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Aspectos populacionais do bivalve invasor *Mytilopsis leucophaeata* (Dreissenidae): padrão de ocorrência em áreas nativas e não-nativas e co-ocorrência com a fauna nativa

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Orientador: Dr. Igor Christo Miyahira
Coorientadora: Dra. Raquel de Almeida Ferrando Neves

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Antonio Jailson de Sousa Rodrigues

ASPECTOS POPULACIONAIS DO BIVALVE INVASOR *MYTILOPSIS LEUCOPHAEATA* (DREISSENIDAE): PADRÃO DE OCORRÊNCIA EM ÁREAS NATIVAS E NÃO-NATIVAS E CO-OCORRÊNCIA COM A FAUNA NATIVA

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“Despite it all there is still love”

(As I Lay Dying)

Resumo

Espécies de *Mytilopsis* Conrad, 1857 vêm apresentando diversos registros em áreas não-nativas nas últimas décadas, provocados principalmente pela água de lastro e bioincrustação. Nestas localidades, estas espécies são capazes de provocar diversos impactos negativos, como a redução da biodiversidade e alterações nas características físico-químicas da coluna d’água, ocasionadas especialmente pelas elevadas densidades de indivíduos de *Mytilopsis* spp. presentes nestes ambientes. Umas das espécies de *Mytilopsis* amplamente estudadas é *Mytilopsis leucophaeata* (Conrad, 1831), cuja distribuição nativa é restrita ao Golfo do México e costa leste do E.U.A. No entanto, essa espécie é registrada como invasora em diversas localidades dos continentes europeu, asiático e América do Sul, incluindo o Brasil. Para o Brasil, esta espécie é somente registrada em duas lagoas costeiras situadas no município do Rio de Janeiro: Lagoa Rodrigo de Freitas e Lagoa de Marapendi. Assim, este estudo visa avaliar aspectos biológicos (e.g. densidade, comprimento dos indivíduos, biomassa) e fatores ambientais de sistemas aquáticos nos quais *Mytilopsis* spp. ocorrem ao redor do mundo em áreas nativas e não-nativas a partir de um banco de dados, como também, identificar a dinâmica populacional do bivalve *Brachidontes darwinianus* (d'Orbigny, 1842), uma espécie nativa presente na Lagoa Rodrigo de Freitas e que ocorre em simpatria com o bivalve invasor *M. leucophaeata*. Através da plataforma Google Acadêmico, um banco de dados contendo aspectos biológicos de *Mytilopsis* spp. e fatores ambientais associados a essas populações em áreas nativas e não-nativas foi compilado e analisado posteriormente para identificar possíveis diferenças significativas na composição dessas populações entre essas áreas. Enquanto que para a população de *B. darwinianus* presente na Lagoa Rodrigo de Freitas, parâmetros populacionais (e.g. taxa de crescimento, mortalidade) foram estimados e comparados ao do bivalve invasor *M. leucophaeata*. As densidades de *Mytilopsis* spp. geralmente foram bastante elevadas em áreas não-nativas, além de *M. leucophaeata* e *Mytilopsis sallei* (Récluz, 1849) apresentarem uma ampla distribuição ao redor do mundo nestas áreas. Provavelmente, essas elevadas densidades afetam negativamente espécies nativas, como *B. darwinianus*, visto que as taxas de crescimento de *M. leucophaeata* foram superiores as do bivalve nativo, o que pode facilitar o processo de invasão na Lagoa Rodrigo de Freitas por essa espécie. Além disso, as densidades do bivalve nativo foram consideravelmente inferiores durante todo o estudo, fato que deve ser analisado com cautela, visto que a população de *M. leucophaeata* pode estar competindo por recursos (e.g. espaço, alimento) com *B. darwinianus*. Outro fator que pode influenciar na predominância de *Mytilopsis* spp. é a sua ampla tolerância a variações de fatores ambientais, como temperatura e salinidade, que são bastante variáveis principalmente em ambientes estuarinos. Portanto, é necessário que estudos sobre aspectos biológicos e fisiológicos de *Mytilopsis* spp. sejam desenvolvidos nos mais diversos âmbitos justamente para entender o processo de invasão que tem sido recorrente nos últimos anos, como também, os impactos que podem ser gerados para as espécies nativas nesses ambientes. Além disso, medidas mitigatórias devem ser elaboradas para prevenir a dispersão de *Mytilopsis* spp., visto o seu alto poder de dispersão e impactos gerados após o estabelecimento.

Palavras-chave: bioinvasão; *Brachidontes darwinianus*; dinâmica populacional; falso-mexilhão; Lagoa Rodrigo de Freitas.

Abstract

Mytilopsis Conrad, 1857 species have been presenting several records in non-native areas in the last decades, mainly caused by ballast water and biofouling. In these locations, these species are capable of causing several negative impacts, such as the reduction of biodiversity and changes in the physical-chemical characteristics of the water column, caused especially by the high densities of *Mytilopsis* spp. individuals present in these environments. One of the widely studied *Mytilopsis* species is *Mytilopsis leucophaeata* (Conrad, 1831), whose native distribution is restricted to the Gulf of Mexico and the east coast of the U.S.A. However, this species is recorded as invasive in several localities in Europe, Asia and South America, including Brazil. For Brazil, this species is only recorded in two coastal lagoons located in the municipality of Rio de Janeiro: Rodrigo de Freitas and Marapendi. Thus, this study aims to evaluate biological aspects (e.g. density, individual length, biomass) and environmental factors of aquatic systems in which *Mytilopsis* spp. occur around the world in native and non-native areas from a database, as well as to identify the population dynamics of the bivalve *Brachidontes darwinianus* (d'Orbigny, 1842), a native species present in Rodrigo de Freitas Lagoon and occurring in sympatry with the invasive bivalve *M. leucophaeata*. Through the Scholar Google platform, a database containing biological aspects of *Mytilopsis* spp. and environmental factors associated with these populations in native and non-native areas was compiled and further analyzed to identify possible significant differences in the composition of these populations between these areas. While for the *B. darwinianus* population present in Rodrigo de Freitas Lagoon, population parameters (e.g. growth rate, mortality) were estimated and compared to that of the invasive bivalve *M. leucophaeata*. Densities of *Mytilopsis* spp. were generally high in non-native areas, and *M. leucophaeata* and *Mytilopsis sallei* (Récluz, 1849) have a wide distribution around the world in these areas. These high densities probably negatively affect native species such as *B. darwinianus,* since the growth rates of *M. leucophaeata* were higher than those of the native bivalve, which may facilitate the invasion process in Rodrigo de Freitas Lagoon by this species. Furthermore, densities of the native bivalve were considerably lower throughout the study, a fact that should be analyzed with caution, since the *M. leucophaeata* population may be competing for resources (e.g. space, food) with *B. darwinianus*. Another factor that may influence the predominance of *Mytilopsis* spp. is its wide tolerance to variations in environmental factors, such as temperature and salinity, which are variable mainly in estuarine environments. Therefore, it is necessary that studies on biological and physiological aspects of *Mytilopsis* spp. be developed in the most diverse areas, precisely to understand the invasion process that has been recurrent in recent years, as well as the impacts that can be generated for native species in these environments. In addition, mitigation measures should be designed to prevent the dispersal of *Mytilopsis* species, given their high dispersal power and impacts generated after establishment.

Keywords: bioinvasion; *Brachidontes darwinianus*; false mussel; population dynamics; Rodrigo de Freitas Lagoon.

SUMÁRIO

Introdução geral

A crescente globalização vem causando o aumento no número de espécies invasoras no ambiente aquático, provocada principalmente pela liberação da água de lastro e bioincrustação (Seebens *et al.* 2013, Anil e Krishnamurthy 2018), induzindo frequente impacto ao ambiente estuarino (Nehring 2006). Entre os principais impactos, os mais recorrentes são a redução da biodiversidade, com o deslocamento de espécies nativas, alterações na dinâmica trófica e ciclagem de nutrientes, além de problemas econômicos e de saúde pública (Gallardo *et al.* 2016, Boltovskoy *et al.* 2018, Ojaveer *et al.* 2018, Kumar *et al.* 2019, Neves *et al.* 2020).

Os bivalves invasores são reconhecidos como um dos grupos de organismos com maior potencial de impacto nos ambientes na qual são introduzidos. Estes animais costumam apresentar colonização bem-sucedida devido ao seu crescimento rápido, plasticidade fenotípica, ampla tolerância à variação de fatores ambientais, assim, são capazes de colonizar diversos tipos de habitats (e.g. Verween *et al.* 2010, Boltovskoy e Correa 2015, Fernandes *et al.* 2020). Além disso, espécies deste grupo são capazes de atuar como engenheiras de ecossistemas, alterando a estrutura e o funcionamento do ambiente (Sousa *et al.* 2009).

A família Dreissenidae (Bivalvia) é conhecida por apresentar diversas espécies com histórico de bioinvasão ao redor do mundo. Dentre as mais conhecidas, estão as incluídas nos gêneros *Dreissena* e *Mytilopsis,* registradas em ambientes dulcícolas e estuarinos; e que possuem larvas livre-natantes, alcançam elevadas densidades e são capazes de provocar diversos impactos nos ambientes invadidos (Verween *et al.* 2010, Kennedy 2011, Enders *et al.* 2019, Neves *et al.* 2020). Apesar de algumas espécies fósseis serem marinhas, a maioria das espécies são estuarinas e de água doce, sendo todas epifaunais e se prendendo ao substrato por filamentos de bisso (Nuttall 1990, Fernandes *et al.* 2018). A taxonomia de *Mytilopsis* não é bem resolvida, no entanto, duas espécies são as mais conhecidas em função da ampla distribuição e impactos provocados: *Mytilopsis leucophaeata* (Conrad, 1831) e *Mytilopsis sallei* (Récluz, 1849) (e.g. Marelli e Gray 1983, Marelli e Gray 1985, Kennedy 2011, Cai *et al.* 2014, Rizzo *et al.* 2014, Tan e Tay 2018, Fernandes *et al.* 2020).

Esta dissertação está dividida em dois capítulos, sendo o primeiro oriundo de dados secundários de áreas de ocorrência nativa e não-nativa do gênero *Mytilopsis* e o segundo contempla um estudo de caso após a introdução de *M. leucophaeata* em uma lagoa urbana costeira impactada. O primeiro capítulo, que deu origem a um manuscrito (apresentado na íntegra) submetido em março de 2021 para o periódico *Biological Invasions*, e atualmente encontra-se publicado, visa avaliar aspectos biológicos, condições ambientais e o uso do habitat associado às populações nativas e não-nativas de *Mytilopsis* ao redor do mundo. Enquanto no segundo, o objetivo é analisar a dinâmica populacional do bivalve nativo *Brachidontes darwinianus* (d'Orbigny, 1842) que ocorre em simpatria com o bivalve invasor *M. leucophaeata* na Lagoa Rodrigo de Freitas, Rio de Janeiro, Brasil.

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Capítulo 1

Wide tolerance to environmental conditions and substrate colonization mediates the invasion of false mussels (Bivalvia: Dreissenidae) in brackish systems

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Abstract

False mussels are recognized as the brackish water equivalent of zebra mussels, although the abiotic and habitat conditions that mediate these invaders' success are barely known. In this context, we aimed to evaluate the native and non-native geographical distribution of *Mytilopsis* species worldwide and assess biological traits, environmental condition, and habitat associated with false mussels in native and invaded systems. Our hypothesis is that *Mytilopsis* invasion is driven by species tolerance to environmental conditions and substrate use in brackish systems, where the colonization of non-native populations is favored by great availability of artificial substrates and tolerance to wide ranges of environmental conditions. In addition, this study provides the range and distribution patterns of *Mytilopsis* species within their introduced and native areas and documents the spread of introduced populations worldwide. Considering the five species evaluated, *M. leucophaeata* and *M. sallei* are the most widespread, while *M. adamsi*, *M. trautwineana*, and *M. africana* showed more restricted geographic distribution. In the last decades, *M. leucophaeata* and *M. sallei* consolidated and expanded their distributions. Environmental conditions were significantly different between native and non-native areas, where *Mytilopsis* populations presented significantly higher densities. Non-native populations exhibited remarkable plasticity concerning habitat colonization that was more frequent on artificial substrata. *Mytilopsis* populations presented significant differences in their biological traits, habitat environmental conditions, and substrate use between native and non-native areas. These species seem to adapt to the conditions of invaded systems, changing their preferences, which may reflect plasticity and a potential shift of their realized niches.

Keywords: *Mytilopsis leucophaeata*, *Mytilopsis sallei*, Invasive species, Biofouling, Environmental conditions.

Introduction

In the last decades, the globalization process has increased the number of biological invasions in marine and brackish environments (Paavola et al. 2005; Hulme 2009; Seebens et al. 2013; Anil and Krishnamurthy 2018; Teixeira and Creed 2020). Currently, the main routes for the dispersal of invasive marine species encompass ports located in the United States of America, Europe, and East Asia, resulting in impacts where these species are introduced (Ruiz et al. 1997; Kaluza et al. 2010; Ojaveer et al. 2018). The most common impact of invasive species is the loss of native biodiversity induced by new-established interactions with the receiving community (e.g. competition, predation), leading to changes in the structure of ecosystems and also possibly modifying their physical and chemical features (Anil 2006; Kalchev et al. 2013; Ojaveer et al. 2018; Neves et al. 2020).

False mussels (Dreissenidae) are among the most notable fresh- and brackish water invaders, including *Dreissena polymorpha* (Pallas, 1771), *Dreissena rostriformis bugensis* Andrusov, 1897, *Mytilopsis leucophaeata* (Conrad, 1831), and *Mytilopsis sallei* (Récluz, 1849) (e.g. Vanderploeg et al. 2002; Verween et al. 2010; McLaughlan et al. 2014; Geda et al. 2018). Members of *Dreissena* and *Mytilopsis* possess a free-swimming larva that allows their dispersal over long distances through ballast water, probably the main vector of their dispersion into new aquatic systems (Chu et al. 1997; Van der Velde et al. 2010; Teixeira and Creed 2020). Moreover, adults attached to hulls can be an important local dispersal agent, carrying reproducing adults to new environments (Minchin et al. 2003; Farrapeira et al. 2007; Richardson and Hammond 2016). The high rates of vessel traffic increase the chance of new introductions of dreissenid species, as evidenced by frequent records of invasions in new geographical locations (Brzana et al. 2017; Zhulidov et al. 2018).

