REVIEW

A review of the non-indigenous Chinese mystery snail, *Cipangopaludina chinensis* (Viviparidae), in North America, with emphasis on occurrence in Canada and the potential impact on indigenous aquatic species

S.E. Kingsbury, Donald F. McAlpine, Y. Cheng, E. Parker, and L.M. Campbell

Abstract: Evidence suggests that the Chinese mystery snail, *Cipangopaludina chinensis*, a freshwater, dioecious, snail of Asian origin has become invasive in North America, Belgium, and the Netherlands. Invasive species threaten indigenous biodiversity and have socioeconomic consequences where invasive. The aim of this review is to synthesize the relevant literature pertaining to *C. chinensis* in Canada. In doing so, we (*i*) describe *C. chinensis* ecosystem interactions in both indigenous (Asia) and non-indigenous habitats (North America and Europe), (*ii*) identify gaps in the literature, and (*iii*) determine where the species potential distribution in North America requires further exploration. We also briefly discuss potential management strategies for this species, as an aquatic invasive species (AIS), in Canada. Due to the much larger relative size of adult *C. chinensis*, multiple feeding mechanisms, and resistance to predation, *C. chinensis* can out-compete and displace indigenous freshwater gastropods and other molluscs. Furthermore, *C. chinensis* can affect food webs through bottom-up interactions with the bacterial and zooplankton communities by changing nitrogen and phosphorous concentrations. Also, the Chinese literature indicates the potential for *C. chinensis* to act as a biotransfer of contaminants between polluted ecosystems and consumers. In its indigenous range, *C. chinensis* was identified as a host for numerous parasites harmful to human and animal consumers alike. A comparison of the Canadian geographical distribution of reported occurrences with that for the United States indicates several potential gaps in Canadian reporting, which merits further investigation and consideration, especially in regard to federal and provincial non-indigenous monitoring and regulations. Southern Ontario had the highest number of reports that were mostly from web-based photo-supported sources. This suggests that interactive citizen science through popular apps backed by well-supported educational campaigns may be a highly effective means of tracking *C. chinensis* spread, which can be complementary to traditional methods using specimen-vouchered taxonomically verified natural-history collections overseen by professional curators.

Key words: *Bellamya chinensis*, Vivipardae, non-native/non-indigenous species, ecosystem impacts.

Résumé : Il semblerait que la vivipare orientale, *Cipangopaludina chinensis*, un escargot d'eau douce dioïque d'origine asiatique, soit devenue envahissante en Amérique du Nord, en Belgique et aux Pays-Bas. Les espèces envahissantes menacent la biodiversité indigène et ont des conséquences socio-économiques aux endroits où elles sont envahissantes. L'objectif de cette étude est de faire une synthèse de la littérature pertinente relative à *C. chinensis* au Canada. Ce faisant, les auteurs (*i*) décrivent les interactions entre *C. chinensis* et de l'écosystème dans les habitats indigènes (Asie) et non indigènes (Amérique du Nord et Europe), (*ii*) identifient les lacunes de la littérature et (*iii*) déterminent les endroits où la répartition potentielle de l'espèce en Amérique du Nord nécessite une exploration plus approfondie. Ils abordent aussi brièvement les stratégies de gestion possibles pour cette espèce à titre d'espèce aquatique envahissante (EAE) au Canada. En raison de la taille relative beaucoup plus importante de *C. chinensis* adulte, de multiples mécanismes d'alimentation et de la résistance à la prédation, *C. chinensis* peut supplanter et déplacer les gastéropodes et les autres mollusques d'eau douce indigènes. En outre, *C. chinensis* peut affecter les réseaux alimentaires par des interactions ascendantes avec les communautés bactériennes et zooplanctoniques en modifiant les concentrations d'azote et de phosphore. De plus, la littérature chinoise indique que *C. chinensis* peut agir comme agent de transfert biologique de contaminants entre les écosystèmes pollués et les consommateurs. Dans son aire de répartition indigène, *C. chinensis* a été identifié comme hôte pour de nombreux parasites dangereux pour les humains et les animaux qui les consomment. Une comparaison de la répartition géographique des signalements au Canada et aux États-Unis révèle plusieurs lacunes potentielles dans les signalements canadiens qui doivent être approfondies et prises en compte, notamment en ce qui concerne la surveillance et la réglementation fédérales et provinciales des espèces non indigènes. Le sud de l'Ontario a enregistré le plus grand nombre de signalements qui provenaient pour la plupart de sources photographiques sur Internet. Cela suggère que la science

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citoyenne interactive, par des applications populaires soutenues par des campagnes d'éducation bien étayées, peut être un moyen très efficace de suivre la propagation de *C. chinensis*, qui serait complémentaire aux méthodes traditionnelles supervisées par des conservateurs professionnels qui utilisent des collections de spécimens témoins vérifiés sur les plans de l'histoire naturelle et de la taxonomie. [Traduit par la Rédaction]

Mots-clés : *Bellamya chinensis*, Vivipardae, espèces non indigènes, impacts sur l'écosystème.

Introduction

The Chinese mystery snail (CMS), *Cipangopaludina chinensis*, is a dioecious, freshwater, potentially invasive, viviparous mollusc introduced to continental North America around 1890 for consumption in the Asian food market (Stephen et al. 2013). The species has also been recently reported as established in the Netherlands (2007) and Belgium (2016) (Matthews et al. 2017; Van den Neucker et al. 2017). *Cipangopaludina chinensis* is reported to be indigenous to Burma, Thailand, South Vietnam, China, Korea, Japan, the Philippines, and the Island of Java (Indiana Department of Natural Resources 2005; Li et al. 2013; Li et al. 2018; but see Fig. 1). Fecundity studies indicate *C. chinensis* produces shelled, fully-formed young at an estimated rate of 27–33 young per female per year (Stephen et al. 2013), although estimates of up to 100 young per female per year have been suggested (Haak 2015). The species feeds on zooplankton and phytoplankton (Indiana Department of Natural Resources 2005), either by grazing using a radula, or by filtering water (Olden et al. 2013). *Cipangopaludina chinensis* excrete large amounts of fecal matter, which can lead to potential re-engineering of nitrogen and phosphorous cycling within aquatic ecosystems (Olden et al. 2013). The ability of *C. chinensis* to resist predation, out-compete indigenous mollusc species for resources, reproduce rapidly, and shift food-webs suggests that *C. chinensis* is an aquatic invasive species.

In Canada, the Department of Fisheries and Oceans (DFO), Canadian Food Inspection Agency, Canada Border Services Agency, Environment and Climate Change Canada, Transport Canada, and the Department of National Defense, in collaboration with every province and territory, have a mandate to monitor and regulate invasive species in fresh and marine waters (Fisheries and Oceans Canada 2018). DFO defines aquatic invasive species (AIS) as: "Fish, animal, and plant species that have been introduced into a new aquatic ecosystem and are having harmful consequences for the natural resources in the native aquatic ecosystem and (or) the human use of the resource" (Fisheries and Oceans Canada 2018). AIS threaten indigenous biodiversity and natural ecosystems, have significant financial consequences, and can become difficult to manage once established. Globally, invasive species are second only to habitat destruction as a cause of species endangerment and extinction (Pejchar and Mooney 2009). Many AIS threaten natural biodiversity through predation, parasitism, competition, and degradation or destruction of invaded ecosystems (Fisheries and Oceans Canada 2019*a*). AIS may carry viruses, bacteria, and parasites and typically entail significant economic costs due to disruption of ecosystem services, impacts on aquaculture and fisheries, and damage to infrastructure (Fisheries and Oceans Canada 2019*a*; Pejchar and Mooney 2009). In both Canada and the United States (USA), *C. chinensis* is not yet federally prioritized as a species needing close management and is labelled in both countries as "non-indigenous" versus "invasive", which may lead to further confusion on the species' status (Fisheries and Oceans Canada 2019*b*; Kipp et al. 2020). Nonetheless, many USA states and a few Canadian provinces do recognize this species as invasive and have taken steps to prevent sale and distribution of *C. chinensis*. In the USA, *C. chinensis* is federally categorized as a non-indigenous aquatic species, although the United States Geological Survey (USGS) does offer a disclaimer that the listing information for this species is preliminary (Kipp et al. 2020).

Cipangopaludina chinensis is reported to be widespread in freshwater systems across the USA and Canada, with recent and more limited distribution in Europe (Clarke 1981; Collas et al. 2017; McAlpine et al. 2016; Jokinen 1982; Kipp et al. 2020; Stephen 2017; Van den Neucker et al. 2017). McAlpine et al. (2016) reported established populations of *C. chinensis* in Canada in southern Ontario, British Columbia, Quebec, New Brunswick, Newfoundland, and Nova Scotia. In the USA, *C. chinensis* has been reported in 32 states (Jokinen 1982; Kipp et al. 2020). The main vectors for the spread of *C. chinensis* in North America are human activities including boating, aquarium releases, and the addition of *C. chinensis* to garden ponds (Haak 2015; Havel 2011; Havel et al. 2014; Indiana Department of Natural Resources 2005; Harried et al. 2015; Solomon et al. 2010). Unlike the USA, Canada has not created a national database for *C. chinensis*, and there is no risk assessment or predictive species distribution model for *C. chinensis* in Canada. Therefore, neither the true nor predicted extent of this species distribution in Canada is known. Furthermore, the current literature dealing with the impact of *C. chinensis* on indigenous freshwater species, habitat preferences, and basic biological information is sometimes contradictory (Haak 2015; Olden et al. 2013; Solomon et al. 2010; Stephen et al. 2013; Sura and Mahon 2011).

Policy makers need to consider AIS effects on each invaded ecosystem in totality when developing management strategies. However, this is difficult to accomplish when information on individual AIS is spread across multiple forums (i.e., peer-reviewed publications, grey literature, on-line and off-line databases, expert knowledge) and political jurisdictions. Literature reviews are therefore increasingly necessary to condense available knowledge, create conceptual frameworks of species impacts on invaded ecosystems, identify potential risk factors that could be important for future species risk assessments, and identify gaps in knowledge (Matthews et al. 2017).

The aim of this review is to synthesize relevant literature dealing with the CMS so as to (*i*) describe *C. chinensis* ecosystem interactions in both indigenous and non-indigenous habitats, (*ii*) identify gaps in the literature, and (*iii*) determine where the species potential distribution in North America requires further exploration, with emphasis on Canada. We also briefly discuss potential management strategies for this species as an AIS in Canada.

Approach

Literature review

Internet searches for published papers written in English, French, or translated to English (often from Chinese), and in Chinese were conducted using Web of Science, Science Direct, PubMed, Google, Google Scholar, and China National Knowledge Infrastructure (CNKI). Reputable internet sources were also examined to determine the extent of information directed at efforts to enlist the public in limiting the species introduction to North American water bodies. References cited within all literature were reviewed, additional papers of interest were noted, and if relevant, included. Additionally, any works that cited these papers were reviewed for relevance. The literature searched spans the years from 1965–2019.

