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Radboud University
Institute for Water and Wetland Research
Department of Animal Ecology and Physiology

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Authors:	F.P.L. Collas, J. Matthews, L. de Hoop, G. van der Velde & R.S.E.W. Leuven
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Project management:	Prof. dr. R.S.E.W. Leuven, Department of Animal Ecology and Physiology, Institute for Water and Wetland Research, Radboud University, Heyendaalseweg 135, 6525 AJ Nijmegen, The Netherlands, e-mail: r.leuven@science.ru.nl
Quality assurance:	Prof. dr. A.Y. Karatayev, Buffalo State University, Great Lakes Center, New York, USA and Ir. D.M. Soes, Bureau Waardenburg BV, Culemborg, The Netherlands
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Contents

Summary

An update of the risk assessment of the alien Chinese mystery snail (*Bellamya chinensis*) was carried out using recent scientific literature and data from field surveys in the European Union (EU). This species originates from Asia and it has recently been recorded as an introduced species in the Netherlands and Belgium. *B. chinensis* is currently also widely distributed in the USA and southern parts of Canada. This snail entered North America through the Asian food markets and as a result of the trade in aquarium animals, and was subsequently intentionally and unintentionally introduced to the wild. The secondary spread of *B. chinensis* is probably facilitated by professional and recreational watercraft, water birds, some mammals and aquarium keepers. Therefore, the NVWA requested an assessment of the ecological risk posed by this species to the Netherlands and the EU.

The present risk assessment is based on a detailed risk inventory of *B. chinensis*, which includes a science based overview of the current knowledge on taxonomy, habitat preference, introduction and dispersal mechanisms, current distribution, ecological impact, socio-economic impact and consequences for public health of the species. Subsequently, a team of experts applied this information to assess and classify the (potential) risks of spread, invasiveness and impact of *B. chinensis* in the EU, using the Harmonia^{+} and Invasive Species Environmental Impact Assessment (ISEIA) protocols. In addition, the report includes a risk assessment of *B. chinensis* with ISEIA that has been undertaken for the Netherlands. This risk assessment allows *B. chinesis* to be compared with other alien species that have been risk assessed for the Netherlands and management interventions to be prioritised.

B. chinensis has been recorded in Europe at 18 locations. In total, 17 of these locations are situated in the Netherlands, and one is located in Belgium. The discontinuous distribution in the Netherlands and Belgium suggests that the current spread may be the result of multiple individual introduction events, possibly as a result of intentional release from domestic aquaria and ponds and from open storage ponds associated with garden and pond centres. The introduced range of *B. chinensis* in the Netherlands and large portions of its introduced range in North America are climatically matched with large parts of Eurasia, including all EU countries. The introduced range of *B. chinensis* in the Netherlands and Belgium is within the Atlantic biogeographical region which is matched with Western Denmark, Germany, France, the far north of Spain, the United Kingdom and the Irish Republic, and some coastal regions in Norway. It is expected that *B. chinensis* will be able to establish in EU habitat type 3150 - natural eutrophic lakes with Magnopotamion or Hydrocharition type vegetation - which is widely available in EU countries. This habitat type in particular is expected to be at high risk of future colonisation by *B. chinensis.* Therefore, it is unlikely that climate, biogeography or habitat availability will restrict the further spread of *B. chinensis* within the EU. In the absence of

management measures, populations of *B. chinensis* in the Netherlands and Belgium may serve as sources of secondary spread within the EU (e.g., to France and Germany). The species may disperse within the EU attached to equipment used in the maintenance of waterways, ship hulls, with fishing equipment, via transportation by waterfowl and aquatic mammals, and through other limited natural dispersal such as water flow and crawling behaviour of the snail.

The impact of predation on established populations appears to have limited the abundance of the species locally. A discrepancy exists between scientific literature and risk assessments describing the perceived impact in North America. The scientific literature often indicates that the species has a limited impact, whereas two existing risk assessments classify the species' impact as high (New York Invasive Species Information, 2017; Jacobs & Keller, 2017). However, the risks associated with extremely high densities of the species are largely unknown and the lack of information on perceived impacts in scientific literature may be attributed to this lack of knowledge. Evidence of negative ecological effects is limited to mesocosm experiments from North America which suggests that *B. chinensis* may outcompete native snail species, increase water clarity and reduce algal biomass due to a high filtration rate, and increase the nitrogen-phosphorus balance (N:P ratio) and benthic-pelagic coupling. Densities of *B. chinensis* in the Eijsder Beemden were similar to the densities used in the mesocosm experiments implying that the aforementioned impacts may be expected. Quantitative information on the current and future economic losses and costs of *B. chinensis* is not available for the EU. *B. chinensis* currently displays a limited recorded distribution in the EU, and is of limited value to the trade in live animals as evidenced by the low number of retail outlets selling the species. Therefore, the economic impact of *B. chinensis* in the EU is likely to be negligible.

The expert team assigned a high risk classification to *B. chinensis* in the EU using the Harmonia^{+} protocol, and a medium risk using the ISEIA protocol.

B. chinensis is currently present in isolated populations in the EU. According to the BFIS list system used in conjunction with the ISEIA protocol, *B. chinensis* classifies as a **B1** species and qualifies for the **watch list**. It is unlikely that future climate change will result in limitation of the species.

The classification of *B. chinensis* by experts based on available knowledge resulted in the following risk scores according to the Harmonia^{+} protocol:

- Probability of introduction: **high** (Confidence: **high**);
- Probability of establishment: **high** (Confidence: **high**);
- Probability of spread: **high** (Confidence: **medium**);
- Probability of environmental impact: **medium** (Confidence: **medium**)
    - Effects on native species through predation, parasitism or herbivory: **medium** (Confidence: **low**);

- Effects on native species through competition: **medium** (Confidence: **medium**);
    - Effects on native species through interbreeding: **no / very low** (Confidence: **high**);
    - Effects on native species through hosting harmful parasites or pathogens: **very low** (Confidence: **medium**);
    - Effects on integrity of ecosystems by affecting abiotic properties: **medium** (Confidence: **medium**);
    - Effects on integrity of ecosystems by affecting abiotic properties: **medium** (Confidence: **medium**);
- Probability of effects on plant cultivation: **low** (Confidence: **high**);
- Probability of effects on domesticated animals and livestock: **low** (Confidence: **high**);
- Probability of effects on public health: **low** (Confidence: **high**);
- Probability of other effects: **high** (Confidence: **low**).

The literature search revealed knowledge gaps that reduced the certainty of elements of the analysis. Specifically, assessments of the potential effects of *B. chinensis* are mainly based on mesocosm experiments and there is a lack of research that analyses direct effects on the aquatic environment. This may be due to the generally low density of *B. chinensis* populations in its introduced range and the density dependent nature of effects. It is recommended that further research is carried out in order to address these knowledge gaps reduce uncertainties whilst carrying out future assessments.

Introduction

1.1 Background and problem statement

The Chinese mystery snail (*Bellamya chinensis*) is an alien species which originates from Asia (i.e., Southeast Asia to Japan and eastern Russia). This species entered the USA and Canada through the Asian food markets and trade in aquarium animals and it was subsequently intentionally and unintentionally introduced to the wild (Wood, 1892; Waltz, 2008; Haak, 2015; McAlpine et al., 2016; Ontario's Invading Species Awareness Program, 2017). The snail is widely distributed and established in the USA and Canada (southern part of Ontario and New Brunswick). It is considered to be an invasive alien species in this region (Minnesota Department of Natural Resources, 2017; Ontario's Invading Species Awareness Program, 2017). The secondary spread of *B. chinensis* is probably facilitated by professional and recreational watercraft (McAlpine et al., 2006).

The Chinese mystery snail was first recorded in the Netherlands in 2007 (Instituut voor Natuureducatie en Duurzaamheid, 2016). The largest population occurs in a floodplain lake in the Eijsder Beemden alongside the River Maas (Meuse) (Soes et al., 2011; Collas et al., 2017). Recently, the species has also been recorded in Belgium (Van den Neucker et al., 2017). Therefore, the Office of Risk Assessment and Research (BuRO) of the Netherlands Food and Product Safety Authority (NVWA) requested an assessment of the ecological risk posed by this species to the Netherlands and other member states of the European Union (EU).

During 2017, additional data was acquired on the spread, density and potential impacts of *B. chinensis* in the EU. In addition, more scientific papers were published with new information regarding *B. chinensis*. Due to the newly available information, a request was made by the Netherlands Food and Product Safety Authority (NVWA) to update the knowledge document and risk assessment of *B. chinensis* of Matthews et al. (2017a), taking this new information into account.

This updated analysis aims to assess the chances of repeated introductions (e.g., through one or more of the pathways described above), further spread and establishment of viable populations. In addition, the potential harm to native nature will be determined, such as the impact of the species on biodiversity, ecosystem functioning, ecosystem services and nature conservation goals. Moreover, further understanding is required of the socio-economic risks and the implications to public health. Therefore, it is important to present a description of the risks for Europe with a solid foundation that meets the criteria set by the EU regulation 1143/2014 for the prevention and management of invasive species eligible for consideration for addition to the Union list of invasive alien species.

The present report presents an updated risk assessment of *B. chinensis* for the EU. Additionally, appendix 1 presents an updated risk assessment for this species that has been undertaken for the Netherlands. The assessments are based on an extensive updated risk inventory. The analyses of available data and risk classifications of the species have been performed by a team of experts using the Harmonia$^+$ and Invasive Species Environmental Impact Assessment (ISEIA) protocols.

1.2 Research objectives

The aim of this study is to conduct a risk assessment of the alien *B. chinensis* for the EU that complies with the criteria for scientific information that will be required for decision making on listing IAS of EU concern described in Regulation 1143/2014. This report analyses the probability of introduction, spread and establishment in habitats with (high) conservation values (e.g., in N2000 areas) and (potential) ecological effects (including N2000 targets), socio-economic consequences and impact on public health.

1.3 Outline and coherence of the research

The coherence between various research activities and outcomes of the study are visualised in Figure 1.1.

The present chapter describes the problem statement, aim of the study and research questions in order to assess and classify the risks of *B. chinensis* in the European Union. Chapter 2 describes the results of the risk inventory, which includes a science based overview of the current knowledge on taxonomy, habitat preference, introduction and dispersal mechanisms, current distribution, ecological impact, socio-economic impact and consequences for public health of the species. A team of experts used the information provided in the risk inventory to assess and classify the (potential) risks of spread, invasiveness and impact of *B. chinensis* in the EU using the Harmonia$^+$ and ISEIA protocols (Branquart, 2009; Branquart et al., 2009; D'hondt et al., 2015; Vanderhoeven et al., 2015). Chapter 3 includes the results of these risk assessments and classifications. Moreover, in this chapter, the results of other available risk classifications are summarized and compared with the results of the risk assessments undertaken in this report. Uncertainties, relevant knowledge gaps and differential outcomes of risk assessments are discussed in chapter 4. Chapter 5 draws conclusions and gives recommendations for further research. Appendix 1 describes the methods used for the inventory (including literature review and data acquisition), and methods of assessment and classification of risks of introduction and spread of this species. Appendix 2 summarizes the results of the risk classification of *B. chinensis* for the Netherlands, using the ISEIA protocol. Finally, details on the outcomes of the peer review procedure for this report are summarized in appendix 3.

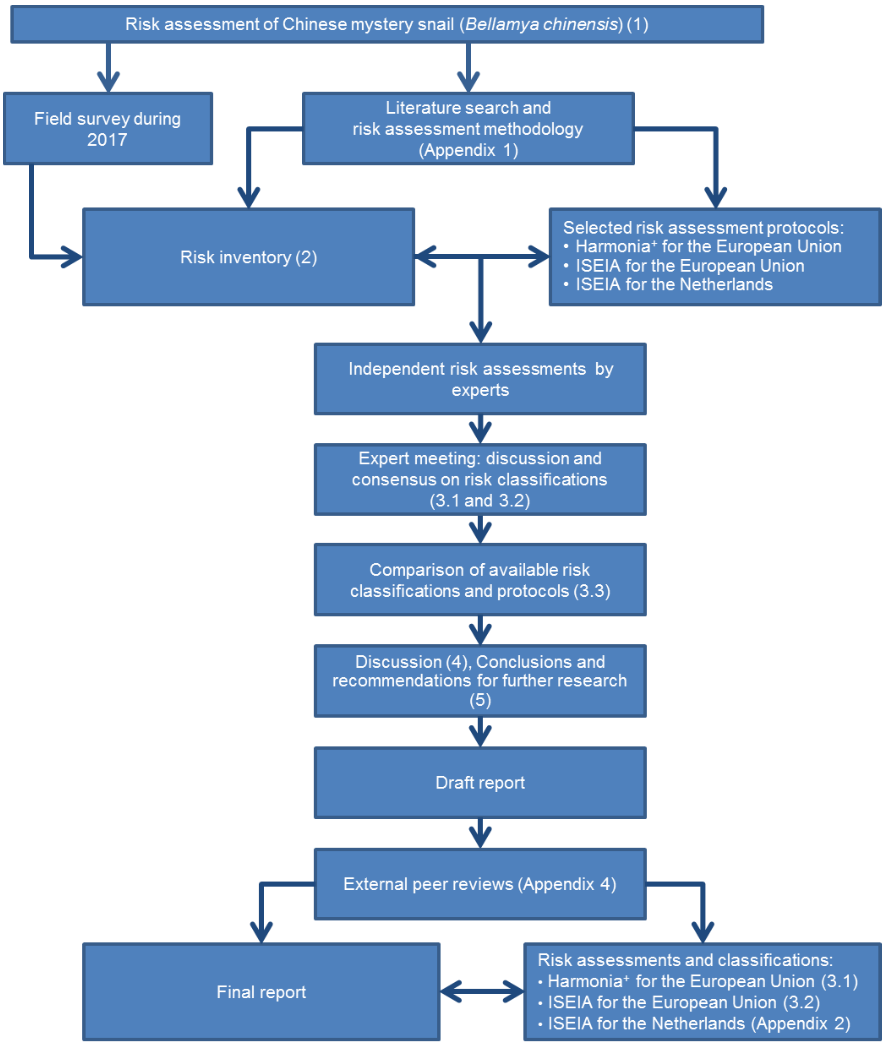


Figure 1.1: Flow chart visualising the coherence of various research activities (chapter numbers are indicated between brackets; ISEIA: Invasive Species Environmental Impact Assessment protocol).

2. Risk inventory

2.1 Species description

2.1.1 Nomenclature and taxonomic status

The nomenclature and taxonomic status of *B. chinensis* are summarized in Table 2.1. Based on the current body of knowledge the species can be regarded as a single taxonomic identity. However, there is some discussion over the correct nomenclature of the species and there are many scientific names in use. The two most common scientific names that are used for the Chinese mystery snail are *Bellamya chinensis* and *Cipangopaludina chinensis*.

Table 2.1: Nomenclature and taxonomic status of the Chinese mystery snail (*Bellamya chinensis*).

Scientific name: *Bellamya chinensis* (Gray, 1834)

Synonyms: *Cipangopaludina chinensis*, *Cipangopaludina malleata*, *Viviparus chinensis*, *Viviparus chinensis malleatus*, *Viviparus malleatus*, *Paludina malleata*, *Viviparus stelmaphora*

Taxonomic tree:
According to Smith (2000):

Domain: Eukaryota
Kingdom: Animalia
Phylum: Mollusca
Class: Gastropoda
Subclass: Caenogastropoda
Order: Architaenioglossa
Superfamily: Viviparoidea
Family: Viviparidae
Subfamily: Bellamyinae
Genus: *Bellamya*
Species: *Bellamya* (*Cipangopaludina*) *chinensis* (Gray, 1834)

Preferred Dutch name: Chinese moerasslak

Preferred English name: Chinese mystery snail

Other Dutch names: Not applicable

Other English names: Oriental mystery snail, trapdoor snail, apple snail, Asian apple snail, Chinese apple snail, rice snail

Native range: China, Taiwan, Korea, Japan, Indonesia, Burma, the Philippines, Asiatic Russia in the Amur region, Thailand and South Vietnam

In older literature the genus name *Cipangopaludina* is commonly used for both the Chinese and Japanese mystery snail species. Smith (2000) uses anatomical data, e.g., the absence of a gill filament typical of *Cipangopaludina*, to advocate the placement of the Chinese mystery snail into the genus *Bellamya*, arguing that *Cipangopaludina* is a subgenus. A recent phylogenetic assessment of the mitochondrial genome of *B. chinensis* places the species outside the genus *Bellamya* and within the genus *Cipangopaludina* (Wang et al., 2017), meaning that the nomenclature of the species remains under debate. However, in the current scientific literature the genus name *Bellamya* is most frequently used. Therefore, the scientific name *Bellamya chinensis* is applied throughout this risk analysis. Two subspecies or varieties of *B. chinensis* are recognized: *chinensis* and *laeta* (AIS, 2005; ITIS, 2009; GISD, 2017; M. Soes, pers. comm.).

2.1.2 Species characteristics

Morphological features

B. chinensis is a large freshwater snail (Fig. 2.1a). Adult snails feature an olive green, greenish brown, brown or reddish brown shell which is globose and has 6 to 7 whorls that are convex with a clear suture (Fig. 2.2a). The inner shell is white to pale blue and features a black lip. The full grown shell is robust and may reach 70 mm in length with a width to height ratio of 0.74-0.82. Collas et al. (2017) found similar width to height ratios for snails sampled at the Eijsder Beemden site in the Netherlands. The maximum wet weights of snails at the Eijsder Beemden and in the Veengoot near Zutphen were 45.0 and 29.2 g, respectively. Individuals grow throughout their entire lifespan, with females living 4 to 5 years and males living 3 to 4 years (Havel, 2011); as a result, females tend to be larger in size (Haak, 2015). Juvenile *B. chinensis* born at the beginning of May showed a growth rate of 0.1 mm.day^{-1} at a temperature of 21°C under laboratory conditions (This study), indicating a high growth rate in juveniles. A hard operculum covers the shell opening when closed. Juveniles exhibit a lightly coloured shell that contains grooves with 20 striae per mm between each groove whose and whorls that display a distinct cartilaginous ridge (carina) (Fig. 2.1b). Juveniles display a patterned periostracum consisting of 2 apical and 3 body whorl rows of hairs terminating with long hooks, distinct ridges and abundant other hairs with short hooks (Kipp et al., 2014).

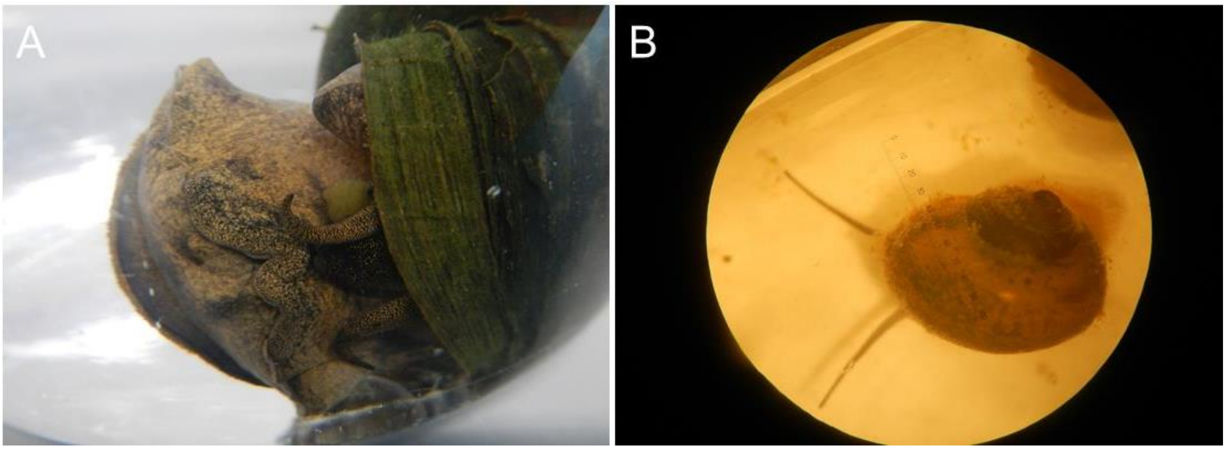


Figure 2.1: A) Live Chinese mystery snail (*Bellamya chinensis*) taken from the floodplain Eijsder Beemden, along the river Meuse in the Netherlands in 2015; **B)** Photo of juvenile female Chinese mystery snail (*Bellamya chinensis*) taken under the microscope (© Photos: F. Collas, 2015; 2017).

Differences between *B. chinensis* and visually similar species have been described in § 2.1.3. However, individuals may vary considerably, and distinct shell variations have been classified as morphotypes that may reflect variations in shell growth in response to differing environmental conditions (AIS, 2005; Prezant et al., 2006; Soes et al., 2011; Kipp et al., 2014; Fig. 2.2b).

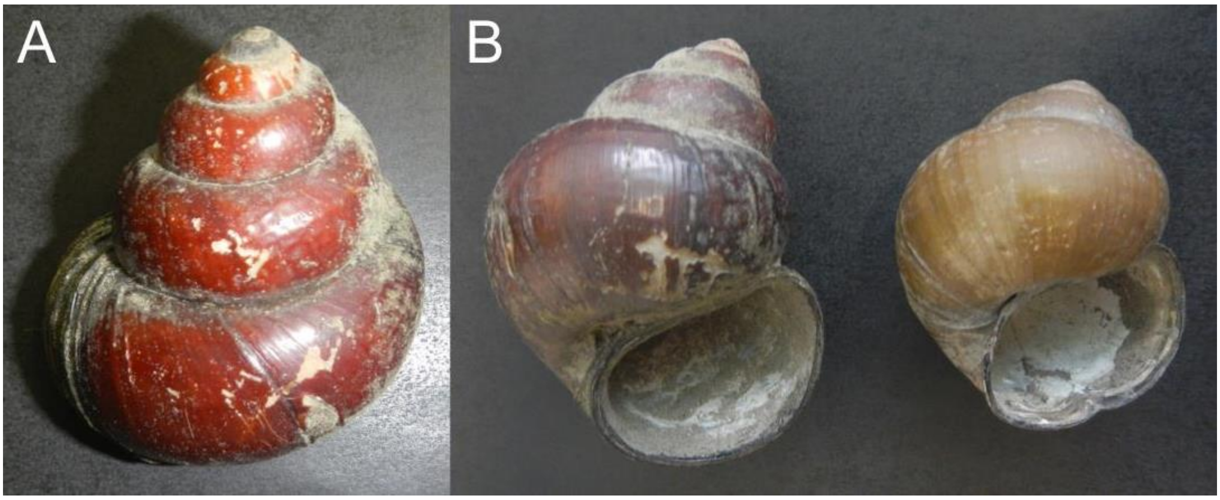


Figure 2.2: A) Adult shell displaying convex whorls and suture, and **B)** Different shell morphotypes of Chinese mystery snail (*Bellamya chinensis*) (© Photos: F. Collas, 2015).

