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Influência da presença de árvores e arbustos não nativos na vegetação ripária sobre o funcionamento de riachos

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Tese submetida ao Programa de Pós-Graduação em Ecologia da Universidade Federal de Santa Catarina como requisito parcial para a obtenção do título de Doutor em Ecologia
Orientador: Prof. Mauricio Mello Petrucio, Dr.
Coorientador: Prof. Luiz Ubiratan Hepp, Dr.

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Lucas Eugenio Fontana

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O presente trabalho em nível de Doutorado foi avaliado e aprovado por banca examinadora composta pelos seguintes membros:

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Certificamos que esta é a **versão original e final** do trabalho de conclusão que foi julgado adequado para obtenção do título de Doutor em Ecologia.

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Prof. Mauricio Mello Petrucio, Dr.
Orientador

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seria muito mais difícil sem vocês!!

RESUMO

Estudos indicam que espécies não nativas (especialmente aquelas consideradas invasoras) podem influenciar diretamente a biodiversidade, estrutura e funcionamento dos ecossistemas além de, potencialmente, causar impactos econômicos. Em riachos florestados (≤ 3ª ordem), especificamente, estudos indicam que a presença de espécies arbóreas e/ou arbustivas não nativas e não nativas invasoras na vegetação ripária pode afetar o funcionamento destes ecossistemas. Neste contexto, esta Tese tem o objetivo de avaliar a influência da presença de espécies arbóreas e/ou arbustivas não nativas e não nativas invasoras na vegetação ripária sobre o funcionamento de riachos. Para isso, utilizei abordagens i) experimental, nos Capítulos I e II, onde avaliei a influência de *Hovenia dulcis* Thunb. (Rhamnaceae) - uma espécie arbórea não nativa e invasora com ampla ocorrência na região sul do Brasil - sobre o funcionamento de riachos subtropicais e; ii) teórica, no Capítulo III, onde realizei um estudo cienciométrico. No Capítulo I, avaliei o efeito da fenologia de *H. dulcis* sobre o padrão temporal anual e a quantidade de entrada de serrapilheira alóctone em riachos subtropicais com e sem a presença da espécie na vegetação ripária. Descobri que a presença de *H. dulcis* na vegetação ripária, quando em altas densidades, é capaz de alterar o padrão temporal e a quantidade sazonal de entrada de serrapilheira alóctone nos riachos, tornando a entrada de serrapilheira alóctone elevada nos meses de transição verão-outono. No Capítulo II, avaliei o efeito das folhas de *H. dulcis*, oferecidas individualmente e associadas a uma mistura de espécies nativas, na decomposição foliar e na comunidade de invertebrados aquáticos em três riachos subtropicais de Mata Atlântica. Descobri que as folhas de *H. dulcis* disponibilizadas individualmente tiveram a maior decomposição e as maiores densidades de invertebrados associados entre os tratamentos foliares. A associação com folhas de *H. dulcis*, entretanto, não modificou a decomposição foliar e a estrutura e composição da comunidade de invertebrados associados as folhas de espécies nativas. No Capítulo III, realizei uma análise cienciométrica para avaliar as principais tendências de publicações abordando os efeitos de espécies arbóreas e/ou arbustivas não nativas e não nativas invasoras sobre a decomposição foliar e as comunidades de microrganismos e invertebrados aquáticos em riachos. Entre 127 artigos publicados no período de 1981 a 2020, descobri que: i) o tema ‘decomposição’ foi abordado em ~64% dos artigos, ‘microrganismos’ em ~24% e ‘invertebrados’ em ~81%; ii) estes artigos foram publicados em 57 periódicos; iii) 29 países tiveram estudos desenvolvidos em seu território; iv) 89 espécies arbóreas e/ou arbustivas não nativas e não nativas invasoras foram avaliadas; v) 4 escalas espaciais de comparação (de folhas à riachos) foram registradas e vi) diferentes métricas foram utilizados para avaliar ‘decomposição’ (n=3), ‘microrganismos’ (n=9) e ‘invertebrados’ (n=18). Esta Tese pode contribuir, primeiro, com o conhecimento existente sobre os efeitos da presença de *H. dulcis* na vegetação ripária sobre a dinâmica de entrada de serrapilheira alóctone, a decomposição foliar e a estrutura e composição das comunidades de invertebrados aquáticos em riachos subtropicais e, segundo, num contexto ecológico mais amplo, com o conhecimento existente sobre à influência da presença de espécies não nativas (especialmente aquelas consideradas invasoras) na vegetação ripária de riachos sobre o funcionamento destes ecossistemas.

Palavras-Chave: Ecossistemas Lóticos. Invasões Biológicas. Espécies Exóticas. *Hovenia dulcis*. Dinâmica de Serrapilheira Alóctone. Decomposição Foliar. Invertebrados Aquáticos

ABSTRACT

Studies indicate that non-native species (especially those considered invasive) can directly influence the biodiversity, structure and functioning of ecosystems, besides to potentially causing economic impacts. In forested streams (≤ 3rd order), specifically, studies indicate that the presence of non-native and non-native invasive tree and/or shrub species in riparian vegetation can affect the functioning of these ecosystems. In this context, this Thesis objectives to evaluate the influence of the presence of non-native and non-native invasive tree and/or shrub species in riparian vegetation on the functioning of streams. For this, I used i) a experimental approach, in Chapters I and II, where I evaluated the influence of *Hovenia dulcis* Thunb. (Rhamnaceae) – a non-native and invasive tree species with wide occurrence in southern Brazil - on the functioning of subtropical streams and; ii) a theoretical approach, in Chapter III, where I carried out a scientometric study. In Chapter I, I evaluated the effect of the phenology of *H. dulcis* on the annual temporal pattern and the allochthonous litterfall input into subtropical streams with and without the presence of this species in riparian vegetation. I found that the presence of *H. dulcis* in riparian vegetation, when at high densities, is capable of change the temporal pattern and the seasonal allochthonous litterfall input into streams, making the input of allochthonous litterfall high in the summer-autumn transition months. In Chapter II, I evaluated the effect of *H. dulcis* leaves, offered individually and associated with a mixture of native species, on leaf decomposition and on the aquatic invertebrate community in three subtropical streams of the Atlantic Forest. I found that *H. dulcis* leaves individually available had the highest decomposition and the highest associated invertebrate densities among the foliar treatments. The association with *H. dulcis* leaves, however, did not change leaf decomposition and the structure and composition of the invertebrate community associated with the leaves of native species. In Chapter III, I performed a scientometric analysis to assess the main trends in publications approaching the effects of non-native and non-native invasive tree and/or shrub species on leaf decomposition and the communities of microorganisms and aquatic invertebrates in streams. Among 127 papers published from 1981 to 2020, I found that: i) the theme 'decomposition' was approached in ~64% of the papers, 'microorganisms' in ~24% and 'invertebrates' in ~81%; ii) these articles were published in 57 journals; iii) 29 countries had studies developed in their territory; iv) 89 non-native and non-native invasive tree and/or shrub species were evaluated; v) 4 spatial scales of comparison (from leaves to streams) were recorded and vi) different metrics were used to assess 'decomposition' (n=3), 'microorganisms' (n=9) and 'invertebrates' (n=18). This Thesis can contribute, first, with the existing knowledge about the effects of the presence of *H. dulcis* in riparian vegetation on the dynamics of allochthonous litterfall input, the leaf decomposition and the structure and composition of aquatic invertebrate communities in subtropical streams and, second, in a broad ecological context, with the existing knowledge about the influence of the presence of non-native species (especially those considered invasive) in the riparian vegetation of streams on the functioning of these ecosystems.

Keywords: Lotic Ecosystems. Biological Invasions. Exotic Species. Hovenia dulcis. Allochthonous Litter Dynamics. Leaf Decomposition. Aquatic Invertebrates.

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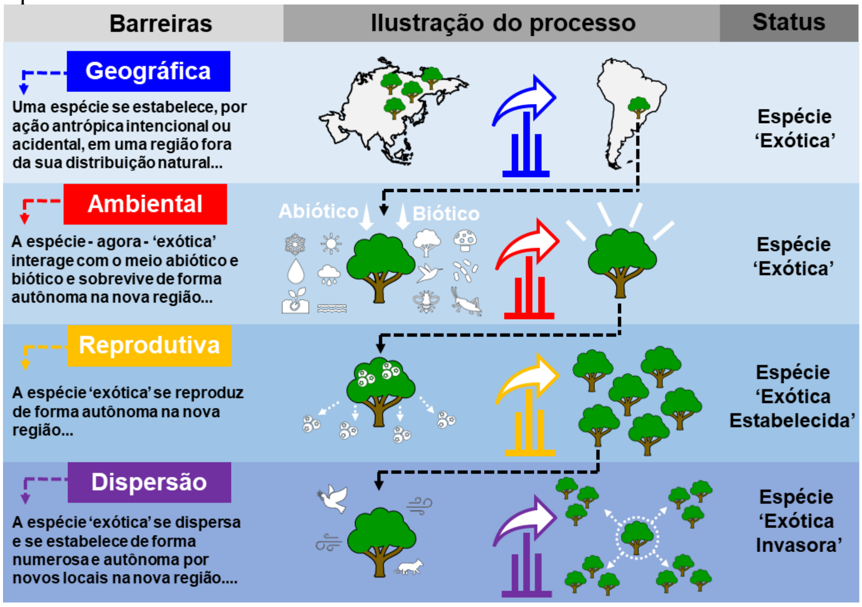
SUMÁRIO

1. INTRODUÇÃO GERAL

1.1 ESPÉCIES EXÓTICAS E INVASORAS

A globalização e os avanços tecnológicos permitiram o constante e rápido transporte de pessoas e cargas entre regiões geograficamente afastadas e possibilitaram, intencional ou acidentalmente, a dispersão de espécies para novos habitats que antes, em teoria, seriam de difícil alcance por mecanismos naturais de dispersão (MEYERSON e MOONEY, 2007). Conceitualmente, para que uma espécie – e neste caso, especificamente plantas – seja considerada ‘exótica’ ou ‘exótica invasora’ ela precisa ser capaz de transpor barreiras geográficas, ambientais, reprodutivas e de dispersão (RICHARDSON et al., 2000; BLACKBURN et al., 2011). Assim, uma espécie é considerada ‘exótica’ (também ‘não-nativa’ ou ‘introduzida’) quando é introduzida, por ação antrópica intencional ou acidental, em uma região fora da sua distribuição natural (i.e., transpõe a barreira geográfica) (veja RICHARDSON et al., 2000 e BLACKBURN et al., 2011). Uma espécie ‘exótica’ pode ser considerada ‘exótica invasora’ quando, uma vez introduzida numa nova região, é capaz de, de forma autônoma, sobreviver (i.e., transpõe a barreira ambiental), reproduzir (i.e., transpõe a barreira reprodutiva) e se dispersar pela nova região habitada (i.e., transpõe a barreira de dispersão) estabelecendo populações autossustentáveis em pontos distantes do ponto de introdução (veja RICHARDSON et al., 2000 e BLACKBURN et al., 2011) (Figura 1).

Figura 1 - Representação esquemática simplificada das principais barreiras superadas por espécies - em especial de plantas - introduzidas em regiões além de sua distribuição geográfica conhecida até serem consideradas ‘espécies exóticas’ e ‘espécies exóticas invasoras’. Baseado em Richardson et al. (2000) e Blackburn et al. (2011). Fonte: Elaborado pelo autor.



Barreiras	Ilustração do processo	Status
Geográfica Uma espécie se estabelece, por ação antrópica intencional ou acidental, em uma região fora da sua distribuição natural...		Espécie ‘Exótica’
Ambiental A espécie - agora - ‘exótica’ interage com o meio abiótico e biótico e sobrevive de forma autônoma na nova região...	Abiótico Biótico	Espécie ‘Exótica’
Reprodutiva A espécie ‘exótica’ se reproduz de forma autônoma na nova região...		Espécie ‘Exótica Estabelecida’
Dispersão A espécie ‘exótica’ se dispersa e se estabelece de forma numerosa e autônoma por novos locais na nova região....		Espécie ‘Exótica Invasora’

Estudos relatam impactos (aqui entendidos como qualquer mudança significativa - aumento ou diminuição - de uma propriedade ou processo ecossistêmico) (SIMBERLOFF et al., 2013) relacionados a espécies exóticas e exóticas invasoras sobre a biodiversidade (e.g., BELLARD et al., 2016; DOHERTY et al., 2016) e a estrutura e funcionamento dos ecossistemas (e.g., MIEHLS et al., 2009; EHRENFELD, 2010). Entretanto, a concepção de que a influência de espécies exóticas e exóticas invasoras sobre a biodiversidade e a estrutura e funcionamento dos ecossistemas é, necessariamente, 'ruim' não é correta (GOODENOUGH, 2010; SIMBERLOFF et al., 2013). Ao contrário, a avaliação dos impactos gerados por espécies exóticas é mais abrangente e complexa, considerando, tanto a esfera ecológica quanto a esfera socioeconômica e, para ambas, os impactos podem ser positivos, neutros ou negativos (KUMSCHICK et al., 2012) e, ainda, apresentar diferentes níveis (classes) de intensidade (BLACKBURN et al., 2014).

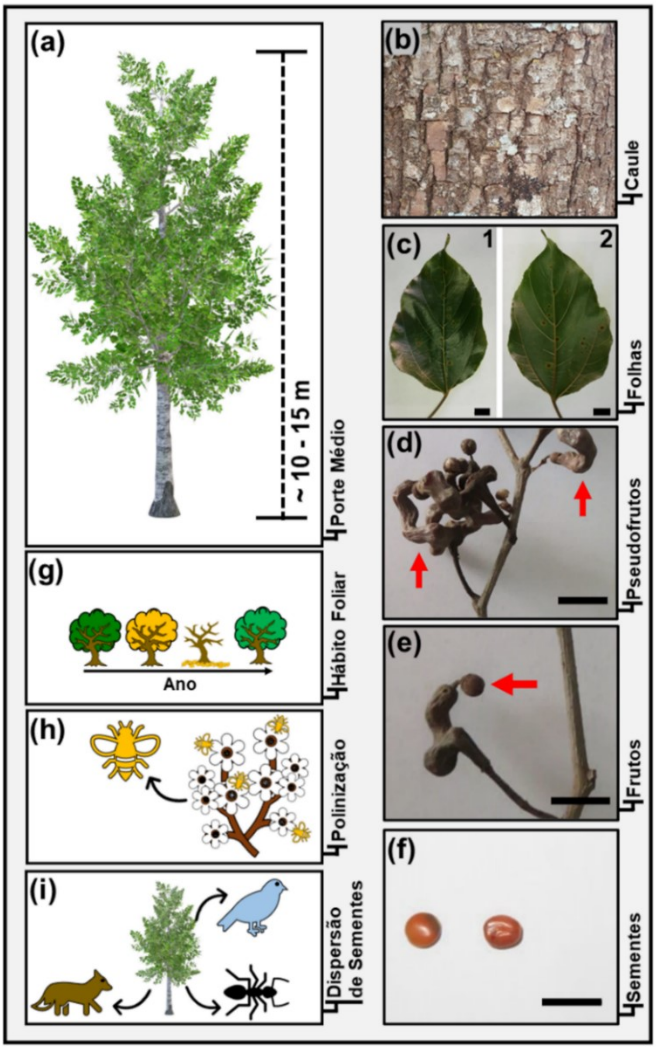
Estimativas indicam a existência de ~1700 espécies de plantas exóticas consideradas invasoras na América do Sul (ZENNI et al., 2022) e, dentre estas, pelo menos 75 delas são árvores e arbustos exóticos que apresentam potencial invasor (veja RICHARDSON e REJMÁNEK, 2011). De fato, os mecanismos exatos que levam ao processo de invasão biológica por plantas não são totalmente compreendidos embora, de forma geral, estejam relacionados isolada ou conjuntamente com i) as características da espécie invasora (e.g., fenologia estendida) (veja SMITH, 2013), ii) as características do ecossistema invadido (e.g., disponibilidade de recursos) (veja DAVIS et al., 2000), iii) as interações bióticas estabelecidas no novo ecossistema (e.g., dispersão de sementes pela fauna) (veja RICHARDSON e REJMÁNEK, 2011), iv) a ação antrópica, acidental ou intencional (e.g., cultivo para silvicultura) (veja RICHARDSON e REJMÁNEK, 2011) e, v) a pressão de propágulos (veja ESCHTRUTH e BATTLES, 2009 e também SIMBERLOFF, 2009). A invasão das zonas ripárias por árvores e arbustos exóticos invasores e os potenciais impactos ecossistêmicos sobre o funcionamento de riachos tem sido relatada em diversas regiões do mundo, incluindo Europa (e.g., MEDINA-VILLAR et al., 2015a; CASTRO-DÍEZ e ALONSO, 2017; SEENEY et al., 2019), América do Norte (e.g., ELLIS et al., 1998; MINEAU et al., 2012; McNEISH et al., 2015; WEAND et al., 2020; LITTLE et al., 2021) Oceania (e.g., McINERNEY et al., 2016; McINERNEY e RESS, 2017) e América do Sul (e.g., GANTES et al., 2011). No sul do Brasil, especificamente, a presença de *Hovenia dulcis* Thunb. (Rhamnaceae) – espécie arbórea, exótica e considerada invasora – na vegetação ripária de riachos também tem sido recentemente relatada (e.g., FONTANA et al., 2020).

1.1.1 *Hovenia dulcis*

Nativa da Ásia, *Hovenia dulcis* Thunb. (Rhamnaceae) é uma espécie arbórea exótica considerada invasora no Brasil (CARVALHO, 1994; ZENNI e ZILLER, 2011), com ocorrência relatada em países de diferentes continentes (BERGAMIN et al., 2022). Além disso, recentemente foi demonstrado que atualmente entre 7% e 8% da área terrestre global (incluindo *hotspots* de biodiversidade) apresentam condições climáticas adequadas para a ocorrência de *H. dulcis* (BERGAMIN et al., 2022). No Brasil, estudos indicam que *H. dulcis* apresenta ampla ocorrência em remanescentes florestais, especialmente na região sul do país, (DECHOUM et al., 2015a; LAZZARIN et al., 2015; PADILHA et al., 2015; SCHMIDT et al., 2020) o que está associado, de forma conjunta, com as características biológicas de *H. dulcis* e com as características dos ecossistemas onde *H. dulcis* foi introduzida. Primeiro, *H. dulcis* parece apresentar alguns eventos fenológicos (e.g., queda de folhas e amadurecimento dos pseudofrutos) temporalmente precoces e/ou distintos (veja REJMÁNEK, 2013; MILANI, 2013; MILANI, 2017 e FONTANA et al., 2020) da maioria das espécies nativas da região sul do Brasil (veja MARQUES et al., 2004; ATHAYDE et al., 2009 e FERRERA et al., 2017). Segundo, folhas e pseudofrutos de *H. dulcis* podem apresentar atividade alelopática negativa sobre a germinação e desenvolvimento de plântulas de outras espécies vegetais (WANDSCHEER et al., 2011; RIBEIRO et al., 2019; SCHMIDT et al., 2021). Terceiro, os pseudofrutos de *H. dulcis* são consumidos pela fauna (i.e., aves, mamíferos e formigas) (GIARETTA et al., 2013; DE LIMA et al., 2015; PODGAISKI et al., 2022). Quarto, *H. dulcis* é relativamente tolerante ao sombreamento, sendo capaz de germinar e estabelecer plântulas em remanescentes florestais de diferentes estágios sucessionais (DECHOUM et al., 2015b). Conjuntamente, estes aspectos podem proporcionar para a *H. dulcis* algum tipo de vantagem adaptativa (e.g., obtenção de radiação solar, de nutrientes do solo ou dispersão de sementes) em relação as espécies nativas, facilitando o seu estabelecimento em novos ecossistemas.

A espécie *H. dulcis* pode atingir, em média, 10 a 15 de altura (Figura 2a). O caule é, em geral, reto, cilíndrico, com copa aberta e arredondada e possuí casca acinzentada com aspecto fissurado (Figura 2b). As folhas têm filotaxia alternada, são simples, em geral lisas, ovadas, com margens serrilhadas, ápice agudo e base arredondada (5 a 15 cm de comprimento x 7 a 12 cm de largura) (Figura 2c). As flores são hermafroditas, pequenas, branco-esverdeadas e dispostas em inflorescências com panículas terminais e axilares de ~10 cm. Os ramos das flores se intumescem, passando a ter consistência suculenta e sabor agridoce formando, assim, os pseudofrutos (Figura 2d). Na extremidade dos pseudofrutos formam-se os frutos verdadeiros que são globosos e do tipo cápsula (até 7 mm de diâmetro) (Figura 2e). Cada fruto possui 2 a 4 sementes que são duras, achatadas e vermelho-alaranjadas (~ 4 a 8mm de diâmetro) (Figura 2f) - descrição botânica baseada em Carvalho (1994) e Lorenzi et al. (2018) (consulte detalhes nas referidas obras). Sob o aspecto ecológico, *H. dulcis* é uma espécie decídua com período de queda foliar ocorrendo, em geral, entre abril e agosto (CARVALHO, 1994) e, no sul do Brasil, aparentemente com picos de queda foliar entre março e maio (FONTANA et al., 2020) (Figura 2g). A polinização é entomofílica e, em geral, realizada por abelhas (Figura 2h) enquanto a dispersão da sementes é zoocórica e, em geral, realizada por aves, mamíferos (veja CARVALHO, 1994 e DE LIMA et al., 2015) e, possivelmente, também por formigas (veja PODGAISKI et al., 2022) (Figura 2i).

Figura 2 - Aspectos botânicos (a - f) e ecológicos (g - i) de *Hovenia dulcis*. (a) ilustração do porte médio; (b) aspecto fissurado do caule; (c) folhas [c1 – face adaxial; c2 – face abaxial]; (d) pseudofrutos; (e) frutos; (f) sementes; (g) ilustração do hábito foliar decidual; (h) ilustração da principal forma de polinização das flores e (i) ilustração da principal forma de dispersão de sementes. Em (d) e (e) as setas destacam a estrutura indicada. Em (c), (d), (e) e (f) a barra de escala é de 10 mm. Baseado em Carvalho (1994) e Lorenzi et al. (2018). Fonte: Elaborado pelo autor.



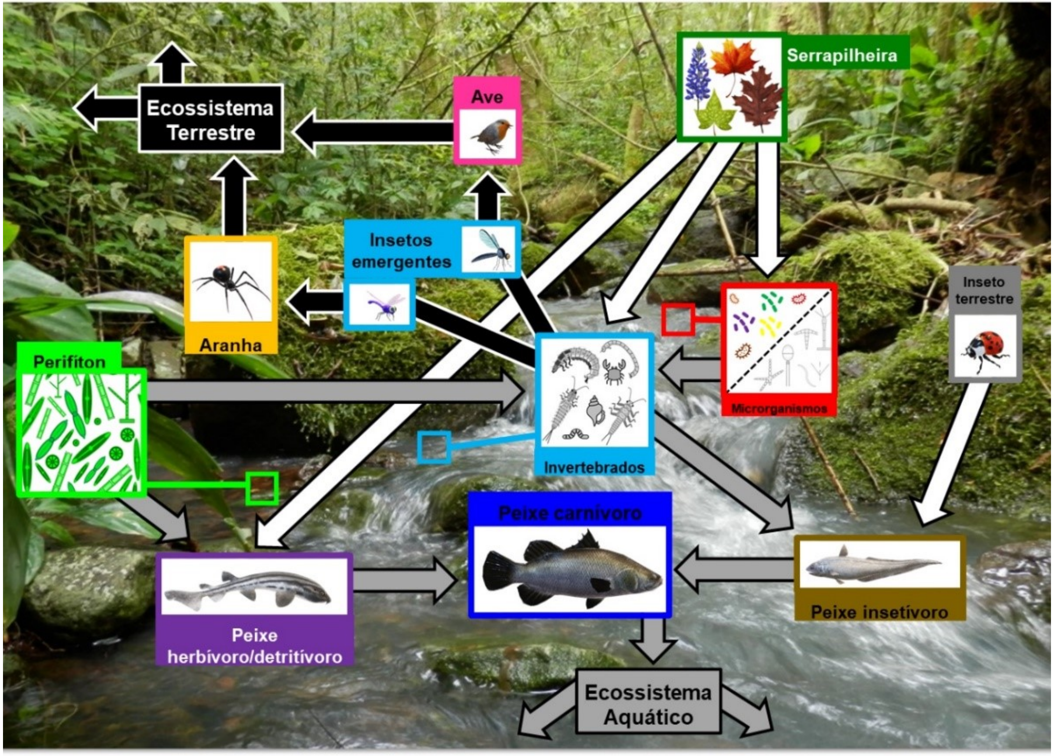
1.2 FUNCIONAMENTO DE ECOSSISTEMAS

O conceito de ecossistema como um sistema integrado e composto tanto pelos organismos quanto pelos fatores físicos (ambiente) foi proposto por Arthur George Tansley no artigo *The Use and Abuse of Vegetational Concepts and Terms* de 1935 (TANSLEY, 1935). Assim, num contexto amplo, 'ecossistema', pode ser entendido como o conjunto de todos os organismos que habitam um determinado local interagindo direta ou indiretamente entre si e com o meio abiótico (TANSLEY, 1935; PICKETT e CADENASSO, 2002; RICKLEFS e RELYEA, 2016). Em muitos casos, entretanto, os limites espaciais dos ecossistemas são de difícil determinação e, muitas vezes, a determinação de um ecossistema (e.g., um riacho, uma floresta) é uma convenção prática considerando o contexto ecológico avaliado. Assim, ao invés de sistemas isolados, os ecossistemas são sistemas abertos e conectados uns aos outros por fluxos de energia, matéria e organismos através de seus limites espaciais constituindo, dessa forma, meta-ecossistemas (veja LOREAU et al., 2003). Assim, o 'funcionamento dos ecossistemas', é determinado pelo fluxo de matéria e energia que ocorre através de processos ecológicos envolvendo seus componentes bióticos (e.g., organismos) e abióticos (e.g., atmosfera, hidrosfera, litosfera) (DÍAZ e CABIDO, 2001; GESSNER et al., 2010). Neste contexto, o funcionamento dos ecossistemas é caracterizado por diferentes processos ecológicos chave, incluindo a produção primária, a decomposição da matéria orgânica e as relações tróficas entre os organismos (DÍAZ e CABIDO, 2001) que são sustentados, total ou parcialmente, pela biodiversidade (e.g., taxonômica e funcional) (HOOPER et al., 2005; GESSNER et al., 2010; CARDINALE et al., 2011; TILMAN et al., 2014; BARNES et al., 2018).

1.2.1 Funcionamento de riachos e a interface riacho - zona ripária

Os ecossistemas lóticos de pequeno porte (i.e., riachos florestados de cabeceira ≥ 3^a ordem – daqui em diante apenas 'riachos') e a zona ripária constituem um meta-ecossistema (veja LOREAU et al., 2003) com uma intensa e mútua troca de energia na forma de matéria orgânica e organismos (BAXTER et al., 2005; TOLKKINEN et al., 2020). No sentido zona ripária – riacho, ocorre uma elevada entrada de energia através de i) invertebrados terrestres que podem ser recurso nutricional para peixes (e.g., MAZZONI et al., 2010; SILVA et al., 2012) e ii) serrapilheira (i.e., folhas, galhos, flores, frutos e sementes) oriunda da vegetação ripária adjacente (e.g., FRANÇA et al., 2009; GONÇALVES e CALLISTO, 2013) que constitui uma importante fonte nutricional para microrganismos e invertebrados aquáticos (VANNOTE et al., 1980) (e veja também BAXTER et al., 2005; TOLKKINEN et al., 2020). Por outro lado, no sentido riacho - zona ripária, insetos emergentes adultos se deslocam em direção à zona ripária onde são recurso nutricional para artrópodes (e.g., aranhas) e vertebrados (e.g., aves) terrestres (BAXTER et al., 2005; BARTRONS et al., 2013) (Figura 3).

Figura 3 - Representação esquemática simplificada do fluxo de energia bidirecional entre os riachos e a zona ripária. As setas indicam o fluxo de energia no sentido zona ripária – riacho (cor branca), dentro do riacho (cor cinza) e no sentido riacho – zona ripária (cor preta). Baseado em Vannote et al. (1980), Baxter et al. (2005) e Cummins (2016). Fonte: Elaborado pelo autor.



