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Cefas contract report C5524 - Final

Review of invasion pathways and provisional pathway management plan for non-native ponto-caspian species of potential invasion risk to Great Britain

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For Defra, Protected Species and Non-native Species Policy Group

Executive Summary

- A review was conducted to identify species from the Ponto-Caspian region that pose the greatest threat of introduction and/or spread in Great Britain.
- From the review, 14 species were identified as posing the greatest threat.
- The potential (and in some cases actual) impact that each species would have in GB were also reviewed.
- The list consisted of two species of mollusc, three fish, eight crustacean and one cnidarian of which five are recognised as currently being present in GB waters.
- A literature review conducted on each of the species identified pathways and mechanisms of introduction and translocation most commonly associated with the species.
- Shipping was recognised as the most common pathway of introduction into GB, with ballast water being the most common mechanism and then (in descending order) inland boating (a secondary pathway of dispersal), hull fouling (a mechanism of the shipping pathway), unintentional stocking (secondary pathway), anglers gear (secondary pathway), air/land transport (primary pathways) and intentional release (secondary pathway).
- The most common forms of dispersal (secondary pathway) were identified as natural dispersal and recreational activities (*e.g.* anglers, recreational boats and canoes). Other secondary pathways identified were the ornamental trade, unintentional stocking and incidental transport.
- The main drivers behind the predominant mechanisms/pathways were global trade/economic growth and recreation.
- Species identified as being associated with more pathways and mechanisms are thus more likely to gain entry into GB and/or disperse within GB once introduced. It is therefore possible to rank the spread in order of ‘threat’ posed.
- An outline of a provisional Pathway Management Action Plan (PAP) is provided outlining:
    - Stakeholders that should be involved in the further development of the PAP
    - Target areas of development to prevent further introductions
    - Actions to aid in preventing further spread
    - Recommendations on the development of biosecurity, control and/or eradication plans
    - The development of contingency plans for rapid response

Contents

1.0. Introduction and rationale

Prevention is increasingly recognised as the most effective means of avoiding or mitigating the impacts associated with unwanted non-native species. Indeed, the guiding principles in the non-native species management espoused by the Convention on Biological Diversity (CBD) are hierarchical in structure (Wittenberg & Cock 2001) and emphasize preventive measures primarily over eradication, containment, control and mitigation. A key component of prevention is the forecasting of (or horizon scanning for) the pathways by which potentially invasive species are likely to be introduced, as this is a means of reducing the potential success of subsequent invaders (Holeck *et al.* 2004). These pathways may involve either accidental or intentional movement of species as a consequence of human activities (Ruiz & Carlton 2003; Copp *et al.* 2005, 2007), and the risks of new introductions increase with the continued globalisation in trade, transport and tourism.

Ballast water and hull fouling (the attachment of organisms to boat bottoms) have been identified as mechanisms of translocation along inland waterways (Ahnelt *et al.* 1998; Wiesner 2005), and in the case of hull fouling, overland as well (Johnson *et al.* 2001; Ricciardi 2001; Holeck *et al*. 2004; David & Gollasch 2008; Bailey *et al.* 2011). All vessels require ballast in one form or another to operate. Owing to the continually changing loads of freight vessels, water is used as ballast as it can easily be taken on or discharged as cargo is loaded or unloaded. Organisms in the water column can be take up into the vessels ballast tanks and then discharged at the vessels point of destination, where the water is discharged as cargo is taken on. This can result in the translocation of aquatic organisms over considerable distances. Cargo vessels are involved in the global transfer of an estimated 8–10 billion tons of ballast water per year and the daily transport by ships of 3000–4000 species (Carlton & Geller 1993; Gollasch 1996).

The creation of transport routes to facilitate trade, *e.g.* canal construction during the 18th century, has connected previously-separated biogeographic regions (*e.g.* contiguous drainage basins). This inter-connection of drainage basins has facilitated the range expansions of many species in Europe (*e.g.* Jazdzewski 1980; Copp *et al.* 2005; Grabowska *et al.* 2010). These invasions are not restricted to species that have been introduced unintentionally via canals and/or river boat traffic. Intentional introductions by humans have a long history (Copp *et al.* 2005) and this is a continuing pattern (Bailey *et al.* 2008); with intentionally released or escaped specimens making use of the connections between river basins to expand their introduced ranges.

Biofouling of ship hulls is probably the most recognised form of species dispersal via attachment to hard substrata. However, biofouling of other, static human-made structures can act as a 'stepping stone' means for a non-native species to overcome biogeographical barriers. For example, in the marine environment the foundations of offshore wind farms and oil rigs in a given area are likely to be colonised by locally present organisms (Olenin *et al.* 2011), which can then invade new areas when part or all of the off-shore structure or shipping-lane marker buoy is taken for use or repair in a currently un-infested area (Bouma *et al*. 2011). It is likely that hard structures found along invasion routes in Europe, such as ports, harbours and marinas have acted as stepping stones Ponto-Caspian species. The recent discoveries of *Dikerogammarus villosus* in the UK, including a very recent report of the species in the Norfolk Broads, are believed to have occurred as overland translocations via contaminated boat hulls and/or angling gear (Non-native Species Secretariat 2012; Norfolk Broads Authority 2012).

In developing a non-native risk mitigation strategy, the key steps are: 1) the identification of non-native species invasion pathways into Great Britain (GB); 2) the specification of source regions; and 3) the evaluation of species from the specified region(s) that have invaded neighbouring European countries and therefore pose a potential risk to GB. The two principal source regions for invasive species in Europe during the previous decades have been Asia (*e.g.* goldfish *Carassius auratus*, gibel carp *Carassius gibelio*, topmouth gudgeon *Pseudorasbora parva,* Japanese or Manilla clam *Venerupis philippinarum,* Chinese mitten crab *Eriocheir sinensis*) and more recently the Ponto-Caspian region, with the so-called 'killer' shrimp *Dikerogammarus villosus* being the species that has attracted the most media attention. In contrast to most of the pathways followed by Asian species to arrive in Europe, which have been relatively simple, the pathways involved in Ponto-Caspian invasions of Continental Europe have been largely unintentional and complex in character, being linked to ballast water exchanges and most notably biological corridors (*i.e.* canal systems) built by humans to facilitate the transport of goods between river basins.

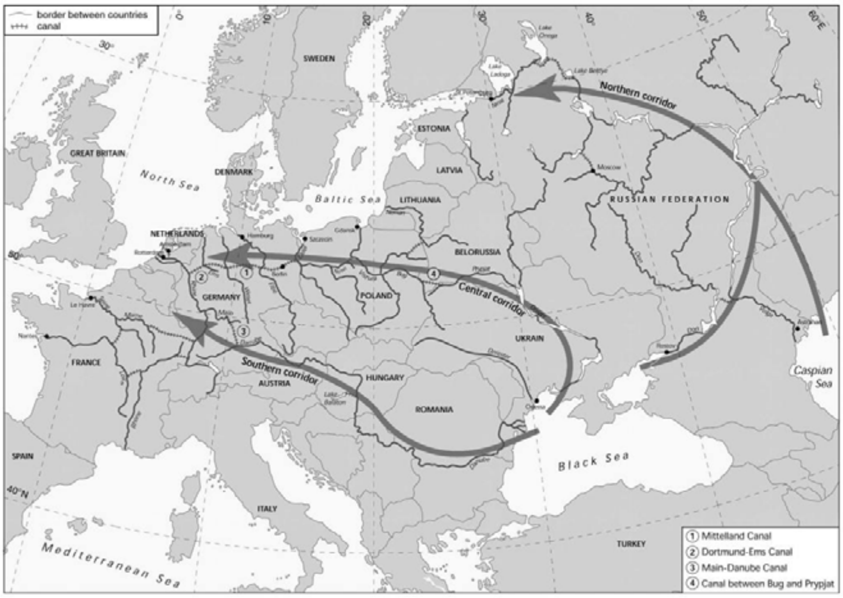

Figure 1. Map illustrating the principal invasion routes by which Ponto-Caspian species have invaded Europe (reproduced with permission from Bij de Vaate *et al.* 2002).

Three principle inland corridors by which Ponto-Caspian species expanded outwards into Central and then Western Europe have been identified (Figure 1). These corridors are a complex, associated network of important trade harbours and canal connections, which have permitted range extensions of aquatic organisms (Bij de Vaate *et al.* 2002). The first corridor was via the north, connecting the River Dnieper to the Kuronian Lagoon (Baltic Sea) via the Oginskiy Canal and the River Neman. This initial northern corridor, which opened in the 18th century, was the pathway probably used by the zebra mussel *Drissena polymorpha* for its range expansion in the 1930s, but this corridor no longer exists (Bij de Vaate *et al.* 2002), and the current northern corridor, which is a more recent route, involves connections between the following hydrosystems: River Volga, Lake Beloye, Lake Onega, Lake Ladoga, River Neva and the Baltic Sea. The main migration route prior to 1992 was the central corridor (Figure 1), which connects (in consecutive order) the river basins Dnieper, Vistula, Oder, Elbe and Rhine (Bij de Vaate *et al.* 2002). Since 1992, the southern corridor became the principle migration route for range expansions to the west due to the connection of the Rhine and Danube basins via the Main-Danube Canal, facilitated by water level management practices that involved the use of water from the Danube for lock opening/closing operations (Bij de Vaate *et al.* 2002).

The value of assessing introduction pathways and mechanisms prior to invasions is well demonstrated by the work of Balon *et al.* (1986), which predicted invasions of the River Rhine basin by Danube fishes prior to the Main-Danube Canal opening, including at least one of the Ponto-Caspian gobies, the freshwater tubenose goby *Proterorhinus semilunaris* (formerly *P. marmoratus*), which is described in a subsequent section (3.1.11). Other Ponto-Caspian gobies, such as the round goby *Neogobius melanostomus*, bighead goby *Neogobius kessleri* and monkey goby *Neogobius fluviatilis* have also expanded up from the lower Danube (the westernmost extent of their native ranges) and invaded the River Rhine via the Main-Danube Canal (Borcherding *et al.* 2011). They were not included amongst the invaders predicted by Balon *et al.* (1986), mainly because they had only just begun to expand up the lower Danube in the 1970s (for a review, see Jurajda *et al.* 2005), *e.g. N. kessleri* was first observed in Austria and Slovakia in the early-to-mid 1990s (Zweimüller *et al.* 1996; Černý *et al.* 2003). More importantly, they are the 'type of species' predicted by Balon *et al.* (1986) that would invade the Rhine system.

In a recent 'horizon-scanning' exercise for England specifically, aquatic species of Ponto-Caspian origin, including the gobies, have already been identified as potential new invaders of England (Parrott *et al.* 2009). However, a wider, Great Britain (GB) perspective is needed – one that focuses on both primary and secondary invasion pathways. Indeed, the GB Non-native Species Strategy (Defra 2008) provides a number of recommendations for the management of non-native species issues from a GB perspective, and Key Action 6.5 calls for the development of Pathway Action Plans (PAPs) to minimise the risks associated with the pathways but the Strategy does not specify what a PAP should involve. Although no PAP has yet been developed for GB, there have been some Invasive Species Action Plans (ISAPs) developed, which are short (2–3 pages) action-focussed documents that are informed by a risk assessment and linked to a formal risk management process. Casting a wider net than ISAPs, PAPs are considered a sensible means of assessing a broader range of species than considered in ISAPs.

As a first step towards the development of PAPs, a working group of stakeholders at the 7th Non-native Species Stakeholders Forum (Defra 2010) was assigned the task of providing guidelines as to what a PAP should contain.

The aim of the current project was to address these needs, with the specific objectives to:

1. Identify which aquatic species from the Ponto-Caspian region are most likely to be potential invaders of inland waters;
2. Rank the species by their potential, relative risk;
3. Summarize potential impacts of the highest ranked species, based on a literature review;
4. Undertake pathway assessments to evaluate the likelihood of entry to the UK, ranking the pathways in terms of their relative risk;
5. Assess mechanisms and the likelihood of dispersal within GB, ranking these in terms of their relative risk;
6. Identify, where feasible, the drivers for the current spread of Ponto-Caspian species across Western Europe;
7. Identify the sectors and interest groups that may be potentially involved in transmission, ranking these in terms of their relative risk;
8. Provide recommendations about possible appropriate precautionary measures and priority areas for risk reduction as a basis for informed policy development; and
9. Produce a provisional Pathway Management Action Plan (PAP) to assist in identifying which sectors (stakeholders) should be represented in agreeing and coordinating future actions as regards each invasion pathway.

2.0. Methodology

To generate a list of potentially invasive species from the Ponto-Caspian region, a literature survey and ranking of species was undertaken in a hierarchical manner using the search functions available at www.google.com. For scientific-based investigations, the ‘Scholar’ search function is particularly useful in that it restricts the search outcomes (*i.e.* ‘hits’) to scientific papers that contain the selected search terms. First, a general search was undertaken using the terms ‘Ponto-Caspian’ +

‘invasive’ to generate a list of potentially invasive species. Secondly, to rank organisms by their potential (relative) risk, a Google Scholar search was undertaken, species-by-species, using ‘species name’ + ‘invasive’ as search terms. The various species were then ranked in descending order according to the number of ‘hits’ reported for their search and the list evaluated for anomalies. In the case that the same species appeared twice, due to changes in their genus or species names, the number of hits from both searches was added to those for the species’ current valid name.

A frequency distribution of Google hits was then evaluated to identify a cut-off point for the detailed species literature reviews, based on the premise that species that have attracted the most scholarly attention are mostly likely to be species of potential, adverse environmental risk, and that these groups of species should be distinguishable from other by a notable drop (or ‘break’) in the number of ‘hits’. A ‘short-list’ of species for detailed literature review, revealed by the frequency distribution, was then evaluated to determine whether or not any species had been subject to a pre-screening or full non-native species risk assessment. Species that had been assessed as ‘low’ or ‘medium’ risk were removed from the ‘short list’ and the remaining species were then subjected to a literature review that focused on adverse impacts and invasion pathways.

