nutrients available to primary producers, the activity creates micro-habitats and physical and chemical gradients in the substrate, which diversify conditions for the fauna. Crayfish are species that do this in large degree. Correia and Ferreira (1995) recorded burrowing activity of *Procambarus clarkii* in Portugal. With high density of crayfish (burrow densities up to 6 burrows m^{-2}) and deep burrows (typically 0.28-0.58 m, but some >4 m), this represented a large volume of excavated soil per unit area. Crayfish in lotic systems also have a substantial role in increasing sediment transport and its subsequent deposition (Statzner et al., 2001, Statzner and Sagnes, 2008, Johnson et al., 2010).

Crayfish act as ecological engineers, able to affect turbidity and sediment transport; shred and consume aquatic macrophytes, and accelerate nutrient cycling by processing of detritus. They are capable of modifying the ecosystem they have invaded in such a way that conditions can become less suitable for various indigenous species, whereas the invasive species can cope with the modified conditions. This confers an additional competitive advantage on the invasive species. The types of impacts of non-indigenous crayfish are summarized in Figure 1.3.

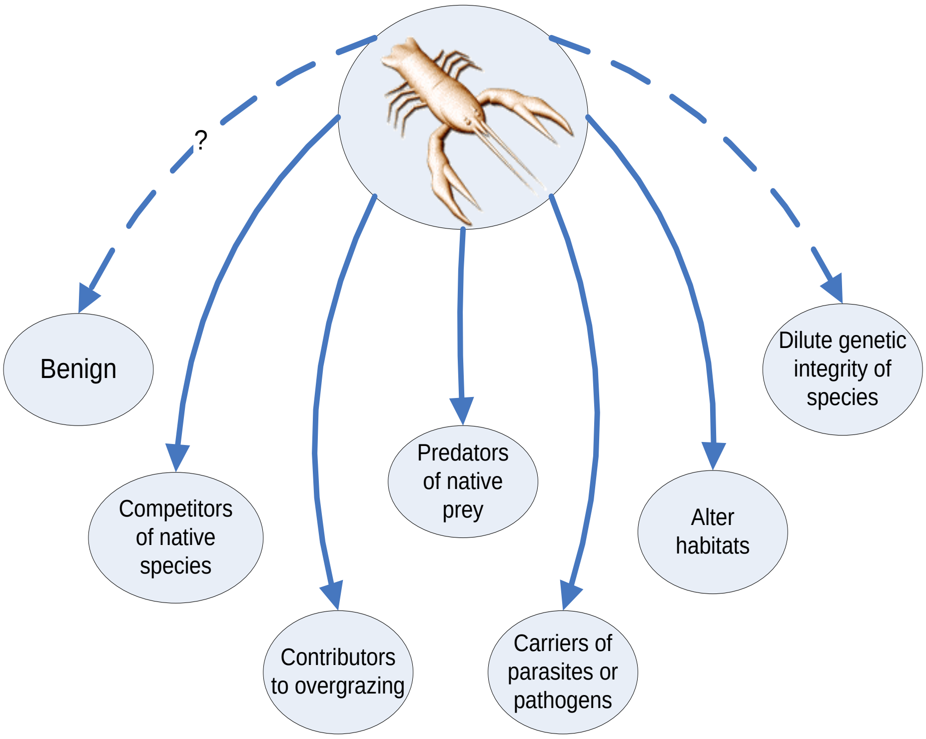

Figure 1.3 The impacts of invasive non-indigenous crayfish

1.3 Impact of invasive crayfish on indigenous crayfish

Invasive crayfish compete with indigenous crayfish for food and for habitat, and often exclude them. For example, even without the effect of crayfish plague, the non-indigenous signal crayfish is able to displace white-clawed crayfish (Peay and Rogers, 1999) and noble crayfish *Astacus astacus* (Westman et al., 2002). One of the factors that may enable an introduced species of crayfish to replace an indigenous one is greater metabolic efficiency. For example, rusty crayfish consumed twice as much food as similarly sized virile crayfish due to a higher metabolic rate (Hamr, 2002). The latter is out-competed by rusty crayfish. Nyström (2005) found that signal crayfish grew faster than noble crayfish in the same habitat and it had twice the width of trophic niche compared to noble crayfish (Olsson et al., 2009). Furthermore, as populations of both species in the same stream had similar diets, the implication from the wider trophic niche is that signal crayfish showed greater plasticity in diet, switching to the most favourable diet in the local environment. Evidence of similar plasticity in diet was shown in indigenous crayfish species in New Zealand (Parkyn et al., 2001). Guan and Wiles (1996) determined that the growth rate of signal crayfish in a lowland river was greater than the rates had been for white-clawed crayfish in the same river (Pratten, 1980) before the population of indigenous crayfish was extirpated by crayfish plague, carried by the invading non-indigenous crayfish. Guan and Wiles (1996) also found the estimated biomass for signal crayfish was greater (it averaged 60.7 g m^{-2} for crayfish > 30 mm carapace length only and hence was an under-estimate of the total biomass) and was similar to the average biomass of fish in the river (at 43.8 g m^{-2}).

Better foraging ability can also help a non-indigenous crayfish outcompete an indigenous congener, for example whilst virile crayfish in its natural range consumes fish eggs, two invasive crayfish species have been found to be better predators of fish eggs, rusty crayfish (Morse et al., 2013) and the Northern Clearwater crayfish *Orconectes propinquus* (Mason and Evans, 2011). In another example, non-indigenous signal crayfish consumed more macrophytes and invertebrates than indigenous noble crayfish in a study in Sweden (Nyström et al., 1999)

Greater size of an invasive species also confers an advantage, because larger crayfish are better able to compete for shelter or defend against predatory fish. For example, Mather and Stein (1993) found that greater body size benefited rusty crayfish over an indigenous congener, Sanborn's crayfish *Orconectes sanborni*, as the larger rusty crayfish were more successful at avoiding predation. Avoidance of predation by displacement of other crayfish from shelter was found to be an important factor in laboratory and field studies of assemblages of *Orconectes rusticus*, *O. virilis* and *O. propinquus* (Butler and Stein, 1988, Garvey et al., 1994, Davis and Huber, 2007). The more aggressive rusty crayfish commandeered the most favourable cobble habitat for shelter by day, when fish were active, with the other two species spending more time on macrophyte-dominated habitats that offered less shelter (Hill and Lodge, 1994).

Aggressive interaction is usually an important factor in the replacement of indigenous crayfish by invaders, as mentioned regarding the rusty crayfish above. Invasive crayfish are also more aggressive in Europe; indeed, Hudina et al. (2011a) found that there was competition between co-occuring invaders signal crayfish and spiny cheek crayfish *Orconectes limosus*, with signal crayfish being consistently more successful in staged encounters and when in direct competition for resources. Experimental studies indicated the dominance of signal crayfish in competition with white-clawed crayfish (Holdich and Domaniewski, 1995, Gherardi and Cioni, 2004); noble crayfish (Söderbäck, 1994, Söderbäck, 1995) and stone crayfish *Austropotamobius torrentium* (Vorburger and Ribi, 1999). However, this competitive effect only occurs where the invading species is not carrying crayfish plague, which is lethal to European species of crayfish.

Crayfish plague (*Aphanomyces astaci* Schikora, 1903) is an oomycete pathogen of crayfish. It is a fungus-like organism, which grows as aseptated hyphae and produces asexual, biflagellate primary zoospores to transmit infection (Diéguez-Uribeondo *et al.,* 2006). On contact with a crayfish, a zoospore germinates, extending a new hypha into the cuticle of the crayfish. In North American species, an immune response is triggered, which produces melanisation around the invading hyphae, limiting the pathogen to a minor infection in otherwise healthy crayfish. Nonetheless, some growth of can occur in the cuticle of the crayfish, including production of zoospores, making the North American chronic carriers of crayfish plague. Infection with *Aphanomyces astaci* does increase the vulnerability of signal crayfish to stress from environmental factors or other pathogens, and it is possible for mortality to occur (Söderhall and Cerenius, 1999), although this is uncommon.

In European crayfish, germinating zoospores penetrate deeply into the crayfish, the hyphae grow rapidly throughout the body and death occurs within a few days, accompanied by massive sporulation. Hence, whenever populations of North American crayfish have a proportion of infected individuals, there is a high probability that contact with any population of European crayfish will trigger a lethal epidemic. The susceptibility of European crayfish to this recent pathogen means that entire populations in large waterbodies have been extirpated by crayfish plague across Europe. Although the most common route of infection is directly from an infective carrier, if a zoospore does not come into contact with a crayfish within a few hours it can form a secondary cyst, enabling it to survive for a few days as a ‘dormant’ cyst, before it becomes a zoospore again, a process that can be repeated up to three times before the cyst is unviable. This increases the infective period for *Aphanomyces astaci* to several days in water and several weeks in mud (Longshaw, 2011) and means that the disease can be transmitted in water with stocked fish. Matthews and Reynolds (1992) identified crayfish plague in Ireland, in the absence of non-indigenous crayfish, and suggested the likely route was on wet angling gear brought from elsewhere in Europe.

In Europe, transmission of disease by non-indigenous crayfish is the dominant factor in the loss of indigenous crayfish (Söderhall and Cerenius, 1999). In the presence of North American crayfish most of the losses have been due to crayfish plague, rather than competition (Holdich, 1999, Holdich et al., 2002), although there are some exceptions, where populations of signal crayfish have been free from the disease. For example in a Finnish lake there was only very slow replacement of noble crayfish by the similar-sized non-indigenous signal crayfish (Westman and Pursianen, 1973, Westman and Savolainen, 1995). In England, a mixed population of signal crayfish and white-clawed crayfish in a lake was described by Holdich and Domaniewski (1995). In the absence of crayfish plague, the progressive replacement of white-clawed crayfish by signal crayfish in a river in Yorkshire was estimated to take 4 - 7 years (Peay and Rogers, 1999). More recent advances in the ability to detect crayfish plague in crayfish samples have shown non-indigenous crayfish populations with varying proportions of the population carrying crayfish plague. For example, in France, about half the populations of invasive crayfish sampled carried plague, with about 20% of individuals sampled, but the incidence of detectable infection ranged from 0-80% (Filipova et al., 2013) and apparently plague-free populations of spiny cheek crayfish have also been found (Schrimpf et al., 2013b). Yet, as new species of North American crayfish appear in Europe, they have been found to carry crayfish plague, most recently the calico crayfish *Orconectes immunis* (Schrimpf et al., 2013a). All the North American species currently established in Europe have been confirmed as capable of carrying crayfish plague.

Recent work on the genetics of *Aphanomyces astaci* has revealed several strains present in Europe, with differing virulence (Makkonen et al., 2012a). There are the first indications of partial resistance to at least one strain of the pathogen in populations of noble crayfish (Viljamaa-Dirks et al., 2011, Makkonen et al., 2012b). Recently, chronic persistence of crayfish plague has been reported in populations of narrow-clawed crayfish *Astacus leptodactylus* in the Danube delta (Parvulescu et al., 2012) and in Turkey (Kokko et al., 2012). Whilst encouraging for the future in some respects, this is an indication of the increasing impact of non-indigenous crayfish, as outbreaks of crayfish plague are now occurring in areas of eastern Europe hitherto unaffected by the disease. Furthermore, in noble crayfish, the first confirmation that apparently healthy noble crayfish in Finland are carrying one of the strains (Jussila et al., 2011) potentially complicates the management regarding re-stocking.

Another mechanism for replacement of indigenous species by an invader is reproductive interference or hybridization, which has not been observed in Europe, but does occur in North America when species are introduced outside their natural range, for example the hybrid between rusty crayfish and Northern Clearwater crayfish showed hybrid vigor compared to its indigenous parent (Capelli and Capelli, 1980, Perry et al., 2001).

Non-indigenous crayfish can cause substantial changes in aquatic ecosystems and replace indigenous species. They have biological traits that confer advantages in new aquatic systems and they can benefit from the absence of some of the predators, parasites and diseases in the natural range; whilst their diet-switching ability allows them to take advantage of new prey. But even if the process is understood, there is the issue of what, if anything can or should be done about it. From being abundant in streams and lakes throughout Europe, populations of indigenous crayfish in Europe are plummeting, whilst non-indigenous crayfish are rapidly extending their range. Management options are needed to prevent further invasions, eradicate or, failing that, control populations of non-indigenous crayfish. Or, where none of those options is feasible, other alternatives are needed if populations of indigenous crayfish are to have a longterm future, in at least some parts of their natural range.

1.4 Assessing risk of invasive crayfish

As described above, biological invasions are becoming increasingly frequent globally and due to their omnivory and adaptability in freshwater ecosystems (section 1.2), crayfish species present risks when introduced into new areas beyond their natural range. However, (as discussed in 1.3 above) not all crayfish are equally competitive and there is a need to investigate the potential risks of introduction of new crayfish species, by the pathways mentioned (in 1.1 above) and the potential for spread and impact from crayfish once they have established. The issue of the risk of invasive crayfish is investigated in this study (Chapter 2).

1.5 Eradication and control of invasive crayfish

Potential methods of eradication or control of crayfish have been considered by various authors (Holdich et al., 1999, Peay, 2001, Freeman et al., 2010, Gherardi et al., 2011). Methods have been categorised according to broad types: mechanical removal of crayfish, physical methods, chemical and biological and an overview of these methods is given in this section.

1.5.1 Mechanical removal

These methods include trapping and manual removal.

Trapping of crayfish is used routinely as a harvesting method around the world. Most studies have been concerned with acceptable harvest rates, but there are some examples of use of trapping to try to control invasive crayfish. No cases have been reported where trapping has eradicated crayfish of any species and it is generally considered as a control measure, although there have been very few studies where any mitigation of impact of an invasive crayfish species has been found. Trapping is highly size selective, catching predominantly the larger size classes (e.g. Brown and Brewis, 1978, Abrahamsson, 1981, Fjalling, 1995) and there is a risk that adequate control will not be achieved if the minimum size for reproduction is below the minimum size that can be caught in traps. Signal crayfish are capable of reproducing at sizes below the effective range of traps, e.g. crayfish of 25 mm carapace length (CL) have been found with eggs in England (Peay, unpublished), whereas traps catch proportionately few crayfish below 30 mm CL and some types seldom catch crayfish below 40 mm CL (Peay, 2004).

There are various examples where trapping has reduced the catch of large crayfish over several years. In two sites in Bavaria (Keller 1999a, 1999b) very intensive trapping led to substantial reductions in large-sized noble crayfish (indigenous there), but the total biomass harvested increased rather than decreased, due to a large increase in the number of smaller crayfish caught. A similar effect was seen in Finland, when two harvesting regimes with different minimum size limits were monitored for 7 years, compared with a no trapping zone. Marketable catches (>50 mm CL) declined markedly, but the proportion and the overall Catch Per Unit Effort (CPUE) of small-sized crayfish increased (Tulonen et al., 2008). Frutiger et al. (1999) tried to control red swamp crayfish by trapping in Switzerland, where the climate is suboptimal for the species. They found that intensive trapping reduced the catch (from 4.1 to 1.3 crayfish trap night^{-1}), but it was rapidly replaced by the rapid recruitment of this species. In England, a decade of continuous trapping of signal crayfish at the River Stour (Wright and Williams, 2000, Wright, 2009) reduced seasonal catch per unit effort, but the invasion continued beyond the trapped area. The results were similar on the River Clyde (Reeve, 2004, Sinclair, 2009). In the River Bachawy in Wales, which was trapped intensively for at least seven years large numbers of crayfish were removed, with little effect on catch per unit effort (Dyson, 2012). By contrast, trapping on the River Lark (West, 2010) has produced a trend of decreasing catch in a stretch of river within 10 years and is reported to have reduced angling nuisance and bank erosion.

A rare case of a trapping control programme that has had a measurable benefit is the case of Sparkling Lake, Wisconsin, a large (64 ha) lake, where rusty crayfish had invaded the lake, which had two indigenous congeners. Two of the fish species present, smallmouth bass *Micropterus dolomieu* and rock bass *Ambloplites rupestris* feed on indigenous crayfish species in Wisconsin lakes, but catches of the fish by anglers had reduced the predation pressure. A substantial trapping programme (> 14,000 trap nights year^{-1}) and a reduction in the take of fish by anglers led to a substantial decrease in the catch of crayfish over a period of five years; trap catches reduced from 11 crayfish per trap night to 0.5 crayfish (c. 95%) (Hein et al., 2006, Hein et al., 2007). After eight years the trapping programme ceased, but the rusty crayfish population did not recover in the following four years. Instead, following reduction of density of invasive rusty crayfish, the aquatic plants recovered and sunfish (*Lepomis spp*.) that had been predated by the crayfish increased. Effects on invertebrates varied, due to the recovering fish population switching to the more readily available invertebrates (Hansen et al., 2013). As yet, there are no comparable cases of effective control of invasive crayfish by trapping in Europe.

Manual removal has seldom been carried out because it is so labour intensive. There are two notable examples of attempted control of signal crayfish by manual removal from England and one from Spain. In the three cases the intensive and prolonged programmes of removal achieved some reductions in catch per unit effort over time compared to the initial catches, (50 to 80% lower catches). In the River Gwash in Leicestershire, England a population of signal crayfish was thought to be present only within a 800 m length of the stream. A section 1300 m long was subject to intensive manual removal (Judson, 2003). Sections were repeatedly worked by a line of people, who removed crayfish by trawling with hand-held nets and additional hand search of the bed and banks. A minimum of 100 man days effort was repeated every summer and early autumn for six years. After two years the catch was reduced to approximately half that in the first session, but no further reduction was obtained during the next four years and the invasion continued. Another stream in Leicestershire, Gaddesby Brook, was give the same intensive treatment and although by the second year large crayfish were reduced in abundance, the number of young of year caught was five times greater than in the previous year (Sibley, 2000, Sibley, 2001). In Spain, a spring-fed stream approximately 1 km in length had a control programme for signal crayfish using a combination of trapping and manual searching continuously (c. 40 man days month^{-1}). In the first year the catch was approximately 30 crayfish per man day, but it fell to about a third by second year and thereafter varied seasonally, but was sustained at the same level (Dana et al., 2010).

Electro-fishing, which is used in fisheries survey, often has a bycatch of crayfish, but the method generally has lower catches than either trapping or manual survey, so it has no advantages over them.

1.5.2 Physical methods of control

These methods include modification of habitat.

Draining a pond temporarily is not effective, because signal crayfish can survive out of water, as was reported by Holdich and Reeve (1991). This issue will be discussed further in Chapter 5. Infilling of a small pond would be likely to achieve eradication of population and in some of the cases where biocide treatment was considered (see Chapter 4), it would have been a very feasible option to infill the bed and banks of a pond and subsequently reinstate the pond, if required. Indeed, in some cases this would be a less expensive option than biocide treatment. The method is destructive, however, so is not likely to be acceptable as an eradication treatment in most cases, but it has an advantage that there is no effect outside the treated area and should not be entirely discounted.

Freezing is another physical method with potential to kill crayfish. Kozak and Policar (2003) found some crayfish survived in a largely dewatered pond through the winter, when air temperatures regularly reached -20^{o}C, presumably because some individuals were able to avoid freezing by hiding below the surface. Artificial freezing is used as an engineering technique that is sometimes employed during the construction of tunnels, because substrates can be frozen, allowing tunnel-boring to proceed. In principle, such a treatment of a drained waterbody would kill crayfish, because the process would ensure that the substrate was entirely frozen beyond the maximum extent of burrows, but the cost would be likely to be prohibitive, in addition to the impacts on non-target flora and fauna. It is unlikely to be used in practice.

Electric shock treatment is a potential eradication or control method, which is addressed in detail in a field trial reported in Chapter 6.

Physical barriers do not remove or kill crayfish, but may control crayfish by preventing or delaying invasion. Major waterfalls and culverts prevented upstream invasion by red swamp crayfish (Kerby et al., 2005) and dams were barriers to signal crayfish (Light, 2003), although if the surface was broken, or there was vegetation on the dam face, this did provide an access route (Wirka, 2006). Physical barriers do have an important function, especially with respect to the protection of isolated sites for indigenous crayfish, ark sites, which are discussed in Part 3 of this study (Chapters 8 and 9). The disadvantage of physical barriers is that if they are sufficient barrier against crayfish, they may also impede movement of fish. Another use of barriers is to prevent invasive crayfish in a small site escaping. A catchpit design with a bell-mouth overflow was installed in the drainage outfall from a pond at West Tanfield in Yorkshire (Rogers and Loveridge, 2000, Peay, 2001) and catchpits have also been built on outfalls from a group of fishing ponds with signal crayfish at a site near Leeds in 2013.

1.5.3 Chemical methods

These methods include chemical attractants and repellents; biocide treatments; endocrine disruptors, and microbial biotoxins.

Chemical attractants and repellents. Stebbing et al. (2003) investigated whether sexual pheromones were an attractant to signal crayfish. Male crayfish did respond, whereas female crayfish did not and the attractants were much less effective than food bait. When sexually receptive red swamp crayfish were used to lure other crayfish to traps, again it was males that responded rather than females (Aquiloni and Gherardi, 2010). Even if the attractants were extracted and commercially produced, it would add to the cost of trapping and it is unlikely that the results would be any better. If a safe chemical repellent was available for crayfish it might be of some value for angling on sites where signal crayfish cause nuisance by taking baits. In principle, a repellent might also have a value in discouraging crayfish from approaching a barrier or outlet, although it is uncertain whether any chemical would be acceptable for continuous dosing of a waterbody, or whether crayfish would become habituated to it, reducing its effectiveness.

Biocide treatments are chemicals with a high toxicity to crayfish, including organophosphate insecticides, natural pyrethrum and synthetic pyrethroids. There are none available that are selective for crayfish, so they have impacts on non-target fauna. This limits their use to eradication treatments on relatively small sites. The use of biocide treatment is featured in this study and is discussed in more detail in Chapters 4 and 6.

Endocrine disruptors. The avermectin emamectin benzoate was developed for use in fish feed as a treatment for farmed salmon against sea lice, ectoparasitic copepods. The product does not seem to have been tested on freshwater crayfish, but it has been tested on American lobster *Homarus americanus*. Burridge et al. (2004) found both adult and small juvenile lobsters were capable of surviving relatively high doses of emamectin benzoate (LC_{50} >589 µg $g^{-1}$for juveniles), which meant it was unlikely to pose an acute lethal threat to lobsters foraging around salmon farms, where it was used to kill sea lice. Subsequently, it was found that the product could induce premature moulting in ovigerous lobsters and the lowest observed effect level has been calculated at 0.22 µg g^{-1} lobster (Waddy et al., 2002, Waddy et al., 2006). If crayfish showed a similar response, an ovigerous female weighing 10 g would have to consume at least 0.4 g medicated fish food with emamectin benzoate at 5 µg g^{-1} feed. Application would have to be targeted carefully, because ovigerous female crayfish are relatively inactive over winter. They do feed in the spring before the eggs hatch and can be seen in night-viewing surveys and traps (Peay unpublished), so if the product does cause abortion in crayfish, application in May or early June might reduce the number of juveniles hatched. The effects on non-target species would be of concern. Treatment with emamectin benzoate or similar product would have to be carried out every year to reduce the population, so other aquatic species would potentially be exposed to it.

Microbial biotoxins. Endotoxins from *Bacillus thuringiensis* ssp. *Israelensis* (Bti) and *B. cereus* have been used to protect agricultural crops and their action has been reviewed in detail (Whalon and Wingerd, 2003, Bravo et al., 2006). The endotoxins are relatively species-specific, which has proved useful in the management of mosquitoes. Effects of Bti insecticide have been recorded on crustaceans in some cases, notably reduced fitness of *Daphnia magna* fed on Bt transgenic maize (Bøhn et al., 2008). At present there are no strains of *Bacillus thuringiensis* available for use on crustaceans, although it could be of value in the control of non-native crayfish if a suitable strain is found. The low persistence of Bti insecticides referred to by Whalon and Wingerd (2003) means it would require regular application to suppress populations of crayfish, as it is more likely to achieve a high level of control, rather than eradication. The species selectivity would be important, particularly the susceptibility of non-target crustaceans and other aquatic invertebrates. One concern is the evidence of development of resistance to biotoxins in crop pests, notably when the Bt i has been used as a spray application to field crops, with diamond back moth, *Plutella xylostella* increasingly resistant (Whalon and Wingerd, 2003). This might be a problem with any toxin developed for crayfish too.

1.5.4 Biological methods

These methods include predation, autocidal methods (male sterilization) and pathogens.

Predation. Wherever indigenous fish and crayfish co-occur, predatory fish will take advantage of crayfish as part of the benthic macroinvertebrate community. Crayfish have developed strategies for avoidance; by detection of fish by scent (Appelberg and Odelström, 1988); use of shelters (Stein and Magnuson, 1976, Mather and Stein, 1993) by day and foraging at night (Abrahamsson, 1981, Kozak et al., 2002), and defence, by use of their chelae (Keller and Moore, 2000), or by escape-swimming using tail-flipping (Blake and Hart, 1995). Nonetheless, as described by Hein et al. (2007), some fish are able to have a regulating effect on crayfish populations.

Englund (1999) found that green sunfish *Lepornis cyanellus* strongly reduced the density of the Appalacian brook crayfish *Cambarus bartoni.* Nyström (2002) observed that some Scandinavian lakes with a high density of eels *Anguilla anguilla* had a low density of noble crayfish. Crayfish are predated by eel and perch *Perca fluviatilis* (Blake and Hart, 1995) and crayfish can also form a sizeable portion of the diet of large perch (Nyström et al., 2006). Pike *Esox lucius* are also predators of crayfish (Elvira et al., 1996). However, Holdich and Domaniewski (1995) recorded increasingly high density of signal crayfish in a lake fishery heavily stocked with brown trout, rainbow trout *Oncorhynchus mykiss*, perch and carp *Cyprinus carpio,* all of which predate crayfish. Similar cases, of high densities of signal crayfish in coarse angling lakes that are heavily stocked with large carp, are common throughout England.

Aquiloni et al. (2010) found that eels were capable of reducing the abundance of juvenile red swamp crayfish, but due to their limited gape and the defensive chelae of the crayfish, eels were not able to affect adult crayfish and in laboratory tests eels avoided large crayfish. Nonetheless, in an enclosed site if there was some potential to keep eels at high enough density, Aquiloni et al. (2010) considered there might be scope for a combination of trapping and predation to control the abundance of a population of the crayfish. The introduction of eels and pike into a 1.5 ha pond in Switzerland reduced the abundance of an invading population of red swamp crayfish to less than 10% of its initial abundance (trapping catch per unit effort from 3.44 to 0.69), although gut contents analysis showed few crayfish consumed by pike, suggesting the eels had the main effect (Frutiger and Muller, 2002). Retaining the population of eels appears to have been difficult at that site, because of the 250 eels stocked, only one was recaptured subsequently.

Male sterilization. X-ray treatment has been used to produce male crayfish that are capable of mating, but produce fewer young (43% reduction) (Aquiloni et al., 2009). Whilst this has been moderately successful in the laboratory, it is likely to pose significant difficulties for field use. One problem is that even if a large stock of sterilized crayfish is available, it would have no direct effect on sexually mature females and there would still be fertile males available to them in the extant population. There is uncertainty about the effects of density-dependent processes in these circumstances.

The other problem with male sterilization is that at field scale the most likely approach would be to use farmed sterilized stock, which would increase the density in the treatment area and potentially accelerate the invasion, due to increased competition and subsequent emigration of crayfish. Severing the gonapods of male crayfish might reduce mating ability until they could be regenerated after subsequent moults. If so, crayfish could be trapped and released without the need for adding stock. Even so, it would be difficult to reduce the population below the Allee threshold, the critical density below which the population would continue to decline to extinction, especially with on-going expansion of range. The most likely outcome would be a population at reduced density, similar to one with sustainable wild harvesting.

Crayfish plague and other pathogens. Crayfish are beset by a wide range of pathogens, ranging from viral infections, to bacteria, fungi and metazoans, as reviewed by Evans and Edgerton (2002), Diéguez-Uribeondo et al. (2006) and Longshaw (2011). Movement of crayfish to new regions has brought pathogens, notably crayfish plague, to naïve populations, but the non-indigenous species also encounter new pathogens and parasites in their introduced range. This has happened in Europe, where signal crayfish have been exposed to the microsporidian pathogen porcelain disease *Thelohania contejeani* and infection can occur (Dunn et al., 2009). This chronic disease affects muscle tissue in white-clawed crayfish and, in visibly affected individuals, causes mortality after a year or two (Imhoff et al., 2009), with usual prevalence in white-clawed crayfish populations ranging from 0.2 – 10%, although higher incidence has been reported (Imhoff et al., 2009). Horizontal transmission occurs, mainly from consumption of infected tissue (Imhoff et al., 2012). Although infection occurs in signal crayfish, sometimes at rates higher than found in white-clawed crayfish (Dunn et al., 2009), the infected signal crayfish show lesser or no visible symptoms and no evidence of sporulation has been seen. Hence, it is unlikely that porcelain disease will have much effect on the invading populations of signal crayfish and indeed the invasive signal crayfish may act as a sink, in effect diluting the parasite where white-clawed crayfish and signal crayfish occur sympatrically.

A major difficulty in selecting a pathogen against signal crayfish, or other American crayfish species, is the risk of transmission of a new lethal agent to indigenous crayfish. The White Spot Syndrome Virus (WSSV), an intra-nuclear bacilliform virus, moved from commercial culture of marine shrimps to red swamp crayfish in commercial aquaculture feedstock and other American species are susceptible (Evans and Edgerton, 2002), but so are European species.

Mass mortalities of signal crayfish in Europe appear to have occurred in Sweden, Finland and Spain, likely due to crayfish plague (Edgerton et al., 2004b) when signal crayfish carrying crayfish plague were immuno-suppressed due to environmental stress. Lethal infection of crayfish plague in signal crayfish has been produced in laboratory conditions (Söderhall and Cerenius, 1999) and in a recent review of potential pathogens, Freeman et al. (2010) considered that further research on susceptibility of non-indigenous crayfish to crayfish plague would be useful. Crayfish plague has been used to eradicate enclosed populations of the Australian yabby, *Cherax destructor* at two sites in Spain (Holdich et al., 2006a).

As shown in this section, there is no easy solution for the management of invasive crayfish. Apart from trapping, relatively few trials been carried out on eradication or control. Part 2 of the thesis investigates two potential methods for eradication or control in limited areas: biocide treatment (Chapter 4) and electric shock (Chapter 6). It also includes an assessment of the costs of biocide treatment compared to trapping or re-stocking (Chapter 7).

1.6 Conservation of indigenous crayfish

Whilst there are some 640 crayfish species globally, many of them are threatened (Crandall and Buhay, 2008). In his detailed and wide-ranging review of conservation of crayfish Horwitz (2010) considered the basis for concern and summarised the threatening processes to crayfish globally as: water regime change, climate change, habitat quality change, biological invasion, human predation and combinations of these factors. Ironically, the signal crayfish, which is so invasive within its introduced range in Europe, is threatened by other species of invasive crayfish that have been introduced within its natural range (Larson and Olden, 2011).

Within Europe, there are only five indigenous crayfish species, but the growing threats to them from crayfish plague, non-indigenous crayfish, pollution and habitat degradation and in some areas, over-harvesting have been recognised as being of serious concern. Legislation has been introduced to protect crayfish species both at European and national scales (Peay, 2009b). In the preface to the Atlas of Crayfish in Europe, Souty-Grosset et al. (2006) highlighted the importance of indigenous crayfish, in terms of their provision of a range of ecosystem services: for their role in biological diversity, their economic and recreational value as food sources, as flagship species for environmental quality in aquatic ecosystems, for their value in education and encouraging involvement of people in the natural world and in research. These wider human values placed on European crayfish; recreational, cultural, ethical, aesthetic, scientific and educational have been recognised by other authors too, e.g. Gherardi (2011).

Nonetheless, two of the European species, white-clawed crayfish and stone crayfish are now globally threatened (Füreder et al., 2010a, Füreder et al., 2010b). There is a need to conserve the species at all scales, from European to site (Peay and Fuereder, 2011). Part 3 of the thesis looks at the issue of conservation and presents two new decision-making tools to assist in planning for conservation action (Chapters 8 and 9).

1.7 Overview of thesis

The aim of this study has been to address problems related to crayfish that face ecologists and other resource managers in Great Britain, although the approaches may be relevant to other areas where there are threats to indigenous crayfish. The study set out to provide practical guidance to help resource managers to manage areas with indigenous crayfish for many decades to come and, where practicable and effective, to prevent, eradicate or mitigate the impacts of invading non-indigenous crayfish. This thesis presents part of a body of research on the topic of management of crayfish, which was carried out in the period 2006-2013.

The thesis is in three sections. Part 1 considers the risks of invasion by non-indigenous crayfish and their impacts (Chapters 2 and 3). Part 2 is about management of non-indigenous crayfish (Chapters 4 - 7) and Part 3 considers the conservation of indigenous crayfish and measures to help ensure their survival.

1.7.1 Part 1 Invasion ecology: risks and impacts of non-indigenous crayfish

There are several species of non-indigenous crayfish that have already become established in Great Britain and there are others that been recorded elsewhere in Europe (Holdich et al., 2006a, Holdich et al., 2009d). Understanding of how crayfish enter the country and what happens if they do is important to predict future impacts and wherever possible prevent them.

Chapter 2 presents the results of a literature-based risk assessment for five species of non-indigenous crayfish that are now present in Great Britain and discusses the issues of pathways, prevention and prospects for future distribution. It considers the potential for adverse impacts on aquatic habitats and communities from invasive crayfish.

Chapter 3. The range of impacts that arise following introduction of non-indigenous crayfish include the potential for interactions with fish. Although predation of crayfish by fish has been reasonably well studied, much less work has been undertaken on the potential impact of predation of juvenile fish by crayfish. Chapter 3 is a field study that investigated the effect of an abundant invading population of signal crayfish on the fish population in a small headwater stream.

1.7.2 Part 2 Eradication and control of invasive crayfish

Signal crayfish are invading in Great Britain and in much of mainland Europe. A number of strategies for eradication and control have been considered, but concern about impacts on other fauna mean that until quite recently the only methods tried were physical removal by hand or trap. Some experimental work has been carried out on the use of pheromones to improve efficiency of trapping, male sterilization in an attempt to reduce reproductive success and the introduction of fish predators to try to reduce survival of juvenile crayfish. Those methods have had limited success and all have limitations or disadvantages. Part 2 investigates methods for eradication or control. Faced with the reality of invading populations of signal crayfish, the next group of four chapters addresses eradication or control of invasive crayfish populations when they are still relatively localised.

Chapter 4. Control of invasive crayfish is problematic because of the difficulty in finding methods that are effective and have little impact to other biota. Biocide treatment is not selective to crayfish, but several toxicants, notably natural pyrethrum and the related synthetic pyrethroids have the potential to achieve complete mortality of populations within limited areas. It is not a solution to the problem of invasive populations that are already widely dispersed in watercourses, but it could be applied in limited areas, especially as a management response to newly established populations. Only a few treatments have been carried out, and until this study, few of them had been followed up to see whether they had been fully successful. Hence there is a need to assess the success over long period. Chapter 4 investigates biocide treatment of signal crayfish populations in Britain. A feasibility assessment is presented, which practitioners can use to assess the technical feasibility of treatment and the benefits of using a biocide treatment on a site. Based on this, 13 sites were considered and in 6 of them a treatment was then carried out. The chapter presents the outcomes of biocide treatments and the factors that contributed to success and failure are discussed.

Chapter 5. One of the issues in any eradication programme is detecting populations and ensuring all of the population is accessible to treatment. Among the issues identified in Chapter 4 is the problem that some crayfish may be able to avoid biocide treatment because they are not fully immersed in the water at the time of treatment. Chapter 5 deals with the issue of crayfish that may be in exposed refuges prior to an eradication treatment. A laboratory study was used to investigate the responses of signal crayfish to dewatering and to examine how use of burrows and refuges might affect the success of biocide or electric shock treatments. A field trial was also set up at a pond, which was partly dewatered to investigate how readily crayfish vacated exposed burrows.

Chapter 6. Biocide treatment is the only eradication treatment against invasive crayfish that has been used in Great Britain, but its scope is limited, especially in running water, because effective biocides against crayfish have effects on non-target species too. Hence, there are concerns about impacts outside the areas targeted for treatment if biocide-treated water is released before environmental degradation of the biocide has sufficiently reduced its toxicity. As is known from conventional electro-fishing, a standard method for fisheries surveys, electricity can stun or kill fish in water, but unlike biocide treatment, the effects are very localized. Chapter 6 presents a field trial of a novel treatment method using high power electric shock treatment and discusses its scope and limitations.

Chapter 7. One of the problems inherent in either preventing further introductions of non-indigenous crayfish, or carrying out any eradication treatment, is having the resources available for surveillance/prevention and for a rapid appraisal and management action if a new population is found. Resource managers often have little or no experience of the methods and are uncertain about the costs. Chapter 7 presents a simple cost model, which has been developed in this study to allow resource managers to assess the feasibility of biocide treatment and its cost effectiveness at catchment-scale, in comparison with management for control, or the environmental cost of no action.

1.7.3 Part 3 Conservation management for indigenous white-clawed crayfish

Part 3 of the thesis includes two chapters (Chapters 8 and 9) that deal with the prospects for the conservation of the indigenous white-clawed crayfish in Great Britain. In most catchments in England and Wales and several catchments in Scotland, invasive non-indigenous crayfish are already expanding and there is little or no prospect of preventing further expansion of range. Whilst invasive crayfish bring about a range of impacts in aquatic ecosystems and affect the composition of aquatic communities, by far the most severe threat is to indigenous white-clawed crayfish, which were formerly widespread and abundant in most river systems in England and Wales, but are now endangered due to the lethal crayfish plague, or by competition from North American crayfish.

Chapter 8. Further loss of range of white-clawed crayfish is inevitable, so conservation action is essential if the species is to survive in the UK and indeed in any other parts of its natural range in Europe. One of the best management options is to identify existing populations of indigenous crayfish that are isolated from non-indigenous crayfish and, where appropriate, start new populations in suitably isolated sites within the existing, or previous, distribution of the species. There has been little guidance available to practitioners prior to this research. Chapter 8 presents a simple decision-making tool for conservation managers to help them assess the potential threats to existing populations of white-clawed crayfish and identify potential new “ark sites”.

Chapter 9. The whole issue of crayfish in Great Britain needs consideration at more than site-specific scale. Invasive crayfish are already very widely distributed in Great Britain (Rogers and Watson, 2011). Whilst regulation is addressed at country scale, management of crayfish needs to be considered at catchment scale, because even without further human-assisted introductions of non-indigenous crayfish, invasions progress in catchments in which invasive crayfish are already established. Chapter 9 considers the issue of management strategy for crayfish and describes a “toolkit” of management strategy to help practitioners develop action plans for crayfish at catchment scale.

Chapter 10 presents a general discussion of the study, with recommendations for future study.

Some of the tools described in the chapters have been made available in spreadsheets and other guidance for practitioners and these are included as appendix material with the thesis.

Bucket of signal crayfish from a single crayfish trap in a high density population in a stream. There is a risk of small-sized crayfish from trap catches being used for illegal introductions at new sites (see Chapter 2)

Live sale of signal crayfish at Billingsgate Fish Market, London. Surplus crayfish intended for human consumption are supposed to be killed before disposal, but discards are implicated in the start of some invasive populations (see Chapter 2)

Chapter 2
Risk assessments of non-indigenous crayfish in Great Britain

2.1 Introduction

Great Britain (GB) (comprised of England, Scotland and Wales) harbours hundreds of non-indigenous animal and plant species, the vast majority having been introduced either deliberately or accidentally by man relatively recently, i.e. very few have migrated there naturally in the last few hundred years. The majority do not appear to cause any significant harm, either environmentally or economically, and many form the basis of inland fishery, forestry and farming industries.

The government of the United Kingdom (of Great Britain and Northern Ireland), through the Department of Environment, Fisheries and Rural Affairs (Defra) has recently instigated risk assessments of selected established non-indigenous species (NIS), in order to assess the current and future risks from these species. Non-indigenous crayfish species (NICS) are amongst the first for which such risk assessments have been carried out. NICS have not been introduced into Northern Ireland so far, hence this study only deals with the situation in GB. Although all the NICS have been in GB for many years (Holdich et al., 2004) no risk assessments were carried out on them until 2007, even though they have long been considered to be a serious threat to the future survival of the single indigenous species, the white-clawed crayfish *Austropotamobius pallipes* (Lereboullet 1858). The white-clawed crayfish was given legal protection from taking and sale because of perceived threats to its survival. The threats were from NICS and the disease crayfish plague, caused by the oomycete *Aphanomyces astaci* Schikora 1903 that some of the species carry, rather than harvesting (Sibley, 2003, Holdich et al., 2004).

NICS were first introduced into GB in the mid-1970s and now six species are known to be present, plus another kept under aquarium conditions (Holdich et al. 2004) (Table 2.1). Although a whole raft of legislation has been introduced to try and control their spread this has not been very successful (Holdich et al., 2004, Holdich and Pöckl, 2005) and some species continue to spread, often aided by humans (Holdich and Black, 2007). The situation was not helped by a European Court of Justice ruling (C-131/93) that stated that a citizen has the right to import live crayfish and it is then up to member states to regulate what happens thereafter, i.e. to ban the keeping of such species. This means that the authorities have to continually monitor imports for the aquarium trade, as there are very many tempting NICS available to hobbyists via the internet and trade fairs (Table 2.2).

Table 2.1 Non-indigenous crayfish species established in Great Britain[1]

Common name	**Scientific name**	**Date of introduction**	**Natural range**
Signal crayfish	*Pacifastacus leniusculus*	1970s	North America
Red swamp crayfish	*Procambarus clarkii*	1980s	North America
Spiny-cheek crayfish	*Orconectes limosus*	1990s?	North America
Virile crayfish	*Orconectes virilis*	1990s?	North America
Narrow-clawed crayfish	*Astacus leptodactylus*	1980s	Turkey
Noble crayfish	*Astacus astacus*	1980s	Continental Europe
Redclaw[2]	*Cherax quadricarinatus*	1990s	Australia

[1]All of these species are established in England, only the signal crayfish is established in Scotland and Wales. None of these species are known in Northern Ireland (part of the United Kingdom), or the Irish Republic (part of the British Isles) to date.

[2]Redclaw has been introduced into the wild sporadically from aquaria, but is not known to have become established. It is the only species from outside Europe that can be legally imported into GB for aquarium purposes, and then only in England and Wales.

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Table 2.2 Non-indigenous crayfish species illegally imported into UK for aquarium trade 1996-2006 (reported by Centre for Environment, Fisheries and Aquaculture Science (CEFAS))

Common name	**Scientific name**
Signal crayfish	*Pacifastacus leniusculus*
Narrow-clawed crayfish	*Astacus leptodactylus*
Red swamp crayfish	*Procambarus clarkii*
Spiny-cheek crayfish	*Orconectes limosus*
Marbled crayfish	*Procambarus sp.*
Blue crayfish	*Procambarus alleni*
Yabby	*Cherax destructor*
Marron	*Cherax tenuimanus*
Yabby	*Cherax misolocus*
Lorentz yabby	*Cherax lorentzi*
Zebra crayfish	*Cherax papuanus*
Mexican dwarf crayfish	*Cambarellus patzcuarensis*
Acocil crayfish	*Cambarellus zempoalensi*

A small minority of non-indigenous species will become invasive; those that do typically go through a succession of stages to establish, spread and become damagingly invasive. Williamson (1996) postulated a “Three Tens rule”, based on his work on plants, which predicted that only about 10% of species would pass each stage to the next. Gollasch and Nehring (2006) re-iterated this, saying only 10% of introduced, established species are likely to show a significant impact. The DAISIE project reviewed 12,000 non-indigenous species in Europe and found only 15% caused impacts on biodiversity and 15% caused economic damage (DAISIE 2009). However, NICS in GB show a different pattern, as six of the seven species known to have been imported have become established in the wild, and all of those established are considered pests (Holdich, 1999, Holdich et al., 2004). Up to 2008, only three have been declared as species “which may not be released or allowed to escape into the wild” by the government (HMSO, 1992), namely the signal crayfish *Pacifastacus leniusculus* (Dana 1852), noble crayfish *Astacus astacus* (Linnaeus 1758) and narrow-clawed crayfish *Astacus leptodactylus* (Eschscholtz 1823). The establishment of NICS in GB has been aided by the extensive system of interconnected rivers and canals, as has happened in continental Europe (Holdich and Pöckl, 2006). The only introduced NICS that has not been confirmed as breeding in the wild in GB to date is the Australian redclaw *Cherax quadricarinatus* (von Martens 1868).

One fact that is rarely taken into account and which may make risk assessment difficult is that species do not always behave in the same way when they are moved as they do in their home environment. Good examples of this are to be found amongst the non-indigenous crustaceans that have entered inland waters in Western Europe from the Ponto-Caspian basin and from North America, in particular amphipods and crayfish (Holdich and Pöckl, 2007). Many of these species have undergone huge population explosions, to the detriment of the local biota, whilst causing few problems in their home range. An unusual example concerns the North American signal crayfish, which has not been recorded as burrowing to any great extent in North America, but which in Great Britain, in particular, causes great damage to river banks and lake shores (Holdich et al., 1995; Sibley, 2000). Despite these uncertainties, any information available about a species, both within its home range and in areas to which it has been introduced, can be used in a species risk assessment to identify characteristics that may enhance its likelihood of becoming invasive in another geographic area. Whilst this may not identify all potentially invasive species, it does allow early identification of at least some of those that pose severe risks.

2.2 Risk assessment method

The European and Mediterranean Plant Protection Organisation developed a method of screening for potential plant pests (EPPO 1997). Similarly, in Australia and New Zealand Pheloung et al. (1999) produced a questionnaire-based system to vet plant introductions and identify potentially weedy species, so that these could be banned from importation. In the UK, the approach was adapted subsequently for risk assessment of invasive fish (Copp et al., 2005).

Defra in the UK commissioned a further extension of the model so it could be used for risk assessment of any species (CABI Bioscience, 2005). The risk assessment method is divided into sections that follow the stages of invasion: entry, establishment, spread and impact. Each of these stages is investigated by questions to which the assessors respond on a qualitative scale and by additional comments or explanation. The responses are also recorded automatically as a score. The assessors also assign an uncertainty value to their responses, which takes account of the type and quality of information available to make the assessment. For example low uncertainty (i.e. high confidence) is assigned when there is direct evidence of impacts in other highly similar biogeographic areas to those in the area covered by the risk assessment, whereas medium or high uncertainty are recorded if the predictions are solely from characteristics of the species in its original range, or from related species in other areas. Details are included of supporting evidence and references. The scores are aggregated progressively in the assessment, with high scores representing the highest risks.

The section on potential spread of a species takes account of biological and climatic factors, the potential geographic extent of the species, plus the indigenous species and habitats potentially threatened, if known. The assessment of potential impacts is sub-divided into three parts: economic, environmental and social impacts. Any particularly vulnerable features are identified, whether habitats, species or geographic areas. Impacts are rated on a five-point scale from minimal to massive impact. There is also an assessment of the potential for eradication or control and the cost implications of this. A concluding section leads to an overall categorisation of risk and whether the species warrants management action.

The completed risk assessments are peer-reviewed by a technical assessor with knowledge of the species or species group and then submitted for further independent review by a national Risk Analysis Panel. This panel uses the risk assessments to identify the pathways that present the highest risks for entry of non-indigenous species into the UK and the species of greatest concern, so that management action can be taken as part of the UK strategy for non-native species (Defra, 2008).

The risk assessment can be carried out for early screening of potentially invasive species before they arrive, with the aim of introducing appropriate preventative measures, or at later stages after arrival. The crayfish species selected for this species risk assessment were all already established in Great Britain: signal crayfish, red swamp crayfish *Procambarus clarkii*

(Girard 1852), spiny-cheek crayfish *Orconectes limosus* (Rafinesque 1817), narrow-clawed crayfish and noble crayfish. They were included because Defra was conducting a review of legislation on invasive species and updating a list of invasive species already established, but banned from further introduction into the wild. One other North American crayfish species, the virile crayfish *Orconectes virilis* (Hagen 1870), which was discovered after the original risk assessments has subsequently been risk assessed, has a similar risk to that of *O. limosus*.

2.3 Results

2.3.1 Pathways and establishment

The species risk assessments looked at all the possible pathways by which the crayfish species could enter GB. Although all the species had already established, this was to consider whether there was on-going entry into GB, or whether pathways needed to be regulated. The only unintentional pathway identified for freshwater crayfish species was as an accidental contaminant with fish intended for the aquarium trade or ponds. The introduction of control measures on the health of imported fish mean it is likely that those sending fish to GB would spot crayfish during preparation for export, or they would be seen as a contaminant during inspection for fish health at the port of entry (this would also apply to Northern Ireland).

Import of crayfish for angling bait or to provide food for fish in coarse angling lakes is the most likely pathway by which *O. limosus* arrived in England. Another possibility is that individuals brought them back from fishing trips in continental Europe. Further introductions by anglers are still possible, via car ferries and Eurotunnel, but are probably less likely than previously. Use of crayfish as angling bait was made illegal in 2005 and any anglers who flout the law have increasingly easy access to non-indigenous crayfish in GB as invading populations continue to expand.

Import of crayfish for aquarium use is illegal, except for redclaw, the species that Defra allowed to be imported as a concession to the aquarium trade when other NICS were banned from 1996. Redclaw is a tropical species requiring temperature above 23^{o}C to breed (Semple et al., 1995) and it was assumed that the species could not survive winters in England. Karplus et al. (1998) recorded survival of redclaw in earthen ponds despite winter temperatures in the range 7 – 10^{o}C for 10 days and tolerance has been reported as low as 3^{o}C for short periods (Semple et al., 1995). Whilst redclaw discarded from an aquarium have been reported in at least one pond in England (Peay, 2006), there is no evidence of any long-term survival to date. However, they have been reported as breeding in a pond in Germany (Lukhaup, 2007). It is thought that *O. virilis* probably entered GB via the aquarium trade and was then purposely disposed of in a local waterbody by a hobbyist (Ahern et al., 2008). A recent interesting development is that a study of the molecular genetics has revealed that specimens from the single GB population belong to the same lineage as specimens from the only other known population of this species in Europe, i.e. in the Netherlands (Filipova et al., 2009).

Other crayfish species are regularly intercepted by UK Customs service at port of entry. Table 2.2 lists 12 crayfish species found as illegal imports since 1996, when all imports of crayfish for aquaria (except redclaw) were banned. Factors contributing to illegal imports of crayfish into the UK may be the difficulty of identifying crayfish species by aquarium stockists. Also wholesalers, mainly based in continental Europe, may be unaware of the restrictions on sale of crayfish in the UK, as some of the species are not restricted in other European member states. Even with regulation, it is still relatively easy for hobbyists to bring in new species for their aquaria and to subsequently breed, exchange, or trade them. An aquarium stockist caught selling the parthenogenetic marbled crayfish *Procambarus fallax f. virginialis* in England in 2008 claimed to have received the stock from a private individual (Scott, 2007).

Import of live crayfish for human consumption is still legal in England and Wales, but not in Scotland or Northern Ireland, which have slightly different regulations. Although live red swamp crayfish were regularly sold in the fish markets in London, this trade has decreased in the past five years and there is only one regular importer now (Stebbing, 2008). Most of the imported crayfish are now frozen or processed red swamp crayfish from China and Spain. By contrast, although aquaculture of signal crayfish failed commercially in England during the 1980s to 1990s, there is now a growing market for wild-harvested crayfish, mainly signal crayfish, although narrow-clawed crayfish are also sold. Despite there being no legal wild harvest or sale of NICS in Scotland, narrow-clawed crayfish were spotted for sale in some fish shops in Scotland in 2008, supplied from the wholesale market in England (Collen, 2008).

Traders selling live crayfish for food are supposed to take great care to ensure their stock cannot escape and that any unsold stock is killed before disposal and this responsibility is supposed to extend to purchasers too. In practice, there is nothing to prevent people releasing crayfish that were caught or bought for human consumption. Releasing any animal that is not ordinarily resident in GB, or allowing it to escape into the wild is illegal (HMSO 1981), but it is difficult to detect and prosecute. Escape or release of crayfish food stock is thought to have been the source of several introductions of NICS, e.g. a large female narrow-clawed crayfish was recently found in a river downstream of a restaurant in the English Midlands, but no others were found in the vicinity. Narrow-clawed crayfish were seen escaping from crates at Billingsgate fish market in London, heading for the Grand Union Canal; also the introduction of red swamp crayfish into ponds on Hampstead Heath, London was reputedly from a restaurant (Holdich et al., 1995). Noble crayfish in the Chew catchment in South-west England were reported as having been released by fish dealers (Frayling, 2007).

There is an increasing likelihood of wild-harvest of signal crayfish leading to further deliberate introductions within GB. An anonymous telephone call was made to the Environment Agency, in which the caller described how signal crayfish had been stocked into a river in Cumbria in north-west England and into another river in south-west Scotland in anticipation of future wild harvest (Butterill, 2006).

In England and Wales, crayfish trapping is supposed to be done only with a consent issued by the Environment Agency. Nonetheless, fisheries bailiffs and the Environment Agency have been reporting increases in the use of unconsented traps in several regions. Very few individuals trap crayfish commercially as their main business, but there appears to be a growing interest in small-scale operations. This is due, at least in part, to coarse angling clubs, which use trapping to try to alleviate the nuisance from NICS taking angling bait and hence have large numbers of unwanted crayfish. At the same time, there has been promotion of demand for crayfish, albeit unintended. Several popular television chefs have featured wild-caught crayfish on their programmes and some media coverage has even suggested that there is a conservation benefit to indigenous crayfish from harvesting NICS. Soon after one cooking programme promoting signal crayfish, the national fisheries consents office of the Environment Agency received over 700 new enquiries about trapping, more than the total licences granted in the previous year (Stone, 2007).

Other introductions arise from well-meaning ignorance. An employee of the RSPCA, an animal welfare charity, visited a householder in Nuneaton in the West Midlands, England to rescue a signal crayfish, which had been dropped onto the roof of a house, possibly by a grey heron *Ardea cinerea* (Linnaeus 1758). Although well-intentioned, the RSPCA officer introduced the signal crayfish to a wholly enclosed, internationally important reserve for white-clawed crayfish (Daily Telegraph, 2006). There was no immediate outbreak of crayfish plague, as surveys afterwards confirmed the indigenous species was still abundant (Scott Wilson, 2006). Most cases of deliberate releases of NICS are anecdotal, as in the case of the refrigerated lorry carrying live crayfish that broke down on the M25 motorway around London. Because the refrigeration unit could not be operated, the driver released the crayfish into a nearby stream so they didn't die in transit. This release led to a large population of narrow-clawed crayfish developing in a nearby reservoir (Holdich et al., 1995). A well-known journalist recently admitted to having illegally released signal crayfish at his home when he couldn't face killing them to eat. This was despite him knowing the damage done by introductions and yet he was surprised to find one still alive half a mile away the following day (Sunday Times, 2008). In Yorkshire there is riverside area on a historic estate, which attracts thousands of visitors during the summer. The river has a dense population of signal crayfish and, according to staff, visiting children regularly take crayfish away with them, with a high risk of accidental or deliberate release at other sites (Westcot, 2008), including at least one confirmed introduction to a garden pond in the nearby city of Leeds.

The potential mechanisms of spread were assessed as being similar for all the NICS, a combination of natural expansion from populations already established in the risk area (i.e. England, Scotland and Wales) and additional human-assisted introductions.

2.3.2 Spread and impacts

Based on the known distribution of NICS in continental Europe (Souty-Grosset et al., 2006) and parts of GB, climatic conditions and available habitat were considered to be suitable for the spread of any of the crayfish species throughout the risk assessment area, except possibly some of the mountain and bog areas of Wales and the north of Scotland. The potential growth of red swamp crayfish was expected to be limited by the cool summers in northern and western areas, whereas it has been shown to be capable of growth and expansion of its range in south-east England (Richter, 2000, Ellis and England, 2008).

The risk assessments of the potential impacts of NICS differed slightly depending on species. The summarised risks for each species are given in Table 2.3. All five species assessed are potentially able to out-compete the indigenous crayfish. When not carrying crayfish plague, signal crayfish in particular are displacing white-clawed crayfish, especially in northern England (Peay and Rogers, 1999, Bubb et al., 2005). The three American species rate highly for environmental harm because of their ability to transmit crayfish plague, whereas narrow-clawed crayfish and noble crayfish are susceptible to it. A long-established population of narrow-clawed crayfish in the Regents Canal, London has been replaced by red swamp crayfish, although it is uncertain whether this was due to crayfish plague or competition (Ellis, 2008). The few established populations of spiny-cheek crayfish have not been tested for crayfish plague, so it is unclear whether they are carrying it or not, although, as with all American crayfish tested, they are capable of doing so (Souty-Grosset et al., 2006). It is thought that the virile crayfish will pose similar problems to those predicted for the spiny-cheek crayfish.

Table 2.3 Qualitative scale used for the assessment of impacts of non-indigenous crayfish species (after CABI Bioscience, 2005)

Description of impact	**Economic impact (monetary loss and response cost £ yr^{-1})**	**Environmental impact**	**Social impact**
minimal	Up to £10,000	Local, short-term population loss, no significant ecosystem effect	No social disruption
minor	£10,000-£100,000	Some ecosystem impact, reversible changes, localised	Significant concern expressed at

			local level
moderate	£100,000-£1million	Measurable, long-term changes to populations and ecosystem, but little spread, no extinction	Temporary changes to normal activity at local level
major	£1million - £10million	Long-term irreversible ecosystem change, spreading beyond local area	Some permanent change of activity locally, concern expressed over wider area
massive	£10million+	Widespread, long-term population loss or extinction, affecting several species, with serious ecosystem effects	Long-term social change, significant loss of employment, migration from affected area

Signal crayfish, spiny-cheek and red swamp crayfish were also rated as posing major risks to the freshwater environment because of their ability to modify ecosystem composition, reducing the diversity and biomass of macrophytes, macroinvertebrates, amphibians and juvenile fish (see review by Gherardi, 2007a). Signal crayfish are already established in river catchments across the most of the risk area and expanding in all of them, so the impacts are certain to increase in extent too.

