Alien invasive species in Irish water bodies

(2007-W-MS-2-S1)

STRIVE End of Project Report

Prepared for the Environmental Protection Agency

by

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ACKNOWLEDGEMENTS

This report is published as part of the Science, Technology, Research and Innovation for the Environment (STRIVE) Programme 2007–2013. The programme is financed by the Irish Government under the National Development Plan 2007–2013. It is administered on behalf of the Department of the Environment, Heritage and Local Government by the Environmental Protection Agency which has the statutory function of coordinating and promoting environmental research.

Firstly, we would like to thank Dr Alice Wemaere of the EPA. We also thank the Steering Committee: Prof. Steve Ormerod (Cardiff University, UK), Prof. Michael Usher (University of Stirling) and John Lucey (EPA) for guidance and constructive comments throughout the project. We would like to thank Paul McIlwaine for his contribution to the ecological research and also thank the staff and management of the various fisheries boards (now Inland Fisheries Ireland) who provided access to sampling equipment, manpower and necessary local knowledge.

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The EPA STRIVE Programme addresses the need for research in Ireland to inform policy makers and other stakeholders on a range of questions in relation to environmental protection. These reports are intended as contributions to the necessary debate on the protection of the environment.

EPA STRIVE PROGRAMME 2007–2013

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Executive Summary

While many non-native species have negligible negative effects, some cause significant economic and ecological impacts including reductions in biodiversity, decline of commercially important species and the alteration of ecosystems and ecosystem services. When these non-native species become established in existing ecosystems and threaten biodiversity and/or result in economic damage they are referred to as invasive alien species (IAS). IAS are regarded as the second most serious cause of biodiversity loss and environmental change worldwide, affecting freshwater ecosystems in particular due to their isolation and endemism. IAS have an impact on the ecosystem processes that are fundamental to human wellbeing including the wholesale loss or alteration of goods (e.g. fisheries) and services (e.g. clean and plentiful drinking water, culture and recreation).

Less diverse ecosystems, such as those that naturally occur on islands are particularly susceptible to invasion and damage following the introduction and establishment of IAS. Ireland, due to its glacial history and location on the western extreme of Europe is naturally depauperate in terms of its flora and fauna, and has repeatedly undergone invasion by a wide range of taxa, to the extent that many of its freshwater ecosystems are now dominated by IAS. The presence of these IAS in ecosystems can affect the ability of agencies or managers to maintain or improve ecological quality and halt degradation of ecosystem services. This has clear implications for the management of aquatic ecosystems and attainment of good ecological status under the Water Framework Directive (WFD).

As part of the STRIVE Programme 2007-2013, The EPA commissioned this project (Alien invasive species in Irish water bodies) with the aims of improving knowledge on the nature and extent of invasive alien species (IAS) and their impact on natural ecosystems; developing up to date national distribution maps showing the location of aquatic IAS in Ireland; and developing and trial control measures in the context of river basin management. This project has contributed to meeting these aims through a multidisciplinary, inter-institutional study, combining research, policy analysis and GIS database development.

A list of priority IAS of concern was compiled and distribution records collated to produce up to date distribution maps which can be viewed against a range of GIS layers in the National Invasive Species Database (NISD). Assessments of the coverage of the records for each species was made and for the majority of species, all known records are now included in the NISD. Opportunities to integrate IAS into WFD programmes were identified and a series of actions recommended including development of an alert list, inclusion of IAS in monitoring programmes and development of surveillance, recording and reporting protocols. A guidance note for contributors of IAS records was produced. The integration of the NISD with the Environmental Data Exchange Network (EDEN) will enable the delivery of distribution maps, identification of range expansions, species alerts and identification of waterbodies vulnerable to invasion on a River Basin District (RBD) level.

The impact of two IAS were investigated in two different RBD's namely, the ecological impacts of an invasive ecosystem engineer, *Lagarosiphon major* in Lough Corrib and the ecology of Ireland's most recent potential invasive fish, the chub (*Leuciscus cephalus*) in the River Inny. In Lough Corrib, macroinvertebrate community structure differed between invaded and native habitats with greater abundance and biomass in *Lagarosiphon* beds relative to that of the native *Chara* spp. The structure of the macroinvertebrate community also differed with increased abundance of invasive invertebrates, such as the zebra mussel in invaded habitats.

There were no such obvious effects of invasion on fish community structure or production, however, fish captured in invaded habitats differed in several key characteristics including size (roach), growth rate (perch), size at maturity (roach), instantaneous total mortality rates (roach and perch) and fish shape (roach and perch). Stable isotope analyses (SIA) revealed that *Lagarosiphon*, although representing the dominant primary producer in invaded habitats, made very little contribution to the food web of L. Corrib. Many consumers showed reduced isotopic variation in *Lagarosiphon*-dominated habitats, indicating that dietary variation may be reduced following invasion.

Chub were only present in very limited numbers during the study period and there was no evidence of any ecological impact of invasion. However, SIA revealed that the trophic ecology of chub in the River Inny overlaps with that of three important native fishes: eels, brown trout and Atlantic salmon and highlights the need for continued surveillance and control efforts for chub.

The efficacy of control measures was investigated and the use of jute matting to control *Lagarosiphon* and electrofishing to remove chub has delivered promising results. However, it is necessary to prevent and contain any new introductions or range expansions and the key pathways of introduction and vectors of spread for all the priority IAS were identified and include the horticulture, aquaculture, ornamental and aquaria trades and leisure activities such as boating, angling and water sports. A range of policy measures have been recommended including prevention and containment protocols for use at RBD level.

The research on the ecological impacts of *Lagarosiphon* and chub has increased our understanding of the impacts of these species and their interactions with native communities and other non-native species as well as demonstrating new means by which impacts can be measured. The development of the NISD, collation of records and mapping of the up-to-date distributions of aquatic IAS provides a valuable resource for researchers and managers. The demonstration of effective control measures for *Lagarosiphon* and chub will enable rapid reaction to further introductions and range expansions to new waterbodies. The development of proposals for surveillance, monitoring and reporting of IAS and policy measures for prevention and containment will inform the WFD programme of measures and river basin management.

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1 General introduction

1.1 Invasive species, ecosystem services and human wellbeing

While many non-native species have negligible negative effects (Bulleri *et al.*, 2008), some cause significant economic and ecological impacts including reductions in biodiversity, decline of commercially important species and the alteration of ecosystems and ecosystem services (Lodge & Shrader-Frechette, 2003). These non-native species that become established in existing ecosystems and subsequently threaten biodiversity and/or result in economic damage are referred to as invasive alien species (IAS) (Shine *et al.*, 2010). IAS are regarded as the second most serious cause of biodiversity loss and environmental change worldwide, affecting freshwater ecosystems in particular due to their isolation and large rate of endemism (Richter *et al.*, 1997; Dudgeon *et al.*, 2006). Aquatic ecosystems are susceptible to invasions, partly as a consequence of waterborne human activities such as shipping and boating, which represent a major vector for novel introductions (Darrigran, 2002). IAS have an impact on the ecosystem processes that are fundamental to human wellbeing including the wholesale loss or alteration of goods (*e.g.* fisheries) and services (*e.g.* clean and plentiful drinking water, culture and recreation) (Charles & Dukes, 2007; Pejchar & Mooney, 2009).

The evolutionary effects of invasion (Mooney & Cleland, 2001; Strauss *et al.*, 2006) are typically considered in the context of detrimental changes e.g. through introgression and hybridisation or the loss of native biodiversity following extinction or extirpation of native populations (Clavero & García-Berthou, 2005). However, invasion of new habitats by non-native species has also been associated with the formation of novel biodiversity, e.g. through adaptation in the invader itself following ecological release (Nolte *et al.*, 2005; Vellend *et al.*, 2007; Prentis *et al.*, 2008). Furthermore, there is the potential for native or naturalised species in receiving ecosystems to show adaptive responses (Agrawal, 2001), following invasion *e.g.* by an ecosystem engineer that increases habitat complexity or niche availability (Crooks, 2002).

Given the significant ecological and economic costs of IAS, understanding the environmental factors that regulate them has become a major goal for ecologists (Bulleri *et al.*, 2008), and there is a need to identify areas prone to invasions (Ibáñez *et al.*, 2009) or species which might become successful invaders (Van Kleunen *et al.*, 2010). In addition, ecologists have found that even basic field research is increasingly difficult without eventually encountering non-native species, so there is a need to understand the impacts and interactions of non-natives with native communities, as well as how they are transported and become established (Lockwood *et al.*, 2007). A recent study focusing on changes of conservation status of endangered species included in the IUCN Red List concluded that the overall impact of IAS pressure in driving declines in species diversity is increasing although IAS control or eradication can lead to improvement in the status of some endangered species (McGeoch *et al.*, 2010).

1.2 Drivers for IAS management

There are five pressures directly driving biodiversity loss namely, habitat change, overexploitation, pollution, climate change and IAS. In Europe as a whole, the rate of new introductions has risen steadily in recent decades and is still increasing for all taxonomic groups except mammals. The European Union (EU) has recently committed to a

target to halt the loss of biodiversity and the degradation of ecosystem services in the EU by 2020, restore them in so far as feasible, while stepping up the EU contribution to averting global biodiversity loss. Therefore, in addition to the practical need to respond to the impacts of IAS directly there are also a range of policy drivers which require us to take action. These range from the international (e.g. CBD), European (Habitats Directive), National (Wildlife Amendment Act (2000)) to departmental statutory regulations. Among the most critical drivers for addressing the issue of invasive species and conservation of biodiversity are provided in the Habitats Directive by the protection of Natura 2000 sites and features. A number of such features are under direct threat from IAS and introduced genetic material. Recent cases have underlined the importance of taking a pro-active approach to dealing with threats to Natura 2000 features, as opposed to reacting to current damage.

The Convention on Biological Diversity (CBD) requires contracting parties as far as is possible and appropriate, 'to prevent the introduction of, control or eradicate those alien species which threaten ecosystems, habitats or species'. Contracting parties to the CBD are committed under Article 8 to take action to:

- (k) develop or maintain necessary legislation and/or other regulatory provisions for the protection of threatened species or populations;
- (I) where a significant adverse effect on biological diversity has been determined....regulate or manage the relevant process and categories of activities.

Furthermore, Article 13 of the Convention commits contracting parties to 'promote and encourage understanding of the importance of, and the measures required for, the conservation of biological diversity, as well as its propagation through media, and the inclusions of these topics in education programmes...'. The implications of the CBD are that contracting parties have to take account of IAS and aim to prevent introductions, control invasive species and develop legislation.

Negative impacts of IAS on biodiversity can occur through a range of mechanisms such as competition, herbivory, predation, alteration of habitats and food webs, introduction of parasites and pathogens and through the dilution of native gene pools. In Ireland the most prominent of the negative impacts appears to be direct competition with native biota, whilst alteration to habitats and the influence of parasites and pathogens are also important (Stokes *et al.*, 2006). Specific habitat types currently under threat include freshwater rivers and lakes; coastal floodplains, saltmarsh and sand dunes; maritime cliff and slopes; woodland, lowland heath and semi-natural grassland. A variety of protected or native species are also threatened by IAS, including red squirrels, white-clawed crayfish, red deer, earthworms and now the freshwater pearl mussel with the recent introduction of the Asian clam (*Corbicula fluminea*).

Estimates of the number of IAS on the island of Ireland do not always distinguish between non-native species and invasive species and not all non-native species are invasive or have an impact on biodiversity. The total number of alien species has been estimated at 1,259 (DAISIE, 2009); estimates for particular biodiversity groups include 99 non-native animal species (DAISIE, 2009); 716 non-native terrestrial plants (Reynolds, 2002); 112 aquatic IAS and 63 cryptogenic aquatic IAS (Minchin, 2007). A pan-European analysis of presence of the 163 'worst' terrestrial and freshwater IAS threatening biodiversity in Europe showed that in 2006, the island of Ireland had 34 of these species (European Environment Agency, 2009) and since then a further 7 have been recorded. The cumulative number of high impact IAS recorded on the island of Ireland has also continued to grow and many species are expanding their distributions posing a threat to biodiversity and contributing to the degradation of ecosystem services.

1.3 Aquatic invasive species in Ireland

In Ireland, practical management of introduced species is challenging because of the cross-border implications of controlling introductions and spread (Stokes *et al.*, 2006). A pro-active stance is fundamental since prevention of introductions is demonstrably more cost-effective than reactive control or eradication measures. Ireland has a depauperate fauna compared with continental Europe as a result of its geographical isolation. This isolation has given rise to some unique subspecies and gene pools. Species invasions may therefore be particularly damaging if the few native species are adversely affected. Conversely, vacant habitats or niches may allow integration of species with minimal community disturbance and/or rapid recovery (Dick, 1996b).

Over the last decade there has been a growing body of research on aquatic IAS and their impact on the Irish environment. The most extensively studied and high profile invader in recent years has been the zebra mussel (*Dreissena polymorpha*). Like many invasive species, it is an ecosystem engineer (Crooks, 2002), which directly or indirectly controls the availability of resources to other organisms by causing physical state changes in biotic or abiotic materials. Zebra mussels impact on all aspects of the food web and WFD biological parameters, including phytoplankton abundance and composition, macrophyte abundance and composition, benthic invertebrate communities and fish populations.

Other well-documented species impacting on Irish waterbodies include the dace (*Leuciscus leuciscus*), which was restricted to the Munster Blackwater until the 1990s but is now expanding its range. The impacts of aquatic invasive plants on biodiversity are growing with species such as the water fern (*Azolla filiculoides*), parrot's feather (*Myriophyllum aquaticum*) and curly waterweed (*Lagarosiphon major*) of particular concern. There are also riparian species that are impacting biodiversity, the stability of banksides, fish productivity and recreational use of Irish watercourses, such as giant hogweed (*Heracleum mantegazzianum*), Japanese knotweed (*Fallopia japonica*) and Himalayan balsam (*Impatiens glandulifera*), with giant hogweed being the subject of catchment-wide control trials (Caffrey, 2001).

A recently arrived aquatic IAS with potentially serious consequences is the bloody-red shrimp (*Hemimysis* anomala), discovered in Loughs Derg and Ree in the Shannon River Basin District during 2008 and 2010 (Gallagher *et al.*, 2010). Compared to the similar native opossum shrimp (*Mysis relicta*), the invading species is a voracious predator. In Ireland, its prey includes cladocerans, copepods, rotifers and algae. Experiments suggested that the bloody-red shrimp has a significantly greater capacity than the native *M. relicta* species to prey on a range of key invertebrate and zooplankton taxa including *Gammarus pulex*, copepods, *Daphnia pulex* and *Chydorus sphaericus*, as well as on fish eggs and larvae, specifically pollan (*Coregonus autumnalis*). Gallagher *et al.* (2010) concluded that the bloody-red shrimp is highly likely to alter freshwater systems throughout its invasive range in Ireland by predation of other invertebrates, competition for prey with the native opossum shrimp and directly affecting priority fish species by eating eggs and larvae.

Irish waterbodies are increasingly being invaded by more than one high impact invasive species and this may produce unpredictable effects. In some cases, native species or established invasive species appear to facilitate establishment of later-arriving non-indigenous species. Synergistic interactions among invaders may well lead to accelerated impacts on native ecosystems, in an 'invasional meltdown' process (Simberloff & Von Holle, 1999).

Examples include Lough Corrib, which has been invaded by the zebra mussel and a range of non-native plants and fishes.

1.4 Aims of the research

Tackling IAS is complex due to the range of environmental, social, economic, political and technological factors involved and the interactions between them. The EPA consider that IAS represent a significant threat to Ireland's water bodies both in terms of their impact on natural ecosystems and their impact on the beneficial use of our waterways. The 2004 characterisation and analysis of Ireland's River Basin Districts required under Article 5 of the Water Framework Directive (WFD) identified a number of significant knowledge gaps. These included the array and distribution of IAS and their impact on the structure and function of natural ecosystems, particularly for high status water bodies. Better information is needed so that adequate management measures can be put in place and effective control measures developed for the WFD River Basin Management Plans. A call for proposals was issued and with the main aims to:

- Improve our knowledge of the nature and extent of aquatic invasive alien species in Ireland and their impact on natural ecosystems.
- Developing up to date national distribution maps showing the location of alien invasive species in Ireland.
- Develop and trial control measures in the context of river basin management.

This project was designed to fulfil all the elements of the EPA call, contribute to filling the knowledge gap under the WFD and to achieving commitments under the Convention on Biological Diversity (CBD), the Habitats Directive, the target to halt the loss of biodiversity and the degradation of ecosystem services by 2020 and the National Biodiversity Plan. The overall objectives of the project were to improve our knowledge of the nature and extent of IAS in Ireland and their impact on natural ecosystems and to develop and trial control measures for selected species. Three complementary work packages were undertaken by a multidisciplinary team focusing on policy development, IAS impacts and control and monitoring, mapping and recording.

Our policy development goals were to review impacts, vectors and management of high impact aquatic IAS in Ireland and internationally with a focus on a group of prioritised species, in order to inform policy development and management; and to develop strategies for monitoring and reporting and for preventing and containing IAS introductions (Work Package 1).

We aimed to describe the key consequences of invasion on the function and food-web dynamics, and on the community structure of, selected freshwater ecosystems in Ireland using stable isotope analyses, thereby determining the impacts of IAS on community structure and function. Although invasive species have the potential to impact receiving ecosystems (Crooks, 2002; Ward & Ricciardi, 2007), it is extremely difficult to actually quantify the direction and intensity of any subsequent ecological change (e.g. in ecosystem structure and function). One elegant means of assessing the consequences of invasive species is through the use of stable isotope analysis (SIA). (Work Package 2a).

The research was carried out at waterbodies in two River Basin Districts (RBD) which were classified at significant risk of failing to meet WFD objectives, namely Lough Corrib in the Western RBD and the River Inny in the Shannon

International RBD. These sites were selected as they had both been invaded in recent years and pilot studies were being undertaken collecting baseline data in both waterbodies. We undertook to determine the efficacy and consequences of measures aimed to control key invasive species, the invasive plant *Lagarosiphon major* in Lough Corrib, and an invasive fish, chub (*Leuciscus cephalus*), in the River Inny (Work Package 2b). These species were selected as they had a limited distribution and control and eradication programmes were planned which could be used to test proposed eradication methods and the effect on community structure of the removal of the IAS.

We aimed to determine the distribution of a range of priority IAS and put in place mapping and recording infrastructure. Specific objectives were to (a) identify the data requirements for monitoring and reporting strategies in line with national protocols and best practice, producing a guidance note for data contributors; (b) prepare a GIS database of the location of all reported aquatic invasive species displayed by river system and lake catchments units; and (c) make this information publicly available via the National Biodiversity Data Centre web system and to provide an efficient mechanism for this information to be updated (Work Package 3).

2 IAS surveillance, monitoring and recording

2.1 Introduction

It is clear that invasive alien species (IAS) are an increasing pressure on Ireland's freshwater environments, as demonstrated in the previous chapter with new species recorded each year. As measures to prevent the introduction of invasive species will not always be successful, it is important that species are detected early before they can become widely established. This requires surveillance, monitoring, reporting and verification of records, and effective and clear communication protocols.

The Convention on Biological Diversity (CBD) sets out guiding principles for the prevention and mitigation of impacts of IAS that threaten ecosystems, habitats or species (UNEP/CBD, 2002). The CBD guiding principles relating to monitoring state that countries should undertake monitoring of IAS as appropriate, including general and targeted surveys and involve stakeholders. The guiding principles on information exchange state that countries should assist in the development of databases and information systems and disseminate information for use in the context of any prevention, introduction, monitoring and mitigation activities. This information should include incident lists, potential threats to neighbouring countries, information on taxonomy, ecology and genetics of invasive species, as well as national regional and international guidelines, procedures and recommendations.

The European Union is currently developing an EU Strategy on IAS aiming at publication in 2012. The European Commission has proposed a number of possible options for an IAS Strategy which include maximising the use of existing legislation together with voluntary measures, amendment of existing legislation to cover a broader range of potentially invasive organisms or development of a new legal framework for tackling invasive species with independent procedures for assessment and intervention (COM, 2008). The assessment to support the development of the strategy has recommended the development of a dedicated IAS Directive and identified risk assessment, surveillance, monitoring and the development of a European early warning and information sharing system as priority areas (Shine *et al.*, 2010).

2.2 Definitions

In order to develop practical and useful recommendations it is important to clearly define the parameters of any surveillance, monitoring and recording programmes. The following definitions have been adapted from the International Standards for Phytosanitary Measures and are used in this report:

- Surveillance is a process which collects, records, interprets and disseminates data on IAS occurrence, absence or prevalence by survey, monitoring or other procedures.
- Monitoring is an ongoing process and involves regular surveys or ongoing sampling to verify the characteristics
 of IAS populations using standardised protocols.
- Recording is an ongoing process of collection and verification of records concerning the presence or absence of a specific IAS at a particular location at a certain time under described circumstances.

2.3 Role of IAS surveillance, monitoring and recording

Surveillance, monitoring and recording programmes are vital components of the tool-kit of IAS prevention and management. They consist of a range of activities focused at different pathways, taxonomic groups and habitats operating at different spatial scales. An effective programme of early detection leading to rapid response is totally dependent on information being collected, communicated and acted upon. The overall aim of these programmes is to develop and implement effective mechanisms for detection, surveillance, monitoring and recording of new and established invasive species and disseminate the information in a timely way to enable appropriate action to be taken.

2.4 Building on developments to date in Ireland

Risk assessment is a key mechanism to enable the allocation of limited resources to those species which pose the greatest threat to Irish biodiversity and ecosystem services. Ireland's WFD Characterisation and Analysis Report stated that 'alien species risk assessments will require more development during further characterisation' and that 'improved recording of alien species will be required' (EPA, 2005a). The Invasive Species Ireland (ISI) project developed a risk assessment and prioritisation methodology and carried out nearly 600 risk assessments for established and potential IAS which classified a number of species as high risk (see www.invasivespeciesireland.com for details). The risk assessments were carried out in a transparent manner with input from a wide range of IAS experts. The ISI list of high risk freshwater species, the DAISIE (Delivering Alien Invasive Species Inventories for Europe) inventory for Ireland and the Ecoregion 17 list for the Water Framework Directive were used to compile a list of priority species which were the focus of this project. The priority list comprised 17 established species and 14 species considered high risk potential invaders to Ireland (Table 2.1). These species were the focus of the mapping and recording (Chapter 3), the surveillance, monitoring and recording strategy (Chapter 4) and the best practice guide to prevention and containment (Chapter 9).

Table 2.	1: Prio	rity spe	cies list
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Established high impact	Species	Common name
Aquatic plant	Lagarosiphon major	Curly waterweed
	Elodea nuttallii	Nuttall's waterweed
	Myriophyllum aquaticum	Parrots Feather
	Crassula helmsii	New Zealand pigmyweed
	Azolla filiculoides	Water fern
	Lemna minuta	Least duckweed
	Nymphoides peltata	Fringed waterlily
	Hydrocotyle ranunculoides	Floating pennywort
Invertebrate	Dreissena polymorpha	Zebra mussel
	Gammarus pulex and G. tigrinus	Crustacean
	Eriocheir sinensis	Chinese Mitten Crab
Fish	Leuciscus cephalus	Chub
	Leuciscus leuciscus	Dace
Riparian plant	Fallopia japonica	Japanese knotweed
	Impatiens glandulifera	Himalayan balsam
	Heracleum mantegazzianum	Giant hogweed
Potential high impact	Species	Common name
	Ludwigia peploides and L.	
Aquatic plant	grandiflora	Water primrose
Invertebrate	Astacus astacus	Noble crayfish
	Astacus leptodactylus	Turkish crayfish
	Pacificastacus leniusculus	Signal crayfish
	Orconectes limosus	Spiny-cheeked/striped crayfish
	Procambarus clarkii	Red swamp crayfish
	Hemimysis anomala	Bloody red shrimp
	Gyrodactylus salaris	Parasite
	Dreissena bugensis	Quagga mussel
	Corbicula fluminea	Asian clam
Fish	Gymnocephalus cernuus	Ruffe
	Sander lucioperca	Zander
	Pseudasbora parva	Top mouth gudgeon

2.5 **Project objectives and contribution to national research needs**

The EPA alien species risk assessment guidance stated that '*further work will be required to establish, more accurately, the range of each species and assess actual alien species pressures on waterbodies and to design the most appropriate programme of measures*' (EPA, 2005b). This project contributes to this national research need by delivering up to date distribution maps and assessing IAS pressures. There are four project objectives relating to monitoring, surveillance and recording, namely:

- WP1 Objective 2: Develop a monitoring and reporting strategy for high impact invasive species.
- WP3 Objective 1: To identify the data requirements for monitoring and reporting strategies in line with national protocols and best practice.
- WP3 Objective 2: To prepare a GIS database of the location of all reported aquatic invasive species displayed by river system and lake catchments units.
- WP3 Objective 3: To make this information publicly available via the National Biodiversity Data Centre web system and to provide an efficient mechanism for this information to be updated.

3 The National Invasive Species Database

3.1 Introduction

Accurately tracking the movement of invasive species is particularly important as this information can feed directly into effective early warning, rapid response and monitoring and control programmes. The National Invasive Species Database (NISD) has been developed by the National Biodiversity Data Centre and this project has made a major contribution to this important initiative. This element of the project focused on developing the mapping and recording infrastructure necessary to support surveillance and monitoring and provide up to date information on species distributions to support the implementation of programmes to manage IAS as part of achieving favourable conservation status of species and habitats and good ecological status of waterbodies. This involved developing a GIS based database of aquatic invasive species which enables records to be displayed against the backdrop of additional GIS layers, the provision of a mechanism for online submission of verified records and the production of guidance for data contributors that meets requirements for monitoring and reporting strategies in line with national protocols and best practice. The combination of a mechanism for online submission of records and a mapping system has moved the NISD from a static resource into a dynamic database that has the potential to become a vital tool in the identification, monitoring and control of aquatic IAS in Ireland. It also provides the infrastructure for an early warning system which has been deployed four times in 2010. Work undertaken in the National Invasive Species Database project has also contributed to the wider development of IAS information exchange and horizon scanning at the European level largely through NOBANIS - the European Network on Invasive Alien Species (www.nobanis.org).

3.2 Distribution maps of priority aquatic IAS

Records were collated for the 17 established species (see Table 2.1) and entered into the National Invasive Species Database and as of September 2010 this amounted to 5077 records. Three of the potential high impact species that were not known to occur in Ireland when this project began have since been recorded in Ireland namely, the Asian clam (*Corbicula fluminea*), bloody red shrimp (*Hemimysis anamola*) and water primrose (*Ludwigia grandiflora*) and 62 records are in the database for these species. Species distributions have been mapped and made publicly available on the National Biodiversity Data Centre's interactive GIS Biodiversity Maps website: http://maps.biodiversityireland.ie. Not all species have a complete dataset of records i.e. there are records in existence that are not in the database. However, the National Invasive Species Database is a full time project of the Data Centre and so records for these species will be continually updated.

The distribution maps for the established aquatic plant species are shown in Figure 3.1, riparian plant species in Figure 3.2, invertebrate species in Figure 3.3, fish species in Figure 3.4 and the four newly arrived species in Figure 3.5. The records shown in the maps are the records available in the NISD at the time of writing. It is important to know if the distribution shown is reflective of the species known distribution or is deficient. The coverage assessment given is based on knowledge of a species being recorded elsewhere but the records are not in the database. Unless detailed systematic surveys are carried out for each species it is difficult to say if the distribution mapped is an accurate reflection of the species actual distribution so the system used to give the coverage assessment is based on a traffic light system:



Figure 3.1: Distribution maps of the aquatic plant species (a) Azolla filiculoides, (b) Crassula helmsii, (c) Elodea nuttallii, (d) Hydrocotyle ranunculoides, (e) Lagarosiphon major, (f) Lemna minuta, (g) Myriophyllum aquaticum and (h) Nymphoides peltata.



Figure 3.2: Distribution maps of the riparian plant species (a) Fallopia japonica, (b) Heracleum mantegazzianum and (c) Impatiens glandulifera.



Figure 3.3: Distribution maps of the invertebrate species (a) *Dreissena polymorpha*, (b) *Gammarus pulex*, (c) *Gammarus tigrinus* and (d) *Eriocheir sinensis*.



Figure 3.4: Distribution maps of the fish species (a) Leuciscus cephalus and (b) L. leuciscus.



Figure 3.5: Distribution maps of the newly recorded species (a) *Corbicula fluminea,* (b) *Hemimysis anomala* and (c) *Ludwigia grandiflora.*

The distribution maps shown in the above Figures can be viewed against a range of GIS layers which are currently available on the Biodiversity Maps site including geographic (border, counties, localities, townlands), ecological (Special Protection Areas, Natural Heritage Areas, Special Areas of Conservation, Nature Reserves, lakes and rivers) and physical (Corine landcover (2000), bedrock geology, soils, subsoils). In addition, GIS layers for transport, aerial photography, base map and various layers for Northern Ireland are also available. The mapping system will be continually up-dated and developed to optimise its use as a tool for the geographic presentation of observational data on Ireland's biodiversity. Additional GIS layers due to be added to the mapping system which

will contribute to the management of aquatic IAS include detailed rivers and lakes layers with WFD water body codes and the EPA water quality indicator layer.

There are a multitude of benefits from displaying IAS distributional data against GIS layers on an interactive mapping system. The NISD can provide additional contextual information to inform risk assessment and management plans such as natural distribution corridors, soil suitability for establishment conditions and proximity to or presence in designated areas.

3.3 Online submission of IAS records and guidance for contributors

Up to date information on the distribution of invasive species is crucial to support effective management. The provision of an online record submission facility is provided through the Data Centre's invasive species website: http://invasives.biodiversityireland.ie in the form of a customised excel file. This excel file provides a template for inserting invasive species records and highlights the mandatory fields that constitute a valid biological record as well as some desirable data that would enhance the value of the record. Example entries are provided for guidance. The Invasive Species Ireland (ISI) website also includes an Alien Watch page where invasive species records can be submitted (www.invasivespeciesireland.com/sighting/). Periodically, the records are then forwarded onto National Biodiversity Data Centre for inclusion in the National Invasive Species Database.

A Guidance Note for Data Contributors with specific reference to invasive species has been produced. It provides guidance on meeting standards for data capture and reporting in order to provide high quality information and is based on the Data Centre's current Guidance Note for Data Contributors. This is available for download on the Data Centre website and shown in Figure 3.6.



Figure 3.6: Guidance note for contributing IAS data and form showing the minimum data requirements.

3.4 Displaying aquatic IAS data in a biologically meaningful way

This project initially proposed to display aquatic IAS records by river and lake segments with Water Framework Directive (WFD)/EU coding. The relevant WFD/EU coded river and lake layers were obtained from Compass Informatics Ltd. and work was undertaken to map species against these layers. The preliminary results of undertaking this mapping process showed that this was not the most effective way to display this information.

3.4.1 Rivers

The river records for dace (*Leuciscus leuciscus*) were mapped and highlighted by river segment. The outcome of this mapping was not as expected. At the national scale, the distribution pattern is not clear as the segments are small and 'lost' when displayed against the whole river layer (Figure 3.7). In addition not all the river segments with dace records are mapped or have WFD coding so such records cannot be assigned to a river segment and need to be removed from this mapping process giving an incomplete picture of the species distribution.



Figure 3.7: Distribution of (a) Leuciscus leuciscus, (b) mapped by river segment and (c) mapped against river layer.

3.4.2 Lakes

The records for the zebra mussel (*Dreissena polymorpha*) were used to show how lakes with an invasive species can be displayed. Lakes with a positive occurrence of a zebra mussel record are shaded in purple (Figure 3.8). This is visually clearer than the previous example but at the national scale the lakes with the invasive species do not stand out very clearly, especially for the smaller lakes when displayed against the detailed lakes layer.



Figure 3.8: Distribution of (a) *Dreissena polymorpha*, (b) mapped by lake segment and (c) mapped against lake layer.

While it is possible to map and display aquatic species records by river and lake segments there are many disadvantages with displaying records in this manner including:

- Not all river segments are mapped (e.g. not brackish waters) so records of a species in these waters cannot be mapped.
- An automatic spatial join of the record to the nearest river/lake segment requires manual manipulation in most cases.
- Not all river or lake segments have WFD/EU coding.
- It is very time-consuming to manually assign WFD/EU codes to records (where codes are available).
- Being unable to map all records of a species means there is no comprehensive overview of the distribution at a national scale.
- Visually, displaying records in this manner is not very clear especially for river segments.

After evaluating the results it was decided that this method was not feasible and an alternative method would be used. Records of aquatic IAS will continue to be displayed by point location on the mapping system but once the WFD/EU coded detailed river and lake layers are added, it will be possible to view the coding of the river/lake segment and use this information for management and reporting purposes. Static GIS maps for *Dreissena polymorpha* by lake segment will also be displayed on the NISD website once all of the historic records available have been entered into the database.

3.5 Conclusions

The collation and updating of aquatic IAS records will continue beyond the life of this project as the National Biodiversity Data Centre will continue to hold and manage the species distribution data as part of its National Invasive Species Database project. The importance of continuing the management of these data cannot be overstated. The ability to publicly display the current distribution of IAS, track their spread and detect early introductions into new areas is vital to effective invasive species management. However, this requires intensive management of the database with ongoing support and collaboration from a wide range of individuals, organisations and state bodies. By building on the project outputs a number of options for future work can be achieved which include:

- Submission of all species records collected in the project to the National Invasive Species Database so as to improve our understanding of the spread of aquatic IAS.
- The development of an information exchange network that would involve project partners and other organisations submitting IAS records to the Data Centre and the Data Centre in turn, providing regular communication on significant distribution updates to contributors (such as information on early detection of a species to an area).
- Initiation of an all-Ireland survey for the priority species in collaboration with the project partners and others e.g. Botanical Society of Britain and Ireland (BSBI) and the Centre for Data and Recording (CeDAR). Some of the species may not lend themselves to a general public recording initiative and so focusing on plants and easily identifiable species may be more appropriate.
- Development of a way of displaying when species have been removed from sites or eradicated as part of the NISD and promotion of this information.
- Further research to develop of a way of displaying records against river and lake layers.
- Linking of the National Invasive Species Database into existing surveillance and monitoring programmes (see Chapter 4 for detailed recommendations).
- Further development of an early detection and rapid reaction mechanism for Ireland which is essential for IAS management. The Data Centre is in a core position to provide efficient and timely reporting of occurrences of species to new areas but this ability is dependant on up-to-date records being submitted to the National Invasive Species Database. Therefore, awareness of the National Invasive Species Database and its usefulness as a tool for effective IAS management needs to be promoted and utilised by all interested in tackling the threat of invasive species in Ireland.
- Support the development of a European wide Early Warning and Rapid Response information system through continued collaboration with the NOBANIS network and direct liaison with the European Commission.

4 Surveillance, monitoring and reporting strategy

4.1 Introduction

Monitoring programmes are usually undertaken for IAS once the species are already established and are having economic and/or ecological impacts. However, non-native species may be present in an ecosystem for many years before they become invasive and start causing problems, known as a lag phase. Usually it is only when they become a problem that control or eradication attempts are made, often when it is too late. Regular surveillance for IAS is needed so that new invasions or range expansions are detected and control or eradication attempts may be made at an early stage.

A recent report for Great Britain estimated that IAS cost the British economy at least £1.7 billion each year. Twelve economic sectors were analysed with agriculture and horticulture bearing the highest costs through the costs of controlling IAS; quarantine and surveillance programmes; and loss of productivity due to the presence of IAS (Williams *et al.*, 2010). Costs were also calculated for individual species which highlight the benefits of investing in prevention, surveillance and rapid response. The water primrose (*Ludwigia grandiflora*) has recently been found in south-west Ireland (Figure 3.5) and also has a relatively restricted distribution in Britain. The analysis from Britain clearly shows the economic justification for early eradication of this species which has been estimated to cost £73,000 compared to the £242 million it would cost if it became fully established in Britain as it has done in France and Belgium. This demonstrates that investment in surveillance and monitoring can reduce the economic impact of IAS by enabling management to be carried out at an early stage.

This project initially intended to carry out a literature review of aquatic IAS that would be used to inform policy development and management and further develop the Invasive Species Ireland (ISI) recommendations on monitoring and recording into a monitoring and reporting strategy focused on aquatic species. The scope of the ISI work was expanded to include surveillance, and a report recommending monitoring, surveillance and recording programmes for IAS was produced (Maguire, 2009). Therefore there were clear benefits to including surveillance in the scope of this project and building on the ISI work and stakeholder engagement programme which has raised the profile of IAS and provided necessary supporting resources such as identification guides. Developing an effective programme of surveillance, monitoring and reporting will be challenging as it requires the participation of a wide range of stakeholders and for information to be submitted and verified in a timely way. The project tasks were amended and a series of species accounts were produced that are available from the ISI and NISD websites for use by stakeholders and a surveillance, monitoring and reporting strategy has been proposed that focuses on the priority aquatic species and can be used at River Basin District level and incorporated into the Water Framework Directive (WFD) programme of activities.