Life cycle

All developmental stages from fertilised ova to 5 mm long fully shelled juveniles, occur simultaneously inside the uterine sac of females (Prezant et al., 2006). Observations from eastern North American show that after bearing young, females migrate to deeper water in the autumn where they overwinter (Stanczykowska et al., 1971; Jokinen, 1982; Jokinen, 1992; Kipp et al., 2014). Females give birth to juveniles that show allometric growth (Smith, 2000). Adults live for 3 to 5 years and grow to a maximum shell length of 65-70 mm (Jokinen, 1982; Havel, 2011; Kipp et al., 2014). Individuals grow throughout their entire lifespan (Haak, 2015). The maximum shell length discovered at the Eijsder Beemden site in the Netherlands was 63 mm and the maximum recorded shell length in the Veengoot near Zutphen was 51 mm (This study).

Reproduction

B. chinensis has separate sexes (Haak, 2015). Males can be identified by the presence of a modified right tentacle that functions as a penis (GISD, 2017). The species uses internal fertilization and is viviparous, the females giving birth to fully developed juveniles during the breeding season which extends from May to October (Jokinen, 1982; Dillon, 2000; Stephen et al., 2013; Kipp et al., 2014; Haak, 2015). Reproduction occurs in the year following the female's first winter (Jokinen, 1982). *B. chinensis* may also reproduce by parthenogenesis, a form of asexual reproduction, which is common in Viviparidae (Johnson, 1992; Mackie, 2000b).

Females contain upwards of 100 juveniles at varying stages of development and reportedly produce approximately 65 live offspring per year, up to 102 offspring per brood has been reported (Crabb, 1929; Jokinen, 1982; Keller et al., 2007; Stephen et al., 2013; Haak, 2015). Fecundity increases with snail size and females are able to reproduce throughout their lifetime (Stephen et al., 2013). At Eijsder Beemden (the Netherlands), *B. chinensis* had on average 33.2 (SD=28.6; n=9) developing young per snail with a maximum of 78 (Breedveld, 2015). Females bear more young in their 4th and 5th years than in other years (Jokinen, 1992). Environmental conditions influence reproduction rate in *B. chinensis*. Control females have been observed to produce significantly more juveniles than females in the presence of predatory crayfish. However, juveniles born to snails that were accompanied by crayfish were smaller, more variable in size and had a higher organic shell content (Prezant et al., 2006).

Diet

B. chinensis is a facultative filter-feeding detritivore (Olden et al., 2013). The diet of *B. chinensis* depends on its life stage. All life stages graze on periphyton (Dillon, 2000; Johnson et al., 2009; Haak, 2015), but larger individuals also filter-feed on planktonic algae (Dillon, 2000; Olden et al., 2013). A study of gut contents revealed that *B. chinensis* feeds non-selectively on inorganic and organic debris as well as epiphytic and benthic algae which are predominantly diatoms, mostly by scraping (Jokinen, 1982). Carbon stable isotope ratios of *B. chinensis* in North America suggest a clear preference for benthic resources over pelagic resources (Solomon et al., 2010). *B. chinensis* does not feed readily on plants making it popular with aquarists and water gardeners (Mohrman, 2007; Waltz, 2008; Soes et al., 2011). In eastern North America, the species probably gains most of its nutrition from diatoms (Jokinen, 1982). Snails sampled from three habitats revealed that larger and heavier individuals originated from areas with higher concentrations of algae and diatoms in the sediment than smaller snails (Calow, 1975; Jokinen, 1982; Waltz, 2008; Kipp et al., 2014).

Dispersal rate and distance

A natural dispersal rate of 0.1 km.year^{-1} was calculated during field surveys of shallow lakes in the floodplain Eijsder Beemden, the Netherlands (Collas et al., 2017). Evidence for a slow dispersal rate in Europe is supported by the distribution of records of *B. chinensis* in the Netherlands that suggests a number of separate introduction events rather than secondary dispersal from an initial point of introduction (Soes et al., 2016). No further information on dispersal rates specific to individual introductory pathways could be found during a review of available literature.

2.1.3 Differences with visually similar species

B. chinensis may be confused with three other snail species native to the EU and present in the Netherlands: the common river snail (*Viviparus viviparus* (L., 1758), Lister's river snail (*Viviparus contectus* (Millet, 1813) and the for the Netherlands alien Danube river snail (*Viviparus acerosus* (Bourguignat, 1862) (Soes et al., 2009). These species differ from *B. chinensis* as they possess shells with darker coloured bands and different patterns of hairs on the first whorls (Smith, 2000; Figure 2.3). Other species present in the EU, but not in the Netherlands, are *Viviparus ater* (De Cristofori & Jan, 1832), *Viviparus janinensis* (Mousson, 1859) and *Viviparus mamillatus* (Küster, 1852). *B. chinensis* is often confused with the Japanese mystery snail (*Bellamya japonica*) in its native range and in North America, but, at the time of writing, *B. japonica* has not been observed in Western Europe (Soes et al., 2011).

Figure 2.3: Three *Viviparus* species that currently occur in the Netherlands: 1) *Viviparus contectus*; 2) *Viviparus viviparus*, and 3) *Viviparus acerosus* (Adapted from Soes et al., 2009; © Photo: Peter Glöer).

2.1.4 Potential substitute species

Potential alternatives for *B. chinensis* in the aquarium and garden pond trade are *V. viviparus*, *V. contectus* and *V. acerosus*, snail species native to many countries in the European Union. All three species perform a similar function to *B. chinensis.* However, *V. acerosus* is alien to the Netherlands and currently it is unknown whether this species may become invasive in the Netherlands or other introduced areas in the EU. Due to its native status in a part of the EU, this species cannot be included in the Union list of IAS of EU concern. *V. acerosus* is a central European species native to the Danube drainage system that is often sold within the EU for use in aquaria and ponds and is a by-product of Hungarian aquaculture (Soes, pers. comm.). *V. viviparus* is also popular with aquarium hobbyists because of similar feeding and anatomical characteristics. However, *V. acerosus* is more widely sold because it is more suitable to pond habitats (Soes, pers. comm.).

Conclusion

B. chinensis is a large viviparous and facultative filter-feeding detritivorous freshwater snail. The species feeds non-selectively on inorganic and organic debris as well as epiphytic and benthic algae. Females give birth to fully developed juveniles during the breeding season which extends from May to October. *B. chinensis* had on average 33.2 (SD=28.6; n=9) developing young per snail with a maximum of 78 in the floodplain Eijsder Beemden (the Netherlands). Juveniles showed rapid growth during the initial month after their release. In view of the estimation of population size in the Eijsder Beemden, it is likely that the rate of reproduction of *B. chinensis* is high. A natural dispersal rate of 0.1 km.year^{-1} was calculated for lentic and slow flowing water bodies in its introduced range in Europe. However, downstream currents in lotic systems may transport snails in short time over long distances (see also § 3.2.1 and table 2.5).

2.2 Probability of introduction

Evidence relating to the probability of introduction of *B. chinensis* to Europe stems from introductions that have occurred in the Netherlands and Belgium. Currently, these are the only locations in the EU where *B. chinensis* has been recorded. An overview of the potential pathways of introduction of *B. chinensis* to the European Union is shown in table 2.2.

Table 2.2: Active (A) and potential (P) pathways and vectors for introduction of the Chinese mystery snail (*Bellamya chinensis*) to the European Union.

Category[a]	Subcategory[a]	A	P	Examples and relevant information	Reference
Escape from confinement	Pet/aquarium/terrarium species (including live food for such species)	X		Scattered records in the Netherlands suggest isolated introductions from aquaria and ponds	Soes et al. (2016); Collas et al. (2017)
Escape from confinement	Botanical garden/zoo/aquaria (excluding domestic aquaria)	X		Potential escape from open storage ponds owned by fish wholesalers in the Netherlands and Belgium	Soes et al. (2016); Van den Neucker et al. (2017)
Escape from confinement	Live food and live bait		X	Sold in Chinese food markets in North America	Wood (1892); Haak (2015)
Transport contaminant	Contaminant on plants (except parasites, species transported by host/vector)		X	Potential introduction as a passive attachment on ornamental lotus plants in North America	Havel (2011); Haak (2015)
Transport contaminant	Contaminant on animals (except parasites, species transported by host/vector)		X	Potential introduction along with goldfish that were added to a stream in an attempt to control mosquito larvae control in North America	Jokinen (1982); Waltz (2008)
Release in nature	Fishery in the wild (including game fishing)		X	Intentional release to provide extra food source in North America	Waltz (2008)

[a] Classification according to UNEP (2014)

2.2.1 Intentional pathways of introduction

B. chinensis is likely introduced via the aquatic ornamental trade pathway (trade in live animals and intentional disposal from aquaria). This is reflected by the scattered and isolated records of *B. chinensis* in the Netherlands and Belgium. Scattered distribution pattern are expected to be the result of multiple and independent new introduction events (Kroiss, 2005; Strecker et al., 2011; Soes et al., 2016; Collas et al., 2017; Figure 2.4). Evidence for this pathway is further supported by the fact that *B. chinensis* is sold in pond and garden centres in the Netherlands and Belgium

(Soes et al., 2016; Van den Neucker et al., 2017; F.P.L. Collas, personal observation) and online on a single website (www.snailcorner.com). Soes et al. (2016) and Van den Neucker et al. (2017) describe open rearing and stocking ponds at garden and pond centres as a potential point of release of *B. chinensis* to the environment. Species may have been released during maintenance of the ponds (Van den Neucker et al., 2017). Van den Neucker et al. (2017) note that *B. chinensis* are offered for sale at a garden centre in Belgium costing €1.25 per snail, and are therefore also accessible to aquarium and pond hobbyists there. Patoka et al. (2017) note that *B. chinensis* is not traded in the Czech Republic.

Waltz (2008) states that deliberate release from aquariums is a potential vector for the introduction of *B. chinensis*. McAlpine et al. (2016) suggest that boat launches may be convenient locations for aquarium hobbyists to dump the contents of aquaria containing the snail. *B. chinensis* is sold to hobbyists as a species that eats algae also by filter feeding but not water plants and therefore maintains the clarity of the water without damaging the water plants (Soes et al., 2016). Moreover, *B. chinensis* closes its trapdoor, or operculum in the presence of poor water quality to survive, a characteristic that is appreciated by aquarium hobbyists as it serves as an indication of poor aquarium water quality (Waltz, 2008).

Original introductions to North America are thought to be related to the import of *B. chinensis* from the Far East as a live food source and selling in food markets in San Francisco on the west coast (Wood, 1892; Haak, 2015).

2.2.2 Unintentional pathways of introduction

Van den Neucker et al. (2017) speculated that the most probable source of introduction of the Belgian population in the river Laak was a nearby garden centre that specializes in ornamental fish and plants for garden ponds. Van den Neucker et al. (2017) noted that large quantities of *B. chinensis* were offered for sale in this garden centre. Several rearing and stocking ponds owned by the garden centre were situated next to the river Laak. Van den Neucker et al. (2017) suggested that *B. chinensis* may have been unintentionally introduced into the river Laak during maintenance of the ponds and aquaria. It is unlikely that the Belgian record is the result of natural spread from the Netherlands, as it is located 52 km from the nearest record in the Netherlands and is isolated from other known locations.

There is some evidence to suggest that the species may have been introduced unintentionally to North America. *B. chinensis* was first recorded in the state Massachusetts on the east coast of the USA in September of 1917 (Waltz, 2008). It should be noted that this was not the first ever record for North America. This introduction may have occurred through bio-contamination where *B. chinensis* was introduced along with the alien goldfish (*Carassius auratus* (Linnaeus, 1758)) that were added to a stream in an attempt to control mosquito larvae control (Jokinen 1982; Waltz, 2008). *B. chinensis* also likely spreads through the human-mediated movement of aquatic plants and the species may have been transported to the USA as a passive attachment on ornamental lotus plants (Smith, 1995 in Martin, 1999; Havel, 2011; Haak, 2015). Judgements regarding the potential for unintentional introduction of *B. chinensis* to the EU should take into account the distance between the EU nations and the native Asian and alien North American ranges of *B. chinensis* (§ 2.3.1).

2.2.3 Commodities associated with introduction

In the Netherlands, there is a striking similarity between the introduction and establishment of *B. chinensis* and two North American crayfish species which are also sold to the public and have now become established in Dutch open water viz. the viril crayfish (*Orconectes virilis* (Hagen, 1870)), first observed in 2004 in the Netherlands, although probably already widespread at that time; and the white river crayfish (*Procambarus* cf. *acutus* (Girard, 1852)), which was first recorded in 2002 and was subsequently able to expand over large parts of the Vijfheerenlanden and the Alblasserwaard (Koese & Soes, 2011). *B. chinensis* is found together with *O. virilis* at Vinkeveen and at Boven-Hardinxveld with both crayfish species. Since a number of fish wholesalers that use outdoor storage ponds are located in areas where these three species have been simultaneously recorded in the Netherlands, it has been suggested that the three species were imported jointly from North America and introduced via the same pathway (Soes et al., 2016).

2.2.4 Pathway origins and end points

The origin of *B. chinensis* in the Netherlands is likely North America (possibly simultaneous introduction with *O. virilis* and/or *P.* cf. *acutus*; see § 2.2.3). In general, few species have been imported from Japan in the past, although more recently species have started to be imported from Asia (P. Veenvliet pers. comm. in Soes et al., 2016). However, CO1 barcoding data indicates that *B. chinensis* collected in the Netherlands are identical with specimens collected in Japan (Soes et al., 2016). Soes et al. (2016) suggest an introduction pathway from Japan via North America to Western Europe.

2.2.5 Propagule pressure per pathway

In general, the primary introduction pathways of aquatic gastropods are the aquarium, pet, and food trades (Padilla & Williams, 2004). Little information exists describing pathways for invertebrates, including freshwater molluscs, in the ornamental pet trade, despite a noticeably increased interest from hobbyists in recent years (Ng et al., 2016).

Conclusion

Pathways of introductions of *B. chinensis* to the EU have, where recorded, been associated with the trade in live plants and animals. It has been suggested that the species has been introduced in the Netherlands and Belgium from open storage ponds associated with garden and pond centres. The discontinuous distribution in the

Netherlands suggests that the current distribution may be the result of multiple individual introduction events, possibly as a result of intentional and unintentional release from aquaria and ponds.

2.3 Probability of establishment

2.3.1 Current global distribution

Native range in Asia

B. chinensis' native range is said to include China, Taiwan, Korea, Japan, Indonesia, Burma, the Philippines, Asiatic Russia in the Amur region, Thailand, South Vietnam and Bhutan (Pace, 1973; Chiu et al., 2002; GISD, 2017; Yeshi et al., 2017). However, the potential for misidentification of *B. chinensis* in its native range is high due to the presence of visually similar species. This may reduce the certainty of its native range. According to Lu et al. (2014), the native distribution of *B. chinensis* is often reported as China, Taiwan, Korea and Japan (Soes et al., 2016).

Introduced range

European Union

To date, *B. chinensis* has been recorded at 18 locations in the EU, 17 situated in the Netherlands and one in Belgium (Soes et al., 2011; Collas et al., 2017; Van den Neucker et al., 2017; Waarneming.nl; Figure 2.4). For these EU member states, the year of first record and last observation or current status are summarized in Table 2.3.

Table 2.3: First and last observation of the Chinese mystery snail (*Bellamya chinensis*) in EU member states and present populations.

Member state	First observation	Reference	Last observation	Reference	Current population status
The Netherlands	2007	Instituut voor Natuureducatie en Duurzaamheid (2016)	2017	This study	17 Isolated populations; certainly established for several years at four locations
Belgium	2016	Van den Neucker et al. (2017)	2016	Van den Neucker et al. (2017)	Single established population

North America

B. chinensis was introduced in North America at the end of the 19th century. It is established as alien in Canada and the USA. The species was first recorded in North America in the 1890s (Wood, 1892; Haak, 2015). Since then, *B. chinensis* has established populations in 21 of the 34 states in the USA (Jokinen, 1982; Kipp et al., 2014; Haak, 2015). Records in the USA are concentrated in the north-eastern part of the country, but the species has been observed as far south as Cape Coral in Florida and Hawaii (Jokinen, 1982; Kipp et al., 2014). The species has reached high densities in Oregon, North America where management measures resulted in the deaths of approximately 27,000 snails in two ponds (Freeman, 2010). *B. chinensis* has also been recorded in the southern areas of Canada (Havel, 2011), though the species is more abundant in the southernmost parts of its alien range (Solomon et al., 2010). The species has been recorded in Ontario (F.W. Schueler pers. obs. in McAlpine et al., 2016), Québec (Clarke 1981; Tornimbeni et al., 2013), British Columbia (Clarke, 1981), Nova Scotia, New Brunswick and a single location in Newfoundland (McAlpine et al., 2016).

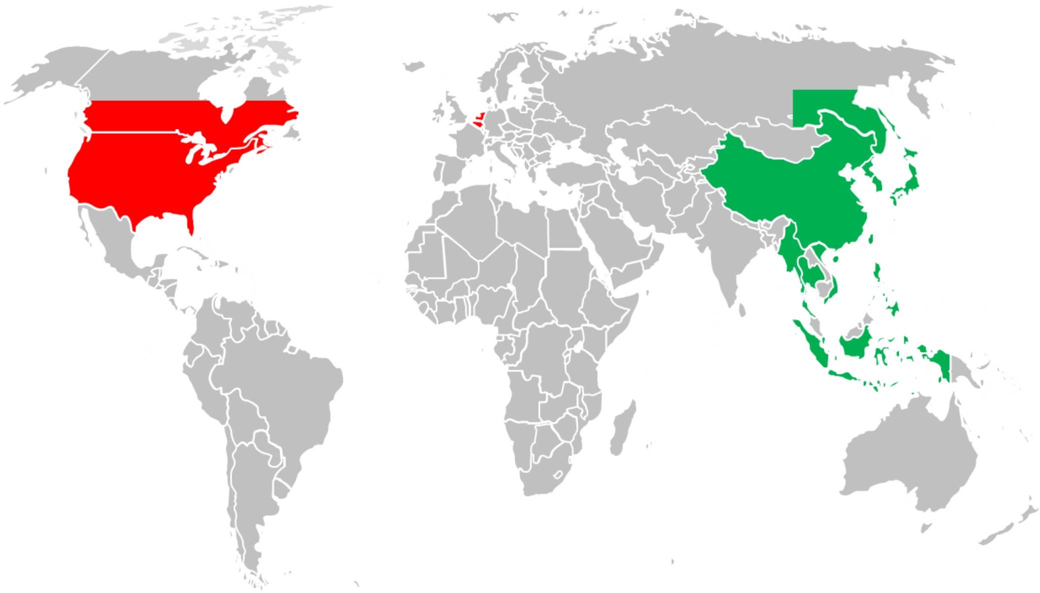


Figure 2.4: Global distribution of Chinese mystery snail (*Bellamya chinensis*) in the native range (green) and introduced range (red) (Sources: Table 2.3 and § 2.3.1 and 2.3.2). (Note that the geographical distribution is visualised at nation state level, however, occurrence in some nation states may be restricted to one or few isolated populations; see text for a detailed description of the occurrence in the USA and southern Canada).

Conclusion

The native range of *B. chinensis* is said to extend to Asia (China, Taiwan, Korea, Japan, Indonesia, Burma, the Philippines, Asiatic Russia in the Amur region, Thailand, Vietnam and Bhutan). However, the potential for misidentification of this species is high, reducing the certainty of this list of native states and areas. According to Lu et al. (2014), the native distribution of *B. chinensis* is often reported as China, Taiwan, Korea and Japan. The current global introduced range, where the species occurs, consists of the USA, Canada, the Netherlands and Belgium.

2.3.2 Current distribution in the EU and neighbouring areas

To date, there are 18 records of *B. chinensis* in the EU, 17 situated in the Netherlands, and one recently recorded established population in Belgium (Figure 2.5; Soes et al., 2011; Collas et al., 2017; Van den Neucker et al., 2017).

The first record was made on the 11th of March, 2007 at the Eijsder Beemden (Instituut voor Natuureducatie en Duurzaamheid, 2016) and this population was still present in 2017 (F. Collas, personal observation). Of the 17 Dutch locations, three sites harbour large persistent populations: lakes in floodplain Eijsder Beemden along the river Meuse, a small river near Giessen-Oudekerk and the stream Veengoot near Zutphen (Soes et al., 2016; Collas et al., 2017). Other locations are ditches near Amsterdam, Almere, Boukoul, s-Gravenzande, Vinkeveen, Neder-Hardinxveld and Boven-Hardinxveld, streams near Maasbree and Nij Beets and a lake near Leidschendam (Figure 2.5; Table 2.4).

The only record made in Belgium is in the small lowland river Laak. Here, 20 juvenile and adult snails were found which may not reflect the actual population size as the murkiness of the water prevented a thorough visual survey (Van den Neucker et al., 2017).

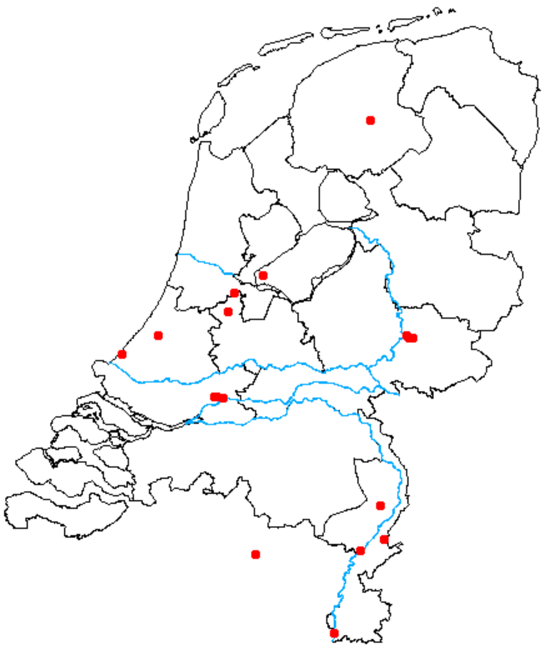


Figure 2.5: Observation of the Chinese mystery snail (*Bellamya chinensis*) at known locations in the Netherlands and Belgium. Data obtained from Collas et al. (2017) and Van den Neucker et al. (2017).

Densities of the species in the Eijsder Beemden ranged between 1.28 and 9.02 ind.m^{-2}, on average. The maximum density recorded in the previous sentence is high compared to densities reported for North America. There are indications that densities in the Veengoot near Zutphen are also high, although mark-recapture studies are necessary to derive accurate population sizes.

Genetic diversity

No information on the genetic diversity of populations of *B. chinensis* in its European non-native range could be found during a search of available literature. However, CO1 barcoding data indicates that *B. chinensis* collected in the Netherlands are identical with specimens collected in Japan (Soes et al., 2016). Forty-one DNA and

RNA sequences are available for *B. chinensis* in Genbank, potentially allowing for eDNA sampling of locations to test for the presence of the species.

Table 2.4: Occurrences of the Chinese mystery snail (*Bellamya chinensis*) in the Netherlands.