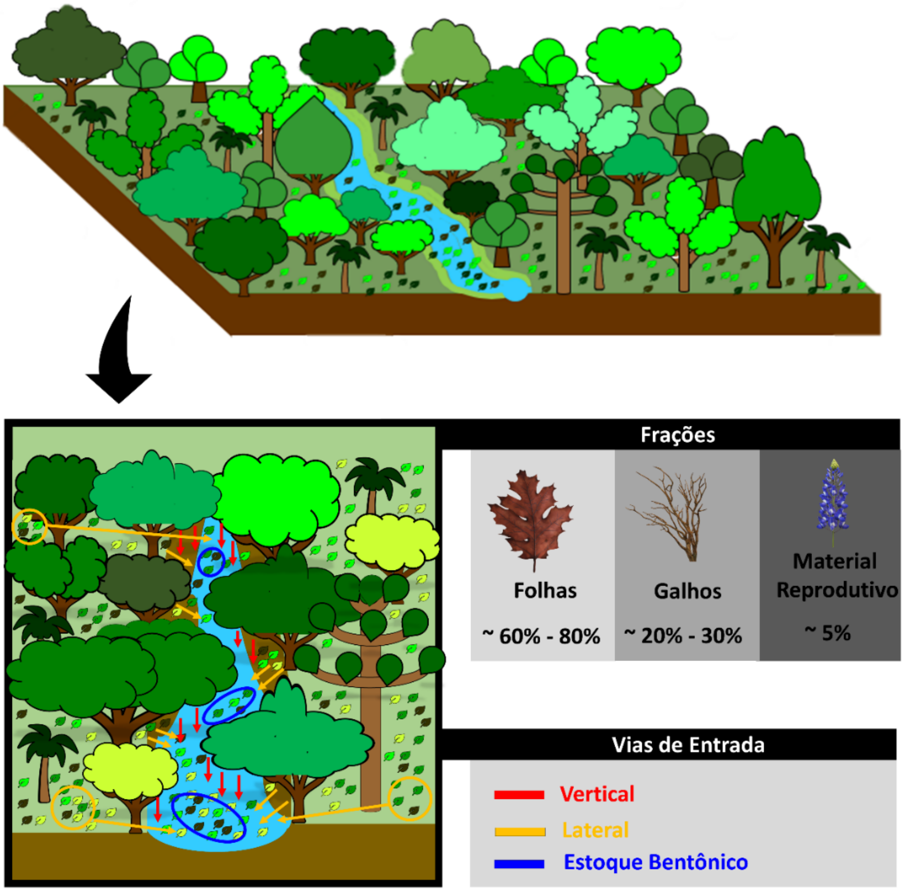
Neste contexto, a vegetação ripária desempenha papel fundamental no funcionamento de riachos pois auxilia na manutenção da qualidade da água (e.g., filtragem de substâncias potencialmente poluentes, prevenção da erosão e entrada de sedimentos e manutenção da estabilidade das margens) (TUNDISI e TUNDISI, 2010; TOLKKINEN et al., 2020) e afeta o metabolismo destes ecossistemas (i.e., limita a produção primária autóctone e, ao mesmo tempo, fornece recursos vegetais alóctones) (VANNOTE et al., 1980). O Conceito do Contínuo Fluvial (*River Continuum Concept*) proposto por Vannote et al. (1980) sugere que os ecossistemas lóticos são estruturados através de um gradiente longitudinal de matéria e energia que organizam, estrutural e funcionalmente, as comunidades aquáticas desde a nascente até a foz. Segundo esta teoria, os ecossistemas lóticos de pequeno porte (i.e., riachos $\leq 3^a$ ordem) são sistemas essencialmente heterotróficos que são sustentados principalmente pelas fontes alóctones (VANNOTE et al., 1980). Num contexto ecológico, o funcionamento de riachos é, portanto, influenciado fortemente pela vegetação ripária, e está vinculado a produtividade primária autóctone e alóctone (fontes de energia) (VANNOTE et al., 1980; WEBSTER e BENFIELD, 1986), a decomposição da serrapilheira alóctone (ciclagem de nutrientes) (CHAUVET et al., 2016) e relações tróficas (fluxo de matéria e energia) entre as comunidades aquáticas (e.g., invertebrados, microrganismos e peixes) (VANNOTE et al., 1980; WALLACE e WEBSTER, 1996; GRAÇA et al., 2016).

1.2.1.1 Produção Primária Autóctone e Alóctone

A produção primária autóctone (i.e., originada dentro do ecossistema aquático) e alóctone (i.e., originada fora do ecossistema aquático) são processos chave no funcionamento de riachos pois constituem as principais formas de entrada de energia nestes ecossistemas (VANOTTE et al., 1980; WEBSTER e BENFIELD, 1986). Nos riachos, a produção autóctone é representada principalmente pelo perifíton (VANNOTE et al., 1980) enquanto a produção alóctone é constituída essencialmente pela serrapilheira oriunda da vegetação ripária (VANOTTE et al., 1980; ABELHO, 2001). O perifíton (biofilme) é composto por algas unicelulares, sobretudo diatomáceas, microrganismos (e.g., bactérias e fungos) e protozoários que, associados, se aderem a substratos submersos (e.g., rochas, troncos e folhas) (VANNOTE et al., 1980; GARI e CORIGLIANO, 2004; BESEMER, 2015). Por outro lado, a serrapilheira alóctone é composta por folhas, galhos e partes reprodutivas oriundas da vegetação ripária adjacente aos riachos (FRANÇA et al., 2009; GONÇALVES et al., 2014a; LISBOA et al., 2015). Nos riachos, a produção primária autóctone é limitada pelo sombreamento e, por isso, é comparativamente menor que a produção primária alóctone (e.g., ABELHO e GRAÇA, 1998). Apesar disso, estudos recentes utilizando isótopos estáveis confirmam a importância das fontes autóctones nas cadeias tróficas aquáticas (e.g., NERES-LIMA et al., 2016; ERDOZAIN et al., 2019). A produção primária alóctone, neste contexto, se torna uma importante fonte de energia dos riachos, sendo essencial para a manutenção dos processos ecossistêmicos (e.g., ciclagem de nutrientes) e das comunidades aquáticas (e.g., invertebrados) (VANNOTE et al., 1980; ABELHO, 2001).

A serrapilheira alóctone (daqui em diante apenas ‘serrapilheira’) pode entrar nos riachos por diferentes vias: vertical (direta), lateral/terrestre (indireta) e estoque bentônico (GONÇALVES e CALLISTO, 2013; GONÇALVES et al., 2014a; HEPP et al., 2020) (Figura 4). No aporte vertical, a serrapilheira oriunda da vegetação ripária localizada acima do leito se desprende das árvores e entra de forma direta nos riachos; no aporte lateral/terrestre, a serrapilheira fica estocada, inicialmente, nas margens adjacentes e, posteriormente, por ação da chuva e vento entra nos riachos; finalmente, o estoque bentônico é caracterizado pela serrapilheira transportada pelo fluxo da água ao longo do curso do riacho e que fica depositada em zonas de remanso, ou estocada em rochas ou troncos presentes no leito do riacho (GONÇALVES e CALLISTO, 2013; GONÇALVES et al., 2014a; HEPP et al., 2020) (Figura 4). A importância relativa das diferentes vias de entrada no aporte total de serrapilheira varia dependendo das características morfológicas (e.g., declividade das margens) e limnológicas (e.g., velocidade de correnteza) dos riachos (FRANÇA et al., 2009; GONÇALVES et al., 2014a; LISBOA et al., 2015). A serrapilheira é composta por diferentes frações vegetais: folhas (em média 70% do aporte total de serrapilheira) galhos (em média 25%) e partes reprodutivas (e.g., flores, frutos e sementes) (em média 5%) (FRANÇA et al., 2009; GONÇALVES e CALLISTO, 2013; GONÇALVES et al., 2014a; LISBOA et al., 2015; REZENDE et al., 2017) (Figura 4).

Figura 4 - Representação esquemática simplificada das vias de entrada (vertical, lateral/terrestre e estoque bentônico) e da contribuição relativa (%) das frações vegetais (folhas, galhos e partes reprodutivas) de serrapilheira alóctone em riachos. Baseado em Fontana et al. (2020) e Hepp et al. (2020). Fonte: Elaborado pelo autor.

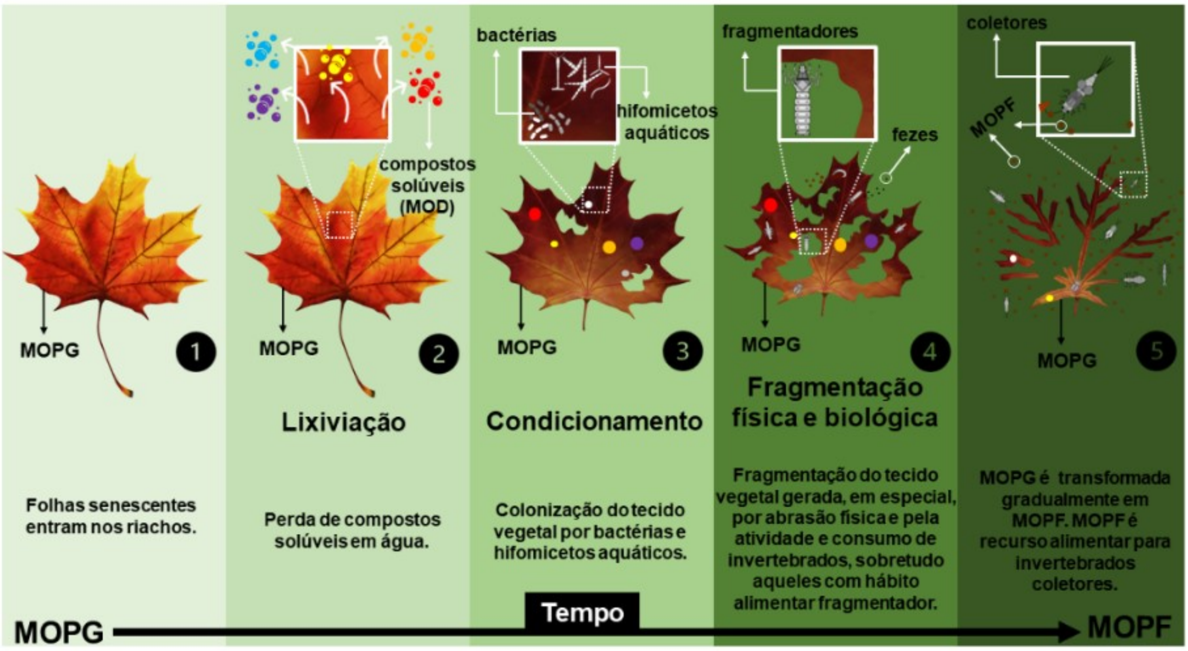


A dinâmica de serrapilheira (i.e., quantidade, diversidade e padrão temporal anual) varia entre riachos, o que está diretamente relacionado a estrutura (SOUZA et al., 2019) e composição da vegetação ripária (GONÇALVES et al., 2014a; REZENDE et al., 2017) e aos padrões climáticos regionais (e.g., regime de precipitação e temperatura) (ZHANG et al., 2014; TONIN et al., 2017). Neste contexto, a dinâmica de serrapilheira apresenta características e padrões anuais geralmente distintos em riachos localizados zonas climáticas temperadas, subtropicais e tropicais (veja HEPP et al., 2020). Em regiões de clima subtropical (i.e., região sul do Brasil) o aporte de serrapilheira apresenta condições intermediárias entre regiões temperadas e tropicais. No sul do Brasil, as estações do ano são bem definidas com invernos frios e verões quentes (temperatura média anual entre 16°C e 18°C) e regime pluviométrico relativamente constante durante o ano (entre 1600 e 2200 mm.ano^{-1}) (ALVARES et al., 2013). Por isso, o aporte de serrapilheira é influenciado tanto pela temperatura quanto pela precipitação (HEPP et al., 2020). Estudos realizados na região indicam que a produção de serrapilheira em remanescentes florestais nativos é ligeiramente maior nos meses de inverno e primavera, o que está associado à fenologia foliar de espécies arbóreas nativas (e.g., queda de folhas na transição outono-inverno e substituição de folhas senescentes na primavera) e, eventualmente, com picos mensais de precipitação (BRUN et al., 2001; KÖNIG et al., 2002; TURCHETTO e FORTES, 2014; FONTANA et al., 2020).

1.2.1.2 Decomposição da Serrapilheira Vegetal

A decomposição da serrapilheira é fundamental ao funcionamento dos riachos pois constituí a principal forma de ciclagem de nutrientes nestes ecossistemas. Nos riachos, a serrapilheira oriunda da vegetação ripária é decomposta e transformada de matéria orgânica particulada grossa (MOPG; maior que 1mm) em matéria orgânica particulada fina (MOPF; entre 1mm e 0,45 μm) e matéria orgânica dissolvida (MOD; menor que 0,45 μm) (ABELHO, 2001; TANK et al., 2010; MARKS, 2019). Folhas representam a maior parte da biomassa da serrapilheira (FRANÇA et al., 2009; GONÇALVES et al., 2014a; REZENDE et al., 2017) e, por isso, o processo de decomposição em riachos é amplamente estudado considerando esta fração da serrapilheira. Durante a decomposição nos riachos, as folhas são transformadas e assimiladas na rede trófica aquática por processos abióticos (e.g., lixiviação e fragmentação física) e bióticos (e.g., condicionamento microbiano e fragmentação biológica) (ABELHO, 2001; MARKS, 2019) (Figura 5). Estes processos, embora descritos individualmente, ocorrem de forma simultânea na maior parte do tempo (veja HIEBER e GESSNER, 2002; GESSNER et al., 2010; GONÇALVES et al., 2014b).

Figura 5 - Representação esquemática simplificada da decomposição foliar em riachos, considerando as etapas de (1) entrada de folhas senescentes no riacho, (2) lixiviação, (3) condicionamento, (4) fragmentação física e biológica e (5) transformação da MOPG em MOPF. Baseado em Marks (2019). Fonte: Elaborado pelo autor.



O processo de lixiviação ocorre, em maior escala, durante as primeiras 48 horas após a entrada das folhas nos riachos, onde até 30% da biomassa foliar é perdida (dependendo das características foliares de cada espécie) (BÄRLOCHER, 2005). Na lixiviação, compostos solúveis em água (e.g., carboidratos, fenóis e aminoácidos) são dissolvidos e, no riacho, passam a constituir parte da MOD (e.g., carbono orgânico e inorgânico dissolvidos) (BÄRLOCHER, 2005; SILVA et al., 2018, MARKS, 2019) (Figura 5). O processo de condicionamento microbiano é caracterizado pela colonização das folhas por microrganismos (e.g., bactérias e fungos) (HIEBER e GESSNER, 2002; MARKS, 2019). Os hifomicetos aquáticos são considerados os principais microrganismos decompositores de ambientes aquáticos (ABELHO, 2001; HIEBER e GESSNER, 2002) e, ao secretarem enzimas sobre o tecido vegetal, decompõem as folhas parcialmente e contribuem com a sua fragmentação (ABELHO, 2001; MARKS, 2019) (Figura 5). O processo de fragmentação consiste, basicamente, na quebra do tecido vegetal em partes gradualmente menores através da abrasão física e atividade de invertebrados (ABELHO, 2001; GONÇALVES et al., 2014b). Na abrasão física, as folhas são fragmentadas por meio da força da correnteza e/ou choque contra rochas e troncos submersos ao serem carregadas pelo fluxo de água (HEARD et al., 1999; ABELHO, 2001). Na fragmentação biológica, as folhas são fragmentadas pelo consumo direto de invertebrados fragmentadores (GRAÇA, 2001; GONÇALVES et al., 2014b; MARKS, 2019). A fragmentação física e, especialmente, biológica contribuem para a transformação da MOPG em MOPF (ABELHO, 2001; MARKS, 2019) que é o principal recurso nutricional de uma parte significativa dos invertebrados aquáticos de riachos (i.e., coletores) (GRAÇA, 2001; CUMMINS et al., 2005; RAMIRÉZ e GUTIÉRREZ-FONSECA, 2014) (Figura 5).

A velocidade da decomposição foliar é influenciada por fatores intrínsecos, relacionados às características estruturais e químicas das folhas (veja FOUR et al., 2019; ZHANG et al., 2019; HEPP et al., 2020), e fatores extrínsecos (veja TANK et al., 2010) relacionados às características dos riachos (e.g., características físicas e químicas da água [veja MEDEIROS et al., 2015; BIASI et al., 2017] e usos da terra nas áreas adjacentes aos riachos [veja ABELHO e GRAÇA, 1996; FERREIRA et al., 2015; HEPP et al., 2016; HEPP et al., 2020; LEMES-SILVA et al., 2020]). As características estruturais (e.g., dureza) e químicas (e.g., concentrações de nutrientes, de compostos secundários e de compostos estruturais) das folhas desempenham, de fato, um papel determinante na decomposição foliar em riachos pois afetam diretamente a atividade de microrganismos e invertebrados fragmentadores (GRAÇA, 2001; GONÇALVES et al., 2014b; ZHANG et al., 2019; HEPP et al., 2020). A dureza das folhas dificulta a quebra do tecido vegetal (GRAÇA, 2001) e, assim, folhas mais duras podem ser menos palatáveis para os invertebrados fragmentadores (e.g., GRAÇA e CRESSA, 2010). As concentrações de nutrientes como nitrogênio e fósforo tornam as folhas mais nutritivas e, dessa forma, folhas com maiores concentrações destes nutrientes podem ser mais atrativas para os invertebrados fragmentadores (e.g., RINCÓN e MARTÍNEZ, 2006). Os compostos secundários (e.g., polifenóis) produzidos pelas plantas auxiliam no combate a herbivoria das folhas (veja SINGH et al., 2021) e, desse modo, folhas com maiores concentrações de polifenóis podem ser menos palatáveis para os invertebrados fragmentadores (e.g., RINCÓN e MARTÍNEZ, 2006). Finalmente, a concentração de compostos estruturais (e.g., lignina e celulose) estão diretamente associadas a dureza das folhas (GESSNER, 2005) e, com isso, folhas com elevadas concentrações de lignina e celulose tendem a ser mais duras. Neste contexto, estudos sugerem que, comparativamente, folhas com características (mas não necessariamente todas estas) como menor dureza (e.g., ARDÓN e PRINGLE, 2008; LI et al., 2009), maiores concentrações de nitrogênio e fósforo (e.g., LECERF e CHAUVET, 2008; FERREIRA et al., 2012) e menores concentrações de polifenóis (e.g., FERREIRA et al., 2012), lignina e celulose (e.g., ARDÓN e

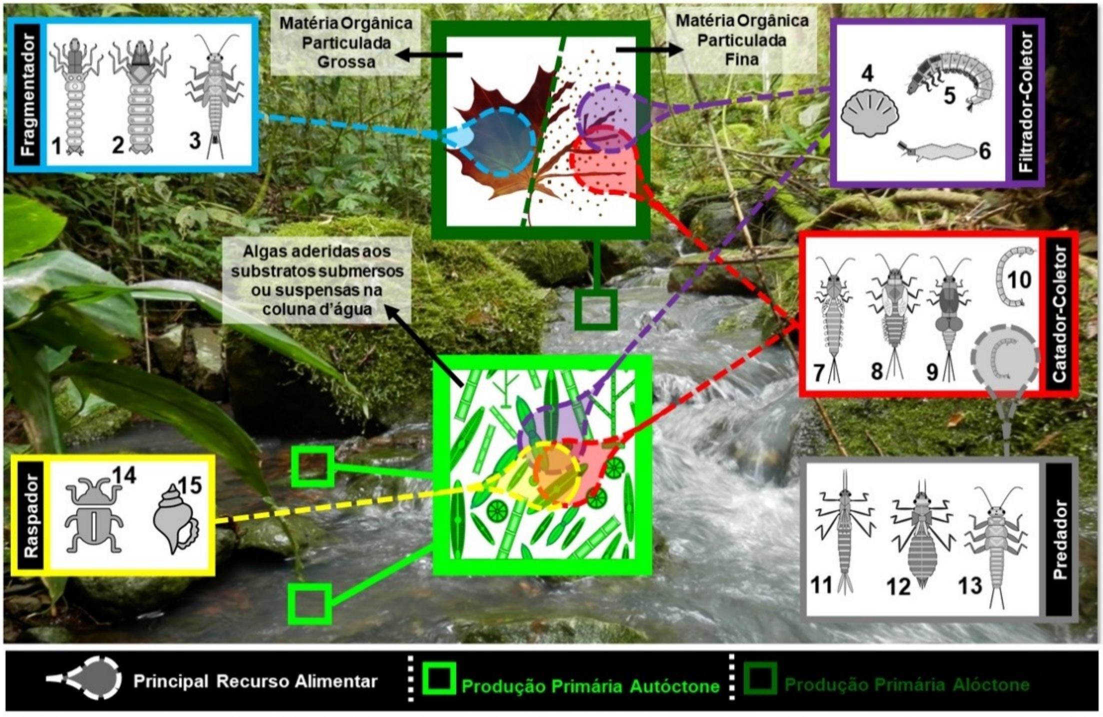
PRINGLE, 2008; ARDÓN et al., 2009) tendem a se decompor mais rápido (GONÇALVES et al., 2014b; HEPP et al., 2020).

1.2.1.3 Comunidades Aquáticas – Invertebrados

Em riachos, as comunidades aquáticas são diversificadas e compostas, principalmente, por microrganismos, algas, invertebrados e peixes (e.g., VANNOTE et al., 1980; MEYER et al., 2007). A comunidade de invertebrados aquáticos (também chamados de ‘macroinvertebrados’) é especialmente diversificada, sendo constituída por insetos (i.e., principalmente em fases imaturas, mas também alguns em fase adulta), anelídeos, moluscos e crustáceos (veja CUMMINS et al., 2005 e MEYER et al., 2007). Os invertebrados desempenham um papel fundamental no funcionamento de riachos e no fluxo de energia das redes tróficas aquáticas pois consomem MOPG, MOPF e perifíton (CUMMINS et al., 2005; RAMÍREZ e GUTIÉRREZ-FONSECA, 2014) e são recurso alimentar para peixes (MAZZONI et al., 2010; SILVA et al., 2012). Além disso, participam também das redes tróficas terrestres, especialmente após a emergência para a fase adulta, quando se deslocam em direção a vegetação ripária e são recurso alimentar para invertebrados (e.g., aranhas) e vertebrados (e.g., aves) (BAXTER et al., 2005; BARTRONS et al., 2013).

Nas últimas décadas, estudos que avaliam a diversidade de invertebrados em riachos passaram a considerar, além da diversidade taxonômica, a diversidade funcional destes organismos (veja SCHMERA et al., 2017). Essa abordagem tornou-se necessária à medida que uma avaliação puramente taxonômica, embora importante, muitas vezes se mostrava insuficiente para compreender alguns processos ecológicos e a inserção dos organismos, numa perspectiva funcional, no funcionamento dos riachos (veja SCHEMERA et al., 2017). Os invertebrados apresentam diferentes atributos funcionais e, para cada um deles, diferentes modalidades (veja uma lista detalhada em USSEGLIO-POLATERA et al., 2000). O hábito alimentar é, por exemplo, um atributo funcional comumente mensurado para os invertebrados em riachos onde, dentro deste atributo, são definidas diferentes modalidades (chamados de ‘grupos tróficos funcionais’). Os invertebrados apresentam hábitos alimentares preferenciais (veja uma lista detalhada em CUMMINS et al., 2005 e RAMIRÉZ e GUTIÉRREZ-FONSECA et al., 2014) embora estudos indiquem que, em alguns casos, os invertebrados também podem apresentar hábitos alimentares facultativos, consumindo itens alimentares pouco comuns ao seu hábito alimentar (veja GUZMÁN-SOTO e TAMARÍS-TURIZO, 2014; NERES-LIMA et al., 2016). Apesar disso, a utilização da classificação de grupos tróficos funcionais tradicionalmente atribuída aos táxons que compõem a comunidade de invertebrados aquáticos continua sendo usada amplamente. De forma geral, se considera que invertebrados ‘fragmentadores’ consomem diretamente o tecido vegetal das folhas (MOPG); ‘coletores-filtradores’ e ‘coletores-catadores’ consomem, respectivamente, biofilme, algas e/ou MOPF suspensas na coluna de água ou acumulada em áreas de remanso; ‘raspadores’ consomem algas perifíticas aderidas ao substrato e ‘predadores’ consomem larvas de organismos aquáticos (CUMMINS et al., 2005; RAMIRÉZ e GUTIÉRREZ-FONSECA et al., 2014) (Figura 6).

Figura 6 - Representação esquemática simplificada dos recursos alimentares utilizados preferencialmente pelos diferentes grupos tróficos funcionais de invertebrados aquáticos em riachos. Números indicam exemplos de táxons (e.g., famílias e classes) de cada grupo trófico funcional. Fragmentador: Leptoceridae (1); Calamoceratidae (2); Gripopterygidae (3). Filtrador-Coletor: Bivalve (4); Hydropsychidae (5); Siimulidae (6). Catador-Coletor: Baetidae (7); Leptophlebiidae (8); Caenidae (9); Chironomidae (10); Predador: Calopterygidae (11); Libellulidae (12); Perlidae (13). Raspador: Elmidae (adulto) (14); Gastropoda (15). Baseado em Cummins et al. (2005) e Marks (2019). Fonte: Elaborado pelo autor.



1.3 ESPÉCIES ARBÓREAS E ARBUSTIVAS NÃO-NATIVAS E O FUNCIONAMENTO DE RIACHOS

A ocorrência de espécies exóticas e exóticas invasoras (e.g., algas, moluscos e peixes) e os seus impactos ecológicos sobre os ecossistemas aquáticos (e.g., rios, lagos e estuários) são amplamente relatados (e.g., STRAYER et al., 2010; GALLARDO et al., 2016). Nesta secção, discutiremos os efeitos da presença de árvores e arbustos exóticos e exóticos invasores (aqui consideradas conjuntamente e, daqui em diante, chamadas apenas de ‘espécies não-nativas’) na vegetação ripária sobre o funcionamento de riachos com enfoque sobre a dinâmica de serrapilheira, decomposição foliar e comunidades aquáticas, em especial invertebrados. De fato, os efeitos da presença de espécies não-nativas na vegetação ripária sobre o funcionamento de riachos são bastante diversificados (e.g., positivos, neutros e negativos) (CASTRO-DÍEZ e ALONSO, 2017) e parecem estar diretamente associados a dois fatores: i) características foliares de palatabilidade e teores nutricionais das folhas (determinada pelas características intrínsecas de cada espécie) e; ii) diversidade e disponibilidade de recursos foliares ofertados nos riachos (determinado pela representatividade da espécie não-nativa na vegetação ripária).

1.3.1. Produção Primária Alóctone e Dinâmica de Serrapilheira

A substituição da vegetação ripária nativa por espécies não-nativas (e.g., implantação de monoculturas arbóreas ou invasão biológica de espécies arbóreas e/ou arbustivas) pode alterar a quantidade (ELLIS et al., 1998; MOLINERO e POZO, 2004; McNEISH et al., 2015), a diversidade/composição (e.g., GRAÇA et al., 2002) ou padrão temporal (ABELHO e GRAÇA, 1996; FONTANA et al., 2020; WEAND et al., 2020) de serrapilheira que entra nos riachos ou se acumula na zona ripária.