4.2 Current developments in Ireland

The ISI analysis identified where IAS can be integrated into ongoing surveillance and monitoring programmes to avoid duplication of resources and recommended an effective system for recording new species and spread of established species so that the information is disseminated for appropriate action. The recommendations were informed by a number of workshops and consultations which included a workshop on monitoring and recording held

as part of the 2008 ISI Forum and the ISI review to identify the way forward for IAS management in Ireland (Maguire & Kelly, 2008).

Maguire (2009) highlighted the need to incorporate surveillance, monitoring and recording programmes into a wider IAS prevention and management framework as actions in areas such as training and education and awareness will contribute to the effectiveness of these programmes. The emphasis in most countries is on developing early detection and rapid response capacity, however it is also important to balance the allocation of resources across activities as monitoring will determine how invasive a population is so that management efforts can be focused on expanding populations rather than those which are unlikely to act as sources of further spread due to habitat constraints.

A range of surveillance, monitoring and recording activities for IAS are already underway in Ireland. The key surveillance programmes are the plant and animal health regimes but these only incorporate IAS to a limited extent. The plant health regime has recently been reviewed and the evaluation has recommended clarifying and possibly expanding the scope to include harmful organisms that affect environmental public goods such as IAS so there may be scope to integrate further with plant health in the future (Shine et al., 2010). Other surveillance activities are specifically targeted at IAS, some are driven by legislative requirements, others are voluntary and they are carried out by a range of organisations at different spatial scales. The ISI review detailed ongoing monitoring and surveillance activities which are mainly focused on aquatic species and plant health intercepts, grey squirrels and other mammals (Maguire & Kelly, 2008). Much of the infrastructure for a recording programme for IAS is already in place with the development of the National Invasive Species Database (NISD) as detailed in Chapter 3. While there has been no formal surveillance programme to date, in the last few years, greater awareness of IAS threats has led to more reporting of new species and new sightings. A system for reporting and verifying records and issuing species alerts is in place through Invasive Species Ireland and the National Invasive Species Database (NISD). This has resulted in verified reports and dissemination of information through species alerts for a number of priority established and potential IAS including the bloody red shrimp (Hemimysis anomala), water primrose (Ludwigia grandiflora) and Asian clam (Corbicula fluminea).

However, surveillance and monitoring for IAS in Ireland currently takes place in a piecemeal way, there is no dedicated monitoring programme and IAS have not yet been integrated into ongoing monitoring programmes in any strategic or systematic manner. Developing and implementing an effective surveillance, monitoring and reporting programme should be viewed as a process as resources are unlikely to be available for additional capacity in the coming years. It may be more effective to start with integrating IAS into ongoing activities and for the priority species which are the focus of this project. The WFD programme of activities offers the greatest scope for integration and potential for development of programmes while requiring minimal additional resources.

4.3 The Water Framework Directive and IAS

The increasing impacts of IAS in the aquatic environment are of growing concern in the context of the WFD which requires member states to achieve at least good status by 2015, aiming at maintaining high status and preventing any deterioration in existing status of waterbodies. The species that were the focus of this project were ones that

are currently impacting or have the potential to impact on meeting WFD objectives by affecting WFD biological parameters, including phytoplankton abundance and composition, macrophyte abundance and composition, benthic invertebrate communities and fish populations. This raises real challenges for policy makers on not only how they can deal with IAS under the WFD but in this time of limited resources, how they can identify and use opportunities that the WFD offers to address threats to biodiversity such as IAS. In this element of the project we have sought to identify those opportunities and make practical recommendations to progress IAS management in tandem with wider WFD objectives.

While the text of the WFD does not explicitly mention IAS, it has been considered that what is listed in Annex II (1.4) under identification of pressures as 'estimation and identification of other significant anthropogenic impacts on the status of surface waters' includes IAS. At the present time there is no common approach to dealing with invasive species under the WFD with only the UK and Ireland issuing guidance on the assessment of alien species pressures. However the establishment and impacts of IAS will hinder progress towards good ecological status in waterbodies. The European Commission's intercalibration process (known as ECOSTAT) is examining the use of IAS in ecological status classification and how the WFD programmes of measures might be used to address IAS. The procedures used by Member States are extremely varied, according to country, biological element and surface water type. Four different approaches are currently used at national or local level: (1) waterbodies are classified using pressure-based classification tools and the classification is then modified in an additional step based on IAS; (2) waterbodies are classified, then modified depending on the abundance or percentage coverage of IAS; (3) no additional assessment of IAS is undertaken on the assumption that impacts of IAS are detected in existing instruments; or (4) a separate risk assessment for IAS undertaken and/or biopollution indices are published alongside the water classification, but do not affect classification (ECOSTAT, 2009).

At an ECOSTAT workshop in June 2009 a range of conclusions was agreed including that as IAS constitute an important pressure and alter the taxonomic composition of biological communities, they must be taken into account when implementing the WFD. Both the text and the spirit of the WFD seem to preclude the presence of established IAS at high status (reference condition) but there is no consensus that amongst Member States on this as some countries might not have any water bodies in high status owing to the high number of IAS present in their waters. This would be problematic for Ireland given the widespread distribution of species such as the zebra mussel. There was also agreement that prevention of further invasions is critically important and that methods for identifying the risk of spread and invasion to previously unaffected water bodies need to be identified for use under Article 5 of the WFD (river basin characterisation). In this way rapid responses can be targeted appropriately under Article 11 (programmes of measures) where water bodies are not in good status due to IAS. It is unlikely that a common European position will be achieved in the short to medium term. The ECOSTAT group has agreed to:

- Evaluate whether sampling methods used for the WFD are effective at recording IAS, and propose improvements.
- Establish a framework and protocol for completing, standardizing, centralizing and maintaining EU lists of IAS for WFD purposes.
- Exchange information on how to address IAS problems through WFD programmes of measures.

The ECOSTAT group also agreed that it is essential that data of sufficient quality and reliability are obtained for describing the effects that IAS have on specific water bodies in order to help in future planning and that where

species not included within the WFD biological elements (e.g. zooplankton) are affected by IAS, it is recommended that an additional method is used to identify IAS pressures.

These future outputs will be useful for Ireland but at the same time actions to manage the ecological impact of IAS in waterbodies still need to be progressed without consensus on these issues. As part of the WFD characterisation and analysis of waterbodies in Ireland, risk assessments were carried out and these included IAS as a shadow assessment as they were reliant on expert opinion. There are five River Basin Districts (RBDs) and three International River Basin Districts (IRBDs) on the island of Ireland and draft River Basin Management Plans (RBMPs) have now been prepared for all of them. The North Eastern RBMP identified IAS as a 'local focus and future issue' and proposed that supplementary measures to address IAS should be undertaken at district level on a 2009-2015 timeframe. Therefore IAS have been identified as a pressure that should be included in the programme of measures for every RBD. This provides an opportunity to integrate IAS into WFD programmes across Ireland.

4.4 The WFD monitoring and reporting programme

A network of monitoring sites for the WFD was established by the EPA in December 2006 covering rivers, lakes, coastal waters and estuaries as well as groundwater. The monitoring is undertaken by several organisations including the EPA, Local Authorities, the Marine Institute and Inland Fisheries Ireland. Monitoring information is collected by the EPA, Inland Fisheries Ireland, Marine Institute, Office of Public Works, National Parks and Wildlife Service, Waterways Ireland and Local Authorities. The EPA is responsible for assessing this monitoring data and assigning status to Ireland's surface waters and groundwaters.

The WFD monitoring programme comprises three types of monitoring; surveillance, operational and investigative and each has different objectives. The objectives of surveillance monitoring include supplementing and validating the risk assessment and the assessment of long-term changes resulting from widespread anthropogenic activity so there is scope for further inclusion and assessment of IAS. The objectives of the operational monitoring are to establish the status of those waterbodies identified as being at risk of failing to meet their environmental objectives and assess any changes in the status of such waterbodies resulting from the programmes of measures. This involves monitoring species and habitat protected areas that are at risk and offers scope to include monitoring of IAS as part of the biological quality elements. The investigative monitoring programme is primarily focused at pollution and offers little scope for inclusion of IAS. Surveillance and operational monitoring programmes will sample waterbodies on a three-yearly cycle (minimum frequency) with over 3000 sites in rivers, 250 in lakes and sites in coastal and transitional waters and canals (EPA, 2006). As the biological communities will mostly be sampled every three years the WFD monitoring programme can provide a useful framework for a monitoring programme for aquatic IAS but it will not cover all priority IAS.

Timely reporting of monitoring results is seen as a key element in the achievement of the aims of the WFD. The direct linking of monitoring results to specific measures within catchments will help to provide ongoing feedback concerning the effectiveness or otherwise of the measures (EPA, 2006). The EPA recognised the importance of data and information management and the WFD monitoring programme states that in order to maximise the use of the data generated, it needs to be collected, managed, analysed and reported in a systematic, efficient and timely

manner, providing the information to those who need it when they need it. The EPA, in conjunction with the Local Government Computer Services Board and the River Basin Districts have developed an Environmental Data Exchange Network (EDEN) which will enable the exchange of data, including WFD monitoring data between environmental agencies in Ireland. In time it is intended that EDEN will be a fully distributed data-sharing network allowing all stakeholders to easily share environmental data (EPA, 2006). This will also include establishing data standards for use by all stakeholders.

4.5 Integration of IAS into WFD programmes

In order to determine the scope for inclusion of IAS into WFD programmes, a key consideration will be whether the ECOSTAT group recommends the assessment of IAS in their own right and whether this is qualitative or quantitative monitoring and reporting. Monitoring for IAS can be incorporated into the biological quality elements, namely, phytoplankton, macrophytes, benthic invertebrate and fish communities. The WFD will not cover IAS outside these groups such as zooplankton species or species which do not affect these elements. However these are less likely to be prioritised as high risk IAS given the emphasis on EU legislative obligations in the risk assessment methodology. An assessment needs to be carried out on a species by species basis whether the WFD monitoring programme would provide the relevant geographical coverage, required frequency of monitoring and type of data collection. This project has focused on a number of priority species (see Table 2.1) and the majority of these species are included in the biological parameters of the WFD as shown in Table 4.1.

Established high impact species	Habitat	WFD Biological Parameter
Lagarosiphon major	Lakes	Macrophyte abundance and composition
Elodea nuttallii	Rivers, lakes	Macrophyte abundance and composition
Myriophyllum aquaticum	Lakes	Macrophyte abundance and composition
Crassula helmsii	Lakes	Macrophyte abundance and composition
Azolla filiculoides	Rivers, lakes	Macrophyte abundance and composition
Lemna minuta	Rivers, lakes	Macrophyte abundance and composition
Nymphoides peltata	Lakes	Macrophyte abundance and composition
Hydrocotyle ranunculoides	Rivers	Macrophyte abundance and composition
Dreissena polymorpha	Rivers, lakes	Benthic invertebrate communities
Corbicula fluminea	Rivers, lakes	Benthic invertebrate communities
Gammarus pulex and G. tigrinus	Rivers, lakes	Benthic invertebrate communities
Hemimysis anomala	Rivers, lakes	n/a
Eriocheir sinensis	Rivers, transitional, coastal	n/a
Leuciscus cephalus	Rivers	Fish community composition
Leuciscus leuciscus	Rivers	Fish community composition
Fallopia japonica	Rivers	n/a
Impatiens glandulifera	Rivers	n/a
Heracleum mantegazzianum	Rivers	n/a
Potential high impact species	Habitat	WFD Biological Parameter
Ludwigia peploides and L. grandiflora ¹	Lakes	Macrophyte abundance and composition
Astacus astacus	Rivers, lakes	n/a
Astacus leptodactylus	Rivers, lakes	n/a
Pacificastacus leniusculus	Rivers, lakes	n/a
Orconectes limosus	Rivers, lakes	n/a
Procambarus clarkii	Rivers, lakes	n/a
Gyrodactylus salaris	Rivers	n/a
Dreissena bugensis	Rivers, lakes	Benthic invertebrate communities
Gymnocephalus cernuus	Rivers, lakes	Fish community composition
Sander lucioperca	Rivers, lakes	Fish community composition
Pseudasbora parva	Lakes	Fish community composition

Table 4.1: Priority IAS (see Table 2.1) and their inclusion in WFD Biological Parameters

¹ L. grandiflora has only been recorded at sites in artificial waterbodies and control action is underway so they have not been classified as established.

While not all the species in Table 4.1 are included in the WFD biological parameters, they all have the potential to impact on the assessment of the biological quality of the waterbody and would be included in surveillance and operational monitoring. Given that the sampling frequency of the WFD monitoring programme is once every three years, for many IAS more frequent monitoring will be needed to detect range expansions especially for species which are in the early stages of establishment.

As the WFD monitoring programme will entail people being out in the field and visiting over 3,250 sites in lakes and rivers this presents a real opportunity for surveillance for new IAS and range expansions of established IAS. The WFD does offer an opportunity to put in place an initial framework for monitoring, surveillance and reporting for the priority aquatic IAS with minimal additional resources.

4.6 Recommended actions for Ireland

Ideally, the surveillance, monitoring and reporting programme would cover all established and potential IAS identified through the ISI risk assessment process as a threat to biodiversity, ecosystem service and the economy. There are many factors which will influence the final design and effectiveness of any surveillance, monitoring and recording programme including policy developments. At an EU level, these include the agreement of a common European position on IAS and the WFD, the development of an EU IAS Strategy (and potentially an IAS Directive) which may result in surveillance, monitoring and reporting requirements; and the review of the plant health regime which may lead to greater involvement of the relevant agencies in IAS surveillance. At a national level, harmonised IAS strategies for the Republic of Ireland and Northern Ireland are currently in development and it is anticipated that the public consultation will take place in spring 2011. These strategies will act as a policy driver for the integration of IAS into current programmes such as the WFD.

The proposals presented here contain recommendations on surveillance, monitoring, recording and reporting protocols and identify synergies with the ISI work programme. This offers the opportunity to move beyond making recommendations to implementation over the next few years.

1. Priority species alert list: It will not be possible to include all aquatic IAS in the programme given the need to prioritise resources and also the practical difficulties in intercepting and identifying IAS. Therefore a prioritised alert list should be developed building on the ISI most unwanted prioritised species list and incorporating the species identified by the EPA in their WFD risk assessment guidance (EPA, 2005b). The ISI risk assessment framework has been reviewed and a new set of risk assessments are currently in preparation and due for publication early in 2011. This should be used to develop and agree an updated priority species list, and as national and local priorities may differ, specific alert lists can also be developed for each RBD which will help identify the risk of invasion to previously unaffected waterbodies.

2. Monitoring: The WFD river and lake surveillance and operational monitoring programmes consist of a number of subnets focused on different elements. All the subnets in the river and lake surveillance monitoring programmes should incorporate the priority species into the monitoring protocols with quantitative data collected on species as part of the monitoring of biological quality elements. Priority IAS should also be included in subnet 4 (monitoring of the effectiveness of measures aimed at retaining high and good status) and subnet 5 (species and habitat protected areas) of the rivers and lakes operational monitoring programmes. Monitoring programmes can be amended during the period of the RBMP and between RBMP cycles so priority IAS could be incorporated before 2015. Power analysis can be used to assess the minimum sampling effort needed but this will need to be carried out for different methods and target species and evaluated against the coverage required by the programme. It will be more beneficial to use methods that are capable of detecting a species at low density to identify new invasions or range expansions. For established species, this will improve our knowledge of species distributions and the data can be used to assess range expansions and identify waterbodies that are vulnerable to invasion by high impact IAS for each RBD.

3. Surveillance: The WFD monitoring programme offers the opportunity for surveillance for new IAS. Once the priority species list has been revised and agreed, ISI will produce materials including identification guides that can be used in the field and if these were supplied to all those involved in the WFD monitoring programme, that would greatly enhance surveillance capacity from current levels right across the island of Ireland.

4. **Recording:** Recording of IAS should be carried out according to the guidance note for contributors of IAS data (see Chapter 3 for more details). This guidance note should be used as the data standard for IAS and the WFD and supplied to all staff involved in the monitoring programme.

5. Reporting: All IAS records should be submitted to the National Invasive Species Database. The Data Centre and ISI are currently developing networks for key species and further developing the alert system. The EPA has committed to submitting all of their biological record information to the Data Centre and to date, the River Biologists digitized dataset (2005-2009) containing 29,587 records for 97 species (native and non-native) covering 1,164 sites has been submitted. Integrating the NISD into EDEN would provide a mechanism to integrate IAS data with WFD data. This would require linking EDEN and the River Basin Management Systems to the NISD and integration of the RBD and catchment GIS layers into the NISD. This will enable IAS records to be displayed in the form of up to date distribution maps on a RBD and catchment basis as well as for individual waterbodies which will provide a useful resource. Figure 4.1 illustrates the outputs of this information management system using the Eastern RBD information management system as an example. Outputs would include distribution maps, identification of range expansions, species alerts and identification of vulnerable waterbodies on a RBD level.



Eastern RBD – River Basin Management System

Figure 4.1: Potential outputs of the integration of the NISD into EDEN and the WFD monitoring programme

The proposals presented here demonstrate the added value of integrating IAS into the WFD and could be progressed over the next few years with supporting measures can be provided by the ISI project. These include the production of priority species alert lists on a national and RBD level and identification guides for use in the field. ISI will also be engaging with catchment stakeholder groups and providing IAS information to support the work of these groups in contributing to the implementation of the WFD. This same information can be supplied at RBD level to technical and advisory councils. This could include information on the distribution of established IAS, recording guidance, identification of potential IAS, best practice management guidance, IAS action plans (which include management, exclusion and contingency plans) and education and awareness materials.

5 The ecological impacts of IAS on freshwater ecosystems

Non-native species dominate ecosystems across the world (Vitousek *et al.*, 1996; Didham *et al.*, 2005) and IAS have been recognised as the second leading cause of biodiversity loss and environmental change affecting freshwater ecosystems (Richter *et al.*, 1997; Dudgeon *et al.*, 2006). The literature on the measured and potential impacts of non-native species on receiving natural ecosystems is large and continually growing (Lockwood *et al.*, 2007). The effects of introduction are diverse, and include impacts at different levels of biological organisation (Landry & Bernatchez, 2010) from genetic effects (Nolte *et al.*, 2005), through interactions between individuals, shifts in consumer diet and trophic position (Britton *et al.*, 2010a) and the provision of new prey resources to native species (Inger *et al.*, 2010) and human populations (Gozlan, 2008). Further impacts of IAS include changes of abiotic and biotic environments (Strayer, 2010) and community structure (Kelly *et al.*, 2003) including in extreme cases, the extinction of native species (Clavero & García-Berthou, 2005). IAS have also been implicated in driving changes in ecosystem function (Vander Zanden *et al.*, 1999; Crooks, 2002; Vander Zanden *et al.*, 2003), which has altered the capacity of natural ecosystems to provide ecosystem goods and services for human use (Charles & Dukes, 2007), with direct financial implications (Pimentel *et al.*, 2001). Although typically considered in a negative context, some non-native species may facilitate native species by providing new habitat, food sources or enemy-free space (Rodriguez, 2006), e.g. through the invasion by autogenic ecosystem engineers (Jones *et al.*, 1994).

Invasion is typically one of a series of factors affecting biotic and abiotic disturbance of natural ecosystems (Didham *et al.*, 2005; Dudgeon *et al.*, 2006; Strayer, 2010) that need to be considered by ecosystem managers. Like similar habitats elsewhere (Brönmark & Hansson, 2002; Dudgeon *et al.*, 2006), Irish freshwaters face the combined effects of habitat degradation, human regulation of water levels, water extraction, nutrient enrichment, overexploitation as well as the introduction of IAS, both accidental and intentional. These factors can interact to modify abiotic and biotic conditions, and may facilitate the successful invasion of non-native species into already stressed ecosystems (Didham *et al.*, 2005; Didham *et al.*, 2007). This further complicates the management of natural ecosystems, where managers have a statutory requirement to maintain, or improve ecological quality (EU, 2000), whilst also reflecting the needs and wishes of stakeholders, including recreational and commercial use and exploitation of aquatic resources (Wilson & Carpenter, 1999). Currently, managers do not have the right tools or access to relevant information to permit the management and control of IAS.

Managers require information on the current distribution of different IAS, as well as the probability of invasion and the likely impacts following the successful establishment of potential IAS. Important factors reflecting the future probability of invasion include current environmental state (see above) and community complexity. There has been a long-running debate within ecology regarding the relative importance of diversity in ecosystem stability (McCann, 2000), and the resistance of ecosystems to invasion. Early ecologists (Elton, 1958) propagated the general view that less diverse communities (e.g. those on islands) were less stable and were therefore more susceptible to invasions compared to more diverse communities (e.g. those located on larger land masses (Paulay, 1994; Lonsdale, 1999)). Although challenged by empiricists and modellers as part of the so called diversity-stability debate (McCann, 2000), there is a general consensus today that the addition or loss of even a single species can have marked effects on community interactions, and therefore ecosystem function. This is of particular importance in island ecosystems, which tend to be less diverse relative to those in adjacent continental areas, and as such are

particularly susceptible to invasion by non-native species. Ireland represents a natural laboratory for the study of the impacts of IAS on receiving ecosystems (Vitousek, 2002).

Ireland is a moderately sized (~85 000 km²) island located off the western extreme of continental Europe, that was fully glaciated during the last glacial maximum (Chiverrell & Thomas, 2010). Due to this glacial history and temperate, wet marine climate, freshwater habitats are a characteristic feature of the often heterogeneous landscape. Ireland's location and glacial history also affected its native biodiversity, which is characteristically depauparate (Wilson, 1986; Griffiths, 1997), indicating potential susceptibility to invasion. The native freshwater fish fauna consist of only 14 euryhaline species that invaded inland lakes, rivers and streams from marine habitats following the retreat of glacial ice. Ireland has undergone a series of historical introductions of non-native species (Thompson, 1856; Went, 1950; Stokes *et al.*, 2006), including 12 species of fish, although the origin and timing of introduction of some species remain unclear (Went, 1950; 1957; Wilson, 1986; Griffiths, 1997). The numbers of freshwater IAS in Ireland have increased markedly in recent years (Griffiths, 1997; Caffrey *et al.*, 2008), with most high-impact invaders being introduced during the last 25 years.

Although invaded, freshwater ecosystems in Ireland continue to support internationally important habitats (EEC, 1992) and species such as Arctic charr (*Salvelinus alpinus*) (Maitland, 1987; Igoe *et al.*, 2001; Igoe *et al.*, 2003) and pollan (*Coregonus autumnalis*) (Harrod *et al.*, 2002). Furthermore, Ireland's freshwaters represent an important tourism resource, and annually attract large numbers of anglers, who contribute substantially to the Irish economy (Solon & Brunt, 2006), but may represent a vector for the spread of IAS (Caffrey *et al.*, 2008). Introductions of IAS in Ireland have been associated with changes in community structure (Dick *et al.*, 1990; Dick, 1996a) including the loss and reduced population size of native fishes (Igoe *et al.*, 2001; Harrod *et al.*, 2002). Although detailed baseline data are not generally available, anecdotal descriptions of fish community structure indicate that after *ca.* 10 000 years of dominance by salmonid and anguillid fishes, the last century has seen many large Irish lakes become dominated by introduced cyprinid and/or percid fishes (Maitland, 1987; Harrod *et al.*, 2002). Lake managers now have to deal with the consequences of interactions between these species, e.g. competition between roach and native species (Winfield *et al.*, 1992). The simplicity of Irish freshwater ecosystems may not only increase their susceptibility to invasion, but may also represent a stage for the generation of novel biological diversity, e.g. hybridisation between non-native bream and roach occurs at an unprecedented frequency in Ireland (Hayden *et al.*, 2010) and the abundance of hybrid progeny can often exceed that of both parental species.

Although invasive species have the potential to impact receiving ecosystems (Crooks, 2002; Ward & Ricciardi, 2007), it can be difficult to actually quantify the direction and intensity of any subsequent ecological change e.g. in ecosystem structure and function (Parker *et al.*, 1999; Strayer *et al.*, 2006). A means of assessing the consequences of IAS that has recently gained wide use in ecology is the use of stable isotope analysis (SIA) (Vander Zanden *et al.*, 1999; Kelly & Hawes, 2005; Bystrom *et al.*, 2007; Britton *et al.*, 2010a). For example, by examining the carbon (δ^{13} C) and nitrogen (δ^{15} N) stable isotope values of consumers and putative food sources, ecologists are able to examine variation in energy utilisation and trophic level within a population (Harrod *et al.*, 2005) or community (Harrod & Grey, 2006). SIA has proved particularly useful in studies examining the consequences of invasion by alien species on the structure and function of aquatic foodwebs (Vander Zanden *et al.*, 1999; Kelly & Hawes, 2005).

This project used SIA to aid our understanding of the ecology of IAS and their impacts (Parker *et al.*, 1999) on receiving Irish freshwater ecosystems as part of a multidisciplinary approach including measures of community structure, trophic ecology and life history responses. We examine the impact of two contrasting IAS on two different river basin districts in Ireland. The first case study examines the ecological impacts of an invasive ecosystem engineer, *Lagarosiphon major* in Lough Corrib, Ireland's second largest lake. The second case study examines the ecology of Ireland's most recent potential invasive alien fish, the chub (*Leuciscus cephalus*) in the River Inny. The rationale for selection of these species and sites is given below in Section 5.2.

5.1 **Project objectives and contribution to national research needs**

The EPA identified the need to improve our knowledge of the impact of IAS on natural ecosystems so that adequate management measures can be put in place. This project contributes to fulfilling this national research need through three objectives relating to the impact of IAS, namely:

- WP2a Objective 1: To describe the key consequences of invasion on the function and food-web dynamics of selected freshwater ecosystems in Ireland including the use of stable isotope analyses.
- WP2a Objective 2: To describe the key consequences of invasion on the community structure of selected freshwater ecosystems in Ireland.
- WP2b Objective 1: To examine the efficacy and consequences of measures aimed to control key invasive species including an invasive plant *Lagarosiphon major*, and an invasive fish (*Leuciscus cephalus*).

5.2 Rationale for site and species selection

The rationale for the approach followed during WP2 was to focus research efforts on high impact species in waterbodies of high ecological value. This reflects the fact that these systems will require management actions in the context of WFD-required programmes of measures. Species of concern were identified by both the EPA and the Invasive Species in Ireland risk assessments have identified the top species of concern (Table 2.1). Given the relatively short term nature of this project, in order to quantify the impact on structure and function of ecosystems, it was necessary to conduct research in waterbodies where the project team held baseline data on the biological parameters of interest. Two species of concern were selected from sites in two separate River Basin Districts (Table 5.1) in order to quantify the impact on invasive species on ecosystem structure and function and to also trial control measures.

Lagarosiphon major, a submerged macrophyte native to southern Africa was first recorded from a single bay in Lough Corrib (Western RBD) in 2005. This fast-growing, stand-forming invasive plant has proven to be a problem e.g. in New Zealand where it has been shown to depress native flora, alter species interactions and preclude angling and recreational use of water bodies (Clayton, 1982; Howard-Williams & Davies, 1988; James *et al.*, 1999; Clayton, 2003), Following its initial appearance in L. Corrib, *L. major* rapidly spread through the upper basin of the lough (Caffrey & Acevedo, 2008), markedly changing the structure of the water column and apparently resulting in the loss of native charophytes. We examined the ecological impacts of the *L. major* invasion of L. Corrib, and the

potential control of this macrophyte due its well-reported ecological effects and potential to spread and impact other waters across Ireland.

In the Shannon IRBD, chub (*Leuciscus cephalus*) were reported to invaded the River Inny in 2005 (Caffrey *et al.*, 2008) and were highlighted as a species of concern due to their potential to act as competitors or predators of native fishes that are ubiquitous to many Irish rivers e.g. *Salmo* spp. Chub are not naturally present in Ireland due to biogeographical reasons, however, if introduced they could be expected to thrive due to the large amounts of suitable riverine habitat. This, combined with the potential opportunity to control an invasion in its initial stages, led to the inclusion of chub within the project as an understanding of the ecological impact of chub and the development of control measures would be of interest to managers across Ireland.

Table 5.1: Waterbodies and species selected for detailed research under WP2.

River Basin District	Waterbody	Designations	WFD risk result	Invasive species present
Western RBD	Lough Corrib	SAC/SPA	1a	Lagarosiphon major
Shannon IRBD	River Inny		1a	Chub

6 Ecological impact of *Lagarosiphon major* in Lough Corrib

6.1 Introduction

Lough Corrib is the second largest lake in Ireland: it is located on the borders of Counties Galway and Mayo (Fig 6.1), with a surface area of 178 km² (Mooney, 1989; Krause & King, 1994; Cannaby, 2005; National Parks and Wildlife Service, 2005). The lough is commonly divided into three sections: the upper, middle and lower lough. The large upper lough is deep and less nutrient enriched; with a maximum recorded depth of 47 m. Littoral areas along the western shore are confined to sheltered bays on siliceous rock. The eastern portion of the lake is less steep and extensive littoral areas are present. The middle lake represents a relatively narrow and shallow corridor that links the upper and lower lakes. It is characterised by countless small islands and prominent reefs. The lower lake is shallow, with few sections supporting a depth greater than 3 m. L. Corrib supports extensive charophyte beds with a number of Chara species mixed with submerged pondweeds (Krause & King, 1994; National Parks and Wildlife Service, 2005). The Chara beds represent an important food source for waterfowl and provide complex habitats for an abundant and diverse macroinvertebrate community and juvenile fishes. Partly due to its charophyte assemblage, L. Corrib is considered to be of major international conservation importance (Ramsar Site No. 846) and includes 14 habitats listed on Annex I of the Habitat Directive (EC, 1992). In addition, three fishes that are listed on Annex II of the Directive, namely Atlantic salmon (Salmo salar), sea lamprey (Petromyzon marinus) and brook lamprey (Lampetra planeri) are present in the lake (O'Keeffe & Dromey, 2004). Other aquatic species listed in Annex II that occur in the Corrib catchment include the freshwater pearl mussel (Margaritifera margaritifera), whiteclawed crayfish (Austropotamobius pallipes) and otter (Lutra lutra) (National Parks and Wildlife Service, 2005). The Lough was designated as a wetland of international importance under the Ramsar convention in 1996 due to its importance to waterfowl. Features of the site which are of greatest interest are the presence of extensive Chara beds and the large wintering populations of wildfowl (e.g. pochard and tufted duck). The Ramsar designation for L. Corrib notes that the lake represents "the best example of a large, limestone lake in Ireland and is one of the best examples in the E.U''.

Lough Corrib represents the principal source of water for domestic and industrial use for the City of Galway, and several other areas of Co. Galway. It also receives treated water from a number of waste water treatment plants. The region attracts large numbers of national and foreign holiday-makers, and tourists spent > €356 million in Co. Galway in 2006 (Fáilte Ireland, 2006). The combination of pristine lakes and wilderness, of which Lough Corrib forms an integral part, attract many of these visitors to the region. However, L. Corrib is a major international tourist destination in itself, as it is an important international and national angling resource (Solon & Brunt, 2006). Long famous (Went, 1942) for its brown trout (*S. trutta*) fishing, today L. Corrib is one of the prime wild brown trout angling lakes in Europe, L. Corrib supports significant stocks of brown trout whilst Atlantic salmon are commonly caught in the lake and use the tributary rivers for spawning purposes. Until recently, commercial netting for European eels (*Anguilla anguilla* L.) was also carried out in the lake and the outflowing Corrib river (Krause and King, 1994).

Although larger than many Irish lakes, L. Corrib can be considered as a good model to examine the response of Irish lakes to invasion by non-native species. The lough is internationally significant for conservation purposes, supports economically-important activities and provides ecosystem goods and services (Costanza *et al.*, 1997) to
Galway, the third largest city in the Republic of Ireland and surrounding areas. Lough Corrib's responses to invasion may therefore extend beyond simple changes in community structure, or ecosystem function, following invasion by IAS through to a loss of conservation value, tourist revenue or key ecosystem services.

In 2005, a single bay located on the western shore of upper L. Corrib was found to include a large (ca. 12 ha) stand of the submerged macrophyte Lagarosiphon major (henceforth referred to as Lagarosiphon). The density was such that it precluded angling or recreational boating and was associated with the almost total loss of native charophyte beds in the infected bay. Lagarosiphon is a member of the Family Hydrocharitaceae, which includes other fastgrowing, large-biomass invading macrophytes such as *Elodea* spp and *Hydrilla* spp. It is native to southern Africa, where it has long been recognised as a problem, as extensive Lagarosiphon beds interfere with boat navigation and recreation (Wager, 1928). Optimal conditions for Lagarosiphon include high light intensity, clear still water with little or no wave action, within a temperature range of 18 - 23 °C. It is a strong competitor under low nutrient and low CO₂ conditions (Rattray et al., 1994) but prevails in high pH water with high nutrient levels. Apart from forming dense stands that restrict the supply of light to the lake bed below, Lagarosiphon also reduces pH and CO₂ concentrations, creating stressful conditions for other submerged vegetation, and it is considered an aggressive competitor (Clayton, 1982; Rattray et al., 1994; James et al., 1999; 2006). The ecological effects of Lagarosiphon on receiving ecosystems have been most studied in New Zealand following its invasion during the 20th Century and indicate that Lagarosiphon has the ability to directly alter ecosystems, species interactions and physical habitat in areas where it has been introduced (Clayton, 1982; Howard-Williams & Davies, 1988; Rattray et al., 1994; Kelly & Hawes, 2005; Bickel & Closs, 2008; 2009). Bickel & Closs (2008) suggested that Lagarosiphon may have been directly subsidising the food web of invaded lakes in New Zealand, however Kelly & Hawes, using stable isotope analysis, showed little evidence that macrophytes from Lagarosiphon-invaded lakes made any significant contribution to the assimilated tissues of fish or invertebrate consumers.

An initial survey following the initial detection of *Lagarosiphon* in L. Corrib in 2005 showed a total of 8 bays to be invaded and a long-term monitoring and control project was initiated by Inland Fisheries Ireland. Field surveys in 2006 revealed the presence of 24 infested sites, whilst following an intensive distribution survey in 2007, a total of 64 invaded bays was recorded (Caffrey *et al.*, 2009). During this same period, the original reported stand of *Lagarosiphon* had extended by ca. 60%, showing its potential for rapid expansion within invaded locations. By 2008, 113 separate invaded sites had been recorded, with a gradual southward shift in distribution toward the as yet *Lagarosiphon*-free lower L. Corrib.

In L. Corrib, seasonal observations have revealed the existence of two distinct parts of the life cycle of *Lagarosiphon* (Caffrey *et al.*, 2009). Unusually for a macrophyte in temperate lakes, the bulk of growth occurs during late autumn, winter and spring (typically October - mid-April). During this period of rapid vertical growth, *Lagarosiphon* stems are erect and buoyant resulting in a structurally-complex water column from lake-bed to the surface, and the formation of a surface canopy. Growth subsequently slows/stops during the summer (May – September), when leaves are shed and the *Lagarosiphon* stems lose their buoyancy, finally collapsing onto the lake bed. During this period, the structurally complex proportion of the water column is restricted to the lower part of the water column. Intermediate stages exist between these two main growth stages, highlighting the dynamic nature of *Lagarosiphon*-invaded habitats.

Changes in the structure of habitats following the invasion of autogenic ecosystem engineers (Jones *et al.*, 1994) *e.g.* submerged macrophytes such as *Lagarosiphon*, can have diverse ecological implications. Carpenter and Lodge (1986) detailed how macrophytes can affect the physicochemical (e.g. light, temperature, hydrodynamics, substrate characteristics, sedimentation rate, nutrient availability) environment of lakes. They also noted the particular effects of macrophytes on nutrient cycling in lakes, due to their often large biomass and nutrient turnover times that are much extended relative to other lake primary producers such as phytoplankton. Submerged macrophytes typically increase habitat complexity in lakes. In turn, this may increase niche diversity, reducing overlap between competitors thus allowing increased production per unit area, which in freshwater lakes typically results in an increase in macroinvertebrate abundance and biomass (Crowder & Cooper, 1982; Diehl, 1992).

Submerged macrophytes represent preferred spawning habitat for many L. Corrib fishes (Maitland & Campbell, 1992) and it is likely that the presence of dense Lagarosiphon stands in spring will improve spawning access for these fishes, once hatched, it is likely that larval and juvenile fishes will display increased survivorship as mortality rates for small bodied fishes tend to be reduced in structured habitats (Werner et al., 1983; Skov & Koed, 2004), leading to these areas acting as predation refugia (Brabrand & Faafeng, 1993; Stansfield et al., 1997). Changes in habitat complexity may affect the abundance and structure of putative prey communities, influencing the capacity of predators to locate and capture prey (Crowder & Cooper, 1982; Manatunge et al., 2000; Warfe & Barmuta, 2006), thus potentially influencing prey survivorship, predator diet, consumption rates and the growth of both predator and prey (Diehl, 1993; Persson & Eklov, 1995). There is empirical evidence that access to submerged vegetation allows fish to feed on more and larger invertebrate prey whilst facing reduced predation risk (Rozas & Odum, 1988). Access to different prey communities may result in marked intraspecific dietary differences between habitats (Svanbäck & Eklöv, 2002), as predators exposed to alternative food types undertake trophic shifts. Such shifts may include the consumption of non-typical prey: for instance pike (Esox lucius) are typically considered obligate piscivores and represent the top predator in many temperate Northern Hemisphere lakes. However, under certain conditions, even large bodied individuals may feed on macroinvertebrates (Treasurer, 1998; Venturelli & Tonn, 2005; Paradis et al., 2008).