The few populations of red swamp crayfish were slow to build up initially, but are now expanding in canals and in the River Lee in the London area (Ellis and England, 2008). Red swamp crayfish may be less able to expand into northern areas due to cool summers, but they have the potential to colonise important wetlands that are currently unoccupied by any crayfish, notably in the fenland of East Anglia, England. Furthermore, connections of canals and other drainage to the rivers means that in the long-term there is at least one potential route to access these wetlands from the River Lee into the Ouse-Ely wetlands.

Compared to pests of crops and livestock, the potential social and economic harm from NICS is minor. The main issue is nuisance, when dense populations of NICS take angling bait. This is a topic often discussed in the angling press in GB and has necessitated changes in fishing methods (Peay and Hiley, 2004). The problem has been serious enough for at least one fishery owner to lose business because anglers refused to fish the infested lake. There is also the potential harm and cost if NICS, particularly signal crayfish, affect recruitment of salmonid fish and hence increase the dependence of socially and economically important recreational angling on stocked fish (Griffiths et al., 2004, Peay et al., 2009).

Burrowing by NICS is more important economically, because it increases the cost of maintenance of canals, rivers and drainage channels. There is growing evidence of damage to river banks from burrowing signal crayfish, especially in the English Midlands (Sibley, 2000). This is expected to increase as crayfish expand their range, especially in the low-lying areas of southern and eastern England, where maintenance of river banks is important for land drainage and flood defence. Current costs of bank protection in maintained rivers range from around £10 m^{-1} for soft works, such as planting of vegetation, to £100-250 m^{-1} for engineered protection, such as channel re-profiling and steel sheet-piling. Existing costs are expected to increase where burrowing reduces the time interval between major maintenance operations in a section of river or canal, although with canals there is the possibility of more extreme damage. To date, no major breaches of canals have been attributed to crayfish, although burrowing by badgers *Meles meles* (Linnaeus 1758) was reported to have caused a breach on the Llangollan Canal in Wales in 2004, which cost £0.5 million to repair (BBC, 2004, British Waterways, 2005), as well as killing the alleged culprits. A more recent breach on the Brecon and Monmouthshire Canal damaged several houses, required the rescue of several people and cost £7.5 million (BBC, 2007).

Signal crayfish would be blocked from access to river banks by steel sheet piling, but evidence from invaded sites in London shows that red swamp crayfish are capable of burrowing down behind bank protection, affecting its stability (Richter, 2000). The low-lying drainage systems and wetlands of

East Anglia, which offer some of the most favourable habitat for red swamp crayfish, if they invade the area, are also among those most dependent on flood defence, because extensive areas lie close to or below sea level. Although not as active as signal crayfish or red swamp crayfish, the spiny-cheek crayfish has also been confirmed as burrowing (Holdich and Black, 2007), so expansion of this species also has potential for costly damage to river banks and lakeshores.

The species risk assessments were carried out for the individual species of NICS, which are all present in the risk assessment area already. They have different current distribution because of the historic pattern of introductions and this will affect future expansion. In addition, there is the potential interaction between NICS where their ranges overlap. From current known distribution and ecological interactions where known, we have made predictions of the potential future distribution of crayfish species in England, Scotland and Wales, as shown in Table 2.4.

Table 2.4 Current and predicted distribution of non-indigenous crayfish species and one indigenous (white-clawed crayfish) in Great Britain (England, Wales and Scotland) and comments on distribution

Species	**Known distribution 2008**	**Predicted distribution by 2050**	**Uncertainty of predicted distribution**
Signal crayfish *Pacifastacus leniusculus*	Very widely distributed and increasingly abundant throughout England and Wales, except west Wales and a few catchments in northern England. Established, but still localised, in most of the main catchments in Scotland.	Expect it to be dominant throughout most catchments in England and Wales and widespread in most of Scotland. May face competition from *Orconectes* species in the future.	Low

Species	**Known distribution 2008**	**Predicted distribution by 2050**	**Uncertainty of predicted distribution**
Red swamp crayfish *Procambarus clarkii*	A few ponds and at least two canals in London. May be a few undetected populations in southeast England.	Expansion into eastern England via river and canals and expect expansion into agricultural drainage system and wetlands. Expected to face competition from virile crayfish and signal crayfish along main route of expansion.	Medium
Spiny-cheek crayfish *Orconectes limosus*	Only a few sites known, but in at least three regions of England, already in at least one river in a major catchment and may have entered another major river catchment, i.e. River Trent.	Expect it to be expanding range rapidly in lowland rivers, especially in the Midlands, in mixed populations with signal crayfish and potentially replacing it in fine-substrate rivers and lakes.	Low
Virile crayfish *Orconectes virilis*	In River Lee, London, spreading fast.	May expand in eastern England, into areas occupied by signal crayfish. May outcompete spiny-cheek if	Medium

Species	**Known distribution 2008**	**Predicted distribution by 2050**	**Uncertainty of predicted distribution**
		expanding ranges overlap.	
Narrow-clawed crayfish *Astacus leptodactylus*	Many sites in southern England, mainly lakes, also a few in central and northern England. Some recent losses due to crayfish plague and competition from other NICS	Expect few populations left, due to further losses from crayfish plague, particularly as most populations (in 2008) are in areas where signal crayfish are carrying crayfish plague and expanding range. However, one new, expanding population was found in the English Midlands in 2008.	Low
Noble crayfish *Astacus astacus*	Southwest England only, in a reservoir and local streams.	Expect no populations, as existing (2008) ones are in an area with expanding signal crayfish populations and frequent outbreaks of crayfish plague.	Low

Species	**Known distribution 2008**	**Predicted distribution by 2050**	**Uncertainty of predicted distribution**
Marbled crayfish *Procambarus sp.*	No known wild populations, although illegal aquarium stock known to have been kept, so there is risk of undetected introductions.	Conditions suitable for establishment and spread. Expect several populations to be confirmed, from aquarium releases. Some might be feasible for eradication with biocide if detected early, but others will probably get into watercourses before detection. Competitive ability against other NICs unknown.	Medium
White-clawed crayfish *Austropotamobius pallipes* (indigenous)	Scattered populations in southern and central England, widespread but sparse in Wales, widely distributed abundant populations in northern England, including a few remaining catchments with no	Absent from southern England except in a few isolated sites. Lost from most watercourses in Wales. Reduced range in northern England, ongoing loss in most catchments, except upstream of major barriers.	Low

Species	Known distribution 2008	Predicted distribution by 2050	Uncertainty of predicted distribution
	known NICS. Two introduced populations in Scotland.	Expect many new isolated populations set up in conservation initiatives.	

Overall, there is likely to be colonisation of most of the river catchments throughout the risk area, except in parts of northwest Scotland and Wales, where combinations of geology and isolation make introductions and colonisation unlikely. Climatic conditions are suitable throughout for all of the species, with the possible exception of red swamp crayfish. Signal crayfish has the advantage of being the first NICS to colonise and hence the greatest range so far, but is expected to face competition in future from spiny-cheek crayfish, virile crayfish and red swamp crayfish, at least in some lowland rivers and lakes.

There are likely to be other crayfish species recorded in future, either as unknown established populations become abundant enough to be detected, or if more illegally stocked aquarium species escape or are released into the wild.

To summarize the risk assessments for the five crayfish species addressed in this study (Table 2.5) there are two species considered to have major risks. One is the signal crayfish, because of its already widespread extent and the developing interest in wild harvest, with associated risks of further introductions. The other is the red swamp crayfish, because of its potential to invade seasonal wetland habitats that cannot be used by the other species of crayfish, its potential reproductive rate, and the scope for its range to increase as climate warms. The other North American species, spiny-cheek crayfish and its congener virile crayfish were rated as species of moderate risk overall because existing populations are in catchments already affected by signal crayfish. Although the *Orconectes* species have the capacity for relatively rapid dispersal along rivers, they are not likely to become preferred species for wild harvest compared to signal crayfish.

Table 2.5 Summary of impacts from species risk assessments of five non-indigenous crayfish established in Great Britain

Potential impacts	**Risk**				
	Pl	**Pc**	**Ol**	**Al**	**Aa**
Potential for environmental harm in risk area	major	major	moderate	minor	minor
Potential for social harm	moderate	minor	moderate	minor	minor
Potential for economic harm	moderate	moderate	moderate	minor	minor
Overall assessment of impacts	major	major	moderate	minor	minor
Ease of control	very difficult	very difficult	very difficult	very difficult, except by crayfish plague *Aphanomyces astaci*	very difficult, except by crayfish plague

Pl *Pacifastacus leniusculus,* **Pc** *Procambarus clarkii,* **Ol** *Orconectes limosus,* **Al** *Astacus leptodactylus,* **Aa** *Astacus astacus.*

No risk assessment has been carried out so far on other NICS intercepted in the aquarium trade (Table 2.2) to see which may be capable of establishing if they enter the wild in GB. A supposed population of *Orconectes limosus* in the River Lee, London, was only recognised as *O. virilis* recently (Ahern et al. 2008). Other *Orconectes* species could be present, but undetected as yet. *Orconectes juvenilis* (Hagen 1870) was only recently identified in France (Churchill and Daudey, 2008). The parthenogenetic marbled crayfish, now considered to be *Procambarus* *fallax* (Hagen 1870) *f. virginalis* (Martin, 2010) has been sold illegally in England (Stebbing, 2007, Practical Fishkeeping, 2007) and may have been introduced already as an aquarium discard, although no populations are known as yet. Populations appear to have established in lakes in Germany and The Netherlands (Souty-Grosset et al., 2006), so it is feasible that marbled crayfish could survive in similar conditions in England too.

There is little doubt that abundant populations of NICS will colonise extensive lengths of river that currently support white-clawed crayfish. They are also well placed to take advantage of the opportunities in rivers where water quality was too poor to support white-clawed crayfish in the past. Major investments in control of pollution mean many watercourses have improved in quality (Environment Agency, 2005), they now support fish and can be expected to support crayfish as soon as the expanding populations of NICS arrive.

Although we can expect the more recently introduced American NICS to partition the available habitats, they are not the only contenders. The Chinese mitten crab *Eriocheir sinensis* (H Milne Edwards 1853) has also established in England and Scotland (see review by Holdich and Pöckl, 2007). Although the species took several decades to establish and expand in the Thames, it has reached the upper tidal limit there (Gilbey et al., 2008). The mitten crab is also expanding rapidly in other estuaries and tidal rivers in the UK, including the Tyne and Humber (Herborg et al., 2005). The rapid expansion in major canals in Europe (Herborg et al., 2005) make it likely that mitten crabs will also extend far upstream in some lowland rivers in England. Their opportunistic diet and rapid growth (Rudnick et al., 2005a) suggest they may be a match for invasive NICS. Their burrowing ability (Rudnick et al., 2005b) is also likely to match or exceed that of signal crayfish.

2.4 Discussion

As pointed out by Gollasch and Nehring (2006), non-indigenous invasive species may threaten indigenous species, lead to habitat change, and even affect the functioning of ecosystems - thus representing a significant risk to the receiving environments. Biological invasions are one of the major global threats to biodiversity (Vermeij, 1996, Richter et al., 1997, Wilcove et al., 1998, Mack et al., 2000, IUCN, 2000, Brönmark and Hansson, 2002). According to the Convention on Biological Diversity (CBD, 2000) non-indigenous invasive species are considered to be the second most important cause of global biodiversity change after direct habitat destruction.

Inland waters appear to be particularly vulnerable to biological invasions (Gherardi, 2007b) and the extinction rate of freshwater species has been estimated as similar to that for tropical rain forests and higher than in most other terrestrial habitats (Riccardi and Rasmussen, 1999). In the United States 35-37% of amphibian and fish species have been classed as vulnerable, imperiled or extinct, whereas among freshwater crayfish and unionid mussels the proportions were 65% and 67% respectively (Richter et al., 1997). Strayer (2006) has estimated that around 10 000 species of freshwater invertebrates are already threatened worldwide.

Freshwater crayfish have a key role in many aquatic ecosystems and those species selected for commercial aquaculture or for aquaria are the most versatile, tolerant and productive (Lindqvist and Huner, 1999). It not surprising therefore that all of the crayfish species that have been introduced into the wild in Europe appear to have established themselves, spread and have become damagingly invasive, in at least part of their new range (Souty-Grosset et al., 2006). Selection of these crayfish species for characteristics favourable for human use pre-selected them for invasive potential and then provided the vectors for their spread.

As mentioned previously, one fact that is rarely taken into account, and which makes any risk assessment difficult, is that species do not always behave in the same way when they are moved as they do in their home environment. Good examples of this are to be found amongst the non-indigenous crustaceans that have entered inland waters in Western Europe and Russia from the Ponto-Caspian basin and from North America, in particular amphipods and crayfish (Berezina, 2007, Holdich and Pöckl, 2007, Zaiko et al., 2007). Many of these species have undergone huge population explosions, to the detriment of the local biota, whilst causing few problems in their home range. Similarly, impacts have occurred with invasion in North America with species from Europe, notably the zebra mussel *Dreissena polymorpha* (Pallas 1771) (Johnson and Padilla, 1996, Ricciardi et al., 1998, Ricciardi and MacIsaac, 2000).

With the possible exception of burrowing behaviour in signal crayfish, the consequences of introducing NICS in GB could have been predicted easily, if only a species risk assessment had been carried out in the 1970s.

Crayfish aquaculture and the import of live crayfish for human consumption or aquaria could have been banned outright at that stage. Yet it took approximately 20 years for any regulation of NICS to be introduced in GB. There was no action until after the international Convention on Biological Diversity, 1992 Article 8h, which set a guiding principle to "Prevent the introduction of, control or eradicate those alien species which threaten ecosystems, habitats or species". There were further delays while there was subsequent enactment of European and UK legislation on biodiversity. Even now, there are still significant loopholes in regulation, which allows movement of live crayfish in England and Wales, ostensibly for food, in areas where any keeping or introduction is banned. Furthermore, the NICS have spread so widely, it is no longer feasible to either control spread from most of the existing areas, nor to entirely prevent deliberate or accidental introductions.

Historically, the introduction of new species in GB and many other developed countries has been seen solely in terms of possible economic benefits, often for a limited group of producers or sellers. There has been little or no consideration given to the potential future environmental, social and economic costs. Even if potential risks are known, there is sometimes an over-optimistic assumption that species kept in aquaculture or aquaria will not escape. Yet sooner or later, accident, malpractice, or deliberate action appears to lead invariably to the species being released into the wild. Nonetheless, the norm has been the precedence of free trade and minimal regulation, unless activities have been demonstrably damaging to other interests in the relevant state. This is the opposite of the precautionary principle (UNEP, 1987).

Deliberate introduction of NICS to new areas for commercial purposes also risks breaching the UNEP Goals (UNEP, 1987) and subsequent European EIA Directive (European Commission, 1985), because the risk of irreparable transboundary impacts from introductions. A classic case was the introduction of NICS in Spain, into river catchments shared with Portugal. Downstream spread of crayfish plague and NICS compromised the conservation efforts in Portugal to encourage recovery of white-clawed crayfish (Gutiérrez-Yurrita et al., 1999). Other major river systems in Europe either form or cross international boundaries. The River Danube is a major corridor for the expansion of spiny-cheek crayfish *Orconectes limosus* to other countries and the inter-catchment links of canals exacerbate the potential for colonisation (Souty-Grosset et al., 2006, Holdich and Pöckl, 2007). Introductions of *Procambarus clarkii* into East African lakes (Foster and Harper, 2007) has allowed colonisation or further introductions between Uganda and Kenya. The start of commercial aquaculture in Mexico with Australian *Cherax quadricarinatus*, without any impact assessmen, is a potential threat to the many indigenous crayfish species in Mexico (Gutiérrez-Yurrita, 2004). It is also a threat to freshwaters in other countries in the region, either by colonisation, or by encouraging yet more introductions.

Genovesi (2007) described the main obstacles to the development of trans-national strategy on biological invasions in freshwater, among which he included: lack of transboundary cooperation, limited ability to detect species early enough, ineffectual or delayed responses to the early stages of invasions, limited tools for eradicating or controlling invasive species in freshwater, the deficiencies and inconsistency of legal provisions and the difficulty of trade regulation.

In principle, there is a legislative mechanism by which NICS could be banned from trade in the European Union. European regulation to implement the Convention on International Trade in Endangered Species of Wild Flora and Fauna, 1973 (CITES) includes scope to prohibit introduction into the Community of: “live specimens of species for which it has been established that their introduction into the natural environment of the Community presents an ecological threat to wild species of fauna and flora indigenous to the Community” EC Regulation No. 338/97 Article 4:(6)(d) (European Commission, 1997). Only a few species have been included in the prohibition so far and no NICS. Agreement on the restriction of international trade appears to be a lengthy process.

Any increase in international regulation within Europe would require coordinated action, either the strengthening of existing regulation in Europe and in member states, or introduction of new European legislation. Both options are being considered in the development of a new, Europe-wide strategy for invasive species (Commission of the European Communities, 2008). The aim is that agreed solutions should be implemented by 2010. With the inexorable spread of NICS throughout Europe, implementation will be difficult.

The number of species moved to new regions has accelerated markedly in the past two centuries, associated with the enormous increase in global trade and transport. A growing number of incidental introductions have occurred, particularly via shipping (reviewed by Gollasch, 2007), but many others have been deliberate, notably introductions of non-indigenous fish and crayfish. Once a non-indigenous invasive species becomes established it is often too late to eradicate or control if it becomes a nuisance. Despite the many thousands of species moved annually, relatively few of those moved actually establish, become invasive and have significant impacts on indigenous species and their environments (Holdich, 1999). It seems only sensible therefore that risk assessments are carried out to predict which species are most likely to become invasive and to do this before they are moved from one country or region to another, an approach strongly promoted by IUCN (2000) and in the practical guidance of Wittenberg and Cock (2001) and others (Genovesi and Shine, 2003, Genovesi, 2007).

Great Britain still has work to do to improve the effectiveness of some regulation on crayfish and a more difficult task on enforcement than would have been required if action had been taken sooner, but it does show the type of action needed, outlined in a new national strategy (Defra, 2008). New Zealand already has a coordinated approach, started in 1993 (New Zealand Legislation, 2008). Other countries or regions without invading populations of NICS, or in early stages of the problems may be able to look at the mistakes made in GB and the subsequent actions and do better for themselves in future.

2.5 Acknowledgements

Thanks go to Paul Stebbing, CEFAS, for his assistance with information and comments during the preparation of the risk assessments and to Niall Moore, Non-Native Species Secretariat, who commissioned the assessments and facilitated the work.

Anglers using landing nets and keep nets. Wet nets and wet clothing are potential routes for transmission of crayfish plague between waterbodies (see Chapter 2)

Wild harvest of signal crayfish (left) and fish farm (right). Wild harvest of crayfish for consumption, or for control of nuisance from crayfish taking angling bait, increases the risks of escape or deliberate introduction of crayfish to new sites. Fish farms that have signal crayfish present, from former crayfish farming or to dispose of dead fish are potential sources of crayfish plague, which can be carried with fish and water to new sites (see Chapter 2)

Chapter 3
The impact of signal crayfish (*Pacifastacus leniusculus*) on the recruitment of salmonid fish in a headwater stream in Yorkshire, England

3.1 Introduction

The signal crayfish *Pacifastacus leniusculus* has been widely introduced in Europe, where it has had significant adverse impacts on European species of crayfish, by competition and by carrying crayfish plague *Aphanomyces astaci*, which is lethal to the European species (Holdich, 1999). Being large, omnivorous invertebrates, introduced crayfish are capable of changing benthic foodwebs by predation, competition and modification of habitat, including shredding and consumption of macrophytes and by burrowing (Nystrom, 1999). Studies have shown that signal crayfish can reduce the abundance of macrophytes (Warner, 1995, Nyström and Stand, 1996, Usio et al., 2009) and similar effects have been found with other crayfish species. A wide range of invertebrates is preyed on by crayfish, with the larger, less mobile invertebrates being significantly reduced, while smaller, fast species are less affected and some species may even benefit from reduced predation by other predatory invertebrates. Adverse impacts of signal crayfish on abundance have been found on snails (Nyström et al., 2001), on chironomids and Trichoptera (Guan and Wiles, 1998), predatory invertebrates and overall invertebrate biomass (Nyström et al. 1996, Stenroth and Nyström, 2003, Crawford *et al.* 2006).

Interactions between fish and crayfish are more complex. Many fish species include crayfish in their diet (reviewed by Foster and Slater, 1995) and this includes signal crayfish, which are predated on by several fish species, including perch *Perca fluviatilis*, eel *Anguilla anguilla* (Blake and Hart, 1995), rainbow trout *Oncorhynchus mykiss* (Nyström et al., 2001) and brown trout *Salmo trutta* (Stenroth and Nyström, 2003). Nonetheless, crayfish have avoidance behaviour, such as increased use of shelter, preferential use of shallow water by juveniles and higher activity at night (reviewed by Nyström, 2002). Some species of crayfish, including signal crayfish, use their outstretched chelae to make themselves too large for the gape of fish, although Nyström et al. (2006) showed perch 25 cm length were able to consume adult crayfish in a lake. Nonetheless, other studies have found that crayfish are able to predate fish eggs (Kempinger, 1988, Savino and Miller, 1991, Dorn and Wodjak, 2004). Competition for shelter has been shown in laboratory conditions between signal crayfish and juvenile Atlantic salmon *Salmo salar* (Griffiths et al., 2004), with the fish having to spend more time out in open water, where they required greater expenditure of energy to keep their position in the flowing water.

Where more than one fish species is present the interactions may vary, for example, small fish species can be displaced from shelter by crayfish, increasing their vulnerability to predation by piscivorous species (Rahel and Stein,1988, Light, 2005); or the fish may show reduced growth in the presence of crayfish (Carpenter, 1995). Benthic fish appear to be particularly vulnerable to the effects of predation or competition by crayfish, with reductions in sculpin species in the USA (Light, 2005), and in bullhead *Cottus gobio* in England (Guan and Wiles, 1997, 1998, Bubb et al. (2009). Whilst Bubb et al. (2009) found some disturbance of bullhead by the indigenous white-clawed crayfish *Austropotamobius pallipes*, non-indigenous signal crayfish were much more aggressive towards the fish in laboratory trials, causing damage to fins and in some cases, mortality. The same authors found reductions in abundance of bullhead in rivers too when signal crayfish were present. Peay (2002 and unpublished) also regularly found bullheads and white-clawed crayfish under the same large cobbles and boulders in several streams in northern England, suggesting that whereas signal crayfish reduce the abundance of bullhead, interactions between the indigenous crayfish and bullhead are minor.

Some studies have not found any evidence of impact of crayfish species on fish. Dietary studies of red swamp crayfish *Procambarus clarkii* have shown this species is not very efficient at catching live fish (Ilheu and Bernardo, 1997) and in a laboratory trial red swamp crayfish did not reduce survival of juveniles of four cyprinid fish species (Xinya,1995). Stenroth and Nyström (2003) set up enclosures with signal crayfish and brown trout fry in a Swedish stream, but found no effect of crayfish on the survival of the fish. Degerman et al. (2007) reviewed data from electro-fishing surveys in 61 streams in southern Sweden that had a period of two years or more when indigenous noble crayfish *Astacus astacus* were present and another when crayfish were absent (generally losses due to crayfish plague), but did not find any reduction of abundance of fish related to either signal or noble crayfish in those streams. Where impacts of crayfish on fish do occur, they may be indirect through modification of aquatic food webs. In a long term study of invasion of a lake in Wisconsin USA by rusty crayfish, *Orconectes rusticus,* Wilson et al. (2004) showed that fish whose diet overlapped with that of the crayfish declined markedly, whereas piscivorous fish did not.

Headwater streams are important spawning grounds for salmonid fish in Britain. Migratory Atlantic salmon and sea trout *S. trutta* return to spawn in their natal rivers and streams after several years at sea and even resident brown trout tend to move upstream into smaller tributaries to find suitable substrates for spawning. If invading signal crayfish have negative impacts on the production of fry or their survival in these streams, this may reduce the population of adult fish over time. It could potentially affect the ability of naturally reproducing populations of brown trout to support recreational angling. This study reports the distribution of an invading population of signal crayfish in a small stream in northern England and presents some evidence for changes in the fish population in the presence of signal crayfish. Possible implications for management of recreational fisheries in rivers are discussed.

3.2 Study area

3.2.1 Description

The study area (Figure 3.1) is in the upland area of England known as the Yorkshire Dales, an area of low hills and glaciated valleys. Bookill Gill Beck is a small headwater stream approximately 5.1 km in length, a tributary of Long Preston Beck, in the catchment of the River Ribble. The solid geology is all in the lower Carboniferous series. At the top of the sub-catchment there is limestone, but where the stream rises, it is overlain by glacial till and peat, at an altitude of approximately 455 m. The geology of the rest of the sub-catchment is primarily sandstone and shales. Bookill Gill Beck is a steep, fast-flowing watercourse. The main study area is a 4.7 km length of Bookill Gill beck, approximately 0.6 km from its source down to its confluence with Long Preston Beck. It has a total fall of 133m, with average gradient of 1:28. The stream is approximately 0.7 m wide at the top, increasing to an average width of 1.9 m at the confluence with Long Preston Beck (Figure 3.2). Long Preston Beck is a larger stream, approximately 4 m wide upstream of the confluence, approximately 3.8 km upstream of the River Ribble.

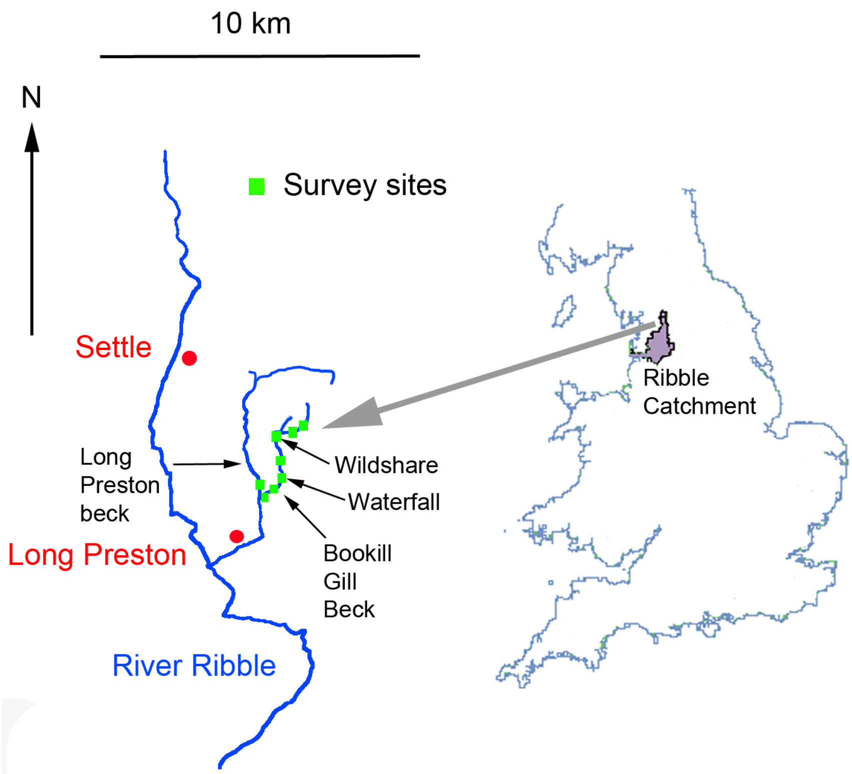

Figure 3.1 Location of study area showing the Ribble catchment and sites on Bookill Gill Beck

Figure 3.2 Bookill Gill Beck at the confluence with Long Preston Beck (Bookill Gill Beck on right side)

The land use in the sub-catchment is unimproved or semi-improved pasture, grazed by sheep and cattle, with extensive seasonally wet areas dominated by rushes, especially soft rush *Juncus effusus*, on the upper slopes and in patches along spring-fed flushes and parts of the valley bottom. A short section of the stream, 120 m, flows through Wildshare Plantation, a conifer woodland, which is the site of introduction of signal crayfish (see section below). There are also some individual broadleaved trees by the streamside in places, mainly in the steepest section, where there are rock outcrops.

There are no farmyards, sheep-dips or domestic properties in the catchment of Bookill Gill Beck to affect the good water quality. A road crosses the stream upstream of the study area, but it is a single-track, rural road with little traffic. There is a farm access track bridleway running part of the way down the valley of Bookill Gill Beck, but much of the valley is inaccessible, except on foot. There are several small bedrock steps in the middle section of the stream, one of which forms a waterfall. There is little growth of macrophytes in the channel, just occasional patches of aquatic mosses, such as *Fontinalis antipyretica* and *Rhynchostegium riparoides,* on some of the more stable boulders and areas of bedrock. Epilithic algae comprise most of the plant growth in the stream.

Rainfall is frequent throughout the year (average monthly rainfall 85 mm in May to 170 mm in December), with typically 40-45 days of rainfall in summer (June-August) (Malham Tarn data, MetOffice, 2009). Major spates large enough to move cobbles and boulders occur in the streams every few years, but during periods of low flow the wetted width of the channel decreases and there are frequent short sections of riffle and run where water is less than 100 mm deep between deeper pools and glides.

3.2.2 Presence of crayfish

Historically, white-clawed crayfish were widely distributed in the catchment of the River Ribble in the main river and the tributaries and in the all the other major catchments of Yorkshire (Don, Calder, Aire, Wharfe, Ure, Swale and Derwent). Now, however, signal crayfish have established at sites in all the catchments (records held by the Environment Agency). Although most populations of signal crayfish in Europe carry crayfish plague, in Yorkshire several populations have established that do not appear to be infected. For example, signal crayfish were stocked into a trout farm at Kilnsey adjacent to the River Wharfe in 1983, from which they have been expanding into a white-clawed crayfish population (Peay and Rogers, 1999, Bubb et al., 2005) and now occupy more than 40 km of main river (Imhoff et al., 2011). A moving zone of mixed population extends over several kilometres of the River Wharfe, yet there have been no outbreaks of crayfish plague recorded there in more than 25 years (to 2008) and no evidence of crayfish plague infection has been found in PCR-tests (Dunn et al., 2009). Similarly, signal crayfish were found in the River Ure in 1997, having escaped from a trout lake and fish farm, and have also expanded into a population of white-clawed crayfish without there being an epidemic of crayfish plague (Bubb et al., 2005).

The River Ribble was affected by crayfish plague in 2001, for which the suspected source was a contaminated consignment of fish stocked into the main river. The spread of the epidemic along the main river and up the tributaries was followed in detail (Bradley, 2009) while it eliminated all of the white-clawed crayfish in the catchment, except in a few semi-isolated parts, one of which was Bookill Gill Beck. During a survey of the stream in 2002, signal crayfish were found in a mixed population with the white-clawed crayfish (Bradley, unpublished). Local information indicates signal crayfish were stocked into the upper part of the stream at Wildshare Plantation (Figure 3.1) in about 1995, with reputedly around 4-12 signal crayfish in the original stock (Handy, 2007). This was an illegal introduction because release of signal crayfish into the wild has been illegal in Great Britain since 1992 (under the Wildlife and Countryside Act 1981 Schedule 9, as amended), except in some areas of southern England.

3.3 Methods

Fish survey was carried out by electro-fishing, using generator-driven electro-fishing gear, LUG AB, flat DC, 1kW in 2007, and a battery-powered electro-fishing gear, Electra Catch International ELBP2, Pulsed DC, 300W in 2008, which allowed easier access to sites with no vehicular access. In all the surveys three consecutive runs were carried out (in accordance with a standardised three-run depletion protocol). Fish were identified to species and measured. Substrate, channel characteristics, pH and conductivity were recorded. Although crayfish were caught and recorded during electro-fishing surveys, this by-catch is not included in the measure of abundance of crayfish, which was done by trapping. One site at the downstream end of Bookill Gill Beck and another on Long Preston Beck were not re-surveyed for fish in 2008, due to disturbance of the channel substrate and fish fry during a separate management operation to remove and re-locate white-clawed crayfish.

Crayfish surveys were carried out using crayfish traps with funnel entrances (LiNi and Trappy Tetra) baited with fish-flavoured cat food. Traps were set for one night and lifted the following morning. Traps were set in the pools and slower-flowing glides, avoiding areas that were too shallow to set the traps, or too fast-flowing for much crayfish activity, based on observations of activity at night (Peay, 2003 and unpublished). The minimum distance between traps was 3 m and the maximum approximately 20 m, depending on the habitat present. A total of 15 traps per site was set immediately prior to the electro-fishing surveys in 2007. There was some variation at sites on Bookill Gill Beck in 2008, where 10-18 traps were used, to utilise sites denoted by the field boundaries. Crayfish surveys were carried out in early September. Trapping was carried out in dry conditions, avoiding rainfall events, which, in this catchment, lead to rapid increases in stream flow and low activity of crayfish. All crayfish caught were recorded for species, sex and size recorded as carapace length (CL) and crayfish abundance at each site was recorded as a Catch Per Unit Effort (CPUE), average number of crayfish per trap. No signal crayfish were returned to the stream (a legal requirement).

Charts were plotted in EXCEL and SPSS. Comparison of fish density and crayfish status was made using non-parametric Kruskall-Wallis tests and between fish density and crayfish abundance using Spearman Rank Correlation tests.

3.4 Results

In 2002 signal crayfish were detected in Bookill Gill Beck, 0.65 km downstream of the suspected point of introduction approximately seven years previously. By 2008 the detected limits were 3.4 km downstream and 0.6 km upstream, using trapping and various intensive manual surveys. This represents a detected rate of expansion of 0.46 km $year^{-1}$ downstream and 0.1 km $year^{-1}$ upstream in the period since 2002, compared to approximately 0.1 km $year^{-1}$ downstream in the initial period of establishment from 1995 -2002.

There were no white-clawed crayfish upstream of the signal crayfish population in 2007 and 2008. It is not certain how far upstream they originally occurred beyond the site of introduction of signal crayfish. There is perennial flow upstream of the site of introduction of signal crayfish, although the flow is low in this section in dry years according to local landowners. White-clawed crayfish were present downstream of the confluence and in Long Preston Beck in all years, confirmed by surveys since 2002.

Figure 3.3 shows the distribution and relative abundance of crayfish (CPUE) recorded in trapping surveys in summer 2007 and 2008 and the total density of fish. At all sites at which signal crayfish were trapped in 2008, the CPUE was higher than in the preceding year (Signed Test, n=7, P<0.01). At the downstream end of Bookill Gill Beck the white-clawed crayfish abundance (CPUE) was typically 2.0 crayfish/trap, but there was a reduction in abundance approximately 1 km upstream of the confluence with Long Preston Beck, corresponding to an increase in the abundance of signal crayfish. This transition from white-clawed crayfish to signal crayfish is evident in the lower CPUE for white-clawed crayfish at 2.09 km, where CPUE decreased from 0.7 in 2007 to 0.06 in 2008, and at 2.38 km downstream, where CPUE was 1.5 and 0.7 in 2008 and 2007 respectively. White-clawed crayfish were absent from traps at sites further upstream, although a white-clawed crayfish was recorded a footpath ford (at 1.7 km) during a manual survey in 2007. The signal crayfish population showed much greater abundance than white-clawed crayfish at any site. In habitat formerly occupied by white-clawed crayfish, CPUE of 7.5 and 8.4 recorded at Wildshare, the site of the introduction (0 km) in 2007 and 2008 respectively. The downstream limit of detection of signal crayfish by trapping was at the site 3.1 km downstream of the introduction, although a few individuals were detected further downstream by intensive manual survey and were confirmed at the confluence by September 2008, approximately 400 m beyond the limited detected in traps. This equates to a lag in detection of about a year by traps compared to manual survey, based on the rate of expansion calculated above.

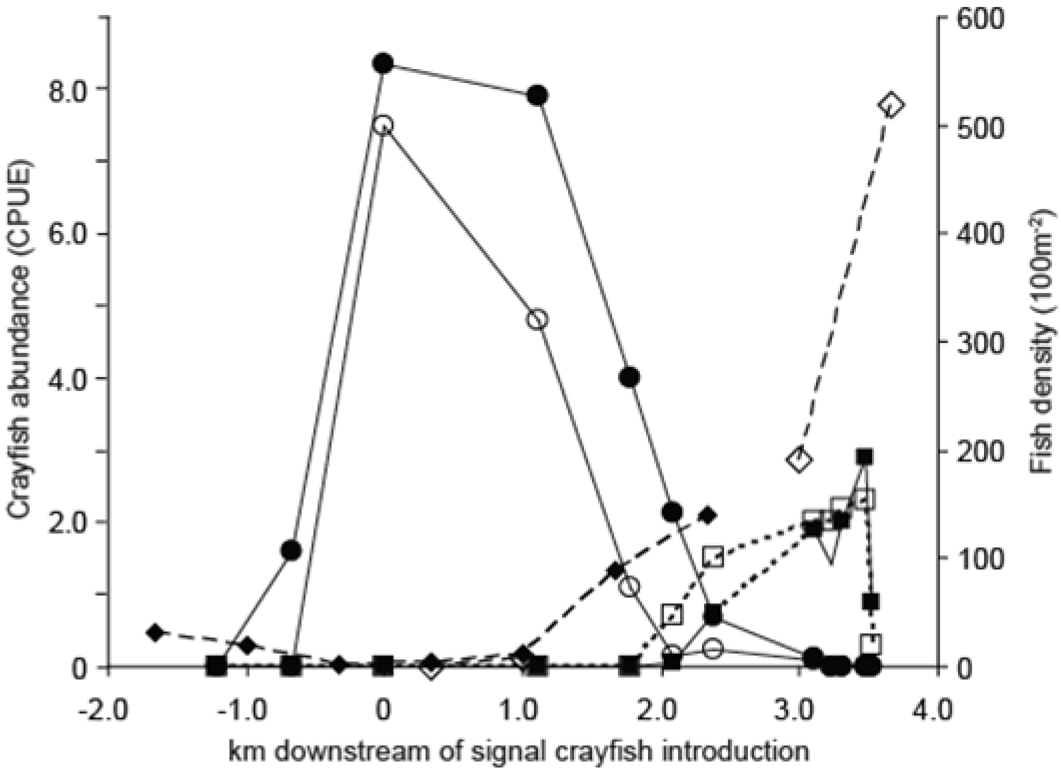

Figure 3.3 Abundance of signal crayfish (circle), white-clawed crayfish (square) (CPUE, average number/trap) and density of fish (diamond) (total density 100 m^{-2}) in 2007 (open legend) and 2008 (filled legend). Error bars have been left out for clarity.

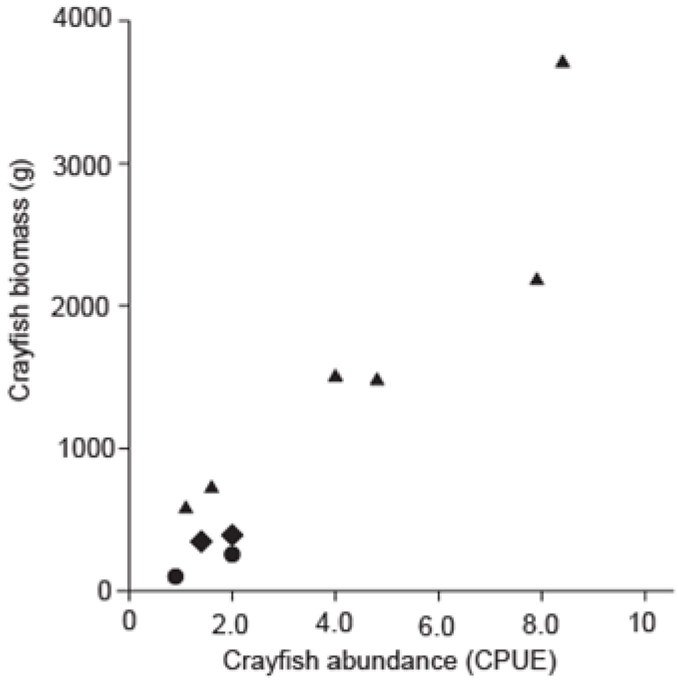

Figure 3.4 Biomass of crayfish (per 15 traps) and total abundance as CPUE (average number crayfish/trap) by crayfish status at sites (white-clawed (circle), mixed (diamond), signal crayfish (triangle).

In addition to higher CPUE being recorded for signal crayfish, individual signal crayfish are able to attain greater size than white-clawed crayfish. This is reflected in the significantly greater cumulative biomass of signal crayfish in traps than white-clawed crayfish (Figure 3.4) (chi-square=6.982, df=2, P<0.03).

The fish population of the stream is principally brown trout, Atlantic salmon and bullhead. Eel is also present in low numbers (Figure 3.5).

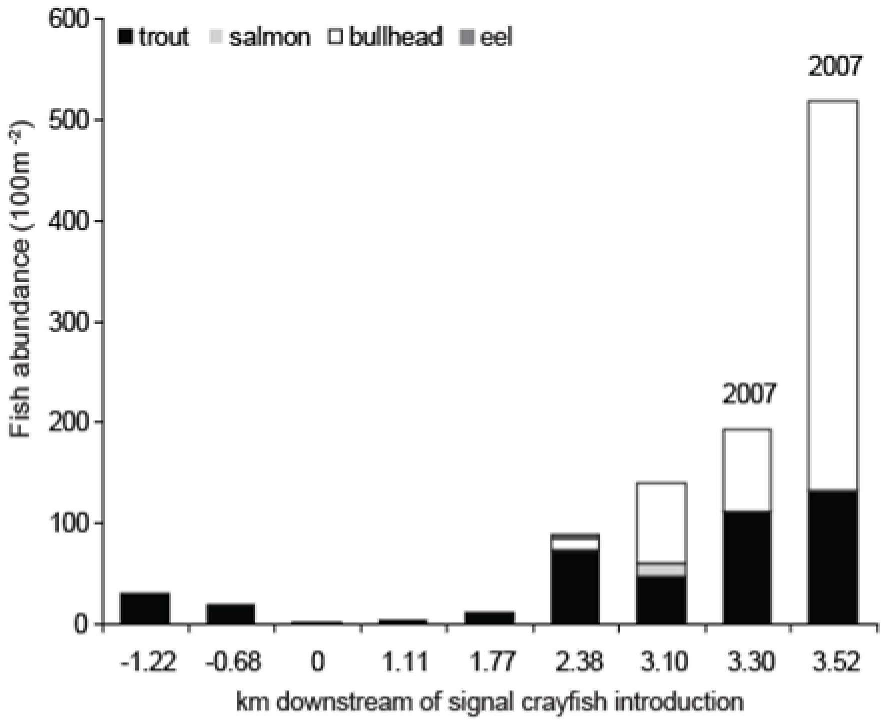

Figure 3.5 Composition of fish catches by species

The land use, rough pasture, is the same throughout, except at Wildshare (0 km), which is conifer plantation. All the sites have abundant stony substrates of varied size, with pools riffles and small rocky steps, plus banks with steep sides, localised erosion and undercutting. There are differences in the proportions of substrates at individual sites and within sites (Figure 3.6), but all are within the range capable of supporting trout in the Ribble catchment. In addition to the substrates in sites in the study area, Figure 3.6 shows the average composition of substrates at other sites surveyed in the Ribble catchment in 2008 where trout fry densities were high (Class A) or good (Class B) (Class A >38 trout fry 100 m^{-2}, Class B 17-38 trout fry 100 m^{-2}, Mainstone et al., 1994).

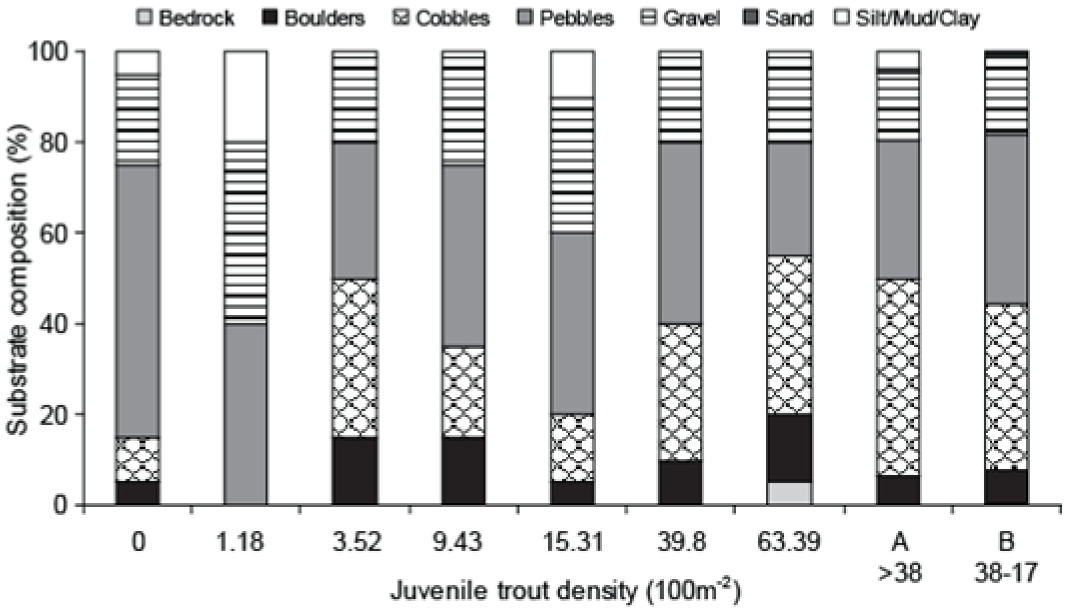

Figure 3.6 Substrate composition (%) at sites in Bookill Gill Beck with the corresponding juvenile trout densities (0-63.39 $100m^{-2}$) and average substrate composition (%) at sites classified as good habitat (Class A 11 sites and Class B 19 sites) for juvenile trout in the Ribble catchment in 2008 and the trout densities that are associated with these habitats.

The density of fish from electrofishing surveys is shown in Figure 3.3 in total and is subdivided by fish species in Figure 3.5. The most widely distributed species was brown trout, present at all sites, from -1.2 km to Long Preston Beck (3.5 km). The furthest upstream record for trout was at -1.4 km, a 150 mm specimen caught as by-catch in a crayfish trap (and so not included in the fish data presented here). Trout were recorded at density in the range 47.5 to 131.9 100 m^{-2} at sites with white-clawed crayfish or mixed crayfish species and at 0 to 18.8 100 m^{-2} at sites with only signal crayfish (Figure 3.5). Bullhead was only recorded in the electrofishing surveys at the site 2.38 km from the introduction site and at increasing abundance downstream. Where present bullhead density exceeded the density of trout at the same sites (Figure 3.5). Juvenile salmon were recorded at the same sites as bullhead in 2008, but were not recorded in 2007. The small waterfall at 1.9 km downstream of the introduction site is considered to be a barrier to migratory salmonids. Most of the trout recorded in 2008 (n=165) were juveniles, with 83% of them less than 100 mm length and only 1.8% more than 200 mm in length when recorded during the electro-fishing surveys in early October 2008. There were no trout caught above 250 mm length.

The by-catch of crayfish during electrofishing is not included in the trapping CPUE or any analysis because the trapping was always carried out in advance, on the previous night in 2007 and variable numbers of days earlier in 2008. Also, the effectiveness of electro-fishing for crayfish is reduced where the presence of boulders and abundant refuges in banks make it more difficult to detect crayfish reliably. Nonetheless the by-catch was 0 to 30 crayfish 100 m^{-2} at sites with white-clawed crayfish or mixed populations, whereas at two sites with dense signal crayfish in 2008 by-catches were 333 (at 1.1 km downstream of the introduction site) and 141.1 (at 1.77 km).

The density of fish differed at sites according to the status of crayfish at sites (signal crayfish, mixed, white-clawed crayfish or no crayfish), for fish overall (chi-square=8.045, df=3, $P<0.045$) and for trout (chi-square=8.328, df=3, $P<0.04$). There were strong negative correlations between the abundance of signal crayfish and the density of trout (Spearman Rank Correlation $r=-0.881$, df=11, $P<0.001$) and total fish ($r=-0.872$, df=11, $P<0.001$) (Figure 3.7), but the weak negative correlations for bullhead and salmon respectively were not significant ($r=-0.334$, df=11, $P<0.3$; $r=0.114$, df=11, $P<0.7$), reflecting the low numbers of sites where salmon and bullhead were recorded.

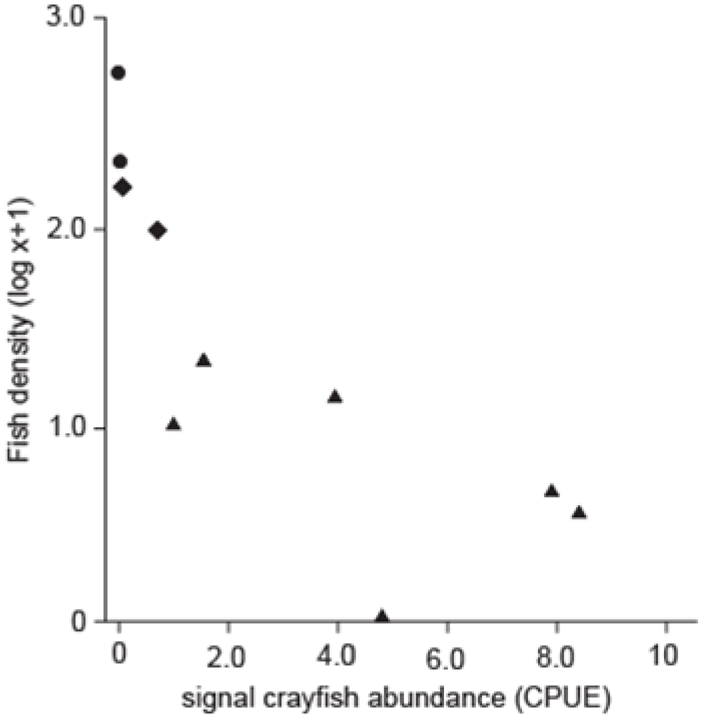

Figure 3.7 Correlation between total density of fish ($100m^{-2}$) in Bookill Gill Beck and abundance of crayfish (CPUE) (white-clawed crayfish (circle), mixed (diamond), signal crayfish (triangle)).

3.5 Discussion

For the first time, at least in Great Britain, there appears to be field evidence that invading populations of signal crayfish can have a significant effect on the recruitment of brown trout in a headwater stream. Where signal crayfish density is high, the density of juvenile brown trout is correspondingly low. In addition bullheads were absent from at least 1.7 km of stream invaded by signal crayfish where they had previously co-existed with white-clawed crayfish. On its own, a negative correlation between fish and signal crayfish does not indicate whether the signal crayfish are causing reduction of the trout, or some other factor causes trout density to be less in a localised stretch of the stream and this allows signal crayfish to increase due to reduced predation. Furthermore, the sites are not entirely independent of each other and a change in one section of the stream may have indirect effects on other areas.

An important consideration is whether the marked change in the fish populations in Bookill Gill Beck could be accounted for by habitat differences, rather than signal crayfish. The waterfall at 1.9 km is considered to be a barrier to migratory salmon and sea trout, which explains the lack of salmon upstream of this feature. The lack of access for sea trout upstream could potentially affect the number of adult trout spawning upstream, although even taking this into account the density of trout fry is less than expected in headwater streams in this catchment when there is a resident population of brown trout upstream. The habitat is suitable for trout upstream of the waterfall; the stream is within the normal range of altitude, slope and width of spawning sites for brown trout and the substrate composition is suitable compared to other sites in the Ribble catchment. There are examples of abundant resident populations of brown trout and bullhead upstream of similar or more severe barriers in the Ribble catchment (Spees, unpublished). Information from local landowners indicates there has been a long-standing population of brown trout upstream of the waterfall. Bullhead and trout were both present upstream of the waterfall at a site 1.1 km downstream of the introduction in 2002, when there was a mixed population of white-clawed crayfish and signal crayfish (Bradley, unpublished). It appears that bullhead have now been lost completely from the sites upstream of the waterfall, despite there having been no changes in environmental conditions since then that might account for the loss, other than the increase in signal crayfish.

Upstream of the introduction site the increase in density of trout corresponds to a reducing abundance of signal crayfish recorded with distance upstream. In general, the abundance of trout would be expected to decrease upstream as the stream became narrower, shallower and steeper upstream, (in this case there are also frequent small steps and shallow gravel runs in the uppermost site), but the data shown in Figure 3.3 and 3.5 do not follow the expected trend and trout are still present and at higher abundance than in the stretch with signal crayfish. The results from the transition zone at the downstream end of the invaded stretch are also indicative. Downstream of the small waterfall, there is a stretch that extends to the confluence with Long Preston Beck with no major barriers to salmonids and where trout density is high. The presence of abundant trout (and other fish species) does not appear to have been able to suppress the increase in abundance of signal crayfish from 2007 to 2008 in the transition zone with white-clawed crayfish, or elsewhere. Hence the reduction in fish recorded in Bookill Gill Beck is most probably due to the high abundance and high biomass of signal crayfish.

Degerman et al. (2007) found no effect of signal crayfish on juvenile trout, even at high density of crayfish, (25-100 crayfish 100 m^{-2}) in electro-fishing catches. Electro-fishing catches of crayfish are not directly comparable with trapping CPUE, but the sites in Bookill Gill Beck with the highest trapping CPUE also had the highest by-catches of crayfish in electrofishing, and these were greater than the highest densities recorded by Degerman et al. (2007). In another study of Swedish streams, Nilsson et al. (2008) had trapping CPUE for signal crayfish at less than 1.0 in 8 out of 10 streams, which is less than the CPUE recorded here, even for white-clawed crayfish. It is not clear why signal crayfish are able to achieve such high abundance in this stream compared to Swedish streams.

In future years, there are likely to be increases in abundance in signal crayfish in the lower part of Bookill Gill Beck and Long Preston Beck and a reduction in the fish population, compared to sites with white-clawed crayfish only, or no crayfish. Signal crayfish in invading populations in England expand their range progressively, although the rate varies, generally slow during establishment and more rapidly thereafter, with typical rates of around 1-2 km $year^{-1}$ in both lowland rivers (Guan and Wiles, 1996) and in the upland rivers of Yorkshire (Peay and Rogers, 1999, Bubb et al., 2005). The current rate of expansion in Bookill Gill Beck is slower (less than 0.5 km $year^{-1}$), but now that the population has reached high abundance in part of its range and has overcome any delaying factor of the small waterfall, it is possible that the rate of expansion may accelerate in the next few years.

The data presented here do not show the mechanism of loss of recruitment in Bookill Gill Beck and it is possible than several factors are in operating in combination. At least some predation of fish by crayfish occurs. Many trout in the zone with signal crayfish had fin damage, or bruising which was unlikely to be attributable to damage during electro-fishing. A dying trout (95 mm length) was caught with a cut in its throat unmistakeably made by the chela of a crayfish but with no other visible indications of damage or poor condition (Figure 3.8). By contrast, Stenroth and Nyström (2003) caged batches of 20 trout fry (average 31.6 mm length) with either 5 or 10 signal crayfish of three size classes (15, 23 and 45 mm carapace length (CL) in a Swedish stream, but did not find any evidence of injuries and no differences in trout survival, length or weight, despite the fish being smaller sizes than those caught at the end of summer in Bookill Gill Beck. Predation of large trout fry and parr by signal crayfish in Bookill Gill Beck may simply be opportunistic, when the fish pass within reach of the crayfish. The opportunities for predation by crayfish may be higher in this stream due to the locally shallow water during periods of low flow, which increases the chance of fish being within grasping range of crayfish, or may be a minor effect compared to other sources of impact.

Shelters that are resistant to high flows are likely to be important to both fish and crayfish in this steep, spate stream. Signal crayfish utilise the refuges under stones in the channel and also make use of undercut banks and burrows that cannot be used by the fish. In the areas with the highest trapping CPUE, signal crayfish appear to occupy almost all the potentially usable refuges in the channel when a manual search is carried out. Griffiths et al. (2004) showed salmon fry had to spend more time swimming in open water when signal crayfish occupied refuges. Reduced access to refuges may make the juvenile fish more vulnerable to being washed away during floods. Floods appears to be an important factor in recruitment of salmonids from year to year, with Ribble Catchment Conservation Trust reporting reduced abundance of juvenile trout in late summer surveys if there have been large or more frequent flood events in the preceding winter and spring. Avoiding the crayfish in refuges may also leave the fish potentially vulnerable to increased predation, especially by grey herons *Ardea cinerea*, which regularly hunt along the stream and have a roost site nearby. Another possibility is that signal crayfish are helping to displace fish downstream and that due to the relatively steep gradient and at least one barrier, fewer fish are able to migrate back to take residence or spawn.

Figure 3.8 Live brown trout fry with jaw cut by signal crayfish (left) and signal crayfish inflicting a cut on a trout parr (right)

There is no information at present on the degree to which signal crayfish in this stream predate fish eggs or emerging fry. Signal crayfish are assumed to be relatively inactive in streams during the winter. Spawning of brown trout occurs in streams in the Ribble catchment in the period late October to December, depending on flows. The late spawning period may reduce the opportunity for predation of eggs by crayfish, but with such high density of signal crayfish there may be pressure to forage even in winter, when there are readily accessible and nutritious fish eggs and larvae. This is particularly so as there has been a pattern of mild winters in northern England in recent years, with only a few days of snow each winter at most. The trout alevins emerge from the gravel in March and April. Depending on the water temperature during incubation, active swimming and avoidance of predators would not be expected until late May or early June. With crayfish in Yorkshire showing increasing activity in April and May, there is the potential for predation when small juvenile trout are at their most vulnerable.

In addition, observations on site suggest there are changes in the composition of the inverterbrate fauna in the signal crayfish zone, such as reduction or loss of *Gammarus pulex*, an important food source for trout. This has not been investigated in any detail as yet, but the findings of reduced invertebrate biomass (Crawford et al., 2006, Stenroth and Nyström, 2003) suggest this is another possible pathway for impact of signal crayfish on fish.

This is a case study of a single stream and as such it cannot be assumed that the effects would be seen in other invaded streams. It may be that there are characteristics of this stream that have allowed it to develop an especially high abundance of signal crayfish. Certainly, the stream is shallow and the density of adult trout is low – the stream is primarily a recruitment area and it is not stocked with reared fish. In addition, in this case, the signal crayfish were introduced near the upstream end of a small tributary, whereas it is more common for introductions of signal crayfish to be made in less remote areas in the larger streams or main rivers, from which they expand slowly up in to the tributaries. Nonetheless there are many similar shallow, stony headwater streams in this catchment and in many other catchments in upland areas of northern England and Scotland, and these are important for recruitment of salmonids. An increasing number of those catchments have signal crayfish populations expanding in one or more areas.