The key words used for the searches were "Chinese mystery snail", "*Cipangopaludina chinensis*", "*Bellamya chinensis*", "*Cipangopaludina leucostoma*", "*Cipangopaludina diminuta*", "*Cipangopaluina wingatei*", "*Paludina malleata*", "*Paludina chinensis*", "*Cipangopaludina chinensis*

Fig. 1. Preliminary indigenous range (dark brown) of *Cipangopaludina chinensis* based on Pace (1973), Köhler et al. (2012), Lu et al. (2014), and the Worldwide Mollusc Species Database (https://www.bagniliggia.it/WMSD/HtmSpecies/1071200630.htm). Question marks indicate areas where *C. chinensis* has been reported, but we consider indigenous status as uncertain or suspect. The confirmed range presented may be conservative, but it cannot be determined fully until taxonomy within the genus *Cipangopaludina* is resolved. This map was generated using ArcMap 10.7 with country polygon layers (ArcGIS Hub 2020) obtained from the DIVA-GIS repertoire (Diva-GIS 2020) and the world base map was sourced from ArcGIS Hub UIA World Countries Boundaries.

malleata", "*Cipangopaludina malleata*", "*Viviparus chinensis*", "*Viviparus malleata*", and "mystery snail". The literature searches in Chinese were conducted using those terms commonly used for *Cipangopaludina chinensis*: "田螺科", "中华圆田螺", "中华圆田螺+天敌", "中华圆田螺+饲料", and "中华圆田螺".

For *C. chinensis*, condensing knowledge from older literature can be problematic because the taxonomy of the CMS has been contentious, with various scientific names applied to the species over the past few decades. Regardless of the name used in each reference, we follow Integrated Taxonomic Information System (ITIS) conventions here and use "Chinese mystery snail" and "*Cipangopaludina chinensis*" (ITIS Taxonomic Serial No. 70329) in this review. It is noted that *C. chinensis* and *B. chinensis* are the more commonly used terms in modern English-language literature (Haak 2015; Haak et al. 2017; Johnson et al. 2009; Olden et al. 2013; Prezant et al. 2006; Smith 2000; Solomon et al. 2010; Stephen et al. 2013; Twardochleb and Olden 2016; Waltz 2008), while *V. chinensis*, *V. chinensis malleata*, *Paludina chinensis*, and *Paludina malleata* reflect an older taxonomy before *C. chinensis* and *Cipangopaludina japonica* (see further) were considered separate species (Jokinen 1982). "Vivipare chinoise" and *V. malleata* were used in French-language literature for *C. chinensis* (Forets, Faune et Parcs Quebec 2018, Stańczykowska et al. 1971). The names *C. wingatei*, *C. leucostoma*,

and *C. diminuta* are synonyms in Asian literature for *C. chinensis* (Lu et al. 2014). Other synonyms used for C. chinensis also have included *C. chinensis malleata*, *C. malleata*, and *V. chinensis* (Kipp et al. 2020; Lu et al. 2014).

Taxonomic confusion between *C. japonica* (ITIS Taxonomic Serial No. 70332) and *C. chinesis* has also led to uncertainty in the North American literature with respect to both species. *Cipangopaludina japonica*, Japanese mystery snail, is another large, non-indigenous, viviparid (also established in North America and potentially invasive; Burks et al. 2016). Older literature has suggested this species is conspecific with *C. chinensis*, each representing different ecophenotypes of the same species (Jokinen 1982; Kipp et al. 2020). Adding to the confusion, *C. japonica* also has an array of various taxonomic names (e.g., Kipp et al. 2020; = *Viviparus japonicus*). However, morphologic and genetic differences between the two species have been reported, and here we follow current consensus as per the ITIS (http://www.itis.gov/) and treat both species as distinct (Chiu et al. 2002; Clench and Fuller 1965; David and Cote 2019; Hirano et al. 2015, 2019; Lu et al. 2014; Smith 2000; Wang et al. 2017).

While much of the older literature (pre-1990s) may contain information relating to one or both species (Clarke 1981; Jokinen 1982; Smith 2000), we have avoided literature that deals explicitly with *C. japonica*, and included literature pertaining to *C. japonica* only

Fig. 2. Literature located for this review (*n* = 93) dealing with *Cipangopaludina chinensis* and organized by nature of content. Publication sources were peer-reviewed journal articles (*n* = 86) and grey literature (*n* = 7). Additionally, source year of publication is plotted using a violin plot to visualize the relevance of the data. Most literature (Supplementary Table S1[1]) were published 2010–2020, suggesting an increased interest in *C. chinensis*.
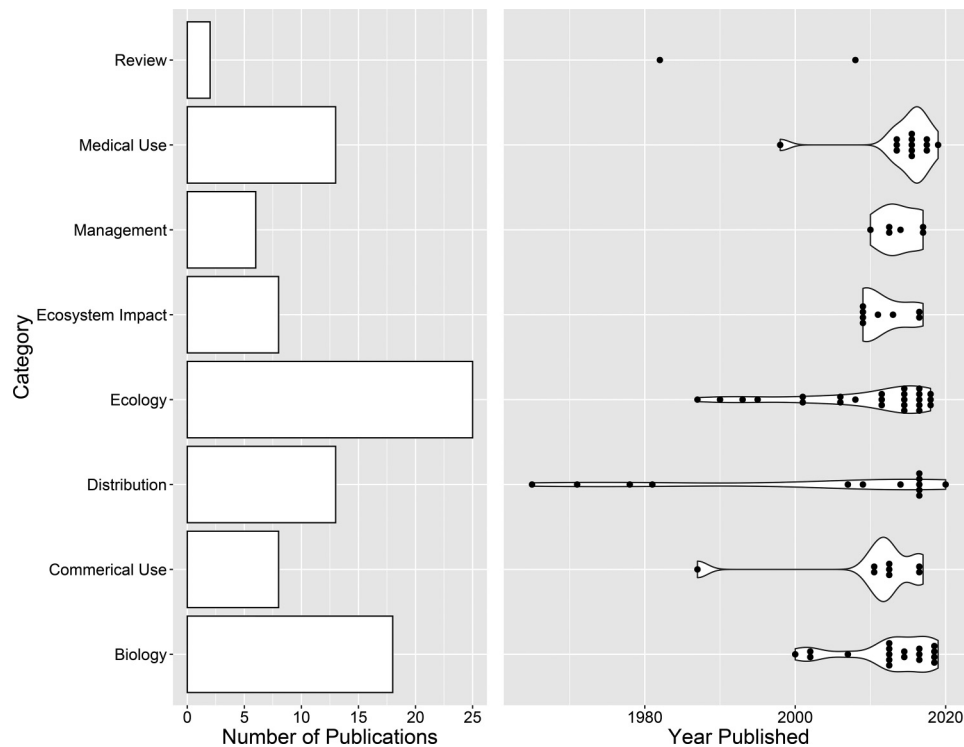
where genetic or morphologic distinction between the two is discussed (Burks et al. 2016; David and Cote 2019; Fox 2007; Hirano et al. 2015). Conversely, across eastern Asia, there are 16 species of Viviparidae, all relatively large. Hirano et al. (2015) identified three species of viviparids in Japan, Chiu et al. (2002) identified two in Taiwan, and Lu et al. (2014) identified 11 species and two subspecies within the genus *Cipangopaludina* in China. We also have avoided inclusion of those indigenous viviparids in our review, keeping the focus on *C. chinensis* in China, where it is abundant, and there is significant scientific literature on its indigenous habitats and ecological interactions.

Our list of references includes peer-reviewed literature, grey literature (e.g., reports, theses), and reputable internet sources. For this review, the term "publication" includes both peer-reviewed and grey literature sources. Together, these publications comprise the literature assessing *C. chinensis* effects on indigenous species in North America and Europe, the species establishment and spread within North America and Europe, and the history and biology of the species in both indigenous (Asian) and non-indigenous (North America and Europe) habitats. The European literature, however, is limited due to the species' very recent establishment on that continent (Van den Neucker et al. 2017).

In regard to terminology, we use "non-indigenous" and "indigenous" as recommended by Colautti and MacIsaac (2004). Publications dealing with *C. chinensis* as an indigenous species in Asia were used to supplement the limited knowledge currently available for North America and Europe. Publications were identified as belonging to one of eight categories (Fig. 2) including (category names are noted in quotation marks, category names are listed in Supplementary Table S1[1]): "Ecology" (ecological studies); "Reviews" (literature reviews); "Management" (species management); "Eco-

system impacts" (disruption of systems by non-indigenous CMS); "Distribution" (both modelled output and absence/presence studies); "Commercial use" (use of CMS for industrial use and for profit); "Medical use" (use of CMS in medical studies and applications); and "Biology" (anatomy, taxonomy, life cycle, morphology, and genetic studies). Reputable internet sources were included in this review to determine the extent and types of information readily available to the public, but these sources were separated from publications because no formal research methodologies were followed or reported. Internet sources were considered "reputable" if they were managed by a group of researchers studying AIS, individual researchers, universities, or government organizations.

Conceptual diagrams

To predict and compare *C. chinensis* ecosystem interactions, conceptual diagrams presented here draw on literature dealing with *C. chinensis* across both its non-indigenous (Fig. 3A) and indigenous (Fig. 3B) ecosystems using literature from the "Ecosystem impacts", "Ecology", and "Biology" categories, as well as suspected vectors of transport as a non-indigenous species in North America. The term "conceptual diagram" was selected because these diagrams provide a visual summary of the main conclusions derived from literature pertaining to *C. chinensis* biological needs, ecosystem impacts, and human interactions. These diagrams were created with two goals: (*i*) to determine if *C. chinensis* meets Canadian federal government criteria as a non-indigenous "invasive" species and (*ii*) to identify ecosystem interactions (either positive or negative) that may occur in North America should *C. chinensis* populations become as widely distributed or as abundant as the species is across its indigenous range.

[1]Supplementary data are available with the article at https://doi.org/10.1139/er-2020-0064.