Specialisation to specific foraging habitats by consumers can result in phenotypic differentiation between consumers exploiting alternative habitats (Hjelm *et al.*, 2001): these differences are often associated with habitatdependent foraging capacities and growth trajectories (Svanbäck & Eklöv, 2002). Although some examples of ecomorphological specialisation are adaptive (Harrod *et al.*, 2010), non-adaptive phenotypic plasticity remains important as it influences ecological interactions e.g. between predator and prey and therefore to ecosystem function (Miner *et al.*, 2005). Although not studied to date, the capacity for invasive plants to radically affect habitat structure may result in ecomorphological specialisation in consumers.

Apart from the modification of foodwebs through changes in consumer behaviour and community structure the invasion of an autogenic ecological engineer such as *Lagarosiphon* can further modify foodwebs through the introduction of a new source of primary production, providing extra energy and nutrients to resident consumers. Although quantifying the contribution of different sources of primary production to higher trophic levels was traditionally difficult, it has been eased considerably through the use of stable isotope analysis (Hamilton *et al.*, 1992; Beaudoin *et al.*, 2001; Finlay & Kendall, 2007) and isotope mixing models (Phillips & Gregg, 2001; Parnell *et al.*, 2010). Stable isotope analysis also provides a means to compare the long-term assimilation patterns and

trophic ecology of consumers from different habitats (e.g. invaded and non-invaded) at the individual, population and community level (Post, 2002; Grey, 2006; Layman *et al.*, 2007a), and can also be used to provide measures of ecosystem function such as food chain length (Cabana & Rasmussen, 1996).

The fish community of Lough Corrib is relative simple, and although internationally famous for trout fishing (Solon & Brunt, 2006), is currently dominated by non-native fishes including roach (*Rutilus rutilus*) and perch (*Perca fluviatilis*). Pike are also non-native, and probably represent the most numerous putative top predator in L. Corrib where they are considered by local anglers and lake managers to play a regulatory role on resident trout populations (Anon., 1980). Each of these fishes can be considered keystone species in European lakes, as they have been shown to affect ecosystem function or community structure under certain conditions, e.g. by removal of zooplankton by juvenile fish resulting in a shift to algal dominance (perch & roach, Brabrand *et al.*, 1986; Kurmayer & Wanzenböck, 1996) or by size-selective piscivory (Grimm & Backx, 1990; Claessen *et al.*, 2000; Lathrop *et al.*, 2002; Winfield & Durie, 2004; Bystrom *et al.*, 2007).

Roach are unusual in the context of European freshwater fish in that they are generalist omnivores. They switch from a juvenile diet of zooplankton, to a diet dominated by macroinvertebrates as they enter sexual maturity, but have the capacity to feed directly (and grow) on aquatic plants including charophytes and macrophytes (Maitland & Campbell, 1992; Michel & Oberdoff, 1995). The generalist diet, efficient zooplanktivory and capacity to rapidly develop abundant populations are all perceived factors in the capacity of roach to successfully invade lakes outside of their natural distribution (Fitzmaurice, 1981; Rask *et al.*, 2000), presumably through competitive superiority over native or other resident fishes (Bergstrand, 1990; Persson, 1990; Persson, 1991). Perch are carnivorous throughout their life cycle, but follow a marked and well reported ontogenetic dietary shift from zooplanktivory, benthivory through to piscivory (Hjelm *et al.*, 2001; Svanbäck & Eklöv, 2002).

It is possible that the Lagarosiphon invasion has facilitated (Rodriguez, 2006) juvenile roach and perch due to their preference for structured environments (Rossier et al., 1996). However, there are a series of potential ecological interactions between roach and perch that under certain conditions can result in competitive asymmetries between the two species which may be affected by the provision of large volumes of new habitat. As juveniles, both species are obligate zooplanktivores, and the greater relative foraging efficiency of roach on this prey in open water habitats can result in significant interspecific competition between roach and perch (Burrough et al., 1979; Byström & García-Berthou, 1999), forcing perch to make a premature switch to benthivory (Persson, 1986; Persson, 1990). This competition-mediated switch effects a bottleneck to subsequent growth as the small-bodied and gape-limited perch are forced to feed on small-bodied, less profitable prey. Perch forced into a premature switch to benthivory display reduced growth, and are unlikely to reach a size at which they are capable of piscivory (Persson, 1990). As large-bodied perch can be important predators of roach, the capacity of roach to regulate perch populations acts as a positive feedback mechanism on roach population dynamics (Persson, 1991). Under certain conditions however, the direction of the competitive asymmetry between juvenile roach and perch can switch, e.g. in structured habitats such as in macrophyte stands, where perch out-perform roach when foraging for a common prey (Diehl, 1988; 1992; 1993). The development of extensive stands of Lagarosiphon in Lough Corrib may therefore have modified interspecific interactions between roach and perch. Pike are closely associated with submerged vegetation throughout their life-cycle (Bry, 1996; Grimm & Klinge, 1996), using it as a spawning substrate, and foraging habitat as juveniles and adults. As such, it is reasonable to expect pike stocks to respond positively to the *Lagarosiphon* invasion.

It is clearly of concern that the importance of L. Corrib as an area of significant natural heritage and conservation is potentially adversely compromised by the presence and expansion of *Lagarosiphon*, an aggressive invasive macrophyte. This study aims to determine the ecological effects of the invasion of L. Corrib by *Lagarosiphon* at several different levels of biological organisation (Parker *et al.*, 1999). At a broad level, comparisons are made of overall macroinvertebrate and fish community structure in areas that are either dominated by native charophytes or invasive *Lagarosiphon*. Due to their dominance of the L. Corrib fish community and the habitat-mediated competitive asymmetry that exists between the two species, a particular focus is made on the ecology of roach and perch, and possible interactions between the two species. Comparisons are made of various population characteristics (size and age structure) and life-history traits (condition, growth, age at maturity and mortality) from both species from native and invaded habitats. Spatiotemporal comparisons were made of the trophic ecology of both invertebrate and fish consumers including the use of stable isotope mixing models to examine the relative contribution of different primary producers to consumer diet and to examine trophic overlap between roach and perch.

The study aimed to examine the potential impact of *Lagarosiphon* on the L. Corrib ecosystem through the examination of a series of questions:

- 1. Did areas of the lough dominated by native charaphytes (Native) and those invaded by *Lagarosiphon* (Invaded) support different biological communities, i.e. fish and benthic macroinvertebrates?;
- 2. Was there any measurable impacts of invasion by *Lagarosiphon* on the population structure (age, size), and life history characteristics (e.g. growth, mortality rate) of the keystone fish species, roach and perch?;
- 3. Was invasion by Lagarosiphon associated with a shift in the diet and trophic niche of macroinvertebrates or fish?; and
- 4. Did Lagarosiphon make any measurable contribution of carbon or nitrogen to the food web of L. Corrib?

6.2 Materials and methods

To investigate the impact of *Lagarosiphon* on the ecosystem of L. Corrib, a sampling protocol was designed to compare fish and macroinvertebrate communities and possible trophic responses by these consumers to invasion. Fish, macroinvertebrates and aquatic vegetation were sampled by small boat and through SCUBA diving from littoral sites located in eight bays located in the upper basin of the lough (Table 6.1; Fig. 6.1). Sampling was conducted over four separate surveys: June 2008, August 2008, October 2008 and May 2009. Five sites were dominated by invasive *Lagarosiphon* and three dominated by native *Chara*. Some bays dominated by *Lagarosiphon*, were originally designated for control trials allowing the ecological responses to restoration to be examined. However, due to operational reasons, control procedures in these bays did not occur in a consistent manner, and it was not possible to undertake a representative study of the effects of control activities and comparisons are limited to those between invaded and native sites.



Figure 6.1 Map of Lough Corrib, showing location of invaded (*Lagarosiphon*-dominated) and Native (*Chara*-dominated) sites sampled during the current study. See Table 1 for more details.

Table 6.1: Location and habitat status	of the different sites on Lou	gh Corrib sampled durin	g the current study
		U I I	

Site number	Grid reference	Dominant vegetation and control status
1	53°29.02'N 9°23.55'W	Lagarosiphon
2	53°27.44'N 9°20.45'W	Chara
3	53°28.03'N 9°13.29'W	Chara
4	53°28.12'N 9°13.46'W	Lagarosiphon
5	53°29.91'N 9°27.41'W	Lagarosiphon: partially cut prior to August 2008, fully cut prior to October
		2008 survey
6	53°28.75'N 9°22.21'W	Lagarosiphon: cut prior to May 2009 survey
7	53°29.55'N 9°30.43'W	Chara
8	53°29.57'N 9°22.38'W	Lagarosiphon

6.2.1 Macroinvertebrates

Originally, attempts were made to quantitatively collect 3 samples of *Lagarosiphon* (if present) and 3 samples of charophytes and associated macroinvertebrates from a depth of 3 m using a custom made diver-operated sampling apparatus. Unfortunately, weather conditions and benthic visibility were only suitable for this during the August 2008 survey. During other surveys only qualitative sampling was possible, which provided suitable samples of vegetation and invertebrates from *Lagarosiphon* and *Chara*-dominated sites for SIA. Macroinvertebrates were removed from plant material by washing them in fresh water and sieving through a 500 μ m sieve. Macroinvertebrates were placed in labelled polythene bags and frozen at -18°C. Once returned to the laboratory, invertebrates were identified to family level, and enumerated and then placed into a drying oven at 60°C to allow dry mass measurements t o be made (± 0.00001 g). Dried samples were then stored prior to preparation for SIA. Molluscs were removed from shells prior to drying. Samples of both charophyte and *Lagarosiphon* and *Chara* samples were oven dried at 60°C for at least 48 hours. Samples were then divided into two, with half being treated with HCl to remove ¹³C enriched inorganic carbonates for analysis of δ^{13} C (Boutton, 1991). The remaining part of each sample was analysed separately for δ^{15} N, to limit any effect of acidification on δ^{15} N.

Where conditions allowed, zooplankton were collected by horizontal tows of a 25 cm diameter x 50 cm long, 250 μ m mesh zooplankton net. Unfortunately, it was impossible to collect zooplankton from *Lagarosiphon*-dominated habitats as the net rapidly became entangled with debris. Bulk zooplankton samples were frozen at -18°C, and returned to the laboratory where they were dried at 60°C for at least 48 hours prior to SIA.

6.2.2 Fish

Fish were collected using a standardised combination of gillnets and fykenets. The net array consisted of three standard monofilament Collins multi-panel gillnets (length 60 m x height 1.5 m) consisting of 12 x 5 metre panels of different mesh sizes (8, 10, 13, 16.5, 19, 22, 25, 30, 33, 38, 45 and 50 mm knot to knot) and one gang of fyke nets (four nets per gang: each net total length of 12.4 m, 8 m leader, 2.2 m long trap end, and hoop diameters of 0.5, 0.45, 0.35 and 0.28 m, mesh 22 mm knot to knot). Two gill nets were surface set, whilst the remaining gillnet was bottom set, allowing representative coverage of the full water column (depth ca. 3 m). In areas invaded by Lagarosiphon, the benthic gillnet and one pelagic net were set within the Lagarosiphon stands, whilst another pelagic gillnet was set adjacent to the edge of the Lagarosiphon stand. Nets were set in a similar configuration in, over and adjacent to Chara-dominated areas. All nets were deployed at dusk and retrieved at dawn, with soak times of ca. 13 hours in areas with depths ca. 3 m. Nets were returned to the shore for processing. Once removed from nets, fish were identified to species and measured (fork length ± 1 mm, blotted wet mass ± 0.1 g). Ageing structures were removed from each fish: scales were taken from cyprinids, opercular bones from perch and pike and sagittal otoliths from eels. The maturity status of individual fish was recorded following visual assessment of gonads following Nikolsky (1963). Fish alimentary canals were dissected out and the contents of the first third of the gut (cyprinids) or of the stomach (other fishes) were preserved in 70 % alcohol for further analysis. A sample (ca. 1 g) of dorsal muscle tissue was excised for stable isotope analysis (SIA) of carbon and nitrogen (δ^{13} C and δ^{15} N) and dried at 60°C for at least 48 hours prior to preparation f or SIA.

6.2.3 Stable isotope analysis

All samples for SIA were ground to a fine powder in an agate pestle and mortar, before weighing into tin-cups. Sample mass varied according to the sample type, and were typically ca. 1.5 mg for primary producers and 0.5 mg for animal tissues. Samples were analysed for δ^{13} C and δ^{15} N following standard protocols (e.g. Harrod *et al.*, 2005) at either the Max Planck Institute for Evolutionary Biology, Germany, the CHRONO Centre for Climate, the Environment, and Chronology at Queen's University Belfast or the School of Biological and Chemical Sciences, Queen Mary, University of London. All laboratories used international recognised secondary standards of known relation to the international standards of Vienna Pee Dee belemnite for carbon and atmospheric N₂ for nitrogen allowing direct comparison of results across different laboratories. Typical precision for a single analysis was \pm 0.1‰ for δ^{13} C and \pm 0.3‰ for δ^{15} N.

6.2.4 Data analysis

As noted above, samples were collected over four sampling occasions from a series of geographically distinct sites that were either dominated by invasive *Lagarosiphon* (max n = 5, min = 3) or by native charophytes (n = 3). Unfortunately, due to continued adverse weather conditions that restricted boat use, sample collection was not always possible at all sites within a particular sampling visit. As such, it was decided to pool samples collected from

different *Lagarosiphon*-dominated and *Chara*-dominated sites within each survey. Although this limited the statistical capacity to examine site-specific variation, by pooling samples collected from a number of distinct sites by their habitat state (i.e. invaded v native), we were able to examine the large-scale ecological differences between habitats dominated by invasive or native aquatic vegetation. This approach also allowed us to examine temporal variation in these putative habitat-associated differences across the four separate surveys.

Due to the unbalanced, often non-normal and heteroscedastic nature of the data generated during the study, statistical comparisons were typically made using distance-based permutation-based analysis of variance (PERMANOVA: Anderson *et al.*, 2008) within the PRIMER 6 framework (Clarke & Gorley, 2006). PERMANOVA is largely analogous to parametric ANOVA or MANOVA, but is robust to deviations from the assumptions of parametric ANOVA (i.e. does not require normality of errors, or homogeneity of variance between groups), and probability (P) values are calculated through permutation. Permutation based tests represent a particularly robust means of examining interaction terms (Fraker & Peacor, 2008). Here, P values were estimated from 10 000 permutations. In each case data were treated by using relevant transformation (i.e. $Log_{10}+x$, square-root) prior to production of similarity matrices. Bray-Curtis distance measures were used for analyses of macroinvertebrate and fish community structure, as well as of fish diet, whilst Euclidian distances were used for all other analyses. In most analyses (i.e. where data were sufficient) we compared the effect of capture habitat (Invaded v Native) on the variable of interest, and how this varied temporally (sampling occasion), and the interaction between habitat type and time. Where data were not sufficient to examine temporal variation, samples were pooled by collection habitat allowing a general comparison between invaded and native habitats.

Comparisons of both macroinvertebrate and fish community structure and fish diet between habitats and survey dates were made using PERMANOVA (as described above) of both abundance and biomass data. Significant differences were examined through the use of SIMPER analysis in PRIMER 6 (Clarke & Gorley, 2006). Due to the known potential for and interspecific dietary competition between roach and perch, the scale of trophic overlap between the species was also examined using PERMANOVA.

6.2.4.1 Fish

Roach were aged from scales, which were viewed using a microfiche, and the anterior radius of the scale measured, along with the number and distance of annuli from the scale focus (Wallin, 1957). Perch were aged following a similar approach using opercular bones (Le Cren, 1947). Fish were placed into age-cohorts with an estimate hatch date of 1 April. In both cases, regression of length at capture on the radius of the ageing structure (data log₁₀-transformed) revealed a slope that differed significantly from unity, showing an allometric relationship between scale/opercular size and fish length. As such, back-calculated lengths at age of individual fish were calculated using the non-linear method originally attributed to Manastryrsky (Francis, 1990). Perch and roach growth rates were compared between habitat types and age cohorts through analysis of back-calculated length at age data in PERMANOVA (data log₁₀-transformed prior to analysis). The condition of roach and perch was estimated through the calculation of residuals of species-specific pooled mass-length regressions (data log₁₀-transformed) (Schulte-Hostedde *et al.*, 2005; Britton *et al.*, 2010a) and compared between habitats and across different surveys. Overall habitat-specific estimates of total adult mortality rates in roach and perch populations captured from native or invaded habitats were generated using age-based catch curves generated from mean Ln-transformed catch per unit effort (CPUE) data (Ricker, 1975; King, 1995).

Length at sexual maturity (L_{mat}) for roach and perch was estimated by calculating the length at which 50 per cent of all perch and roach were sexually mature – taken as when 50 % of all individuals of a certain length had gonads at maturity stage IV or greater (Nikolsky, 1963). Logistic curves were subsequently fitted to the data to allow calculation of mean length at sexual maturity for roach and perch captured from native or invaded habitats (King, 1995).

We utilised landmark-based geometric morphometrics (Parsons *et al.*, 2003; Adams *et al.*, 2004) to characterise shape variation in roach and perch from invaded and native habitats and to examine the existence of putative morphological differences in fish captured from contrasting habitats (Harrod *et al.*, 2010). Images were collected of the left hand flank of each specimen using a Canon EOS D300 digital SLR camera, fitted with an EF-S 18-55mm f/3.5-5.6 lens, mounted on a copy stand with optimal shutter speed and aperture width. Using the program tpsDig 2 (Rohlf, 2008) representative landmarks were placed on images. A total of 17 landmarks were used for perch and 15 landmarks used for roach. Landmark data were then inputted into Morphoj v1.01C (Klingenberg, 2010) where a generalised Procrustes analysis was conducted. Centroid size influenced shape in both roach (R² = 0.09, P < 0.0001) and perch (R² = 0.15, P < 0.0001) indicating ontogenetic variation in shape. We therefore controlled for this by using size-corrected Procrustes distances (Klingenberg, 2010). We examined shape differences between habitats using Discriminant Function Analysis (DFA) in both roach and perch. To demonstrate shape variation along the first discriminant function axis (DFA1), we produced thin-plate deformation grids in MorphoJ.

Diet analysis through direct examination was constrained as many fish examined had empty guts or stomachs. Of those fish with prey in their alimentary canal, diet was examined by calculating the percentage frequency of different prey taxa (Hyslop, 1980). The contents of the first third of the gut (cyprinids) or of the stomach (all other species) were examined under a low-power binocular microscope: prey were identified to the lowest practical taxonomic level and enumerated.

6.2.4.2 Stable isotope data

Stable isotope data were analysed in various ways. Stable isotope ratios were compared between key primary producers, as well as macroinvertebrate and fish consumers collected in invaded or native habitats through multivariate PERMANOVA of δ^{13} C and δ^{15} N. A multivariate approach was preferred over simple univariate comparisons of individual stable isotopes as variation in the location of centroids of δ^{13} C and δ^{15} N values are ecologically informative, providing information on energy source and trophic position. All δ^{13} C values from fish were adjusted prior to analysis to correct for lipid effects following Kiljunen *et al.* (2006). Isotope data were transformed (δ^{13} C log₁₀x+40; δ^{15} N log₁₀x) to stabilise variance prior to analyses.

Individual consumer trophic position (TP) was estimated isotopically following Post (2002). Baseline $\delta^{15}N$ values were estimated by pooling $\delta^{15}N$ values from different putative baseline indicators. $\delta^{15}N$ values were estimated directly from *Lagarosiphon* and *Chara*, whilst epiphyton and phytoplankton $\delta^{15}N$ values were calculated from $\delta^{15}N$ values of grazing or filter feeding molluscs and zooplankton respectively following adjustment for trophic fractionation (Post, 2002; McCutchan *et al.*, 2003). Log₁₀-transformed TP values for all macroinvertebrates, fish and several key taxa were compared between habitats and surveys using PERMANOVA.

Following Bearhop *et al.* (2004), who suggested that isotopic variance could be used as a measure of trophic niche width, Layman *et al.* (2007a) proposed the use of the area of consumer (or community) δ^{13} C- δ^{15} N convex-hull area as a means of examining niche variation. We compared pooled isotopic variation in macroinvertebrate (n taxa = 8) and fish consumers (n taxa = 4) collected from native and invaded habitats by calculating convex hull areas using ImageJ (Abràmoff *et al.*, 2004) and compared relative niche space through Wilcoxon tests. The relative contribution of different primary producers (*Chara, Lagarosiphon*, epiphytic algae and phytoplankton) to consumer assimilated diet was estimated from δ^{13} C and δ^{15} N data through using SIAR, a Bayesian mixing model (Parnell *et al.*, 2010) running within the R statistical framework (R Development Core Team, 2009). Source values were calculated for each different primary producer over each of the survey periods, using relevant trophic fractionation correction factors for macroinvertebrate and fish consumers (McCutchan *et al.*, 2003).

6.3 **RESULTS**

6.3.1 Comparisons of community structure

6.3.1.1 Macroinvertebrate community structure

As noted above, it was only possible to quantitatively sample macroinvertebrates during a single survey (August 2008). During this period, both total abundance (Fig. 6.2A; t-test of log10-transformed data: t = 2.49. d.f. = 11.8, P = 0.028) and biomass (Fig. 6.2B: t = 2.23, d.f. = 11.7, P = 0.046) of vegetation-associated macroinvertebrates per unit area of lough bed were greater in *Lagarosiphon* than in *Chara* dominated habitats. Multivariate comparisons of macroinvertebrate community structure differed between sites dominated by invasive *Lagarosiphon* and those dominated by native charophytes when both abundance data (PERMANOVA: Pseudo-F_{1,12} = 3.4, P = 0.0006) and biomass data (Pseudo-F_{1,12} = 3.9, P = 0.005) were considered. SIMPER analysis indicated that certain taxa were more abundant in invaded habitats (in rank order of dissimilarity: *Bithynia* spp., *Crangonyx pseudogracilis, Pisidium* spp., *Dreissena polymorpha, Segmentina complanata*, Corixidae, Odonata, Siphlonuridae, *Psychomyiidae*, Acari), whilst other taxa (*Planorbis vortex, Gammarus duebeni celticus, Lymnea peregra, Asellus aquaticus*) were more abundant in habitats dominated by native vegetation.



Figure 6.2: Comparison of total macroinvertebrate A) abundance and B) biomass in 0.25 m² Lagarosiphon (invaded) and *Chara* stands during August 2008.

6.3.1.2 Fish community structure

The fish community as sampled during the current study was typical for a lowland Irish lake (Tables 6.2 and 6.3). Survey catches in both invaded and native habitats were dominated by roach and perch, with lesser contributions made by other species (pike, eel, bream (*Abramis brama*), rudd (*Scardinius erythrophthalmus*), roach x bream hybrids, brown trout, and salmon). The contribution to total survey catch made by brown trout was extremely low, with a mean \pm SD hourly capture rate of 0.003 \pm 0. 01 individuals per hourly set of the survey nets. This unfortunately precluded a detailed study of the impacts of *Lagarosiphon* invasion on this important species. Both three-spined (*Gasterosteus aculeatus*) and nine-spined (*Pungitius pungitius*) sticklebacks were encountered during the study (*i.e.* observed by divers, collected inadvertently in vegetation samples), but were not collected by the sampling gears used here.

	Abunda	ince (indiv	iduals net habitat	⁻¹ .h ⁻¹) in inv	vaded	Abunda	ance (indiv	viduals ne habitat	et ⁻¹ .h ⁻¹) in 1	native
Species	June-08	Aug-08	Oct-08	May-09	All	June-08	Aug-08	Oct-08	May-09	All
	1.27	0.6	0.11	4.83	1.43	2.0	0.94	0.19	1.36	1.12
Roach	±1.4	±0.76	±0.19	±4.78	±2.49	±1.63	±0.79	±0.28	±1.63	±1.34
	1.55	1.3	0.02	0.64	1.09	0.9	0.61	0.21	0.83	0.63
Perch	±3.13	±1.72	±0.03	±0.72	±2.20	±0.79	±0.66	±0.29	±1.01	±0.76
	0.04	0.06	0.03	0.02	0.04	0.12	0.06	0.02	0.02	0.06
Pike	±0.05	±0.09	±0.03	±0.03	±0.06	±0.14	±0.07	±0.04	±0.04	±0.1
	0	0	0	0	0.	0.11	0.14	0	0.01	0.06
Bream	±0.01	±0	±0	±0	±0.01	±0.19	±0.26	±0	±0.03	±0.2
Roach x	0	0.01	0	0.05	0.01	0.03	0.1	0.03	0.06	0.05
bream	+0.01	+0.03	+0	+0.07	+0.03	+0.05	+0.18	+0.00	+0.00	+0.00
hybrid	10.01	10.00	10	10.07	10.05	10.00	10.10	10.04	10.11	±0.1
	0.07	0.21	0.01	0.03	0.09	0	0.12	0.01	0.03	0.04
Eels	±0.17	±0.53	±0.02	±0.06	±0.31	±0	±0.16	±0.02	±0.05	±0.1
	0	0.01	0	0.04	0.01	0	0	0.01	0.3	0.08
Rudd	±0	±0.02	±0	±0.09	±0.03	±0	±0	±0.02	±0.91	±0.5
	0	0.01	0	0.01	0.	0	0	0	0	0
Brown trout	±0.01	±0.02	±0	±0.02	±0.01	±0	±0	±0.01	±0.01	±0.01
	0	0	0	0	0.	0	0	0	0	0
Salmon	±0.01	±0	±0	±0	±0.01	±0	±0	±0	±0	±0
	2.94	2.2	0.17	5.61	2.70	3.16	1.97	0.47	2.62	2.05
All fishes	±4.14	±2.3	±0.21	±4.96	±3.68	±2.52	±1.39	±0.61	±2.57	±2.12

Table 6.2: Variation in mean (\pm SD) abundance of fishes captured of fishes per net hour in *Lagarosiphon* (invaded) and *Chara*-dominated (native) habitats (all gears combined).

	Bioma	ass (g net	⁻¹ .h ⁻¹) in ir	nvaded ha	bitat	Biomass (g net ⁻¹ .h ⁻¹) in native habi					itat
Species	June-08	Aug- 08	Oct-08	May-09	All		June-08	Aug-08	Oct-08	May-09	All
	217.7	105.6	54	941.9	251.7		376.5	376.5	51.4	273.9	201.5
Roach	±240	±120.6	±115.2	±869.3	±433.3		±442.1	±442.1	±86.3	±301	±295.0
	208.6	199.2	10.9	95	159.5		326.7	326.7	87.8	81.2	136.7
Perch	±317.4	±285.7	±26.5	±92.1	±262.8		±287.7	±287.7	±182.7	±101.9	±206.0
	12.3	19	8.6	26.4	15.7		79.4	79.4	20.3	32.6	46.5
Pike	±19.7	±32.2	±10.5	±58.4	±29.5		±176	±176	±32.6	±89.1	±104.1
	6.1	0	0	0	2.4		68.9	68.9	0	3.6	30.1
Bream	±23.4	±0	±0	±0	±14.7		±134.9	±134.9	±0	±10.9	±80.3
Roach x Bream	0.8	6.6	0	38.5	7.5		115.4	115.4	19.8	37.7	51.3
hybrid	±3.3	±16.8	±0	±65.6	±26.7		±332.6	±332.6	±37.2	±61.3	168.3
	19.4	30.7	2.6	0	17.8		0	0	1.3	3.5	7.5
Eels	±44.1	±104.9	±6.4	±0	±64.4		±0	±0	±3.8	±7.2	±18.0
	0	0.5	1.5	2.1	0.7		0	0	1	0.2	0.3
Rudd	±0	±1.4	±2.5	±4.7	±2.1		±0	±0	±2.2	±0.7	1.2
	5.4	0.6	0	3.1	2.8		0	0	0.8	0.1	0.2
Brown trout	±20.7	±2.1	±0	±7	±13.2		±0	±0	±2.3	±0.3	1.2
	0	0	0	0	0.01		0	0	0	0	0
Salmon	±0.1	±0	±0	±0	±0.1		±0	±0	±0	±0	±0
	470.3	362.3	77.5	1107.1	458.0		967	967	182.5	432.8	474.2
All fishes	±557.9	±377.5	±139.3	±833.7	±566.9		±1064.9	±1064.9	±301.3	±378.7	±646.3

Table 6.3: Variation in mean (± SD) biomass of fishes captured per net hour in *Lagarosiphon* (invaded) and *Chara*-dominated (native) habitats (all gears combined).

Net type was shown to have a significant effect on survey catch structure for both abundance (Pseudo- $F_{2,61} = 17.3$, P = 0.0001) and biomass data (Pseudo- $F_{2,61} = 34.8$, P = 0.0001). Post-hoc comparisons showed that in all cases, survey catches were similar between surface and bottom set gillnets (P ≥ 0.57), but that catches from fykenets differed significantly (P = 0.0001). SIMPER analyses revealed that these differences reflected the increased capture of eels in fyke nets. Due to this gear effect, further comparisons of fish community structure were made separately for gillnets and fyke nets.

Across the complete study, total gillnet capture rates were similar in both habitats (abundance: Pseudo- $F_{1,42} = 1.63$, P = 0.16; biomass: Pseudo- $F_{1,42} = 1.9$, P = 0.15), but varied significantly between different survey dates (abundance: Pseudo- $F_{3,42} = 6.9$, P = 0.0001; biomass: Pseudo- $F_{3,42} = 4.0$, P = 0.001). There was no evidence of an interaction between habitat and survey date indicating that fish abundance and biomass varied seasonally in a similar manner between invaded and native habitats (abundance: Pseudo- $F_{3,42} = 1.1$, P = 0.40; biomass: Pseudo- $F_{3,42} = 1.88$, P = 0.07).

Two-way comparisons of gillnet survey catch examining the influence of habitat and survey date revealed that capture habitat had no or little effect on fish community structure (abundance: Pseudo- $F_{1,41} = 1.4$, P = 0.23; biomass: Pseudo- $F_{1,41} = 2.6$, P = 0.06). Fish community structure did vary by survey date (abundance: Pseudo- $F_{3,41} = 5.1$, P = 0.0001; biomass: Pseudo- $F_{3,41} = 3.0$, P = 0.005), however there was no interaction between capture habitat and survey date suggesting that the fish community varied in a similar fashion in native and invaded habitats (abundance: Pseudo- $F_{3,41} = 1.1$, P = 0.39; biomass: Pseudo- $F_{3,41} = 0.8$, P = 0.65). Comparisons of total fyke net catches showed no evidence for differences between habitats (abundance: Pseudo- $F_{1,17} = 0.64$, P = 0.48; biomass: Pseudo- $F_{1,17} = 0.67$, P = 0.42). There was no strong effect of survey date on fyke net catch rate in terms of either abundance (Pseudo- $F_{3,17} = 2.54$, P = 0.08) or biomass: Pseudo- $F_{3,17} = 0.95$, P = 0.44), nor interaction effect of

habitat and survey date on total catch rate (abundance: Pseudo- $F_{3,17} = 0.34$, P = 0.79; biomass: Pseudo- $F_{3,17} = 2.85$, P = 0.07). There was little evidence of major differences in fish community structure (as assessed by fyke nets) between different habitat types (abundance: Pseudo- $F_{1,7} = 1.7$, P = 0.18; biomass: Pseudo- $F_{1,8} = 3.8$, P = 0.09). There was an effect of sampling date on the structure of fyke net catches (abundance: Pseudo- $F_{3,7} = 4.2$, P = 0.005; biomass: Pseudo- $F_{3,8} = 4.6$, P = 0.03), but no interaction between habitat and survey date (abundance: Pseudo- $F_{3,7} = 1.9$, P = 0.09; biomass: Pseudo- $F_{3,8} = 3.5$, P = 0.11).

6.3.1.3 Fish life history characteristics

Fish size

The size structure of the roach populations captured from the two different habitats was multimodal, showing the presence of multiple age cohorts (Fig. 6.3). PERMANOVA revealed differences in roach length between capture habitats (Pseudo- $F_{1,2550} = 9.13$, P = 0.002), with roach from invaded habitats tending to be slightly larger on average (median $FL_{Invaded} = 185$ mm, median $FL_{Native} = 175$ mm). Survey date had a greater influence on roach size (Pseudo- $F_{3,2550} = 30.8$, P = 0.0001). A significant interaction between these two factors revealed that the size structure of roach from invaded and native habitats showed separate temporal patterns (Pseudo- $F_{3,2550} = 11.97$, P = 0.0001).



Figure 6.3: Habitat-based and temporal variation in the size structure of Lough Corrib roach captured from invaded (red bars) and native (blue bars) habitats from the four separate surveys. Note the use of different scales on vertical axes.

There was no large overall effect of habitat type on perch length (Fig. 6.4: Pseudo- $F_{1,1388} = 3.5$, P = 0.06) but size varied considerably across the different survey dates (Pseudo- $F_{3,1388} = 41.9$, P = 0.0001). A significant interaction between these two factors revealed that temporal variation in the size structure of perch from invaded and native habitats differed between the two capture habitats (Fig. 6.4: Pseudo- $F_{3,1388} = 22.8$, P = 0.0001).



Figure 6.4: Habitat-based and temporal variation in the size structure of Lough Corrib perch captured from invaded (red bars) and native (blue bars) habitats from the four separate surveys. Note the use of different scales on vertical axes.

Fish age

Comparisons of the age structure of roach and perch populations captured from the two habitat types showed little evidence for habitat-based differences (Fig. 6.5). Roach age structure was similar in both capture habitats (Pseudo- $F_{1,152} = 1.14$, P = 0.29) but differed by survey date (Pseudo- $F_{3,152} = 5.8$, P = 0.001). There was an interaction between capture habitat and survey date (Pseudo- $F_{3,152} = 2.98$, P = 0.03) indicating that different age classes of roach may have utilised the different habitats at different times of the year. Perch age structure was similar in both capture habitats (Pseudo- $F_{1,20} = 0.9$, P = 0.41) and across the four surveys (Pseudo- $F_{3,20} = 0.28$, P = 0.28). There was no interaction between capture habitat and survey date (Pseudo- $F_{3,20} = 0.36$, P = 0.96). The roach population was dominated by age 4 individuals in both habitats (Fig. 6.5A), indicating variable recruitment between cohorts. The age structure of the perch population was noticeably more balanced (Fig. 6.5B), with similar contributions made by a range of different age classes, suggesting consistent recruitment over time.



Figure 6.5: Comparison of mean (\pm 95 % CI) contribution of different age classes to A) roach and B) perch populations inhabiting invaded (red bars) and native habitats (blue bars).

Fish growth

Comparisons of back-calculated length at age data showed that roach collected from habitats dominated by invasive Lagarosiphon showed similar growth patterns to those captured from habitats dominated by native Chara spp.(Table 6.4A; Fig. 6.6A), e.g. mean fork length attained in the first year of life was ca. 50 mm. There was evidence for interannual variation in roach growth, with significant differences shown in size at age 2, 3, 4 and 5. The lack of a significant interaction between capture habitat and age cohort reinforces the lack of overall differences in growth patterns in roach captured in the two different habitats. Conversely, a similar analysis indicated that perch growth rates differed significantly between the two capture habitats at ages 1 to 5 (Table 6.4B; Fig. 6.6B), and that perch captured from Lagarosiphon-dominated habitats had larger back-calculated lengths at age than those from Chara-dominated habitats. The length attained by perch in years 1-5 also varied by cohort, showing interannual variation in growth performance. A significant interaction between capture habitat and age cohort in years 1-4 indicates that interannual variation in growth differed between perch captured from native and invaded habitats. When growth was compared in perch that recruited in years following the invasion by Lagarosiphon the differences between habitats became less clear: length at age 1 did not differ between habitats (Fig. 6.6B: Pseudo-F_{1.462} = 2.0, P = 0.159), nor by cohort (Pseudo-F_{3.462} = 1.6, P = 0.17). Furthermore there was no interaction between habitat and cohort (Pseudo-F_{3,462} = 1.72, P = 0.161). There was a habitat-associated difference in perch length at age 2 in those cohorts originating in years following the Lagarosiphon-invasion (Pseudo-F_{1,296} = 10.8, P = 0.001), although there was differences between cohort (Pseudo- $F_{2,296}$ = 4.13, P = 0.02), and an interaction between cohort and habitat $(Pseudo-F_{2,296} = 11.9, P = 0.001).$

Table 6.4: Variation in mean back-calculated fork length at age for A) roach and B) perch captured from native or invaded habitats. P values reflect the results of PERMANOVA tests examining differences in size at age between habitats, between different age cohorts and the interaction between these terms. Horizontal bars (—) depict comparisons that were not possible due to data limitations.