Since the outbreak of crayfish plague in the River Ribble, the Manchester Anglers Association has changed its management of the fishery from extensive annual stocking of trout to a largely wild fishery. The Ribble Catchment Conservation Trust, which advises angling interests in the catchment, recommends stocking only in compensation for damaging events, such as temporary loss of spawning habitat due to modifications of the river. The Trust and local landowners have invested in a range of habitat improvement measures, including fencing of some stretches to protect river banks from excessive erosion by livestock, dealing with incidents of farm pollution and generally trying to improve the natural production of brown trout and Atlantic salmon in the catchment. The possibility that similar impacts on recruitment of salmonids may be seen in other tributaries over time is a matter of concern to the Ribble Catchment Conservation Trust.

Guan and Wiles (1997) have shown that an invading population of signal crayfish can have an impact on benthic fish; bullhead and stone loach *Barbatula barbatula*. These species are important elements of the overall aquatic biodiversity, but are not of interest for angling. Although the negative impact of signal crayfish on white-clawed crayfish is widely known, there appears to have been little published on the impact of signal crayfish on angling, apart from the nuisance of crayfish taking angling bait in some cyprinid fisheries (Peay and Hiley, 2004). Anglers on the River Wharfe have reported catching brown trout which have eaten juvenile crayfish (Birdsall, 2007) and this has led some of them to assume that signal crayfish solely provide benefits to the recreational fishery.

If impacts that appear to be occurring in Bookill Gill Beck do indeed occur in at least some other watercourses, there is potential for adverse impacts on recreational fisheries which are dependent on recruitment of fish from small headwater streams. Non-indigenous crayfish would be just one of the factors with potential for effects on recruitment, however. Other factors such as land use, water quality, the presence of artificial barriers, the frequency of flood and drought events and fish harvesting regimes may be equally or more important – it is too early to tell, but additional negative impacts from signal crayfish may exacerbate other adverse factors,

Fisheries management policy in Great Britain is increasing encouraging management of natural salmonid fisheries, rather than stocking, so we believe that the potential for invasive non-indigenous crayfish to adversely affect recruitment of fish, including salmonid fish, is a matter that should be investigated further. We hope that this case study will encourage other studies on this topic. Above all, as a precautionary measure to protect both fish and other elements of biodiversity, we hope that those involved in using, managing or regulating recreational fisheries will increase their efforts to prevent further introductions of non-indigenous crayfish in Great Britain and elsewhere.

3.6 Acknowledgements

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Signal crayfish caught in invaded stream, Bookill Gill Beck

Part 2

Management of non-indigenous crayfish

Chapter 4
Biocide treatment of invasive non-indigenous crayfish signal crayfish: successes, failures and lessons learned in the UK

4.1 Introduction

There is a rapidly increasing number of introductions occurring globally (Millennium Ecosystem Assessment, 2005). Within Europe over 11,000 non-indigenous species have been reported (Hulme et al., 2009) and whilst most have no recorded negative effects, at least 10% of introduced species are damagingly invasive (Vila et al., 2010). The proportion with negative impacts is particularly high in aquatic ecosystems (Garcia-Berthou et al., 2005). The number of biological invasions that have economic and or environmental impacts is increasing annually and there is a growing need to manage them (Lodge et al., 2006). The Convention on Biological Diversity requires states to “prevent the introduction of, control or eradicate those alien species which threaten ecosystems, habitats or species” (Article 8(h)) and guiding principles were adopted in 2002, which set a hierarchy of action with prevention as the primary aim, followed by early detection and eradication as the best option where prevention failed, or, if not feasible management for control or mitigation could be considered. These principles have been widely endorsed (e.g. Mack et al., 2000b, Myers et al., 2000, Genovesi, 2005).

Where prevention has failed, in most cases, the window of opportunity to carry out eradication is quite limited and once the invasive species becomes widely established it is often prohibitively difficult, expensive, or unacceptable, due to environmental impacts or social factors, to achieve eradication (Simberloff et al., 2013). If the opportunity is missed, the remaining management strategies are limited to intermittent or continuous control measures, or accepting the impacts of the invasive species as they occur. Yet, whilst concerns about new invasive species are widely publicised and major failures have also been newsworthy, Simberloff (2009) highlighted the growing catalogue of successes and Pluess et al. (2012b) reviewed 173 eradication campaigns and found that about half of them had been successful, across a wide range of taxa and circumstances.

Biocides are among the most frequently used management options for eradication or control, being widespread in agriculture, forestry and aquaculture. They are increasingly used in the management of biological invasions, generally as a rapid response to relatively recently established, or at least localized populations of a non-indigenous invasive species. A global database of eradication campaigns (Kean et al., 2013) included more than 800 eradication campaigns in over 100 countries. Most of the campaigns against plant species and arthropods have included use of bioicides (Wittenberg and Cock, 2001), e.g. the campaigns against malarial vectors (Killeen et al., 2013), but biocides have been used against a wide range of invasive fauna, e.g. against an invasive tunicate *Ciona intestinalis*, which damages beds of blue mussels *Mytilis edulis* (Edwards and Leung, 2009) and the eradication of house mouse *Mus musculus* to protect indigenous flora and fauna on Australian islands (Cory et al., 2011).

Invasive non-indigenous crayfish have been recognised a major source of loss of indigenous crayfish species in Europe, because several species introduced from North America carry crayfish plague *Aphanomyces astaci*, which is lethal to European species of crayfish (Edgerton et al., 2004a, Diéguez-Uribeondo et al., 2006). In addition, as keystone species in freshwater ecosystems, non-indigenous crayfish have modified aquatic communities and habitats throughout their increasingly widespread European range (Gherardi, 2007b, Holdich et al., 2009d). The introduction of crayfish species beyond their natural range has also caused major impacts in North America, through competition with indigenous congeners (Lodge et al., 2000, Lodge et al., 2012) and in Australia (Horwitz, 1990).

Invasive crayfish in Europe are somewhat difficult to control, because of their predominantly nocturnal activity and their use of refuges, both natural and burrows. This makes them difficult to detect at low density in the early stages of establishment. Trapping and manual removal of crayfish has been used traditionally to harvest crayfish and with the indigenous white-clawed crayfish in Italy, over-harvesting is considered to have contributed to reductions in abundance of populations, or even their loss (Brusconi et al., 2008). The crayfish species that were introduced into Europe, however, have all been selected for use in aquaculture and hence are generally more fecund, faster growing and more aggressive than the indigenous species (Holdich et al., 2006a). To date, there do not appear to be any examples of invasive crayfish in Europe having been successfully eradicated by trapping or manual removal. Other options have been considered, including biological control with pathogens (Freeman et al., 2010), male sterilization (Aquiloni et al., 2009), improvement of trapping efficiency with pheromones (Aquiloni and Gherardi, 2010), or predatory fish (Aquiloni et al., 2010). Nonetheless, no single method has been found as yet that appears to be able to eradicate or control established population of invasive crayfish, other than biocide treatment.

There is no biocide currently available that is selective to crayfish alone, or even to crustaceans. Hence, any biocide treatment will kill non-target fauna in the treated area, unless other fauna are removed. Alternatively, non-target fauna will be adversely affected during treatment, but the area will either recover by natural re-colonization, or by re-stocking after treatment. Impact on non-target species is one of the constraints on the use of biocide treatment against non-indigenous crayfish.

Synthetic pyrethroid insecticides are the most toxic pesticides to crayfish (Bills and Marking, 1988, Eversole and Seller, 1997). Eversole and Seller (1997) found the median LC_{50} 24 h for synthetic pyrethroids was 2.5 µg l^{-1} and for ciflutrin LC_{50} 24 h was calculated at 0.13 µg l^{-1} for *Procambarus clarkii* in laboratory tests (Quaglio et al., 2002), compared to 20 µg l^{-1} for natural pyrethrum (Cecchinelli et al., 2012). Natural pyrethrum was selected for use in the UK, however, because of its rapid environmental degradation by photolysis and binding to soils (Leahey, 1979, Palmquist et al., 2012); hence it has lower environmental risks than more persistent alternatives. Natural pyrethrum is not an approved product for use in water in the European Union, so treatments have been carried out under special provisions (an Automatic Experimental Permit, issued by the Health and Safety Executive, which regulates pesticide/biocide use in the UK). Rotenone, another botanical biocide, was already approved in the UK as a treatment against invasive non-native fish, but it has little effect on crayfish, or indeed on other aquatic invertebrates (Morrison and Struthers, 1975).

It is already too late to achieve eradication of non-indigenous crayfish in much of Europe, because species such as spiny-cheek crayfish *Orconectes limosus*, signal crayfish *Pacifastacus leniusculus* and red swamp crayfish *Procambarus clarkii* are already widely established (Holdich et al., 2006a), in many major rivers in the case of the first two species and in extensive wetlands in the third species. Nonetheless, in Great Britain, although the signal crayfish is found in most catchments in England and Wales, there are still catchments in northern England and

Scotland that have not been invaded so far. Regulations are in place to try to prevent human-assisted introductions of non-indigenous crayfish into uninvaded catchments (Peay, 2009b), but where this has failed, if populations have been detected while still in small sites, there is the potential to extirpate the population using a biocide.

The first biocide treatment against crayfish in Europe appears to have been that of Laurent (1995) in ponds with spiny-cheek crayfish, which were treated with the organophosphate fenthion (Baytex®) and appeared to be successful at the time, although the long-term outcome does not appear to have been reported. Kozak and Policar (2003) carried out a chlorine treatment of a pond in a fish farm using chlorinated lime, but in field conditions chlorine did not reach the target dosage and it was not successful. The first biocide treatments against signal crayfish in the UK were at three sites in a catchment in Scotland (Peay et al., 2006a), using natural pyrethrum (Pyblast®).

This study investigates 13 sites where biocide treatment was proposed against invasive non-native signal crayfish in Great Britain in the period 2004 to 2012; six sites where treatment was carried out and seven where treatment was not carried out. The projects are summarized and the outcome is given for the treatments, as determined by post-treatment monitoring. For all the projects the constraints are summarized and compared and the factors that contributed to successes and failures are considered. The assessment of factors is based on the author's experience on all of the projects, apart from the two projects in Cumbria, for which the assessment is based on information provided by Matt Brazier, Non-native Species Technical Adviser, Environment Agency Fisheries Technical Services.

4.2 Methods

4.2.1 Planning and treatment with natural pyrethrum against signal crayfish

The basic method of biocide treatment against signal crayfish using natural pyrethrum was outlined in Peay et al. (2006b) and has been used, with variations, in all the projects described here. In summary there are five stages:

1. appraisal;
2. assessment and planning;
3. preparatory works on site;
4. treatment, and
5. management of post-treatment recovery.

The appraisal stage is an initial appraisal of the feasibility of the project, indicative costs and benefits. This stage usually includes at least some preliminary survey of the extent of the crayfish and assessment of status locally. If a project passes this stage it goes to detailed feasibility and assessment. It is an iterative process that continues through project planning up to the start of work on site and beyond, but a key project milestone is when stakeholders agree the project should go ahead and the required funding is committed. The work potentially required at each stage is summarized in Table 4.1. Small, simple sites require less resourcing than more complex sites, but most elements at each stage are likely to be required, even on simple sites. To avoid impact beyond the area for biocide treatment, treated water needs to be contained within the site until the biocide has degraded and containment may require hydraulic control by flow diversion and/or re-circulation by pumping (see Figures 4.1b and 4.2), which requires more equipment, time and hence greater cost.

Table 4.1 Project outline of work required at each stage of a biocide treatment against invasive crayfish.

Stage	Work potentially required for biocide treatment project
Appraisal	Initial site visit, identification of scale of project, initial view of feasibility, potential benefits depending on status of crayfish in the catchment, identification of main stakeholders and lead agency if project goes to next stage.
Detailed feasibility assessment and project planning	Appointment of project manager; Resources (funding and staff) for detailed assessment; Consultation with all owners and occupiers and neighbours as appropriate; Surveys for extent of crayfish, assess suitability of habitat for crayfish in all areas to be treated; Surveys of other fauna as required; Detailed site survey to identify all inflows, outflows; hydrological surveys and tracer study for groundwater or leakage if required Bathymetric survey to calculate volume, also determine substrate; Preliminary toxicity tests with crayfish, water and substrate from site as required; Plan of technical operation, including materials, equipment, staff, quantity of biocide required, advance works required, operation on site, requirement for any post-treatment management, development of project programme with allowance for weather; Risk assessment and contingency planning, including plan for biomonitoring as appropriate, health and safety, site security if required; Decision on risks and benefits of partial dewatering (storage capacity for treated water if rainfall occurs v. risk crayfish out of water); Consultation with and approvals from relevant statutory agencies for biocide treatment; Funding and other resources for full treatment, also for

Stage	Work potentially required for biocide treatment project
	monitoring outcome; Confirmation of landowner acceptance and agreement on any mitigation measures, compensation provision, or legal provisions; Contract management if contractor used; Orders for supply of biocide, materials, equipment; Communications strategy, before, during, after treatment, with public/media. Health and safety assessment
Preparatory works on site	Prior removal of fish or amphibians if required; Prior diversion of flow, if applicable, partial dewatering well in advance if applicable; Management of vegetation to facilitate biocide application, e.g. mowing, herbicide, partial dredging (with biosecurity to prevent escape of crayfish) as required; Possible test of hydraulic control; Test excavation to check for crayfish above water level; Enabling works, preparation of working area/site compound, if required during complex treatment; Advance delivery/storage of materials and equipment, e.g. material for temporary dams; biocide to secure store; pumps and hoses, boats, sprayers, fuel, clean water for washing; fauna and equipment for bioassays; equipment for emergency use, e.g. first aid, pollution spill-kit etc.; sundry tools and spares.
Treatment	Flow control operating as required; Biomonitoring outside treatment area set up as required; Bioassay/treatment monitoring ready; Equipment, materials readied, all site staff briefed on all procedures; Application of biocide to margins, then to rest of site

Stage	Work potentially required for biocide treatment project
	according to depth plan, additional application in margins before night if required Treatment monitoring with caged crayfish, bioassays or other methods.
Management of post-treatment recovery	Monitoring persistence of toxicity; Management of treated water, e.g. none, dewater to field, removal off-site, accelerated degradation of product; Subsequent monitoring of aquatic recovery with/without restocking. Monitoring for crayfish – 5 years.

Figure 4.1 Illustrations of biocide treatment

a. application of biocide from a boat using a sprayer and handheld boom (top left), b. hosing biocide onto the margins at the lower pond, Ballintuim (top right), c. application of biocide to a stream section, with temporary dams to prevent overspill (middle left), d. plastic cages with live crayfish used to monitor effectiveness of application in each site (middle right), e. tubes with live crayfish buried in the stream bank at Ballintuim to monitor the effectiveness of treatment in burrows (bottom left), f. bioassay cups with *Gammarus pulex* used to estimate the concentration of biocide achieved in field conditions and for monitoring water samples during post-treatment recovery (bottom right).

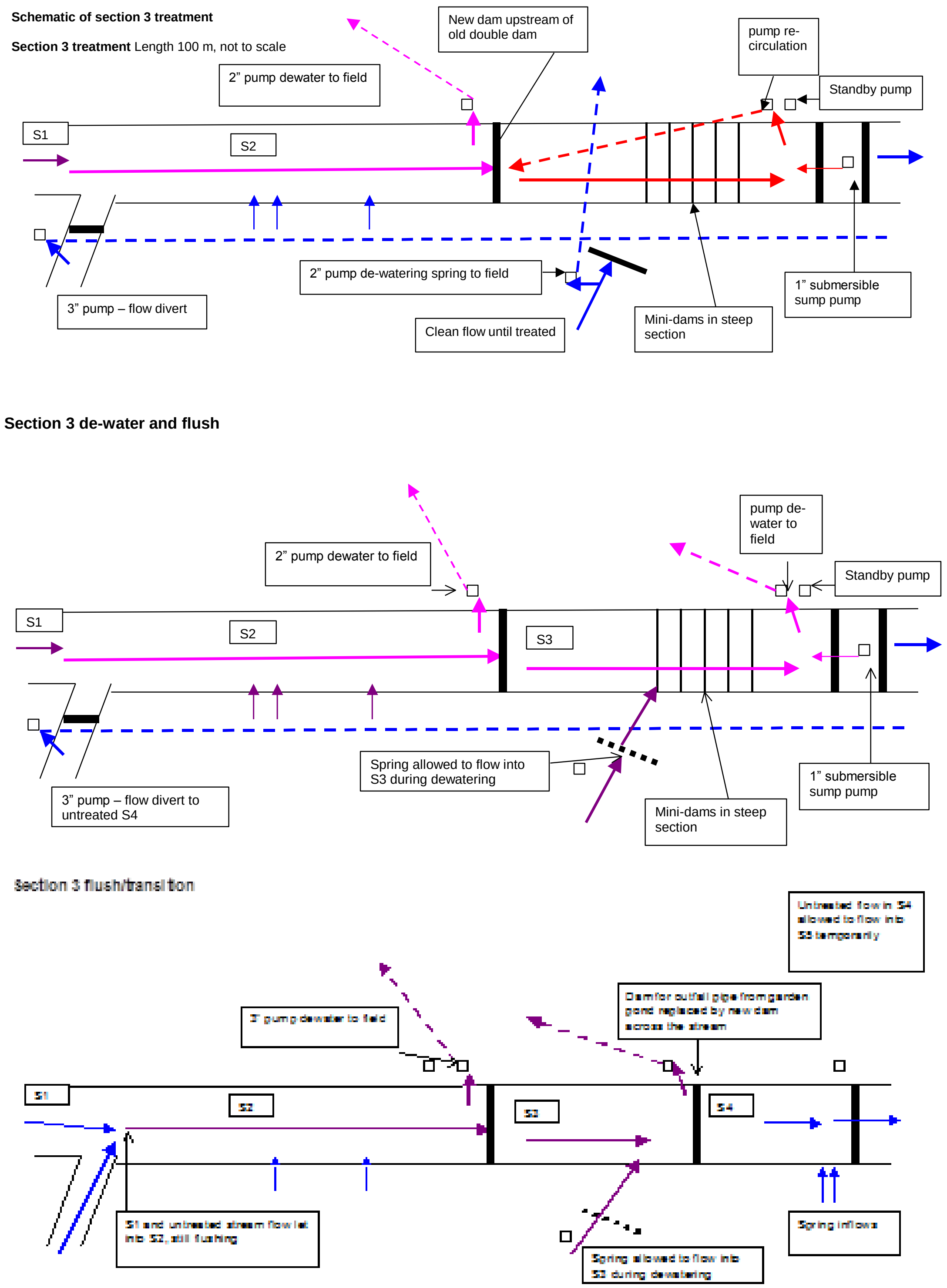

Figure 4.2 Schematic of treatment of a stream showing three stages of treatment on stream section 3, a. treatment, b. dewater and flush, c. flush/transition and preparation for treatment of section 4. Flow is shown as red if newly treated with biocide (Pyblast®), pink then purple during flushing/ degradation of Pyblast and blue if untreated or fully recovered.

In all, 13 sites were assessed for their potential for an eradication treatment with biocide, three were considered unsuitable at an early stage (appraisal), four were not treated for other logistical reasons and six sites were treated (see Figure 4.3 for locations)The projects in which biocide treatment was carried out are summarized in Table 4.2, which shows the year when stages from detection to treatment occurred; the characteristics of each site; the treatment including the target dosage (as active ingredient), the requirement for hydraulic control and staffing; any additional treatment and the outcome from post-treatment monitoring. The projects where biocide treatment was not carried out are shown in Table 4.3. Each project was reviewed in light of the outcome, to determine, where possible, the factors that contributed to success or failure in each case.

4.2.2 Use of biomonitoring

During the treatments the progress of the biocide treatment was monitored by deploying cages with live crayfish distributed within the waterbody undergoing treatment (Figure 4.1d). On the day after treatment if any cages had crayfish that were not dead or severely affected, this would indicate incomplete treatment in the vicinity of the cage. In two cases (lower Mains pond and Ballachulish) additional biocide was applied to a deep area on the morning after the main application. For monitoring the treatment of a stream, mesh-covered tubes with live crayfish were dug into submerged banks prior to treatment (Figure 4.1e), in addition to the cages with crayfish that were placed in the channel.

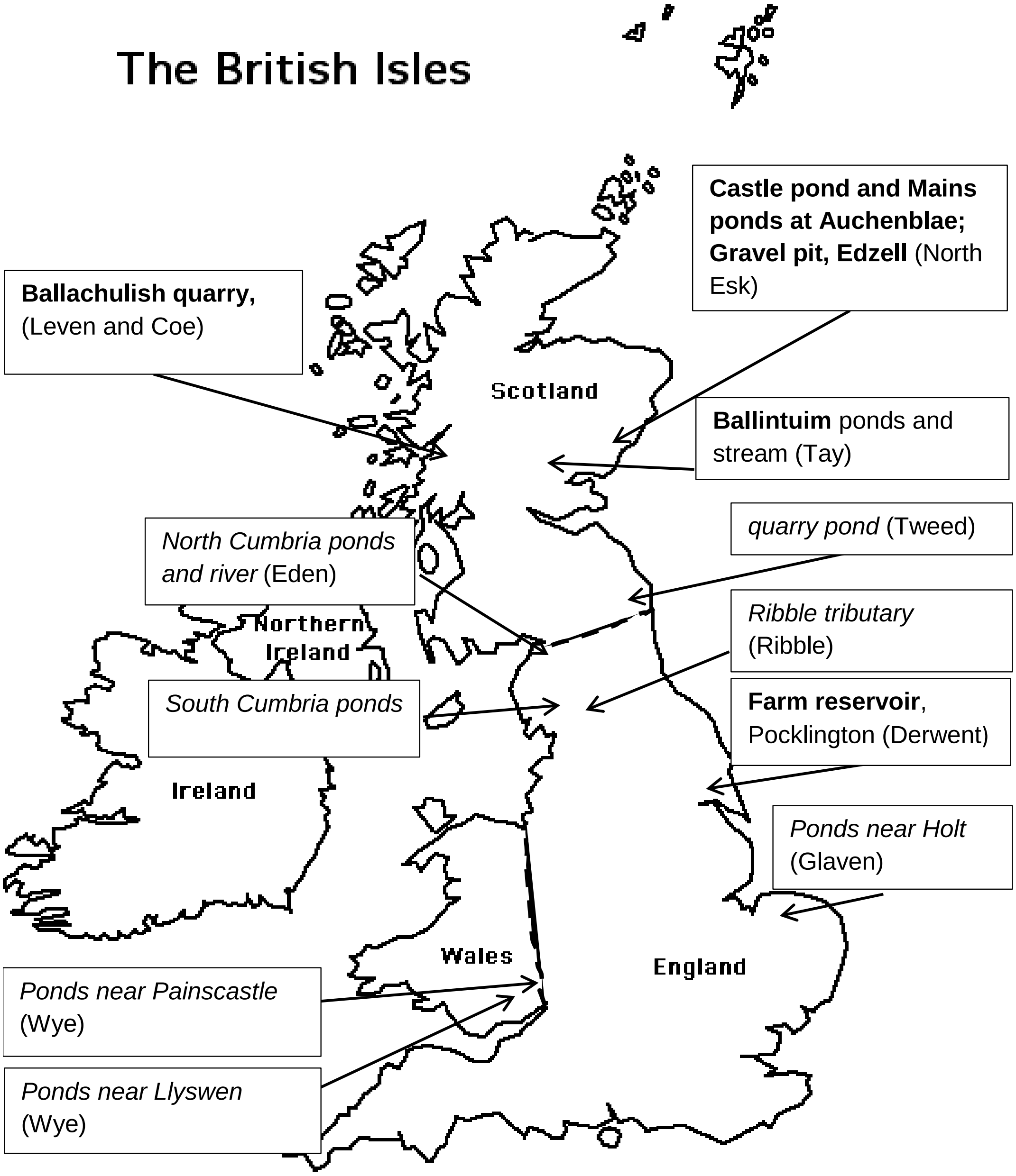

Figure 4.3 Locations of biocide projects and river catchments, names in bold show projects where biocide treatment was carried out, names in italics show projects that did not proceed to biocide treatment

In addition, to check whether the waterbody reached the target concentration of biocide, bioassays were run on water samples that were taken after completion of the application of biocide (Figure 4.1f). The bioassays were run using shrimps *Gammarus pulex*. Although crayfish could have been used as the test organism, the shrimps were more useful for this purpose. There was greater availability of the shrimps locally, they had greater sensitivity to the biocide due to being smaller than the crayfish, but they were large enough for staff to handle easily in field conditions without the need of microscopes and could be kept in tanks in quantity until needed. Crayfish were kept for use in monitoring biocide within the site.

The shrimps were used (10 per test) in a dilution series, with replicates. Mortality in the diluted samples was compared with the toxicity in reference samples of known concentration. The assays were run for 48 hours, but the outcome at 12 hours was the most important, because the information could be used by staff on site to increase the biocide in the event that the site did not reach the target dosage in the areas sampled. Bioassays were carried out at intervals to follow the progressive recovery of sites after biocide treatment. Use of post-treatment bioassays was helpful at some of the sites for confirming to the owners that the biocide had degraded and the site was "safe for shrimps".

To monitor whether there was any release of treated water outside the target areas, on sites that required hydraulic control, aquatic invertebrates were put into monitoring chambers that could be lifted and inspected frequently during the treatment, to see whether mortality occurred and hence whether there was any environmental impact outside the target areas. These monitoring chambers were set in at least two locations downstream of the treated area. *Gammarus pulex* was used in most cases, as it was a component of the fauna at most sites and hence a useful representative of the macroinvertebrate community. At Ballachulish, however, locally-sourced marine copepods *Corophium* were used in the monitoring tubes below an intermittent overflow pipe from the quarry into a sea loch. *Gammarus pulex* would not have been tolerant of the saline conditions there, whereas the *Corophium* were locally abundant.

Table 4.2 Summary of projects where biocide treatment against crayfish was carried out (2004-2012)

Site name	**Gravel pit, Edzell**	**Mains ponds, Auchenblae**	**Castle pond, Auchenblae**	**farm reservoir near Pocklington**	**Ballintuim ponds and stream**	**Ballachulish quarry**
Catchment	North Esk, Scotland	North Esk, Scotland	North Esk, Scotland	Yorkshire Derwent, England	Ardle (Tay), Scotland	Leven and Coe, Scotland
Year stocked with crayfish	c. 1998	c.2002/3	c.2002/3	1992	late 1980s	2000s
Stocked by and reason if known	fishery manager, for fish pond management	fishery manager, for fish pond management	fishery manager, for fish pond management	fishery manager, for fish pond management	owners, as ornamental	children, aquarium release
Year detected	2004	2004	2004	2002	2003	2011
Feasibility study	2004	2004	2004	2003	2005	2011
Year treated	2004	2004 and 2005	2004	2005	2006	2012
Waterbody type	enclosed pond	3 online ponds	1 offline + ditch 100 m	enclosed pond	1 offline pond, 1 online pond, 680 m ditch	enclosed pond

Site name	**Gravel pit, Edzell**	**Mains ponds, Auchenblae**	**Castle pond, Auchenblae**	**farm reservoir near Pocklington**	**Ballintuim ponds and stream**	**Ballachulish quarry**
area, ha	1.0	0.02, 0.15, 0.3	0.54	0.56	garden pond 0.08, lower pond 0.1	2
depth, m	1-2.4	0.2 -2.0	mainly 1.0-2.5	3.4	both 0.5 - 1.8 m, stream 0.1 - 0.25, increased for treatment	0.5-13.0
water source	groundwater	dammed stream	offtake from river	winter fill from river, plus arable land drainage	seasonal inflow, spring seepage	catchment runoff and groundwater
pH	pH 7.0	pH 6.8	pH 7.0	pH 7.8-8.5	pH 7.0	c. pH 7
substrate	gravel	sandy clay	sandy clay	calcareous clay	sandy clay (upper), sand (lower pond)	slate

Site name	**Gravel pit, Edzell**	**Mains ponds, Auchenblae**	**Castle pond, Auchenblae**	**farm reservoir near Pocklington**	**Ballintuim ponds and stream**	**Ballachulish quarry**
siltation	low	peat and silt, up to 0.15 m silt in deep water and wetland	thin silt layer, more at depth	none in steep margins, moderate on bed	upper pond thin silt over butyl lining, lower pond peat silt over sand	low, except in deepest area
vegetation - submerged	moderate cover, 25%	sparse <5%, locally dense>60% at retreatment	<10%, a few patches	sparse <2%	sparse to moderate in garden pond	none
vegetation - emergent	none	locally dense floating and emergent grasses +standing dead trees	scattered clumps of rushes around the margin	tree roots from surrounding willow coppice	*Typha* in garden pond, grass/rushes by stream, wetland at lower pond	minimal <1%

Site name	**Gravel pit, Edzell**	**Mains ponds, Auchenblae**	**Castle pond, Auchenblae**	**farm reservoir near Pocklington**	**Ballintuim ponds and stream**	**Ballachulish quarry**
water temperature at treatment, degrees ^oC, month	9 Late October	15 September	4 December	14 Late September	9-14 October	8-22 June
target dosage, mg/l pyrethrins	0.15	0.2	0.2	0.2	1.0 in ponds, 2.0 in stream	0.5
application method (see notes at foot of table)	margins: 1a, pond: 2a	1st: margins: 1a ponds (1-3): 2a, 2nd: margins: 1a, ponds (2 and 3): 2a,4g, 2d	margins: 1a, pond: 2a	Margins and pond: 2b, 2f	Garden pond: 1a, 2a, 2d; Stream: 1a; Lower pond: 1a, 1d, 2a, 2d,	1a, 1e, 2a, 2c, 2e, 2f
hydraulic control by pumping required	none	inflow diverted; intermittent return pumping from downstream	return pumping of leakage 18-25 l/s from pond. Partly dewatered before treatment	None	complex, recirculation on 5 sections of stream; dewater of ponds post-	intermittent inflows blocked above quarry face

Site name	Gravel pit, Edzell	Mains ponds, Auchenblae	Castle pond, Auchenblae	farm reservoir near Pocklington	Ballintuim ponds and stream	Ballachulish quarry
		sump	(unplanned). Ditch dammed, with recirculation		treatment	
number of days for treatment (excluding prior surveys and monitoring)	3 (x2)	2 (x2)	22	3	26	6
number of staff/day during treatment	2-6	2-7	1-6, + 1 at night	5	main works, 8 - 10 by day, +1 at night, reduced in last phase	4-20
re-treatment[5]	yes, 24 days later (details are for retreatment)	yes, 11 months later (details are for retreatment)	no	No	no	no

Site name	**Gravel pit, Edzell**	**Mains ponds, Auchenblae**	**Castle pond, Auchenblae**	**farm reservoir near Pocklington**	**Ballintuim ponds and stream**	**Ballachulish quarry**
stocked with fish after treatment	no	no	yes	No	yes	no
years of monitoring	5	5	5	5	2	1, ongoing
outcome, crayfish caught in number of years after treatment	0 crayfish, **eradication**	crayfish detected year 2 (middle) and year 3 (upper and lower)	crayfish detected year 2	0 crayfish, **eradication**	0 crayfish in stream and lower pond, crayfish in garden pond year 2 and present year 7	0 crayfish year 1, ongoing

1. <u>Application methods on shallow margins and narrow band on exposed bank</u>: a. backpack sprayers; b. hand-held sprayer boom from boat; c.fixed sprayer boom on boat; d.portable pump to pump treated water from pond onto margins/wetland; e. hand-sousing of margins from treated pond using a bucket.
2. <u>Application methods on pond</u>: a. hand-held sprayer lance or multi-jetboom from backpack sprayers or larger tanks on boat; b. fixed front-mounted sprayer boom on boat; c. pour or pump on biocide via vertical pipe mounted on boat; d. pour-on biocide near the intake of two or more pumps on the bank e. pour or pump on biocide near propeller of boat; f. after

application, two or more pumps on the bank with intake and output hoses set widely apart may be used to induce water circulation to facilitate mixing.

3. Application on stream: as for margins, but natural flow diverted around section under treatment contained between temporary dams, with flow recirculated by pumping throughout treatment period. Stream sections treated in succession downstream overlapped to ensure no gaps in coverage.
4. Timing of application of biocide carried out on one day, but g. top up application used in localised areas next morning if bioassays with crayfish indicate a patch with incomplete application or mixing.
5. Two sites, Gravel pit and Mains ponds had two treatments. First treatment in October 2004 had a preliminary treatment with sodium sulphite to chemically deoxygenate water and encourage emergence of crayfish – effective at test scale, but in field conditions it was poorly dissolved and the residue accelerated degradation of biocide. Re-treatment was carried out with Pyblast alone, as per table.

Table 4.3 Summary of projects where biocide treatment was not carried out

Site name	**ponds near Holt**	**ponds near Painscastle**	**Ribble tributary**	**ponds near Llyswen**	**quarry pond**	**north Cumbria ponds and river**	**south Cumbria ponds**
Catchment	Glaven, England	Bachawy, Wales	Ribble, England	Wye, Wales	Tweed, Scotland	Eden, England	Confidential, England
Year stocked with crayfish	early 1980s	1990s	late 1990s	after 1996	unknown	unknown	unknown
Stocked by	owners, as ornamental	owners, as ornamental	individuals for wild harvest	without owner's knowledge, probably angler	unknown	owners, as ornamental	unknown
Year detected	2007	2005	2002	2012	2009	2012	2011
Feasibility study	2009	2009	2005	2012	2010	2012	2011

Year of work	2009-2010, surveys and detailed planning	2010, planning; 2011-2012, planning, exclusion amphibians	2006-2007 planning, funding, detailed design, surveys; 2008 groundwater tracing, tests on hydraulic control	2012 crayfish survey, cost benefit assessment; experimental dewatering	2011, some planning, 2012 test of partial dewater, re-costed	2012 site appraisal only, considering ark site options for indigenous crayfish	2012 funding, delay, test of partial dewater
Waterbody type	series of small interconnected garden ponds+mill leat	6 ponds (3 linked) dug by river	small headwater stream	two online ponds + ditches and culverts	former quarry	garden pond, small outfall stream, large river	1 site with 2 small ponds, 1 site large
area, ha	0.09, 0.18, + 800m	0.01-0.3	2.5 km	0.5, 1.0	c. 2	<0.3, + watercourses	<0.5 - c. 3
water source	catchment runoff/ spring seepages	catchment runoff/ spring seepages	catchment runoff+baseflow	mill leat+ land drains	groundwater	unknown	catchment runoff/ spring seepages

Table 4.4 Summary of constraints encountered in biocide treatment projects against populations of signal crayfish in Great Britain

Constraints	Gravel pit, Edzell	farm reservoir near Pocklington	Mains ponds, Auchenblae	Ballachulish quarry	Castle pond, Auchenblae	quarry pond	Ballintuim ponds and stream	ponds near Llyswen	ponds near Holt	ponds near Painscastle	Ribble tributary	north Cumbria ponds and river	south Cumbria ponds
not enough benefit	0	0	0	0	0	2	0	3	0	3	0	0	0
large size and/or number of waterbodies to be treated	0	1	1	2	0	2	2	1	2	1	2	3	3
habitat complexity, e.g. wetlands	0	0	1	1	1	1	1	1	2	2	2	1	3
impacts on species in treatment area	0	1	0	0	1	0	1	0	0	2	0	2	1
environmental risks outside treatment area	0	0	1	1	2	0	2	1	1	2	2	3	1
delays due to weather	0	0	0	0	1	1	1	0	0	0	2	0	0
funding insufficient or delayed	0	0	0	0	0	3	0	3	3	1	1	0	3
lack of staff and/or capability	1	1	1	1	1	1	1	1	2	1	1	1	1

Constraints	Gravel pit, Edzell	farm reservoir near Pocklington	Mains ponds, Auchenblae	Ballachulish quarry	Castle pond, Auchenblae	quarry pond	Ballintuim ponds and stream	ponds near Llyswen	ponds near Holt	ponds near Painscastle	Ribble tributary	north Cumbria ponds and river	south Cumbria ponds
difficulties with approvals from statutory agencies	0	0	0	0	1	0	0	0	0	2	3	3	1
landowner or occupier cooperation refused, withdrawn or a constraint on work	0	0	0	0	2	0	2	1	3	0	2	1	2
public objections, actual or potential	0	0	0	0	0	0	0	0	0	0	2	3	2
Total constraints score	1	3	4	5	9	8	10	8	13	11	17	18	18
Stage project reached	4	4	4	4	4	2	4	1	3	3	3	1	1

1. minor, a matter for project planning and action, but not likely to prevent or significantly constrain the project;
2. moderate, a significant constraint causing increased technical difficulty, cost and/or delay;
3. major, sufficient to stop a project on its own

Project stage: 1 appraisal, 2 detailed assessment and planning, 3 preparatory works, 4 biocide treatment

4.2.3 Post-treatment monitoring for crayfish

Post-treatment monitoring of biocide treated sites for the presence of crayfish was carried out by trapping, supplemented at two sites by use of fyke nets with 3 m wings of netting to direct any crayfish to the entrance of the fyke. Details of the traps and results are given in Table 4.5. The effective range of individual crayfish traps in ponds and lakes has been estimated at approximately 12.5 m^2 by Abrahamsson and Goldman (1970) and at 56.3m^2 by Acosta and Perry (2000) in a mark recapture trial in a controlled grid. By area, trap coverage of sites during monitoring (excluding fyke nets) 1 trap in 13 – 100 m^2, generally 1 in 40 – 55 m^2. The worst case (lowest number of traps,12.5m^2) would be 12% simultaneous coverage at the lowest trap density, but assuming the trapping range of Acosta and Perry (2000), monitoring coverage by trapping as a proportion of total area of site was 50-100% or more each year.

4.2.4 Review of factors that influenced the projects

Important constraints that had to be considered during the appraisal and the subsequent stages of each project have been given a qualitative ranking of importance on the following scale:

1. minor, a matter for project planning and action, but not likely to prevent or significantly constrain the project;
2. moderate, a significant constraint causing increased technical difficulty, cost and/or delay;
3. major, sufficient to stop a project on its own

For comparison, constraint scores for projects were summed and ranked. A Spearman rank correlation test was used to test the effect of constraints on the stage a project reached.

4.3 Results and discussion

4.3.1 Biocide treatment

The sites that were treated with biocide were monitored for up to five years to assess the outcome, using a high density of traps each year. The catches before and in successive years after treatment are shown in Table 4.5. Of the six projects where treatment was carried out, two (Gravel pit, Edzell and farm pond near Pocklington) were considered to have been successful in eradicating the crayfish, as no crayfish were caught despite five years of intensive annual trapping. For the third site, no crayfish were trapped in the first year following treatment, but full 5 year post-treatment monitoring has yet to be completed. For the remaining three sites, crayfish were captured during subsequent monitoring, indicating that the population had not been completely eradicated.

The successful projects were also the simplest sites; both were enclosed ponds with no surface water flow and with little or no emergent vegetation and relatively little silt, although the gravel pit had moderately abundant cover of African curly waterweed *Lagarosiphon major*, another introduced invasive species. The sites differed in other characteristics, as the farm reservoir was dug in calcareous clay (pH 8 -8.5), was filled by pumping from a river and received some drainage from arable land. By contrast, the former gravel pit was oligotrophic, with groundwater. The reservoir had a long established, dense population and the banks and willow roots had many burrows. Treatment was effective despite this, possibly because the steep, burrowed banks above the water level were dry at the end of the summer, such that all the crayfish were in refuges in the water.

The third enclosed site was the largest, a former slate quarry. The main issues with that site were the exceptional abundance of potential refuges at and below water level, due to extensive cover of loose slate. The pond extended to nearly 15 m depth and this caused some problems with the application, because of the large temperature difference between surface and deep water, which appears to have slowed the rate at which biocide sank to the bottom. Slow diffusion and degradation of natural pyrethrum in the upper layers risked incomplete treatment, so an additional application was made over the deepest areas, using vertical pipes to place the biocide to 2-3 metres below the surface. Monitoring crayfish in cages and use of an underwater camera suspended above the bottom in the deep holes confirmed that the combined applications were successful in killing all the test crayfish at the bottom. The final outcome on the Ballachulish quarry is not known, although monitoring in summer 2013, one year after treatment (250 trap nights) recorded no crayfish. Monitoring in subsequent years is required to see if the project has been successful.

Table 4.5 Crayfish surveys before and for up to 5 years after biocide treatment with natural pyrethrum (Pyblast) at five sites, a. gravel pit, Edzell; b. Castle pond, Auchenblae; c. Mains ponds, Auchenblae; d. Ballintuim ponds (garden pond and lower pond) and stream, and e. farm reservoir near Pocklington

a. Gravel pit		
Year	Number of crayfish	Number of trap nights (Catch per unit effort)
2003 (29th Aug)	150	10 (15)
2004 (3rd-4th Oct) (before)	241	80 (3)
2005	0	100 (0)
2006	0	111 (0)
2007	0	125 (0)
2008	0	115 (0)
2009	0	108 (0)

b. Castle pond		
Year	Number of crayfish	Number of trap nights (Catch per unit effort)
2003 (29th Aug)	na	na
2004 (3rd-4th Oct) (before)	16	20 (0.8)
2005	0	43 (0)
2006	1	109 (0.01)
2007	0	100 (0)
2008	13	60 (0.22)
2009	5	60 (0.08)

c. Mains ponds, Auchenblae						
	Upper Mains pond		Middle Mains pond		Lower Mains pond	
Year	Number of crayfish	Number of trap nights (Catch per unit effort)	Number of crayfish	Number of trap nights	Number of crayfish	Number of trap nights (Catch per unit effort)
2004 (before)	0	5 (0)	0	5 (0)	2	10 (0.2)
2005 (no retreatment, nearly dry wetland)	na	na	0	10 (0)	3	25 (0.12)

2006	0	8 (0)	0	12 (0)	0	90 (00
2007	1	15 (0.07)	1	45 (0.02)	0	75 (0)
2008	2	8 (0)	5	30 (0.17)	3	75 (0.4)
2009	0	15 (0)	11	30 (0.37)	0	75 (0)

d. Ballintuim						
	garden pond		stream		lower pond	
Year	Number of crayfish	Number of trap nights (Catch per unit effort)	Number of crayfish	Number of trap nights (Catch per unit effort)	Number of crayfish	Number of trap nights (Catch per unit effort)
2006 (before)	109	45 (2.4)	5	manual search	0	38 (0)
2007	0	96 (0)	0	10 (0)	0	40 (0)
2008	1	100 (0.01)	0	10 (0)	0	40 (0)
2013	frequen t	owners report	none seen	owners report	none seen	owners report

e. farm reservoir near Pocklington		
Year	Number of crayfish	Number of trap nights (Catch per unit effort)
2003 (before)	95, 208	51 traps + 6 fyke nets (1.8 traps, 34.6 fyke nets)
2005 (before)	389, 226	8 fyke nets, 2 times (46.2, 44.5)
2006	0	88 traps +9 fyke nets (0, 0)
2007	0	107 traps +6 fyke nets (0, 0)
2008	0	101 (0)
2009	0	100 (0)
2010	0	101 (0)

Three projects did not achieve the target, as some crayfish survived and hence the populations recovered subsequently. It is not possible to state categorically why they failed, but the most likely reason is that in each case some individuals remained in areas that could not be adequately dosed. The likely circumstances illustrate problems that are likely to occur at other sites: difficulty in detecting limits of population, possible extension of range of crayfish prior to treatment, aquatic vegetation and survival in banks at or above water level.

The Mains ponds consisted of three earth dams on a small stream. According to the fishery manager, who carried out the illegal stocking, crayfish were only put in the lower pond. A survey prior to treatment only recorded crayfish in the lower pond, but the other two ponds were treated on a precautionary basis, even though the projecting outfall pipes through the dams between the three ponds made it unlikely that crayfish could use that route to move upstream from the lower pond. In the first biocide treatment at the Mains ponds and the gravel pit, a chemical deoxygenation pre-treatment reduced the effectiveness of the natural pyrethrum and both sites had to be re-treated. Re-treatment at the Mains ponds was delayed until the following year, but during the winter, a problem with the outfall from the middle pond during a flood led to owner excavating a small channel through the dam to release excess water (it was repaired after the re-treatment). When the re-treatment was carried out, with natural pyrethrum alone, the ponds were all heavily vegetated. The small upstream pond had shrunk to little more than a marsh due to the dry summer and was not re-treated, although particular attention was given to application of biocide in the other two ponds, especially in the areas of submerged and emergent vegetation, small marshy islands and standing dead trees in the lower pond.

A crayfish was found in the middle pond and another in the upper pond in the second year after the re-treatment. In the following year three crayfish were removed from the lower pond close to the channel from the middle pond, although none were found in the lower pond in year 4, whereas the population in the middle pond had increased. It is possible that crayfish were in all the ponds from the start and treatment was not complete, probably because the biocide was partly intercepted on floating grasses and other vegetation and did not reach target concentration at the bed in all areas. If the fishery manager was truthful and did indeed only stock the lower pond, crayfish that survived the defective first treatment may have gained access to the middle pond through the new channel cut into the dam and then into the upper pond, a few metres overland via wet grassland. Alternatively, crayfish may have climbed up through damp vegetation into the marshy upper pond even earlier and remained at low density upstream, until they recolonized the middle pond.

The risks of failing to treat an occupied area need to be assessed carefully in project planning. A precautionary approach is recommended, especially at the upstream end of any site to be treated. The difficulties of treatment in dense vegetation could be overcome by carrying out a herbicide treatment in advance, or dredging, provided care is taken to prevent escape of crayfish. Where vegetation could not be removed, high volume application, e.g. with a pumped jet rather than a fine spray, would help biocide-treated water to penetrate through vegetation. Use of a drop pipe to put biocide below the surface would also help to avoid floating-leaved vegetation.

The Ballintuim site had problems with vegetation in the large garden pond, where there were rushes and other herbaceous vegetation overhanging the banks, but cutting back any plant was forbidden by the owner and at best leaves could be only gently moved aside. There was seepage into the pond along one bank. This could have been intercepted by cutting a ditch along a grassy path near the bank, if permitted. A crayfish was found in the second year, in a trap adjacent to a bank with the seepage and large stones, which, by its size, must have avoided or recovered from the biocide treatment. At the lower pond at Ballintuim, where the owner was less concerned about disturbance, the pumped jet was used apply biocide to the margins to improve treatment.

At the third pond, Castle pond, the problem was leakage through old field drain in the bed of the pond, such that continuous pumping was required to prevent treated water escaping to a nearby river. The aim was to maintain the existing level of the pond throughout treatment. Unfortunately, when the owner attempted to block the leaking drains, the works increased the leakage and the water level fell. There was not enough time to fully re-fill the pond prior to treatment, as issues with hydraulic control had already delayed treatment until December and work had to be done in the available window of dry weather. A narrow band of marginal vegetation and pond bed was left exposed. A crayfish was caught in the second year (in the last of 109 traps inspected) and the number caught increased in the following years. Possible reasons for failure at this site are as follows.

If any crayfish were exposed by the reduction in water level a couple of days prior to treatment, the cold temperature (water 4^{o}C) and colder air temperature at night may have discouraged their movement. Unlike the successful case at the farm reservoir, crayfish may have remained above water and avoided biocide treatment. Alternatively, crayfish may have all been in the pond, but the amount of clay and silt disturbed by leakage through the bed, collection and return pumping may have increased the rate of degradation of natural pyrethrum compared to cleaner conditions. Monitoring the treatment using 20 cages of crayfish (n=116) showed relatively slow mortality, only 55% after 48 hours, although all were dead by 96 hours after treatment, with no further exposure. Furthermore, there was complete mortality in a cage of crayfish that was set in the pond three days after biocide treatment, hence toxicity must have persisted until then, despite the pumping. Degradation of natural pyrethrum is slower in cold conditions, there was no sunshine and short day length and as the metabolism of crayfish is also slower in the cold, making recovery less likely. On balance, crayfish left on the exposed margin is the most likely reason for failure in this case.

Only two sites with running water have been treated. One was a length of outfall ditch at Castle pond. The other was nearly 700 m of a small stream at Ballintuim, from upstream of the garden pond to the lower pond. This was by far the most complicated treatment, involving bypassing flow around each section being treated and flushing and dewatering recovering sections, (recovery took 4 - 10 days per section). Five sections were treated, each overlapping slightly to ensure thorough coverage (see treatment of section 3 in Figure 4.2). The banks were mowed along the stream to facilitate treatment here, sandbags were used within sections to raise water level to flood undercut banks and a few seepages on the banks were also treated. Prior to treatment crayfish were recorded at low abundance downstream of a long piped outfall from the garden pond (the drain was later flushed out with biocide). None were found in the two steep sections of watercourse upstream, before or during treatment, indicating recent escape from the garden pond and preferential movement downstream. Approximately 200 m of stream downstream of the pond outfall had been channelized previously, although the sides had slumped since, and the owner asked for the ditch to be re-dug after the biocide treatment. This allowed all of the undercut areas to be dug out, spread onto plastic sheeting and searched for crayfish. No live crayfish were found and only one dead one, the rest having been previously washed out and collected from the channel after biocide treatment. Monitoring by manual search and trapping was continued for two years, with no crayfish found. The excavation combined with subsequent monitoring makes it likely that treatment in the stream was successful, but as crayfish remained in the garden pond, there was potential for future re-colonization.

4.3.2 Post-treatment monitoring

Of the six treatments, two showed no evidence of crayfish five years post treatment, and another site showed no crayfish after one year, with post treatment sampling ongoing. These data suggest that biocide has the potential for effective eradication in an enclosed system.

Deciding when eradication has been achieved is always a problem, because it is only possible to be sure of presence, rather than absence. Nonetheless, there are reasonable grounds to think that in this case the monitoring effort applied for five years would have been sufficient to detect a population at the gravel pit or farm reservoir if they had not been extirpated, for the following reasons. Firstly, the presence of signal crayfish at the Castle pond and Mains ponds was detected within three years of introduction, with a much lower intensity of trapping. When eradication was not achieved at those sites, crayfish were detected in two or three years. At the gravel pit, if biocide treatment had failed, a further two cohorts of crayfish would have been produced by year 5, potentially up to five cohorts. The population would be fast growing in the absence of competition and with no fish present, crayfish would reach maturity in one to two years and those of 1+ or more were caught in the fine-mesh traps used at the unsuccessful sites. With the population expected to be many 1000s within five years, even if trapping efficiency was in the range 1-10%, there would be a high probability of detecting crayfish with saturation-coverage of traps.

With the removal of crayfish by biocide treatment, amphibians increased at the mains ponds and the gravel pit. Several types of crayfish trap were used (see Figure 4.4), but the fine-mesh traps (4 mm mesh, traps with large mesh size did not retain amphibian larvae) retained larvae of common toad (*Bufo bufo*) and palmate newt (*Lissotrichon helveticus*). Bycatch of amphibians is shown in Table 4.6. The proportion of traps with amphibians increased and the relative abundance. Counts of toads in the traps were only estimated in 2006 and 2008, because they were so abundant (Figure 4.5). The apparently lower count of toads in 2009 is a reflection of the surveyors trying to avoid the areas of highest density of toad larvae, which were dense enough along many 10s metres along the southern and eastern margin to completely hide the substrate to about 0.5 m from the edge. The counts of palmate newts in traps were compared for each year using a Kruskall-Wallis test, which showed there was a significant difference between years ($n = 116$, $df = 2$, $K = 42.485$, $p < 0.001$). Increase in common frog *Rana temporaria* was also seen, with numbers increasing from scattered occurrence in 2005 to densities exceeding 10s to 100s m^{-2} of recently emerged young frogs on the exposed margins in 2009. Amphibians bred successfully in the ponds even in the first breeding season after treatment, as the ponds had recovered from biocide application. Whilst full prior amphibian surveys were not carried out at the North Esk sites, due to the short time (less than 3 months) between site appraisal and treatment at the first site, the increase in amphibians at the sites is likely to be due to the cessation of predation by crayfish and fish and possibly due to the short-term reduction in predatory invertebrates.

Table 4.6 Recorded bycatch of amphibian larvae in fine-mesh crayfish traps at the gravel pit, Edzell, in years following biocide treatment

Year	**2006 (year 2)**	**2008 (year 4)**	**2009 (year 5)**
Survey date	09 June	21 June	29 July
Number of fine-mesh traps recorded (total number of traps)	64	35	17
Number of traps with toad larvae	43 (67%)	28 (80%)	17 (100%)
Total number of toad larvae	Low 1000s	>3000	1300
Number of traps with palmate newt larvae	4 (6%)	17 (50%)	13 (76%)
Total number of palmate newt larvae	4	35	36
Average number of newts per trap	0.06	0.97	2.12

Figure 4.4 North side of Castle pond, Auchenblae with traps used for post-treatment monitoring. Types shown are Trappy® (yellow cylinder), Trappy Tetra® (pyramid) and GB Nets fine-mesh (green cylinder)

Figure 4.5 Margin of the gravel pit, Edzell during post-treatment monitoring, densely covered by tadpoles

4.3.3 Project constraints overcome, or not

Any large field-scale project is likely to encounter potential problems, which need to be overcome if the project is to progress from planning to action and successful outcome. Where the technique is new, experience has to be gained by practice. The problems encountered in the crayfish biocide projects are summarized in Table 4.4. All the projects had one or more constraints and even two of those where the project progressed to treatment had constraints rated as moderate, i.e. causing a significant increase in technical difficulty, cost or delay. The frequency of different categories of constraint is shown in Figure 4.6.

The most frequently occurring constraint was staff experience, in that with the exception of the three sites in the North Esk catchment, which were run jointly, there was a different project leader and team for each one and as a novel technique, none of them had any prior experience. The constraint was kept to minor because the author provided continuity as an adviser to all of the projects (except the appraisals in Cumbria). Furthermore, in the most recent example (Ballachulish), the project was used as a training opportunity for staff of several of the Rivers Trusts in Scotland, to increase their capacity to assess and respond to future cases. In one project, the limited availability of the project manager was a moderate constraint, but the reason the project could not be carried out was the funding was not available to fund the project at the scale required. In that project, treatment was considered in two phases, but there were two landowners involved; one who did not give consent and one, at the site of introduction, who withdrew consent shortly before the treatment. Having lost the opportunity to eradicate signal crayfish from that catchment, effort is now focused on salvaging at least part of the catchment as an ark site for the white-clawed crayfish.

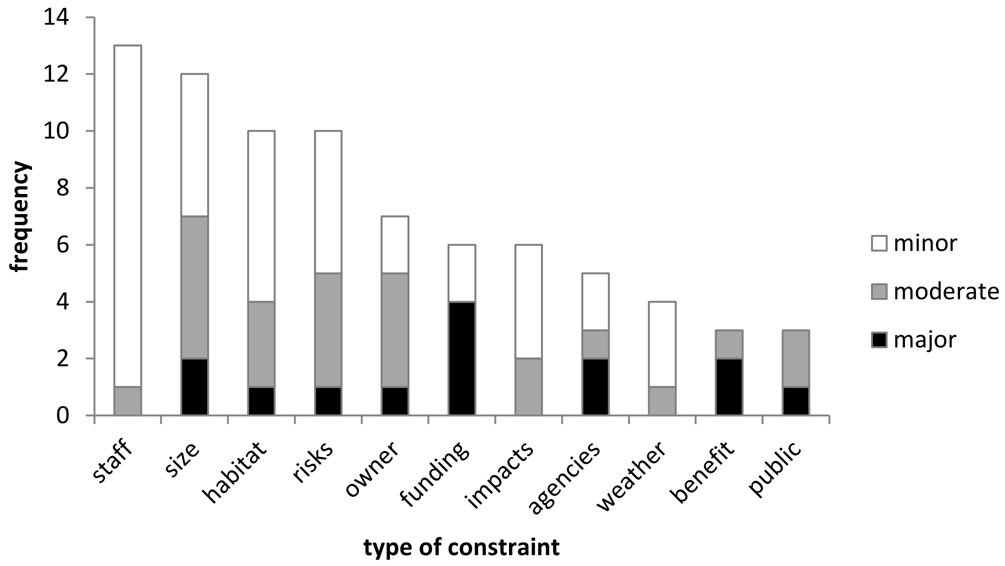

Figure 4.6 The frequency of occurrence of project constraints on biocide treatment by type and magnitude

Type:
staff = insufficient staff and/or lack of capability;
size = waterbody large, multiple and/or extensive;
habitat = habitats requiring treatment complex, e.g. extensive wetland;
risks = environmental risks outside the treatment area/pollution risk;
owner = landowner or occupier cooperation refused, withdrawn or a constraint on work;
funding = funding insufficient and/or delayed;
impacts = on species in treatment area, e.g. protected species present;
agencies = difficulties with approvals from statutory agencies limit or delay work (usually due to other issues);
weather = work prevented or delayed due to poor weather;
benefit = not enough benefit compared to cost or risk, e.g. due to other populations of invasive species locally that cannot be treated;
public = public objection, actual or potential, to biocide treatment, generally due to lack of understanding and/or concern about risks/benefits
Magnitude:
major = sufficient to prevent a project on its own;
moderate = significant constraint causing increased technical difficulty, cost and/or delay;
minor = requires project planning and action, but not likely to prevent or significantly constrain the project.

Obtaining funding was a major factor in several projects (in England), but in combination with the scale or complexity of the projects. This was exacerbated by funds having to be sought regionally, rather than nationally, which incurred delays of 6 months to more than 2 years in some cases. In the case of the Ribble catchment, which was the most ambitious in tackling a stream, substantial funding was obtained in within two years, but further delays occurred due to two particularly wet summers, although some technical work was carried out late in both seasons in preparation. Obtaining the cooperation of approximately 20 landowners and occupiers was an achievement on the part of the project leader, but an issue raised by a land agent about a water supply necessitated further investigation and yet more delay occurred, until it could be demonstrated that there was no risk to public water supply. In the meantime, the invasion continued, the crayfish reached the confluence with a larger watercourse, after which the goal of eradicating signal crayfish from the whole catchment could no longer be achieved and the project was abandoned, six years after the population was detected. In retrospect, if the knowledge gained from this and other biocide treatments had been available when the population was first detected, there is a reasonable chance that eradication could have been achieved.

As the number and magnitude of constraints increased, so the projects were less likely to proceed (Figure 4.7). A Spearman rank correlation test showed a significant correlation between the sum of constraint scores and the stage reached (Spearman Rank correlation: $n = 13$, $r = -0.633$, $P = 0.02$). The projects that had longer times between detection and a feasibility study also had longer times between feasibility and the treatment or the decision not to proceed further (Spearman rank correlation: $n = 13$, $r = 0.846$, $P = 0.01$).