The colonization of invasive dreissenid mussels can drastically alter the functioning of a newly invaded ecosystem, causing economic and ecological problems (e.g. Burlakova et al. 2000; Ward and Ricciardi 2007; Verween et al. 2010; Therriault et al. 2013; Cai et al. 2014). The major ecological consequences include the collapse of native mussel populations (e.g. through space and food competition, and overgrowth), reductions in phytoplankton biomass and changes in water transparency through water filtration, and physical changes in the benthic substrates through biofouling (reviewed in Vanderploeg et al. 2002; Neves et al. 2020). Moreover, dreissenid colonization has implications for the nutrient dynamics in aquatic systems through the removal of nutrients by filtration but also recycling nutrients by mussel excretion, mainly dissolved phosphorus and nitrogen (Arnott and Vanni 1996; James et al. 2001; Conroy et al. 2005; Naddafi et al. 2008). The changes in nutrient dynamics and water transparency can be a trigger to excessive growth of benthic macroalga and macrophytes (Hecky et al. 2004; Wilson et al. 2006; Ozersky et al. 2009). Despite the widely known negative effects, some apparent beneficial impacts, as an improvement in water transparency, have been also described after the dreissenid invasion (Graczyk et al. 2004; Higgins and Vander-Zanden 2010; McLaughlan and Aldridge 2013; Neves et al. 2020).

Most of the dreissenid studies are focused on the impacts of zebra mussels (*D. polymorpha*), and its congener the quagga mussel (*D. rostriformis bugensis*), in North America lakes and rivers, especially the Laurentian Great Lakes (e.g. Idrisi et al. 2001; reviewed in Vanderploeg et al. 2002; Strayer et al. 2004; Fahnenstiel et al. 2010; Kelly et al. 2010). Invasive dreissenid mussels often filter large volumes of water (e.g. Horgan and Mills 1997; Baldwin et al. 2002; Vanderploeg et al. 2010) and attach to a variety of natural and artificial hard substrata using byssal threads, reaching high densities in invaded areas (Ricciardi et al. 1997; Sousa et al. 2009; Kennedy 2011; Rizzo et al. 2014; Tan and Tay 2018). Dreissenid mussels may also tolerate wide ranges of temperature and salinity (Rajagopal et al. 2005; Verween et al. 2007, 2010; Van der Gaag et al. 2016), which makes them successful aquatic invasive species.

Mytilopsis is known as a brackish water equivalent of zebra mussels (Verween et al. 2010), although the optimal abiotic conditions of *Mytilopsis* species and detailed aspects of its invasive process are barely known. The taxonomy of the genus *Mytilopsis* is not well solved, but five recent species are generally recognized (Marelli and Gray 1983, 1985; Nuttall 1990a, b; Kennedy 2011): *M. leucophaeata*, *M. sallei*, *Mytilopsis adamsi* Morrison, 1946, *Mytilopsis trautwineana* (Tryon, 1866), and *Mytilopsis africana* (Van Beneden, 1835). The first four species are known as “invasive species” (e.g. Marelli and Gray 1983; Tan and Morton 2006; Aldridge et al. 2008; Rizzo et al. 2014; Fernandes et al. 2018; Wangkulangkul 2018), and *M. africana* (described upon specimens from African coast) is considered by some authors as a synonymy of *M. sallei*, that was possibly historically introduced into Africa (Morton 1981; Nuttall 1990a, b; Le Loeuff 1999). Regarding the scarcity of ecological information on *Mytilopsis* species within their invaded systems (e.g. Neves et al. 2020; Fernandes et al. 2020; Rodrigues et al. 2021), tracking of environmental and ecological impacts caused by *Mytilopsis* colonization is virtually impossible.

In this context, the present study aims to evaluate the native and non-native geographical distribution of *Mytilopsis* species worldwide and assess biological traits, environmental conditions, and habitat use associated with native and non-native populations of false mussels. More specifically, our goal was i) characterize the abiotic conditions (i.e. physical and chemical variables: salinity, temperature, chlorophyll *a*, dissolved oxygen and transparency) in native and non-native areas, ii) identify and distinguish the environmental conditions and biological traits (such as size, density and biomass) among native and non-native *Mytilopsis* populations, iii) test for differences in the use of substrates for colonization by native and non-native *Mytilopsis* populations, and iv) identify the main sessile epibenthic taxa that co-occur with *Mytilopsis* species. For that, literature data focusing on the geographical occurrence of five extant *Mytilopsis* species (*M. leucophaeata*, *M. sallei*, *M.*

adamsi, M. trautwineana, and *M. africana*) was compiled within their native and invaded brackish systems simultaneously to other biological data (e.g. density and individual size of *Mytilopsis* species, and the occurrence of other sessile epibenthic taxa). The main hypothesis is that *Mytilopsis* invasion is driven by species tolerance to environmental conditions and substrate use in brackish systems, where the colonization of these bivalves is favored by the great availability of artificial substrates and species tolerance to wide ranges of environmental conditions. Besides the disclosure of ecological aspects of these invasive mussels, we describe the range and distribution patterns of *Mytilopsis* species within their native and non-native regions and document the spread of introduced populations around the world.

Material and Methods

Species selection and data acquisition

Because the specific taxonomy of *Mytilopsis* is inconclusive, we accepted the identifications provided by the author of each work. An electronic survey was performed to compile all published data from the Google Scholar (https://scholar.google.com.br/) database for the five recognized species of the genus using the following keywords: "*Mytilopsis* sp." AND "distribution" OR "occurrence", where the "sp." was replaced by the specific epithet, i.e., *leucophaeata, sallei, adamsi, trautwineana,* and *africana*. The search included scientific articles, book chapters, and grey literature (e.g. technical reports and theses). Moreover, references that were cited in retrieved studies for occurrence data not detected in the first survey were also evaluated. Our database included all the retrieved studies that provided geographic records for the assessed *Mytilopsis* species. In seven localities, the false mussel was identified as *M.* cf. *sallei* (see Fernandes et al. 2018 for further information) and considered together with those of *M. sallei* for the purposes of this analysis.

For each study, available data were compiled on the occurrence sites, *Mytilopsis* species (according to author identification), geographical coordinates, population density and biomass, shell length of individuals, the substrate used for attachment, and the presence and identification of co-occurring benthic species. The type of substrate used by false mussel populations was divided into eleven categories: existing benthic fauna (i.e. other fouling animals), mangrove roots, other aquatic or emergent vegetation, rocky substrates, soft substrates where settlement began on a hard object (see Fernandes et al. 2020), human-created concrete structures (e.g. piers, marinas, and others), metals structures, plastic materials (including other petroleum products, such as Styrofoam), vessels, wood fragments (usually experimental plates, with human treatment, not natural), and ropes and meshes (e.g. mooring ropes, fish cages).

A habitat characterization was also performed for each occurrence site (georeferenced) by compiling physical and chemical data from the available literature. For each brackish system where *Mytilopsis* spp. was recorded, data were obtained of area (km^2), and mean, minimum, and maximum values of five variables, viz. (1) surface water salinity, (2) surface water temperature (°C), (3) chlorophyll *a* in surface water (mg L^{-1}), (4) dissolved oxygen in surface water (mg L^{-1}), and (5) water transparency (cm). When abiotic data was not available in the reviewed papers, additional literature searches were performed for the georeferenced localities to obtain all the environmental data. For that, electronic surveys were performed in the Google Scholar (https://scholar.google.com.br/) database applying a combination of keywords with the aquatic system name and the abiotic variable of interest (e.g. “Rodrigo de Freitas Lagoon” AND “water transparency”).

Data analysis

The densities of *Mytilopsis* were plotted on a distribution map, using 1 as density value for the records without a density value. Considering species record limitations for the other three species of the genus, only the occurrence data of *M. leucophaeata* and *M. sallei* were plotted on the distribution map.

A Principal Component Analysis (PCA) was performed on the abiotic matrix to assess the relationship among environmental variables and how they were associated with the native and non-native sites in which *Mytilopsis* species were recorded. A Hellinger transformation was used to control for the different scaling measures and unities among variables, and the broken-stick model was applied to select which principal components were significant for explaining sample distribution. PCA was performed using the PC-ORD v 6.0 software. A Canonical Analyses of Principal Coordinates (CAP) were performed to test for differences in environmental conditions and substrate types among native and non-native populations of *Mytilopsis* species. Hellinger distance and 9,999 permutations were applied in CAP, following Anderson and Willis (2003). Finally, Generalized Additive Models (GAMs) were applied to assess the smooth terms of the pairwise relationships between biological descriptors (log10-transformed density and shell-length of *Mytilopsis* species) and the environmental variables which could affect the distribution of false dark mussels. GAMs are an extension of generalized linear models that, unlike more conventional regression methods, do not assume a functional relationship between the response variable and the predictors (Lepš and Šmilauer 2003). Model complexity of GAMs was chosen by the stepwise selection procedure using the Akaike Information Criterion (AICc) corrected for small sampling size ($N \leq 30$), and normal data distribution and identity as linkage function were chosen to broaden the selection of either linear as non-linear responses. GAMs were performed using the software CANOCO 4.5.

Considering data limitation for the five species distinguished, statistical analyses were applied for the genus *Mytilopsis* (all species together), but considering the difference in geographical distribution, i.e., native and non-native populations. Student t-test was applied to test for differences in the density of native and non-native *Mytilopsis* populations (without addressing differences for each species separately), to test for differences in shell length between *M. leucophaeata* and *M. sallei* (the two most widespread species), and to test for differences in the proportion of occurring taxa with *Mytilopsis* populations in native and non-native areas.

Results

Geographic distribution of Mytilopsis species

In total, 158 occurrences of *Mytilopsis* species were obtained from the literature survey (*n* = 50 for native and *n* = 108 for introduced areas). *Mytilopsis* records by specific locality are fully presented in Supplementary Material S1. Among the five *Mytilopsis* species, *M. leucophaeata* and *M. sallei* are the most widespread, while *M. adamsi*, *M. trautwineana*, and *M. africana* were reported to be geographically more restricted in distribution. *Mytilopsis adamsi* were reported in few localities on its historical native range, while its introduced range varied from North America (Mexico) to the south of Africa (Mauritius), and Asia (Philippines and Thailand), with an overall latitudinal variation from 23°N to 20°S (*n* = 11 locations). *Mytilopsis trautwineana* (*n* = 3 locations) was reported as associated with shrimp farms in Cartagena, Colombia, a non-native area for this species, despite other unprecise records, as a generically mention for to the eastern Pacific coast of Colombia and Ecuador (Aldridge et al. 2008), within its native area. *Mytilopsis africana* were generically reported to the west coast of Africa, from Ivory Coast to Gabon, from 5°N to 0°, on the native range (Le Loeuff 1999).

The native range of *M. leucophaeata* has been reported as brackish systems in the Chesapeake Bay (USA) and Gulf of Mexico (latitudinal variation from 39°N to 18°N), while its invaded range extends from South America (Brazil), Eurasia (from Spain to Iran), and north of Africa, following a latitudinal variation from 60°N to 23°S (Fig. 1; *n* = 88 occurrence records). The longitudinal range was from 96°W to 76°W for its native range, and from 74°W to 49°E for invaded range. The native range of *M. sallei* has been reported as portions of the Gulf of Mexico, Caribbean Islands, and the North of South America, with a latitudinal variation from 28°N to 10°N. However, its invaded range included the North of Africa (Egypt), Asia (from India to Japan), Oceania (Australia), and Pacific Islands (Fiji), following a latitudinal range from 35°N to 18°S (Fig. 1; *n* = 46 occurrence records). The longitudinal variation was from 88°W to 61°W, and the introduced range was almost all the globe around, from 30°E to 34°W (considering the records for *M.* cf. *sallei*, *n* = 7).

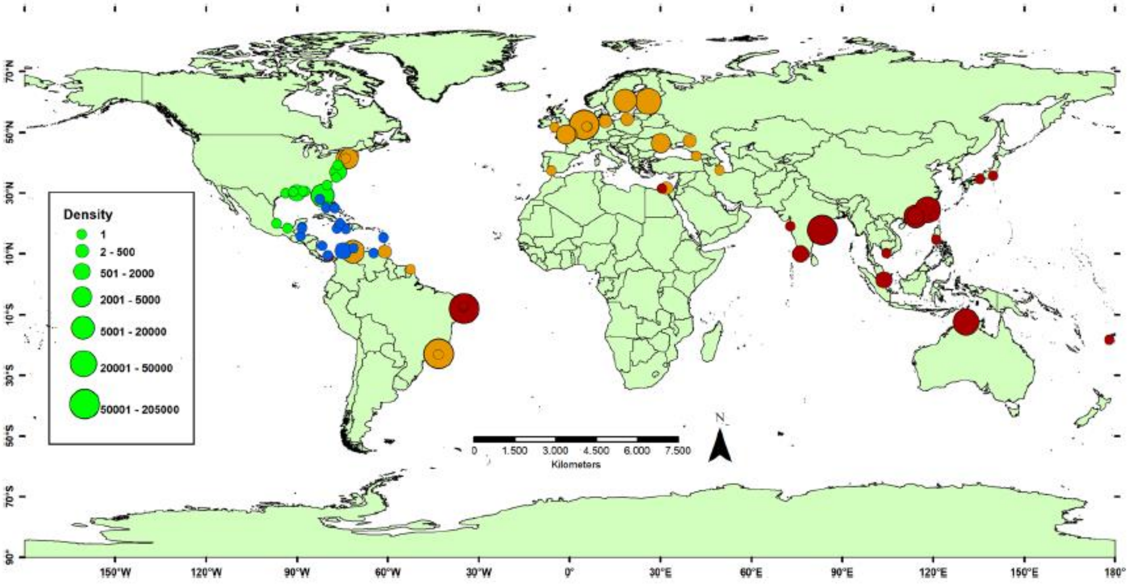

Fig. 1. World map indicating the distributions of *Mytilopsis leucophaeata* (*n* = 88) and *M. sallei* (*n* = 53, including *M.* cf. *sallei*) in native (green and blue circles, respectively) and non-native (orange and red circles, respectively) areas. Differences in population density are expressed by distinct circle sizes.