As implicações ecológicas destas alterações, entretanto, nem sempre são claras e algumas alterações podem ser mais significativas para o funcionamento dos riachos que outras. Primeiro, a diminuição da quantidade de serrapilheira que entra ou se acumula na zona ripária de riachos (e.g., ELLIS et al., 1998; MOLINERO e POZO, 2004; McNEISH et al., 2015) pode representar uma diminuição na disponibilidade dos recursos alóctones (e.g., folhas) que sustentam a ciclagem de nutrientes e as comunidades aquáticas destes ecossistemas. Entretanto, é necessário considerar que, em alguns casos, a produção total de serrapilheira em zonas ripárias compostas unicamente por espécies nativas (veja BAMBI et al., 2017) varia em termos de quantidade ao longo dos anos. De forma semelhante, a produção de folhas da serrapilheira em zonas ripárias compostas por espécies nativas e não-nativas (veja MEDINA-VILLAR et al., 2015a) também varia quantitativamente de um ano para outro. Assim, a detecção de uma diminuição da quantidade de serrapilheira que entra ou se acumula na zona ripária de riachos com a presença de espécies não-nativas em um intervalo anual não significa, necessariamente, um padrão constante em um intervalo de tempo maior. Segundo, a diminuição da diversidade (especialmente), e a alteração da composição da serrapilheira que entra nos riachos, de fato, pode diminuir a diversidade de características foliares dos recursos alóctones disponíveis e, consequentemente, afetar a decomposição foliar e as comunidades aquáticas (e.g., microrganismos e invertebrados) sustentadas por elas (e.g., FERREIRA et al., 2016a). Isso parece especialmente claro, por exemplo, em riachos inseridos em áreas de monoculturas arbóreas (e.g., ABELHO e GRAÇA, 1996; FERREIRA et al., 2015), entretanto, mesmo nestes casos, os efeitos (e.g., sobre os invertebrados aquáticos) podem ser mais evidentes em uma região, mas não em outra (e.g., FERREIRA et al., 2015). Terceiro, alterações temporais de entrada de serrapilheira parecem estar diretamente relacionadas as características fenológicas (e.g., queda de folhas) das espécies não-nativas presentes na vegetação ripária (e.g., ABELHO e GRAÇA, 1996; FONTANA et al., 2020). Em riachos onde a entrada de serrapilheira ocorre, naturalmente, de forma sazonal (e.g., na Floresta Decidual com grandes aportes nos meses de outono) o ciclo de vida de alguns invertebrados pode estar sincronizado com o período de maior disponibilidade de recursos e, assim, mudanças temporais de entrada de serrapilheira (associadas a mudança na diversidade e qualidade nutricional das folhas) podem afetar este processo (ABELHO e GRAÇA, 1996). Por outro lado, em riachos onde o aporte de serrapilheira é relativamente constante ao longo do ano (e.g., região subtropical do Brasil), o ciclo de vida de insetos aquáticos provavelmente não está vinculado a disponibilidade de recursos alóctones.

1.3.2. Decomposição Foliar e Comunidade de Invertebrados Aquáticos

Considerando a importância dos invertebrados no processo de decomposição foliar e importância das folhas na estruturação da comunidade de invertebrados em riachos, a seguir discutiremos os efeitos da presença de árvores e arbustos não-nativos na vegetação ripária sobre a decomposição foliar e a estrutura e composição da comunidade de invertebrados conjuntamente. De fato, os resultados são diversificados assim como as abordagens experimentais usualmente utilizadas.

Numa das abordagens mais utilizadas, a decomposição foliar e a comunidade de invertebrados associados são avaliadas com base na comparação de folhas de espécies nativas e não-nativas incubadas nos mesmos riachos. Nesta abordagem, os estudos simulam os efeitos da presença de espécies não-nativas na vegetação ripária ao compararem a decomposição foliar e associação da comunidade de invertebrados em folhas de espécies nativas e não-nativas de forma direta em tratamentos foliares que são ofertados, em geral, em *litter bags*. Neste contexto, as características foliares (e.g., palatabilidade e teores nutricionais) são um fator mais relevante na determinação da decomposição foliar e associação da comunidade de invertebrados do que origem das folhas (nativa *vs* não-nativa). Assim, em geral, espécies foliares não-nativas apresentam taxas de decomposição foliar e uma associação de invertebrados semelhante as observadas a espécies nativas com características foliares de palatabilidade e teores nutricionais semelhantes (e.g., CASAS et al., 2013; CLAESON et al., 2014; KÖNIG et al., 2014; MEDINA-VILLAR et al., 2015b; KUGLEROVÁ et al., 2017; FOGELMAN et al., 2018). Nestes casos, alterações consideradas negativas sobre a decomposição foliar (e.g., diminuição das taxas de decomposição) e a comunidade de invertebrados (e.g., diminuição da abundância/densidade e/ou riqueza da comunidade como um todo ou de algum grupo trófico funcional específico como os fragmentadores), quando existentes, são mais evidentes apenas para espécies não-nativas com folhas de baixa palatabilidade e/ou teores nutricionais (ALBARIÑO e BALSEIRO, 2002; CASAS et al., 2013; LARRANÃGA et al., 2014; HEPP e PASTORE, 2021). De forma semelhante, em experimentos de preferência alimentar com invertebrados fragmentadores, discos foliares não-nativos ofertados podem ser preteridos em relação a discos foliares com origem nativa devido a sua menor palatabilidade (CASOTTI et al., 2014). Em outros casos, entretanto, mesmo quando consumidos, os discos foliares de espécies não-nativas não constituíram, necessariamente, um recurso alimentar de qualidade afetando o crescimento ou a sobrevivência das larvas (KIFFER et al., 2018; MORETTI et al., 2020).

Em outra abordagem comum, por outro lado, a decomposição foliar e a comunidade de invertebrados são avaliadas com base na comparação entre trechos de riachos ou entre riachos com e sem a presença (parcial ou majoritária) de espécies não-nativas na vegetação ripária. Nesta abordagem, os estudos consideram os efeitos tanto das características foliares, quanto da diversidade e da disponibilidade de recursos foliares disponíveis nos riachos sobre a decomposição foliar e a comunidade de invertebrados. Como resultado, diminuição nas taxas de decomposição foliar (ABELHO e GRAÇA, 1996; MARTÍNEZ et al., 2013; FERREIRA et al., 2015) e na abundância (ou densidade) e/ou riqueza (ou diversidade) da comunidade de invertebrados (ABELHO e GRAÇA, 1996; McNEISH et al., 2015; McNEISH et al., 2017; SEENEY et al., 2019; LITTLE et al., 2021) aquáticos são reportados. Apesar disso, mesmo com esta abordagem, os resultados se mostram variados: riachos com presença de espécies não-nativas na vegetação ripária podem apresentar uma evidente diminuição da densidade de invertebrados e de fragmentadores em uma determinada região mas nenhuma alteração em outra (FERREIRA et al., 2015). Em outros casos, são observados impactos pontuais com a diminuição da densidade e riqueza apenas dos fragmentadores mas não da comunidade de invertebrados de como um todo (MARTÍNEZ et al., 2013).

Num contexto quantitativo, o número de estudos avaliando o efeito da presença de espécies não nativas na vegetação ripária sobre o funcionamento de riachos é expressivo. A maior parte destes estudos utiliza abordagens experimentais *in situ* (i.e., experimentos realizados em ambiente natural) (e.g., MENÉNDEZ et al., 2013; FERREIRA et al., 2015; SEENA et al., 2017) e/ou *ex situ* (i.e., experimentos realizados em ambiente laboratorial e/ou microcosmo/mesocosmo e, em geral, sob condições controladas) (e.g., CANHOTO e GRAÇA, 1999; SILVA et al., 2018; MORETTI et al., 2020). Além disso, uma parcela menor destes estudos utiliza abordagens teóricas (e.g., GRAÇA et al., 2002; CASTRO-DÍAZ e ALONSO, 2017; FERREIRA et al., 2021) ou de meta-análise (e.g., FERREIRA et al., 2016b; KENNEDY e EL-SABAAWI, 2017) para avaliar padrões ecológicos mais amplos (e.g., em escala regional ou global) com base na compilação de dados existentes na literatura. Estudos com uma abordagem cienciométrica, por outro lado, são praticamente inexistentes embora, nos últimos anos, a utilização de estudos cienciométricos para avaliar tendências ecológicas associadas ao funcionamento de riachos (LUIZA-ANDRADE et al., 2017; BRASIL et al., 2020; GAYER et al., 2021) e invasões biológicas (e.g., BARBOSA et al., 2012; FREHSE et al., 2016; PINTO et al., 2020) tenha crescido, especialmente na última década. Neste contexto, uma abordagem cienciométrica considerando estudos que avaliam o efeito de árvores e arbustos não-nativos sobre o funcionamento de riachos podem ser ecologicamente relevantes para o entendimento das principais tendências ecológicas associadas tanto a temática ‘funcionamento de riachos’ quanto à temática ‘invasões biológicas’.

1.3.3. *Hovenia dulcis* e o Funcionamento de Riachos

Apesar da ampla ocorrência nos remanescentes florestais na região sul do Brasil, estudos que avaliam os efeitos da presença de *H. dulcis* na vegetação ripária e sobre o funcionamento de riachos são pouco numerosos e recentes. Considerando a produção primária, foi demonstrado que, quando presente em elevadas densidades na vegetação ripária, *H. dulcis* é capaz de alterar o padrão temporal do aporte de serrapilheira alóctone (FONTANA et al., 2020). A contribuição relativa das diferentes frações da serrapilheira (e.g., folhas, galhos e partes reprodutiva) de *H. dulcis* sobre o aporte de serrapilheira total e o período do ano no qual elas apresentam as maiores contribuições de biomassa vegetal são, entretanto, ainda desconhecidos. Estas informações são ecologicamente relevantes pois, ao longo do ano, a serrapilheira alóctone, especialmente as folhas, é utilizada como recurso nutricional, abrigo ou substrato por microrganismos (e.g., SALES et al., 2015; GOMES et al., 2016; LEMES-SILVA et al., 2016) e invertebrados (e.g., CUMMINS et al., 2005; MORETTI et al., 2009; LEMES-SILVA et al., 2016; BIASI et al., 2019; REZENDE et al., 2019). Além disso, a composição química difere entre as frações de serapilheira de *H. dulcis* (SCHUMACHER et al., 2008) e, portanto, medir apenas a contribuição total (todas as frações juntas) pode gerar uma compressão incompleta da entrada de nutrientes nesses ecossistemas (MEDINA-VILLAR et al., 2015a).

Considerando a decomposição foliar, foi demonstrado que as folhas de *H. dulcis* apresentam elevadas taxas de decomposição, com perda de, em média, 40% da massa foliar após 15 dias de incubação nos riachos (KÖNIG et al., 2014; BIASI et al., 2020; HEPP e PASTORE, 2021; FONTANA et al., 2022). Embora o padrão de rápida decomposição foliar seja, aparentemente, constante, o efeito de *H. dulcis* sobre a estrutura e composição da comunidade de microrganismos e invertebrados é, aparentemente, contrastante. Considerando os microrganismos (neste caso, hifomicetos aquáticos), observamos que as taxas de esporulação tendem a ser menores na serrapilheira de *H. dulcis* se comparado com uma mistura de espécies nativas (BIASI et al., 2020). Por outro lado, a presença de folhas de *H. dulcis* em riachos parece não afetar significativamente a comunidade de invertebrados aquáticos cuja estrutura e composição é similar a espécies nativas com características foliares semelhantes (KÖNIG et al., 2014; HEPP e PASTORE, 2021; FONTANA et al., 2022). Neste contexto, o conhecimento existente sobre a influência de *H. dulcis* na decomposição e colonização da comunidade de invertebrados aquáticos se concentra na comparação das caraterísticas foliares (i.e., folhas nativas vs. folhas de *H. dulcis*) (KÖNIG et al., 2014; BIASI et al., 2020; HEPP e PASTORE, 2021; FONTANA et al., 2022). Embora ecologicamente importante, esta abordagem desconsidera os efeitos da interação entre as folhas de *H. dulcis* com as folhas das espécies nativas. Isso pode ser ecologicamente relevante para um entendimento amplo da influência de *H. dulcis* sobre a decomposição foliar e sobre a associação de invertebrados nas folhas por diferentes fatores. Primeiro, a interação entre folhas de diferentes espécies e com características foliares contrastantes pode acelerar ou retardar a decomposição foliar nos riachos através de mecanismos químicos (e.g., troca de nutrientes via lixiviação) ou físicos (e.g., a combinação de folhas diferentes cria micro-habitats diversificados) (veja GARTNER e CARDON, 2004). Segundo, ao contrário de monoculturas arbóreas (predomínio de uma espécie) (ABELHO e GRAÇA, 1996; MOLINERO e POZO, 2004), *H. dulcis* ocorre associada a espécies arbóreas nativas (DECHOUM et al., 2015a; FONTANA et al., 2020).

Nesta Tese, avaliei o efeito da presença de espécies arbóreas e arbustivas não nativas e não nativas invasoras na vegetação ripária sobre o funcionamento de riachos testando, especificamente, os efeitos destas espécies sobre as fontes de energia alóctones, a ciclagem de nutrientes e a rede trófica aquática destes ecossistemas. Para isso, utilizei uma abordagem experimental (Capítulos I e II) e uma abordagem cienciométrica (Capítulo III). No Capítulo I, considerei como modelo biológico a espécie *Hovenia dulcis* - uma espécie arbórea, não nativa e considerada invasora no sul do Brasil - e avaliei os efeitos da sua presença na vegetação ripária sobre a dinâmica de serrapilheira alóctone (i.e., fonte de energia) em riachos. No Capítulo II, considerei novamente como modelo biológico a espécie *H. dulcis* e avaliei os efeitos da sua presença na vegetação ripária sobre a decomposição foliar (i.e., ciclagem de nutrientes) e a estrutura e composição da comunidade de invertebrados aquáticos (i.e., rede trófica aquática) em riachos. No Capítulo III, considerei diferentes espécies arbóreas e arbustivas não nativas e não nativas invasoras e avaliei, sob uma abordagem cienciométrica, estudos que abordaram os efeitos da presença destas espécies na vegetação ripária sobre a decomposição foliar (i.e., ciclagem de nutrientes) e a estrutura e composição da comunidade de microrganismos e invertebrados aquáticos (i.e., rede trófica aquática). A partir disso, e considerando a abordagem de cada um dos capítulos, eu formulei os objetivos desta Tese.

2. OBJETIVOS

2.1 OBJETIVO GERAL

O objetivo geral desta Tese foi avaliar a influência da presença de árvores e arbustos não nativos e não nativos invasores na vegetação ripária sobre o funcionamento de riachos.

2.2 OBJETIVOS ESPECÍFICOS

Os objetivos específicos desta Tese foram:

i) Avaliar o efeito das características fenológicas de *H. dulcis* no padrão temporal e na quantidade de entrada de serrapilheira alóctone em riachos por meio da contribuição relativa de folhas, galhos e partes reprodutivas de espécies nativas e de *H. dulcis* em riachos com e sem a presença de *H. dulcis* na vegetação ripária **(Capítulo I).**

ii) Avaliar o efeito das folhas de *H. dulcis* disponíveis individualmente e/ou em conjunto com folhas de espécies nativas na decomposição foliar e na estrutura e composição da comunidade de invertebrados aquáticos associados em riachos **(Capítulo II).**

iii) Avaliar as principais tendências de estudos abordando os efeitos de espécies arbóreas e arbustivas não nativas e não nativas invasoras sobre a decomposição foliar e sobre as comunidades de microrganismos e invertebrados aquáticos em riachos **(Capítulo III).**

3. ESTRUTURA DA TESE

Considerando os aspectos ecológicos discutidos e os objetivos propostos, estruturei esta Tese em 3 capítulos que foram construídos a partir de uma abordagem experimental (Capítulos I e II) e cienciométrica (Capítulo III). Na sequência, descrevo brevemente cada capítulo, informo em que cada capítulo pode contribuir com o conhecimento existente e o periódico para o qual cada capítulo está e/ou será submetido. No **Capítulo I** *("When and how much a non-native tree species changes the temporal patterns and biomass of litterfall input in subtropical streams")*, avaliei o efeito da fenologia de uma espécie arbórea não nativa e invasora - *Hovenia dulcis* Thunb. (Rhamnaceae) - sobre o padrão temporal anual e a quantidade de entrada de serrapilheira alóctone (i.e., folhas, galhos e partes reprodutivas) em riachos subtropicais. Este manuscrito, inova exatamente ao quantificar mensalmente a contribuição relativa de folhas, galhos e partes reprodutivas de *H. dulcis* na entrada de serrapilheira alóctone em riachos o que, até onde tenho conhecimento, ainda era desconhecido. Este manuscrito foi publicado no periódico *Marine and Freshwater Research*. No **Capítulo II** *("Effect of a non-native invasive species on leaf decomposition and associated invertebrates in subtropical Atlantic Forest streams")* avaliei o efeito das folhas de *H. dulcis* oferecidas individualmente e associadas a uma mistura de espécies nativas, sobre a decomposição foliar e a comunidade de invertebrados aquáticos associados em três riachos subtropicais de Mata Atlântica. Este manuscrito inova ao avaliar o efeito da associação de folhas de *H. dulcis* com espécies foliares nativas sobre decomposição foliar e a associação de invertebrados aquáticos em riachos o que, até onde tenho conhecimento, ainda não havia sido estudado. Este manuscrito encontra-se submetido ao periódico *Aquatic Sciences*. No **Capítulo III** *("Effects of non-native plants on leaf decomposition and on communities of microorganisms and invertebrates in streams: a scientometric and systematic approach")*, realizei uma cienciometria e uma revisão sistemática para avaliar as principais tendências de publicações abordando os efeitos de espécies arbóreas e/ou arbustivas não nativas e não nativas invasoras sobre a decomposição foliar e as comunidades de microrganismos e invertebrados aquáticos em riachos. Este manuscrito inova ao utilizar uma amostragem quantitativa das principais tendências associadas a estudos que avaliam os efeitos de espécies arbóreas e/ou arbustivas não nativas e não nativas invasoras sobre a decomposição foliar e as comunidades de microrganismos e invertebrados aquáticos em riachos o que, até onde tenho conhecimento, é uma contribuição cienciométrica inédita. Este manuscrito será submetido ao periódico *Ecology and Evolution*.

4. CAPÍTULO I

When and how much a non-native tree species changes the temporal patterns and biomass of litterfall input in subtropical streams

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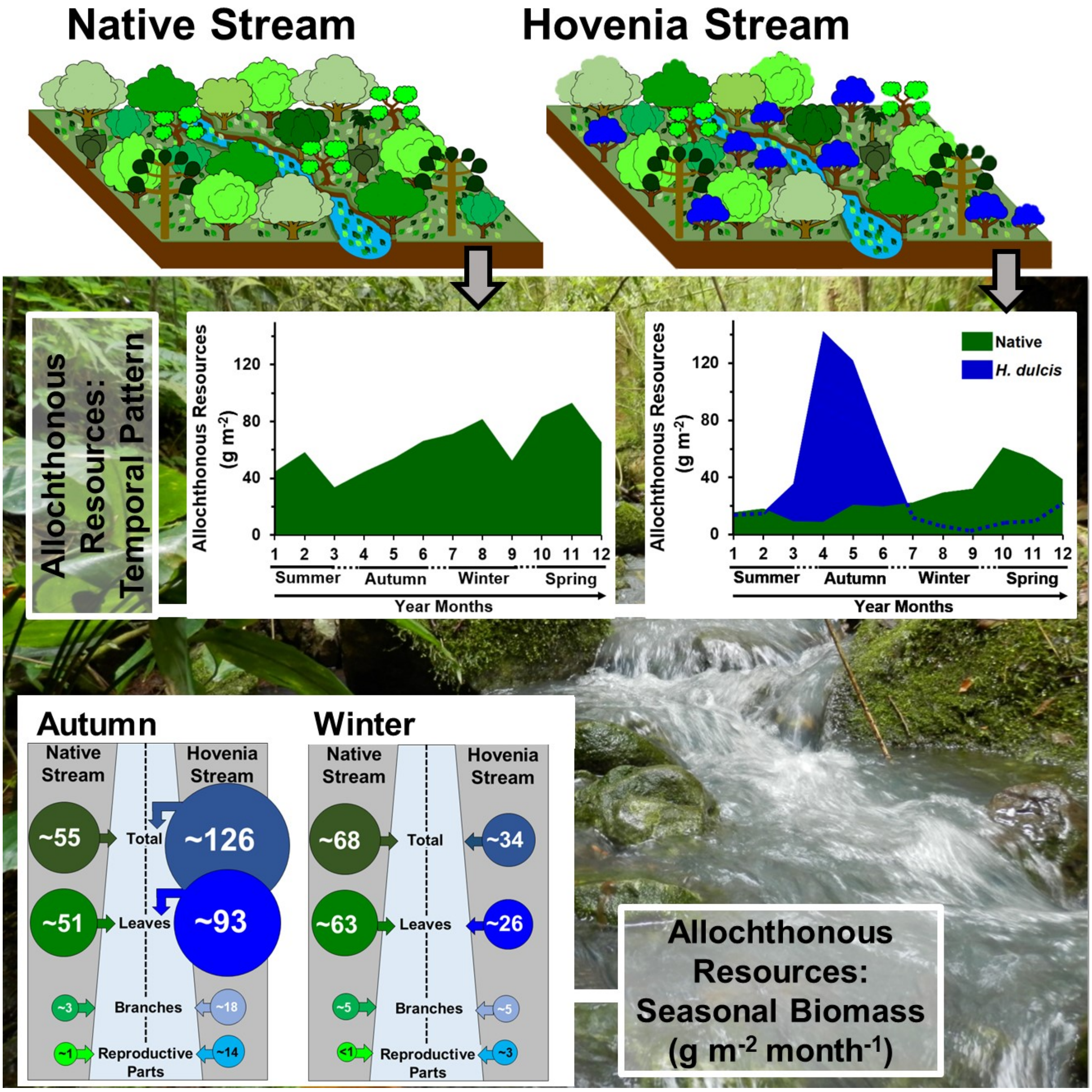
Graphical Abstract



Abstract. Context: In headwater streams, allochthonous litterfall input is an essential process to maintain the ecosystem functioning. The presence of non-native species in the riparian vegetation, with phenological characteristics distinct from those of most native tree species, can alter this process. **Aims:** We evaluated the effect of the phenology of a non-native species (*Hovenia dulcis*) on the temporal patterns and biomass of litterfall input into streams by comparing one stream with (*Hovenia stream*) and another without (*Native stream*) *H. dulcis* in their riparian vegetation. **Methods:** We quantified the litterfall input from native tree species and *H. dulcis* for 1 year by using buckets suspended above the streambeds. **Key results:** The temporal pattern of litterfall input changed between streams, with quantitative differences between them during autumn and winter. In the *Native stream*, litterfall input was slightly higher in spring (~79 g m^{-2} $month^{-1}$), followed by winter (~68 g m^{-2} $month^{-1}$) and autumn (~54 g m^{-2} $month^{-1}$), whereas in the *Hovenia stream*, it was concentrated in autumn (~126 g m^{-2} $month^{-1}$). **Conclusions and implications:** Our results indicated that the presence of *H. dulcis* in the riparian vegetation, when in high density, changes the temporal pattern and biomass of litterfall input into subtropical streams.

Additional Keywords. Atlantic Forest, coarse particulate, exotic species, Japanese raisin tree, non-native plant species, riparian zone, stream functioning, tree invasions.

Introduction

Ecosystem functioning is associated with the flow of matter and energy through its biotic and abiotic components (Díaz and Cabido 2001). In forest headwater streams, primary production is limited by shading, and the input of allochthonous litterfall is essential to maintain functioning ecosystem aquatic communities and processes (Vannote *et al.*1980; Abelho 2001). Allochthonous litterfall input from riparian vegetation consists of leaves (between 60 and 80%), branches, and reproductive parts (e.g. flowers, fruits, and seeds) (Gonçalves and Callisto 2013; Gonçalves *et al.* 2014; Rezende *et al.* 2017). Litterfall production is strongly related to the structure (Souza *et al.* 2019) and composition of riparian vegetation (Gonçalves *et al.* 2014; Rezende *et al.* 2017) and seasonal climatic variations (Zhang *et al.* 2014). Seasonal variations in rainfall (e.g. wet and dry seasons) (e.g. Gonçalves and Callisto 2013) and temperature (e.g. year seasons) (e.g. Abelho and Graça 1998) affect litterfall production because they influence the phenological processes of tree species (e.g. in this case, leaf fall) (see also Tonin *et al.* 2017 and Zhang *et al.* 2014).

In southern Brazil (subtropical climatic zone) (Alvares *et al.* 2013), typical forests are composed of deciduous, semi-deciduous and perennial tree species (e.g. Ruschel *et al.* 2005; Athayde *et al.* 2009). In these subtropical native forests, litterfall production varies throughout the year, with slightly higher inputs during winter and spring. This is associated with tree species leaf phenology, especially the fall of deciduous leaves (in the cold seasons) (see Athayde *et al.* 2009; Ferrera *et al.* 2017) and the replacement of senescent leaves (in the early growing season) (see Brun *et al.* 2001; König *et al.* 2002; Turchetto and Fortes 2014). Moreover, litterfall production is also related to monthly rainfall peaks (e.g. Fontana *et al.* 2020).

Riparian forests are highly susceptible to invasion by non-native plant species, which can affect stream functioning (e.g. Mineau *et al.* 2012; Castro-Díez and Alonso 2017; Seeney *et al.* 2019; Biasi *et al.* 2020). Effects of invasive tree species on stream functioning can be mediated by changes in the characteristics of allochthonous litterfall input into streams, which may differ from native species in terms of phenology, biomass, and typology (Mineau *et al.* 2012; Fontana *et al.* 2020). Native to Asia, *Hovenia dulcis* Thunb. (Rhamnaceae) (Carvalho 1994) is a non-native, invasive tree species in Brazil (Zenni and Ziller 2011). *H. dulcis* is a deciduous tree species (Carvalho 1994) and, in southern Brazil, presents extended leaf phenology, with young leaves growing earlier than for most native deciduous tree species (Rejmánek 2013). Additionally, *H. dulcis* pseudo-fruits are consumed by the local fauna (e.g.

mammals and birds) (Lima *et al.* 2015), which possibly contributes to the dispersion of the species. *H. dulcis* occurrence has been reported in several forest remnants in southern Brazil (Dechoum *et al.* 2015; Padilha *et al.* 2015; Schmidt *et al.* 2020), including in the riparian vegetation of streams (König *et al.* 2014; Biasi *et al.* 2020; Fontana *et al.* 2020).

Recently, it has been shown that the presence of *H. dulcis* in high density in riparian vegetation can change the allochthonous litterfall input into streams, with a higher input especially during autumn (Fontana *et al.* 2020). Despite this, little is known about the relative contribution of *H. dulcis* leaves, branches, and reproductive parts to the composition of this temporal pattern. This paper presents fundamental data on the impact of *H. dulcis* as a non-native invasive riparian species, which is lacking in the literature for this species. This is the first study to quantify the monthly contribution of *H. dulcis* leaves, branches, and reproductive parts in the total allochthonous litterfall in subtropical streams. This can be ecologically important because aquatic communities (e.g. microorganisms and invertebrates) use *H. dulcis* allochthonous resources, especially its leaves, both as food and substrate (König *et al.* 2014; Biasi *et al.* 2020). In addition, chemical composition differs between the fractions of *H. dulcis* allochthonous litterfall (e.g. between leaves and branches) (see Schumacher *et al.* 2008). Thus, measuring only total litterfall (i.e. all fractions together) can generate a limited comprehension of the nutrient input into these ecosystems (Medina-Villar *et al.* 2015).

In this study, we evaluated the effect of *H. dulcis* presence in the riparian vegetation on the temporal pattern and biomass of allochthonous litterfall inputs into streams. We also quantified the relative contribution of *H. dulcis* leaves, branches, and reproductive parts to the litterfall. Comparisons were made between a headwater stream with (*Hovenia stream*) and one without (*Native stream*) *H. dulcis* in the riparian vegetation. We expected that *H. dulcis* leaf fall would be the main phenological process associated with the temporal pattern and biomass of allochthonous litterfall inputs into the *Hovenia stream*. Thus, we hypothesised that (1) leaves would be the main plant fraction of the litterfall in both streams and allochthonous litterfall input into the *Hovenia stream* would be greater during autumn than that into the *Native stream*, and (2) allochthonous litterfall input into the *Hovenia stream* would be smaller than that into the *Native stream*, because the greater *H. dulcis* leaf fall occurs during a shorter period of the year (autumn months).

Material and Methods

Study Area

This study was conducted in the northern region of the state of Rio Grande do Sul, southern Brazil (Fig. S1). This region is part of the Atlantic Forest biome located in the transition zone between the Seasonal Forest and the Araucaria Rain Forest (Oliveira-Filho *et al.* 2015). In the study region, native forest formations are diverse (Leyser *et al.* 2012; Loregian *et al.* 2012) and characterised by the presence of deciduous, semi-deciduous and perennial tree species (Ruschel *et al.* 2005; Athayde *et al.* 2009). Studies conducted in this region indicate the presence of ~70 tree species and ~30 botanical families, with Fabaceae, Lauraceae, Meliaceae, Myrtaceae and Sapindaceae being especially abundant and diverse (for more details, see Leyser *et al.* 2012; Loregian *et al.* 2012; Mélo *et al.* 2013). Climate in the study region is humid subtropical, Cfa according to the Köppen classification, with annual mean temperature of 16–20°C and annual rainfall ranging from 1600 to 2200 mm (Alvares *et al.* 2013). Considering *H. dulcis* occurrence in several forest fragments in this region (see Padilha *et al.* 2015), two low-order forest streams (≤3rd order) with similar riparian vegetation widths (~30 m) and different *H. dulcis* densities in the riparian zone were selected. The streams were ~7 km apart (in a straight-line estimation) and are part of the Dourado River hydrographic basin.