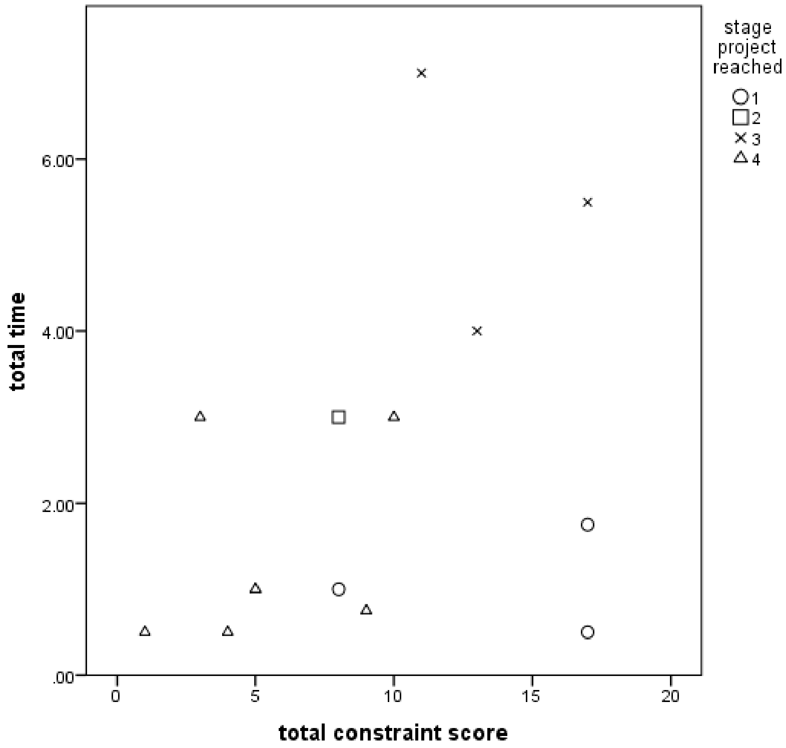

Figure 4.7 The effect of constraints on the total time in years between detection and the treatment on site or the decision not to proceed and the stage reached in biocide treatment projects

Stage 1: appraisal, stage 2: detailed feasibility and planning, stage 3: preparatory works on site, stage 4 biocide treatment. Constraints in 11 categories scored as 1 minor, 2 moderate, 3 major.

Projects that were clearly too difficult or not feasible were generally dismissed at an early stage (stage 1 appraisal). Those with only minor constraints, or moderate ones that could be overcome during planning, were able to proceed to treatment in a relatively short time, 3 (the sites in the North Esk) in less than a year and the others less than 3 years between detection and treatment. A third group of projects had potential to be treated, but were beset by problems, such as owner cooperation or funding constraints and these incurred the longest delays, in some cases with problems increasing over time, until eventually a decision had to be taken not to proceed with the project.

The issue of benefits and impacts arose in some projects. Where there was the chance to free a river catchment from invasive crayfish, the long term benefit was obvious. In the case of the Ballachulish quarry, the part of the catchment potentially susceptible to invasion was minimal, as there was only occasional overflow from the quarry site via an outfall into a sea loch, which would be unlikely to support a breeding population of signal crayfish.

Nonetheless, the site is popular for public recreation, hence the main concern was the risk of illegal removal of crayfish and their introduction to other catchments, this being the only known site with crayfish on the west coast of Scotland. It was agreed by all the stakeholders as a worthwhile project to help protect other catchments and it had the benefit of offering a training opportunity in biocide treatment, to increase the potential capability of staff in other areas of Scotland.

Gaining experience to deal with future projects was the main reason for considering the two projects in Wales, because signal crayfish were already beyond control in the river. As it became apparent that treatment would be more costly than budgeted, the agency involved decided not to pursue the projects. One of the constraints that increased the cost was the requirement to carry out a translocation of a population of great crested newt *Triturus cristatus*, a European protected species, which has strict legal protection, although it is widespread in the Wye catchment. Removal of signal crayfish would have benefited conservation of the newts and other amphibians in the ponds in the medium term, because of the removal of the impact of predation by invasive crayfish; as seen in this study, following biocide treatment in Scotland and in other studies with red swamp crayfish (Cruz et al., 2008, Nunes et al., 2010, Ficetola et al., 2012). This net benefit would have justified the short term fully recoverable impact, but the statutory agencies decided it did not remove the requirement to carry out the translocation. This increased the cost of the project and delayed work by at least a year. In that project, it did not affect the overall outcome, but in other circumstances the delay incurred could have reduced or lost the window of opportunity for eradication.

In north Cumbria the signal crayfish population was detected in the catchment of the River Eden Special Area for Conservation, a river which is designated under the Habitats Directive of the European Union as an area of international importance and supports a wealth of biodiversity, including what is probably the most extensive population of white-clawed crayfish remaining in England. There would be an overwhelmingly strong case for its protection. If the population of signal crayfish had been detected sooner, the ornamental pond where they were introduced and an outfall and small stream would have been capable of being treated with biocide. The crayfish remained undetected until identified in a routine fisheries survey in a larger tributary. The size of this watercourse and the uncertain extent of the population in it would make biocide treatment very expensive and there would be a considerable risk of incomplete treatment. Given the designated status of the river and the expected public objection to an expensive, high impact treatment on a river of high quality, the environmental agencies recognised that biocide treatment was not a feasible management option. The culprits may avoid prosecution for their illegal introduction, because of concerns that it would draw public attention to the population. Wild harvesting of signal crayfish is not allowed anywhere in Northwest England, but unconsented harvesting is a potential risk, with its associated possibility of introductions to other sites, or the spread of crayfish plague to white-clawed crayfish elsewhere in the catchment.

As an aid to help project managers assess possible future projects, the lessons learned from these projects have been used in two schematics prepared in this study. Figure 4.8 shows a basic appraisal of the likelihood of being able to extirpate crayfish at a site where a population has been newly detected. As the responses to questions in the chart progress to the right the projects become more difficult. The outcomes shown at the bottom of the flowchart are projects that should be technically feasible, but they become more difficult and expensive to the right. The outcomes along the top are greatest in difficulty, cost and risk and are likely to be ruled out at an early stage unless there are exceptional circumstances.

Figure 4.9 follows on from Figure 4.8 on the basis that a project appears to be technically feasible. This flowchart gives guidance on assessing the benefit of treatment, largely based on the existing status of crayfish in the catchment and the opportunity to avoid negative impacts on features of importance for biodiversity. The outcomes at the bottom have the highest priority for action and it would be recommended that all such cases would be taken to detailed feasibility and if possible to treatment. Progressing to the right in the flowchart the gains are less and the priorities would shift towards mitigation measures, such as ark sites for indigenous species and public awareness-raising to improve biosecurity. Whilst this does not preclude biocide treatment, decisions may need to be taken to determine management action that will provide the best short term and long term benefits overall.

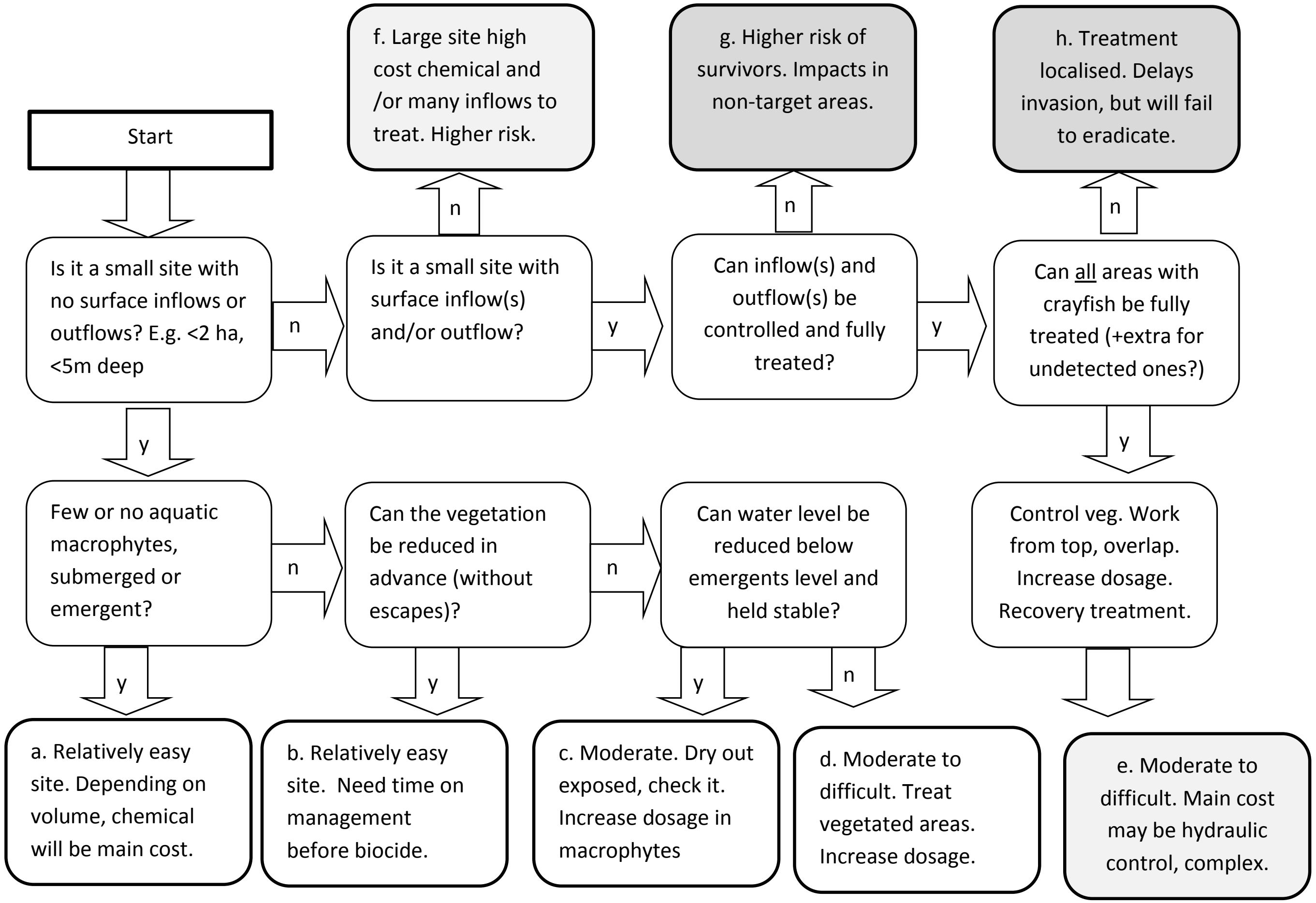

Figure 4.8 Schematic flowchart for initial appraisal of potential to carry out a biocide treatment on a site with an unwanted population of signal crayfish

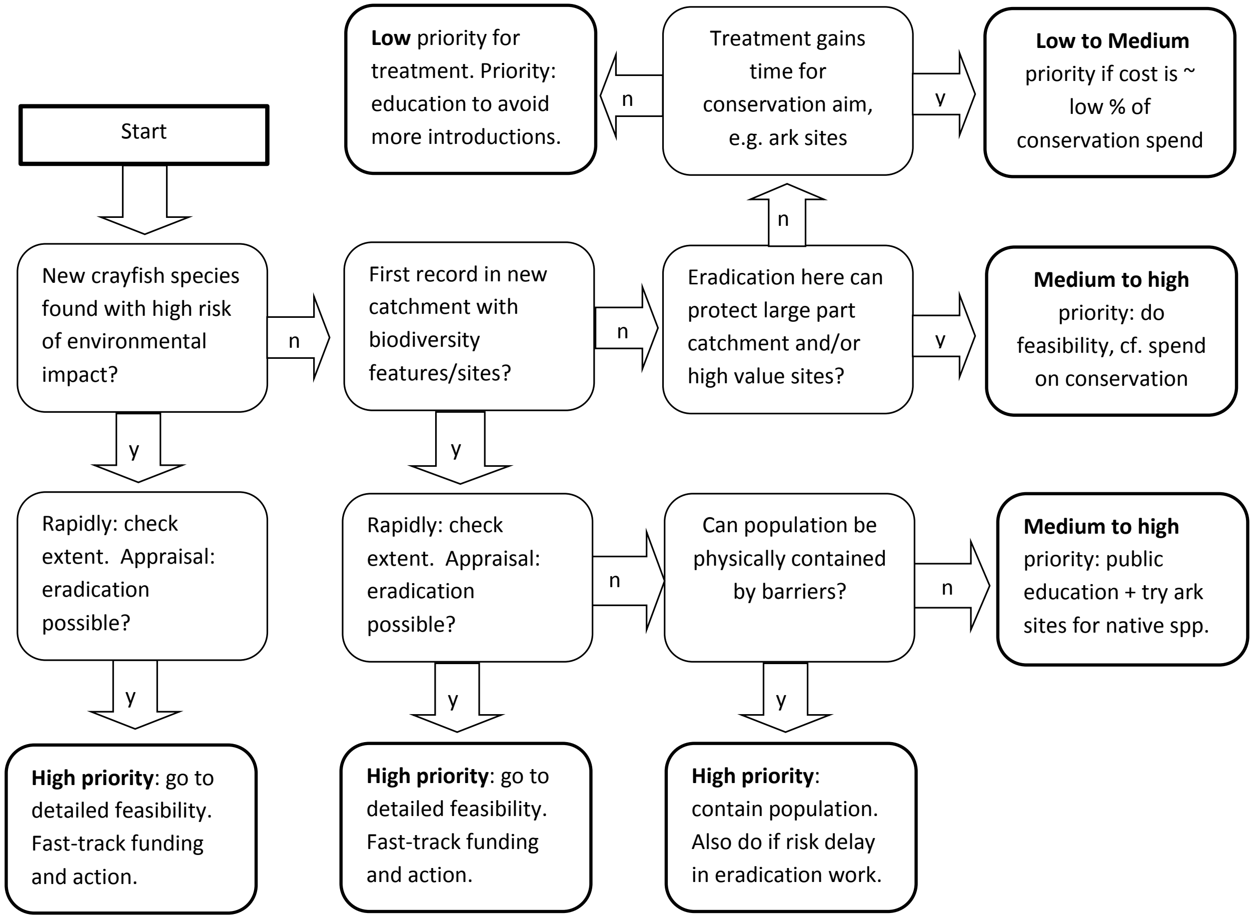

Figure 4.9 Schematic flowchart for prioritising a potential project of biocide treatment against unwanted populations of crayfish

4.4 Conclusions

Hulme (2006) levied the criticism that too much time was spent in invasion biology quantifying the problem and not enough time on delivering solutions. This study has taken a pragmatic approach to developing a method that environmental agencies and other resource managers can use, in at least some cases, to eradicate populations of one of the major aquatic biological invaders in Europe. Hulme (2006) recommended that in attempting to manage a biological invasion factors to consider should include the technical options available, the ease with which the species can be detected and targeted, the risks associated with management, the likelihood of success and the extent of public concern and stakeholder involvement. This study has confirmed the importance of these factors to management of invasive crayfish. Likewise, Myers et al. (2000) considered that there were six requirements for successful eradication, summarized here as: 1. Funding for the whole programme, 2. Authority to act, 3. Susceptibility to treatment, 4. Prevention of re-invasion, 5. Detectability at low density and 6. Awareness of possible future restoration or management requirements after treatment.

There have been good examples of such factors operating in this study. For example, when project funding was increased in some cases when it became evident the scope of work was greater than expected, although in other cases the time to obtain funding caused serious delay. Secondly, authority to act has been a constraint on several of the projects, when one or more landowner was not willing to cooperate, or imposed restrictions on actions and timing. Now, however, the Wildlife and Natural Environment (Scotland) Act 2011 provides new powers to control the release or keeping of invasive species, which could be used to require compliance if landowners are unwilling to cooperate with approved control or eradication projects. The third and fourth factors of susceptibility and prevention of re-invasion relate to the initial decision on the extent to be treated and the effectiveness of treatment within that area. This study showed the importance of making sure all individuals are in places that are susceptible to treatment, in order to prevent re-invasion, which was the reason for failure in the three unsuccessful projects. The fifth factor, detection at low density is important both in defining the areas to be treated and in assessing the outcome of treatment in subsequent years. Finally, although not the focus of this study, there were indirect effects of biocide treatment. The benefits to amphibians have been described, but there were other indirect effects. At the mains ponds the biocide treatment was followed by an increase in growth of submerged macrophytes, which had been scarce when crayfish were present. This increase was probably mainly due to the reduction in crayfish, because the on-line ponds were quickly recolonized by aquatic invertebrates. Although the ponds were not re-stocked with farmed brown trout *Salmo trutta*, there was a change of management to use as release ponds for ducks that had been reared for shooting and a pair of swans bred on the site. Grazing by the wildfowl led to reduction of all the increased growth of macrophytes within a year and increased growth of algae through eutrophication. The impact of wildfowl on aquatic macrophytes is as seen in other studies, (e.g. Hidding et al., 2010, Gayet et al., 2011, Kim et al., 2013).

Drawing from a wide range of eradication projects around the world, Pluess et al. (2012a) found that only the spatial extent of invasions was significantly related to outcome and that is in accord with the findings here. It did not matter that the population at the farm reservoir had been established for more than a decade before it was detected, because it was still within the confines of the site, whereas it was important to attempt to treat Castle pond before the population reached a critical density, because there was only about 100 m of small watercourse before the confluence of a larger river. From their wide-ranging review, Pluess et al. (2012b) also emphasised the value of starting a campaign within four years of detection. Although this study deals with individual sites, the comparison of projects that proceeded to treatment and those that did not showed that similar delay of three or four years meant a project probably had too many difficulties to proceed.

The projects in this study show that it is possible to carry out an eradication treatment against non-indigenous signal crayfish using natural pyrethrum, provided that they are still in a relatively small site. Although few in number as yet, the successful treatments are encouraging and the experience gained from unsuccessful treatments has highlighted the difficulties in complex sites and the scope to improve the likelihood of success in future projects.

Scotland has the greatest number of un-invaded catchments and hence the greatest need to tackle newly established populations if the opportunity arises. There is also public support for eradication projects in Scotland (Bremner and Park, 2007). The River and FisheriesTrusts Scotland aim to make use of the experience gained from these projects to develop the capability to assemble project teams rapidly and effectively to tackle new cases, with the support of Scottish Natural Heritage, which initiated the biocide treatment work and the Scottish Environmental Protection Agency, which now has responsibilities for aquatic invasive species. This approach has worked in England, with the expertise developed on treatment of populations of invasive fish (Britton et al., 2011).

Of the regions within the UK, only Northern Ireland and the neighbouring Republic of Ireland remain completely free from invasive crayfish to date (Reynolds et al., 2002b, Holdich et al., 2009c) and efforts to prevent introductions of non-indigenous crayfish there are important for both the UK and the Republic and in the context of conservation of white-clawed crayfish within Europe, but if the worst happens, all necessary resources should be applied to eradicate any population of crayfish there with the greatest possible speed.

Even though there may be some reluctance to tackle an eradication project where there is some uncertainty about future success and hence the benefits are not guaranteed, nonetheless, the precautionary approach is to make use of the window of opportunity for eradication while it is there, because with invasive crayfish, it can be predicted with confidence that once established and beyond the feasible range of eradication treatment, the population will continue to expand until it occupies all accessible areas of a catchment, with all the associated impacts. Alas, this is an all too common theme with aquatic invaders.

4.5 Acknowledgements

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Chapter 5
The response of the invasive signal crayfish *Pacifastacus leniusculus* to experimental dewatering of burrows and its implications for management of crayfish

5.1 Introduction

The introduction and spread of invasive species into new regions and habitats is one of the major threats to global biodiversity (Millennium Ecosystem Assessment, 2005), as invasive species have modified habitats and ecosystem processes and outcompeted indigenous species, especially so in freshwater ecosystems (Sala et al., 2000, Gherardi, 2007a). In Europe, the introduction of several species of crayfish from North America has led to progressive loss of populations of indigenous crayfish (Holdich et al., 2006a), mainly due to transmission of crayfish plague *Aphanomyces astaci* Schikora, 1903 (Diéguez-Uribeondo et al., 2006, Holdich et al., 2009d). For example, in the UK non-indigenous crayfish are now present in most of the river catchments in England and Wales (Rogers and Watson, 2011) and are present in some in Scotland too (Gladman et al., 2009). The non-indigenous crayfish have also been found to have a wide range of impacts on habitats and communities by burrowing activity, grazing of aquatic macrophytes, and predation of benthic invertebrates and juvenile fish (Gherardi, 2007b).

Measures to prevent further introductions of non-indigenous crayfish have had limited success in many European countries and human-assisted introductions continue (Peay, 2009b). Measures have been proposed to eradicate localized populations of invasive crayfish, or control population density with the aim of mitigating the adverse impacts of invasion. Measures considered include mechanical removal, biocide treatment, male sterilization and biological control with pathogens or parasites (Holdich et al., 1999, Freeman et al., 2010, Gherardi et al., 2011). Of the various trials of methods so far, however, only non-selective biocides have eradicated any populations of non-indigenous crayfish. Some of the projects have been successful against populations of signal crayfish *Pacifastacus leniusculus* (Dana, 1852) (Peay et al., 2006a, Sandodden and Johnsen, 2010), but results have not been consistent between sites (Chapter 4).

One of the problems for any eradication method is reaching all of the population. When a biocide is used as an eradication treatment against crayfish, or indeed any other invertebrates, all individuals need to be exposed to a lethal dose. This is made more difficult with crayfish because of their use of refuges. Whereas invasive fish species are typically found in open water, many crayfish species use natural refuges or excavated burrows to shelter from predatory fish or birds by day and emerge to forage at night. If a biocide treatment is used, the chemical product may not reach the target dosage within refuges, especially if they are deep and if the product used has relatively short environmental persistence. Consequently, there is a risk that the application of biocide may be less effective if crayfish are in burrows and especially if they are out of water, because the refuges need to be inundated for the biocide to come in contact with the crayfish. A trial of natural pyrethrum on red swamp crayfish *Procambarus clarkii*, (Girard, 1852) was unsuccessful because the deep burrows made by this species above the water level could not be treated effectively and individuals in burrows survived, even though mortality was high in treated water (Cecchinelli et al., 2012). A population of signal crayfish in an online pond that was treated with natural pyrethrum late in 2004 (Peay et al., 2006a), was not successful. One probable reason for the lack of success was that, unfortunately, the site was partly dewatered by the owner shortly before the application of the biocide. Potential refuges along the margins were left exposed in the cold conditions during the treatment and this increased the risk of an incomplete treatment of the population, if crayfish remained in torpor in refuges and did not come into contact with the biocide-treated water or surfaces. Crayfish vary considerably in their ability to utilize terrestrial habitats. Many species are aquatic species or largely so, moving into hyporheic zone for short periods if drought occurs (DiStefano et al., 2009). Others occupy seasonal wetlands with alternating dry and flooded conditions (Huner, 1995), whereas some, notably in North America and Australia are terrestrial burrowing species, which spend almost all of their lives below ground (classed as primary burrowers) (Hobbs, 1981, Hogger, 1988). Burrowing crayfish are adapted to exposure and in dry conditions many of them either seal their burrows, or construct chimneys at the entrances. Crayfish species of perennial waterbodies, such as the signal crayfish and the European white-clawed crayfish *Austropotamobius pallipes* (Lereboullet, 1858), which is being replaced by invading populations of signal crayfish in much of its range, are less well adapted to exposure in air. The white-clawed crayfish is particularly sensitive to exposure. For example, Taylor and Wheatly (1981) found that white-clawed crayfish became immobile within 48 hours when exposed in air (at 70-80% relative humidity, 15^{o}C) and moribund within 72 hours, associated with hypoxia-induced acidosis, although the acidosis was reduced by temporary storage of lactate in the carapace (Jackson et al., 2001). White-clawed crayfish can walk over land, e.g. if stranded by floods (Lewis and Morris, 2008), or to cross in-stream barriers (author unpublished). If their daytime refuges are exposed by falling water level during the day, white-clawed crayfish change their usual behaviour and soon emerge (Peay, 2003, Holdich et al., 2006b).

The signal crayfish lives in perennial streams in its indigenous range, although it also occupies lentic habitats in its introduced ranges in Europe and other regions. The signal crayfish can tolerate much longer periods out of water than the white-clawed crayfish, whose range it has invaded. Individuals survived up to 13 weeks out of water in drought conditions in a dried out tributary of the River Thames in 1990 (Holdich et al., 1995) and for as long in another dried out tributary in 2003 (author unpublished). Signal crayfish released on land have been reported as walking up to 1 km from the point of release (Peay et al., 2010). When 100 m sections of headwater stream in Yorkshire, England were artificially dewatered during the day, there was little response from signal crayfish, which remained in their refuges, but when similar lengths were dewatered downstream, to facilitate translocation of white-clawed crayfish to an ark site away from the invading population of signal crayfish, the indigenous crayfish responded rapidly by day, emerging in the period 0.1-2 hours after exposure (author unpublished).

If their aquatic refuges are exposed by a reduction of water level (natural or induced), crayfish are likely to either opt to wait in a refuge in case conditions improve, or seek more favourable conditions elsewhere. Exit from a refuge onto land leaves a crayfish potentially at greater risk from predation, especially from predatory birds and mammals, but remaining in a refuge for an extended period potentially risks dehydration, loss of ability to walk and death. Loss of as little as 10-15% of body water reduced the terrestrial walking speed by half in rusty crayfish *Orconectes rusticus* (Girard, 1852)(Claussen et al., 2000) and similar effects of exposure and associated hypoxia occur in other decapods, including lobsters (Vermeer, 1987, Ridgway et al., 2006). If the water level and hence wetted area is reduced in a waterbody, whether by climatic conditions, or by a regulated release of water, the crayfish remaining in the wetted areas, together with those that re-enter from exposed refuges, will be at greater density than before. This increases the likelihood of agonistic interactions. The largest and most aggressive individuals are expected to hold territories (Figler et al., 1995, Bergman and Moore, 2003, Fero and Moore, 2008) and exclude sub-dominant individuals from the best refuges. If aggression levels are high, some of the subordinate crayfish may stay in poorer quality, exposed refuges, rather than face aggression from the dominant individuals.

For resource managers considering use of biocide to eradicate an unwanted population of signal crayfish, reducing the volume of a pond prior to treatment is seen as a potential way to reduce the quantity and cost of the biocide, as well as facilitating application of the biocide; yet survival of crayfish outside the zone of effective treatment would leave a treated site accessible for re-colonization. It is therefore important to know whether the crayfish leave their refuges in response to de-watering and how long it takes for them to emerge. This study involved two elements: an experiment in tanks to investigate the response of signal crayfish to exposure of their burrows and a field trial to see how long it took crayfish to leave exposed burrows made in the earth bank of a partly dewatered pond.

5.2 Methods

5.2.1 The response of crayfish to exposure of burrows in experimental tanks

To investigate how readily crayfish would leave burrows, experimental tanks were set up, with artificial burrows to mimic those in the densely populated submerged banks of a pond, so that half the tanks could be partly dewatered in the first stage of treatment and all fully dewatered in a second stage. Opaque white plastic tanks were used (width 28 cm, length 52 cm and height 40 cm). For each tank a block of florists foam (width 32 cm, length 18 cm, height 23 cm) was trimmed to fit the width of the tank and a wedge of foam was cut from the longest face, to create a steep front face of the foam at approximately 70^{o} when the block was fitted tightly against the back of the tank. The florists foam was used because, as with natural burrows, it retained moisture for many days after reduction of water level. A plastic sheet was fastened to the top of the block to reduce moisture loss from the block of foam and to keep the top completely dry, to make it unattractive to crayfish. A 2 cm projecting lip of plastic sheet was intended to make it more difficult for crayfish to climb on top.

Horizontal (or slightly upward sloping) burrows were cored out of the foam to a depth of 8 cm, using a short length of 30 mm diameter plastic pipe. Five burrows were made in the lower half of the foam block and five in the upper half. The tanks were filled with water above the level of the upper burrows. The foam blocks were pre-conditioned by soaking them for two weeks prior to the trial with several replacements of water, because the pH of newly soaked foam was in the range pH 4-5. During the experiment itself, de-chlorinated tap water was used, with a bicarbonate buffer added and with freshly chipped limestone gravel loosely strewn across the floor of the tank (28 cm wide, 30 cm to the foot of the foam block) to buffer the water to pH 7.2-6.8. A filter pump was attached to the side of the tank.

Initially, the lower five burrows were covered by a screen of 6 mm plastic mesh pinned to the foam block to exclude crayfish. White lights for day use and red lights for night were suspended above the tanks, set for 13 hour day, 11 hour night, with the switch to red light occurring at approximately local sunset time. An overlap period of 0.5 hours was given when red lights were on before white light was switched off and a similar overlap was given after white light was switched on. A pole was suspended over the tanks as a camera gantry, with a small webcam camera set 0.9 m above the floor of each tank. The cameras were each given a foil hood to shield them from overhead lighting and they were connected to a laptop computer with video-recording facility.

Signal crayfish for the trial were caught from an upland stream by trapping, but only those in the size range 30-45 mm carapace length were used, due to the size of experimental burrows. Crayfish in each replicate were size-matched to within 5 mm difference in carapace length. Crayfish with missing or recently regenerated chelae were not used, because chelae loss disadvantages crayfish in agonistic interactions (Gherardi et al., 1998, Gherardi et al., 2000). Five crayfish were selected by random number for each tank. Crayfish were individually measured and numbered on the carapace, with a number from 1 to 5, using a yellow paint marker (Dykem Britemark®). Chelae were also marked to facilitate identification of individual crayfish when they were within burrows. Crayfish were fed ad libitum with wafers of an aquarium fish and crustacean food. On the first evening (setup), crayfish were placed in the tank and left to interact and established themselves in the available upper burrows. Most of the crayfish had settled in burrows well before the end of the setup night.

Tanks were given two stages of dewatering, involving partial and then full exposure of the burrows. At the first stage tanks given dewater treatment (dewater1) were bailed out to reduce the water level to half way between the upper and lower burrows and the mesh over the lower burrows was removed. The control treatment tanks were bailed without removal of water and the mesh over the lower burrows was removed. The treatment was started c. 24 hours after the initial setup, approximately 1.5 hours before dark. Activities in the treatment and control tanks were video-recorded overnight and the final positions of crayfish at the end of the night were noted.

In order to see the effect of increasing competition for inundated refuges, tanks previously given the dewater1 and the control treatments were both given the second stage of treatment, dewater2. The water level was reduced to 4 cm deep, fully exposing the upper and lower burrows. To provide alternative refuges for crayfish, three 30 mm plastic tubes 15 cm long were laid on the floor with the ends facing the exposed lower burrows. The treatment is shown in Figure 5.1 (at the end of section 5.2). If crayfish remained in exposed burrows beyond the first night of treatment, those that had already vacated the exposed burrows were removed either on the second day (n=11 tanks), or on third, fifth or eighth day (n=2, 1, 2 tanks respectively). The positions of crayfish were recorded daily until all the crayfish had left the exposed burrows. Tanks were then flushed out and re-set for the next batch of treatment. Each batch of the experiment was run with either one or two pairs of treatment and control tanks, run with freshly caught crayfish each time. For all tanks, the time to first exit was recorded, and the daily position of each individual crayfish.

All tanks were video-recorded, but intermittent technical problems with power supply, camera connections, video quality and lighting meant that useful full first night recordings were only analysed for five pairs of dewater1 and concurrent control tanks. The data extracted from video included the entries and exits from burrows, time spent walking on top of the block or on top of the pump and the fights between crayfish, plus crayfish leaning head and body out of their burrow, which was generally a foraging behaviour or a precursor to attack or defence from a burrow. The location and duration of all the individual spells in burrows, out on the tank floor or on top or pump were calculated for each crayfish on the first night of treatment (dewater1/control). As the recording was carried out mainly in the dark (red light), not all of the submissive interactions were visible, only the higher intensity interactions. These involved an aggressor rushing at a defender within a body length, striking or grappling. Crayfish often put a chela into a burrow to see if it was occupied, but this was not counted as a fight, unless the crayfish grappled, or an occupying crayfish immediately left the burrow. Fight outcomes for aggressor and defender were recorded as wins and losses when the loser fled, adopted a submissive posture, or was evicted from a burrow. Successful defence of a burrow from an attack involving grappling counted as a win for the defender. In some cases a draw was recorded because the outcome was uncertain, for example if the fight was disturbed by another crayfish. There were no injuries or mortalities in the tanks.

5.2.2 Statistical analysis response of crayfish to exposure of burrows in experimental tanks

The number and outcome of fights was used to assess the social rank of each crayfish in its group within each tank. Several metrics of rank were made for each crayfish as follows: a) an *a priori* prediction of rank made on the basis of size and sex, with the larger crayfish expected to rank higher than smaller ones, but with female crayfish subordinate unless 2 mm larger than a male crayfish; b) rank derived from the pair-wise comparison of winners and losers of interactions by each crayfish with every other crayfish in the group; c) rank from a dominance activity index (Martin and Moore, 2008), calculated as (DAI = log[(p+0.1)2/N+0.1]; p=sum of dominance interactions, wins in this case, N= sum of submissive interactions, losses; d) rank from the proportion of fights won, and e) rank from the number of times as the aggressor in interactions. An analysis of correlation of the ranking by different metrics was carried out, using IBM SPSS Statistics 20 for this and other analyses. Spearman Rank Correlation coefficients are shown in Table 1. There was no significant correlation between e) rank by the number of times as aggressor and the other metrics, but the others were all significantly correlated with each other. The predicted rank by size and sex had lower correlation coefficients with the other metrics of rank than did the closely correlated metrics by proportion won, dominance index and pairwise fights. The pairwise ranking by fights was adopted as the ranking method thereafter, because it aided ranking of the subordinate crayfish, especially those in intermediate to lowest ranks (3 to 5).

Table 5.1 Spearman Rank Correlation of the ranks of crayfish calculated using five different metrics of social rank

Ranking by metric	n=50	a	b	c	d	e
Predicted from size and sex	a	1.000	0.628**	0.542**	0.488**	0.151 NS
Pairwise from fights	b		1.000	0.755**	0.678**	0.113 NS
Dominance index	c			1.000	0.827**	0.177*
Proportion of fights won	d				1.000	0.215**
Number of times as aggressor	e					1.000

Correlation is significant at the 0.01 level (2-tailed): **

Correlation is significant at the 0.05 level: *

Not significant: NS

Data on times were tested for normal distribution and homogeneity of variance. As the time to first exit of crayfish did not meet the requirements for parametric tests, with or without transformation, non-parametric tests were used to compare the effects of sex, rank and treatment on time to first exit. To compare the total time spent outside burrows and the time spent climbing on top of the block or pump by sex, rank and treatment the data were log-transformed and analyses of variance were carried out. The counts of spells in lower burrows were compared between treatments after square root transformation using a Student t-test, but non-parametric comparison was required for other comparisons of spells of burrow occupation. The distribution of crayfish between the upper and lower burrows was compared for individual crayfish using a Wilcoxon signed rank test. The time taken for individual crayfish to first exit their burrow at the first stage of treatment (dewater1 and control) was compared with the time taken at the second stage (dewater2), again using a Wilcoxon signed rank test.

The effect of competition on the time to complete evacuation of crayfish from burrows during dewater2 treatment was investigated using a generalized linear model. A Poisson log linear model was used with the number of nights before all crayfish fully vacated their exposed burrows as the dependent variable, with number of nights to the start of removal as the covariate and with prior treatment at the first stage and batch number (i.e.

when the tank experiment was run) as factors. The full model was checked for over-dispersion, using a Pearson Chi-Squared Goodness of Fit test, (Pearson Chi-Square = 2.430, df = 7, value/df = 0.347), but was not over-dispersed. The test of model effects showed there was no significant effect of prior treatment (dewater1 or control) on the number of crayfish that remained in exposed burrows after the first night, nor of batch number, so both the terms were removed from the model.

5.2.3 The response of crayfish exposed in burrows in the bank of partly dewatered pond

The study site used in August 2012 was a 0.54 ha fishing pond near Llyswen in Wales (UK Ordnance Survey reference SO 1390 3755; latitude 52.029723 longitude -3.2565987), which has an established population of signal crayfish. The site was formerly a mill pond and the water level could be controlled by blocking the inflow and by opening and closing sluice gates that regulated the outflow at the mill dam. The water level was to be reduced to facilitate netting of the pond for assessment of the stock of fish and this provided an opportunity to carry out a trial on the response of the crayfish in exposed refuges.

Prior to dewatering, crayfish were caught for use in the trial by setting crayfish traps overnight. The traps were Fladen®cylindrical folding traps, 30 cm diameter and 60 cm long with 12 mm mesh and with the funnel apertures at both ends restricted to 55 mm. They were baited with about 150 g cat food in a 6 mm mesh bag. Traps were lifted the next morning and the crayfish were measured for sex, size and condition (loss or regeneration of chelae or other injuries). Crayfish were kept in the pond while the trial area was prepared. The water level was reduced, in order to expose the pond banks and margins and provide a working area for the trial. The south side of the pond had a steep bank and margin (gradient of 50^o to 90^o), part of which had large stones, undercut sections, roots of trees and emergent plants. This side of the pond offered abundant refuges favourable for crayfish, but the bank was too steep and complex to use in the field trial. By contrast, the north side of the pond had a shallow margin with a relatively uniform gradient (5^o to 10^o), with only sparse cobbles, plus a few patches of amphibious bistort *Persicaria amphibia (L.) Delarbre*, whose stout roots and stems on the bed provided some shelter for crayfish, as was confirmed during the dewatering period. The shallow margin was bordered by a near vertical earth bank 0.2-0.4m high, where the closely-grazed pasture around the pond had been eroded by wave action when the pond was full. This bank was inspected carefully for any crayfish burrows, but none were found.

To see the response of crayfish in exposed burrows, five partial enclosures were set up along the exposed shallow margins. Each enclosure comprised an 8 m length of the exposed earth bank, forming the base of a triangle and a V-shaped barrier running from each end of the bank to an apex approximately 2 m nearer the pond. The barrier was intended to direct any crayfish that left refuges and walked towards the pond into a trap, where they could be recorded. The barrier was made of translucent plastic sheeting stapled to a series of fence posts. The bottom of the sheeting was turned inwards at a right angle below the surface and carefully back-filled, to avoid leaving any gaps that would allow crayfish to walk or burrow beneath it. The sheeting was cut at the apex of the triangle and a crayfish trap was inserted at ground level and sealed into the sheeting with adhesive tape, such that only the entrance of the trap was within the enclosure. The end of the trap outside the barrier was closed to prevent any exit. The sheeting at the ends of each enclosure was extended to the top of the earth bank, but the 8 m long bank top was left unfenced. Any crayfish that climbed into the pasture would have little refuge, as the average sward height was less than 7 cm. The site layout is shown in Figure 5.2 (at the end of section 5.2).

The surface substrate within the enclosures was largely gravel and pebble and the few stones present greater than10 cm were removed before construction of the enclosures. Hence, because the earth bank lacked crayfish-constructed burrows, the enclosures were free from crayfish before the start of the trial. Artificial burrows were constructed just above the bottom of the damp earth bank using a surveyors ranging pole, to create burrows approximately 3 cm wide, to a target depth of 20 cm (actual range 17-32 cm). Burrows were a minimum of 15 cm apart, 25 per enclosure.

In the late afternoon, 2-3 hours before sunset, crayfish that had been caught overnight were randomly assigned into five batches for use in the trial, one for each enclosure. Crayfish were marked individually with numbers on the carapace, using a yellow paint marker (Dykem Britemark®). Each marked crayfish was immediately installed tail-first into a burrow, 114 crayfish in total. Only crayfish in good condition with intact chelae were used. If a crayfish tried to leave the burrow as soon as it was released (within 1 minute), the burrow was checked to make sure it was wide enough for the crayfish and widened slightly if not. The crayfish was put back in the burrow again, but left alone thereafter.

The position of crayfish in each enclosure was recorded the following morning; those in the traps, in the open, in burrows, or (by subtraction) missing from the enclosures. Crayfish were removed once they had entered the traps, any crayfish out on the bed were noted but left. Where crayfish were not readily visible in the burrows, the occupancy status of each burrow was checked quickly with a torch and/or a grass stem.

To facilitate the fisheries work (not part of this study), the water level in the pond was reduced further on the second morning and this allowed a second batch of crayfish to be collected by hand from the newly exposed areas, especially from the steep margin. An additional 99 crayfish were marked and added to the enclosures late in the afternoon, into burrows that had been vacated during the previous night, or additional ones where necessary. The size distributions of crayfish in the two batches used in the trial are shown in Figure 5.3, obtained by trapping (for batch 1) and caught by hand on the newly exposed margins of the pond (for batch 2), excluding the small crayfish (< 25 mm) which were not used. There was no significant difference in the median size of crayfish in the two batches (chi-squared text: $n = 194$, $df = 1$, chi-squared = 0.713, NS) nor in the size distributions (Mann-Whitney U test: $n = 194$, $U = 5129$, NS). The number of all marked crayfish recaptured or remaining in burrows was recorded on the subsequent day.

The enclosures were kept in place for 6 nights in total, from 15th-21st August, after which they were dismantled and the pond was allowed to re-fill. Weather conditions were typically varied during the trial. There was rain prior to setting up the enclosures, but several hours of dry sunny conditions occurred just before the start of the trial (23^{o}C, 74% relative humidity) and it remained dry but humid on the first night. The second day was overcast, but remained dry through the day and most of the night, with some drizzle. The third day had prolonged rain. Conditions remained unsettled for the rest of the trial, with sunny intervals, overcast conditions and intermittent showers.

5.2.4 Statistical analysis of the response of crayfish exposed in burrows in the bank of partly dewatered pond

The data on counts of crayfish recaptured and nights to recapture were tested for the assumptions of normal distribution with Shapiro-Wilks test and homogeneity of variance with Levene's test, but even with log and square root transformations they did not accord with normal distribution, so non-parametric tests were used; Mann-Whitney U tests to compare the effects of sex and batch on time to recapture and Kruskall Wallis tests to check for any differences in time to recapture due to size or enclosures.

Figure 5.1 Experimental tank during dewater2 treatment

Figure 5.2 Site layout for field trial of expose burrows

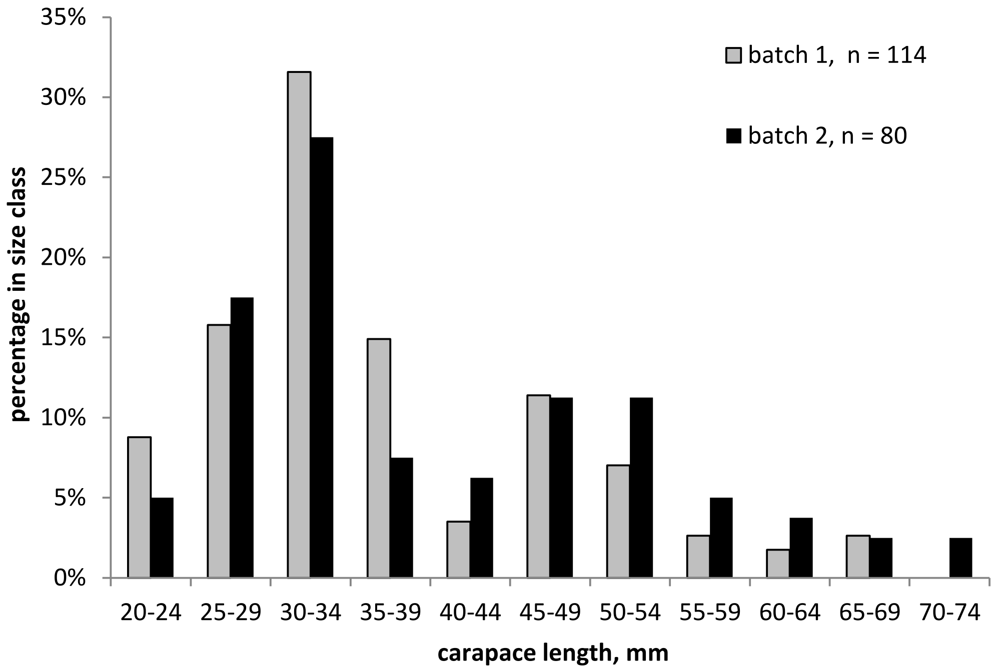

Figure 5.3 Size distribution of signal crayfish used in the field trial of exposure of signal crayfish. Two batches of crayfish were used, batch 1 from crayfish traps (Fladen® cylindrical folding traps, with 12 mm mesh; and two funnel apertures restricted to 55 mm; baited with cat food) and batch 2 hand-caught on newly exposed margins during dewatering of the pond.

5.3 Results

5.3.1 Results of test of the response of crayfish to exposure of burrows in experimental tanks

The crayfish varied markedly in the time it took for them to first exit their burrows after the start of treatment (n = 90, from 9 pairs of tanks) (Figure 5.4). Those out on the floor at the start of treatment (11%, n = 10) were excluded from the time of first exits. Some crayfish emerged from their burrows in less than 0.25 hours, in some cases even before completion of the bailing out, which took a couple of minutes. Those that made an early exit in the dewater1 tanks represented 34% (n=14) of crayfish in burrows, whereas in the control tanks it was only 2.6% (n = 1). The number of early responders in the control tanks was too small to allow non-parametric tests of significance to be used, but the difference between treatments was reversed in the group that emerged 0.25-2 hours after the start (approximately the interquartile range of time). In all, 74% of crayfish in the control tanks emerged in the period 0.25-2 hours, compared to 34% of those in the dewater1 tanks, a significant difference (Mann-Whitney U test: n = 63, U = 629.50, P = 0.038). Furthermore, more of the crayfish in the dewater1 treatments stayed in burrows for more than 2 hours than did those in the control tanks, another significant difference (Mann-Whitney U test: n = 22, U = 89, P = 0.043). There was no difference in time of first exit found between sexes (Mann-Whitney U test: n = 45, df = 1, U = 305, NS), nor according to rank (Kruskal-Wallis test: n = 45, df = 4, K = 2.618, NS).

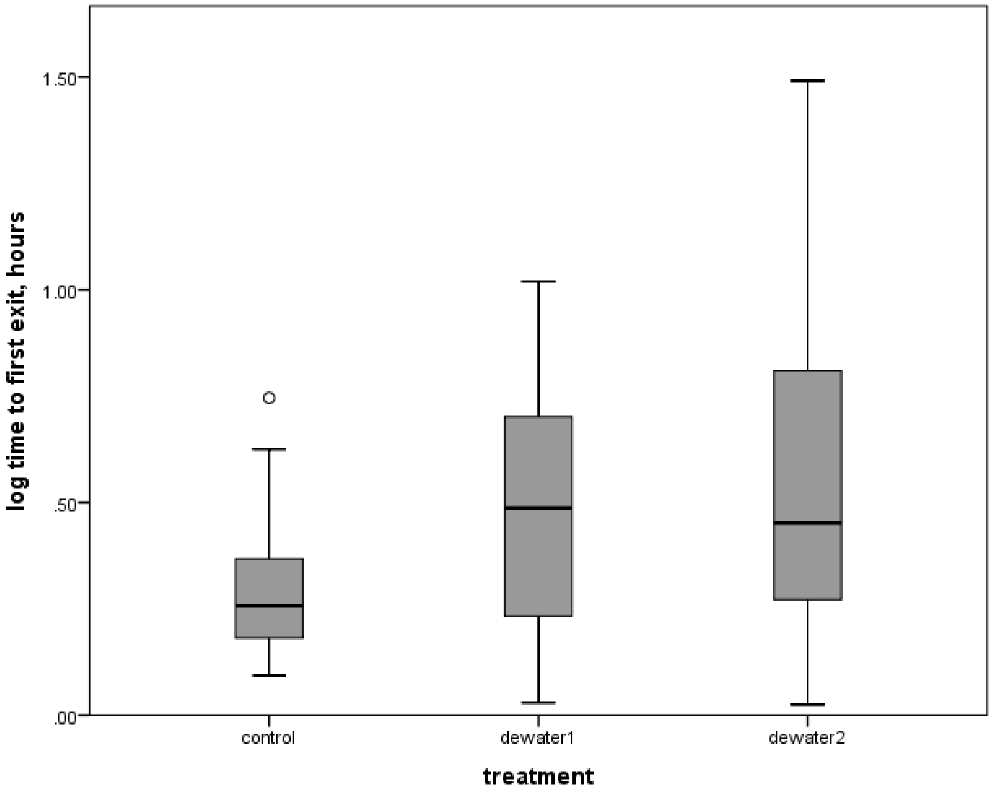

Figure 5.4 Time taken in hours for crayfish to first exit a burrow during stage 1 of treatment (dewater1 and control) and stage 2 (dewater2)

Overall activity in the tanks was low at the start, but it increased after lights off, with a peak of activity about 3 hours after the start (Figure 5.5). There was significantly more activity during dark hours than light. The median events/hour were 21 in light and 51 in dark (chi-squared test: n = 130, df = 1, chi-squared = 8.451, P = 0.004). The number of fights by individual crayfish was correlated with rank (Table 1, Figure 5.6) and evictions from burrows occurred in every tank during the first night of treatment (median 5 evictions). During the first night of treatment there was no significant difference in the total number of fights per tank between control and dewater1 treatments (106.2 ± 18.0 control and 102.4 ± 16.1 dewater1; Student t test: n = 10, df = 8, t = 0.156, NS).

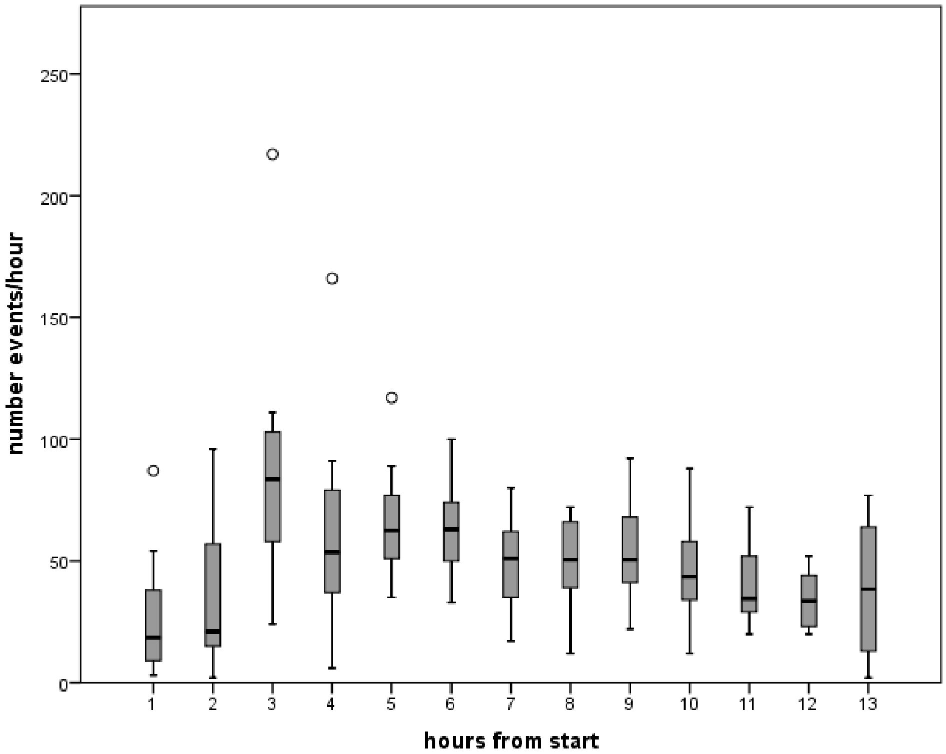

Figure 5.5 Crayfish activity during stage 1 first night, total number of recorded events per hour (entries and exits from burrows, fights, climbing, and leaning out of burrows)

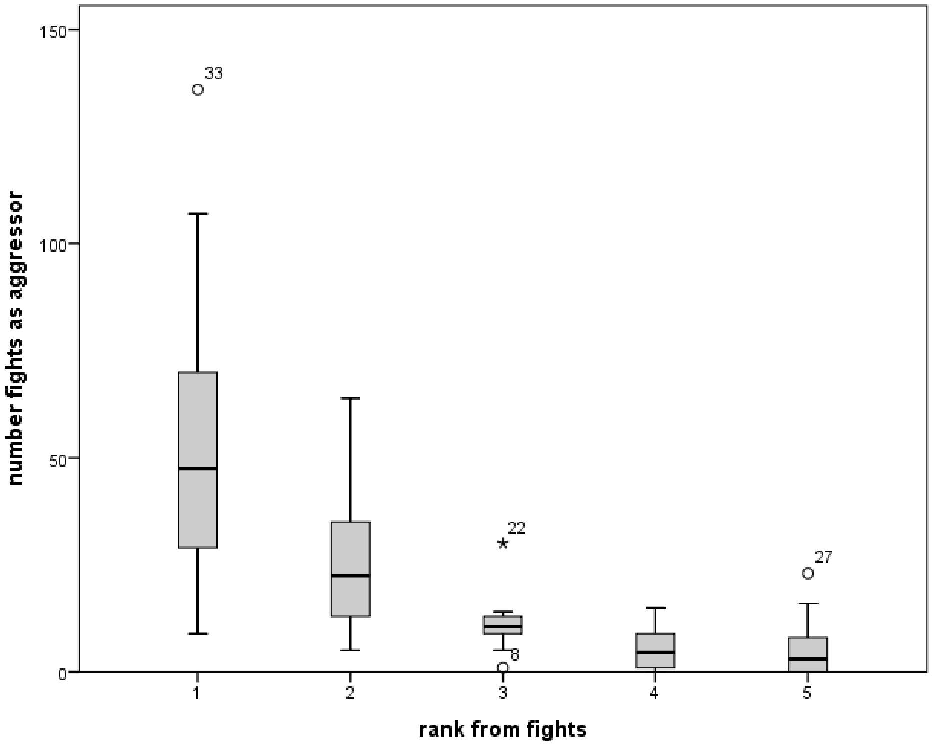

Figure 5.6 Number of fights as aggressor according to rank from fight outcomes (ranked 1 to 5 from highest to lowest)

There was no significant difference in the total number of entries and exits per tank between the control and dewater1 (290.6 ± 32.4 control and 203.2 ± 41.9 dewater1; Student t-test: n = 10, df = 8, t = 0.1647, NS). During the first night of treatment crayfish spent 23% of the time out of burrows on average. The total time spent out was not significantly affected by treatment (log-transformed data, one-way ANOVA: n = 50, df = 1, F = 0.205, NS), or rank (ANOVA: n = 50, df = 4, F = 0.403, NS) or sex (ANOVA: n = 50, df = 1, F = 0.119, NS). The crayfish explored the newly available lower burrows in both treatments.

As well as their time in burrows and on the floor of the tank, crayfish climbed out onto dry exposed areas, on top of the block of burrows and on the pump. The actual time spent in these exposed areas was low. Overall, 58% of crayfish were recorded on top of the block of burrows (n = 19), on the pump (n = 13), or both (n = 3). Among the crayfish that climbed, the average time spent climbing was 5% of the total time they spent outside burrows. There was no significant difference in the number of climbing crayfish between sexes or treatments. When the total time spent climbing in exposed areas was considered, however, there was an effect of rank (log-transformed data, one-way ANOVA: n = 26, df = 4, F = 2.986, P = 0.039). The dominant crayfish (rank 1) spent less time climbing.

During the first night of the treatment in the dewater1 tanks, 64% of the crayfish (n = 16) revisited (fully entered) at least one exposed (upper) burrow, but only 16% (n = 4) spent more than 0.25 hours in any exposed burrow. All of those were low ranking crayfish (rank 3 to 5). There was a significant difference between the number of spells spent in the upper burrows in the dewater1 (exposed burrows) and control treatments (submerged burrows) (Mann-Whitney U test: n = 50, U = 46, P < 0.001), (Figure 5.7). In the control tanks there was also some bias toward spells in lower burrows (square root transformed data, Student t-test: n = 25, df = 24, P < 0.001). In both treatments most spells in burrows were short, only about 34% were more than 0.25 hours duration (control 0.32 ± 0.029; dewater1 0.37 ± 0.03), with the short spells occurring when crayfish were choosing burrows, involved in fights, or making forays to feed. The median number of spells in burrows was 25 per crayfish in the first 13 hours, but it ranged from 2 to 82. Overall, fidelity to burrows was low, with the median number of burrows occupied per crayfish per night being 6 (of 10 total) for dewater1 and higher, 8, for control (Mann-Whitney U test: n = 50, U = 142.5, P = 0.001).

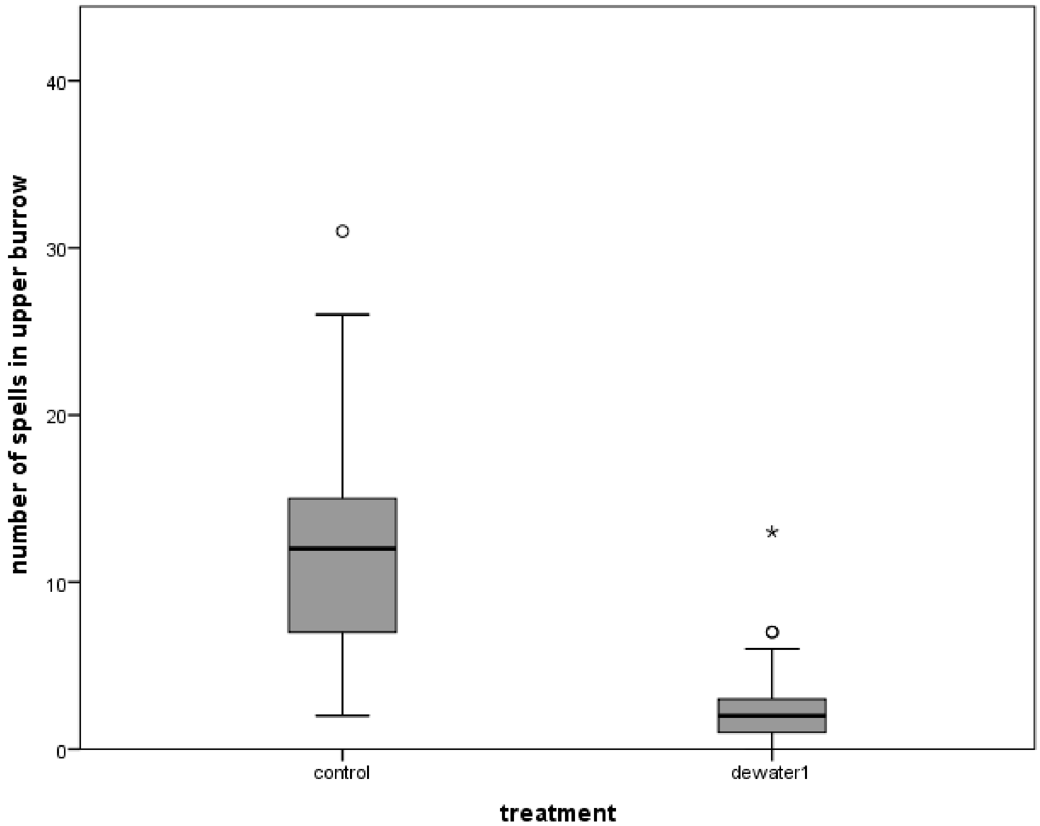

Figure 5.7 Number of spells spent by crayfish in upper burrows during the first night in control (submerged burrows) and dewater1 (exposed burrows) treatments

The positions occupied by crayfish differed between treatments. Just before the start of treatment (stage 1), all but 11.1% (10 crayfish in 18 tanks) had settled in an upper burrow (lower burrows were blocked). In the dewater 1 tanks after the first night of the treatment, there was only 4.4% left in exposed (upper) burrows (2 crayfish in 9 tanks), the rest had moved out to submerged locations. In the control tanks crayfish also changed burrow during the first night; only 8.5% of crayfish were found in the same burrow in the morning after the start of the treatment. In addition, in both the dewater1 and control tanks more of the lower ranking crayfish stayed out on the floor in the light, both before the start of treatment and after the first night; 80% were intermediate to lowest rank (ranks 3 to 5) and the only tank in which a dominant crayfish (rank 1) remained out was one which had mated a female overnight and was guarding her.