Fig. 3. (A) Conceptual diagram summarizing *Cipangopaludina chinensis* ecosystem impacts within the non-indigenous range in North America and Europe, based on a review of current North American literature. (B) Conceptual diagram summarizing ecosystem interactions for *C. chinensis* across indigenous range based on current Asian literature. Arrows represent *C. chinensis* impacted by the ecosystem (towards) or impacting the ecosystem (away), while line thickness indicates the severity of the impacts (thicker arrow = increased effect).
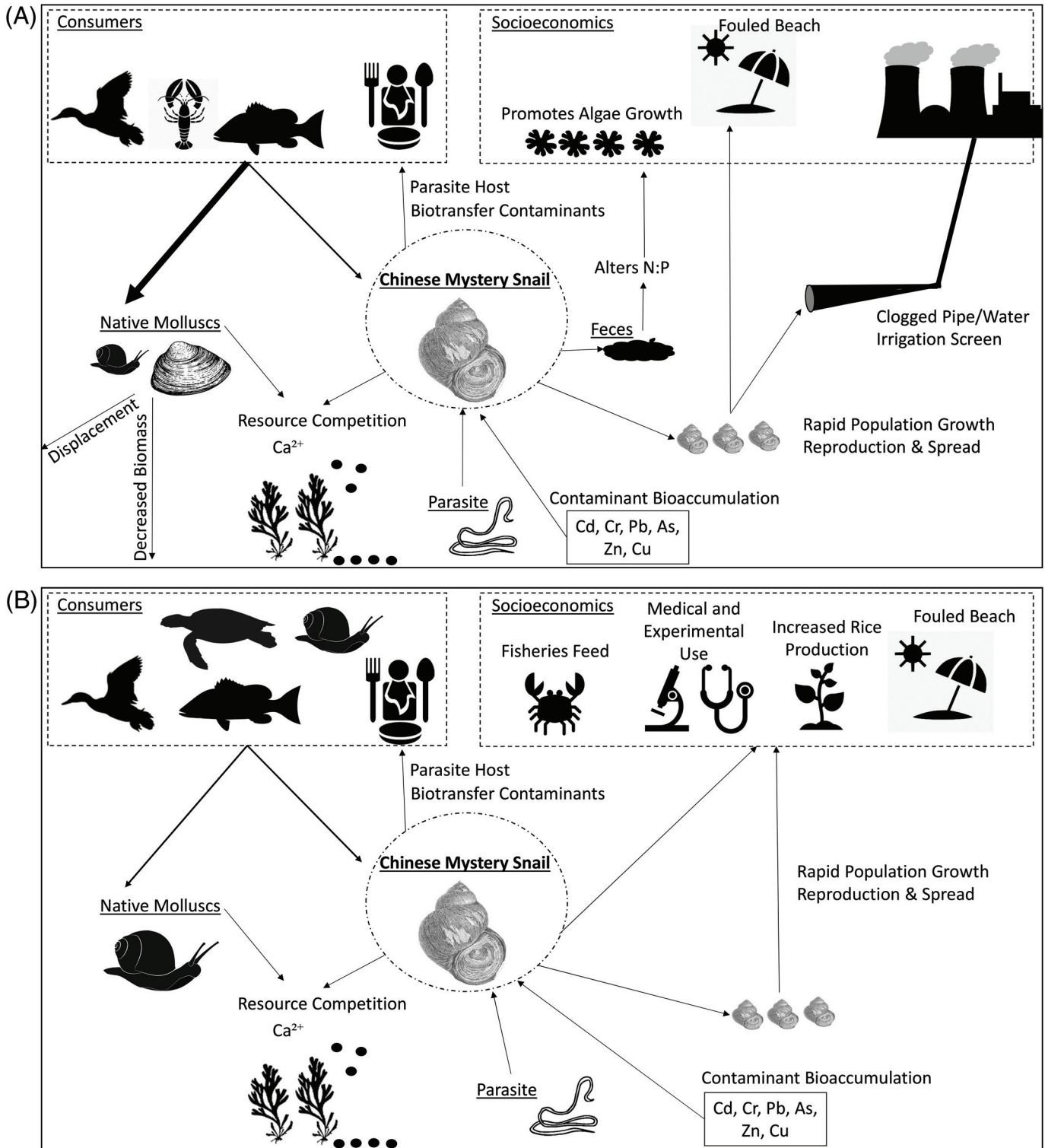
Geographical distribution of Cipangopaludina chinensis in Canada

We contacted multiple Canadian sources to gain access to and build a database of *C. chinensis* reported occurrences in Canada. An effort was made to include all Canadian reports of *C. chinensis* occurrence, regardless of year reported (i.e., both historical and more recent reports). Natural history collections databases from across Canada were accessed for records of *C. chinensis*. The following are the natural history museums contacted (and response to our query if they housed CMS reports): Royal British Columbia Museum (yes), Royal Alberta Museum (no), University of Saskatchewan Museum of Natural Sciences (no), Manitoba Museum (no), the Canadian Nature Museum (yes), the Royal Ontario Museum (yes), Ontario Natural Heritage Information Centre (yes), Quebec Biodome (no), New Brunswick Museum (yes), Nova Scotia Museum of Natural History (yes), and the Provincial Museum of Newfoundland and Labrador (yes). Additional reports were shared with us from the British Columbia Conservation Data Centre, Alberta Department of Environment and Parks, Environment and Climate Change Canada, Wildlife Systems Research, Fragile Heritage (Ontario), the Maritime Aboriginal People's Council (Nova Scotia, New Brunswick, and Prince Edward Island), and our own unpublished records (Kingsbury et al. 2021). The popular online citizen-science natural-history social networks, iNaturalist and Early Detection and Distribution Mapping System (EDDMapS) Ontario, both of which include an on-line app for smartphones, were also searched for confirmed observations of *C. chinensis*. All reports in our geographical distribution maps were verified from specimen vouchers, photographs, or from sources regarded as reliable, and geo-tags (latitude and longitude) confirmed. The geo-tagged report occurrences in Canada were compiled as a data layer in ArcGIS and compared with the USA *C. chinensis* data layer maintained by the USGS (https://nas.er.usgs.gov/viewer/omap.aspx?SpeciesID=1044). In keeping with our focus on Canada and continental North America, we also checked Mexican reports for *C. chinensis*. We did not find any "verified" species reports or reported presences in Mexican waterbodies. *Cipangopaludina chinensis* is not on the United Mexican States invasive species list or the Mexico Invasive Alien Species Action Plan (Mexico Secretariat of the Convention of Biological Diversity, n.d.).

Findings

Literature review

This review includes 123 sources across 93 publications (Fig. 2), including English-, French- and Chinese-language sources, and thirty reputable internet sources (Supplementary Table S1[1]). Literature reporting *C. chinensis* occurrences for Canada was all published prior to 2000s (Clarke 1981; Clench and Fuller 1965; Jokinen 1982; Smith 2000; Stańczykowska et al. 1971), except an article reporting the species for Atlantic Canada (McAlpine et al. 2016). The categories "Ecology", "Biology", "Medical use", and "Distribution" included two thirds of the publications available with 26, 18, 13, and 13 papers, respectively (Fig. 2). Publications reporting the "Ecosystem impacts" of *C. chinensis* on indigenous species (eight), species management– "Management" (six), and "Commercial use" (seven) were relatively few.

The least common publication category was "Reviews" (two publications). These include a grey-literature review (Waltz 2008) and an older second review, which, although peer-reviewed, was written prior to the publication of the majority of the North American literature on *C. chinensis* (Jokinen 1982). Both reviews have been widely cited in a number of grey-literature documents, including federal government documents and theses (Bobeldyk 2009; Haak 2015; Rivera and Peters 2008), government and expert-run websites (Kipp et al. 2020), and peer-reviewed journal articles (Collas et al. 2017; Matthews et al. 2017; Rothlisberger et al. 2010).

The 30 reputable internet sources found were from the USA (17), Canada (10), Australia (1), and international organizations such as the International Union for Conservation of Nature and Global Invasive Species Database (2020) (2) (see Supplementary Table S2[1] for a full list of internet links). Much of the on-line information available to the public appears to be reputable (i.e., written by a scientific specialist) and is presented on government supported websites. These websites offer concise biological information, scientific and common names, descriptions of indigenous and non-indigenous range, and likely vectors of dispersal for *C. chinensis* in North America. Generally, internet sources tend to focus on the biology of the snail, often including notes on its indigenous range, and typically target recreational boaters in an effort to enlist the public in limiting the species introduction to water bodies where *C. chinensis* is non-indigenous and potentially invasive.

Biology

Life cycle

Literature from both indigenous and non-indigenous ecosystems indicate similar biological characteristics in all freshwater habitats, with a single *C. chinensis* female carrying embryos capable of rapidly establishing a population. Female *C. chinensis* give birth to live, shelled, young that are between 3–5 mm in shell diameter. Mature females can produce numerous offspring per brood (Fox 2007). Females reach sexual maturity at 6–12 months (Bobeldyk 2009; Zhang et al. 2017) and studies in North America estimate fecundity to be 27.2–33.3 young/female/year (Stephen et al. 2013). However, snail uteri have been documented holding as many as 102 embryos (Jokinen 1982; Waltz 2008). Chinese studies indicate that *C. chinensis* may reproduce 2–7 times/year (April–October) and brood 20–50 fertilized eggs at a time (Zhang et al. 2017). In North America, females are believed to release juvenile snails from June to late October (Jokinen 1982). Haak (2015) found that adult *C. chinensis* exposed to colder temperatures (≤ 12 °C) ceased reproduction, while those held at warmer temperatures (27 °C) increased reproduction when compared to those at ambient temperatures (20 °C). The number of brooded embryos appears to be directly correlated to female shell size in *C. chinensis* (Stephen et al. 2013). *Cipangopaludina chinensis* is believed to be iteroparous, with females expected to produce multiple broods over a lifetime (Fox 2007; Jokinen 1982), estimated at 5 years (Jokinen 1982; Stephen et al. 2013). Thus, once introduced to a site, *C. chinensis* may undergo exponential population growth, especially in the absence of density-dependent factors that might limit the population (Stephen et al. 2013).

Although habitat can influence the size and morphology of *C. chinensis*, this species is one of the largest freshwater snails among the Viviparidae (Liu et al. 1995). Growth of *C. chinensis* is indeterminate, with both sexes of the species reaching much larger maxima (40–60 mm) than any indigenous North America freshwater gastropod (Liu et al. 1995). Females appear to live longer than males (5 years versus 3–4 years) and therefore grow to larger size (Jokinen 1982). The larger shell size likely enables non-indigenous *C. chinensis* to avoid some predators that otherwise feed on aquatic North American gastropods (e.g., yellow perch, *Perca flavescens*) (Twardochleb and Olden 2016).

Morphology and genetics

Both morphology and genetics suggest that *C. chinensis* is a species distinct from *C. japonica* (Fig. 4; Burks et al. 2016; Chiu et al. 2002; Clarke 1978, 1981; Clench and Fuller 1965; David and Cote 2019; Hirano et al. 2015; Lu et al. 2014; Smith 2000; Wang et al. 2017). Nonetheless, the nomenclatural status of snails currently placed within the genus *Cipangopaludina* remains contentious (Clench and Fuller 1965; Smith 2000). This is especially the case across eastern Asia, where multiple subspecies of *C. chinensis* have been recognized and other species of *Cipangopaludina* co-occur

Fig. 4. The three mystery snail species, Chinese (*Cipangopaludina chinensis*), Japanese (*C. japonica*), and banded (*Viviparus georgianus*), established in continental North America. The shell of *C. japonica* (6–7 whorls, ≈65 mm shell length, 65–80° spire angle) is reported to have a taller spire, a more acute spire angle and finer carina (arrow) on the shell surface than compared to *C. chinensis* (7–8 whorls, ≈50–65 mm shell length, 50–55° spire angle). *Cipangopaludina chinensis* is broader than *C. japonica*. However, these distinguishing features appear to be plastic and shell morphology may vary considerably. Molecular genetic methods are the most reliable method of distinguishing *C. chinensis* from *C. japonica*. *Viviparus georgianus* is the most distinguishable of the three, with a yellow-green shell, dark brownish-red bands, 4–5 whorls, ≈45 mm shell length, and 70–85° spire angle. Catalogue numbers of illustrated specimens: CM58-143 (*C. chinensis*), CM47465 (*C. japonica*), and CM62.6630 (*V. georgianus*). Photo by Dr. T.A. Pearce, Section of Mollusks, Carnegie Museum of Natural History.