The information compiled in these ‘species briefs’ was combined with that from existing literature sources in order to summarize their potential impacts in GB and to inform pathway assessments, which considered both principal and secondary pathways (*e.g.* shipping, air transport, land transport) as well as the mechanisms (*e.g.* ballast water, hull fouling, consignment contamination) associated with each pathway, and the likelihood of entry to and dispersal within GB in order that they be ranked in terms of their relative risk. It should be noted that the ‘species briefs’ were based wholly on the literature available to ensure that the study was evidence based rather than speculative or biased by opinion. To calculate relative risk, the questions and response options, as well as the scoring scheme, from the ‘pathway’ assessment module of ‘NAPRA’ (2012), the GB non-native risk assessment scheme (Mumford *et al.* 2010; Booy *et al*. 2011), were re-constructed in a purpose-built Excel® assessment tool (Figure 2); the module offers drop-down menus for selecting the appropriate response score (very unlikely = 1, unlikely = 2, moderately likely = 3, likely = 4, very likely = 5), and the assessor’s level of confidence (low = 1, medium = 2, high = 3, very high = 4) based on the available information (see Appendix 1 for details). The Pathway Module tool automatically calculates the mean score for the responses and for the confidence levels.

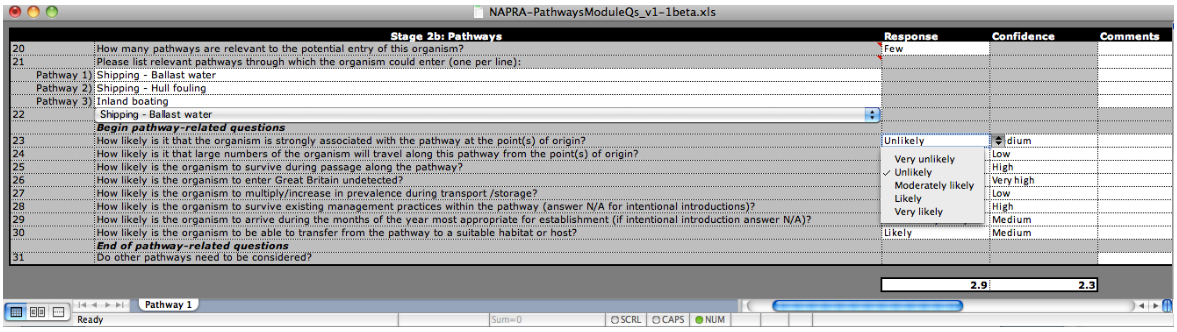

Figure 2. Screen grab of the purpose-built Excel® assessment tool, which contains questions 20–31 from the ‘Pathway’ assessment module of ‘NAPRA’ (2012) the GB non-native risk assessment scheme (Mumford *et al.* 2010; Booy *et al.* 2011). Example responses and confidence levels (see Appendix 1) are given for illustration purposes only.

Assessment of each species began with the listing of known pathways (shipping, air transport, land transport) with respect to reported introduction mechanisms (*e.g.* ‘ballast water’ and ‘hull fouling’ for the pathway ‘shipping’) as identified from the bibliographic reviews. Each ‘pathway × mechanism’ combination, as well as the secondary pathways (*i.e.* dispersal within GB once introduced) was assessed separately, with the assessment carried out by the same person who undertook the literature review for the species being assessed. As such, a species known to be associated with more ‘pathway × mechanism’ combinations (and/or with secondary pathways) attracts a higher cumulative score, and thus is more likely to gain entry into GB and/or to disperse within GB once introduced.

As a further assessment of the potential risks of impacts to GB of Ponto-Caspian invasions, the drivers behind the current spread of these species across Western Europe were identified along with the sectors and interest groups (stakeholders) that may be potentially involved in the transmission of Ponto-Caspian species.

The concept of a Pathway Management Action Plan (PAP) *per se* is relatively new, though initiatives such as the guidelines and convention on ballast water management (IMO 1997, 2004) and the subsequent guidance on biofouling (IMO 2011) represent *de facto* the basis of action plans for specific mechanisms associated with the shipping pathway. The only available guidance on what a PAP should comprise comes from a working group of the Defra Non-native Species Forum (Defra 2010), which recommended that PAPs be short, functional and include an overall aim, specified priorities, estimated risk levels, maps of GB and the world, risk communication/education, relevance to legislation and a means of measuring of progress and/or successes. It was further recommended that PAPs consider the following aspects:

- Introduction pathways into GB.
- Dispersal pathways within GB, including railways, roads and waterways.
- Use of risk analysis to underpin the PAPs (*e.g.* GB and European risk schemes).

- Involve stakeholders at an early stage in the PAP development process.
- Using a hierarchical approach, with indicator species as examples of common pathways.
- Involve a horizon scanning perspective.
- Consider receptor analysis to identify areas or habitats in GB that are most at risk.
- Consider lapsed or under-used legislation while drafting PAPs.

In absence of the recommended 'stakeholder consultation' (see Section 1), and not to prejudice the outcome of the consultation, a provisional 'draft' Pathway Management Action Plan (PAP) was developed to assist in the formulation of recommendations on possible appropriate precautionary measures, to identify priority areas for risk reduction as a basis for informed policy development, and to provide the basis of recommending which sectors should be represented in agreeing and coordinating future actions, *i.e.* what actions need to be taken on each of the pathways to address the problem.

3.0. Results

A list of 99 species was compiled from the initial search using the terms 'Ponto-Caspian' + 'invasive', ranging from over 5000 to zero Google hits (Figure 3; Appendix 2). This can be used as an index to indicate the overall 'invasiveness' of the species listed. This is based on the assumption that the more studied species are the more invasive and/or have a significant impact.

3.1. Identification and literature reviews of potentially invasive species

The frequency distribution of Google hits (Appendix 2) revealed two principal break points (Figure 3), the first at species 7 (*Cercopagis pengoi*) and the second at species 18 (*Limnomysis benedeni*), the latter distinguishing the most prominent species, *i.e.* those warranting a literature review as potential future invaders, from all other species. Four of the top 18 species are fish (beluga sturgeon *Huso huso,* sterlet *Acipenser ruthenus,* stellate sturgeon *Acipenser stellatus*) or decapod (Turkish crayfish *Astacus leptodactylus*) that have been subjected to pre-screening and full risk assessments, using the Fish Invasiveness Scoring Kit (Copp *et al.* 2009) or a modular risk analysis scheme for non-native species, either the UK scheme (Baker *et al*. 2008), which in its current version is known as the GB scheme (Mumford *et al.* 2010), or ENSARS (Copp *et al.* 2008) – the European Non-native Species in Aquaculture Risk Assessment Scheme developed for the EU Regulation on the use of alien species in aquaculture. In all cases, the species were assessed as being of either low or medium risk of being invasive and therefore exerting adverse environmental impacts (Copp *et al.* 2009; Copp, Almeida, Merino-Aguirre, Godard *et al.*, unpublished data). The remaining species (Table 1) are reviewed here below.

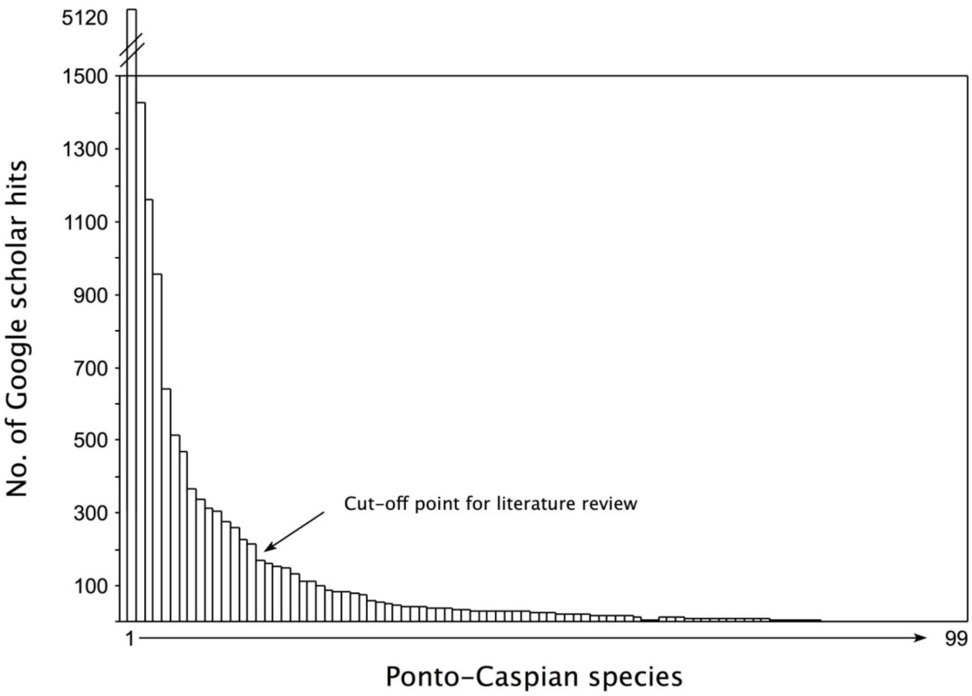

Figure 3. Number of Google Scholar hits for each of the 99 Ponto-Caspian species listed (in same order) in Table 1, with indication of the cut-off point for prominent species, *i.e.* those warranting a literature review as potential future invaders.

Table 1. Top fourteen Ponto-Caspian species from the initial search (see Appendix 2, Figure 3) retained for thorough literature review, which excludes three fish species (*Huso huso*, *Acipenser stellatus*, *Acipenser ruthenus*) and one crayfish (*Astacus leptodactylus*) that have been risk assessed for GB and found to pose low (three species) or medium (one species) risk. The species status in GB is indicated (* = not present; † = early stage of establishment; ¥ = well established).

No.	Latin species name	Taxonomic group
1	*Dreissena polymorpha*¥	Mollusc
2	*Dreissena bugensis/rostriformis**	Mollusc
3	*Neogobius melanostomus**	Fish
4	*Dikerogammarus villosus*†	Crustacean
5	*Chelicorophium/Corophium curvispinum*¥	Crustacean
6	*Cercopagis pengoi**	Crustacean
7	*Hemimysis anomala*†	Crustacean
8	*Echinogammarus ischnus**	Crustacean
9	*Neogobius fluviatilis**	Fish
10	*Cordylophora caspia*†	Cnidaria (hydroid)
11	*Proterorhinus semilunaris**	Fish
12	*Pontogammarus robustoides**	Crustacean
13	*Dikerogammarus haemobaphes**	Crustacean
14	*Limnomysis benedeni**	Crustacean

The review of these species and their potential impacts suggests that, in the short term, most introductions result in increased local species richness rather than the replacement of native species. As such, the impact on diversity is functional rather than taxonomic, and this is of ecological importance (Ojaveer *et al.* 2002), with Ponto-Caspian organisms becoming predominant in both benthic and pelagic food webs, encompassing various trophic levels (herbivores, detritivores, consumers). Large-scale ecosystem impacts have been reported for the Great Lakes of North America, but impacts in Europe have been variable, being less pronounced in some locations, more restricted spatially, and partially overshadowed by other longer-term ecological impacts, with eutrophication being of particular note. Ponto-Caspian invaders can alter energy flows as well as multiple abiotic and biotic components within ecosystems, including the introduction of non-native infectious agents, and the most important impacts of these species in cases are on economically important activities such as fishing and aquaculture. Therefore, the long term impacts that these species may have are significant, in many cases leading to a reduction in species richness. It is essential that the introduction and/or spread of these species are prevented. Additional research is needed to assess their impacts in greater detail.

3.1.1. Zebra mussel (*Dreissena polymorpha*)

Distribution history- Originally from the rivers of Southern Russia, including the Ural, Volga and Dnieper, as well as the Black and Caspian Sea basins, this mussel has a long history of introductions. In Europe it was first reported in Hungary in 1794 (Clarke 1952), United Kingdom in the 1820s (Kerney & Morton 1970), Holland in 1827, Hamburg in 1830 (Morton 1979), USSR in 1845 (Mackie *et al.* 1989), Scandinavia in the 1940s (Nowak 1974), and is now found in Italy, Finland and Ireland (Mackie *et al.* 1989). *D. polymorpha* was first recorded in the Great Lakes of North America in 1986; it is now found in 23 states and 2 Canadian provinces (O'Neill & Dextrase 1994) covering most of the eastern seaboard of North America.

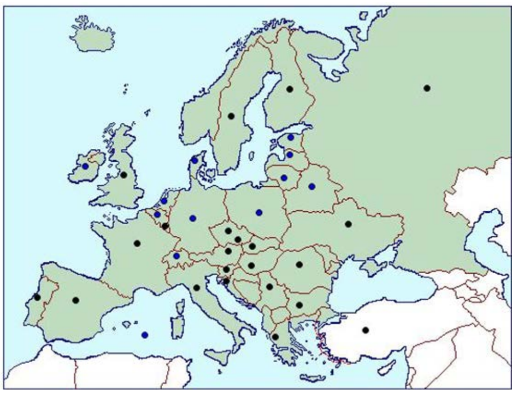

Figure 4. Map showing countries in Europe where *D. polymorpha* is currently found (black dots = present, blue dots = wide spread). Taken from CABI website.

Introduction and dispersal- The dispersal of this species in Europe has mainly been facilitated by human-mediated mechanisms such as hull fouling on commercial and recreational vessels, ballast water transfer, and by timber rafting (Nowak 1971; Karataev *et al.* 1998; Olenin *et al.* 1999; Bij de Vaate *et al.* 2002; Pollux *et al.* 2003), all associated with the development of Europe's canal system. It is thought that the first introduction into the Great Lakes was during a ballast water exchange of a commercial cargo ship (McMahon 1996) with the first report of an established population coming from Lake St. Clair in 1988. Ussery & McMahon (1994) suggested that it is possible that adult mussels may survive the transoceanic voyage by attaching themselves to anchors or their chains of vessels in European harbours, and stored onboard with sufficient moisture to allow the mussels to survive.

The most recent introduction into Ireland was via a trailer-transported pleasure boat arriving by ferry from England (Orlova *et al.* 2004; Pollux *et al.* 2003). The movement of boats overland on trailers between rivers and lakes is the main means by which recreational vessels have dispersed *D. polymorpha* in New Zealand, the US and Ireland (Johnstone *et al.* 1985; Johnson *et al.* 2001; Minchin *et al.* 2003; Benson and Raikow, 2012). Another identified route of transmission is on plants which have *D. polymorpha* attached to them, and then snagged on outboard motors and trailers, which are then transported to different waterbodies (Minchin *et al.* 2006).The rapid colonisation of this species in North America was facilitated by natural mechanisms, such as passive drifting of the larval stage, but also with the assistance of more mobile animals such as ducks (Johnson & Carlton 1996). *D. polymorpha* attached to a hull will not survive out of water for more than five days (Johnson & Padilla 1996), however in cooler and damper conditions of Ireland they can survive for up to 18 days (Pollox *et al.* 2003). There is a further risk that larval *D. polymorpha* are being carried in live bait wells and thus transmitted between water bodies with anglers (Johnson *et al.* 2001).