When the second stage, dewater2, was carried out, the median time to first exit was greater than during stage 1 (Wilcoxon signed rank test: n = 43, df = 1, T =-2192, P = 0.028), increasing from 0.87 to 1.38 hours. After the first night of dewater2 treatment, most tanks (14 of 16) still had crayfish in exposed burrows (ranging from 1-3 crayfish per tank), 32.5% of the total crayfish (26 of 80) (Figure 5.8). The location of individual crayfish (tube, floor or exposed burrow) was correlated with rank (Spearman's rank correlation test: $n = 60$, $r = 0.437$, $P < 0.001$), with the dominant crayfish preferentially occupying the submerged tubes and the lowest ranked crayfish predominantly in the exposed burrows.

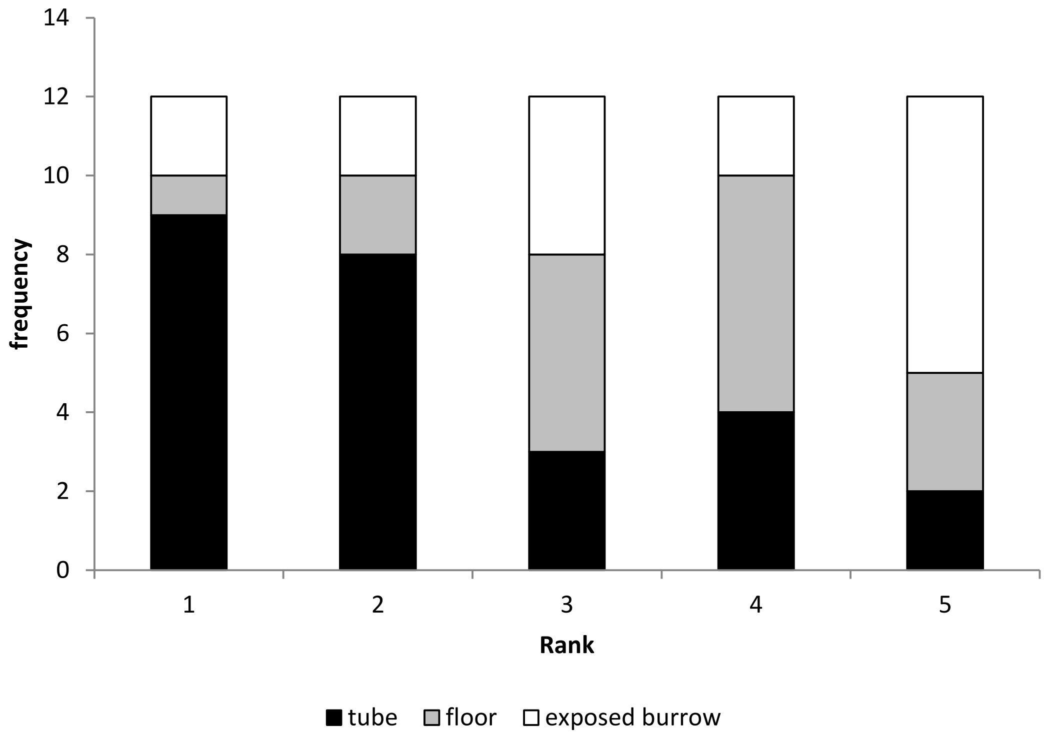

Figure 5.8 Locations of crayfish after first night of dewater2 treatment (all burrows exposed) according to crayfish rank (1 highest to 5 lowest)

During the dewater2 treatment, the time until all the crayfish vacated exposed burrows was affected by the competition for refuges on the floor of the tanks. There were only two (of 16) tanks in which all the crayfish evacuated the burrows during the first night of dewater2 treatment. In the other tanks, the crayfish that stayed in exposed burrows after the first night generally occupied an exposed burrow until the crayfish that had already occupied wetted refuges had been removed. There was a significant relationship between the number of nights before any removal of crayfish and the time taken for all the crayfish to vacate the exposed burrows in a tank (generalized linear model with log Poisson distribution: $n = 16$, $df = 15$, $P < 0.001$, $\text{Ln } y = (0.233 \pm 0.0269\, x)+(0.576 \pm 0.1174)$, where y = number of nights for all crayfish to vacate burrows, x = number of nights before start of removal of crayfish that had vacated previously). There was no significant effect of prior treatment (dewater1 or control) on the number of crayfish that remained in exposed burrows after the first night, so this term was removed from the model. Many of the tubes were unoccupied after the first night of the dewater2 treatment (9 of 16 tanks had unoccupied tubes available). Of the crayfish that moved to the submerged area on the first night of dewater2, 39.5% of them stayed on the floor of the tank, rather than the tubes.

5.3.2 Results of test of the response of crayfish exposed in burrows in the bank of partly dewatered pond

The field trial was to investigate how long it took for crayfish to vacate burrows in the bank of a partially dewatered pond. Of the total 194 crayfish used in the trial, 47.9% (n = 93) were recaptured in the traps in the enclosures within the week of the trial. There was 3.6% known mortality (n = 7) and 48.4% (n = 94) were not recaptured, due to escape via the steep earth bank or other unknown loss. Overall, 35% remained in the burrows for more than one night and 10% for more than two nights. At the end of the trial, there was still one live crayfish in a burrow (representing 0.5%), which had remained in the enclosure for either 5 or 6 nights (the crayfish was not dug out to identify the individual and hence batch).

Most of the crayfish waited until after dark before emerging from the exposed burrows. Approximately 9% exited burrows during daylight soon after the start of the trial (13 of 114 in the first batch), although some of those walked around and then returned to the burrow without entering the trap.

There was no significant effect of sex on the time until recapture, whether the whole dataset was compared (Mann Whitney U test: $n = 187$, $U = 4520.5$, NS), or only the individually identified recaptured crayfish (Mann Whitney U test: $n = 90$, $U = 1208.5$, NS) (Figure 5.9), so crayfish of both sexes were pooled for subsequent analyses. Three crayfish that had lost marks were not identified individually at recapture and are not included in the analyses below. There was no significant effect of crayfish size categories (< 30 mm, 30 – 39 mm, 40 -49 mm, > 49 mm) on the time until recapture (Kruskall Wallis test: $n = 187$, $df = 3$, $K = 4.407$, NS) (see Fig. 5.3 for size distribution). There was no significant difference between enclosures, whether for all crayfish (Kruskall Wallis test: $n = 187$, $df = 4$, $K = 3.438$, NS), or only those recaptured (Kruskall Wallis test: $n = 90$, $df = 4$, $K = 6.295$, NS).

There was an effect of batch (Mann Whitney U test: $n = 90$, $U = 139.5$, $P < 0.001$), with significantly more crayfish from batch 2 remaining in the burrows for two or more nights, whereas the median time to recapture for batch 1 was one night (Median test: n = 90, df = 1, chi-squared = 15.188, P<0.001) (Figure 5.10).

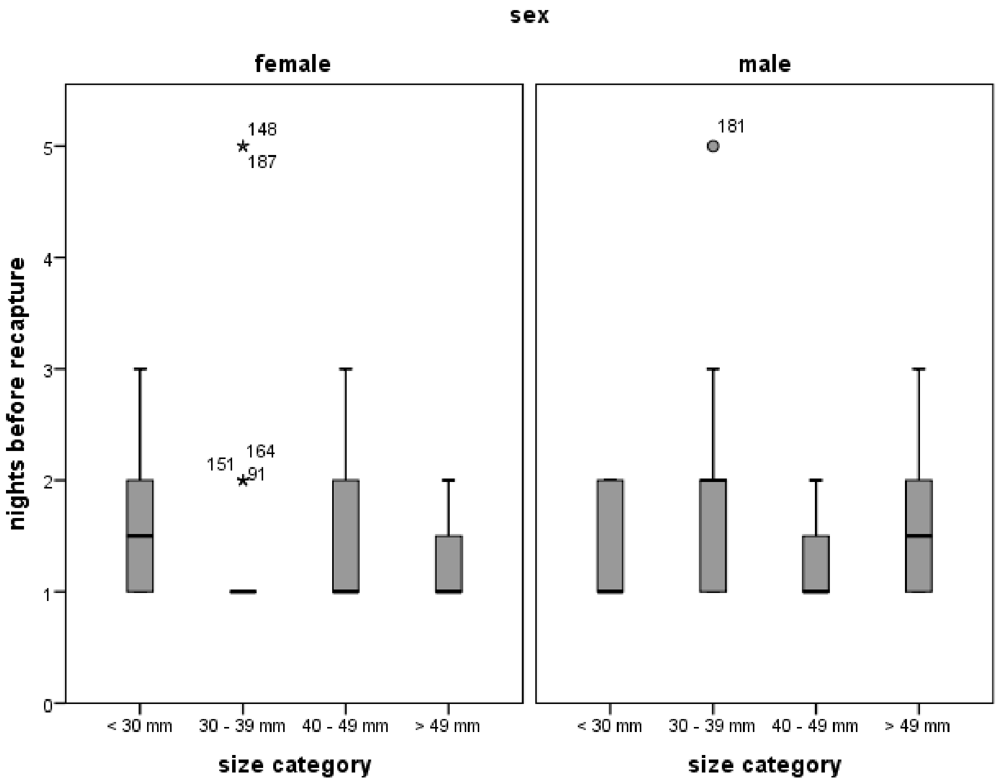

Figure 5.9 Number of nights until recapture, grouped by sex and size class

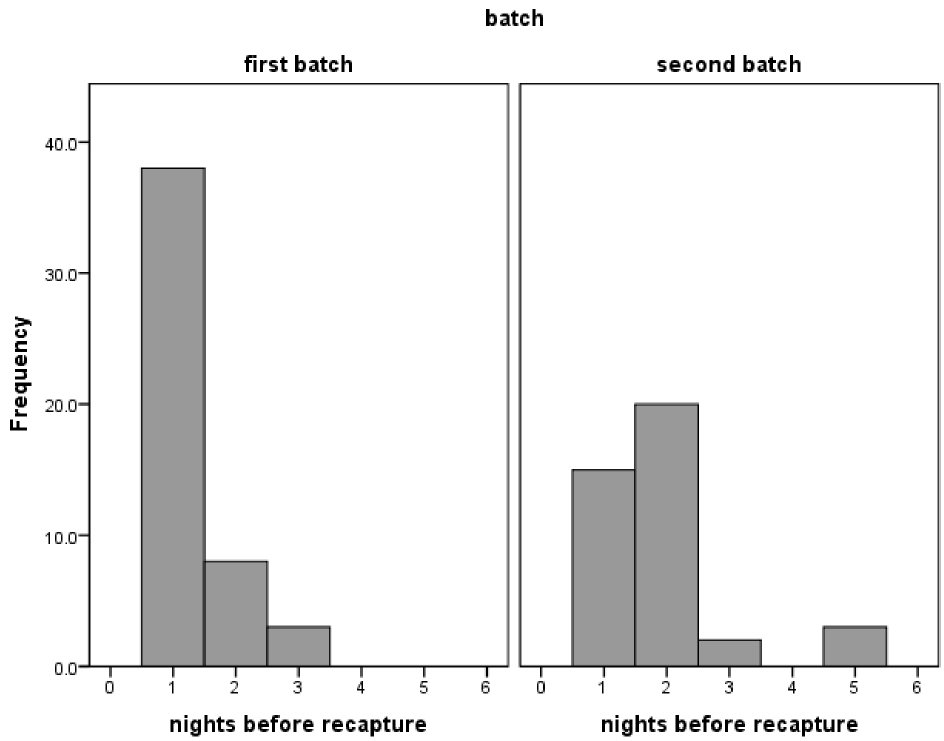

Figure 5.10 Frequency chart of the number of nights until recapture for two batches of crayfish in exposed burrows (batch 1 n=50, batch 2 n=40)

Weather conditions may have influenced the time taken to leave burrows between batch 1 and 2. Crayfish were seen responding to rainfall in two of the enclosures on the third day of the trial (batch 2 had one night in burrows). There was heavy rain in the morning for about three hours, during which there were no crayfish seen out of burrows, but when it decreased to light rain or drizzle by 13:00, five crayfish in enclosure 1 and one in enclosure 2 emerged and started walking about and some were seen drinking in small puddles. Other crayfish moved to the entrance of their burrows, where water had collected. By the time the light rain had ceased just over an hour later, all the crayfish had returned to burrows.

Of the crayfish that escaped by climbing up the steep earth bank, none were found predated on the pasture or exposed margins. There was no significant difference in median size between the crayfish that escaped and those that were recaptured. An unknown proportion of the marked crayfish that escaped from enclosures during the first night re-entered the pond, because several were found sitting on newly exposed margins when the water level was lowered further in the morning (for a survey of fish) and other marked crayfish were seen walking in shallow water that morning, presumably having just left the newly exposed areas.

5.4 Discussion

5.4.1 Crayfish behaviour

This study showed that individual signal crayfish all vacated burrows in response to exposure, but they differed in the time of response, both in tanks and in the field. Although some individuals left their refuges in daylight soon after being exposed, the majority waited until after dark before emerging. This is in keeping with the expected pattern of predominantly nocturnal behaviour in signal crayfish (Abrahamsson, 1981, Gherardi, 2002). The proportion of crayfish emerging in light in the experimental tanks was greater than in the field trial (34% compared to 9%), although this may simply have been due to the proximity to water within touching distance in the tanks. In contrast, in the field trial, burrows were at least 4 m from the reduced level in the pond and the pond was screened from sight except through the trap set to intercept the direct route to the pond. Movement of the barrier fencing in the wind may also have discouraged crayfish from exiting until after dark, if the movement was perceived as a threat from potential predators and so contributed to the lower initial response.

In both the tank and field experiments, the majority of crayfish vacated exposed refuges during the first night. When alternative submerged burrows were available in the tanks (dewater1), only 4% of the crayfish remained in an exposed (upper) burrow after the first night. When wetted refuges of good quality were not available, more crayfish remained in exposed refuges. This occurred in the tanks when all the burrows were exposed (dewater2 treatment), when 29% stayed in the burrows for more than one night, as well as in the field trial, when 35% remained in the burrows for more than one night and 10% remained for more than two nights.

Even when the signal crayfish vacated exposed burrows, some of them remained on the floor of the tanks, rather than occupy the open-ended tubes, suggesting the tubes were less attractive as refuges. Crayfish in the tubes were frequently displaced by another crayfish entering from the other end and the tubes could be moved if crayfish climbed over them. These less attractive submerged refuges probably contributed to subordinate crayfish choosing to remain in damp, exposed burrows, rather than compete with the dominant crayfish for refuges in the submerged area. A preference for refuges with a single entrance was found by Martin and Moore (2008) among rusty crayfish *Orconectes rusticus* (Girard, 1852), which used refuges with a single entrance for significantly longer periods than those with two or more entrances. Another factor making the tubes less attractive as refuges may have been their white colour, which tends to be less attractive to crayfish than dark refuges (Alberstadt et al., 1995). Nonetheless, interaction between crayfish appears to be the main factor, because in most tanks the dominant crayfish moved to a tube on the first night of the dewater2 treatment (all burrows exposed).

In this study, the dominant crayfish frequently blocked access of lower ranking crayfish to submerged burrows, by leaning out and striking, or emerging to attack. As in other studies, aggressors won most fights (Copp, 1986, Figler et al., 1999, Ahvenharju and Ruohonen, 2007) and within each group there was one clear dominant, plus four subordinates that were much more closely ranked together, as seen in the study with red swamp crayfish by Issa et al. (1999). Subordinate crayfish tended to enter burrows away from the dominant, or wait until the dominant had withdrawn deep into its burrow before making a hasty entry into a burrow nearby. Whilst the dominant crayfish evicted subordinate crayfish, sometimes repeatedly, subordinates did so too in some cases in this study, as seen by Ranta and Lindstrom (1993). Subordinates also successfully defended burrows from larger, higher ranking crayfish in this study, a prior residency effect seen in other studies too (Ranta and Lindstrom, 1993, Peeke et al., 1995, Blank and Figler, 1996). When opponents were closely matched for size, less than 15% size difference, Klar and Crowley (2012) found no advantage for prior residents in competition for shelters, but Tricarico and Gherardi (2010) found that crayfish remembered shelter occupancy and previous agonistic interactions with other crayfish for at least two days and this affected the likelihood of fights and success in the previously successful crayfish. In this study, crayfish had time to establish rankings in the set-up period before the start of treatment and may have had prior contact in the field prior to being caught for the trial. The size difference between the largest and smallest crayfish in tanks was in the range 6.1 – 15.9% of the size of the smallest crayfish and whilst the largest crayfish were dominant, size was not a good indicator of rank among the subordinate crayfish in this study.

Both the tank and field experiments showed that when access to submerged refuges is restricted, either by competition for shelter, as in the tanks, or by a barrier to access, as in the field trial, more than half the crayfish were willing to climb out onto dry terrestrial areas that lacked refuges. In the tanks the spells of climbing were of short duration, seconds to minutes at a time, but in field conditions the time spend wandering on the adjacent pasture is likely to have been longer, depending on the time taken for crayfish to scale the steep bank and find an alternative route to the pond. The shortest path via the steep bank and around the barrier directly to the pond was about 5 m, so if the walking pace on land was similar to that found for rusty crayfish, 0.05 m s^{-1} (Claussen et al., 2000),and 0.015 m s^{-1} for white-clawed crayfish (Pond, 1975), the minimum time to water would be 1.5 – 5 minutes. It was probably longer in practice. When one of the crayfish was seen climbing the near vertical earth bank of the pond (Figure 5.11) it made several pauses and took more than 3 minutes to climb 0.3 m. Even so, it was feasible for crayfish to return to the pond during the night. Up to half the crayfish climbed out of the enclosures and the presence of marked crayfish in the pond confirmed the successful escape of some, probably most of them. In this study there were no other waterbodies in the vicinity, but where sites had wetlands associated with the open water, there would be the potential for signal crayfish to remain in vegetated wetland for a time, rather than following a receding water level to open water.

Figure 5.11 signal crayfish climbing onto pasture, having vacated an exposed burrow during the field trial of dewatering at a pond

Although not known to burrow extensively in their indigenous range, signal crayfish in their introduced range in Europe are known for their burrowing behaviour (Guan, 1994). In the tanks, crayfish were able to modify the burrows slightly by digging into the florists foam and some of the subordinate crayfish attempted to burrow at the side of the block of burrows. Crayfish were also observed carrying food back to their burrows, as noted by Goddard (1988). Unusually, there were also instances of crayfish taking a piece of limestone gravel (up to 18 mm in length) into burrows; once into an upper burrow, twice into lower burrows and once into a tube. A female crayfish was also recorded picking up two pieces of gravel and transporting them to other positions on the floor of the tank. Carrying gravel may have been an attempt by crayfish to block the entrance of a burrow, albeit not effectively. Another, more likely, possibility is that crayfish were consuming calcium carbonate from the surface of the gravel to make up the calcium lost during moulting, which has to be replenished in the inter-moult period (Wheatly and Ayers, 1995). Furthermore, water in the burrow is likely to have been more acidic than the rest of the tank, due to the properties of the florists foam and/or due to respiration.

A few of the crayfish in the field trial also modified exposed burrows. Two crayfish were found with tail facing out after one night and another one after the second night of the trial, a position characteristic of crayfish when burrowing (Guan, 1994). Two other vacated burrows were found to have many small pellets of soil piled up at the entrance, indicative of burrowing activity, which is unusual to see out of water in the signal crayfish, although such behaviour is common among burrowing crayfish species (Hobbs, 1981).

The role of climatic conditions in the time taken for signal crayfish to emerge from burrows may be significant, as is suggested in the field trial by the delayed emergence of crayfish in the second batch and in the diurnal emergence of crayfish during rainfall. In warm dry conditions, crayfish sitting in exposed burrows run the risk of dehydration, immobility and death, so they may have to leave their burrows soon after exposure, unless they are able to burrow into moist substrate, for example if there is seepage of groundwater. If rainfall occurs at near daily frequency, the crayfish would have a much lower risk of dehydration. In cold conditions, crayfish would be able to stay out of water in torpor in a damp refuge, provided they didn’t freeze. Kozak and Policar (2003) found that some signal crayfish survived in burrows in a fully drained pond for three months in winter, with air temperatures as low as -20 C^{o}. In northwest England in autumn 2012, two ponds were dewatered in cool, damp conditions and approximately one month later, examination of sections of the exposed banks showed there were some signal crayfish remaining in burrows, which were up to 0.3 m deep (Brazier, 2013). Yet in summer, risk of dehydration is an incentive for crayfish to vacate burrows. At a farm irrigation reservoir in Yorkshire, England, when signal crayfish burrows were excavated along several metres of exposed clay bank that was only slightly damp below the surface, there were no crayfish present (author, unpublished). Similarly, (Guan, 1994) found that burrows excavated below the normal winter water level in a lowland river were unoccupied in summer when the water level fell.

5.4.2 Management implications

The primary purpose of this study was to determine the response of signal crayfish to dewatering, to provide guidance to managers as to suitability of reducing the water level in a pond prior to application of a biocide to try to eradicate an unwanted population of invasive signal crayfish. The field trial shows that if dewatering is carried out shortly before the start of biocide treatment, there is a significant risk that crayfish will be present above water level and out of reach of biocide treatment.

From this study it is recommended that if dewatering is done, the water level should be held at a stable level for more than one week, or until the bank and any exposed refuges are dry, which may take longer. The maximum time required for all signal crayfish to vacate their refuges is not known and it is likely to be affected by the population density, the type and distribution of refuges, the water retention of the substrate and climatic conditions. There is still uncertainty about the degree of influence these factors have individually and cumulatively. On free-draining sites with little vegetation and with relatively dry conditions, the crayfish would be expected to vacate exposed refuges readily. If the population is still in the establishment phase and below carrying capacity, crayfish would be likely to be able to move down into the water to alternative submerged refuges without experiencing enough competition to make them retreat to exposed areas. By contrast, there is much more of a risk of individuals remaining out of the pond if there is high density in the pond before dewatering, there are heavily vegetated, complex margins offering damp cover, water-retentive substrate, or continuous seepage of groundwater. Experience from an unsuccessful biocide treatment in winter (Peay et al., 2006a) and dewatering in autumn in Cumbria (Brazier, 2013) suggests signal crayfish may not respond readily to dewatering outside the season of high activity.

The ideal case in dewatering a pond before biocide treatment may be to reduce the water level to a stable level in summer for a period long enough for exposed refuges to dry out, but then increase the level slightly immediately before treatment to ensure that any partly-covered refuges are submerged for the biocide application. If the margins have abundant crayfish and the dewater area has little cover, it may help to provide some temporary refuge material in the water to encourage crayfish to move into the water.

It is recommended that representative samples of exposed margins are excavated prior to biocide treatment to check for the presence of crayfish. Presence of any live crayfish above water means the treatment should not proceed until the water level is raised and all potential refuges are inundated. Any exposed live crayfish would be highly likely to recolonize after the waterbody had recovered from biocide treatment and in that case the eradication treatment would fail. A two-stage treatment could be used, with an initial treatment of a partly dewatered pond, followed by a repeated treatment, after any surviving exposed crayfish had time to move to the recovered pond. This may be a useful approach anyway, increasing the likelihood of achieving complete eradication, but it would certainly increase the financial cost and the other resource requirements for the treatment.

Fishing ponds and ornamental lakes are periodically drawn down to facilitate management such as de-silting, liming, or the repair of dams and other reinforced banks. Where such ponds are occupied by invasive crayfish the dewatering may stimulate the search for new habitats, as seen in this field trial, when about half the crayfish walked over dry ground to try to find suitable habitat. In most cases, by the time the presence of a population of invasive crayfish is known to environmental agencies, the crayfish have already escaped via outfalls into the wider catchment. Where crayfish have established beyond the pond, roaming in response to dewatering is unlikely to make any difference to future invasion. If the crayfish have not yet escaped, however, there is an increased risk of them doing so if a pond is completely dewatered. Mitigation measures are recommended in that case. For example, a plastic sheet barrier, or other permanent barrier fencing around the site, would potentially eliminate the risk of overland escape of crayfish during restoration work on a pond, although modifications to any inlets and outfalls to a waterbody would also have to be considered carefully for longer term biosecurity. This study shows the importance of understanding the behaviour as well as the ecology of an invasive non-native species in planning any measures for its control.

5.5 Acknowledgements

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Chapter 6
Use of electric shock treatment to control invasive signal crayfish in streams

6.1 Introduction

The most important drivers of global biodiversity loss are habitat loss, climate change, overexploitation, pollution and invasive alien species (Millennium Ecosystem Assessment, 2005). Several species of freshwater crayfish have been introduced beyond their natural ranges, for food, angling bait, or aquaria (Lodge *et al.*, 2000) and have become invasive (Holdich *et al.*, 2009, Peay *et al.* 2010). They have significant impacts on species assemblages by competition and predation; have produced modification of habitats by grazing and burrowing, and within Europe the introduction of North American crayfish species has caused loss of many populations of native crayfish, due to transmission of the disease crayfish plague (Gherardi, 2007).

Where prevention has failed, land managers and other decision makers often seek measures to eradicate local populations of invasive species, or to limit their spread. Various unsuccessful attempts have been made to eradicate populations of invasive crayfish by trapping (Holdich *et al.*, 1999), although population density has been reduced in some cases (Bills and Marking, 1988, Hein *et al.*, 2007). Biocides have been used against invasive crayfish in still water sites, where treated water can be contained (Peay *et al.* 2006a, Sanddoden and Johnsen, 2010), but controlled treatment of running water is more challenging, as the available biocides are not specific to crayfish. Hence there is a growing need to eradicate invasions in watercourses, or failing that to control invasive crayfish populations effectively.

A potential method for control of crayfish is electrofishing (Holdich *et al.*, 1999). Electrofishing has been used to survey crayfish in shallow water (Westman *et al.*, 1979, Alonso, 2001); but as generally used (Beaumont *et al.*, 2002), it does not kill crayfish (Reeve, 2004). Electrofishing uses a relatively high voltage (typically c. 300 volts), but with only low power, in order to stun fish for a brief period. The pulsed electric shock initially causes movement towards the anode in fish (galvanotaxis). At closer proximity and hence greater strength of electric field galvanonarcosis occurs, i.e. muscle relaxation and temporary loss of swimming ability. This facilitates capture of fish in a hand-held net. If the power setting is not adjusted properly for the conductivity of the water, high voltage shock produces galvanic muscle spasm and can cause bruising of the fish and even permanent damage to the spine (Beaumont *et al*., 2002). By contrast, Westman *et al.* (1979) found that crayfish were less susceptible to electrofishing and often behaved in unexpected ways, sometimes using tail flipping to swim away, or else walking slowly out of refuges. Compared to electro-fishing surveys for fish, Westman *et al.* (1979) found it necessary to use equipment with higher voltage (500-700 volts) and power (0.5 – 1.5 amps, or 5 - 7 amps). Burba (1993) carried out tank tests and found that the response of crayfish to an increasing electric field was an initial physical movement of antennae and limbs at 0.03 – 0.3 volts cm^{-1}, including escape response; electrotaxis and increased locomotion, at pulse frequencies up to 10 Hz in an electric field of 0.1-1.0 volts cm^{-1}; followed by the onset of electronarcosis, at 1.0-1.7 volts cm^{-1}. Burba (1993) did not find any lethal effects at these field strengths, but the response to the highest voltages included loss of claws, a response also seen during electrofishing surveys by Westman et al. (1979).

Other aquatic macroinvertebrates have been found to respond to electrofishing by drifting downstream (Elliot and Bagnall, 1972, Blisson, 1976). Elliot and Bagnall (1972) found that three passes with conventional electrofishing gear were not lethal to macroinvertebrates, but led to approximately 5% loss of the benthos.

Electrical equipment has been developed to stun or kill lobsters and crabs in brine, prior to boiling for human consumption (Ogawa *et al.*, 2007, Neil, 2010). Electric shock treatment is also increasingly used in freshwater aquaculture to kill fish prior to processing for consumption (Lines *et al.*, 2003).

Here we examine the efficacy of a recently developed portable apparatus that is capable of delivering high power electric pulses in freshwater. The aim of this project was to test the efficiency of the apparatus in field conditions and determine whether an invading population of signal crayfish *Pacifastacus leniusculus* could be eradicated from a defined area in a stream.

6.2 Methods

The effectiveness of electric shock treatment at killing crayfish was examined by applying electric shocks to two enclosed sections of a small stream during late summer, when crayfish are active. Two treatments were carried out in the same stream sections: low intensity treatment, and then high intensity treatment using higher power equipment for a longer period. Stream sections were then de-watered, crayfish were removed and total mortality was determined. In addition, cage tests were used to measure mortality in response to increasing exposure to treatment.

Authorisation for trapping and removal of crayfish and use of the apparatus and process (UK patent no. 2480437) was obtained from the Environment Agency, subject to safety measures to protect staff, livestock and wildlife.

The study site was a headwater stream in North Yorkshire, England (at 54^{o} 02’ 33.16’’ N, 2^{o} 14’ 05.50’’ W) average width 1.5 m, average depth 0.17 m. It had steep earth banks, substrate mainly of cobble with underlying gravel, and it lacked aquatic vegetation, except mosses on some stones (Figure 6.1). Fish were not removed in advance, because previous surveys (Peay *et al.*, 2009) indicated that signal crayfish had replaced the fish population in the stream.

6.2.1 Equipment

The electrical equipment (designed and built by Electro Fishing Services Ltd) had three main parts: 1. a power supply, (portable 5 kW frame generator and a separate 230 volt generator for the pulse unit); 2. a capacitor unit to deliver power to a pulse unit, and 3. the pulse unit, which controlled the power, frequency and duration of DC (direct current) pulses (Figure 6.2). Metallic tape electrodes laid in flexible strips along the bed of the stream, delivered shocks in the water. Two different pulse units were used; the low power unit delivered 500 v pulses of approximately 20 kW; the ‘high power’ unit delivered *ca*. 96 kW; output, typically 1600 volts, 57.8 amps, at 7 Hz, with square pulses of width 4.4ms. For comparison, conventional electrofishing in this type of stream would deliver 300 volt pulses of *ca.* 0.5 kW.

6.2.2 Low intensity treatment

The low intensity treatment was applied to consecutive *ca.* 7 m long sections, 0.07-0.25 m deep in mid-August 2009 during conditions of low flow. Both sections had a slow-flowing glide or pool and a shallower run or riffle, typical of the stony stream. Two parallel electrodes were laid along the bed of the stream to treat both sections with shock cycles in immediate succession. To prevent crayfish escaping and to capture any stunned crayfish that were washed downstream, 4 mm mesh stop-nets were set across the upstream and downstream ends of the sections.

The treatment was carried out at night, when crayfish were seen to be active. Both sections were given a shock cycle of 2-minute shock, then 20 minutes rest, then a second shock cycle (section 2 given a shock immediately after section 1). The rest period was to allow crayfish to leave their refuges if they received only a non-lethal shock, which might make them more susceptible to a subsequent shock. When the equipment had been disconnected for safety, crayfish seen lying on the bed were collected by hand. The following morning the stream was dewatered and hand searched thoroughly. Crayfish sex, size (carapace length (CL), mm) and condition was recorded (as in Peay *et al.*, 2006a; Table 6.1). Surviving crayfish from the treated sections were taken off site, grouped according to their condition, to minimise agonistic interactions and placed in aerated 70 l tanks with shelters. Condition of crayfish was recorded again after 48 hours, although any dead crayfish were removed twice daily. Crayfish were also collected by hand from a control untreated area of the stream ca. 20 m downstream and their condition was recorded at collection and after 48 hours.

Table 6.1 Qualitative scale of crayfish condition

Reference	**Condition**	**Comments**
SR	Self-righting	Normal condition, no apparent effect, easily turns over if placed on back, in water turns over within a few seconds
SSR	Slow self-righting	Movement slow and often stiff, may take 1 minute or more to turn over if placed on back in water
NSR	Not self-righting	Lying on back, but still making voluntary movements of limbs, in water or air
T	Torpid	No voluntary movement, lying on back, will show slight movement of limbs when touched, but in more advanced stages may only show minimal response, or only movement of mouthparts, or eye-stalk response

D	Dead	No response to touching, no eye-stalk response, loss of rigor

6.2.3 High intensity treatment

The high intensity treatment was carried out in mid-August 2011, in the same two sections used for the low intensity treatment. The new high power (96 kW) equipment was used with three electrodes laid along the streambed (Figure 6.3), rather than two, to increase the power delivered (electrical field strength) during shock cycles. In order to compare efficiency of removal of crayfish and the incidence of damage among crayfish, a similar control section 50m downstream was given the same de-watering and removal of crayfish, without shocks.

It was expected that some (live or dead) crayfish would remain in banks where the crayfish could not be removed manually. Therefore crayfish were trapped and marked prior to the shock treatment. If live marked crayfish were trapped after the treatment, it would confirm that part of the population was inaccessible to manual removal and that there had been partial survival in the shock-treated sections. On the night prior to the start of treatment, crayfish traps (folding traps, 30 cm diameter, 60 cm long, two apertures 50 mm, mesh size 6 x 12 mm, baited with catfood) were set ca. 2-4 m apart in the glides in the stream sections. Four traps were set for one night in the area to be shock-treated, four in the control section and four between them. Trapped crayfish were marked on the carapace (using a yellow Dykem Brite-Mark® paint-marker) and then released into the section from which they had been caught, after the installation of stop nets at each end of the sections. The yellow marks were visible at night, when transects were walked to observe whether crayfish were active prior to the start of treatment, and subsequently.

Starting at night, the two sections were treated concurrently with cycles of 2-minutes high power shock and 25 minutes rest, during a period of 4 days, (total 49 shock cycles, cumulative shock time 98 minutes). The number of treatment cycles varied in each 24 hour period, for safety when other activities were carried out on site. There were minor variations in the power delivered during the treatment, due to natural variation in the conductivity of the stream. Conductivity rose slowly from 105 $\mu S\ cm^{-1}$ to 220 $\mu S\ cm^{-1}$ with the (natural) recession of flow during the period of treatment, whereas pH remained stable at pH 8.4.

Section 2 was dewatered after the initial 49 shock cycles and the crayfish were removed by thorough hand-searching. Sex, size, damage, mass and condition of crayfish (as in Table 6.1) were recorded, plus details of any marks.

Section 1 was given an additional treatment of 14 cycles of 15-minute high power shocks with 35-minute rests (a cumulative total of 63 cycles, 308 minutes). The rest period was increased because no crayfish had been seen emerging during the rest periods of 25 minutes in the first phase of the treatment. Section 1 (see Figure 6.4) and the control section were de-watered and crayfish were removed and recorded. Flow was restored in each section immediately after removal of the crayfish.

As it was possible that some live crayfish remained in inaccessible banks in the two shock-treated sections after manual removal had been completed, another session of dewatering was carried out on the last night of the trial and the sections were observed to see whether any crayfish appeared from refuges in the banks.

To investigate the effects of shock cycles on caged crayfish, crayfish were obtained from traps set ca. 100 m downstream of the study area. Batches of 15 individually numbered crayfish were put into cages (6 mm plastic mesh, 19 cm high, rectangular base 30 x 45 cm, with a loose covering of gravel and pebbles inside). The cages were placed in the stream for one or more shock cycles of one of the following: a) 2 minutes at high power, b) 2 minutes at low power, c) 15 minutes at high power, or d) no shock treatment, with rest intervals as for the treatment of the stream sections (Figure 6.3). Cumulative time of treatment of cages ranged from 2 to 30 minutes. In total, 19 cages of crayfish were treated with shock cycles, but two were excluded from analysis, one cage with survivors from the first night, i.e. three prior 2-minute shocks, and another cage that had interrupted treatment, leaving 8 with high power and 9 with low power cycles. After exposure to shock cycles, the crayfish were removed and their condition was recorded. Any survivors were kept in a cage in an untreated part of the stream and observed at twice daily for 48 hours. Marked survivors from the cage tests (n = 60) were released into the stream 3 m downstream of the shock-treated sections after all treatments had been completed. Traps were set three weeks later; any recaptures would confirm extended survival, albeit without allowing the proportion surviving to be estimated.

Figures 6.1 – 6.4 illustrate features of the field trial.

Figure 6.1 Stream section being prepared for electric shock treatment

Figure 6.2 Pulse unit for electric shock treatment

Figure 6.3 Test cages with crayfish set in channel for exposure to treatment

Figure 6.4 Dewatered channel and manual search for crayfish after high intensity treatment

6.2.4 Trapping after high intensity treatment

Traps were set for 1 night 20 days after the high intensity treatment, at the same positions used prior to treatment, in the shock-treated and control sections (4 traps in each), plus another 4 traps between.

6.2.5 Analysis

To compare mortality in the different treatments, G-tests were carried out with Williams corrections (Sokal and Rohlf, 2012). Mortalities of males and females and of different size classes were compared. G-tests were also used to compare the incidence of damaged claws in crayfish in shocked and un-shocked sections and then the incidence of damage in crayfish that survived shock treatment compared to those that did not. To investigate the mortality of crayfish in the cage tests, the cumulative exposure was calculated as number of cycles x duration of shock at low or high power. A generalized linear model with a binomial distribution was fitted to the dataset, to test the effects of total shock time (2 to 30 minutes), shock power (high or low) and duration of shock cycles (2 or 15 minutes) on crayfish mortality in cages, (expressed as number dead from 15). Analysis was carried out using SPSS 20.0. Goodness of fit was assessed and because the data were over-dispersed (Pearson Chi-square value/df = 2.269), a model was used with the scale parameter set as a Pearson Chi square distribution, rather than a fixed value. As the terms of shock power and duration of shock cycle were not significant, they were dropped from the model..

6.3 Results

6.3.1 Low intensity treatment

After the low intensity treatment (two low power shocks of 2 minutes), 357 signal crayfish were removed from the treated sections. When their condition was assessed 12 hours after the shock treatment, there was no significant difference in overall mortality between sexes (G test: $G = 0.76$, $df = 1$, $P = 0.44$). Crayfish mortality differed between the two sections (G test: $G = 37.75$, $df = 1$, $P < 0.001$): 13% in section 1 and 42% in section 2; likely to be due to the wider spacing of the two electrodes (0.5-1.0 m) through a pool in section 1 and hence reduced power.

The mortality of crayfish of different sizes was assessed to see whether there was any size-associated variation in susceptibility to shock treatment. Frequency distributions of crayfish were plotted in 5mm size classes (Figure

6.5), but for statistical analyses it was necessary to aggregate the classes into three classes (5-9 mm, 10-29 mm, 30-54 mm). There was no obvious effect of size on the frequency of mortality in either of the sections (section 1 G test: G = 0.87, df = 2, P = 0.64; section 2 G = 3.16, df = 2, P = 0.20).

The crayfish (n = 248, 70%) that survived the low intensity treatment were examined after a further 48 hours in order to assess whether their condition changed over time (Table 6.2). Delayed mortality occurred in 24% (N= 59) of the crayfish alive 12 hours after treatment, whereas only 10% (N=2) died in the control treatment; however the difference was not significant (G = 1.17, df = 1, P = 0.27). The surviving crayfish that were in poor condition at 12 hours (not self-righting, or torpid) were much more likely to die subsequently (86% mortality) than were those rated as normal (8% died, G = 81.6, df = 1, P <0.001). Cumulatively, the mortality of crayfish given the low intensity treatment increased from 30% at 12 hours to 46% by 60 hours post-treatment.

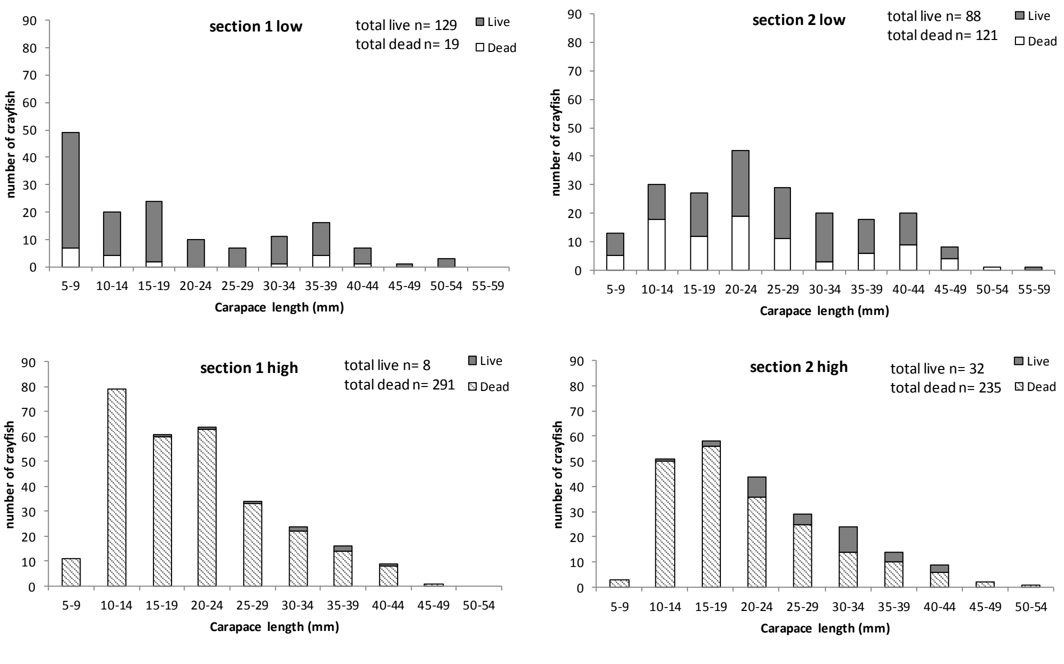

Figure 6.5 Mortality of signal crayfish in two stream sections given low intensity and high intensity electric shock treatment

Low intensity: 20 kW, total shock time 4 minutes, section 1 low, section 2 low); high intensity treatment: section 1 high: 96 kW, total shock time 308 minutes; section 2 high : 96kW, total shock time 98 minutes; total mortality by size class (carapace length, mm). Sections are the same sites, but high intensity treatment is 2 years later.

Table 6.2 Changes in condition of surviving crayfish between 12 hours and 60 hours after exposure to low intensity treatment (two low power shocks of 2 minutes)

	12 hours after exposure	**60 hours after exposure**				
condition	no. crayfish	Self-righting	Not Self-righting	Torpid	Dead	% mortality
Self-righting (control: no shock)	20	18	0	0	2	10%
Self-righting (after shock treatment)	203	186	0	0	17	8%
Not Self-Righting (after shock treatment)	21	2	0	1	18	86%
Torpid (after shock treatment)	24	0	0	0	24	100%

Note: see Table 6.1 for descriptions of crayfish condition

6.3.2 High intensity treatment

Mortality was significantly higher in the high intensity treatments (86 - 97%) than the low intensity treatment (13 – 42%), ($G = 404$, $df = 1$, $P < 0.001$), (Figure 6.5). None of the crayfish in the control section died (n=614). The mortalities of crayfish in the two sections given high intensity treatments were different in section 2 (86.4%) and section 1(97.4%) ($G = 10.922$, $df = 1$, $P < 0.001$), likely reflecting the longer programme of treatment in section 1.

As in the low intensity treatment, mortality rates for high intensity treatment were similar for males and females ($G = 0.295$, $df = 1$, $P = 0.44$), and results were pooled for subsequent analyses. Juvenile crayfish <30 mm predominated in both sections (85% and 80% in sections 1 and 2 respectively). Crayfish were aggregated into two size classes (5-29 mm and

30-54 mm) for a test of difference (as the number of survivors was too low to allow analysis with more size classes). Mortality was greater in smaller crayfish. In section 2, where overall mortality was 86.4%, mortality in the larger crayfish was only 66%, significantly lower ($G = 3060$, $df = 1$, $P < 0.001$) than the 92% mortality in smaller crayfish. In section 1, where overall mortality was 97.4%, the larger crayfish also had lower mortality (90%) than smaller ones (99%), but the difference was not significant ($G = 8.66$, $df = 1$, $P = 0.28$). Following dewatering, similar proportions of marked crayfish were retrieved by manual searching from the two sections given high intensity treatments; (16 of 22 marked crayfish recaptured in section 1, 15 of 21 in section 2; $G = 0.007$, $df = 1$, $P = 0.93$). Recovery of 71-73% of the marked crayfish indicates that the shocking and de-watering did not remove all crayfish, e.g. crayfish (live or dead) may have been missed from inaccessible crevices.

To investigate whether shock treatment caused crayfish to lose their chelae, the frequency of lost and regenerating chelae was compared in crayfish from the two shock treated sections ($n = 668$) and those in the control section ($n = 606$). In the control, 17% ($n = 101$) of crayfish had one or both chelae missing, but among shocked-treated crayfish the proportion was higher, 70% ($n = 469$) ($G = 371$, $df = 2$, $P < 0.001$). The loss of chelae could have been a direct effect due to shock treatment, or a response to it, i.e. autotomy as part of an escape response by the crayfish. Among the 60 crayfish that survived two or more high power shocks, the incidence of loss of chelae was 75%, which was not significantly different from the 71% loss of chelae in crayfish that died ($G = 0.29$, $df=1$, $P = 0.59$), suggesting loss of chelae was likely to be an effect of treatment.

Cage tests were used to investigate the effects of increasing total shock time. Total shock times in the range 2 – 30 minutes produced mortality in the range 0 – 100%. The mortality of the caged crayfish increased with total shock time (Figure 6.6). The fitted model was as follows: $y = 0.0089\ x \pm 0.015 - 1.555 \pm 0.242$, where $y = \mathrm{Ln}\ (n/1-n)$, n = number of dead crayfish, x = total shock time; Wald chi-squared = 14.862, $df = 1$, $p < 0.001$.

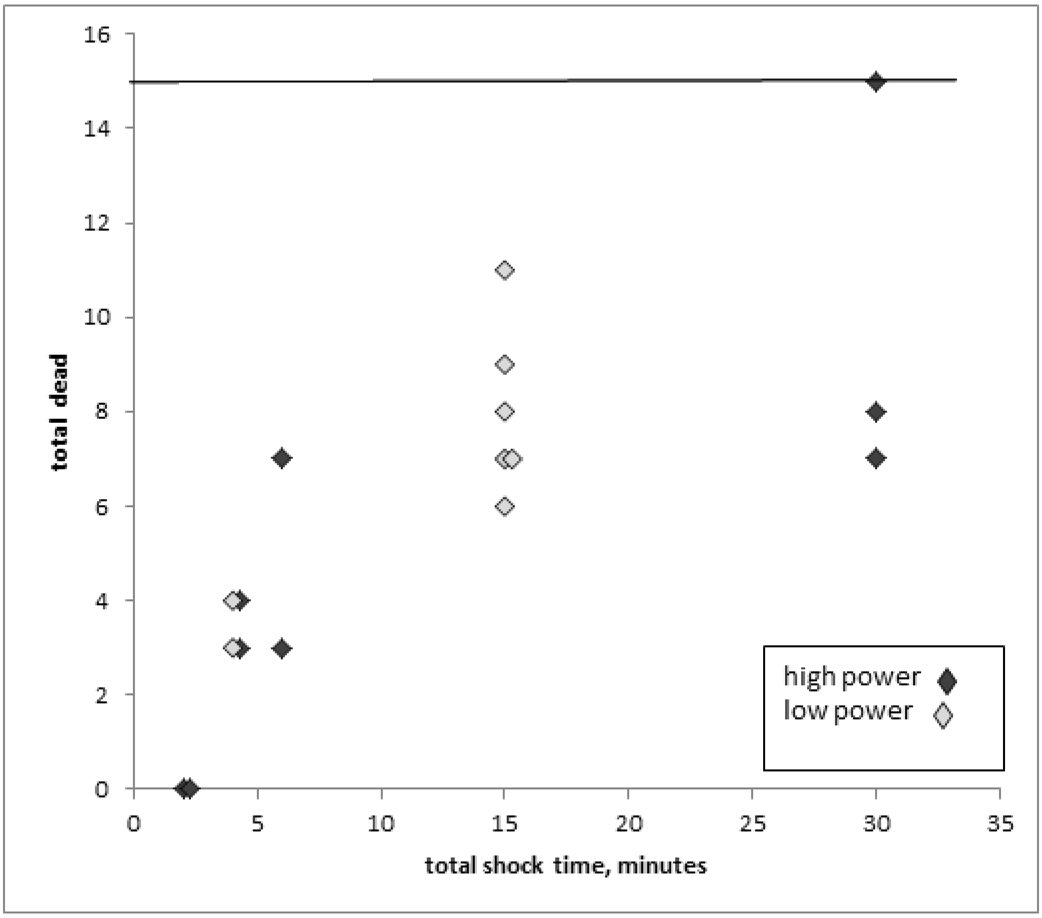

Figure 6.6 Mortality of crayfish (n = 15 per cage) exposed to different cumulative total shock times in cage tests at high power (96 kW) or low power (20 kW).

6.3.3 Trapping after high intensity treatment

Nine (15%) of the (marked) survivors of cage tests that were released after the period of shock treatment were recaptured during trapping three weeks after treatment and all were normal (self-righting).

The trapping 20 days after treatment was also used to examine re-colonisation of the stream sections, following shocking and removal of crayfish (Table 6.3). Despite the removal of crayfish during the treatment, there was no significant difference in the total catches between the combined shock-treated sections and the control (4 trap sites in each) before and after treatment ($G = 0.03$, $df = 1$, $P = 0.85$). Trapping after shock treatment captured higher numbers (e.g. 42 crayfish in one trap) at the upstream end, but only a few (1-3 crayfish per trap) in the others, suggesting re-colonisation from upstream into a section depleted of crayfish. This effect showed in differences in the catches in individual traps before and after treatment in the shock-treated sections ($G = 44.2$, $df = 3$, $P < 0.001$) and the control section (G = 8.5, df = 3, P =0.035). The crayfish trapped before and after treatment showed similar size distributions (G = 5.68, df = 4, P = 0.22; sizes aggregated into five classes for analysis).

Table 6.3 Crayfish trapped before and after treatment

Crayfish per trap						
	Shock treated (Section 1 and 2)		Control		Stream between shock-treated and control	
Trap number	before	after	before	after	before	after
1	10	42	12	6	10	10
2	9	3	7	16	13	10
3	10	2	27	13	9	5[1]
4	16	1	22	34	9	ND[2]
Total	45	48	68	69	41	25

[1] Location shock-treated during cage tests at low power

[2]ND no data, trap displaced overnight, probably by cattle

6.4 Discussion

The aim of the this trial was to investigate whether an electric shock treatment of invasive crayfish could be carried out under field conditions and achieve eradication, i.e. complete mortality of the population within the treated area, using the portable apparatus. High intensity treatment achieved high mortality (86-97%) of signal crayfish, but low intensity treatment was much less effective (13-42% mortality). The low power treatment would not be sufficient to provide effective control of the population, due to the high reproductive capacity of the crayfish. Cage tests were used during the treatment of the stream to investigate whether short sessions of treatment were as effective as the extended trial at high power. This showed that mortality increased with the cumulative shock time, both at low and high power. There was some delayed mortality in the crayfish within 48 hours, but other individuals appeared to fully recover from treatment and survived in the stream for at least three weeks. Although the high intensity treatment was run for several days, the cage tests suggest that the maximum achievable mortality could be produced with a lower total shock time at high power than was applied in the field trial. The apparatus has the capacity to treat longer sections of watercourse in succession than the short lengths in this study.

Most of the crayfish that were found still alive after the full programme of high power treatment were in refuges in the banks, or sheltered by boulders up to 1m across. This problem has also been reported in conventional electrofishing (Reeves, 2004). Mark and recapture work showed that approximately 27% of crayfish in the enclosed stream sections were not found during dewatering and manual search and some marked survivors were recaptured three weeks after the treatment. Hence, the true mortality is likely to be lower than the 97% that was recorded in the high intensity treatment using manual removal. Nonetheless, even if 27% of the crayfish were hidden and all were assumed to be alive, the minimum mortality would be 77% of the total population.

Immigration from adjacent untreated areas occurred soon after treatment (within three weeks) and this is in accord with rapid immigration of crayfish into areas with reduced density recorded by Moorhouse and MacDonald (2011). Mortality was not complete and survivors could also re-start the population. With annual recruitment and at least one year for juvenile signal crayfish to reach maturity, usually two to three years (Guan and Wiles, 1999; Lewis, 2002), recovery of the population would be expected to take two years or longer.

This type of treatment should be carried out in summer, when crayfish are active and more vulnerable to shock treatment than in the winter when they stay in refuges (Peay unpublished). Two options for treatment efficiency are: increasing the power and improving treatment of the banks. An increase in power could be achieved by (a) using additional, more closely spaced, electrode tapes,(b) higher power equipment (c) reducing the water level provided refuges remain submerged, (d) increasing conductivity (e.g. by addition of any electrolyte). The treatment of the banks by electric shock will be influenced by the substrate, its water retention and ion content. Banks with wet sand would have higher conductivity than those with a high content of stone or clay, and sand artificially saturated with brine might allow use of additional electrodes spiked into the bank, but further investigation would be required to assess the scope for improvement. Unless the issue of bank treatment can be addressed successfully, electric shock treatment seems unlikely to achieve eradication of populations of invasive crayfish species.

Where eradication of an invasive species is not achieved within a defined period, a treatment or other measure can only be considered to be a control measure (IUCN, 2000). Crayfish trapping is a control measure, rather than an eradication treatment (Gherardi *et al.*, 2011). Electric shock treatment has advantages over trapping in being effective against all sizes of crayfish, rather than just large crayfish (Brown and Brewis, 1978, Abrahamsson, 1981) and because it requires only one short session, rather than long-term annual effort, as with trapping (Hein *et al.*, 2007). A disadvantage of electric shock treatment is that, like biocide treatments, it has impacts on non-target fauna. Biocide treatment is the only successful eradication method so far against signal crayfish populations (Chapter 4). The main advantage of electric shock treatment over biocide treatment is that it has no impacts outside the treated area, but in contrast to biocide treatment, it is only suitable for very shallow waters. It may have some potential as a pre-treatment for a high density population, prior to use of a longer-term control measure that has lower impact. Electric shock could also be considered for periodic treatment of a localised area in conjunction with a physical barrier to upstream invasion by crayfish, in that keeping the invading population at low density downstream of the barrier would reduce the risk of it being overcome.

Electric shock treatment does not provide a simple solution to the problem of non-native crayfish in the increasingly invaded catchments across Europe, but it does offer a new tool that may have some specific applications in the management of invasive crayfish. It may also have greater potential for eradication of invasive aquatic species that are not as refuge-dependent, for example, invasive fish or amphibians in shallow waterbodies.

6.5 Acknowledgements

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Chapter 7
A cost model for decision-making on management of signal crayfish: to eradicate or not

7.1 Introduction

Where there is potential for introduction, establishment and spread of invasive non-indigenous species, best practice recommendations (IUCN, 2000, Hulme, 2006) give a hierarchy of management response: prevention, eradication and if these are unsuccessful, control to reduce spread or mitigate impacts may be appropriate. Whether eradication should be viewed as a potential approach to invasive exotic species depends both on an evaluation of the costs and benefits of programs, and on their potential to be successful (Myers et al., 2000). There is a need to act rapidly and decisively when populations of invasive species are first detected to have the best chance of successful eradication or control (Mack et al., 2000b, Simberloff, 2009, Pluess et al., 2012b). Yet resource managers do not necessarily have decision-making tools to help them decide on effective and cost-effective action; sometimes there is only indicative guidance at best and this is seen as a constraint on effective action (Anderson, 2005, Britton et al., 2011). Decision-making about surveillance for invasive species and response to invasion is often limited by the availability of resources.

Cost models have long been used in agriculture for decision-making on the control of pests (Mumford and Norton, 1984), but there has been an increasing use of such tool for various biological invasions to assess the relative costs and benefits of management action, e.g. for mink *Mustela vison* in Spain (Zuberogoitia et al., 2010), feral ungulates in Australia (McMahon et al., 2010), zebra mussels *Dreissena polymorpha* in the Great Lakes (Leung et al., 2002) and rusty crayfish *Orconectes rusticus* in lakes in Wisconsin (Keller et al., 2008).

Crayfish species from North America introduced into Europe and elsewhere have become highly invasive, causing significant impacts on indigenous species (Gherardi, 2007b, Holdich et al., 2009d). The three most widespread and damaging of these invasive species are the signal crayfish *Pacifastacus leniusculus*, red swamp crayfish *Procambarus clarkii* and spiny-cheek crayfish *Orconectes limosus*. To date, management options to eradicate localized populations of invasive crayfish appear to be limited to complete physical destruction of habitat, e.g. infilling a pond, or else the use of a non-selective biocide. Both are expensive and have impacts on non-target species, so the feasibility of eradication treatment, its acceptability and costs need to be considered carefully, especially when the work is funded by public bodies. Consideration of costs and benefits of local eradication is all the more important when an invasive species is already too widely distributed to make it feasible to eradicate or control at national scale, as is the case with the signal crayfish. This species was introduced into Europe in the second half of the 20th century, is present in most countries in Europe (Holdich et al., 2009d) and is now invading most of the river catchments in England (Rogers and Watson, 2011) and several catchments in Scotland (Gladman et al., 2009).