The *Native stream* runs through an Atlantic Forest remnant with riparian vegetation composed only of typical native species. The *Hovenia stream* runs through an Atlantic Forest remnant with riparian vegetation composed of typical native species and *H. dulcis*. In the stretches (~100 m; see below) where the buckets were suspended, the *Native stream* is 1–2 m wide and has a maximum depth of 0.20 m, whereas the *Hovenia stream* is 3–5 m wide with a maximum depth of 0.30 m. Both streams present predominantly rocky beds. In the *Hovenia stream*, *H. dulcis* density in the riparian zone (see below) was estimated by counting and measuring the circumference of all live *H. dulcis* individuals with diameter at breast height (~1.50 m above the ground) of ≥ 0.10 m in 10 plots of 10 × 4 m (40 m^2 per plot; total area sampled: 400 m^2; 0.04 ha). On each stream bank, five of these plots were established, separated from each other by at least 5 m, with the 10 m side running parallel to the stream. We also estimated the basal area of each *H. dulcis* tree using the equation $BA = \pi \times r^2$, where BA = basal area, $\pi = 3.1415$, and r = radius estimated from the circumference. In the *Hovenia stream*, the *H. dulcis* estimated density was ~650 individuals ha^{-1} and the estimated basal area of *H. dulcis* individuals was 0.070 ± 0.080 m^2 $individuals^{-1}$.

Dynamics of Allochthonous Litterfall

The effect of *H. dulcis* invasion on litterfall input into the streams was assessed by quantifying the contribution of allochthonous litterfall for 1 year (April 2019 – March 2020) by using a methodology adapted from Gonçalves and Callisto (2013) and Fontana *et al.* (2020). We considered only the vertical (direct) input, because the lateral (indirect) input is associated with stream morphology aspects (e.g. slope; Lisboa *et al.* 2015) and may not reflect an association between leaf fall and allochthonous litterfall input into streams. In each stream, a stretch of ~100 m was defined and divided into three collection points ~20–30 m apart from each other. The allochthonous litterfall input was quantified using buckets (with a perforated bottom to drain rainwater) suspended ~1 m above the streambed. At each collection point, two rows of five buckets (10 buckets) were placed, totaling 30 buckets per stream (area: 0.04 m^2 $bucket^{-1}$; total area: 1.2 m^2 $stream^{-1}$).

All the plant material retained in the buckets was collected monthly (30 ± 5 days), individually packed in plastic bags, and taken to the laboratory, where it was dried in an oven with air circulation at 40 ± 5°C for 72 h. Afterwards, the plant litter was separated into material from native species and that originating from *H. dulcis*. Subsequently, each group was sorted into leaves, branches, and reproductive parts (i.e. flowers, fruits, and seeds) and weighed separately. Finally, we estimated the daily allochthonous litterfall input on the basis of the monthly data and multiplied the daily values by 30 for standardisation (considering months with 30 days). In this study, we defined the months corresponding to each season in the southern hemisphere, with autumn occurring from April to June, winter from July to September, spring from October to December, and summer from January to March.

Data Analysis

Initially, the data were evaluated to verify the assumptions of normality (Shapiro–Wilk test) and homoscedasticity (Levene's test) and transformed into log (x + 1) or square-root transformed when necessary to meet the assumptions for the use of parametric tests. First, we tested the variation of allochthonous litterfall input between the seasons in the *Native stream* and the *Hovenia stream* (in the latter, determining the litterfall of native species and *H. dulcis* together) individually using ANOVA with repeated measures (repeated-measures ANOVA). After that, we tested the variation of allochthonous litterfall input from native species and *H. dulcis* between the seasons in the *Hovenia stream* also using repeated-measures ANOVA. To this end, in both cases, we considered the input of leaves, branches, reproductive parts, and total litterfall as response variables (quantitative), the seasons as explanatory variable (qualitative), and the collection points of each stream (qualitative) as the fixed repeated factor. When the repeated-measures ANOVA showed a difference between the seasons, we used, *a posteriori*, paired comparisons between the groups (seasons), with *P*-values adjusted by the Bonferroni correction method, to determine differences between the seasons. We compared the biomass of allochthonous litterfall input into streams between each season individually, using Student's *t*-tests. To this end, we considered the input of leaves, branches, reproductive parts, and total litterfall for each season (quantitative) as response variables and the streams as explanatory variables (qualitative). The statistical analyses were performed on the R statistical software (ver. 3.6.1, R Foundation for Statistical Computing, Vienna, Austria, see https://www.R-project.org/) using the packages 'stats' (ver. 3.6.1, R Core Team 2019, see https:// www.R-project.org/), 'car' (ver. 3.0-4, Fox and Weisberg 2019, see https://socialsciences.mcmaster.ca/jfox/Books/Companion/) and 'carData' (ver. 3.0-2, Fox *et al.* 2018, see https://CRAN.R-project.org/package=carData).

Results

Allochthonous Litterfall Input: Temporal Patterns

In the *Native stream*, leaves represented ~82% of the total input (a mean between seasons), whereas branches represented ~13%, and reproductive parts accounted for ~5% of the input. The input from leaves was highest during winter (~64 g m^{-2} $month^{-1}$), followed by spring (~57 g m^{-2} $month^{-1}$) and autumn (~51.4 g m^{-2} $month^{-1}$; $F_{3,30} = 6.0$, $P = 0.002$). The input from branches was highest ($F_{3,30} = 8.9$, $P < 0.001$) during spring (~18 g m^{-2} $month^{-1}$), whereas the input from reproductive parts was highest ($F_{3,30} = 9.1$, $P < 0.001$) in summer (~5 g m^{-2} $month^{-1}$) and spring (~4 g m^{-2} $month^{-1}$). The total allochthonous litterfall input varied throughout the year and was relatively constant during spring (~79 g m^{-2} $month^{-1}$), winter (~68 g m^{-2} $month^{-1}$) and autumn (~54 g m^{-2} $month^{-1}$), but lower in summer (~46 g m^{-2} $month^{-1}$; $F_{3,30} = 4.1$, $P = 0.014$) (see Fig. 1a, Table 1, Table S1 for details).

In the *Hovenia stream*, leaves (from native species and *H. dulcis* together) represented ~75% of total input (a mean between seasons), branches accounted for ~16%, and reproductive parts ~9% of input. The input from leaves was highest ($F_{3,30} = 3.5$, $P = 0.025$) in autumn (~94 g m^{-2} $month^{-1}$), which was also observed for branches ($F_{3,30} = 5.3$, $P = 0.004$; ~18 g m^{-2} $month^{-1}$). The input from reproductive parts was slightly higher in autumn (~14 g m^{-2} $month^{-1}$); however, differences were not statistically significant between the seasons ($F_{3,30} = 1.5$, $P = 0.215$). The total allochthonous litterfall input varied throughout the year and was highest in autumn (~126 g m^{-2} $month^{-1}$; $F_{3,30} = 18.1$, $P < 0.001$) (see Fig. 1b, Table 1, Table S1 for details). The contribution of *H. dulcis* to the allochthonous litterfall input was highest in autumn and summer, representing ~87 and ~57% of the total input respectively. In winter and spring; however, allochthonous litterfall input from *H. dulcis* was <25% (see Fig. 1b, Table 2, Table S2 for details).

Fig. 1. Monthly litterfall input (g m^{-2}; mean ± s.d.) from native species and *H. dulcis* observed in the *Native Stream* (*a*) and *Hovenia stream* (*b*). L = Leaves; B = Branches; R = Reproductive Parts. Initial “A” = “April”.

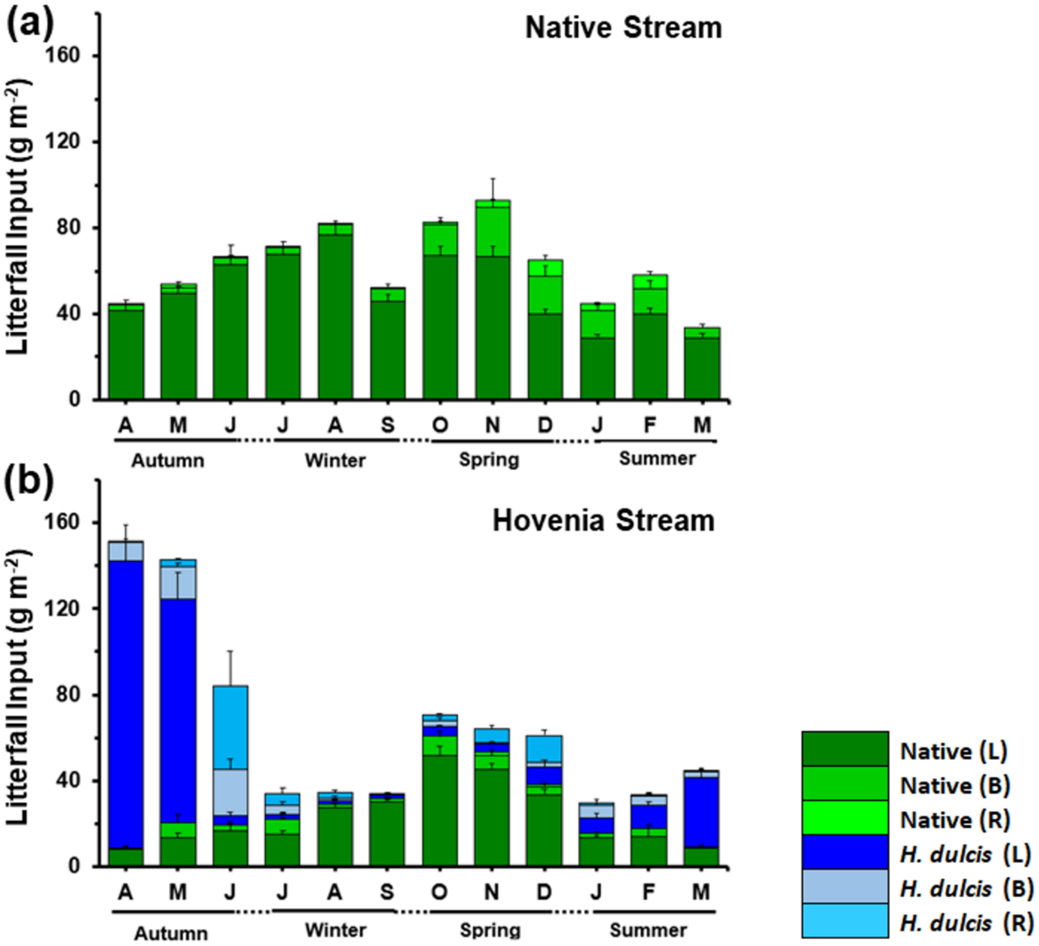


Table 1. Allochthonous litterfall input (g m^{-2} month^{-1}; mean ± s.d.) of leaves, branches, reproductive parts, and total in the studied streams during the seasons. Values described in parentheses indicate the approximate relative contribution (%) in relation to the total litterfall contribution in each season. The test values (F), degrees of freedom (d.f.), and test significance (P) are described. For detailed statistical results, see Table S1. (*) Different letters indicate statistical differences between seasons ($P < 0.05$).

Native Stream	Autumn		Winter		Spring		Summer		RM ANOVA (d.f. = 3,30)
Leaves	51.4 ± 24.2^{ab}	(93.5%)	63.5 ± 22.0^{a}	(93%)	56.9 ± 23.3^{a}	(72%)	32.3 ± 6.7^{b}	(70%)	$F = 6.0, P = 0.002$*
Branches	2.7 ± 0.8^{b}	(5%)	4.5 ± 1.9^{b}	(6.5%)	18.1 ± 15.7^{a}	(23.0%)	8.8 ± 6.7^{ab}	(19%)	$F = 8.9, P < 0.001$*
Reproductive Parts	0.8 ± 1.8^{bc}	(1.5%)	0.1 ± 0.2^{c}	(0.5%)	3.6 ± 5.0^{ab}	(5%)	5.1 ± 4.2^{a}	(11%)	$F = 9.1, P < 0.001$*
Total Litterfall Input	54.9 ± 25.6^{ab}	(100%)	68.0 ± 21.7^{ab}	(100%)	78.8 ± 35.3^{a}	(100%)	46.3 ± 11.6^{b}	(100%)	$F = 4.1, P = 0.014$*
H. dulcis Stream									
Leaves	93.5 ± 67.0^{a}	(74%)	25.7 ± 7.7^{b}	(75%)	45.7 ± 20.1^{ab}	(72%)	29.8 ± 13.2^{ab}	(80%)	$F = 3.6, P = 0.025$*
Branches	18.3 ± 9.4^{a}	(14.5%)	5.5 ± 6.4^{b}	(16%)	9.9 ± 7.3^{ab}	(15.5%)	6.5 ± 4.0^{b}	(17.5%)	$F = 5.4, P = 0.004$*
Reproductive Parts	14.2 ± 30.5	(11.5%)	3.1 ± 3.5	(9%)	7.9 ± 9.4	(12.5%)	1.0 ± 0.7	(2.5%)	$F = 1.6, P = 0.215$
Total Litterfall Input	126.1 ± 56.7^{a}	(100%)	34.3 ± 7.3^{c}	(100%)	63.5 ± 21.9^{b}	(100%)	37.3 ± 13.4^{c}	(100%)	$F = 18.1, P < 0.001$*

Table 2. Allochthonous litterfall input (g m^{-2} month^{-1}; mean ± s.d.) of leaves, branches, reproductive parts, and total from native species and *H. dulcis* in the *Hovenia stream* during the seasons. Values described in parentheses indicate the approximate relative contribution (%) in relation to the total litterfall contribution in each season. The test values (F), degrees of freedom (d.f.), and test significance (P) are described. For detailed statistical results, see Table S2. (*) Different letters indicate statistical differences between seasons ($P < 0.05$).

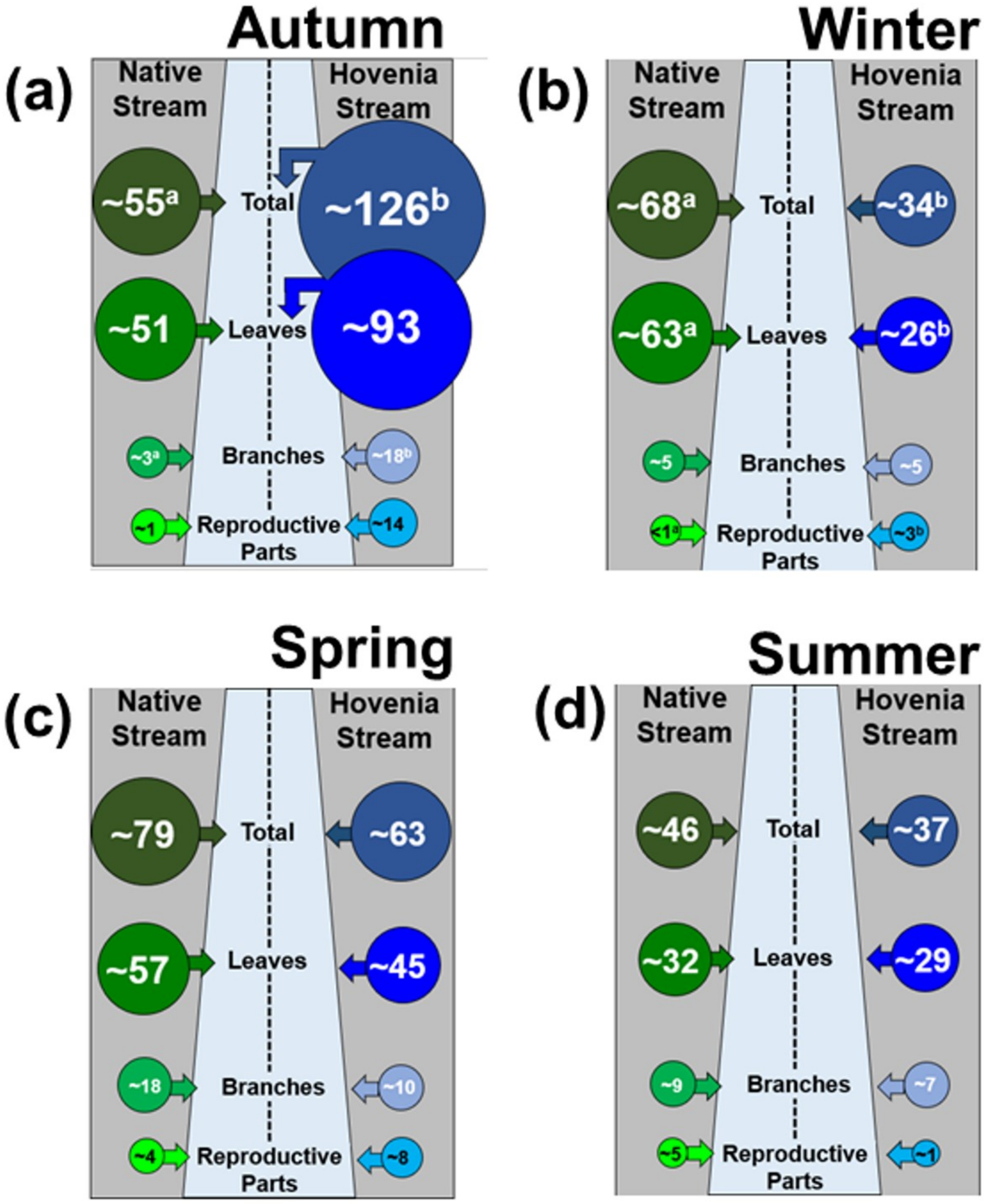
Origin	Autumn		Winter		Spring		Summer		RM ANOVA (d.f. = 3,30)
Native Leaves	12.9 ± 8.9	(10%)	24.2 ± 8.5	(70%)	40.7 ± 19.7	(64%)	13.5 ± 9.3	(36%)	F = 2.9, $P = 0.052$
Native Branches	3.4 ± 4.9	(3%)	3.5 ± 4.5	(10%)	7.3 ± 6.1	(11.5%)	2.7 ± 2.9	(7%)	F = 2.6, $P = 0.073$
Native Reproductive Parts	0.0 ± 0.0^{b}	(0%)	0.2 ± 0.2^{b}	(1%)	1.1 ± 1.3^{a}	(1.5%)	0.1 ± 0.1^{b}	(0.5%)	F = 5.6, $P = 0.003$*
H. dulcis Leaves	80.5 ± 71.8^{a}	(64%)	1.5 ± 1.7^{c}	(4%)	5.0 ± 4.5^{bc}	(8%)	16.3 ± 14.1^{ab}	(43.5%)	F = 14.9, $P < 0.001$*
H. dulcis Branches	14.8 ± 7.11^{a}	(12%)	1.9 ± 2.4^{b}	(6%)	2.5 ± 2.6^{ab}	(4%)	3.9 ± 2.6^{b}	(10.5%)	F = 13.6, $P < 0.001$*
H. dulcis Reproductive Parts	14.2 ± 30.5	(11%)	2.9 ± 3.53	(9%)	6.9 ± 8.7	(11%)	0.9 ± 0.6	(2.5%)	F = 1.5, $P = 0.242$
Native Litterfall Input	16.4 ± 10.9^{b}	(13%)	27.9 ± 8.4^{ab}	(81%)	49.1 ± 20.9^{a}	(77%)	16.2 ± 29.8^{b}	(43.5%)	F = 10.6, $P < 0.001$*
H. dulcis Litterfall Input	109.6 ± 63.1^{a}	(87%)	6.6 ± 6.5^{c}	(19%)	14.5 ± 9.5^{bc}	(23%)	21.0 ± 13.9^{b}	(56.5%)	F = 24.3, $P < 0.001$*
Total Litterfall Input	126.1 ± 56.7^{a}	(100%)	34.3 ± 7.3^{c}	(100%)	63.5 ± 21.9^{b}	(100%)	37.3 ± 13.4^{c}	(100%)	F = 18.1, $P < 0.001$*

Allochthonous Litterfall Input: Biomass

Comparatively, allochthonous litterfall input did not differ between the streams when considering the entire year ($t = -0.6$, *d.f.* = 70, $P = 0.535$), but differences were observed in autumn and winter. In autumn, the allochthonous litterfall input in the *Hovenia stream* (~126 g m^{-2} $month^{-1}$) was 2.3 times greater than that in the *Native stream* (~55 g m^{-2} $month^{-1}$; $t = 3.6$, *d.f.* = 16, $P = 0.002$) (see Fig. 2a, Table S3 for details). However, in winter, the allochthonous litterfall input in the *Native stream* (68 g m^{-2} $month^{-1}$) was two times greater than in the *Hovenia stream* (34 g m^{-2} $month^{-1}$; $t = -5.3$, *d.f.* = 16, $P < 0.001$) (see Fig. 2b, Table S3 for details).

In the *Native stream*, the contribution of different fractions of litterfall originating from native species was more evenly distributed throughout the year. The input of leaves was constant from April to December, the input of branches occurred mainly from October to February (spring and summer), and the input of reproductive parts occurred especially from November to February (spring and summer) (Fig. 1a, Fig. 2a–d, Table 1). However, in the *Hovenia stream*, the peaks of allochthonous litterfall input from the different fractions originating from *H. dulcis* occurred in autumn, including leaves (~64%, especially in April and May), branches (~12%, especially in May and June), and reproductive parts (~11%, especially in June) (Fig 1b, Fig. 2a, Table 2).

Fig. 2. Comparative allochthonous litterfall input (g m^{-2} month^{-1}) of leaves, branches, reproductive parts, and total in the *Native Stream* and *Hovenia stream* in autumn (*a*), winter (*b*), spring (*c*), and summer (*d*). For statistical details, see Table S3. Different letters indicate statistical differences between streams ($P < 0.05$).



Discussion

In both streams, the total allochthonous litterfall input was determined especially by leaf fall and, in the *Hovenia stream*, was concentrated during autumn, corroborating our first hypothesis. The biomass of allochthonous litterfall that entered both streams was similar considering the whole year (although amounts differed during autumn and winter), which contradicted our second hypothesis that biomass input would be lower in the *Hovenia stream*.

Allochthonous Litterfall Input: Temporal Patterns

Different temporal patterns for allochthonous litterfall input were observed between the *Native stream* and *Hovenia stream*. This variability was directly related to the phenological differences between most native tree species and *H. dulcis*. Comparatively, some *H. dulcis* phenophases, such as leaf fall and, possibly, the ripening of pseudo-fruits (see Milani 2013; Milani 2017), occurred earlier than in most native tree species (see Marques *et al.* 2004; Athayde *et al.* 2009; Ferrera *et al.* 2017).

The leaf fall peak occurred earlier in *H. dulcis* than in most native species. In the *Hovenia stream*, the input of leaves was higher between March and May, which is related to the period of *H. dulcis* leaf fall (Carvalho 1994; Fontana *et al.* 2020). The low and constant input of *H. dulcis* leaves during the other months occurred because this species punctually replaced senescent leaves with younger leaves (Milani 2013). In contrast, in the *Native stream*, the input of leaves showed little variation throughout the year, being slightly higher in winter, followed by spring and autumn. In autumn and winter, this input was related to the partial (semi-deciduous trees) or total (deciduous trees) leaf fall of native species, possibly owing to the typical decrease in temperature during these seasons (Marques *et al.* 2004; Athayde *et al.* 2009; Ferrera *et al.* 2017; Perina *et al.* 2019). In spring, the high input of leaves may be related to the replacement of senescent leaves with younger leaves, which occurs at the beginning of the growth season (Brun *et al.* 2001; König *et al.* 2002; Turchetto and Fortes 2014).

A different temporal phenological pattern was also observed for the fall of *H. dulcis* reproductive parts, particularly pseudo-fruits. In the *Hovenia stream*, the contribution of reproductive parts, although statistically similar between the seasons, showed peaks mainly in June and December. This corresponded, possibly, to the periods of ripening of pseudo-fruits and flowering in *H. dulcis* respectively (Carvalho 1994; Milani 2013; Milani 2017). In contrast, in the *Native stream*, the input of reproductive parts increased during spring and summer (November to February), which coincides with the flowering and fruiting periods of most native tree species in subtropical forests in southern Brazil (Marques *et al.* 2004; Athayde *et al.* 2009; Ferrera *et al.* 2017).

Allochthonous Litterfall Input: Biomass

Biomass of allochthonous litterfall input was similar in both streams when considering the whole year, but it differed during autumn and winter. During autumn, the allochthonous litterfall input was higher in the *Hovenia stream*, when there is greater input of leaves, branches, and reproductive parts originating from *H. dulcis*. This highlights the importance of evaluating the different litterfall input fractions and corroborates findings from Fontana *et al.* (2020), who showed that a higher litterfall input in autumn was attributed principally to *H. dulcis* leaf fall. However, allochthonous litterfall input was higher in the *Native stream* in winter, when a higher input of leaves from native tree species occurs. This significant decrease in allochthonous litterfall input in the *Hovenia stream* during the winter months can be associated with the presence of *H. dulcis* in the riparian vegetation. Moreover, studies have demonstrated that presence of *H. dulcis* in forest remnants in southern Brazil does not seem to affect species richness (Dechoum *et al.* 2015; Schmidt *et al.* 2020). Therefore, although allochthonous litterfall input in winter was lower in the *Hovenia stream*, it was not necessarily less diversified.

Implications for Stream Functioning

Presence of non-native species in the riparian vegetation of streams may cause changes in leaf decomposition and structure of aquatic communities (Castro-Díez and Alonso 2017; Ferreira *et al.* 2021). In the specific case of *H. dulcis*, the changes in temporal pattern and allochthonous litterfall input generated by the presence of this species in the riparian vegetation can affect stream functioning. For example, the high decomposition rates of *H. dulcis* leaf litter (König *et al.* 2014; Biasi *et al.* 2020) may increase litter processing in the streams during autumn, when high allochthonous litterfall input of *H. dulcis* leaves occurs. Moreover, considering the aquatic communities, studies have shown different microorganism (e.g. aquatic hyphomycetes) and invertebrate responses to the presence of *H. dulcis* material in streams. Regarding aquatic hyphomycetes, the sporulation rates and diversity tend to be lower in *H. dulcis* leaf litter compared with that from native tree species (Biasi *et al.* 2020). As for aquatic invertebrates, the structure and composition of aquatic communities tend to be similar in leaf litter from *H. dulcis* and native tree species, with similar structural and chemical characteristics (König *et al.* 2014). In this context, future studies should evaluate whether variation in leaf litter availability over the seasons (especially in autumn), as demonstrated in this study and in

Fontana *et al.* (2020), affects the functioning of streams with *H. dulcis* present in their riparian vegetation.

Hovenia dulcis *Representativeness in Forests of Southern Brazil: A Parallel*

In this study, changes in temporal patterns and allochthonous litterfall input were observed in a stream where the density of *H. dulcis* individuals in the riparian vegetation was high. This finding corroborates the study by Fontana *et al.* (2020), who reported that changes in allochthonous litterfall input were also related to the high density of *H. dulcis* in the riparian vegetation of streams. In our study, the annual (sum of monthly means) litterfall input from *H. dulcis* was ~450 g m^{-2} in the *Hovenia stream* (~650 individuals ha^{-1}), which is proportionally smaller than the annual litterfall input from *H. dulcis* reported by Fontana *et al.* (2020), namely, ~330 and 530 g m^{-2} in streams with density of ~360 and ~230 individuals ha^{-1} respectively. This demonstrated that, besides the *H. dulcis* density in riparian vegetation, other factors (e.g. basal area, size or age of *H. dulcis* trees) can determine the biomass of allochthonous litterfall input into streams. Therefore, our results strengthened the idea that the presence of *H. dulcis* in the riparian vegetation can affect the dynamics of allochthonous litterfall in streams, but only when *H. dulcis* is present at high densities in riparian vegetation (e.g. ~230 and ~360 individuals ha^{-1} in Fontana *et al.* (2020) and ~650 individuals ha^{-1} in this study).