			A) Ro	ach		B) Perch				
	Invaded	Native	Habitat	Cohort	Interaction	Invaded	Native	Habitat	Cohort	Interaction
A go 1	49 ± 11	53 ± 14	D - 0.67	D = 0.67	P = 0.27	76 ± 11	71 ± 10	P = 0.0001	P = 0.00	B _0.0001
Age I	(401)	(229)	F = 0.07	F = 0.07	P = 0.27	(542)	(383)	F = 0.0001	F = 0.09	F =0.0001
A	98 ± 19	100 ± 23		D 0.001	D 0 106	119 ± 16	116 ± 14	D 0.0001	D 0.005	
Age 2	(398)	(227)	P = 0.26	P = 0.001	P = 0.106	(425)	(332)	P = 0.0001	P = 0.005	P = 0.002
A	153 ± 26	149 ± 31	D 0.50	D : 0.001		163 ± 19	158 ± 17		D 0.0001	
Age 3	(362)	(194)	P = 0.52	P < 0.001	P = 0.52	(355)	(274)	P = 0. 0002	P = 0.0001	P = 0.03
A	192 ± 21	188 ± 34	D 0.04	D : 0.001	D 0.71	195 ± 19	188 ± 21	D 0.0001	D 0.0001	
Age 4	(294)	(163)	P = 0.94	P < 0.001	P = 0.71	(303)	(210)	P = 0.0001	P = 0.0001	P = 0.03
	216 ± 36	210 ± 39	D = 0.25	P = 0.01	P = 0.47	215 ± 19	212 ± 20	P = 0.000	P = 0.001	D = 0.22
Age 5	(73)	(76)	P = 0.35	P = 0.01	P = 0.47	(241)	(128)	P = 0.009	P = 0.001	P = 0.33
A 90 6	241 ± 32	232 ± 38	D _ 0.91	D = 0.26		231 ± 18	229 ± 19	P = 0.12	P = 0.07	D = 0.25
Age 6	(44)	(47)	F = 0.01	F = 0.20	F = 0.00	(159)	(74)	P = 0.12	F = 0.07	F = 0.25
A go 7	255 ± 36	249 ± 35	D _ 0.91	D = 0.26		245 ± 21	242 ± 18	D = 0.27	D _ 0.22	P = 0.40
Age /	(19)	(26)	F = 0.01	F = 0.20	F = 0.00	(100)	(40)	F = 0.27	F = 0.55	F = 0.40
	263 ± 37	257± 29				261 ± 20	262 ± 19	D _ 0.97		
Age o	(9)	(13)	_	_	_	(39)	(23)	F = 0.07	_	_
	267 ± 35	264 ± 20				276 ± 25	275 ± 19			
Age 9	(4)	(6)	_	_	_	(14)	(14)	_	_	_
Ago 10	274	279 ± 15				279 ± 20	295			
Age 10	(1)	(4)		_	—	(2)	(1)	_	_	



Figure 6.6: Comparison of mean back-calculated length at age for A) roach and B) perch cohorts captured from native (blue) or invaded (red) habitats during the current study. Filled markers relate to cohorts hatched prior to the *Lagarosiphon* invasion, open markers to those subsequent to the invasion. Numbers to the right of markers reflect different age classes.

Fish condition

Analysis of residuals from the least-squares regression of log_{10} -transformed mass-length data though PERMANOVA showed that condition was similar in roach from both habitat types (Pseudo-F_{1,2268} = 1.11, P = 0.28).

However, condition varied considerably between survey dates (Pseudo- $F_{3,2268} = 66.74$, P < 0.0001), reflecting a seasonal cycle of condition in Lough Corrib roach. A significant interaction between capture habitat and survey date (Pseudo- $F_{3,2268} = 5.11$, P = 0.002) indicated that the seasonal cycle of condition differed in roach captured from the two contrasting habitats. Capture habitat had a small effect on perch condition, with perch from *Chara*-dominated habitats having slightly greater condition on average (Pseudo- $F_{1,1083} = 4.17$, P = 0.04), particularly in June 2008 and May 2009. Sampling period was associated with the most marked variation in perch condition (Pseudo- $F_{3,1083} = 24.1$, P = 0.01). The interaction between capture habitat and survey date was on the threshold of conventional statistical significance (Pseudo- $F_{3,1083} = 2.76$, P = 0.06), indicating that condition cycles were largely similar between the two different habitats.

Size at maturation

Length at maturity was increased in both roach and perch collected from invaded habitats (Fig. 6.7). Roach from *Lagarosiphon*-invaded habitats were ca. 14% larger at maturity compared to their conspecifics from native habitats. Differences were less clear in perch, with a ca. 5% difference in length at maturity between habitats.



Figure 6.7: Comparison of the length at which 50% maturity was reached by A) roach and B) perch from native and invaded habitats during the current study. Logistic curves are fitted to observed data (markers).

Adult mortality rates

Estimates of the instantaneous total adult mortality rates encountered by roach captured from native ($Z = 0.78 \text{ y}^{-1}$) and invaded ($Z = 0.89 \text{ y}^{-1}$) habitats (Fig. 6.8A) indicated that roach mortality rates were higher in the structurally complex *Lagarosiphon*-dominated habitats. Perch instantaneous mortality rates (Fig. 6.8B) were lower than that of roach, but again estimated mortality rates were elevated in perch inhabiting habitats dominated by *Lagarosiphon* ($Z = 0.58 \text{ y}^{-1}$) compared to conspecific from native *Chara*-dominated habitats ($Z = 0.49 \text{ y}^{-1}$).



Figure 6.8: Dynamic catch curve plot estimating mean instantaneous total mortality rate from Ln-transformed mean CPUE of different A) roach and B) perch age groups captured from habitats dominated by native (blue markers) or invasive (red markers) vegetation.

6.3.1.4 Fish shape

Roach shape differed significantly between native and invaded habitats (DFA: $T^2 = 98.4$, P = < 0.0001). Comparison of DF scores (Fig. 6.9A) indicated that more shape variation was displayed by roach captured from habitats dominated by invasive *Lagarosiphon*. Deformation plots (Fig. 6.9A) indicated that roach captured from invaded habitats tended to be deeper bodied, and have reduced caudal peduncle length relative to conspecifics captured from native habitats. Habitat-associated differences in shape were more striking in perch with marked differences in perch captured from *Lagarosiphon* and *Chara*-dominated habitats (Fig 6.9B: $T^2 = 200.0$, P < 0.0001). Deformation plots indicated that perch captured in habitats dominated by native *Chara* tended to have a characteristic pelagic form: they were more fusiform, with relatively elongate caudal peduncles and larger eyes. Perch from structurally-complex invaded habitats were deeper bodied, with smaller eyes, more superior mouthparts, and pectoral fins located anteriorly relative to those from conspecifics captured from native habitats.



Figure 6.9: Variation in A) roach and B) perch shape associated with capture in habitats dominated by invasive or native vegetation (deformation plots = mean value x 4 to highlight differences).

6.3.1.5 Fish diet

Many fish had empty guts or stomachs, limiting sample sizes for comparisons of fish diet between invaded and native habitats. At the whole community level, there was no evidence for an overall difference in fish diet between the two capture habitats (Pseudo $F_{1,588} = 1.02$, P = 0.45), but there was evidence for a strong seasonal influence on diet, with differences in diet between survey dates (Pseudo $F_{3,588} = 8.71$, P = 0.0001). Interestingly, there was a small, but significant interaction between capture habitat and survey date (Pseudo $F_{3,588} = 1.6$, P = 0.03) revealing that seasonal variation in fish diet was different in fish inhabiting *Lagarosiphon* and *Chara*-dominated habitats.

Detailed analyses of diet were restricted to the two most numerous fishes, roach and perch. Comparisons of fish diet using PERMANOVA showed significant overlap in the diet of roach captured in the two contrasting habitat types (Pseudo $F_{1,176} = 1.32$, P = 0.241). However, there was a strong seasonal effect on roach diet (Pseudo $F_{3,176} = 5.12$, P = 0.001): SIMPER analysis revealed that differences were driven by a dominance of cladoceran prey in Visit 1, followed by a switch to benthic macroinvertebrates (particularly gastropods) in subsequent surveys. There was no interaction between capture habitat and survey date, indicating that similar seasonal shifts in diet were displayed by roach from the two habitats (Pseudo $F_{2,176} = 1.17$, P = 0.28). In contrast, perch diet differed considerably between the two habitats (Pseudo $F_{1,333} = 4.49$, P = 0.002), although sampling date was associated with a larger level of dietary variation (Pseudo $F_{3,333} = 11.06$, P = 0.0001). There was a small (i.e. on the threshold of conventional statistical significance) interaction between capture habitat and survey date on perch diet (Pseudo $F_{2,333} = 1.8$, P = 0.05) indicating that although there small seasonal differences in perch diet in *Lagarosiphon* and *Chara*-dominated habitats. SIMPER analysis indicated that the inter-habitat differences in perch diet were driven by increased consumption of cladoceran zooplankton, *Mysis relicta, C. pseudogracilis*, and *G. duebeni celticus* in *Lagarosiphon*-dominated habitats, and increased consumption of larval Ephemoptera, Trichoptera, and Coleoptera, chironomid larvae and pupae and *A aquaticus* in *Chara*-dominated habitats.

Comparisons of the influence of capture habitat and survey date on the level of dietary overlap between roach and perch were made using PERMANOVA. As assessed by GCA, the two species had distinct diets in Lough Corrib (Pseudo- $F_{1,519} = 14.87$, P = 0.0001). The influence of habitat on the diet of roach and perch was less marked than that of species (Pseudo- $F_{1,519} = 5.3$, P = 0.001). However, there was a significant interaction between species and capture habitat (Pseudo- $F_{1,519} = 9.83$, P = 0.0001), probably driven by the inter-habitat differences in perch diet.

Dietary comparisons in the other fishes encountered during the current study were constrained by the number of non-empty stomach or guts encountered. Dietary comparisons could only be made across habitat types for pike, bream, eels whilst insufficient numbers of rudd, roach x bream hybrids, brown trout or salmon contained prey to permit analyses. There was no evidence for inter-habitat differences in pike diet from stomach contents (Pseudo $F_{1,23} = 1.78$, P = 0.12): pike stomach contents included a mixture of fish (perch, 9 and 3 spined stickleback, perch, trout, eel) and invertebrate prey, with the invertebrate component dominated by Odonata and *A. aquaticus*). Of the few bream (n = 6) with gut contents, there was complete overlap in diet between habitats (Pseudo $F_{1,4} = 1.0$, P = 0.41). Statistical comparisons of diet of the 38 eels examined were on the threshold of statistical significance (Pseudo $F_{1,36} = 1.9$, P = 0.07). Eel stomach contents were dominated by invertebrates (chironomid larvae, gastropods, isopods, amphipods, and trichopteran larvae) but also included perch and fish eggs.

6.3.2 Stable isotope analysis

The distribution of δ^{13} C and δ^{15} N values for primary producers and consumers sampled from Lough Corrib was typical of that from temperate freshwater lakes (Fig. 6.10). δ^{13} C values varied between -35.6 and 19.6 ‰ and δ^{15} N values ranged between 0.8 and 16.9 ‰. Detailed analyses are reported below for different taxonomic groups.



Figure 6.10: Variation in carbon (δ^{13} C) and nitrogen (δ^{15} N) values for primary producers and consumers sampled from Lough Corrib during the current study. Convex hulls encompass individuals belonging to different taxonomic groups. The value for the otter reflects a young road kill individual sampled from the West shore of the Lough in June 2008.

6.3.2.1 Baseline isotopic variation

Stable isotope ratios (δ^{13} C from acid-treated samples, δ^{15} N from untreated samples) were measured directly from *Lagarosiphon* and *Chara* over each of the sampling dates. Other macrophytes were present in the survey area; however, they were extremely patchily distributed and where present, were in relatively small amounts. Values for other key putative sources of primary production were estimated indirectly through the analysis of filter feeding bivalves (phytoplankton) and grazing gastropods (epiphytic algae) following Post (2002). These data were used to estimate baseline δ^{15} N values (to calculate TP) and the relative contribution of different primary producers to consumer assimilated diets (see below).

There was considerable overlap in both δ^{13} C and δ^{15} N values (Fig. 6.11) between *Chara* and *Lagarosiphon* which dominated the survey areas sampled during the current study. However, *Lagarosiphon* tended to be ¹³C and ¹⁵N enriched relative to *Chara* (Fig. 6.11A). Evidence for isotopic differences between the two vegetation types was provided by PERMANOVA, which revealed strong species-specific differences in location of the δ^{13} C- δ^{15} N centroids of *Chara* and *Lagarosiphon* (Pseudo-F_{1,89} = 14.8, P = 0.0001). There was a significant effect of survey date (Pseudo-F_{1,89} = 6.6, P = 0.0001) but no evidence for an interaction between these two main effects (Pseudo-F_{3,89} = 1.6, P = 0.13), indicating that the seasonal shifts in the location of δ^{13} C- δ^{15} N centroids were similar for both species.



Figure 6.11: Variation in stable isotope ratios (δ^{13} C & δ^{15} N) in *Lagarosiphon* and *Chara* sampled during the current study.

6.3.2.2 Isotopic baselines for estimation of TP

In order to estimate baseline $\delta^{15}N$ for the calculation of consumer TP (Post, 2002), we compared $\delta^{15}N$ values measured directly from *Lagarosiphon* and *Chara*, and estimated $\delta^{15}N$ values for epiphyton and phytoplankton from fractionation-corrected $\delta^{15}N$ values of gastropod and bivalve consumers respectively (Fig. 6.12). PERMANOVA of log₁₀-transformed baseline respectively showed no overall differences in baseline $\delta^{15}N$ between habitats (Pseudo- $F_{1,531} = 1.5$, P = 0.43). However, baseline $\delta^{15}N$ values varied considerably between survey dates (Pseudo- $F_{3,531} = 22.9$, P = 0.0001) with a marked ¹⁵N enrichment in October 2008, whilst a significant interaction between habitat and survey date (Pseudo- $F_{3,531} = 4.5$, P = 0.004) showed that the temporal shifts in baseline $\delta^{15}N$ values differed between habitats dominated by invaded and native vegetation. As such, estimates of baseline $\delta^{15}N$ for calculation of consumer TP (Post, 2002) were calculated separately for each habitat and sampling date.

6.3.2.3 Macroinvertebrate consumers

Although there were clear isotopic differences between *Chara* and *Lagarosiphon* (see above), the $\delta^{13}C-\delta^{15}N$ centroids of macroinvertebrates from invaded and native habitats overlapped (Fig. 6.12A: Pseudo-F_{1,3.6} = 0.87, P = 0.87). Isotopic variance (as estimated from isotope convex hulls, Layman *et al.*, 2007a) exhibited by macroinvertebrates from the two habitats was identical (Fig. 6.12A). There was a strong temporal effect however, with marked differences in macroinvertebrate $\delta^{13}C-\delta^{15}N$ values between sampling periods (Pseudo-F_{3,672} = 13.5, P = 0.0001). The strength of the interaction between sampling habitat and period (Pseudo-F_{3,672} = 3.52, P = 0.002) indicated that macroinvertebrates in the two different habitats displayed different temporal isotopic shifts.



Figure 6.12: A) Isotopic biplot comparing variation in δ^{13} C and δ^{15} N in Lough Corrib macroinvertebrates collected from *Lagarosiphon*-dominated (invaded) and *Chara*-dominated habitats during the current study. Temporal variation in B) δ^{13} C and C) δ^{15} N are also shown.

The relative contribution of four different primary producers (*Chara, Lagarosiphon*, phytoplankton and epiphyton) to the δ^{13} C and δ^{15} N assimilated by the pooled macroinvertebrate community was examined through the use of an isotopic mixing model. It indicated (Table 6.5) considerable temporal flexibility in the utilisation of different sources of C and N. However, there was less evidence for marked differences between habitats with the bulk of energy and nutrients being sourced through phytoplankton (mean contribution = 51 %) and epiphytic algae (40 %). Overall, neither *Lagarosiphon* (5 %) nor *Chara* (8 %) contributed greatly to the assimilated diet of macroinvertebrates from the two collection habitats. However, both made increased contributions to the diet of macroinvertebrates from native habitats during certain survey periods (*Lagarosiphon* = 15% in June 2008 and *Chara* = 34 % in August 2008).

Survey		A) Inv	aded habitats			B)	Native habitats	
date	Chara	Lagarosiphon	Phytoplankton	Epiphytes	Chara	Lagarosiphon	Phytoplankton	Epiphytes
lun-08	0.05	0.08	0.49	0.38	0.03	0.15	0.44	0.38
5011-00	± 0.08	± 0.07	± 0.20	± 0.19	± 0.05	± 0.06	± 0.15	± 0.15
Aug-08	0.11	0.03	0.68	0.17	0.34	0.03	0.50	0.13
Aug-08	± 0.08	± 0.06	± 0.22	± 0.28	± 0.09	± 0.06	± 0.18	± 0.22
Oct-08	0.04	0.02	0.64	0.30	0.06	0.05	0.36	0.53
001-00	± 0.06	± 0.04	± 0.34	± 0.39	± 0.08	± 0.07	± 0.34	± 0.37
May-00	0.01	0.02	0.45	0.52	0.01	0.02	0.49	0.48
Way-09	± 0.01	± 0.04	± 0.14	± 0.14	± 0.01	± 0.04	± 0.13	± 0.15

Table 6.5: Mean (± 95% Bayesian credibility interval) proportional contribution of different primary producers to the assimilated δ^{13} C and δ^{15} N of macroinvertebrate consumers inhabiting A) *Lagarosiphon* or B) *Chara*-dominated habitats.

6.3.2.4 Fish

Across the study, fish stable isotope values varied considerably, by species (Table 6.6), as well as by capture habitat and by sampling date (see below): δ^{13} C values varied between -34.1 and -23.5 ‰ and δ^{15} N values between 8.2 and 16.9 ‰.

Table 6.6: Summary statistics for δ^{13} C and	d δ ¹⁵ N values	for Lough	Corrib fishes	captured in	Lagarosiphon-
dominated (invaded) and Chara-dominated h	nabitats.				

	δ ¹³ C i	invaded	δ ¹³ C	native	δ ¹⁵ N i	nvaded	δ ¹⁵ N	native
Species	2	Mean	2	Mean	2	Mean	2	Mean
	11	± SD		± SD	11	± SD	11	± SD
9-spine stickleback	1	-29.2	13	-26.4	1	13.5	13	14.0
	1	—	15	± 0.6	1	—	15	± 0.4
Bream	4	-28.3	32	-28.1	4	13.1	32	11.9
	4	± 0.9	52	± 1.1	7	± 1.4	52	± 0.9
Brown trout	6	-27.4	7	-26.8	6	13.9	7	12.9
	0	± 1.0	'	± 2.2	0	± 1.0	'	± 2.7
Eel	21	-27.3	6	-26.5	22	13.3	6	15.1
	51	± 1.1	0	± 1.8	55	± 1.0	0	± 1.5
Roach x bream hybrid	16	-29.6	22	-28.3	16	12.7	22	11.8
	10	± 1.5	22	± 1.0	10	± 0.7	22	± 0.9
Perch	100	-28.7	2/18	-28.3	199	13.7	250	13.2
	100	± 1.0	240	± 1.0	100	± 0.7	230	± 0.9
Pike	37	-27.9	40	-28.1	37	14.0	49	13.8
	57	± 0.7	-5	± 0.9	57	± 0.7	45	± 0.7
Roach	252	-28.7	288	-28.8	224	12.6	280	12.2
	202	± 1.1	200	± 1.4	224	± 0.7	203	± 1.0
Rudd	7	-27.9	7	-29.9	7	13.4	7	13.1
	'	± 1.3	'	± 1.4	'	± 0.4	1	±1.2
Salmon	3	-26.7			3	13.5		
	5	± 1.0		_	5	± 08	_	

When examined as a community, δ^{13} C- δ^{15} N centroids from fish collected from invaded and native habitats were distinct (Fig. 6.13A: Pseudo-F_{1,1171} = 34.3, P = 0.0001), with a much (-60 %) reduced isotopic variation (as measured by convex-hull area) in fishes inhabiting *Lagarosiphon*-dominated habitats. Fish δ^{13} C- δ^{15} N values were variable over the different survey dates Fig. 6.13A-C): Pseudo-F_{3,1171} = 7.4, P = 0.0001), and fish collected from the two different habitats showed different temporal shifts in isotope values (Pseudo-F_{3,1171} = 5.1, P = 0.0003). Univariate comparisons of δ^{13} C values showed no overall effect of capture habitat (Pseudo-F_{1,1171} = 0.09, P = 0.76), but varied by sampling date (Fig. 6.13B: Pseudo-F_{3,1171} = 4.1, P = 0.007). δ^{13} C values recorded from fish captured in *Lagarosiphon* and *Chara*-dominated habitat: Pseudo-F_{3,1171} = 8.2, P = 0.001). Univariate PERMANOVA revealed a large statistical effect of capture habitat on fish δ^{15} N (Pseudo-F_{1,1171} = 50.2, P = 0.0001), with fish from invaded habitats having elevated δ^{15} N values relative to their counterparts from native habitats (Fig. 6.13C). Fish δ^{15} N varied

by survey date (Pseudo- $F_{3,1171} = 9.4$, P = 0.0001), and fish captured from *Lagarosiphon* and *Chara*-dominated habitats showed small, but significant differences in their $\delta^{15}N$ values over the different surveys (Pseudo- $F_{3,1171} = 3.3$, P = 0.02). Although there was a large habitat-associated difference in fish $\delta^{15}N$ values (see above), when corrected for differences in baseline $\delta^{15}N$, this effect was lost, and there was no difference in TP in fish captured from invaded or native habitats (Fig. 6.13D: Pseudo- $F_{1,1171} = 0.1$, P = 0.81). However, fish TP differed across the different survey periods (Pseudo- $F_{3,1171} = 10.8$, P = 0.0001), with TP values showing different temporal patterns across the two capture habitats (Pseudo- $F_{3,1171} = 23.7$, P = 0.0001).



Figure 6.13: A) Isotopic biplot comparing variation in δ^{13} C and δ^{15} N in fish collected from *Lagarosiphon*-dominated (invaded) and *Chara*-dominated (native) habitats during the current study. Temporal variation in B) δ^{13} C, C) δ^{15} N and D) trophic position is also shown.

Estimates of the isotopic contribution by four different primary producers (*Chara, Lagarosiphon*, phytoplankton and epiphyton) to the diet of the fish community (Table 6.7) indicated that assimilation patterns were generally similar between fish inhabiting habitats dominated by *Lagarosiphon* or *Chara* (but note the increased mean consumption of phytoplankton derived materials in fish captured from invaded habitats in October 2008). There was very little evidence that *Lagarosiphon* (mean contribution = 4 %) or *Chara*-derived (1 %) energy (δ^{13} C) or nutrients (δ^{15} N) was assimilated into fish somatic tissues. The bulk of δ^{13} C and δ^{15} N in fish originated from phytoplankton (43 %) or epiphytic algae (51 %).

Table 6.7: Mean (± 95% Bayesian credi	bility interval) proportional	I contribution of diffe	erent primary producers to
fish consumers inhabiting either A) Lag	arosiphon or B) Chara-don	ninated habitats.	

Survey		A) Inv	aded habitats			B)	Native habitats	
date	Chara	Lagarosiphon	Phytoplankton	Epiphytes	Chara	Lagarosiphon	Phytoplankton	Epiphytes
	0.01	0.01	0.61	0.38	0.01	0.01	0.59	0.39
Juli-00	± 0.01	± 0.01	± 0.1	± 0.1	± 0.01	± 0.01	± 0.1	± 0.1
	0.01	0.07	0.08	0.84	0.01	0.11	0.10	0.77
Aug-08	± 0.01	± 0.03	± 0.07	± 0.06	± 0.01	± 0.03	± 0.08	± 0.08
Oct 09	0.02	0.08	0.22	0.68	0.02	0.04	0.06	0.88
001-08	± 0.04	± 0.05	± 0.19	± 0.18	± 0.03	± 0.04	± 0.12	± 0.09
Mov 00	0.01	0.01	0.93	0.05	0.01	0.01	0.88	0.10
May-09	± 0.01	± 0.01	± 0.13	± 0.1	± 0.01	± 0.01	± 0.12	± 0.21

Roach

Multivariate PERMANOVA comparisons revealed isotopic differences between roach captured from the two different habitats (Fig. 6.14A: Pseudo- $F_{1,503}$ = 15.3, P = 0.0001). Comparisons of the areas of the δ^{13} C- δ^{15} N convex-hulls revealed that roach captured from these invaded habitats displayed much reduced (-39%) isotopic variance relative to conspecifics from native, *Chara*-dominated habitats. There was a strong temporal influence on isotopic

variation, with the δ^{13} C- δ^{15} N space inhabited by roach shifting across the different survey periods (Fig. 6.14B & C: Pseudo- $F_{3,503}$ = 9.3, P = 0.0001). A significant interaction effect between capture habitat and survey date indicated that the seasonal shift in roach isotope values differed between the two habitats (Pseudo- $F_{3.503} = 6.3$, P = 0.0001). There was no apparent correlation between roach size (as fork length) and δ^{13} C in either Lagarosiphon (r = 0.1, n = 252, P = 0.14) or Chara-dominated habitats (r = -0.1, n = 287, P = 0.09). Univariate PERMANOVA comparisons of roach δ^{13} C values (Fig.6.14B) showed small but statistically significant differences between the two capture habitats (Pseudo-F_{1.503} = 4.9, P = 0.025). Roach δ^{13} C values differed temporally (Pseudo-F_{3,503} = 10.7, P = 0.001), with fish captured from Chara and Lagarosiphon-dominated habitats showing distinct temporal patterns in δ^{13} C values (Pseudo- $F_{3.503}$ = 13.7, P = 0.001). Roach from both capture habitats showed positive correlations between roach fork length and $\delta^{15}N$ (invaded: r = 0.2, n = 224, P = 0.003; native: r = 0.4, n = 288, P < 0.0001). Univariate PERMANOVA of roach $\delta^{15}N$ values showed a marked habitat-associated effect (Pseudo-F_{1,503} = 18.0, P = 0.0001), with roach from invaded habitats having slightly elevated δ¹⁵N values relative to conspecifics from *Chara*-dominated habitats (Fig. 6.14C). Roach $\delta^{15}N$ values differed between the different sample dates (Pseudo-F_{3.503} = 9.3, P = 0.001), with individuals from Lagarosiphon and Chara-dominated habitats showing small, but significantly different temporal responses in $\delta^{15}N$ values (Pseudo-F_{3.503} = 3.0, P = 0.02). Once roach $\delta^{15}N$ values were baseline-corrected, there was no overall differences in TP values from roach collected in the two capture habitats (Fig. 6.14D: Pseudo- $F_{1.503}$ = 2.0, P = 0.16). There was no evidence of a correlation between roach size and TP in fish captured from invaded habitats (r = -0.03, n = 224, P = 0.68); however, there was a relatively strong positive relationship in individuals associated with native habitats (r = 0.36, n = 288, P < 0.0001). Roach TP varied between sampling date (Pseudo- $F_{3.503} = 81.7$, P = 0.0001), and followed distinct temporal patterns of variation in roach captured in the two habitats $(Pseudo-F_{3.503} = 20.5, P = 0.0001).$



Figure 6.14: A) Isotopic biplot comparing variation in δ^{13} C and δ^{15} N in roach collected from either *Lagarosiphon*-dominated (invaded) or *Chara*-dominated (native) habitats during the current study. Temporal variation in B) δ^{13} C, C) δ^{15} N and D) trophic position is also shown.

Mixing-model derived estimates of the relative contribution made by the four different primary producers (*Chara*, *Lagarosiphon*, phytoplankton and epiphyton) showed little overall differences in the ultimate sources of energy and nutrients fuelling roach from the two different capture habitats (Table 6.8). Although the overall mean contribution of *Lagarosiphon* (7 %) derived δ^{13} C and δ^{15} N was low, it made a considerable contribution (18 %) to roach captured from both habitats in June 2008. *Chara* made little (2 %) contribution to roach somatic tissues, with the bulk of δ^{13} C and δ^{15} N in fish originating from either phytoplankton (50 %) or epiphytic algae (41 %).

Survey		A) Inv	vaded habitats			B)	Native habitats	
date	Chara	Lagarosiphon	Phytoplankton	Epiphytes	Chara	Lagarosiphon	Phytoplankton	Epiphytes
lun_08	0.02	0.18	0.45	0.34	0.03	0.18	0.45	0.35
Jun-08	± 0.04	± 0.05	± 0.11	± 0.11	± 0.04	± 0.05	± 0.10	± 0.11
	0.01	0.02	0.47	0.50	0.04	0.09	0.32	0.55
Aug-08	± 0.02	± 0.03	± 0.14	± 0.14	± 0.06	± 0.05	± 0.17	± 0.15
Oct 09	0.04	0.02	0.31	0.62	0.01	0.01	0.20	0.78
001-08	± 0.04	± 0.03	± 0.27	± 0.26	± 0.02	± 0.02	± 0.22	± 0.21
May 00	0.01	0.02	0.89	0.07	0.01	0.01	0.94	0.04
iviay-09	± 0.03	± 0.04	± 0.11	± 0.15	± 0.02	± 0.03	± 0.06	± 0.08

Table 6.8: Mean (± 95% Bayesian credibility interval) proportional contribution of different primary producers to roach inhabiting either A) *Lagarosiphon* or B) *Chara*-dominated habitats.

Perch

Unlike roach, the effect of capture habitat on perch δ^{13} C- δ^{15} N values was small, and differences between centroids being on the threshold of conventional statistical significance (Fig. 6.15: Pseudo- $F_{1,430}$ = 3.2, P = 0.07). However, total isotopic variance (total convex hull area) was reduced by 29% in perch from Lagarosiphon-dominated habitats. δ^{13} C- δ^{15} N values differed by survey date (Pseudo-F_{3.430} = 53.6, P = 0.0001), with perch displaying distinct isotopic responses to sampling date in the two capture habitats (Pseudo- $F_{3,430}$ = 10.9, P = 0.0001). There were positive correlations between perch size (fork length) and δ^{13} C in both invaded (r = 0.35, n = 187, P < 0.0001) and native (r = 0.32, n = 248, P < 0.0001) habitats, showing an ontogenetic isotopic shift from more ¹³C depleted (e.g. zooplankton) to ¹³C enriched (e.g. benthic invertebrates) prey. Univariate PERMANOVA comparisons of perch δ^{13} C showed small, but significant differences associated with capture habitat (Fig. 6.15B: Pseudo- $F_{1,430} = 5.0$, P = 0.026), with individuals captured in Lagarosiphon-dominated habitats being slightly ¹³C-depleted relative to those from native, *Chara*-dominated habitats. Survey date affected perch δ^{13} C values (Pseudo-F_{3.430} = 4.8, P = 0.003), but the response differed in individuals captured from native and invaded habitats (Pseudo- $F_{3,430}$ = 3.9, P = 0.01). Correlations between perch size (fork length) and $\delta^{15}N$ were positive in both capture habitats, but stronger in invaded (r = 0.41, n = 187, P < 0.0001) relative to native (r = 0.22, n = 248, P = 0.0006) habitats. Univariate PERMANOVA indicated that Lagarosiphon-associated perch were significantly ¹⁵N enriched relative to conspecifics captured from Chara-dominated habitats (Fig. 6.15C: Pseudo-F_{1,430} = 55.0, P = 0.0001). Perch δ^{15} N values were generally similar across sampling dates (Pseudo- $F_{3,430} = 2.4$, P = 0.07), but there was a significant interaction between sampling data and capture habitat on perch $\delta^{15}N$ (Pseudo-F_{3,430} = 7.0, P = 0.0004), indicating that temporal variation in $\delta^{15}N$ differed between the two habitats. Correlations between perch size (fork length) and estimated TP were positive in both invaded (r = 0.25, n = 187, P = 0.0006) and native (r = 0.19, n = 248, P = 0.003) habitats. Univariate PERMANOVA of perch TP values (Fig. 6.15D: Pseudo-F_{1,430} = 3.2, P = 0.07) indicated that the highly significant differences in perch $\delta^{15}N$ values (see above) reflected differences in baseline $\delta^{15}N$ between the two capture habitats rather than major differences in trophic ecology. There was a very strong effect of sampling date on perch TP (Pseudo-F_{3,430} = 53.2, P = 0.0001), with perch from invaded and native habitats showing different temporal patterns of variation in TP (Pseudo- $F_{3,430} = 10.9$, P = 0.0001).



Figure 6.15: A) Isotopic biplot comparing variation in δ^{13} C and δ^{15} N values from perch captured from *Lagarosiphon*-dominated (invaded) or *Chara*-dominated (native) habitats during the current study. Temporal variation in B) δ^{13} C, C) δ^{15} N and D) trophic position is also shown

Estimates of the relative contribution made by the four different primary producers (*Chara, Lagarosiphon*, phytoplankton and epiphyton) to the assimilated diet of perch showed some differences between fish associated with either *Lagarosiphon* or *Chara* (Table 6.9). *Chara* made little (2 %) contribution to perch somatic tissues. Phytoplankton (44 %) and epiphytic algae (44 %) made similar overall contributions to perch somatic tissues, but mean estimated phytoplankton contribution was typically (i.e. in three out of four surveys) slightly larger in invaded habitats, whilst epiphyton-derived δ^{13} C and δ^{15} N made slightly larger relative contributions in perch from native habitats. However, 95% credibility intervals for these estimates often overlapped. The overall mean contribution of *Lagarosiphon* (10 %) derived δ^{13} C and δ^{15} N was low, and it only made a major (>10 %) contribution to perch from (10 – 20 %) to perch from *Chara*-dominated, native habitats between June and October 2008.

Survey		A) Inv	aded habitats		B) Native habitats					
date	Chara	Lagarosiphon	Phytoplankton	Epiphytes		Chara	Lagarosiphon	Phytoplankton	Epiphytes	
Jun-08	0.03	0.14	0.45	0.38		0.02	0.20	0.40	0.37	
our oo	± 0.05	± 0.07	± 0.16	± 0.15		± 0.04	± 0.06	± 0.14	± 0.13	
	0.01	0.05	0.24	0.70		0.01	0.16	0.08	0.75	
Aug-08	± 0.02	± 0.04	± 0.21	± 0.19		± 0.02	± 0.03	± 0.08	± 0.08	
Oct 09	0.04	0.06	0.39	0.52		0.04	0.10	0.21	0.65	
001-08	± 0.05	± 0.06	± 0.28	± 0.29		± 0.07	± 0.06	± 0.25	± 0.21	
May 00	0.02	0.03	0.86	0.10		0.01	0.02	0.92	0.06	
iviay-09	± 0.03	± 0.05	± 0.12	± 0.14		± 0.02	± 0.03	± 0.07	± 0.09	

Table 6.9: Mean (± 95% Bayesian credibility interval) proportional contribution of different primary producers to perch captured from either A) Lagarosiphon or B) Chara-dominated habitats.

Roach-perch interactions

Multivariate comparisons of δ^{13} C- δ^{15} N values from roach and perch captured from *Chara* and *Lagarosiphon*dominated habitats revealed clear isotopic differences between the two species (Pseudo-F_{1,933} = 160.3, P = 0.0001). There was an overall effect of habitat (Pseudo-F_{1,933} = 40.6, P = 0.0001), as well as survey date (Pseudo-F_{3,933} = 7.2, P = 0.0001). There was a significant species x habitat interaction (Pseudo-F_{1,933} = 5.7, P = 0.004), indicating that the isotopic relationships between the species differed in the two capture habitats. Perch TP was typically higher than that of roach (Pseudo-F_{1,933} = 267.9, P = 0.001). There was no effect of habitat (Pseudo-F_{1,933} = 0.1, P = 0.801), but a strong effect of survey date on TP exhibited by perch and roach (Pseudo-F_{3,933} = 130.6, P = 0.001). There was a significant species x habitat interaction, (Pseudo-F_{1,933} = 5.1, P = 0.03), with a reduced interspecific difference in TP in perch and roach captured in native habitats.