Signal crayfish and other invasive crayfish species can have impacts on ecosystem processes by physical modification of habitats and by modification of energy flow within aquatic systems (Gherardi, 2007b). The impacts arise from perturbation of substrates; the shredding and consumption of aquatic macrophytes, increased cycling of nutrients through processing and consumption of detritus, and predation of aquatic invertebrates. Crayfish are consumed by predatory birds and mammals and by fish, however it is increasingly recognised that the interaction with fish species can be two way, with crayfish capable of having negative impacts on fish populations.

There are several pathways for effects on fish: direct predation of fish eggs and fry; competition with fish for shelter; competition for food resources where there is overlap in trophic niche; modification of habitat that makes it less suitable for one or more life-stage of fish, for example by reduction of macrophytes used by fish for spawning or shelter, or by release of fine sediments by burrowing, which subsequently deposit in gravel beds used for spawning, reducing oxygen exchange and hence reducing survival of eggs.

Crayfish predate fish eggs and fry. Signal crayfish appear to have difficulty finding buried eggs of Atlantic salmon *Salmo salar* (Gladman et al., 2012), but do predate eggs when they have access to them and newly emerged alevins (Edmonds et al., 2011, Hayes, 2012, Findlay, 2013). The eggs and fry of other fish species are also consumed, including brown trout *Salmo trutta* (Findlay, 2013). Peay et al. (2009a) found injuries and mortality in juvenile brown trout consistent with attacks from signal crayfish. Crayfish can compete with fish for shelter, especially benthic fish (Carpenter, 2005, Light, 2005), such as sculpin species. Reduced density of bullhead *Cottus* *gobio* in the presence of signal crayfish has been recorded in several studies in England (Guan and Wiles, 1997, Bubb et al., 2009, Findlay, 2013). Griffiths et al. (2004) showed that signal crayfish were able to displace juvenile salmon from shelters, and lack of shelter reduces the growth and survival of salmon fry (Finstad et al., 2007).

In addition to direct predation, crayfish can also indirectly affect fish populations through predation of invertebrates and habitat modification. Burrowing by abundant populations of crayfish can increase downstream sediment transport (Statzner et al., 2001). Subsequent accumulation of sediment can reduce the suitability of spawning sites for salmonids (Sear, 1993, Soulsby et al., 2001) and Findlay (2013) found reduced survival of trout eggs due to deposition of fine sediment due to the activity of crayfish.

These factors in combination can affect the populations of fish in the presence of invasive crayfish. For example, the growth and condition of chub *Squalius cephalus* were reduced in the first four year classes when signal crayfish were present, although large individuals benefited from the availability of crayfish prey (Wood, 2008, Hayes, 2012). Holdich et al. (1995) found that sites on a lowland stream with well-established signal crayfish had lower density of trout overall than those with white-clawed crayfish or no crayfish and fewer 0+ brown trout and bullheads.

There are some studies, however, that have not detected any significant effect of signal crayfish on salmonid fish. Degerman et al. (2007) did not find any significant impact of non-native signal crayfish or native noble crayfish *Astacus astacus* in fisheries surveys in streams in Sweden, compared to streams without crayfish (>0 – >25 crayfish 100 m^{-2}), albeit at lower abundance than that recorded in some watercourses in the UK. Stenroth and Nystrom (2003) found that brown trout survived equally well in enclosures with signal crayfish as those without crayfish. In the margins of a large boreal lake in the early stage invasion by signal crayfish, Ruokonen et al. (2012) found that, although there was a large dietary overlap between benthic fish and crayfish, there was no significant difference in the abundance of fish compared to uninvaded sites in the lake.

By contrast, Peay et al. (2009a) (see Chapter 3) found a strong negative relationship between abundance of signal crayfish and density of brown trout and bullhead, in a shallow headwater stream in northern England. In areas with a dense population of signal crayfish there was extirpation of bullhead and subsequent loss of recruitment by brown trout. Subsequent monitoring of fish and crayfish populations in this stream has shown impacts on fish extending downstream over time as the crayfish invasion progresses (Peay et al. unpublished) . A smaller population of resident adult trout in the stream than in the lake, and hence less predation of juvenile crayfish, may be possible factor in the greater impact of crayfish in the stream. Similar reductions in juvenile brown trout and bullhead have been recorded by Findlay (2013). Hence, although in large waterbodies there may be limited effects, there is growing evidence of impacts of signal crayfish in upland nursery streams for salmonid fish in England. This is discussed further in section 7.2.2 below.

The impact of crayfish may be masked in part by density-dependent growth and mortality in brown trout populations, with reduced growth and survival at high density and compensatory growth at reduced density (Jenkins et al., 1999, Lobón-Cerviá, 2009). Similar growth response occurs in salmon (Davidson et al., 2010, Imre et al., 2010), although temperature and stream discharge are the major factors, affecting both spawning and subsequent survival and growth. In addition to any impact of invasive crayfish, there is predation of eggs by parr during spawning (Youngson, 2007), competition within 0+ year class (Gee et al., 1978, Elliott, 1996) and between year classes (Egglishaw and Shackley, 1985). These factors make it difficult to predict relationships between breeding stock and subsequent recruitment to juvenile and adult year classes from year to year.

Hence, crayfish affect fish populations both directly, through predation and competition for shelter and indirectly, by disturbance of habitat and thus can reduce the recruitment of fish. This has potential for social and economic impacts on fisheries and anglers. Despite the difficulties in quantifying the impact of invasive crayfish in detail, in principle, where adverse impact of invasive crayfish occurs, it could be mitigated by stocking with farmed fish to meet the demand for recreational angling, although in practice the benefit of doing so is not certain.

This study provides the first cost-benefit analysis of the control of invasive crayfish populations using either biocide or trapping. The cost of invasion is based on an estimate of the environmental impact of crayfish on salmonid fish. In this study, the cost of restocking invaded waterbodies with Atlantic salmon and brown trout was used to provide a surrogate cost for the environmental impacts of invading signal crayfish.

The cost model was developed for Scottish Natural Heritage, the statutory agency which advises government in Scotland on biodiversity; to help environmental authorities and resource managers assess the costs and benefits of attempting to eradicate invasive crayfish if a population is found in a catchment for the first time. Hence, it is a tool for risk assessment of potential invasion, or as part of a rapid response when new populations are detected.

The model was used to predict the environmental cost of invasion, calculated as the cost of mitigating the impact of signal crayfish on the recruitment of salmonid fish by annual rearing and restocking of invaded watercourses. Control of invasive crayfish is difficult, not least because so many populations are well established and the opportunity for eradication has already been missed, such that crayfish will be able to extend their range within each invaded catchment, with corresponding impacts. In this study short term and long term costs of eradication treatment, a control measure and an environmental cost of no control.

Three scenarios were compared:

1. no control, i.e. impact and restocking cost;
2. eradication using a biocide treatment, and
3. control using trapping.

The management measures of biocide treatment and removal of crayfish by trapping have both been carried out in Scotland, but they have different applications. Biocide treatment has been used when there was the potential to treat and kill the whole population of crayfish. Trapping of crayfish is not permitted in Scotland, except with the consent of the Scottish Executive. Apart from authorised use of traps as part of surveillance for new populations of invasive crayfish, only two trials of crayfish control using trapping have been allowed, one in the River Clyde (Reeve, 2004) and another in Loch Ken (Gladman, 2012).

Biocide treatment has been used against signal crayfish (Peay et al., 2006a, Sandodden and Johnsen, 2010), with some success. Projects in Britain have been solely with natural pyrethrum, which degrades rapidly after use (Peay et al., 2006a), although the treatment in France (Laurent, 1995) used fenthion, an organophosphate and in Norway (Sandodden and Johnsen, 2010) cypermethrin, a synthetic pyrethroid. The synthetic pyrethroids are by far the most toxic to crayfish, but take longer to degrade than natural pyrethrum. None of the biocides is selective to crayfish, which means there are always impacts on non-target fauna.

Physical removal of crayfish using traps has commonly been used to harvest crayfish for human consumption. Trapping is generally considered to be a control measure, at best, not an eradication method (Gherardi et al., 2011), but it is frequently proposed as a management option when dealing with invasive crayfish. It is usually assumed that by increasing harvesting effort the total population of invasive crayfish could be reduced. If a large enough reduction was achieved, it would have the potential to mitigate the impacts of crayfish in invaded areas or perhaps even reduce the rate of invasion. Studies showing a reduction of ecological impact of signal crayfish as a result of trapping for control are somewhat lacking. Nonetheless, in one example trapping in a length of English river for eight years reduced a population of signal crayfish to a level at which nuisance to anglers was alleviated and erosion of banks by burrowing was reported as having been reduced (West, 2010). There is little evidence that trapping for control has any effect on the rate of invasion. Moorhouse and MacDonald (2011) found that immigration/emigration rates of signal crayfish remained the same through a zone where trapping for control was used. Nonetheless, trapping has been considered here as a management option.

7.2 Methods: development of a cost model for the assessment of eradication treatment or control programme

The cost model developed in this study simulated the expansion of an invading population of crayfish along the watercourses of a catchment over time and applied an environmental cost to the total length of watercourse occupied by crayfish each year. A biocide treatment was included as a once-only cost, i.e. successful eradication. Control by trapping was included as an annual cost in the total length of invaded watercourse. The model was set up as a spreadsheet, to make it simple for users to modify variables for individual catchments or projects and see the workings as effects on the costs annually and cumulatively. Annual inflation of costs and discounting to Net Present Value were not used in the model.

Rates of invasion and costs of management were derived from literature, used as variables in the model and tested in a case study.

7.2.1 Catchment model

The cost model was developed to show the progressive invasion of a theoretical catchment. Rates of invasion by signal crayfish were obtained or calculated from literature, as shown in Table 7.1 and ranged from less than 0.1 km y^{-1} to 12.9 km y^{-1}. Consistently slower rates were recorded upstream than downstream and in higher energy streams than in larger lowland rivers. Rates were lowest during the establishment phase of a new population. The effect of rate of invasion was tested in the model using three different rates (Table 7.2).

Table 7.1 Observed rates of upstream and downstream invasion by signal crayfish in rivers

Rate km yr^{-1} downstream	**Rate km yr^{-1} upstream**	**Years**	**Location**	**Author**
0.18	0.06	1996 (introduction?) - 2003 – 7 years	River Ure, North Yorkshire	(Bubb et al., 2005)
0.41	0.11	1997-2007 11 years	River Glen, tributary of River Stour, Suffolk	(Wright, 2009)
0.46	0.1	2002-2008 6 years	Bookill Gill Beck, River Ribble, North Yorkshire	(Peay et al., 2009b)
0.56	nd	1998-2001 3 years (introduction 1976)	Broadmead Brook, Wiltshire	(Spink and Rowe, 2002)
0.65	0.25	1990 (introduction) to 2002 12 years	River Clyde	(Reeve, 2004)

Rate km yr^{-1} downstream	Rate km yr^{-1} upstream	Years	Location	Author
1.5		2002 -2009	River Clyde	(Sinclair, 2009)
0.26, 0.87	0.05; 0.25	2010	River Clyde tributaries (calculated from radio-tagging)	(Gladman, 2012)
0.75	0.43	1983 (introduction) to 1993 8 years	River Bain, Lincolnshire	(Holdich et al., 1995)
1	nd	1984 (introduction) to 1993 9 years	River Great Ouse, Bucks.	(Guan and Wiles, 1996)
1	nd	1992-2000 9 years	Gaddesby Brook, Leicestershire	(Sibley, 2000)
1.6	nd	1996	River Thame	(Ibbotson et al., 1997)
1.27 2.4 1.5	nd 0.47, 0.35 0.3	1985 (introduction) to 1997 12 years 1997-2002 6 years 2002-2009	River Wharfe, Yorkshire	(Peay and Rogers, 1999) (Bubb et al., 2005) (Imhoff et al., 2011)
2.3	0	1991-1993	River Stour,	(Wright and Williams,

Rate km yr^{-1} downstream	**Rate km yr^{-1} upstream**	**Years**	**Location**	**Author**
			Norfolk	2000)
0.9	0.26	1993-1997		
0?	0.77	1997-2000		
1.1	?	2000-2007		
0.5-1	nd	2001 to 2005 – (introduction 1995)	Lake Geneva southern shore	(Dubois et al., 2006)
12.9	nd	2003 to 2009 (expansion from Slovenia)	River Mura, Croatia	(Hudina et al., 2011b)

Table 7.2 Example rates of invasion used in the cost model

Invasion rate scenario	**Invasion rate downstream km y^{-1} (d)**	**Invasion rate upstream km y^{-1} (u)**
typical	1.5	0.5
slow	0.5	0.2
fast	2.5	1

The model was set up assuming that crayfish established in the main river, spread in both directions and invaded each tributary progressively at the upstream rate, increasing the cumulative length of invaded river each year, but with only one order of tributaries from the main river. Variables are shown in Table 7.3.

Table 7.3 Model variables for catchment invasion

Model variables	
Rate of invasion, km y^{-1} downstream, upstream,	d, u
Total length of watercourses in catchment, r	w
Tributary factor, average distance on main river between tributaries, km	t
Number of tributaries reached during invasion, downstream, upstream	n, m
Years elapsed since introduction	y
Total length of invaded main river, Total invaded length of watercourses	r l

In year y the distance invaded along the main river was:

$r = dy + uy$ (1)

The number of tributaries reached by the invasion in downstream (n) and upstream (m) were:

$n = dy/t$ (2)

$m = uy/t$ (3)

Total progress of invasion upstream of the introduction in year y was the sum of the total length of invaded main river upstream, plus the invaded length on each tributary, in which invasion had also progressed at upstream rate u. The total length of watercourse invaded upstream, ($l_{upstream}$), was:

$l_{upstream} = uy+uy(m+1)/2$ (4)

In the downstream direction, the progress of invasion was similarly calculated, using the downstream rate (d) on the main river, and the upstream rate (u) in the tributaries, with a subtraction of a tributary term to avoid duplication of calculation in the first tributary.

$l_{downstream} = dy+uy[(n+1)/2]-uy$ (5)

Combining invaded lengths upstream and downstream (equations 4 and 5) and substituting the number of tributaries (m and n) gave a total length of invaded watercourse at year y as follows:

$ly = (dy+uy)(1+uy/2t)$ (6)

The tributary distance was not measured as actual distance in individual catchment, but was used as an adjustment factor to distribute the tributaries and hence the invasion within the modelled catchment. Examples from actual catchments were used in calibrating the model (Table 7.4). The calibration method involved setting the model with known main river length and total river length for a catchment, and then adjusting the tributary distance factor incrementally until the year in which both main river and catchments were fully invaded was approximately the same. Actual distance between tributaries would vary within each catchment, although the average tributary distance is related to the drainage density in the catchment, i.e. the average length of channel required to drain a unit of catchment area. A tributary distance factor of 0.9 km was used for cases in which only the length of main river was available. The effect of varying the tributary distance factor is shown in Table 7.5.

Table 7.4 Example catchments and tributary distance factors for cost model

Catchment	**Catchment area, km^2**	**Length of main river, km**	**Total length of watercourse**	**Tributary distance factor**
Tay	4,970	193	4980	0.99
Clyde	312,000	170	4244	0.85
Spey	3,000	157	3650	0.87

Source of river data:

1 Tay District Salmon Fisheries Board

2 Clyde River Protection

3 Spey Fisheries Management Plan

Table 7.5 Effect of variation in tributary distance factor

Tributary distance factor	**Effect on invasion**
calibrated value	Main river and tributaries fully invaded in same year
greater value	Invasion of tributaries continues after main river fully invaded
lower value	Total length of watercourse invaded reaches maximum r before full invasion of the main river (or this is an option to increase total length of tributaries to compensate for only one order of tributaries)

7.2.2 Cost of impact using restocking as a measure

The application of environmental impact cost in an invaded catchment was applied in two different ways, both using the restocking cost of salmonid fish. The first approach, for migratory fish, was to use estimates of the total number of returning adult fish in the catchment and apportion the restocking cost required to replace that number of adults in future across the watercourses of the catchment by stocking large juvenile fish (smolts). The second approach was to apply a restocking cost per kilometre for non-migratory fish, in this case brown trout (disregarding the migratory component of the population, the sea trout).

For migratory salmon, the approach used was to consider the number of returning adults and estimate the number of hatchery-reared smolts required to produce the returning adults. There is uncertainty about return rates of salmon, although estimates are made by the Fisheries Trusts or District Salmon Fisheries Boards in catchments in Scotland, based on surveys of fry, angling catch returns, net catches and fish counters. It has been suggested that marine mortality has increased leading to fewer returning adult fish from stocking smolts than from stocking at smaller sizes (Youngson, 2007). McGinnity et al. (2004) found poorer survival (smolt to returning adults) of stocked smolt (3%) compared to wild smolt (11%) in Ireland and similar low return occurred in a river in Norway; Saltveit (2006) recorded 2% survival. For the cost model a ratio of 100 hatchery-reared smolts per returning adult was used.

For brown trout, the assumption made in the model is that if recruitment of brown trout is reduced in the presence of signal crayfish, parr would be stocked at 10 cm length or more into tributaries. Fin damage from signal crayfish is seen in fish of that size (Peay, 2009b), but parr may be better able to survive than small fry. The National Fisheries Classification (Mainstone et al., 1994, Wyatt, 2002) gives expected densities of trout for different grades, from A to F, where A is good and F has no fish, as shown in Table 7.6, which shows stream areas and corresponding fish populations for trout parr at the minimum density for each grade. Densities of trout parr differ between streams, but densities equivalent to A and B grades are recorded in Scotland (e.g. 19-33 of 1+ trout per $100m^2$ in Shelligan Burn, Perthshire (Egglishaw and Shackley, 1985). From the table, a nominal stocking rate of 400 trout km^{-1} in the model would represent restocking of streams less than 5 m with grade B as the target or a lower proportion of the expected population in larger streams. By contrast, stocking rates recommended by Aprahamian et al. (2003) for trout in British rivers were in the range 2-10 m^{-2}, equivalent to 5,000 to 25,000 km^{-1} for streams as shown in Table 7.6.

Table 7.6 Estimated minimum population of trout parr per km, derived from National Fisheries Classification grades A to D

	Grade	**A**	**B**	**C**	**D**
		minimum density trout parr per $100m^2$ for Grade			
		21	12	5	2
Stream width m	Stream area m^2 km^{-1}	Number of trout parr km^{-1}			
1	1000	210	120	50	20
2	2000	420	240	100	40
3	3000	630	360	150	60
4	4000	840	480	200	80
5	5000	1050	600	250	100
10	10000	2100	1200	500	200

As loss of salmonid recruitment would not be 100% in most cases, a factor for loss due to signal crayfish was included in the model, i.e. the proportion of annual recruitment in reaches invaded by crayfish compared to uninvaded reaches. There is limited information on the actual loss of salmonid recruitment in nursery streams in the presence of signal crayfish and as such the calculations are based on estimated losses, although simple adjustment of the loss factor would allow the model to be customized to individual catchments, or in light of better estimates of loss.

Based on Peay et al. (2009, see Chapter 3), the loss factor used was 0.1, representing potential losses with high density populations (e.g. with trap catches >1.0 crayfish trap^{-1} night^{-1}) in headwater streams. A limitation of Peay et al. (2009) is that it was a study of a single stream in the Ribble catchment in North Yorkshire.

Since then, a study has been carried out by Findlay (2013) on a series of small tributaries of the River Tees, another upland river with hard water, with similar headwater streams, dominated by bullhead and used as nursery areas by brown trout. Findlay carried out surveys of fish and crayfish at 20 sites and by using a range of environmental variables and the densities of fish and crayfish tested a series of models to find those that would provide the best and most parsimonious fit to the observed data. For the density of 0+ (young of year fry) brown trout the best model predictors, which accounted for 83% of the variation, were habitat factors (% unembedded gravel and % riffle and cascade) together with the density of signal crayfish. From the model, the density of brown trout fry was 9.27 times higher in the absence of signal crayfish at their highest observed densities. Similarly, a model for bullhead also showed a positive correlation with habitat (% glide) and a negative relationship with signal crayfish. The best model predicted bullhead density 3.85 times higher in the absence of signal crayfish. The findings of Findlay (2013) are similar to those of Peay et al. (2009), but from more streams in a similar invaded catchment.

The study by Degerman et al. (2007) in Swedish streams shows that impacts of signal crayfish on fish are not evident in all cases and indeed, they may not be in some streams and larger rivers in Great Britain either. Nonetheless, even if the scale of the impact varies within individual catchments, the cost model provides an indication of the cost implications of the effect.

At lower densities, such as the leading edge of an invading population, there is insufficient information on impacts to be sure about loss factor. In practice, the loss factor in an actual catchment might vary temporally and spatially. Model sensitivity was tested with loss factors ranging from 0.05 to

0.9. Lag time between impacts on juvenile fish and the resulting adult stock was not included in the model.

Variables used in the restocking cost model are shown in Table 7.7, together with examples for the Tay catchment.

Table 7.7 Cost model variables: restocking costs

Restocking cost variables	**term**	**Value applied**	**Example for River Tay**
Number of returning adult salmon	a	User enters number of returning salmon	Est. 50,000 salmon[1].
Cost per salmon smolt stocked	f_s	£0.50	£0.50 locally reared smolt
Reared smolt per returning adult	b	100	
cost to restock full catchment with salmon smolts	v	v calculated	Stock with 100x locally reared smolt @ £0.50 = £2.5 million
Cost per 10 cm trout stocked	f_t	£0.75 (sample price range £0.43-£0.95)	
Trout stocking rate, number km^{-1} (full restocking)	q	User selects from table 9.6., e.g. 400 trout km^{-1}	Stock with 400 trout km^{-1} = £300 km^{-1}, or £1.49 million
Proportion of watercourse for trout	p	1	
Proportion natural recruitment lost due to crayfish	s	0.1	

[1]source: (Summers, 2009),

The cost of restocking a whole catchment with salmon (v) was derived from the number of returning salmon (a), the unit cost of reared smolt (f_s) and a restocking factor (b):

$$v = af_sb \qquad (7)$$

The loss factor for crayfish was then applied to the total length of invaded watercourses, as calculated in equation (6). The annual cost at year y is then:

C_{sy} = ([1-s]v/r)(dy+uy)(1+uy/2t) (8)

For brown trout, the restocking cost was calculated in similar fashion, with the proportion of length as trout habitat included. There are differences in habitat used by brown trout and Atlantic salmon (Bremset and Heggenes, 2001, Armstrong et al., 2003), partly due to interspecific competition. When both salmon and brown trout were included in the model, using a reduction factor for the proportion of length as trout habitat gave the option of including some partition of habitat within the catchment. Hence the cost per kilometre was calculated as the product of stocking rate, cost per trout and proportion of watercourse used or stocked:

$w = qf_tp$ (8)

This cost (8) was applied to the length of invaded watercourse from equation (6) and the loss factor for crayfish was applied

C_{ty} = ([1-s]w/r)(dy+uy)(1+uy/2t)

The environmental impact (as a restocking cost) were compared over time under different scenarios. Different management scenarios for signal crayfish invading a catchment have been used: an eradication treatment, using a biocide on a localized population in the early stage of invasion prevents all further costs of environmental impact from crayfish in subsequent years; a control treatment, using crayfish trapping to reduce the abundance of the population, but without preventing expansion of range; or no management, i.e. restocking cost only.

7.2.3 Cost of biocide treatment

The costs of biocide treatment consist of the cost of the chemical product, the cost of any hydraulic control required to prevent biocide being released into areas not intended for treatment, and labour costs. Chemical costs depend on the size of waterbody to be treated and the dosage applied. Hydraulic control costs may not apply on wholly enclosed site with no risk of leakage, but may be high when large-scale continuous pumping is required, especially if control is required throughout treatment and recovery. Labour costs depend on the scale and complexity of the project and the unit costs, which vary depending on use of contractors, staff and volunteers.

The costs of previous projects are shown in Table 7.8 where known, with the total cost of chemical used (based on the price at the time of treatment), the cost incurred (outlay cost), including materials, equipment, paid labour and any other outlay costs. An estimated cost (standardised cost) was calculated by taking the labour input and costing it on a standardized basis (£400/day for management/science and £300/day other labour). There was no adjustment for inflation from year of treatment. Costs included preparation for treatment on site, but not prior surveys, or project management. The cost of post-treatment monitoring in subsequent years to check the success of treatment is also omitted.

The cost model for biocide treatment was set up in a spreadsheet within the cost model workbook as an aid to project appraisal. Indicative costs for generic types of biocide treatment project are given in Table 7.9 below, based on the costs from actual projects.

With very small sites the additional cost of detailed project planning and subsequent monitoring may represent a significant proportion of the overall cost. Even so, it may be possible to treat small enclosed sites (e.g. < 0.5 ha) at relatively low costs of £1000s. By contrast, large and complex treatment programmes with several waterbodies and hydraulic control may require in the order of £100,000-£500,000.

Table 7.8 Cost of biocide treatments against crayfish, all treatments against *Pacifastacus leniusculus,* unless stated otherwise

Site, **Year of treatment**	**Description, volume, surface area**	**Biocide, application, target dosage**	**Outlay cost total (biocide cost)**	**Standardized cost, excluding monitoring**
Gravel pit Scotland 2004	Isolated pond. 9,250 m^3, 6,000 m^2	Pyblast[1], spray on, 0.15 mg l^{-1}	Total for 3 sites £29,969 (£8,569)	Total for 3 sites £69,755
Castle pond Scotland 2004	Lake with leakage and flow through. 6,000 m^3, 54,500 m^2	Pyblast, spray on, 0.2 mg l^{-1}	See above	See above
Mains ponds Scotland 2005	3 ponds with flow through, vegetation. 6,500 m^3, 6,100 m^2	Pyblast, spray on, and pour on 0.3 mg l^{-1}	See above	See above
Farm reservoir England 2005	Isolated pond. 19,000 m^3, 5,640 m^2	Pyblast, spray on, 0.18 mg l^{-1}	£11,200 (£4,200)	£32,680
Ballintuim Scotland 2006	2 ponds connected by 700m stream. Flow through. 3,300 m^3, 2,700 m^2	Pyblast, spray on, and pour on 1 mg l^{-1} 2 mg l^{-1} in stream	£22,929 (£6,700)	£103,599
Ballachulish Scotland 2012	Isolated pond. 46,000 m^3, 19,500 m^2	Pyblast, spray on, min. 0.3 mg l^{-1}	£73,100 (£33,835)	£78,917

Site, Year of treatment	**Description, volume, surface area**	**Biocide, application, target dosage**	**Outlay cost total (biocide cost)**	**Standardized cost, excluding monitoring**
Dammane Norway 2008	5 connected ponds, largest 3154 m^3	Betamax Vet[3], spray on, 0.02 mg l^{-1}	£86,600 (£10,600)	Insufficient to estimate
Gotland Sweden 2008	3 isolated quarry ponds (details not known)	Decis[2], spray on, 0.0006 mg l^{-1}	£78,139 (£131)	Insufficient to estimate
Lorraine France 1990	3 ponds. 16,636 m^3, 15,495 m^2	Baytex[4] PM40, spray on, 0.06 mg l^{-1} - 0.13 mg l^{-1}	unknown	unknown
Wisconsin[6] USA 1988	1 pond. 9,000 m^3, 4,500 m^2	Baythroid[5], pour on, 0.025 mg l^{-1}	unknown	unknown
Vodnany Czech Republic 2001	1 pond. 1,600 m^2	Chlorinated lime, pour on, 78 g m^{-2}	unknown	unknown
Wisconsin[7] USA	2 ponds.	Sodium hypochlorite, spray on, 50 mg l^{-1}	unknown	unknown

Notes on Table 7.8:

Products used and costs (at 2009 rates), prices are indicative only.

[1]Pyblast, 3% natural pyrethrum, Agropharm Ltd. £50 l^{-1}

[2]Decis, 2.8% deltamethrin, 25 g l^{-1}, Bayer Ltd. £28 l^{-1}

[3]Betamax Vet, 50 g l^{-1} cypermethrin, Novartis Ltd. £1000 l^{-1}

Agrocypa, 10% cypermethrin, Agropharm Ltd. £51 l^{-1}

Note that Betamax Vet is a formulation for vetinary use in treatment of fish lice, formulations of cypermethrin for crop use, e.g. Agrocypa, see above, are less expensive, but are not authorised for use in aquatic environments.

[4]Baytex PM40, fenthion Bayer Crop Science, not available

[5]Baythroid, 12.5 g l^{-1} cyfluthrin, Bayer Crop Science. C. £35 l^{-1}

[6] Target crayfish population *Orconectes rusticus*

[7] Target crayfish population *Procambarus clarkii*,

Table 7.9 Indicative unit costs for biocide treatment using natural pyrethrum by type of waterbody

Type	**Unit cost (est.)**	**Description**
Pond	1.7 £ m^{-3}	Contained pond, no complications.
Pond flow through	£70,000	Extra cost assumed for flow interception and pump back, with natural recovery, no removal of treated water.
Small stream	£25,000 km^{-1}	0-20 l s^{-1}
Large stream	£30,000 km^{-1}	20-100 l s^{-1}

7.2.4 Cost of trapping

Examples of crayfish control projects by trapping or manual removal were used to calculate costs (Table 7.10), based on the number of trapping days or other labour and using standardized labour rates (as for the biocide treatments). The reduction in catch per unit effort (CPUE) is shown between two reference years where available. The variables used in the cost model to calculate total trapping cost per km per year are shown in Table 7.11.

The variables are used in the cost model to calculate total trapping cost per km per year. All of the variables can be altered in the model as required, but those used in the case study here are shown in Table 7.11.

It is assumed that for efficiency a trapping campaign would operate in the period of greatest activity of signal crayfish, which would be from May to September inclusive in the UK. The number of traps per kilometre was calculated from an average spacing of 5 m, based on the range of influence of the traps, estimated as 2.5 m (Acosta and Perry, 2000). Combining the values shown in Table 7.11, the unit costs per trap are £3.50 per trap day and £15,400 km^{-1} $year^{-1}$.

Table 7.10 Crayfish trapping and removal case studies, cost and outcome

Location	**Description** **Length used in comparison**	**Treatment** **T = trapping,** **M = manual removal**	**(Comparison Year1)** **Total catch** **or CPUE**	**(Comparison Year 2)** **Total catch** **or CPUE**	**Estimated** **annual cost (standardized labour cost)**	**Spread** **Prevented**	**Reduction of catch or CPUE between comparison years (est.)**
River Clyde[1] Scotland	5km of main river	T	(2003/4) 9625	(2004/5) 7177	£181,000	No, 0.9 km y^{-1}	25%
Bachawy[2] Wales	Hatchery Ponds	T	(June 2006) 10.9 trap CPUE	(June 2007) 7.7 trap CPUE	£41,600	No	25%
Wixoe[3] England	250 m of River Stour	T	(6 months 1998) 680	(6 months 1999) 1014	£15,100	No	None
Gaddesby Brook[4] England	500 m of stream at Newbold	M	(1998) 3.0 trap CPUE monitoring (Total no. 3553	(2000) 1.5 trap CPUE monitoring (Total no. 3069	£34,750	No	50%

			+1123 in 1999)	summer 2000)			
River Gwash[5] England	1.3km stream	M	(summer 1999) 2412 total no. (345 >30 mm CL)	(summer 2000) 1009 total no. (171 no. >30 mm CL)	£63,500	No	43%
Rio Frio[6] Spain	900m of stream	M	(2005/2006) 17,400, 30.4+3.2 /man-day	(2008/2009) 2086 9.8+1.7/ man-day	£222,850	No data	88% by count, 68% by CPUE

Sources:1: Reeve (2004), 2: Sibley (2000), 3: Wright and Williams (2000), Wright (2009), 4: Sibley (2001), 5: Judson (2003), 6: Dana et al. (2010)

Table 7.11 Trapping cost model, variables and values assigned

Variables used in trapping model	**Values assigned in model**
Traps deployed per km[1]	200
Frequency of setting/lifting of traps (as a proportion of days in the period)[2]	0.2
Total number of trapping days per year	110
Number of traps worked per man day	100
Trap depreciation and bait cost, cost per trap night, £	0.50
Labour cost £ per man day	300

1. A frequency of 0.2 is equivalent to trapping for one night per (5-day working) week, or emptying static traps weekly
2. Trapping would be from both banks and selective placing of traps would exclude fast-flowing sections or other areas with limited refuges
3. The number of traps worked per day (100 traps man-day^{-1}) is practicable in readily accessible areas, but is an ambitious target for headwater streams, which are often in rough terrain, with poor access.

7.2.5 Case study of a cost model for the North Esk catchment

The North Esk catchment was used as a case study to test the model. It supports both salmon and brown trout, with approximately 88% of the waters in the catchment assessed as providing habitat for salmon (MacLean et al., 2006). Catchment parameters are shown in Table 7.12. Using the method for calculating stocking cost for salmon, the total restocking cost for the catchment would be £938,160, or £1,285 km^{-1} y^{-1}.

In order to compare different management options for crayfish, the North Esk catchment was used as a case study for the cost model, using the scenarios shown in Table 7.13. The ponds included in the biocide scenarios are the ones that were actually treated in the North Esk catchment (first three cases in Table 7.8), with some modifications to improve the treatment in two of the sites (Mains pond and Castle pond).

Table 7.12 Characteristics of North Esk catchment

Catchment area:	732 km^2	(Centre for Ecology and Hydrology, 2013)
main river length	69 km	(Esk Rivers and Fishery Trust, 2008)
total length of watercourse	730 km	estimated for cost model
total wetted area in catchment	4.76 km^2 (0.62% catchment)	(MacLean et al., 2006)
Rating of abundance of juvenile salmon and trout, fisheries surveys	"reasonable", "good" or "abundant"	(Esk Rivers and Fishery Trust, 2008)
Number of returning adult salmon per year (1981-1997 at Logie Mill fish counter)	20,846 average (range 13,006 to 27,688)	(MacLean, 2007)

Table 7.13 Cost model: management scenarios in the North Esk catchment

	Management of crayfish	**Year of treatment**	**Rate of invasion upstream, downstream**	**Extent of treatment**
1	Biocide treatment	1	0	Three ponds and short lengths (<200m ditches at two of them)
2	Biocide treatment	5	0.2,0.5 until year 5	As scenario 1 plus 7.75 km small stream, minor inflows and 3.6 km Luther Water
3	Biocide treatment	5	0.5, 1.5 until year 5	As scenario 1 plus 15.8 km small streams and 8.8 km Luther Water
4	Trapping	1 ongoing	0.2, 0.5	All invaded watercourses annually
5	Trapping	1 ongoing	0.5, 1.5	All invaded watercourses

	Management of crayfish	Year of treatment	Rate of invasion upstream, downstream	Extent of treatment
				annually
6	Trapping	1 ongoing	0.1, 0.25	All invaded watercourses annually
7	None (restocking)	1 ongoing	0.5, 1.5	All invaded watercourses annually

In summary, the estimated costs for the biocide treatments were:

- Scenario 1: £135,000
- Scenario 2: £433,000
- Scenario 3: £784,000

A schematic diagram of the three biocide scenarios is shown in Figure 7.1. The gravel pit (wholly enclosed) lies to the west of the sites shown in the Figure and is not shown.

Using the method described above, the maximum annual restocking cost for the whole catchment was calculated at £938,160 per year, assuming a loss factor due to signal crayfish of 0.1. Apportioned across 730 km of watercourses, this represented £1285 km^{-1} for salmon. Potential restocking for trout was not included in the case study, as the proportion of catchment suitable for trout recruitment was not estimated, but if included it would add £300 km^{-1} of invaded catchment.

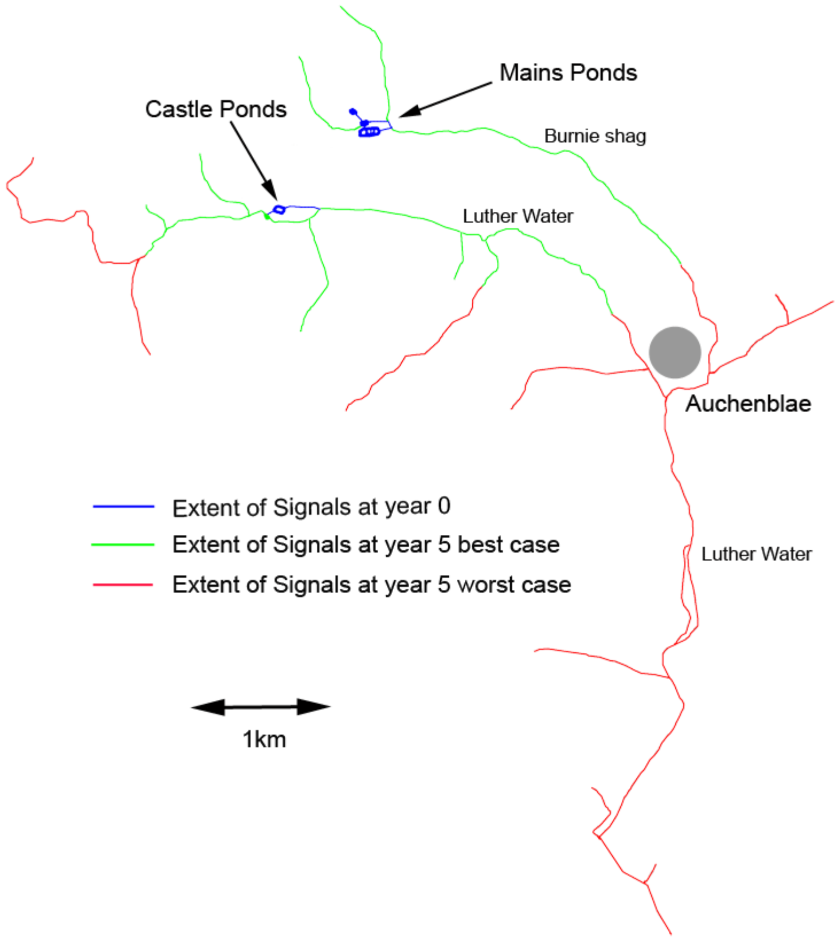


Figure 7.1 Schematic diagram of part of North Esk catchment showing three scenarios for biocide treatment

7.3 Results of a cost model for the assessment of eradication treatment or control programme

7.3.1 Rate of invasion of a catchment and the cost of impact

When calibrating the model the effect of changing the rate of invasion was assessed on the time to complete colonization of a modelled catchment. The Clyde catchment, with 4200 km of watercourses, was used because studies have shown the actual rate of invasion in this catchment (Table 7.1). Rates have differed over time and between the main river and tributaries, in the range 0.26 – 1.5 km year^{-1} downstream and 0.05 – 0.25 km year^{-1} upstream.

Figure 7.2 shows the total length of invaded watercourse against years for three rates of invasion: a) fast rate (2.5 km year^{-1} downstream and 1 km year^{-1} upstream), b) 'typical' rate (1.5 km year^{-1} downstream and 0.5 km year^{-1} upstream) and c) slow rate (0.5 km year^{-1} downstream and 0.2 km year^{-1} upstream), rates which are within the range recorded (Table 7.1).

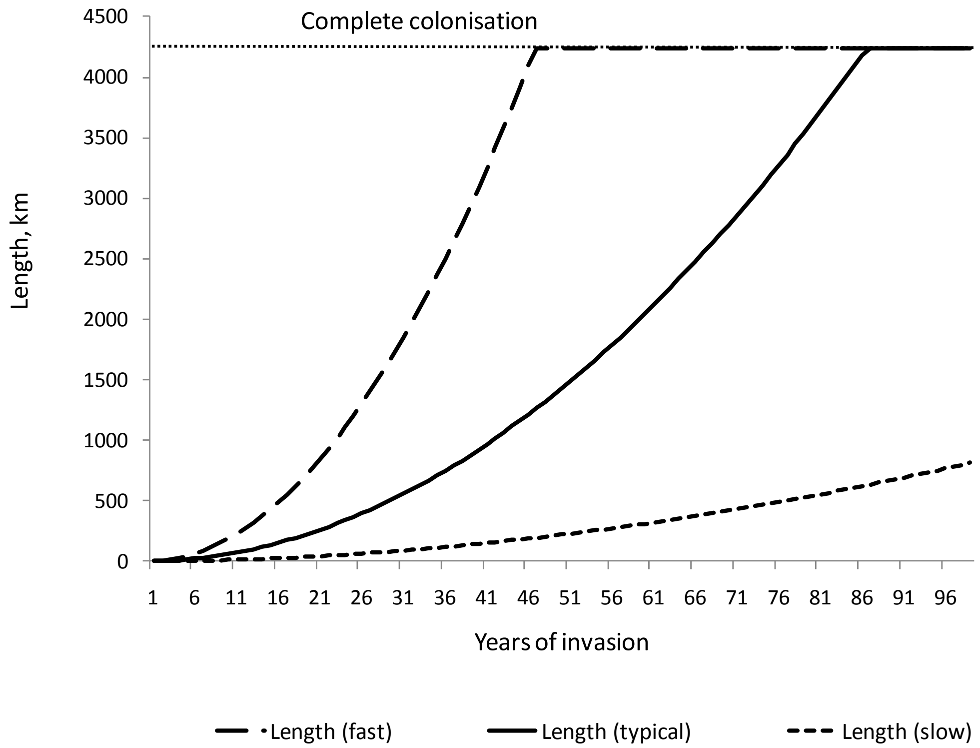


Figure 7.2 Invasion in signal crayfish in a model catchment and time to complete colonisation at three different rates (upstream and downstream rates in km year^{-1}), a) fast (2.5, 1.0) b) typical (1.5, 0.5) c) slow (0.5, 0.2)

At the ‘typical’ rate, it would take 86 years for signal crayfish to fully colonize the catchment, compared to the fast rate of 46 years and slow rate of 230 years.

The model shows that restocking cost would increase progressively as a catchment became increasingly invaded. Figure 7.3 shows the cumulative costs of restocking, based on a restocking cost for salmon of £450 km^{-1}, or £300 km^{-1} for trout and Figure 7.4 shows the increasing annual cost. The cumulative-stocking cost for salmon in year 10 would be £146,000, rising to £72.5 million by year 93. The costs for trout would be £97,000 and £48 million in the corresponding years. By year 93 the whole catchment would be colonized, but the cumulative cost would continue to increase linearly at £2.25million per year, having reached the maximum annual restocking cost for salmon. If both salmon and trout were stocked at the modelled rates in this fully invaded catchment the maximum annual restocking cost would be £3.74 million per year. All future restocking costs are shown at present day prices without any adjustment for inflation or discounting to net present value.

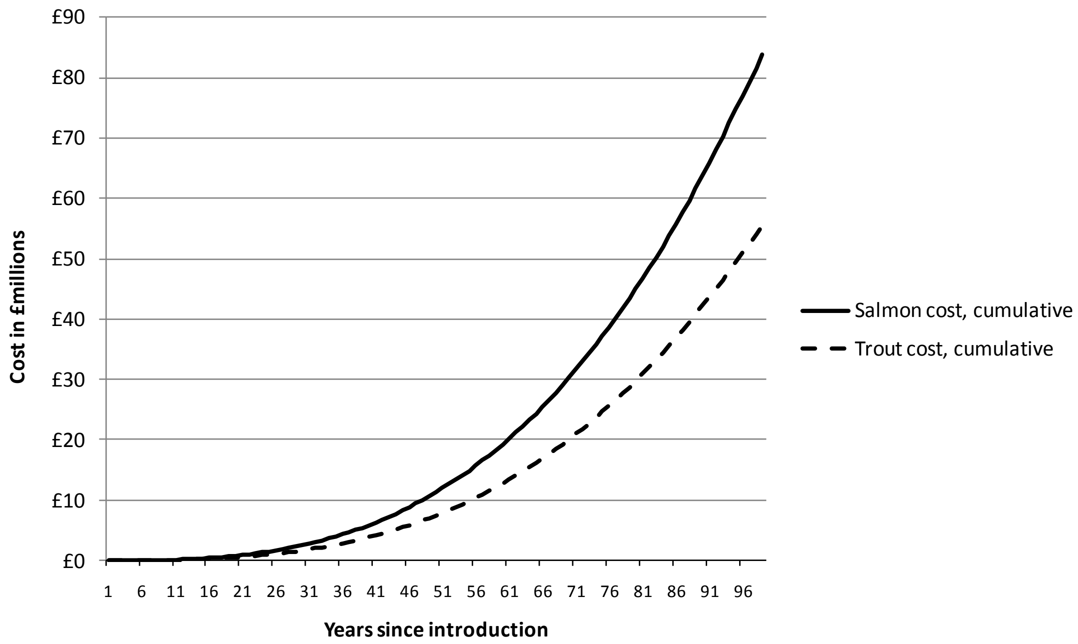


Figure 7.3 Modelled cumulative re-stocking costs for salmon and trout in an invaded catchment

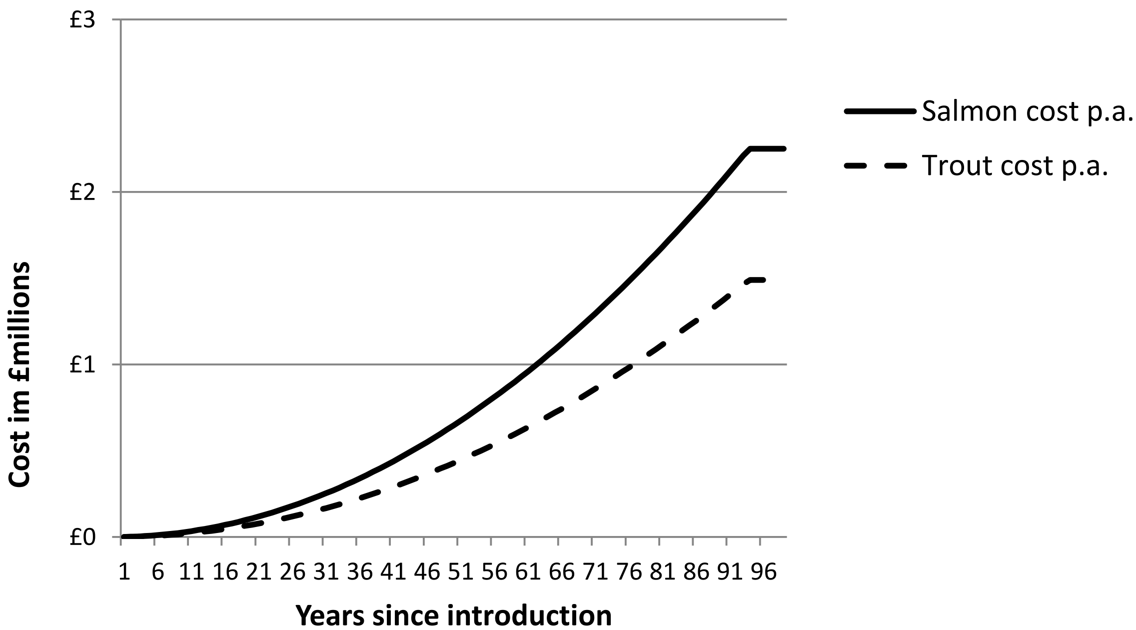


Figure 7.4 Modelled annual re-stocking costs for salmon and trout in an invaded catchment

The same modelled example was used to show the number of years before the cost of a biocide treatment would be exceeded by the restocking cost, effectively the time to break even for the investment in biocide treatment carried out at the start.

Figure 7.5 shows the cumulative restocking cost, for salmon alone, for the first 16 years of the invasion. The costs of two examples of biocide treatment are also shown, using the indicative costs from Table 7.9; treatment of a small enclosed waterbody at £20,000 and a more complex treatment at £200,000, for example a similar-sized pond with a throughflow, plus a short length of watercourse. The cost of biocide treatment would be exceeded by the cumulative restocking cost in year 5 for Scenario 1 and in year 12 Scenario 2. The model shows that even the annual restocking cost for salmon would exceed the total cost of the biocide treatment by year 8 for Scenario 1 and by year 27 for Scenario 2 and with both salmon and trout restocking included, the break-even points would be years 4 and 11 for the cumulative restocking costs of Scenarios 1 and 2 respectively.

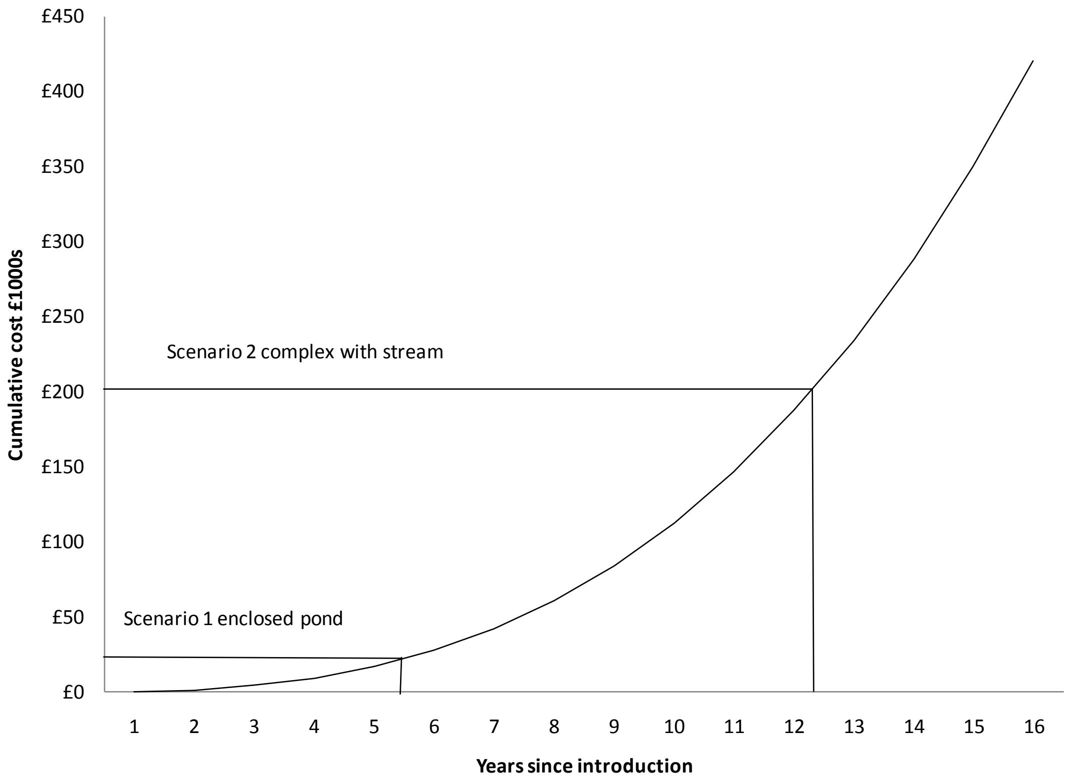


Figure 7.5 Cumulative re-stocking cost or salmon over the first 16 years of invasion compared to the total cost of treatment with biocide for a) a simple treatment of an enclosed waterbody and b) a complex including a length of stream

The simple cost model does not account for four potential sources of delay or lag time. Firstly, the rate of invasion is likely to be slower in the establishment phase, for the first few kilometres invaded and this phase seems to be in the order of 5 to 20 years, based on examples such as the River Clyde in Scotland and River Wharfe in England (Table 7.1). Secondly, it would probably take several years before any impacts were noticed in field conditions, due to the time required for crayfish to reach high abundance, plus a further lag time until any impacts were detected in fisheries surveys. For example, with salmon (MacLean, 2007) showed that for the North Esk individuals spent 1 to 4 years in freshwater and 1-3 years in the sea, which mean that salmon spawning in year *i* would produce recruits returning in years *i+3* to *i*+8, diffusing the effects of impacts in year *i,* although impacts would continue to year *i*+8 and beyond. Thirdly, the location at which an introduction occurs in a catchment could also create a lag, because of the time taken for an invading population starting on a tributary to reach the main river and access other tributaries. If modelling an “actual” invasion in a catchment, the number of years of lag time would be added to the number of years before restocking cost exceeded the cost of a biocide treatment. Fourthly, there would be the time between detection of a population of invasive crayfish and biocide treatment.

In the example (Figure 7.5), each year of delay would bring the population closer to its potential to expand rapidly in the main river, with its associated non-linear increases in annual costs as soon as another tributary is invaded. This means that even though the treatment would more expensive, the cost would be exceeded by the cumulative restocking cost in a similar period. The model in Figure 7.5 shows that in the early stages of invasion would take 12 years of cumulative restocking cost to match an expenditure of £200,000 on a biocide treatment, but the cumulative restocking cost would reach £400,000 in the following three years. This means that even a delayed treatment may be justified, provided it still has a high likelihood of achieving eradication.

Varying the loss factor proportionately decreases the annual increase in impact of signal crayfish expressed as a re-stocking cost (Figure 7.6). Changing the loss from e.g. 0.1 for headwater streams to 0.9, to represent only a minor loss as may be the case in large rivers, would increase the time before the outlay cost of biocide treatment was exceeded by the cumulative or annual cost of re-stocking. For example, in the example above, instead of

12 years of cumulative re-stocking to match £200,000 on biocide treatment, it would take 33 years.

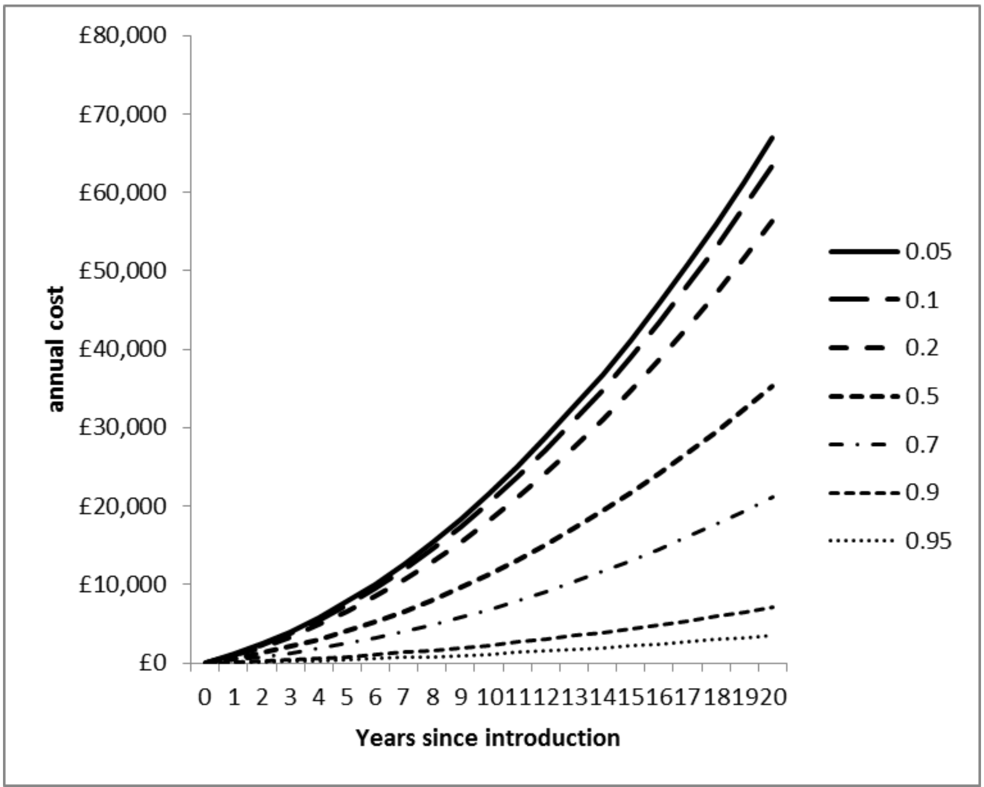


Figure 7.6 Annual cost of a restocking with salmon in an invaded catchment ('typical' invasion rate, 1.5 km y^{-1} downstream, 0.5 km y^{-1} upstream) with different loss factors applied, ranging from 0.05 (only 5% salmonid recruitment, 95% replacement required) to 0.95. (95% salmonid recruitment, 5% replacement required)

7.3.2 Modelling the costs of trapping

As with the restocking cost, the cost of trapping would increase annually with the invasion and then continue indefinitely at the maximum cost once the catchment was fully invaded. The invasion cost model was tested for the effect of trapping on annual cost of restocking when trapping halved the invasion rate or when it reduced impact but did not reduce the rate of invasion (see Figure 7.7 and Table 7.14 for annual costs at year 10). Because of the high unit cost of trapping compared to annual restocking cost, the reduction in signal loss factor had little effect on cost. The cost of trapping was proportionally less if the rate of invasion was reduced, but the annual cost was still much greater than the restocking cost alone. If trapping achieved the highly optimistic target of removal of impact (loss factor 1.0), the total annual cost would be £227,578 in that year. As trapping would be unable to prevent invasion, even if, in the best possible case it reduced erosion, the model showed the annual cost would increase until the invasion reached the limits of the catchment, in this example to £76.5 million per year in the fully invaded catchment (compared to £2.5 million for restocking).