Other overland human-assisted mechanisms are movements between waterways of dredging or waterway maintenance equipment. Also aquarium enthusiasts are encouraged to use *D. polymorpha* as "biological filters" (Tippit 2004) and as shown in Copp *et al.* (2005), with other non-natives species, released into nearby waters when no longer required. Scuba divers are also suspected of transporting *D. polymorpha* long distances and introducing them into quarries to improve water clarity (Bossenbroek *et al.* 2007). In the USSR and Britain, artificial canals have aided in the dispersal of *D. polymorpha* (Kerney & Morton 1970).

Impact- There are several documented adverse impacts associated with this species in North America. For instance, dense groups of *D. polymorpha* (up to 700,000 m^{-2}; Griffiths *et al.* 1991, Pimentel *et al.* 2005) are known to compete with native North American unionids by settling on the shells of these species. *D. polymorpha* either smothers the clams with the weight of the numerous mussels, or by not allowing them to feed, leading to excessive mortalities or complete extirpation of these populations. Feeding by *D. polymorpha* also results in reductions of phytoplankton and suspended inorganic matter in the water, leading to an increase in water clarity, this increase then leads to an increase in both density and biomass of benthic macrophytes (Hebert *et al.* 1991) as a result of decreased shading from phytoplankton. In addition, *D. polymorpha* accumulate environmental contaminants, with predators of this species showing elevated contaminants in their tissues and lower reproductive success (Maclsaac *et al.* 1996). However, the most documented and expensive impact of this species is its tendency to invade and clog water intake pipes, water filtration, and electric generating plants causing an estimated \$1 billion yr^{-1} in damages and control costs (USACE Army 2002; Pimentel *et al.* 2005).

In stark contrast to the negative attitude held by North Americans, *D. polymorpha* have been intentionally stocked as biomanipulation tools in lakes with poor water quality in the Netherlands (Reeders & Bij de Vaate 1990; Noordhuis *et al.* 1992; Maclsaac *et al.* 1996).

3.1.2. Quagga mussel (*Dreissena rostriformis bugensis*)

Distribution history- This mussel has an original native range in the Lower River Dniepr and Southern River Bug (Son 2007) and is similar to *D. polymorpha* in terms of geographic ranges and life-history strategies, but their invasion histories in Europe and North America differ (Mills *et al.* 1996; Baldwin *et al.* 2002; Orlova *et al.* 2005). In Europe, *D. rostriformis bugensis* were first described in the 1890s in the Ukraine and maintained a relatively restricted distribution until 1992, when they were observed in the River Volga system (Therriault *et al.* 2005). Their range now encompasses a 3000 km stretch of the River Volga from the Uglich and Rybinsk Reservoirs in the north to the River Volga Delta, including the northern Caspian Sea shallows (Therriault *et al.* 2005). The relatively rapid expansion of their range within the Ponto-Azov region resulted from construction of irrigation canals and reservoir impoundment in the basin's watershed (Mills *et al.* 1996; Orlova *et al.* 2004). In 2004, *D. rostriformis bugensis* was reported in the Romanian stretch of the Danube (Popa & Popa 2006), this was considered to be the most westerly point in its range until 2006, when it was recorded in the Hollands Diep, Netherlands (Molloy *et al.* 2007), where it occurred in relatively low numbers between the *D. polymorpha* (1% were *D. rostriformis bugensis*; Bij de Vaate 2010). The species has spread further in Holland and has now been reported in Belgium (Marescaux 2012). It is considered to have arrived as late as 2004 through the Maine-Danube Canal after its opening in 1992. *D. rostriformis bugensis* invaded the Great Lakes area of North America around the same time as the *D. polymorpha* (1987), most likely as larvae via ballast water discharge (van der Veldel 2007), however *D. rostriformis bugensis* was not recognised as a separate species in the Great Lakes until 1991 (Spidle *et al.* 1994).

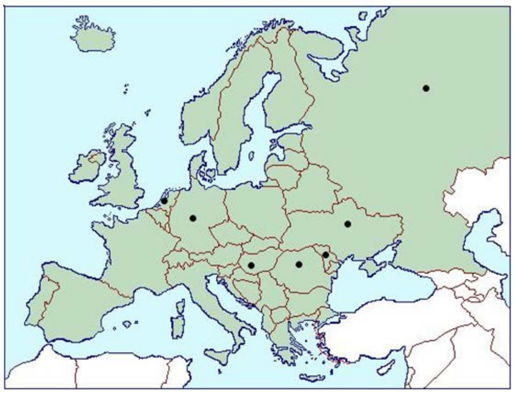

Figure 5. Map showing countries in Europe where *D. rostriformis bugensis* is currently found (black dots = present, blue dots = wide spread). Taken from CABI website.

Introduction and dispersal- A lot of speculation exists on the mechanisms responsible for the rapid range expansion of this species throughout Europe. It is suggested that human-mediated activities such as channelisation and reservoir impoundment have facilitated dispersal, enabling this species to circumvent previously impassable barriers (van der Velde & Platvoet 2007). Establishment of *D. rostriformis bugensis* in most of its introduced range, as well as most upstream movements were made possible with human-mediated vectors, such as shipping, fishing, boating, and scientific expeditions (Orlova & Shcherbina 2002). In addition to these human mediated vectors, conversion of areas from riverine to lacustrine habitat appears to have facilitated the species' spread (Orlova & Shcherbina 2002; Orlova *et al.* 2004). In North America, it is suggested that *D. rostriformis bugensis* could have been rapidly spread via movement of infested boats with natural dispersal vectors including larval drift contributing significantly to a downstream range expansion. Estimates show that veligers were transported at least 306 km downstream before settlement (Stoechel *et al.* 1997). The species is thought to have been discharged in ballast water at the port of Antwerpe resulting in a large number of the species being present (Marescaux 2012).

Impact- Impacts of this species on its introduced waters are similar to that of *D. polymorpha*, with dramatic and significant changes in the lower food web through declines in phytoplankton productivity, as well as increased water clarity (Fahnenstiel *et al.* 2010). However, the most significant difference between these species is the ability of *D. rostriformis bugensis* to colonise deeper (up to 130 m) offshore regions than *D. polymorpha* (Mills *et al.* 1996), and *D. rostriformis bugensis* is now shown to be the more dominant mussel (of these two species) in nearshore regions as well as in large offshore colonies (Nalepa *et al.* 2009). Similar to *D. polymorpha* (USACE Army

2002; Pimentel *et al.* 2005), *D. rostriformis bugensis* is likely to exert expensive economic impacts on water intake pipes and related systems (Hosler 2011), especially those that take water from deeper layers of the water column.

3.1.3. Round goby (*Neogobius melanostomus*)

Distribution history- This Ponto-Caspian goby was first discovered in 1990 in both the Gulf of Gdansk of the Baltic Sea (Skora & Stolarski 1993) and the Great Lakes of North America (Jude *et al.* 1992) and had spread to all five lakes within five years (Jude *et al.* 1992, Charlebois *et al.* 1997). The species has now been recorded in Slovakia (Stráňai & Andreji 2004; Jurajda *et al.* 2005), the Netherlands (van Beek 2006), Austria (Corkum *et al.* 2004), Serbia, Yugoslavia (Simonović *et al.* 1998) and Estonia (Ojaveer 2006).

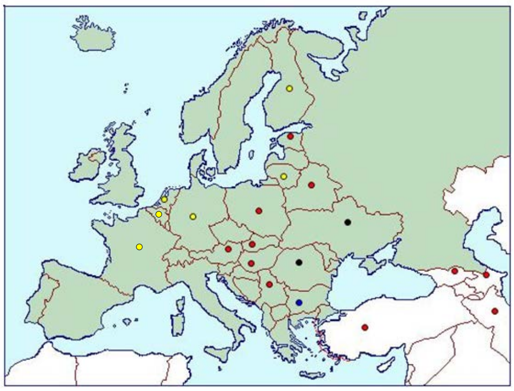

Figure 6. Map showing countries in Europe where *N. melanostomus* is currently found (black dots = present, blue dots = wide spread, red dots = localised, yellow = occasional few reports). Taken from CABI website.

Introduction and dispersal- Round goby are believed to have been introduced into the Baltic and the North American Great Lakes region in the ballast waters of ships from the Black and Caspian Seas. The spread of the round goby in upstream locations in the USA likely occurred via bait bucket transfers (Jude 2001), however in Europe the dispersal of this species may have also been via the Northern Corridor (Figure 1). For example, in the Gulf of Gdasnk, round gobies were found to migrate up to 40 km upstream in the River Vistula (Corkum *et al.* 2004). The secondary movements between sites has potentially been facilitated by abandoned eggs in nests deposited in crevices on freight vessels moving from port to port (Ray & Corkum 2001). Once the gobies have become established in a new area, the fish are likely to disperse naturally (Ray & Corkum 1997; Ray & Corkum 2001; Sapota & Skora 2005).

Impact- There are numerous impacts associated with this species such as competition for food and habitats (van Beek 2006) and predation on native species (French & Jude 2001; Dubs & Corkum 1996; Janssen & Jude 2001) as well as their eggs and young of the year (Chotkowski & Marsden

1999; Charlebois *et al.* 2001; French & Jude 2001; Nichols *et al.* 2003). It is also suggested that predation on invertebrates by the gobies causes increased algal biomass (Kuhns & Berg 1999) as well as bottom up changes in the food web which are likely to occur through the transfer of energy and contaminants (Morrison *et al.* 2000) by gobies.

3.1.4. Freshwater 'killer' shrimp (*Dikerogammarus villosus*)

Distribution history- A gammarid amphipod, the so-called 'killer' shrimp *D. villosus* is native to the Ponto-Caspian region, and has recently invaded most of Western Europe. *D. villosus* was first reported in the Danube in the 1990s and is now found in numerous countries including Austria (Mayer *et al.* 2009), Netherlands (Bij de Vaate & Klink 1995; Dick & Platvoet 2000), Belgium (Messiaen *et al.* 2010), Germany (Nesemann *et al.* 1995), France (Devin *et al.* 2004), Italy (Casellato *et al.* 2006), Belarus (Mayer *et al.* 2009) Poland (Grabowski *et al.* 2007) and England (Defra 2011). The first record from GB was in September 2010, when it was found to be present in high numbers throughout Grafham Water, a reservoir in Cambridgeshire (MacNeil *et al.* 2010). In November 2010 it was found at two further UK sites, in Cardiff Bay and in Eglwys Nunydd Reservoir near Port Talbot (Madgwick & Aldridge 2011) and more recently in the Norfolk Broads, but no further populations had been discovered in the UK or Ireland up to September 2011 (Non-Native Species Secretariat 2011). However, *D. villosus* is considered likely to continue to disperse in Great Britain, facilitated by its broad environmental tolerances, its climatic suitability, and the extensive connectivity, both natural and artificial, of the hydrological network (Gallardo *et al.* 2011).

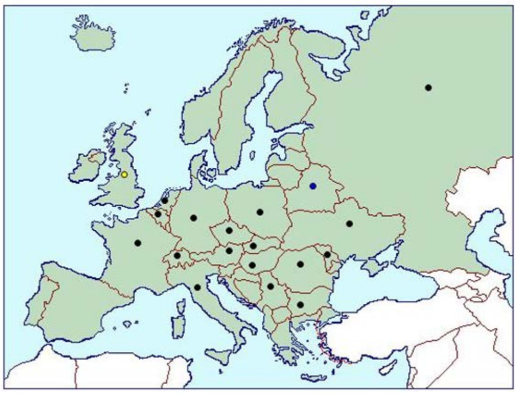

Figure 7. Map showing countries in Europe where *D. villosus* is currently found (black dots = present, blue dots = wide spread, yellow = occasional few reports). Taken from CABI website.

Introduction and dispersal- The dispersal of *D. villosus* is believed to have been facilitated by the opening of canals to connect river basins, such as the Main–Danube–Rhine canal, making them continuously navigable (Dick & Platvoet 2000; Casellato *et al.* 2006). *D. villosus* is thought to have spread initially either via ship ballast water tanks or as hull foulants (Mayer *et al.* 2009), followed by natural dispersal, with secondary invasion into inland lakes (as in Italy) via recreational vessels, associated equipment or equipment used by other water users from Western European lakes and rivers (Casellato *et al.* 2006). As a consequence of these movements, it is predicted that *D. villosus* could extend its range upstream range by up to 40 $km \cdot yr^{-1}$, *i.e.* ≈ 100 $m \cdot day^{-1}$ (Josens *et al.* 2005).

Impact- The numerous impacts associated with killer shrimp has lead to this species being named as one of the 'Top 100' invasive alien species in Europe (www.europe-aliens.org). *D. villosus* is a voracious predator that has replaced a number of native species, such as *Gammarus duebeni* (Dick & Platvoet 2000). This species has a number of ecological attributes which have helped its invasive potential such as: 1) top predator in food webs (Van Riel 2006); 2) a short generation time (Gruszka & Wozniczka 2008); 3) broad salinity and temperature tolerances (Muskó 1992; Bij de Vaate & Klink 1995); 4) rapid growth with an early sexual maturity (Grabowkski *et al.* 2007); 5) high reproductive capacity (Pöckl 2007); and 6) ability to colonise a wide range of substrata (Dedju 1967; Devin *et al.* 2003).

3.1.5. Caspian mud shrimp (*Chelicorophium curvispinum*)

Distribution history- A tubicolous crustacean, the Caspian mud shrimp *Chelicorophium curvispinum* (formerly *Corophium curvispinum*) has spread rapidly from the Ponto-Caspian region throughout European fresh waters. It has only been recorded twice in the British Isles — once in the River Avon at Tewkesbury in 1935, and at Stourport on the Severn in 1962 (Moon 1970). It has now been recorded in Estonia (Herku & Kotta 2006), Germany (Bij de Vaate *et al.* 2002), Poland (Grabowski *et al.* 2007), Hungary (Borza 2011), Belarus (Mastitsky & Makarevich 2007) and is also found in both the River Danube and the Rhine (Baur & Schmidlin 2007).