In the *Hovenia stream*, *H. dulcis* density in the riparian vegetation was considerably high, which has also been observed in other subtropical forest remnants in southern Brazil (Fontana *et al.* 2020; Schmidt *et al.* 2020). However, *H. dulcis* presence in forest remnants in southern Brazil at high densities (similar to what was observed in this study) is unusual. Scenarios where the representativity of *H. dulcis* (i.e. density) is considerably lower have been more commonly reported (Valério *et al.* 2008; Loregian *et al.* 2012; Schorn *et al.* 2014; Turchetto *et al.* 2015; Freitas *et al.* 2017; Gerber *et al.* 2018).

Conclusions

Our results indicated that, owing to the phenological characteristics of *H. dulcis*, its presence in riparian vegetation can alter the temporal pattern of allochthonous litterfall input into subtropical streams. Comparatively, some *H. dulcis* phenophases, especially leaf fall, occurred earlier than those of most native tree species. Given this, in the stream where *H. dulcis* was present in the riparian vegetation, higher allochthonous litterfall input was observed in autumn, but not in winter and spring, as would typically be expected in the native forest formations studied. However, we emphasise that these results were observed in a stream where the density of *H. dulcis* individuals in the riparian vegetation was high (in this study, ~650 individuals ha^{-1}). Therefore, it is not possible to know whether the changes observed in this study for allochthonous litterfall input also occur in streams where the density of this species in riparian vegetation is lower.

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Declarations

Declaration of Funding

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Declaration of interests

The authors declare having no conflicts of interest.

Data Availability Statement (DAS)

The data used to generate the results in this paper are not available.

Authors Contributions

LEF, LUH and MMP conceived the study design; LEF and CB performed the field activities; LEF performed the laboratory processing of plant materials; RMR and LUH provided the laboratory structure necessary for processing plant material; LEF performed the data analysis and led the manuscript writing with the help of LUH and MMP. All authors revised critically the manuscript and agreed with its submission.

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Supplementary Material List

Fig S1. Localization of *(a)* Brazil, *(b)* state of Rio Grande do Sul, and *(c)* study area and studied streams. Prepared using QGIS 3.4 (creation of maps) and Microsoft PowerPoint 365® (formatting and presentation of maps). The spatial data were obtained from the database provided by DIVA-GIS (https://www.diva-gis.org/Data) (the countries in *(a)*), by Instituto Brasileiro de Geografia e Estatística - IBGE (https://www.ibge.gov.br) (Brazil and the Brazilian states in *(b)* and *(c)*), and by Fundação Estadual de Proteção Ambiental Henrique Luís Roessler - FEPAM (http://www.fepam.rs.gov.br) (hydrographic network in *(c)*).

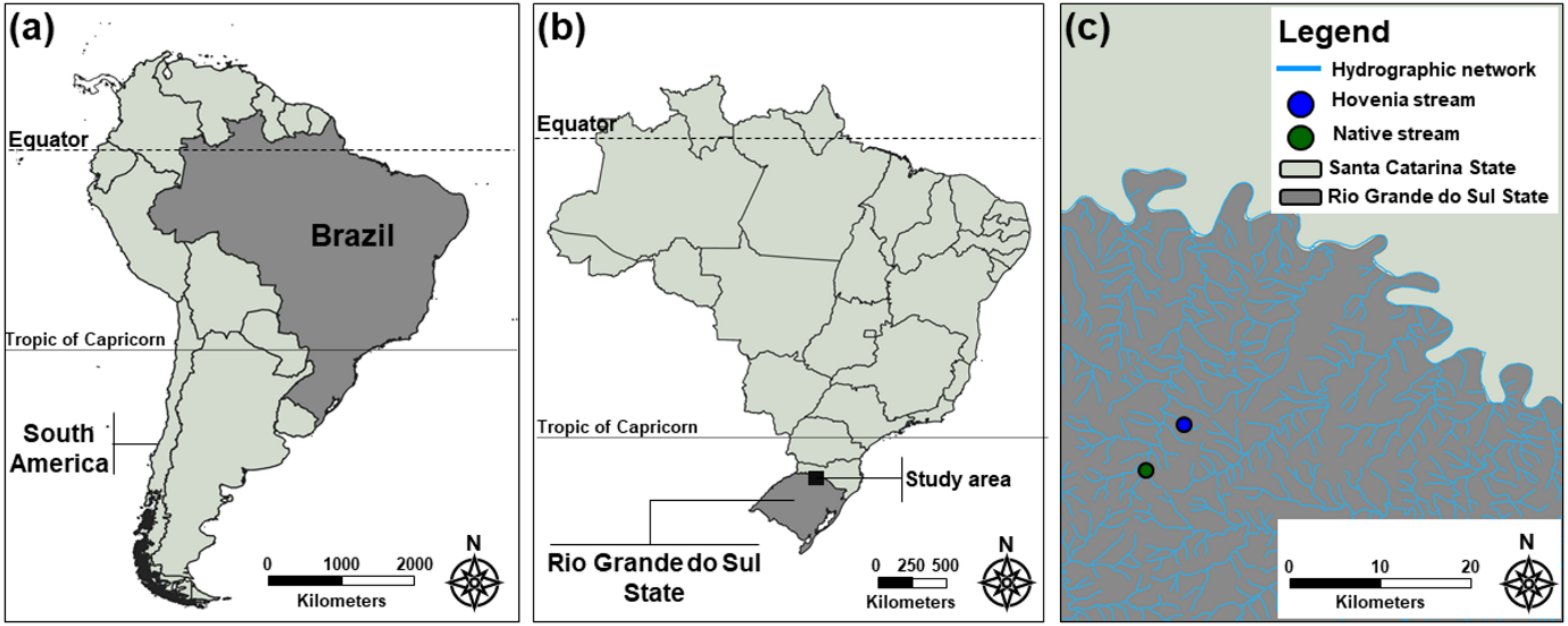

Table S1. Results of repeated measures ANOVA evaluating the variation of allochthonous litterfall inputs throughout the seasons for the studied streams. The degrees of freedom (d.f.), sum of squares (SS), mean squares (MS), test value (*F*), and test significance (*P*) are described. For paired comparations, see Table 1. (*) Statistical differences between seasons ($P < 0.05$).

Factors	d.f.	SS	MS	*F*	*P*
Native Stream					
Leaves					
Error: Stream collection point	2	0.428	0.214		
Seasons	3	1.993	0.664	6.0	0.002*
Residuals	30	3.297	0.1099		
Branches					
Error: Stream collection point	2	0.272	0.136		
Seasons	3	9.814	3.271	8.9	< 0.001*
Residuals	30	11.021	0.367		
Reproductive Parts					
Error: Stream collection point	2	3.371	1.686		
Seasons	3	11.830	3.943	9.1	< 0.001*
Residuals	30	12.980	0.433		
Total					
Error: Stream collection point	2	0.692	0.346		
Seasons	3	1.301	0.433	4.1	0.015*
Residuals	30	3.166	0.105		
Hovenia Stream					
Leaves					
Error: Stream collection point	2	0.479	0.239		
Seasons	3	5.397	1.798	3.6	0.025*
Residuals	30	15.139	0.504		
Branches					
Error: Stream collection point	2	1.399	0.699		
Seasons	3	7.769	2.589	5.4	0.004*
Residuals	30	14.465	0.482		
Reproductive Parts					
Error: Stream collection point	2	2.558	1.279		
Seasons	3	5.600	1.866	1.6	0.215
Residuals	30	35.48	1.183		
Total					
Error: Stream collection point	2	0.154	0.077		
Seasons	3	8.362	2.787	18.1	< 0.001*
Residuals	30	4.619	0.154		

Table S2. Results of repeated measures ANOVA evaluating the variation of allochthonous litterfall inputs from native tree species and *H. dulcis* in the *Hovenia stream*. The degrees of freedom (d.f.), sum of squares (SS), mean squares (MS), test value (*F*), and test significance (*P*) are described. For paired comparations, see Table 2. (*) Statistical differences between seasons ($P < 0.05$).

Factors	d.f.	SS	MS	*F*	*P*
Native Leaves					
Error: Stream collection point	2	1.662	0.803		
Seasons	3	6.251	2.084	2.9	0.052
Residuals	30	21.722	0.724		
Native Branches					
Error: Stream collection point	2	8.798	4.399		
Seasons	3	3.721	1.240	2.6	0.073
Residuals	30	14.523	0.484		
Native Reproductive Parts					
Error: Stream collection point	2	0.012	0.006		
Seasons	3	1.658	0.553	5.6	0.003*
Residuals	30	2.968	0.098		
Native Total					
Error: Stream collection point	2	4.200	2.100		
Seasons	3	9.183	3.061	10.6	< 0.001*
Residuals	30	8.652	0.288		
H. dulcis Leaves					
Error: Stream collection point	2	4.485	2.242		
Seasons	3	42.750	14.252	14.9	< 0.001*
Residuals	30	28.640	0.955		
H. dulcis Branches					
Error: Stream collection point	2	0.371	0.185		
Seasons	3	19.200	6.399	13.6	< 0.001*
Residuals	30	14.111	0.470		
H. dulcis Reproductive Parts					
Error: Stream collection point	2	2.382	1.191		
Seasons	3	5.170	1.722	1.5	0.242
Residuals	30	35.130	1.171		
H. dulcis Total					
Error: Stream collection point	2	2.946	1.476		
Seasons	3	39.810	13.271	24.3	< 0.001*
Residuals	30	16.400	0.547		

Table S3. Comparative allochthonous litterfall inputs (g m^{-2} month^{-1}; mean ± s.d.) of leaves, branches, reproductive parts, and total in the studied streams. The test value (t), degrees of freedom (d.f.), and test significance (P) are described. (*) Statistical differences between streams ($P < 0.05$).

Litterfall Fractions	*Native Stream*	*Hovenia Stream*	t test
Autumn (d.f. = 16)			
Leaves	51.4 ± 24.1	93.3 ± 67.0	$t = 1.06$, $P = 0.302$
Branches	2.7 ± 0.8	18.3 ± 9.4	$t = 8.55$, $P < 0.001$*
Reproductive Parts	0.8 ± 1.8	14.2 ± 30.5	$t = 2.04$, $P = 0.057$
Total	**54.9 ± 25.6**	**126.1 ± 56.7**	**$t = 3.57$, $P = 0.002$***
Winter (d.f. = 16)			
Leaves	63.5 ± 22.0	25.7 ± 7.7	$t = -5.26$, $P < 0.001$*
Branches	4.6 ± 1.9	5.5 ± 6.4	$t = -0.18$, $P = 0.857$
Reproductive Parts	0.1 ± 0.2	3.1 ± 3.6	$t = 2.75$, $P = 0.013$*
Total	**68.3 ± 21.7**	**34.3 ± 7.3**	**$t = -5.31$, $P < 0.001$***
Spring (d.f. = 16)			
Leaves	56.9 ± 23.3	45.7 ± 20.1	$t = -1.11$, $P = 0.281$
Branches	18.2 ± 15.7	9.8 ± 7.3	$t = -1.47$, $P = 0.158$
Reproductive Parts	3.7 ± 5.0	7.9 ± 9.4	$t = 1.00$, $P = 0.329$
Total	**78.8 ± 35.3**	**63.5 ± 21.9**	**$t = -1.09$, $P = 0.291$**
Summer (d.f. = 16)			
Leaves	32.3 ± 6.7	29.8 ± 13.0	$t = -0.90$, $P = 0.377$
Branches	8.9 ± 6.8	6.5 ± 4.0	$t = -0.57$, $P = 0.578$
Reproductive Parts	5.1 ± 4.2	0.9 ± 0.6	$t = -3.49$, $P = 0.002$*
Total	**46.3 ± 11.6**	**37.3 ± 13.5**	**$t = -1.52$, $P = 0.147$**
Year (d.f. = 70)			
Leaves	51.1 ± 22.7	48.6 ± 43.6	$t = -1.75$, $P = 0.084$
Branches	8.5 ± 10.2	10.1 ± 8.5	$t = 0.98$, $P = 0.329$
Reproductive Parts	2.4 ± 3.9	6.5 ± 16.2	$t = 1.65$, $P = 0.102$
Total	**62.1 ± 27.1**	**65.3 ± 47.9**	**$t = -0.62$, $P = 0.535$**

5. CAPÍTULO II

Effect of a non-native invasive species on leaf decomposition and associated invertebrates in subtropical Atlantic Forest streams

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A formatação das referências e de alguns pontos do manuscrito obedecem as normas deste periódico.

Abstract: Allochthonous litterfall input is the main energy source for small forested streams. The non-native and invasive tree *Hovenia dulcis* Thunb. (Rhamnaceae) occurs widely in forest remnants, in southern Brazil. We evaluated the effect of *H. dulcis* leaves on leaf decomposition and on the associated aquatic invertebrate community in three subtropical streams. We structured treatments where leaves of *H. dulcis* and *Nectandra megapotamica* (Spreng). Mez (Lauraceae) (native reference species) were offered individually (*Hovenia* vs. *Nectandra*) and associated with a mixture native species leaves (*Hovenia+mix* vs. *Nectandra+mix*). The leaves were inserted in litter bags, incubated in the streams and collected after 7 and 15 days (due the fast *H. dulcis* leaf decomposition). The leaf decomposition was ~2.8 times greater in *Hovenia* than *Nectandra* and ~1.3 times greater in *Hovenia+mix* than *Nectandra+mix* (but the leaf decomposition of native species associated not differed between both). The total invertebrates density, and shredders, collectors and predators densities were between 4 and 8 times higher in *Hovenia* than *Nectandra.* Richness, Shannon Index, evenness, and scrapers density differed between *Hovenia* and *Nectandra* but these variations were depend on streams and sample times. These metrics, at large, were similar between *Hovenia+mix* and *Nectandra+mix* and, when different, the variation was dependent on streams and sample times. The communities composition differed only between *Hovenia* and *Nectandra.* The *H. dulcis* leaves change the leaf decomposition (both individually and associated with native species) and the associated invertebrates (only individually) in the streams, but evident negative effects on these were not observed.

Key-words: *Hovenia dulcis*; biological invasions; exotic species; benthic macroinvertebrates; nutrients cycling.

Introduction

The functioning of ecosystems is related to the flow of matter and energy through their biotic and abiotic components (Diaz and Cabido 2001). In small forest streams where autochthonous productivity is limited, allochthonous litterfall input (i.e., leaves) is the main source of energy for the maintenance of ecosystem processes and aquatic communities (Vannote et al. 1980; Abelho 2001; França et al. 2009; Gonçalves et al. 2014). In streams, the leaves go through abiotic (e.g., physical abrasion and leaching) and biotic (e.g., colonization of microorganisms and invertebrates) processes that result in their transformation and incorporation into the aquatic trophic network (Abelho 2001; Graça 2001). The activity of microorganisms (e.g., aquatic hyphomycetes) partially decomposes plant tissues and increases the palatability and nutritional quality of leaves, which facilitates their consumption by shredders (Abelho 2001; Graça 2001; Marks 2019). The activity of the shredders contributes to the transformation of coarse particulate organic matter (CPOM) into fine particulate organic matter (FPOM) (Abelho 2001; Marks 2019). The FPOM is the main nutritional resource of a significant share of stream aquatic invertebrates (i.e., collectors) (Graça 2001; Cummins et al. 2005; Ramiréz and Gutiérrez-Fonseca 2014).

The structure (Souza et al. 2019) and composition (França et al. 2009; Gonçalves et al. 2014) of the riparian vegetation affects the diversity of leaf species entering streams. Leaves of different species usually have distinct structural and chemical characteristics (e.g., Bakker et al. 2011; Zukswert and Prescott 2017; López-Rojo et al. 2021; Ramos et al. 2021). Leaf characteristics affect (individually or together) the leaf decomposition in streams (Zhang et al. 2019) because they have a direct influence on the activity of microorganisms and invertebrates (Abelho 2001; Graça 2001; Gessner et al. 2010). In streams, leaf decomposition and the associated invertebrate community appear to be more affected by the identity (i.e., composition) than by the number (i.e., richness) of leaf species (see Swan and Palmer 2004; Lecerf et al. 2007; Moretti et al. 2007; Taylor et al. 2007; Abelho 2009; Ferreira et al. 2012; but see also Ferreira et al. 2016). Some characteristics make leaves more nutritious (e.g., higher nitrogen concentrations) and palatable (e.g., lower toughness), which facilitates, for example, their consumption by shredders (Rincón and Martínez 2006; Graça and Cressa 2010; Foucreau et al. 2013; Casotti et al. 2015) and results in higher decomposition rates (Quinn et al. 2000; Gonçalves et al. 2007; Li et al. 2009; Hepp and Pastore 2021). The interaction among leaves of different species with contrasting characteristics can accelerate or retard leaf decomposition in streams through chemical (e.g., nutrient exchange via leaching) or physical mechanisms (e.g., the combination of different leaves creates diversified microhabitats) (Gartner and Cardon 2004; Lecerf et al. 2007; Lecerf et al. 2011).

The replacement of native riparian vegetation caused by tree monocultures implantation or invasive non-native species of tree and shrubs invasions can change the quantity, composition, or temporal pattern of the allochthonous litterfall input to streams (Abelho and Graça 1996; Molinero and Pozo 2004; Mineau et al. 2012; Medina-Villar et al. 2015; Fontana et al. 2020; Weand et al. 2020). The influence of non-native species on leaf decomposition and on the structure and composition of the aquatic invertebrate community, however, does not show a clear pattern and varies widely (Castro-Díez and Alonso 2017). At some cases, even the influence of a same non-native species (e.g., *Eucalyptus*) on leaf decomposition (e.g., Ferreira et al. 2019) and on the structure and composition of the aquatic invertebrate community (e.g., Ferreira et al. 2015) may vary between different regions, which may be associated with different characteristics of native species leaves present in the riparian zone of local streams. In this context, studies point out some negative impacts caused by non-native species on leaf decomposition (e.g., comparatively slower decomposition) and on the structure of the aquatic invertebrate community (e.g., decreased abundance and richness, especially shredders) (Albariño and Balseiro 2002; Larranãga et al. 2009; Casas et al. 2013; Claeson et al. 2014; Larranãga et al. 2014; Little et al. 2021). These impacts, however, are usually more evident only in non-native species that have, when compared to native species, leaves with low palatability and nutrient concentration (see Casas et al. 2013; Martínez et al. 2013; Hepp and Pastore 2021).

Native from Asia, *Hovenia dulcis* Thunb. (Rhamnaceae) (Carvalho 1994) is a non-native tree species considered invasive in Brazil (Zenni and Ziller 2011). *H. dulcis* occurs widely in forest remnants in the subtropical region of Brazil (Dechoum et al. 2015; Padilha et al. 2015; Schmidt et al. 2020), including in the riparian vegetation of streams (König et al. 2014; Biasi et al. 2020). Studies evaluating the effect of *H. dulcis* on leaf decomposition and aquatic invertebrate community in streams are recent. In general, *H. dulcis* leaves have high palatability (e.g., low toughness) and, therefore, present rapid decomposition (König et al. 2014; Biasi et al. 2020; Hepp and Pastore 2021; Fontana et al. 2022). Furthermore, the presence of *H. dulcis* leaves in streams appears not to affect significantly the aquatic invertebrate community whose structure and composition are similar to native species with similar leaf characteristics (König et al. 2014; Hepp and Pastore 2021; Fontana et al. 2022). The existing knowledge about the influence of *H. dulcis* on leaf decomposition and aquatic invertebrate community focuses on the i) comparison between leaves of *H. dulcis* vs. leaves of native tree species individually

(König et al. 2014; Hepp and Pastore 2021) or ii) comparison between *H. dulcis* leaves individually vs. a mixture of leaves from native tree species (Biasi et al. 2020; Fontana et al. 2022). Although ecologically important, this approach disregards the effects of interaction among the leaves (i.e., *H. dulcis* with native species), which is relevant because unlike tree monocultures (predominance of one species) (Abelho and Graça 1996; Molinero and Pozo 2004) *H. dulcis* occurs associated with native tree species (Dechoum et al. 2015; Fontana et al. 2020). Furthermore, a recent study suggests that currently about of 7% the terrestrial area worldwide (including biodiversity hotspots) has potentially suitable habitats for the occurrence of *H. dulcis* (Bergamin et al. 2022). Thus, although inserted in the ecological context of subtropical streams of Atlantic Forest in Brazil, the findings of this study may be relevant and extrapolated to other regions of the world where *H. dulcis* can establish itself as a tree invasive species.

Thus, this study evaluated the leaf decomposition and the structure and composition of the aquatic invertebrate community associated on *H. dulcis* leaves that were available individually and together with native species leaves in subtropical streams. For this purpose, we structured four different leaf treatments where leaves of *H. dulcis* and *Nectandra megapotamica* (Spreng). Mez (Lauraceae) (a reference native species; a tree common in the studied region) were available, at same time, individually (treatments comparation: *Hovenia* vs. *Nectandra*) and associated with a mixture of native species leaves (treatments comparation: *Hovenia+mix* vs. *Nectandra+mix*). Thus, we expect that i) the *H. dulcis* leaves present a rapid decomposition and, consequently, accelerate the decomposition of native species leaves associated with it (i.e., leaf decomposition will be higher in *Hovenia* than *Nectandra* and in *Hovenia+mix* than *Nectandra+mix*); ii) *Hovenia* (when compared to *Nectandra*) and *Hovenia+mix* (when compared to *Nectandra+mix*) present the higher density of invertebrates, especially shredders (due to *H. dulcis* leaves high palatability) and collectors (due to greater availability of FPOM originated from leaf decomposition of *H. dulcis*) and, as result, also present a higher invertebrates diversity and a different taxonomic composition.

Material and Methods

Study Area

This study was carried out in three low order streams (<3rd-order) located in Rio Grande do Sul state, southern Brazil (Fig. S1) during the Summer (January and February) of 2020. The study region is inserted in the Atlantic Forest domain (Oliveira-Filho et al. 2015) at 400 - 800 meters above the sea level (Alvares et al. 2013) on the Serra Geral formation. In the studied region, the native forest remnants are composed by deciduous, semi-deciduous and perennial tree species (Athayde et al. 2009) with the occurrence of ~ 60 - 70 tree species and ~20 - 30 botanical families, with Fabaceae, Lauraceae, Meliaceae, Myrtaceae and Sapindaceae being especially abundant and diverse (for more details see Leyser et al. 2012; Loregian et al. 2012; Mélo et al. 2013 and Turchetto et al. 2015). The region climate is classified as humid subtropical (Köppen classification Cfb) with mean annual temperature ranging from 16 to 18°C and precipitation from 1900 to 2200 mm/year (Alvares et al. 2013).

The studied streams are located in forest remnants predominantly composed by native tree species (personal observation), have a predominantly rocky bed and are morphologically similar in terms of bed width (range: 1 - 4 meters) and depth (range: 0.05 - 0.20 meters) in the stretch (~100 meters) where the litter bags were incubated. In the streams, throughout the study period, the temperature (range: 18.8 - 22.3°C), electrical conductivity (range: 76 - 170 μS.cm^{-1}), pH (range: 6.2 - 7.5), dissolved oxygen (range: 8.3 - 8.9 mg.L^{-1}), total dissolved solids (range: 51 - 112 μ.L^{-1}) and turbidity (range: 7.0 - 19.9 UNT) were measured with a multiparameter analyzer (Table S1). The total dissolved nitrogen concentration (range: 1.19 - 2.18 mg.L^{-1}) was also measured (Table S1). During the study period, rainfall was 49.4 mm (Instituto Nacional de Meteorologia 2020).

Selection and Structural and Chemical Characterization of Leaves

Leaves of *H. dulcis* and native species with contrasting leaf characteristics and decomposition rates (Hepp and Pastore 2021) were selected to represent part of the diversity found in the native forest remnants of the study region. Tree leaves with decomposition considered slow (*Cryptocarya aschersoniana* Mez (Lauraceae) and *Inga marginata* Willd. (Fabaceae)), intermediate (*Cedrela fissilis* Vell. (Meliaceae) and *Nectandra megapotamica* (Spreng.) Mez (Lauraceae)) and fast (*Eugenia uniflora* L. (Myrtaceae), *Handroanthus heptaphyllus* (Vell.) Mattos (Bignoniaceae)) (see Hepp and Pastore 2021) were selected and collected (Table S2). These leaf species belong to tree families commonly found in the mixed ombrophilous forest and/or seasonal deciduous forest (Vibrans et al. 2013a; Vibrans et al. 2013b) and have been reported in forest remnants of the subtropical region of Brazil where our study was carried out (Piroli and Nascimento 2008; Santos et al. 2012; Capellesso et al. 2013; Mélo et al. 2013).

The leaves of each species were dried at room temperature (~20º C) for about 15 days and then structurally and chemically characterized (Table S2). For the structural characterization, we measured: toughness (expressed in g; method adapted from Graça and Zimmer 2005), thickness (mm; measurement of leaf discs with 12 mm in diameter using a digital caliper) and specific leaf area ($mm^2.mg^{-1}$; area and weight of leaf discs with 12 mm in diameter). Chemically, we measured: carbon (%, sample incineration), nitrogen (%, adapted from Flindt and Lillebø 2005), phosphorus ($mg.g^{-1}$; adapted from Flindt and Lillebø 2005) and polyphenols ($mg.g^{-1}$; adapted from Bärlocher and Graça 2005) concentrations. For *Hovenia+mix* and *Nectandra+mix* (see below), we considered the mean of structural and chemical leaf characteristics of the species included in each treatment.

Preparation of Treatments and Field Procedures

Litter bags were prepared with four treatments, where leaves of *H. dulcis* and *N. megapotamica* (a native reference species) were incubated individually and associated with a mixture of native species leaves (Fig. 1). The treatment *Hovenia* was composed only by leaves of *H. dulcis* (3.0 ± 0.1 g), and the treatment *Nectandra*, only by leaves of *N. megapotamica* (3.0 ± 0.1 g) (Fig. 1). The *Hovenia+mix* treatment was composed by leaves of *H. dulcis* and a mixture of native species leaves (0.5 ± 0.1 g of each species; 3.0 ± 0.1 g of leaves in total), whereas the *Nectandra+mix* treatment was composed by leaves of *N. megapotamica* and a mixture of native species leaves (0.5 ± 0.1 g of each species; 3.0 ± 0.1 g of leaves in total) (Fig. 1). The mixture of native species was composed of *C. fissilis, C. aschersoniana, E. uniflora, H. heptaphyllus* and *I. marginata* (Fig. 1). *N. megapotamica* was defined as the reference native species based on its occurrence in the native forest remnants in the studied region and on its leaf characteristics. First, *N. megapotamica* is a common and comparatively more abundant native tree in the forest remnants of the studied region in relation to the other selected native species (see Piroli and Nascimento 2008; Leyser et al. 2012; Loregian et al. 2012; Mélo et al. 2013). Second, a Principal Component Analysis (PCA) was used, where the native species were ordered considering leaf characteristics (data were standardized due to their distinct nature) and

N. megapotamica presented intermediate leaf characteristics in relation to the other selected native species (Fig. 2).

The treatments were placed in 72 coarse-mesh litter bags (10 x 20 cm; 10 mm mesh opening) and incubated in the streams. Twenty-four litter bags were randomly incubated in each stream (6 litter bags of each treatment) (Fig. 1). After 7 and 15 days of immersion, three litter bags of each treatment were removed from the streams, placed in plastic bags, and transported to the laboratory. Considering the high decomposition rates of *H. dulcis* (König et al. 2014; Biasi et al. 2020; Hepp and Pastore 2021; Fontana et al. 2022), we defined the last sampling time at 15 days to ensure that ~50% of the weight would remain in the *Hovenia* treatment.