Location	Water-type	Latitude	Longitude	Date	Current population	Observer	Source
Eijsder Beemden	Floodplain ponds	50.7931	5.6970	11-03-2007	living adults and juveniles	J. Piters	Piters (2007)
Amsterdam	Ditch	52.2949	4.9833	07-08-2008	living adult	A. Klink	Soes et al. (2016)
's-Graven-zande	Ditch	52.0144	4.1731	02-11-2009	living adults and juveniles	S. Vlaar-dingerbroek	Soes et al. (2010)
Vinkeveen	Ditch	52.2120	4.9362	28-06-2010	living adults	W. Teunissen	Soes et al. (2010)
Maasbree	Stream	51.3581	6.0193	05-07-2010	living adults and juveniles	A. Klink	Soes et al. (2016)
Boven Hardinxveld	Polder water	51.8216	4.8941	2012	living adults	A. Blokland	Soes et al. (2016)
Giesen-Oudekerk	River	51.8394	4.8648	2013	living adults	A. Blokland	Soes et al. (2016)
Leidschendam	Lake	52.1032	4.4344	25-10-2013	empty shell	R. Geling	Waarneming.nl
Neder-Hardinxveld	River	51.8307	4.8383	16-07-2014	living adult	D.M. Soes	Soes et al. (2016)
Zutphen	Stream	52.1001	6.2197	14-11-2014	living adults	M. Klemann	Soes et al. (2016)
Wessem	Stream	51.1625	5.8782	14-10-2015	living adult	E. Binnendijk	Soes et al. (2016)
Vorden	Channel	52.0960	6.2610	25-09-2016	living adult	M. Klemann	Waarneming.nl
Wichmond	Stream	52.0940	6.2766	11-05-2017	living adult	R. Heusink-veld	Macrofauna-nieuwsmail 140 (2018)
Almere	Ditch	52.3745	5.1844	29-05-2017	living adult	J. Mulder	Macrofauna-nieuwsmail 140 (2018)
Boukoul	Ditch	51.2080	6.0470	22-07-2017	living adult	G. Majoor	Waarneming.nl
Vierakker	Stream	52.0950	6.2440	20-08-2017	empty shell and living juveniles	M. Klemann	Waarneming.nl
Nij Beets	Stream	53.0570	5.9740	02-09-2017	living juvenile	B. Koese	Waarneming.nl

Conclusion

The alien *B. chinensis* has been recorded in Europe at 18 locations. In total, 17 of these locations are situated in the Netherlands, with large persistent populations in floodplain lakes along the river Meuse, a small river near Giessen-Oudekerk, and a stream near Zutphen. One location is situated in Belgium.

2.3.3 Habitat description and physico-chemical conditions

B. chinensis is found in stagnant or slow flowing waters featuring a soft and nutrient rich substrate in its European introduced range (Soes et al., 2016). In the

Netherlands, the species appears to favour substrates that are visibly rich in algae and organic matter (Figure 2.6).

Figure 2.6: Chinese mystery snail (*Bellamya chinensis*) habitats in floodplain Eijsder Beemden along the river Meuse, the Netherlands (© Photos: F. Collas, 2015).

B. chinensis usually occurs in large lentic or slow-moving lotic systems with soft, muddy or silty bottoms (Schmeck, 1942; Stanczykowska et al., 1971; Jokinen, 1982; Distler, 2003; Waltz, 2008; Kipp et al., 2014; GISD, 2017). Such habitats include ponds, lakes, rivers, streams, roadside ditches, irrigation canals and rice paddies (Pace, 1973; Jokinen, 1982; AIS, 2005). *B. chinensi*s has been found in similar habitats in the European Union (Appendix 4). However, the species was also recorded at a location near Zutphen where flow velocities of 0.54 m/s were observed (Table 2.5; Waarneming.nl, 2017). In the Netherlands, the species is present in highly nutrient rich waters with high algal abundance and muddy substrates (Soes et al. pers. comm.).

Waltz (2008) reports that the habitat types of *B. chinensis* are characterized by a high frequency of occurrence of rooted aquatic vegetation. North American populations have been documented living on artificial riprap as well as on submerged vegetation (Chaine et al., 2012; Haak, 2015). Adult snails are often found on the surface or (partially) buried under mud or silt. Juveniles often live in crevices or under rocks (Prezant et al., 2006; GISD, 2017). *B. chinensis* favours waterbodies with relatively high non-anthropogenic turbidity, such as lakes with high densities of phytoplankton (Solomon et al., 2010; Haak, 2015). The species has been found on top of dense aquatic vegetation in the Eijsder Beemden.

In the EU, *B. chinensis* may occur in endangered and protected nature areas (e.g., Natura 2000 sites) which are classified according to the EU Habitat Directive

(European Commission, 2013, European Environment Agency, 2016) into fresh water habitats:

- Standing water (HT3100): HT3150, natural eutrophic lakes with Magnopotamion or Hydrocharition - type vegetation;
- Running water (HT3200).

Habitat type HT3150 overlaps with the EUNIS classification C1.3 – permanent eutrophic lakes, ponds and pools – which is expected to provide the most suitable conditions for *B. chinensis* establishment.

Adaptability to physiological conditions facilitating species establishment
Available data on the physico-chemical conditions at which *B. chinensis* was found are summarized in Table 2.5. *B. chinensis* is a species from a temperate climate region and has a wide temperature tolerance: lower limit of 0°C and upper limit of 30°C (Karatayev et al., 2009; Basten et al., 2012; Kipp et al., 2014; Haak, 2015; GISD, 2017; own data). Natural populations protect themselves from cold winter conditions by burrowing into the substrate and emerging in the spring (Jokinen, 1982). At Eijsder Beemden, the Netherlands, *B. chinensis* emerged after a week of warm weather at the end of April 2017 and within a few days the snails were producing young (F. Collas, personal observation). According to Wong et al. (unpublished data in Haak, 2015), adult individuals of *B. chinensis* are able to survive acute heating to approximately 45°C. This data is brought into question because the duration of exposure was not quantified in the original reference material. Moreover, the acuteness and short-term nature of the exposure may not reflect natural conditions, and the data was obtained during one laboratory experiment, the results of which were not peer reviewed. However, the 45°C figure emphasises the ability of this species to resist extreme temperature conditions. This suggests that high water temperature will not limit its further spread because the water temperatures of most waterbodies within the EU do not exceed 45°C. *B. chinensis* survives freezing water temperatures for over 24 hours (Wong et al., unpublished data in Haak, 2015). The species is highly resistant to desiccation and has survived air exposure for more than nine weeks (Havel, 2011; Unstad et al., 2013; Havel et al., 2014).

During an eight week laboratory experiment examining reproduction, snails were exposed to water at temperatures of 12, 20 and 27°C (Haak, 2015). No juveniles were born at 12°C, 212 juveniles were birthed at 20°C, and 418 juveniles were birthed at 27°C (Haak, 2015). However, there was no significant difference in the mean number of juveniles birthed per live adult between 20°C and 27°C. Snails at 27°C showed the lowest survival rates but allocated more energy to reproduction than growth (Haak, 2015). Fecundity at 27°C peaked in the first week but subsequently declined to zero. Surviving females exposed to 27°C contained only two juveniles at the end of the experiment, suggesting that snails were stressed at 27°C. In contrast snails exposed to 20°C contained significantly higher numbers of juveniles than those exposed to 27°C suggesting that reproduction was likely to continue at 20°C. At 12°C, more energy was allocated to maintenance and growth than to offspring production. Adult individuals of *B. chinensis* survived well at 12°C but did not reproduce (Haak, 2015).

In its non-native range, *B. chinensis* has been found in habitats featuring depths of 0.15 - 4.5 m, flow rates of 0.0 - 0.54 m/s, pHs of between 6.5 and 9.3, conductivities of 63 to 1705 µS/cm, total dissolved solids (TDS) of 71 to 850 ppm, salinities of 0.1 to 0.86 ppt; and concentrations of calcium between 5 - 97 ppm, magnesium 13 - 31 ppm, oxygen 7 - 11 ppm and sodium 2 - 49 ppm (Jokinen, 1982; Jokinen, 1992; Basten et al., 2012; Haak, 2015; Collas et al., 2017; This study; Table 2.5). In its native range, *B. chinensis* has been reported to occur in waters with pHs of 4.0 - 9.0, conductivities of < 30 - > 195 µS/cm, calcium concentrations of < 2 - > 20 ppm, water temperatures of 8.1 - 26.8°C and flow rates of 0.17 - 0.31 m/s (Chiu et al., 2002; Pan et al., 2017; Table 2.5). The species has been observed to tolerate stagnant conditions near septic tanks (Perron & Probert, 1973).

Table 2.5: Ranges of physico-chemical conditions measured in water bodies inhabited by the Chinese mystery snail (*Bellamya chinensis*).

Parameter	Laboratory	Introduced range	Native range
pH	NA	6.5 - 9.3[1,7,8]	4.0 - 9.0[6,10]
Conductivity (µS/cm)	NA	63 - 1705[2,7]	< 30 - > 195[6]
Total dissolved solids (ppm)	NA	71 - 850[7]	NA
Calcium (ppm)	NA	5.0 - 97[1]	< 2 - > 20[6]
Magnesium (ppm)	NA	13 - 31[1]	NA
Sodium (ppm)	NA	2 - 49[1]	NA
Oxygen (ppm)	NA	7 - 11[1]	NA
Oxygen (mg/l)	NA	4.4 - 11.3[7]	NA
Water temperature survival (°C)	45[9]	0 - 30[3,7,8]	8.1 - 26.8[10]
Water temperature - no reproduction observed (°C)	≤12[4]	NA	NA
Salinity (ppt)	NA	0.1 - 0.86[5,7]	NA
Flow rate (m/s)	NA	0.0 - 0.54[5,7]	0.17 - 0.31[10]
Secchi depth (m)	NA	0.3 – 6.5[7,11]	NA
Depth (m)	NA	0.15 - 4.50[1,7,12]	NA

[1]Jokinen (1982); [2]Jokinen (1982), Breedveld (2015), Collas et al. (2017); [3]Karatayev et al. (2009), Breedveld (2015), Haak (2015), Collas et al. (2017); [4]Haak (2015); [5]Breedveld (2015), Collas et al. (2017); [6]Chiu et al. (2002); [7]Own data; [8]Basten et al. (2012); [9]Acute heating, duration of exposure not recorded (Wong et al. unpublished data in Haak, 2015); [10]Pan et al. (2017); [11]Twardochleb and Olden (2016); [12]Wisconsin Lake Partnership (2014); NA: not available.

Facilitation of its establishment by capacity to spread and parthenogenesis

B. chinensis has a low natural dispersal rate (§ 2.1.2). This low natural dispersal rate is supported by Soes et al. (2016) who suggests that the natural ability of *B. chinensis* to spread in the Netherlands appears limited due to the isolated nature of current records. The wide-ranging distribution of the species in North America is likely caused by new introductions and not by natural spread (Kroiss, 2005). However, a single gravid female is capable of founding a population which may facilitate dispersal (Waltz, 2008). Moreover, parthenogenesis is common in Viviparidae, which may facilitate dispersal and establishment following colonization by one female individual. However, it is not clear if *B. chinensis* is able to reproduce in this way (Mackie, 2000b).

Population establishment and genetic diversity

Despite potential founder effects, population establishment has successfully occurred outside *B. chinensis*' native range in the Netherlands, Belgium, and extensively in the northern half of the USA and southernmost areas of Canada.

Effects on establishment through competition or predation with other species

Soes et al. (2016) suggested that in the Netherlands, observed predation of mammals and crayfish, and assumed predation by fish and birds may regulate population expansion of *B. chinensis*.

Figure 2.7: Evidence of predation on the Chinese mystery snail (*Bellamya chinensis*) in the Netherlands. (© Photo: F. Collas, 2015).

Adult shells have been found in Amsterdam showing evidence of assaults by rodents, presumably alien brown rats (*Rattus norvegicus* (Berkenhout, 1759)) (Figure 2.7). In the floodplain Eijsder Beemden along the river Meuse a relative lack of young *B. chinensis* suggests that birds, fish and/or other animals may selectively predate on juveniles before adult snails. Moreover, native black crows (*Corvus corone* (Linnaeus, 1758)), are able to break open the shells of *B. chinensis* (Soes et al., 2016). Alien crayfish species present in the Netherlands may also predate on smaller *B. chinensis* individuals as it is known they feed on other freshwater snail species (Koese & Soes, 2011; Soes et al., 2016). Predation by alien crayfish increased during the summer of 2017 due to extremely low water levels (pers. comm. J. Piters). Fish species that may predate on *B. chinensis* in its European introduced range are the native common roach (*Rutilus rutilus* (Linnaeus, 1758)), common bream (*Abramis brama* (Linnaeus, 1758)), tench (*Tinca tinca* (Linnaeus, 1758)), the European perch (*Perca fluviatilis* (Linnaeus, 1758)), and the alien pumpkinseed sunfish (*Lepomis gibbosus* (Linnaeus, 1758)) and largemouth bass (*Micropterus salmoides* Lacepède, 1802) (Keller & Ribi, 1993; Soes et al., 2016; Twardochleb & Olden, 2016; Linzmaier et al., 2018); various bird species may predate on the snails including the native Eurasian coot (*Fulica atra* (Linnaeus, 1758)) and a number of duck species (SOVON, 2002; Soes et al., 2016). However, *B. chinensis* can grow to 70 mm, which is larger than native snails. This size difference may decrease the vulnerability of adult *B. chinensis* to predation relative to native species (Waltz, 2008; Johnson et al., 2009), but also makes them attractive to aquatic mammals.

Predators, parasites or pathogens affecting establishment
In its native range, *B. chinensis* is often infected by trematode species (Bury et al., 2007; Collas et al., 2017). To date, in Europe no extensive research has been carried out with regard to parasites and pathogens in *B. chinensis*. Only the native commensal or parasitic oligochaete worm *Chaetogaster limnaei limnaei* Von Baer, 1827 has been sampled at very low densities from snails removed from Boven-Hardinxveld in the Netherlands (Soes et al., 2016). Two subspecies of this worm have been described. The subspecies *C. limnaei vaughini* lives parasitic inside snails of a certain minimum size and feeds mainly on the host's kidney cells (Gruffyd, 1965). The other subspecies (*C. limnaei limnaei*) attaches itself externally to the body of snails or to the inside of their shell and can freely move (Buse, 1974). This so called commensal ectosymbiotic species has been recorded in many snail genera, among which *Lymnaea*, *Physa*, *Ancylus* and *Australorbis*.

The low incidence of *C. limnaei limnaei* observed on *B. chinensis* in the Netherlands is replicated in other areas of its alien range. In North America both field and experimental research suggests that *B. chinensis* is characterised by a low rate of trematode infection (Harried et al., 2015). Of 147 *B. chinensis* removed from 22 lakes across Wisconsin, USA, only two individuals hosted trematode parasites (Harried et al., 2015). Moreover, experimental exposure to other trematode species in laboratories indicated a lower infection rate than in other snail species. McLaughlin et al. (1993) showed following experimental exposures of *B. chinensis* to the trematode *Sphaeridiotrema pseudoglobulus* McLaughlin, Scott & Huffman, 1993, a parasite implicated in waterfowl die-offs, that a significantly lower infection level resulted in comparison with the snail species *Physella gyrina* (Say, 1821) and *Bithynia tentaculata* (Linnaeus, 1758) (Harried et al., 2015). Individual parasites that were able to infect *B. chinensis* were frequently encased in the snail shells in a non-viable state (Harried et al., 2015). Trematodes can lower reproduction and survival in snails at high infection rates (Harried et al., 2015). Therefore, lower infection rates relative to native species may give *B. chinensis* a competitive advantage in alien ranges (Harried et al., 2015; Collas et al., 2017).

Establishment under protected conditions

No information could be found on population establishment of *B. chinensis* under protected conditions in Europe, such as private or public areas in which the environment is artificially maintained (e.g., zoological gardens, wildlife parks, glasshouses and aquaculture facilities). Zoological gardens and wildlife parks must take measures to prevent escapes of their animals, discourage intentional disposal and are subject to controlled public access. According to the Directive 1999/22/EC on the keeping of wild animals in zoos an operating licence will be required. In order to obtain an operating licence, zoos must for instance 1) prevent animals from escaping in order to avoid possible ecological threats (e.g., invasive alien species) to native species, as well as to prevent the intrusion of outside pests, and 2) keep up-to-date records of the animals in the establishment which vary according to the species. So, probability of escapes and disposal of individuals from establishment populations under protected conditions is expected to be low. It is likely that the main pathway of introduction of *B. chinensis* to the EU is via intentional disposal from aquaria and ponds by hobbyists (see § 2.2.1 and 2.2.2).

Availability of suitable habitat in the EU

B. chinensis usually establishes in lakes and slow flowing river and streams that have silty or muddy substrates in the littoral zone. However, the species can resist relatively high flow rates and has been observed in water bodies with current velocities up to 0.54 m/s (Table 2.5). Lakes with average temperatures ranging from 17 to 22°C during the spring and summer months are ideal habitat for this species (Haak, 2015). Rip-rap or rocky substrates may provide refuge from predation for juveniles particularly (Haak, 2015). *B. chinensis* thrives in eutrophic environments where there is abundant diatom and other periphyton available for forage (Haak, 2015).

A habitat type of EUNIS that is expected to be suitable for *B. chinensis* establishment is habitat type C1.3: Permanent eutrophic lakes, ponds and pools. Habitat type C1.3 overlaps with the Annex I habitat type 3150 of the EU habitats directive. This habitat type is characterized by the European Environment Agency as 'lakes and pools with mostly dirty grey to blue-green, more or less turbid, waters, particularly rich in nutrients (nitrogen and phosphorus) and dissolved bases (pH usually > 7). Moderately eutrophic waters can support dense beds of macrophytes, but these disappear when pollution causes nutrient levels to rise further (European Environment Agency, 2017)'. This habitat type is widely available in the EU.

Ratio of colonized and available habitat in the EU

The actual geographical distribution of *B. chinensis* in the EU is still limited compared to the area that can potentially be colonised by this species (i.e., less than 5% of the potential area; see figure 2.10 in paragraph 2.3.5).

Conclusion
The preferred habitat of *B. chinensis* is widely available in the EU. Potentially, the species can colonise large parts of the EU that constitute suitable habitat.

2.3.4 Climate match and biogeographical comparison

B. chinensis has established several isolated persisting populations in the Netherlands and Belgium (§ 2.3.3). This means that the climate and certain habitats in the Netherlands and Belgium are suitable for *B. chinensis* establishment. This is supported by Soes et al. (2016) who observed that *B. chinensis* survived well and produced offspring in garden ponds in the Netherlands. All nine snails caught in the Eijsder Beemden, the Netherlands had developing young inside (Collas et al., 2017).

A biogeographical comparison using the biogeographic classification system of the European Environment Agency shows that the introduced range of *B. chinensis* in the Netherlands and Belgium is classified within the Atlantic biogeographical region of the EU (European Environment Agency, 2016; Figure 2.8). The Atlantic biogeographical region extends to Western Denmark, Germany, France, the far north of Spain, the United Kingdom and Ireland, and some coastal regions in Norway.

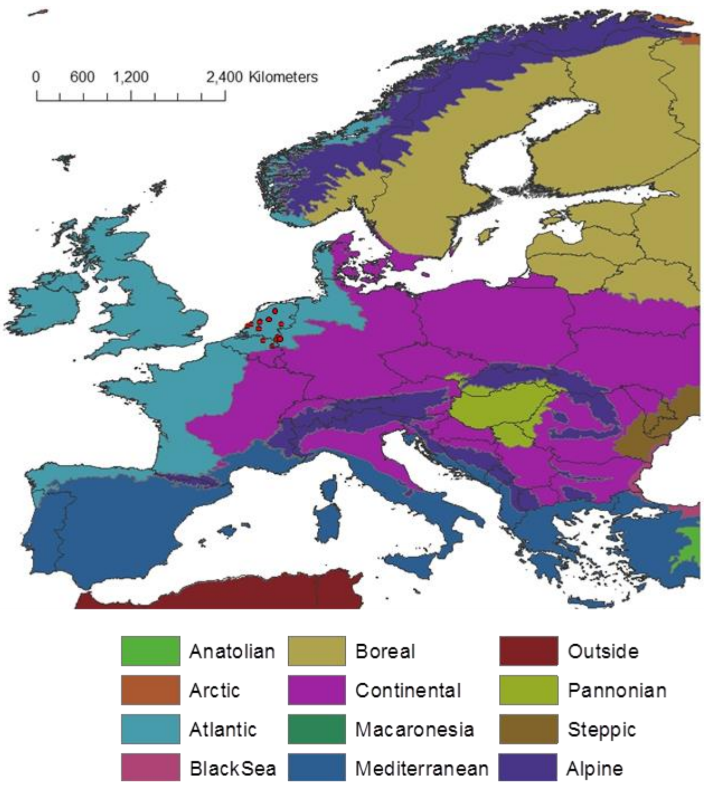
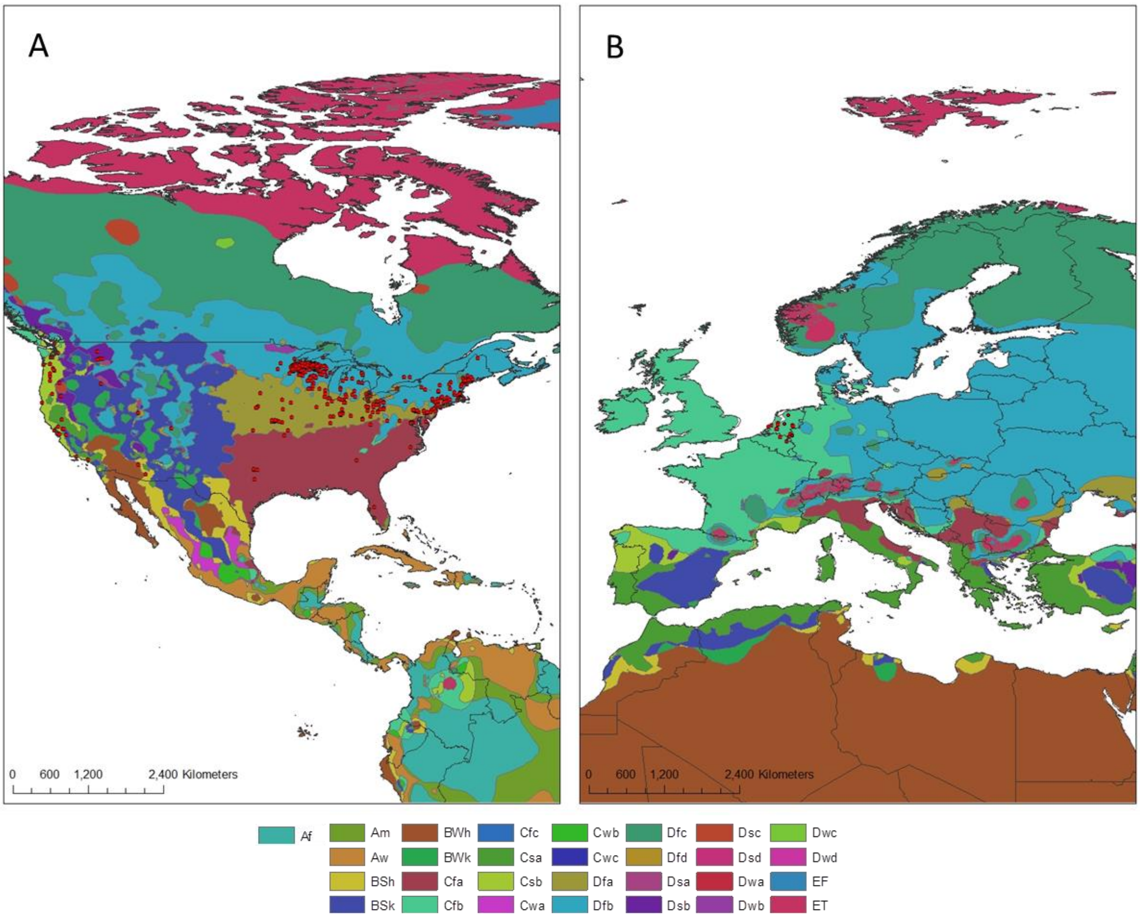


Figure 2.8: Current European Union records of the Chinese mystery snail (*Bellamya chinensis*) (red dots) matched with biogeographic regions in Europe (European Environment Agency, 2016).