Pike

Multivariate PERMANOVA of pike δ^{13} C- δ^{15} N data indicated overlap in the centroids from fish from both capture habitats (Fig. 7.16A: Pseudo-F_{1.78} = 0.1, P = 0.87). However, isotopic variation (as estimated by the area of δ^{13} C- δ^{15} N convex-hulls) was much (-56 %) reduced in individuals captured from invaded, Lagarosiphon-dominated habitats. There was a small overall effect of survey date (Pseudo- $F_{3,78}$ = 2.97, P = 0.02), but fish from Lagarosiphon and Chara-dominated habitats showed distinct isotopic shifts over time (Pseudo-F_{3.78} = 5.38, P = 0.0004). There were moderate positive correlations between pike δ^{13} C and fork length in both invaded (r = 0.33, n = 37, P = 0.04) and native (r = 0.51, n = 48, P = 0.0002) habitats. Univariate PERMANOVA comparisons provided no evidence (Fig. 7.16B: Pseudo-F_{1.78} = 0.16, P = 0.69) for habitat-associated differences in δ^{13} C values for pike collected in invaded and native habitats. The effect of survey date on pike $\delta^{13}C$ was small, and bordered conventional statistical significance (Pseudo-F_{3,78} = 2.57, P = 0.07), but the interaction between capture habitat and survey date (Pseudo- $F_{3.78}$ = 6.7, P = 0.0006) indicated that temporal variation in pike δ^{13} C values followed different patterns in the two habitats. Pike $\delta^{15}N$ was similar between invaded and native habitats (Fig. 6.16C: Pseudo-F_{1.78} = 0.06, P = 0.80). Pike displayed clear positive correlations between δ^{15} N values and fork length in both habitats (invaded: r = 0.69, n = 37, P < 0.0001; native: r = 0.39, n = 48, P = 0.006). Survey date had a small, but significant effect on pike $\delta^{15}N$ (Pseudo- $F_{3.78}$ = 3.15, P = 0.032), with a significant interaction between survey date and capture habitat (Pseudo- $F_{3.78}$ = 5.15, P = 0.004). Positive correlations between pike TP and fork length were recorded in both invaded (r = 0.44, n = 37, P = 0.006) and native (r = 0.27, n = 48, P = 0.06) habitats. Overall, median pike TP was slightly higher in Lagarosiphon-dominated, invaded habitats (3.7) relative to Chara-dominated, native habitats (3.6) (Pseudo- $F_{1.78}$ = 6.45, P = 0.011). However, general comparisons of pike TP between habitats were complicated by a marked effect of survey date (Pseudo-F_{3.78} = 49.9, P = 0.0001) with much decreased TP in October 2008, and an interaction between survey date and capture habitat on pike TP (Fig. 6.16D: Pseudo-F_{3.78} = 12.01, P = 0.0001).



Figure 6.16: A) Isotopic biplot comparing variation in δ^{13} C and δ^{15} N values from pike captured from *Lagarosiphon*-dominated (invaded) or *Chara*-dominated (native) habitats during the current study. Temporal variation in B) δ^{13} C, C) δ^{15} N and D) trophic position is also shown.

Bayesian mixing model results indicated some spatiotemporal variation in the relative contribution made by the four different primary producers (*Chara, Lagarosiphon*, phytoplankton and epiphyton) to pike somatic tissues (Table 6.9). Across the study, *Chara*-derived δ^{13} C and δ^{15} N made a moderate (9 %) mean contribution to pike, varying between 4 and 19 %. The overall mean contribution of *Lagarosiphon* was 16 %, with the invasive macrophyte making a slightly increased mean contribution in invaded (18 %) relative to native (14 %) habitats. However, credibility intervals around mean estimates were typically large (see Table 9). The overall contributions made to pike somatic tissues by phytoplankton (37 %) and epiphytic algae (36 %) were similar, but varied considerably by habitat and survey date (range_{phytoplankton} 25 – 67 %; range_{epiphyton} = 23 – 48%), again with large credibility intervals.

Survey	A) Invaded habitats				B) Native habitats			
date	Chara	Lagarosiphon	Phytoplankton	Epiphytes	Chara	Lagarosiphon	Phytoplankton	Epiphytes
Jun-08	0.10	0.26	0.34	0.30	0.05	0.05	0.42	0.33
	± 0.15	± 0.14	± 0.21	± 0.19	± 0.08	± 0.08	± 0.16	± 0.16
Aug-08	0.07	0.27	0.25	0.35	0.09	0.16	0.33	0.42
	± 0.11	± 0.28	± 0.26	± 0.31	± 0.14	± 0.16	± 0.26	± 0.30
Oct-08	0.07	0.13	0.32	0.48	0.07	0.13	0.33	0.48
	± 0.11	± 0.12	± 0.28	± 0.31	± 0.11	± 0.12	± 0.27	± 0.30
May-09	0.04	0.06	0.67	0.23	0.19	0.23	0.31	0.28
	± 0.06	± 0.11	± 0.26	± 0.27	± 0.23	± 0.24	± 0.29	± 0.25

Table 6.10: Mean (± 95% Bayesian credibility interval) proportional contribution of different primary producers to pike captured from either A) *Lagarosiphon* or B) *Chara*-dominated habitats.

Brown trout

Brown trout were only captured in relatively limited numbers (n = 13) during the study. As such, only basic statistical comparisons could be made, i.e. between capture habitats, and no examination of temporal variation could be made. Brown trout δ^{13} C values varied between -29.6 and -24.1‰, and δ^{15} N values between 8.4 and 15.1 ‰. Two individuals captured in native habitats had markedly enriched δ^{13} C values and depleted δ^{15} N values relative to other individuals, probably reflecting recent migration to the lake from inflowing rivers. Multivariate analysis of δ^{13} C- δ^{15} N data through PERMANOVA showed that brown trout captured from invaded and native habitats were isotopically similar (Pseudo-F_{1,11} = 0.86, P = 0.42). As expected, there was no difference in δ^{13} C (Pseudo-F_{1,9} = 0.07, P = 0.79) or δ^{15} N (Pseudo-F_{1,9} = 1.18, P = 0.34) values in trout from the two habitats. Furthermore, there was no effect of habitat on TP (Pseudo-F_{1,9} = 0.96, P = 0.36).

Due to the importance of brown trout to Lough Corrib anglers, an extended effort was made to obtain extra samples (capture location unknown), which were kindly provided by anglers and Inland Fisheries Ireland. From a sample of 19 individuals, mean (\pm SD) δ^{13} C was -28.2 \pm 1.2 ‰ and δ^{15} N 14.0 \pm 0.8 ‰. The precise capture location (and therefore invasion status) for these individuals were unknown, whilst their capture dates were not associated with our survey dates. As such, it was not possible to make reliable estimates of TP or the relative contribution of different primary producers to the assimilated diet to trout, both of which require information on baseline δ^{15} N. However, δ^{13} C- δ^{15} N values of brown trout of unknown origin largely overlapped with those from individuals collected during the current study (Pseudo-F_{2,29} = 2.2, P = 0.08).

Eel

During the current study, eel made a limited contribution to survey catches, especially in native, *Chara*-dominated habitats. A total of only 39 individuals were available for isotope analysis, limiting statistical comparisons. Across habitats, eel δ^{13} C values varied between -29.7 and -23.5 ‰, and δ^{15} N values between 10.8 and 16.9 ‰. Comparisons of δ^{13} C- δ^{15} N centroids through multivariate PERMANOVA showed that eels collected from the two habitats were isotopically similar (Pseudo-F_{1,36} = 3.5, P = 0.09), although close to the threshold of conventional statistical significance. Isotopic variance (convex-hull size) was reduced by -9% in eels captured in invaded habitats, even though there was a marked imbalance in sample size between the two habitats (invaded n = 31, native n = 6).

6.3.2.5 Other consumers

A single mammal was sampled during the current study – a young male otter (*Lutra lutra*) which was observed to be hit by a car whilst crossing a road adjacent to the western shore of the North basin of Lough Corrib during the June 2008 survey. From its isotope values (δ^{13} C = -28.1, δ^{15} N = 12.7 ‰), the otter sat relatively central to the overall foodweb (as sampled here) with an estimated trophic position of 3.4.

6.3.2.6 Ecosystem-level effects of invasion

Food chain length

The length of the aquatic food chain (as estimated by mean maximum estimated trophic position) was slightly lower in invaded habitats (mean = 3.9, range = 3.4 - 4.2) compared to native habitats (mean = 4.1, range = 3.8 - 4.3).

Isotopic effects of invasion - total niche area

We compared the total δ^{13} C- δ^{15} N isotopic space inhabited by consumers from 12 different taxa collected from *Lagarosiphon*-dominated (invaded) and *Chara*-dominated (native) habitats. Paired comparisons of the isotopic niche area showed a consistent (median reduction = -40 %) and significant (Wilcoxon test: Z = 3.1, P = 0.002) reduction of isotopic niche area in invaded habitats.

6.4 Discussion

This project aimed to examine the impacts of the invasion by a non-native species on several key aspects of the ecology of Lough Corrib. Lough Corrib represents only one of the > 12, 500 lakes in Ireland (Graham & Harrod, 2009), and it is likely that the majority of these lakes are also responding to or facing invasion by one or a suite of invasive species. As such, this project aimed to present ways in which to examine the impact of invasion on lacustrine ecosystems in Ireland rather than to categorically define the impacts of invasive species on receiving ecosystems. Here we demonstrated a range of impacts of the invasion of Lough Corrib by *Lagarosiphon major*, an autogenic ecosystem engineer (Jones *et al.*, 1994). Impacts of *Lagarosiphon* elsewhere in its introduced distribution are well reported and are typically considered detrimental, e.g. reduction in access, replacement of native macrophytes and changes in waterfowl community structure (Howard-Williams & Davies, 1988; Rattray et al., 1994; Bickel & Closs, 2009). However, several authors have reported that a small bodied endemic fish (the common bully, *Gobiomorphus cotidianus*) responded positively to the invasion by Lagarosiphon (Kelly & Hawes, 2005; Bickel & Closs, 2008) indicating that under certain circumstances the invasive macrophyte can facilitate native species (Rodriguez, 2006).

The effects of invasion by *Lagarosiphon* include marked changes in the form of the littoral habitats of the Lough Corrib ecosystem, with a shift from *Chara* meadows that include both structured benthic habitats, and an overlaying unstructured pelagic zone to a state where, depending on the time of year, the whole water column can consist of monospecific stands of the invasive macrophyte (Caffrey *et al.*, 2009), forming structured habitat and limiting light penetration to the lake bottom. Here we have shown a series of ecological impacts of invasion that include changes in macroinvertebrate community structure and production, differences in key life history traits of the two dominant fishes of Lough Corrib and marked differences in some measures of consumer trophic ecology.

Our analysis of macroinvertebrate abundance/biomass was limited due to our inability to routinely (and safely) sample vegetation for macroinvertebrates throughout the study. As such, balanced, detailed data were only available for one of four survey dates. However, analyses of data collected during August 2008 revealed that both the total abundance and biomass of invertebrates per unit area of lake bed were increased in *Lagarosiphon*-dominated habitats and that macroinvertebrate community structure differed between invaded and native habitats. Analysis of community structure through SIMPER showed that the differences in macroinvertebrate community structure between *Lagarosiphon* and *Chara*-dominated habitats stands were driven by an association between *Lagarosiphon* and increased densities of native gastropods (*Bithynia* spp.), bivalves (*Pisidium* spp.), insect larvae (Trichoptera, Ephemoptera, Odonata, Chironomidae) and mites, as well as invasive invertebrates including the amphipod *Crangonyx pseudogracilis* and the zebra mussel *Dreissena polymorpha*. Certain native taxa were found in increased densities (and biomass) in native, *Chara*-dominated habitats (*Planorbis vortex, Gammarus duebeni celticus, Lymnea peregra, Asellus aquaticus*). Our results support an earlier analysis that revealed differences in macroinvertebrate abundance and community structure in invaded and native vegetation in Lough Corrib (Baars *et al.*, 2008) as well as studies conducted elsewhere in the invasive distribution of *Lagarosiphon* (Kelly & Hawes, 2005; Bickel & Closs, 2008; Bickel & Closs, 2009).

The increased macroinvertebrate biomass in invaded habitats partly reflects the inclusion of inorganic materials incorporated into the shells of certain abundant taxa, e.g. gastropod and bivalve molluscs. However, *Lagarosiphon* was also associated with the increased abundance of other macroinvertebrate taxa, showing that this is not simply an effect of increased mollusc production. *Chara* and *Lagarosiphon* have distinct morphologies, and the greater complexity and individual size of individual *Lagarosiphon* plants results in a greater volume of available habitat per unit area of lake bed in invaded areas (Kelly & Hawes, 2005; Baars *et al.*, 2008; Bickel & Closs, 2009). Assuming that these macroinvertebrates are available to fish consumers in the structured habitats represented by *Lagarosiphon* (Crowder & Cooper, 1982; Diehl, 1988; Diehl, 1992), the increased production of putative prey is likely to lead to increased growth potential for fish associated with the invasive macrophyte.

Although this access to increased habitats has been reflected by increased abundances of several native species, IAS including zebra mussels and *C. pseudogracilis* were also associated with *Lagarosiphon* stands. Subsequent to our study, densities of zebra mussels have increased considerably, raising the issue of future invasional meltdown as described elsewhere (Simberloff & Von Holle, 1999; Ricciardi, 2001) through the well described detrimental effects of zebra mussels (Mayer *et al.*, 2001; Maguire & Grey, 2006; Ward & Ricciardi, 2007). The lake bed of Lough Corrib includes significant areas of hard-bottomed substrate, suggesting that zebra mussel recruitment habitat will not be limited if *Lagarosiphon* is finally successfully controlled in the lake (Baars *et al.*, 2010; Caffrey *et al.*, 2010).

Bickel & Closs (2008) suggested that *Lagarosiphon* may have been directly subsidising the food web of invaded lakes in New Zealand. At the start of the current study, we estimated that between 8-15 % of the C and N assimilated by macroinvertebrates was *Lagarosiphon*-derived. However, over the entire study, this fell to a mean of 5 %, suggesting that at a community level, little of the macroinvertebrates biomass was derived from *Lagarosiphon* (Bickel & Closs, 2008). Our results indicated that most Lough Corrib macroinvertebrates had assimilated energy and nutrients from epiphyton and pelagic phytoplankton (estimated indirectly through grazing and filter feeding organisms respectively). This further suggests that both *Lagarosiphon* and *Chara* are utilised by macroinvertebrates

as habitat rather than food. This reflects previous work in *Lagarosiphon*-invaded lakes (Kelly & Hawes, 2005), and is of interest with regard to the long debate about the interactions between macrophytes and putative macroinvertebrate herbivores and the contribution of macrophytes to higher trophic levels (Newman, 1991; James *et al.*, 2000). In the case of Lough Corrib, it also suggests that native consumers have a limited capacity to act as natural biological control of this invasive plant, and that other non-native taxa would be required to fulfil this role, if required by lake managers (Baars *et al.*, 2010).

At the time of sampling, the Lough Corrib fish community was dominated by non-native fishes (e.g. roach, perch, pike, roach x bream hybrids, bream), with relatively small contributions made by native fishes apart from eels. The capture rate of brown trout was very low in all habitats, which given the status of the lake as a noted salmonid fishery (Solon & Brunt, 2006) is of concern. No Arctic charr were captured during the study, adding to the evidence of the extirpation of this conservationally important fish in Lough Corrib (Igoe *et al.*, 2001). Roach, which were first recorded in the Lough Corrib system in the 1980s have rapidly become the dominant fish in the system, which, when considering their capacity to affect water quality (Brabrand *et al.*, 1986; Bergstrand, 1990) and to regulate populations of other fishes (Persson, 1986; Persson, 1990; Persson, 1991) is of concern. This invasive species now dominates the fish community of most of the large lakes in Ireland, including Loughs Corrib, Neagh, Erne, Derg and Ree (Harrod *et al.*, 2002; Inger *et al.*, 2010).

Of the fishes encountered, we recorded strong seasonal effects on both total capture rates and the structure of the Lough Corrib fish community, a typical pattern recorded from temperate lake ecosystems. However, we did not observe overall differences in capture rates, fish biomass or fish community-structure between native, *Chara*-dominated habitats and those invaded by *Lagarosiphon*. The homogeneity of fish community structure and capture rates between habitats were unexpected in the light of previous studies that have reported increased fish abundance in *Lagarosiphon*-infested habitats in New Zealand lakes (Bickel & Closs, 2008; Bickel & Closs, 2009). However, these studies focussed on a single species of small-bodied fish, and utilised a single sampling approach (fish-traps). Conversely, we examined the wider fish community (Hayes, 1989; Jackson & Harvey, 1997; Millar & Fryer, 1999), using a more representative method (multipanel gillnets combined with fykenets) that is less selective than traps. Although our results were unexpected, Duffy and Baltz (1998) reported similar results in their study of the impacts of the invasion by the submerged macrophyte Eurasian milfoil *Myriophyllum spicatum* of Lake Pontchartrain: i.e. fish community structure was similar in invaded and native habitats, but showed a strong pattern of seasonal variation.

At a community level, the invasion of Lough Corrib by *Lagarosiphon* had some effects on the trophic ecology of Lough Corrib fish. Direct analyses of diet were restricted by the large proportion of empty guts and stomachs, a common problem in fish ecology (Bowen, 1996; Polunin & Pinnegar, 2002), and a community-level analysis of fish diet showed no major effect of capture habitat on observed diet. However, the seasonal pattern of diet variation characteristic in temperate fishes (Nilsson, 1960; Sinha & Jones, 1967; Vijverberg *et al.*, 1990) was reflected differently in fish associated with *Lagarosiphon* and *Chara*-dominated habitats. Further evidence for differences in the trophic ecology of fish from the two contrasting capture habitats was provided through the use of SIA. Isotopic results contrasted with results from direct observations of diet, as fish from the two habitats were isotopically (δ^{13} C- δ^{15} N) distinct throughout the study. Fish from *Lagarosiphon*-invaded habitats showed much reduced (-60%) isotopic variation relative to those from native habitats, indicating a reduced niche width in invaded habitats (Bearhop *et al.*,

2004; Layman *et al.*, 2007a), and δ^{15} N values were typically higher in fishes from invaded habitats. Isotope data provided further evidence for seasonal differences in the diet of fish as well as different temporal shifts exhibited in fish collected in the two capture habitats. When δ^{15} N data were recalculated as measures of individual TP by adjusting for different baseline δ^{15} N values, there was no overall habitat-associated difference in TP in L. Corrib fishes. Strong seasonal differences in TP suggested that the ecosystem role of the L. Corrib fish community is not temporally constant; furthermore, fish from the two different capture habitats followed distinct temporal patterns further highlighting differences in the trophic ecology of fish associated with invasive or native vegetation. Isotope mixing models (Parnell *et al.*, 2010) revealed that epiphyton and phytoplankton provided the bulk of energy and nutrients assimilated by the fish community in both habitats, with little contribution of either *Chara* or *Lagarosiphon*, as shown in New Zealand (Kelly & Hawes, 2005). Seasonal differences were most apparent in October 2008, when fish from invaded habitats had increased mean estimated consumption of phytoplankton relative to those from native habitats. This may reflect increased foraging on filter-feeding organisms (e.g. bivalves or zooplankton) or consumption of taxa feeding on phytoplankton deposited onto *Lagarosiphon*.

More focused comparisons of the effects of invasion were made on the two dominant fishes of the Lough Corrib fish community, roach and perch. As described in the introduction, these fishes have well-described interspecific interactions that could potentially be influenced by the *Lagarosiphon*-invasion including competition and predation (Persson, 1986; Persson, 1990; Persson, 1991; Persson *et al.*, 1991). As a generalist omnivore, with the capacity to consume and assimilate plant material (Maitland & Campbell, 1992; Michel & Oberdoff, 1995), it could be predicted that the *Lagarosiphon* invasion represented a trophic opportunity for roach. Conversely, the competitive asymmetry between roach and perch that exists with regard to foraging for common prey (i.e. zooplankton) has been shown to be reversed in structured habitats, where perch outperform roach (Nilsson, 1960; Diehl, 1988; Persson, 1990), potentially allowing Lough Corrib perch to enjoy a refuge from inter-specific competition and to gain an individual size that permits piscivory, and possible control of sympatric roach populations (Persson, 1986; Persson, 1990; Hargeby *et al.*, 2005).

The size structures of roach and perch populations were extremely dynamic during the study, reflecting differences in habitat use and also the recruitment of juvenile fish. Over the complete study, roach were larger in invaded habitats, but there was no clear habitat-related difference in perch lengths. However, both species displayed marked temporal changes in size structure that followed different patterns in *Lagarosiphon* or *Chara* habitats. Small bodied roach were particularly abundant in *Lagarosiphon* in October 2008, whilst small-bodied perch were notably abundant in the same habitat in June 2008 and October 2008. The age structure of the roach population was similar in the two habitats, but varied considerably between different survey dates, with different age classes using *Chara* and *Lagarosiphon* habitats at different times. Although dominant numerically in the lake, the dominance of a single year class (age 4) and reduced contribution by younger cohorts indicates that roach recruitment was variable in L. Corrib, a common feature in roach and other lacustrine fishes (Goldspink, 1978; Mills & Mann, 1985; Wyatt, 1988). Perch age structure was generally similar in the two habitats, with little evidence for seasonal variation. Unusually for lacustrine populations of perch in Ireland or Great Britain, perch recruitment appears to have been relatively constant in recent years (Goldspink & Goodwin, 1979; Craig & Kipling, 1983; Winfield *et al.*, 1998; Paxton *et al.*, 2004), indicating a likely match between perch requirements and current biotic and abiotic conditions.

Fish growth is commonly used as a means of assessing spatial or temporal variation in the performance of fish stocks (Francis, 1990; Britton, 2007). Growth of both roach and perch in Lough Corrib was slightly above the mean calculated for a series of European lakes (Jamet & Desmolles, 1994). Growth of roach was largely similar between habitats; however, for a given age class, back-calculated sizes at age perch were typically larger in perch collected from *Lagarosiphon*-dominated habitats. Differences in back-calculated length at age extended to years prior to the widespread invasion of the invasive macrophyte, suggesting that perch with different growth trajectories were associated with the different habitats (Hjelm *et al.*, 2001; Svanbäck & Eklöv, 2002; Svanback & Eklov, 2003), rather than differences in growth being driven by *Lagarosiphon* itself. This is further supported by a breakdown in the clear habitat-associated differences in the length at age in perch from cohorts that recruited subsequent to the invasion by *Lagarosiphon*.

Used by fish and other ecologists (Bolger & Connolly, 1989; Jakob *et al.*, 1996), condition indices provide a further measure of individual performance or fitness to analyses of growth. Individuals in poor condition may have reduced likelihood of maturation (Morgan, 2004) or may face the risk of overwintering mortality, and roach are typically more sensitive to this in Irish waters than perch (Griffiths & Kirkwood, 1995). During the current study, both roach and perch showed the seasonal variation in condition typical in fish from temperate regions (Jamet & Desmolles, 1994), but there was no obvious habitat-associated difference in roach condition. Perch condition was slightly elevated in those individuals captured in *Chara*-dominated habitats, and both species showed a capture date x habitat interaction suggesting that the seasonal cycle in condition differed between the two contrasting habitats.

Observations of maturation status of roach and perch captured in the two vegetation types showed that in both species, individuals associated with *Lagarosiphon*-dominated habitats matured at a larger individual size than conspecifics from *Chara*-dominated habitats. Although the differences were small (roach = 14%, perch = 5%), they provide another indication of the effects of invasion on the fundamental ecology of the fish of Lough Corrib. Increases in length at maturation are typically associated with the reduced risk of juvenile mortality (Stearns, 1992), suggesting that juvenile mortality rates (not measured here) were reduced in *Lagarosiphon*-dominated habitats. Such reductions in juvenile mortality rates fit well with empirical studies that show that piscivore foraging success is reduced in structured habitats such as those represented by *Lagarosiphon* stands (Brabrand & Faafeng, 1993; Persson & Eklov, 1995). Interestingly, our estimates of adult mortality rates in both roach and perch were both higher in *Lagarosiphon* than *Chara*-dominated habitats, possibly reflecting increased foraging success by piscivorous fishes in invaded habitats (Eklöv, 1997).

Fish ecologists have recently increased their use of geometric analysis of shape to examine population structuring in fishes (Elmer *et al.*, 2010; Harrod *et al.*, 2010) including perch (Svanbäck & Eklöv, 2002; Svanback & Eklov, 2003). Here, we showed significant shape differences in both roach and perch captured from *Lagarosiphon* or *Chara*-dominated habitats. Differences were more substantial in perch than roach, but in both species, individuals collected from native habitats were more fusiform, with large eyes, whilst individuals from *Lagarosiphon*-dominated, invaded habitats had the deep bodied form typical of benthivorous fish (Svanbäck & Eklöv, 2002; Svanback & Eklov, 2003).

Lough Corrib roach and perch had distinct diets, as estimated from analysis of gut/stomach contents (GCA), and SIA, suggesting that trophic overlap between the species in not a major regulating factor in either native or invaded

habitats in Lough Corrib. However, the level of differentiation was affected by capture habitat, suggesting that the presence of invasive *Lagarosiphon* has modified interspecific interactions between the two dominant fishes of the Lough Corrib fish community.

Roach diets (as assessed by GCA) were similar between the two habitats, but showed strong seasonal shifts, with a shift from zooplanktivory at the start of the study to a benthivory in subsequent surveys. Perch diets did differ between the two capture habitats: individuals from invaded habitats consumed more zooplankton, mysid shrimps, and amphipods, whilst individuals from native, *Chara*-dominated habitats were associated with increased consumption of insect larvae and isopods. Again, there were strong seasonal effects on diet in perch, revealing the dynamic nature of the trophic role of perch in L. Corrib.

Stable isotope analysis revealed a similar pattern of trophic differences between the species, with clear differences between species, but an effect of capture habitat and sampling period on the isotopic differences between the species. Roach were isotopically distinct between native and invaded habitats, with reduced (ca. -40%) isotopic variation apparent in roach captured from Lagarosiphon-dominated habitats. Isotopic differences in perch from the two habitat types were less obvious, although again, isotopic variation was less (-30%) in perch from invaded habitats. Both roach and perch showed temporal variation in isotope values, with significant interactions between habitat and survey date. Consumer isotope values are rarely consistent over time in temperate aquatic ecosystems, reflecting seasonal shifts in abiotic and biotic conditions (Grey et al., 2001; Perga & Gerdeaux, 2005; Harrod & Grey, 2006). Usually, such variation is examined in a single habitat (e.g. pelagic zone), but here we have shown that different seasonal patterns in isotopic variation may exist in two contrasting habitats within the same lake, further highlighting the need to collect relevant samples from both consumers and putative prey. Although isotopic differences existed, the results of mixing models that showed both roach and perch had largely assimilated energy and nutrients originating from phytoplankton and epiphytic algae. Lagarosiphon made a small contribution to both perch (mean = 10 %) and roach (7 %) in both habitats, especially in June 2008, but Chara made little contribution. The contribution by Lagarosiphon in both invaded and native habitats suggests that this material may have been fed on by invertebrate consumers in particulate form (Carpenter & Lodge, 1986; James et al., 2000), possibly following the loss of leaves during the shift to the collapsed phase of Lagarosiphon in late spring.

Examination of the effects of the *Lagarosiphon* invasion on other species was constrained by small sample sizes. Analysis of pike stomach contents was constrained by large numbers of empty stomachs. However, our results indicated no differences between capture habitats, and revealed that Lough Corrib pike fed on a mixture of fish and invertebrate prey, as noted by several other authors (Beaudoin *et al.*, 1999; Venturelli & Tonn, 2005; Paradis *et al.*, 2008). Isotopic analyses were conducted on a larger sample size, but again showed no major isotopic differences in pike from *Chara* and *Lagarosiphon*-dominated habitats, apart from a 56% reduction in isotopic variance in pike from invaded habitats. Again, there were marked seasonal differences in isotope values and TP in pike, and interactions between capture date and habitat.

Unfortunately, insufficient brown trout were captured to allow a full examination of the effects of the invasion by *Lagarosiphon* on this economically-important species. SIA revealed that brown trout showed isotopic similarities with other carnivorous fishes in Lough Corrib, including eels, perch and pike, suggesting a mixed diet of fish and

invertebrates. SIA also revealed that some individuals captured in the lake had isotope values indicative of recent migration into the lake, showing the importance of non-lacustrine production in this important species.

In terms of the wider Lough Corrib ecosystem, it is clear that the impacts of the Lagarosiphon invasion varied considerably. There was no obvious effect of Lagarosiphon on the abundance, biomass or structure of the fish community. However, during this study, we have revealed marked, habitat-associated differences for a number of ecological measures between native Chara-dominated habitats compared to Lagarosiphon-invaded sites (e.g. invertebrate biomass, abundance, and community structure; perch growth rates; size at maturity in both roach and perch; adult mortality rates; fish trophic ecology). However, many effects of the invasion were relatively subtle, and were revealed as part of an interaction between survey date and capture habitat on the measure of interest. The inability to identify differences between invaded and native habitats may possibly reflect limited statistical power (Peterman, 1990; Fairweather, 1991; Cheruvelil et al., 2000). However, our analyses typically include large sample sizes, suggesting that some effects of invasive species are indeed less marked, and furthermore vary over time. There were ecosystem-level differences between native and invaded habitats, ranging from different macroinvertebrate communities, to a reduction in the length of the food chain in Lagarosiphon-dominated habitats. Further evidence for wide-ranging effects of invasion was the much reduced level of consumer isotopic variation in Lagarosiphon-dominated habitats (median reduction relative to Chara-dominated habitats = -40 %). Such reduction in isotopic variation has been associated with habitat degradation in aquatic ecosystems and is thought to reflect reduced niche utilisation by consumers through a reduction in putative prey resources (Layman et al., 2007b; Syväranta & Jones, 2007). This may reflect that consumers from native habitats fed on more isotopically-variable food sources than their counterparts in invaded habitats, or that there was a homogenisation of sources of energy and nutrients in Lagarosiphon-dominated habitats. An alternative explanation is that the reduced isotopic variation displayed by consumers from Lagarosiphon-dominated habitats fed on less prey taxa. Although examination of trophic niche breadth of fish consumers was complicated by the large number of empty stomachs we encountered, niche breadth (Levins, 1968) across the study was generally similar in both roach (invasive Levins' B'= 0.042, native B'= 0.050) and perch (invasive B'= 0.074, native B'= 0.072) from the two capture habitats. Although this argues against differences in niche breadth, and for a homogenisation effect of Lagarosiphon, our isotope mixing model results only showed small differences in the source of energy and nutrients assimilated by consumers from the two habitats. Although we used a powerful Bayesian mixing model (Parnell et al., 2010), this likely reflects an effect of overlap in the isotopic values of putative energy sources or inaccurate estimates of trophic fractionation leading to reduced mixing model sensitivity.

Of the fishes examined here, perch appeared to be most sensitive to the invasion of Lough Corrib by *Lagarosiphon*. Comparison of fish captured in native and invaded habitats revealed differences in several key life history characteristics including maturation patterns, growth and mortality rates. Perch from the two capture habitats also had distinct shapes, diets and were isotopically distinct. There is an increasing interest in phenotypic plasticity and adaptive responses to change (Agrawal, 2001), and ecosystems invaded by ecosystem engineers such as *Lagarosiphon* may provide novel habitats that generate biological variation through phenotypic plasticity, that may also lead to increased population differentiation. In other systems, perch have been seen to show significant population sub-structuring associated with utilisation of different habitats (Gerlach *et al.*, 2001; Behrmann-Godel *et al.*, 2004; Bergek & Björklund, 2007; Bergek *et al.*, 2010). These differences shown here suggest that perch populations in Lough Corrib have displayed a measure of habitat-associated segregation and may represent an
adaptive response (Agrawal, 2001) by a phenotypically plastic species (Svanbäck & Eklöv, 2002) due to the invasion of an ecological engineer (Gerlach *et al.*, 2001; Bergek *et al.*, 2010). The evolutionary effects of invasion (Strauss *et al.*, 2006; Andvik *et al.*, 2010) are typically considered in the context of loss of native populations (e.g. through introgression), but have also been linked to the formation of biological diversity, e.g. through adaptation in the invader itself (Nolte *et al.*, 2005; Vellend *et al.*, 2007; Prentis *et al.*, 2008; Andvik *et al.*, 2010). It appears probable that this may be the case in Lough Corrib, and is worthy of future study. It should be considered in any management of perch in Lough Corrib.

Some ecological measures (size, maturation rate, adult mortality rate, shape and stable isotope ratios) showed clear differences, suggesting considerable differences in roach captured from invaded and native habitats. However, habitat-associated differences were less marked in roach, with similar diet, growth rate, age structure, and condition between fish captured from the two capture habitats. This difference between the two species may be expected as roach, unlike perch are generalist omnivores with the capacity to consume and assimilate aquatic vegetation (Maitland & Campbell, 1992; Michel & Oberdoff, 1995) and are characteristically flexible with regard to environmental variation (Lappalainen *et al.*, 2008). Indeed, access to large amounts of invasive vegetation reflects a potential trophic opportunity to L. Corrib roach, in particular to the larger bodied individuals that typically consume aquatic macrophytes (Horppila, 1999). However, analysis of gut contents and stable isotopes provided little evidence that roach directly consumed *Lagarosiphon*. This may reflect the low nutritional status of *Lagarosiphon* (James *et al.*, 2006), but access to abundant alternative prey resources in Lough Corrib probably limits the need for roach to consume macrophytes.

Although experimental studies examining control of Lagarosiphon conducted subsequent to the current study have been successful (Caffrey et al., 2010), the invasion of Lough Corrib by Lagarosiphon represents only one of a series of perturbations of the Lough Corrib ecosystem (Strayer, 2010). Lough Corrib, like other Irish lakes, is a system in flux following changes in land use in the surrounding catchment, and in the community of the lake. Although diffuse nutrient inputs remain a growing issue, the lake remains mesotrophic according to the OECD classification (Clabby et al., 2008). However, changes in the lake, such as shifts in fish community structure to a dominance by nonsalmonid fishes (including the likely loss of Arctic charr (Igoe et al., 2001)), point to fundamental ecological shifts (Nümann, 1972; Persson et al., 1991). Roach were first noted to be abundant in the system in the 1980s (Fitzmaurice, 1981), but now dominate the lake. The recent invasions by the zebra mussel and Lagarosiphon point to continued ecological change in the L. Corrib system. Furthermore, there is a strong possibility that the ecological changes that have taken place in the lake over the last century have also included an influence of climate change (Graham & Harrod, 2009). These factors highlight problems with defining suitable reference conditions for measures of future change, as well as the low probability of returning to an unperturbed state. Although we were unable to examine the effects of control during the current study, our data will allow a future assessment of the effects of restoration/control of Lagarosiphon on the Lough Corrib ecosystem (Vander Zanden et al., 2003; Schmidt et al., 2007).

6.5 Conclusions and recommendations

Like other studies, this work has revealed that the impacts of invasion on receiving ecosystems can be extremely variable. Here, we showed that impacts differed between taxa, but we have also revealed that across many of the

measures examined, the invasion by *Lagarosiphon* was associated with seasonal shifts in the ecology of L. Corrib taxa that differed with those shown in native, *Chara*-dominated habitats. These differences are likely to combine to affect the overall ecology and function of invaded systems such as L. Corrib, and are likely to affect the assessment of ecological status of the waterbody in terms of the WFD.

Macroinvertebrate community structure differed between invaded and native habitats: macroinvertebrate abundance and biomass was greater in Lagarosiphon beds relative to that of the native *Chara* spp. and structure of the macroinvertebrate community differed between invaded and native habitats, including the increased abundance of invasive invertebrates, such as the zebra mussel.

There were no such obvious effects of invasion on fish community structure or production, as the abundance and biomass of fishes were similar in invaded and native habitats. However, fish captured in invaded habitats differed in several key characteristics including size (roach), growth rate (perch), size at maturity (roach), instantaneous total mortality rates (roach and perch) and fish shape (roach and perch). In many statistical comparisons of these characteristics, there was evidence that fish showed different seasonal responses to seasonal differences in invaded and native habitats.

Comparisons of macroinvertebrate (based on SIA) and fish (SIA and gut content analysis) captured from invaded and native habitats provided evidence that invasion affected consumers trophic ecology to different degrees. Some taxa showed clear overall differences between habitats (e.g. perch), whilst differences in other taxa were driven by individuals from invaded and native habitats displaying distinct seasonal patterns in diet. Stable isotope analyses revealed that *Lagarosiphon*, although representing the dominant primary producer in invaded habitats, made very little contribution to the food web of L. Corrib. Stable isotope analysis revealed that many consumers showed reduced isotopic variation in *Lagarosiphon*-dominated habitats, indicating that dietary variation may be reduced following invasion.

Assuming that *Lagarosiphon* remains in the system, and is not fully managed through the control methods described in Section 8.4 the effects of invasion detailed here are likely to have implications for the lake's status in the light of several WFD quality elements, e.g. macrophytes, macroinvertebrates and fish. The most obvious effect is the loss of native *Chara* species associated with the invasion by *Lagarosiphon*. The increased abundance/biomass of invasive macroinvertebrates in invaded habitats e.g. *C. pseudogracilis*, *D.polymorpha*) indicates the potential for invasional meltdown in Lough Corrib.

The experimental approach used here was aimed to provide an understanding of the effects of invasion by *Lagarosiphon* (and by extension, other invasive taxa) on receiving ecosystems. It performed well, and could be extended to studies of invasion at other sites: we recommend that it should be continued in future as a means of quantifying the potential long-term effects of invasion or responses to the successful control of *Lagarosiphon* in Lough Corrib.