Habitat environmental conditions in native and non-native areas

Characterization of environmental conditions associated with the native and non-native geographical distribution range of *Mytilopsis* species is presented as Supplementary Material S2. The multivariate analysis (PCA) applied on the environmental dataset (i.e. latitude, longitude, water temperature, dissolved oxygen, chlorophyll *a*, salinity, and transparency) evidenced preferential conditions for native and non-native *Mytilopsis* populations (Fig. 2). The first two PCA axes were selected by broken-stick (BS) method, with axis 1 and 2 explaining, respectively 36.6% (BS eigenvalue = 2.829) and 22.4% (BS eigenvalue = 1.829) of total variance. PCA axis 1 accounted for most of the differences between native and non-native geographical locations. Most of the samples from non-native populations (i.e. red color in Fig. 2) were located at the right side of the biplot and correlated with high values of water transparency (r = 0.856). This difference in environmental conditions between samples from native and non-native systems was overall supported by canonical analysis of principal coordinates (CAP trace statistics, $p = 0.035$). Moreover, PCA axis 2 accounted for the sample's separation among *Mytilopsis* species. Most of the samples from *M. leucophaeata* distribution were located at the lower side of biplot and correlated with higher values of dissolved oxygen, both for mean (r = - 0.260) and range (r = - 0.520) values, latitude (r = - 0.774) and temperature range (r = - 0.649); while samples from the other three *Mytilopsis* species (i.e. *M. sallei*, *M. adamsi*, and *M. trautwineana*) were exclusively located at the upper side of biplot and correlated with higher values of salinity (r = 0.269), mean temperature (r = 0.407), longitude (r = 0.513), and chlorophyll *a* (r = 0.383). These differences in the environmental conditions between samples of *M. leucophaeata* distribution and samples from the other *Mytilopsis* species were statistically significant (CAP trace statistics, $p < 0.0001$).

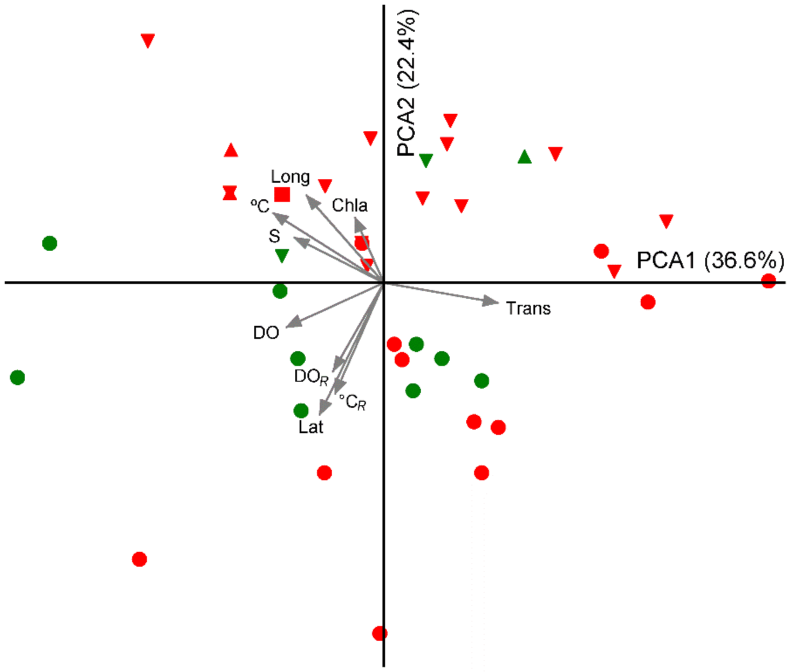

Fig. 2. PCA biplot ordination diagram of environmental conditions for *Mytilopsis* species within its native (green color) and non-native (red color) geographical range of distribution. Environmental dataset of geographical distribution of four *Mytilopsis* species were considered: *M. leucophaeata* (●), *M. sallei* (▼), *M. adamsi* (▲), and *M. trautwineana* (■). Environmental variables analyzed were: latitude (Lat), longitude (Long), water temperature (°C), salinity (S), chlorophyll *a* (Chla), dissolved oxygen (DO), and transparency (Trans). Values of data range (i.e. maximum-minimum values) were only used for temperature and dissolved oxygen since no significant correlation was found between mean and range data (Pearson *p*-value > 0.05).

Populational traits of Mytilopsis within native and introduced systems

The maximum values found for the populational traits (i.e. density and shell length) of *Mytilopsis* species are presented in Table 1. Moreover, all the compiled values (mean, minimum and maximum) of population traits for four *Mytilopsis* species (except by *M. africana* for which no data was found) are presented in Supplementary Material S1.

Table 1. Maximum values of density and individual size (shell length) found for *Mytilopsis* species on native (N) and introduced (I) areas. No information was found for *M. africana*. A fully detailed compilation of populational traits is shown in Supplementary Material S1.

Species	**Density (ind. m^{-2})**		**Size (mm)**	
	N	I	N	I
M. leucophaeata	14,150	204,000	22.0	31.7
M. sallei	900	83,000	25.6	35.0
M. adamsi	-	569,000	-	32.0
M. trautwineana	-	-	-	24.0

(-) no data available

The density of *Mytilopsis* species in introduced aquatic systems was significantly higher than in native ones (Table 1; t-test, $p = 0.012$). Significant relationships between the density of native and invasive populations and environmental data were only found for two variables: values of longitude (Linear AIC = 44.26; F = 8.11, $p < 0.01$) and temperature range (Non-linear AIC = 47.88; F = 6.56, $p = 0.017$) (Fig. 3). A non-linear model for *Mytilopsis* density in relation to the range in water temperature was selected by AIC, where invasive populations (red color) showed higher densities, but was more restricted to a lower temperature range (i.e. lower variation), with a tendency to decrease after a range of 15°C (Fig. 3b). A linear model was chosen by AIC for *Mytilopsis* species density in relation to the longitude of the sampling site, where *M. leucophaeata* was more restricted to lower longitudinal areas and lower density within its native range and extended its longitudinal distribution and density on the introduced range (Fig. 3a). In contrast, the distribution of *M. sallei* was more restricted to higher longitudinal areas, where this species is invasive, with higher densities (Fig. 3a). Independently of the distribution range, populations of *M. sallei* showed significantly higher densities than *M. leucophaeata* (t-test, $p = 0.025$).

No significant difference was found for false mussel size (in shell length) between the most widespread species, *M. leucophaeata* and *M. sallei* (t-test, $p = 0.207$). However, significant relationships (GAMs) between size and environmental data were found for two variables: water transparency range (Non-linear AIC = 480.18; F = 9.87, $p < 0.01$) and temperature range (Non-linear AIC = 641.34; F = 8.75, $p < 0.01$) (Fig. 3). A unimodal response was found for the relationship of mussel size and with the range of temperature (Fig. 3d) and transparency (Fig. 3c), in which a maximum shell length was reached close to 15°C of temperature variation and 200 cm of water transparency, and then decreasing towards higher temperature range (i.e. broader variation) and water transparency.

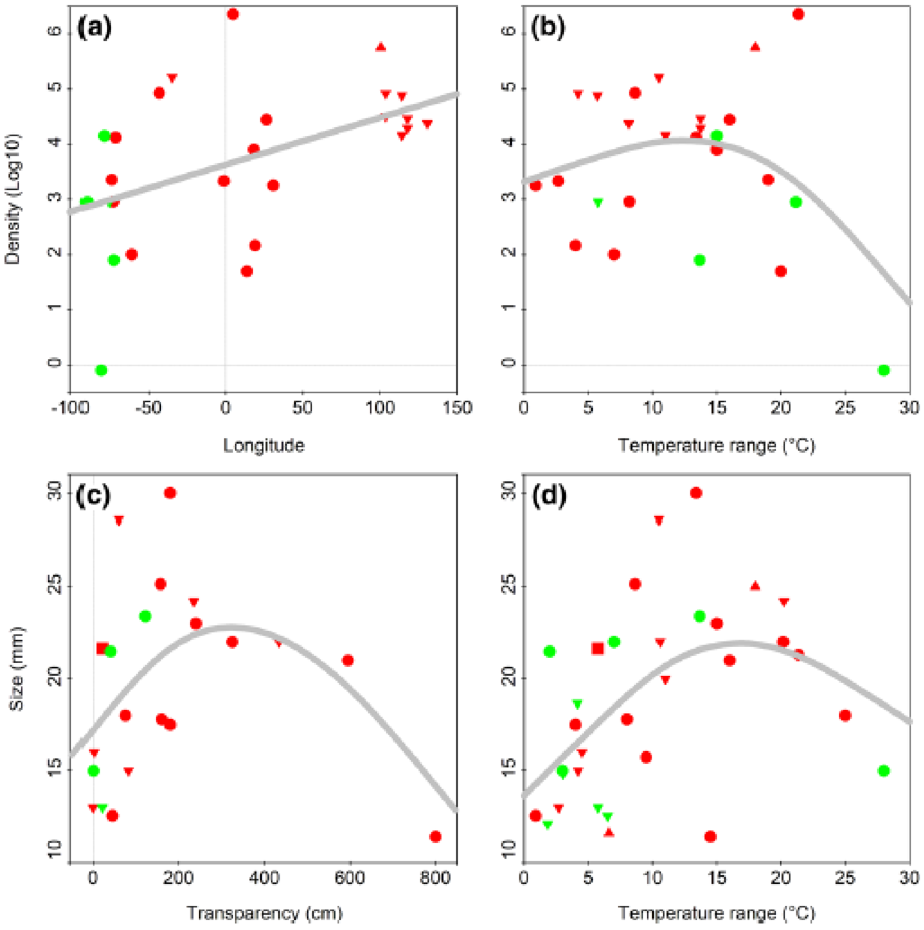

Fig. 3. Generalized Additive Models (GAMs) selected for Akaike Information Criterion (AIC) for biological traits of invasive (red color) and native (green color) *Mytilopsis* populations to environmental variables. Relationships of the population density (Log_{10}-transformed) to longitude of georeferenced species distribution (a) and temperature range (°C; b), and of the false mussel size (in shell length, mm) to water transparency (c) and temperature range (d) were presented. Samples were coded by species: *M. leucophaeata* (●), *M. sallei* (▼), *M. adamsi* (▲), and *M. trautwineana* (■).

Taxa co-occurring with native and invasive populations of Mytilopsis spp.

Information on the epibenthic taxa that co-occur with *Mytilopsis* populations in native and invaded systems was restricted to four species: *M. adamsi* (*n* = 3), *M. leucophaeata* (*n* = 25), *M. sallei* (*n* = 30), and *M. trautwineana* (*n* = 1). No information about co-occurring species with *M. africana* was found in the literature. A significant difference was found in the proportion of epibenthic taxa that co-occur with *Mytilopsis* populations between native and invaded areas (t-test, $p < 0.0001$). *Mytilopsis* has been recorded occurring with the greatest variety of epibenthic taxa in non-native areas. Seventeen epibenthic groups of ten different phyla were found to co-occur with *Mytilopsis* populations (Fig. 4), in which four groups have co-occurred exclusively with non-native false mussels (Nematoda, Nemertea, Platyhelminthes, and Porifera). Bivalvia was the most frequent taxa associated with *Mytilopsis* populations, regardless of the origin of the false mussel populations (i.e. native or non-native) and considering all the co-occurring groups, followed by Gastropoda for native populations and Cirripedia for non-native ones (Fig. 4).

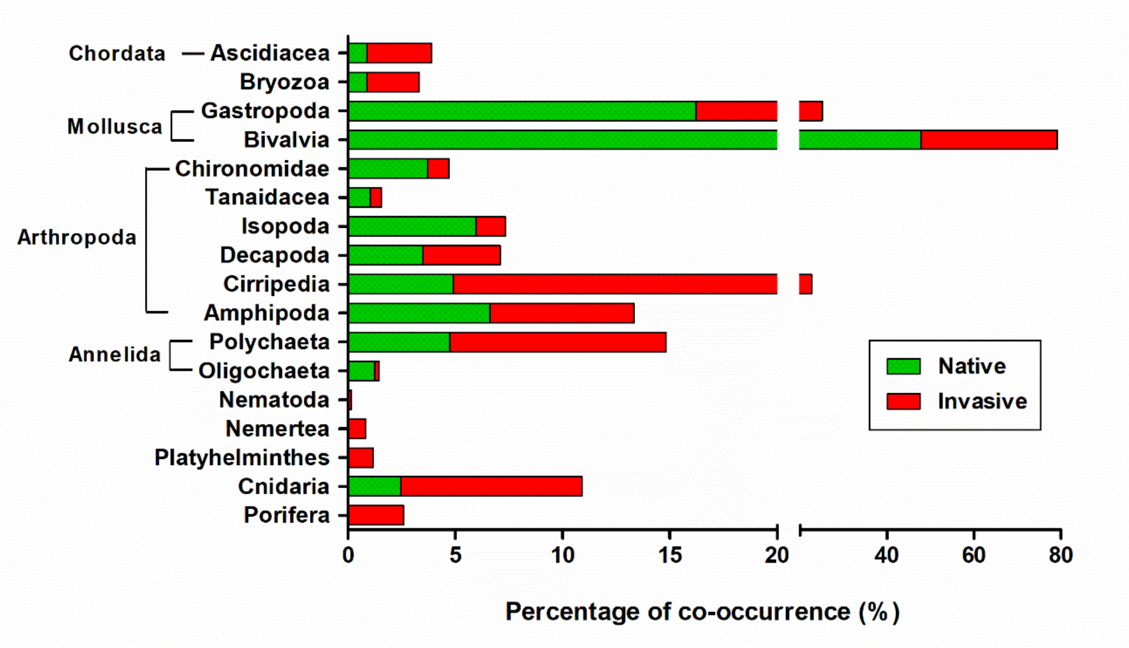

Fig. 4. Percentage of occurrence of epibenthic taxa with native (green bar, *n* = 16) and invasive (red bar, *n* = 43) populations of *Mytilopsis* in brackish systems. Data are shown as the mean percentage of taxa occurrence.

Substrate colonization by native and non-native Mytilopsis populations

Non-native *Mytilopsis* populations colonized a wider variety of substrates, including the artificial ones, in their invaded systems compared to individuals within their native geographical distribution (Fig. 5). In the native area, *M. leucophaeata* was found mainly on soft sediment samples (42%), over benthic fauna (31%), and rocks (17%); while in the invaded systems, the most colonized substrates were human constructions (28%), rocks (24%), and plastic materials (13%). In the native range, *M. sallei* was found attached on mangrove roots (56%), rocks (19%), and sediment samples (19%); whereas in the non-native areas, the species occurred mainly on human constructions (35%), wood fragments (13%), and plastic materials (12%). No information regarding substrate colonization was found for *M. adamsi* within its native occurrence, but the species was mostly found attached to plastic materials (34%) within invaded systems. No information about substrate occurrence was found for *M. africana* and only two records for *M. trautwineana,* thus not displayed in Fig. 5.