Table 7.14 Comparison of annual costs of restocking and trapping 10 years after start of invasion at different rates of invasion

Annual cost of management at year 10	**Restock only**	**Trapping and restock**		
Loss factor	0.1	0.1	0.2	1.0
Invasion rate typical	34,205	1,197,761	1,193,960	1,163,556
Invasion rate slow	6,690	234,268	233,525	227,578
Invasion rate typical for restock, reduced by half for trapping	34,205	378,704	377,502	367,889
Invasion rate slow for restock, reduced by half for trapping	6,690	86,309	86,035	83,844

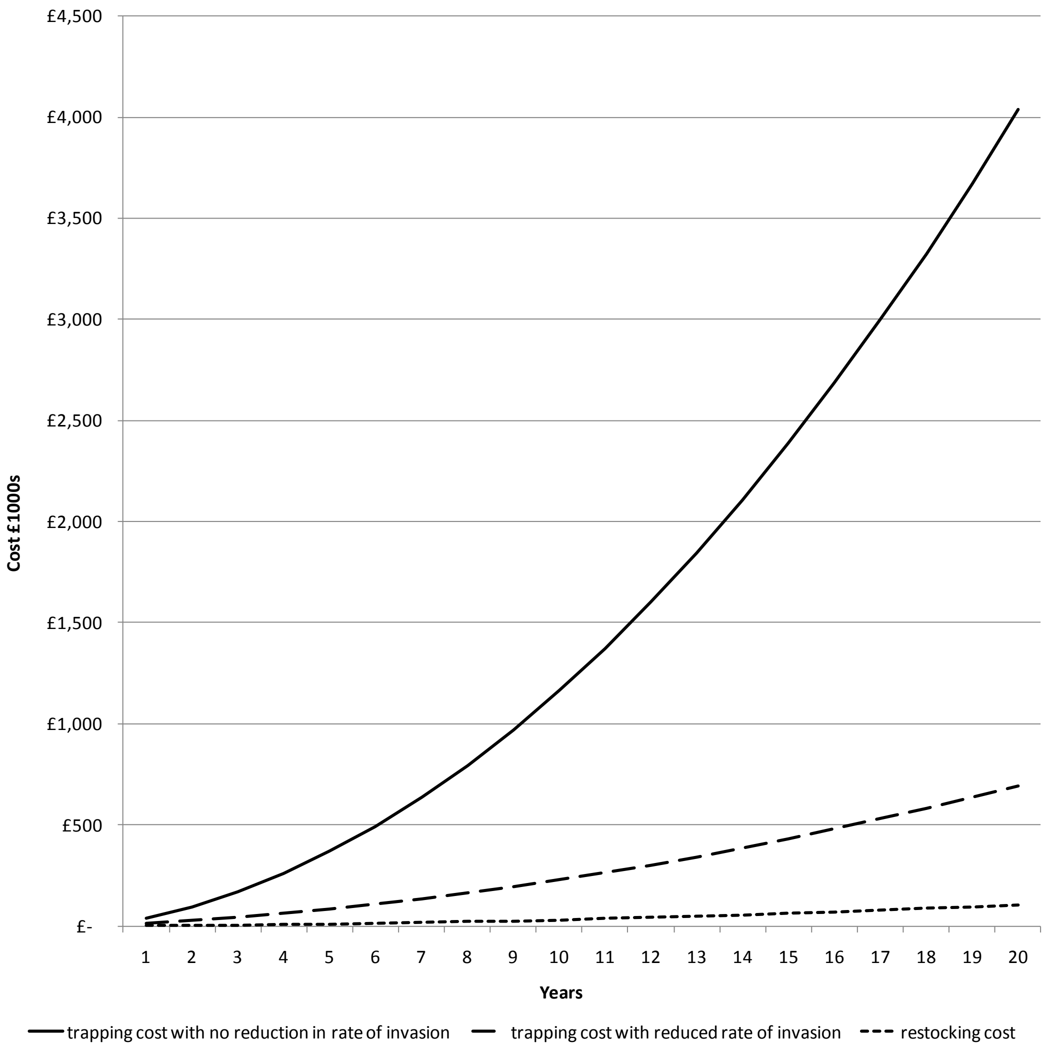


Figure 7.7 Annual cost of a trapping programme with a) cost of restocking only b) trapping without reduction of impact (0.1) c) trapping with reduction impact by 50% (0.2) d) trapping with reduction of impact by 100% (1.0), e) trapping with invasion rate reduced by half, reduction of impact 100%

7.3.3 Results of the case study of a cost model for the North Esk catchment

The cost model was used to compare the cost of biocide treatment of sites in the North Esk with those of a trapping programme to reduce the abundance of crayfish in the invaded reaches. If successful, after the biocide treatment there would be no further invasion and hence no restocking cost, whereas with no biocide treatment, the cost of impact of signal crayfish (expressed as restocking cost) would increase annually until the catchment was fully invaded and continue at a fixed annual rate thereafter. The time taken until restocking cost exceeded the cost of biocide treatment was calculated for three scenarios as follows: treatment in year 0 (scenario 1), treatment in year 5 with a slow rate of invasion (scenario 2) and treatment in year 5 with a typical rate of invasion (scenario 3) (Table 7.15).

With typical invasion rates for crayfish shows that the cumulative restocking cost, i.e. the environmental impact foregone, exceeded the cost of biocide treatment under Scenario 1 in year 7. Indeed by year 13 even the annual restocking cost exceeded the total cost of Scenario 1. With lower rates of invasion assumed, the cumulative restocking costs accumulated more slowly, but exceeded Scenarios 1 in year 12.

The annual cost of impact would continue to increase until the catchment was fully colonized, which for the North Esk was estimated to be at £938,070 year^{-1} after 35 years, or 93 years if the slow rate of invasion is assumed throughout the invasion, rather than just in the initial period. By the time the catchment was fully invaded, the cumulative restocking cost (impact cost) would £12 million for the typical rate of invasion (£30 million for the slow rate), with no discounting or adjustment for inflation rates.

Table 7.15 Cost model for North Esk catchment showing the number of years until management for mitigation of impact of crayfish (restocking) or control by trapping exceeds the cost of biocide treatment

		Year in which restocking exceeds cost of treatment			
Scenario	Biocide treatment cost £	Cumulative cost, typical rate invasion	Cumulative cost, slow rate invasion	Annual cost, typical rate invasion	Annual cost, slow rate invasion
1: biocide treatment all sites in year 0	135,000	7	12	13	33
2: biocide treatment in year 5, slow rate (0.2, 0.5 km year^{-1})	433,000	11	20	23	62
3: biocide	784,000	13	25	32	85

treatment in year 5 typical rate (0.5,1.5 km year^{-1})					
		Year in which trapping cost exceeds cost of biocide treatment			
		Cumulative cost of trapping exceeds scenario 1	Annual cost of trapping exceeds scenario 1	Cumulative cost of trapping exceeds scenario 2	Annual cost of trapping exceeds scenario 2
4: trapping at slow rate (0.2, 0.5 km year^{-1})		3	7	7	15
5: trapping at typical rate (0.5,1.5 km year^{-1})		2	3	4	6
6: trapping at half of slow rate(0.1,0.25 km year^{-1})		6	13	11	30

The trapping model for the North Esk (200 traps/km, on a weekly rotation, , throughout May to September inclusive), would cost £15,400 km^{-1} of invaded catchment trapped, significantly more than the £1285 km^{-1} of invaded catchment re-stocked. If a trapping programme started at year 1, i.e. as an alternative to a biocide treatment, the cumulative cost of trapping would exceed the cost of scenario 1 biocide treatment in 2 to 6 years, depending on the rate of invasion, and the annual cost would exceed it in 4-11 years. The annual trapping cost would continue to increase each year thereafter, to a potential maximum of £11,242,000 per year in a fully invaded catchment, from year 35 onwards if invasion proceeded at a typical rate of invasion, or year 99 at a slow rate of invasion.

7.4 Discussion

This study demonstrated how an environmental cost of a biological invasion could be estimated over time at the scale of an individual catchment. It showed how comparisons could be made between the costs of short term eradication and a long term control programme. Issues to consider here include the implications of the increasing costs for management of invasive crayfish, the importance of avoiding delay and the limitations and scope of the cost model.

7.4.1 The benefits of early eradication if feasible

The cost model developed in this study used information gained from actual projects to eradicate or control crayfish to provide a simple tool to help resource managers understand the costs and benefits of management of invasive crayfish in future cases. It indicates that if it is technically feasible, an eradication treatment is the lowest cost option, by avoiding the increasing cumulative environmental costs of a progressively invaded catchment. The model shows how, in the absence of eradication or control, the cost of environmental impact accelerates once the invasion starts to progress up tributaries, so avoiding the cost of impact represents a benefit that soon balances the cost of an early eradication.

Delay in carrying out an eradication treatment against a population of invasive crayfish leads to increases in the cost and complexity of treatment when the invasion extends along the watercourses, with increasing risk that eradication cannot be achieved. Lag time between introduction and increased expansion and impact is a common pattern in invasion biology (Crooks, 2005), which has the benefit of providing a limited window of opportunity for action to eradicate or control the invasion, even in a small watercourse, but conversely, the lag also delays the development of the cost of environmental impact.

7.4.2 The cost and effects of trapping

Trapping or other physical removal of crayfish is an expensive option, even when the most optimistic assumptions are made about efficacy. There is a lack of evidence to indicate that the rate of invasion can be reduced by reducing the abundance of the population in a watercourse, which makes modelling of control by trapping somewhat speculative. Moorhouse and MacDonald (2011) found that immigration/emigration rates of signal crayfish remained the same through a zone where trapping was used.

There are circumstances in which trapping may be effective in limiting the impact of an invasive crayfish, although probably only in conjunction with other management. Trapping reduced a population of invasive rusty crayfish *Orconectes rusticus* in a lake in Wisconsin, when combined with reduced harvesting of large predatory fish and a reduction of favourable habitat due to reduced water level. In that case, the crayfish remained at low abundance after the campaign and the reduction in the number of fish taken by anglers. Aquatic macrophytes, some benthic invertebrates and small fish species recovered (Hein et al., 2007, Hansen et al., 2013). Trapping for eight years reduced a population of signal crayfish in a length of English river, to a level at which nuisance to anglers was alleviated and erosion of banks by burrowing was reported as having been reduced (West, 2010).

By contrast, the control campaigns against signal crayfish shown here (Table 7.8) were able to reduce the population to a relatively stable annual yield obtained after about 1-3 years. None of them prevented continued expansion of range. There is uncertainty about the effects of trapping on the population of signal crayfish. Insufficient removal of crayfish means that the reduction can be compensated by recruitment and increased growth with reduced competition (Skurdal and Ovenild, 1986), as happened in a stream in Andalucia (Dana et al., 2010) and in the River Clyde, i.e. a sustainable harvest. By contrast, removal of a high proportion of large crayfish may lead to large increases in survival of juvenile crayfish and potentially even a net increase in total biomass, as has been found in intensively harvested ponds with noble crayfish (Keller, 1999a, Keller, 1999b) and with Australian yabby *Cherax albidus* (Lawrence et al., 2006). An 'ideal' level of trapping for control between these two scenarios may be hard to find – and variable in different habitats and from year to year.

Nonetheless, from the model, even if trapping can be shown to mitigate impact on aquatic communities, and at much lower cost and effort than appears likely to be required, the need to expand the management to the whole catchment in perpetuity means that it is unlikely to be sustainable.

7.4.3 Limitations of the cost model

One of the limitations of the current cost model is that it assumes an introduction into the main river and can only predict fastest and slowest progress (upstream plus downstream expansion rates (default), or all upstream) and in reality, as the upstream invasion rate is generally slower than downstream rate, the time until invasion of the catchment is complete will be strongly affected by the point of introduction. The cost model does not simulate invasion of a specific catchment in geographic detail, however, if a hydraulic model of a river catchment already exists, for example for water quality modelling, a cost model of the type shown here could be attached to provide a more realistic forecast of an invasion.

Another limitation of the model is that the cost of the environmental impact has been calculated conservatively by using a re-stocking cost for recreational salmonid fisheries and this is likely to be an under-estimate of the ‘true’ environmental cost. Re-stocking cost is often used in the UK when legal cases are brought against polluters following pollution incidents in rivers, but it is not necessarily an appropriate management response to a reduction of salmonid productivity in the presence of invasive crayfish. The strong homing instincts in anadromous salmon (Stabell, 1984) and the isolation of populations of species such as Arctic charr *Salvelinus alpinus* (Adams et al., 2006) mean that local races cannot readily be replaced if reduced or lost and this biodiversity value is not taken into account in the cost model. Furthermore, salmonid fish are the only species included in the model at present and this disregards other species with importance for biodiversity in habitats invaded by crayfish. A surrogate cost could be included for some indigenous species, based on the management cost of running captive breeding and release programmes, but even so would not provide an adequate valuation of the total impact on biological assets.

Re-stocking is carried out when fisheries experience significant environmental impacts, due to the economic and social importance of recreational fisheries, but it does not include any direct measure of social importance. The perceived value of a recreational fishery to anglers may be affected by crayfish. For example a fishery may considered less desirable because local races of fish have been replaced by re-stocked equivalents, or abundant crayfish frequently cause nuisance to (coarse) anglers by taking angling bait and this may reduce the willingness of anglers to pay to use a fishery (Peay et al., 2010). Contingent valuation based on the willingness of anglers to pay for fishing in waters unaffected by an invasive species has been used to justify the cost of spending on measures to prevent introductions of rusty crayfish into lakes in Wisconsin (Keller et al., 2008).

The impact of burrowing activity on rates of erosion and siltation in waterbodies is another potential element that could be included in cost-benefit analysis, due to the greater cost of maintenance or restoration of river banks, which Peay et al. (2010) estimated could range from about £10 m^{-1} for small scale restoration of banks to £100-£250 m^{-1} for major re-profiling and strengthening of the banks of rivers and canals. These additional factors could be included in a cost benefit analysis of the impact of aquatic invasion by crayfish and would provide additional support for investment in prevention or eradication. A cost-benefit analysis of management action on invasive species in then Doñana wetlands in Spain Garcia-Llorente et al. (2011) showed there was popular support for treatments to eradicate damaging invasive species, although stakeholders were less willing to pay for prevention than for eradication treatment.

In a cost model for invasion by a highly mobile species, the gypsy moth *Lymantria dispar*, Sharov and Liebhold (1998) showed the increasing cost of control along an expanding front of invasion and showed how the optimal strategy changed over the time of invasion from highly cost-effective effort on early eradication, through control measures to limit or slow the spread and eventually to no action due to the proportionately high cost of control.

7.4.4 Justifying the investment in prevention

For signal crayfish, which in most cases cannot cross from one catchment to another except with human assistance, the current position in Great Britain is that in many catchments the invading population is either too extensive, or is in a waterbody that is too large to treat. Nonetheless, prevention and in some cases eradication are still feasible at the scale of individual catchments. The cost model in this study provides a potential tool for agencies to justify resourcing action on prevention and eradication, especially in Scotland, which still has many catchments that are still free from invasive crayfish.

The government agency that advises on biodiversity in Scotland, Scottish Natural Heritage and the Rivers and Fisheries Trust Scotland, which represents a range of angling interests recognises the value of investment in increasing awareness of the risks of invasive crayfish. They have started providing training for local fisheries staff about invasive crayfish, which is being communicated to other local stakeholders both to discourage further illegal introductions and encourage reporting of new sites.

Ricciardi et al. (2011) advocated preparing for biological invasions as if they were natural disasters, with rapid response strategies and adequate resourcing ready at what may be relatively short notice. Homans and Smith (2013) drew parallels between the costs of biological invasion in aquatic systems and the prevention and clean-up of pollution. Yet in most cases of chemical pollution, after the initial event and dissipation, environmental processes ameliorate the effects over time, whereas with biological invasions the impact increases spatially with time. Helping resource managers and others understand the long-term potential costs of biological invasion may encourage contingency planning for rapid response, while there is still time to deal effectively with a localized population of an invading aquatic species. Invasion of watercourses by non-indigenous crayfish has been used in this case, but a similar approach could be applied to the invasion of catchments by other aquatic invaders. Even a relatively simple cost model can be used as part of the process of risk assessment.

7.5 Acknowledgement

Thanks go to Scottish Natural Heritage for funding this project, to Colin Bean for advice and to Paul Bryden for assistance in collating the cost data.

Part 3
Conservation management for indigenous white-clawed crayfish

White-clawed crayfish

Chapter 8
Selection criteria for “ark sites” for white-clawed crayfish – a management tool

8.1 Introduction

The white-clawed crayfish *Austropotamobius pallipes* (Lereboullet) is our only indigenous freshwater crayfish and it is under threat in England and Wales (Holdich et al., 2009a), due to the introduction of non-indigenous species of crayfish, especially the American signal crayfish *Pacifastacus leniusculus* (Dana) (Holdich and Sibley, 2009). White-clawed crayfish are out-competed by signal crayfish and they are also highly vulnerable to crayfish plague *Aphanomyces astaci* Schikora, which is often carried in populations of signal crayfish and is completely lethal to all white-clawed crayfish. The progressive decline in populations of white-clawed crayfish has been reported by Sibley et al. (2002) and continues throughout England and Wales, with yet more white-clawed crayfish populations fragmented or lost due to outbreaks of crayfish plague or by competition from invading populations of non-indigenous crayfish. In many catchments where white-clawed crayfish were formerly abundant, there are only a few remnant and threatened populations left. For example, only three catchments in Essex have any indigenous crayfish left (Pugh et al., 2008) and the last known population of white-clawed crayfish in Bedfordshire was lost in 2006 (Peay et al., 2006). Holdich and Sibley (2009) showed the inexorable spread of signal crayfish in South-west England since 1975, due to new introductions and progressive expansion of the established populations.

Once non-indigenous crayfish are established and expanding in a watercourse, there is limited scope, at best, to eradicate or control them. Whenever signal crayfish establish in any part of a catchment white-clawed crayfish will eventually be replaced by them in the long term, unless there are barriers preventing invasion of the whole catchment. The entire population of white-clawed crayfish in a catchment may be lost much sooner if it is infected with crayfish plague, which can eliminate white-clawed crayfish within weeks (Holdich et al., 2009a). This means that even where there are locally abundant populations of white-clawed crayfish in watercourses at present they are all potentially vulnerable.

Sibley (2003) forecast that most rivers in England and Wales might lose their populations in the next 30 years. Local or regional extinction from river systems seems increasingly likely in even shorter time in some areas, including several rivers in South-west England (Holdich and Sibley, 2009). White-clawed crayfish can only survive in isolation from signal crayfish and other invasive non-indigenous crayfish species.

In developing a conservation strategy for white-clawed crayfish it is important to prevent any more introductions of non-indigenous crayfish and provide isolation for white-clawed crayfish. Holdich et al. (2004) recommended a strategy of introductions of white-clawed crayfish to new, isolated sites that would provide a basis for conservation of the indigenous species, which they described as “ark sites”.

A protocol was developed for re-introduction of white-clawed crayfish to rivers from which they had been lost historically (Kemp et al., 2003). At that time attention was being given to the potential to restock rivers where white-clawed crayfish had been lost, for example due to pollution or outbreaks of crayfish plague due to contaminated fish or angling nets as part of the Life in UK Rivers initiative. However, Holdich et al. (2004) highlighted the threat of invading signal crayfish in rivers and proposed introductions to isolated sites that had not had white-clawed crayfish previously, especially relatively recently created still water sites. The potential of new minerals sites for this is being promoted by Buglife (Whitehouse et al. 2009, Kindemba and Whitehouse, 2009). Whilst the introduction protocol (Kemp et al., 2003) provided useful principles, more detail is needed about how to select potential ark sites in practice.

The aim of this work has been to provide a simple, but flexible, tool for land managers and ecologists and other practitioners, to help them assess potential ark sites. It has been designed for use in England and Wales, although a similar approach may be of value in other parts of the range of white-clawed crayfish in Europe.

A simple set of selection criteria has been developed to help those seeking potential ark sites for white-clawed crayfish. By avoiding unsuitable sites, efforts and resources can be concentrated on those most likely to succeed. This will benefit the conservation of white-clawed crayfish locally, regionally and nationally. Potential ark sites will be found at local level as individual isolated sites, but they need to be considered in the context of individual catchments and within the region or River Basin District as a whole (as now used by the Environment Agency under the Water Framework Directive).

The search for ark sites is likely to proceed from region and catchment scales, based on information on the distribution of crayfish species. Where are the existing populations of white-clawed crayfish? What are the threats to those existing populations? If non-indigenous crayfish are invading a catchment how far and how fast can they spread? Are there any barriers to invasion in the medium to long term? What opportunities are available for potential ark sites, or can be actively sought? The regional approach is ideally represented by the South West Crayfish Conservation Strategy (Nightingale et al., 2009, Nightingale, 2009).

At the same time as strategies are being developed for whole regions and catchments, individual sites may be suggested by landowners, or developers such as mineral operators who are aiming to provide benefits for nature conservation. Recently worked quarries and other mineral sites may offer excellent opportunities for new ark sites, as promoted by Buglife (Whitehouse et al., 2009, Kindemba and Whitehouse, 2009).

But which sites have the best chance of succeeding in the long term? The approach is a risk-based one, as it is the risks that are most likely to determine the success of an ark site in the medium to long term. The key factors are:

- Is the site at risk from colonisation by non-indigenous crayfish?
- Is there a significant risk of crayfish plague?
- Are there any other adverse factors?

Guidance on selecting ark sites will be useful at different scales and stages. The intention is that users will utilise them for different purposes according to regional or local needs. Examples of potential uses are as follows:

- In initial desk studies at the scale of region, River Basin District, catchment or administrative district.
- To help select and prioritise potential sites identified from a desk study.
- As an aid to recording relevant features during an appraisal on site, for later evaluation of a potential ark site.
- To help assess the risks for a potential ark site and its likelihood of success.

- To help assess the risks for an existing population of white-clawed crayfish, which may be in a site considered to be an existing ark site, or may be under threat – can those threats be reduced?
- To record the basis for deciding whether a site is considered to have potential to become an ark site. This information would be used in a more detailed feasibility study and could be included as part of the information supplied to support an application to statutory agencies to introduce white-clawed crayfish into a potential ark site.
- To encourage recording of relevant features of sites as an aid to future reviews of success of ark sites, for an evidence-based approach to improving best practice guidance on ark sites in future.

The selection criteria are the same for different uses but the information about individual sites that is used in the process and the decision-making itself may differ.

8.2 Method: description of the selection criteria

Selection criteria for ark sites have been prepared as a spreadsheet tool, which guides a user through the criteria, compiling information about a site and its suitability. It then gives guidance on how to use the information, to decide whether to consider the site as a potential ark site. If the site is considered to be a potential ark site, the user will proceed to detailed assessment of the site and, if feasible, to preparation for the introduction of white-clawed crayfish to the site. This detailed stage is not covered by the spreadsheet at present, but there is some existing guidance in Kemp et al. (2003).

The selection criteria are set up in two stages, followed by decision-making. A user wanting to use the selection criteria starts by obtaining information about a site. If done at a regional scale, information may be from maps, aerial photographs, local plans, information held by the Environment Agency and other sources on crayfish distribution, water quality etc. In other cases there will be more detailed information on a site and its environs from a recent field survey or local knowledge.

The first stage of the selection criteria is a coarse filter of five questions that allows any obviously unsuitable sites to be excluded, on grounds of: 1. the known presence of non-indigenous crayfish, 2. lack of permanent water, 3. insufficient physical isolation to avoid colonization by non-indigenous crayfish, or 4. poor water quality.

In addition, one question excludes sites that already have white-clawed crayfish, as they are not classed as potential ark sites. There is a presumption against the introduction of white-clawed crayfish to any isolated site that already has a population present, mainly on grounds of biosecurity. The risks of multiple stockings were shown in Finland in re-stocking projects with noble crayfish *Astacus astacus (*Linnaeus) (Jussila et al., 2008). A site with an existing population of white-clawed crayfish may be assessed as an established ark site, or it may be an existing population at risk. A user can utilise the criteria to help assess or re-assess the threats to the population.

The second stage of the selection criteria is a series of nine tables, each with a different topic, as listed in Table 8.1 below. In each table there is a series of descriptions, each of which is listed against a qualitative rating: Best, Good, Possible, Poor. The user selects the description that best matches the site being assessed and ticks the corresponding box, which assigns a rating and copies it to a summary table. There is space for a user to add descriptive text to explain the basis for the choice or any limitations. Each table has explanatory notes and references to guide the user. The criteria considered to be most important are listed early in the series of tables, so users can opt to screen out unfavourable sites early, or continue to a full assessment. The first three tables assess the likely effectiveness of barriers to colonization by non-indigenous crayfish. The fourth table deals with the availability of water year round and its quality. The next four tables are mainly related to human activity and the likelihood that this will lead to introduction of crayfish plague or the release of non-indigenous crayfish into the site. Broadly, sites with high levels of angling and other general public use are considered to have greater risks than sites with little public access, or where management has conservation objectives. The last table deals with physical habitat in six sections. Although these are rated, they are considered to be only minor elements in the decision-making process, because it is relatively simple to create or improve physical habitat for white-clawed crayfish (Peay, 2003) and other crayfish species with similar habitat preferences (Johnsen and Taugbøl, 2008).

Table 8.1. Key to tables of selection criteria for potential ark sites.

Criteria Table number	Topic
1	Degree of enclosure
2	Terrestrial barriers: proximity to watercourses with potential for colonization by non-indigenous crayfish species
3	Aquatic barriers: for sites not wholly enclosed
4	Water quality and quantity
5	Non-indigenous crayfish and crayfish plague – local status
6	Angling
7	Usage and risks from access
8	Ownership
9	Physical habitat

Once ratings for a site have been obtained, the user reviews the compiled summary table of ratings, together with two other tables, entitled “Action” and “Rationale”, which guide the decision-making process. The “Action” table recommends whether to proceed or not, based on the number of ratings from best to poor. It suggests “go”, “improve then go”, “possible go”, or “no go unless other options limited”. There is no strict threshold for accepting a site as a potential ark site; it depends on the acceptable level of risk. There is no numerical scoring or aggregation of the qualitative ratings from the tables, because this would risk masking relevant factors. In addition, a scoring system would encourage adoption of some threshold of pass or fail for potential ark sites instead of an evaluation of relative risk.

The “Rationale” table asks the user to consider the site in a local and regional context of risks and conservation objectives. Different levels of risk will be accepted depending on the circumstances and this is best determined as part of a local or regional conservation strategy for white-clawed crayfish conservation. Current abundance of white-clawed crayfish varies markedly in different regions. The same threats apply everywhere, but the immediacy varies. In addition, the number of options for potential ark sites differs and the resources available to develop them.

Where users consider they have a largely suitable site, but with some risk factors, they can use a Table entitled ”Improvements” to set out their own plans for improvements to reduce risk or increase habitat quality at individual sites. This also allows a user to re-consider a site, which may be sub-optimal at present but may be more favourable if improvements are implemented. .

When the selection criteria have been used and a decision is made to proceed with a potential ark site, this does not mean a selected site should be stocked immediately. There should be a detailed feasibility study to: check that the introduction would not have any significant adverse impacts on existing features of high importance for biodiversity; secure the agreement of relevant stakeholders; identify an appropriate source of donor stock, and secure the resources necessary to set up the site and to monitor its success subsequently. Only if a potential ark site is confirmed as suitable at this detailed stage should an introduction be made. If the introduction is successful in achieving a breeding population it can then be classed as an established ark site.

The selection criteria have been issued as a spreadsheet tool, which can be downloaded from the Buglife website: http://www.buglife.org.uk/sites/default/files/Buglife%20Toolkit%20for%20Crayfish%20Strategy.pdf.

8.3 Results and discussion

The selection criteria require users to consider the rationale for action on potential ark sites for white-clawed crayfish and the level of risk that is appropriate to the local and regional circumstances. In many areas there will be few “best” sites and it will be necessary to accept some risks in efforts to conserve the indigenous crayfish. For example, in a catchment where existing populations of white-clawed crayfish are small, fragmented and being lost rapidly, delaying starting ark sites until ideal potential ark sites are available may mean that there are few or no populations left from which donor stock can be obtained within a catchment by the time sites have been selected and approved. In those circumstances, it may be better to start one or more potential ark sites where barriers may not be effective in the medium to long term, but nonetheless stocks of crayfish can be maintained and increased in the wild so that there is donor stock available when more or better sites are found.

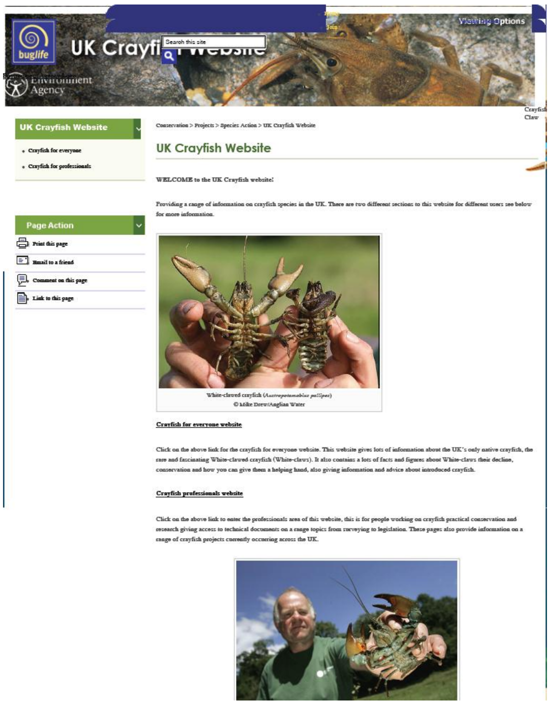
This approach has already been applied in North-west England in the Ribble catchment. White-clawed crayfish were formerly abundant and present in most parts of the upper catchment, but only three small partly isolated populations survived when crayfish plague swept through the catchment, apparently introduced as a contaminant with a consignment of stocked fish (Guthrie and Bradley, 2006, pers. comm.). One of the surviving populations is being lost progressively, due to an apparently plague-free population of introduced signal crayfish expanding down the small headwater stream. White-clawed crayfish were rescued from the leading edge of the invading signal crayfish population in 2007 and stocked into another tributary, where white-clawed crayfish had been lost some years previously due to crayfish plague. In the long term, that watercourse is not safe from colonization, but once the population develops, it can be used to stock other more secure sites once they are identified. An isolated length of watercourse was identified in 2008 as a good potential ark site in the catchment and after a feasibility study and the necessary approvals, another rescue operation was carried out to stock this site, which has lower risks in the long term. Crayfish populations in all three populations are being monitored (Peay, unpublished).

Wildlife Trusts, the Environment Agency, ecological consultants and others who may want to contribute to the conservation of white-clawed crayfish need guidance about potential ark sites now. If potential ark sites are not found and new populations established quickly, there is a risk that white-clawed crayfish will become extinct in individual catchments or whole River Basin Districts before alternative ark sites are found for the threatened populations. Only a handful of ark sites have been established so far, all of them recent (e.g. Sibley et al., 2006, Sibley et al., 2007, Peay and Hiley 2007a, 2007b, Peay and Guthrie, 2008), which means it has not been possible to thoroughly field-test the selection criteria at this stage. The selection criteria have been compiled with the knowledge available at present, from the literature and the experience of the author and others working with white-clawed crayfish in England, but there are uncertainties, which are acknowledged in the selection criteria. The purpose of providing the selection criteria now is to encourage people to take action locally and that completed assessments should be kept and compiled regionally. Over time when there are more established ark sites, the intention is to review the case studies and to revise the criteria and other guidance based on the evidence of outcomes.

Users who have used the selection criteria are recommended to lodge them with the Environment Agency locally and with Buglife. Completed assessments of sites and records of decisions taken would be of value. Which sites were screened out, and which were considered to be potential ark sites? Of those selected, which were taken forward to the introduction stage and what was the outcome for white-clawed crayfish, both in the short term and long term? Answers to these questions from plenty of future case studies will help to develop evidence-based guidance and conservation strategies for white-clawed crayfish in future.

It is clear that non-indigenous crayfish will continue to expand their range in England and Wales and that much of the existing range of white-clawed crayfish in watercourses and some still waters will be lost as non-indigenous crayfish gain ground. However, identification of potential ark sites and setting up those confirmed as suitable gives grounds for some optimism that the indigenous crayfish white-clawed crayfish will not be lost from the fauna of England and Wales. Ark projects are relatively simple to do and can provide achievable and measurable conservation benefits for indigenous crayfish. Action to find potential ark sites now will help to ensure white-clawed crayfish are there for future generations to appreciate.

Example of high quality habitat for white-clawed crayfish, with abundant limestone cobbles and boulders, submerged undercut banks and tree roots; good features for Ark sites



Examples of public communications about crayfish: a) screen shot from the UK crayfish website, hosted by Buglife the Invertebrate Conservation charity (top left); b) guidance on conservation strategy – the 'toolkit' (top right, see Chapter 9); c) information leaflet on biosecurity, produced by Environment Agency (bottom left); d) volunteers and children involved in a white-clawed crayfish conservation project at an ark site (see Chapter 8)

Chapter 9
Developing conservation strategy for white-clawed crayfish at catchment scale in England and Wales – a way forward?

9.1 Introduction

The white-clawed crayfish *Austropotamobius pallipes* (Lereboullet 1858) is the only native freshwater crayfish in Britain and is a species under threat in streams, rivers and lakes throughout its range in England and Wales, as well as in its wider distribution in parts of mainland Europe (Holdich et al., 2009b). Its status has already deteriorated to 'globally endangered' (Füreder et al., 2010a, Sibley et al., 2011). It is likely that only positive and urgent conservation action can safeguard populations for the future. The big issue in Britain now is what can be done to conserve the white-clawed crayfish, where should it be done and how can the most conservation benefit be obtained from the resources available?

Across its range the white-clawed crayfish has been reduced in abundance and extent by reductions of habitat quality. Adverse factors have included organic pollution (e.g. Foster and Turner, 1993, Demers and Reynolds, 2003, Trouilhé et al., 2003, Bramard et al., 2006), toxic pollution (Füreder et al., 2003, Howell and Slater, 2003), modification of watercourses by channelization or damming (Bramard et al., 2006), intensification of land use and loss of riparian vegetation (Füreder et al., 2003, Souty-Grosset et al., 2010), afforestation (Garcia-Arberas et al., 2009) and drought or water abstraction (Alonso et al., 2000, Nardi et al., 2005, Renai et al., 2006, 2008). These factors have caused fragmentation or loss of some populations. Yet, overall, the greatest threat to white-clawed crayfish is likely to be the disease crayfish plague, which is caused by the fungus-like organism *Aphanomyces astaci* Schikora 1903, and is usually carried by populations of non-native crayfish species from North America, of which Europe has eight species (Holdich et al., 2009) and Britain four species (Holdich and Sibley, 2009). The pathogen rarely has any significant effect on the non-native crayfish, but causes mass mortality in white-clawed crayfish (Diéguez-Uribeondo, 2006a). Even without the crayfish plague, populations of signal crayfish *Pacifastacus leniusculus* (Dana 1852) out-compete those of the native species (Holdich and Domaniewski, 1995, Peay and Rogers, 1999) and are the greatest threat to white-clawed crayfish together with the crayfish plague they carry.

This chapter discusses the challenges in developing conservation strategy for white-clawed crayfish and recommends an approach to developing action plans. It deals with the situation in England and Wales, but threats to white-clawed crayfish are similar in most of its European range (Holdich et al., 2009), so effective conservation management is just as important in other areas, for white-clawed crayfish and indeed for other European crayfish species, which are becoming increasingly threatened. There are 590 crayfish species known, of which twenty-four percent have already been assessed as threatened (Sibley et al., 2011) and there are potentially other threatened species among those classed as data deficient in the IUCN Red List (IUCN, 2010). Many of the threatened crayfish species are also affected by loss of habitat and the impacts of introductions of crayfish species far beyond their native range. The need for development and implementation of action plans for crayfish species seems likely to increase in future.

Within England and Wales, whilst some populations of white-clawed crayfish were likely to have been lost from lengths of lowland river due to urban and industrial pollution around major towns and cities in the 19th and 20th centuries, it still had ‘somewhat continuous distribution throughout the country’ in the early 1970s (Thomas and Ingle, 1971). It was the stocking of signal crayfish into fish farms and other ponds from the mid 1970s that started the severe decline of white-clawed crayfish. The first outbreaks of crayfish plague and mass mortality of white-clawed crayfish were recorded only a few years after the first introductions of signal crayfish (Alderman 1993) and signal crayfish soon escaped from nominal captivity and established invasive populations in watercourses (Holdich et al., 1999b, Sibley et al., 2002).

Action plans to conserve the native white-clawed crayfish need to take account of the current and future distribution of non-native crayfish and the management issues posed by them to have the best likelihood of achieving their objectives. The situation described here for England and Wales illustrates the problems and opportunities.

9.2 Policy and regulation

Concern about crayfish plague and observed losses of extensive populations of white-clawed crayfish during the 1980s led to the species being given protection under the Wildlife and Countryside Act 1981 (as amended) from 1988 (Holdich et al., 2004). The legal protection from taking or sale of white-clawed crayfish was not directly beneficial, because there was little or no harvesting of white-clawed crayfish at the time and it did nothing to protect populations from either crayfish plague or invading signal crayfish. It did, however, raise the profile of white-clawed crayfish and give a policy incentive for a species action plan for white-clawed crayfish and for the regulation of keeping or stocking of non-native crayfish.

The protected status given to white-clawed crayfish was followed by preparation of a species action plan as part of the UK Biodiversity Action Plan (UK Biodiversity Steering Group, 1995). Signal crayfish and two other species of non-native crayfish (noble crayfish *Astacus astacus* (Linnaeus 1758) and narrow-clawed crayfish *Astacus leptodactylus* (Eschscholtz 1823) were added to Schedule 9 of the Wildlife and Countryside Act in 1992, making it an offence to release them or allow them to escape. This could be viewed as either a valuable tool to help prevent further introductions, or a record of the failure to prevent damagingly invasive species from becoming so widespread prior to regulation that they were already considered to be established or 'ordinarily resident', as described in Schedule 9. Following public consultation in 2007-2008, the Department of Environment, Food and Rural Affairs (Defra, 2009) added two other non-native crayfish species to Schedule 9 (red swamp crayfish *Procambarus clarkii* (Girard 1852) and spiny-cheek crayfish *Orconectes limosus* (Rafinesque 1817)).

Despite the introduction of legislation against introduction of non-native crayfish, further accidental and deliberate introductions of signal crayfish and other non-native crayfish continued throughout the 1990s and indeed in 2000s, in addition to the natural extension of range by established populations. This may have been due, at least in part, to lack of public understanding about the impacts of non-native crayfish. The main pathways for introduction of non-native crayfish were probably: introduction for future wild harvest; introduction to fishing lakes or fish farms for weed control, fish food, or to clean up dead fish; discard of surplus live crayfish intended for human consumption; discard of unwanted aquarium stock, accidental introduction with fish for stocking, and crayfish used as angling bait (Peay et al., 2010). It is sometimes difficult to be certain about the actual reason for individual introductions.

Additional regulations on crayfish were introduced in 1996, to prevent keeping of crayfish, or stocking them, but with some exemptions. Crayfish can be kept and sold for live food in England and Wales, provided surplus crayfish are killed before disposal, but not in Scotland or Northern Ireland. One species of crayfish can be kept in aquaria, redclaw *Cherax* *quadricarinatus* (von Martens 1868), an Australian species, which is thought not to be able to breed in the wild in Britain. Keeping of this species in aquaria is not allowed in Scotland or Northern Ireland. A nationwide ban on use of crayfish as angling bait was introduced in 2005 and the Environment Agency introduced restrictions on the use of traps for crayfish from 2007, requiring individual consents to be obtained for use of crayfish traps.

With signal crayfish already so widespread in southern England before the start of regulation in 1996 and with new areas being invaded all the time, riparian landowners in many catchments soon found themselves with non-native crayfish, whether they wanted them or not. Under the Keeping of Fish (Crayfish) regulations 1996, Britain was divided into two zones: “go” areas, where keeping and harvesting of signal crayfish would be permitted, covering most of southern England and “no go” areas, where keeping or harvesting of signal crayfish would not be allowed, across central and northern England, most of Wales and all of Scotland (Department for Environment, Food and Rural Affairs, 2008). As regulation could not be applied retrospectively, where signal crayfish were already established in “no go” areas before 1996 it would not be an offence to continue to keep them.

9.3 Developing strategy for management of crayfish

9.3.1 How conservation strategy for white-clawed crayfish has developed

The UK Biodiversity Action Plan (BAP) for white-clawed crayfish (UK Biodiversity Steering Group, 1995) had as its primary objective: ‘attempt to maintain the present distribution of this species by limiting the spread of crayfish plague, limiting the spread of non-native species, and by maintaining appropriate habitat conditions’. Since the publication of the BAP non-native crayfish have increased their range in the many catchments in England and Wales where they were already established and have been found in additional catchments too, with corresponding reductions in the extent of white-clawed crayfish. The objective of maintaining present distribution has not been achieved. Further losses of range of white-clawed crayfish seem to be inevitable. Even so, many conservation agencies do consider that there is scope to keep natural populations of white-clawed crayfish in most regions, by planning and implementing local action plans. There has been growing interest in the conservation of native crayfish in England and Wales in the past five years or so, with the formation of groups to further crayfish conservation, for example in south-west England

(Nightingale, 2009), Kent, Norfolk, Derbyshire, Nottinghamshire (Holdich and Jackson, 2011), Staffordshire, Sheffield (Dangerfield, 2011) and Cumbria (Backshall, 2011). The composition of the groups varies in different areas, but most include the statutory agencies, i.e. the Environment Agency and either Natural England or Countryside Council for Wales; plus some local authorities and generally one or more non-statutory agencies, such as the Wildlife Trusts, Rivers Trusts, Buglife and other groups, often with help from local volunteers.

The Environment Agency has maintenance of biodiversity in freshwaters as part of its remit for protection and enhancement of the environment. In 2009 the Environment Agency developed a five-year Biodiversity Strategy for white-clawed crayfish (Christmas, 2009), which updated the UK BAP for white-clawed crayfish, set aims, specific goals and measures of success for the agency. The Environment Agency's strategy recognized the importance of having a coordinated approach to the conservation of white-clawed crayfish and emphasised the need for statutory agencies to adopt the same conservation approach and to 'target resources to get more effective outcomes more efficiently'. To deliver the strategy in England and Wales would require 'regional delivery plans appropriate to the needs of the regional geography and conservation need' (Christmas, 2009). One of the goals set in the strategy was to 'provide catchment-scale decision-making tools to guide conservation effort.' This has led to the development of a 'toolkit' for developing a catchment-scale conservation strategy for crayfish (Peay et al., 2011).

The toolkit does not specify the conservation strategy for each catchment, but instead gives some guidance on how to prepare action plans. It was issued as a consultation draft early in 2011, so potential users could provide feedback on the material, with a final version expected later in 2011, after approval by other conservation agencies in England and Wales. This, together with a range of other guidance and general information on crayfish in Britain has recently been made available through a new dedicated website (www.crayfish.org.uk), operated by the invertebrate conservation charity Buglife (Kindemba, 2011). Both of these initiatives contribute to the strategic aims 'to ensure best practice is freely available to all, so we work strategically to conserve and enhance native crayfish populations' (Christmas 2009).

9.3.2 A toolkit for conservation strategy

Conservation strategy for white-clawed crayfish in England and Wales is only likely to be effective if it thoroughly considers the current and future distribution of non-native crayfish and crayfish plague, as well as the distribution of white-clawed crayfish. The toolkit for developing a catchment-scale conservation strategy for crayfish recommends assessing catchments as whole units from source to estuary. This means some are relatively large units, which may cover several administrative districts, but the whole river catchment is an appropriate scale for conservation strategy here. This is because wherever non-native crayfish have established in a watercourse, they can be expected to colonize all connected waterbodies in the catchment eventually, unless there are significant barriers to invasion. These factors and other existing or potential threats need to be assessed as part of each catchment risk assessment for white-clawed crayfish. The threats and opportunities need to be considered for whole catchments, but action plans for individual sub-catchments may be easier or more efficient to implement, e.g. if dealing with habitat quality and land use issues.

The toolkit recommends involving local agencies in the development of catchment-based action plans, coordinating data management to make best use of available information and carrying out appropriate action at regional and local scale. The approach suggested is to compile the available data on the distribution of crayfish species, any past records of crayfish plague, together with environmental data on water quality, land use, physical barriers in watercourses and other anthropogenic factors, and use this to carry out a catchment risk assessment. Then, based on the status and the current and future threats, develop specific action plans to target conservation effort in the catchments. The priorities will differ in the catchments.

The toolkit offers an approach to catchment risk assessment for catchments with and without white-clawed crayfish. It takes the user through a series of questions and choices in a succession of simple flowcharts to make qualitative assessments of the conditions in the catchment and to use this to identify actions that may be applied in a catchment of this type. The topics reviewed in the flowcharts include: 1. the existing status and trend of white-clawed population in the catchment; 2. the incidence or risk of crayfish plague; 3. the risk from non-native crayfish in the catchment; 4. the water quality, especially from anthropogenic influences and 5. the condition of physical habitat.

In each case the status of these risk factors is given a grading on a four-point scale, ranging from excellent (favourable) to bad (unfavourable). The suggested actions and priority actions identified individually under each topic can be pooled together by those carrying out catchment risk assessment and action planning, and used to help develop specific targeted actions for the individual catchment. Descriptions are given to help assign a grading to each risk factor, although they are necessarily general and should be considered on the basis of local conditions and best judgement.

Although the focus here is on conservation of white-clawed crayfish, catchment risk assessment is also recommended for catchments that do not have white-clawed crayfish, to identify potential for invasion by non-native crayfish and the threats this poses to other biodiversity. Preventing establishment of signal crayfish in new catchments is already recognized as a high priority in Scotland under the Scottish Species Action Framework, in which the signal crayfish is listed as one of the invasive species posing a significant threat to biodiversity (Scottish Natural Heritage, 2007). Scotland is beyond the natural geographic range of white-clawed crayfish, however, two established populations exist in lakes, whereas it already has at least 15 catchments being invaded by signal crayfish (Gladman, 2009).

The toolkit includes a table with summary recommendations for catchments, based on the status of crayfish in the catchment, i.e. white-clawed crayfish only, mainly white-clawed crayfish, mainly non-native crayfish, only non-native crayfish, or no crayfish. The recommendations are grouped under a series of topics in three main categories: protection of white-clawed crayfish, management issues with non-native crayfish and overall issues. ‘Protection of white-clawed crayfish’ deals with recommendations on site protection and designation of ark sites (i.e. running water or still water sites with white-clawed crayfish, which are isolated from non-native crayfish and crayfish plague), and monitoring. The topic ‘Management issues with non-native crayfish’ deals with: fisheries management; harvest/control of non-native crayfish, and eradication of non-native crayfish (or how to decide whether a technically feasible treatment is worth doing). ‘Overall issues’ deals mainly with education and promotion, and a concluding section suggests possible priorities within each category of catchment (based on status of crayfish). The summary table is an alternative, or supplement to the flowchart approach. It lists possible actions for a scenario based on overall status of crayfish, whereas the flowcharts suggest individual action points based on the risk factors in the catchment risk assessment. Either or both can be used.

Existing biological and geographic datasets provide the starting point for catchment risk assessments. Distribution data on crayfish are held in various local biological records centres, in Environment Agency offices and Natural England or Countryside Council for Wales, where they have been submitted as a condition of licences to carry out surveys of the protected species white-clawed crayfish. There was formerly no coordinated system for regularly compiling crayfish records nationally, although the Environment Agency held the most comprehensive dataset. The import of crayfish data into the National Biodiversity Network (NBN) (http://data.nbn.org.uk/) (Harding, 2003) provided publicly accessible distribution data and this dataset is developing progressively. Agreements are being developed so that data-holding agencies will copy all the records of crayfish to NBN, although there is likely to be a time lag for records to reach the database. In 2010 the Department of Food, Environment and Rural Affairs (Defra) commissioned a study on crayfish plague in which existing datasets on distribution of crayfish species were combined with historic records of the original stockings with signal crayfish and recorded outbreaks of crayfish plague by catchment (Rogers and Watson, 2011). A compiled dataset may be a potentially useful starting point for catchment risk assessments, if it is made available to potential users in the regions in future and is kept up to date.

Part of the process of considering the status of white-clawed crayfish in a catchment risk assessment is considering the extent of potential habitat for white-clawed crayfish in the watercourses and still water sites. Excluding stretches that are too acidic, too high energy, lacking in perennial wetted habitat, or upstream of major natural barriers such as waterfalls, white-clawed crayfish probably occupied most of the catchments in which they have been recorded, as far downstream as the tidal limit. For example, white-clawed crayfish occurred as far downstream as the weirs at the tidal limits in the Yorkshire Ouse and Wharfe in the mid-1990s (Peay and Rogers, 1999), although were lost from the polluted lower stretches of other rivers, e.g. in the Aire, Calder, Rother and Don (Christmas, 2009, pers. comm.).

If the white-clawed crayfish population is currently sparse or fragmented compared to the potential historic range, the difference between the potential maximum extent and the current distribution represents the loss of range. Where there has been no occurrence of non-native crayfish or crayfish plague in a sub-catchment, the loss of range is generally due to land-use factors. For at least some types of habitat degradation, remediation is possible and indeed has occurred; for example due to improvement of the quality of effluents from waste-water treatment works, or by fencing stream banks to protect them from damage by livestock (Peay, 2003a). By contrast, losses of range due to non-native crayfish are progressive and generally irremediable, as although many control methods have been considered (Holdich et al., 1999a, Freeman et al., 2010), no effective measures are available for established, extensive populations.

Grading of the risk factors in a catchment is likely to help determine priorities for action. The critical factors are generally the least favourable ones. For example, a population in a sub-catchment may have generally favourable factors, with some localised issues of poor habitat quality due to trampling of banks by livestock or small polluting discharges. The habitat could be improved to make conditions more favourable. If, however, there is an extensive population of signal crayfish in the watercourse downstream, that may be the most severe threat. An action plan needs to focus on whether there is an effective barrier to upstream invasion, or whether one can be provided in the predicted time available before the invading population arrives. The time until contact between the two species depends on the rate of invasion and how effectively the leading edge of the population can be detected. In these circumstances, if a barrier to invasion is not a feasible and acceptable option, *in situ* conservation of native crayfish is unlikely to be possible in that watercourse. Enclosed still-water sites, which can keep populations of white-clawed crayfish safely isolated from non-native crayfish and crayfish plague (i.e. ‘ark sites’) (Holdich et al., 2004) may be the only option locally in some cases. Guidance has been produced on selecting ark sites in England and Wales (Kemp et al., 2003, Peay, 2009a, Peay et al., 2009).

9.3.3 Ark site and donor stock

The recommended conservation strategy for white-clawed crayfish is to defend the largest extent of population, but have contingency plans in place to minimise further losses if one of the barriers to loss is overcome in future. The goal of keeping whole river basin districts free from non-native crayfish and plague has already been lost in England and Wales. There are still some whole catchments without non-native crayfish, however. Every effort should be made to prevent entry of non-native crayfish or crayfish plague, by public education and use of regulation. If prevention fails and a population of non-native crayfish is found, or an outbreak of crayfish plague occurs, good planning for these contingencies may mean that at least some of the sub-catchments can be kept wholly, or largely, safe for white-clawed crayfish. If conditions deteriorate, pre-prepared ark sites within the catchment may provide long-term security for at least some populations within the catchment or individual sub-catchments, i.e. a tiered approach is needed; planning to keep all, but defending progressively smaller populations where necessary.

If there are no adequate barriers to block an invasion and there are no existing or potential ark sites available within the catchment in time, stakeholders have three options: 1. do nothing and leave the white-clawed crayfish to be replaced by non-native crayfish; 2. translocate a portion of the white-clawed crayfish population to a potential ark site in another catchment within the river basin district or region; or 3. take stock for captive-breeding and release progeny as soon as potential ark sites are available (e.g. Bradley 2010, Nightingale and Rudd 2011). The second and third options are not necessarily mutually exclusive. Option 1 ‘do nothing’ may be unavoidable in some cases, if there is not the time, the will and the resources locally to act effectively to save the population. In some cases it may be necessary to make difficult choices and concentrate effort on the sites or areas that have the best chance of success.

Captive breeding has some advantages, especially if crayfish in donor populations are fairly sparse, because a small number of crayfish can be used. If necessary, the original adult stock can be returned to the source population, either when the juvenile crayfish detach from the female (Policar et al., 2010), or by removal and rearing of eggs (Carral et al., 2003). Captive breeding has start-up and operational costs that are likely to be higher than simple translocation of wild stock because of the equipment and time-scale required for captive breeding. Souty-Grosset and Reynolds (2009) considered it preferable to use a good donor stock and harvest it heavily, than to take a few crayfish from a feeble population, but where all potential donor populations are weak, captive bred stock is needed. There would be potential conflict if running a captive breeding facility used up all the resources that would otherwise be used for finding and establishing ark sites, because the primary aim is maintaining wild populations. If captive breeding is carried out, there should be a clear plan for how stock will be used in future, although as well as stock for (re-)introductions, it could include provision of white-clawed crayfish for exhibits in zoos and public aquaria where this can contribute to the conservation message, as is being done at Bristol Zoo Gardens (Nightingale, 2009).

Unlike the populations of white-clawed crayfish in some areas of mainland Europe, where there can be significant genetic differences between populations in some adjacent catchments, populations in England and Wales are considered to be similar across their range and closely related to those in NW France (Souty-Grosset et al., 2003). Even so, keeping new ark populations within their source catchments, or at least the same river basin district, is recommended as a precautionary measure (Kemp et al., 2003). Souty-Grosset and Reynolds (2009) recommended genetic studies to assess the variation within populations and this may be especially relevant in parts of Europe where there can be significant variations between catchments due to past patterns of connection and isolation of watercourses after successive glacial periods. In principle, source populations that have high natural genetic variability may have the best chance of adapting to new conditions. Reynolds et al. (2002a) detected slight genetic differences between a donor population in an Irish stream and a lake site only 80 years after it was stocked from the stream. This may be due loss of some of the founders shortly after stocking, or to local adaptation to a different habitat over time. Where stocking is being done in an ark site and the action to start a new population is because the donor population is under imminent threat, genetic studies are of value, but should not delay necessary conservation action. Samples can be investigated later if required, whereas delaying setting up the ark site may mean the donor population is severely reduced or lost before stock can be moved.

A white-clawed crayfish population can be used as donor stock, even if it is in the process of being invaded and out-competed by signal crayfish, i.e. where it appears that there is no crayfish plague. Donor stock was taken from a mixed population to start an ark site in Ribbledale, North Yorkshire (Peay et al., 2009) and subsequent monitoring indicates that white-clawed crayfish have survived the first two years in the new site (Peay, unpublished). Clearly, if both species are present, there is a need for multiple checking of every crayfish to ensure only white-clawed crayfish are translocated, which was done in the case described. There is no concern about taking as much donor stock as practicable from the wild in these conditions, because any white-clawed crayfish left *in situ* will not survive more than a few years at most.

If there is a significant risk that invading signal crayfish are carrying crayfish plague, any salvage of the white-clawed crayfish population has to be done before the signal crayfish come into contact. The time until contact may be difficult to predict when the invading crayfish are at low abundance initially. There are several options 1. transfer white-clawed crayfish to a holding facility for a quarantine period; 2. transfer white-clawed crayfish directly to an enclosed ark site and hold a proportion of the donor stock in cages in the site, to see if they survive; 3. take white-clawed crayfish only from upstream of a temporary barrier, such as a small cascade, to increase the confidence that white-clawed crayfish are still uninfected at the time the stock is removed. Option 1, quarantine, is a potentially useful approach, but it has some potential disadvantages too: cost; smaller founder population can be taken; there is potential for losses in the holding facility other than from crayfish plague, and it causes delay, during which the rest of the white-clawed crayfish population may be lost. Rescue of as many white-clawed crayfish as is practicable direct to a new ark site offers the advantage of a large founder population; but, in addition, it may be worth having a small stock in a holding facility, in case any infection was present in the main batch. If so, the captive stock gives the option of a re-introduction after the plague outbreak.

Records of donor populations and corresponding stocking of white-clawed crayfish are held by the Environment Agency nationally and this information will need to be kept up to date with the action plans for white-clawed crayfish in the catchments and regions/river basin districts.

Ideally, all the catchments currently with white-clawed crayfish would have at least some populations surviving in areas considered secure enough to be classed as existing ark sites in both watercourses and still waters. In addition, there would be several or many other potential ark sites available and various others expected in future. This would give flexibility to cope with occasional impacts and subsequent re-stocking from other sites if necessary. For example, there might be a pollution incident, or a need to drain and dig out an amenity lake to restore it. Maintaining public awareness of the need for biosecurity may be easier if there are clusters of ark sites in a catchment or adjacent catchments. When populations develop sufficiently in the new ark sites they can be used as local donor stock for additional sites, although it may be 10-15 years, or more, before the population is large enough to use for other introductions or re-stocking. Having many populations in a range of sites of different types and sizes within a catchment or region is recommended as a way of maximizing the number of populations that survive.

In practice, there are likely to be catchments where it will be difficult to find any potential ark sites, because signal crayfish have already been introduced into so many ponds, lakes and watercourses and there are few or no sites coming forward from mineral workings. If a lone ark site is set up in an area already completely dominated by signal crayfish, it will need a high degree of biosecurity if it is to survive in the long term, especially protection from crayfish plague. Souty-Grosset and Reynolds (2009) reviewed projects in which there was re-stocking of white-clawed crayfish, where they had been lost due to crayfish plague and others involving introductions into new sites. In the projects in six countries across Europe the authors found 26 of 59 were successful. Projects where there was re-stocking after an outbreak of crayfish plague may have been at more risk, because the factors that led to a site becoming infected in the first instance may still have been present, making re-infection more likely. In the selection criteria for ark sites, Peay, (2009a) recommended considering factors such as the type and frequency of public access to sites and the status of crayfish plague in the surrounding area.

9.3.4 Action planning in catchments with different status of native crayfish

Of the catchments still solely with white-clawed crayfish, Cumbria in the Northwest River Basin District has the greatest concentration and can be considered to be of very high importance for white-clawed crayfish in England. The county has several river systems with white-clawed crayfish throughout the catchment, most notably the Eden and Kent catchments, which are thought to be completely free from non-native crayfish at present. These are both river systems of high quality and designated as Special Areas for Conservation (SAC), of European importance for nature conservation (Joint Nature Conservation Committee, 2011). Any invasion of non-native crayfish in these rivers would be expected to have a potentially significant adverse impact on the integrity of some of the features for which the sites are designated: including white-clawed crayfish, brook lamprey *Lampetra planeri* (Bloch 1784) and river lamprey *Lampetra fluviatilis* (Linnaeus 1758), Atlantic salmon *Salmo salar* (Linnaeus 1758), bullhead *Cottus gobio* (Linnaeus 1758), and several aquatic vegetation types (*Littorelletea uniflorae*, *Ranunculion fluitantis* and *Callitricho-Batrachion* vegetation). The importance of these Cumbrian rivers for white-clawed crayfish is even more significant now that another of the designated rivers with abundant white-clawed crayfish in northern England has recently been found to have non-native crayfish in its upper tributaries, the Yorkshire River Derwent SAC (Penn, 2009, pers. comm.), which means that the populations in that river system are now threatened.