Cipangopaludina chinensis   *Cipangopaludina japonica*   *Viviparus georgianus*
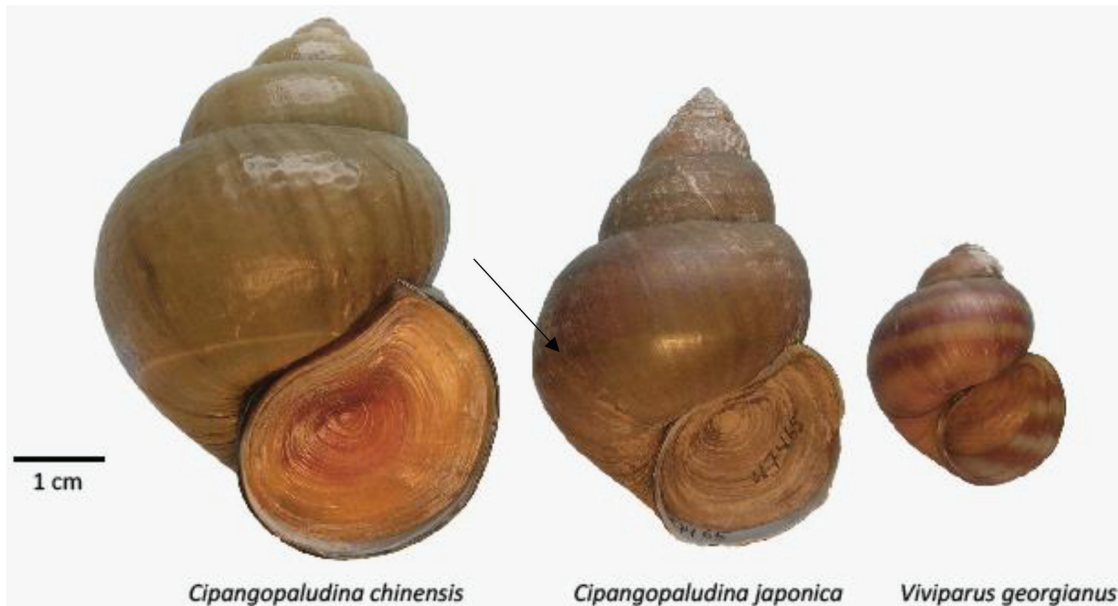
(Lu et al. 2014). Lu et al. (2014) has develop an identification key for the 18 species of *Cipangopaludina* spp. recognized in China, which may be useful for tracking potentially invasive *Cipangopaludina* spp. in North America. Morphologically, two shell phenologies have been identified in *C. chinensis*; tall-spire and short-spire (Chiu et al. 2002). Short-spired shells are found in waterbodies with low pH and low calcium concentrations. Tall-spired shells are found in higher pH, higher calcium concentration lakes (Chiu et al. 2002). The differences in *C. chinensis* shell morphology further confounds identification issues between *C. chinensis* and *C. japonica* in North America, suggesting that one of the species may be more prevalent than previously believed. Recent research, using the mitochondrial cytochrome *c* oxidase subunit I (COI) gene, has established that *C. chinensis* is genetically distinct from *C. japonica* (Burks et al. 2016). To ensure correct species identification we recommend genetic testing, when possible, of mystery snails found in areas where both *C. chinensis* and *C. japonica* occur.

Ecology

Indigenous habitat

Cipangopaludina chinensis (Chinese taxonomic name: 中国圆田螺) is wide-spread within its indigenous range and is an important prey option to indigenous predators and a staple-food item in the human diet in many regions of China and eastern Asia. The indigenous range of *C. chinensis*, commonly referred as the "mud snail" in China (Chinese common name: 田螺) extends across eastern Asia, although the true extent of range remains unknown due to taxonomic uncertainty (Fig. 1). *Cipangopaludina chinensis* occurs naturally in rice paddies, rivers, and lakes, where it is common and where positive ecosystem interactions have been documented (Dewi et al. 2017; Kurniawan et al. 2018; Liu et al. 1995; Nakanishi et al. 2014). As a result, this species is an important component of managed rice paddy ecosystems where the presence of mud snails is correlated

with a greater abundance and diversity of terrestrial arthropods, which leads to greater rice yields (Dewi et al. 2017). Also, within its indigenous range, *C. chinensis* is prey for multiple species (Supplementary Table S3[1]), including waterfowl such as Muscovy duck (*Cairina moschata*, 番鸭), domestic duck (*Anas platyrhynchos domesticus*, 北京鸭), and mallard duck (*Anas platyrhynchos*, 绿头鸭), Asian carp, such as black carp (*Mylopharyngodon piceus*, 青鱼) and Amur carp (*Cyprinus rubrofuscus*, 鲤鱼), and Chinese softshell turtle (*Pelodiscus sinensis*, 中华鳖). The species is also sold in local wet markets for human consumption (Luo et al. 2012; Tian et al. 2012; Yan 2002; Zhang et al. 2017). *Cipangopaludina chinensis* also competes ecologically with the golden apple snail (*Pomacea canaliculata*, 福寿螺), which is a non-indigenous invasive species in China (Luo et al. 2012; Zhang et al. 2017). In North America, species reported preying on *C. chinensis* include largemouth bass (*Micropterus salmoides*), pumpkinseed sunfish (*Lepomis gibbosus*), signal crayfish (*Pacifastacus leniusculus*) (Olden et al. 2009; Twardochleb and Olden 2016), and ringed crayfish (*Faxonius neglectus*) (Kelley 2016). As predators, these species, and others as yet identified or investigated (e.g., freshwater drum *Aplodinotus grunniens*), warrant further examination as potential biological controls of non-indigenous *C. chinensis*.

Ecological and biological tolerances

Due to its ecological plasticity, *C. chinensis* has proven to be extremely adaptable to a variety of environments (Table 1). Burnett et al. (2018) tested thermal tolerance in *C. chinensis*, establishing temperature tolerances ranging from 0 to 45 °C, with 0 °C the lowest temperature tested (and not necessarily the lower thermal limit). These experiments assessed critical maximum temperature and incipient lower lethal temperature over acute exposures (i.e., relatively short-term exposure). *Cipangopaludina chinensis* survival decreases where oxygen concentration is below 15 mg/L, and 4-day survival rate drops to 38.3% at pH 5.5, suggesting the species is

Table 1. Environmental and biological information for *Cipangopaludina chinensis* habitats derived from English-, French-, and Chinese-language literature.

Parameter	Range	Literature reference	Additional notes
Thermal tolerance	Upper limit: between 40 and 45 °C Lower limit: <0 °C	Burnett et al. 2018	Researchers did not determine lower thermal tolerance because experimental minimum was set at 0 °C. Exposure time varied because treatments were either heated or cooled (1 °C/h) until water temperature reached the testing temperature, then aquaria were removed from testing chambers and allowed to recover to room temperature for 48 h.
Water temperature	0–30 °C	Karatayev et al. 2009	Environmental water temperatures where species detected; no laboratory experiments were conducted to establish these thresholds.
	19–21.4 °C	Collas et al. 2017	Environmental water temperatures where species detected; no laboratory experiments were conducted to establish these thresholds.
Air exposure	Adult + mid-large size juveniles: +4 weeks Small juveniles: <2 weeks	Havel 2011	Havel's initial work found that *C. chinensis* could survive extended air exposure. These discoveries were later updated by Havel's student, Unstad.
	>9 weeks	Unstad et al. 2013	
CaCO$_3$ saturation	0.00015	Latzka et al. 2015	Environmental values for this study were determined based on a set of lakes in the North Highland District of Wisconsin. No laboratory experiments were conducted to establish these thresholds.
Max depth	<15 m		
Conductivity	≥67 μs/cm		
Distance to highway	<28 km		
Total P	No significant difference between lakes with *C. chinensis* and lakes without	Twardochleb and Olden 2016	Environmental values where the species was detected.
Water clarity			
Predators	Largemouth bass and pumpkinseed sunfish are able to eat *C. chinensis*, but yellow perch may not.		
pH	6.5–8.4	Jokinen 1982	Environmental pH values where the species was detected; no laboratory experiments were conducted to establish these thresholds.
	4–10	Haak 2015	Part of a larger PhD dissertation. Exposure duration: 4 weeks
	~4.2–8	Fraser et al. unpublished	Exposure duration: 14 days
Salinity	0.03–0.08 ppt	Collas et al. 2017	Environmental salinity values where the species was detected; no laboratory experiments were conducted to establish these thresholds.
	0–10 ppt	Fraser et al. unpublished	Exposure duration: 14 days
Calcium concentration	5–97 ppm	Jokinen 1982	Environmental calcium concentration range at sites of species occurrence; no laboratory experiments were conducted to establish these thresholds.
	<2–>20 ppm	Chiu et al. 2002*	Environmental calcium concentration range at sites of species occurrence; no laboratory experiments were conducted to establish these thresholds.
	2–120 ppm	Haak 2015	Part of a larger PhD dissertation. Exposure duration: 4 weeks
Conductivity	63–400 μS/cm	Jokinen 1982	Environmental conductivity range at sites of species occurrence; no laboratory experiments were conducted to establish these thresholds.
	4–9 μS/cm	Chiu et al. 2002*	Environmental conductivity range at sites of species occurrence; no laboratory experiments were conducted to establish these thresholds.
	140–437 μS/cm	Collas et al. 2017	Environmental conductivity range at sites of species occurrence; no laboratory experiments were conducted to establish these thresholds.
Magnesium concentration	13–31 ppm	Jokinen 1982	Environmental magnesium range at sites of species occurrence; no laboratory experiments were conducted to establish these thresholds.
Sodium concentration	2–49 ppm	Jokinen 1982	Environmental salinity range at sites of species occurrence; no laboratory experiments were conducted to establish these thresholds.

Table 1 (concluded).

Parameter	Range	Literature reference	Additional notes
Oxygen concentration	7–11 ppm	Jokinen 1982	Environmental dissolved oxygen concentration at sites of species occurrence; no laboratory experiments were conducted to establish these thresholds.
Flow rate	0.03–0.08 m/s	Collas et al. 2017	Hypothesized tolerance of water flow rate for habitats inhabited by *C. chinensis*.
Snail density (individuals/m^2)	2.62–3.92	Stephen et al. 2013	
	<1–40	McCann 2014	
	100	Karatayev et al. 2009	
	38	Solomon et al. 2010)	
	<0.5	Soes et al. 2011	Reference from Collas et al. (2017).
	0.33	Collas et al. 2017	
	0.25–30	Nakanishi et al. 2014*	
Filtration rate	106–113 mL/snail × h (max of 471 mL/snail × h)	Olden et al. 2013 **Note:** this study also suggests that *C. chinensis* is related to Chlorophyll-*a* concentrations.	Similar to zebra mussel (22–375), quagga mussel (40–310), Asian clam (347–567), golden mussel (19–350), and blue mussel (17–2767). Study points out difficulty in comparing filtration rates among studies due to varied experimental designs.
Reproduction rate based on fecundity	27.2–33.3 young/year/female	Stephen et al. 2013	Value varies greatly and has been estimated to be as high as 100 young/year/female.
Feeding mechanism	Grazer or filter feeder	Olden et al. 2013	
Taxa tolerance values for Viviparidae	Liao River, Northeast China: 8.4 East China: 5 Idaho, USA: 6	Zhao et al. 2015*	A comparison of the abundance of various viviparid snails commonly found in China and the USA in different habitats of varying quality. Tolerance values (TV) are used for bioassessments where different species assemblage can be indicative of habitat health (e.g., areas of high contamination may have low biodiversity and are only inhabited by more tolerant species). The TV for *C. chinensis* indicates that in the Northeast Liao River of China, the species shows intermediate tolerance to poor water quality.
Taxa tolerance talues for *Cipangopaludina chinensis*	Liao River, Northeast China: 4.6		
Land use	No correlation found between different land-use (state protected forest versus developed housing) and *C. chinensis* presence	Owens 2017	

Note: Many of the references report ecological data for locations where the species has been discovered and are not actual ecological threshold experiments, and as such should be interpreted carefully. There is a need for laboratory studies to assess the true ecological thresholds for this species in order to assess potential habitats (see Kingsbury et al. 2021). Literature references with an asterisk (*) denote references from *C. chinensis* indigenous range (i.e., Eastern Asia).

sensitive to oxygen and pH levels (Zhang et al. 2017). Molluscicides, such as copper sulphate and rotenone, are not effective at controlling this species (Haak et al. 2014). Although an aquatic mollusc, initial air exposure experiments found that adult snails can survive at least 4 weeks of air exposure and juvenile snails survive 3–14 days (Havel 2011). Other desiccation experiments suggest that adult *C. chinensis* can survive exposure for longer than 9 weeks (Unstad et al. 2013). Finally, the concentration of calcium required (Table 1) by *C. chinensis*, at < 2 ppm, is very low compared to other molluscs, such as the zebra mussel (*Dreissena polymorpha*), quagga mussel (*Dreissena rostriformis bugensis*), and other calcifying AISs such as banded mystery snail and rusty crayfish (*Faxonius rusticus*) (Chiu et al. 2002; Latzka et al. 2015). Our conceptual diagrams therefore represent a conservative estimate of *C. chinensis* ecosystem impacts in North America. Notably, the ecological and biological data available (Table 1) is often based on opportunistic accounting of where the species has been discovered and does not reflect statistically robust ecological threshold experiments. Therefore, further research is needed to determine the ecological thresholds of this species.