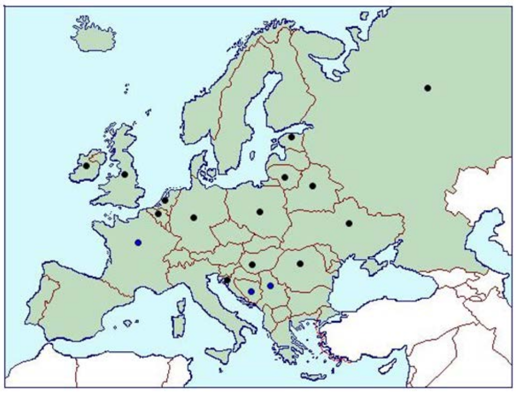

Figure 8. Map showing countries in Europe where *C. curvispinum* is currently found (black dots = present, blue dots = wide spread). Taken from CABI website.

Introduction and dispersal- Similar to a lot of invasive Ponto-caspian species, ballast water and hull fouling are thought to have been the main vectors for dispersal (Leppäkoski & Olenin 2001; Bij de

Vaate *et al.* 2000; Van Riel *et al.* 2010) with floating substratum also mentioned as an aid in possible transmission (Taylor & Harris 1986a, 1986b; Van den Brink *et al.* 1993).

Impact- *Chelicorophium curvispinum* has had a profound impact on the habitats of other benthic fauna. It is an efficient filter feeder, and also very markedly affects its environment by building networks of mud tubes (up to 4 cm thick) that cover the bottom (Van der Velde *et al.* 2002). *C. curvispinum* also cover other benthic animals that are an important source of food for bottom dwelling fish. Competition with native species (and other introduced non-natives such as the *D. polymorpha*) for particulate food has been shown to have occurred, with displacement of species such as the caseless caddisfly larvae *Hydropsyche* spp. in the Rhine (Ricciardi & Rasmussen 1998). Reductions in total organic carbon and suspended material in stretches of the lower Rhine have also been attributed to the population growth and filtration activity of this amphipod (Van den Brink *et al.* 1993). For instance, specimens of *D. polymorpha* were observed to completely overgrow *C. curvispinum* (Baur & Schmidlin 2007). The colonisation success of this species in the Rhine is due to the species having opportunistic characters such as rapid growth, early maturity, ability to produce several generations per year, and high fecundity (den Hartog *et al.* 1992).

3.1.6. Fish-hook waterflea (*Cercopagis pengoi*)

Distribution history- The fish-hook waterflea *Cercopagis pengoi* (synonym *C. ossiani*) is a euryhaline species with a native range restricted to the Caspian, Azov and Aral seas, together with the lower reaches of the rivers entering these water bodies, and coastal lakes in Bulgaria and Turkey (Birnbaum 2006). The species was first recorded in the Baltic Sea region in 1992, in both the Gulf of Riga and the Gulf of Finland (Leppäkoski & Olenin 2000), and by September 1995 comprised 25% of the zooplankton biomass at some stations in the Gulf of Riga (Ojaveer *et al.* 1998). Since then, the distribution has expanded through the western, southern and northern Baltic Sea, and the first record from Germany was reported in 2004 (Birnbaum 2006). In 1998, *C. pengoi* was first recorded in North America, in Lake Ontario, and then spread rapidly through the Great Lakes, appearing in Lake Michigan and the Finger Lakes of New York in 1999 (MacIsaac *et al.* 1999, Makarewicz *et al.* 2001) and reaching Lake Erie by 2001 (Therriault *et al.* 2002) and Lake Huron by 2002 (Benson *et al.* 2012).

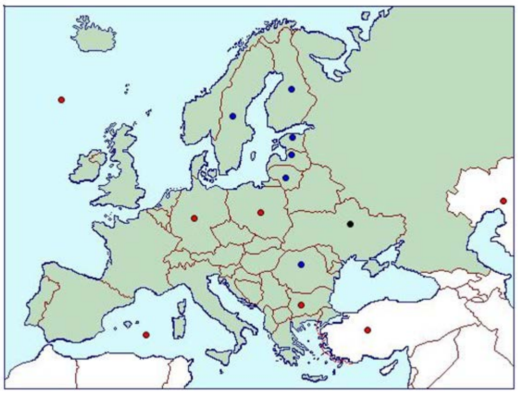

Figure 9. Map showing countries in Europe where *C. pengoi* is currently found (black dots = present, blue dots = wide spread, red dots = localised). Taken from CABI website.

Introduction and dispersal-Key life history characteristics of *C. pengoi* that facilitate dispersal and rapid population growth are asexual reproduction, high fecundity and the ability to produce diapausing eggs (Makarewicz *et al.* 2001). Diapausing eggs are considered to be a major dispersal mechanism for many zooplankton species (Panov *et al.* 2004). Adult *C. pengoi* are tolerant of high concentrations of salinity and could potentially be transported in ballast tanks flushed with sea water. Jacobs and MacIsaac (2007) suggest that entry to North America as diapausing eggs would have been more likely.

The presumed pathway of introduction to the Baltic Sea region was with ballast water in cargo vessels. Genetic studies indicate that the colonization of North America was a secondary introduction from the Baltic Sea populations (Cristecu *et al.* 2001). MacIsaac *et al.* (1999) consider it almost certain that *C. pengoi* was introduced to North America via ballast water transfer, and Makarewicz *et al.* (2001) state that invasion of Lake Michigan almost certainly resulted from movement of contaminated Lake Ontario ballast water by a commercial vessel.

Potential methods of secondary dispersal include ballast water transfer, fishing gear, bait buckets, and trailer-drawn boats, in plumage or digestive tracts of waterfowl or fish, or on contaminated plankton nets (MacIsaac *et al.* 1999). Trials using dead waterfowl in Lake Ontario demonstrated that *Cercopagis* could foul plumage, but although waterfowl could have been a dispersal factor to New York lakes, accidental transfer by sport or commercial fishermen, recreational boaters or researchers is considered more significant (Makarewicz *et al.* 2001). In four trials of brands of fishing lines in Lake Ontario, significant fouling by *C. pengoi* was recorded, with up to 106 diapausing eggs per line (Jacobs & MacIsaac 2007).

Impact- *Cercopagis* has quickly formed large populations in its non-native range (up to 1800 individuals per m^3 at some sites in the Gulf of Finland (Leppäkoski *et al.* 2002)). It is a predator on smaller zooplankton, and its arrival in the eastern Gulf of Finland coincided with declines in cladoceran diversity (Leppäkoski *et al.* 2002). Significant declines were noted in the previously dominant small cladoceran, *Bosmina*, following the spread of *C. pengoi* to the Gulf of Riga (Ojaveer *et al.* 2004). The effects on fish communities are mixed. In the Baltic, it is preyed upon by fish such as herring *Clupea harengus* and smelt *Osmerus eperlanus* (at certain times and locations constituting a substantial portion of their diet), but *C. pengoi* is also considered a competitor with the larvae of these species, preying on the same small cladocerans (Ojaveer & Lumberg 1995; Ojaveer *et al.* 1998, 2004; Gorokhova *et al.* 2004, 2005; Kotta *et al.* 2006). Similarly, studies in Lake Ontario have shown that *C. pengoi* can dramatically reduce summer abundance of zooplankton, by predation on the three dominant smaller species, and it is suggested that *C. pengoi* may therefore be an important competitor of planktivorous fish such as juvenile alewife *Alosa pseudoharengus* as well as forming an increasing component of the diet of adult alewifes (Benoit *et al.* 2002; Bushnoe *et al.* 2003; Laxson *et al.* 2003; Warner *et al.* 2006). Another concern is that by adding another link to the food web, *C. pengoi* invasions could increase contaminant concentrations (such as mercury and PCBs) in fish (Vanderploeg *et al.* 2002).

While *C. pengoi* may at certain times be a significant prey item for commercial fish species, it also has negative economic impacts through the clogging of fishing nets, particularly during its mass occurrences in late summer (Leppäkoski & Olenin 2000). Losses at a fish farm in the eastern Gulf of Finland from 1996 to 2000 exceeded USD 50,000 due to a decline in fish catches as a result of net fouling (Panov *et al.* 2003).

3.1.7. Bloody-red shrimp (*Hemimysis anomala*)

Distribution history- A mysid shrimp, *Hemimysis anomala* originates from brackish coastal waters of the Ponto-Caspian region (Black Sea, Caspian Sea and Sea of Azov) and up to 50 km upstream in the rivers entering these waters. Its spread throughout Europe has been well documented, and has occurred via two routes. In the 1950s and 1960s, the bloody-red shrimp was one of a number of invertebrate species intentionally introduced to reservoirs of the former Soviet Union and Hungary by fisheries scientists to enhance food for fish stocks. One of these introductions, from the lower River Dnieper to a Lithuanian reservoir in 1960, is believed to be the source from which it spread into the Baltic Sea, 200 km down river (Främmande Arter 2006; Audzijonte *et al.* 2008). Following the first record in the Baltic Sea in 1992, in the Gulf of Finland (Salemaa & Hietalahti 1993), *H. anomala*'s tolerance of brackish water allowed the species to spread around the Baltic coast, with first records from Sweden in 1995 and Estonia in 2009 (Kotta & Kotta 2010).

Concurrently, *H. anomala* also spread from the Danube Delta through the canal network of continental Europe, spreading via the Danube, the Main-Danube Canal and the Rhine, with the first record from the Netherlands occurring in the Rhine Delta in September 1997. Further new introductions of *H. anomala* are believed to account for its appearance in 1997 at several locations in the western Netherlands, all in the vicinity of international ports, suggesting arrival in ballast water (Bij de Vaaate *et al.* 2002). *H. anomala* was first recorded in Belgium in 1999 (Verslycke *et al.* 2000), and in the River Rhone in France in 2003, and had spread down the Rhone to the Mediterranean Sea by 2007 (Wittmann & Ariani 2009). In GB, this species was first recorded in November 2004 in Nottinghamshire in the Erewash Canal, which empties into the River Trent, and in 2005 it was found at a number of sites in the River Trent and associated canals.

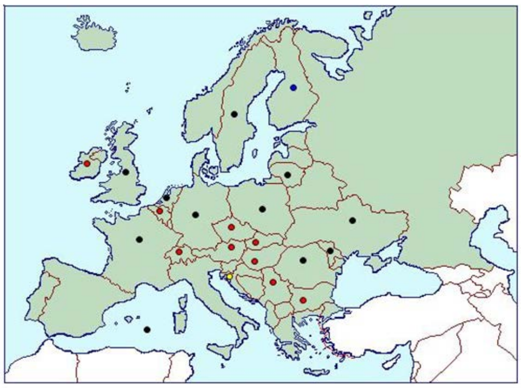

Figure 10. Map showing countries in Europe where *H. anomala* is currently found (black dots = present, blue dots = wide spread, red dots = localised, yellow = occasional few reports). Taken from CABI website.

Introduction and dispersal- It is not known whether the GB populations come from deliberate introductions, or accidental introduction, possibly with foreign boats used at a rowing centre (Holdich *et al.* 2006; Stubbington *et al.* 2008). In 2008, *H. anomala* was discovered in the Shannon River system in Ireland (Minchin & Holmes 2008). Again the method of arrival remains unknown, but Minchin & Boelens (2010) suggest a number of possible pathways – transport by leisure craft (in bait boxes, ship bilge or toilet waters), deliberate release into ponds as live food for fish, or accidental transfer in water involved in the stocking of imported fishes, or in imports of aquatic plants. The first record of *H. anomala* in North America was in Lake Michigan in 2006 (Pothoven *et al.* 2007), although Marty (2008) suggests that the species was likely to have been present unnoticed for a number of years. By 2009, *H. anomala* occurred in four of the Great Lakes of North America (Marty *et al.* 2010). Genetic studies have confirmed that both the English and North American invasions are from the Danube lineage, as opposed to a Baltic Sea origin (Audzijonte *et al.* 2008). The arrival in North America is considered very likely to be the result of ballast water release from transoceanic ships (Kipp & Ricciardi 2007), and its subsequent spread around the Great Lakes region is likely to be the result of both natural dispersal and anthropogenic methods. One example of the latter is the appearance of *H. anomala* in a lake in New York, 53 km upstream of the suspected source population in Lake Ontario but separated by several large rapids, locks and dams. Consequently, the likely vector of introduction is considered to be pleasure boat or light commercial boat traffic using the canal, or overland transport (Brooking *et al.* 2010).

Impact- There is limited information on ecological impacts of *H. anomala*, which is an opportunistic omnivore, feeding primarily on zooplankton, but also preying on benthic invertebrates, and scavenging dead animal matter. Younger individuals feed mainly on phytoplankton (Ketelaars *et al.* 1999; Borcherding *et al.* 2006). Studies in reservoirs in the Netherlands suggested a dramatic effect on zooplankton biomass and diversity (Ketelaars *et al.* 1999). *H. anomala* is considered an energy-rich food source for planktivorous fish species (Kipp & Ricciardi 2007), and they were found to be an important prey item of alewife in the Great Lakes (Lantry *et al.* 2010). Studies in the River Rhine suggest that *H. anomala* may become an important link between primary/secondary production and higher trophic levels (Borcherding *et al.* 2006), and similar concerns have been expressed about potential impact to food webs in the Great Lakes of North America (Walsh *et al.* 2010).

3.1.8. Ponto-Caspian amphipod (*Echinogammarus ischnus*)

Distribution history- This gammarid is native to the drainage systems of the Black and Caspian seas (Cristecu *et al.* 2004), and has spread through Europe using the invasion corridor connecting the River Dnieper with the Baltic Sea basin via the rivers Vistula and Neman. The first record outside its native range was in the River Vistula below Warsaw in 1928. In the late 1970s and 1980s, *E. ischnus* was discovered in the North Sea drainage basin (in canals joining the Elbe, Weser and Ems rivers, and the Rhine-Herne and Wesser-Dattel canals) (Cristecu *et al.* 2004). *E. ischnus* was first recorded in the Rhine in 1989, reaching the Netherlands in 1991 and the French stretches of the Rhine in 2009 (Labat *et al.* 2011). Colonisation of North America has also occurred, probably via ballast water transfer, with the first records in 1995 in the Detroit River, although archived samples suggest that *E. ischnus* had been present for at least a year prior to discovery (Witt *et al.* 1997; van Overdijk 2003). Within a few years, Dermott *et al.* (1998) reported that it was widespread in the Great Lakes.

Figure 11. Map showing countries in Europe where *E.ischnus* is currently found (black dots = present). Adapted from CABI website.