A large portion of North-Western Europe is characterized by the Köppen-Geiger climate region Cfb which matches its current European non-native range in the Netherlands. Climate region code C means that the air temperature of the warmest month is higher or equal to 10°C, and the temperature of coldest month less than 18°C but higher than –3°C. The codes f and b indicate that precipitation is evenly distributed throughout year and the temperature of each of four warmest months is 10°C or above but warmest month less than 22°C, respectively. Region Cfb covers the Netherlands, Belgium, western Germany, France, the United Kingdom and the Irish Republic, and Northern Spain (Figure 2.9b). Currently, *B. chinensis* occurs in the EU in the Netherlands and Belgium. Given the current climate of its native and introduced ranges, and assuming no management measures to prevent introduction or spread on a European scale, the species may potentially establish in all other EU member states.



Main climates		Precipitation		Temperature		
A: Equatorial	D:Snow	f: Fully humid	w: Dry winter	T:Polar tundra	h:Hot arid	b:Wwarm summer
B: Arid	E: Polar	m:Monsoon	W:Desert	F:Polar frost	k:Cold arid	c:Cool summer
C: Warm		s: Dry summer	S: Steppe		a:Hot summer	d:Extremely continental

Figure 2.9: Records of the Chinese mystery snail (*Bellamya chinensis*) (red dots) matched to Köppen-Geiger climate zones of Kottek et al. (2006) in **A)** The United States of America and Ontario, Canada (Kipp et al., 2014); **B)** The European Union (§ 2.3.3).

A comparison of the Köppen-Geiger climate regions in *B. chinensis*' North American introduced range and their prevalence in Eurasia suggests that climate will form no barrier to the establishment of the species in large parts of Europe. The North American introduced range is dominated by the Köppen-Geiger climate regions Dfa and Dfb in Eastern USA and Csb on the West coast (Figure 2.9a). However, the species is also present in regions Dfc, CSa, Dsb, Bsk, Bwh and Bsh.

Large portions of Eastern Europe are dominated by climate zone Dfb (Figure 2.9b). Climate zone Dfb occurs mostly in Germany, northern Denmark, southernmost parts of Finland and southern Sweden, Poland, the Czech Republic, Hungary, Austria, Slovakia, Romania, Moldova, Ukraine, Belarus, Lithuania, Latvia, Estonia, and Western parts of Russia. Also, small parts of Northern Italy and Spain, the Netherlands, Switzerland, France, central and southern Norway, Bulgaria, Serbia, Slovenia, Bosnia and Herzegovina, and the Former Yugoslav Republic of Macedonia are classified under climate zone Dfb. Parts of north-western Spain, Northern Portugal and small parts of southern France and central Italy are classified as climate zone Csb. Small parts of Slovakia, Hungary, Romania, Bulgaria, Moldova, and southern Ukraine are classified as climate zone Dfa.

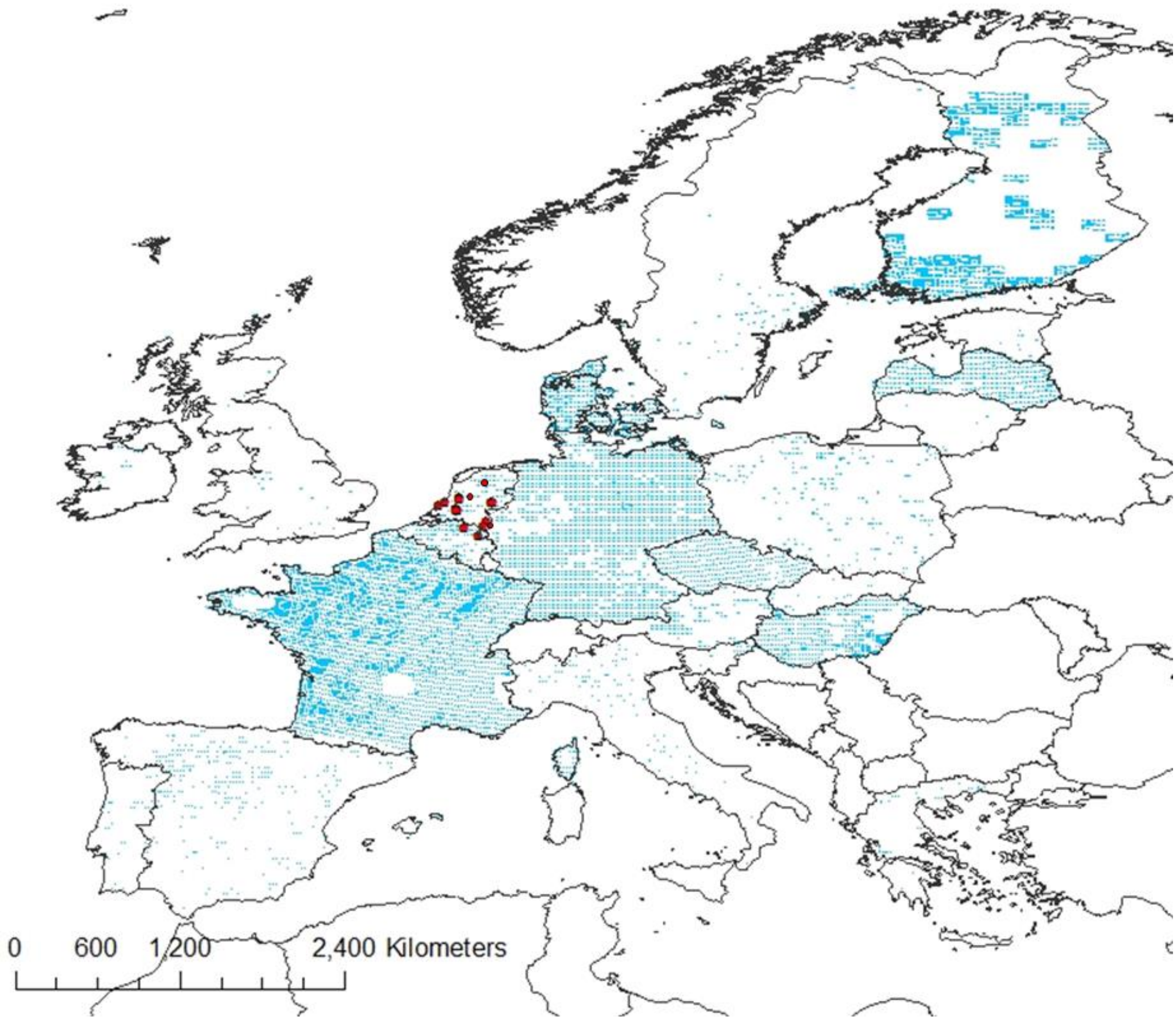


Figure 2.10: Records of the Chinese mystery snail (*Bellamya chinensis*) matched with EU habitat types 3150 (Natural eutrophic lakes with Magnopotamion or Hydrocharition type vegetation) indicated in blue. Suitable parts of habitat types 3200 (rivers) for future establishment of *B. chinensis* were not mapped due to lack of spatial explicit data on stream velocity, depth and substrate of littoral zones of rivers and streams.

Conclusion

The introduced range of *B. chinensis* in the Netherlands and large portions of its introduced range in North America are climatically matched with large parts of Eurasia, including all EU countries. The introduced range of *B. chinensis* in the Netherlands and Belgium is within the Atlantic biogeographical region which is matched with Western Denmark, Germany, France, the far north of Spain, the United Kingdom and the Irish Republic, and some coastal regions in Norway.

2.3.5 Potential influence of climate and habitat change on native range

B. chinensis has a very wide climate tolerance which is illustrated by its current native and alien distributions and broad environmental tolerances (§ 2.3.4). Therefore, it is unlikely that future climate change will result in limitation of the species.

2.3.6 Potential future distribution in the EU and endangered areas

It is expected that EU habitat type 3150 provides most suitable conditions for the establishment of *B. chinensis* (§ 2.3.3). Areas within the EU that are classified under this habitat type are shown in Figure 2.10 with an indication of the current distribution of the species in the Netherlands and Belgium. These areas in particular are expected to be at high risk of future colonisation by *B. chinensis.*

Moreover, running waters (EU habitat types 3200) will be partly suitable for future establishment of *B. chinensis* (i.e., slow flowing and shallow littoral zones of freshwater sections of lowland zones of rivers and streams). It was not possible to delineate the endangered areas of habitat type 3200 due to lack of spatial explicit data on stream velocity, depth and substrate of littoral zones of rivers and streams.

Conclusion

It is expected that *B. chinensis* will be able to establish in EU habitat type 3150. These areas in particular are expected to be at high risk of future colonisation by *B. chinensis.*

2.3.7 Influence of management practices

Facilitation of establishment by current management practices

It is unknown if current conservation practices, such as general river habitat improvement, encourage the establishment of *B. chinensis* in the EU. However, management practices leading to habitat change that increase the availability of suitable habitat may encourage the establishment of *B. chinensis* (see Haak, 2015). For example, a reduction of flow velocity and increase in silty or muddy substrates in the littoral zone will allow the snails to burrow in winter or addition of patches of rip-rap or rocky substrates may provide refuge for developing juveniles and adults.

Current 'Room for the River' measures in the Netherlands are expected to increase the habitat availability for *B. chinensis*, owing to transitions of (semi)terrestrial areas to aquatic ecotopes, such as slow flowing side channels with inlet work of rip-raps and relatively high sedimentation rates (Straatsma et al., 2009).

Conclusion

It is unknown if current conservation practices encourage the establishment of *B. chinensis* in the EU. However, current river and lake rehabilitation measures that increase the availability of suitable habitat may encourage the establishment of *B. chinensis* in the EU.

2.4 Pathways and vectors for spread within the EU

2.4.1 Intentional pathways of introduction

No evidence of intentional pathways relating to secondary dispersal, i.e., dispersal that occurs following initial introduction, within the EU was found during a search of the available literature.

Table 2.5: Active (A) and potential (P) pathways and vectors for secondary spread of the Chinese mystery snail (*Bellamya chinensis*) in the European Union.

Category[a]	Subcategory[a]	A	P	Examples and relevant information	Reference
Transport - contaminant	Transportation of habitat material (e.g., sediment and vegetation)		X	Movement of mud within or between waterbodies as a result of maintenance	D.M. Soes (Pers. comm.)
Transport - stowaway	Ship/boat hull fouling		X	*B. chinensis* presence has been correlated with boating activity in North America	Havel (2011); Havel et al. (2014);
Transport - stowaway	Hitchhikers on ship/boat (excluding ballast water and hull fouling)		X	*B. chinensis* presence has been correlated with boating activity in North America	Havel (2011); Havel et al. (2014); Haak (2015)
Transport - stowaway	Angling/fishing equipment		X	*B. chinensis* presence has been correlated with fishing activity in North America	Haak (2015)
Unaided	Natural dispersal across international borders of invasive alien species that have been introduced through other pathways	X	X	*B. chinensis* displays limited natural dispersal. Downstream currents may transport snails over long distances. Waterfowl and aquatic mammals may disperse *B. chinensis*	Mackie (2000b); Bury et al. (2007); Collas et al. (2017)

[a] Classification according to UNEP (2014)

2.4.2 Unintentional pathways of introduction

Records of *B. chinensis* in the EU are scattered and expected to be the result of new introduction events (Kroiss, 2005; Strecker et al., 2011; Soes et al., 2016; Collas et al., 2017). This is supported by the observed low natural dispersal ability of *B. chinensis* in the EU of 0.1 kilometres.year^{-1} calculated during field surveys in the floodplain Eijsder Beemden, the Netherlands (Collas et al., 2017; § 2.1.2). However, it is possible that dispersal by human action has occurred at recorded locations in Giessen, Eijsder Beemden and Zutphen in the Netherlands (F. Collas, pers. comm.). In the Netherlands, maintenance of water systems (e.g., dredging and weed control in ditches and canals) and the transfer of sediment, plant material or equipment containing snails between or within waterbodies is probably an important future pathway of secondary spread. (D.M. Soes, pers. comm.). However, there is no available information regarding the frequency of dispersal via these pathways. Secondary dispersal may be facilitated by the presence of several human mediated dispersal vectors that increase the likelihood of spread of *B. chinensis* in Europe (Table 2.5).

Recreational boats could be a vector for dispersal of *B. chinensis* (Havel, 2011; Havel et al., 2014). Secondary spread may be facilitated by boater interactions between invaded and non-invaded lakes via bait-buckets, live wells, fishing gear, and the boat itself (Waltz, 2008). The likelihood of *B. chinensis* occurrence has been shown to decrease with increasing distance to a boat launch in a Wisconsin lake in the USA (Solomon et al., 2010). Haak (2015) found during research of Wisconsin lakes that the species was more often found in lakes with more public boat landings and in deeper lakes that are more likely to attract sports fishermen and recreational boaters. Local fishermen indicated that they have occasionally caught *B. chinensis* individuals in the Eijsder Beemden. Moreover, Solomon et al. (2010) found a significant positive relationship between the distance to human habitation and snail abundance in the same lake.

B. chinensis can survive long periods of drought, and tolerance of air exposure for more than nine weeks has been observed (Havel, 2011; Unstad et al., 2013; Havel et al., 2014). Within the EU there are a high number of overland boat transports during summer holiday periods. Therefore, overland dispersal in entangled aquatic plant fragments attached to recreational boats is also a possibility. However, if the species does not become entangled in plant fragments and is directly attached to the hull, it will drop off when the operculum is closed on exposure to air.

Natural dispersal vectors include waterfowl and aquatic mammals (e.g., otters, brown rats, muskrats) that can probably disperse *B. chinensis* between water bodies (Mackie, 2000b). Moreover, Collas et al. (2017) states that if *B. chinensis* is able to colonise lowland rivers, it can be expected that the downstream dispersal rate strongly increases due to water flow (especially applicable to the high discharge that occurred during the winter of 2017-2018), recreational boating and shipping. Therefore, spread by natural means is likely to increase in the absence of management measures. According to physico-chemical ranges of the species' habitat (Table 2.5) and distribution in its native and introduced ranges (§ 2.3.1), it is not likely that seasonal factors will affect the survival of *B. chinensis* in the EU. The species is buried in sediment in winter and early spring thereby potentially avoiding the high discharges and low temperatures that occur in this period.

2.4.3 Commodities associated with introduction

B. chinensis probably spreads through the human-mediated movement of aquatic plants (Havel, 2011; Haak, 2015). In the Netherlands, *B. chinensis* has been recorded with alien crayfish *O. virilis* at Vinkeveen, and both *O. virilis* and *P.* cf. *acutus* at Boven-Hardinxveld. Large fish importers that store fish in open air ponds are situated close to these locations and it has been suggested that these importers are the source of introductions of all these species originating from North America (Soes et al., 2016).

2.4.4 Pathway origins and end points

The 17 known locations of *B. chinensis* in the Netherlands and single record in Belgium are currently the origins for potential further natural spread within the EU (Figure 2.10). Potential endpoints of this pathway within the EU are France and Germany resulting from upstream dispersal in the rivers Rhine and Meuse or overland dispersal facilitated by several vectors, e.g., watercraft and fishing equipment (see also § 2.4.5). Suitable habitat for *B. chinensis* is available in large parts of the EU.

2.4.5 Propagule pressure per pathway

The current EU distribution of *B. chinensis* has likely resulted from multiple independent intentional and unintentional introductions (Kroiss, 2005; Strecker et al., 2011; Soes et al., 2016; Collas et al., 2017). In North America, recreational activities such as boating are suspected to contribute to the species' secondary spread (Van den Neucker et al., 2017). However, no evidence describing how influential this pathway is to *B. chinensis* secondary dispersal in the EU could be found during a review of available literature.

Conclusion

B. chinensis are likely to be introduced to the EU as part of the international trade in live animals via garden and aquarium centres and as a potential food source for people particularly of Asiatic origin. The recorded pathways of introduction and spread of *B. chinensis* to North America and Europe suggest that further spread within the EU may result from isolated introductions resulting in a discontinuous distribution. Isolated introductions may result from the disposal of the contents of ponds and aquaria. In the absence of management measures, populations of *B. chinensis* in the Netherlands and Belgium may serve as sources of secondary spread within the EU (e.g., to France and Germany). The species may disperse within the EU attached to shipping and with fishing equipment, via transportation by waterfowl and aquatic mammals, and through limited natural dispersal.

2.5 Impacts

2.5.1 Environmental effects on biodiversity and ecosystems

In general, *B. chinensis* has generally not been seen as a problematic in North America where densities are relatively low (Mackie, 2000a; Solomon et al., 2010; McAlpine et al., 2016). Specifically, the species has exerted no recorded impacts in the North American Great Lakes (Kipp et al., 2014; McAlpine et al., 2016). However, the risks associated with extremely high densities of the species are largely unknown and the lack of perceived threat may be attributed to this lack of knowledge (Mackie, 2000a; Bury et al., 2007; Waltz, 2008; Breedveld, 2015). An assessment of densities of adult *B. chinensis* in two floodplain lakes in the Eijsder Beemden was made as part of this study. Average densities ranged between 1.28 and 9.02 ind.m^{-2}. The total surface area of the floodplain lakes was applied in combination with these species densities to estimate adult population size which ranged roughly between 26,000 and 460,000 individuals. The filtration capacity of the entire population of adult *B. chinensis* in both lakes ranged between 4,400 and 192,000 litres per hour. Therefore, the water contained in the lakes could potentially be filtered by the *B. chinensis* population every 22 to 190 days. The filtration rate depends on season, temperature, food availability, local conditions and is relative to densities of other filter feeding species. The aforementioned values are an indication of potential filtration only and may not reflect the actual filtration occurring in the lakes. The *B. chinensis* biomass ranged between 850 and 1,225 kilograms in an isolated lake and to 14,400 and 15,000 kilograms in a lake connected to the river Meuse. Such a high snail biomass can be expected to influence water chemistry, water transparency and food webs (Hall et al., 2003; Chaine et al., 2012). Moreover, the species received a high invasiveness score in the New York Invasiveness Ranking System (New York Invasive Species Information, 2017). This high invasiveness score can be explained due to the high scores given to the species' biological characteristic and dispersal ability (27 out of a possible 30), ecological amplitude and distribution (28 out of a possible 30), and difficulty of control (7 out of a possible 10). The New York Invasiveness Ranking System gave *B. chinesis* a score of 21 out of 30 for ecological impact. An assessment of invasive aquatic species impacts was made by sending a survey to aquatic ecologists in the State of Illinois (Jacobs & Keller, 2017). Respondents were asked to assign ecological impacts on a four point scale. *B. chinensis* received an average score of 3, which classified it as a high impact species (species where classified as high impact when the average score is between 2.5 and 3.49).

Abiotic impacts

Water quality and chemistry

Mesocosm experiments have demonstrated that *B. chinensis* grazing has been found to alter periphyton algal species composition, reduce algal biomass and increase the N:P ratio in the water column (Johnson et al., 2009). Johnson et al. (2009) observed that the addition of *B. chinensis* significantly increased the N:P molar ratio by approximately 25 % over control values. Changes in the N:P ratio can significantly affect algal community structure in natural systems (Collas et al., 2017). A higher N:P ratio might be beneficial for recreation as higher ratios stimulate the growth of diatoms rather than blue algae (Wetzel & Likens, 2000). This may result from *B. chinensis*' low excretion of phosphorus compared to *P. gyrina*, *L. stagnalis* and *Helisoma trivolis* Say, 1817, native snails in North America (Johnson et al., 2009; Kipp et al., 2014; Collas et al., 2017). Therefore, an increase in the N:P ratio of the system is expected if *B. chinensis* replaces these native snails, or if adult densities are as high as 9.02 ind.m^{-2} as was found in one of the floodplain lakes in the Eijsder Beemden. Of these species *L. stagnalis* is native to Western Europe (Fauna Europaea, 2017).

Biotic impacts

Competition

A number of authors have examined the potential impacts of *B. chinensis*. Evidence from experimentation suggests that *B. chinensis* may reduce native snail populations through competitive exclusion, altering nutrient cycling and decreasing algal biomass (Clark, 2009; Johnson et al., 2009; McAlpine et al., 2016). In mesocosm experiments, *B. chinensis* caused considerable declines in the abundance and growth of *L. stagnalis* probably through competition for food (Johnson et al., 2009; Kipp et al., 2014). However, field studies have not yet confirmed any negative impacts on native gastropod assemblages (Kipp et al., 2014). Solomon et al. (2010), found no difference in snail assemblages with respect to *B. chinensis* presence or abundance in North American lakes. However, it has also been reported that some snail species in North America, tend not to occur where *B. chinensis* is abundant, including native *L. stagnalis* (Solomon et al., 2010). A mesocosm experiment demonstrated that *B. chinensis* at densities of circa 10 individuals m^{-2} had stronger effects on *L. stagnalis* (reduced survival) than on *P. gyrina* (reduced growth) (Johnson et al., 2009). Similar adult *B. chinensis* densities of 9.02 ind.m^{-2} have been found in the Eijsder Beemden in a lake where *L. stagnalis* occurs. Therefore, *L. stagnalis* may be negatively influenced by the presence of *B. chinensis* in this lake. Moreover, it has been suggested that the size advantage of *B. chinensis* over native species may decrease predator vulnerability of *B. chinensis* (Waltz, 2008; Johnson et al., 2009). The lower vulnerability of *B. chinensis* may result in selective predation of the native common river snail (*V. viviparus*), a species that occurs at the Eijsder Beemden. *B. chinensis* has a filtration rate that is similar to invasive *Dreissena polymorpha* (Pallas, 1771), *Dreissena rostriformis bugensis* Andrusov, 1897 and *Limnoperna fortunei* (Dunker, 1857; Olden et al., 2013). Dreissenid mussels have high filtering rates and have great potential to decrease the phytoplankton biomass in ecosystems (MacIsaac et al., 1992; Bunt et al., 1993). Dreissenids are absent from several sites where *B. chinensis* currently occurs (e.g., Amsterdam, Balen, Zutphen, Maasbree, Maasbracht and an isolated lake in the Eijsder Beemden). The presence of *B. chinensis* subjects these sites to higher filtration pressure, potentially influencing ecosystem functioning.

