SOCIO-ENVIRONMENTAL IMPACTS OF NON-NATIVE AND TRANSPLANTED AQUATIC MOLLUSC SPECIES IN SOUTH AMERICA. WHAT DO WE REALLY KNOW?

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ABSTRACT

The impacts of biological invasions remain poorly known for some habitats, regions and taxa. To date, there has been no comprehensive effort to review and synthesize the impacts of invasive mollusc species in South America (SA). In this paper, we provide a synoptic view on what is known on documented socio-ecological impacts of aquatic no-native mollusc species (NNMS) and transplanted mollusc species (TMS) from SA. An expert group involving malacologists and taxonomists from different countries, the “South America Alien Molluscs Specialists” (eMIAS), shared and summarized the scientific literature, databases, and published and unpublished information on confirmed impacts of NNMS and TMS in SA. Three broad categories, non-mutually exclusive were used as a framework: “Environmental / Biodiversity impacts”, “Economic and social impacts”, and “Human health impacts”. Some 21 NNMS and 7 TMS have documented impacts on at least one of those three categories. We encourage targeting the less known areas of research, such as economic valuation of human health (and veterinary) impacts attributable to NNMS or TMS and expand our knowledge of environmental impacts for the species listed in this study.

Key words: gastropods, bivalves, freshwater species, marine species, invasive species

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INTRODUCTION

Humans are completely dependent on the goods and services provided by Earth ecosystems, such as food, water, disease management, climate regulation and even for the intrinsic value it provides such as spiritual fulfilment and aesthetic enjoyment (Millennium Ecosystem Assessment, 2005). In the last 50 years, humans have changed ecosystems faster and more extensively than in any other comparable period in human history, in large part to meet humans' demands for ecosystem services. The harmful effects from that practice are causing a persistent decline in the ability of ecosystems to provide such services (Millennium Ecosystem Assessment, 2005).

Biological invasions are a significant aspect of the Anthropocene (Campinha et al., 2015; Pyšek et al., 2017) and a constant threat to biodiversity (IPBES, 2019). Humanity has introduced thousands of species to areas outside their native ranges, and while most of these fails to establish viable populations, invasive non-native species have been traditionally identified as one of the main drivers of biodiversity loss worldwide, but their impacts on ecosystem services, sustainable development, and human well-being are poorly quantified and understood (IPBES, 2019). Further, the magnitude of the threat to endangered species is still controversial, due to a scarcity of empirical data and a high degree of uncertainty (Dueñas et al., 2018; Gurevitch & Padilla, 2004). This knowledge gap is more pronounced for some regions and taxa.

The impacts of invasive species have been studied, and reviewed more often for temperate latitudes of the Northern hemisphere in comparison to the Southern hemisphere, for terrestrial rather than aquatic ecosystems, and for plants and insects (together accounting for two-thirds of the studies), in comparison with other taxa (Pyšek et al., 2008). Molluscs, the second most diverse metazoan phylum (Darrigran et al., 2020) are no exception, and there is a direr situation concerning non-native mollusc species (NNMS) in aquatic environments. Molluscs account for only 5% of global studies, and South America is among the regions with fewer studies concerning this topic (Speziale et al., 2012; Thomsen et al., 2014). For example, Thomsen et al. (2014) reported only 10 studies that quantified impacts of 13 aquatic non-native species from a review of 259 papers published between 1972–2012, but no information on aquatic molluscs seems to be included. Similarly, a recent paper addressing the economic cost of biological invasions worldwide (Diagne et al., 2021) found that 79% of the information regarding impacts was gathered form studies performed in North America, Oceania and Europe. These biases affect our understanding and management of this pressing issue.

Non-native aquatic molluscs play important roles in the ecosystems where they are introduced (e.g., as consumers, competitors, hosts or prey). Despite their potential environmental importance, the distribution patterns of NNMS in South America and their entry points have only recently been documented by Darrigran et al. (2020), who listed 86 NNMS distributed in 152 (out of 189) terrestrial, freshwater and marine ecoregions of South American continent. Of those, 30 were aquatic (16 in freshwater and 14 in marine environments). More recently, 20 aquatic transplanted mollusc species (TMS), i.e., native mollusc species introduced deliberately or accidentally beyond their natural range, were recognised in South America (Darrigran et al., 2022).

To date, there have been no comprehensive efforts to review and synthesise the impacts of NNMS and TMS in South American ecosystems, and thus a synoptic picture on the impacts of NNMS and TMS in the region is still lacking. One of the underlying reasons is the greater attention given to *Corbicula fluminea* (Müller, 1774) and *Limnoperna fortunei* (Dunker, 1857), which have been the subject of numerous studies and important reviews (Penchaszadeh, 2005; Darrigran & Damborenea, 2006, 2009; Dreher Mansur et al., 2012; Boltovsky, 2015a). In this work, a synthesis of the known impacts documented in South America for all registered NNMS and TMS is presented. Both *C. fluminea* and *L. fortunei* are included, without claiming an exhaustive review of all published information. Such synthesis aims to provide a better understanding of the present situation on the continent and grant insights for future monitoring and policies, including limiting new introductions. Therefore, in this study we synthesize and provide examples of socio-economic effects and environmental impacts of marine and freshwater NNMS and TMS in South America, highlighting avenues for future research.