Introduction and dispersal- The rapid dispersal of *E. ischnus* through the canal systems of Europe, and its spread from the west end of Lake Erie to the east end in just two years would suggest that this species is very active in the water column (Nalepa *et al.* 2001). Ballast water from transatlantic ships is recognised as the source of its introduction to the Great Lakes of North America and ports of the Lower River Rhine are considered a more likely source than the Black Sea region due to a greater volume of ship traffic and shorter transit times (Cristecu *et al.* 2004). Within the Great Lakes region, its spread appears to have been largely due to natural dispersal, although two 'jumps' into Lakes Michigan and Superior were likely to have resulted from interlake movement of ballast water by commercial ships (Nalepa *et al.* 2001; Vanderploeg *et al.* 2002; Cristecu *et al.* 2004). The previous invasion of the Great Lakes by *D. polymorpha* and *D. rostriformis bugensis* has facilitated the invasion of this species, by providing optimal habitat where *Echinogammarus* can survive (Stewart *et al.* 1998).

Impact- There is evidence from the Great Lakes that *E. ischnus* competitively displaces other amphipod taxa in rocky habitats (Dermott *et al.* 1998; Nalepa *et al.* 2001; van Overdijk 2003), and particularly in those areas dominated by the non-native *D. rostriformis bugensis* (Duggan & Francoeur 2007). Some studies have reported that *E. ischnus* replaces the native amphipod *Gammarus fasciatus* on rocky habitats in a number of rivers (Dermott *et al.* 1998), whereas other studies report co-existence of the two species, as a result of differences in microhabitat use (van Overdijk *et al.* 2003; Gonzalez & Burkart 2004). In Europe, displacement of native *Gammarus* species has also been noted for the Vistula River, Poland, where native *G. pulex* and *G. varsoviensis* were replaced in the 1920s by *Chaetogammarus ischnus* (syn. *Echinogammarus ischnus*), which was itself out-numbered by subsequent invasions of *Dikerogammarus haemobaphes* and *Pontogammarus robustoides* (Jazdzewski *et al.* 2004).

3.1.9. Monkey goby (*Neogobius fluviatilis*)

Distribution history- The native range of the monkey goby *Neogobius fluviatilis* is fresh and brackish waters in the basins of the Black Sea and the Sea of Marmara (Neilson & Stepien 2011). It was discovered in Polish rivers in the mid 1990s, and since the beginning of this century have been an abundant component of the fauna of the lower River Vistula (Kakareko *et al.* 2005), expanding 836 km downstream in just seven years (Grabowska *et al.* 2010). The monkey goby has also moved up the River Danube, colonising areas at least 219 km upstream of Lake Balaton, Hungary within 31 years of the first record there, in 1970 (Holčík *et al.* 2003). Genetic studies confirmed that the Danube and Vistula populations originate in the north-western Black Sea area (Neilson & Stepien 2011). The species was first recorded in Germany in 1998 (in Duisberg Harbour), and in the Netherlands (tributaries of the Rhine) in the following year (Van Kessel 2009). Its discovery in Greece in August 2011, in the River Evros, is likely to be the result of an introduction, although the possibility of it being a previously overlooked native species cannot be discounted (Zogaris & Apostolou 2011).

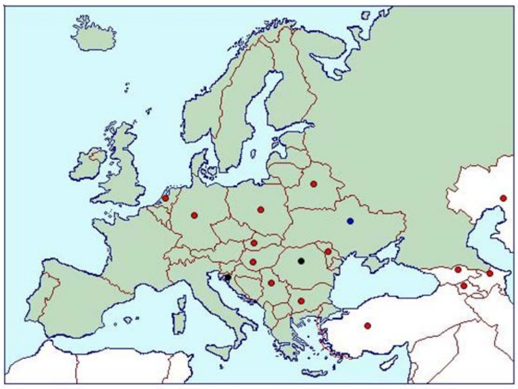

Figure 12. Map showing countries in Europe where *N. fluviatilis* is currently found (black dots = present, blue dots = wide spread, red dots = localised, yellow = occasional few reports). Taken from CABI website.

Introduction and dispersal- The spread of *N. fluviatilis* through central Europe is probably due to a combination of natural dispersal through rivers and canals, and transport in ballast water (Ahnelt *et al.* 1998). Natural dispersal may have been triggered in the last few decades by increasing water temperature, in part due to canalization, reservoirs and industry (Harka & Bíró 2007).

Impact- Owing to its more limited habitat requirements (with a strong specialization for sandy substrata), *N. fluviatilis* is considered to have less invasion potential and likely impact than other Ponto-Caspian goby species such as *N. melanastomus* and *N. kessleri* (Jurajda *et al.* 2005; Capova *et al.* 2008). Compared to the other Ponto-Caspian gobies, the impact of the monkey goby on native fish and invertebrate communities has been little studied. One concern is the potential of this species to act as a reservoir for non-native parasites and diseases, such as *Gyrodactylus proterorhini* in the River Vistula (Grabowska *et al.* 2010; Mierzejewska *et al.* 2011).

3.1.10. Colonial hydroid (*Cordylophora caspia*)

Distribution history- *Cordylophora caspia* is a colonial hydroid (growing up to 10 cm high), native to estuaries and brackish coastal lagoons in the Ponto-Caspian region (Olenin 2006). It was introduced to the Baltic Sea in the early 1800s, and spread rapidly through Europe, reaching Ireland by 1842. *C. caspia* is now known from temperate and tropical regions of every continent (excluding Antarctica), with first records from North America in 1860, Australia in 1885 and the Panama Canal in 1944 (Olenin 2006). It has a wide range of salinity tolerance, and can establish freshwater populations (Smith *et al.* 2002), although recent genetic studies suggest that freshwater populations may belong to a different cryptic species (Folino-Rorem *et al.* 2009). In Britain, *C. caspia* has a sporadic distribution, associated mainly with areas of low salinity in estuaries and brackish lagoons (Tyler-Walters & Pizzolla 2007).

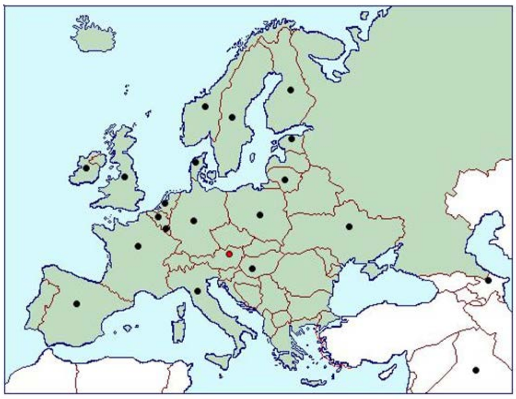

Figure 13. Map showing countries in Europe where *C. caspia* is currently found (black dots = present, red dots = localised). Taken from CABI website.

Introduction and dispersal- *Cordylophora caspia* is capable of spreading by a number of different mechanisms, including local colony expansion through vegetative growth, sexual population expansion by either natural or anthropogenic dispersal of planulae, and asexual expansion by dispersal of drifting or fouling menonts (colony fragments containing living tissue) (Darling & Folino-Rorem 2009). Menonts are highly resistant to changes in temperature and can survive exposure to sea water (Bij de Vaate 2002) as well as a number of biofouling control efforts (Folino-Rorem & Indelicato 2005).

Impact- This hydroid forms large dense colonies, competing with native species for space, and potentially competing with larval fish for prey (Olenin & Leppäkoski 1999). The filamentous structure may facilitate the settlement and recruitment of invasive dreissenid mussel larvae (Folino-Rorem *et al.* 2006). As a fouling species it can have an economic impact, particularly in clogging up industrial cooling systems and water treatment plants (Folino-Rorem & Indelicato 2005; Mant *et al.* 2011).

3.1.11. Freshwater tubenose goby (*Proterorhinus semilunaris*)

Distribution history- A similar feature of all Ponto-Caspian gobies that expanded up the River Danube in the 1990s is a native distribution that encompassed the lower Danube. However, *P. semilunaris* is unique in that its native range includes middle sections of the Danube in Slovakia and Austria (Copp *et al.* 2005). Until recently, the identity of the *Proterorhinus* species was confused, but DNA sequencing now suggests that *P. marmoratus* actually comprises a complex of at least three cryptic species: the marine tubenose goby *P. marmoratus* native to estuaries and brackish waters of the Black Seas; the freshwater tubenose goby *P. semilunaris* native to the lower Danube, the rivers Dniester and Dnieper; and at least one further taxon from the Caspian Sea basin (Stepien & Tumeo 2006; Neilson & Stepien 2008). It is the freshwater species, *P. semilunaris* that has proved to be invasive, being first found in Lake St. Clair of the North American Great Lakes around 1990, with range extension into Lake Erie by 2007 (Jude *et al.* 1992; Kocovsky *et al.* 2011). Although its rate of spread has proved to be slower than that of the round goby, *P. semilunaris* has expanded its range up the Danube in recent decades, first up tributaries of the Middle Danube in the early 1990s (see Prášek & Jurajda 2005), and then invading the Rhine after the opening of the Rhine-Main-Danube in 1992 to Germany by 1999 (reviewed by Borcherding *et al.* 2011) and the Netherlands in 2002 (Van Kessel *et al.* 2009), and then up the Moselle in 2005 (von Landwüst 2006) and the French Rhine in 2007 (Manné & Poulet 2008). Indeed, *P. semilunaris* is the only species of the Ponto-Caspian gobies predicted (Balon *et al.* 1986), and then observed (Borcherding *et al.* 2011), to invade the Rhine via the Main-Danube Canal. Elsewhere in Europe, *P. semilunaris* was also recorded in the Baltic Sea region for the first time in 2006 (Antsulevich 2007).

Figure 14. Map showing countries in Europe where *P. semilunaris* is currently found (black dots = present). Adapted from CABI website.

Introduction and dispersal- A number of invasion pathways are known for *P. semilunaris*, in addition to natural dispersal attributed to drift associated with nocturnal vertical migration of larvae (Kocovsky *et al.* 2011). Transoceanic transport in ballast water is assumed to be responsible for its arrival in North America (Wonham *et al.* 2000). It can also be spread by transport of egg clutches on the hulls of ships, and by accidental stocking with other fish species (von Landwüst 2006). Its recent arrival in the Czech Republic is attributed to use as live bait fish by anglers (Prášek & Jurajda 2005), and it has been suggested that this vector could explain its discontinuous range in Lake Erie (Kocovsky *et al.* 2011).

Impact- Studies suggest that *P. semilunaris* could potentially compete with a number of native fish species for food and space. In the Great Lakes, there is substantial overlap in diet and habitat with the rainbow darter *Etheostoma caeruleum*, and potential competition with the johnny darter *E. nigrum* for spawning sites on the underside of rocks (French & Jude 2001; Kocovsky *et al.* 2011). In Europe, particular concern has been expressed about the possible impact on bullheads *Cottus*, following observations of a decline in European bullhead *Cottus gobio* population density in the Slovak River Danube after invasion by several goby species in the 1990s (Jurajda *et al.* 2005). *C. gobio* were absent from the headwater of a weir populated by *P. semilunaris* on the River Moselle (von Landwüst 2006), and preliminary experiments have shown that *C. gobio* may be outcompeted for shelter, moving to less optimal habitats (van Kessel *et al.* 2011; also G.H. Copp, personal observation).

3.1.12. Ponto-Caspian amphipod (*Pontogammarus robustoides*)

Distribution history- The native range of the gammarid amphipod *Pontogammarus robustoides* encompasses the lower sections of the rivers Volga, Don, Dnieper, Dniester and Danube, and brackish and freshwater lakes around the Black Sea (Jazdzewski 1980). In the 1960s *P. robustoides* was introduced into a number of Ukrainian, Caucasian and Lithuanian artificial dam reservoirs. It subsequently spread into the Baltic Sea catchment area, reaching the Gulf of Finland in the 1990s (Berezina & Panov 2003). By a combination of spread around inshore Baltic waters and through river/canal systems, it has now reached a number of waterways in Poland, Belarus and Germany (Bij de Vaate *et al.* 2002, Grabowski *et al.* 2007; Mastitsky & Makarevich 2007), as well as Lake Ladoga in Russia (Kurashov & Barbashova 2008).

Figure 15. Map showing countries in Europe where *P. robustoides* is currently found (black dots = present). Adapted from CABI website.

Introduction and dispersal- The first recorded movements out of its native range were intentional introductions into reservoirs in the former Soviet Union to improve food for fish stocks (Arbačiauskas *et al.* 2010). Its subsequent spread through Europe is attributed to a combination of shipping, probably attached to fouling organisms on hulls, and natural dispersal assisted by newly created canals (Jazdzewski *et al.* 2002; Mastitsky & Makarevich 2007).

Impact- *P. robustoides* is a large, competitive and aggressive amphipod species, which preys on invertebrates, and may pose a threat to native amphipods (Grabowski 2006). In parts of the River Vistula, Poland, native species of *Gammarus* are now outnumbered (or entirely replaced) by *P. robustoides* and other invasive gammarid species (Jazdzewski *et al.* 2004; Grabowski *et al.* 2006; Surowiec & Dobrzyca-Krahel 2008). Studies in Lithuanian lakes found a significant reduction in species richness and community diversity of benthic invertebrates in the habitats where *P. robustoides* was numerous (Gumuliauskaitė & Arbačiauskas 2008). It also grazes on algae, and in the eastern Gulf of Finland has been shown to reduce biomass of filamentous *Cladophora* algae (Berezina *et al.* 2005).

3.1.13. Ponto-Caspian amphipod (*Dikerogammarus haemobaphes*)

Distribution history- The gammarid *Dikerogammarus haemobaphes* occurs naturally in the lower and middle stretches of Black and Caspian Sea rivers, and brackish lagoons. It spread up the southern corridor, using the River Danube and the Main-Danube canal (first records in 1993) to reach the River Rhine (Grabowski *et al.* 2007). It was first recorded in France in 2008, in the Meuse and Seine rivers (Labat *et al.* 2011). Expansion has also occurred along the central corridor, leading to the first records in the Polish stretches of the River Vistula in 1997 (Grabowski *et al.* 2007) and in Belarus in 2006 (Mastitsky & Makarevich 2007). It has recently reached the Baltic Sea, with records from the Gulf of Gdansk in 2010 (Dobrzyca-Krahel & Rzemykowska 2010).

Figure 16. Map showing countries in Europe where *D. haemobaphes* is currently found (black dots = present). Adapted from CABI website.

Introduction and dispersal- It is likely that this species has used similar pathways to other invasive amphipods, *i.e.* dispersal through newly-created canal systems aided by shipping. The prior spread of zebra mussels has assisted the establishment of *D. haemobaphes* in newly invaded areas by providing its preferred habitat (Kobak *et al.* 2009). Conversely the arrival of the killer shrimp *D. villosus* can lead to displacement of *D. haemobaphes* (Müller *et al.* 2002; Kinzler *et al.* 2009).