Parasitology

In North America it appears that parasite prevalence in *C. chinensis* is much lower than among indigenous gastropod species (Karatayev et al. 2012). Of 147 necropsied *C. chinensis* from lakes in Wisconsin, USA, only two contained trematode parasites (Harried et al. 2015).

One snail contained *Cyathocotyle bushiensis*, a parasite of waterfowl known to cause mortality in dabbling ducks (Hoeve and Scott 1988), and the other contained nine aspidogastrean flatworms, *Aspindogaster conchicola* (Harried et al. 2015), a widespread parasite (North American, Europe, Africa) of freshwater bivalves, fish, and turtles. Karatayev et al. (2012) found that of 30 *C. chinensis* collected from the Great Lakes area, Canada, none were infected with parasitic helminths. In laboratory experiments where *C. chinensis* and two indigenous snail species, *Bithynia tentaculata* and *Physa gyrina*, were exposed to *Sphaeridiotrema pseudoglobulus*, an important waterfowl parasite, *C. chinensis* contained significantly fewer metacercariae than either of the two indigenous snail species (Harried et al. 2015). Moreover, those metacercarial cysts found in *C. chinensis* were typically encased within the mollusc shell and likely no longer viable (Harried et al. 2015). This limited information suggests that *C. chinensis* may be less vulnerable to infection by endoparasites commonly affecting indigenous gastropod species in North America. However, it is concerning that *C. chinensis* has been found hosting *A. conchicola* as this parasite has been linked to decreased reproduction in North American freshwater mussels (Gangloff et al. 2008), a diverse group of organisms for which there is considerable conservation concern (Williams et al. 1993).

East Asian research demonstrates that *C. chinensis* may act as an intermediate host for a number of human endoparasites of medical importance, including *Echinostoma cinetorchis* (an Asian intestinal

fluke) and *Angiostrongylus cantonensis* (a nervous system nematode) (Chao et al. 1993; Chung and Jung 1999; Jokinen 1982; Lü et al. 2006; Sohn and Na 2017). Although of Asian origin, *A. cantonensis* has been identified as an emerging zoonotic pathogen in North America (York et al. 2015).

Biotransfer of contaminants

There have yet to be studies published on the potential for *C. chinensis* in North America to transfer contaminants from environments with elevated contaminant concentrations to human consumers. Nonetheless, several North American studies have shown that this species bioaccumulates contaminants present in polluted sediments, specifically mercury, arsenic, iron, manganese, zinc, copper, nickel, lead, cadmium, and chromium (Chapman et al. 2020; Tornimbeni et al. 2013). The study of the biotransmission of contaminants from sediments to *C. chinensis* and thence to consumers has been a key focus of multiple studies in Asia (Cui et al. 2012; Fang et al. 2001; Kurihara et al. 1987; Luo et al. 2016; Wu et al. 2001). Mainly, this is because of human health concerns due to the wide-spread consumption of *C. chinensis* in Asia. For example, of 14 edible mollusc species purchased at food markets originating in the Pearl River Delta, China, *C. chinensis* was among those with concentrations of cadmium, copper, zinc, lead, nickel, chromium, antimony, and tin that exceeded the regulatory human consumption limits. Chromium, most notably, exceeded the Chinese daily consumption limits of 1 μg/g (Fang et al. 2001). Snails from the highly polluted Zhalong Wetland, China, also had elevated lead (average concentration = 45.32 μg/kg), cadmium (average concentration = 2.45 μg/kg), and arsenic (average concentration = 11.48 μg/kg), due to grazing over highly contaminated sediments. (Luo et al. 2016). Additionally, *C. chinensis* has been documented to bioaccumulate zinc and copper from sewage sludge used to fertilize rice paddy fields (Kurihara et al. 1987). While the ability to bioaccumulate contaminants may prove useful in bioremediation, or allow *C. chinensis* to serve as a bioindicator, there is also concern that where the species is a regular part of the human diet, that *C. chinensis* may transfer contaminants to consumers. Even though *C. chinensis* is consumed far less frequently by humans in North America than in Asian countries, further North American research on bioaccumulation and contaminant transfer is required. Some North American wildlife certainly consume *C. chinensis* and may bioaccumulate contaminants that may impair North American wildlife populations or eventually find their way to human consumers.

Ecosystem impacts

Inter-mollusc competition

Where *C. chinensis* is present in North America, especially where other freshwater invasive calcified species are established (i.e., gastropods, bivalves, crayfish), there is evidence of impacts on indigenous species. Studies by Sura and Mahon (2011) found that the presence of *C. chinensis* caused snails (*Helisoma trivolvis*) to increase feeding rates and suggested cascading changes to community structures due to higher rates of algal consumption by *C. chinensis* (Sura and Mahon 2011). Johnson et al. (2009) found that indigenous snails (*H. trivolvis*, *Lymnaea stagnalis*, and *Physa gyrina*) decreased in mass during in-laboratory exposure to *C. chinensis*. In the presence of a second invasive species, the rusty crayfish, one indigenous mollusc species became extinct (Johnson et al. 2009). The thick shell and larger size of *C. chinensis* relative to indigenous gastropods leaves *C. chinensis* more resistant to predation by rusty crayfish (Johnson et al. 2009). However, rusty crayfish and other invasive crayfish species, such as red swamp crayfish (*Procambarus clarkii*) and northern crayfish (*Faxonius virilis*), will consume *C. chinensis*, although apparently taking only smaller, juvenile snails. Indigenous signal crayfish seem to be particularly prone to prey on *C. chinensis* (Olden et al. 2009). Predation experienced by *C. chinensis* in mesocosm experiments with red swamp,

virile, and signal crayfish decreased with increased *C. chinensis* shell size (Olden et al. 2009). As a result, *C. chinensis* has been documented to have an impact on inter-species dynamics in regard to competition, stressors, and as a potential prey source for a limited number of indigenous predator species.

There is evidence that the diet, feeding mechanisms, and excretions of *C. chinensis* may alter freshwater bacterial and algal communities leading to changes in water chemistry and food webs (Bobeldyk 2009; Chen et al. 2011; Johnson et al. 2009; Olden et al. 2013). Previous studies have shown that *C. chinensis* ingest benthic organic matter, inorganic matter, and algae, primarily using the radula. Stomach contents of *C. chinensis* show that this species favours diatoms, but other studies indicate that *C. chinensis* does not feed selectively (Jokinen 1982; Olden et al. 2013; Plinski et al. 1978; Stańczykowska et al. 1971). Additionally, larger adult *C. chinensis* (≥ 44 mm) can filter feed and will do so selectively when inter-snail competition is high (Olden et al. 2013). The ability to switch between two feeding mechanisms has previously been noted as a competitive advantage (Brendelberger and Jurgens 1993). Freshwater gastropods are typically grazers and freshwater bivalves are filter feeders (Brendelberger and Jurgens 1993). However, the ability of *C. chinensis* to compete for nutrition via both feeding mechanisms gives it a competitive advantage. Also, due to the significant amounts of nitrogen and phosphorous this large gastropod excretes via fecal matter, it is possible for *C. chinensis* to alter the algal community in habitats occupied (Olden et al. 2013). Mesocosm experiments with high *C. chinensis* densities showed significant changes in bacterial community composition (Olden et al. 2013). Changes in bacterial community composition led to a decrease in bacterial community variability, but did not decrease bacterial abundance (Olden et al. 2013).

Nutrient cycling

Filtration rates of large *C. chinensis* (maximum 471 mL/snail/h) are comparable to high-profile invasive freshwater and marine bivalves, including zebra mussel, quagga mussel, Asian clam (*Corbicula fluminea*), golden mussel (*Limnoperna fortunei*), and blue mussel (*Mytilus edulis*) (Olden et al. 2013). Additionally, mesocosm studies with *C. chinensis* present at low and high densities showed decrease of 54% in the amount of suspended chlorophyll-α concentrations at "low" snail density and a 115% decrease at "high" snail density (Olden et al. 2013). Substrate chlorophyll-α increases observed were likely due to elevated nitrogen:phosphorus (N:P) concentrations ratios produced by *C. chinensis* excreta that promote periphyton production (Johnson et al. 2009; Olden et al. 2013). Previous research suggests that *C. chinensis* increase N:P ratio in the water column due to low P excretion (Johnson et al. 2009). Future research should further investigate the food web implications of bacterial community shifts and the species of algae produced (whether *C. chinensis* presence may lead to higher probabilities of toxic algal blooms). Previous studies link *C. chinensis* nutrient cycling, particularly nitrogen (N) and phosphorus (P) compounds to downstream eutrophication, with higher temperatures producing greater release rates of total-N, dissolved-N, total-P, dissolved-P, ammonia, and phosphate (Bobeldyk 2009; Chen et al. 2011). These studies suggest that *C. chinensis* may play an important role in eutrophication. This warrants further research to determine *C. chinensis* population densities required to cause eutrophication. More importantly, aquatic systems already under pressure of elevated nutrient levels, such as from farmland run-off, should be protected against *C. chinensis* introduction to ensure that this species does not become the tipping point that leads to eutrophication of these vulnerable ecosystems.

Commercial use

An array of human uses for *C. chinensis* have been documented in East Asia, although in North America *C. chinensis* has not yet

been exploited commercially. The species has been used as feed for larval fireflies (*Luciola ficta*) (Ho et al. 2010) and hatchery-cultured mud crab (*Scylla paramamosain*) (Gong et al. 2017). Fish farms in China use viviparid snails as commercial feed for Chinese carp (*M. piceus* and *Cyprinus rubrofuscus*) (Yan 2002), Chinese softshell turtles (Tian et al. 2012), and Chinese golden-coin turtles (*Cuora trifasciata*) (Luo et al. 2012). *Cipangopaludina chinensis* has been used to remediate rice paddy fields contaminated with sewage with elevated metals (Fujibayashi et al. 2016; Kurihara and Suzuki 1987; Kurihara et al. 1987; Xing et al. 2016).