MATERIAL AND METHODS

An expert group involving malacologists and taxonomists from different countries of South America (Argentina, Brazil, Chile, Ecuador, Peru, Uruguay, Venezuela), the "South American Alien Molluscs Specialists'' (eMIAS; https://emiasgroup.wixsite.com/emias), reviewed and shared scientific literature (including "grey" literature), collection data, databases and experiences on the subject through a virtual forum. Additionally, the group compiled published information on confirmed impacts of non-native mollusc species (NNMS) and transplanted mollusc species (TMS) in South America. The list of NNMS and TMS was presented in previous contributions of the eMIAS (Darrigran et al., 2020, for NNMS; Darrigran et al., 2022, for TMS). Each contributor provided information based on published evidence and/or research experience according to their expertise, familiar taxa and region. The database on species and impacts was completed with a literature search on Scopus and Google Scholar, with an open search period. Keywords used in the search strategy include "species name," and "impacts" in English, Spanish and Portuguese, identifying those publications relevant to the current study, according to the criteria stated below.

Definitions

We define non-native mollusc species (NNMS) in South America as species introduced outside their natural geographical range through human action, that are able to maintain a self-sustaining population (Darrigran et al., 2020). Transplanted mollusc species (TMS) are defined as species native to South America that underwent changes in their natural distribution within the continent, either through human action or due to human–induced environmental factors (Darrigran et al., 2022). In our discussion, if a given species has an evident impact on the environment and human well-being and livelihoods, it is dubbed an "invasive species" (irrespectively of being a NNMS or a TMS). Cryptogenic species *sensu* Carlton (1996) were not considered.

In the present study, an impact is considered to be a measurable change in the state of a given indicator of an invaded ecosystem, which can be attributed to non-native or transplanted species (Ricciardi, 2003). This definition of impact includes any change in ecological or ecosystem properties but takes no position on whether a given impact is positive or negative value (Jeschke et al., 2014). Therefore, only the effects on human well-being and livelihoods caused by invasive species are considered either as positive or negative. Examples of ecosystem impacts include increased risk of extinction of native species, changes in the genetic composition of native populations, modification of the phylogenetic and functional diversity of invaded communities and food webs, changes in the productivity of ecosystems, nutrient cycling and pollutants (e.g., Pyšek et al., 2020). We acknowledge, however, that ecosystem impacts can also directly or indirectly affect human well-being (e.g., Martinez-Juarez et al., 2015) and that species redistributions itself may impair economic development, livelihoods, food security, human health and culture (Pecl et al., 2017).

There are several frameworks to assess the impacts or effects of aquatic non-native species (e.g., Dextrase & Mandrak, 2006; Everard et al., 2009; Thomsen et al., 2014; Doherty-Bone et al., 2019; Pyšek et al., 2020), which can be grouped into three broad and non-mutually exclusive categories: "Environmental impacts or Ecological impacts" or "Biodiversity impacts" (i.e., impacts on "wild" populations, communities, species or ecosystems). Similarly, effects on human well-being and livelihoods are often divided in "Economic and social impacts" and "Human health impacts". The latter two pertain to different dimensions of human well-being and, although effects in human health can also be considered Economic and Social impacts, we maintained these two categories separated (cf. Martuzzi, 2005; Ebi et al., 2006; Zeimes et al., 2012; Pedersen et al., 2014).

Herein, we focused on the documented impacts and effects of invasive species in South America, not considering possible risks and potential threats. Therefore, studies reporting range expansion and first records of a given species in a certain area were not considered if they lack significant observations on local impacts, even though those studies often include a list of potential impacts based on what is known from elsewhere. Thus, we did not consider impacts that have been reported from other continents where the same species has been introduced. When available, some experimental results were included, although we did not necessarily affirm that the reported interactions are occurring in nature.

For the category of environmental impacts, it may be argued that just by the arrival of a NNMS or TMS there is a modification of the biogeographic distribution of native taxa, causing a change in several community-level attributes, such as local species composition, diversity and evenness of local communities. In this study, we focused mainly on conspicuous changes in local community structure driven by the abundance of NNMS and TMS, and/or their incorporation into food webs. These effects may be particularly relevant in human-modified ecosystems invaded by bivalves (Burlakova et al., 2022). Other documented impacts may include changes on abiotic conditions directly attributable to the presences of NNMS or TMS, or genetic interaction with local species (e.g., hybridization). Correlational evidence for some impacts was accepted as a “documented” impact, but these cases clearly deserve further experimental analysis to elucidate the underlying mechanisms or to confirm cause-effect relationships.

Socio-economic effects include direct and indirect monetary costs associated with the action of invasive species (Burlakova et al., 2022 and Adelino et al., 2021; Diagne et al., 2021, respectively). For example, reduction or loss of profits due to the effect of mollusc-borne parasites in domestic cattle may be particularly difficult to quantify or even estimate. We thus considered primarily those reports highlighting the interaction of NNMS and TMS species with economic activities. Furthermore, some species were introduced for the development of commercial aquaculture, and some accidentally introduced species may also be commercially exploited.

Some NNMS can cause the spread of new and/or existing diseases acting as vectors of pathogens. In the public health category, we were more liberal, so any reports documenting the presence of a human or veterinary parasite or pathogen in a NNMS or TMS in South America were considered. Other potential effects include allergic reaction, ingestion of toxins, loss of aesthetical value or mechanical harms of several sorts (Mazza et al., 2013). Clearly, public health effects further include an economic dimension, which should be considered elsewhere.