Interaction with other alien species

In the Netherlands, *B. chinensis* has been recorded with *O. virilis* at Vinkeveen, and both *O. virilis* and *P.* cf. *acutus* at Boven-Hardinxveld (Soes et al., 2016). An experimental study undertaken in Washington, USA, suggests that *B. chinensis* may facilitate the establishment and ecological impacts of *O. virilis* by providing an abundant food resource (Olden et al., 2009; Kipp et al., 2014).

When *B. chinensis* occurs with invasive crayfish species the negative effects on the occurrence of some native snail species in North America are synergistic (Solomon et al., 2010). A mesocosm demonstrated that impacts on native snail species of predation by the rusty crayfish (*Orconectes rusticus* (Girard, 1852)) and competition of *B. chinensis* were more severe than either species alone (Kipp et al., 2014). The abundance of *L. stagnalis* decreased by 32 % and 100 % with the addition of *B. chinensis* alone, and with *B. chinensis* and *O. rusticus* combined, respectively (Johnson et al., 2009). The abundance of *B. chinensis* was also reduced following the addition of *O. rusticus*, though the species' total biomass remained the same (Johnson et al., 2009). *B. chinensis* is less vulnerable to predation by *O. rusticus* compared to other snail species due its larger size and thicker shell (Kipp et al., 2014). *B. chinensis* may also benefit from an increased availability of food that would otherwise be consumed by competing snail species (Johnson et al., 2009; Kipp et al., 2014). In mesocosm experiments performed in Washington State (USA) comparing different crayfish species indicated that the American signal crayfish, *Pacifastacus leniusculus* (Dana, 1852), an alien species widespread in the EU, consumed more *B. chinensis* compared to red swamp crayfish, *Procambarus clarkii* (Girard, 1852), and virile crayfish, *O. virilis*, all alien species that have also been introduced to Europe (Olden et al., 2009).

Information on the effects of *B. chinensis* on other alien species apart from crayfish is limited. Twardochleb & Olden (2016), observed that the alien pumpkinseed sunfish (*Lepomis gibbosus* (Linnaeus, 1758)) and largemouth bass (*Micropterus salmoides* Lacepède, 1802), species that are invasive and accidentally occur in the Netherlands, respectively, preyed on *B. chinensis* in urban lakes in the USA and may therefore benefit from the availability of extra food if *B. chinensis* establishes. The pumpkinseed sunfish and largemouth bass occur throughout the European Union. In its native range, the species has been found to have a high protein:lipid ratio, which was beneficial for the growth of hatchery-cultured crabs (Gong et al., 2017). It is possible that native or alien European species that consume *B. chinensis* may benefit from this ratio as well. Shells of *B. chinensis* are often found covered with algae at the locations where the species occurs in the European Union. These algae serve as a food source for *B. chinensis*, increasing growth (Fujibayashi et al., 2016). It is not clear whether the attached algae are native to the European Union, or have been introduced together with *B. chinensis*. The quagga mussel (*D. rostriformis bugensis*), an invasive species in the EU, was found attached to 27% of the *B. chinensis* collected from a lake connected to the river Meuse at the Eijsder Beemden. On average, 1.4 dreissenids were found attached to *B. chinensis* individuals, possibly exerting a negative influence on *B. chinensis* (This study, Matthews et al., 2014). *B. chinensis* can potentially transport dreissenids to sites where the dreissenids have previously not occurred, though the natural dispersal capacity of dreissenids is higher than that of *B. chinensis*.

Hybridization

Genetic introgression becomes a problem when alien and native species hybridize successfully. No evidence of hybridization with European native species was discovered during a search of the available literature. No native congeners of *B. chinensis* exist in the EU.

Parasites and pathogens

To date, no extensive research has been carried out with regard to parasites and pathogens in *B. chinensis* in Europe. However, field and laboratory research indicate that *B. chinensis* has a low rate of parasitic infection in its alien North American range (Harried et al., 2015). Moreover, Mastitsky et al. (2010) state that there is no evidence of *B. chinensis* transporting native parasites to its introduced environment (Haak, 2015). In its native range, *B. chinensis* often carries trematode species (Bury et al., 2007). However, until now only the commensal or parasitic oligochaete worm *C. limnaei limnaei* has been sampled from snails removed from Boven-Hardinxveld in the Netherlands. This oligochaete species is very common in the Netherlands and exist in a wide spectrum of freshwater ecosystems (Soes et al., 2016).

In North America, the leech *Macrobdella decora* (Say, 1824) has been found to feed on *B. chinensis* in a laboratory setting, ultimately resulting in *B. chinensis* mortality (Canfield et al., 2016). Due to the limited experimental set-up used by Canfield et al. (2016), additional research is required to fully grasp the effect of leeches on *B. chinensis*. To date, no records of leeches that feed on *B. chinensis* in the European Union exist.

In general, *B. chinensis* is a host to the trematode *Aspidogaster conchicola* Von Baer, 1827, a parasite with a wide European distribution that parasitizes unionid bivalves (Huehner & Etges, 1977; Bury et al., 2007; Waltz, 2008; WoRMS, 2017). *B. chinensis* commonly hosts *A. conchicola* in its introduced North American range (Michelson, 1970; Kipp et al., 2014). This parasite does not appear to be of economic or medical importance and has consequently not been well studied. Experiments from North America indicate that *B. chinensis* hosts *Sphaeridiotrema pseudoglobulus* (Rudolphi, 1814), a trematode parasite implicated in waterfowl die-offs, at a significantly lower infection level than the snail tadpole physa (*P. gyrina*), an alien species introduced to the EU, and mud bithynia (*B. tentaculata*), an originally European species (McLaughlin et al., 1993; Harried et al., 2015).

However, at locations where the frequency of infection is high, such as in its native range of China, parasites of *B. chinensis* may pose a risk to birds and mammals through predation (Chao et al., 1993).

Haak (2015) sounds a note of caution by stating that we still do not know if *B. chinensis* transports parasites from their native range, and that some of these parasites can be deadly to humans. *B. chinensis* is a second intermediate host of *Echinostoma cinetorchis* Ando & Ozaki, 1923 and *Echinostoma marcrorchis*, parasitic trematode flatworms that are a potential source of human infection in Korea (Chung & Jung, 1999; Waltz, 2008; Sohn & Na, 2017). *E. cinetorchis* is recorded in Korea, Japan, Taiwan, and Java. Sohn & Na (2017) observed that infection rates of *B. chinensis* with *E. macrorchis* varied between 60 and 100%, with the number of metacercariae (encysted larva) ranging between 2 and 412 per individual. *B. chinensis* is an intermediate host for *Angiostrongylus cantonensis* (Chen, 1935), a parasitic nematode (roundworm) that causes eosinophilic meningitis in humans in Taiwan (Lv et al., 2009; Haak, 2015). *B. chinensis* infected with *A. cantonensis* have been found in restaurants in Taiwan (Lv et al., 2009). *B. chinensis* has been reported to serve as a vector for several parasites, which are however exotic to the Netherlands (Jokinen, 1982; Chai et al., 2009). Most of these parasites, like *A. cantonensis* (Chen, 1935) and several members of the family Echinostomatidae need high temperatures or primary hosts that are not present in Europe (H. Cremers, personal communication to D.M. Soes).

Ecosystem alteration

Unfortunately, little is known about interactions between *B. chinensis* and native aquatic species, which severely limits our ability to understand the potential large scale impacts of this snail on ecosystems in its alien range (Chaine et al., 2012; Harried et al., 2015). Waltz (2008) suggests that *B. chinensis* may exert a bottom-up control of some food webs which may in turn reduce the amount of fish predators prized by anglers. A slight alteration in the composition of the microbial community may occur as a result of *B. chinensis*' feeding behaviour or their excretion products (Olden et al., 2013). *B. chinensis* can support native consumers by serving as a source of prey in degraded ecosystems (Twardochleb & Olden, 2016). In addition, *B. chinensis* may serve as a non-typical food source for parasitic leeches following the loss of piscivores in lake systems (Canfield et al., 2016).

Haak (2015) used a combination of social and ecological network modelling to predict the impact of *B. chinensis* in the USA. Population development from the point of introduction in reservoirs was simulated over a 25 year period using the Ecopath and Ecosim models (Polovina, 1984; Christensen & Pauly, 1995; Haak, 2015). The results indicated that the introduction of *B. chinensis* by boat and bank anglers did not cause significant changes to ecosystem functioning in flood-control reservoirs in south-eastern Nebraska but did often cause changes in the biotic composition of the community, with mid-trophic-level fishes most often negatively affected. Additionally, after introduction, primary production was predicted to greatly increase (Haak, 2015).

Impacts on species and ecosystem functioning in the EU in the future

Future impacts of *B. chinensis* in the EU will be dependent on the density of snails that occurs in individual water bodies. Effects on native snail species and ecosystem functioning are expected if high densities of *B. chinensis* occur. Despite the presence of multiple dispersal vectors (§ 2.4), and the presence of suitable habitat (§ 2.3.3), the potential for the development of high densities of individuals locally may be limited. Evidence from the Netherlands suggests that at the few locations where *B. chinensis* has become established, a variety of native and alien predator species are able to make use of *B. chinensis* as a food source and that expansion of populations of *B. chinensis* may be limited (Soes et al., 2016). Evidence of predation by brown rats (*Rattus norvegicus*), and a relative lack of juveniles at the Eijsder Beemden suggests that birds, fish and/or other animals may predate on the snails (Soes et al., 2016). Predictions of future impact should also consider the lack of perceived impacts in *B. chinensis*' North American introduced range (Mackie, 2000a; Solomon et al., 2010; Kipp et al., 2014; McAlpine et al., 2016). The importance of impacts will depend on the reference conditions, ecological status and conservation goals of areas that will be colonized by *B. chinensis*.

Declines in conservation status of nature areas

An analysis of the current distribution of *B. chinensis* in the Netherlands and Belgium indicates that the species has been recorded in two Natura 2000 areas. In the river IJssel floodplain, south of the town of Deventer, in the Netherlands; and in Belgium, in the headwaters of the river Grote Nette near Zammelsbroek, Langdonken and Goor. Three samples in the Netherlands lie in close proximity to the Biesbosch Natura 2000 area in the Netherlands and one sample is situated close to the IJsselmeer Natura 2000 area (Figure 2.11). Therefore, the species should be able to colonise areas of high conservation value.

Declines in conservation status of nature areas now and in the future caused by *B. chinensis* (e.g., changes in ecological status of water bodies according to Water framework Directive classification or effects on habitat types or target species in Natura 2000 areas) will be dependent on the density with which the species occurs locally. In its North American alien range where densities are low, *B. chinensis* is perceived to have little impact on native biodiversity (Mackie, 2000a; Solomon et al., 2010; McAlpine et al., 2016). There are indications that *B. chinensis* densities are lower in developed areas than in undeveloped areas (Twardochleb & Olden, 2016). The species is able to reach high densities in its European introduced range (§ 2.3.2). However, the risks associated with extremely high densities of the species are largely unknown (Mackie, 2000a; Bury et al., 2007; Waltz, 2008; Breedveld, 2015). The importance of changes in status of nature areas resulting from introductions of *B. chinensis* depends on the reference conservation goals of these areas.

Where environmental impacts are likely to occur in the EU

The impacts on biodiversity and ecosystem functioning such as competition with native snails, alterations to algal species composition, and increase of the N:P ratio in the water column, are likely to occur in all introduced ranges where the species achieves high densities.

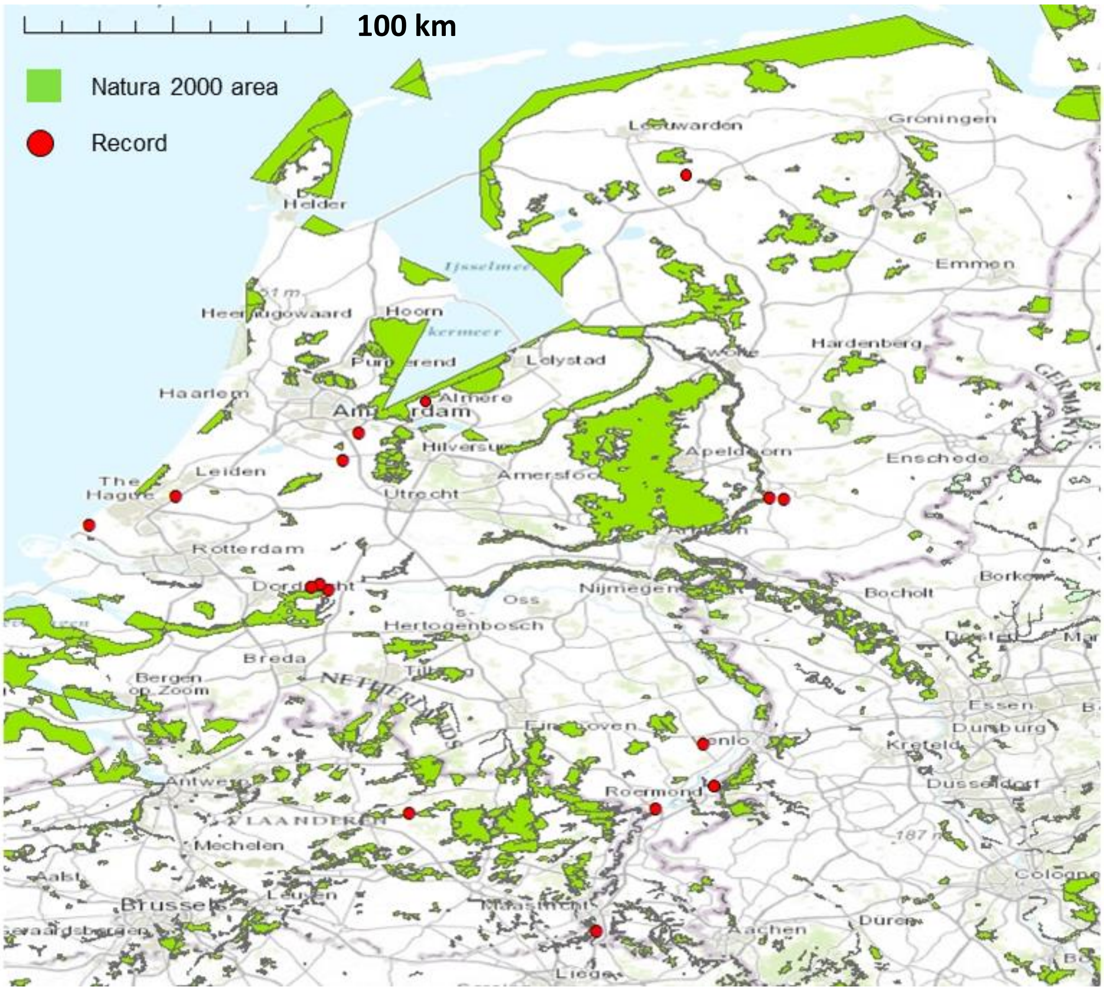


Figure 2.11: Records of the Chinese mystery snail (*Bellamya chinensis*) matched with Natura 2000 areas highlighted in green in the Netherlands and Belgium.

2.5.2 Effects on cultivated plants

Scientifically sound information describing damage to cultivated plants by *B. chinensis* is not available. *B. chinensis* eats algae by grazing and filter feeding. The species does not feed on water plants (Soes et al., 2016). Therefore, it is unlikely that negative impacts would occur on cultivated water plants.

2.5.3 Effects on domesticated animals

Scientifically sound information concerning the effects of *B. chinensis* on domesticated animals in native or introduced ranges is not available. It is unlikely that production animals or companion animals will be affected by *B. chinensis* as a result of parasitism, hosting pathogens or parasites, or via the biological, physical and / or chemical properties of *B. chinensis* that are harmful upon contact.

2.5.4 Effects on public health

No evidence of effects of *B. chinensis* on public health in the species' introduced range could be found in the available literature. Some authors state that there is no record of parasitic infection of humans in the USA as a result of contact with *B. chinensis* (Bury et al., 2007; GISD, 2017). However, there is evidence that the species is eaten in the USA, the route by which human infection of parasites carried by *B. chinensis* normally occurs if not sufficiently cooked (JCC, 2017). Indeed, the consumption of *B. chinensis* is actively encouraged through the presence of recipes available on websites (Invasivore.org, 2017; JCC, 2017), albeit with recommendations for thorough cooking to reduce the chances of infection (Invasivore.org, 2017). The absence of observed human infection may be related to the low rate of *B. chinensis* infection observed in its introduced ranges (Mastitsky et al., 2010; Harried et al., 2015; Soes et al., 2016).

Residents of areas with lakes inhabited by *B. chinensis* in North America are worried that the snail may carry the swimmer's itch parasite which, in theory, it could (Wood & Blumer, 2017). However, there are no records of this parasite being hosted by *B. chinensis*.

In its Asian native range *B. chinensis* carries a variety of parasites that may potentially infect humans. This snail is also the intermediate host for several helminth parasites that affect humans (Kipp et al., 2014). These include *Echinocasmus elongatus* Miki, 1923; *Echinochasmus redioduplicatus* Yamaguti, 1933; *Echinochasmus rugosus* Yamaguti, 1933; *Eupariphium ilocanum*; *Eupariphium recurvatum* (Linstow, 1873); *Echinostoma macrorachis* and *Echinostoma cinetorchis* Ando and Ozaki, 1923 (Pace, 1973; Chao et al., 1993; Sohn & Na, 2017). Moreover, *B. chinensis* hosts a trematode parasite that causes eosinophilic meningitis in humans in Taiwan, and hosts echinostome larvae in the Kinmen islands (Chao et al., 1993; Chung & Jung, 1999; Lv et al., 2009; Kipp et al., 2014; Haak, 2015). *B. chinensis* is an intermediate host for the trematode *Echinostoma gotoi* Ando and Ozaki, 1923 (Chao et al., 1993). Human infection of *E. gotoi* and subsequent echinostomiasis occurs following the consumption of raw or undercooked snails and symptoms include abdominal pain, diarrhoea and anorexia (Graczyk & Fried, 1998). Echinostomiasis can be simply treated with Praziquantel, a widely available oral medication (CDC, 2017). However, this will require sufficient knowledge on presence of this parasite in the Netherlands and other EU member states (e.g., general practitioners and doctors) and availability of diagnostics (H. Sprong, personal communication).

According to the Dutch National Institute for Public Health and the Environment (RIVM) it is unknown whether *B. chinensis* may act as an intermediate host for the native common liver fluke (*Fasciola hepatica* L., 1758) in Europe (H. Sprong, personal communication). However, known intermediate hosts of *F. hepatica* mainly belong to air-breathing freshwater snails from the family Lymnaeidae. Due to knowledge gaps on transmission of parasites *B. chinensis*, we recommend to study presence of zoonotic agents in established populations of this snail in the Netherlands.

2.5.5 Socio-economic effects

Socio-economic loss and costs: native geographic range

No information describing economic losses relating to *B. chinensis* in its native range could be found during the literature survey.

Socio-economic loss and costs: introduced geographic range

No information describing economic losses relating to *B. chinensis* in the EU could be found during the literature survey. The current populations in the Netherlands are too small to inflict any measurable social or economic impact. However, cyclical die-offs of *B. chinensis* occur frequently (Bury et al., 2007). In Minnesota, USA, dead and decaying shells have become a nuisance to local residence because they wash up on shores in high abundances (Bury et al., 2007). In the Laurentian Great Lakes of North America, fisherman have previously complained of making large hauls containing “2 tons” of snails in dragnets, which were likely *B. chinensis* or *B. japonica* (Wolfert & Hiltunen, 1968). *B. chinensis* may clog water intake pipes and inhibit water flow, incurring economic costs relating to reduced efficiency and the need for clean-up operations (Kipp et al., 2014). Impacts on water intake systems (e.g., obstruction of water flow small in pipes and pumps) may already occur at low numbers of individuals due to large size of these snails in comparison with other biofouling organism (e.g., dreissenid mussels).

Current and future economic costs in EU (excluding management costs)

The actual and future economic costs resulting from the introduction of *B. chinensis* in the EU (excluding management costs) are unknown. Scientific cost-benefit analyses are not available. The current costs in the EU are probably low or negligible due to the species’ restricted geographical distribution. Estimations of future cost should consider the lack of perceived impacts in North America where *B. chinensis* has been established for more than 120 years and is widespread, but recorded at low densities (Mackie, 2000a; Solomon et al., 2010; McAlpine et al., 2016). The risks associated with extremely high densities of the species are largely unknown (Mackie, 2000a; Bury et al., 2007; Waltz, 2008; Breedveld, 2015). Soes et al. (2016) reported that expansion of populations of *B. chinensis* in the Netherlands may be limited by predation of birds, fish and/or other animals (e.g., rats). If predation were to control population densities in the EU then impacts may be limited in a similar way to North America.

2.5.6 Effects on ecosystem services

In general, little is known about the impacts of invasive freshwater snails on ecosystem services, despite many successful invasions, partially because they are difficult to measure (Parker et al., 1999; Strayer et al., 2006; Simberloff, 2011; Haak, 2015). The effects of establishment of *B. chinensis* populations on various categories of ecosystem services are summarized in Table 2.6.

Table 2.6: Potential effects of Chinese mystery snail (*Bellamya chinensis*) populations on ecosystem services.

Service	Sub-category	Effect
	Provisioning Services	
Food	Crops	0
	Livestock	0
	Capture fisheries	+ / -
	Aquaculture	0
	Wild plant and animal food products	+
Fibre	Timber	0
	Cotton, hemp, silk	0
	Wood fuel	0
Genetic resources		0
Bio-chemicals, natural medicines, and pharmaceuticals		+
Fresh water		0
	Regulating Services	
Air quality regulation		0
Climate regulation	Global	0
	Regional and local	0
Water regulation		0
Erosion regulation		0
Water purification and waste treatment		+ / -
Disease regulation		0
Pest regulation		+ / -
Pollination		0
Natural hazard regulation		0
	Cultural Services	
Cultural diversity		0
Spiritual and religious values		0
Knowledge systems		0
Educated values		0
Inspiration		0
Aesthetic values		-
Social relations		0
Sense of place		0
Cultural heritage values		0
Recreation and ecotourism		+ / -
	Supporting services	
Soil formation		0
Photosynthesis		0
Primary production		-
Nutrient cycling		+
Water cycling		0

+ = positive effect, - = negative effect, +/- = positive and negative effects, 0 = no effect, ND = no data found in literature.

The potential effect scores are mainly based on the best professional judgement of the authors owing to a lack of (quantitative) data. *B. chinensis* is likely to have mixed positive and negative effects on capture fisheries and animal food products if it provides an alternative or supplementary food source and is accidently caught in fishermen's nets (§ 2.5.5). In addition, positive effects are expected as a number of pharmaceuticals can be extracted from the species (§ 2.5.9). Mixed positive and negative effects on water purification and waste water treatment are expected because of the species' capacity for biofouling and a high filtration rate; mixed positive and negative effects on pest regulation are expected as the species' carries fewer parasites than native snails, but may carry parasites that infect humans if ingested; negative impacts on aesthetic values, and mixed positive and negative effects on recreation and ecotourism are expected if large scale mortality occurs in rivers and lakes, but a positive effect will occur if blue algae growth is reduced. Supporting services will be subject to mixed effects due to a negative impact on primary production and a positive impact on nutrient cycling due to benthic grazing and filtration activities (benthic-pelagic coupling of nutrients). However, these last two effects appear to conflict due to conflicting evidence in literature that suggests (1) a large increase in modelled primary production in lakes in Nebraska, USA which may lead to eutrophication (Haak, 2015), and (2) increases in the N:P ratio and reductions in algal biomass following the introduction of *B. chinesis* in experimental mesocosms by Johnson et al. (2009).