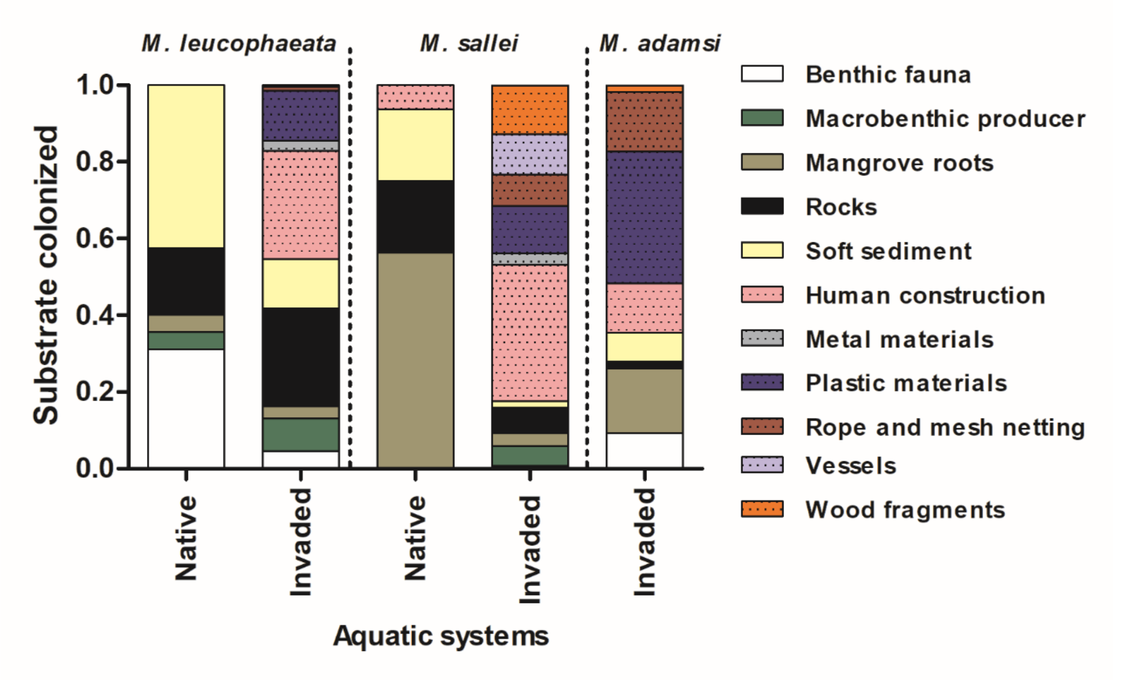

Fig. 5. Substrate type colonized by *Mytilopsis* species within native and invaded systems. Artificial substrates are distinguished by a dotted pattern. Data are shown as the mean proportion of substrate occurrence by species according to data availability in the literature, in which data for native and invaded systems were: *M. leucophaeata* (*n* = 22 and *n* = 35, respectively), *M. sallei* (*n* = 9 and *n* = 29, respectively), and *M. adamsi* (*n* = 8 for invaded systems).

A significant difference in the colonization of natural and artificial substrates was found between native and non-native populations of *Mytilopsis* (CAP trace statistics, $p < 0.001$; Fig. 6). Non-native populations of *Mytilopsis* used significantly more artificial substrates for colonization in introduced areas. Most samples of substrate colonization within the invaded systems (red color) were located at the left side of CAP axis 1 and highly correlated with artificial substrates: human construction (r = - 0.64), rope and mesh netting (r = - 0.48), plastic material (r = - 0.47), and vessels (r = - 0.37). However, most samples of substrate colonization within the *Mytilopsis* native systems (green color) were located at the right side of CAP axis 1 and highly correlated with natural substrates: soft sediments (r = 0.47) and benthic fauna (r = 0.26).

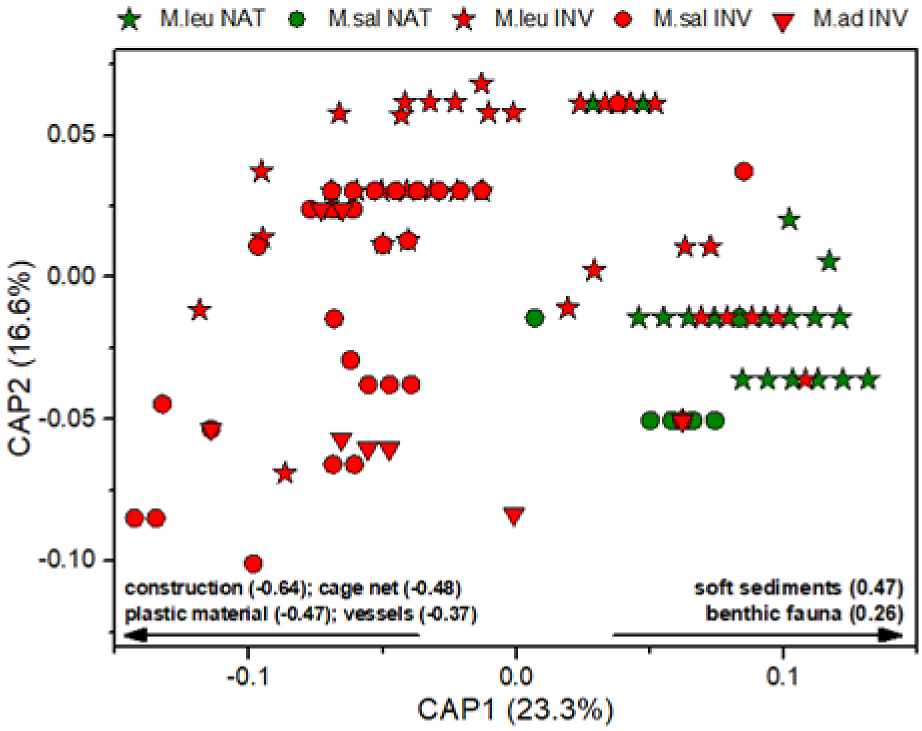

Fig. 6. Biplot ordination diagram (CAP) of substrate colonization within the *Mytilopsis* native (green color, *n* = 31) and invaded (red color, *n* = 72) systems. Samples were categorized by species: *M. leucophaeata* (M.leu, star symbol), *M. sallei* (M.sal, circle symbol), and *M. adamsi* (M.ad, down triangle symbol).

Discussion

This study evaluated the worldwide distribution of five *Mytilopsis* species and revealed distinct patterns of environmental conditions and habitat colonization between native and non-native areas. There are some divergences in the theories with respect to the native distribution range of *M. leucophaeata* (Kennedy 2011); some authors (e.g. Pathy and Mackie 1993; Richardson and Hammond 2016) consider the Hudson River (41°12'49"N; 73°57'50"W) as an invaded area, while Marelli and Gray (1983) indicated based on literature records the Hudson River estuary as a native area. In his historic overview, Kennedy (2011) pointed that the first authors end the distribution of *M. leucophaeata* on Chesapeake Bay, limit range adopted by us, as also by other authors (e.g. Pathy and Mackie 1993; Richardson and Hammond 2016). The introduced ranges of *M. leucophaeata* and *M. sallei* described by us were overall similar to those presented by Marelli and Gray (1983) but updated considering recently published studies. The latitudinal variation of native occurrence is similar between *M. leucophaeata* and *M. sallei* (Δ Lat = 21° and 18°, respectively) but, in the last four decades, both species consolidated and expanded their distributions to non-native areas. Considering our compilation of georeferenced occurrence records for *Mytilopsis* species, the updated distribution of *M. leucophaeata* ranges from almost the polar seas of Finland (Forsström et al. 2016) to the tropical climate of Brazil (Rizzo et al. 2014) (Δ Lat = 83°). In parallel, the updated distribution of *M. sallei* ranges from the temperate climate of Japan (Otani 2002) to the tropical climate of Fiji Island (Marelli and Gray 1983) (Δ Lat = 53°). These latitudinal variations highlight that *M. leucophaeata* has spread more along a latitudinal gradient than *M. sallei*. However, the longitudinal distribution range of *M. sallei* is wider (Δ Long = 296°) than the observed for *M. leucophaeata* (Δ Long = 145°). The updated geographical occurrence of *M. adamsi* extends from Mexico (23°N) to Mauritius (20°S), also a high latitudinal variation (Δ Lat = 43°) but only inside the tropical zone, indicating more limited environmental distribution. *Mytilopsis trautwineana* and *M. africana* were recorded in few native and non-native areas, and this narrow distribution can be related to the lack of available studies for those species, which have reduced our analytical power but may also indicate lower invasiveness potential of these species. The taxonomy of *Mytilopsis* is not well solved and some of these two lineages (*africana* and/or *trautwineana*) may not represent valid species. However, despite some morphological similarities, some studies have shown that may be considerable hidden molecular differences between different lineages of *Mytilopsis* (Fernandes et al. 2020). Another source of debate is the identification of *Mytilopsis* found in the Pacific (e.g. Marelli 2021). A broad morphological and molecular study is desirable to move forward in this question.

The macro-scale perception of *Mytilopsis* distribution patterns indicates a clear geographical spreading of invasive species, especially *M. leucophaeata* and *M. sallei*, across brackish systems of the world. This wide distribution range suggests high species tolerance to a range of environmental factors that varies along latitudinal and longitudinal gradients, such as water temperature. These dreissenids are known to tolerate changes in temperature and salinity (Rajagopal et al. 2005; Verween et al. 2007, 2010; Astudillo et al. 2017; Sa-Nguansil and Wangkulangkul 2020), which reinforces *Mytilopsis* ability to colonize new areas, as evidenced by our geographic distribution map, and reflects their invasiveness potential in aquatic systems worldwide. Moreover, environmental conditions in non-native areas provided beneficial conditions for population establishment and growth, which could be reinforced by significantly higher densities of *Mytilopsis* in non-native areas. Higher density values were reported for *M. leucophaeata* and *M. sallei* on their non-native range (e.g. Pati 2011; Van der Gaag et al. 2017), in addition, high recruitment rates were also observed in these areas (Van der Gaag et al. 2014). As evidenced by our study, invasive populations of *M. sallei* reach significantly higher densities than *M. leucophaeata*. The biomass proved to be difficult to compare considering the different methodologies employed by the authors, which made unfeasible any analysis in the present study. However, high biomass values have been found for *M. leucophaeata* (e.g. Rajagopal et al. 2002) and *M. sallei* (e.g. Shetty et al. 1989) on their introduced range. All the available data of biomass or secondary production for *Mytilopsis* species found in the literature were included in our Supplementary Material S1. There are several theories to explain the success of invasive species, like the release of predators and pathogens, absence of competitors, among others (Fagan et al. 2002; Simberloff et al. 2013). The wide tolerance evidenced here for *Mytilopsis* false mussels, mainly *M. leucophaeata*, are expected to contribute to the successful establishment of this species across non-native ranges, thus increasing its invasiveness potential.

As discriminated by Principal Coordinates Analysis (PCA), native and non-native areas of *Mytilopsis* distribution were modulated by different environmental conditions, overall, statically supported by Canonical Analysis of Principal Coordinates (CAP). Environmental conditions in non-native areas showed a high correlation with increased values of water transparency, and this significant marked difference between native and non-native areas is known as a consequence of *Mytilopsis* invasion in brackish systems (Vanderploeg et al. 2002; Neves et al. 2020). High populational densities, associated with high biomass, may lead to habitat modifications (e.g. higher water transparency, reductions in chlorophyll *a*, and phytoplankton density) promoted by dark false mussel's filtration (Neves et al. 2020). Moreover, as evidenced by PCA and statistically supported by CAP analysis, the environmental conditions most closely related to *M. leucophaeata* distribution areas were significantly different from the other three species tested (*M. sallei, M. adamsi*, and *M. trautwineana*). The similar habitat environmental conditions shared by these three *Mytilopsis* species suggests a higher tendency of co-occurrence in aquatic systems, mostly related to high values of salinity, mean temperature, longitude, and chlorophyll *a*. In contrast, geographic areas of *M. leucophaeata* distribution showed a high correlation with ranges in dissolved oxygen and temperature in surface water, which suggests that this species has great plasticity for extreme values and shifts in these environmental conditions.

Seventeen different taxa were found co-occurring with *Mytilopsis* populations, in which four have occurred exclusively with non-native populations. In invaded systems, Bivalvia (31.28%) and Cirripedia (17.72%) were the most frequent taxa associated with false mussel clusters. These two groups are the commonest sessile invertebrates on the hard substrate of brackish areas around the world (Grzelak and Kuklinski 2010; Oganjan et al.

2017; Sokołowski et al. 2017). Our results presented a snapshot of the distribution and co-occurrence of *Mytilopsis* populations with several epibenthic taxa, but our data is not appropriate to indicate changes in the pattern of epibenthic communities related to false mussels' invasion. However, more detailed effects, on a local scale, have more diverse outputs. Cai et al. (2014) observed a reduction in the density, biomass, and richness of other co-occurring fauna in the presence of *M. sallei*. Rodrigues et al. (2021) have found variations of sympatric populations of *M. leucophaeata* and *Brachidontes darwinianus* (d'Orbigny, 1842) (the native ecological equivalent) on a two years follow-up. After two years, neither *M. leucophaeata* nor *B. darwinianus* were excluded. Similarly, *M. sallei* and *Brachidontes variabilis* (Krauss, 1848) were found co-occurring in Hong Kong, although the invasive species was dominant (Astudillo et al. 2017). Other species, like gastropods, can take advantage of the cluster created by invasive *Mytilopsis* species (Boltovskoy and Correa 2015; Fernandes et al. 2020), using the clusters as refuge area and hard substrate habitat. More detailed studies are needed for a better comprehension of the ecological impacts of *Mytilopsis* invasion on native epibenthic species, as well as shifts in the ecological interactions of brackish benthic communities after false mussels' invasion.

The availability of suitable colonization substrates is one of the main ecological requirements for epibenthic species, which seems to be crucial for the transport, introduction, and establishment of bivalves in new habitats (Rajagopal and Van der Velde 2012; Zhulidov et al. 2018). Non-native *Mytilopsis* populations exhibited remarkable plasticity concerning habitat colonization. The colonization of non-native *Mytilopsis* populations was significantly more frequent on artificial substrata (e.g. human construction, plastic material, cage net, vessels), but occurred mainly on natural substrata on its native range (e.g. benthic fauna, soft sediments). In anthropogenically modified systems, non-native species may have an advantage over the natives (Tyrrel and Byers 2007; Dafforn et al. 2009; Johnston et al. 2009), as natives are now existing in conditions different than those in which they have evolved. This hypothesis - called selection regime alteration - establishes that an environment becomes more prone to receive invasive species due to the man-made modifications, modifying the selection pressures that already exist (Byers 2002; Riquet et al. 2013). One of those human modifications is the creation of new artificial substrata on brackish environments (i.e. human constructions), where the false mussels have been colonizing and performing well. McCarthy et al. (2006) observed a change in substrate use by an invasive crayfish. Similarly, the present study showed that *Mytilopsis* spp. were able to take advantage of the new opportunities in the invaded area, shifting some of their preferences, as substrate use. Invasive species can occur abundantly on artificial substrates, having a preference for these substrates or using them in an opportunistic way (Creed and Paula 2007; Neves et al. 2007; Tyrrel and Byers 2007). *Mytilopsis* can also grow over soft substrata, although a hard nucleus is necessary (Fernandes et al. 2020), potentializing the areas for species distribution, but this strategy was more frequently used in the native area. Another invasive dreissenid, *D. polymorpha* also started to explore soft substrata after colonizing the available hard substrata (Strayer and Malcom 2006). Moreover, several estuaries were naturally poor on hard substrata, thus without (or only with few) native species that explore this new, hard, and artificial substrate, ‘leaving the way clear’ for the invaders (Tyrrel and Byers 2007). The initial record of *M. leucophaeata* in Rio de Janeiro (Brazil) was done mainly on man-made substrata (Rizzo et al. 2014), therefore the species was also recorded in natural substrata (Maia-Neto 2018; Fernandes et al. 2020), which highlights the wide species plasticity for colonization within invaded systems. Noteworthy that our data were based on available literature records, a systematic sampling effort to investigate the substrate used by native and invasive populations can return a different result.