RESULTS

The information on confirmed impacts and effects of NNMS and TMS in South America is synthesised in Tables 1 and 2. A total of 28 mollusc species was documented as having impacts in South America, 21 of them are NNMS (nine freshwater and 12 marine) and seven are TMS (two freshwater and five marine). All marine TMS are bivalves, *Leiosolenus aristatus* (Dillwyn, 1817)*, Mytella strigata* (Hanley, 1843), *Mytilopsis trautwineana* (Tryon 1866), *Argopecten purpuratus* (Lamarck, 1819) and *Tawera elliptica* (Lamarck, 1818) and cause economic and social impacts. *A*. *purpuratus* is included on the basis of being transplanted for commercial aquaculture. In the freshwater environment, *Anodontites trapesialis*

(Lamarck, 1819) causes economic and social effects. On the other hand, a species of gastropod, *Pomacea canaliculata* (Lamarck, 1822) has documented impacts on the three categories (Biodiversity, Economic and social effects, Human health effects). However, this species can also exhibit positive effects as a potential control of the NNMS *Physa acuta* Draparnaud, 1805. Altogether, negative effects are more commonly documented than positive effects. Finally, several NNMS are listed in more than one category (see Tables 1 and 2). In the marine environment, nine NNMS cause Biodiversity impacts, nine Economic and social effects and two Human health effects. In freshwater, seven NNMS cause Biodiversity impacts, seven Economic and social effects, while three effects Human health.

DISCUSSION

Documented Impacts on Biodiversity

Marine Ecosystems

There are 14 aquatic NNMS that were intentionally introduced to develop commercial marine aquaculture (Darrigran et al., 2020). However, only two of these species (the abalones *Haliotis discus* Reeve, 1846 and *Haliotis rufescens* Swainson, 1822) are not established in natural environments, and they have been highlighted as a threat to native and cultured species as a vector that facilitates the spread of boring polychaetes (Moreno et al., 2006; Diez et al., 2011). The remaining 12 species are currently distributed in coastal environments along South America, where they are at least modifying species composition and relative abundances within communities, which can be viewed as a primary impact on biodiversity. For some species, [e.g., *Perna viridis* (Linnaeus, 1758), *Isognomon bicolor* (Adams, 1845), *Magallana gigas* (Thunberg, 1793) [=*Crassostrea gigas* (Thunberg, 1793)] and *Eualetes tulipa* (Rousseau in Chenu, 1843)] there are studies quantifying densities or abundances, and that provide a description of community structure after the arrival of NNMS. Often, those species that increase the heterogeneity of native environments (e.g., *M. gigas* reefs on mudflats) cause shifts in the occurrence and abundances of associated species (Melo et al., 2010; Ludwig et al., 2011; Mendez et al., 2015), thus increasing alpha diversity at a local scale. Studies on some other encrusting, hard-bottom species (e.g., *I. bicolor*) have likewise been carried out, and most include occurrence reports and abundance estimates (Ignacio et al., 2010; Dias et al., 2013; Agostini & Ozorio, 2016; Oricchio et al., 2019).

Other impacts at the functional level include the incorporation of NNMS in local food webs. For example, *Rapana venosa* (Valenciennes, 1846) seems to be an important food item for Loggerhead turtle, *Caretta caretta* (Linnaeus 1758), in the Río de la Plata estuary (Carranza et al., 2010a). Another example is *I. bicolor* that has been shown to cause changes of food habit in the gastropod *Stramonita haemastoma* (Linnaeus, 1758), which fundamentally preyed on the mussel *Perna perna* (Linnaeus, 1758), native species according to Darrigran et al. (2020), before the arrival of *Isognomon* (López et al., 2010). There are other interactions reported, such as the massive fouling of *R. venosa* on green sea turtles *Chelonia mydas* (Linnaeus, 1758) (Lezama et al., 2013), although the effects on individual fitness are yet to be confirmed. Similarly, due to the predatory role and high local abundances of *R. venosa*, this species could be significantly affecting some ecological properties of their intertidal habitat, such as mussel coverage on rocky bottoms (Carranza et al., 2010b), but no studies have quantified the extension of this presumably environmental impact.

Freshwater Ecosystems

Reports of biodiversity impacts of NNMS/TMS are available for only eight species in freshwater environments. One of the best studied species is the golden mussel, *Limnoperna fortunei*, which is the most aggressive aquatic invasive in South America. The rapid spread of *L. fortunei* populations in hydrographic basins have been attributed to human-mediated dispersal (Belz et al., 2012; Boltovskoy, 2015b; Borges et al., 2017; Ludwig et al., 2021).

Populations of *L. fortunei* are found on virtually any natural hard surface available (e.g., logs, water vegetation, and compact sandy silt), as well as any artificial structure and substrate (e.g., walls, piers, pipes, glass, nylon) (Darrigran & Damborenea, 2005). De Lucía et al. (2023) recommend conservation efforts given the constant advance of urbanization, with environmental impact studies prior to coastal reforms, and implementation of density control strategies for *Limnoperna fortunei* in protected areas. Considering the serious problems that it causes, it is astonishingly overlooked by society and governments in South America. The golden mussel modifies environmental conditions of invaded South American inland freshwater environments, altering both abiotic and biotic variables affecting ecosystem services, with large environmental and socio-economic impacts (Darrigran & Damborenea, 2011; Boltovskoy & Correa, 2015). Impacts of *Limnoperna fortunei* are difficult to interpret due to the multiple interactions with the biotic and abiotic components and their dynamics and to the regional environmental conditions. So, the impacts are variable in the medium and long terms, and in both local and regional scale. The impacts and the effects are reflected in the high number of publications. Boltovskoy (2015a) and Burlakova et al. (2022) summarized the scale and variety of the environmental impacts and economic and human well-being effects caused by the golden mussel. In this contribution we only have addressed the most conspicuous effects, such as fouling on native molluscs and other macroinvertebrates (including *Anodontites trapesialis* and *C. fluminea* [Darrigran, 2002], the crabs *Trichodactylus borellianus* [Rojas Molina & Williner, 2013] and *Aegla platensis*, and the gastropod *Pomacea canaliculata* [Darrigran & Damborenea, 2005; Silva et al., 2021a], impacts on benthic communities, fish communities, bioaccumulation of metals, impacts in water column, nutrient cycling, and on plankton communities and cyanobacteria blooms (Table 1). In summary, *L. fortunei* is a very effective ecosystem engineer, altering both the structure and function of the ecosystem (Darrigran & Damborenea, 2011; Boltovskoy, 2015a).