Impact-In comparison to *D. villosus,* there is little information available on its ecological impact. However *Dikerogammus haemobaphes* is now the dominant amphipod in stretches of the River Vistula (Poland), displacing the previous invader, *Echinogammarus ischnus*, which had itself replaced native *Gammarus* species (Jazdzewski *et al.* 2004).

3.1.14. Mysid Shrimp (*Limnomysis benedeni*)

Distribution history- The mysid shrimp *Limnomysis benedeni* is native to brackish and fresh water in the Caspian, Black, Azov and Marmara Seas and the rivers entering them. It has spread through Europe using two routes – the southern corridor and the central corridor. Deliberate introduction by Soviet fisheries scientists into the Kaunas Reservoir (Lithuania) in 1960 provided the source population from which it spread downstream to the Baltic Sea (Arbačiauskas *et al.* 2010). In the southern corridor, *L. benedeni* appeared in a harbour on the Danube in Budapest, Hungary in 1946, 1200 km beyond its native range, and over the next fifty years spread upstream to reach the Main-Danube Canal by 1998. Further range expansion (Wittmann 2011) has since occurred to the Netherlands (1997), France (1998) and Belgium (2005).

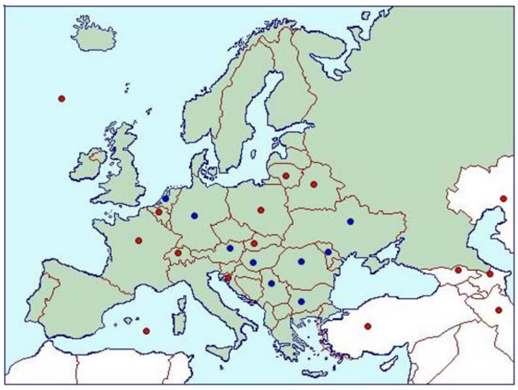

Figure 17. Map showing countries in Europe where *L. benedeni* is currently found (black dots = present, red dots = localised). Taken from CABI website.

Introduction and dispersal- The spread through European river and canal systems has been attributed to unintentional transport by ships, with genetic analysis suggesting the spread up the Danube stems from multiple invasion events (Audzijonyte *et al.* 2009). Secondary dispersal pathways may also be locally important, and Borza *et al.* (2011) consider that its wide distribution in isolated fishing lakes in Hungary is evidence for accidental introduction with stocked fish acting as an important dispersal mechanism. Accidental introduction with aquatic plants or by overland transport of boats have also been suggested as likely dispersal mechanisms (Wittmann 2011).

Impact- *Limnomysis benedeni* is omnivorous, and in diet studies has been described as predominantly a detritivorous herbivore (Aßmann *et al.* 2009). Few ecological impacts have been reported, although outdoor mesocosm experiments demonstrated selective predation on components of the zooplankton, leading to reduced densities of cladocerans, rotifers and copepod nauplii (Fink *et al.* 2012).

3.2. Assessment of primary invasion pathways and mechanisms

The findings of the literature review (Table 2) provide a list of the primary pathway identified as being used by each species, the mechanism (e.g. hull fouling), the driver for the pathway (e.g. commercial) and the relevant sector. These latter two will be used to inform on the development of the PAP. Secondary pathways are also listed, indicating possible mechanisms of dispersal (Table 2). Risk assessments of each species with regard to the primary pathways of introduction for each mechanism and the secondary pathways are also summarized (Table 3).

The primary pathways of introduction of Ponto-Caspian species recongised in the literature are shipping and air/land transport. Based on the species listed within this report, the literature listed shipping as the main pathway in comparison to air/land transport (see Figure 18).

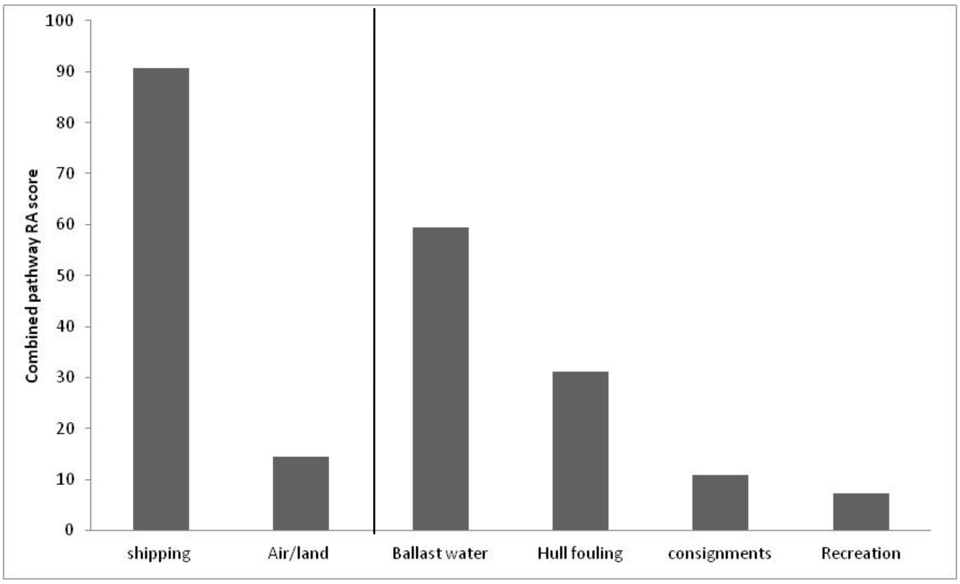

Figure 18. Graph showing the combined pathway RA score (Table 3) for the 1° pathways of introduction and associated mechanisms.

Both of the pathways can be further broken down into the consituent mechanisms. For example possible mechanisms associated with the trans-location of Ponto-Caspian species by shipping are ballast water and hull fouling. The combined pathway RA scores (calculated based on the evidence provided in the literature) suggest that ballast water is the predominant mechanism of introduction, followed by hull fouling, contaminated consignments, and then recreational activities.

Table 2. Top fourteen Ponto-Caspian species from the initial search (see Appendix 2, Figure 3) retained for thorough literature review, which revealed the primary (1°) pathways, the associated mechanisms, drivers, sectors as well as secondary (2°) dispersal pathways relevant to GB. Possible but less likely mechanisms for GB are given in parenthesis.

No.	Species (Latin) name	1° Pathway	Mechanism	Drivers	Sectors	2° Pathway
1	*Dreissena polymorpha*	Shipping	BW, HF	GT, EG, REC	CT, L	ND, IB, IS
2	*Dreissena bugensis/rostriformis*	Shipping	BW, HF	GT, EG, REC	CT, L	ND, AG, IB, IS
3	*Neogobius melanostomus*	Shipping	HF (BW)	GT, EG, REC	CT, L	ND, AG, IB, IS
4	*Dikerogammarus villosus*	Shipping	BW	GT	CT	ND, AG, IB, IT
		Air/Land transport	CC	GT, EG	O/A, L	ND, AQ
5	*Chelicorophium/Corophium curvispinum*	Shipping	BW, HF	GT, EG	CT, L	ND
6	*Cercopagis pengoi*	Shipping	BW	GT	CT	AG, IB, IT
7	*Hemimysis anomala*	Shipping	BW	GT	CT	IB, AQ, US
8	*Echinogammarus ischnus*	Shipping	BW	GT	CT	ND
9	*Neogobius fluviatilis*	Shipping	HF (BW)	GT	CT	ND, IB, US, IR
		Air/Land transport	CC	GT, EG, REC	O/A, L	ND, AQ
10	*Cordylophora caspia*	Shipping	BW, HF	GT	CT	ND, IB
11	*Proterorhinus semilunaris*	Shipping	HF (BW)	GT	CT	ND, US, IR
		Air/Land transport	CC	GT, EG, REC	O/A, L	ND, IB, US, IR
12	*Pontogammarus robustoides*	Shipping	BW, HF	GT	CT	ND
13	*Dikerogammarus haemobaphes*	Shipping	BW	GT	CT	ND
14	*Limnomysis benedeni*	Shipping	BW	GT	CT	ND, US, IB

Key:

1° Pathways: shipping, air transport, land transport
Mechanisms: ballast water (BW), hull fouling (HF), consignment contamination (CC)
Drivers: global trade (GT), economic growth (EG), recreation (REC)
Sectors: commercial trade (CT), leisure (L), ornamental/aquarist trade (O/A)
2° Pathways: natural dispersal, often aided by canal construction (ND), angler's gear (AG), inland boating (IB), aquatics trade (AQ), incidental transport by birds, mammals, vehicles (IT), un-intentional stocking with fish or plants (US), intentional release, e.g. discard of live bait by anglers (IR),

.

Table 3. Risk Assessment scores for top fourteen Ponto-caspian species based on combined total of all pathways. Species already present in GB are shown in **bold.**

No.	Species (Latin) name	Pathway	Mechanism	Score	Confidence	Total Score	Total Confidence
1	***Dreissena polymorpha***	1° Shipping	BW	4.3	2.6		
		1° Shipping	HF	4.0	2.6		
		2°	IB	3.4	2.1		
		2°	IR	3.2	2.1		
						14.9	**9.4**
2	*Dreissena bugensis/rostriformis*	1° Shipping	BW	4.3	2.6		
		1° Shipping	HF	4.0	2.6		
		2°	AG	3.1	2.1		
		2°	IB	3.4	2.1		
		2°	US	3.6	2.1		
						18.4	**11.5**
3	*Neogobius melanostomus*	1° Shipping	BW	4.0	2.9		
		1° Shipping	HF	3.4	2.6		
		2°	AG	3.1	2.1		
		2°	IB	3.4	2.1		
		2°	US	3.6	2.1		
						17.5	**11.8**
4	***Dikerogammarus villosus***	1° Shipping	BW	4.3	2.9		
		1° Shipping	HF	4.3	2.9		
		2°	AG	3.1	2.1		
		2°	IB	3.4	2.1		
		2°	IT	3.2	2.1		
		1° Air/Land transport	CC	3.6	2.6		
						21.9	**14.7**
5	***Chelicorophium/Corophium curvispinum***	1° Shipping	BW	4.6	2.9		
		1° Shipping	HF	3.9	2.9		
						8.5	**5.8**
6	*Cercopagis pengoi*	1° Shipping	BW	4.3	2.3		
		2°	AG	3.1	2.1		
		2°	IB	3.4	2.1		
		2°	IT	3.2	2.1		
						22.5	**14.4**
7	***Hemimysis anomala***	1° Shipping	BW	4.5	2.9		
		2°	IB	3.4	2.1		
		2°	AQ	3.1	2.1		
		2°	US	3.6	2.1		
						14.6	**9.2**

No.	Species (Latin) name	Pathway	Mechanism	Score	Confidence	Total Score	Total Confidence
8	*Echinogammarus ischnus*	1° Shipping	BW	4.0	2.0		
						4.0	**2.0**
9	*Neogobius fluviatilis*	1° Shipping	BW	4.1	2.1		
		2°	IB	3.4	2.1		
		2°	US	3.6	2.1		
		2°	IR	3.2	2.1		
		1° Air/Land transport	CC	3.6	2.6		
		2°	AQ	3.1	2.1		
						21.0	**13.1**
10	***Cordylophora caspia***	1° Shipping	BW	4.1	2.7		
		1° Shipping	HF	3.6	2.6		
		2°	IB	3.4	2.1		
						11.1	**7.4**
11	*Proterorhinus semilunaris*	1° Shipping	BW	4.3	2.1		
		1° Shipping	HF	3.9	2.1		
		2°	US	3.6	2.1		
		2°	IR	3.2	2.1		
		1° Air/Land transport	CC	3.6	2.6		
		2°	IB	3.4	2.1		
		2°	US	3.6	2.1		
		2°	IR	3.2	2.1		
						28.8	**17.3**
12	*Pontogammarus robustoides*	1° Shipping	BW	4.1	2.1		
		1° Shipping	HF	4.1	2.1		
						8.2	**4.2**
13	*Dikerogammarus haemobaphes*	1° Shipping	BW	4.3	2.1		
						4.3	**2.1**
14	*Limnomysis benedeni*	1° Shipping	BW	4.3	2.1		
		2°	US	3.6	2.1		
		2°	IB	3.4	2.1		
						11.3	**6.3**

Key:

1° Pathways: shipping, air transport, land transport
Mechanisms: ballast water (BW), hull fouling (HF), consignment contamination (CC)
2° Pathways: natural dispersal, often aided by canal construction (ND), angler's gear (AG), inland boating (IB), aquatics trade (AQ), incidental transport by birds, mammals, vehicles (IT), un-intentional stocking with fish or plants (US), intentional release (IR)

3.3. Assessment of secondary invasion pathways and mechanisms

Based on the combined pathway RA scores as derived from the literature it is possible to indicate which pathways are the most likely means of secondary movement of Ponto-Caspian species once they have been introduced and established in a country. The primary means of dispersal recognised in the literature was inland boating (see Figure 19), with unintentional stocking with fish and plants being the second major pathway. Intentional release and angling gear have a very similar combined score, but are of a lesser risk. Incidental transport and aquatics trade also are considered to be of a lesser risk and also have similar scores.

Natural dispersal was not included as part of the assessment. Despite being recognised as a major secondary pathway for several species and playing a significant role within Europe and North America it is difficult to differentiate between natural spread and other distribution mechanisms.

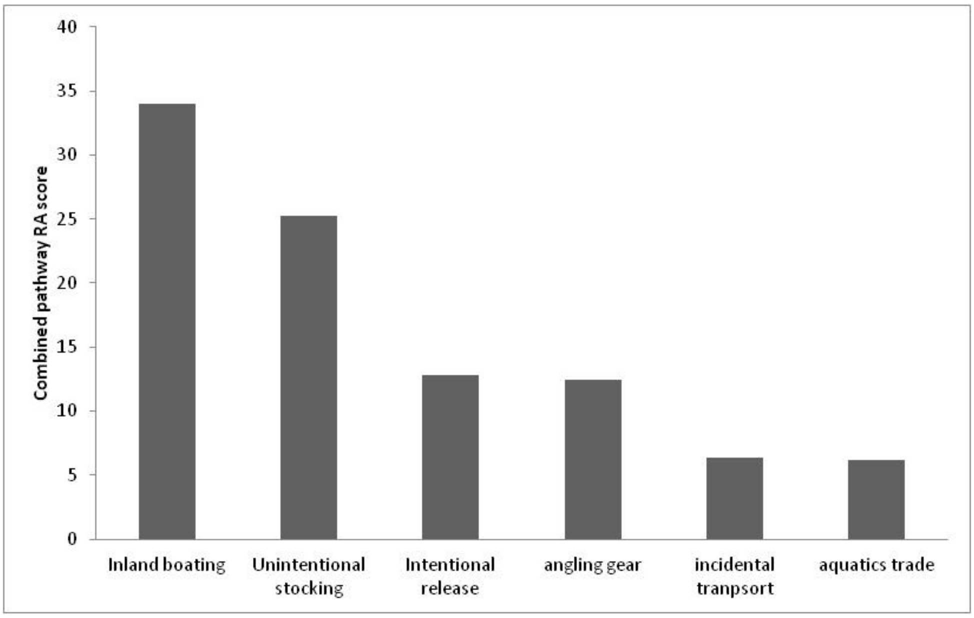

Figure 19. Graph showing the combined pathway RA score (Table 3) for the 2° pathways of spread.