Invasive species are known to alter their habitats and life histories within native and introduced ranges (Côté and Maljkovic 2010; Petanidou et al. 2012). *Mytilopsis* populations presented significant differences in their biological traits, environmental conditions, and substrate use between native and non-native areas. Moreover, these species seem to easily adapt to the new conditions faced on invaded systems, significantly changing their preferences (e.g. substrate type, environmental conditions), which may reflect wide plasticity and a potential shift in their realized niches (i.e. novel biotic and abiotic conditions) in their non-native ranges. Niche plasticity is certainly a feature that helps species to spread to new habitats, and that was usually related to invasive species (Davidson et al. 2011). Evidence of invasive species success has been related to shifts in the species' realized niche, as opposed to evolutionary shifts in range limiting traits (Tingley et al. 2014; Escobar et al. 2016; Gallego-Tévar et al. 2018). Therefore, we recommend further studies on distribution modelling (e.g. niche modelling) and biological aspects of *Mytilopsis* invasive populations to understand their ecological and economic impacts on introduced brackish systems and to propose measures for the management and control of invasive *Mytilopsis* populations and on comparative and taxonomic studies of all *Mytilopsis* species.

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Supplementary Material S1

Wide tolerance to environmental conditions and substrate colonization mediates the invasion of false mussels (Bivalvia: Dreissenidae) in brackish systems

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Table S1. Values of density (ind/m²), size (mm) and biomass/secondary production found in native (N) and invaded (I) areas for *Mytilopsis* species.

Species	**N/I**	**Aquatic System**	**Lat**	**Long**	**Density (ind/m²)**			**Size (mm)**			**Biomass/Secondary Production**			**References**
					Mean	***Min***	***Max***	***Mean***	***Min***	***Max***	***Mean***	***Min***	***Max***	
Mytilopsis leucophaeata	**N**	USA - Upper Chesapeake Bay	38°14'03"N	76°16'26"W	-	-	-	-	-	-	-	-	-	Allen 1962 apud Kennedy 2011a
	N	USA - Potomac River, Chesapeake Bay	38°06'30.0"N	76°31'28.3"W	-	-	-	-	-	-	-	-	-	Frey 1946 apud Kennedy 2011a
	N	USA - Lower Potomac River, Maryland	38°06'30.0"N	76°31'28.3"W	-	-	-	-	-	-	-	-	-	Pfitzenmeyer and Drobeck 1963
	N	USA - Chesapeake Bay	37°33'02"N	72°39'53"W	-	-	80	-	-	-	-	-	-	Jordan and Sutton 1984
	N	USA - Estuary James River	37°01'34.9"N	76°32'49.4"W	-	-	-	-	-	-	-	-	-	Larsen 1985

Species	**N/I**	**Aquatic System**	**Lat**	**Long**	**Density (ind/m²)**			**Size (mm)**			**Biomass/Secondary Production**			**References**
					Mean	***Min***	***Max***	***Mean***	***Min***	***Max***	***Mean***	***Min***	***Max***	
	N	USA - Estuary James River, Virginia	37°01' 34.9" N	76°32'49.4"W	-	-	-	-	-	-	-	-	-	Diaz 1989
	N	USA - Beaufort, North Carolina	34°45' 36.6" N	76°42'22.1"W	-	-	-	-	-	-	-	-	-	Wells 1961
	N	USA - Trent River, North Carolina	35°05' 24.8" N	77°02'57.8"W	-	-	-	-	-	-	-	-	-	Wolfe and Petteway 1968
	N	USA - Escambia River, Sabine River, Neches River (Gulf of Mexico)	24°45' 53"N	85°35'39"W	-	-	-	-	-	-	-	-	-	Wurtz and Roback 1955
	N	USA - Florida Bay	25°01' 02.1" N	80°57'03.9"W	-	-	-	-	-	-	-	-	-	Brewster-Wingard and Ishman 1999
	N	USA - Peters Creek, St. Johns River, Florida	30°02' 40.5" N	81°43'05.4"W	-	-	-	-	-	-	-	-	-	Deaton et al. 1989
	N	USA - Lower St. Johns River	29°41' 09.2" N	81°38'17.9"W	-	-	-	-	-	-	-	-	-	Mason Jr. 1998
	N	USA - St. Johns River	29°52' 06"N	78°47'17"W	14,150	-	-	-	-	-	-	-	-	Cooksey and Hyland 2007
	N	USA - Gulf of Mexico	24°45' 53"N	85°35'39"W	0.8	-	-	-	-	-	-	-	-	Montagna et al. 2008
	N	USA - Florida Bay	25°17' 26.0" N	80°59'59.1"W	-	-	-	-	-	-	-	-	-	Tabb and Manning 1961

Species	**N/I**	**Aquatic System**	**Lat**	**Long**	**Density (ind/m^2)**			**Size (mm)**			**Biomass/Secondary Production**			**References**
					Mean	***Min***	***Max***	***Mean***	***Min***	***Max***	***Mean***	***Min***	***Max***	
	N	USA - North River, Florida	25°19'16"N	80°56'01"W	-	-	-	-	6	18	-	-	-	Odum and Heald 1972
	N	USA - Virginia Key, Florida	25°44'08"N	80°10'27"W	-	-	-	22	-	-	-	-	-	Siddall 1980
	N	USA - Lake Pontchartrain, Louisiana	30°13'57"N	89°39'30"W	1,500	-	-	-	-	-	-	-	-	Brammer et al. 2007
	N	Gulf of Mexico	24°45'53"N	85°35'39"W	-	-	-	-	-	-	-	-	-	Britton and Morton 1989 apud Kennedy 2011a
	N	USA - South Coast	37°00'25.0"N	76°11'54.3"W	-	-	-	-	-	-	-	-	-	Conrad 1831 apud Kennedy 2011a
	N	USA - Virginia Rivers	37°00'25.0"N	76°11'54.3"W	-	-	-	-	-	-	-	-	-	Conrad 1857-1858 apud Kennedy 2011a
	N	Mexico - Chica-Grande System, Veracruz	20°02'00.0"N	96°36'00.0"W	-	-	-	-	-	-	-	-	-	García-Cubas et al. 1992
	N	USA - Patapsco River, Chesapeake Bay	39°14'58.0"N	76°36'20.1"W	-	-	-	-	-	-	-	-	-	Pathy and Mackie 1993
	N	USA - Chesapeake Bay	38°21'28.8"N	76°05'16.8"W	-	-	-	-	-	-	-	-	-	Kennedy 2011b
	N	Mexico - Mecoacán Lagoon	18°24'04.0"N	93°07'55.4"W	-	-	-	-	-	-	-	-	-	García-Cubas et al. 1990

Species	**N/I**	**Aquatic System**	**Lat**	**Long**	**Density (ind/m^2)**			**Size (mm)**			**Biomass/Secondary Production**			**References**
					Mean	***Min***	***Max***	***Mean***	***Min***	***Max***	***Mean***	***Min***	***Max***	
	N	USA - Kiawah Island, South Carolina	32°37'11.4"N	80°03'41.4"W	-	-	-	-	-	-	-	-	-	Flemming et al. 2008
	N	USA - Mobile-Tensaw Delta, Gulf of Mexico	30°37'31"N	87°56'10"W	-	-	-	-	-	-	25 g AFDM m^2	-	-	Chaplin and Valentine 2009
	N	Gulf of Mexico	24°45'53"N	85°35'39"W	-	-	-	-	-	-	-	-	-	Britton and Morton 1989 apud Kennedy 2011a
	N	Gulf of Mexico	24°45'53"N	85°35'39"W	-	-	-	-	-	-	-	-	-	García-Cubas 1981 apud Kennedy 2011a
	N/I	Belgium - Antwerp; USA - from Hudson River to Colorado River; Mexico - from Tampico to Veracruz	-	-	-	-	-	Belgium: 21.48; Texas, USA: 8.38; Louisiana, USA: 8.52; Florida, USA: 12.65	-	-	-	-	-	Marelli and Gray 1983
	N	USA - Chesapeake Bay to Florida	38°21'28.8"N	76°05'16.8"W	-	-	-	-	-	-	-	-	-	Johnson 1934 apud Kennedy 2011a
	I	USA - Hudson River, Haverstraw	41°11'31.3"N	73°57'00.5"W	-	-	-	-	-	-	-	-	-	Rehder 1937 apud Kennedy 2011a

Species	**N/I**	**Aquatic System**	**Lat**	**Long**	**Density (ind/m^2)**			**Size (mm)**			**Biomass/Secondary Production**			**References**
					Mean	***Min***	***Max***	***Mean***	***Min***	***Max***	***Mean***	***Min***	***Max***	
	I	USA - Hudson River, New York	40°43' 27.1" N	74°01'01.0"W	-	-	-	-	-	-	-	-	-	Jacobson 1953 apud Kennedy 2011a
	I	USA - Housatonic River and Charles River	41°11' 30.0" N	73°07'01.0"W	-	-	-	-	-	-	-	-	-	Smith and Boss 1996 apud Kennedy 2011a
	I	USA - Mississipi River and Missouri River	38°48' 52.1" N	90°07'14.4"W	-	-	-	-	-	-	-	-	-	Koch 1989 apud Kennedy 2011a
	I	USA - Hudson River Estuary	40°39' 47.0" N	74°02'36.9"W	-	-	-	-	-	-	-	-	-	Weiss et al. 1978
	I	USA - Hudson River	41°12' 49"N	73°57'50"W	-	-	3,200; 1,851	-	-	-	-	-	-	Menzie 1980
	I	USA - Hudson River Estuary	41°12' 49"N	73°57'50"W	-	-	2,000	-	-	-	-	-	-	Walton 1996
	I	USA - Hudson River Estuary	40°39' 47.0" N	74°02'36.9"W	-	-	-	-	-	-	-	-	-	Strayer and Malcom 2007
	I	USA - West River, Connecticut	41°18' 51.2" N	72°57'22.4"W	913.6	141	2,168	-	-	-	-	-	-	Richardson and Hammond 2016
	I	USA - Mississippi River and Hudson River	30°N, 42.2° N	90°W, 73.5°W	-	-	-	-	-	-	-	-	-	Stepien et al. 1999
	I	Trinidad and Tobago - Salybia River	10°43' 29.3" N	61°02'01.3"W	-	20	100	-	-	-	-	-	-	Mohammed et al. 2018

Species	**N/I**	**Aquatic System**	**Lat**	**Long**	**Density (ind/m^2)**			**Size (mm)**			**Biomass/Secondary Production**			**References**
					Mean	***Min***	***Max***	***Mean***	***Min***	***Max***	***Mean***	***Min***	***Max***	
	I	Venezuela - Lake Maracaibo, Zulia	9°48'57" N	71°33'24" W	13,200	-	> 20,000	-	1	30	-	-	-	Lodeiros et al. 2019
	I	French Guiana - Montsinéry	4°53'36.0"N	52°29'22.8"W	-	-	-	-	-	-	-	-	-	Massemin et al. 2011
	I	Brazil - Rodrigo de Freitas Lagoon, Rio de Janeiro	22°57'02"S	43°11'09"W	84,560	-	-	-	1	31.7	1.2 g AFDM m^2	-	-	Maia-Neto 2018
	I	Brazil - Rodrigo de Freitas Lagoon, Rio de Janeiro and Recife	22°58'45"S, 08°04'44"S	43°12'55"W, 34°53'25"W	-	-	-	-	-	-	-	-	-	Fernandes et al. 2018
	I	Brazil - Rodrigo de Freitas Lagoon, Rio de Janeiro	22°57'02"S	43°11'09"W	-	-	-	18.59	6.85	22.15	-	-	-	Rizzo et al. 2014
	I	Belgium - Port of Antwerp	51°14'33.0" N	4°24'20.3"E	-	-	-	-	-	-	-	-	-	Nyst 1835 apud Kennedy 2011a
	I	Belgium - Schelde River	51°21'31.2" N	4°17'30.4"E	-	-	-	-	-	-	-	-	-	Verween et al. 2007b
	I	Belgium - Port of Antwerp	51°14'33.0" N	4°24'20.3"E	-	-	-	-	-	-	-	-	-	Verween et al. 2007a
	I	Belgium - Schelde River	51°23'59"N	4°00'01"E	-	-	-	-	-	26	-	-	-	Verween et al. 2006

Species	**N/I**	**Aquatic System**	**Lat**	**Long**	**Density (ind/m²)**			**Size (mm)**			**Biomass/Secondary Production**			**References**
					Mean	***Min***	***Max***	***Mean***	***Min***	***Max***	***Mean***	***Min***	***Max***	
	I	Belgium - Schelde River	51°23'59"N	4°00'01"E	-	-	-	-	-	-	-	-	-	Verween et al. 2009
	I	Spain - Guadalquivir River	36°49'59.9"N	6°21'15.5"W	-	-	-	-	-	-	-	-	-	Escot et al. 2003
	I	Spain - Guadalquivir River	37°24'15.0"N	5°59'55.2"W	-	-	-	-	-	-	-	-	-	Clusa et al. 2017
	I	Spain - Guadalquivir River	37°24'15.4"N	6°00'48.6"W	-	-	-	-	-	-	-	-	-	Escot et al. 2007
	I	Germany - North Sea coast	53°31'49.7"N	8°33'55.8"E	-	-	-	-	-	-	-	-	-	Nehring and Leuchs 2000
	I	Germany - Mecklenburg-Vorpommern, Baltic Sea	54°37'51"N	13°40'50"E	-	-	50	-	-	-	-	-	-	Darr and Zettler 2000 apud Laine et al. 2006
	I	The Netherlands - North Sea Canal	52°27'51"N	4°39'0"E	-	-	204,000	-	1	24	-	-	-	Van der Gaag et al. 2017
	I	The Netherlands	51°19'16.4"N	3°49'24.0"E	-	-	-	-	-	-	-	-	-	Kuijper 2000
	I	The Netherlands - Waal River near Nijmegen	51°51'39.8"N	5°49'59.1"E	-	-	-	-	-	-	-	-	-	Kelleher et al. 1999
	I	The Netherlands	52°27'51"N	4°39'0"E	-	-	-	-	-	-	-	-	-	Wolff 2005