Medicinal use

In Asia, there have been recent efforts to develop pharmaceutical uses for *C. chinensis*. *Cipangopaludina chinensis* has been long used in traditional Korean medicine and is a component of traditional Korean medical knowledge used to treat indigestion (Kim et al. 2018). More recent medical studies have explored the efficacy of compounds derived from various *C. chinensis* tissues for protection against liver damage (Jiang et al. 2013; Wang et al. 2015*a*; Xiong et al. 2019; Yang et al. 2012), as an anti-inflammatory for joint pain-caused immune responses (Lee et al. 1998; Maoka et al. 2012; Shi et al. 2016), and as a supplement that protects against plaque build-up in arteries (Xiong et al. 2013, 2017, 2019). Additionally, *C. chinensis* has been used as a model organism in neurophysiological, hepatological, and reproductive studies testing pharmaceutical products and supplements (Swart et al. 2017; Wang et al. 2014; Wang et al. 2015). Finally, waste shells have been recycled to extract calcium that can be used for promoting bone growth (Zhou et al. 2016). These methodologies have the potential for transfer to, and development in, North America as a means of reducing non-indigenous *C. chinensis* populations.

Management

Commonly employed molluscicides, including rotenone and copper sulphate, are not effective in controlling or eliminating *C. chinensis* (Haak et al. 2014) and can be toxic to other invertebrates. Drawdowns, in which the water is removed from a specific water body, will likely not eradicate *C. chinensis* because the species can survive periods of desiccation exceeding 9 weeks (Unstad et al. 2013). Furthermore, long periods of air exposure, or increased predator pressure, has only been found to elevate *C. chinensis* reproductive rates (Prezant et al. 2006; Unstad et al. 2013; and personal observation of laboratory cultures). Juvenile *C. chinensis* typically hide under rocks and burrow into sediment, so it is impractical to manually cull populations of *C. chinensis* (Jokinen 1982).

Nonetheless, approaches to management, either through eradication and (or) limiting further introductions, have not been thoroughly explored, with only five publications available (Collas et al. 2017; Haak et al. 2014; Matthews et al. 2017; Rothlisberger et al. 2010; Unstad et al. 2013). Only two of these publications conducted experiments to examine *C. chinensis* response to commonly used AIS management approaches (Haak et al. 2014; Unstad et al. 2013). Two publications have focused on public education, placing controls on species movements, and early species detection (Matthews et al. 2017; Rothlisberger et al. 2010). Matthews et al. (2017) noted that there are no ongoing case studies for eradication of *C. chinensis,* perhaps due to the perception that *C. chinensis* in North America is always present at relatively low densities and is nonproblematic. Currently, species monitoring of *C. chinensis* in North America is lacking in terms of monitoring population density. Population density likely plays an important role in how *C. chinensis* impacts indigenous species. As previously noted, *C. chinensis* at relatively low densities is able to overwhelm nutrient release of indigenous molluscs (Bobeldyk 2009). Eradication of AIS (by chemical, manual, biological, or mechanical means) is generally only effective when a species is recently introduced to an aquatic system (Government of Canada 2019*a*), supporting the need for an effective monitoring and modelling program across Canada. A single study of *C. chinensis* ecological preference for slow moving water noted that perhaps increasing water flow via installation of a culvert between connected water bodies may limit the upstream spread of *C. chinensis*. However, these results were only observational and require further investigation (Rivera and Peters 2008). As we note elsewhere in this review, *C. chinensis* has been documented as established in a variety of freshwater, and even brackish habitats (see section titled "Geographical distribution of reported *C. chinensis* occurrences in Canada"), and under various flow regimes.

Some Canadian government agencies have taken steps to limit the spread of *C. chinensis* by preventing new introductions. In Canada, Manitoba and Alberta both have laws controlling the sale, distribution, and reporting of *C. chinensis* and other non-indigenous viviparid snail species. These Canadian provinces have also implemented required boat cleaning and AIS reporting programs to ensure AIS are not accidently transferred between waterbodies and to improve early AIS detection (Government of Alberta 2015; Government of Manitoba 2015). The DFO, with federal oversight of AIS in Canada, operates under the *Fisheries Act* and enforces AIS management through the *Aquatic Invasive Species Regulations*. Section 10 of the *Aquatic Invasive Species Regulations* prohibits any person from introducing a species that is non-indigenous to any region or water body frequented by fish unless authorized under provincial or federal law (Canada Legislative Services Branch 2015). *Cipangopaludina chinensis* is listed under the *Aquatic Invasive Species Regulations*, and is therefore subject to management under these regulations. However, a previous risk assessment by DFO has assessed *C. chinensis* impacts to be negligible in Canada and this has led to reduced management priority for *C. chinensis* when compared to other AISs (Schroeder et al. 2013). In the USA, examples of governmental control of *C. chinensis* include Missouri and Minnesota. *Cipangopaludina chinensis* is listed as a "regulated invasive species" in Minnesota, which allows for possession, sale, purchase, and transportation of this species, but prohibits the introduction of live *C. chinensis* into new ecosystems, such as through aquarium releases or via garden ponds (Minnesota Department of Natural Resources 2020). In Missouri, *C. chinensis* is designated as a "prohibited species", meaning that no person may possess, purchase, sell, transport, import, or export this species (Missouri Department of Conservation 2020). Within the European Union a risk assessment was completed that recommended targeting the pet and aquarium trade and increasing public education/engagement to assist in controlling the further spread of *C. chinensis* (Matthews et al. 2017). It is clear that regulatory action to prevent the sales or transport of viviparid snail species via the pet, aquarium, and water garden trade, and public education and engagement will be essential for early detection and slowing the spread of *C. chinensis* in North America.

In Canada, public education concerned with invasive and potentially invasive species, including *C. chinensis*, is limited (Matthews et al. 2017; Office of the Auditor General of Canada 2019). This has led to a lack of public engagement in terms of monitoring invasive species across much of Canada, with the possible exception of Ontario. This may explain some of the gaps in distribution currently present in Canada. A more complete understanding of *C. chinensis*' impacts on freshwater ecosystems in North America is required, especially in terms of determining where *C. chinensis* is present, and which ecosystems are likely to be most affected. Also, species managers should consider expanding the terminology used to describe *C. chinensis* from merely "non-indigenous" or "non-native" to incorporate "invasive". This should encourage greater public support for, and assistance with, species containment and geographical analyses.

Conceptual diagrams

Our conceptual diagram of *C. chinensis* ecosystem interactions in North America (and Europe) further supports the argument that

Table 2. Estimated *Cipangopaludina chinensis* densities and social and environmental parameters thought to impact density.

Location	*C. chinensis* density (individuals/m²)	Parameter	Possible link	Reference
Takashima, Shiga, central Japan-rice paddy fields	0.25–30	Mean water depth	Rice paddy fields with deeper waters had higher snail density	Nakanishi et al. 2014
		Mean mud depth	Rice paddy fields with deeper mud had greater snail density	
		Reduced-pesticide regime	Rice paddy fields that used a quarter of the "standard" pesticide use in Japan had greater snail density	
		No-pesticide regime	Rice paddies with no-pesticide use had lower snail densities	
		Earthen ditch presence	Rice paddies with earthen ditches resulted in higher snail density	
		Soil crevice presence	Soil crevice presence in rice paddies indicated habitual dryness and resulted in lower snail density	
Lancaster County, Nebraska, United States of America – Wild Plum Lake	5.2	Depth	Hypothesized to impact snail density estimates as few snails were found in the deeper portions of the mark-recapture experiment transects (∼3 m)	Chaine et al. 2012
Lancaster County, Nebraska, United States of America – Wild Plum Lake	38.58	Mean flow values for consumption	May represent a biological limitation if *C. chinensis* consumes more than is naturally available.	Haak et al. 2017
		Presence of boaters and anglers	The rate of transmission may be impacted by accidental boater and bait bucket transfers. An estimate transfer rate of 0.12% was assumed in this study.	
Long Island, New York, United States of America – Setauket Mill Pond	<1–40	Water depth	2 m	McCann 2014
		Dissolved phosphate (summer)	0.02–0.05 mg/L	
		Dissolved nitrate (summer)	0.11–1.47 mg/L	
		Dissolved ammonia (summer)	0.04 mg/L	
		pH	6.4–9.4 (average 7.9)	
		Conductivity	131.8–269.3 μs/cm (average 213.7 μs/cm)	
		Filamentous algae abundant during spring, submerge vegetation abundant in summer		
		Ecosystem composition	Leeches, giant water bug (belastomatids), dragonfly larvae (odonates), bluegill (*Lepomis macrochirus*), common snapping turtle (*Chelydra serpentina*), and other turtles and waterfowl (not specified)	
		Surface area	1.46 ha	
Northern Highlands Lake District, Wisconsin, United States of America – 44 lakes within survey area	0.16–4	Surface area	14–1400 ha (median 130 ha)	Solomon et al. 2010
		Water clarity	Lower water clarity was positively correlated with greater snail density	*We downloaded the Solomon et al. 2010 Supplementary Material data[1] and statistically analysed these data to determine if each of these parameters impacted the density of *C. chinensis* found at each site. We did not find any statistically significant correlations when the Otter Lake outlier was excluded.
		Shoreline development	Greater shoreline development = greater snail density 1.00 ± 0.40 ($p = 0.005$)	
		Proximity to population centre	Closer proximity = greater density −0.71 ± 0.37 ($p = 0.04$)	
		Conductivity		
		Secchi disc depth	−0.84 ± 0.4 ($p = 0.02$)	
		Boat lunch presence	Positive correlation	

Table 2 (concluded).

Location	C. chinensis density (individuals/m^2)	Parameter	Possible link	Reference
Northern Highlands Lake District, Wisconsin, United States of America – Otter Lake	38	Surface area = 70 ha Distance to pop centre = 38 694 m Shoreline development = 28.19 building/km Conductivity = 72 µS/cm Rusty crayfish = present Lake access = public boat ramp	*These are the specific environmental parameters measured for Otter Lake, which was an outlier from the Solomon et al. (2010) Supplementary Material Data[1]	Solomon et al. 2010

Note: The information from each reference is specific to the geographic study area, which is also noted. Unfortunately, there are no studies comparing species density in multiple geographic locations (e.g. China, USA, Canada, and Europe).

C. chinensis is an AIS on the continent (Fig. 3A). *Cipangopaludina chinensis* negatively affects indigenous aquatic species by increasing environmental pressures (e.g., greater resource competition, changes in food web structure, changes to nutrient cycling), leading to decreased biomass and even localized extinction of indigenous molluscs (Johnson et al. 2009; Karatayev et al. 2009; Sura and Mahon 2011). Indirect, negative, impacts on indigenous aquatic species may be caused by *C. chinensis* through shifts in food web structure (Olden et al. 2013), changes in water chemistry (Olden et al. 2013), and eutrophication (Bobeldyk 2009). Also, *C. chinensis* has become a vector for indigenous *A. conchicola*, which parasitize North American unionid bivalves of conservation concern (Harried et al. 2015).

The indigenous (Asian range) conceptual diagram identifies areas where North American literature is incomplete and flags some future concerns for growing *C. chinensis* populations and further distribution in North America (Fig. 3B). In particular, consumption of *C. chinensis* by North American wildlife species, and perhaps humans, is potentially problematic because there is good evidence of *C. chinensis* transferring parasites and contaminants to both human and wildlife consumers in Asia (Chao et al. 1993; Chung and Jung 1999; Cui et al. 2012; Fang et al. 2001; Kurihara et al. 1987; Lu et al. 2018; Luo et al. 2016; Sohn and Na 2017; Tornimbeni et al. 2013; Wu et al. 2001). Further research is required to assess the importance of *C. chinensis* in hosting parasites or transferring contaminants into North American wildlife consumers that are part of freshwater food webs, or to humans.