Four NNMS of the genus *Corbicula* were recorded in South America [*C. fluminea*, *C. largillierti* (Philippi, 1844), *C. fluminalis* (Müller, 1774) and *Corbicula* sp.] (Mansur et al., 2011). Among these species, *C. fluminea* causes a severe impact on the environment. This species invaded ecosystems around the world, being present between 39° South and 53° North. In less than 100 years, it has invaded all continents except Antarctica, being one of the most successful invasive species in aquatic ecosystems (Crespo et al., 2015). In the hydrographic basins of South America, the macroinvertebrates assemblages are mainly impacted by displacement and reduction of available habitat (Darrigran et al., 2020; Labaut et al., 2021). Thus, like the golden mussel, *C. fluminea* often plays a role of ecosystem engineer, causing physical disruptions wherever it establishes and changing the structure of macroinvertebrate benthic communities (Reshaid et al., 2017). Labaut et al. (2021) observed that on the Limay River, in the Argentinean Patagonia, *C. fluminea* impacts the abundance of some taxa, due to the competition for resources in a low productivity ecosystem. The faeces and pseudo-faeces of C. *fluminea* deposited on the sediment enrich their organic content. However, they compete for food with benthic macroinvertebrates. Sites invaded by *C. fluminea* showed a tendency towards homogenization of species and functional composition (Labaut et al., 2021). However, in other cases, the evidence for competitive displacement of native species is not always strong. Clavijo and Carranza (2014), analysing the correlation between the critical reduction of the distribution of the native *Cyanocyclas* spp. and the spread of *Corbicula* in Uruguay, proposed the interplay between a) the direct adverse effect of interspecific competition with the Asiatic clam, and/or b) the degradation of environmental conditions leading to the disappearance of the native species and their replacement by opportunistic species. Both hypotheses should be regarded as extremes of a continuum, with several intermediate scenarios likely to coexist.

Reproductive studies offer a solid basis for predictive trends of the invasion of populations of *C. fluminea*. The reproductive features (Pigneur et al., 2014; Ludwig et al., 2014; Cao et al., 2017) facilitate the survival of *C. fluminea* from Venezuela (10°10'S - 63°30'W) to Patagonia Argentina (39°28'S - 68°58'W) (Labaut et al., 2021), being present in about half of the South American freshwater ecoregions (Darrigran et al., 2020). The rapid spread of *C. fluminea* in South America has involved humans as vectors, either transporting individuals in the bilge water of crafts, with or as fish bait, in dredged river sand, as juveniles attached to boat hulls, and by aquarium hobbyists (McMahon, 2000; Belz et al., 2012; Labaut, 2021).

Other NNMS freshwater species with reported impacts in South America are *Melanoides tuberculata* (Müller, 1774) and the New Zealand mud snail *Potamopyrgus antipodarum* (Gray, 1843). In Brazil, *M. tuberculata* has negatively affected native populations of *Pomacea lineata* (Spix *in* J. A.

Wagner, 1827) in Rio de Janeiro state, *Biomphalaria glabrata* (Say, 1818) in Minas Gerais and Rio de Janeiro states, *Biomphalaria straminea* (Dunker, 1848) in Minas Gerais, and *Aylacostoma tenuilabris* (Reeve, 1860) in the Tocantins River, Goiás (Guimarães et al., 2001; Giovanelli et al., 2002; Fernandez et al., 2003). Similarly, Collado et al. (2019) reported correlational evidence of competitive displacement of native gastropods by *P. antipodarum* in Chile. Interactions of native species with NNMS or TMS are also worth evaluating. Maldonado & Martin (2019) experimentally evaluated the effects of *Pomacea canaliculata*, *Melanoides tuberculata* and *Physa acuta* Draparnaud, 1805 on native snails [*Heleobia parchappii*, (d'Orbigny, 1835), *Biomphalaria peregrina* (d'Orbigny, 1835), and *Chilina parchappii* (d'Orbigny, 1835)], showing negative interactions including reduced fecundity in *P. acuta* and *B. peregrina*, although the NNMS *Melanoides tuberculata* was not affected by *P. canaliculata*. Thus, the impact of *P. canaliculata* in recently colonised regions of South America deserves further attention.

Documented Socio-Economic effects

Marine Ecosystems

So far, there are few documented negative effects on economic activities by NNMS in South American marine ecosystems. The mytilids *Mytella strigata* and the false mussels *Mytilopsis* spp. have been reported to produce a trophic imbalance in culture pools, decreasing production in shrimp farming as well as fouling in some structures (Aldridge et al., 2008; Lodeiros et al., 2019, 2021). Similarly, the boring TMS *L. aristatus* caused damage to shells of the cultured scallop *Nodipecten nodosus*, producing serious scars, deformations and even death, in a marine farm in São Paulo state (Brazil; Simone & Gonçalves 2006). Additionally, *Talonostrea talonata* Li & Qi, 1994 [=*Crassostrea talonata* (Li & Qi, 1994)] may outcompete *Crassotrea tulipa* (Lamarck, 1819) [=*Crassostrea gasar* (Lamarck, 1819)], being a nuisance species in oyster culture (Cavaleiro et al., 2019). Finally, the vermetid *Eualetes tulipa* fouls power plant turbines in Venezuela (Miloslavich & Penchaszadeh, 1992).