3.4. Assessment of high risk Ponto-Caspian species

From the risk assessment it is possible to rank the species in order of those that pose the greatest threat of introduction and subsequent spread in GB (Figure 20). Those species that are recognised as being already present within GB waters are to the right of the graph while those that are not currently in GB are to the left. The risks of introduction of those species not currently found in GB are ranked in the following order (from high to low risk):

1. *Proterorhinus semilunaris* (freshwater tubenose goby)
2. *Cercopagis pengoi* (fish hook water flea)
3. *Neogobius fluviatilis* (monkey goby)

4. *Dreissena bugensis/rostriformis* (quagga mussel)
5. *Neogobius melanostomus* (round goby)
6. *Limnomysis benedeni* (Mysid shrimp)
7. *Pontogammarus robustoides* (Ponto-Caspian amphipod)
8. *Dikerogammarus haemobaphes* (Ponto-Caspian amphipod)
9. *Echinogammarus ischnus* (Ponto-Caspian amphipod)

The species currently found in GB are risked in the following order based on the risk assessement:

1. *Dikerogammarus villosus* (killer shrimp)
2. *Dreissena polymorpha* (zebra mussel)
3. *Hemimysis anomala* (blood-red shrimp)
4. *Cordylophora caspia* (colonial hydroid)
5. *Chelicorophium/Corophium curvispinum* (Caspian mud shrimp)

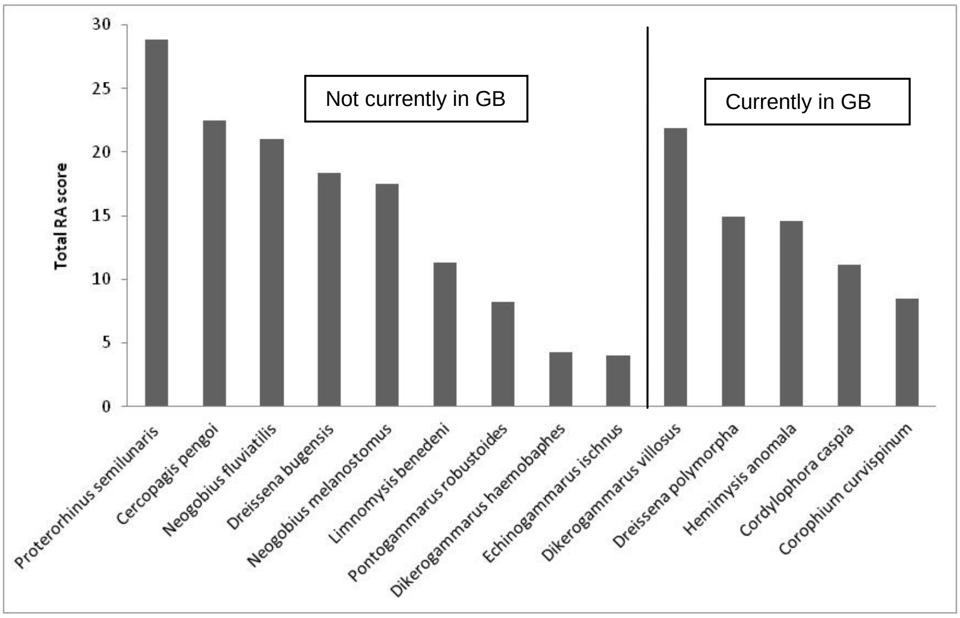

Figure 20. Graph showing the total RA score (Table 3) for the species identified as being of highest risk.

4.0. Evaluation of the listed species

A list of invasive Ponto-Caspian species was generated (see Appendix 2). The number of papers published that contained the species name and the term 'invasive' was used as a proxy to indicate the invasiveness of this species in a global context; the logic being that the more papers published on a species relating to invasive issues then the more likely the species is to be high risk. The majority of papers published in relation to invasive Ponto-Caspian species originate from Europe (especially where Ponto-Caspian species are invasive), including GB, but there are also a large number of publications from North America; this is because the Great Lakes are becoming heavily impacted by invasive Ponto-Caspian species causing significant concern. It is, therefore, of no great surprise that species such as *Dressena spp.* appear as the first two species (Table 1) given their significant impact in both Europe and North America.

The inclusion of North American publications in this review is particularly important when considering modes and pathways of introduction that Ponto-Caspian species may follow when entering GB. North America and GB are similar in their geographical relation to Europe in the sense that they are separated by a marine barrier. The English Channel and the North Sea are much narrower than the Atlantic and easier to traverse. Therefore, it can be argued that if a species is able to invade across such an expanse of water as the Atlantic, then it is also likely to cross the English Channel or North Sea, especially considering the extensive network of freight vessels going between Northern Europe and GB.

4.1. Evaluating primary pathways of invasion

The main pathway of introduction of Ponto-Caspian species into North America has been identified as shipping, with ballast water and hull fouling as the mechanisms. Shipping is, therefore, likely to be a major proponent of the transfer of Ponto-Caspian species into GB. This could occur in two ways: ships moving directly from the Ponto-Caspian area or from countries in Northern Europe where Ponto-Caspian species have established. The latter is considered in the scientific literature to be the more likely. However, there is a significant lack of information concerning the use of ballast water in freshwater systems. Although it is recognised there is shipping activity between freshwater ports in Europe and the UK, the full extent and modes of use of ballast water by this sector needs more detailed investigation. Past and recent reviews of ballast water issues in GB ports have all concentrated on its use in the marine environment (Fisheries Research Services 1994; Directorate of Fisheries Research 1995; Enshaei & Mesbahi 2009). These studies have found that an estimation of 25.7 million tonnes of ballast water discharged into Scottish ports (n = 12) each year and an estimated 16.8 million tonnes discharged annually at ports in England and Wales. However, it is unclear what proportion of this is freshwater ballast, and what proportion of that is being discharged into either freshwater or brackish environments. Several species listed in the report as high risk (see Table 1) are likely to have entered GB by this mechanism (*Chelicorophium curvispinum* and *Cordylophora caspia*).

The review by Enshaei & Mesbahi (2009) came to the following conclusions:

1. In general, all UK ports collect and store information related to their operations. Apart from the Port of Sullom Voe, no other UK port collects information relevant to the ballasting and de-ballasting operations of ships arriving or leaving their ports;

2. Differences exist among ports in the methods of collection and storage of data. The use of *ad-hoc* and non-consistent databases are particularly unhelpful when aiming to arrive at an overall conclusion regarding port operations in UK;
3. The dead weight and gross tonnage of ships are closely correlated to their ballast water capacity, and this has been quantified for various ship types (tankers, liquid natural gas carriers, bulk carriers, container ships, general cargo, roll-on-roll-off, ferries, cruise liners and dredgers), and subsequently used for estimating the ballast water capacity of ships arriving at targeted UK ports;
4. In almost all UK ports, a clear and accurate correlation exists between ballast water operations and freight operations, and this yields a "ballast water proportion coefficient" with which to estimate the ballast water operations of ports for which information was not available (*i.e.* not provided or due to small size of operations);
5. Based on the total volume of discharged water, 20 UK ports were identified as the main recipients of ballast discharges, with the ports of Grimsby and Immingham at the top with >20 MMT of annual discharged ballast water;
6. In UK ports, 46% of ballast water operations related to the export of ballast water to non-UK destinations and 36% of operations led to a discharge in UK ports of ballast water from non-UK origins;
7. Based on information received, the origin of ballast water discharged at any port is as important as its volume, revealing that the majority of ballast water discharged into UK ports originates from Northern Europe (77%), North America (12%) and the Mediterranean (5%);
8. Of the ballast water taken up in UK ports, ≈ 81% is discharged at Northern European destinations, followed by North America (6%) and Central & South America (5%).

Overall, the review by Enshaei & Mesbahi (2009) indicates a high quantity of water exchange between GB and Northern Europe, where a large number of invasions by Ponto-Caspian species have been reported, if not well documented (*e.g.* Leppäkoski *et al.* 2002; Grabowski 2006; Kotta *et al.* 2006). The highest coefficients for estimating ballast water capacity of different types of ships were observed with tankers and bulk vessels (Enshaei & Mesbahi 2009), the latter being particularly likely to discharge ballast water into ports (T. McCollin, personal communication). Given the number of species able to survive the crossing of the Atlantic, it would seem likely that they would also be able to cross the English Channel and North Sea. The risks of undetected introduction are emphasized by the fact that there are currently no management or reporting requirements in place to deal with ballast water issues in GB (Enshaei & Mesbahi 2009). The International Maritime Organisation Ballast Water Management Convention (IMO 2004) still awaits ratification and there are difficulties in compliance that need to be overcome for the Convention to be effective (Wright 2012). The Convention applies to all vessels and ports, including those in freshwater. Through ratification and compliance to the Ballast Water Management Convention, this control mechanism could also apply to the prevention of invasion by freshwater/brackish organisms from the Ponto-Caspian regions via mainland Europe. However, it is not clear how the Convention will be applied in the freshwater environment.

Hull fouling has also been highlighted as potential mechanisms for the introduction of species into GB. Although it would seem unlikely for freshwater species to be able to tolerate the exposure to marine water for prolonged periods of time, it is still possible for some euryhaline species, such as the fish-hook water flea *Cercopagis pengoi* to survive.

Less likely, but still feasible are land and air transportation, in decreasing order of likelihood. Aquatic organisms are transported into GB via both land and air, and Ponto-Caspian species may be introduced inadvertently with consignments that contain an intentionally imported species or group of species. For example, the Asian cyprinid fish, topmouth gudgeon *Pseudorasbora parva*, has invaded most of Europe from infested fish farms in Romania, which had received the species accidentally in consignments of Asian carp species (Simon *et al.* 2011). Another fish species to have entered the UK as a consignment is the sunbleak *Leucaspius delineatus* (Pinder & Gozlan 2003). However, like topmouth gudgeon, the arrival of sunbleak occurred before (or during) the implementation of stricter controls on non-native freshwater fish imports (see Copp *et al.* 2005). Once these controls took effect, the only new records of fish in the wild have been of ornamental species (Zięba *et al.* 2010). Therefore, the most likely species to be introduced via air and land transport pathways are plants and invertebrates as contaminants of consignments transported into GB.

Recreational activities have also been associated with the introduction of Ponto-Caspian species. Owing to the methodology used in the present study (see Section 4.3. for further discussion) this mechanism of introduction was calculated as being of low risk. However, recreation within the aquatic environment has grown significantly in recent decades due to increasing affluence and people having more available time to pursue their interests or hobbies. Therefore, it is likely that the risks posed by the pathway as a mechanism for introduction will increase. There are few studies examining recreational activities as pathways, and this may have resulted in an under estimation of the risk it poses within this study. The lack of studies conducted on this sector may be due to a number of reasons: i) the low relative visibility of this sector in contrast to others, for example, commercial shipping, has led to a focus on other sectors, ii) a lack of data relating to recreational pathways, such as the number of anglers visiting Europe from GB, makes quantification of the pathways difficult and therefore problematic to study and iii) the previous lack of recognition of the importance of recreational activities to link locations.

4.2. Evaluating secondary pathways of invasion

Ships used in the transport of goods *via* European waterways have already been identified as the main pathway associated with intra-continental biological invasions (Panov *et al.* 2007), as well as for potential future invasions of GB (Parrott *et al.* 2009). Although canal systems in GB are used less frequently for goods transport than for leisure craft, the principle mechanism of this pathway (*i.e.* attachment to a vessel that eventually moves through the canal network) remains the same. There is a considerable network of navigable water systems in GB (as illustrated in Figure 21) that supports leisure craft traffic and to a lesser extent freight barges.

There is the risk that internal spread could occur through contamination of consignments produced in GB at locations (*e.g.* aquacultural facilities) infested by Ponto-Caspian species *via* another primary or secondary pathway. For example, new initiatives regarding the use and disposal of non-native aquatic plants (PlantWise initiative) recognise the risk of biological invasions via land transport and subsequent secondary pathways.

Of the potential secondary pathways, overland-transport of recreational boats and of angling gear (*e.g.* Zięba *et al.* 2010) is generating increasing concern. The use of fish as live baits has also been problematic, most notably for the translocation of species native to southern England but not northern England or Scotland (Winfield *et al.* 2010), but new regulations against this angling practice are reducing the risk of new invasions. A major facilitator of biological invasions is increasingly believed to be habitat degradation, both for freshwater and marine environments. For example, homogenized and disturbed environments (*e.g.* marine dredging and inland reservoir/waterway construction) are found to favour invaders over native species, and fixed structures (wind turbines, oil-rig platforms) provide substrata for foulants, acting as stepping-stones from one location to another. It is commonly recognised that diverse habitats are more robust and are less likely to be invaded than degenerated ecosystems.

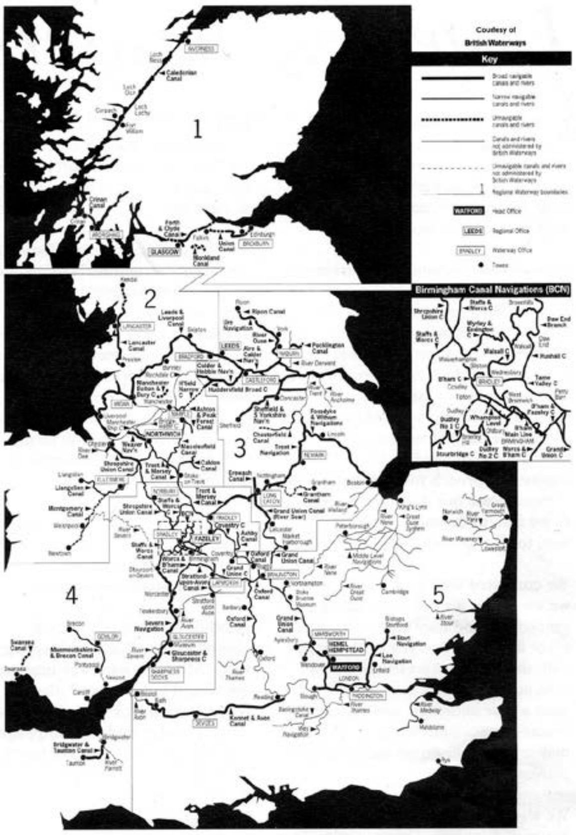

Figure 21. Map of navigable waterways in GB (from the Canal & River Trust, formerly British Waterways).