Additionally, our literature review and conceptual diagrams uncovered gaps in current knowledge for *C. chinensis* distribution and management across Canada. There are indications that *C. chinensis* is an invasive species with the potential to disrupt North American ecosystems and food webs and to reduce the economic benefits of freshwater habitats (Fig. 3A). Unfortunately, available data are based on a limited number of laboratory and field experiments and surveys (Bury et al. 2007; Chaine et al. 2012; Clarke 1981; Haak et al. 2017; Karatayev et al. 2009; Kipp et al. 2020; Latzka et al. 2015; McAlpine et al. 2016; Rothlisberger et al. 2010; Solomon et al. 2010; Stephen 2017). Also, there is limited literature regarding *C. chinensis* in its indigenous ranges that can be used to supplement the English-language literature and support management decisions. While, in theory, formal risk assessments may allow one to accurately assess the risks associated with AIS in North America, data and studies are still needed to support any such risk assessments for *C. chinensis*. A systematic, continental-scale, record-keeping approach to *C. chinensis* in North America would better enable predictions of habitat overlap with indigenous species of conservation concern (e.g., species at risk).

The conceptual diagrams indicate that *C. chinensis*' role in North American habitats (Fig. 3A) versus its role in Asian habitats (Fig. 3B) are similar. However, the outcomes of interactions between *C. chinensis* and its indigenous ecosystems are less extreme in the species Asian range than in North America. Presumably, this is evidence of a current lack of ecological equilibrium in North America between *C. chinensis* and the ecosystems it occupies on the continent. It is also worth noting that research priorities involving *C. chinensis* in Asia and North America differ and that the conceptual diagrams presented here are visual summaries of current available literature. In North America (and Europe), research has typically focused on determining the ability of *C. chinensis* to disperse and become established, and whether the species exhibits invasive behaviours. In Asia, work on *C. chinensis* has often been concerned with issues of environmental contamination and monitoring and possible commercial or medicinal use of *C. chinensis* (Kurihara and Suzuki 1987; Li 2012). In summary, the introduction of a non-indigenous species into an ecosystem can have unpredictable outcomes. It is likely that the ecosystem impacts *C. chinensis* exerts on invaded habitats may be density dependent. Currently, no studies exist that compare snail density in its native range versus the non-indigenous range. However, a few studies (Table 2) do suggest certain environmental parameters that may impact *C. chinensis* successful establishment and population growth (Chaine et al. 2012; Collas et al. 2017; Haak et al. 2017; McCann 2014; Nakanishi et al. 2014; Solomon et al. 2010). Our "non-indigenous (North America and Europe)" and "indigenous (Asia)" conceptual diagrams should be useful for assessing potential scenarios, in they summarize previously published information on *C. chinensis* from both indigenous and non-indigenous ranges. Based on the conceptual diagram presented here and applying the DFO definition for an AIS (Fisheries and Oceans Canada 2018), *C. chinensis* in North America should be regarded as an invasive snail.

Geographical distribution of reported *C. chinensis* occurrences in Canada

We located 278 reports documenting *C. chinensis* in Canada (Fig. 5), with the greatest number of reports from Ontario (77.7%). Canadian reports of *C. chinensis* occurrence by province included: British Columbia (12); Alberta (one); Ontario (216); Quebec (10); New Brunswick (seven); Nova Scotia (30); Prince Edward Island (one); and the island of Newfoundland (one). We located no reports of *C. chinensis* from Saskatchewan, Manitoba, Yukon, Northwest Territories, Nunavut or Labrador (Table 3). The first reported occurrence in Canada was at St. John's, Newfoundland, in 1905, the second from Kawkawa Lake, British Columbia (1953). Since then there has been only sporadic reporting in Canada (Clarke 1981; Clench and Fuller 1965; Kalas et al. 1980; McAlpine et al. 2016; Plinski

Fig. 5. *Cipangopaludina chinensis* occurrences in Canada and the continental United States of America (USA). USA records (blue) presented are taken from Kipp et al. (2020). Hawaii is excluded, but it is the site of the first reported case of *C. chinensis* in North America (year 1800). There are four reports from the Hawaiian Islands (Hawaii, Kauai, Molokai, and Oahu), but none since 1997 (Kipp et al. 2020). Canadian records (red) were assembled for this study (see text for sources). The gaps in species occurrence reports suggest investigator bias (i.e., only reported by individuals with knowledge of this species and access to reporting resources such as phone apps). Therefore, this map is likely a conservative estimate of the species' true distribution. Species occurrence points are aggregated to within 10 km. This map was generated using ArcMap 10.7 with base maps sourced from Statistics Canada (2019) (Canada base map, rivers, lakes, and coastal waters) and ArcGIS (2012, 2013) (USA and Mexico base maps). This is the first collected and mapped national database of *C. chinensis* occurrences in Canada (see subsection titled "Geographical distribution of *C. chinensis* in Canada" in section "Approach" section for details) and the USA point data were sourced from United States Geographical Services (Kipp et al. 2020).
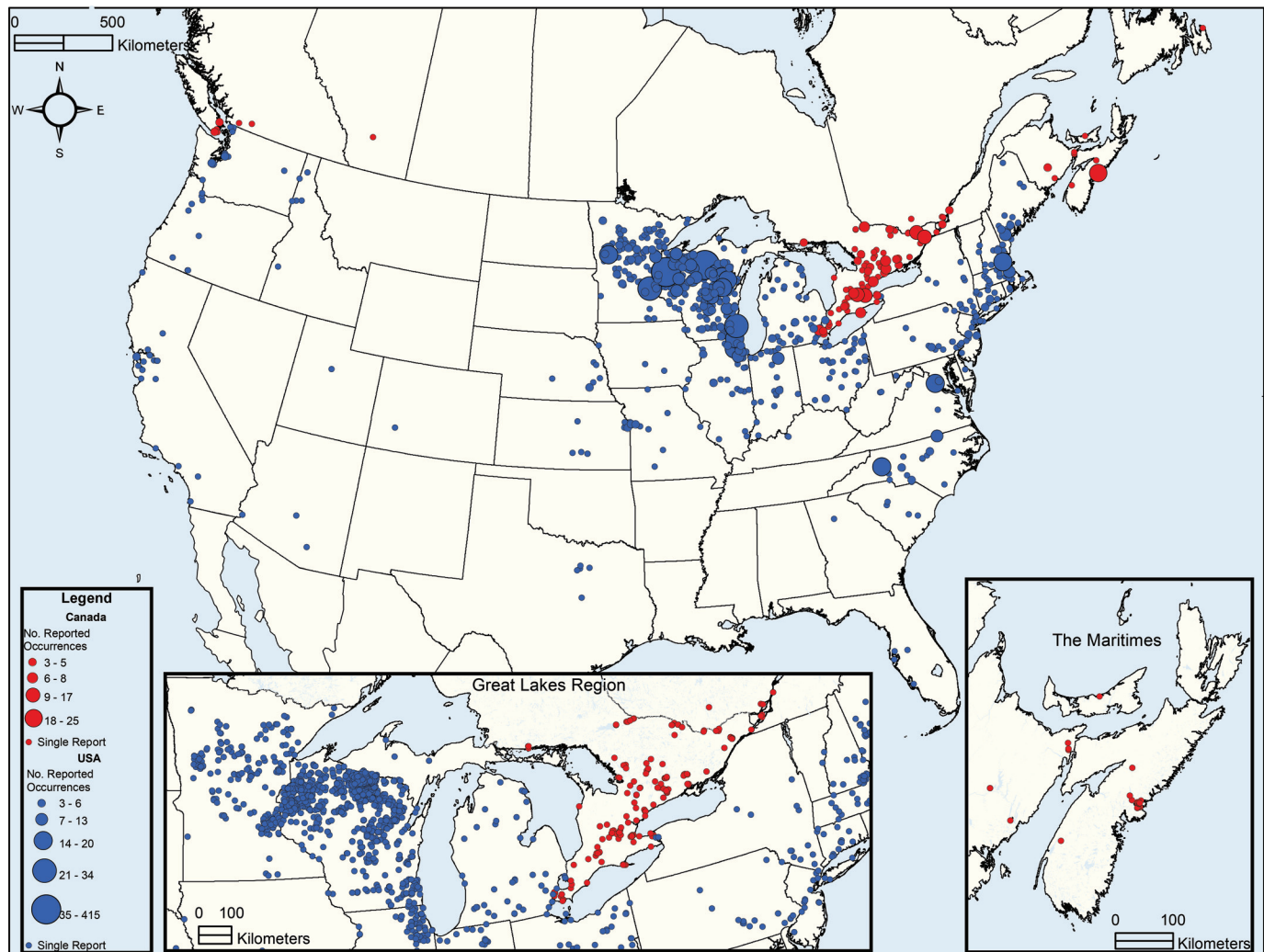
et al. 1978; Stańczykowska et al. 1971). The greatest number of Canadian *C. chinensis* occurrence reports (Fig. 6) were made between 2010–2019 (215 reports), 1990–1999 (33), followed by 2000–2009 (26). Very few reports pre-date 1990 (nine). For example, *C. chinensis* was first noted in Quebec in 1971 (Jokinen 1982), but there have been very few subsequent reports for the province. Also, in Ontario, where monitoring and reporting of *C. chinensis* occurrences has been most consistent, reports indicate that the species is spreading quickly to new water bodies from invaded ones (Fig. 6). This is concerning due to the species wide range of ecological tolerances and suggests that this species is likely to continue spreading.

The source of *C. chinensis* reports varied and is associated with the level of public awareness in each province, and whether there is an easily accessible means of reporting. In British Columbia, *C. chinensis* reports were derived entirely from museum records.

The Prairie provinces (Alberta, Saskatchewan, Manitoba) had no museum records or published reports at the outset of preparing this review. However, Alberta Environment recently confirmed *C. chinensis* as well established in Lake McGregor in southern Alberta, and the provinces has now started an awareness campaign (Alberta Invasive Species Council 2018). This suggests that all Prairie provinces may need to add this species to their non-indigenous monitoring programs. Ontario reporting methods include a preponderance of internet and phone app reports, likely the result of a high level of public awareness of invasive species through educational campaigns. Publicly accessible phone apps and internet databases, such as the Ontario Department of Natural Resources online database (EDDMaps Ontario; 33% of our reports) and iNaturalist (21% of our reports), were the most frequently used tools for reporting public sightings of *C. chinensis* in

Table 3. Canadian *Cipangopaludina chinensis* reports by province, and the total number of water bodies with documented *C. chinensis* occurrence by province.

Provincial data		
Province or territory	Number of reports	No. of invaded waterbodies
British Columbia	12	11
Alberta	1	1
Saskatchewan	0	0
Manitoba	0	0
Ontario	216	105
Quebec	10	9
New Brunswick	7	6
Nova Scotia	30	16
Prince Edward Island	1	1
Island of Newfoundland	1	1
Labrador	0	0
Yukon	0	0
Northwest Territories	0	0
Nunavut	0	0
Total:	**278**	**150**

Ontario. In Quebec, the majority of *C. chinensis* reports were derived from previously published literature (71.4%) with a limited number of reports available at iNaturalist (28.6%). In Atlantic Canada (New Brunswick, Nova Scotia, Prince Edward Island, Newfoundland, and Labrador) scientists researching freshwater lakes (50%) were the most common source of *C. chinensis* records, followed by museum records (25%), and iNaturalist (11%).