2.5.7 Influence of climate change on impacts

Currently, studies that model the effects of climate change on *B. chinensis* are not available for the EU. *B. chinensis* tolerates a wide range of different climates and has been recorded in southern parts of Canada, and in the southernmost regions of the USA (Florida). The species non-native range has been climate matched to regions within all EU countries. Therefore, climate change defined as a temperature increase of 2 °C according to agreements made at the Paris climate conference – COP21 in the next fifty to hundred years (European Commission, 2017), will probably allow the potential distribution range of *B. chinensis* to expand north as northerly waterbodies exceed the 12 °C limit on reproduction.

2.5.8 Positive effects

Limited evidence for North America and the EU suggests that *B. chinensis* carries fewer parasites in its introduced range compared to its native range (Harried et al., 2015; Soes et al., 2016). Consumption of *B. chinensis* by native species is, therefore, less likely to lead to infection relative to native snail species. *B. chinensis* has been used in North America to supplement food stocks for game fisheries. There is evidence that *B. chinensis* increases the N:P ratio thereby favouring the growth of diatoms over blue algae (Wetzel & Likens, 2000). Reduced blue algae growth will positively impact recreation.

2.5.9 Known economic uses

Ornamental use and the selling of *B. chinensis* in garden and pond centres, and online in the Netherlands and Belgium are regarded as potential sources of species records in these countries (Soes et al., 2016; Van den Neucker et al., 2017, www.snailcorner.com). Release from overstocked aquaria and ponds, and escape from ponds, are likely routes of introduction of this species. The species is attractive to hobbyists as it maintains the clarity of the aquarium water and closes its trapdoor, or operculum in the presence of poor water quality (Waltz, 2008; Soes et al., 2016).

Individual examples of *B. chinensis* are sold at a Belgian garden centre for 1.25 euro (Van den Neucker et al., 2017), and in the Netherlands for approximately 2.95 euro (Frank Collas, pers. comm.). At the time of writing, the species was being sold online for € 7.50, though it is currently out of stock (www.snailcorner.com).

B. chinensis is an edible snail that was initially imported in North America as a food product sold on Chinese markets (Haak, 2015). There was no evidence in the available literature to suggest that the snail is sold for human consumption in the EU. However, the availability of recipes online which includes *B. chinensis* as an ingredient (e.g., Invasive.org, 2017), suggests that the snails may still be consumed by the public to some degree.

Purified polysaccharides may potentially be extracted from the flesh of *B. chinensis*. These polysaccharides have been shown to have a strong anti-inflammatory and anti-angiogenic effect (Xiong et al., 2017), potentially making *B. chinensis* a species that can be used to develop new medicines that reduce infections or inhibit the development of blood vessels that would otherwise supply blood to tumours. The species can also be used to extract hydroxyapatite (Zhou et al., 2016), a substance that facilitates blood clotting and bone regeneration (Yang et al., 2017). *B. chinensis* is widely used in traditional medicine in Asia (Yeshi et al., 2017; Kim et al., 2018).

Conclusion

A search of available literature found no evidence of current recorded or potential future impacts of *B. chinensis* in the EU. This may be related to the current distribution of the species which is limited to 17 records in the Netherlands and a single record in Belgium, and the impact of predation on established populations which appears to have limited the abundance of the species locally. In general, *B. chinensis* has not been seen as problematic in North America where densities are relatively low. However, the species has been seen to reach high densities in the Eijsder Beemden (9.02 $ind.m^{-2}$). At these densities and observed levels of biomass, and taking into account the filtration capacity of the species, ecosystem effects can be expected. The effect of grazing by *B. chinensis* remains unknown. As the risks associated with extremely high densities of the species are largely unknown, the lack of perceived threat may be attributed to this lack of knowledge. Quantitative information on the current and future economic losses and costs of *B. chinensis* is not available for the EU. *B. chinensis* currently displays a limited recorded distribution in the EU, and is of limited value to the trade in live animals as evidenced by the low number of retail outlets selling the species. Therefore, the current economic impact of *B. chinensis* in the EU is likely to be negligible. Medicinal extracts from the species may have an economic value in future.

3. Risk assessment

3.1 Risk assessment and classification with the Harmonia+ protocol

3.1.1 Classification for the current situation

Table 3.1 presents an overview of the updated risk assessment of *B. chinensis* derived using the Harmonia+ protocol. The expert team exchanged arguments relating to risk scores and came to a consensus. Evidence supporting the risk classification is given in the following paragraphs. The risk scores and confidence levels relate to both the current and expected future situations in the European Union.

Probability of species introduction

The probability that individuals of *B. chinensis* will enter the EU's wild from outside the EU through natural pathways within the time span of a decade is scored low with a high confidence level. The probability that the species will be introduced into the EU's wild by unintentional human actions is scored high with a medium confidence level. During the last 12 years the species has been introduced 18 times to the EU's wild, resulting in a high probability of introduction (> 10 events per decade). Based on current knowledge it is impossible to differentiate between unintentional and intentional introductions with absolute certainty resulting in a medium confidence score. However, there are several indications that unintentional introductions may have occurred. A number of fish wholesalers that use outdoor storage ponds are located in areas where crayfish species (*O. virilis* and *P.* cf. *acutus*) and *B. chinensis* have been simultaneously recorded in the Netherlands, it has been suggested that *B. chinensis* may have been accidently imported to the EU as a contaminant with intentionally imported crayfish species from North America. Moreover, *B. chinensis* may have been unintentionally introduced into the river Laak during maintenance of the ponds and aquaria associated with garden centres (§ 2.2.2; 2.2.3). However, the frequency of introductions via these potential pathways is unknown. In the future unintentional introductions are expected to increase due to increased shipping, dredging and maintenance of water systems. The probability that the species will be introduced into the EU's wild by intentional human action is scored high with a medium confidence level. The species has been recorded 18 times in the EU since 2007 (18 times in 12 years, approximately), several of these records are thought to have originated from isolated introductions occurring as the result of the trade in live animals and intentional disposal from aquaria (§ 2.2.1). However, based on current knowledge, it is impossible to differentiate between unintentional and intentional introductions resulting in a medium confidence score.

Probability of establishment

The climate and habitat of the EU are scored as optimal for the establishment of the species, with high confidence. At least one established reproducing population has been present for more than 10 years, which classifies the species as '2b Introduced:

10-100 years independent survival' according to the Dutch species register (Nederlands Soortenregister, 2018).

Table 3.1: Consensus risk scores for Chinese mystery snail (*Bellamya chinensis*) with confidence levels for both the current and future situations in the European Union, using the Harmonia+ protocol.

Context		
A01. Assessor(s)	**Consensus scores of four experts**	
A02. Species name	**Chinese mystery snail (Bellamya chinensis)**	
A03. Area under assessment	**European Union**	
A04. Status of species in area	**Alien and established within the area's wild**	
A05. Potential impact domain	**Environmental domain**	
Risk category	**Risk**	**Confidence**
Introduction		
A06. Probability of introduction by natural means	Low	High
A07. Probability of introduction by unintentional human actions	High	Medium
A08. Probability of introduction by intentional human actions	High	Medium
Establishment		
A09. Climate for establishment	Optimal	High
A10. Habitat for establishment	Optimal	High
Spread		
A11. Dispersal capacity within the area by natural means	Very high	Low
A12. Dispersal capacity within the area by human actions	High	Medium
Impacts: environmental targets		
A13. Effects on native species through predation, parasitism or herbivory	Medium	Low
A14. Effects on native species through competition	Medium	Medium
A15. Effects on native species through interbreeding	No / very low	High
A16. Effects on native species by hosting harmful parasites or pathogens	Very low	Medium
A17. Effects on integrity of ecosystems by affecting abiotic properties	Medium	Medium
A18. Effects on integrity of ecosystems by affecting biotic properties	Medium	Medium
Impacts: plant targets		
A19. Effects on plant targets through herbivory or predation	Inapplicable	High
A20. Effects on plant targets through competition	Inapplicable	High
A21. Effects on plant targets through interbreeding	Inapplicable	High
A22. Effects on integrity of cultivation systems	Very low	High
A23. Effects on plant targets by hosting harmful parasites or pathogens	Inapplicable	High
Impacts: animal targets		
A24. Effects on animal health or production through parasitism or predation	Very low	Low
A25. Effects on animal health or production by properties hazardous upon contact	Very low	High
A26. Effects on animal health or production by parasites or pathogens	Low	Low
Impacts: human health		
A27. Effects on human health through parasitism	Inapplicable	High
A28. Effects on human health by properties hazardous upon contact	Very low	High
A29. Effects on human health by parasites or pathogens	Low	Low
Impacts: other targets		
A30. Effects by causing damage to infrastructure	High	Low
Ecosystem services		
A31. Effects on provisioning services	Moderately positive	Medium
A32. Effects on regulation and maintenance services	Neutral	Medium
A33. Effects on cultural services	Moderately negative	Medium
Effects of climate change		
A34. Introduction	No change	Medium
A35. Establishment	Increase moderately	High
A36. Spread	Increase moderately	Low
A37. Impacts: environmental targets	No change	Medium
A38. Impacts: plant targets	No change	Medium
A39. Impacts: animal targets	No change	Medium
A40. Impacts: human health	No change	Medium
A41. Impacts: other targets	No change	Medium

A climate match between the North American non-native range of *B. chinensis* and the EU shows that particularly Eastern Europe is climatically suitable for the species. However, *B. chinensis* has established persistent populations outside the limits of this climate match in the Netherlands and Belgium suggesting a broader climatic tolerance. These regions are classified within the Atlantic biogeographical region of the EU which extends to Western Denmark, Germany, France, the far north of Spain, the United Kingdom and Ireland, and some coastal regions in Norway. *B. chinensis* favours eutrophic lakes, ponds and low velocity streams and rivers with a soft substrate. EU habitat type 3150 is expected to be the most suitable habitat for the establishment of this species. Moreover, running waters EU habitat types 3200 will be partly suitable for future establishment of *B. chinensis* (slow flowing and shallow littoral zones in freshwater sections of lowland rivers and streams). Both habitat types (HT3150 and HT3200) are widely available within the EU (See § 2.3.6).

Probability of spread

The capacity of the species to disperse within the EU by natural means is scored very high, with low confidence. The snail was found to crawl with a natural dispersal rate of 0.1 kilometres·year^{-1} measured during field surveys of shallow lakes in the floodplain Eijsder Beemden, the Netherlands (§ 2.4.2). However, as the species occurs at locations connected to the rivers Meuse and IJssel, can withstand flow velocities of up to 54 cm.s^{-1} (§ 2.3.3), and may be carried in the water current, the capacity to disperse by natural means is potentially very high. Moreover, if the species were to become widespread, the potential for natural long distance dispersal via waterfowl and aquatic mammals becomes very high. A low confidence level is given because these types of natural dispersal have, at the time of writing, not been validated with field data. The risk of spread within the EU by human actions is scored high, with medium confidence. It is possible that dispersal by human action has occurred at several locations in the Netherlands. However, the frequency at which dispersal via human action occurs in the EU is unknown. Secondary spread may be facilitated by boater interactions between invaded and non-invaded lakes via bait-buckets, live wells, fishing gear, and the boat itself (§ 2.4.2).

Environment

The probability of effects of *B. chinensis* on native species through predation, parasitism or herbivory is medium, considering that mesocosm experimentation suggests that *B. chinensis* may decrease algal biomass. However, this risk score is given with low confidence as these effects have not been observed in nature and that the risks associated with high densities of the species are largely unknown (§ 2.5.1). The probability of effects of *B. chinensis* on native species through competition is medium, with medium confidence. Mesocosm experimentation suggests that *B. chinensis* may reduce native snail populations through competitive exclusion. Also, mesocosm experiments have shown that when introduced with invasive crayfish species, synergistic effects, such as reduced abundance of native snails occurs due to selective predation by crayfish on the native species. However, field studies have not yet confirmed any negative impacts on native gastropod assemblages (§ 2.5.1). There is no risk of interbreeding as there is an absence of native congeners in the EU (§ 2.5.1). Therefore, this risk classification is assigned a high confidence level.

The probability of hosting harmful parasites is assessed as very low, with medium confidence. Only the commensalistic or parasitic worm *C. limnaei limnaei* has been sampled from snails removed from Boven-Hardinxveld in the Netherlands. This species is very common in the Netherlands and exists in a wide spectrum of freshwater ecosystems so no additional risk relating to this parasitic species is expected. However, individuals of *B. chinensis* sampled at Boven-Hardinxveld may not be representative for the entire EU. Moreover, information regarding parasites originating from outside the EU is conflicting (§ 2.5.1). In addition, schistosomiasis is not entered in the Wildtool list which is used by Harmonia$^+$ protocol to prioritize wildlife-borne pathogens to different target groups within Belgium (Tavernier et al., 2011).

The probability of impact on the abiotic properties of ecosystems is expected to be medium with a medium confidence. The probability of impact on biotic properties is classified as medium with a medium confidence. Abiotic and biotic impacts of *B. chinensis* have been found, though risks associated with high densities of *B. chinensis* are largely unknown, limiting the confidence levels that relate to the risk classifications. *B. chinensis* has a filtration rate that is similar to invasive *D. polymorpha*, *D. rostriformis bugensis* and *L. fortunei*. Moreover, changes to the ratio of nutrients in the water column as a result of *B. chinensis* have been observed experimentally. *B. chinensis* may exert bottom-up control in some food webs due to high levels of phytoplankton filtration. A slight alteration in the composition of the microbial community may occur as a result of *B. chinensis*' feeding behaviour or excretion products. Information on biotic properties has mainly been suggested by researchers or resulted from modelling studies leading to a lower confidence score. Risk scores for both abiotic and biotic impacts were deemed medium since a sudden disappearance of the species would result in a recovery of both the abiotic and biotic conditions.

Plant crops
The criterion on effects through herbivory, competition, interbreeding, parasites or pathogens is inapplicable. The probability of effects on the integrity of cultivation systems is very low with a high confidence level. *B. chinensis* does not feed on plants.

Human health
The category considering effects of *B. chinensis* on human health through parasitism is inapplicable. This is also the case for effects due to properties that are hazardous upon contact, but the Harmonia$^+$ protocol does not provide an 'inapplicable' option for this criterion and therefore it was scored as very low with a high level of confidence. The probability of effects by parasites or pathogens is scored as low with a low confidence. There is a risk that if eaten following inadequate cooking, the species may transfer parasites that are harmful to humans. However, even if densities of *B.*

chinensis were to achieve high levels in the EU, the frequency with which this species is consumed by humans in the EU will probably remain at the same low level. Therefore, the probability of parasitic transfer will also remain low. Moreover, limited evidence from the Netherlands and other parts of the species' introduced range suggests that parasitic load is low and consists of species already widespread in the EU (§ 2.5.1).

Infrastructure
The category 'effects of *B. chinensis* on infrastructure' is scored high with a low level of confidence considering that *B. chinensis* reduces the aesthetic, recreational and ecotourism values of recreational infrastructure when large scale mortality occurs in rivers and lakes that have recreational functions such as bathing (§ 2.5.5 and § 2.5.6). Cyclical die offs of *B. chinensis* have been reported and local residents have indicated that they viewed *B. chinensis* as a nuisance due to dead and decaying snails that form large windrows on shore (Bury et al., 2007). In addition, the species may clog water intake pipes and inhibit water flow.

Ecosystem services
Effects on ecosystem services are scored as moderately positive in the case of provisioning services, neutral in the case of regulation and maintenance services, and moderately negative in the case of cultural services. The positive effect on provisioning services results from the use of *B. chinensis* as an animal food product providing an alternative or supplementary food source, and the extraction of pharmaceuticals. These are likely to outweigh the negative effects on capture fisheries (§ 2.5.5). The overall effect on regulation and maintenance services is neutral due to conflicting effects resulting from high filter feeding rates and benthic-pelagic coupling that may lead to increased water clarity, but also encourages (extensive) growth of submerged water plants. Moderately negative impacts on aesthetic values, and recreation and ecotourism s are expected when large scale mortality occurs in rivers and lakes (§ 2.5.6).

Risk classification
The calculated invasion score (geometric mean of all introduction, establishment and spread scores; table 3.1) is high (Table 3.2). The overall impact score is calculated on the basis of the maximum impact score. This score is high due to the high score allocated to other targets, specifically the probability of damage to infrastructure. The overall risk score of *B. chinensis* to the European Union is classified as medium. This score is derived by multiplying the invasion score with the impact score.

3.1.2 Classification for future situation

Climate change may moderately increase the risk of secondary spread because of a potential increase in recreational activities such as boating and fishing that are dispersal vectors for *B. chinensis*. However, this moderate increase is stated with a low degree of confidence. A moderate increase in establishment risk may occur at the northern most limits of *B. chinensis*' climate range if the 12 °C limit in water temperature for reproduction is exceeded more frequently (§ 2.3.4; table 2.4). This moderate increase in establishment is stated with a high degree of confidence. However, introduction and impact scores for this species are expected to remain the same (Table 3.1 and 3.2).

Table 3.2: Risk classification and maximum impact scores for *Bellamya chinensis* with confidence levels in the European Union calculated with the online version of the Harmonia+ protocol.

Risk category	Risk classifciation	Risk score	Confidence	Confidence score
Introduction[1]	High	1.00	High	1.00
Establishment[1]	High	1.00	High	1.00
Spread[1]	High	1.00	Medium	0.50
Impacts: environmental targets[1]	Medium	0.50	Medium	0.50
Impacts: plant targets[1]	Low	0.00	High	1.00
Impacts: animal targets[1]	Low	0.25	High	1.00
Impacts: human health[1]	Low	0.25	High	1.00
Impacts: other targets[1]	High	0.75	Low	0.00
Invasion score[2]	High	1.00	NA	NA
Impact score	High	0.75	NA	NA
Risk score (Invasion x Impact)	High	0.75	NA	NA

3.2 Risk assessment and classification with the ISEIA protocol

3.2.1 Classification for the current situation

The expert team discussed the risk scores of *B. chinensis* and came to a consensus. The experts allocated a high risk score (score 3) to the category dispersion potential and invasiveness and a high score (score 3) to the category colonisation of high conservation value habitats (Table 3.3).

The categories adverse impact on native species and alteration of ecosystem functions were both categorised as likely (score 2). The total score for the environmental risk of this species is ten. *B. chinensis* is classified as a B1 species according to the list system proposed by the Belgian Forum on Invasive Species (BFIS; Figure 3.1). This is because of its medium environmental risk and recorded distribution in the EU (isolated populations). *B. chinensis* qualifies for the watch list according to the BFIS list system.

Dispersion potential or invasiveness

The risk score is 3 (**High**). The dispersal potential of a species is expressed by combining both the spread and reproduction potential. Except when assisted by man, *B. chinensis* does not colonise remote places. A natural crawling dispersal rate of 0.1 kilometres·year^{-1} was calculated during field surveys of shallow lakes in the floodplain

Eijsder Beemden, the Netherlands. However, natural downstream dispersal may be high during extreme river discharges, and waterfowl and aquatic mammals may carry the species across large distances. Therefore, the natural dispersal rate can be more than 1 km·year^{-1} and complies with the ISEIA criterion for a high dispersal risk.

Table 3.3: Consensus risk scores for alien *Bellamya chinensis* for the current and future situations, including the potential effects of climate change, for the European Union, using the ISEIA protocol.

Risk category		Consensus scores
Dispersion potential and invasiveness		3
Colonisation of high conservation value habitats		3
Direct or indirect adverse impacts on native species		2
1. Predation/herbivory	2 (Likely)	
2. Interference, exploitation competition	2 (Likely)	
3. Transmission of parasites and diseases	DD	
4. Genetic effects (e.g., hybridisation / introgression with natives)	1 (Unlikely)	
Direct or indirect alteration of ecosystem functions		2
1. Modification of nutrient cycling or resource pools	2 (Likely)	
2. Physical modifications of habitat	2 (Likely)	
3. Modification to natural succession	DD	
4. Disruption to food webs	2 (Likely)	
Total score		10
Range of spread		Isolated populations
Risk classification		B1

The distribution of records of *B. chinensis* in the Netherlands suggests a number of separate introduction events rather than secondary dispersal from an initial point of introduction, though dispersal is expected to increase in the future. This type of distribution pattern is also reflected in other parts of the snail's introduced range, i.e., North America, where the species is widespread but densities are also generally low. However, the species has a wide climate tolerance and suitable habitat is widely available in the EU. Moreover, the species can become locally invasive because of a strong reproduction potential. The species has reached high densities in Oregon, North America where management measures resulted in the deaths of approximately 27,000 snails in two ponds. In an isolated lake and a hydrologically connected lake at the Eijsder Beemden in the Netherlands, adult population size was found to range roughly between 26,000 - 38,000 and 443,000 - 460,000, respectively. According to the Köppen-Geiger climate zones, this area is climate matched with south-east Spain, large parts of Italy and Greece.

Colonization of habitats with high conservation value

The risk score is 3 (**High**). *B. chinensis* will often colonise high conservation value habitats that satisfy its habitat requirements (§ 2.3.4), i.e., most of the sites of a given habitat are likely to be readily colonised by the species when source populations are present in the vicinity. Currently, *B. chinensis* is mainly limited to sites that are not designated as habitats with high conservation value in the Netherlands. The species has mostly been introduced and is limited to areas near human populations and favours eutrophic habitats. However, the species has been recorded in two Natura

2000 areas, one in the Netherlands (the river IJssel floodplain, south of the town of Deventer), and one in Belgium (headwaters of the river Grote Nette near Zammelsbroek, Langdonken and Goor). The species is able to disperse rapidly downstream in river currents and, therefore, will probably spread from these locations to other aquatic habitats with high conservation value.

Adverse impacts on native species
The risk score is 2 (**Likely**). *B. chinensis* likely causes adverse effects on native species with respect to the subsection predation/herbivory and interference and exploitation competition. There is no available evidence of adverse effects on native species due to predation or herbivory from field observations in the EU or from climatically similar regions. Evidence from mesocosm experiments in North America indicates that plankton is removed from the water column due to the high filtering capacity of this species. *B. chinensis* filtration capacity is similar to that of other species of invasive molluscs known for their impacts on plankton communities, i.e., *D. polymorpha*, *D. rostriformis bugensis* and *L. fortunei.* Additionally, mesocosm experiments have suggested that *B. chinensis* may out-compete native snail species. However, it is unknown what effects high densities of *B. chinensis* may have on plankton and native snail communities in the EU. There is insufficient data to allow an expert judgement of the probability of adverse effects of parasites and diseases carried by *B. chinensis* (DD). Conflicting evidence that suggests that parasites may impact other species in the native range, that parasitic load is low in *B. chinensis* in its North American range, and limited evidence of low parasitic loads within individuals found at one location in the Netherlands, prevents clear conclusions from being drawn. According to expert knowledge, no congener of *B. chinensis* occurs in the EU, therefore, genetic effects such as hybridisation or introgression with native species are unlikely.