On the other hand, positive economic return is associated with commercial cultures of *Haliotis discus* and *Haliotis rufescens* (Flores Aguilar et al., 2007; Castilla & Neill, 2009; SUBPESCA, 2021) and the Pacific oyster *Magallana gigas* (Furse et al., 2004; dos Santos & Costa, 2016; Martínez-García et al., 2021). This kind of introductions for commercial aquaculture often presents positive social effects such as direct income, increased employment and associated research. In this line, the development of experimental aquaculture may also be considered as a positive effect associated with the green mussel *Perna viridis* in Venezuela, since it provides new employment opportunities for local researchers and workers (Acosta-Balbás et al., 2019).

Another interesting effect to be more carefully analysed is the claim that NNMS act as vectors of boring polychaetes. Once marine species are introduced to new areas for aquaculture, their associated epibionts can also be accidentally introduced. This may pose a risk both to the economic activity and the native biodiversity, since non-native epibionts may be able to exploit new native hosts (e.g., Kuris & Culver 1999). This effect could change population and community composition and dynamics (Grosholz et al., 2000), but this phenomenon remains poorly understood in South America. However, Moreno et al. (2006) pointed out that aquaculture activities may be the primary introduction vector for boring polychaete species in Chile. Similarly, spionid polychaetes heavily parasitize and destroy the shell of the invading *Rapana venosa* in Uruguay (A. Carranza, unpublished), and in certain areas it may be exerting some control of the invader species. However, the identity and biogeographic origin of the polychaete species involved is hard to elucidate.

Freshwater Ecosystems

Limnoperna fortunei easily invades water transfer tunnels and attaches to tunnel walls and structures with extremely high density, resulting in biofouling and being responsible for negative effects on hydropower generation, water quality, and damages in man-made structures (Adelino et al., 2021). The effect on turbine components occurs by hydro-abrasion; the abrasiveness of the golden mussel shell was compared with that of silicon carbide (SiC) and the wear mechanisms acting on the SiC tests are the same as for the mussels (de Castro et al., 2019). Additionally, the consequences of the establishment of *L. fortunei* also include reduction in pipe diameter or outright blockage of pipes, water contamination by massive mortality of individuals, and obstruction of cooling systems (Darrigran, 2010; Boltovskoy, 2015a). Rebelo et al. (2018) estimated that the cost of monitoring and maintenance due to golden mussel fouling in the infrastructure of hydroelectric power plants in Brazil ranges between USD 6.9 and 8 million annually, and the economic losses in that country due to the stoppage of a turbine are in the order of USD 120 million a year. For Argentina, Duboscq-Carra et al. (2021) indicated a cost of around USD 2 million from three reports on management, while Haubrock et al. (2022) reported a total of USD 40.5 million between 2001 and 2020 for South America.

In contrast with the effects of *Corbicula fluminea* reported from North America (McMahon, 2000), in South America the only known report come from a hydroelectric power station in the Rio Grande do Sul state, Brazil, where it fouled heat exchangers in 1988 (dos Santos et al., 2012).

Another socio-economic issue is reported for *Anodontites trapesialis*, a TMS whose larvae heavily parasitize some fish cultures in South America (Silva-Souza & Eiras, 2002; Felipi & Silva-Souza, 2008; Agudo-Padrón, 2019). Furthermore, the mollusc-borne fluke *Philophthalmus gralli* Mathis and Leger, 1910 (Digenea, Philophthalmidae; hosted by *Melanoides tuberculata*) can infect poultry causing profit loss (Pinto & de Melo, 2010). Well-documented direct economic effects of *Pomacea* spp. in rice cultures has also been reported (Wiryareja & Tjoe-Awie, 2006; Agudo-Padrón et al., 2010; Horgan et al., 2014a, 2014b; Correoso Rodriguez et al., 2017).

Pseudosuccinea columella (Say, 1817) and *Galba truncatula* (O. F. Müller, 1774) are vectors for the trematodes *Fasciola hepatica* Linnaeus 1759 (Digenea, Fasciolidae) and *Cotylophoron cotylophorum* (Fischoeder, 1901) (Digenea, Paramphistomidae), which can infect domestic cattle, resulting in deteriorated condition of infected individuals and consequent economic losses (Ueta, 1980; Heinzen et al., 1994; Mas-Coma et al., 2001; Salazar Jaramillo et al., 2006; Lopez et al., 2008; Prepelitchi & Wisnivesky-Colli, 2013).

Impacts on Public Health

Marine ecosystems

No public health issues or even risks are reported associated with most marine NNMS. The only exception pertains to the sea slug *Pleurobranchaea maculata* (Quoy & Gaimard, 1832), which can carry neurotoxins that affect human and domestic animals (Bökenhans et al., 2019). The presence of the bacteria *Xenohaliotis californiensis* and the probable presence of *Bonamia* sp. in abalone cultures is also worth noting (Campalans & Lohrmann, 2009).

Freshwater ecosystems

At least four NNMS can be hosts of pathogen parasites that cause human diseases. The liver fluke *Fasciola hepatica*, that causes human fasciolosis, has been reported in the lymnaeid snails *Pseudosuccinea columella* (Ueta, 1980; Heinzen et al., 1994; Salazar Jaramillo et al., 2006; Prepelitchi & Wisnivesky-Colli, 2013) and *Galba truncatula* (Mas-Coma et al., 2001; Esteban et al., 2002). In the Bolivian Altiplano, where endemic fasciolosis has been reported since 1984, the transmission to humans appears to be linked with the ingestion of aquatic plants infected with metacercariae, and the prevalence of the disease is correlated with the presence of snails (Marcos et al., 2006; Parkinson et al., 2007). Genetic evidence from individuals of *Fasciola hepatica* and *Galba truncatula* suggest a recent introduction from Europe (Mas-Coma et al., 2001), and concomitantly, prevalence and intensity of human fasciolosis in the northern Bolivian Altiplano are the highest reported to date.