Species	**N/I**	**Aquatic System**	**Lat**	**Long**	**Density (ind/m^2)**			**Size (mm)**			**Biomass/Secondary Production**			**References**
					Mean	***Min***	***Max***	***Mean***	***Min***	***Max***	***Mean***	***Min***	***Max***	
	I	The Netherlands - Zijkanaal C	52°25'30"N	4°42'07"E	-	-	-	-	-	-	-	-	-	Verhofstad et al. 2013
	I	The Netherlands - North Sea Canal	52°27'51"N	4°39'0"E	-	-	-	-	-	-	-	-	-	Rajagopal et al. 1997
	I	The Netherlands - North Sea Canal, Velsen	52°27'49.0"N	4°38'07.7"E	-	-	-	-	-	-	-	-	-	Rajagopal et al. 2003
	I	The Netherlands - North Sea Canal, Velsen	52°27'49.0"N	4°38'07.7"E	-	-	-	-	-	-	-	-	-	Rajagopal et al. 2005
	I	The Netherlands - Electricity plant near Nijmegen	51°54'16"N	5°31'01"E	-	-	-	-	3.1	17.8	-	-	-	Kelleher et al. 1997
	I	The Netherlands - North Sea Canal, Velsen	52°27'49.0"N	4°38'07.7"E	-	-	-	-	-	-	-	-	37 kg/m^2	Rajagopal et al. 2002
	I	The Netherlands - North Sea Canal	52°27'51" N	4°39'0"E	-	-	> 100,000	-	-	-	-	-	-	Van der Gaag et al. 2014
	I	The Netherlands - North Sea Canal, near Velsen	52°27'49.0"N	4°38'07.7"E	-	-	-	-	-	-	-	-	-	Van der Gaag et al. 2018

Species	**N/I**	**Aquatic System**	**Lat**	**Long**	**Density (ind/m^2)**			**Size (mm)**			**Biomass/Secondary Production**			**References**
					Mean	***Min***	***Max***	***Mean***	***Min***	***Max***	***Mean***	***Min***	***Max***	
	I	The Netherlands - North Sea Canal	52°27' 51" N	4°39'0" E	-	-	-	-	-	-	-	-	1.8 g	Stekhoven et al. 2015
	I	France - Caen	49°20' 10.3" N	0°27'00 .8"W	-	-	-	-	-	-	-	-	-	Chemin 1910-1911 apud Kennedy 2011a
	I	France - Bretagne	48°38' 33.7" N	2°03'20 .2"W	-	-	-	-	-	-	-	-	-	Le Mao 2003 apud Kennedy 2011a
	I	France - Normandy	49°17' 06" N	0°14'45 "W	2,145	-	-	-	-	-	-	-	0.23 g	Charles et al. 2018
	I	England - Cardiff Docks	51°27' 28" N	3°09'59 "E	-	-	-	-	-	20	-	-	-	Oliver et al. 1998 apud Verween et al. 2010
	I	Wales - Pembrokeshire	51°40' 53"N	4°55'24 "W	-	-	-	12	-	-	-	-	-	Oliver 2015
	I	England - Thames River	51°29' 02" N	0°27'03 "E	-	-	-	-	-	15.2	-	-	-	Bamber and Taylor 2002 apud Verween et al. 2010
	I	Poland - Vistula Delta	54°38' N	19°06' E	-	-	69	-	5	22	-	-	-	Brzana et al. 2017
	I	Poland - Gulf of Gdańsk	54°38' N	19°06' E	-	-	222	-	1.4	4.9	-	-	-	Dziubińska 2011
	I	Poland - Gulf of Gdańsk	54°38' N	19°06' E	-	-	-	-	8	13	-	-	-	Brzana and Janas 2016
	I	Finland - Archipelago Sea, Baltic Sea	59°42' 59"N	22°29'3 1"E	-	-	-	-	2.06	11.42	-	-	-	Forsström et al. 2016

Species	**N/I**	**Aquatic System**	**Lat**	**Long**	**Density (ind/m²)**			**Size (mm)**			**Biomass/Secondary Production**			**References**
					Mean	***Min***	***Max***	***Mean***	***Min***	***Max***	***Mean***	***Min***	***Max***	
	I	Finland - Island of Hästholmen, Loviisa, Gulf of Finland	60°22' N	26°21' E	-	-	28,000	-	-	21	-	-	9.8 kg/m²	Laine et al. 2006
	I	Sweden - Bothnian Sea	60°27' N	18°12' E	-	-	8,000	-	1	23	-	-	-	Florin et al. 2013
	I	Georgia - Patara Paliastomi Lake	42°07' 21"N	41°46'12"E	-	-	-	-	6.3	16	-	-	-	Mumladze et al. 2019
	I	Ukraine - Dniester Liman, Black Sea	46°18' 11.1" N	30°14'15.9"E	-	-	-	-	-	-	-	-	-	Therriault et al. 2004
	I	Russian - Taganrog Bay, Azov Sea	47º05' 07"N	39º13'27"E	-	-	-	-	12.8	15.4	-	-	-	Zhulidov et al. 2015
	I	Ukraine and Russian - Black Sea and Caspian Sea	46º10' 15"N; 45º38' 10"N	30º24'13"E; 47º42'52"E	-	-	2,400	-	3.8	24.5	-	-	-	Zhulidov et al. 2018
	I	Iran - Caspian Sea	37°28' 54.97" N	49°27'37.01"E	-	-	-	13.15	5.5	22	-	-	-	Heiler et al. 2010
	I	Egypt - Mediterranean Sea, Damietta	31°24' 50"N	31°48'35"E	101	-	-	-	-	-	-	-	-	Zyadah et al. 2004
Mytilopsis sallei	**N/I**	USA - from Tampa to Miami; Caribbean	-	-	-	-	-	Laguna Bacelar, Mexico: 12.71;	-	-	-	-	-	Marelli and Gray 1983

Species	**N/I**	**Aquatic System**	**Lat**	**Long**	**Density (ind/m^2)**			**Size (mm)**			**Biomass/Secondary Production**			**References**
					Mean	***Min***	***Max***	***Mean***	***Min***	***Max***	***Mean***	***Min***	***Max***	
		Islands - from Bahamas Islands to Dominica; Western Atlantic Mainland - from Mexico to Venezuela; Indo-Pacific: Fiji and India						11.66; Lake Izabal, Guatemala: 18.67; St. Mary, Jamaica: 24.09; Roseau, Dominica: 15.04; Torbeck, Haiti: 25.57						
	N	Mexico - Chetumal Bay	17°53' N	87°51' W	-	-	-	-	5.57	12.54	0.1174 g	-	-	González et al. 2009
	N	Colombia - Santa Marta, Cartagena and Coveñas	-	-	-	-	-	-	-	-	-	-	-	Osorio 2013
	I	Panama Canal	8°55'36.8"N	79°33'27.2"W	-	-	-	-	-	-	-	-	-	Cohen 2006
	I	India - Visakhapatnam Harbour	17°41' N	83°20' E	-	-	-	-	-	-	-	-	-	Devi 1996
	I	India - Visakhapatnam Harbour	17°41'34"N	83°17'45"E	-	-	-	-	2	31	-	-	-	Morton 1981
	I	India	19°01'37.0" N	72°59'35.9"E	-	-	-	-	-	-	-	100 kg/m^2/year	120 kg/m^2/year	Shetty et al. 1989

Species	**N/I**	**Aquatic System**	**Lat**	**Long**	**Density (ind/m²)**			**Size (mm)**			**Biomass/Secondary Production**			**References**
					Mean	***Min***	***Max***	***Mean***	***Min***	***Max***	***Mean***	***Min***	***Max***	
	I	India - Visakhapatnam Harbour	17°41' 34"N	83°17'45"E	-	-	56,000	-	-	-	-	-	-	Pati et al. 2012
	I	India - Visakhapatnam Harbour	17°40' N	83°16' E	-	-	-	-	-	-	-	-	-	Pati et al. 2013
	I	Hong Kong - Government Dockyard	22°16' 59"N	114°10' 08"E	-	-	-	-	2	28	-	-	228.6 g	Morton 1989
	I	Hong Kong - Government Dockyard	22°16' 59"N	114°10' 08"E	-	-	-	-	-	-	-	-	-	Morton 2008
	I	Hong Kong - Kwun Tong	22º18' N	114º13' E	-	-	14,400	-	-	-	-	-	3,372 g/m²	Astudillo et al. 2014
	I	Hong Kong - Kwun Tung, Kowloon	22°18' 39.4" N	114°12' 55.8"E	-	-	-	-	-	-	-	-	-	Astudillo et al. 2017
	I	Singapore - Sungei Pandan and others	1º18'N	103º45' E	-	-	31,800	-	-	29	-	-	-	Tan and Morton 2006
	I	Malaysia - River Sungei Sekudai, Johor Bahru	1º29'N	103º42' E	-	-	83,000	-	-	24	-	-	-	Tan and Morton 2006
	I	Australian - Cullen Bay Marina, Darwin Harbour	12°27' S	130°49' E	-	-	23,650	-	-	-	-	-	-	Willan et al. 2000
	I	Australia - Cullen Bay	12°27' S	130°49' E	-	-	-	-	-	-	-	-	-	Russel and Hewitt 2000

Species	**N/I**	**Aquatic System**	**Lat**	**Long**	**Density (ind/m^2)**			**Size (mm)**			**Biomass/Secondary Production**			**References**
					Mean	***Min***	***Max***	***Mean***	***Min***	***Max***	***Mean***	***Min***	***Max***	
		Marina, Darwin Harbour												
	I	China - Yundang Lagoon, Xiamen	24°36' N	117°55' E	-	-	29,005	-	-	-	-	-	5,943 g/m^2	Cai et al. 2014
	I	China - Yundang Lagoon, Xiamen	24°36' N	117°55' E	-	-	-	-	-	-	-	-	-	Magni et al. 2019
	I	China - Maluan Bay, Xiamen	24°33' 00.0" N	118°01' 00.0"E	-	-	-	-	-	-	-	-	-	He et al. 2016
	I	Hong Kong - Shing Mun River, Shatin	22°23' 26"N	114°11' 44"E	-	-	76,000	-	3.5	28	-	-	-	Morton and Leung 2015
	I	India - Cochin Harbour	9°50'4 3.9"N	76°17'1 7.2"E	-	-	748	-	5	24	-	-	-	Jayachandran et al. 2018
	I	Vietnam - Ba Tai Lake, Kien Giang Province	10°12' 27"N	104°36' 11"E	-	-	-	-	-	35	-	-	-	Lutaenko et al. 2019
	I	India - Mumbai Harbour	18°54' 00.0" N	72°40'0 0.0"E	-	-	-	-	-	-	-	-	-	Gaonkar et al. 2010a
	I	India - Mumbai Harbour	18°54' 00.0" N	72°40'0 0.0"E	-	-	-	-	-	-	-	-	-	Gaonkar et al. 2010b
	I	Taiwan - Port of Kaohsiung	22°39' 39.0" N	120°21' 08.1"E	-	-	-	-	3	29	-	-	-	Minchin et al. 2016

Species	**N/I**	**Aquatic System**	**Lat**	**Long**	**Density (ind/m^2)**			**Size (mm)**			**Biomass/Secondary Production**			**References**
					Mean	***Min***	***Max***	***Mean***	***Min***	***Max***	***Mean***	***Min***	***Max***	
	N	Mexico - Chetumal Bay	17°53' N	87°51' W	-	-	-	-	5.07	12.54	-	0.034 g	0.413 g	Llanes-Baeza and González 2002
	N	Colombia - Archipelago of San Andres and Old Providence	12°32' 48"N	81°43'5 7W	-	-	-	12.1	-	-	-	-	-	Vilardy and Polanía 2002
	N	Colombia - Hooker and Honda Bays, Sand Andrés Island	12°33' 48"N	81°43'5 7W	-	-	-	-	-	-	-	-	-	Murillo 2004
	I	Egypt - Mouth of River Nile, Dumyat	30°32' 12"N	33°10'0 1"E	-	-	-	-	-	22	-	-	-	Hoffman et al. 2006
	N	Panama Canal - Gatún Locks	9°19'0 1.0"N	79°55'1 3.6"W	-	-	-	-	-	-	-	-	-	Jones and Rützier 1975
	N	Panama - Las Minas Bay	9°22'5 1.5"N	79°49'5 3.9"W	-	-	-	-	-	-	-	-	-	Garrity and Levings 1993
	N	Panama - Las Minas Bay	9°22'5 1.5"N	79°49'5 3.9"W	-	-	-	-	-	-	-	-	-	Keller and Jackson 1991
	I	Japan - Tokio Bay	45°10' N	141°15' E	-	-	-	24.20	-	-	0.38 g	-	-	Takeuchi et al. 2009
	N	Colombia - Cispatá Bay	9°24'1 5.2"N	75°50'1 0.7"W	-	-	-	-	-	-	-	-	-	Moreno-Ríos 2007
	N	Colombia - Cartagena Bay	10°23' 58.99" N	75°30'5 1. 98"W	-	-	900	-	-	13	-	-	-	Suarez-Mozo 2011
	I	Philipines - Manilla Bay	14°34' 43.1" N	120°58' 18.7"E	-	-	-	-	-	-	-	-	-	Vallejo et al. 2017

Species	N/I	Aquatic System	Lat	Long	Density (ind/m^2)			Size (mm)			Biomass/Secondary Production			References
					Mean	*Min*	*Max*	*Mean*	*Min*	*Max*	*Mean*	*Min*	*Max*	
	I	Philipines - Manilla Bay	14°34'43.1"N	120°58'18.7"E	-	-	-	-	-	-	-	-	-	Vallejo et al. 2019
	I	Philipines - Manilla Bay	14°34'43.1"N	120°58'18.7"E	-	-	-	-	-	-	-	-	-	Trinidad et al. 2019
	I	Japan - Osaka Bay	34°27'35.9"N	135°11'27.7"E	-	-	-	-	-	-	-	-	-	Otani and Willan 2017
	I	Japan - Osaka Bay	34°27'35.9"N	135°11'27.7"E	-	-	-	-	-	-	-	-	-	Otani 2002
	I	Panama - Pedro Miguel Locks	9°01'20.3"N	79°37'10.1"W	-	-	-	-	-	-	-	-	-	Hildebrand 1939
	N	Cuba - Santiago de Cuba	19°58'00.7"N	75°52'20.4"W	-	-	-	-	-	-	-	-	-	García and Capote 2013
	N	Venezuela - Alta Guajira, Zulia	11°33'10.0"N	71°58'01.0"W	-	-	-	-	-	-	-	-	-	Reyes et al. 2007
	N	Colombia - Caribe and Magdalena-Cauca	10°43'43.4"N	74°25'39.1"W	-	-	5,000	-	-	-	-	-	-	Puyana 1995 apud Linares et al. 2018
	I	China - Yundang Lagoon, Xiamen	24°36'N	117°55'E	-	-	> 10,000	-	-	-	-	-	163.98 g	Xie et al. 2019
Mytilopsis* cf. *sallei	**I**	Brazil - Recife Port, Pernambuco	8º04'53"S	34º53'18"W	-	3,600	176,800	-	0.6	27.22	-	-	-	Souza et al. 2005
	I	Brazil - Pina Basin and	8º3'48''S;	34º52'15"W;	-	-	119,300	-	1.1	38.02	-	-	-	Freitas 2009