Alteration of ecosystem functions
The risk score is 2 (**Likely**). *B. chinensis* likely causes alterations to ecosystem functions with respect to the subsections modification of nutrient cycling or resource pools, physical modifications of habitat, and disruption to food-webs. The densities found at Eijsder Beemden indicate an influence on nutrient cycling due to high filtration capacity that is reversible. According to expert opinion, modification of nutrient cycling or resource pools will certainly occur if *B. chinensis* reaches high population densities (D.M. Soes, pers. comm.). However, the effects on ecosystem functioning are reversible, resulting in a risk score of 2. There is a discrepancy between scientific literature and risk assessments in North America with respect to perceived risk. In the literature, the species is often referred to as having a limited impact whereas two existing risk assessments classify the impact of the species as high (New York Invasive Species Information, 2017; Jacobs & Keller, 2017). There is insufficient data to allow an expert judgement of the risk of modification to natural succession (DD).

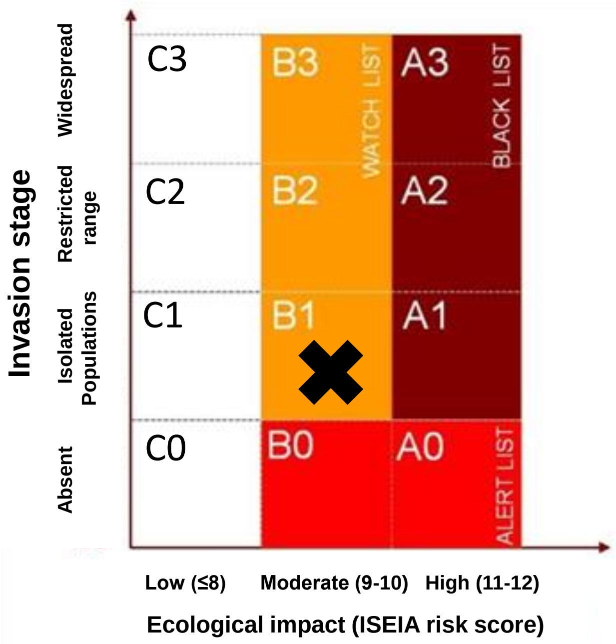


Figure 3.1: The risk classification of the alien Chinese mystery snail (*Bellamya chinensis*) for the current situation in the European Union according to the BFIS list system.

3.2.2 Classification for the future situation

The future risks of *B. chinensis* were also assessed and classified using the ISEIA protocol. It is unlikely that future climate change will result in limitation of the species in the EU. Therefore, the ecological impact in the future situation is expected to remain the same as the ecological impact in the current situation (Score 10, moderate risk score; table 3.3). However, if management policy remains the same, *B. chinensis* is expected to become widespread in the EU. Therefore, *B. chinensis* is classified as a B3 species in the future situation (BFIS; Figure 3.2). *B. chinensis* qualifies for the watch list according to the BFIS list system in the future situation.

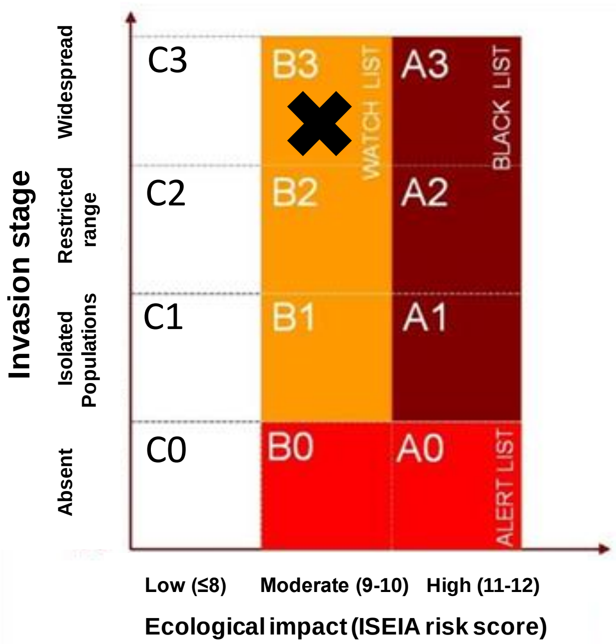


Figure 3.2: The risk classification of the alien Chinese mystery snail (*Bellamya chinensis*) for the future situation in the European Union according to the BFIS list system.

3.3 Other available risk assessments

Table 3.4 summarizes available risk assessments and classifications of *B. chinensis*. Two previous assessments assessing the risk of *B. chinensis* were carried out by Breedveld (2015). Breedveld (2015) applied both the Harmonia$^+$ and ISEIA protocols using the same information. The risk prioritisation using the ISEIA protocol resulted in a score of 10 for *B. chinensis* (moderate environmental risk). This score is the same as the risk score allocated to *B. chinensis* during the ISEIA assessment in this report. However, Breedveld (2015) allocated a high (3) score to the subcategory interference and exploitation competition. Breedveld (2015) justifies this high score by stating:

> 'The reason this section scored a 3 was that there are indications for strong interference and exploitation competition from North America (Johnson et al. 2009; Olden et al. 2013), also the species is possibly a competitor for unionids in the Netherlands.'

However, we challenge this rationale because indications for strong interference and exploitation competition are based on mesocosm experiments only, and no direct evidence has been obtained from field observations. Moreover, there is no evidence to suggest that unionid mussels are or will be effected in the Netherlands. Due to the lack of direct evidence we allocated a likely (2) score to this category. Compared to Breedveld (2015), we allocate a high score to dispersal potential compared to the medium score given by Breedveld (2015). The rationale for this high score is that the natural dispersal of the species can be more than > 1 km.year due to riverine flow and transport by waterfowl and aquatic mammals.

The application of the Harmonia$^+$ protocol by Breedveld (2015) resulted in a low risk classification for the Netherlands whereas we classified *B. chinensis* as a high risk species using the same protocol. This difference in classification may be explained by Breedveld (2015) choosing to apply the average rather than maximum score when calculating the overall impact score. By applying the maximum impact score we follow the precautionary approach as prescribed by principle 15 of the Rio Declaration on Environment and Development (UNEP, 2017). Collas et al. (2017) and Matthews et al. (2017) used the Harmonia$^+$ protocol resulting in a medium impact for this species. Our score differs due to the very high natural dispersal potential allocated by us due to water flow, waterfowl, aquatic mammals and other vectors.

Another available risk classification found was for New York, USA. The outcome of this assessment indicates that the establishment of *B. chinensis* will pose a very high risk for negative effects on native biodiversity. Overall, the outcomes of this assessment present a higher risk than our own results. This may be clarified by the different criteria applied in the New York assessment and how these were scored. For example, the New York Invasive Ranking System also includes criteria that assess species traits and the effectiveness of potential management interventions.

In the state of Illinois, an assessment of aquatic invasive species impacts was made by sending a survey to aquatic ecologists in which respondents were asked to assign ecological impacts on a four point scale. *B. chinensis* received an average score of 3, which classified its impact as high (species where classified as high when the average score was between 2.5 and 3.49).

Table 3.4: Other available risk assessments of the Chinese mystery snail (*Bellamya chinensis*).

	New York, USA	Illinois, USA	The Netherlands	The Netherlands	The Netherlands	The Netherlands
Scope	Risk prioritisation	Impact assessment	Risk assessment	Risk prioritisation	Risk assessment	Risk prioritization
Method	New York fish and aquatic invertebrate invasiveness ranking system	Survey of Illinois aquatic ecologists regarding impact of several species	Harmonia^{+}	ISEIA	Harmonia^{+}	ISEIA
Risk classification	Very high	High	Low	Moderate (10)	Medium	Moderate (9)
Source	New York Invasive Species Information (2017)	Jacobs and Keller (2017)	Breedveld (2015)	Breedveld (2015)	Collas et al.(2017); Matthews et al. (2017)	Matthews et al. (2017)

4. Discussion

4.1 Classification of risks

The expert team classified *B. chinensis* as an alien species with a high risk of invasiveness for the EU when the maximum score per impact category is considered. Although there are no data proving that the species is currently invasive and has much impact in Europe, the species has the potential to become problematic due to the high probability of human introduction, optimal habitat and climate matches, and high probability of impacts to infrastructure. Evidence of negative ecological effects is limited to mesocosm experiments from North America which suggests that *B. chinensis* may outcompete native snail species, increase water clarity and reduce algal biomass due to a high filtration rate, and increase the N:P balance and benthic-pelagic coupling. Densities found in the invaded range in the European Union make it highly likely that the species has a negative effect on abiotic and biotic properties. Though, these effects are in theory easily reversible. A lack of recorded ecological impacts of *B. chinensis* on North American aquatic ecosystems, including the Great Lakes, may be related to the generally low species densities observed there.

Since it was first recorded in the Netherlands in 2007, 18 records of *B. chinensis* have been made in the EU. In total, 17 of these records are sited in the Netherlands and one recent record is located in Belgium. The scattered distribution in these two countries suggest that isolated introductions from aquaria and potential escape from open storage ponds owned by fish wholesalers have been the main pathways of introduction to the EU. This theory is further supported by records in the Netherlands and Belgium that are located close to open storage ponds. The optimal climate and habitat matches between *B. chinensis*' North American introduced range, the species habitat requirement and the EU, suggests that continuing introductions (high propagule pressure) will result in a widespread distribution in the future situation.

4.2 Knowledge gaps and uncertainties

Establishment of populations could not be validated at some locations which were difficult to sample. eDNA can be used in future research to assess the occurrence of the species at these locations.

Assessments of the potential effects of *B. chinensis* are based on mesocosm experiments alone and there is a lack of research that analyses direct effects on the aquatic environment. This may be due to the generally low density of *B. chinensis* populations in its alien range and the density dependent nature of effects. For example, the potential for *B. chinensis* to increase water clarity and reduce planktonic biomass is based on a comparison of filtration rates with other mollusc species that are known to increase the clarity of the water column when present at high densities (*D. polymorpha*, *D. rostriformis bugensis* and *L. fortunei*). However, no direct measurements recording the impact of high densities of *B. chinensis* on water clarity have been made. Additionally, conflicting evidence from the native and introduced ranges with respect to the composition and abundance of parasites carried by *B. chinensis* reduce the certainty of classifications of risk to native species. Further research regarding the potential effect of predators and parasites on *B. chinensis* is recommended.

5. Conclusions

Current presence in the EU

- The alien *B. chinensis* has been recorded in Europe at 18 locations. In total, 17 of these locations are situated in the Netherlands, and one is located in Belgium.

Probability of introduction

- *B. chinensis* may be introduced to the EU as part of the international trade in live animals for aquaria and ponds and as a potential human food source.
- The discontinuous distribution in the Netherlands and Belgium suggests that the current distribution may be the result of multiple individual introduction events, possibly as a result of intentional release from domestic aquaria and ponds or unintentionally from open storage ponds associated with garden and pond centres.
- The recorded pathways of introduction and spread of *B. chinensis* to North America and Europe suggest that further spread within the EU may result from isolated introductions resulting in a discontinuous distribution.

Probability of establishment

- The introduced range of *B. chinensis* in the Netherlands and large portions of its introduced range in North America are climatically matched with large parts of Eurasia, including all EU countries.
- The introduced range of *B. chinensis* in the Netherlands and Belgium is within the Atlantic biogeographical region which is matched with Western Denmark, Germany, France, the far north of Spain, the United Kingdom and the Irish Republic, and some coastal regions in Norway.
- It is expected that *B. chinensis* will be able to establish in EU habitat types 3150. This habitat type in particular is expected to be at high risk of future colonisation by *B. chinensis.*
- It is unknown if current conservation practices encourage the establishment of *B. chinensis* in the EU. However, river and lake rehabilitation measures that increase the availability of suitable habitat may encourage the establishment of *B. chinensis* in the EU. Densities of *B. chinensis* in its North American invaded range were found to be higher in natural areas compared to rural regions suggesting that conservation practices might facilitate the spread of *B. chinensis*.

Probability of spread

- In the absence of management measures, populations of *B. chinensis* in the Netherlands and Belgium may serve as sources of secondary spread within the EU (e.g., to France and Germany).
- The species may disperse within the EU attached to equipment used in the maintenance of waterways, ship hulls, recreational boats and with fishing

equipment, via natural dispersal by waterfowl and aquatic mammals, and through limited natural dispersal by crawling.

Probability of impact

- A search of available literature found no evidence of current recorded or potential future impacts of *B. chinensis* in the EU. This may be related to the current limited distribution of the species, and the impact of predation on established populations which appears to have limited the abundance of the species locally.
- A discrepancy exists between scientific literature and risk assessments describing the perceived risks in North America. In several scientific papers the species is indicated to have a limited impact whereas two risk assessments for North America classify the potential impact of the species as high.
- Evidence of negative ecological effects is limited to mesocosm experiments from North America which suggests that *B. chinensis* may outcompete native snail species, increase water clarity and reduce algal biomass due to a high filtration rate, and increase the N:P balance and benthic-pelagic coupling. With densities found in the Eijsder Beemden similar to those applied in mesocosm experiments, these effects can potentially occur at this specific location. Additionally, there are indications that densities at other locations (e.g., Veengoot) may also be similar to those observed at Eijsder Beemden.
- Quantitative information on the current and future economic losses and costs of *B. chinensis* is not available for the EU. *B. chinensis* currently displays a limited recorded distribution in the EU, and is of limited value to the trade in live animals as evidenced by the low number of retail outlets selling the species. Therefore, the economic impact of *B. chinensis* in the EU is likely to be negligible.

Risk classification

- The expert team assigned an overall **high** risk classification for the risks of *B. chinensis* in the EU using the Harmonia$^+$ protocol and a **medium** risk using the ISEIA protocol.

- *B. chinensis* is currently present in isolated populations in the EU. According to the BFIS list system used in conjunction with the ISEIA protocol, *B. chinensis* classifies as a **B1** species and qualifies for the **watch list**. It is unlikely that future climate change will result in limitation of the species.

- The classification of *B. chinensis* by experts based on available knowledge resulted in the following risk scores according to the Harmonia$^+$ protocol:
  - Probability of introduction: **high** (Confidence: **high**);
  - Probability of establishment: **high** (Confidence: **high**);
  - Probability of spread: **high** (Confidence: **medium**);
  - Probability of environmental impact: **medium** (Confidence: **medium**)
    - Effects on native species through predation, parasitism or herbivory: **medium** (Confidence: **low**);

- Effects on native species through competition: **medium** (Confidence: **medium**);
- Effects on native species through interbreeding: **no / very low** (Confidence: **high**);
- Effects on native species through hosting harmful parasites or pathogens: **very low** (Confidence: **medium**);
- Effects on integrity of ecosystems by affecting abiotic properties: **medium** (Confidence: **medium**);
- Effects on integrity of ecosystems by affecting abiotic properties: **medium** (Confidence: **medium**);

- Probability of effects on plant cultivation: **low** (Confidence: **high**);
- Probability of effects on domesticated animals and livestock: **low** (Confidence: **high**);
- Probability of effects on public health: **low** (Confidence: **high**);
- Probability of other risk effects: **high** (Confidence: **low**).

Knowledge gaps

- Assessments of the potential effects of *B. chinensis* are based on mesocosm experiments alone and there is a lack of research that analyses direct effects on the aquatic environment. This may be due to the generally low density of *B. chinensis* populations in its alien range and the density dependent nature of effects. It is recommended that further research is carried out in order to address these knowledge gaps reduce uncertainties whilst carrying out future assessments.

- The presence of zoonotic agents in established populations of *B. chinensis* in the Netherlands and Belgium.

- There is limited information available on the natural spread of *B. chinensis* in the Netherlands and Belgium. Quantitative information on predation of this species in the European Union is lacking.

- Establishment of populations could not be validated at some locations which were difficult to sample. eDNA can be used in future research to assess the occurrence of the species at these locations.

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Appendix 1 – Materials and methods

A1.1 Risk analysis components

The present risk assessment of the Chinese mystery snail (*Bellamya chinensis*) in the European Union includes analyses of the probability of introduction, establishment and spread within the EU. Also the available literature on the ecological and socio-economic effects, impact on public health was analysed. The background information and data collected in the risk inventory are presented in chapter 2 and used as basis for the risk assessments and classification in chapter 3.

Subsequently, an ecological risk assessment and risk classification of the species in the EU was made using the Harmonia$^+$ protocol (D’hondt et al., 2015). The novel internet version of this protocol includes criteria for an ecological risk assessment as well as modules for the assessment of (potential) impacts on human health, infrastructure and ecosystem services, and a module to assess effects of climate change on the risks posed by alien species. The earlier version of Harmonia$^+$ was nearly compliant with criteria for risk assessment of IAS of EU-concern derived from Regulation 1143/2014 on the prevention and management of the introduction and spread of IAS (Roy et al., 2014). We assumed that the current internet version of Harmonia$^+$ is compliant with these criteria due to the addition of modules concerning the impacts on ecosystem services and the potential effects of climate change on future impacts of alien species.

In addition, a risk assessment was performed using the Invasive Species Environmental Impact Assessment (ISEIA) protocol (Branquart, 2009; Branquart et al., 2009; Vanderhoeven et al., 2015).

A1.2 Risk inventory

An extensive literature review was carried out to compile a science based overview of the current knowledge on taxonomy, habitat preference, introduction and dispersal mechanisms, current distribution, ecological impact, socio-economic impact and consequences for public health of the species. In addition, data on the current distribution in EU member states were acquired. In this risk inventory internationally published knowledge in scientific journals and reports was described. If relevant issues mentioned in the format for this risk inventory could not sufficiently be supported by knowledge published in international literature, ‘grey literature’ or ‘best professional judgement’ was used. In the latter case, this has been indicated in the report to clearly identify which arguments may be prone to discussion. Uncertainties and knowledge gaps are also addressed in the discussion.

A1.2.1 Literature review

The Web of Science and Google Scholar search engines were used to find general information on *B. chinensis* and more specific information on its distribution, tolerances, habitat characteristics and other aspects indicated by the search terms given in (Table A1.1). For the original risk assessment report all hits of the Web of science searches and the first 150 hits of the Google scholar searches were screened for relevance. A second search was performed for the update of the risk assessment report. The second search was performed on February the 6th, 2018 and used the previously indicated search terms and methodology, except results were filtered for the years 2017 and 2018.

Table A1.1. Search strategy to retrieve scientific literature on the invasion biology of the Chinese mystery snail (*Bellamya chinensis*).

Search engine	Search terms
Web of Science (All databases)	*Bellamya chinensis*, *Cipangopaludina chinensis*, Chinese mystery snail
Google Scholar	*Bellamya chinensis*, *Cipangopaludina chinensis*, Chinese mystery snail

A1.2.2 Data acquisition on current distribution

Scientific publications retrieved with search engines (Table A1.1) and online databases (i.e., Global Invasive Species Database, Invasive Species Compendium and NOBANIS; table A1.2) were used to acquire data on the current distribution of *B. chinensis* (native and introduced range).

Table A1.2: Overview of search engines and websites used during data acquisition.

Organisation / database	Web address
European Alien Species Information Network (EASIN)	https://easin.jrc.ec.europa.eu/
Invasive Species Compendium	http://www.cabi.org/isc/
Global invasive species database	http://www.issg.org/database/welcome/
Web of Science	http://apps.isiknowledge.com
DAISIE (European alien species information)	www.europe-aliens.org/
NOBANIS	http://www.nobanis.org/
GB non-native species secretariat	http://www.nonnativespecies.org/home/index.cfm
Invasive species in Belgium	http://ias.biodiversity.be/species/all
Integrated taxonomic information system (ITIS)	http://www.itis.gov/
Google	www.google.com
Google Scholar	http://scholar.google.nl/

A1.3 Risk assessment and classification

A1.3.1 Selection of risk assessment methods

One of the aims of this project is to provide insight into the risks of *B. chinensis* to biodiversity and ecosystems in the EU. Assessments of ecological risks were therefore required and it was decided to apply both the Harmonia^{+} and the ISEIA protocol for this purpose. In the current study, the Harmonia^{+} protocol was used as it includes the assessment of impacts on socio-economic aspects, public health, infrastructure and ecosystem services, as well as the effects of climate change on the establishment, spread, and impacts of alien species. Moreover, the Harmonia$^+$ protocol complies with the criteria of the EU regulation 1143/2014. The ISEIA protocol requires less detailed information on impacts to obtain a risk classification than Harmonia$^+$ and focuses on ecological impacts only. In the Netherlands, the ISEIA protocol has been most frequently used for the risk classification of alien species.

Harmonia$^+$ and ISEIA are protocols for risk screening and are primarily developed for assessing the negative effects of alien species. They do not consider positive effects, except the module on ecosystem services in the Harmonia$^+$ protocol. However, available information on positive effects of alien species has been included in the risk inventory (Chapter 2).

A1.3.2 Harmonia$^+$ ecological risk assessment protocol

The Harmonia$^+$ protocol includes procedures for the risk assessment of potentially invasive alien plant and animal species. This protocol stems from a review of the ISEIA protocol and incorporates all stages of invasion and different types of impacts. The online version of the Harmonia$^+$ protocol (D'hondt et al., 2015) was used for the risk assessment of *B. chinensis*. All risk scores were calculated using this online version. This risk assessment method comprises 41 questions grouped in the following modules:

A0. Context (assessor, area and organism);
A1. Introduction (probability of the organism to be introduced into the area);
A2. Establishment (does the area provide suitable climate and habitat);
A3. Spread (risks of dispersal within the area);
A4. Potential impact on the following subcategories:
- A4a. Environmental effects: wild animals and plants, habitats and ecosystems;
- A4b. Effects on cultivated plants;
- A4c. Effects on domesticated animals;
- A4d. Effects on human health;
- A4e. Effects on infrastructure;

A5a. Effects on ecosystem services;
A5b. Effects of climate change on the impact of the organism.

Each module contains one or more risk assessment questions and provides options for risk scores in each question. The protocol provides guidance for all questions and includes explanations and examples that serve as a reference for attributing risk scores.

Table A1.3 shows the formulas used for the calculation of various risk scores. The protocol allows the assignment of various weighing factors to impact categories (i.e., weighing risks within and between categories). In order to prevent averaging of risks and to keep the highest score of each risk category visible, the highest score was always used to calculate final effect scores for a specific impact category. This 'one out all out' principle has also been used in other risk assessments of alien species (e.g., in ISEIA and the EPPO prioritizing schemes) and other policy domains (such as ecological status assessments of water bodies according to the European Water Framework directive). The default value 1 was always used for weighing between various impact categories (i.e., equal weighing). The product of the introduction, establishment and spread was used to calculate the invasion score. The maximum of the different impact scores was used to calculate the aggregated impact score.

Table A1.3: Concepts and definitions for risk assessments and classifications of alien species with the Harmonia+ protocol (D'hondt et al., 2015).