The freshwater gastropod *Melanoides tuberculata* can also act as an intermediate host of the trematode *Centrocestus formosanus* Nishigori, 1924 (Digenea, Heterophyidae) (Hernández et al., 2003; Velásquez et al., 2006; Pinto et al., 2018), which can infect humans through ingestion of raw or undercooked parasitized fish, causing gastric pain and indigestion accompanied by diarrhea (Chai et al., 2013). However, there are no reported cases in South America.

In 2008, the presence of *Angiostrongylus cantonensis* (Chen, 1935) (Nematoda, Angiostrongylidae) was reported for the first time in Ecuador, as well as the first cases of an emerging disease caused by the larval stage, eosinophilic meningitis. Several authors have highlighted the apple snail *Pomacea canaliculata* as an intermediate host of *A. cantonensis* in Ecuador (Solórzano Álava et al., 2014; Correoso Rodriguez et al., 2017; Thiengo et al., 2017). In 2015 an experimental infection of *Pomacea canaliculata* with *Angiostrongylus vasorum* (Baillet, 1866), which infects the heart and pulmonary artery of domestic and wild canids was reported (Mozzer et al., 2015).

On the other hand, *Schistosoma mansoni* Sambon, 1907 (Digenea, Schistosomatidae) is a blood fluke causing schistosomiasis in humans, depending on Planorbidae snails as intermediate hosts. This tropical disease is largely neglected but ranks amongst the most prevalent in humans: in 2021, the World Health Organisation reported 236.6 millions of people diagnosed with schistosomiasis in Africa, the Middle East, the Caribbean, Brazil, Venezuela and Suriname. In this case, *Marisa cornuarietis* (Linnaeus, 1758) has been regarded as a biological control of schistosomiasis vectors, thus providing an example of a positive impacts of a NNMS in the Public Health dimension.

Finally, the New Zealand mud snail *Potamopyrgus antipodarum* is another NNMS known to host parasites of veterinary and human health relevance, such as *Sanguinicola* sp. (Bacteria)*,* *Paracardicoloides yamagutii* Martin, 1974 (Digenea, Aporocotylidae) and *Notocotylus gippyensis* (Beverley-Burton, 1958) (Digenea, Notocotylidae) (Hine, 1978; Morley, 2008), but no study has yet analysed their prevalence in South America.

CONCLUDING REMARKS

Twenty-eight NNMS and TMS are known to have documented impacts and effects on at least one of the three dimensions here considered. Given that South America is a large and heterogeneous continent, it is unclear how impacts or effects (positive or negative) of a NNNMS or TMS can be distributed along a species distribution range. However, this contribution provides a synoptic view of the literature at a continental scale, and thus can be useful to direct future research priorities. The first interesting fact emerging from our study, is that 70% of all NNMS from marine and freshwater habitats in South America (30 species according to Darrigran et al., 2020) had documented impacts and effects compared with only 41% of all TMS (14 species according to Darrigran et al., 2022). Thus, the overall impact of NNMS exceeds that of TMS, and/or alternatively, there may be a bias towards documenting impacts of exotic, well known invasive species. This putative bias should be further investigated, since there are known biases towards reporting negative over positive effects (e.g., Boltovskoy et al., 2021, 2022). This provides interesting avenues for new research, and to disentangle if this perceived pattern is correlated with biological reality or a publication bias. Notice, however, that we were not able to compare the relative magnitude of these impacts and effects. Besides, impacts have different levels of certainty. Studies reporting correlational evidence were often included as an impact (e.g., Clavijo & Carranza, 2018), in particular when direct quantitative estimates were lacking. Further work should focus on a deeper analysis of these claimed or suggested impacts. Finally, there is not a clear relationship between the direct impact of NNMS and TMS in aquatic environments of South America and losses in native biodiversity, in line with previous work suggesting that the main threat drivers are habitat loss, overharvesting and habitat disturbances (e.g., Dueñas Gurevitch et al., 2018; Gurevitch & Padilla, 2004).

Except for costs associated with the control of *Limnoperna fortunei*, quantified direct economic effects are scarce in the available data and literature. Our results provide an underestimation of the environmental impacts of NNMS and TMS in South America, due to both underreporting and the often-considerable lag between first record, identification and communication of new NNMS and TMS (Pires Teixeira & Creed, 2020). Among the ecosystem services recognised (Millennium Ecosystem Assessment, 2005), the results of this work show the alterations caused by NNMS and TMS in South America directly on provisioning, regulation, and supporting services (Tables 1 and 2), but do not often consider the cultural services of ecosystems. However, there is evidence that indicates that both directly (e.g., injuries caused in bathers by mussel colonies in recreational waterbodies) and indirectly (e.g., enhancing cyanobacterial blooms), NNMS and TMS can affect recreational, aesthetic, and spiritual services.

The effective control of established invasive species remains a pressing challenge for most South American ecosystems. If this control is not achieved, it is very likely that the dispersal of species mediated by humans will cause the breakdown of biogeographic barriers and that, not only climate, but also to some extent, socio-economic relations will define biogeography in an era of global change (Campinha et al., 2015). A lot of work remains to be done concerning the impact of NNMS and TMS in South America. In this vein, it is worth noting that the listed impacts and positive or negative effects for all established categories may be based on a single study for a given region. We encourage targeting less explored areas of research, such as economic valuation of human health (and veterinary) effects attributable to NNMS/TMS, and expanding the knowledge of environmental impacts for all the species listed here. We hope that this review will help direct efforts of the research community in South America and beyond to achieve a multidisciplinary approach in investigating the socio-ecological effects of biological invasions in aquatic habitats.