4.3. Evaluation of drivers and sectors

A major driver in Europe for all transport pathways is economic growth, with the selection of ports related to cost and proximity to major industrial and production centres. A recent report on the balance of container traffic amongst European ports (Newton *et al.* 2011) indicates that Europe's northern seaports remain the most economically and environmentally efficient gateways to large parts of central Europe, offering numerous economic and geographic advantages for shippers relative to those along the Mediterranean coast. Amongst the advantages cited are easy access to inland waterways, which act as high-capacity, low-cost corridors, the use of large container vessels to reduce shipping costs between Northern Europe ports and the Far East, and the ability of principal northern ports to link up overseas and inland transport routes, which environmentally reduces pollution loads (per tonne per km). Further investigation is required to full understand the extent of the use of ballast water in the freshwater environment and to full quantify the risk that this pathway may pose.

Other major drivers in Europe include the diversification of protein/food production, which in aquaculture focuses primarily on fishes, crustaceans and molluscs, as well as the distribution of these products. An increasingly important driver is recreation, which is characterised by economic importance of angling amenity, and to a lesser extent the pet industry (noting however that fishes are the most popular type of pet). These drivers relate directly to the food industry, leisure and ornamental/aquarist sectors of local and national economies.

The increasing use of bodies of water and waterways for leisure activities, combined with greater capacity to travel has created new links between previously unconnected areas. The relative lack of visibility of this sector may have resulted in a delayed recognition of the importance that this sector may provide in the potential control of future introductions of non-indigenous species.

5.0. Pathway management Action Plan (PAP) criteria

The **purpose** of this part of the review is to identify criteria for an action plan with which to address the pathways into GB most directly relevant to potential invasions by Ponto-Caspian species.

The three-stage hierarchical approach adopted by the Convention on Biological Diversity (CBD), and consequently in the GB Non-native Species Strategy (2008) involves (in decreasing order of importance): prevention, detection/surveillance and control/eradication. Prevention of non-native species invasions aims to reduce the risk and likelihood of adverse impacts, and hence the associated economic costs. Prevention is therefore the main **priority** of any PAP associated with non-native species. As an island, Great Britain has a geographical advantage over most other countries in that it can potential implement preventive measures to avert invasions, and therefore prevention is given the highest priority.

The main **objective** of a PAP is to reduce the risk of biological invasions by reducing the likelihood of entry to the country of organisms from the source region (in the present case, the Ponto-Caspian Region). There are many factors associated with each of the three main pathways (shipping, land transport, air transport) associated with Ponto-Caspian organisms, and therefore these need to be identified and assessed in detail, taking into account any onward dispersal via secondary pathways within GB *via* railways, roads and waterways, noting that due to the existence of the Channel Tunnel, secondary pathways include both rail and road transport into GB from other Continental locations. Given that economic growth and diversification are the main drivers, and that the latter of these involves an on-going search for new species to exploit in the various sectors (food production, recreation), an underlying objective of the PAP is horizon scanning for future potential invaders along the identified pathways.

Proposed actions for developing a Ponto-Caspian Pathway Action Plan

- Open a consultation process to involve stakeholders (*e.g.* shipping companies, port authorities, angling and aquatics trade associations, recreational boating clubs, etc.) at an early stage in developing a draft PAP into a comprehensive plan. The following list contains organisations suggested for initial involvement in further development of the PAP:

    - Commercial trade:
        - Department for Transport
        - Associated British Ports
        - Association of Inland Navigation Authorities
        - The Canal & River Trust/Scottish Canals (formerly British Waterways)
        - Department for Environment, Food and Rural Affairs
        - Environment Agency
        - Inland Waterways Association
        - Local Government Association
        - Maritime and Coastguard Agency
        - Port of London Authority/United Kingdom Major Ports Group
        - Commercial Boat Operators' Association
        - Centre for Environment, Fisheries and Aquaculture Science
        - Non-Native Species Secretariat
        - Royal Yacht Association
        - The Green Blue
        - British Sub-Aqua Club
        - British Canoe Union
        - The Angling Trust
        - Ornamental Aquatic Trade Association
- Develop a greater understanding of some of the pathways highlighted in this report and how they relate to the potential introduction and spread of non-indigenous species in GB. For example, i) a greater understanding of the use of ballast water in the freshwater environment, both between Europe and GB, and within GB; ii) increase our understanding of the extent of connectivity by recreational activities between freshwater locations in Europe and GB.
- Examine potential methods by which these pathways could be more effectively managed. For example, one method to increase control over the discharge of ballast water would be

to ratify and implement compliance to the IMO 2004 Ballast Water Management Convention, significantly reducing the risk of the introduction of Ponto-Caspian species such as those reviewed within the report. However, the extent of the application of this pathway to the freshwater environment is not realised yet.

- The quantification of the main pathways of introduction and spread will facilitate the production of a network analysis by which high risk nodes can be identified.
- Evaluate areas and/or habitats in GB that are most at risk from Ponto-Caspian species invasions. This will aid in the development of region specific Biosecurity Action Plans, such as for protected environments. This information can be combined with the network analysis to indicate where the species are most likely to be found, assisting in the development of more targeted surveillance schemes.
- Implement a public awareness and education programme to enhance the role of private individuals and groups in reducing the risk of secondary pathways (boating, angling). This is already being conducted for certain species present in GB (killer shrimp), however, this could be developed further.
- Further research is required to produce robust guidance on the implementation of biosecurity measures *e.g.* disinfectants and protocols, to help prevent introduction and translocation.
- Consider the use of existing or new legislative powers to reduce the risk of invasions via the pathways discussed. A review of potential legislation is beyond the scope of this document, but should be conducted to assess potential control measures.
- Many of the species recognised as high risk in this report are also capable of dispersal through natural means, it is therefore important that control and/or eradication methods are developed to prevent further spread of those already present and contingency plans for those not currently found in GB.
- Improved co-ordination and co-operation within and among relevant government agencies.

6.0. Conclusion

From the present report, 14 species originating from the Ponto-Caspian region have been identified as likely to pose the greatest potential risk of being introduced and/or dispersed within GB. This list comprises two species of mollusc, three fishes, eight crustaceans and one cnidarian. Five of these species are recognised as currently being present in GB waters, and therefore a priority for management, either containment or where feasible eradication. The potential for further dispersal of these species can be recognised in the pathways by which these species have already spread, both in GB and elsewhere. Hence, it is possible to rank the mechanisms of translocation (overland) and/or dispersal (along waterways). Shipping was identified as the most important pathway for new or repeated introductions, encompassing ballast water as the main mechanism followed by hull fouling – note that the risk of establishment increases with increasing propagule pressure (the frequency and size of introductions). The most important secondary pathway was inland boating, followed by unintentional stocking. Therefore, the main pathways of introduction to GB is via shipping, mainly commercial but also recreational, with the latter being a major secondary (dispersal) pathway. It is recommended that measures should be taken to reduce the possible risk posed by these pathways as soon as possible to prevent the introduction of novel species from the Ponto-Caspian region as well as to reduce or eliminate further, repeated introductions of species already present. To prevent the further spread of species already introduced into GB, containment measures are needed to prevent transfers of species by recreational users of water ways (*e.g.* boaters, anglers) to uninfected waters, both coastal and inland. Because many of these species are also capable of natural dispersal, control and/or eradication methods should be developed in conjunction with contingency plans for rapid response, using the mechanism for rapid response to non-native species infestations recently developed for, and coordinated by, the Non-native Species Secretariat. The main drivers behind the introduction of species were identified as global trade/economic growth and recreational, with the main driver of spread being recreational. A draft outline of a provisional PAP has been produced as an initial step to the development of a full plan, which would serve to assist in the reduction of invasion risks posed by the identified pathways and their associated mechanisms.

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Appendix 1.

Questions 20–31 from the 'Pathways' assessment module of NAPRA (2012)(see also Baker *et al*. 2008; Mumford *et al*. 2010). In questions 23–30, the response options are (very unlikely, unlikely, moderately likely, likely, very likely) and the confidence levels are (low, medium, high, very high).

Stage 2b: Pathways

20 - How many pathways are relevant to the potential entry of this organism?
Guidance: For organisms which are already present in Great Britain, only complete the entry section for current active pathways of new entry.

Response options: no active pathways, very few, few, moderate number, many, very many

21 - Please list relevant pathways through which the organism could enter (one per line).
Guidance: Give details about specific origins and end points of the pathways (where possible) in the comment box.

List of pathways: 1), 2), 3)…

22 - Please select the pathway:

Pathway 1: Pathway 2: Pathway 3:

Begin pathway-related questions

23 - How likely is it that the organism is strongly associated with the pathway at the point(s) of origin?

24 - How likely is it that large numbers of the organism will travel along this pathway from the point(s) of origin?

25 - How likely is the organism to survive during passage along the pathway?

26 - How likely is the organism to enter Great Britain undetected?

27 - How likely is the organism to multiply/increase in prevalence during transport /storage?
very unlikely

28 - How likely is the organism to survive existing management practices within the pathway (answer N/A for intentional introductions)?

29 - How likely is the organism to arrive during the months of the year most appropriate for establishment (if intentional introduction answer N/A)?

30 - How likely is the organism to be able to transfer from the pathway to a suitable habitat or host?

End of pathway-related questions

31 - Do other pathways need to be considered?

Appendix 2

The order number (No.) of each named Ponto-Caspian species (Latin name) and the number of hits on Google Scholar using the search terms ‘Ponto-Caspian’ + ‘invasive’.

No.	Species	Hits	No.	Species	Hits
1	*Dreissena polymorpha*	5150	51	*Paranais frici*	24
2	*Dreissena bugensis/rostriformis*	1429	52	*Chelicorophium sowinskyi*	23
3	*Huso huso*	1160	53	*Katamysis warpachowskyi*	22
4	*Neogobius melanostomus*	957	54	*Nitocra incerta*	22
5	*Dikerogammarus villosus*	640	55	*Cornigerius maeoticus maeoticus*	20
6	*Chelicorophium/Corophium curvispinum*	515	56	*Pontogammarus maeoticus*	20
7	*Cercopagis pengoi*	470	57	*Hypanis colorata*	18
8	*Hemimysis anomala*	365	58	*Monodacna colorata*	18
9	*Echinogammarus ischnus*	339	59	*Cyclops kolensis*	17
10	*Acipenser stellatus*	313	60	*Corophium mucronatum*	14
11	*Astacus leptodactylus*	307	61	*Gmelina costata*	14
12	*Acipenser ruthenus*	277	62	*Heterocope appendiculata*	14
13	*Neogobius fluviatilis*	262	63	*Paramysis baeri*	14
14	*Cordylophora caspia*	226	64	*Potamothrix heuscheri*	14
15	*Proterorhinus marmoratus*	214	65	*Caspihalacarus hyrcanus*	13
16	*Pontogammarus robustoides*	172	66	*Cornigerius lacustris*	12
17	*Dikerogammarus haemobaphes*	162	67	*Heterocope caspia*	12
18	*Limnomysis benedeni*	152	68	*Hypaniola kowalewskii*	12
19	*Acipenser gueldenstaedtii*	151	69	*Paramysis ullskyi*	12
20	*Neogobius kessleri*	133	70	*Pterocuma pectinata*	11
21	*Neogobius gymnotrachelus*	114	71	*Manayunkia caspica*	10
22	*Hypania invalida*	112	72	*Potamothrix bedoti*	10
23	*Lithoglyphus naticoides*	99	73	*Cystobranchus fasciatus*	9
24	*Chaetogammarus ischnus*	90	74	*Gmelina pusilla*	9
25	*Jaera istri*	86	75	*Hypanis pontica*	9
26	*Paramysis lacustris*	85	76	*Podonevadne camptonyx*	7
27	*Obesogammarus crassus*	78	77	*Theodoxus pallasi*	7
28	*Abramis sapa*	76	78	*Podonevadne trigona ovum*	6
29	*Daphnia cristata*	61	79	*Cornigerius bicornis*	5
30	*Clupeonella cultriventris*	57	80	*Podonevadne angusta*	5
31	*Obesogammarus obesus*	51	81	*Polyphemus exiguous*	5
32	*Maeotias marginata*	48	82	*Ectinosoma abrau*	4
33	*Acipenser gueldenstaedti*	44	83	*Evadne prolongata*	4
34	*Chaetogammarus warpachowskyi*	44	84	*Pontogammarus aralensis*	4
35	*Jaera sarsi*	44	85	*Pontogammarus subnudus*	4
36	*Maeotias inexspectata*	39	86	*Pterocuma rostrata*	4
37	*Potamothrix moldaviensis*	37	87	*Stenocuma cercaroides*	4
38	*Umbra krameri*	37	88	*Corophium chelicorne*	3
39	*Pontogammarus obesus*	36	89	*Stenogammarus carausui*	3
40	*Echinogammarus trichiatus*	35	90	*Lanceogammarus andrussovi*	2
41	*Corophium nobile*	32	91	*Obesogammarus aralensis*	2
42	*Dendrocoelum romanodanubiale*	32	92	*Amathilina cristata*	1
43	*Calanipeda aquaedulcis*	31	93	*Kuzmelina kusnetzowi*	1
44	*Dikerogammarus bispinosus*	31	94	*Limnodrillus newaensis*	1
45	*Paramysis intermedia*	31	95	*Mesomysis kowalevskii*	1
46	*Potamothrix vejdovskyi*	31	96	*Palaeodendrocoelum romanodanubialis*	1
47	*Clupeonella caspia*	29	97	*Psammoryctes deserticola*	1
48	*Corophium robustum*	28	98	*Schizopera bobrutzkyi*	1
49	*Caspiobdella fadejewi*	27	99	*Paraleptastacus spinicaudus triseta*	0
50	*Echinogammarus warpachowskyi*	25			

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