7 Ecological implications of the invasion of chub (*Leuciscus cephalus*) in the River Inny

7.1 Introduction

In recent years, a number of non-native fish species have become invasive in Ireland (Griffiths, 1997) and are now present in rivers across the island. The most widespread of these, roach were first introduced to the Munster Blackwater, a river in the south of Ireland in 1889 (Went, 1950; Fitzmaurice, 1984) and are now found across Ireland. The initial spread of this species was gradual, but by the mid 1970's, roach were becoming invasive and their distribution was increasing at a rapid rate. Currently, roach are present in most river catchments in Ireland and their introduction has had significant consequences. Like chub, it is a fast-growing, prolific and generalist species (Maitland & Campbell, 1992). In coarse fisheries, the closely related rudd (*Scardinius erythrophthalmus*), an economically important angling species has been the most adversely affected species (Cragg-Hine, 1973), although a decline in bream populations in some watercourses may also be a attributed to direct or indirect competition with roach or hybridisation (Hayden *et al.*, 2010; Toscano *et al.*, 2010).

Dace (*Leuciscus leuciscus*) another non-native cyprinid that is very closely related to chub, was also introduced to the Munster Blackwater from specimens taken from Britain in 1889. However, until the early 1980s dace were more or less endemic to this catchment (Caffrey *et al.*, 2007). The species has since been recorded in the lower reaches of the River Shannon, in Doon Lake, Co. Clare and a number of other major river catchments in the south of Ireland. In these systems dace have established large, sustainable populations and like roach, threaten to compete directly with resident fish species (e.g. brown trout and salmon) for food and habitat (Caffrey *et al.*, 2007).

Like roach and dace, chub are cyprinid fishes with a native distribution that extends from Britain northward to Scandinavia and eastwards into the former Soviet Union (Hellawell, 1971; Cowx, 2001). Chub, are larger than both roach and dace, and are a highly sought after angling species in Britain and Europe. The absence of chub from the rivers of Ireland, many of which may provide suitable putative habitat for the species (Maitland & Campbell, 1992) and excellent conditions for the angler, have provoked considerable controversy among the visiting angling fraternity. Chub prefer lotic habitats, with shallow water and gravel banks with moderate to high water flow for spawning. The preferred substrate of chub is coarse gravel (Lelek & Lusk, 1965; Cowx & Welcomme, 1998). Juvenile chub form schools in shallow water, while larger specimens tend to be solitary, often residing under overhanging trees or roots (Maitland & Campbell, 1992; Cowx, 2001). Chub have a catholic diet: fry and young juvenile fish feed extensively on small aquatic invertebrates, while fish up to five years of age consume mainly vegetation and aerial insects (Hellawell, 1971; Mann, 1976). Chub exhibit an ontogenetic shift and become piscivorous when they reach a length of *ca.* 150 mm after which the proportion of fish in their diet increases directly with length (Mann, 1976; Vitali & Braghieri, 1984). Several authors have noted the similarity in chub diet with salmonids (brown trout and salmon) and eels (Hellawell, 1971; Mann, 1976). Chub typically have a life expectancy of 10 – 12 years (Maitland & Campbell, 1992).

In 2005, a number of chub were caught in the River Inny by anglers and officially identified by fisheries scientists from Inland Fisheries Ireland. This species had most likely been illegally introduced to the river by anglers with a view to establishing a population of this popular angling species (Maitland & Campbell, 1992) in Ireland (Caffrey *et*

al., 2008). In 2006 and 2007 IFI conducted baseline surveys to establish the distribution of chub in the river Inny. Chub were positively identified at two sites. This project aimed to build on the findings of these previous surveys and to understand the ecology of invasive chub in Ireland through the examination of a series of research questions:

- 1. What was the current status of the chub population in the River Inny?;
- 2. Did sections of the River Inny inhabited by chub support different biological communities, i.e. fish and benthic macroinvertebrates, compared to non-invaded areas?;
- 3. Did the population structure (age, size), and life history characteristics (growth) of different fish species shift in areas of the River Inny invaded by chub?;
- 4. Was invasion by chub associated with a shift in the diet and trophic niche of macroinvertebrates or fish and did chub diet overlap with native and established fishes?; and
- 5. Did chub make large scale-movements in the River Inny that may impact the efficacy of control measures?

7.2 Materials and methods

7.2.1 Site description

The River Inny, a 5th order river is one of the major tributaries to the River Shannon (Fig 7.1). The river is 88.5 km long and occupies a catchment area of 782.46 km² (O'Reilly, 2002). The River Inny rises in Co. Westmeath and flows through Loughs Sheelin, Kinale, Derravaragh and Iron before discharging into Lough Ree. The river between Loughs Derravaragh and Ree is wide, varying between 25 and 30 m. The adjacent land use for much of the river length is lowland pasture. From Lough Derravaragh to Abbeyshrule and from *ca*. 50 m downstream of Shrule Bridge to Lough Ree, the river is relatively deep (mean depth = 2.3 m) and moderately slow flowing. The predominant substrate type is mud and silt. From Abbeyshrule to *c*. 100 m downstream of Shrule Bridge the river is shallow (*ca*. 0.5 m), fast flowing and dominated by coarse gravels and boulders.

7.2.2 Fish survey

The fish community in the river Inny was sampled on three occasions. During the initial sampling period (April 2008), the entire 42 km-long channel between Lough Derravarragh and Lough Ree (Fig. 7.1) was intensively electrofished over a four day period. A short section of river (*ca.* 2 km long) between survey stretch 2 and 3 was not sampled due to inaccessibility and hazardous river conditions. Qualitative information relating to the abundance and community structure of the fish populations in the river was gathered during these surveys. During subsequent visits (August 2008 & May 2009) quantitative timed electrofishing was conducted at three survey stretches (Table 7.1; Fig. 7.1) on the river to provide reliable estimates of community structure and the presence/absence of invasive chub, without impacting the wider fish community. These stretches were located at Ballinalack, Abbeyshrule and Shrule Bridge. At each stretch electrofishing was conducted upstream for 7 minutes and all fish were retained for laboratory analyses.



Figure 7.1: Schematic map of the three sections of the River Inny sampled during the current study. During the first survey (April 2008) fish were collected from the entire survey stretch. During subsequent surveys (August 2008 and May 2009) fish were collected during 7 minute semi-quantitative electrofishing runs conducted in the middle section of the three survey stretches. See Table 7.1 for details.

Site	Local name	Grid reference: Start of section	Grid reference: End of section
number			
1	Ballinalack	53° 37.833' N 7° 28.461' W	53°37.930' N 7°28.195' W
2	Abbeyshrule	53°35.460' N 7°39.039' W	53°35.649' N 7°38.924' W
3	Shrule	53° 33.149' N 7° 47.820' W	53° 33.413' N 7° 47.730' W
	Bridge		

Table 7.1: Location of the different sites on the river Inny sampled for fish during the current study

Electrofishing surveys were conducted with a crew of 15 people. A land based support crew was always in attendance. The electrofishing equipment comprised of a 16 hp generator that produced a maximum 600v pulsed direct current, with a maximum current of 30 amps. Electricity was discharged into the water through an insulated hand-held anode and a 6 m-long trailing copper cathode. Each boat had a 3-man crew. One steered the 25hp outboard motor that drove and directed the boat. The second stood on the bow and held the anode ring in the water ahead of the boat. The third crew member stood alongside or behind the anode operator, equipped with a landing net, and retrieved stunned fish from the water (Fig. 7.2).



Figure 7.2: Aerial view of a single electrofishing crew on the River Inny

During the initial survey, representative subsamples of the different fish species captured were retained for subsequent laboratory analysis. Two further semi-quantitative electric fishing surveys were conducted in August 2008 and May 2009, when three study sites were selected and a 7 minute upstream electric fishing survey was conducted at each stretch. All fish (except chub) captured during the semi-quantitative surveys were retained for laboratory analyses. Individual fork lengths (\pm 1mm) and mass (\pm 0.1 g) were recorded. Ageing structures (scales: cyprinids, salmonids; opercular bones: perch, pike) were removed for analysis of age and growth. Growth rates were determined through the estimation of back calculated length at age using the non-linear method originally attributed to Manastryrsky (Francis, 1990).

In all fish a longitudinal ventral incision from the vent to a line level with the pectoral fin was made. This allowed a detailed examination of the body cavity, allowing sex to be determined and stomach/gut contents to be collected and preserved in 70% alcohol prior to identification and enumeration. Diet analysis through direct examination was constrained as many fish examined had empty guts or stomachs. Of those fish with prey in their alimentary canal, diet was examined by calculating the percentage frequency of different prey taxa (Hyslop, 1980). The contents of the first third of the gut (cyprinids) or of the stomach (all other species) were examined under a low-power binocular microscope: prey were identified to the lowest practical taxonomic level and enumerated.

A sample (ca. 1 g) of dorsal muscle tissue was excised for stable isotope analysis (SIA) of carbon and nitrogen (δ^{13} C and δ^{15} N) and dried at 60°C for at least 48 hours prior to p reparation for SIA. Archived chub samples collected during previous surveys conducted by IFI were also analysed for δ^{13} C and δ^{15} N.

Throughout the three surveys, all chub (N=2) encountered were fitted with a VHF radio transmitter (ATS F1520: Fig. 7.3A) and re-released to the site of capture. It was anticipated that these fish might assist in the location of any shoals or aggregations of chub that might be present in the river. Fish were anaesthetised in a phenoxy-2-ethanol solution and a ca. 35 mm incision was made on the ventral surface, anterior to the right pelvic fin. A biopsy needle (16 gauge) was used to pass the trailing antenna (200 mm) through the abdominal wall of the fish and the transmitter unit was placed in the abdominal cavity between the abdominal wall and the peritoneum. The wound

was closed using 3-4 interrupted absorbable sutures with square knots (Fig. 7.3B). Fish were allowed to recover in oxygenated water for 1 hour before being released to the site of capture. Fish movements were monitored ca. weekly using a VHF receiver (ATS model R2000) and a four element Yagi antenna and the location of the fish was determined through triangulation and co-ordinates recorded using a Garmin Etrex GPS system (± 5 m).



Figure 7.3: A) ATS F1520 VHF transmitter used to track chub. B) Chub recovering following surgery, showing site of tag-insertion.

7.2.3 Macroinvertebrates

Macroinvertebrates were collected from single sites located within each of the three survey stretches (Table 7.2). We originally aimed to sample macroinvertebrates monthly, however high river levels and flow rates due to extraordinary precipitation made it hazardous to collect samples for much of the year. Semi-quantitative kick-samples were collected in April 08 (2 sites only), May 08, August 08, Jan 09, and May 09. Sampling consisted of a four minute counter-current kick-sample using a 250 μ m mesh kick net followed by a four minute hand-pick. Macroinvertebrates were placed in labelled polythene bags and frozen at -18°C. Once returned to the laboratory, invertebrates were identified to the highest resolution possible and enumerated.

Table 7.2: Macroinvertebrate collection sites on the River Inny.

Stretch	Local name	Grid reference
1	Ballinalack	53° 37.833' N 7° 28.461' W
2	Abbeyshrule	53°35.422' N 7°39.066' W
3	Shrule bridge	53° 35.422' N 7° 47.824' W

7.2.4 Stable isotope analysis (SIA)

Samples were dried (60°C for min. of 48 h), ground to a fine powder using an agate pestle and mortar and stored in 1.5 ml centrifuge tubes in a silica desiccator, before weighing into tin-cups. Sample mass differed according to the sample type but were typically ca. 1.5 mg for macroinvertebrates and 0.4 – 0.6 mg for fish tissues. Samples were analysed for δ^{13} C and δ^{15} N following standard protocols (Harrod *et al.*, 2005) at either the Max Planck Institute for Evolutionary Biology (Plön, Germany) or the CHRONO Centre for Climate, the Environment, and Chronology at Queen's University Belfast. All laboratories used international recognised secondary standards of known relation to the international standards of Vienna Pee Dee belemnite for carbon and atmospheric N2 for nitrogen allowing direct comparison of results across different laboratories. Typical precision for a single analysis was ± 0.1‰ for δ 13C and ± 0.3‰ for δ 15N.

7.2.5 Control of invasive chub

In addition to the scientific surveys, 8 electrofishing operations aimed at controlling / eradicating chub in the River Inny took place between 2008 and 2010. On four occasions (June 2006, April 2007, April 2008 and September 2010) the majority of the channel length between Loughs Derravaragh and Ree was examined (Fig. 7.1). On the other four sampling occasions, electrofishing was confined to specific stretches of the River Inny. In September 2007 and May 2009 two stretches, each measuring *c*. 2 km in length, at Shrule Bridge and Ballinalack were electrofished. The river at Shrule Bridge was also electrofished in August 2008 and May 2010 as radio-tagged chub were present in this location.

7.2.6 Data analysis

Due to the unbalanced, often non-normal and heteroscedastic nature of the data generated during the study, statistical comparisons were typically made using distance-based permutation-based analysis of variance (PERMANOVA: Anderson *et al.*, 2008) within the PRIMER 6 framework (Clarke & Gorley, 2006). PERMANOVA is largely analogous to parametric ANOVA or MANOVA, but is robust to deviations from the assumptions of parametric ANOVA (i.e. does not require normality of errors, or homogeneity of variance between groups), and probability (P) values are calculated through permutation. Permutation based tests represent a particularly robust means of examining interaction terms (Fraker & Peacor, 2008). Here, P values were estimated from 10 000 permutations. In each case data were treated using relevant transformation (i.e. $Log_{10}+x$, square-root) prior to production of similarity matrices. Bray-Curtis distance measures were used for analyses of macroinvertebrate and fish community structure, as well as of fish diet, whilst Euclidian distances were used for all other analyses. In most analyses we compared the effect of capture location (Stretch) on the variable of interest, and how this varied temporally (Survey Date), and the interaction between habitat type and time. Where data were not sufficient to examine temporal variation, or were only recorded during a simple sample (e.g. SIA data), data were polled to permit general comparisons between survey stretches or species.

Comparisons of both macroinvertebrate and fish community structure and fish diet between survey stretches and dates were made using PERMANOVA (as described above) of abundance data. Significant differences were examined through the use of SIMPER analysis in PRIMER 6 (Clarke & Gorley, 2006). Trophic overlap between

different fish species was also examined using PERMANOVA. Variation in macroinvertebrate community structure were visualised through the use of nonmetric multidimensional scaling ordination plots.

7.2.7 Stable isotope data

Stable isotope data were analysed in various ways. Stable isotope ratios were compared between key primary producers, as well as macroinvertebrate and fish consumers collected in invaded or native habitats through multivariate PERMANOVA of δ^{13} C and δ^{15} N. A multivariate approach was preferred over simple univariate comparisons of individual stable isotopes as variation in the location of centroids of δ^{13} C and δ^{15} N values are ecologically informative, providing information on energy source and trophic position. All δ^{13} C values from fish were adjusted prior to analysis to correct for lipid effects following Kiljunen *et al.* (2006). Isotope data were transformed (δ^{13} C log₁₀x+40; δ^{15} N log₁₀x) to stabilise variance prior to analyses.

Individual consumer trophic position (TP) was estimated isotopically following Post (2002) using baseline δ^{15} N values from molluscs collected from the survey stretches. Log₁₀-transformed fish TP values were compared between stretches using PERMANOVA.

Following Bearhop *et al.* (2004), who suggested that isotopic variance could be used as a measure of trophic niche width, Layman *et al.* (2007a) proposed the use of the area of consumer (or community) δ^{13} C- δ^{15} N convex-hull area as a means of examining niche variation. We compared pooled isotopic variation in macroinvertebrate and fish consumers collected from the different survey stretches by calculating convex hull areas using ImageJ (Abràmoff *et al.*, 2004).

7.2.8 Movement and behaviour of invasive chub

Robust estimates of summary statistics (mean ± 95% confidence intervals) of various different measures of chub behaviour (location relative to datum, movements and swimming velocity) measured from telemetry data were calculated through bootstrapping (10 000 permutations) of the original data in SYSTAT 13 (Efron & Tibshirani, 1993). The size of the chub population in the River Inny was estimated using catch depletion data gathered over subsequent control surveys conducted by the CFB. Population size was estimated following Carle and Strub (1978).

7.3 Results

7.3.1 Macroinvertebrate community structure

Multivariate analyses indicated that macroinvertebrate community structure (as abundance per 4 minute kick sample) was similar in the different survey stretches (Fig. 7.4; PERMANOVA: Pseudo- $F_{2,15} = 0.82$, P = 0.62), but differed considerably across the different survey dates (Pseudo- $F_{4,13} = 4.2$, P = 0.0001).



Figure 7.4: Nonmetric multidimensional scaling ordination of macroinvertebrate community structure. Each point represents the results of an individual kick sample, with markers reflecting different sampling date, and numbers reflecting the location of the sample on the three different survey stretches. The proximity of individual markers reflects the relative similarity of different samples (close together = increased similarity).

7.3.2 Fish community structure

A total of 13 fish species were recorded during the surveys (Table 7.3). These were bream (*Abramis brama*), roach (*Rutilus rutilus*), perch (*Perca fluviatilis*), pike (*Esox lucius*), brown trout (*Salmo trutta*), *Atlantic salmon (Salmo salar*), European eel (*Anguilla anguilla*), roach x bream hybrids, *gudgeon (Gobio gobio*),minnow (*Phoxinus phoxinus*), *stoneloach (Barbatula barbatula), brook lamprey (Lampetra planeri)* and chub (*Leuciscus cephalus*). Roach was the most abundant species recorded, followed by brown trout and pike.

Comparative differences of fish community structure (presence-absence data) between the three survey stretches across the study were on the threshold of conventional statistical significance (Pseudo- $F_{2,6}$ = 2.5, P = 0.05). Pairwise comparisons showed that the largest differences were between Site 2 and Site 3 (P = 0.09). There was no evidence of seasonal variation in fish community structure (Pseudo- $F_{2,6}$ = 1.2, P = 0.38).

Table 7.3: Variation in fish community structure across the three different survey stretches as A) numerical CPUE for different species as n.min⁻¹ captured during 7 minute electrofishing runs and B) biomass CPUE as g.min⁻¹ captured during 7 minute electrofishing runs. Note that during Survey 1, fish were collected using qualitative electrofishing and as such data are shown as Y = species present in stretch, - = species not present in stretch.

A) Abundance		0.4			0.4			0.4	
(n.min [*])	Survey	Site 1	Survey	Survey	Site 2	Survey	Survov	Site 3	Survov
Species	Survey 1	2	3	Survey 1	2	3	3urvey 1	2	3
Bream	—	0.0	0.0		0.0	0.0	Y	0.0	0.0
Bream x roach hybrid	Y	0.0	0.3	Y	0.3	0.0	Y	0.1	0.0
Brook lamprey	Y	0.0	0.0	_	0.0	0.0	_	0.0	0.0
Brown trout	Y	0.1	0.0	Y	2.7	0.0	Y	14.6	1.4
Chub	—	0.0	0.0	_	0.0	0.0	Y	0.1	0.3
Eel	Y	0.4	0.1	_	0.1	0.0	Y	0.6	0.0
Gudgeon	Y	0.0	0.0		0.0	0.0	Y	7.4	0.4
Minnow	—	0.0	0.1	Y	0.6	4.1	_	11.7	0.0
Perch	Y	1.1	3.0	Y	1.1	0.4	Y	1.7	0.3
Pike	Y	0.0	0.3	Y	0.6	0.1	Y	0.6	0.0
Roach	Y	5.3	23.6	Y	6.6	2.6	Y	4.4	9.4
Salmon	—	0.0	0.0	Y	1.6	0.0	Y	0.0	0.0
Stoneloach		0.1	0.0		0.3	0.0	_	0.6	0.0
n individuals		147	243	_	390	134	_	485	83

B) Biomass (g.min ⁻¹)		Site 1			Site 2			Site 3	
Species	Survey 1	Survey 2	Survey 3	Survey 1	Survey 2	Survey 3	Survey 1	Survey 2	Survey 3
Bream	_	0	0	—	0	0	Y	0	0
Bream x roach hybrid	Y	0	0	Y	16	0	Y	94	0
Brook lamprey	Y	0	35	_	0	0	_	0	0
Brown trout	Y	11	0	Y	1 313	0	Y	835	219
Chub	—	0	0	—	0	0	Y	66	180
Eel	Y	10	8	—	58	0	Y	186	
Gudgeon	Y	0	0	—	0	0	Y	106	8
Minnow	—	0	1	Y	1	12	—	23	0
Perch	Y	50	172	Y	161	56	Y	84	16
Pike	Y	0	396	Y	875	187	Y	209	0
Roach	Y	131	310	Y	822	164	Y	148	1852
Salmon	—	0	0	Y	12	0	Y	0	0
Stoneloach	_	1	0	_	1	0	_	5	0
Total mass (g)	_	1 420	6 443	_	22 807	2 938	_	12 296	15 924

7.3.3 Diet

Many fish had empty guts or stomachs, which was a limiting factor regarding sample sizes for comparisons of fish diet at each site. Detailed analyses of diet were restricted to brown trout, roach, perch and pike. An initial comparison of fish diet (from analysis of gut and stomach contents) at a community level through multivariate PERMANOVA showed a strong effect of both survey stretch (Pseudo- $F_{2,383} = 2.7$, P = 0.0017) and survey date (Pseudo- $F_{2,6} = 3.9$, P = 0.0002) on the prey consumed by fish. There was a significant and strong interaction between survey stretch and date (Pseudo- $F_{4,383} = 3.5$, P = 0.0001) indicating that seasonal variation in diet differed between survey stretches. Comparisons of overall dietary differences are shown in Table 7.4, and indicate that chub diet (n = 2) overlapped with a number of species.

Table 7.4: Probability (P values) of pairwise PERMANOVA comparisons of dietary overlap (from gut and stomach contents) between different fishes. Comparisons that are statistically similar ($P \ge 0.05$) are shown in bold, and reflect potential dietary overlap (i.e. there was overlap in the observed diet between the two species). Note that samples were pooled across sites and surveys to provide an overall comparison of the ecology of chub and other species in the River Inny. Fish species names are abbreviated as: bream = Br; brown trout = Bt; chub = Ch; Eel = Ee; gudgeon = Gu; roach x bream hybrid = RoxBr; brook lamprey = Bl; minnow = Mn; perch = Pe; pike = Pi; roach = Ro and salmon = Sa.

	Bt	Ch	Ee	Gu	RoxBr	Pe	Pi	Ro	Sa
Br	0.002	0.016	0.1	0.0012	0.09	0.03	0.0001	0.0001	0.0001
Bt	—	0.041	0.103	0.0001	0.02	0.0001	0.0001	0.0001	0.0001
Ch		_	0.06	0.05	0.07	0.06	0.009	0.035	0.02
Ee			—	0.007	0.24	0.22	0.04	0.12	0.002
Gu				—	0.016	0.0001	0.0001	0.0001	0.0006
RoxBr					—	0.03	0.089	0.98	0.0014
Pe						—	0.0001	0.0001	0.0001
Pi							_	0.0001	0.0001
Ro								—	0.0001
Sa									—

7.3.3.1 Roach

Roach diet varied both across survey stretch (Pseudo- $F_{2,99} = 2.6$, P = 0.0011) and date (Pseudo- $F_{2,99} = 4$, P = 0.0001). SIMPER analysis showed that differences between stretch 1 and stretch 2 were driven by an increased consumption of filamentous algae (*Cladophora*) in stretch 1 whilst *Bithynia* (gastropod) and gammarids were the dominant prey items in stretch 2. Differences between stretch 1 and 3 related to an increased amount of unidentifiable (due to mastication) invertebrates in stretch 1 and increased consumption of gammarids and gastropods in stretch 3. Dissimilarities between stretch 2 and 3 were due to a high presence of *Bithynia* and *A. aquaticus* from stomachs taken from fish at stretch 2 and a high consumption of trichopteran larvae in stretch 3.

Temporal differences in roach diet between visit 1 and 2 were directly due to high consumption of gammarids in visit 1 and increased consumption of *A. aquaticus* and *Cladophora* in visit 2. Dissimilarities between visits 1 and 3 reflect a high consumption of *Cladophora* during visit 3 while invertebrate prey was the predominant food items in visit 1. Differences in roach diet between visits 2 and 3 again are driven by an increase in consumption of *Cladophora* during visit 3 and an increased consumption of *A. aquaticus* and *Bithynia* in visit 2. Species level comparisons indicated that overall roach diet was statistically similar to eels and roach-bream hybrids in the River Inny.

7.3.3.2 Perch

The interaction between study site and survey date was examined for perch but did not show a significant difference (Pseudo- $F_{4,63} = 1.4$, P = 0.10). However, perch diet did differ between stretches (Pseudo- $F_{2,63}=2.1$, P=0.011) and sampling visit (Pseudo- $F_{2,63}=2.1$, P = 0.018). SIMPER analysis revealed that differences between stretches 1 and 2 were due to an increased presence of *A. aquaticus* from stomachs removed from fish from stretch 1 and a greater consumption of ephemeropteran larvae and gammarids in stretch 2. Dietary differences between stretch 1 and 3 were related to a higher consumption of Ephemeroptera in stretch 1 and 3 were driven by an increased predation of Ephemeroptera larvae in visit 2 and an increased predation of Trichoptera larvae in visit 3.

SIMPER analysis showed that temporal differences in perch diet between visit 1 and 2 related to a greater consumption of *Gammarus* in visit 1 coupled with a greater consumption of ephemeropteran and trichopteran larvae in visit 2. Dissimilarities between stomach contents from visits 1 and 3 reflected an increased consumption of gammarids in visit 1 and an increased consumption of ephemeropteran and trichopteran larvae in visit 3. A greater consumption of trichopteran larvae in visit 2 and the greater consumption of ephemeropteran larvae A. aquaticus and the presence of brook lamprey ammocoetes in stomach contents from fish sampled at site 3 resulted in a statistical difference. Perch diet overlapped with that of eels and chub.

7.3.3.3 Pike

Pike diet differed between survey sites (Pseudo- $F_{2,72}$ =4, P=0.005). No comparisons of temporal variation in pike diet were possible as only a single pike with stomach contents was available from the second survey and none from the third survey. SIMPER analysis revealed that differences between stretch 1 and 2 during the first survey were largely driven by increased consumption of roach and perch at site 1 and a small increase in consumption of brook lamprey at site 2. Lamprey and roach were the dominant prey items for sites 1 and 3 however; consumption of lamprey was far greater at site 3. Again, lamprey and roach were the dominant prey items at both stretch 2 and 3 although consumption of lamprey was somewhat decreased at site 2.

7.3.3.4 Brown Trout

Brown trout diet varied both across survey stretch (Pseudo- $F_{2,91} = 2.3$, P = 0.005) and date (Pseudo- $F_{2,91} = 2.1$, P = 0.005). Stretch 1 trout were distinguished from Stretch 2 individuals by an increased consumption of aerial invertebrates (adult dipterans) and *Theodoxus fluviatilis* (gastropods) in Stretch 1. Dissimilarities between stretch 1 and stretch 3 were again driven by increased presence of dipterans and *T. fluviatilis* from stomachs at stretch 1 and

increased presence of trichopteran larvae in stomachs from trout in Stretch 3. Trout from Stretch 3 consumed more trichopteran larvae, *A. aquaticus*, *Apelocheirus* and adult Diptera than conspecifics from Stretch 1. Consumption of A. aquaticus was increased in stretch 2 and decreased in stretch 3.

Temporal differences in brown trout diet between visit 1 and 2 reflected an increased consumption of *A. aquaticus* and adult dipterans in visit 1 and trichopteran larvae and *Apelocheirus* in visit 2. Dissimilarities in stomach contents from visit 1 and 3 were driven by increased feeding on adult dipterans in visit 1 whilst during visit 3 brown trout were largely feeding on simulid and ephemeropteran larvae. Differences between visit 2 and 3 reflected an increased consumption of ephemeropteran nymphs in visit 3 whilst trichopteran larvae were the dominant prey item in visit 2.

7.3.4 Stable isotope analysis

7.3.4.1 Community level isotopic variation

Macroinvertebrate and fish consumers collected from the River Inny during the April 2008 survey displayed considerable variation in both δ^{13} C and δ^{15} N (Figs. 7.5 and 7.6). δ^{13} C values varied between -36.5 and -20.9 ‰ indicating the contribution of a diverse range of energy sources. River Inny consumer δ^{15} N values varied between 8.2 and 18.4 ‰, which assuming a typical Δ^{15} N value of 3.4‰ between consumers and their diet, suggest the existence of ca. three different trophic levels within the macroinvertebrate and fish community. However, comparison of δ^{15} N values from molluscan baseline indicators revealed significant differences in δ^{15} N values between the three survey stretches along the River Inny (PERMANOVA: Pseudo-F_{2,9} = 8.58, P = 0.011). Consumer trophic position was therefore corrected for baseline variation following Post (2002).

When considered at a community level (Fig. 7.5), macroinvertebrate δ^{13} C and δ^{15} N values were similar across the three survey stretches (PERMANOVA Pseudo-F_{2,173} = 1.53, P = 0.20). However, the δ^{13} C- δ^{15} N centroids for fishes from the three stretches differed (Pseudo-F_{2,328} = 4.1, P = 0.011). Pairwise tests indicated that fish δ^{13} C- δ^{15} N centroids overlapped between stretch 1 and 2 (t = 0.66, P = 0.62) but differed between stretches 1 and 3 (t = 2.6, P = 0.006) and 2 and 3 (t = 2.2, P = 0.02).



Figure 7.5: Stable isotope biplots comparing variation in δ^{13} C- δ^{15} N in fish and invertebrates collected from each of the three survey stretches of the River Inny, A) = Stretch 1, B) = Stretch 2 and C) = Stretch 3. Data from chub collected prior to the survey (n = 3, Stretch 1; n = 3, Stretch 3) are combined with the two individuals captured during the study provide an indication of the long-term diet and ecology of the invasive chub in the River Inny.

Comparisons of mean (±SD) δ^{13} C and δ^{15} N for each of the fish species encountered from the three survey stretches (Fig. 7.6) showed a general structuring by different species. Roach were typically ¹³C depleted whilst chub were typically ¹³C enriched. Pike were typically the most ¹⁵N enriched of the species encountered, but showed considerable individual variation. The δ^{13} C- δ^{15} N isotope space occupied by chub was similar to that of salmonids, rather than to other cyprinid fishes (note overlap with brown trout (Fig. 7.6A and C) and salmon (Fig. 7.6C). Adult brook lampreys collected during their spawning migration were notably ¹⁵N depleted and ¹³C enriched relative to other fishes (Fig. 7.6A).



Figure 7.6: Stable isotope biplots comparing variation in mean (±SD) δ^{13} C and δ^{15} N in fishes collected from each of the three survey stretches of the River Inny, A) = Stretch 1, B) = Stretch 2 and C) = Stretch 3.

In order to gather a general impression of isotopic differences between species encountered during the study, and to understand the ecology of chub relative to native (eel, brown trout, salmon, brook lamprey), and other non-native fishes (roach, perch, pike, minnow, gudgeon, bream, roach-bream hybrids), data were pooled for all stretches and compared using PERMANOVA. There was a strong species effect on δ^{13} C- δ^{15} N location (Pseudo-F_{11,319} = 21.8, P = 0.0001), indicating that at a larger geographical scale, the fish species examined here differed isotopically. Pairwise comparisons of isotopic differences between species (Table 7.5) showed significant differences in 44 of the 66 individual comparisons. Significant isotopic overlap was indicated in 22 species comparisons. Invasive chub were statistically indistinguishable from three native and conservationally-important species: brown trout, salmon and

eels. Brown trout overlapped isotopically with eels, gudgeon, roach-bream hybrids, minnows, and salmon. Salmon δ^{13} C- δ^{15} N centroids were statistically similar to bream, eels, gudgeon, roach-bream hybrids, minnows and perch.

Roach showed significant overlap with only two taxa: gudgeon and roach-bream hybrids. Pike showed unexpected isotopic overlap with bream. The sample of bream analysed here was collected from stretch 3. All individuals were large bodied (fork length range = 427 - 580 mm) individuals and were in spawning conditions (*i.e.* they displayed spawning tubercles). Their elevated δ^{15} N values likely indicates that they entered the River Inny from the eutrophic (and presumably ¹⁵N-enriched) Lough Ree (located directly downstream of their capture site) rather than that they had a piscivorous diet (Anderson & Cabana, 2005).

Table 7.5: Probability (P values) of pairwise PERMANOVA comparisons of δ^{13} C- δ^{15} N isotope overlap between different fishes. Comparisons that are statistically similar (P ≥ 0.05) are shown in bold, and reflect potential dietary overlap (i.e. there was overlap in δ^{13} C- δ^{15} N centroids between the two species). Note that samples were pooled across sites to provide an overall comparison of the ecology of chub and other species in the River Inny. Fish species names are abbreviated as: bream = Br; brown trout = Bt; chub = Ch; Eel = Ee; gudgeon = Gu; roach x bream hybrid = RoxBr; brook lamprey = Bl; minnow = Mn; perch = Pe; pike = Pi; roach = Ro and salmon = Sa.

	Bt	Ch	Ee	Gu	RoxBr	BI	Mn	Pe	Pi	Ro	Sa
Br	0.02	0.0001	0.005	0.002	0.08	0.0001	0.003	0.15	0.06	0.017	0.07
Bt	—	0.11	0.09	0.11	0.17	0.0001	0.71	0.0001	0.0001	0.0001	0.65
Ch		—	0.45	0.0003	0.0008	0.0001	0.007	0.0008	0.0004	0.0042	0.08
Ee			_	0.005	0.009	0.0002	0.006	0.006	0.004	0.02	0.09
Gu					0.13	0.0001	0.008	0.58	0.0009	0.35	0.31
RoxBr						0.0001	0.02	0.96	0.03	0.22	0.34
BI						_	0.0001	0.0001	0.0001	0.0001	0.0001
Mn							-	0.003	0.0001	0.008	0.66
Pe									0.0001	0.001	0.05
Pi										0.0001	0.0001
Ro										_	0.05
Sa											_

7.3.4.2 Species-level spatial isotopic variation

Within species variation in δ^{13} C- δ^{15} N and TP (corrected for baseline variation) was examined across the different stretches for those species with large sample sizes (brown trout, pike, perch and roach).

Roach

Differences in the location of roach δ^{13} C- δ^{15} N centroids were on the threshold of conventional statistical significance, suggesting overall isotopic similarity across the three different survey stretches (Figs. 7.6 & 7.7A; Pseudo-F_{2.65} = 3.11, P = 0.05). Pairwise comparisons indicated that roach from stretch 1 and 3 were isotopically distinct (P = 0.03). Comparison of total isotopic variation, i.e. relative area of δ^{13} C- δ^{15} N convex hulls (Layman *et al.*, 2007a) indicated that roach were most isotopically variable in stretch 1 (99 704 pixels), intermediate in stretch 3 (44 844) and least variable in stretch 3 (13 644).

Roach did not feed at the same trophic level across the three survey stretches (Fig. 7.7B: Pseudo- $F_{2,65}$ = 27.8, P = 0.0001): pairwise comparisons revealed differences between stretches 1 and 3, and stretches 2 and 3 (both P = 0.0001), but roach TP overlapped in individuals from stretch 1 and 2 (P = 0.31). When compared across all stretches, roach fed at a slightly higher estimated TP than that of chub (Fig. 7.7B: t = 2.5, P = 0.015).



Figure 7.7: Variation in roach A) δ^{13} C- δ^{15} N and B) estimated trophic position (from δ^{15} N data) in the three survey stretches on the River Inny. NB: Arrows in B) show estimated TP for chub in the two different stretches.

Perch

Comparison of perch δ^{13} C- δ^{15} N centroids through PERMANOVA (Figs. 7.6 & 7.8A) indicated spatial differences between the survey stretches (Pseudo-F_{2,44} = 3.31, P = 0.023). Pairwise comparisons showed isotopic overlap in perch from stretch 2 and 3 (P = 0.12), and small isotopic differences between stretch 1 and 2 (P = 0.029) and stretch 1 and 3 (P = 0.039). Perch showed the greatest level of relative isotopic variation (Layman *et al.*, 2007a) from stretch 3 (convex hull area = 103 987 pixels), whilst individuals from stretch 1 (44 628 pixels) and stretch 2 (57 589 pixels) both showed similar and reduced amounts of δ^{13} C- δ^{15} N variation. Perch trophic level varied considerably across the three survey stretches (Pseudo-F_{2,44} = 33.7, P = 0.0001). Pairwise comparisons revealed overlap in TP in perch collected from stretch 1 and 2 (P = 0.13). TP differed between perch from stretches 1 and 3, and stretches 2 and 3 (both P = 0.0001). When data were pooled across survey stretches, perch fed at a higher estimated TP than chub (t = 2.88, P = 0.006).



Figure 7.8: Variation in perch A) δ^{13} C- δ^{15} N and B) estimated trophic position (from δ^{15} N data) in the three survey stretches on the River Inny. NB: Arrows in B) show estimated TP for chub in the two different stretches.