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TABLE 1. List and summary of documented impacts and effects of freshwater species in South America, according to the Environmental / Biodiversity, Socio-Economic and Public Health categories. *NNMS: non-native mollusc species; **TMS: transplanted mollusc species. (-) negative effect, (+) positive effect.

TAXA	ENVIRONMENTAL/ BIODIVERSITY IMPACTS	SOCIO-ECONOMIC EFFECTS	PUBLIC HEALTH EFFECTS
BIVALVIA			
Anodontites trapesialis ** (Mycetopodidae)		**(-)** Effects in fish cultures via glochidiosis [1-3]	
Corbicula fluminea * (Cyrenidae)	Competitive displacement of native bivalves [4-6] and other invertebrates [7]. Empty shells provide shelter and substrate for other species [7].	**(-)** Macrofouling in heat exchangers, hydroelectric power station [8]. **(+)** Bioindicator [9]; **(+/-)** Bioacumulate lead, cadmium and copper [10].	
Corbicula largillierti * (Cyrenidae)	Competitive displacement of native bivalves [11].	**(+)** Bioindicator [12] and biomarker of Chlorothalonil (CLT) [13]. **(-)** Obstruction of the refrigeration system of power generation facilities [8].	
Limnoperna fortunei * (Mytilidae)	Overgrowth of other organisms [14-17]. Impacts on benthic communities [18-25]. Predation by larval and adult fishes [24, 26-33]. Impacts on the water column – nutrient recycling [34-37]. Water clarification and plankton grazing [27, 34-44], enhancement of Cyanobacteria [41, 43].	**(-)** Fouling on a wide array of human infrastructure: affects water supply sources for drinking water treatment plants, industrial refrigeration systems, fire protection systems and power plants [14, 15, 44-52]. **(-)** Fish-farming [53, 54]. **(+)** Bioindicator [54]. **(+/-)** Bioaccumulation of heavy metals [55].	
GASTROPODA			
Galba truncatula * (Lymnaeidae)		**(-)** Vector of *Fasciola hepatica* [56, 57] and *Cotylophoron cotylophorum* [58].	**(-)** Vector of *Fasciola hepatica* [56, 57, 59, 60].
Marisa cornuarietis * (Ampullariidae)	Competition with and predation of native vector snails [61].	**(+)** Pet trade [61].	**(+)** Control of *Schistosoma mansoni* vectors [61].

Melanoides tuberculata * (Thiaridae)	Competitive displacement of local gastropods [62-65].	(-) Vector of *Philophthalmus gralli* (Digenea, Philophthalmidae) [66].	
Physa acuta * (Physidae)	Incorporation in local food webs [67].		
Pomacea canaliculata ** (Ampullariidae)	Potential control of *Physa acuta* [67].	**(-)** Effects on rice culture [61, 68-71]. **(-)** Vector of *Fasciola hepatica* and *Cotylophoron cotylophorum* [58, 74].	**(-)** Vector of *Angiostrongylus cantonensis* [71-73]. **(-)** Vector of *Fasciola hepatica* [67, 74, 75].
Pseudosuccinea columella * (Lymnaeidae)		**(-)** Vector of *Philophthalmus gralli* [66].	**(-)** Vector of *Centrocestus formosanus* [76-78].
Potamopyrgus antipodarum * (Tateidae)	Competitive displacement of local gastropods [79].		

[1] Silva-Souza & Eiras, 2002; [2] Felipi & Silva-Souza, 2008; [3] Agudo-Padrón, 2019; [4] Pereira et al., 2013; [5] Reshaid et al., 2017; [6] Clavijo & Carranza, 2018; [7] Labaut et al., 2021; [8] dos Santos et al., 2012; [9] Guimarães & Barbujiani Sígolo, 2008); [10] Cataldo et al., 2001; [11] Clavijo, 2014; [12] Reyna et al., 2019; [13] Reyna et al., 2021; [14] Darrigran, 2002; [15] Darrigran & Damborenea, 2005; [16] Silva et al., 2021a]; [17] Rojas Molina & Williner, 2013; [18] Darrigran et al., 1998; [19] Sylvester et al., 2007a; [20] Sardiña et al., 2008; [21] Sardiña et al., 2011; [22] Sylvester & Sardiña, 2015 and references therein; [23] Duchini et al., 2018; [24] Silva et al., 2021b; [25] Silva Bertão et al., 2021; [26] Penchaszadeh et al., 2000; [27] Boltovskoy et al., 2006; [28] García & Montalto, 2006; [29] Paolucci et al., 2007; [30] Sylvester et al., 2007b; [31] González-Bergonzoni et al., 2010; [32] Cataldo, 2015; [33] Paolucci & Thuesen, 2015; [34] Cataldo et al., 2012b; [35] Boltovskoy et al., 2009; [36] Boltovskoy et al., 2015 and references therein; [37] Burlakova et al., 2022; [38] Rojas Molina & José de Paggi, 2008; [39] Rojas Molina et al., 2010, [40] Rojas Molina et al., 2015 y referencias en el mismo; [41] Cataldo et al., 2012a; [42] Rojas Molina et al., 2012; [43] Boltovskoy et al., 2013; [44] Darrigran & Pastorino, 1995; [45] Darrigran & Damborenea, 2011; [46] Brugnoli et al., 2005, [47] Brugnoli et al., 2006; [48] Darrigran et al., 2007; [49] Boltovskoy & Correa, 2015; [50] Resende et al., 2014; [51] de Castro et al., 2019; [52] Hermes-Silva et al., 2021; [53] Costa et al., 2018; [54] Besen & Garcia Marengoni, 2021; [55] Marengoni et al., 2013; [56] Salazar Jaramillo et al., 2006; [57] Prepelitchi & Wisnivesky-Colli, 2013; [58] Lopez et al., 2008; [59] Ueta, 1980; [60] Heinzen et al., 1994; [61] Horgan et al., 2014b; [62] Fernandez et al., 2001; [63] Fernandez et al., 2003; [64] Guimarães et al., 2001; [65] Giovanelli et al., 2002; [66] Pinto & de Melo, 2010; [67] Maldonado & Martín, 2019; [68] Wiryareja & Tjoe-Awie, 2006; [69] Agudo Padrón et al., 2010; [70] Horgan et al., 2014a; [71] Correoso Rodriguez et al., 2017; [72] Solózano Álava et al., 2014; [73] Thiengo et al., 2017; [74] Mas-Coma et al., 2001; [75] Esteban et al., 2002; [76] Hernández et al., 2003; [77] Velásquez et al., 2006; [78] Pinto et al., 2018; [79] Collado et al., 2019.