Across Canada, established populations of *C. chinensis* have been reported from 146 locations (Table 3): lakes (64), rivers (29), and ponds (25) were the most frequently reported habitat of *C. chinensis*. However, there were some reports of *C. chinensis* occurrence in marshes (1), bays (5), and harbours (1). The number of freshwater systems with reported *C. chinensis* occurrences by province include British Columbia (11), Alberta (1), Ontario (105), Quebec (9), New Brunswick (6), Nova Scotia (16), Prince Edward Island (1), the island of Newfoundland (1), (Table 3). The number of invaded water bodies per province followed the same trend as the number of reported occurrences of *C. chinensis* with the highest number of invaded water bodies in Ontario (Table 3). Previous studies have suggested that the habitat preference of *C. chinensis* as slow or stagnant freshwater (Matthews et al. 2017; McAlpine et al. 2016; Rothlisberger et al. 2010). Our review of Canadian *C. chinensis* reports indicate that *C. chinensis* also inhabit rivers, harbours, and bays, which agrees with Asian literature on this species use of habitat across its indigenous range. Assumptions concerning *C. chinensis* habitats in North America can be risky. For example, in 2019, *C. chinensis* was confirmed as present in a brackish tidal stream (conductivity = 349.8 μs/cm) directly connected with Cow Bay (conductivity = 36 700 μs/cm) in the Halifax Regional Municipality (Fraser, personal observation 2019). This leads to questions about salinity tolerance and thresholds for this species in locations where seasonal salt and freshwater mixing may occur.

This is the first country-wide summary for *C. chinensis* in Canada. A federally-maintained Canadian database would allow researchers and managers to identify what will likely be an expanding distribution, gaps in distributional survey effort, and most importantly, will contribute to predictive distribution models (Haak et al. 2017; Papes et al. 2016; Kingsbury et al. 2021) that may highlight areas of concern (e.g., species and habitats at risk, areas of ecotourism, recreational fishing spots). Species distribution modelling for *C. chinensis* in Atlantic Canada and other parts of North America suggest that water chemistry and ecosystem composition are important factors for successful *C. chinensis* establishment, and human-mediated transfers between waterbodies are significantly correlated with *C. chinensis* introductions (Haak 2015; Haak et al. 2017; Kingsbury et al. 2021; Latzka et al. 2015; Papes et al. 2016). Environmental parameters (Table 1) that may impact *C. chinensis* establishment include calcium concentration, flow rate, pH, maximum water depth, and distance to highway (Chiu et al. 2002; Collas et al. 2017; Fraser et al., unpublished; Haak 2015; Jokinen 1982; Latzka et al. 2015), and there may be others as yet unidentified. Studies focused on factors influencing successful establishment of *C. chinensis* have been to date limited in North America, both in terms of geographic area and in modeling success. In particular, a lack of knowledge of ecological thresholds and ecosystem interactions between *C. chinensis* and other indigenous or invasive species, inconsistency in water chemistry databases, and inconsistent coverage of water chemistry databases across political jurisdictions (e.g., a whole province or state) has limited research outcomes. Research and improved monitoring efforts, both for species occurrence and habitat characteristics, are clearly needed to expand our current understanding of how ecological thresholds influence habitat selection in *C. chinensis*, as these will influence presence/absence surveys and predictive distributional modeling for the species.

Not surprisingly, current Canadian *C. chinensis* distribution is concentrated in areas where human population is most dense. This may reflect actual distribution, but it may also be reflective of reporting opportunity. The distribution of *C. chinensis* in Canada is essentially a northward extension of the species range in the USA, so spatial trends for this species south of the Canadian border merit consideration when examining Canadian trends. The geographical distribution of North American *C. chinensis* reports in both Canada and USA identified a few likely reporting gaps in western Canada, around Lake Superior in Ontario, southern Quebec, and eastern Canada (Fig. 5). For example, *C. chinensis* was first noted in Quebec in 1971 (Jokinen 1982); but there have been very few subsequent reports for the province, even though the snail is commonly found in some areas of the Ottawa and St. Lawrence rivers (A. Ricciardi, McGill University, personal communication, 2020). The prevalence of pleasure craft, developments that redirect and change water flow, and the significant likelihood for aquarium releases suggest a wider establishment for *C. chinensis* in southern Quebec since 1971 (Matthews et al. 2017). Public education of *C. chinensis* as a potential problem for indigenous ecosystems, in parallel with an accessible reporting system (e.g., smartphone apps, email, mobile texting, and phone contacts), should lead to increased reports of occurrence. We recommend that a Canada-wide database of *C. chinensis* occurrences, using the compilation we present here as a foundation, be established, and perhaps maintained by DFO. These occurrence reports can then be included with other geo-tagged data layers for non-indigenous species, vulnerable species and species-at-risk, and vulnerable habitats to determine areas of conservation management priority.

Conclusion

We reviewed 123 literature and internet sources (86 peer-reviewed journal articles, seven grey literature, and 30 internet sources) for both indigenous and non-indigenous *C. chinensis* habitats, compiled North American reported occurrences, and created two conceptual diagrams for both indigenous (Asia) and non-indigenous habitats (North America and Europe). Evidence suggests that *C. chinensis* meets the current federal Canadian definition of an aquatic invasive species.

While we believe the geographical distribution of reported occurrences that we present under-represents actual *C. chinensis* distribution in Canada, this is the first Canada-wide compilation. The data indicate that *C. chinensis* is already widespread in Canada. Our conceptual diagrams suggest freshwater ecosystems in North America may be vulnerable to the negative impacts of

Fig. 6. The number of *Cipangopaludina chinensis* occurrences reported at 5-year intervals across Canada (plot on the left) follows a similar trend to the number of water bodies reported invaded by *C. chinensis* each year in Canada in each province (plot on the right). The frequency of reports made may be increasing due to advancements in technology (e.g., smart phone apps), which may also impact the number of water bodies reported invaded each year. Regardless, the number of water bodies invaded by *C. chinensis* in Canada is increasing. AB, Alberta; BC, British Columbia; NB, New Brunswick; NFL, Newfoundland and Labrador; NS, Nova Scotia; ON, Ontario; PEI, Prince Edward Island; QC, Quebec.
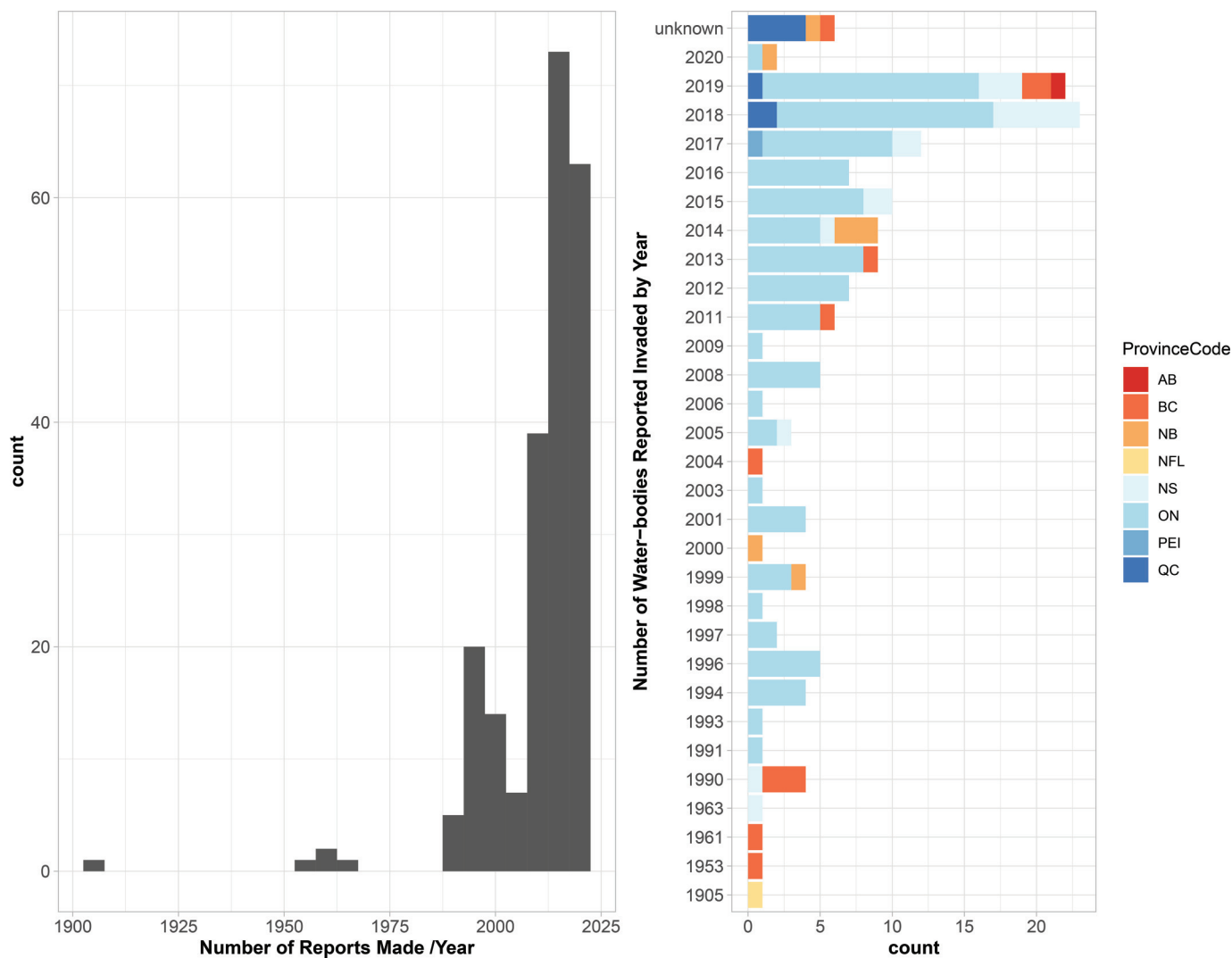
C. chinensis. Our indigenous (Asian range) conceptual diagram suggests consumption of *C. chinensis* by North American wildlife species or humans is problematic, as there is good evidence of *C. chinensis* transferring parasites and contaminants to both humans and wildlife consumers in Asia. With global temperatures predicted to rise, and given the species adaptability and fecundity, *C. chinensis* can be expected to expand its range northward in North America. Additionally, rising temperatures are predicted to lead to an increase in rates of release of nitrogen and phosphorous compounds, impacting food webs by altering bacterial community structure and potentially leading to an increased risk of eutrophication where *C. chinensis* becomes established and abundant.

Due to the difficulty of eradicating *C. chinensis* once established, preventing introduction of the species is the most cost-effective management strategy. This will require increased public education and improved regulations that includes the enforcement of existing AIS regulations on the sale of *C. chinensis* in the aquarium

and water garden trade and prohibition on the release of the species into the environment in Canada. Canada lags in terms of research on *C. chinensis*, with most of the northern temperate literature based on studies from the USA and Europe. While further research is needed in North America to better document *C. chinensis* distribution and to establish predicted impacts, impacts may vary geographically. Those in Canada may depart from impacts documented to date in the southern USA. In summary, we recommend that *C. chinensis* be recognized as an aquatic invasive species (AIS) in Canada (and across the USA) and that the species be managed through a combination of public education and enforcement of regulations.

Acknowledgements

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