Species	N/I	Aquatic System	Lat	Long	Density (ind/m^2)			Size (mm)			Biomass/Secondary Production			References
					Mean	*Min*	*Max*	*Mean*	*Min*	*Max*	*Mean*	*Min*	*Max*	
		Capibaribe River, Pernambuco	8º3'46''S	34º24'4"W										
	I	Brazil - Mamanguape and Paraiba stuaries	6°46'08.2"S, 7°00'10.6"S	34°55'57.7"W, 34°50'40.3"W	-	-	-	-	-	-	-	-	-	Dolbeth et al. 2016
	I	Brazil - Port of Recife, Pernambuco	8°03'19.0"S	34°52'28.3"W	-	-	-	-	-	-	-	-	-	Farrapeira et al. 2007
	I	Brazil - Capibaribe River, Pernambuco	8º3'48''S; 8º3'46''S	34º52'15"W; 34º24'4"W	-	-	130,000	-	-	-	-	-	-	Freitas-Galeão and Souza 2015
	I	Brazil - Capibaribe River Estuary, Pernambuco	8º4'46''S	34º53'26"W	-	-	-	25.66	23	30	-	-	-	Maciel 2011
	I	Brazil - Port of Natal, Rio Grande do Norte	5°45'57.1"S	35°12'37.7"W	-	-	-	-	-	-	-	-	-	Farrapeira et al. 2010
Mytilopsis adamsi	**I**	Thailand - Haad-kaew Lagoon	7°14'14.35" N	100°33'47.04" E	-	-	569,000	25	-	-	-	-	-	Wangkulangkul 2009
	I	Thailand - Songkhla Lagoon System	7°10'34.40" N	100°34'00"E	-	-	-	-	2	32	-	-	-	Wangkulangkul 2018
	I	Thailand - Pak Phanang Estuary	8°23'58.10" N	100°08'54.75" E	-	-	-	-	2	31	-	-	-	Wangkulangkul and Lheknim 2008

Species	N/I	Aquatic System	Lat	Long	Density (ind/m²)			Size (mm)			Biomass/Secondary Production			References
					Mean	*Min*	*Max*	*Mean*	*Min*	*Max*	*Mean*	*Min*	*Max*	
	I	Thailand - Thale Sap Songkhla	7°10'34.40"N	100°34'00"E	-	-	-	-	3	24	-	-	-	Wangkulangkul and Lheknim 2008
	N	Panama - Pacific coast	8°39'13.5"N	79°45'49.9"W	-	-	-	-	-	-	-	-	-	Morrison 1946 apud Salgado-Barragán and Toledano-Granados 2006
	I	Mexico - Urías Estuary	23°10'36"N	106°20'00"W	-	-	-	-	2	21.2	-	-	-	Salgado-Barragán and Toledano-Granados 2006
	N	Mexico - Urías Estuary	23°10'36"N	106°20'00"W	-	-	-	-	-	-	-	-	-	Okolodkov et al. 2007
	I	Philipines - Manilla Bay	14°34'43.1"N	120°58'18.7"E	-	-	-	-	-	-	-	-	-	Vallejo et al. 2017
	N	Peru - Chimbote, Ancash	9°04'52.8"S	78°35'45.9"W	-	-	-	-	-	-	-	-	-	Paredes et al. 2016
	I	Mauritius - Port Louis	20°09'21.0"S	57°29'45.4"E	-	-	-	-	-	-	-	-	-	Awad 2018
Mytilopsis trautwineana	**I**	Colombia - Cartagena	10°23'58.99"N	75°30'51.98"W	-	-	-	-	4	24	-	-	-	Aldridge et al. 2008
	I	Colombia - Cartagena	10°08'00.0"N	75°32'00.0"W	-	-	-	-	-	-	-	-	-	Aguado 2010
	N	Ecuador	0°36'09.2"S	80°24'43.6"W	-	-	-	-	-	-	-	-	-	Linares et al. 2018
Mytilopsis africana	**N**	West coast of Africa - from	5°15'07.5"N,	3°59'43.2"W,	-	-	-	-	-	-	-	-	-	Le Loeuff 1999

Species	**N/I**	**Aquatic System**	**Lat**	**Long**	**Density (ind/m²)**			**Size (mm)**			**Biomass/Secondary Production**			**References**
					Mean	***Min***	***Max***	***Mean***	***Min***	***Max***	***Mean***	***Min***	***Max***	
		Ivory Coast to Gabon	1°47'00.3"S	9°11'40.3"E										

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Supplementary Material S2

Wide tolerance to environmental conditions and substrate colonization mediates the invasion of false mussels (Bivalvia: Dreissenidae) in brackish systems

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Table S2. Geographical data for aquatic systems where *Mytilopsis* spp. occur, as native (N) or invasive (I) species, and their key surface water variables.

	N/ I	Aquatic System	Lat	Long	Salinity (S)			Temperature (°C)			Chlorophyll *a* (mg L^{-1})			Dissolved oxygen (mg L^{-1})			Transparency (cm)			References
					X	*Min*	*Max*	*X*	*Min*	*Max*	*X*	*Min*	*Max*	*X*	*Min*	*Max*	*X*	*Min*	*Max*	
M. adamsi	**I**	México - Urias Estuary	23°10'36"N	106°20'00"W	31.37	17.40	40.70	23.26	23.35	29.95	0.01335	0.00410	0.03130	3.95	2.30	4.36	100	-	-	Guerrero-Galván et al. 1999; Barranco-Ramírez et al. 2002; Ochoa-Izaguirre et al. 2002
	N	Panama - San Jose Island	24°52´N	110°43´W	34.90	34.69	35.10	24.80	18.5	31.00	-	-	-	7.60	6.65	8.64	-	-	-	Villegas-Sánchez et al. 2009; Barjau-González et al. 2012
	N	Peru - Chimbote Bay	9°07'22''S	78°35'20"O	35.04	34.60	35.50	28.75	16.35	23.05	0.00200	0.00050	0.00600	6.71	5.37	8.11	185	120	250	Guillén et al. 1977
	I	Philippines - Manila Bay	14°34'43.06"N	120°58'18.69"E	28.78	21.80	31.40	27.90	25.9	31.30	0.00510	0.00020	0.02550	5.15	0.79	7.25	25.08	13.54	36.99	Tamura et al. 2003; Jacinto et al. 2011; Vallejo et al. 2017; Trinidad et al. 2019; Vallejo et al. 2019
	I	Thailand - Haad-kaew Lagoon	7°14'14.35"N	100°33'47.04"E	20.55	13.35	31.75	27.70	18.7	36.70	-	-	-	8.20	2.90	13.50	-	-	-	Wangkulangkul and Lheknim 2008; Wangkulangkul 2009

	I	Thailand - Pak Phanang Estuary	8°23'58.10"N	100°08'54.75"E	21.40	0.95	25.50	30.15	25.75	34.60	0.01510	0.00610	0.02400	4.90	4.0	6.20	22.50	20	25	Foulkes et al. 2007; Islam et al. 2007; Shinnaka et al. 2007; Wangkulangkul and Lheknim 2008; Wangkulangkul 2009
	I	Thailand - Thale Sap Songkhla Lake	7°10'34.40"N	100°34'00"E	13.33	0.50	31.00	29.25	26.00	33.00	0.01950	0.00250	0.04250	9.20	-	-	0.85	30	150	Yamaguchi et al. 1994; Angsupanich 1997; Angsupanich and Kuwabara 1999; Wangkulangkul and Lheknim 2008; Wangkulangkul 2018
M. leucophaeata	**I**	Belgium - Scheldt Estuary (Port of Antwerp)	51°23'59°N	4°00'01"E	16.25	7.0	30.00	12.00	0	25	0.01000	0	0.05800	4.00	1.00	20	75	25	100	Somville and De Pauw 1982; Kromkamp and Van Engeland 2010
	I	Brazil - Rodrigo de Freitas Lagoon	22°57'02"S	43°11'09"W	15.05	11.77	18.69	25.25	20.79	29.42	0.05750	0.00053	0.48680	7.23	0.80	13.52	78.70	17.50	175	Van Weerelt et al. 2012; Maia-Neto 2018; Neves et al. 2020
	I	Egypt - Damietta Branch	31°24'50"N	31°48'35"E	32.00	9.87	42.00	19.95	17	35.80	0.01600	0.00300	0.02000	8.16	6.45	10	100	50	150	Zyadah et al. 2004; Abdo 2010; Hamouda 2014
	I	Finland - Archipelago Aland, North Baltic Sea	59°42'59"N	22°29'31"E	6.0	1.0	30	16	5.5	20	0.002975	0.00145	0.00450	3.0	1.5	6.0	407.5	90	890	Carstensen et al. 2005; Lundberg 2005
	I	Finland - Hastholmen Island, Gulf da Finland	60°22.1'N	26°21.3'E	4.15	0	6.0	13.5	5.0	21	-	-	-	5.75	4.3	7.2	279.42	15	610	Pitkänen et al. 2001; Laine et al. 2006; Ilus 2009
	I	France - Carentan Channel	49°18'N	1°14'W	16.79	9.78	23.8	7.335	6.0	8.67	-	-	-	-	-	-	-	-	-	Charles et al. 2018
	I	France - Cherbourg Rade	49°38'N	1°37'W	30.36	26.06	34.65	13.11	8.1	17.25	0.00750	0.0050	0.01	9.88	8.82	10.95	22.23	8.09	32.13	Charles et al. 2018
	I	Georgia - Patara Paliastomi Lake	42°07'21"N	41°46'12"E	5.50	1.0	10	-	0	-	-	-	-	-	-	-	-	-	-	Mumladze et al. 2019

	I	Germany - Vorpommersche Boddenlandschaft National Park	54°37'51"N	13°40'50"E	10.15	3.5	12	10	0	20	78.25	27	245	6.3	5.5	6.8	-	-	-	Schiewer et al. 1999; Schumann et al. 2006; Heyl 2016
	I	Great Britain - Pembroke River, Pembrokeshire, South Wales	51°40'53"N	4°55'24" W	30.0	26	34	14.75	10	19.5	-	-	-	-	-	-	-	-	-	Nelson-Smith 1965
	I	Iran - Caspian Sea	37°28'54.97"N	49°27'37.01"E	9.76	6.97	12.94	18.36	8.275	28.45	0.00563	0.00050	0.0534	7.3225	4.5	6.99	285	135	460	Kideys and Moghim 2003; Kideys et al. 2008; Heiler et al. 2010; Bagheri et al. 2012
	N	Mexico - Lagoon System Chica-Grande, Veracruz	20°06'S	96°42'W	7.0	5.25	8.7	29.95	23.2	31.3	0.05116	0.01585	0.09822	5.847	4.96	7.08	0.1	0.05	0.2	García-Cubas et al. 1992; Mecalco-Hernández 2010
	N	Mexico - Mecoacán	18°25'N	93°10'W	20.0	4.5	35.5	28	25	31	0.02860	-	-	5.44	-	-	25.6	-	-	George-Zamora and Aranda 2005; Lanza-Espino et al. 2010; García-Cruz et al. 2018
	I	Netherlands - North Sea Canal (Noordzeekanaal)	52°27'51" N	4°39'0" E	5.57	2.95	9.1	16.5	3.7	25.05	0.02116	0.00089	0.0655	6.3	-	-	-	-	-	Rajagopal et al. 1997; Rajagopal et al. 2002; Rajagopal et al. 2003; Rajagopal et al. 2005; Kennedy 2011; Verhofstad et al. 2013; Van der Gaag et al. 2014; Stekhoven et al. 2015; Van der Gaag et al. 2018
	I	Netherlands - Waal River close to Nijmegen	51°54'16"N	5°31'01" E	0.22	0.2	0.3	16	12	20	0.03500	0.02000	0.0500	-	-	-	77.5	40	200	Spaink et al. 1998; Van de Wolfshaar et al. 2010; Spikmans et al. 2020
	I	Poland - Gulf of Gdansk	54°38'54°41' N	18°93' - 19°06' E	2.30	1.17	3.57	12	10	14	0.10875	0.00190	0.2156	11.19	5.71	19.03	110	20	200	Renk et al. 2001; Zgrundo and Bogaczewicz-Adamczak 2004; Zajączkowski et al. 2010
	I	Russia - Taman Bay	45°16'26"N	36°46'59"E	13.30	0.1	26.5	19	6	32	-	-	-	-	-	-	-	-	-	Zhulidov et al 2015; Spiridonov et al. 2016