Fig. 1 Flow diagram of the study experimental design. The composition, availability, and diversity of resources and, the number of litter bags of each treatment and litter bags incubated in each stream are described. Ced = *Cedrela fissilis*, Cry = *Cryptocarya aschersoniana*, Eug = *Eugenia uniflora*, Han = *Handroanthus heptaphyllus*, Ing = *Inga marginata*, Nec = *Nectandra megapotamica* and Hov = *Hovenia dulcis.*

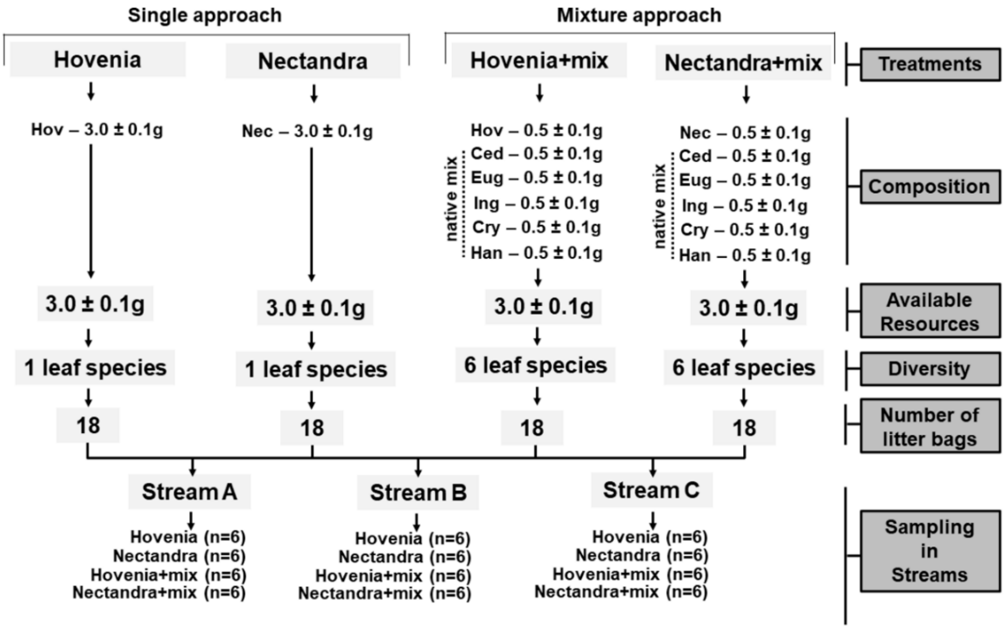
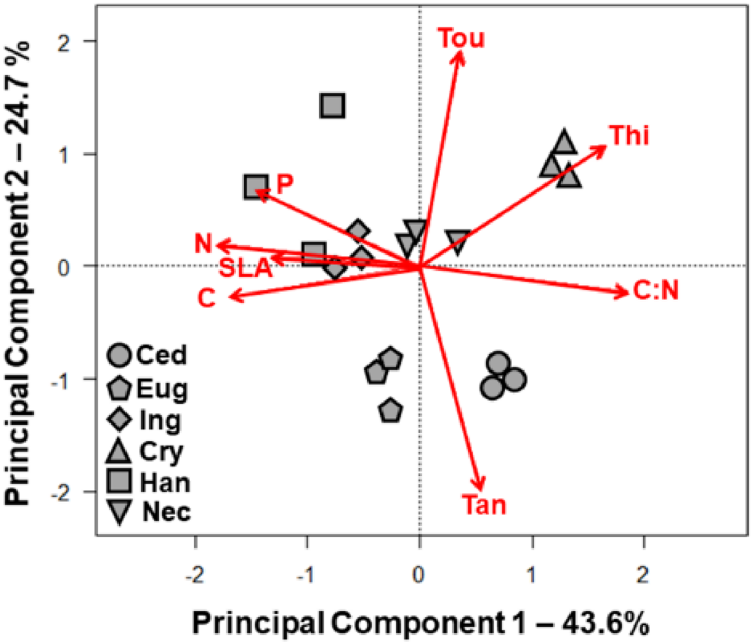


Fig. 2 Principal Component Analysis demonstrating the ordering of native species in relation to the structural and chemical characteristics of the leaves. Ced = *Cedrela fissilis*, Cry = *Cryptocarya aschersoniana*, Eug = *Eugenia uniflora*, Han = *Handroanthus heptaphyllus*, Ing = *Inga marginata*, Nec = *Nectandra megapotamica*. Thi = Thickness; Tou = Toughness; SLA = Specific Leaf Area; C = Carbon; N = Nitrogen; C:N = Carbon:Nitrogen; Tan = Tannins; P = Phosphorous.



Laboratorial Procedures

In the laboratory, the leaves were gently washed in a tray with water to remove the sediment and associated invertebrates. The washing water was passed through a 250-µm mesh sieve to retain the invertebrates. The leaf litter was dried in an oven with air circulation (40 ± 5° C/72 h) and, subsequently, weighed in an analytical balance to determine the remaining mass and decomposition rates. In the *Hovenia+mix* and *Nectandra+mix* treatments, the mass loss and decomposition rates of the native species composing the mixture of natives leaves were also measured individually.

The invertebrates were fixed in 70% ethanol and sorted, counted and identified to the taxonomic level of family (for Arthropoda) and class (for Annelida and Mollusca) with a stereomicroscope (up to 80x magnification) according to the key proposed by Mugnai et al. (2010). The organisms were further classified into their respective functional feeding groups (shredders, collectors-gatherers, collectors-filterers, scrapers and predators) according to the specific literature (Cummins et al. 2005; Ramírez and Gutiérrez-Fonseca 2014). In the data analysis, we chose to consider collectors-gatherers and collectors-filterers together (as collectors), since both consume the same food items (e.g., FPOM) suspended in the water column or deposited in depositional areas (Cummins et al. 2005).

Data Analysis

The leaf decomposition was determined using the leaves remaining weight (in percentage) measured at days 7 and 15. The leaf decomposition was estimated, first, for each treatment (for that, in the *Nectandra+mix* and *Hovenia+mix* treatments, the remaining weight of all species that composed each one treatments were considered together). Afterwards, were also measured the leaf decomposition for each native species of the mixture of natives in the *Nectandra+mix* and *Hovenia+mix* treatments (for that, in each treatment, the leaves of each one native species were individually weighed). To evaluate the invertebrate community structure, density data (individuals.g^{-1}) were used to reduce the effect of variation in final remaining weight among treatments (range: 4 - 92 %) and rarefied richness (based on lower abundance) to reduce the effect of variation of the abundance of invertebrates in litter bags (range: 27 - 496 individuals per litter bag). Additionally, the Shannon Index (H') and evenness (J) were calculated (based on the ratio between H' and the natural log of richness) using the PAST statistical software (version 4.04) (Hammer, Harper and Ryan 2001). Both metrics were considered because they allow to evaluate the uniformity of the aquatic invertebrate community (Melo and Hepp 2008).

As to the statistical tests, the data were tested regarding the assumptions of normality (Shapiro-Wilk test) and homoscedasticity (Levene's test). When necessary, the data that did not present normal distribution were transformed into ln(x+1), aiming at the best possible fit for the performance of parametric statistical tests. To test the proposed hypotheses, the statistical tests used were concentered on the comparison between treatments: first between *Hovenia* vs. *Nectandra* and then between *Hovenia+mix* vs. *Nectandra+mix*. To test our first hypothesis, an Analysis of Covariance (two-way ANCOVA) was used to assess the variation in remaining weight between treatments and streams (with time as cofactor). Afterwards, we were used the same test to assess the variation in remaining weight of each species of the mixture of natives between the treatments (i.e., between *Hovenia+mix* and *Nectandra+mix*) and streams (with time as cofactor). Additionally, a test-t was used to test the variation of leaf characteristics between treatments. Finally, simple linear regressions were used to evaluate the influence of leaf characteristics on the final remaining weight of the four treatments (we choose not to use multiple linear models due to the high collinearity between different leaf characteristics). To test our second hypothesis, a three-way ANOVA was used to assess the variation in invertebrate density, rarefied richness, Shannon Index, evenness, and density of each functional feeding group between treatments, streams and sample times. Moreover, a Multivariate Analysis of Homogeneity of Groups Dispersions (PermDISP), and a Permutational Multivariate Analysis of Variance (PerMANOVA) were used to test the invertebrate community distribution within each treatment, stream and, sample times and between treatment, stream and, sample times respectively. Subsequently, associated both, a Non-Metric Multidimensional Scaling Analysis (nMDS) was used to verify the invertebrate community distribution between treatments, streams and sample times from a Bray-Curtis similarity matrix considering the abundance of each taxon. For PermDISP, nMDS and PerMANOVA analysis, the abundance was log+1 transformed to minimize the effect of very abundant taxa.

The statistical analyses were performed in the statistical software R (R Core Team 2019) using the packages “vegan” (Olksanen et al. 2019), "stats" (R Core Team 2019), "car" (Fox and Weisberg 2019) and “carData” (Fox et al. 2018). To create the graphics, we used the Microsoft Excel (2021), PAST version 4.04 (Hammer, Harper and Ryan 2001) and R (R Core Team 2019) software.

Results

Leaf Decomposition

The leaf decomposition was different between treatments (Fig. 3; Table S3). *Hovenia* (27.8 ± 14.2%; mean ± standard deviation) presented, on average, a remaining weight ~2.8 times lower ($F_{(1;47)} = 54.7$; $P < 0.001$) than *Nectandra* (79.4 ± 5.7%) (Fig. 3a; Table S3). At the same time, *Hovenia+mix* (41.7 ± 12.5%) presented, on average, a remaining weight ~1.3 times lower ($F_{(1;47)} = 11.1$; $P = 0.002$) than *Nectandra+mix* (55.3 ± 7.3%) (Fig. 3b; Table S3). Despite this, the species that composed the mixture of natives, individually, presented similar leaf decomposition between *Hovenia+mix* and *Nectandra+mix* (Fig. 3c; Table S3).

The leaf characteristics differed between the treatments (Table 1). *Hovenia* presented lower thickness, nitrogen e phosphorous concentrations and higher SLA and C:N ratio when compared with *Nectandra* (Table 1). Similarly, *Hovenia+mix* presented lower thickness and nitrogen concentrations and higher C:N ratio when compared with *Nectandra+mix* (Table 1).

The linear models indicated that the leaf decomposition (remaining mass) of the treatments were significantly influenced ($P < 0.05$) by all leaf characteristics measured with the exception of tannin concentration (Table 2). The most influential leaf characteristics ($R^2_{adj} > 0.60$) on the leaf decomposition of the treatments were thickness (straight slope (t) = 8.4; $R^2_{adj} = 0.67$), specific leaf area ($t = - 10.4$; $R^2_{adj} = 0.75$) and phosphorous concentration ($t = 7.9$; $R^2_{adj} = 0.64$) (Table 2).

Fig. 3 Final remaining weight (% at day 15) (a) for *Hovenia* vs. *Nectandra*, (b) for *Hovenia+mix* vs. *Nectandra+mix* and, (c) for species constituting of the mixture of natives. Ced = *Cedrela fissilis*, Cry = *Cryptocarya aschersoniana*, Eug = *Eugenia uniflora*, Han = *Handroanthus heptaphyllus*, Ing = *Inga marginata*; HM = *Hovenia+mix*, NM = *Nectandra+mix.* Different letters indicate the statistical differences between treatments ($P < 0.05$). For statistical details, see Table S3.

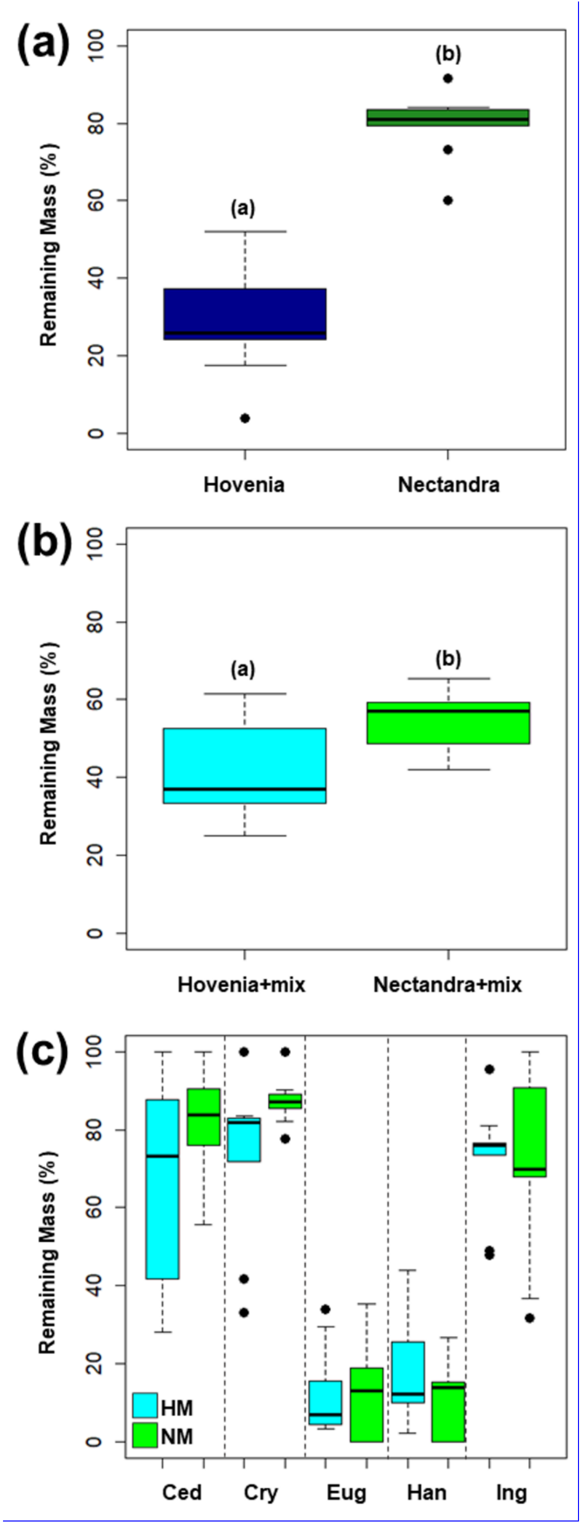


Table 1 Structural and chemical characterization (mean ± standard deviation) of the leaf treatments studied. In the *Hovenia+mix* and *Nectandra+mix* treatments, the value of each structural and chemical variable was obtained by the mean of each species composing the treatment (for the structural and chemical characterization of each species, see Table S3). The test value (t), the degrees of freedom (df) and the test significance (P) are described.

Variables	Hovenia	Nectandra	t test ($df = 4$)	Hovenia+mix	Nectandra+mix	t test ($df = 4$)
Thi (mm)	0.15 ± 0.01	0.24 ± 0.01	$t = -9.3$; $P < 0.001$***	0.20 ± 0.01	0.21 ± 0.01	$t = -5.3$; $P = 0.006$**
Tou (g)	224.0 ± 170.4	299.6 ± 13.4	$t = -1.0$; $P = 0.360^{ns}$	314.8 ± 55.0	327.4 ± 24.4	$t = -0.3$; $P = 0.756^{ns}$
SLA (mm^2.mg^{-1})	15.3 ± 0.24	10.64 ± 0.51	$t = 3.7$; $P = 0.021$*	13.5 ± 1.99	12.77 ± 1.87	$t = 0.4$; $P = 0.672^{ns}$
C (%)	43.15 ± 0.34	44.29 ± 1.00	$t = -1.8$; $P = 0.134^{ns}$	44.10 ± 0.10	44.29 ± 0.21	$t = -1.4$; $P = 0.233^{ns}$
N (%)	1.11 ± 0.03	1.55 ± 0.03	$t = -20.0$; $P < 0.001$***	1.51 ± 0.02	1.58 ± 0.02	$t = -3.9$; $P = 0.017$*
C:N	38.95 ± 1.01	28.65 ± 0.72	$t = 15.4$; $P < 0.001$***	29.19 ± 0.49	27.97 ± 0.58	$t = 2.9$; $P = 0.041$*
Tan (mg.g^{-1})	0.43 ± 0.06	0.47 ± 0.02	$t = -1.0$; $P = 0.352^{ns}$	0.52 ± 0.01	0.53 ± 0.01	$t = -1.2$; $P = 0.286^{ns}$
P (mg.g^{-1})	4.82 ± 0.20	5.54 ± 0.28	$t = -3.5$; $P = 0.023$*	5.26 ± 0.25	5.38 ± 0.26	$t = -0.6$; $P = 0.604^{ns}$

Thi = Thickness; Tou = Toughness; SLA = Specific Leaf Area; C = Carbon; N = Nitrogen; C:N = Carbon:Nitrogen; Tan = Tannins; P = Phosphorous.
Asterisks indicate the significance: $P \leq 0.001$ (***), $P \leq 0.01$ (**), $P \leq 0.05$ (*), $P > 0.05$ (^{ns}).

Table 2 Linear regression models evaluating the influence of leaf characteristics on the leaf decomposition (i.e., final remaining weight at day 15) of the treatments. The degrees of freedom (*df*), the slope of the line (*t*), the regression value (*F*), the explanation of the adjusted model (R^2_{adj}) and the significance of the test (*P*) are described.

Leaf characteristics (*df* = 1;34)	*t*	*F*	R^2_{adj}	*P*
Thickness	8.4	72.0	0.67	<0.001***
Toughness	3.3	11.2	0.22	0.002**
Specific Leaf Area	- 10.4	108.0	0.75	<0.001***
Carbon	5.6	31.1	0.46	<0.001***
Nitrogen	4.9	23.8	0.39	<0.001***
Carbon : Nitrogen	- 4.8	22.8	0.38	<0.001***
Tannins	1.2	1.4	0.01	0.249^{ns}
Phosphorous	7.9	63.2	0.64	<0.001***

The asterisks indicate the significance: $P \leq 0.001$ (***), $P \leq 0.01$ (**), $P \leq 0.05$ (*), $P > 0.05$

Invertebrates

A total of 10022 aquatic invertebrates associated with the leaf treatments were identified (Table S4). Chironomidae (3250 individuals; ~32% of the total); Leptophlebiidae (1834 ind.; ~18%) and Gastropoda (1234 ind.; ~12%) were the most abundant taxa which represent, together, ~62% of the total of aquatic invertebrates sampled (Table S4). Comparatively, *Hovenia* (325.7 ± 753.7 ind.g^{-1}; mean ± standard deviation) presented a total invertebrates density ~7 times higher ($F_{(1;24)} = 45.8$; $P < 0.001$) than *Nectandra* (45.1 ± 46.8 ind.g^{-1}) (Fig. 4a; Table S5). Rarefied richness ($F_{(2;24)} = 7.8$; $P = 0.002$), Shannon Index ($F_{(2;24)} = 10.7$; $P < 0.001$) and evenness ($F_{(2;24)} = 4.6$; $P = 0.002$) also were differed between *Hovenia* and *Nectandra* but this variation depended on the stream and sample time considered (interaction treatment:stream:time) (Fig. 4b to 4d; Table S5). At the same time, the total invertebrates density, rarefied richness, Shannon Index, and evenness were similar between *Hovenia+mix* and *Nectandra+mix* (Fig. 4a to 4d; Table S5).

Considering the aquatic invertebrates trophic structure, the collectors were the most abundant functional feeding group (7186 individuals; ~72% of the total of invertebrates), followed by scrapers (1412 ind.; ~14%), shredders (1119 ind.; ~11%) and predators (305 ind.; ~3%) (Table S4). Chironomidae (3250 ind.; ~45% of the collectors), Gastropoda (1234 ind.; ~87% of the scrapers), Calamoceratidae (745 ind.; ~66% of the shredders) and Calopterygidae (72 ind.; ~24% of the predators) were, respectively, the most representative taxa of each functional feeding group (Table S4). The shredders density was ~ 4.6 times higher ($F_{(1;24)} = 21.4$; $P < 0.001$) in *Hovenia* (24.6 ± 32.8 ind.g^{-1}) than in *Nectandra* (5.3 ± 5.0 ind.g^{-1}) and the collectors density was ~ 8.6 times higher ($F_{(1;24)} = 42.0$; $P < 0.001$) in *Hovenia* (268.4 ± 667.1 ind.g^{-1}) than in *Nectandra* (31.1 ± 38.4 ind.g^{-1}) (Fig. 5a; 5b; Table S5). The scrapers density was higher in *Hovenia*, but this variation depended on the streams (interaction treatment:stream) and on the sample time considered (interaction treatment:time) (Fig. 5c; Table S5). The predators density was ~5.7 times higher ($F_{(1;24)} = 8.1$; $P = 0.008$) in *Hovenia* (9.1 ± 23.9 ind.g^{-1}) than in *Nectandra* (1.6 ± 1.5 ind.g^{-1}) (Fig. 5d; Table S5). At the same time, shredders, collectors and predators densities were similar between *Hovenia+mix* and *Nectandra+mix* (Fig. 5a; 5b; 5d; Table S5). The scrapers density was higher ($F_{(2;24)} = 6.5$; $P = 0.005$) in *Hovenia+mix*, but this variation depended on the stream and sample time considered (interaction treatment:stream:time) (Fig. 5c; Table S5).

The invertebrates communities composition differed between *Hovenia* and *Nectandra* ($F_{(1;35)} = 4.0$; $P = 0.007$) but, despite this, the main factor of variation was the streams ($R^2 = 0.43$) (Fig. 6a; Table S6). At the same time, the invertebrates communities composition was similar between *Hovenia+mix* and *Nectandra+mix* (Fig. 6b; Table S6).

Fig. 4 (a) Invertebrate density (ind g^{-1}); (b) rarefied richness (taxa), (c) Shannon Index and (d) evenness observed between the treatments. Different letters indicate the statistical differences between treatments ($P < 0.05$). Note the adjustment (-----) on the y-axis scale in *a*. For statistical details, see Table S5.

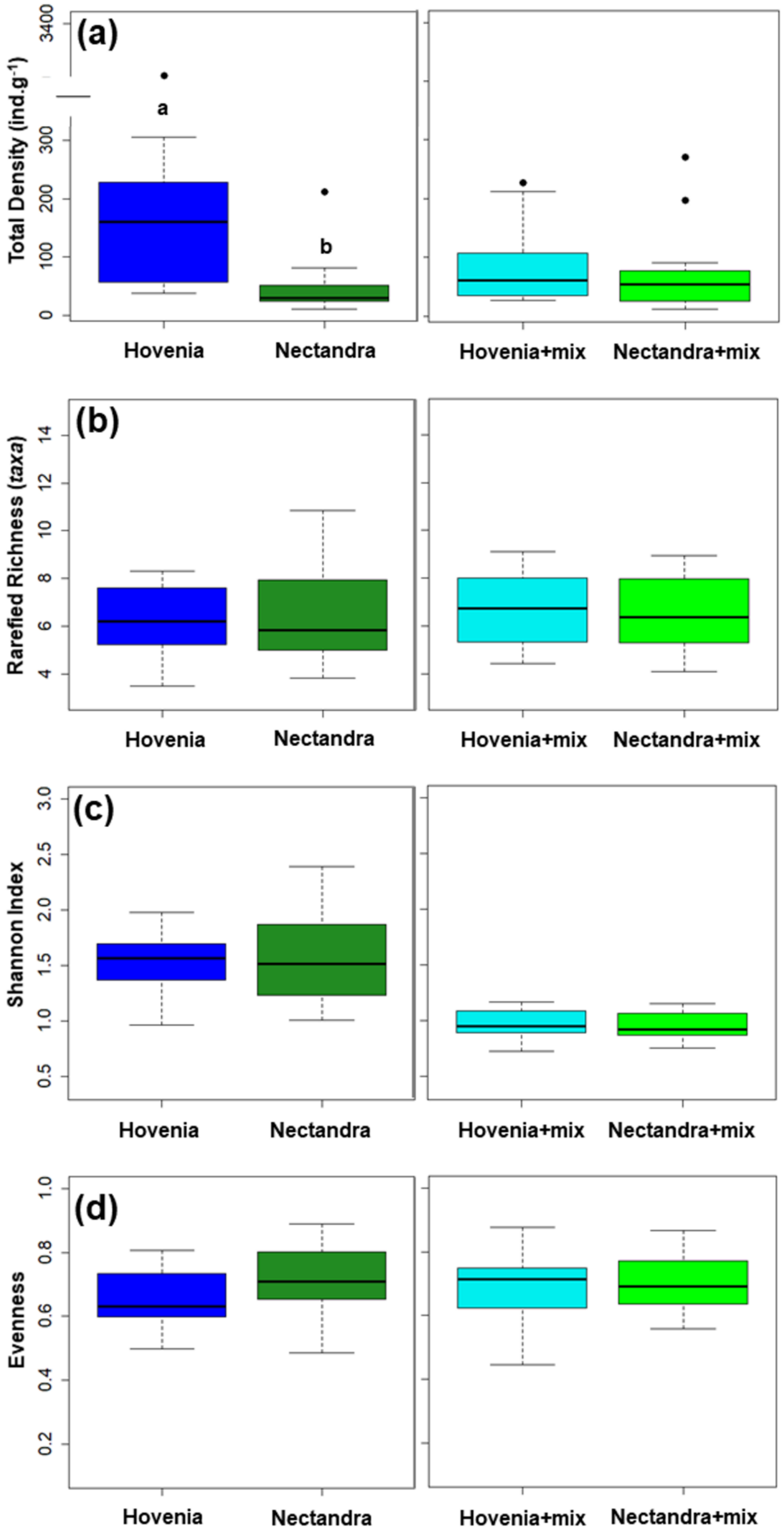


Fig. 5 Density (ind g^{-1}) of (a) shredders, (b) collectors, (c) scrapers and (d) predators observed between the treatments. Different letters indicate the statistical differences between the treatments ($P < 0.05$). Note the adjustment (-----) on the y-axis scale in *a, b,* and *d*. For statistical details, see Table S5.

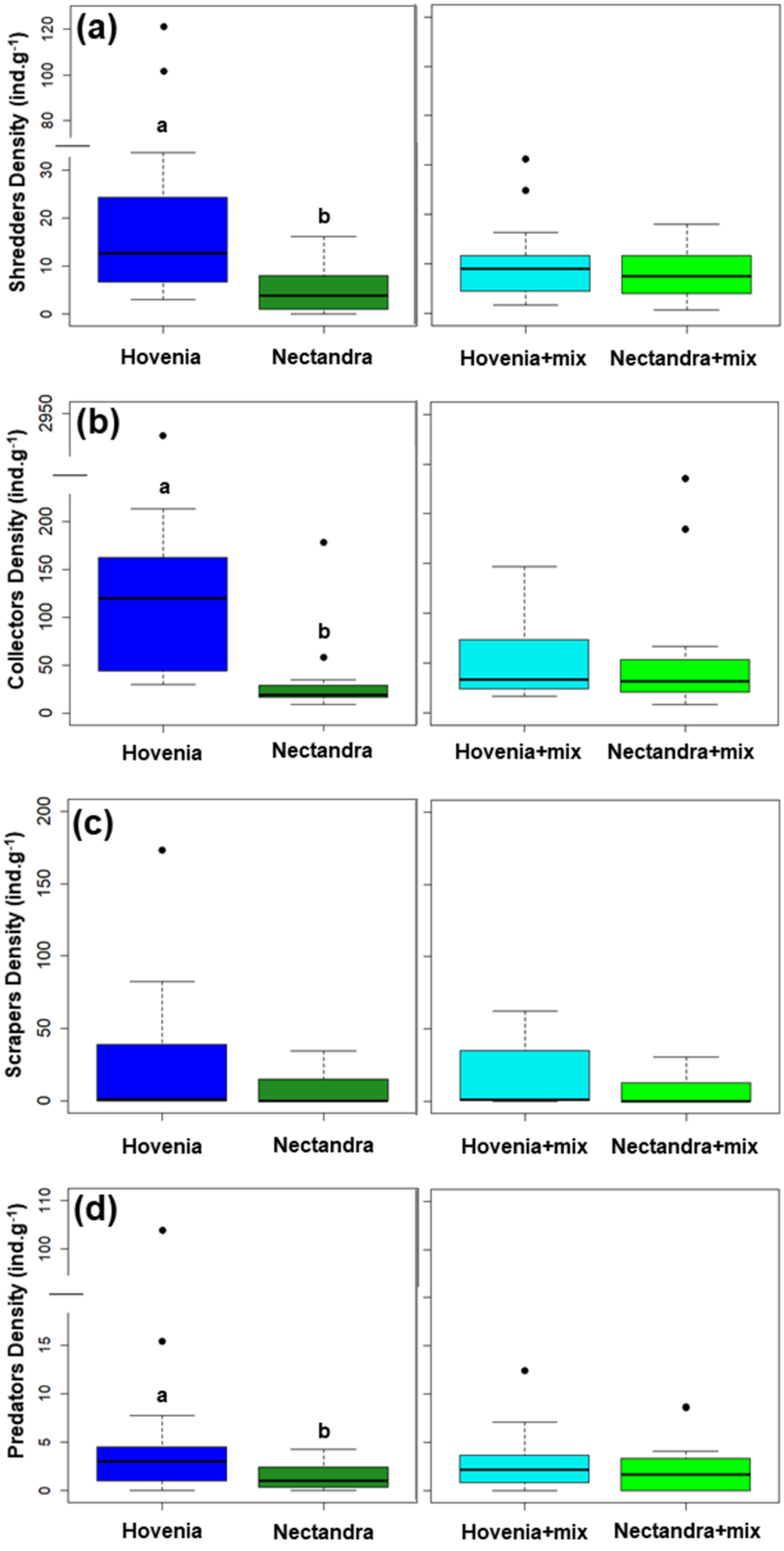
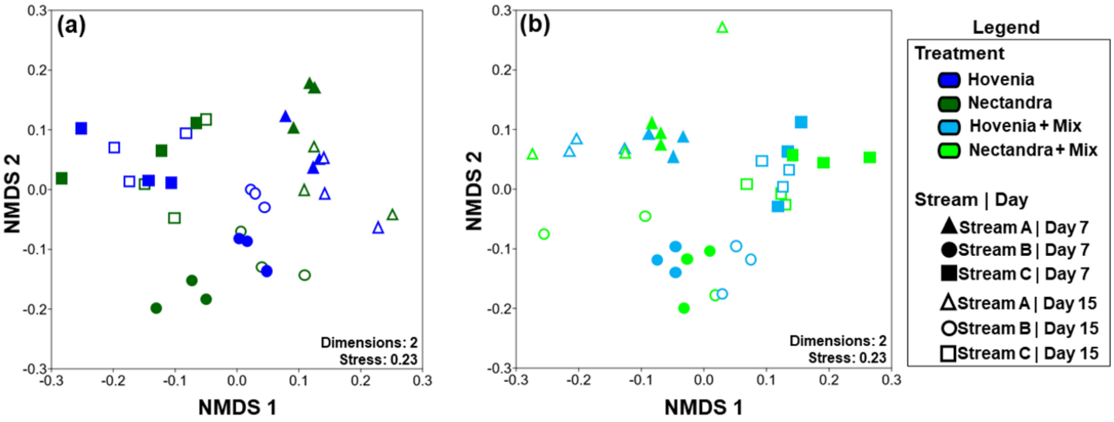


Fig. 6 Non-Metric Multidimensional Scaling (nMDS) ordering based on the taxonomic composition of invertebrates associated with each treatment in the different sample times (*7* and *15*) and in the three streams studied (*A*, *B* and *C*). For statistical details, see Table S6.