Pike

Pike were isotopically distinct (δ^{13} C- δ^{15} N centroids) between the three survey stretches (Figs. 7.6 & 7.9A: Pseudo-F_{2,90} = 34.52, P = 0.0001). This overall comparison was supported by multivariate pairwise comparisons of pike δ^{13} C- δ^{15} N values that showed no overlap between pike from all three stretches (all comparisons: P = 0.0001). Isotopic variation (Layman *et al.*, 2007a) was reduced in individuals from stretch 3 (convex hull area = 39 470 pixels), compared to individuals from stretch 1 (83 773 pixels) and stretch 2 (79 877 pixels). Pike did not feed at the same trophic level across the three survey stretches, (Fig. 7.9B: Pseudo-F_{2,90} = 66.97, P = 0.0001). Pike fed at the greatest TP at survey stretch 1, and the lowest in stretch 3: pairwise comparisons revealed differences from each of the stretches (all comparisons P = 0.0001). As expected, pike TP values were elevated relative to chub (Fig. 7.9B: t = 5.6, P = 0.0001).



Figure 7.9: Variation in pike A) δ^{13} C- δ^{15} N and B) estimated trophic position (from δ^{15} N data) in the three survey stretches on the River Inny. NB: Arrows in B) show estimated TP for chub in the two different stretches.

Brown trout

Brown trout δ^{13} C- δ^{15} N centroids differed slightly across the different survey stretches (Fig. 7.6 & Fig. 7.10A; Pseudo-F_{2,44} = 3.1, P = 0.03), but pairwise comparisons indicated that differences in centroids were only apparent between trout captured in Stretch 2 and 3 (P = 0.01). Isotopic variation (as measured by δ^{13} C- δ^{15} N convex hulls) was greatest in fish collected from stretch 2 (131 035 pixels), intermediate in stretch 1 (90 381 pixels) and lowest in stretch 3 (54 337 pixels). Comparisons of brown trout TP across the three survey stretches showed considerable spatial differences (Fig. 7.10B: Pseudo-F_{2,44} = 13.7, P = 0.0001). Pairwise comparisons revealed that TP was higher in fish from stretch 2 relative to both stretch 1 (t = 2.15, P = 0.038) and stretch 3 (t = 4.47, P = 0.0004). TP was higher in trout collected from stretch 1 relative to stretch 3 (t = 3.75, P = 0.001).



Figure 7.10: Variation in brown trout A) δ^{13} C- δ^{15} N and B) estimated trophic position (from δ^{15} N data) in the three survey stretches on the River Inny. NB: Arrows in B) show estimated TP for chub in the two different stretches.

7.3.5 Fish age and length

7.3.5.1 Roach

Roach captured from the River Inny ranged in fork length 19 to 304 mm and in age 1 to 3 years. PERMANOVA revealed differences in roach fork length (Fig. 7.11) between study sites (Pseudo- $F_{2,213} = 3.85$, P = 0.023), although survey date had a greater influence on roach size (Pseudo- $F_{2,213} = 19.7$, P = 0.0001). A significant interaction between these two factors revealed that roach size structure from the different survey sites showed separate temporal patterns (Pseudo- $F_{4,213} = 9.7$, P = 0.001). There was no apparent effect of either location (Pseudo- $F_{2,213} = 0.2$, P = 0.66) on roach age structure during the study. There was evidence for interannual variation in roach growth between sites with significant differences in lengths shown at ages 1 to 3 (Table 7.6).

Age	Survey stretch	Survey date	Interaction
1	Pseudo- $F_{2,213} = 9.31$	Pseudo-F _{2,213} = 5.3	Pseudo- $F_{4,213} = 44.0$
	P=0.001	P =0.011	P = 0.001
2	Pseudo- $F_{2,201} = 9.67$	Pseudo-F _{2,201} = 9.02	Pseudo-F _{4,201} =23.70
	P = 0.001	P = 0.002	P = 0.001
3	Pseudo- $F_{2,176} = 4.82$	Pseudo-F _{2,176} =22.16	Pseudo-F _{4,176} = 11.30
	P = 0.01	P = 0.001	P = 0.001
4	Pseudo-F _{2,127} = 1.27	Pseudo-F _{2,127} = 21.31	Pseudo-F _{4,127} = 7.26
	P = 0.284	P = 0.001	P = 0.001
5	Pseudo-F _{2,79} = 0.80	Pseudo-F _{2,79} = 8.83	Pseudo-F _{3,79} = 2.14
	P = 0.45	P = 0.102	P = 0.102
6	Pseudo-F _{2,44} =2.88	Pseudo-F _{2,44} = 1.91	Pseudo-F _{1,44} = 1.12
	P = 0.055	P = 0.143	P = 0.288
7	Pseudo- $F_{2,26} = 2.0$	Pseudo-F _{2,26} = 2.30	$Pseudo-F_{1,26} = 0.38$
	P = 0.141	P = 0.112	P = 0.576

Table 7.6: PERMANOVA table of results	showing effect of location,	sampling date and the in	nteraction between
age and sampling date on roach length a	t age.		



Figure 7.11: Variation in mean (\pm SD) back calculated length at age for roach collected from the three survey stretches: A) Stretch 1, B) Stretch 2 and C) Stretch 3.

7.3.5.2 Perch

Perch sampled from the River Inny ranged in length 59 to 287 mm and in age from 1 to 7 years (Fig. 7.12). PERMANOVA analyses revealed overall differences in perch fork length between study sites (Pseudo- $F_{2,83}$ =3.43, P = 0.042) but provided no evidence for significant temporal differences (Pseudo- $F_{2,83}$ = 1.01, P=0.332). Interannual variation of length between stretches and survey dates was only evident at age 4 (Table 7.7).

Table 7.7: PERMANOVA	table of results	s showing effect	of location a	and sampling d	ate on perch	back-calculated
length at age.						

Age	Survey stretch	Survey date
1	Pseudo-F _{2,82} = 2.24, P =0.109	Pseudo- $F_{2,82}$ = 2.01, P = 0.14
2	Pseudo- $F_{2,61} = 0.34$, P = 0.68	Pseudo- $F_{2,61} = 0.67$, P = 0.50
3	Pseudo-F _{2,32} = 0.05, P = 0.93	Pseudo- $F_{2,32} = 0.04$, P = 0.94
4	Pseudo-F _{2,9} = 11.52, P = 0.003	Pseudo- $F_{2,9}$ = 4.643, P = 0.039



Figure 7.12 - Variation in mean (\pm SD) back calculated length at age for perch collected from the three survey stretches: A) Stretch 1, B) Stretch 2 and C) Stretch 3.

7.3.5.3 Pike

Pike taken from the River Inny ranged in fork length 194 to 1000 mm and in 0+ to 10 years. Pike exhibited no significant spatial differences in fork length between the different survey stretches (Pseudo- $F_{2,52} = 1.92$, P = 0.16 or between survey date (Pseudo- $F_{2,52} = 0.15$, P = 0.87) (Fig. 7.13). There was however evidence for interannual variation in growth between sites with significant differences in lengths shown at ages 1 to 4 (Table 7.8).

Age	Survey stretch	Survey date
1	Pseudo- $F_{2,52} = 2.39$	Pseudo- $F_{2,52}$ = 2.39
	P = 0.101	P = 0.024
2	$Pseudo-F_{2,41} = 1.58$	Pseudo-F _{2,41} = 5.68
	P = 0.216	P = 0.005
3	$Pseudo-F_{2,28} = 0.61$	Pseudo- $F_{2,28} = 6.39$
	P = 0.5986	P = 0.004
4	$Pseudo-F_{2,16} = 1.79$	Pseudo- $F_{2,16} = 8.50$
	P = 0.172	P = 0.0023

Table 7.8: PERMANOVA table of results showing effect of location and sampling date on pike length at age.



Figure 7.13 - Variation in mean (\pm SD) back calculated length at age for pike collected from the three survey stretches: A) Stretch 1, B) Stretch 2 and C) Stretch 3.

7.3.5.4 Brown trout

Brown trout captured during the electrofishing surveys ranged in fork length from 68 to 611 mm and in age from 1 to 9 years. Brown trout generally exhibited slow growth in the River Inny (Fig. 7.14). Brown trout collected from the second survey stretch included a sub-component of rapidly growing individuals (shown separately in Fig. 7.14B). When compared by PERMANOVA, there was no apparent effect of location (Pseudo- $F_{2,94} = 0.52$, P = 0.52) or survey date (Pseudo- $F_{2,94} = 0.58$, P = 0.53) on brown trout fork length. However, there was a large effect of survey date on mean age (Pseudo- $F_{2,94} = 16.72$, P = 0.001). It was also noted that survey date had an effect on the length of brown trout in all age classes (Table 7.9).

Table	7.9:	PERMANO	A table	of	results	showing	effect c	of location	and	sampling	date	on brown	trout	length	at
age.															

Age	Survey stretch	Survey date
1	Pseudo-F _{2,94} = 0.38	Pseudo- $F_{2,94} = 4.80$
	P = 0.668	P = 0.022
2	Pseudo- $F_{2,74} = 3.77$	Pseudo-F _{2,74} =28.75
	P = 0.029	P = 0.001
3	Pseudo- $F_{2,42} = 0.63$	Pseudo-F _{2,42} = 22.81
	P = 0.534	P = 0.001
4	Pseudo- $F_{2,24} = 0.62$	Pseudo-F _{1,24} = 11.667
	P =0.539	P = 0.004
5	$Pseudo-F_{0,10} = 0.35$	_
	P = 0.731	



Figure 7.14: Variation in mean (\pm SD) back calculated length at age for brown trout collected from the three survey stretches: A) Stretch 1, B) Stretch 2 including overall mean (\oplus), fast growing (\blacktriangle), and slow growing (∇)individuals) and C) Stretch 3.

7.3.5.5 Chub

A previous study (Caffrey *et al.* 2008) found that chub growth in the River Inny exceeded known rapid growth rates of chub from British waters (Cowx, 2001). A further 2 chub were captured during this study: these measured 326 and 400 mm respectively, and were found to be 5 and 10 years of age and followed the same growth patterns described by Caffrey *et al.* (2008).

7.3.6 Movement and behaviour of invasive chub

In April and August 2008 two mature male chub (fork length 381 and 311 mm respectively) were captured in stretch 3 (upstream of Shrule Bridge). Both individuals were fitted with VHF radio transmitters and returned to the same section of river from where they had been captured. Regular monitoring over the next 9 months provided no evidence of any long-distance movements. The chub remained reasonably proximal to each other and occupied a normal range of *ca.* 600 m (Fig. 7.15).

In May 2009 one of the tagged chub was recaptured upstream of Shrule Bridge, in almost the same location where it was originally captured almost 13 months previously. It was the intention to remove these tagged fish, along with any other chub present, before they had an opportunity to spawn. There was no evidence of the second tagged chub on this occasion, although the 2 km long stretch at Shrule Bridge was electrofished on 4 separate occasions over two days. The recaptured chub measured 326 mm and weighed 568 g. This represented an increase in length and mass over the 9 month release period of 15 mm and 106 g, respectively. In May 2010 another tagged chub was captured during an electrical fishing operation at Shrule Bridge. This was the first of the two chub that were tagged in 2008. This fish had been at liberty for 25 months and had increased in fork length by 19 mm and in mass by 137g. Both fish were humanely euthanized.



Figure 7.15: Schematic map showing the capture / release sites and relative movements of radio – tagged chub in the vicinity of stretch 3.

Combined measures of chub location (relative to a fixed datum) showed very little evidence of large-scale movements by either chub. For much of the study, both fish were located ca. 109 m (\pm 95% CI = 35) upstream of the datum. However, following a period of intense rainfall and a shift in water levels, both fish made a relatively long-range upstream movement (Fig 7.16A). During this period, they utilised an area ca. 1 021 \pm 20 m upstream of the datum. Combining data from these two periods, bootstrapped estimates indicate that 95% of locations of the two tagged chub were made within an area of only 164 m. Movements by chub between individual tracking surveys (Fig. 7.16B) were typically small (mean \pm 95 % CI = 183 \pm 105 m) but included several large (>900 m) movements, as chub responded to increased water levels following floods. When these values were corrected for variation in the period between individual location estimates, the mean bootstrapped estimate of movement velocity (Fig. 7.16C) was revealed to be very small (13 \pm 7.5 m.d⁻¹).

Although based on only two individuals, taken together, these data indicate that chub in the River Inny showed considerable fidelity to a small area of river, and even considering movements in response to flood, the bulk of their activities were located in a stretch of < 200 m in length.



Figure 7.16: Frequency histograms showing variation in various measures of chub activity and habitat use estimated through telemetry: A) Location of radio tracked chub (n = 2) above a fixed datum; B) movement between individual location estimates and C) estimated velocity of chub movements in m.d⁻¹ between individual location estimates.

7.4 Discussion

This study aimed to examine the key impacts of the introduction of chub a potential IAS on the community and function of the River Inny. As such, this study aims to present methods in which to examine the adverse impacts of invasive species on riverine ecosystems in Ireland (Richter *et al.*, 1997). Williamson (1996) suggested that approximately 1% of freshwater fish introductions are likely to result in serious adverse ecological effects. There is always an inherent risk that potentially catastrophic and irreversible ecological consequences will result after the introduction of a non-native fish (Reinthal & Kling, 1994; Vander Zanden *et al.*, 1999; Baxter *et al.*, 2004; Britton *et al.*, 2010a; Britton *et al.*, 2010c). However, we showed no obvious effects of chub largely due to successful and timely control of this potential invasion.

Our results showed no obvious ecological effects of the chub invasion. Although our analysis of macroinvertebrate abundance was limited by an inability to routinely sample each survey stretch due to hazardous water conditions, analysis of these samples indicated that macroinvertebrate community structure did not differ significantly between survey stretches but as expected there was a strong seasonal effect between survey dates. No obvious response to chub predation pressure was detected.

The fish assemblage of the River Inny consisted of 13 species (including roachxbream hybrids) and was heavily dominated primarily by roach, itself a non-native fish – with contrasting habitat requirements to those of chub. However native brown trout were also abundant. Of the fishes encountered, we recorded strong seasonal variation of CPUE data at all sites. This was likely caused by the intense movement and spawning migration of cyprinids during survey 3. Stomach content analysis of the fishes in the Inny indicated that there was a strong seasonal and spatial variation as well as strong temporal–spatial interactions at a community level. Further results from the stomach content analysis hint at a potential dietary overlap between chub, eels, gudgeon, roach x bream hybrids and perch: however, the reliability of these results is questionable due to the limited sample size of chub. The use of stable isotope analysis of a larger sample of chub raised concerns as it indicated that the long term assimilated diet of chub was generally similar to that of the native and conservationally important brown trout, eel and salmon.

Potential overlap between chub and these species has also been noted in other studies (Hellawell, 1971; Mann, 1976; Caffrey *et al.*, 2008).

Fish growth is often used as a means of assessing temporal and spatial variation in the performance of fish stocks (Francis, 1990; Britton, 2007). Analyses from this study revealed that brown trout, perch and roach exhibited relatively slow growth in the Inny (Maitland & Campbell, 1992). Chub captured from the Inny initially displayed slow growth until the age of 5, but then growth increase at an accelerated rate that exceeded that of fast growing chub from English waters (Cowx, 2001). Growth remained linear even up to the age of 10, when the River Inny chub achieved a fork length of 371 mm, considerably greater than the mean length at this age attained by chub from English rivers (348).

Following the first reports of the presence of chub in the River Inny, a total of 28 chub were removed from the river. Twenty-six of these were mature (19 male and 7 female). Only two immature chub (age 3+) were captured, both in 2006 at Shrule Bridge and it is unknown whether the represented fish spawned in the River Inny. No female chub were recorded in the river after September 2007. It appears that the rapid control activities by the relevant fisheries boards coupled with an initial apparent low propagule pressure (Lockwood *et al.*, 2005) and a limited population size prevented the successful establishment of chub as an invasive alien species in the River Inny. All chub removed from the River Inny to date were collected from two discrete stretches of river – at Shrule (stretch 3, N = 26) and Ballinalack (stretch 1, N = 2): both areas are characterised by rapid flow and shallow water depth, with substrates comprised mainly of gravel and cobble. In the native range of chub, such sections could be described as optimum summer habitat for this species (Maitland & Campbell, 1992; Cowx & Welcomme, 1998). The River Inny has been heavily modified from dredging operations and is mainly comprised of deep (> 2 m) and slow flowing sections, and access to areas where shallow water flowing over gravel substrate (e.g. survey stretch 3) is limited. This combined with the small population size, and the limited movements displayed by tagged chub in the river, may have been a natural inhibiting factor which prevented the dispersal of chub further afield.

The link between the River Inny and the River Shannon catchment through Lough Ree and ultimately the River Erne system, was of serious concern to fisheries biologists in terms of possible migration of invasive chub. However the rapid assessment made by IFI allowed infested sites to be located, and control was promptly applied, rapidly minimising an already limited chub population size. As the population size diminished, a telemetry survey was conducted were 2 chub were radio-tagged and their movements were monitored regularly. The intention was to use these fish to characterise movements and the location of shoals or congregations. The telemetry survey showed little evidence of long distance movement in the River Inny, Chub typically occupied a core area of < 200 m, however movements of a further 600 m where observed during flood events. These findings consistent with previous work on chub in British waters (Bolland *et al.*, 2008) and indicate that they may be susceptible to control.

In the River Inny it appeared that within an 850 metre section of river, chub could avail of suitable habitat to survive in high/low water and rapid/slow flow. Control surveys in 2009 and 2010 removed the 2 tagged chub from the river plus one other untagged chub. More recent surveys conducted by Inland Fisheries Ireland have failed to find any chub in the River Inny (see Section 9.2) indicating the species has been eradicated from the River Inny.

7.5 Conclusions

We examined several key aspects of the ecology of three different stretches of the River Inny, allowing a comparison of the ecology of macroinvertebrates and fish in invaded and non-invaded stretches. Chub were only present in very limited numbers during the survey period and there was no evidence of any ecological impact of invasion. However, using stable isotope analysis, we demonstrated that the trophic ecology of chub in the River Inny to overlap with that of three important native fishes: eels, brown trout and Atlantic salmon. This combined with the suitability of many Irish rivers for chub and the known overlap in habitat preferences (Maitland & Campbell, 1992; Caffrey *et al.*, 2008) highlights the considerable potential for chub to become an IAS in Ireland if not controlled, with implications for the ecology, management (e.g. WFD) and conservation (e.g. Habitats Directive) status of Irish freshwaters.

Given the potential for chub to impact Irish waters, thought needs to be directed at the provision of surveillance for the introduction and establishment of chub (see Chapter 4) both in the River Inny and other waters across Ireland. As the potential for reintroduction remains, annual electrofishing monitoring and control surveys following the methods used in the current study are recommended, and should limit the capacity of chub to become established in the River Inny.

8 Preventing and containing IAS introductions: case studies of *Lagarosiphon* and chub

8.1 Policy context

Preventing new introductions of IAS is particularly important in the aquatic environment given the difficulty in eradicating or managing introductions once species have become established. The Convention on Biological Diversity (CBD) sets out 15 guiding principles for the prevention and mitigation of impacts of IAS that threaten ecosystems, habitats or species (UNEP/CBD, 2002). These principles provide an international framework for Governments and other organisations to develop effective strategies while recognising that every country faces unique challenges and will need to develop context-specific solutions. The precautionary approach and the emphasis on prevention underpin the guiding principles. Their purpose is to assist Governments in combating invasive species recognising the extent to which they can be implemented depends on available resources. The CBD requires the Irish government 'as far as is possible and appropriate, to prevent the introduction of, control or eradicate those alien species which threaten ecosystems, habitats or species'. In October 2010, the CBD Parties approved the following IAS target that 'by 2020, invasive alien species and pathways are identified and prioritised, priority species are controlled or eradicated, and measures are in place to manage pathways to prevent their introduction and establishment'.

At a European level, the Habitats Directive and Birds Directives contain explicit IAS prevention obligations (EC, 1992). The Habitats Directive requires Member States to 'ensure that the deliberate introduction into the wild of any species which is not native to their territory is regulated so as not to prejudice natural habitats within their natural range or the wild native fauna and flora and, if they consider it necessary, prohibit such introduction' (Article 22b). The Birds Directive requires Member States to 'see that any introduction of species of bird which do not occur naturally in the wild state in the European territory of the Member States does not prejudice the local flora and fauna' (Article 11). At a national level, the National Biodiversity Plan contains a target to 'reduce the risk of establishment and spread of new species and control harmful invasive alien species'.

8.2 Economic case for investment in prevention

Recent estimates of the cost of IAS in the EU through lost outputs, health impacts and expenditure to repair IAS damage were at least \in 12 billion per year over the past 20 years, of which costs identified for key economic sectors have been estimated at over \in 6 billion per year (Kettunen *et al.*, 2009). This is a conservative estimate as many costs were unable to be included. A recent report for Great Britain estimated that invasive species cost the British economy at least £1.7 billion each year (Williams *et al.*, 2010). Little information exists on the economic impact of many species so this is likely to be significantly less than the full economic cost since many indirect costs, such as the damage to ecosystem services and loss of biodiversity, cannot be readily quantified. Of particular relevance to this project was the cost to the leisure and tourist industry. The growth of this sector and ecotourism is an important part of the Irish Government's economic renewal plans and is vulnerable to IAS impacts on navigable waterways, boating and angling with the resulting reduction in the amenity value of waterbodies. In Britain, the cost of IAS to

recreational boating was estimated at £30.45 million, with an additional £21.86 million cost to waterway management and £4.89 million to angling per annum.

Costs were also calculated for many of the priority species which are the focus of this project, further highlighting the benefits of prevention, surveillance and rapid response. Of the riparian plants species, Japanese knotweed (Fallopia japonica) which is widely established in Ireland has the highest annual cost to the British economy at £179 million, of which, £5.6 million is incurred for control in riparian habitats. Giant hogweed (Heracleum mantegazzianum) costs £965 thousand and over £1 million is spent controlling Himalayan balsam (Impatiens glandulifera) annually. Annual costs have also been estimated for six of the priority aquatic plants species. Floating pennywort (Hydrocotyle ranuculoides) which has a relatively restricted distribution in Ireland, is more widely established in England and Wales where it costs the economy £25 million per year. New Zealand pigmyweed (Crassula helmsii), water fern (Azolla filiculoides), parrot's feather (Myriophyllum aquaticum) and curly leaved waterweed (Lagarosiphon major) are estimated to have an annual cost of £3 million, with Nuttall's pondweed (Elodea nuttallii) costing £4 million per annum. Costs to the British economy have also been estimated for species which are potential invaders to Ireland IAS including signal crayfish (Pacificastacus leniusculus) at £2.69 million per annum. The successful eradication of five populations of topmouth gudgeon (Pseudoboras parva) has cost £190,000 and the species is currently controlled in England at an average annual cost of £50,000 (Britton et al., 2010b). What these and other estimates of the economic cost of IAS demonstrate is the clear economic justification for investment in preventing species invasions. Putting in place a strong framework with an emphasis on prevention will bring more benefits through avoided costs than the actions will cost to implement. The economic impacts of IAS in Ireland are currently being estimated in a similar analysis to Williams et al. (2010) with a report due in autumn 2011 (Maguire et al., in prep).

8.3 **Project objectives and contribution to national research needs**

There are a range of key prevention measures that can be put in place such as legislative provisions banning import, sale and spread of species; risk assessment and the development of priority/alert lists; identification and management of key vectors and pathways; voluntary measures such as Codes of Practice and industry standards; and an early warning system linked to interactive information portals that also shares information with neighbouring countries or at a regional level. All of these measures have been developed to different extents in Ireland to date including legislative provisions, risk assessment, alert lists, exclusion and contingency plans for species, Codes of Practice and the National Invasive Species Database (see Chapter 9 for further details).

However there are still significant knowledge gaps about the relative importance of the pathways which facilitate spread of invasive species and what the key prevention actions are that will reduce the risk of IAS introductions. This project has sought to address this for the priority aquatic IAS and to build on developments to date by identifying the key pathways and vectors and measures and prevention measures for each (Chapter 9). Containment and control measures have been evaluated for the priority species and management actions are recommended. Control measures for chub and *Lagarosiphon* have also been evaluated in the field. Together these outputs will provide an important resource to inform the design and implementation of programmes of measures to reduce IAS pressures on Irish waterbodies. There are two project objectives relating to prevention and containment, namely:

- WP1 Objective 3: Develop a strategy for containing and preventing high impact aquatic IAS introductions.
- WP2 Objective 3: To examine the efficacy and consequences of measures aimed to control key invasive species including an invasive plant *Lagarosiphon major*, and an invasive fish (*Leuciscus cephalus*).

8.4 *Control of Lagarosiphon major* in Lough Corrib

The project had intended to examine the efficacy and consequences of control measures for *Lagarosiphon major* in Lough Corrib. The resources provided by this project contributed to on-going control work being undertaken by Inland Fisheries Ireland (IFI). It soon became clear that resources were needed on a much greater scale to control *Lagarosiphon* and IFI secured funding from the National Parks and Wildlife Service and the European Union's LIFE+ programme for a large scale project to control *Lagarosiphon in* Lough Corrib and aquatic IAS in the Grand Canal and Barrow Navigation. Therefore, the limited project resources were refocused on further investigating the ecological aspects of *Lagarosiphon*. A summary of the work undertaken to date and to which the project contributed is provided below.

8.4.1 Summary of IFI Lagarosiphon control programme

8.4.1.1 Manual removal by divers

Manual removal of *Lagarosiphon* by SCUBA divers is operated at sites where the plant has just recently invaded and where the level of vegetative abundance is low. It is also used in areas that are inaccessible to weed cutting or spraying boats and at the fringes of *Lagarosiphon* beds where other control methods have been used. While this method of control can be tedious, it is acknowledged to be an effective long-term strategy for the control/eradication of *Lagarosiphon* in such circumstances (Clayton, 2003).

Between 2007 and 2010 some 73 sites in the lake that supported small and isolated populations of *Lagarosiphon* were identified and cleared using hand removal. Subsequent monitoring revealed that little or no regrowth of *Lagarosiphon* was recorded at these sites in the months following treatment.

8.4.1.2 Mechanical removal and harvesting

Mechanical cutting provides a fast method for removing obstructive vegetation. However, the method can be shortterm, however, as the weed commonly grows back rapidly and can again present obstructive stands within weeks of the initial cut. In 2007 a series of trials were conducted on Lough Corrib. Paired V-blades or trailing knives were attached to a length of chain (normally 8m long) and trailed behind a weed cutting boat. The outer edges of the blades were blunt so that the targeted vegetation was ripped from the substrate rather than being cleanly cut. This caused greater trauma to the plant and also removed a considerable amount of root material. The cut weed in all cases was immediately harvested and brought ashore. The trials revealed that regrowth in the cut plots was less than 10%, and much of this resulted from the regrowth of plant fragments that floated into the plots (Caffrey & Acevedo, 2008).

In 2008 a weed cutting boat was purchased and dedicated to *Lagarosiphon* control on the upper lake. Previous research had demonstrated that the most efficient cut is achieved where the weed is cut during its active growing

phase, in winter (Caffrey *et al.*, 2009). At this time the stems are buoyant which makes it easy for the boat operator to see the area of lake that requires cutting and renders the weed easier to cut. In addition, because the stems are buoyant at this stage in the plant's growth cycle, the cut weed floats to the water surface and can be efficiently harvested.

8.4.1.3 Herbicide treatment

Up to March 2010 dichlobenil was the only approved herbicide available for use against submerged weed species in Ireland. Research has demonstrated that, when used in strict accordance with the manufacturers' recommendations, this herbicide has minimal adverse impact on water quality, non-target plants, macroinvertebrates or fish (Caffrey, 1993a; 1993b). From here it is taken up by the target plants *via* root uptake.

Dichlobenil was used to treat a number of localised trial sites infested with *Lagarosiphon* in Lough Corrib. Where the invasive weed grew vigorously and presented a surface canopy, the granules were unable to penetrate to the lake bed and a low level of weed control was achieved (Caffrey & Acevedo, 2008). Where the canopy layer was removed, however, sufficient chemical to effectively kill the weed was absorbed by root uptake. This demonstrated that *Lagarosiphon* is susceptible to the activity of dichlobenil but that treatment will only be effective when the weed bed is sufficiently open to allow the granules to reach the substrate.

The herbicide is used to best effect in discrete areas that are inaccessible to boats, in the vicinity of small marinas or harbours or to deal with localised plant stands that were missed during weed cutting operations. In the last three years, 103 such locations have been treated with dichlobenil. This represents a total area of 7.1 ha. The results have been excellent and subsequent monitoring has revealed almost total control in treated areas.

8.4.1.4 Light exclusion

The exclusion of light from submerged vegetation will inhibit the life-giving process of photosynthesis and result in the death of affected plants. In early 2008 a light excluding material made of natural fibre was trialled on *Lagarosiphon* in Lough Corrib (Caffrey *et al.*, 2010). Rolls of jute matting (5 m wide × 900 m long, *circa* 200 gm⁻²) were sourced and deemed suitable for these trials. Jute is a natural vegetable fibre and the matting used is loose-woven and durable. Another beneficial characteristic of the product for application in large lake situations is the fact that it saturates on contact with water and rapidly sinks, thus limiting the impact that wind can have on its accurate placement.

Seven *Lagarosiphon* dominated sites were treated with jute matting between August 2008 and October 2009. The areas of lake bed covered varied between 100 and 5000 m². The length of time the jute matting was in place ranged from 4 to 17 months. At the end of the trial period a series of measurements were recorded at each site. These data were recorded by divers using five randomly placed 0.25 m2 quadrats at each site. The status of *Lagarosiphon* beneath the matting was assessed *in situ* by making a small incision in the material and inspecting the plant material, both visually and manually. The extent of sedimentation was assessed visually.

At all seven treated sites the growth of *Lagarosiphon* was stopped by the placement of the light exclusion barrier (Caffrey *et al.*, 2010). The invasive weed was completely decomposed beneath the matting, even where the

matting was in place for only four months. At all of the sites where seven or more months had elapsed since the jute matting was put in place, some level of growth among native macrophyte species was recorded through the weave of the jute. The charophyte *Nitella flexilis* was present at each of these sites. This is one of the keystone species present in lower Lough Corrib, where *Lagarosiphon* remains unrecorded. Three other charophyte species, namely *Chara globularis*, *Chara rudis* and *Chara virgata* were also recorded growing through the jute matting. At five of the sites an appreciable percentage cover among native charophyte species was recorded.

Because of the success of these trials the primary method for control of *Lagarosiphon* is jute matting. To date, 116 discrete *Lagarosiphon* infested areas in the lake, covering *c*. 1.6 ha, have been treated. Excellent results have been achieved and *Lagarosiphon* has never been observed to grow through the matting while the native vegetation is recolonising the treated areas.

The control programme for Lagarosiphon in Lough Corrib will only be successful if new introductions or reinvasion can be prevented. At present Lagarosiphon is still legally on sale and other populations will act as a source of new inoculations. It was not possible during the present study to determine if changes in Lagarosiphon distribution in Lough Corrib represent expansion within the lake system or new inoculations from outside.

8.5 Control of invasive chub

In addition to the scientific surveys described in Chapter 7, a total of 8 electrofishing operations were conducted by Fisheries Board staff between 2008 and 2010. These operations followed a similar methodology to that described in Chapter 7, but aimed at controlling and eradicating chub in the River Inny. On four occasions (June 2006, April 2007, April 2008 and September 2010) the majority of the channel length between Loughs Derravaragh and Ree was examined (Fig. 7.1). On the other four sampling occasions, electrofishing was confined to specific stretches of the River Inny. In September 2007 and May 2009 two stretches, each measuring *c*. 2 km in length, at Shrule Bridge and Ballinalack were electrofished. The river at Shrule Bridge was also electrofished in August 2008 and May 2010 as radio-tagged chub were present in this location.

During these control operations a total of 28 chub were removed from the river. Using these data (Fig. 9.1), an estimate of the mean \pm 95% abundance of chub in the River Inny was made following the Carle & Strub (1978) depletion model. The estimated population size was 28 \pm 0 individuals, indicating that the population of chub in the River Inny was very small. The overlap between the population estimate with the number of chub removed to date provides support to the view indicating the likelihood that the species has been eliminated from the River Inny.



Figure 8.1: Depletion in the numbers of chub in the River Inny in following subsequent electrofishing control runs.

9 National Best Practice Guide on Preventing and Containment of Aquatic IAS

9.1 Introduction

All the River Basin Management Plans (RBMP) except the North Eastern identified IAS as one of the locally focused and future issues for which supplementary measures should be undertaken at River Basin District (RBD) level on a 2009-2015 timeframe. The North Eastern RBMP identified IAS as a main pressure and has prepared a programme of measures. The other RBMP's all identify the need to introduce new legislation to prevent the introduction and control possession of IAS which are detrimental to native species and state that further supporting measures will be identified by Invasive Species Ireland and local investigations at RBD level. The Western RBMP additionally identified the need to support the control programme for curly leaved pondweed (*Lagarosiphon major*) in Lough Corrib and support the Biosecurity Action Plan for the Mask and Carra catchments. As this study has focused on a range of priority species which include the eight identified by the EPA and included in the WFD risk assessments (EPA, 2005b) it can inform and support the development of measures for the RBD's.

Climate change was also included as a future issue and the RBMPs highlight that temperature changes may give IAS a competitive advantage increasing the impact on native biodiversity. There is a general scientific consensus that climate change is likely to favour IAS, leading to new invasions and spread of already established species. A review on climate change and invasive species prepared for the standing committee of the Bern Convention concluded that there is increasing evidence that climate change will affect the processes underlying biological invasions but that the current state of knowledge is not robust enough to make specific predictions (Capdevila-Arguelles & Zilletti, 2008). The interacting effects of IAS and climate change are complex and will vary by region and species (Bierwagen *et al.*, 2008). Predictions will need to be made on a species by species basis taking into account the biology and ecology of the species, the susceptibility of the habitat to invasion, the vulnerability of native biodiversity to climate change and the interactions between ecosystems and human activities (Coll *et al.*, 2009). IAS can also exacerbate the impacts of climate change by impacting biodiversity in advance of direct climatic impacts. In addition, there is a need to ensure that conservation policies adopted in response to climate change to increase ecosystem resilience such as increasing connectivity and assisted migration do not inadvertently facilitate species invasions.

There are a wide range of potential preventative measures which could be put in place, including legislation, risk assessment, developing Codes of Practice and standards for key sectors, all underpinned by education and awareness campaigns. It is important to focus on pathways and vectors as well as species and include measures to address intentional and unintentional introductions and have exclusion plans coupled with containment protocols. In order to successfully formulate and implement an effective toolkit to prevent and contain IAS introductions, these key elements need to be in place.

9.2 Developments in Ireland on prevention and containment

While the focus of this guide is on measures that can be put in place to address aquatic IAS at a national level, there is added value in identifying actions that can be taken at a RBD level to inform the development of any supplementary actions or programmes of measures. Therefore the proposals presented here have aimed to maximise synergies with ongoing projects such as Invasive Species Ireland which will be delivering outputs that can be used to help implement these proposals over the coming years.

9.2.1 Legislation provisions

One of the key elements of prevention is an effective legislative framework. In addition to the requirements under the WFD, Habitats and Birds Directives outlined in previous chapters, a range of other European legislative instruments and Directives have a role to play in preventing introductions of IAS, including the plant and animal health regimes, Wildlife Trade Regulation, Aquaculture Regulation and the Marine Strategy Framework Directive. However they have limited coverage of IAS in general and in particular the priority species which are the focus of this project. At a national level the main provisions are contained within the Wildlife Act (1976) and Wildlife (Amendment) Act 2000 and under Section 52(6) the Department of the Environment, Heritage and Local Government has the powers to adopt regulations prohibiting the introduction or possession of any species of wild bird, wild animal or wild flora or any part, product or derivative or such species which may be detrimental to native species. Section 52(7) also prohibits the release of 'exotic' species of wild animals and plants. The initiation of regulatory and legal gap analysis is a first step in development of an effective legislative framework. This has already been completed for Ireland and the main gaps and needs have been identified, including inconsistent application of the polluter pays principle and the need to harmonise provisions with Northern Ireland (Turner, 2008). Many of the recommendations from this review have been incorporated into the European Communities (Birds and Natural Habitats) Regulations which are currently being finalised and will prohibit the possession and sale of listed IAS except under licence. The majority of the priority species in Table 2.1 have been listed in the schedule and during the consultation the project team recommended the inclusion of all the species which are the focus of this project. The regulations are due for publication in early 2011 and will provide an important measure to prevent introductions and continued sale of high impact IAS.

9.2.2 Risk assessment and listing

Risk assessment is a fundamental part of prevention as it provides an objective and transparent evidence base for policy and decision making. An updated risk assessment for Ireland will be published in early 2011 and the outputs could be further developed by ISI in conjunction with the EPA, Data Centre, Northern Ireland Environment Agency, CeDAR and co-ordinators of the RBD's into alert lists and distribution maps tailored for each RBD.