	I	Spain - Guadalquivir River, Sevilla	37°13'20"N	5°54'01"W	30.0	0	50	20	10	27	0.00100	0.0003	0.0037	6.0	3.0	11	138.41	63.59	162.03	Mendiguchía et al. 2007; Acker et al. 2009; Díaz-Delgado et al. 2010
	I	Sweden - Öregrundsgrepn - Forsmark Nuclear Power Plant. Bothnian Sea	60°27'N	18°12'E	5.50	5	6	12	3	18	0.00500	0.001	0.015	-	-	-	200	180	420	Eriksson et al. 1977; Erichsen et al. 2010
	I	Trinidad and Tobago - Dry River / Salybia River, Toco	10°43'29.3”N	61°02'01.3”W	20.0	5	35	28.5	25	32	-	-	-	-	-	-	-	-	-	Belford et al. 2019
	I	Ukraine - Dniester Estuary, Black Sea	46°07'N	30°45'E	10.90	0.5	17	17.52	10.7	11.6	0.01000	0.008	0.023	-	-	-	37.5	15	60	Kiseliova 2008; Goncharov et al. 2018
	N	USA - Chesapeake Bay	37°33'02"N	72°39'53"W	11.78	5.08	14.2	17.83	11	24.67	0.03000	0.009	0.0774	3.4	0.5	7.9	80	28	150	Karlsen et al. 2000; Gitelson et al. 2007
	N	USA - Dan River, Peters Creek, North Carolina	36°28"26"N	80°17'04"W	-	-	-	5.7	4.2	7.19	0.02000	0.010	0.03	3.1	0.1	6	-	-	-	Cappelin 2019
	I	USA - Edgewood Park Pond, Lower West River, Connecticut	41°18'51.2"N	72°57'22.4"W	17.50	10	25	21.7	19.3	27.5	-	-	-	0	0	4.8	-	-	-	Casagrande 1997; Richardson and Hammond 2016
	N	USA - Escambia River, Florida	30°37'23"N	87°09'58"W	16.20	10	21.6	20.5	18.5	22.5	0.00884	0.00114	0.02323	6.5	4.3	8.7	40.87	23.47	53.72	Lores and Sprecht 2001; Han and Jordan 2005
	N	USA - Florida Bay	25°1'59"N	80°45'36"W	31.70	26.4	36.2	26	12	40	0.00600	0.0004	0.0140	6.5	5	9	1.0	0.6	1.5	Costello et al. 1986; D'sa et al. 2000; Melesse et al. 2008
	N	USA - Gulf of Mexico	24°45'53"N	85°35'39"W	35.50	35	36.5	28.5	27.5	29.5	0.00070	0.00004	0.00135	6.1	4.6	7.6	12.6	1.0	42	Signoret et al. 2006; Lugo-Fernández et al. 2008
	N	USA - Harlowe Creek, Newport River Estuary	34°45'35"N	76°40'46"W	25.05	11.9	33.8	19	8.65	27.95	0.00238	0.00069	0.00675	9.5	3.6	14.4	60.77	0	89.59	Palumbo and Ferguson 1978; Coulliette and Noble 2008
	I	USA - Haverstraw Bay, Upper Hudson	41°12'49"N	73°57'50"W	3.60	0	7.1	10.7	1.2	20.2	0.02640	0.0032	0.0746	9.3	5.5	13.1	142	70	150	Sirois and Fredrick 1978; Sweka et al. 2007; Michelena et al. 2014

		River Estuary, New York																		
	N	USA - Kiawah Creek, South Carolina	32°36'N	80°08'W	16.90	13.75	19.2	18.55	8.5	27.25	-	-	-	7	3.4	13.6	30.15	15.69	42.45	Manzi et al. 1977; Flemming 2006
	I	USA - Mississippi River	33°51'59"N	58°58'14"W	0.05	0	1.0	20	3	30	0.02100	0.004	0.038	5.3	-	-	35	18	80	Allen et al. 1999; Lane et al. 2007
	I	USA - Missouri River	44°39'45"N	85°35'15"W	5.50	1.0	10	15	2	30	0.02360	0.0045	0.107	6.7	3.5	9.9	44.5	7.6	81.3	Berner 1951; Knowlton and Jones 2000
	N	USA - Neches River, Texas	29°58'59"N	93°52'57"W	2.70	0.4	9.0	21.1	9.1	34.4	0.00300	0.00100	0.00600	5.40	3.10	8.05	86.90	53.72	112.16	Bianchi et al. 1997
	N	USA - Pontchartrain Lake, Louisiana	30°13'57"N	89°39'30"W	5.63	1.75	11.25	21.6	11.05	32.20	0.00400	0.00030	0.0069	9.20	8.30	10.5	96.4	55.8	144.2	Francis et al. 1994; Bianchi and Argyrou 1997; Duffy and Baltz 1998
	N	USA - Sabine River, Texas	30°50'16"N	91°33'16"W	2.70	0.4	9.0	20.55	9.55	31.2	0.00300	0.00100	0.00600	6.50	2.80	10.47	136.71	79.25	179.30	Bianchi et al. 1996; Bianchi et al. 1997
	N	USA - St. Johns River, Florida	29°52'06"N	78°47'17"W	15.20	4.3	26	19.3	11.8	26.8	0.00990	0.00390	0.0141	6.15	4.30	7.95	53.19	29.46	48.34	Tagatz 1968; Cooksey and Hyland 2007; Environmental Protection Board 2008; Fisher et al. 2009
	N	USA - Trent River, North Carolina	35°03'55"N	77°19'04"W	3.0	0	13	20	2	35	0.01093	0.00200	0.06000	7.83	4.50	14	-	-	-	Wool et al. 2003
	N	USA - Virginia Key, Miami, Florida	25°44'08"N	80°10'27"W	15.0	8.0	22	16.5	13	20	1.30000	0.40000	2.20000	-	-	-	-	-	-	Siddall 1980; Skjoldal 1982
	I	Venezuela-Maracaibo Lake	9°48'57" N	71°33'24" W	4.42	2.1	6.8	25.17	17.93	31.32	0.01280	0.00350	0.02200	5.60	2.34	8.86	210	120	300	Gardner et al. 1998; Montiel et al. 2013; Lodeiros et al. 2019
	I	Venezuela-Neveri River, Barcelona, Anzoátegui	10°10'30"N	64°40'04"W	36.10	36.01	36.23	27.7	27.57	27.87	-	-	-	-	-	-	-	-	-	Lorenzoni et al. 2009
M. salle	**I**	Australia - Darwin Cullen Bay Marina	12°27'S	12°27'S	21.38	8.67	36.33	29.0125	24.92	33.07	-	-	-	7.05	6.68	7.43	119.25	57.90	161.87	Russell and Hewitt 2000; Marshall et al. 2003; Williams et al. 2006; Bax et al. 2002
	I	China - Hong Kong, Mirs Bay	22°32'00"N	114°23'43"E	31.50	30.3	32.5	24	18	30	0.00080	0.0003	0.0065	7.1	6.5	7.5	90	30	150	Wear et al. 1984; Lie and Wong 2010; Li et

i																				al. 2014; Nazeer and Nichol 2016
	I	China - Hong Kong, Mun River	22°23'26"N	114°11'44"E	15.40	5.4	25.4	23.6	20.7	26.4	0.03014	-	-	9.1	6.0	11.9	110	-	-	Hodgkiss and Yim 1995; Chen et al. 2000; Morton and Leung 2015
	I	China - Hong Kong, Tai Tam Bay	22°12'17"N	114°12'30"E	27.30	17.33	33.33	23.4	15.2	30.4	0.00300	0	0.03	8.08	-	-	-	-	-	Chiu et al. 1994; Chan and Morton 2001; Wangkulangkul and Lheknim 2008; Astudillo et al. 2014; Cai et al. 2014; Astudillo et al. 2017; Wangkulangkul 2018
	I	China - Hong Kong, Victoria Harbour	22°16'59"N	114°10'08"E	30.0	26	32	24	17	28	0.00700	0.00100	0.013	6.0	1.0	12	-	-	-	Xu et al. 2010
	I	China - Maluan Bay	24°31'51"N	118°00'37"E	23.70	22.1	26.17	23.95	21.5	26.4	0.01290	0.00280	0.0271	10.2	7.075	12.715	-	-	-	Wang et al. 2011; Wang et al. 2012; Lutaenko et al. 2019
	I	China - Yundang Lagoon, Xiamen	24°36'N	117°55'E	27.17	24.5	28	22.22	15.4	29.14	0.27000	0.15000	0.39	5.1775	3.105	7.25	-	-	-	Cai et al. 2014; Zheng et al. 2015; Magni et al. 2019; Xie et al. 2019
	I	Colombia - Ciénaga Grande de Santa Marta, Magdalena delta River	10°51'10"N	73°58'32"W	21.0	9.0	33	26.5	25.3	28	0.00300	0.001	0.005	6.2	2.0	6.2	70	-	-	Ramírez and Rueda 1999; Polanía et al. 2001; Rueda-Roa and Muller-Karger 2013
	N	Colombia - San Andrés, Honda Bay	12°32'48"N	81°43'57W	25.33	19.67	31	28.83	27.9	29.73	-	-	-	7.47	5.2	9.67	-	-	-	Vilardy and Polanía 2002; Murillo 2004
	N	Cuba - Santiago de Cuba Bay	19°59'27"N	75°45'03"W	35.60	35	36.2	32.25	31.5	33.9	0.01830	0.01180	0.025	5.75	4.4	6.95	130	110	160	Gómez et al. 2001; Regadera et al. 2010
	I	Egypt - Suez Canal and Nilo Delta	30°32'12"N	33°10'01"E	41.90	41.6	42.5	26.85	17.6	28.2	0.00187	0.00079	0.00404	7.6	6.67	9.15	383	233	667	Fahmy et al. 2005; Swami and Udhayakumar 2010
	I	Fiji - Viti levu Island	17°31'21"S	179°35'29"W	36.0	34	37.6	27.45	24.4	30.5	0.00200	0.00100	0.003	6.1	1.8	8.2	190	0	300	Taloiburi 2009; Buhadi et al. 2013
	N	Guatemala - Dulce River	08°23'-08°45'N	83°05'-83°29'W	27.50	20	35	15.65	1.3	30	-	-	-	-	-	-	-	-	-	Acuña-González et al. 2006
	N	Guatemala - Lake Izabal	15°26'46"N	89°15'96"W	122	22	36	26.325	25.05	29.2	-	-	-	7.075	5.24	7.86	-	-	-	Michot et al. 2002; Pérez et al. 2011; Romero-Viana et al. 2013

	I	India - Mumbai Port	18°54'N	72°40'E	32.92	24.97	37.03	28.33	25.06	31.13	5.90000	3.20000	8.60000	4.33	2.19	6.72	110	40.0	180	Gupta et al. 2005; Sawant et al. 2007; Gaonkar et al. 2010
	I	India - Thevara channel, Kochin backwaters	9°50'43.9"N	76°17'17.2"E	12.87	6.2	27.4	29.56	28.7	32.95	3.70000	3.00000	4.40000	6.66	5.05	9.15	80.25	56.5	104	Qasim and Reddy 1967; Astudillo et al. 2014; Deepa and Magudeswaran 2014; Jayachandran et al. 2018; Joseph et al. 2018
	I	India - Visakhapatnam Harbour, Andra Pradesh	17°41'°34"N	83°17'45"E	32.05	24.6	32.785	28.57	26.45	30.95	25.20000	2.85000	82.2000	5.99	4.71	8.72	1.70	0.60	3.20	Umamaheswara Rao and Mohanchand 1988; Satyanarayana et al. 1992; Devi 1996; Mukherjee et al. 2009
	I	Japan - Osaka Bay	34°39'00"N	135°819'50"E	26.13	20.33	31.86	19.97	8.9	29.13	0.04233	0.01370	0.04863	7.1	2.45	10.7	369.5	245	480	Tsujimoto et al. 2006; Yasuhara et al. 2007; Okuda et al. 2016
	I	Malaysia - Sungai Sekudai, Johor Bahru	1°29.4'N	103°42.2'E	138	121	155	28.6	26.5	30.7	-	-	-	11.1	8.9	13.3	117.99	70.76	153.45	Tan and Morton 2006; Naubi et al. 2016
	N	Mexico - Chetumal Bay, Quintana Roo	17°53'N	87°51'W	11.50	6.67	18.0	27.66	24.5	31	0.00250	0.00150	0.0055	5.31	3.315	6.12	-	-	-	Ortiz-Hernández and Sáenz-Morales 1999; Morales et al. 1996 apud Morales-Vela et al. 2000; Llanes-Baeza and González 2002; González et al. 2009; Herrera-Silveira et al. 2009
	N	Mexico - Laguna Bacalar. Quintana Roo	18.7°N	88.3°W	6.75	2.075	9.60	28.77	27.1	30.13	9.90000	9.50000	14.2000	103	-	-	-	-	-	Johnson et al. 2018; Zayas 2018; Valdez-Moreno et al. 2019
	N	Panama - Las Minas Bay	09°23'00"N	79°50'00"W	20.0	11.0	29.0	29.0	-	-	-	-	-	-	-	-	-	-	-	Dangremond 2013
	I	Panama - Panama Canal	8°57'44''N	79°15'36"W	12.50	5.0	20.0	26.75	24.45	29.0	0.00398	0.00384	0.0041	5.70	3.80	4.10	81	52	111	Vilardy and Polanía 2002; Dung and Siang 2011; Carrasco et al. 2019
	I	Philippines - Manila Bay	14°34'43.06"N	120°58'18.69"E	28.78	21.80	31.4	27.9	25.9	31.3	0.00510	0.0002	0.0255	5.15	0.79	7.25	25.08	13.54	36.99	Tamura et al. 2003; Jacinto et al. 2011; Vallejo et al. 2017; Trinidad et al. 2019; Vallejo et al. 2019

	I	Taiwan - Kaohsiung Port	22°39'39.0"N	120°21'08.1"E	22.10	9.10	35.0	27.8	22.9	32.6	0.01067	0.00025	0.0463	4.90	2.93	6.92	97.8	35.0	178	Chen et al. 2016; Minchin et al. 2016; Putri et al. 2020
	N	Venezuela - Neveri River, Barcelona, Anzoátegui	10°10'30"N	64°40'04"W	36.10	36.01	36.23	27.7	27.57	27.87	-	-	-	-	-	-	-	-	-	Lorenzoni et al. 2009
	I	Vietnam - Nha Trang	120°42'N	109°11’17”E	32.10	33.54	34.5	24.55	13.65	35.4	0.00676	0.00041	0.13351	6.70	6.4	7.04	400	-	-	Pavlov et al. 2004; Dung and Siang 2011; Quang et al. 2017
	I	Vietnam - Ba Tai Lake	10°12'27"N	104°36'11"E	17.50	15.0	20.0	37.50	35.0	40.0	-	-	-	-	-	-	-	-	-	Lutaenko et al. 2019
M. cf sallei	**I**	Brazil - Estuary of Rio Potengi	5°46'24''S	35°12'20"W	32.30	27.17	37.20	27.25	25.8	30.05	-	-	-	3.0	2.80	3.20	-	-	-	Chellappa et al. 2005; Farrapeira et al. 2010; Souza et al. 2010
	I	Brazil - Recife Port	08°04'03"-08°05'06"S	34°52'16"-34°53'08"W	17.90	10.09	25.65	22.25	17.25	27.75	0.04192	0.00449	0.14500	1.40	0.10	4.40	33.50	10	70	Farrapeira et al. 2007; Otsuka et al. 2018
M. trautwineana	**I**	Colombia - Cartagena Bay	10°23'58.99"N	75°30'51. 98"W	25.58	16.11	35.04	30.40	27.50	33.25	0.01700	0.00025	0.03430	4.51	1.79	7.25	46.85	33.55	60.15	Morton 1981; Morton 1989; Aldridge et al. 2008; Aguado 2010; Astudillo et al. 2017

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O capítulo 2 está restrito porque ainda não foi publicado em periódico.