Discussion

Our first hypothesis was only partially supported because the leaf decomposition of *Hovenia* was higher than *Nectandra* and *Hovenia+mix* was higher than *Nectandra+mix,* however, the leaf decomposition of the species that composed the mixture of natives was similar between *Hovenia+mix* and *Nectandra+mix*. Our second hypothesis was few supported: *Hovenia* presented a higher invertebrate density (i.e., shredders and collectors) than *Nectandra* and the invertebrates community composition differed between both, however, we did not observe a diversity pattern clearly higher in *Hovenia* than *Nectandra.* Moreover, these same metrics was similar between *Hovenia+mix* and *Nectandra+mix*.

Leaf Decomposition

The leaf decomposition rates differed between treatments, where *Hovenia* > *Nectandra* and *Hovenia+mix* > *Nectandra+mix*. The high leaf decomposition rates of *Hovenia* corroborate previous studies where *H. dulcis* showed rapid decomposition with a loss of ~40 to 50% of weight in the first two weeks (König et al. 2014; Hepp and Pastore 2021; Fontana et al. 2022). The leaf decomposition of the species that composed the mixture of natives was similar between *Hovenia+mix* and *Nectandra+mix*. Thus, the highest leaf decomposition observed in *Hovenia+mix* was generated, probably, only due the higher leaf decomposition rates of *H. dulcis* when compared to *Nectandra megapotamica*. The similar leaf decomposition of the species in the mixture of natives may have been caused by different factors (individually or in combination). First, it is possible that the representativeness of *H. dulcis* (~16%) in the mixture of leaves was insufficient to accelerate leaf decomposition of native species with intermediate and slow decomposition. In fact, studies indicate that some species can affect the leaf decomposition of others, but this seems to be more common in leaf mixtures where species representativeness is, comparatively, higher (i.e., mixtures with fewer leaf species) (see Swan and Palmer 2004; Kominoski et al. 2007). Second, it is also possible that, in *Hovenia+mix*, the occurrence of three leaf species with rapid leaf decomposition (*E. uniflora*, *H. heptaphyllus* and *H. dulcis*) and three species with intermediate or slow leaf decomposition (*C. fissilis* and *I. marginata*, *C. aschersoniana*) (Hepp and Pastore 2021) has mutually nullified the potential effects of one species over another on their leaf decomposition (Schindler and Gessner 2009).

The linear models indicated that thickness, specific leaf area and phosphorus concentration were the leaf characteristics that most affected leaf decomposition. Plant tissue toughness is the leaf structural characteristic most commonly considered in studies about leaf decomposition in streams. Thus, as they are considered in fewer studies, the influence of thickness and specific leaf area on leaf decomposition in streams is not clear, although it has been reported in studies carried out in terrestrial ecosystems (see Santiago 2007; Bakker et al. 2011; Zukswert and Prescott 2017). In terrestrial ecosystems, thicker leaves can delay mass loss via leaching and hinder leaf colonization and processing by microorganisms and invertebrates (Zukswert and Prescott 2017). Furthermore, leaves with larger specific leaf areas can are more cost-effective for the consumers who can consume the leaves more efficiently, processing less leaf material per area or volume unit (Bakker et al. 2011). It is not possible to determine, however, whether this also applies to leaf decomposition in streams (although, for thickness, see Cornut et al. 2015). However, if so, leaves that are less thick and have a larger specific area would present higher leaf decomposition rates, which explains our results. Thus, leaf thickness and specific leaf area can be relevant and easy-to-measure leaf characteristics for understanding leaf decomposition in streams. Regarding nutrients, the initial concentration of nitrogen and phosphorus present in leaves is generally positively associated with leaf decomposition in streams (see Lecerf and Chauvet 2008; López-Rojo et al. 2021), as it affects the activity of microorganisms and invertebrates. In our study, however, we observed a negative relationship between initial phosphorus concentrations and leaf decomposition (also observed in Ferreira et al. 2012) which may be associated with a rapid decrease in phosphorus concentrations in the early stages of leaf decomposition (Tonin et al. 2014).

Invertebrates

The structure and composition of the invertebrates community differed only between *Hovenia* and *Nectandra*. In this context, possibly the similarity between *Hovenia+mix* and *Nectandra+mix* may be explained by, as well as for the leaf decomposition, the low representativity of *H. dulcis* and *N. megapotamica* in the mixture of species (i.e., in this proportion, both species were not able to generate differences).

The density of invertebrates, especially of shredders and collectors, was higher in *Hovenia* than *Nectandra.* The higher density of shredders in *Hovenia* possibly results from its greater palatability compared to *Nectandra.* When compared with *Nectandra*, *Hovenia* presented lesser thickness, higher SLA and also lower concentrations of nitrogen (higher C:N ratio) and phosphorus. These leaf characteristics, considered together, may make *H. dulcis* leaves more palatable (e.g., lesser thickness) (see Cornut et al. 2015) but, at the same time, them less nutritional (e.g., lower concentrations of nitrogen and phosphorus) (see Graça 2001; Rincón and Martínez 2006) for shredders. In this context, although our results support the assumption that *H. dulcis* leaves have high palatability (König et al. 2014), they indicate that this does not always reflect high nutritional levels. This finding is ecologically relevant as it indicates that the influence of *H. dulcis* on leaf decomposition and, especially, on associated invertebrates may vary spatially due to intraspecific variation in the leaf characteristics of the species (see Lecerf and Chauvet 2008). The higher collectors density in *Hovenia* corroborates König et al. (2014) and, was possibly associated with greater availability of FPOM (they main food resource) resulting from fast *H. dulcis* leaf decomposition (Graça 2001; Cummins et al. 2005).

The rarefied richness, Shannon Index and evenness differed between *Hovenia* and *Nectandra,* but this variation does not seem to be a clear effect of *H. dulcis* leaves because it was dependent on the interaction with streams and sample time. In a same pattern, the invertebrates communities composition differed between *Hovenia* and *Nectandra*, but the major source of variation was the streams and not the treatments. In this context, the statistical interaction with streams and sample times, indicates that these changes were more influenced by factors intrinsic to each stream and by the temporal variation of invertebrates colonization in submerged leaves. In fact, we reflect if it the taxonomic level used to identify invertebrates (i.e., family) may have influenced the similarity of diversity patterns observed between *Hovenia* and *Nectandra*. Studies indicate, however, that the general diversity patterns observed in aquatic invertebrate communities identified at the family taxonomic level often reflect the same general diversity patterns observed for lower taxonomic levels (e.g., genus) (Melo 2005; Heino and Soininen 2007; Brito et al. 2018). Thus, the similarity of the general diversity patterns observed in our study possibly would be maintained even with a more accurate taxonomic identification. Corroborating this hypothesis, a previous study found that the taxonomic diversity of associated aquatic invertebrates was similar in leaves of native species and *H. dulcis* (both with similar leaf characteristics) even with the identification of some organisms at the taxonomic level of genus (König et al. 2014).

Final Remarks and Future Perspectives

In this study, the presence of *H. dulcis* leaves, individually or associated with other native species, did not negatively affect the leaf decomposition and the structure and composition of associated aquatic invertebrates, corroborating and complementing previous studies with *H. dulcis* (König et al. 2014; Hepp and Pastore 2021; Fontana et al. 2022).

In this context, although relevant, our results should be viewed with caution and do not exclude the possibility of the presence of *H. dulcis* in riparian vegetation changing the functioning of streams. A several questions ecologically relevant for the functioning of streams remain unknown or little explored for *H. dulcis***.** First, in *Hovenia+mix* the representativeness of *H. dulcis* leaves in the leaf mixture may have been insufficient to generate significant changes in the leaf decomposition of associated native species leaves and, in the structure, and composition of invertebrates community. We cannot disregard, however, that significant effects be observed in leaf mixtures where *H. dulcis* is more representative (e.g., mixtures of leaves with fewer species). Second, the rapid decomposition of *Hovenia* suggests, but does not ensure, that *H. dulcis* leaves are more assimilated or a nutritious food source for shredders. In some cases, invertebrates showed greater assimilation of carbon and nitrogen from slow decomposing leaves (Siders et al. 2018; Siders et al. 2021). Furthermore, the leaves of some non-native species, although consumed, do not necessarily constitute a nutritious food resource affecting larvae growth or survival (Kiffer et al. 2018; Moretti et al. 2020). Third, this study we just simulated the effects of *H. dulcis* leaves presence in streams exploring the influence of its leaf characteristics on leaf decomposition and associated invertebrates both individually and associated with leaves of native species. We disregard, however, the influence of variation in the availability of *H. dulcis* leaves in streams throughout the year (see Fontana et al. 2020) which is ecologically relevant for a comprehensive understanding of their influence in the stream functioning. These questions were partially explored recently (Fontana et al. 2022), but many questions remain little known. Four, in a broader ecological context, the set of existing studies (including this one) seems to be suggested that the *H. dulcis* leaves may be characterized as an ephemeral resource in the subtropical southern Brazilian streams due its fast leaf decomposition (see König et al. 2014; Biasi et al. 2020) and availability in large quantities only during three or four months of the year (Summer-Autumn transition) (see Fontana et al. 2020). This seems to be reinforced the importance of the presence of a diversified riparian vegetation to ensure, throughout the year, a diversified supply of allochthonous litterfall.

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Declarations

Authors Contribution

Conceptualization: LEF, LUH and MMP (equal for all); **Data curation:** LEF (lead) and CB (supporting); **Formal analysis:** LEF (lead), LUH and MMP (supporting); **Investigation:** LEF (lead) and CB (supporting); **Methodology:** LEF (lead), LUH and MMP (supporting); **Project administration:** LEF, LUH and MMP (equal for all); **Resources Provision** (laboratorial structure): LUH; **Supervision:** LUH and MMP (equal); **Writing – original draft:** LEF (lead); **Writing – review & editing:** LEF, LUH, MMP and CB (equal for all)

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Ethics Declarations

Conflicts of interest/Competing interests: the authors declare having no conflicts of interest.

Availability of data and material: not applicable.

Code availability: not applicable.

Ethics approval: not applicable, this study does not involve experiments with humans and/or vertebrate animals.

Consent to participate: not applicable, this study does not involve experiments with humans.

Consent for publication: not applicable, this study does not involve experiments with humans.

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Supplementary Material List

Fig. S1 Localization of Brazil (a), state of Rio Grande do Sul (b), and study area and studied streams (c). Prepared using QGIS 3.4 (creation of maps) and Microsoft PowerPoint 365® (formatting and presentation of maps). The spatial data were obtained from the database provided by DIVA-GIS (https://www.diva-gis.org/Data) (the countries in (a)), by Instituto Brasileiro de Geografia e Estatística - IBGE (https://www.ibge.gov.br) (Brazil and the Brazilian states in (b) and (c)), and by Fundação Estadual de Proteção Ambiental Henrique Luís Roessler - FEPAM (http://www.fepam.rs.gov.br) (hydrographic network in (c)).

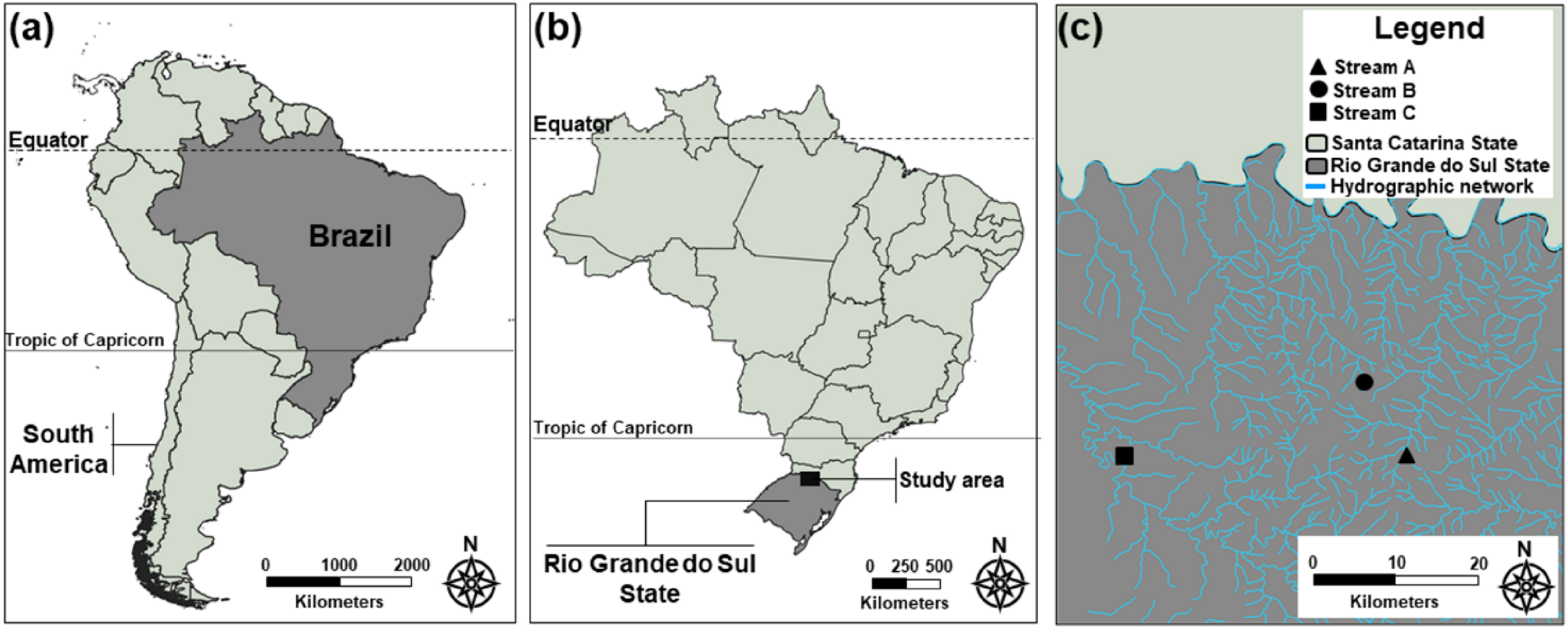

Table S1 Physical and chemical (mean ± standard deviation) water characteristics measured in the streams studied during the experiment. The variation in water characteristics among streams and sampling times was evaluated using an Analysis of Variance (two-way ANOVA). The test value (*F*), the degrees of freedom (*df*) and the test significance (*P*) are described.

Variables	Streams			ANOVA		
	Stream A	Stream B	Stream C	Streams ($df = 2;18$)	Time ($df = 2;18$)	Streams:Time ($df = 4;18$)
Temp (°C)	18.88 ± 0.77	19.04 ± 0.22	22.34 ± 0.68	$F = 677.6$; $P < 0.001$***	$F = 30.4$; $P < 0.001$***	$F = 24.1$; $P < 0.001$***
EC ($\mu S.cm^{-1}$)	170 ± 11	117 ± 4	76 ± 4	$F = 957.5$; $P < 0.001$***	$F = 1.6$; $P = 0.231^{ns}$	$F = 10.4$; $P < 0.001$***
pH	7.51 ± 0.14	7.35 ± 0.34	6.21 ± 1.95	$F = 17.1$; $P < 0.001$***	$F = 19.2$; $P < 0.001$***	$F = 15.8$; $P < 0.001$***
DO ($mg.L^{-1}$)	8.71 ± 0.82	8.31 ± 1.30	8.92 ± 1.29	$F = 5.5$; $P = 0.013$*	$F = 38.4$; $P < 0.001$***	$F = 28.5$; $P < 0.001$***
TDS ($\mu g.L^{-1}$)	112 ± 7	77 ± 2	51 ± 2	$F = 610.4$; $P < 0.001$***	$F = 0.4$; $P = 0.665^{ns}$	$F = 3.2$; $P = 0.038$*
Turb (NTU)	7.00 ± 3.19	12.01 ± 6.38	19.89 ± 4.22	$F = 42.3$; $P < 0.001$***	$F = 5.8$; $P = 0.011$*	$F = 7.9$; $P = 0.001$***
TDN ($mg.L^{-1}$)	2.18 ± 0.41	1.19 ± 0.48	1.75 ± 0.53	$F = 18.8$; $P < 0.001$***	$F = 2.7$; $P = 0.091^{ns}$	$F = 5.9$; $P = 0.003$**
Elev (m.a.s.l)	582	646	505			

Temp = Temperature; EC = Electrical Conductivity; DO = Dissolved Oxygen; TDS = Total Dissolved Solids; Turb = Turbidity; TDN = Total Dissolved Nitrogen; Elev = Elevation; m.a.s.l = meters above sea level

The asterisks indicate the significance: $P \leq 0.001$ (***), $P \leq 0.01$ (**), $P \leq 0.05$ (*), $P > 0.05$ (^{ns}).

Table S2 Structural and chemical (mean ± standard deviation) characterization of the leaves of native species composing the mixture of natives and of *H. dulcis* leaves.

Leaf Characteristics (n = 3)	Species						
	Ced	Cry	Eug	Han	Ing	Nec	Hov
Thickness (mm)	0.22 ± 0.02	0.31 ± 0.02	0.14 ± 0.01	0.16 ± 0.02	0.21 ± 0.00	0.24 ± 0.01	0.15 ± 0.01
Toughness (g)	188.2 ± 14.9	525.5 ± 56.7	229.6 ± 42.3	424.1 ± 213.7	297.3 ± 56.9	299.6 ± 13.4	224.0 ± 170.4
Specific Leaf Area ($mm^2.mg^{-1}$)	11.24 ± 2.04	7.74 ± 0.35	14.54 ± 1.57	23.52 ± 13.31	8.93 ± 0.24	10.65 ± 1.65	15.33 ± 1.44
Carbon (%)	43.96 ± 1.12	43.12 ± 0.18	44.48 ± 0.43	45.02 ± 0.12	44.86 ± 0.75	44.29 ± 1.00	43.15 ± 0.34
Nitrogen (%)	1.19 ± 0.00	1.21 ± 0.10	1.66 ± 0.00	1.76 ± 0.04	2.14 ± 0.01	1.54 ± 0.03	1.11 ± 0.03
Carbon : Nitrogen	36.95 ± 0.95	35.86 ± 2.70	26.75 ± 0.26	25.65 ± 0.60	20.96 ± 0.42	28.65 ± 0.57	38.95 ± 1.01
Tannins ($mg.g^{-1}$)	0.66 ± 0.02	0.47 ± 0.01	0.65 ± 0.02	0.43 ± 0.02	0.48 ± 0.01	0.47 ± 0.02	0.43 ± 0.06
Phosphorus ($mg.g^{-1}$)	4.96 ± 0.29	4.40 ± 0.19	4.96 ± 1.16	7.51 ± 0.80	4.91 ± 0.16	5.54 ± 0.28	4.82 ± 0.20

Ced = *Cedrela fissilis*, Cry = *Cryptocarya aschersoniana*, Eug = *Eugenia uniflora*, Han = *Handroanthus heptaphyllus*, Ing = *Inga marginata*, Nec = *Nectandra megapotamica*, Hov = *Hovenia dulcis*

Table S3 Results of the Analysis of Covariance (Two-way ANCOVA) testing the variation of the remaining weight between treatments and streams (time as cofactor) and the variation of the remaining weight of each species of the mixture of natives between treatments and streams (time as cofactor). The test value (*F*), the degrees of freedom (*df*) and the test significance (*P*) are described.

Factors	*df*	*F*	*P*
Treatments			
Hovenia *vs.* Nectandra			
Treatments	1	54.7	< 0.001***
Streams	2	0.1	0.861^{ns}
Time	1	101.6	< 0.001***
Treatments:Streams	2	0.4	0.636^{ns}
Residuals	47		
Hovenia+mix *vs.* Nectandra+mix			
Treatments	1	11.1	0.002**
Streams	2	0.4	0.642^{ns}
Time	1	477.0	< 0.001***
Treatments:Streams	2	0.25	0.775^{ns}
Residuals	47		
Mixture of natives			
Cedrela fissilis			
Treatments	1	1.9	0.177^{ns}
Streams	2	1.3	0.277^{ns}
Time	1	34.8	< 0.001***
Treatments:Streams	2	0.5	0.608^{ns}
Residuals	47		
Eugenia uniflora			
Treatments	1	0.0	0.984^{ns}
Streams	2	0.8	0.435^{ns}
Time	1	697.7	< 0.001***
Treatments:Streams	2	0.3	0.716^{ns}
Residuals	47		
Inga marginata			
Treatments	1	0.0	0.939^{ns}
Streams	2	0.0	0.993^{ns}
Time	1	38.8	< 0.001***
Treatments:Streams	2	0.8	0.449^{ns}
Residuals	47		
Cryptocarya aschersoniana			
Treatments	1	0.6	0.438^{ns}
Streams	2	0.5	0.589^{ns}
Time	1	11.1	0.002**
Treatments:Streams	2	0.4	0.697^{ns}
Residuals	47		
Handroanthus heptaphyllus			
Treatments	1	1.0	0.310^{ns}
Streams	2	1.5	0.225^{ns}
Time	1	728.7	< 0.001***
Treatments:Streams	2	1.1	0.330^{ns}
Residuals	47		

The asterisks indicate the significance: $P \leq 0.001$ (***), $P \leq 0.01$ (**), $P \leq 0.05$ (*), $P > 0.05$ (^{ns}).

Table S4 Abundance (individuals litter bag^{-1}) (mean ± standard deviation) of invertebrates associated with treatments at different sampling times (7 and 15). Differentiation of the taxonomic hierarchy presented: **PHYLUM** – CLASS – **Order** – Family. Life stage: larvae (l); adult (a).

Taxa	FFG	H7	H15	N7	N15	HM7	HM15	NM7	NM15
ANNELIDA									
HIRUDINEA	Pr	0.9 ± 2.0	0.4 ± 0.9	1.3 ± 2.2	0.2 ± 0.4	0.7 ± 1.0	0.2 ± 0.4	0.4 ± 1.0	1.2 ± 3.7
OLIGOCHAETA	CG	2.4 ± 2.3	10.7 ± 14.5	0.7 ± 1.3	7.1 ± 14.8	1.0 ± 2.1	14.1 ± 20.1	1.2 ± 2.6	14.7 ± 20.9
ARTHROPODA									
ARACHNIDA									
Hydrachnidae	Pr	-	0.3 ± 0.7	-	0.6 ± 0.7	0.1 ± 0.3	0.4 ± 1.0	-	0.6 ± 0.9
INSECTA									
Diptera									
Ceratopogonidae	Pr	-	-	0.1 ± 0.3	0.1 ± 0.3	-	0.2 ± 0.4	0.1 ± 0.3	0.8 ± 1.6
Chironomidae	CG	47.1 ± 26.9	67.7 ± 58.6	28.0 ± 9.8	54.6 ± 67.2	29.7 ± 11.7	53.2 ± 57.9	26.6 ± 17.9	54.3 ± 47.2
Empididae	Pr	0.3 ± 0.7	0.6 ± 1.3	-	0.6 ± 1.0	0.1 ± 0.3	0.7 ± 1.4	0.1 ± 0.3	0.2 ± 0.7
Psychodidae	CG	-	-	-	-	-	-	-	0.2 ± 0.7
Simulidae	CF	-	0.4 ± 1.4	0.1 ± 0.3	0.2 ± 0.7	0.1 ± 0.3	0.3 ± 1.0	-	1.7 ± 4.6
Tipulidae	Sh	-	-	-	-	-	-	-	0.4 ± 1.0
Coleoptera									
Dytiscidae (l)	Pr	-	-	-	-	-	-	0.2 ± 0.4	0.1 ± 0.3
Dryopidae (l)	Sh	-	0.1 ± 0.3	-	-	-	-	-	-
Elmidae (a)	Sc	0.1 ± 0.3	0.6 ± 0.7	-	0.2 ± 0.4	0.2 ± 0.4	0.3 ± 0.7	-	0.3 ± 0.5
Elmidae (l)	CG	1.9 ± 1.5	4.1 ± 3.6	0.7 ± 1.0	4.2 ± 4.7	1.6 ± 1.2	0.3 ± 2.4	1.6 ± 0.7	10.6 ± 19.5
Hydrophylidae (a)	CG	-	-	-	0.2 ± 0.7	-	2.1 ± 2.4	-	-
Hydrophylidae (l)	Pr	0.2 ± 0.7	-	-	0.1 ± 0.3	-	0.1 ± 0.3	-	0.1 ± 0.3
Psephenidae	Sc	-	-	-	-	-	-	-	0.3 ± 0.7
Ephemeroptera									
Baetidae	CG	3.1 ± 3.1	0.8 ± 1.4	0.7 ± 1.7	2.4 ± 2.5	2.2 ± 4.6	1.9 ± 1.8	0.9 ± 2.0	2.0 ± 4.0
Caenidae	CG	3.6 ± 5.0	20.3 ± 27.1	1.3 ± 1.9	1.8 ± 2.1	1.6 ± 3.3	2.0 ± 3.3	1.2 ± 1.3	10.9 ± 20.5
Leptophlebiidae	CG	44.3 ± 49.2	32.0 ± 23.8	17.7 ± 16.6	13.9 ± 9.4	26.2 ± 18.5	17.4 ± 14.2	32.1 ± 33.4	21.1 ± 21.6

H = *Hovenia*; HM = *Hovenia+mix*; N = *Nectandra*; NM = *Nectandra+mix*.

FFG = Functional Feeding Group; CG = Collector-Gatherer, CF = Collector-Filterer, Pr = Predator, Sh = Shredder, Sc=Scraper.