9.2.3 Management plans

Plans have been developed for a range of species to prevent, contain and manage IAS and their impacts. These plans address the range of vectors for which management actions are necessary to prevent new introductions and further spread of established IAS. They also review and assess control options for these groups of established species and set out protocols to be followed in the event of a new introduction or further spread to contain and

where possible eradicate the species. Many aquatic IAS have common vectors of both initial introduction and secondary spread and control programmes for aquatic plant IAS uses common mechanical and chemical control methods. Exclusion strategies and contingency plans have been prepared for non-native crayfish species (*Astacus astacus, A. leptodactylus, Pacificastacus leniusculus, Oronectes limosus* and *Procambrus clarkii*) and water primrose (*Ludwigia* spp). Best practice management guidance has been developed for widely established species such as Japanese knotweed (*Fallopia japonica*), giant hogweed (*Heracleum mantegazzianum*) and Himalayan balsam (*Impatiens glandulifera*) which contain template management plans that can be used at a local level. Invasive Species Action Plans (ISAPs) have been developed for a range of aquatic plant species such as parrot's feather (*Myriophyllum aquaticum*), New Zealand pigmyweed (*Crassula helmsii*), water fern (*Azolla filiculoides*), fringed water lily (*Nymphoides peltata*) and floating pennywort (*Hydrocotyle ranunculoides*) which also contain template management plans that con developed for chub (*Leuciscus cephalus*) and Chinese mitten crab (*Eriocheir sinensis*). These ISAPs are available to download from the ISI website (www.invasivespeciesireland.com/downloads) and will be updated during 2011, with a range of new plans developed in the coming years based on the 2011 species risk assessment.

9.2.4 Codes of Practice and industry standards

Voluntary measures such as Codes of Practice (COP) and industry standards, coupled with their adoption into procurement policies are important ways of raising awareness with those industries whose activities act as pathways and vectors for IAS. COPs have been developed for the horticulture sector and marina operators and an aquaculture COP is currently in development. Ireland is currently developing a Green Public Procurement Action Plan and the inclusion of adherence to IAS COPs will be an important way of increasing the uptake amongst the target sectors. Additional COPs will also be developed in the coming years and although no decisions have yet been made on the target sectors or activities, it is likely that this will include the pet and aquaria trade.

9.2.5 Early warning and information system

Having an early warning system linked to interactive information portals that also shares information with neighbouring countries or at a regional level is another key mechanism to enable rapid response and containment of IAS. The development of the National Invasive Species Database (NISD) is a vital tool for this as outlined in Chapter 3.

9.2.6 Education and awareness materials

The general public and many stakeholder organisations, including parts of Government, have limited awareness and understanding of the threats posed by IAS. Improved awareness and understanding of the issues surrounding IAS is a vital element of prevention and in addition to raising general awareness, there is a need for initiatives targeted at key groups whose activities can contribute to the problem or be part of minimising the risk from IAS. The Invasive Species Ireland (ISI) website acts as a central information resource and ISI provide advice upon request on IAS. Specific stakeholder groups such as ports authorities, garden centres and the wider horticultural sector are currently being targeted with the development of dedicated materials and training. An identification guide that includes many of the priority species and guidance for boat owners are available to download. Inland Fisheries Ireland also has developed education and awareness materials on aquatic IAS and published biosecurity
guidance for anglers aimed at preventing further IAS introductions and spread (<u>www.fisheriesireland.ie/Invasive-Species/invasive-species.html</u>).

9.3 Pathway and vector analysis

9.3.1 Classification of pathways and vectors

The vectors and pathways by which IAS are transported are numerous and result from the diverse array of human activities which operate over a range of scales. In the aquatic environment primary introductions often result from the accidental transport of species, for example via hull fouling or ballast water. Secondary introductions result from the expansion of IAS from the initial place of establishment. Secondary spread will normally include a wider range of vectors that may act either separately or together. The challenge is to identify which pathways and vectors present the highest risks, develop efficient and timely responses and ensure that contingency plans for management, control and prevention of spread are feasible. While there are many uncertainties, in summary, the risk of invasions is determined by (Shine *et al.*, 2010):

- The invasiveness of species which depends on species or genotype traits, resource flows (trade, transport and travel) and what measures to detect and intercept introduced species are in place.
- The invasibility of habitats which depends on climatic and environmental conditions in the recipient areas and on the degree of habitat disturbance and fragmentation.
- The ease of introduction which is determined by factors including the openness of the economy, composition of trade flows, the effectiveness of regulatory regimes and the importance of agriculture, forestry or tourism (Perrings *et al.*, 2010).

In order to identify the key prevention measures for the priority species their pathways of introduction and vectors of spread need to be analysed. There is no standardised classification of pathways or vectors for use in risk assessment or this type of analysis as the field of pathway risk assessment is at an early stage in development. The CBD only distinguishes two pathways, intentional and unintentional and to date the main focus has been on assessing the risk associated with particular pathways such as the introduction of IAS as pets, aquarium and terrarium species, live bait and live food which is the task of an ad-hoc technical expert group convened under the CBD in October 2010. The Global Invasive Species Programme (GISP) identifies 35 main pathways (Wittenberg & Cock, 2001) and the US National Invasive Species Council developed guidance which defined pathways for unintentional introductions of aquatic IAS. It identified 76 pathways and sub-pathways categorised under three broad headings: transportation related (23), living industry related (43) and miscellaneous (10) (National Invasive Species Council, 2007). Hulme *et al.* (2008) identified three broad mechanisms by which IAS enter a new region: the importation of a commodity, the arrival of a transport vector, and natural spread from a neighbouring region. They proposed that these three mechanisms result in six principal pathways reflecting a gradient of human involvement: release, escape, contaminant, stowaway, corridor and unaided.

There is some confusion with the terminology in the literature with the terms pathways and vectors often used interchangeably. As the aim of this analysis was to identify both pathways of introduction and vectors of spread and some processes are both a pathway of initial introduction and a vector of secondary spread e.g. shipping, an integrated approach was taken and a classification developed that can be used for both (Table 10.1). This is more disaggregated than that proposed by Hulme *et al.* (2008) whose pathway classification was designed to assist the

development of overarching legislation that targets shared attributes of pathways and facilitates comparisons across taxa and biomes. The six pathways are broad classifications and each pathway comprises of multiple sectors so policy measures would have to separately target all of the sectors in each pathway. Therefore the classification proposed by Hulme *et al.* (2008) was of limited use in informing the development of targeted policy measures but there is correspondence between the two classification systems and those used by GISP as illustrated in Table 10.1.

	Pathway category	Sub-pathway/vector	Hulme et al.	GISP
		category	(2008)	pathways
Intentional introductions	Trade	Ornamental plant	Release Escape	Ornamental plants
		Aquaria	Escape	'Pets released into the wild' and 'aquarium trade'
		Live food	Escape	Mammals released on islands as food resource
		Pet	Escape	'Pets released into the wild' and 'aquarium trade'
	Industry	Aquaculture/mariculture	Escape	Aquaculture and mariculture
	sectors	Forestry	Escape	Foreign plants grown for forestry use
		Wildfowl/game stocking	Release	Birds and mammals released for hunting
		3	Escape	purposes
		Agriculture	Escape	Plants introduced for agricultural purposes
		Fur farming	Escape	Farmed mammals
		Fisheries (release and	Release	Fishery releases
		stock movement)	Escape	
	Miscellaneous	Biological control	Release	Biological control
		Intentional release	Release	Releases to enrich the native flora and fauna
		Scientific research	Escape	Research and introductions through research institutes
		Unknown	n/a	n/a
Unintentional	Transport	Vessels (hull fouling)	Stowaway	Hull fouling
introductions	corridors	Vessels (ballast water	Stowaway	'Ballast soil' and 'ballast water of ships' and
		and sediments)		'ballast sediment in ballast water tanks'
		Air travel	Stowaway	Hitchhikers in or on planes
		Canals	Corridor	Human-made structures which enhance spread of alien species
		Land (road, rail)	Corridor	'Hitchhikers in or on mail and cargo' and 'machinery, equipment, vehicles, military, etc'.
	Trade contaminants	Agricultural products	Contaminant	Contaminants of agricultural produce
		Horticulture (plants)	Contaminant	Seed and invertebrate contamination of nursery plants
		Horticulture (growing medium)	Contaminant	Soil inhabiting species
		Packaging	Contaminant	'Hitchhikers in or on package material' and 'organisms in or on timber'
		Other imports	Contaminant	Hitchhikers in or on mail and cargo
	Industry sectors	Aquaculture/mariculture	Contaminant	Parasites, pathogens and hitchhikers of aquaculture and mariculture
		Fisheries	Contaminant	n/a
		Forestry	Contaminant	n/a
		Agriculture	Contaminant	n/a
	Leisure	Boating/water sports	Stowaway	n/a
	activities	Angling	Stowaway	n/a
		Tourism	Stowaway	Tourists and their luggage/equipment
	Miscellaneous	Scientific research	Stowaway	Research and introductions through research institutes
		Waterways maintenance	Stowaway	Machinery, equipment, vehicles, military, etc.
		Escape from captivity	Escape	Escapes from captivity such as zoos and botanical gardens
		Natural dispersal	Unaided	n/a
		Unknown	n/a	n/a

Table 10.1 Pathway and vector classification

9.3.2 Analysis of pathways and vectors

The pathways of introduction and the main vectors of secondary spread for the priority species were identified from the literature for Ireland (Minchin, 2007 and references therein), Great Britain (Keller et al., 2009) and the project team's own unpublished records and field studies. The pathways and vectors for potential IAS have been identified for Britain as usually these species appear in Britain before occurring in Ireland (Minchin & Eno, 2002).

Established high impact	Pathway of introduction	Main vectors of secondary spread
species		
Lagarosiphon major	Ornamental plant/aquaria trade	Ornamental plant/aquaria trade, Natural dispersal, Boating/water sports, Angling
Elodea nuttallii	Ornamental plant/aquaria	Ornamental plant/aquaria trade, Natural dispersal,
	trade	Boating/water sports, Angling
Myriophyllum aquaticum	Ornamental plant/aquaria trade	Ornamental plant/aquaria trade, Natural dispersal, Boating/water sports, Angling
Crassula helmsii	Ornamental plant/aquaria trade	Ornamental plant/aquaria trade, Natural dispersal, Boating/water sports, Angling
Azolla filiculoides	Ornamental plant/aquaria trade	Ornamental plant/aquaria trade, Natural dispersal, Boating/water sports, Angling
Lemna minuta	Ornamental plant/aquaria trade	Ornamental plant/aquaria trade, Natural dispersal, Boating/water sports. Angling
Nymphoides peltata	Ornamental plant/aquaria	Ornamental plant/aquaria trade, Natural dispersal, Boating/water sports, Angling
Hydrocotyle ranunculoides	Ornamental plant/aquaria	Ornamental plant/aquaria trade, Natural dispersal, Boating/water sports, Angling
Dreissena polymorpha	Vessels (bull fouling)	Natural dispersal Boating/water sports Angling
Corbicula fluminea	Unknown ¹	Natural dispersal, Boating/water sports, Angling
Gammarus pulex	Fisheries (release and	Intentional release Natural dispersal Boating/water sports
	stocking)	Angling
Gammarus tigrinus	Vessels	Natural dispersal, Boating/water sports, Angling
Hemimysis anomala	Unknown ²	Natural dispersal, Boating/water sports, Angling
Eriocheir sinensis	Unknown ³	Natural dispersal
Leuciscus cephalus	Intentional release	Intentional release, natural dispersal
Leuciscus leuciscus	Intentional release	Escape from captivity, Intentional release, natural dispersal
Fallopia japonica	Ornamental plant trade	Horticulture, natural dispersal
Impatiens glandulifera	Ornamental plant trade	Horticulture, natural dispersal
Heracleum mantegazzianum	Ornamental plant trade	Horticulture, natural dispersal
Potential high impact species	Most likely pathway into GB	Main vectors of secondary spread
Ludwigia peploides and L. grandiflora	Ornamental plant trade	Ornamental plant trade, Natural dispersal, Boating/water sports, Angling
Astacus astacus	Aquaculture, aquaria trade	Escape from captivity, Aquaria trade, Intentional release, Natural dispersal
Astacus lentodactulus	Aquaculture, aquaria	Escape from captivity, Aquaria trade, Intentional release,
Astacus teptodactytus		Escape from cantivity Aquaria trade Intentional release
Pacificastacus leniusculus	trade	Natural dispersal
Orconectes limosus	Aguaria trade	Aguaria trade, Intentional release, Natural dispersal
Procambarus clarkii	Aguaria trade	Aquaria trade, Intentional release, Natural dispersal
	Fisheries (release and	
Gyrodactylus salaris	stocking)	Angling, Natural dispersal
Dreissena bugensis	Vessels (hull fouling)	Natural dispersal, Boating/water sports, Angling
Gymnocephalus cernuus	Intentional release	Intentional release, Natural dispersal
Sander lucioperca	Intentional release	Intentional release, Natural dispersal
Pseudasbora parva	Aquaria trade	Aquaria trade, Intentional release, Natural dispersal

Fable 10.2 Pathways of introduction and vectors	of secondary spread of the priority	species in Table 2.1
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¹ most likely pathway into GB was the aquaria trade, pathway for Ireland is unknown at present. ² most likely pathway into GB was boating, pathway for Ireland is unknown at present. ³ most likely pathway into Ireland was vessels.

In many cases it is not possible to definitively state what the pathway of introduction was due to a lack of documented evidence. This particularly applies to identifying the vectors of secondary spread which can be multiple so the main ones have been listed in Table 10.2. The key pathways of introduction of those species which are already established in Ireland are the ornamental and aquaria trades, vessels (either commercial or leisure) and intentional release for fisheries. These are also the key pathways by which the potential high risk IAS would be introduced with the addition of aquaculture. Therefore any policy measures to prevent aquatic IAS introductions must address the risks associated with these industry sectors, vessels and intentional releases to enhance fisheries. The key vectors of secondary spread for all the priority species are also the ornamental and aquaria trades; leisure activities such as boating, water sports and angling; intentional releases; escapes from captivity and natural dispersal. Therefore any policy measures to contain IAS introductions must address the risk associated with these vectors and identify the contingency and management actions required.

All of the established plant species were originally introduced by the ornamental and/or aquaria trade. In Britain the ornamental trades were responsible for the greatest percentages of freshwater IAS that have intentionally (73%) and unintentionally (34%) become established (Keller *et al.*, 2009). In many cases they were not known to be invasive at the time of initial introduction but subsequently became established outside the areas they were originally planted in such as gardens and ponds or artificial waterbodies. This can happen in many ways, through flooding which washes plant fragments into nearby watercourses or transfer by birds; intentional release by disposing of plants and garden waste into the wider countryside or watercourses; and accidental transfer via boating and angling equipment such as trailers and keep nets. However, many of these plant species are still sold and their use promoted despite the evidence of their impacts on the economy and native biodiversity.

It is not known for certain how the established invertebrate species were all originally introduced but evidence suggests they were probably accidental introductions with vessels (commercial and leisure) with the exception of *Gammarus pulex* which was intentionally introduced for fish food. The fish species were intentional introductions; dace was originally in confinement as live bait and subsequently released or escaped. The illegal enhancement of fisheries is the primary way that new fish species become established in Ireland. Secondary spread of these species tends to occur naturally or unintentionally although there is anecdotal evidence that zebra mussels were spread intentionally to waterbodies to improve water clarity.

9.4 **Prevention measures for key pathways and vectors**

The risks associated with the key pathways and vectors are not always fully understood. Reducing the risks will require the use of a combination of policy instruments, a regulatory and voluntary approach and will be more successful if developed in partnership with stakeholders. The majority of the priority species will be introduced and spread unintentionally therefore completely preventing their introductions and spread will be impossible, but risk can be managed and reduced.

There are a range of measures needed to prevent IAS introductions which are outlined below. These address both intentional and unintentional pathways and many of these measures will reduce the risk of all IAS introductions not just aquatic.

Legislation banning the import and sale of listed IAS: The implementation of legislation banning the import and sale of known IAS will be one of the most important measures to prevent new introductions via industry sectors. There are already provisions in the Wildlife (Amendment) Act 2000 under which it is an offence to deliberately release IAS into the natural environment such as intentional introductions of new fish species, however to date there have been no prosecutions and enforcement of current legislative provisions is weak. The provisions contained in the forthcoming (see 10.2.1) European Communities (Birds and Natural Habitats) Regulations will provide an important way to prevent intentional introductions of the priority IAS species, however, the key will lie in enforcement of these regulations and engagement with the relevant industry sectors in their implementation.

Ratification of the IMO Ballast Water Convention: Shipping and releases of untreated ballast water and hull fouling are important pathways and vectors for IAS therefore Ireland should ratify the IMO Ballast Water Convention to reduce the risk of IAS introductions from this pathway through the implementation of the ballast water discharge standards, designation of ballast water exchange areas, provision of reception facilities for sediments and introduction of monitoring and compliance requirements.

Pre-import risk screening for new species: There are many IAS which could be introduced by industry sectors and potentially become established in Ireland. The ISI risk assessments due for publication in early 2011 will identify high risk potential IAS and horizon scanning exercises have been carried out for Britain which are also relevant for Ireland. A system of pre-import risk screening would reduce the risk from industry pathways as once species are kept in Ireland; the risk of secondary spread is always present. Schedule 3 in the forthcoming European Communities (Birds and Natural Habitats) Regulations lists the plant, animal and vector material to which the regulations apply. It does contain species which are not yet established in Ireland, however this list is incomplete and updating and revising schedules does not always occur in a timely manner so an additional measure of risk screening will be required.

Licensing and permitting: The introduction of a system of licensing and permitting for possession of high impact IAS is an important way of reducing the risk from pathways and vectors such as escape from captivity and intentional release. The provisions in the forthcoming European Communities (Birds and Natural Habitats) Regulations enable the Minister to grant a license for possession of IAS. A license should only be granted where the introduction of the species is deemed to be unlikely to cause harm to ecosystems, habitats or species on the island of Ireland or indeed pose a risk to neighbouring countries. The burden of proof should be with the applicant and the anticipated benefits of introduction should strongly outweigh potential adverse effects and related costs.

Introducing licence conditions that are focused on preventing escape from captivity and make it compulsory to have contingency plans in place and carry out monitoring will reduce the risk from this pathway. A system of licensing will need resourced and could be combined with pre-import risk screening for new species with charges for the importer or applicant that cover the costs of the risk assessment and licensing system. The use of quarantine for vector materials should also be considered. This will require cooperation and coherence with plant and animal health systems. Consideration should be given to the use of a 'duty of care' similar to that used in waste management legislation. Such a duty of care would mean that liability for the IAS could not be avoided, rules governing 'fitness' to hold import licenses would apply, and only registered carriers could be permitted to transport IAS.

Codes of practice and industry standards: As it is currently legal to possess or sell the priority plant IAS, voluntary measures such as Codes of Practice and industry standards can be used to reduce the risk of further spread by educating and encouraging sectors not to trade in these species. The Horticulture COP recommends measures such as proper labelling of species as IAS and encourages traders to provide their customers with advice

on how to dispose of these species to prevent their establishment in the wider countryside. Limiting factors for the success of voluntary efforts to prevent introductions of IAS by the horticulture sector include ineffective information dissemination and lack of outreach (Burt *et al.*, 2007). COP's can also be used to reduce risk from pathways and vectors such as boating and angling. COP's have already been developed that can be adopted such as the draft European charter on Recreational Fishing and Biodiversity (Brainerd, 2010); the ISI COP for water users and the IFI biosecurity guidance (see 10.2.6).

Public procurement: The adoption of Codes of Practice by public bodies for use in procurement will enable Government to target their purchasing at species which are not IAS and act as an incentive for industry to adopt the standards.

Integration of IAS into border inspection control: There is limited capacity for the integration of IAS into current border control as many of the species arrive in Ireland from intra-EU trade. However customs and border control can be involved in surveillance and reporting for potential IAS if they are provided with training and the resources they need to do this. These would include focusing on those IAS that are contaminants of imports with identification guides for staff and clear reporting protocols.

Stakeholder partnerships with key industry sectors: Having an effective policy mix that reduces risk also requires building partnerships with key industry sectors. Strong legislative provisions will not be effective without enforcement and awareness of those provisions by stakeholders and industry support for voluntary measures. **Awareness raising and communications:** Awareness of the impacts of IAS, control measures and where to report sightings and get advice can help address vectors and pathways such as intentional release and accidental spread through leisure activities. Awareness programmes can encourage the uptake of targeted guidance and should also include raising awareness of legislation and penalties.

9.5 Control and containment measures for priority species

Prevention will not always be successful so at a national level measures to effectively control and contain IAS must be in place. These should include targeted monitoring for new introductions and range expansions of IAS into vulnerable waterbodies (see Chapter 4 for proposals) coupled with an early warning and information system (see Chapter 3 for proposals) and a rapid response mechanism. For the majority of aquatic IAS once they have become established, eradication, i.e. the elimination of the entire population including any resting stages is not feasible. Before a decision is made on whether eradication is feasible an analysis of the costs (including indirect costs) and likelihood of success must be made and eradication should only be pursued when both funding and commitment of all stakeholders is secured.

When eradication is not feasible, the aim should be to control and contain the species to prevent further spread. Containment plans should have clearly defined goals e.g. to protect particular waterbodies from invasion and identify the area beyond which the species should not spread. An important part of any containment plan will be to put in place the ability to detect new invasions or spread which can be challenging as these new infestations can be at very low densities. Containment measures are most likely to be successful for those species which spread slowly over short distances and where effective natural dispersal mechanisms are absent. Ireland's main river basins are connected by canals so containment of species in these systems is unlikely to be successful. However it may be possible to contain some species in isolated waterbodies in the absence of effective natural dispersal mechanisms. Whether new introductions or range expansions of the priority species can be contained will depend on the location. For many of these species, particularly the invertebrates, effective control measures in the natural environment are not available. In addition, increasing restrictions on the use of herbicides and concerns over the impact on non-target species may make the relative impact of control measures unacceptable to managers.

The species accounts (see Section 4.1) detail the key prevention, control and management actions for each species and are available from the ISI and NISD websites and so are not repeated here. An integrated approach should be taken to IAS control incorporating physical, chemical and biological. Many of the established aquatic plant species can be controlled through a combination of physical and chemical control but as some species spread by fragmentation, effective control is challenging. In general, control programmes manually remove growth, lay light occluding geotextile fabric and/or spray with herbicide. As shown in Chapter 8, jute matting can be used to control Lagarosiphon effectively and its biological control is currently being investigated. Biological control of Azolla filiculoides has been carried out in South Africa since 1995 using a frond feeding weevil Stenopelmus rufinasus. Although not deliberately released, the weevil is known to occur in the UK since 1921, where it offers periodic control of the water fern and was first reported in Ireland in 2007 and so could potentially control this species here. Fish species can generally be controlled by electrofishing and netting as demonstrated by the chub control programme (see Chapter 6 for details). Britton et al. (2010c) evaluate a range of control and containment method for invasive fish species, all of which have limitations depending on the waterbody and economic and political constraints. For example, electrofishing does not always remove all of the population and chemical control methods such as removal of native species followed by application of rotenone are generally only acceptable in isolated waterbodies where managers consider the benefits of removal of the IAS to outweigh the non-target impacts.

9.6 **Protocols for use at RBD level**

The key prevention measures need to be put in place at a national level and while the exact actions taken in response to a new invasion or range expansion will depend on the species, location and nature of the incursion; it is possible to set out protocols that can be used at RBD level with the aim of reducing IAS pressures. Following these protocols will ensure that the information is provided in a timely way and that any available resources are supplied to local managers.

- Step 1: If a new sighting of an IAS occurs, the information should be recorded in accordance with the guidance note for contributors of IAS data (see Section 3.3).
- Step 2: The record should be submitted online through the NISD or ISI websites. The record will then be verified and if required a species alert will be issued and the relevant organisations notified.
- Step 3: Consult with ISI as the IAS management tool-kit available on the ISI website provides resources for use at a local level to inform and support management. ISI will also facilitate contact with species experts.
- Step 4: If a decision is taken to control the species then a management plan should be developed. The ISAP's provide a template plan that can be adapted.
- Step 5: The management plan should be implemented and ISI will assist in disseminating the outcomes to key stakeholders to enable learning.

9.7 Conclusions

The proposals presented here are best practice for preventing and containing aquatic IAS at a national level and set out simple protocols for use at RBD level. For many aquatic IAS prevention and containment will be challenging if not impossible due to the nature of the pathways and vectors of spread and lack of effective control methods. There is also an opportunity to further develop the prevention actions in the coming years as the ISI project is carrying out pathway risk assessment and developing management strategies for the highest risk pathways. This work can support the actions being taken at RBD level and it is suggested and recommended that the EPA facilitate dissemination of the project outputs to those involved with the WFD programme supported by the project partners.

10 Conclusions and Recommendations

10.1 Introduction

The EPA commissioned this project with the aims of improving knowledge on the nature and extent of IAS and their impact on natural ecosystems; developing up to date national distribution maps showing the location of aquatic IAS in Ireland; and developing and trial control measures in the context of river basin management. This project has contributed to meeting these aims through a multidisciplinary project combining research, policy analysis and GIS database development.

The research on the ecological impacts of *Lagarosiphon* in Lough Corrib and chub in the River Inny has increased our understanding of the impacts of these species and their interactions with native communities and other nonnative species as well as demonstrating new means by which impacts can be measured. The development of the National Invasive Species Database, collation of records and mapping of the up-to-date distributions of aquatic IAS provides a valuable resource for researchers and managers. The demonstration of effective control measures for *Lagarosiphon* and chub will enable rapid reaction to further introductions and range expansions to new waterbodies. The development of proposals for surveillance, monitoring and reporting of IAS and policy measures for prevention and containment will inform the WFD programme of measures and river basin management.

10.2 Nature and distribution of IAS in Ireland

Determining the distribution and tracking range expansions of IAS is fundamental to effective management. Given the large number of non-native species present in Irish waterbodies, it was important to focus efforts at those species which have the greatest impact. The development of a prioritised list of species based on, amongst other factors, their potential to affect the ecological status of Irish waterbodies has enabled efforts to be targeted. The development of a GIS database of aquatic IAS as part of the National Invasive Species Database enables up-todate distribution maps for 21 priority species to be produced. Although it was not possible to display aquatic IAS records by lake and river segment with WFD coding, the distribution maps are displayed against a backdrop of GIS layers that provide important contextual information which assists risk assessment and the identification of waterbodies vulnerable to invasion. The addition of the detailed rivers and lakes layer with WFD waterbody codes and the EPA water quality indicator layer will enhance the functionality of the database as a tool for management of aquatic IAS.

The provision of a mechanism for online submission of verified records and the production of guidance for data contributors also ensures that this output can be used in any monitoring and reporting programme and has moved the NISD from a static resource into a dynamic database that has the potential to become a vital tool in the identification, monitoring and control of aquatic IAS in Ireland.

A number of areas for future work have been identified that can build on these outputs including the development of an information exchange network; surveys for the priority species to improve the coverage and quality assessment of the distribution maps; and further development of the NISD as the information infrastructure for an early detection and rapid response mechanism. Ensuring that information on the distribution of IAS is up-to-date and that new invasions are detected early requires surveillance, monitoring, recording and reporting programmes. The assessment of current developments and how IAS can be integrated into the WFD monitoring programmes has enabled the development of proposals for these programmes in Ireland. All of the priority species will either be included in the biological parameters of the WFD, or have the potential to impact on the assessment of biological quality and so will be included in WFD surveillance and operational monitoring. While the WFD monitoring programme does not cover all aquatic IAS, it does provide a useful framework for a monitoring programme and in addition, it offers the opportunity to greatly enhance surveillance capacity across the island of Ireland.

The EPA has highlighted the importance of data information and management so that the data generated by the monitoring programme is collected, managed, analysed and reported in a systematic, efficient and timely manner. The integration of the NISD into the Environmental Data Exchange Network (EDEN) will enable a range of outputs to be delivered that can inform river basin management. These include IAS distribution maps, identification of range expansions, species alerts and identification of waterbodies vulnerable to invasion at a RBD level.

10.3 Analysis of IAS pressures

Developing effective programmes of measures for RBDs will require greater understanding and analysis of IAS pressures. The species accounts produced for the priority species highlight their potential impacts on WFD objectives. However, IAS are just one pressure and many waterbodies are subject to the combined effects of habitat degradation, human regulation of water levels, physical modification, water extraction and nutrient enrichment. It can be challenging to actually quantify the direction and intensity of ecological change as a result of a species invasion and for the wide range of IAS that are currently established in Irish waterbodies. As such, the project aimed to present ways in which to examine the impact of invasion on ecosystems rather than categorically define the impacts of IAS on receiving ecosystems.

The introduction of two relatively recent IAS, *Lagarosiphon* and chub, and the lack of knowledge about their potential ecological and economic impacts raised concerns, particularly as they have the potential to become widely established in Ireland. The study of the ecological effects of the invasion of Lough Corrib by *Lagarosiphon* investigated several different levels of biological organisation. The effects of invasion by *Lagarosiphon* include marked changes in the form of the littoral habitats of the Lough Corrib ecosystem, with a shift from *Chara* meadows that include both structured benthic habitats, and an overlaying unstructured pelagic zone to a state where, depending on the time of year, the whole water column can consist of monospecific stands of the invasive macrophyte (Caffrey *et al.*, 2009), forming structured habitat and limiting light penetration to the lake bottom. The research showed a series of ecological impacts of invasion that include changes in macroinvertebrate community structure and production, differences in key life history traits of the two dominant fishes of Lough Corrib and marked differences in some measures of consumer trophic ecology.

Lagarosiphon invasion was associated with increased macroinvertebrate biomass and changes in community structure which may reflect increased habitat availability; however other IAS including zebra mussels were also associated with Lagarosiphon beds. In terms of the wider Lough Corrib ecosystem, it is clear that the impacts of the

Lagarosiphon invasion varied considerably. There was no obvious effect of Lagarosiphon on the abundance, biomass or structure of the fish community. However we have demonstrated marked, habitat-associated differences for a number of ecological measures between native *Chara*-dominated habitats compared to *Lagarosiphon*-invaded sites (e.g. invertebrate biomass, abundance, and community structure; perch shape and growth; size at maturity in both roach and perch; adult mortality rates; fish trophic ecology). However, many effects of the invasion were relatively subtle; suggesting that some effects of invasive species are indeed less marked, and furthermore vary over time. There were ecosystem-level differences between native and invaded habitats, ranging from different macroinvertebrate communities, to a reduction in the length of the food chain in *Lagarosiphon*-dominated habitats.

As the study of the ecological implications of the introduction of chub in the River Inny was also combined with a control programme aimed at eradicating the population, the focus of the research was on understanding the ecology of chub to improve our understanding of the implications of further introductions of this species. No obvious ecological effects of the chub introduction were detected although the stable isotope analysis results raised concerns as they indicated that the long term diet of chub was similar to that of the native and conservationally important brown trout, eel and salmon.

The analysis of IAS pressures on waterbodies of these two species was hindered by logistical and financial restraints. However the results clearly show that *Lagarosiphon* is impacting on the biological quality elements in Lough Corrib, in particular, macrophyte and benthic invertebrate abundance and community composition.

10.4 Prevention and control measures

Preventing new introductions is particularly important in the aquatic environment given the difficulty in eradicating or managing IAS once they have become established. The analysis of the vectors and pathways for the priority species identified the ornamental and aquaria trades, vessels (either commercial or leisure) and intentional release for fisheries as the key pathways of introduction. The key vectors of secondary spread were also the ornamental and aquaria trades; leisure activities such as boating, water sports and angling; intentional releases; escapes from captivity and natural dispersal. A range of policy measures were proposed that addressed the risk associated with these pathways and vectors, and identified the contingency and management actions required to reduce risk. Simple protocols for use at RBD level with the aim of reducing IAS pressures and ensuring that available resources are supplied to local managers.

Control measures for *Lagarosiphon* and chub were evaluated in the field and showed that effective control measures could be identified for use with new introductions and range expansions even if *Lagarosiphon* is too widely established in Lough Corrib for eradication to be successful. Tracking the movement of chub showed that their movement was limited, allowing control, assuming the response is suitably rapid.. It appears that such rapid control activities *i.e.* electrofishing, combined with a low propagule pressure (i.e. number of fish introduced) and a limited population size enabled the removal and putative eradication of this new IAS to Ireland.

10.5 Implications for the WFD programme of measures

The outputs from this project will contribute to how IAS are managed in the context of the WFD programme of measures and the RBMPs. While the agreement of a common European position on how IAS should be dealt with in ecological status classification and how WFD programmes of measures might be used to address IAS is important, this project clearly shows that aquatic IAS are a growing problem in Ireland. We now know that most of the major Irish lakes have established IAS populations: therefore actions will need to be progressed without European consensus on these issues.

All the RBMPs have identified IAS as either a main pressure or an issue for which programmes of measures need to be developed. Here we have proposed how the outputs from this project and follow on actions can assist in this process, namely:

- Further development of the NISD with the addition of detailed lakes and rivers layers with WFD coding to provide up-to-date distribution maps of IAS at an RBD level which will enable the identification of range expansions and waterbodies vulnerable to invasion.
- Integration of the NISD into EDEN so that IAS information is supplied in a timely way and the adoption of the guidance note as the IAS data standard.
- Integration of IAS surveillance and monitoring into WFD monitoring programmes.
- Development of alert lists at a national and RBD level and provision of ID guides for use in the field.
- Provision of Invasive Species Action Plans and template management plans for use at local level.
- Analysis of IAS pressures for a range of species and provision of this information online in this report and in the species accounts.
- Identification of control measures that can be used for new introductions or range expansions of *Lagarosiphon* and chub (and similar taxa).
- Development of protocols for use at RBD level to enable rapid reaction and containment of IAS.

10.6 Recommendations

This project has produced research and analysis that provides an evidence base for policy development and decision making on aquatic IAS management in Ireland. We have attempted to present the research findings in a way which will facilitate their uptake and use. As such, the outputs of this project can provide an evidence base for decision making for a range of stakeholders, examples of which are set out below:

- Policy makers to underpin the development of the WFD programme of measures and target resources to address aquatic IAS impacts.
- Industry sectors whose activities are the key pathways of introduction of IAS to demonstrate the need to change practises to reduce the risk of IAS introductions and spread.
- Local Authorities to provide information on what IAS are present in their areas and to encourage their participation in surveillance.
- Development and support of a research community to further our understanding of the ecological impacts of aquatic IAS in Ireland.

A range of recommendations have been developed for further work, divided into three main areas, policy, research, education and capacity building.

10.6.1 Policy

- The EPA should routinely liaise with Invasive Species Ireland and other relevant organisations to update the list of species that need to be considered in the WFD risk assessments and programmes of measures.
- A programme of surveillance and monitoring should be developed and implemented by integrating aquatic IAS into the WFD monitoring programme supported by guidance on surveillance, monitoring, recording and reporting for use by staff.
- The proposals in the best practice guide on preventing and containing aquatic IAS should be implemented to reduce the risk of new introductions of IAS.

10.6.2 Research

- Further research on the impact of multiple IAS in Irish freshwater systems is required to further understanding of IAS pressures, *e.g.* interactions between invasive species and the potential for invasional meltdown.
- The ecological effects of control measures should be investigated further to inform their refinement and proposals for ecological restoration, e.g. through the monitoring of ecological responses to control, such as the removal of *Lagarosiphon* from Lough Corrib.
- The efficacy of jute matting as a control method should be evaluated for other aquatic plant IAS.

10.6.3 Education and capacity building

- The EPA should convene a workshop with the RBD co-ordinators, technical and stakeholder councils, project partners and Invasive Species Ireland to enable targeted dissemination of the project outputs and further development of resources for use at RBD level.
- The NISD should be further developed as the information infrastructure which underpins an early warning and rapid response mechanism for Ireland.
- The NISD should be promoted to and used by those involved in the WFD programme.
- Targeted education and awareness initiatives should be developed that increase awareness of IAS amongst those involved in activities which act as pathways and vectors of introductions and spread.
- Education and training materials should be produced targeted at those involved in recording to improve knowledge of IAS distributions.

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Acronyms and Annotations

BSBI	Botanical Society of the British Isles
CBD	Convention on Biological Diversity
CeDAR	Centre for Environmental Data and Recording
DAISIE	Delivering Alien Invasive Species Inventories for Europe
ECOSTAT	European Commission intercallibaration process for the WFD
EDEN	Environmental Data Exchange Network
GIS	Geographical Information Systems
IAS	Invasive Alien Species
IRBD	International River Basin District
IFI	Inland Fisheries Ireland
ISI	Invasive Species Ireland
NBDC	National Biodiversity Data Centre
NISD	National Invasive Species Database
NOBANIS	European Network on Invasive Alien Species
PERMANOVA	Permutation-based Analysis of Variance
PRIMER	Plymouth Routines in Multivariate Ecological Research
QUB	Queen's University Belfast
RBD	River Basin District
RBMP	River Basin Management Plan
SIA	Stable Isotope Analysis
SIMPER	Similarity Percentage Analysis
WFD	Water Framework Directive