Conceptual framework
Invasion= *f*(*Introduction; Establishment; Spread; Impact*$_{a\text{-}g}$)
Risk = *Exposure* x *Likelihood* x *Impact*

Invasion = risk?
Exposure ≡ f_1(*Introduction;Establishment;Spread*) = ***Invasion score***
Likelihood x *Impact* ≡ f_2($Impact_a$; $Impact_b$; $Impact_c$; $Impact_d$; $Impact_e$; $Impact_f$; $Impact_g$) = ***Impact score***
a: environment (biodiversity and ecosystems); b: cultivated plants; c. domesticated animals; d. human health; e: other; f: ecosystem services; g: climate change

Total risk = *Exposure* x *Likelihood* x *Impact* ≡ f_3(*Invasion score*; *Impact score*) = **Invasion**

Mathematical framework
f_1 : (weighed) geometric mean or product
f_2 : (weighed) arithmetic mean or maximum
f_3 : product

The degree of certainty associated with a given risk was scored as a level of confidence. The level of confidence of risk scores has been consistently reported using low, medium and high, in accordance with the framework of Mastrandrea et al., (2010, 2011). Harmonia+ attributes values of 0, 0.5 and 1 to low, medium and high confidence, respectively, to calculate confidence levels for various impact categories. The cut-off values for risk scores and confidence levels used for the risk classification of *B. chinensis* in the EU are summarized in Tables A1.4 and A1.5.

Table A1.4: Cut-off values for risk scores used for the risk classification of the Chinese mystery snail (*Bellamya chinensis*) in the EU, using the Harmonia+ protocol.

Colour code risk	Risk classification	Risk score*
(green)	Low	< 0.33
(orange)	Medium	0.33 ≤ RS ≤ 0.66
(red)	High	> 0.66

*: Arbitrary cut-off values.

Table A1.5: Confidence levels used for the risk classification of the Chinese mystery snail (*Bellamya chinensis*) in the EU, using the Harmonia+ protocol.

Colour code confidence	Confidence	Confidence score*
	Low	< 0.33
	Medium	0.33 ≤ CS ≤ 0.66
	High	> 0.66

*: Arbitrary cut-off values.

A1.3.3 ISEIA ecological risk assessment protocol

The ISEIA protocol assesses risks associated with dispersion potential, invasiveness and ecological impacts only (Branquart et al., 2009).

The ISEIA protocol contains twelve criteria that match the last steps of the invasion process (i.e., the potential for spread establishment, adverse impacts on native species and ecosystems). These criteria are divided over the following four risk sections: (1) dispersion potential or invasiveness, (2) colonisation of high conservation habitats, (3) adverse impacts on native species, and (4) alteration of ecosystem functions. Definitions for risk classifications relating to the four sections contained within the ISEIA protocol are presented in Table A1.6. Section 3 contains sub-sections referring to (i) predation / herbivory, (ii) interference and exploitation competition, (iii) transmission of diseases to native species (parasites, pest organisms or pathogens), and (iv) genetic effects such as hybridization and introgression with related native species. Section 4 contains sub-sections referring to (i) modifications in nutrient cycling or resource pools, (ii) physical modifications to habitats (changes to hydrological regimes, increase in water turbidity, light interception, alteration of river banks, destruction of fish nursery areas, etc.), (iii) modifications to natural successions and (iv) disruption to food-webs, i.e., a modification to lower trophic levels through herbivory or predation (top-down regulation) leading to ecosystem imbalance.

Each criterion of the ISEIA protocol was scored by six experts (§ 1.3.4). The scores range from 1 (low risk) to 2 (medium risk) and 3 (high risk). If information obtained from the literature review was insufficient for the derivation of a risk score, then the risk score was based on best professional judgement and field observation leading to a score of 1 (unlikely) or 2 (likely). If no answer could be given to a particular question (no information) a score of 1 was given (DD - deficient data). This is the minimum score that can be applied in any risk category. In cases with data or knowledge limitations, periodical review of new literature and updates of risk scores will be recommended. Finally, the highest score within each section was used to calculate the total ISEIA risk score for the species.

Consideration was given to the future situation assuming no changes in management measures that will affect the invasiveness and impacts of this invasive plant. The risk assessment and classification of *B. chinensis* for the future situation was performed, with the assumption of a temperature increase of 2 ^{o}C in 2050, which reflects the IPCC scenarios for Climate Change (IPCC, 2013) and unchanged policies on exotics in the EU member states.

Subsequently, the Belgian Forum Invasive Species (BFIS) list system for preventive and management actions was used to categorise the species of concern (Branquart et al., 2009). This list system was designed as a two dimensional ordination (Ecological impact * Invasion stage; Figure A1.1). The BFIS list system is based on guidelines proposed by the Convention on Biological Diversity (CBD decision VI/7) and the European Union strategy on invasive alien species.

Table A1.6: Definitions of criteria for risk classifications per section used in the ecological risk assessment protocol (Branquart et al., 2009).

1. Dispersion potential or invasiveness risk	
Low	The species does not spread in the environment because of poor dispersal capacities and a low reproduction potential.
Medium	Except when assisted by man, the species doesn't colonise remote places. Natural dispersal rarely exceeds more than 1 km per year. However, the species can become locally invasive because of a strong reproduction potential.
High	The species is highly fecund, can easily disperse through active or passive means over distances > 1km / year and initiate new populations. Are to be considered here plant species that take advantage of anemochory, hydrochory and zoochory, insects like *Harmonia axyridis* or *Cemeraria ohridella* and all bird species.
2. Colonisation of high conservation habitats risk	
Low	Population of the alien species are restricted to man-made habitats (low conservation value).
Medium	Populations of the alien species are usually confined to habitats with a low or a medium conservation value and may occasionally colonise high conservation habitats.
High	The alien species often colonises high conservation value habitats (i.e., most of the sites of a given habitat are likely to be readily colonised by the species when source populations are present in the vicinity) and makes therefore a potential threat for red-listed species.
3. Adverse impacts on native species risk	
Low	Data from invasion histories suggest that the negative impact on native populations is negligible.
Medium	The alien is known to cause local changes (<80%) in population abundance, growth or distribution of one or several native species, especially amongst common and ruderal species. The effect is usually considered as reversible.
High	The development of the alien species often causes local severe (>80%) population declines and the reduction of local species richness. At a regional scale, it can be considered as a factor for precipitating (rare) species decline. Those alien species form long standing populations and their impacts on native biodiversity are considered as hardly reversible. Examples: strong interspecific competition in plant communities mediated by allelopathic chemicals, intra-guild predation leading to local extinction of native species, transmission of new lethal diseases to native species.
4. Alteration of ecosystem functions risk	
Low	The impact on ecosystem processes and structures is considered negligible.
Medium	The impact on ecosystem processes and structures is moderate and considered as easily reversible.
High	The impact on ecosystem processes and structures is strong and difficult to reverse. Examples: alterations of physicochemical properties of water, facilitation of river bank erosion, prevention of natural regeneration of trees, destruction of river banks, reed beds and / or fish nursery areas and food web disruption.

Ecological impact of the species was classified into a group represented by the letters A, B or C, which was based on the total ISEIA risk score: low ecological risk score 4-8 (C), moderate ecological risk score 9-10 (B - watch list) and high ecological risk score 11-12 (A - black list) (Figure A1.1). This letter was then combined with a number representing the invasion stage: (0) absent, (1) isolated populations, (2) restricted range, and (3) widespread. A cross was used to indicate the risk classification of the assessed species within the BFIS system. A green cross indicates a low risk species that should not appear on any list within the BFIS system. A black cross indicates a species that should appear on either the watch, alert or black list of the BFIS system.

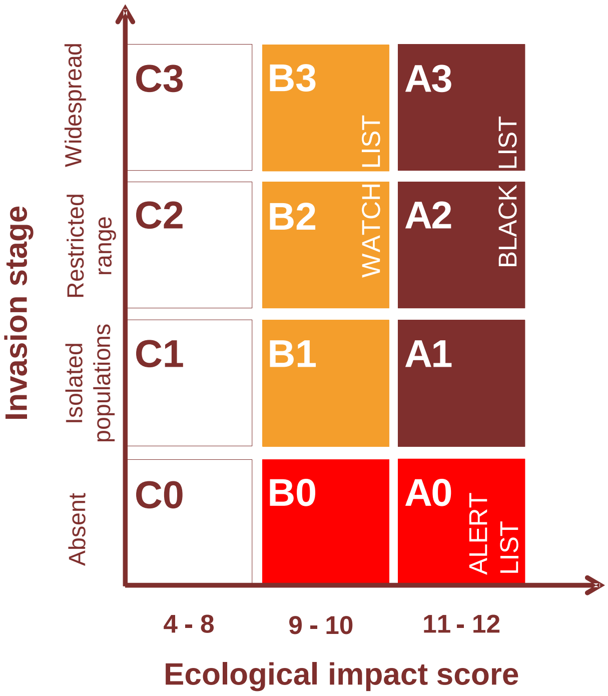

Figure A1.1: BFIS list system to identify species of most concern for preventive and mitigation action (Branquart et al., 2009; score 4-8: low risk; score 9-10: medium risk; score 11-12: high risk).

A1.3.4 Expert meeting on risk classification

The original risk assessments of *B. chinensis* have been performed by a team of five experts (F.P.L. Collas MSc, Dr. L. de Hoop, Prof. Dr. R.S.E.W. Leuven, Dr. G. van der Velde and Dr. J. Matthews), using the ISEIA and Harmonia⁺ protocol. Each expert thoroughly reviewed the risk inventory (knowledge document). Subsequently, experts independently assessed and classified current and future risks of *B. chinensis*, using both protocols. Future risks were determined with respect to the potential effects of climate change on the introduction, establishment, spread and impacts of the species.

Following the individual assessment of experts, the entire team met, elucidated differences in risk scores, discussed diversity of risk scores and interpretations of key information during a risk assessment workshop. Discussion during the workshop led to agreement on consensus scores and a risk classification relating to both protocols. The consensus scores, risk classifications and justifications for the scores were described in a draft report that was reviewed by the project team, assuring full agreement with the outcomes of the risk assessments.

The update of the risk assessment of *B. chinensis* has been performed by a team of four experts (F.P.L. Collas MSc, Prof. Dr. R.S.E.W. Leuven, Dr. G. van der Velde and Dr. J. Matthews), using the ISEIA and Harmonia$^+$ protocol. Updating the risk assessments consisted of a critical evaluation of the former risk assessment of Matthews et al. (2017a) based on the new information and results of field surveys carried out in 2017. Subsequently, the entire team met and discussed the proposed changes in risk scores during a workshop on February 27th 2018.

A1.3.5 Other available risk assessments and classifications

A specific literature search using Web of Science and Google (Scholar) was performed to retrieve other available risk assessments and classifications of *B. chinensis* (Table A1.1). Search terms applied were the scientific species name and English name combined with the following terms: risk, risk assessment, risk analyses and risk classification. The outcomes of these risk assessments and classifications were included in this report and compared for consistency with our risk classifications.

A1.4 Peer review by independent experts

The quality of the former risk assessment, Matthews et al. (2017a), was assured by an external peer review procedure. The final draft of that report was reviewed by two independent experts:

1. Prof. Dr. A.Y. Karatayev (Buffalo State University, Great Lakes Center, New York, USA).
2. Ir. D.M. Soes (Bureau Waardenburg BV, Culemborg, the Netherlands).

Both experts critically reviewed the available data and information described in the risk inventory as well as the outcomes of the risk assessments. Special attention was focused on the justification of the risk classification and relevant scientific uncertainties. Appendix 3 summarizes all comments of the reviewers and how their remarks and suggestions were dealt with in this risk assessment. All remarks and suggestions made by the peer reviewers were again checked and, when necessary, dealt with in the updated risk assessment.

Appendix 2 – Risk assessment for the Netherlands

The expert panel has also classified the risks of the Chinese mystery snail (*Bellamya chinensis*) for the Netherlands using the ISEIA protocol (Table A2.1). For a detailed methodological explanation of this assessment protocol see Appendix A1.3.3. Risk scores were allocated for both the current situation and for a future climate change situation.

Table A2.1: Risk classification of the Chinese mystery snail (*Bellamya chinensis*) using the ISEIA protocol for the current and future situation in the Netherlands.

Risk category		Consensus scores
Dispersion potential and invasiveness		3
Colonisation of high conservation value habitats		3
Direct or indirect adverse impacts on native species		2
1. Predation/herbivory	2 (Likely)	
2. Interference, exploitation competition	2 (Likely)	
3. Transmission of parasites and diseases	DD	
4. Genetic effects (e.g., hybridisation / introgression with natives)	1 (Unlikely)	
Direct or indirect alteration of ecosystem functions		2
1. Modification of nutrient cycling or resource pools	2 (Likely)	
2. Physical modifications of habitat	2 (Likely)	
3. Modification to natural succession	DD	
4. Disruption to food webs	2 (Likely)	
Total score		10
Range of spread		Isolated populations
Risk classification		B1

Present situation

The risk of dispersion and invasiveness was scored **high** (Table A2.1). Except when assisted by man, *B. chinensis* does not colonise remote places. A natural crawling dispersal rate of 0.1 kilometres·year^{-1} was calculated during field surveys of shallow lakes in the floodplain Eijsder Beemden, the Netherlands. However, natural downstream dispersal may be high during extreme river discharges and waterfowl and aquatic mammals may transfer the species across large distances. Therefore, the natural dispersal rate can exceed more than 1 kilometre·year^{-1} and complies with the ISEIA criterion of high risk for dispersion and invasiveness. The distribution of records of *B. chinensis* in the Netherlands suggests a number of separate introduction events rather than secondary dispersal from an initial point of introduction, though dispersal is expected to increase in the future. The species has a wide climate tolerance and suitable habitat is widely available in the Netherlands.

The risk of colonization of valuable habitats has been assessed **high** by the expert team. Currently, *B. chinensis* is mainly limited to sites in the Netherlands that are not designated as habitats with high conservation value (see § 2.5.1, figure 2.10). The species has mostly been introduced and is limited to sites near urban areas and favours eutrophic habitats. However, the species has also been recorded in one Natura 2000 area in the Netherlands (the river IJssel floodplain, south of the town of

Deventer). The species is able to disperse rapidly in fast flowing river currents and, therefore, will spread from these locations to other aquatic habitats with high conservation value.

The probability of negative effects on native species is classified '**likely**' (score 2). *B. chinensis* likely causes adverse effects on native species with respect to the subsections predation/herbivory, and interference and exploitation competition. There is no evidence of effects on native species from field observations in the Netherlands or climatically similar regions. However, evidence from mesocosm experiments in North America indicates that plankton is removed from the water column due to the high filtering capacity of this species and that *B. chinensis* may outcompete native snails. Densities at several locations in the Netherlands are similar to the densities used in the aforementioned mesocosm experiments implying that impacts are likely.

The probability of changes in ecosystem functions is classified '**likely**' (score 2). *B. chinensis* likely causes adverse effects on ecosystem functioning with respect to the subsections modification of nutrient cycling or resource pools, physical modifications of habitat and disruption to food webs. There is no available evidence of effects of *B. chinensis* on ecosystem functioning from field observations in the Netherlands or climatically similar regions. However, at several locations in the Netherlands the species reaches high densities implying that impacts are likely. *B. chinensis* has a filtration rate that is similar to invasive *D. polymorpha, D. rostriformis bugensis* and *L. fortunei*. Dreissenid mussels have high filtering rates and have great potential to decrease the phytoplankton biomass in ecosystems. High filtration rates may lead to increases in the transparency of the water column. Addition of *B. chinensis* during mesocosm experiments significantly increased the N:P molar ratio by approximately 25 % over control values.

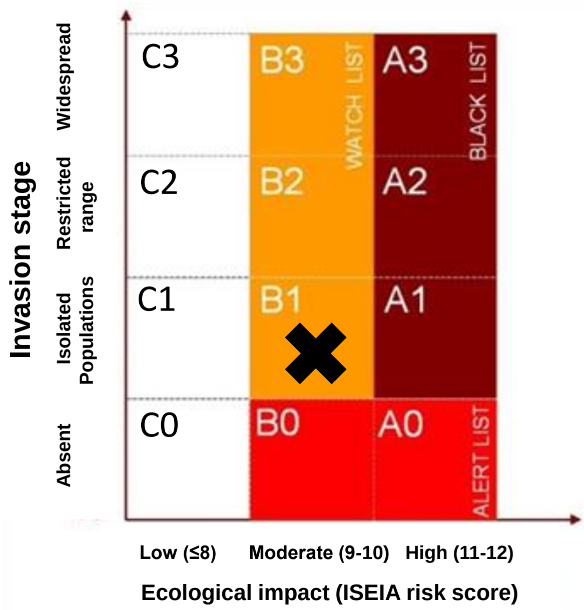

Figure A2.1: The risk classification of the alien Chinese mystery snail (*Bellamya chinensis*) for the current situation in the Netherlands according to the BFIS list system.

The overall risk classification is moderate (score 10) for the present situation. In view of the moderate risk of this species, which is recorded in isolated populations in the Netherlands, eligible for placement on the watch list of the BFIS system (Classification: B1; figure A2.1). The rationale for the allocation of risk scores to the Netherlands is the same as for the EU as a whole. Therefore, additional supporting information for this risk classification can be found in § 3.2.

Future situation

Climate change in the event of unchanged management policies will not influence the risk classification for the Netherlands. In the event of unchanged policy, the species will probably become widely spread in the Netherlands. In view of the moderate risk of this species, *B. chinensis* remains eligible for placement on the watch list of the BFIS system (Classification: B3; figure A.2.2).

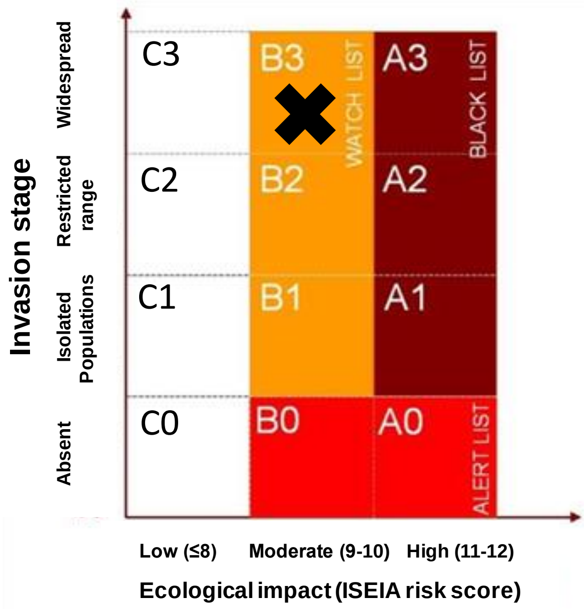

Figure A2.2: The risk classification of the alien Chinese mystery snail (*Bellamya chinensis*) for the future situation in the Netherlands according to the BFIS list system.

Comparison with risk classification for EU

The classification of the risks of *B. chinensis* for the Netherlands corresponds with the classification for the EU.

Appendix 3 – Quality assurance by peer review

The quality of this risk assessment was assured by an external peer review procedure. The independent experts Prof. Dr. A.Y. Karatayev (Buffalo State University, Great Lakes Center, New York, USA) and Ir. D.M. Soes (Bureau Waardenburg BV, Culemborg, the Netherlands) reviewed the final draft of this report. They assessed the available information used for the former risk assessments and the outcome of the assessments (Matthews et al., 2017a), including the justifications for the risk classifications and scientific uncertainties. As part of the update of the risk assessment, previous comments made by the reviewers were assessed taking into account newly available information.

The external reviewers emphasised the thoroughness and scientific rigour of the risk analyses. The reviewers delivered useful comments and suggestions for improvement of the risk inventory and assessment. All remarks and suggestions of the reviewers were implemented in the final version of this report. Specifically, references to Russia as the native range of *B. chinensis* were made more specific in the text by referring to 'Asiatic Russia in the Amur region' and by clearly indicating the area where the species is known to occur (Figure 2.4). In response to one reviewer's comment regarding potential misidentification of *B. chinensis*, descriptions of the native range of the species have been qualified in section § 2.3.1. Moreover, the reviewers comments led to useful additions to sections dealing with subspecies, substitute species, habitat preference and pathways of introduction and parasites. The maximum survivable temperature of adults of *B. chinensis* during acute heating in the laboratory was found to be 45°C by Wong et al. (unpublished data). However, this temperature was judged to be too high and potentially misleading by one of the reviewers. The duration of exposure to this high temperature was not quantified in the original reference material. In response to this, we emphasised that this temperature was obtained during one laboratory experiment the results of which were not peer reviewed. Moreover, we emphasise the acuteness and short-term nature of the exposure that probably does not reflect natural conditions, and the fact that the duration was not quantified. However, we have not removed the data because it suggests that high water temperature will not limit the further spread of this species as most waterbodies do not exceed 45°C in the EU. This reasoning has been added to the text in § 2.3.4.

With respect to the risk scoring and classification, one reviewer suggested that the 'environmental impact risk' derived using the Harmonia^{+} protocol should be low instead of medium, however, no rationale was given in support of this change. We are aware that *B. chinensis* is generally regarded as a low risk species in North America due to a lack of observed environmental impacts. However, two risk assessments performed in North America concluded that the species poses a high risk due to high scores for potential impacts (New York Invasive Species Information, 2017; Jacobs & Keller, 2017). In addition, we have decided to maintain our assessment of environmental impact risk at medium for the following reasons. Despite the lack of observed negative effects in its North American introduced range, the species has been shown to alter algal composition and outcompete native snail species during mesocosm experiments (Clark, 2009; Johnson et al., 2009; McAlpine et al., 2016). In addition, while *B. chinensis* is widely distributed in North America, its population density is generally low which may limit potential effects. The species is only present in isolated populations in the EU. However, densities have been found to be high at several locations in its European introduced range (This study; § 2.3.3), further research is recommended to explore potential effects at these locations. The Harmonia$^+$ protocol allocates the maximum score from all subcategories to the overall environmental impact risk. These indications, combined with the effects on algal biomass and native snail species observed during laboratory experimentation led to the allocation of a medium risk score to the environmental impact risk category.

The second reviewer suggested that the certainty of the high risk score applied to the category of Harmonia$^+$ relating to impacts on infrastructure should be low rather than high. The rationale given for this change was that there is no evidence to suggest that *B. chinensis* establishment leads to biofouling. Negative impacts on aesthetic values, recreation and ecotourism of recreational infrastructure are expected if large scale mortality occurs in recreational rivers and lakes (§ 2.5.6), however, again there is no evidence of this actually occurring. In view of this we have reduced the certainty of this risk score to low. The same reviewer stated that modification of nutrient cycling or resource pools will certainly occur if *B. chinensis* reaches high population densities, thereby suggesting that the 'likely' classification allocated to this section of the ISEIA assessment should be altered. However, this classification was applied because ISEIA only allows a 'likely' or 'unlikely' classification where expert judgement is used in cases of lack of evidence. In this case there was no available published evidence and the likely score was allocated as this is the 'maximum' allowable score in cases using expert opinion. Therefore, the classification remains unchanged.

Appendix 4 – Locations where *B. chinensis* has been found in the EU

Figure A4.1: Photos of locations where *B. chinensis* has been found: A) Amsterdam, B) 's Gravenzande, C) Maasbracht, D) Vinkeveen, E and F) Eijsder Beemden, G and H) Zutphen, I) Balen, J) Maasbree.