The whole county of Cumbria was thought to be entirely free from non-native crayfish, but a population of signal crayfish was discovered in 2005 in a tributary of the Cumbrian River Derwent (entirely separate from the Yorkshire River Derwent), which is part of the River Derwent and Bassenthwaite Lake SAC. The location by a road bridge in a rural area with no dwellings nearby suggests the crayfish were either introduced accidentally with stocked fish, or put into the river deliberately. Although there are no white-clawed crayfish in the Cumbrian River Derwent, invasion by signal crayfish has the potential for other ecological impacts in the river and lakes downstream. Of future concern is that with 8.3 million day visitors a year to the Lake District of Cumbria (Cumbria Tourism, 2008) there is a risk that people may start to move the signal crayfish around. This would be a potential risk to the nearby Eden and Kent catchments, if, or rather when, the signal crayfish become abundant in the popular riverside areas near Keswick on the River Derwent, where they would become readily accessible to the public. Human-assisted introductions of signal crayfish have been a significant source of losses of white-clawed crayfish (e.g. Alderman, 1993, Diéguez-Uribeondo, 2006b).

The toolkit has already been used in its draft form to help develop a conservation strategy for white-clawed crayfish in the Eden catchment (Backshall, 2011) tailored to include specific proposed actions and targeted outcomes.

Since 1996 much of the focus of conservation effort on the River Eden and its tributaries has been on improvement of river habitat quality, for example by fencing off riparian habitats from excessive grazing and trampling by sheep and cattle, or dealing with local issues of pollution from farms. Much of this was carried out as part of agri-environment schemes or with assistance from other funds, for example work carried out by the Eden Rivers Trust. Such measures protect the habitat of white-clawed crayfish and other species and are proposed to be extended to other areas of the catchment in future where there are local problems. They have the potential to increase the abundance of white-clawed crayfish and extend their range the within the catchment.

Nonetheless, the benefits of these measures for white-clawed crayfish are dependent on the whole catchment remaining free from non-native crayfish and crayfish plague in future. The toolkit on conservation strategy recommends investing in prevention and contingency planning. Promoting public awareness of the need for biosecurity is recommended as a high priority for the dwindling number of catchments where the status of white-clawed crayfish is assessed as ‘good’ or ‘excellent’. The success in keeping catchments and individual sites free from non-native crayfish and crayfish plague will depend on the environmental regulators (especially the Environment Agency), conservation groups and other local people, regarding non-native crayfish, fisheries management and biosecurity, at catchment scale and regionally. Where stocking with fish is carried out in ark sites, maintaining biosecurity might require stocking only from fish farms without non-native crayfish, or at least handling measures to ensure no accidental transfer of crayfish and disinfection to avoid transmission of crayfish plague.

Encouraging local appreciation of populations of white-clawed crayfish is important. Public awareness-raising is likely to be needed in a wider area than the immediate vicinity of an ark catchment or site. This is especially so, if the ark site is relatively small, in an area where catchments nearby are extensively invaded. Work on awareness-raising and education has already started in Cumbria and the North-West region, with the production of leaflets on crayfish by the Environment Agency and recent features in press and television. Similar initiatives have started in some of the other regions in England.

Another recommended priority for catchments or sub-catchments with only white-clawed crayfish is preparing contingency plans to deal with an outbreak of crayfish plague, if prevention fails. Rapid action could make the difference between complete eradication of the population and retention of a significant part of it. In the Porter Brook in Sheffield, South Yorkshire, a stock of white-clawed crayfish was rescued from ahead of an outbreak of crayfish plague in 2009, kept in captivity and then returned to the watercourse upstream of a barrier (Bradley, 2009, pers. comm.; Dangerfield, 2011). Modifications to existing weirs are proposed to improve biosecurity in future (Dangerfield, 2011). Even if there is some risk to the population in the long term, it keeps a local population going while new ark sites are being sought. This action would not have been possible if there had not been an active group involved in crayfish conservation locally and a facility to house the rescued crayfish available at short notice.

In Cumbria the Eden River Trust has propose a four-tiered approach to ark sites in the Eden catchment (Backshall, 2011). Firstly, the whole Eden catchment is considered as an ark site, as it is considered to be free from non-native crayfish; secondly, the best sub-catchments for white-clawed crayfish have been identified e.g. Rivers Leith and Lyvennet, Hoff and Helm Becks, and others, where effort will be concentrated on enhancement; thirdly, parts of the headwaters of these sub-catchments are being identified upstream of barriers, which might make good barriers to crayfish plague if an outbreak occurred, and fourthly, isolated still water sites will be identified as ark sites, with existing populations, or with potential to introduce them in several sub-catchments. In addition, contingency plans will be put in place, so that all the local agencies can act quickly and effectively in the event of an outbreak of crayfish plague, or the detection of non-native crayfish.

Prompt investigation of any reports of non-native crayfish from the public is another important element of contingency planning in any catchments or sub-catchments with white-clawed crayfish, or without previous records of non-native crayfish. If a new population of non-native crayfish is confirmed and the catchment changes status to 'mainly white-clawed crayfish', there is rarely any effective action that can been taken to reverse it.

To date, the only treatments to eradicate populations of signal crayfish have been carried out using biocides, (Peay et al., 2006a, Sandodden and Johnsen, 2010), at relatively small sites, where the entire population could be fully treated. The biocides available are not specific to crayfish, so there are localised environmental impacts, albeit fully recoverable. Some of the initial trials using natural pyrethrum have been successful, as confirmed by monitoring for 5 years (see Chapter 4). The larger and more complex the site, the higher the cost of treatment and the greater the risk that eradication is not achieved because some crayfish evade or survive the treatment. Biocide treatment is a potentially useful tool for rapid response, where a new population of non-native crayfish is detected soon enough, it is within a relatively small or manageable site, and there are significant benefits from eradicating the population.

There would be significant benefit where a treatment protected a whole catchment or sub-catchment from invasion by non-native crayfish, especially if the catchment had features considered to be of high value, for example if it was an internationally or nationally designated site, e.g. SAC or SSSI, where the ecology was potentially vulnerable to non-native crayfish (e.g. the Eden catchment in Cumbria). By contrast, there would be little benefit for biodiversity in treating a fishing lake that would be re-invaded from a connected watercourse within a year or two. Eradication treatment is only recommended as a rapid response. Delaying treatment in order to try selective control methods, such as physical removal of crayfish, or else waiting until there is unequivocal evidence of spread and impact greatly increases the risk that the opportunity for success will be lost. Once non-native crayfish extensively invade lakes and rivers there is no current prospect of eradication and little or no option for any effective control at present.

Where non-native crayfish are present in a catchment where there are white-clawed crayfish, i.e. partly invaded catchments, in situ conservation is still recommended as a priority, where it is feasible, but it needs to be assessed realistically. Parts of watercourses upstream of barriers or enclosed waterbodies may be able to sustain white-clawed crayfish as existing ark sites, although there is still a risk that some will be lost due to crayfish plague, or because a population of non-native crayfish is detected upstream of the barrier in future, or there are unexpected events such as major pollution incidents. Action plans for white-clawed crayfish should therefore work towards replacing populations by re-stocking and/or new ark sites to mitigate the impact of losses that cannot be avoided.

The toolkit recommends biosecurity measures to keep as many sub-catchments as possible free from non-native crayfish, by identifying barriers to invasion and especially by encouraging people to avoid moving non-native crayfish around. The priorities for action plans in partly invaded catchments are likely to involve identifying the areas where existing populations of white-clawed crayfish have the best likelihood of surviving, enhancing the likelihood where practicable and by increasing future provision of ark sites as well. Strategically, the most effective time to do this is early on in the invasion, before white-clawed crayfish populations are severely reduced, when there is sufficient time to carry out the conservation actions required.

There can be a significant lead-in time for new ark sites, especially former quarries, where ark sites may form part of the restoration proposals and therefore be constrained by the time taken for the approval process, completion of mineral working and any habitat remediation required. In addition, potential ark sites need to be surveyed to assess the potential impacts of any introduction of white-clawed crayfish.

Changes in ecology due to white-clawed crayfish can be expected to be minor compared to those of non-native crayfish, although studies in a lake in Ireland (Matthews and Reynolds, 1992) found that charophytes showed an increase in growth after white-clawed crayfish died of crayfish plague and there was an increase in the abundance of planorbid molluscs and freshwater shrimps. If sites are already of high importance for nature conservation due to rare aquatic invertebrates that could be reduced in abundance by white-clawed crayfish, they may not be appropriate as new ark sites. If a very precautionary approach is taken with all sites, however, no ark sites will be started, or the time taken to carry out an ecological assessment of the sites may be so long that the potential donor populations are severely reduced, or lost from entire catchments, before any translocation is carried out. A sensible balance is needed in surveying and assessing the suitability of sites.

We can afford to accept some uncertainty about ark sites, even if interactions between white-clawed crayfish and all of the species in the waterbody are not fully known. In an extreme example, suppose that after stocking of an ark site in a small lake, a species was found, which subsequent studies indicated might be significantly affected by white-clawed crayfish, and it was decided that conservation of the other species had higher priority than that of white-clawed crayfish at that site. In such a case, there would be an option to move some of the white-clawed crayfish to an alternative site and eradicate the rest using crayfish plague. This would be very unlikely to be carried out in practice and it is only mentioned here to make the point that decisions on ark sites need not be irrevocable, whereas introductions of North American non-native crayfish usually are so.

In England and Wales, a high proportion of clean still-water sites in lowland areas support breeding populations of one or more amphibian species. Among these is the great crested newt *Triturus cristatus* (Laurenti 1768), a European priority species. It is widely distributed and in some areas is relatively abundant. Historically, it suffered major reductions due to habitat fragmentation, introductions of fish to breeding ponds and lack of habitat management, as well as loss of ponds due to development and agriculture. Although the decline has been greatly reduced (Joint Nature Conservation Committee, 2010), there is uncertainty as to whether the national population is now recovering. Compensatory provision of habitat and translocation of populations is required whenever ponds with great crested newts are affected by development and many other new ponds are being created, although there are still populations being reduced or lost due to introductions of fish into ponds. O'Neill and Whitehouse (2011) have made the case that the risk of adverse impacts on great crested newts from introductions of white-clawed crayfish is low and the optimal sites for great crested newts are not favourable for white-clawed crayfish, because they dry out occasionally. Hence, they argue, sites with great crested newts need not be discounted automatically as potential ark sites for white-clawed crayfish.

Changes of the status of crayfish species in river catchments continue in England and Wales, as new populations of non-native crayfish are discovered each year. For example, records compiled by the Environment Agency in the Humber River Basin District show that four of the major sub-catchments of the River Ouse (Aire, Nidd, Swale and Derwent) have had populations of non-native crayfish identified in them within the last ten years, losing their previous status as 'white-clawed crayfish only'. The River Ribble, which has its headwaters in Yorkshire, but is in the Northwest River Basin District, was also found to have changed status in the same period. This means that none of the main river catchments within the Yorkshire region is wholly free from non-native crayfish, although there are tributary sub-catchments that are still free from invading populations and have locally abundant white-clawed crayfish.

In addition, signal crayfish have become established in parts of the canals in the Aire and Calder catchments in Yorkshire as well and this is likely to facilitate invasion of some of the smaller sub-catchments. The canals offer a faster route for invasion than would be so if the invasion was solely via one tributary, to the main river and then progressively upstream and downstream on the main river to each confluence and from there up the tributaries. Feeder streams and overflows from the canals may offer shortcuts. The Yorkshire region was one of the strongholds for white-clawed crayfish in England in the 1990s. It remains so to a large extent, because there are still locally abundant and extensive populations in many rivers and streams, but the deteriorating status is of significant concern, both within the region and nationally.

Several of the populations of signal crayfish invading the rivers in North Yorkshire have been replacing white-clawed crayfish by competition, rather than by immediate loss due to crayfish plague, e.g. in the Wharfe (Peay and Rogers, 1999, Bubb et al., 2005), Ure (Bubb et al., 2005) and a tributary of the Ribble (Peay et al., 2009). There have been outbreaks of crayfish plague, in the Rother/Don catchment in Sheffield, South Yorkshire, most recently in 2009 (Bradley, 2009, pers. comm.). The signal crayfish in that area are likely to be carrying crayfish plague and this may increase the risks of accidental transmission to other sites with white-clawed crayfish in the Rother/Don catchment and beyond. This is why a local action group for crayfish conservation, Crayfish Action Sheffield, has been set up and, with the help of a part-time project officer, is engaging with angling groups, especially young anglers, and with local societies to increase awareness of crayfish issues (Dangerfield, 2011).

The Crayfish Action Sheffield project has identified four potential ark sites already in the Rother/Don catchment, although consent has only been obtained for one of them so far. Bradford District Council, which is mainly within the Aire catchment, has prepared an action plan for its area. Signal crayfish have recently reached Bradford district via the River Aire and there are a few still waters reported to have them too, but there is at least one existing ark site, several others upstream of dams that require confirmation of status and others are being sought. There have been some recent surveys in some of the other tributaries in the Aire catchment, by the Environment Agency and others, but there is no action plan for the whole catchment as of yet. A catchment risk assessment and preliminary search for potential ark sites is underway in the Nidd catchment as a student project in 2011 (Slingsby, 2011). There are extensive and abundant populations of white-clawed crayfish in some parts of the Nidd catchment, but there are at least three known populations of signal crayfish invading watercourses (Penn, 2010). In addition, there are many gravel pits and ponds, which may be either opportunities (as existing or potential ark sites) or threats (as further sources of non-native crayfish and crayfish plague).

Some catchments are already dominated by non-native crayfish with very few populations of white-clawed crayfish remaining, or none and this is the future trend for most catchments where non-native crayfish have established. In the 1970s white-clawed crayfish were present in most tributaries of the River Thames except in polluted stretches. There have been major improvements in water quality in the Thames catchment since the 1970s, but the spread of non-native crayfish has led to the population of white-clawed crayfish becoming fragmented. Relict populations still remain in some of the tributaries of the Rivers Cherwell, Evenlode and Windrush, plus a tributary of the upper Thame (Ellis, 2009; Scholley, 2011, pers.

comm.), but as invasion of the catchment proceeds it is likely to be difficult to retain those populations *in situ*. The few populations in still waters may be more likely to survive, but may still potentially be under threat, due to the risks of accidental transmission of crayfish plague, or possible accidental or deliberate introductions of non-native crayfish. The latter may be a risk because the Thames catchment has the most wild harvesting activity, 64% (436) of the trapping consents granted by the Environment Agency in England and Wales in 2009 (Sadler, 2010, pers. comm.).

In catchments dominated by non-native crayfish, the recommended actions in the toolkit include maintaining good biosecurity for the ark sites, and utilising white-clawed crayfish populations under imminent threat of loss as donor stock for new ark sites in the catchment (if they are sufficiently isolated and have good biosecurity), or elsewhere in the River Basin District. Public-awareness campaigns would still important, but would be focused on preventing spread of non-native crayfish or crayfish plague to unaffected areas and on education and enforcement of trapping regulations, where applicable. In SW England an action plan for white-clawed crayfish is already being implemented, which involves setting up many ark sites for white-clawed crayfish to retain the species in the region (Sibley, 2006, Kindemba and Whitehouse, 2009, Robbins,2011, Sibley et al., 2011) and active efforts on public awareness-raising to minimize the risk of non-native crayfish or crayfish plague being spread to areas that do not have them (Rees, 2011).

9.3.5 The role of barriers

Substantial physical barriers do not affect established populations of non-native crayfish directly, but can either prevent further invasion, or at least significantly delay it. This is only likely to be feasible for preventing invasion upstream as crayfish can spread downstream over barriers (Peay and Rogers, 1999, Light, 2003, Kerby et al., 2005). Barriers do not prevent impact in areas that have already been invaded downstream, nor do they prevent ongoing expansion in other parts of the catchment. Nonetheless, barriers do have a significant role in conservation action for white-clawed crayfish in protecting semi-isolated populations from crayfish plague in cases where an outbreak occurs downstream. They may also have a role in protecting at least some parts of catchments from the other ecological impacts of high-density populations of signal crayfish. Identification and assessment of barriers from aerial photographs and site visits is an activity that is already being carried out in some catchments in England, e.g. in

Sheffield (Dangerfield, 2011), the Eden catchment in Cumbria (Backshall, 2011), and in the Nidd catchment in Yorkshire (Slingsby, 2011, pers. comm.). With appropriate training, surveys of existing barriers and the preliminary desk-studies of possible new ark sites can be done by local volunteers and students even if they have little or no experience of surveying crayfish.

The barriers posed by dams, weirs and culverts are artificial modifications of watercourses and are counted as adverse features under the Water Framework Directive, especially because of their effects on access to the upper parts of catchments to migratory fish such as Atlantic salmon, sea-trout and lamprey. Old industrial weirs with sloping faces are likely to pose only a temporary barrier to upstream invasion by signal crayfish, because crayfish can readily climb rough or algae-covered surfaces, in the slow-flowing margins or even on wet rocks at the edges. If those are removed to facilitate movement of fish, it will not affect the outcome of the invasion by non-native crayfish. In the short term, however, even partial barriers may have a value in preventing transmission of crayfish plague upstream by crayfish to crayfish contact. Partial barriers may be sufficient to ensure at least part of a population of native crayfish survives an outbreak of crayfish plague long enough for action to be taken to conserve the remaining population, as was the case in the Porter Brook in Sheffield and the Ribble catchment. Where a physical barrier includes one large vertical face, preferably with an overhanging lip, or else a series of vertical steps, the natural or artificial barrier has a much greater likelihood of blocking invasion upstream.

To date, no wholly new barriers have been created in watercourses in England to protect sub-catchments from invasion by non-native crayfish. Modification of existing barriers has been carried out in Sheffield to Porter Brook (Dangerfield, 2011). A series of three small weirs was installed in a headwater stream in the Ribble catchment, to prevent white-clawed crayfish moving over a natural barrier into an area affected by crayfish plague, and so allow time for the infection to die out completely (Bradley, 2010). A barrier composed of two new weirs has also recently been installed in a tributary of the River Clyde in Scotland with the aim of prevent signal crayfish from invading up a small stream onto a wet plateau, from which there may be a risk that crayfish could cross the watershed and invade the adjacent catchment (Bean 2010, pers. comm.). In the case-study in Scotland the rationale for the barrier was protection of the spawning areas of

Atlantic salmon and brown trout *Salmo trutta* (Linnaeus 1758), plus populations of lamprey species *Lampetra* spp. and freshwater pearl mussel *Margaritifera margaritifera* (Linnaeus 1758), juveniles of which may be potentially vulnerable to predation high-density populations of signal crayfish.

Where prevention has not stopped the introduction of non-native crayfish into a river catchment, the presence of natural or artificial barriers will be critical in determining how much of the catchment can be invaded by the non-native crayfish and how much of the range of any native crayfish in that catchment can be retained in the long term. Physical isolation from non-native crayfish is likely to be a fundamental requirement in any conservation strategy for native crayfish in Britain or mainland Europe. The approach of ark sites as isolated areas for native crayfish secure from non-native crayfish or crayfish plague was proposed by Holdich et al. (2004) and has been discussed as a pragmatic response to conservation management among crayfish scientists and managers in Europe (Taugbøl and Peay, 2004, Holdich et al., 2009, Souty-Grosset and Reynolds, 2009).

Being part of an island, England and Wales have many small, short watercourses going directly to the sea. Some of these have white-clawed crayfish and are existing ark sites for white-clawed crayfish. Nonetheless, even relatively small catchments, or isolated individual sites may be at risk from human-assisted introduction of non-native crayfish and/or crayfish plague, so the status of white-clawed crayfish, non-native crayfish and crayfish plague in adjacent areas may be significant. Policy on the management of non-native crayfish in one area, especially regarding wild harvesting, has the potential to affect management in adjacent areas.

9.3.6 The issue of crayfish trapping

In England, most of the original aquaculture enterprises of the 1970s and 1980s were not commercially successful and there was little interest in crayfish as food. Since then the growing popularity of cooking programmes on television and the greater variety of foods readily available has encouraged many people to try new foods. The combination of celebrity chefs promoting wild foods and publicity about invasive non-native crayfish and the threat to native crayfish has raised the profile of crayfish generally. It has helped to generate some public interest in harvesting of signal crayfish as a wild food. In 2009 there were 756 applications submitted to the Environment Agency for consent to trap crayfish, 83% of which were granted (Sadler, 2010) and although this included some applications for surveys, most of the applications were for wild harvesting. The number of people trapping crayfish illegally is not known.

Harvesting is popularly assumed to provide a degree of control of the crayfish population and so have some benefits, either in helping native crayfish, or as recreation and food from an unwanted species. Whether any wild-harvesting/control operation, even a very intensive one, can achieve any benefits for nature conservation is unclear. There are certainly reports from anglers that intensive trapping can reduce the nuisance of signal crayfish taking angling bait (Peay and Hiley, 2004), but there is little evidence so far of reductions of ecological impact due to trapping projects in Britain. For example, there has been continuous daily trapping and removal of signal crayfish from the invaded stretch of the River Clyde for more than 8 years in an attempt to control the population (Reeves, 2004, Mitchell, 2006, pers. comm.; Gladman, 2009, pers. comm.), yet signal crayfish continued to spread. Crawford et al. (2006) found that signal crayfish altered the macroinvertebrate community in the River Clyde compared to uninvaded areas; reducing the species diversity and the abundance of invertebrates, but they did not find any significant relationship with abundance of crayfish, despite sampling in the area where the control by trapping was being carried out. The implication is that whilst the trapping reduced the catch per unit effort compared to the initial trap catches, it did not have any significant benefit for the macroinvertebrate community.

Large scale, intensive trapping was trialled in Loch Ken (Ribben and Graham, 2009). This involved five months effort with around 450 traps per day. Over 719,000 signal crayfish were removed, weighing approximately 20 t, in what is probably the largest controlled removal of crayfish from a waterbody in a single season in the UK. There was a reduction in the catch per unit effort in male crayfish, in the largest size classes (60-64 mm carapace length (CL) and above), but little effect on the catch per unit effort for female crayfish. Furthermore, monitoring by trapping after the intensive removal showed no significant difference overall in the catch per unit effort in heavily trapped areas compared to control zones, despite some depletion of the largest male crayfish. Although intended for control, the programme may represent a sustainable harvest from the loch.

Nonetheless, wild-harvesting/control does have its proponents in England (Stancliffe-Vaughan, 2007, Rogers and Watson, 2011) and a recent report (West, 2011) described an apparent reduction in the rate of erosion of the banks by burrowing crayfish in the River Lark, Suffolk. West (2011) also reported the on-going immigration of crayfish into the control zone from areas outside. This immigration into trapped areas was also found in an experimental trapping study by Moorhouse and Macdonald (2011) in Oxfordshire and a mark-recapture exercise during the Loch Ken study (Ribben and Graham, 2009) also showed significant movement of some crayfish, with distances of 400-800m travelled within 14 days.

An opposing view on trapping, which is common among the conservation agencies in Britain, is that wild harvesting increases the risk of transmission of crayfish plague and the risk of accidental or deliberate introductions of signal crayfish to new sites or new catchments, because of the incentive of future wild harvest. Hence, opponents of wild harvesting of non-native crayfish in Britain consider that trapping should be discouraged because of the risks they consider it poses to remaining populations of white-clawed crayfish from crayfish plague, introductions and escapes of crayfish. Several groups have lobbied the UK government (Defra) for a ban on the sale of any live non-native crayfish, including the Wildlife Trusts, Buglife and the Salmon and Trout Association (2009), and the UK BAP Steering Group for White-clawed crayfish.

Supporters of trapping argue that trapping responsibly is not a threat and should not be restricted just because illegal activity also occurs. Opponents refer to examples in various parts of Europe where harvesting has been authorised in one area and not in others and this has been followed by further stocking and harvesting progressively beyond the designated areas, leading to expansion of range of non-native crayfish, for example in Spain (Alonso et al., 2000) and Sweden (Edsman, 2004). There are examples from Britain too. In the early 1990s, a landowner in Yorkshire introduced signal crayfish into his former gravel pit near the River Ure (Peay, 2001). Once the population developed in the gravel pit, the landowner said he gave batches of signal crayfish to other local landowners for their own ponds. No permitted stocking of crayfish was allowed in Yorkshire at the time and no wild harvesting. In Scotland, there have been illegal introductions of crayfish into several catchments, despite there being no harvesting of crayfish permitted anywhere in the country (Bean et al., 2006, Gladman et al., 2009).

The IUCN guidance on non-native species (IUCN, 2000) takes the position that simply reducing the abundance of a population of an invasive non-native species is not sufficient to justify the control action. To be worthwhile in terms of cost and effort, it should have demonstrable benefits, such as a significant reduction in the ecological impacts of the non-native species. In the case of invasive crayfish, when a control method does not prevent further expansion of range, the population of non-native crayfish will continue to expand until it fills all accessible parts of a catchment. Any areas where there was reduced density would be continuously re-colonized from adjacent areas, as shown by Moorhouse and MacDonald (2010), and reduced competition in those areas may allow compensatory growth in the population. There is insufficient evidence from field studies so far about compensatory growth, however, small egg-bearing signal crayfish have been found, e.g. 25 mm CL (carapace length) in size (Peay, unpublished), which is below the effective lower size limit of traps. For example, in the Loch Ken project (Ribbens and Graham, 2009) less than 9% of the crayfish caught were below 35 mm CL in size. This suggests that recruitment could still continue during intensive trapping, even if trapping had been effective in reducing the abundance of female crayfish in larger size classes.

The debate on trapping for wild harvest/control is likely to continue in Britain for some time, although consented trapping seems likely to persist, at least in areas of southern England where it has been allowed previously. Trapping crayfish is mainly a small-scale recreational activity (there are very few commercial trappers). It has also been used, albeit with uncertain success, as a localized coping measure for some invaded waterbodies. On the issue of use of trapping as a measure to mitigate the environmental impacts of established populations of signal crayfish, there is not enough evidence about the benefits to consider whether the on-going cost of an intensive trapping could be justified, even in a limited area. The author considers that trapping does not contribute to the conservation of white-clawed crayfish, because it does not prevent further spread, nor reduce the risk of crayfish plague, nor allow white-clawed crayfish to re-occupy lost range. Hence, trapping and removal of signal crayfish is not likely to be a worthwhile component of any conservation action plan for white-clawed crayfish.

9.4 Discussion

The white-clawed crayfish is a protected species in Britain, but legal protection alone is not enough to ensure survival of the species, due to the extent of non-native crayfish already present. Whilst no loss of range of the species is a goal of the UK Biodiversity Action Plan, it is very unlikely to be achievable and further loss of range continues each year. Yet initiatives that can be taken by statutory agencies, conservation groups and interested local people can protect at least some of the existing populations, start new, potentially ‘defendable’ ark sites, and provide alerts about new populations of non-native crayfish, or about activities that increase the likelihood of further introductions of non-native crayfish.

Working within individual catchments is recommended to give a local focus and encourages involvement by a wide range of local stakeholders. There is a need to get clear messages to all those potentially involved with crayfish and the general public. Success in maintaining existing populations and re-stocking or starting selected new sites encourages participation in conservation efforts, making it easier to carry out additional work in future, even when there are some setbacks. There is also a need to coordinate action in the context of whole catchments and in wider regions or River Basin Districts. Management of white-clawed crayfish cannot be considered separately from the issues of non-native crayfish and crayfish plague and this emphasises the importance of regulators and other agencies in coordinating management decisions, for example on control/harvest of non-native crayfish and on the stocking of fish within ark sites for white-clawed crayfish. Losses of white-clawed crayfish and the increasing extent and impacts of non-native crayfish are ecological changes in the aquatic ecosystems, but it is the behaviour of people that has a major influence on where native and non-native crayfish will occur in future.

We are likely to need many existing and new isolated sites in order to be able to conserve white-clawed crayfish. The time available in which to act is limited, although rates of upstream invasion by non-native crayfish are relatively slow, provided there is no human assistance in moving them. With the growing interest in crayfish in general and action for conservation of the native crayfish in particular, there seems to be a good chance of keeping white-clawed crayfish in England and Wales for decades to come. Conservation strategy needs to operate at a range of scales, from international scale, with respect to regulation within Europe and on the introduction of new potentially invasive species; to national and regional policies on protection of native crayfish and management of non-native crayfish; to catchment-scale and more local areas, to benefit native crayfish population, where a range of local groups are involved in planning and implementing projects.

The challenges in conserving the white-clawed crayfish in England and Wales will be shared by those trying to conserve other native crayfish species in Europe and other parts of the world, as anthropogenic influences on aquatic environments increase and so do the impacts of invasive non-native species. In this case, we have a species whose formerly widespread natural distribution is now fragmented and shrinking fast. We cannot undo the damage wrought by the introduction of the non-native species, but by forecasting and contingency planning now, before we reach the worst case, we can aim to keep core areas and many individual populations of white-clawed crayfish. As with the re-positioning of coastal defences in England to mitigate the impact of sea level rise on coastal habitats, so conservation strategy for native crayfish could be considered to be a case of ‘managed retreat’.

Surveyors catching white-clawed crayfish at a donor site in Yorkshire, UK threatened by signal crayfish (left); the receptor site in a stream nearby in the same catchment, re-stocked with white-clawed crayfish from the donor site after an outbreak of crayfish plague (right). This re-introduction forms part of a catchment strategy for conservation of white-clawed crayfish (see Chapter 9).

Chapter 10
General Discussion

10.1 Crayfish in Britain – a historic context

One of the first scientists to investigate crayfish in any detail was T. H. Huxley, a leader in the development of the study of zoology and a staunch supporter of the work of Charles Darwin. He considered the white-clawed crayfish *Austropotamobius pallipes* was an ideal animal for study and aimed "*to show how the careful study of one of the commonest and most insignificant of animals, lead us, step by step, from everyday knowledge to the widest generalizations and the most difficult problems of zoology; and indeed of biological science in general*" (Huxley, 1880). Huxley would never have considered that such a common species would have started on a path of critical decline only a hundred years after the publication of his seminal monograph. Whilst Huxley looked at anatomy and physiology, as this study shows, crayfish have a contemporary role in the study of ecology, invasion biology and conservation biology. It shows how an invasive keystone species can replace a similar indigenous species and modify aquatic ecosystems. Furthermore, as seen in this study, the case of biological invasion by non-indigenous crayfish illustrates the technical and human problems involved in management of an invasive species and the difficulties in management to conserve a threatened species, when the threat cannot be removed; only, at best, avoided.

In Huxley's time it was the pollution of rivers that was the environmental issue of growing concern, and indeed, as the rivers in the industrial areas of England were grossly polluted, they would have been unable to support white-clawed crayfish. Following work of Snow on the epidemiology of a cholera outbreak in London (Snow, 1857) and the 'Great Stink' of the River Thames in 1858, legislative measures were introduced into the UK to tackle the problem, with the Rivers Pollution Act 1876. Another development in environmental protection in the UK was the passing of the Control of Pollution Act 1974, which set new requirements for the control of waste and the prevention of pollution of air, land and water. It was an exemplar in its time of national legislation on the topic of pollution and marked a turning point in the quality of rivers in England. Water quality improved from the 1970s onwards (Johnstone and Horan, 1996) and has continued to improve, with approximately 80% of rivers in England classed as good quality by 2009, from 62% in 1990 (Department for Environment Food and Rural Affairs, 2010). Hence, indigenous crayfish should have been able to recover much of their range lost to historic pollution, in the absence of confounding factors.

The legislation on pollution in 1974 followed soon after the first global conference on the environment, the United Nations Conference on the Human Environment in Stockholm 1972. The Stockholm conference produced a Declaration, with 26 guiding principles, which introduced the ideal of sustainable development. It is worth highlighting three of these fundamental principles, as this international agreement was the starting point for the subsequent international and national law on the environment:

"*Principle 2: the natural resources of the earth including the air, water, land, flora and fauna and especially representative samples of natural ecosystems must be safeguarded for the benefit of present and future generations through careful planning or management as appropriate.*

Principle 3: the capacity of the earth to produce vital renewable resources must be maintained and, wherever practicable, restored or improved.

Principle 4: Man has a special responsibility to safeguard and wisely manage the heritage of wildlife and its habitat which are now gravely imperilled by a combination of adverse factors. Nature conservation including wildlife must therefore receive importance in planning for economic development."

These principles provide an international basis for conservation of biodiversity at global and national scales. In the UK, a contentious political process during the late 1970s led to the Wildlife and Countryside Act 1981, which is still the legal basis for protection of species and habitats for nature conservation and which, eventually, provided some legal protection for white-clawed crayfish, when it was added to Schedule 5 of the Wildlife and Countryside Act in 1986.

It is ironic, therefore, that in the early 1970s, around the same time as the Stockholm conference and the Control of Pollution Act set a framework that would help protect rivers and wildlife such as the white-clawed crayfish, the decision was made by MAFF (Ministry of Agriculture, Fisheries and Food) to allow the start of aquaculture of signal crayfish *Pacifastacus leniusculus* (Holdich and Reeve, 1991), which was the launch of what would become an unstoppable biological invasion. Indeed, such human-instigated invasion has been likened to biological pollution (Homans and Smith, 2013).

The decision to allow the introduction of invasive crayfish is galling, because it was based on the false assumption that white-clawed crayfish in Great Britain had already been lost to crayfish plague. It can be difficult to predict which species are likely to become invasive if introduced into a new region, but with crayfish there was plenty of evidence available on the behaviour of non-indigenous crayfish as existing invaders to allow the risk assessment (undertaken in Chapter 2) to be carried out. Enough information was potentially available even in the 1970s, if risk assessment methodology had been available and used.

10.2 The risks of invasive crayfish

The study in Chapter 2 served as a useful test of the FiSK assessment methodology on species other than fish, for which it was originally developed (Copp et al., 2009). This study predicted the further spread of signal crayfish within existing invaded catchments and future competition from more recent invaders, the spiny cheek crayfish *Orconectes limosus* and virile crayfish *Orconectes virilis*, albeit in catchments where signal crayfish is already present. Red swamp crayfish *Procambarus clarkii* still has only limited extent in the London area, but the main concern with this species is the potential for it to invade wetlands that are not currently occupied by any crayfish species. This species may become more invasive over time as it is likely to be favoured if warmer summers occur due to climate change.

Since the completion of the risk assessments on crayfish in 2008 for this study, two more crayfish species have undergone risk assessment for the UK, the marble crayfish *Procambarus fallax f. virginialis* and virile crayfish (Non Native Species Secretariat, 2013a) and many other species risk assessments have been added, now totally nearly 60 completed assessments. In addition, the application of the FiSK methodology has subsequently been extended to a larger suite of 37 crayfish species, including those known to be in the aquarium trade in parts of mainland Europe (Tricarico et al., 2010). The later work also ranked signal crayfish and red swamp crayfish as the species of greatest risk, with spiny cheek crayfish as another species with high risks, albeit not as great as the other two species. The impacts of signal crayfish on the ecosystems into which it has been introduced, have been well studied, particularly the transmission of crayfish plague; its aggressive behaviour, which facilitates its competition with indigenous crayfish, and its impacts in reducing abundance of aquatic macrophytes and the abundance and diversity of other aquatic invertebrates. One aspect that had not been well studied prior to this work is the impact on fish, which is addressed in Chapter 3. There is a need to investigate the potential impacts of other crayfish species that the study identifies as having future risks. For example, much of the work on red swamp crayfish has been in Mediterranean countries. Less work has been done on populations of red swamp crayfish in the cooler northern Europe, although growth and reproduction have recently been investigated in the Netherland (Chucholl, 2011). The species is increasingly frequent in France and the Netherlands and may become so in England. Of particular concern is the potential for red swamp crayfish to move over land and potentially access new sites that way. More information on the factors that influence terrestrial movement in red swamp crayfish would be helpful.

10.3 The impacts of invasive crayfish

One of the impacts identified is the potential for invasive crayfish to have impacts on the recruitment of fish species and this was investigated in the field study in Chapter 3. A finding of this study was that signal crayfish are capable of reaching very high abundance, even in a relatively low nutrient, high energy headwater stream. This type of stream is typical of the spawning sites for brown trout and salmon. The study showed a negative correlation between increasing abundance of crayfish as invasion progresses and the reduction of recruitment of brown trout *Salmo trutta*. It also showed the decline in bullhead *Cottus gobio*, which was also reported by Guan and Wiles (1997), but this time from an upland stream, rather than the productive lowland river studied by Guan.

Prior to this study, there was only limited interest among angling groups about invasive crayfish. The work of Guan and Wiles (1997) provided evidence for the impact of signal crayfish on benthic fish, yet benthic fish were of little interest to anglers. From the perspective of many anglers, the decline of white-clawed crayfish was unfortunate, but not relevant to recreational angling; fish were predators of crayfish and there was little risk of catching crayfish if fly-fishing. Hence biosecurity to prevent crayfish plague and prevent further introductions was of little concern (Peay and Hiley, 2004), except with respect to avoiding nuisance to coarse angling.

This study showed that there was potential for an adverse effect of signal crayfish on juvenile salmonid fish and hence on the interests of angling for salmon and trout. Even prior to the publication of a paper from Chapter 3, the Salmon and Trout Association was aware of the research and became concerned about this new issue, so much so that it joined with Buglife to lobby Defra on the need to regulate signal crayfish more effectively (Salmon and Trout Assocation and Buglife, 2009). Having met with local Salmon and Trout Association members to talk about invasive crayfish early in 2009, it was partly in response to them that the paper was published in the open-access journal Knowledge and Management of Aquatic Ecosystems, as it allowed people who normally have little access to papers in journals the opportunity to see the study in full. It was among the top ten most downloaded papers in the journal in 2011 and 2012.

Following this study, other researchers have investigated the issue of predation of salmon eggs; which showed that signal crayfish did not readily find buried eggs (Gladman et al., 2012), but the crayfish consumed eggs at the surface (Findlay, 2013), and once buried salmon eggs hatched, the alevins were predated as they emerged (Edmonds et al., 2011). Findlay (2013) carried out a field survey of fish and crayfish, similar to the study in Chapter 3, on several tributaries streams in the upper Tees catchment. He also found a strong negative relationship between the density of signal crayfish and that of brown trout and bullhead, which provides additional support for the conclusions here. Another point of interest in that study was the finding that even small crayfish, one year old (1+, 10-16 mm CL size), reduced the survival of buried trout eggs by a combination of predation and the fine sediment produced by crayfish. This is relevant because crayfish of that size are not generally caught in traps and small crayfish are much more abundant than the larger ones. Usio et al. (2009) found that although large crayfish (> 30 mm CL) caused the rapid loss of most of the macrophytes within enclosures, small crayfish (< 30 mm CL) had as much impact, but took about three weeks longer to have the same impact as the larger crayfish.

Although this study and other studies have shown impacts of crayfish at high abundance on the recruitment of fish, there is still some uncertainty as to the range of habitats in which impacts occur. Are the effects less in larger rivers, where consumption of juvenile crayfish by predatory fish may go some way to mitigating the impacts? If so, are there habitat-related critical densities of crayfish that could be used to identify waterbodies where the magnitude of impact is likely to be significant? Indeed, using methods of environmental impact assessment, thresholds for significance of effects would need to be defined. Furthermore, do such effects persist over time? The effects of invasive crayfish in different habitats and their relationships with fish could be the subject for other research projects.

Variations in populations over time are more difficult to assess. Matched surveys of fish and crayfish are rarely carried out. There are programmes of fisheries monitoring in most catchments; in England and Wales these are generally carried out by the Environment Agency and Rivers Trusts, but there is little long term monitoring of crayfish, even of indigenous crayfish (Peay 2003b). Limited resources mean that there is unlikely to be any great increase in monitoring of crayfish in future. One of the problems of temporal study is the range of year to year variation due to climatic factors, which is particularly marked in salmonid fish. Any trend in fish population in watercourses invaded by signal crayfish needs to be identified from the background of annual variation and this can only be obtained in the context of comparable surveys in uninvaded watercourses. Factors such as stocking with reared fish or wild harvesting of crayfish would potentially mask trends, so any such studies would need to be carried out in catchments without these factors and allowance made for any other confounding factors such as pollution incidents.

Since the work presented in this study (Chapter 3) several more years of matched fisheries and crayfish survey have been carried out in the same stream and further work is planned to analyse fisheries data across a range of sites in the Ribble catchment. This will potentially provide a baseline for other future studies on the relationship between fish and crayfish in the catchment. Other catchments that may offer scope for future work on this topic include the Clyde, Tees and Wharfe.

10.4 Management of invasive crayfish – biocide treatment

Part 2 of this study focused on methods to eradicate or control signal crayfish. It has been recognised from the start that eradication treatments for invasive crayfish are only likely to be used on populations that are still localized. Use of biocide treatment was investigated in Chapter 4 and followed on from previous studies, namely a preliminary experiment with different biocides (Hiley and Peay, 2006) and a field trial in the North Esk catchment (Peay et al., 2006a).

This study (Chapter 4) included additional cases and investigated the outcome of the work undertaken. The study shows that populations of crayfish can be eradicated successfully from small sites using natural pyrethrum, but a critical factor is the extent of the treatment. If the full extent of the population is not treated, there is scope for survival of some individuals, leading to regrowth of the population. Careful assessment of habitat is required to identify areas that may harbour crayfish - and how best to ensure all are adequately exposed to treatment. With only six sites in Britain having proceeded to treatment to date, the technique is still in development and there is the opportunity to learn from these cases to improve the prospects for successful treatment in future. Treatment did not achieve eradication in three cases, but the study showed some of the problems such as marginal vegetation, seepages, reductions in water level and deep water and how these problems could be overcome in some cases in future projects. Most treatments will be in small still water sites, because introductions are often made into fish ponds and because large sites would be prohibitively expensive, difficult and uncertain of outcome. There are likely to be cases when a pond can be treated but there is also a need to treat a length of ditch or other small watercourse downstream of the pond. In one of the cases in the study a treatment was carried out progressively in carefully controlled sections of a small stream. Although this increased the difficulty of the project and the time required, it was done successfully without polluting areas outside the zone designated for treatment. Use of this method potentially increases the range of sites that could be tackled successfully in future.

The study showed the constraints encountered in each project, which were common to projects that were treated, as well as those that did not proceed to full treatment. Factors that contributed to success were having a strong project manager, a ‘project champion’ with the determination to make the project succeed, adequate resources of staff and funding being made available rapidly, as soon as the decision was made to proceed. Ideally, a rapid response plan would include funds that could be called on at short notice if a future project met pre-determined criteria. This study suggested how to prioritise projects, with the protection of a whole catchment taking the highest priority, together with the first arrival of new invasive crayfish species, such as marbled crayfish. Northern Ireland has no known invasive crayfish at present and because as an island, Ireland (Northern Ireland (UK) and the Republic of Ireland) is isolated, it has a key role in the conservation of white-clawed crayfish in a European context. It must be given the highest priority for prevention and for incisive and well-resourced action if the worst happens and prevention fails.

This study showed that another major factor was landowner cooperation and at least one project of high priority and technical feasibility (at least at the stage at which it was first mooted) did not proceed to treatment because one or more landowners did not cooperate. Partial cooperation was a source of delay or difficulty in some projects, even when treatment was permitted, but various constraints were imposed on timing or actions. Clearly, consensus is highly desirable and some projects have had the cooperation of owners and, indeed, in some cases active support and contributions in kind. New legislation in Scotland provides additional powers to secure cooperation to carry out eradication treatments, but similar provision to deal with invasive non-native species is not yet in place in the other countries of the United Kingdom. It may come in future, due to a forthcoming European Union Directive on invasive non-indigenous species (Non Native Species Secretariat, 2013b), but EU Directives take years to implement in national regulation and it is highly desirable that regulation is developed in other parts of the UK much sooner. Landowner cooperation is likely to be a recurrent issue with eradication campaigns for any species in the meantime.

10.5 Implications of crayfish out of water

Chapter 5 addressed a problem that arose from the biocide treatments, namely the potential for signal crayfish to remain out of water for a period if the level was reduced prior to a biocide treatment and hence avoid the treatment. The experiments in tanks and in a field trial showed that crayfish generally vacated burrows on the first or second night after exposure. The experiments in tanks showed that at high density aggression from dominant crayfish led to some crayfish remaining out of water, rather than compete with higher ranking crayfish for submerged refuges. In field conditions, there was also a response to climatic factors, in a trade-off between vacating burrows as soon as possible to find submerged habitat, or waiting for damp conditions that would reduce the risk of dehydration. In both tanks and field conditions, more than half the crayfish climbed onto dry areas in search of alternative refuges. In the field trial, with access to water partly constrained by a barrier and trap, live crayfish were able to survive in exposed burrows for at least 6 nights (0.5%) and observations from one of the projects considered for biocide treatment suggests that in autumn persistence out of water may be longer. By contrast, the successful biocide treatment of a farm reservoir showed that, in summer, if the burrows dried out sufficiently, they were all vacated by crayfish. In that case, the crayfish were in submerged habitats and exposed to biocide treatment.

There would be a benefit from further investigation of the behaviour of signal crayfish out of water. Future work could investigate the effects of soil moisture and humidity on the time for signal crayfish to emerge from burrows in mesocosm and field tests. Work could also consider how far over land signal crayfish can and do walk and how readily they leave water to seek more favourable conditions. Such work would have several management purposes. Firstly, if biocide treatment is being considered and partial dewatering is being considered as a pre-treatment to reduce costs or facilitate application, how long in advance should dewatering be carried out and how dry does the exposed area have to be to give confidence that crayfish have vacated exposed refuges? Secondly, if a pond is fully dewatered for purposes of management, what is the risk that crayfish will leave the site in search of alternatives? If there is a significant risk of crayfish colonizing uninvaded areas by this means, then mitigation by the installation of barrier fencing may be required. Finally, in running water, how readily will signal crayfish climb out of water in order to circumvent an in-stream barrier? This latter question is important to the conservation of white-clawed crayfish that may survive in ark sites upstream of natural or manmade barriers.

10.6 Shock tactics

In the biocide treatments (Chapter 4) an important issue was avoiding impacts outside the area targeted for treatment. Although managed successfully in this study, where there is a need to treat a length of stream beyond a pond, the treatment would be made easier and potentially cheaper (due to shorter time required for treatment and full recovery), if a method could be found that both killed invasive crayfish effectively and had no potential for impact outside the treated area. In this study (Chapter 6) a new method was trialled using specially developed equipment for high power electric shock treatment, to deliver repeated high power shocks to a watercourse to kill crayfish. Within the channel, high intensity treatment was highly effective, 86-97% mortality, but did not achieve complete mortality. The limitation on the treatment was the difficulty of treating crayfish within the banks. When crayfish were in deep refuges in the banks, or beneath large rocks, some of them survived. Hence, the method did not achieve complete eradication of the population and in its current form can only be considered to be a control measure. Nonetheless, even with its current limitations, there is potential for localised use of the method, for example as a periodic measure to prevent crayfish abundance increasing to a high density, which might increase the risk of them overcoming a physical barrier to upstream invasion. The risk of incomplete treatment would be highest in streams with complex habitat in the banks and a high proportion of large rocks in the channel, but in small streams or ditches other substrates, the treatment would be more effective, even in its current form.

The treatment of banks is the main uncertainty at present and there is some potential for further improvement with modification of the methods, although this would require further trials. To date there has been no funding to develop the equipment or the method, other than a grant provided towards the cost of data collection in this study. A future field trial to develop the method on an invaded stream is being considered in Scotland, subject to funding. In addition, there is a lack of information about the shock treatment required to produce complete mortality rapidly and some physiological research on this would be required.

10.7 The cost of management action or inaction

All attempts to eradicate or control invasive crayfish, or indeed any other invasive species, require funding for labour, material and equipment. Difficult decisions may need to be made about what is funded, according the benefits to be gained, the resources required and the likelihood of success. In this study (Chapter 7) a cost model was developed as a potential tool to aid such decision-making about invasive crayfish, and whether to carry out a biocide treatment or not. It adopted a new approach in using a simple model of invasion of watercourses in a catchment and applying an annual environmental cost. The environmental cost was applied using the cost of re-stocking invaded reaches with reared salmonid fish. Whilst not recommended as an actual response to the impacts of signal crayfish on fish (described in Chapter 3), an advantage of the use of this surrogate cost is that restocking is familiar to resource managers as a management tool and it is not dependent on subjective value, which is a problem with contingent valuation and Willingness to Pay. As used in this study, the environmental cost is conservative, in that it is only based on two species, but the cost factors applied could readily be extended to include additional cost factors related to impacts on biodiversity, habitat or other ecosystem services.

As shown in this study, the cost of an eradication treatment in the early stage of invasion is soon balanced by the accumulation of annual environmental costs if no treatment is carried out. Indeed, even the annual environmental costs may increase to and exceed the total cost of an initial eradication treatment in a large catchment. Using the example of a catchment in which the costs of biocide treatment are known, the cost model showed how delay in treatment, and hence increase in the extent of treatment required, would increase costs. The cost of a delayed treatment could still be justified, however, because the lag time until treatment would bring it closer to the period of accelerating environmental costs and hence high benefits from avoiding those costs. Even so, because of the greater uncertainty about complete success of treatment in larger and more complex areas (Chapter 4), this study strongly recommends that the resources required for an eradication treatment are provided soon after detection, to give the best likelihood of success.

The study showed the environmental cost of invasion if eradication was not carried out and the even higher annual cost of control, in this case using trapping. It was evident that whilst control might be affordable in the initial stages of invasion, the cost would soon become prohibitive, because of the need to continuously increase the length of watercourse treated in line with the invasion until the catchment was fully invaded. The constraint was the rate of invasion in the presence of the control measure. Little work has been done on this, but there does not appear to be any evidence to indicate that physical removal of crayfish has any effect on reducing the rate of invasion, let alone preventing it.

An exception to this demonstration of poor value of management for control would be the provision of effective physical barriers to invasion by signal crayfish. In that case, the cost model would show the accumulating benefit of preventing further upstream invasion. Scottish Natural Heritage was able to use the cost model from this study as a tool in the decision to fund the first purpose-built barrier to invasion of signal crayfish in the UK. This barrier was a pair of dams, which were installed on a headwater stream of the River Clyde in south-west Scotland, with the aim of preventing upstream invasion of signal crayfish and the potential expansion across the wet plateau of the watershed into the catchment of the Nith (Scottish Natural Heritage, 2011). The project cost approximately £50,000, but compared to the long term environmental costs arising from invasion of the catchment of the River Nith, this was a good investment. Constructing two barriers close together means that even if one is overcome at some time in future, it would be a relatively simple procedure to divert flow around the section between the dams and use a biocide treatment to eradicate any crayfish between the two barriers.

10.8 How to find a refuge for white-clawed crayfish in England and Wales

Part 3 of this study dealt with conservation of white-clawed crayfish and the aim of finding secure areas for the species, away from invasive non-indigenous crayfish. Successive compilations of the data on the distribution of crayfish in the UK have shown the progressive expansion of invasive crayfish species and losses of white-clawed crayfish at each data compilation (Holdich and Reeve, 1991, Sibley et al., 2002, Holdich et al., 2009d, Rogers and Watson, 2011) until only a few catchments with white-clawed crayfish are still free from signal crayfish and all those remaining are highly vulnerable to future introductions. Rogers and Watson (2011) summarized the situation in England and Wales in 2010 as 81 sub-catchments with records of white-clawed crayfish only, 115 with both species and 275 with signal crayfish; i.e. 390 sub-catchments in total with signal crayfish, compared to 96 prior to 1990.

Because so many populations of signal crayfish are carrying crayfish plague and even if not they can consistently out-compete white-clawed crayfish, there is no possibility of recovering the historic range of white-clawed crayfish in England and Wales, nor even of maintaining the existing range. White-clawed crayfish are still locally abundant, but the distribution is becoming more fragmented over time. A pragmatic approach is needed – a managed retreat from the inexorable invasion by non-indigenous crayfish.

This study (Chapter 8) used a literature-based review and first-hand experience of white-clawed crayfish in England to assess the threats to white-clawed crayfish and the habitat requirements for the species. A series of selection criteria for ark sites were then developed. The assessment of the risks considered invasive crayfish, disease and other adverse factors at existing and potential sites. The criteria then addressed characteristics of the site and whether there was any need to improve the habitat. It is a two-stage process, the first to rule out obviously unsuitable sites, the second to assess a site in more detail and to compare and rank sites. Provided the water chemistry of a site is adequate for white-clawed crayfish, the barrier to invasion was recommended as the most important characteristic of an ark site, whether existing or proposed.

It will take time to build up a database of case studies of ark sites to see their effectiveness over time. Souty-Grosset and Reynolds (2009) compiled results from 59 reintroductions of white-clawed crayfish from across Europe (France, Ireland, UK, Spain, Italy and Austria), of which 26 (44%) were successful, the rest unsuccessful or unknown. In the UK, ark site introductions only started in 2006, so most projects are in their early stages. It is important that details of project methods are recorded to allow an evidence-based review in future, possibly in 2015.

The lead time for projects should not be under-estimated. Based on experience in Yorkshire, a typical programme might be to carry out a desk study in winter and a preliminary appraisal of sites early in the season. A promising site would have to undergo surveys for presence of crayfish, assessment of other fauna and flora and then go through the approvals process. In most cases, there is insufficient time to carry out an introduction in the same year, so introduction is more likely the following summer, hence, about 18 months if there is no funding requirement, i.e. no preparation is required on site and all work is done by volunteers. Projects make take two to three years to reach the stage of an introduction and at one quarry in Yorkshire, the use of a site as a bargaining piece in negotiations over a planning application meant it took five years before an introduction could be done.

An ark site project was started by Buglife in south west England in 2009. It screened 39 aggregates sites for suitability, using the criteria from this study, identified 15 sites with some potential, surveyed and assessed 5 and proceeded with 3 in 2011 (O'Neill and Whitehouse, 2011). Additional stocking may be required in one or two additional years if the founder stock is small and monitoring to determine outcome is likely to be over a period of 3 – 5 years or more. Projects are also underway in other parts of England. In Nottinghamshire, surveys and mapping of distribution of crayfish by volunteers has been carried out (Holdich and Jackson, 2011) and this is being followed by identification of potential ark sites (D. Holdich, 2013 pers. comm.). Hence, it appears that the selection criteria for ark sites in this study are being used by practitioners.

The desk studies and initial appraisal of sites are recommended as future student projects, although full implementation may on too long a time scale.

Further research is needed on the topic of barriers that can block upstream invasion of a watercourse by crayfish. The work on the Clyde has been mentioned above and there has been a barrier installed in Spain to try to isolate white-clawed crayfish from red swamp crayfish (Dana et al., 2011). Recently Frings et al. (2013) reported on the way signal crayfish were able to climb a sloping face of different surface roughness in a flume chamber, which provides some useful information on the ability of crayfish to walk against a flow. It indicates why sloping weirs are not an effective barrier against crayfish. Although they report that crayfish were able to overcome a barrier by an escape-flip action, it should be noted that the barrier being used was only 35 cm long, which is not much of an impediment. Further work is needed in the field to see what natural and artificial barriers have been overcome by signal crayfish in an upstream direction, taking into account the risk of human intervention.

The most substantial barriers to upstream invasion by non-indigenous crayfish are water supply reservoir dams. Although not forming a part of this study, a project was carried out for Yorkshire Water to assess the issues associated with crayfish in water supply reservoirs (URS, 2012a, b). While supporting the principle of conservation of a threatened native species, the company was concerned as to whether the presence of white-clawed crayfish would be a future constraint or additional cost on their operations. Reservoirs (111) were screened for their potential to support white-clawed crayfish and operational guidelines were developed to allay concern regarding crayfish and operations; hence there is now scope for ark sites in or upstream of reservoirs in Yorkshire in future. Work with volunteers on crayfish distribution and potential ark sites is underway in the Wharfe and Aire catchments in Yorkshire in 2013. The examples here show some of the future opportunities for ark sites and hence the need to use and, in due course, update the selection criteria developed in this study.

10.9 Putting it all together – conservation strategy

The ‘toolkit’ for developing catchment scale conservation strategy for white-clawed crayfish, which was prepared as part of this study (Chapter 9), was intended to be a practical manual to help users to prepare their own plans for conservation. It recommended a risk-based approach at catchment scale. Recommendations were based on following topics. 1. protection of white-clawed crayfish (through site protection, ark sites and monitoring); 2. management issues with non-native crayfish; including reducing the risks of introductions and crayfish plague due to fisheries management; regulation, enforcement and education, to try to prevent expansion of wild harvesting into areas where it risks white-clawed crayfish, and determining when eradication of non-native crayfish is feasible and appropriate; 3. Overall issues of increasing public awareness and promoting biosecurity. The topics were applicable to all catchments, but the priorities recommended for action plans varied, depending on the degree to which the catchment was invaded and the status of white-clawed crayfish.

In parallel with the ‘toolkit’, the Crayfish UK website was developed in association with Buglife with the support of the Environment Agency. Having two channels “Crayfish for everyone” and “Crayfish for professional” it offers both general information and more detailed guidance for practitioners. It has become a resource for ecologists and others involved with crayfish to access guidance, including the ‘toolkit’ and the selection criteria for ark sites (2011, 11,000 website ‘hits’; 2012, 35,000 hits).

To conclude, this study has ranged across a wide range of topics, from the behaviour of crayfish to strategic planning at catchment scale. It has all been to address questions raised by fellow practitioners about how best to conserve white-clawed crayfish in the UK and to encourage participation in crayfish conservation. Surely T.H. Huxley would approve of the interest and concern of so many people in this ‘*commonest and most insignificant of animals’.*

White-clawed crayfish – indigenous European crayfish now Endangered

Appendices: supplementary material on CD

Chapter 7: A cost model for decision-making on management of signal crayfish: to eradicate or not

Content: model worksheets with costs of biocide treatment and costed invasion model

File name: costmodel for crayfish management v2.xlsx

Chapter 8: Selection criteria for "ark sites" for white-clawed crayfish - a management tool

Content: spreadsheet-based decision-making tool to help users select ark sites for white-clawed crayfish.

File name: criteria for whiteclaw ark site v1a 09April2009.xls

Publication available from the Crayfish UK web-site, hosted by Buglife – The Invertebrate Conservation Trust. www.buglife.org.uk

Chapter 9: Developing conservation strategy for white-clawed crayfish at catchment scale in England and Wales – a way forward?

Content:

Peay S., Kindemba V., Attwood F. and Christmas M. (2011). A toolkit for developing catchment-scale conservation strategy for White-clawed crayfish. Version 1 October 2011
Buglife – The Invertebrate Conservation Trust, Peterborough.

ISBN 978-1-908657-00-8

Publication downloaded from the Crayfish UK web-site, hosted by Buglife – The Invertebrate Conservation Trust. www.buglife.org.uk [Accessed: 19th September 2013]

File name: BuglifeToolkitforCrayfishStrategy.pdf

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Signal crayfish Pacifastacus leniusculus