TABLE 2. List and summary of documented impacts and effects of marine species in South America, according to the Environmental / Biodiversity, Socio-Economic and Public Health categories. *NNMS: non-native mollusc species; **TMS: transplanted mollusc species. (-) negative effect; (+) positive effect.

TAXA	ENVIRONMENTAL/ BIODIVERSITY IMPACTS	SOCIO-ECONOMIC EFFECTS	PUBLIC HEALTH EFFECTS
BIVALVIA			
Argopecten purpuratus* * (Pectinidae)		**(+)** Commercial aquaculture [1].	
***Isognomon bicolor* *** (Isognomonidae)	Habitat modification [2]. Incorporation in local food webs [3].	**(-)** Fouling on pipeline monobuoys [4].	
Leiosolenus aristatus** (Mytilidae)		**(-)** Boring in shells of cultured scallops [5]	
***Magallana gigas* *** **[=*Crassostrea gigas*]** (Ostreidae)	Habitat modification [6, 7]. Increased diversity of macrofaunal benthic assemblages [8]. Probable vector for boring polychaetes infecting native mollusc species [9].	**(+)** Commercial aquaculture [10, 11]. **(-)** Probable vector for introduced boring polychaetes infecting cultured species [9, 12].	
Mytella strigata* * (Mytilidae)		**(-)** Fouling in culture structures and trophic imbalance in the culture pools [13].	
***Mytilopsis leucophaeata* *** (Dreissenidae)		**(-)** Fouling in culture structures and trophic imbalance in the culture pools, [14]	
Mytilopsis trautwineana** (Dreissenidae)		**(-)** Fouling in culture structures and trophic imbalance in the culture pools	

		[15]. Calculated incurred cost in South America of USD 0.007 billion [16].	
***Mytilus galloprovincialis* *** (Mytilidae)	Hybridization with local Mytilidae [17, 18].	**(-)** Fouling in culture structures [19].	
***Perna viridis* *** (Mytilidae)	Habitat modification [20].	**(+)** Experimental aquaculture [21].	
***Saccostrea cucullata* *** (Ostreidae)	Probably reducing available habitat in mangrove ecosystems [22].		
***Talonostrea talonata* *** **[=*Crassostrea talonata*]** (Ostreidae)		**(-)** Nuisance species for oyster *Crassostrea tulipa* culture (space competition) [23].	
Tawera elliptica* * (Veneridae)		**(+)** Commercial aquaculture [24]	
GASTROPODA			
***Eualetes tulipa* *** (Vermetidae)		**(-)** Fouling on power plant turbines [25].	
***Haliotis discus* *** (Haliotidae)	Substrate for native boring polychaetes [10].	**(+)** Commercial aquaculture [26-28].	
***Haliotis rufescens* *** (Haliotidae)	Substrate for native boring polychaetes [10].	**(+)** Commercial aquaculture [26-28]. **(-)** Probable vector for introduced boring polychaetes infecting cultured species [10]. **(-)** Probable presence of *Bonamia* sp. [29].	**(-)** Presence of the bacteria *Xenohaliotis californiensi* [29].

Pleurobranchaea maculata * (Pleurobranchaeidae)	Predation on native benthic species [30].		**(-)** Presence of neurotoxins that affect human and domestic animals [30].
Rapana venosa * (Muricidae)	Predation on native bivalves; [31-34]. Fouling on green turtles [31] Incorporation in local food webs [35-37].		

[1] Von Brand et al., 2016; [2] Breves-Ramos et al., 2009; [3] López et al., 2010; [4] Agostini & Ozorio, 2016; [5] Simone & Gonçalves 2006; [6] Melo et al., 2010; [7] Mendez et al., 2015; [8] Bazterrica et al., 2022; [9] Moreno et al., 2006; [10] Furse et al., 2004; [11] dos Santos & Costa, 2016; [12] Diez et al., 2011; [13] Lodeiros et al., 2021; [14] Lodeiros et al., 2019; [15] Aldridge et al., 2008; [16] Haubrock et al., 2022; [17] Westfall & Gardner, 2013; [18] Zbawicka et al., 2018; [19] Belz et al., 2020]; [20] Villafranca & Jiménez, 2006; [21] Acosta et al., 2006; [22] do Amaral et al., 2020; [23] Cavaleiro et al., 2019; [24] Oliva & Durán 2012; [25] Miloslavich & Penchaszadeh, 1992; [26] Flores-Aguilar et al., 2007; [27] Castilla & Neil, 2009; [28] SUBPESCA, 2021; [29] Campalans & Lohrmann, 2009; [30] Bökenhans et al., 2019; [31] Carranza et al. 2010a; [32] Carranza et al. 2010b; [33] Giberto et al., 2011; [34] Lanfranconi et al., 2013; [35] Lezama et al., 2013; [36] Bonelli et al., 2016; [37] Spotorno-Oliveira et al., 2020.