Table S4 Continuation…

Taxa	FFG	H7	H15	N7	N15	HM7	HM15	NM7	NM15
Hemiptera									
Veliidae	Pr	-	-	0.0 ± 0.0	0.1 ± 0.3	-	0.1 ± 0.3	0.0 ± 0.0	0.0 ± 0.0
Megaloptera									
Corydalidae	Pr	-	-	0.0 ± 0.0	0.1 ± 0.3	-	0.1 ± 0.3	0.0 ± 0.0	0.0 ± 0.0
Odonata									
Aeshnidae	Pr	-	0.4 ± 0.9	0.0 ± 0.0	0.3 ± 0.7	-	0.1 ± 0.3	0.0 ± 0.0	0.8 ± 1.6
Calopterygidae	Pr	0.9 ± 1.3	1.9 ± 2.2	0.1 ± 0.3	1.2 ± 1.7	0.6 ± 0.9	2.0 ± 1.5	0.2 ± 0.7	1.1 ± 1.8
Gomphidae	Pr	0.1 ± 0.3	0.1 ± 0.3	0.0 ± 0.0	0.0 ± 0.0	-	0.1 ± 0.3	0.0 ± 0.0	0.0 ± 0.0
Libellulidae	Pr	-	0.2 ± 0.4	0.0 ± 0.0	0.1 ± 0.3	0.1 ± 0.3	0.0 ± 0.0	0.2 ± 0.7	0.1 ± 0.3
Coenagrionidae	Pr	0.8 ± 1.4	0.9 ± 1.5	0.8 ± 1.6	0.7 ± 0.9	0.3 ± 0.5	1.1 ± 2.4	1.0 ± 1.5	0.9 ± 1.4
Plecoptera									
Gripopterygidae	Sh	1.8 ± 4.2	2.0 ± 3.6	1.4 ± 4.0	4.2 ± 6.2	2.9 ± 6.5	2.3 ± 3.1	1.2 ± 2.4	2.3 ± 4.1
Perlidae	Pr	0.3 ± 0.7	0.2 ± 0.7	0.8 ± 1.7	0.7 ± 1.7	0.8 ± 1.6	0.4 ± 1.0	0.8 ± 2.3	0.0 ± 0.0
Trichoptera									
Calamoceratidae	Sh	10.8 ± 7.6	18.1 ± 22.5	6.9 ± 12.1	6.3 ± 3.9	10.2 ± 7.0	8.8 ± 2.3	9.9 ± 7.2	11.8 ± 7.5
Hydrobiosidae	Pr	0.1 ± 0.3	-	0.0 ± 0.0	0.0 ± 0.0	-	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0
Hydropsychidae	CF	-	3.8 ± 9.3	0.0 ± 0.0	10.8 ± 27.6	-	6.8 ± 13.3	0.0 ± 0.0	10.0 ± 30.0
Hydroptilidae	Sc	2.4 ± 4.8	3.0 ± 4.9	1.6 ± 2.4	3.6 ± 7.9	0.8 ± 0.8	5.4 ± 10.7	0.7 ± 1.0	0.2 ± 0.7
Leptoceridae	Sh	1.4 ± 1.6	1.7 ± 1.6	0.3 ± 0.5	1.4 ± 1.5	0.8 ± 1.1	1.4 ± 2.2	0.7 ± 1.0	1.3 ± 0.9
Odontoceridae	Sh	0.9 ± 1.1	1.8 ± 1.8	1.0 ± 3.0	3.3 ± 4.2	0.7 ± 1.1	2.7 ± 3.6	1.3 ± 1.6	2.0 ± 3.9
Philopotamidae	CF	-	-	0.0 ± 0.0	0.2 ± 0.7	-	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0
MOLLUSCA									
BIVALVIA	CF	10.7 ± 22.3	21.8 ± 36.2	1.9 ± 3.8	5.3 ± 16.0	5.6 ± 10.4	8.0 ± 19.7	5.9 ± 9.0	3.2 ± 7.0
GASTROPODA	Sc	27.5 ± 36.3	8.9 ± 19.2	22.8 ± 35.8	8.8 ± 15.3	31.6 ± 50.6	16.9 ± 25.5	19.4 ± 30.0	6.4 ± 9.8

H = *Hovenia*; HM = *Hovenia+mix*; N = *Nectandra*; NM = *Nectandra+mix*.

FFG = Functional Feeding Group; CG = Collector-Gatherer, CF = Collector-Filterer, Pr = Predator, Sh = Shredder, Sc=Scraper.

Table S5 Results of the Analysis of Variance (three-way ANOVA) testing the variation in invertebrate density, rarefied richness, Shannon Index, evenness, and shredder, collector, scraper and predator density between treatments, streams and sampling times evaluated. The test value (*F*), the degrees of freedom (*df*) and the test significance (*P*) are described.

Factors	Hovenia vs. Nectandra			Hovenia+mix vs. Nectandra+mix		
	df	F	P	df	F	P
Structural Metrics						
Invertebrates Density						
Treatments	1	45.8	<0.001***	1	1.4	0.244^{ns}
Streams	2	9.2	0.001**	2	7.0	0.004**
Time	1	15.6	<0.001***	1	11.0	0.002**
Treatments:Streams	2	0.2	0.799^{ns}	2	0.5	0.601^{ns}
Treatments:Time	1	3.6	0.071^{ns}	1	0.0	0.865^{ns}
Streams:Time	2	0.5	0.605^{ns}	2	0.2	0.838^{ns}
Treatments:Streams:Time	2	0.3	0.741^{ns}	2	0.2	0.227^{ns}
Residuals	24			24		
Rarefied Richness						
Treatments	1	1.1	0.296^{ns}	1	0.9	0.359^{ns}
Streams	2	15.8	<0.001***	2	4.1	0.028*
Time	1	14.3	<0.001***	1	16.0	<0.001***
Treatments:Streams	2	1.7	0.192^{ns}	2	0.7	0.487^{ns}
Treatments:Time	1	6.3	0.018*	1	0.2	0.636^{ns}
Streams:Time	2	4.5	0.021*	2	2.9	0.070^{ns}
Treatments:Streams:Time	2	7.8	0.002**	2	2.0	0.151^{ns}
Residuals	24			24		
Shannon Index						
Treatments	1	0.2	0.680^{ns}	1	0.5	0.484^{ns}
Streams	2	28.4	<0.001***	2	6.5	0.005**
Time	1	15.8	<0.001***	1	19.9	<0.001***
Treatments:Streams	2	1.1	0.340^{ns}	2	0.8	0.440^{ns}
Treatments:Time	1	3.8	0.062^{ns}	1	0.2	0.684^{ns}
Streams:Time	2	2.3	0.122^{ns}	2	4.4	0.022*
Treatments:Streams:Time	2	10.7	<0.001***	2	1.1	0.344^{ns}
Residuals	24			24		
Evenness						
Treatments	1	5.0	0.034*	1	0.1	0.733^{ns}
Streams	2	11.6	<0.001***	2	11.1	<0.001***
Time	1	0.1	0.725^{ns}	1	8.3	0.008**
Treatments:Streams	2	0.8	0.458^{ns}	2	0.3	0.711^{ns}
Treatments:Time	1	0.4	0.548^{ns}	1	0.3	0.572^{ns}
Streams:Time	2	0.4	0.681^{ns}	2	8.3	0.001**
Treatments:Streams:Time	2	4.6	0.020*	2	2.5	0.102^{ns}
Residuals	24			24		

Table S5 Continuation…

Functional Feeding Groups						
Shredders Density						
Treatments	1	21.4	<0.001***	1	1.2	0.287 ^{ns}
Streams	2	0.0	0.974 ^{ns}	2	0.5	0.637 ^{ns}
Time	1	13.1	0.001**	1	8.4	0.008**
Treatments:Streams	2	0.9	0.406 ^{ns}	2	1.8	0.191 ^{ns}
Treatments:Time	1	0.9	0.338 ^{ns}	1	0.2	0.692 ^{ns}
Streams:Time	2	0.3	0.723 ^{ns}	2	0.4	0.697 ^{ns}
Treatments:Streams:Time	2	1.5	0.233 ^{ns}	2	0.2	0.850 ^{ns}
Residuals	24			24		
Collectors Density						
Treatments	1	42.0	<0.001***	1	0.5	0.496 ^{ns}
Streams	2	3.9	0.031*	2	2.5	0.106 ^{ns}
Time	1	13.9	0.001**	1	11.5	0.002**
Treatments:Streams	2	0.5	0.636 ^{ns}	2	0.3	0.738 ^{ns}
Treatments:Time	1	3.1	0.090 ^{ns}	1	0.0	0.880 ^{ns}
Streams:Time	2	0.2	0.820 ^{ns}	2	0.1	0.935 ^{ns}
Treatments:Streams:Time	2	0.4	0.684 ^{ns}	2	2.6	0.091 ^{ns}
Residuals	24			24		
Scrapers Density						
Treatments	1	8.5	0.007**	1	19.9	<0.001***
Streams	2	144.7	<0.001***	2	397.5	<0.001***
Time	1	1.5	0.232 ^{ns}	1	0.1	0.758 ^{ns}
Treatments:Streams	2	4.1	0.029*	2	8.9	0.001**
Treatments:Time	1	5.9	0.023*	1	0.0	0.954 ^{ns}
Streams:Time	2	2.4	0.107 ^{ns}	2	2.0	0.154 ^{ns}
Treatments:Streams:Time	2	0.6	0.538 ^{ns}	2	6.5	0.005**
Residuals	24			24		
Predators Density						
Treatments	1	8.1	0.008**	1	0.7	0.412 ^{ns}
Streams	2	11.0	<0.001***	2	3.3	0.053 ^{ns}
Time	1	8.5	0.007**	1	10.7	0.003**
Treatments:Streams	2	2.9	0.076 ^{ns}	2	0.4	0.682 ^{ns}
Treatments:Time	1	2.1	0.162 ^{ns}	1	0.6	0.439 ^{ns}
Streams:Time	2	0.1	0.889 ^{ns}	2	0.2	0.835 ^{ns}
Treatments:Streams:Time	2	0.9	0.389 ^{ns}	2	0.0	0.998 ^{ns}
Residuals	24			24		

The asterisks indicate the significance: $P \leq 0.001$ (***), $P \leq 0.01$ (**), $P \leq 0.05$ (*), $P > 0.05$ (^{ns}).

Table S6 Results of the Multivariate Analysis of Homogeneity of Groups Dispersions (PermDISP) testing the invertebrate community composition distribution within each treatment, stream and sampling time and the Permutational Multivariate Analysis of Variance (PermANOVA) testing the distribution of invertebrate community composition among treatments, streams and sample times. The test value (*F*), the degrees of freedom (*df*), the model explanation (R^2) and the test significance (*P*) are described.

Test	**Treatments**			**Treatments**		
PermDISP	Hovenia vs. Nectandra			Hovenia+mix vs. Nectandra+mix		
Factors	*df*	*F*	*P*	*df*	*F*	*P*
Treatments						
Groups	1	2.9	0.099^{ns}	1	1.6	0.207^{ns}
Residuals	34			34		
Streams						
Groups	2	0.9	0.404^{ns}	2	1.5	0.229^{ns}
Residuals	33			33		
Time						
Groups	1	1.4	0.252^{ns}	1	0.9	0.350^{ns}
Residuals	34			34		

Test	**Treatments**				**Treatments**			
PermANOVA	Hovenia *vs.* Nectandra				Hovenia+mix *vs.* Nectandra+mix			
Factors	*df*	*F*	R^2	*P*	*df*	*F*	R^2	*P*
Treatments	1	4.0	0.05	0.007**	1	1.4	0.02	0.212^{ns}
Streams	2	16.4	0.43	0.001***	2	15.2	0.43	0.001***
Time	1	3.7	0.05	0.006**	1	3.8	0.05	0.003**
Treatments:Streams	2	1.1	0.03	0.424^{ns}	2	1.0	0.03	0.407^{ns}
Treatments:Time	1	1.4	0.02	0.210^{ns}	1	0.5	0.01	0.859^{ns}
Streams:Time	2	2.5	0.07	0.005**	2	2.8	0.08	0.002**
Treatments:Streams:Time	2	1.2	0.03	0.284^{ns}	2	1.6	0.04	0.085^{ns}
Residuals	24		0.32		24		0.34	
Total	35		1.00		35		1.00	

The asterisks indicate the significance: $P \leq 0.001$ (***), $P \leq 0.01$ (**), $P \leq 0.05$ (*), $P > 0.05$ (^{ns})

6. CAPÍTULO III

Effects of non-native plants on leaf decomposition and on communities of microorganisms and invertebrates in streams: a scientometric and systematic approach

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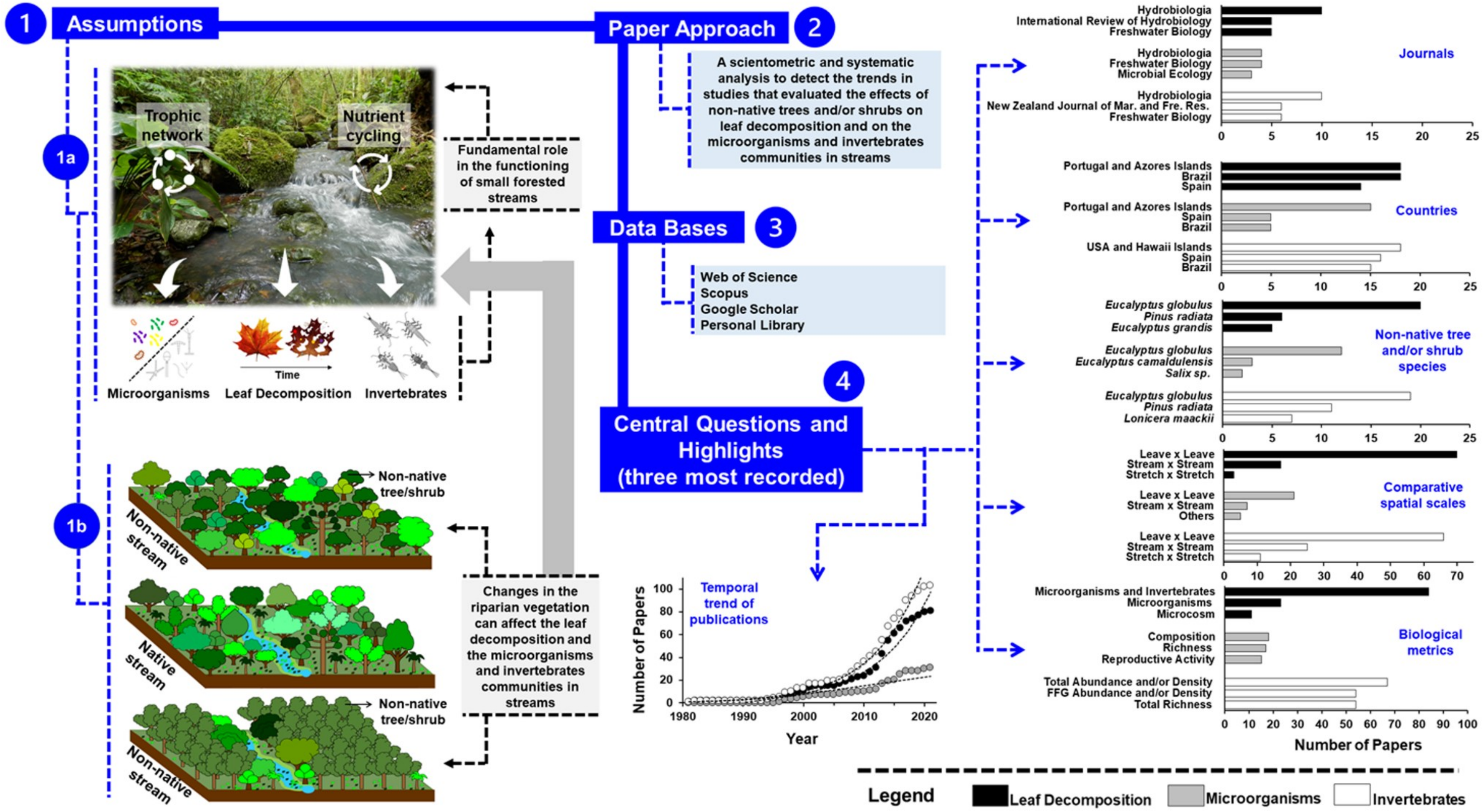
A formatação das referências e de alguns pontos do manuscrito obedecem as normas deste periódico.

Graphical Abstract



1 **Assumptions**

1a

Fundamental role in the functioning of small forested streams

1b

Changes in the riparian vegetation can affect the leaf decomposition and the microorganisms and invertebrates communities in streams

Paper Approach 2

A scientometric and systematic analysis to detect the trends in studies that evaluated the effects of non-native trees and/or shrubs on leaf decomposition and on the microorganisms and invertebrates communities in streams

Data Bases 3

Web of Science
Scopus
Google Scholar
Personal Library

4

Central Questions and Highlights (three most recorded)

Legend ■ Leaf Decomposition ▨ Microorganisms ☐ Invertebrates

Abstract: In small forested streams, leaf decomposition and trophic relationships between microorganisms and aquatic invertebrates are part of the functioning of these ecosystems. The presence of non-native trees and/or shrubs in the riparian zone can alter the functioning of streams. We performed a systematic review and scientometric analysis to assess the main trends in publications regarding the effects of trees and/or shrubs on leaf decomposition, aquatic microorganisms and aquatic invertebrates in streams. We used the Web of Science, Scopus and Google Scholar databases to search for publications of interest and, for this, we use a combination of words associated with 'streams', 'leaf decomposition', 'aquatic microorganisms', 'aquatic invertebrates', 'tree', 'shrub' and 'non-native species'. We quantified the number of studies and listed the most journals publising on this topic, countries, non-native trees and/or shrubs, spatial scales of comparison and biological metrics indicators. We considered 127 papers published between 1981 and 2020 in 57 journals. *Decomposition* was approached in ~64% of the papers, *microorganisms* in ~24% and *invertebrates* in ~81%. Among 29 countries, ~75% of the analyzed papers were focused on 5 countries. This also occurred for non-native trees and/or shrubs, with only 6 species, among 89, concentrating ~50% of papers. Most studies compared native *vs.* non-native leaves and native *vs.* non-native streams. Leaf decomposition driven by microorganisms and invertebrates (for *decomposition*), structure, composition and reproductive activity (for *microorganisms*), and structure and feeding habits (for *invertebrates*) were the most recorded biological metrics. We believe that the determination of these trends can help future studies to formulate new questions about ecological aspects still little explored in the evaluation of the effects of non-native tree and/or shrub species on leaf decomposition and the communities of microorganisms and invertebrates in streams.

Key-words: Biological Invasions, Exotic Species, Lotic Ecosystems, Macroinvertebrates, Aquatic Hyphomycetes, Scientometric Study

Introduction

The functioning of ecosystems is constituted by a set of ecological processes (e.g., production, decomposition and trophic relationships) (Díaz & Cabido, 2001) fully or partially supported by biodiversity (Gessner et al., 2010; Cardinale et al., 2011; Barnes et al., 2018). In small forested streams, autochthonous and allochthonous production (i.e., energy sources) (Webster & Benfield, 1986), litterfall decomposition (i.e., nutrient cycling) (Chauvet et al., 2016) and trophic relationships between communities of aquatic microorganisms and aquatic invertebrates (i.e., energy flow) (Wallace & Webster, 1996; Graça et al., 2016) constitute a fundamental part of the functioning of these ecosystems.

In forested streams, the autochthonous primary productivity (e.g., periphytic algae) is limited by the shading generated by the riparian vegetation and, therefore, the allochthonous primary productivity (e.g., litterfall, especially leaves) becomes the main source of energy for these ecosystems (Vannote et al., 1980; Abelho, 2001; Graça, 2001). Within streams, leaves are gradually decomposed and incorporated into the trophic network by chemical and physical (e.g., leaching and physical fragmentation) and biological (e.g., colonization and consumption by microorganisms and aquatic invertebrates) processes (Abelho, 2001; Marks, 2019). Aquatic hyphomycetes - the main decomposing microorganisms in streams - secrete enzymes that partially decomposes leaf tissue during their nutrition process (Abelho, 2001; Hieber & Gessner, 2002; Marks, 2019). Aquatic invertebrates, on the other hand, consume allochthonous resources directly (e.g., shredders; coarse particulate organic matter) or indirectly (e.g., collectors; fine particulate organic matter), autochthonous resources (e.g., scrapers; periphytic algae) or even larvae of other organisms (e.g., predators) (Graça, 2001; Cummins et al., 2005; Ramiréz & Gutiérrez-Fonseca, 2014; Marks, 2019).

The structure and composition of native riparian vegetation in streams can be modified through biological invasion (e.g., Mineau et al., 2012; Medina-Villar et al., 2015a) or implementation of monocultures (e.g., Abelho & Graça, 1996; Molinero & Pozo, 2004) by non-native trees and/or shrubs (hereinafter only 'non-native species'). Changes in the structure and composition of riparian vegetation caused by the presence of non-native species can alter the quantity (Molinero & Pozo, 2004; McNeish et al., 2015), the temporal pattern (e.g. Abelho & Graça, 1996; Fontana et al., 2020) and the composition (Graça et al., 2002) of the allochthonous litterfall input to streams, which, in many cases, can affect the functioning of these ecosystems. Thus, in recent decades, a significant number of studies have evaluated, separately or in combination, the effects of different non-native species on leaf decomposition, as well as on communities of aquatic microorganisms and aquatic invertebrates in streams around the world (e.g., Abelho & Graça, 1996; Quinn et al., 2000; Bärlocher & Graça, 2002; Swan et al., 2008; Menéndez et al., 2013; König et al., 2014; Medina-Villar et al., 2015b; Ferreira et al., 2019; Biasi et al., 2020). As a result, a diverse set of positive and negative effects, or even the absence of significant changes mediated by the ocurrence of non-native species on leaf decomposition, as well as on communities of aquatic microorganisms and aquatic invertebrates in streams has been reported (Castro-Díaz & Alonso, 2017; Ferreira et al., 2021).

The distinct ecosystem responses observed seem to vary depending, independently or collectively, on factors such as i) the non-native species (e.g., Martínez et al., 2013a; König et al., 2014; Medina-Villar et al., 2015b); the spatial scales of comparison (e.g., at leaf level, stretches of streams, or streams) (e.g., Canhoto & Graça, 1996; Menéndez et al., 2013; McNeish et al., 2015); iii) the biological indicators (i.e., some metrics may be more sensitive than others as to the effects of non-native species) (e.g., Gonçalves et al., 2012; Martínez et al., 2013b; Menéndez et al., 2013; Gonçalves et al., 2017), and iv) the studied regions (i.e., countries) (e.g., Boyero et al., 2015; Ferreira et al., 2019). In this context, recent studies using meta-analysis (Ferreira et al., 2016; Kennedy & El-Sabaawi, 2017) and theoretical reviews (Castro-Díez & Alonso, 2017; Ferreira et al., 2021) have sought to understand the effects of non-native species on leaf decomposition, as well as on communities of aquatic microorganisms and aquatic invertebrates in streams, both at regional and global scales. Despite this, quantitative data listing countries, non-native species, spatial scales of comparison and the most used biological metrics are still incipient on a global scale. The determination of these data (trends) can help in the formulation of new questions still little explored in the evaluation of the effects of non-native tree and/or shrub species on leaf decomposition and on these aquatic communities of in streams.

In this study, a systematic review and scientometric analysis (hereinafter only 'scientometrics') were carried out to detect the main trends of studies evaluating the effects of non-native trees and/or shrubs on leaf decomposition, as well as on communities of aquatic microorganisms and aquatic invertebrates in streams. To do this, different information from publications were collected to answer the following questions: i) What is the number and time trend of such publications?; ii) Which journals have the highest number of publications?; iii) Which countries have the highest number of published studies carried out, in whole or in part, in their territory?; iv) Which non-native species are the most studied in the referred publications?; v) What are the spatial scales of comparation most used in those publications?; and, vi) What are the most used biological metrics in said publications?

Material and Methods

Research Sources

The present study sought publications that approached, under different ecological aspects, the influence of non-native trees and/or shrubs on leaf decomposition, as well as on communities of aquatic microorganisms and aquatic invertebrates in streams. For this, the search for publications of interest was carried out using the databases of Web of Science (https://www.webofscience.com), Scopus (https://www.scopus.com), Google Scholar (https://scholar.google.com.br/) and, in specific cases, the authors personal libraries.

Search of Publications

At the outset, the search for publications of interest was carried out collectively in July/2021, using the Web of Science and the Scopus databases. Due to the keyword limitation, publications concerning the theme 'leaf decomposition' were searched first, followed by publications approaching the themes 'aquatic microorganisms' and 'aquatic invertebrates' together. For both, the same set of keywords were used and papers written in English, Spanish and Portuguese were considered. For the theme 'leaf decomposition', the search was performed by combining the keywords (*stream** OR *headwater** OR *low order river** OR *lotic ecosystem**) AND (*non-native** OR i*ntroduced** OR *exotic** OR *invasive** OR *non-indigenous**) AND (*leaf** OR *litter** OR *leaves** OR *plant** OR *vegetal** OR *tree** OR *shrub**) AND *(riparian vegetation** OR *riparian zone** OR *invaded riparian zone* OR *monoculture** OR *afforestation** OR *replacement**) AND (*decomposition** OR *breakdown** OR *decay*), present in the title, and/or in the abstract, and/or in the keywords of the related publications. For the themes 'aquatic microorganisms' and 'aquatic invertebrates' the search was conducted by combining the keywords (*stream** OR *headwater** OR *low order river** OR *lotic ecosystem**) AND (*non-native** OR *introduced** OR *exotic** OR *invasive** OR *non-indigenous**) AND (*leaf** OR *litter** OR *leaves** OR *plant** OR *vegetal** OR *tree** OR *shrub**) AND (*riparian vegetation** OR *riparian zone** OR *invaded riparian zone* OR *monoculture** OR *afforestation** OR *replacement**) AND (*microorganism** OR *microbe** OR *bacteria** OR *fungi** OR *hyphomycetes** OR *invertebrate** OR *macroinvertebrate** OR *aquatic** *communities** OR *aquatic** *insects** NOT *fish**), present in the title, and/or in the abstract, and/or in the keywords of said publications. In both cases, boolean operators (i.e., ‘AND’, ‘OR’ e ‘NOT’) and special caracters (i.e., ‘*’ and ‘( )’) were used to, respectively, combine and search for keyword variations in order to broaden the scope of our research.

A total of 442 papers were obtained. Of these results, 165 were related to the theme 'leaf decomposition' (Web of Science: 100 results; Scopus: 65) and 277 were related, collectively, to the theme 'aquatic microorganisms' and 'aquatic invertebrates' (Web of Science: 169 results ; Scopus: 108). Afterwards, all search results were merged into a single dataset and all duplicated records were excluded, so that only a single record was kept for each publication. Then, a total of 146 publications remained, which were selected considering the established criteria (see below), mainly through reading the respective abstracts.

Subsequently and in a complementary way, in October/2021 the Google Scholar database was used to search for papers of interest written in English, Spanish and Portuguese with the combination of words - *non native, exotic, leaf litter, streams, decomposition, microorganisms and invertebrates*. The results obtained were also selected by reading the papers abstract. We included in this scientometrics only papers that: i) met all the established criteria (see below); ii) were not registered in searches carried out in the Web of Science and Scopus databases; and, iii) appeared in the first 50 pages of Google Scholar results (~500 resulting publications). Finally, papers from the authors personal libraries were also included in this scientometrics, provided they also i) met all the established criteria (see below); and, ii) were not listed in any of the other databases used.

Criteria for the Inclusion of Publications

Only publications that necessarily met all of the following criteria were included in this scientometrics: i) being a paper published in a journal (papers published in annals of events were disregarded); ii) having been published until December 2020 (papers from 2021 made available 'online first' by December 2020 were also included); iii) being a paper with an experimental approach in the field (*in situ*) and/or in microcosm/mesocosm (*ex situ*) (theoretical review or meta-analysis papers were disregarded); iv) having the experimental approach carried out in streams (≤ 5th order) and/or microcosm/mesocosm simulating this ecosystem (papers carried out in large rivers [≥ 6th order], lentic ecosystems or microcosms/mesocosms simulating these ecosystems were disregarded); v) evaluating, under different ecological aspects, leaf decomposition, and/or communities of aquatic microorganisms and/or communities of aquatic invertebrates; vi) evaluating the effects of at least one non-native tree and/or shrub species (studies approaching only non-native grass and/or herbaceous species were disregarded).

Thematic Organization, Extraction of Variables and Data Analysis

After the search and selection, the collected papers were united into a single set and organized into three thematic groups: i) papers approaching the theme 'leaf decomposition' (hereinafter only *decomposition*); ii) papers approaching the theme 'aquatic microorganisms' (hereinafter *microorganisms*); and, iii) papers approaching the theme 'aquatic invertebrates' (hereinafter *invertebrates*). Papers covering more than one of these themes were included in more than one thematic group (e.g., Gonçalves et al., 2012; Menéndez et al., 2013; Ferreira et al., 2015; Seena et al., 2017; Biasi et al., 2020).

Ensuing the thematic organization, the extraction of data from each paper was performed. All variables were extracted from information present in the papers, based on the central questions that structure this scientometrics (see Table 1). For each paper included in this scientometrics, information was extracted according to: i) the year of publication (papers from 2020 and papers from 2021 'online first' until December 2020 were considered together) (Question 1); ii) the journal where the paper was published (Question 2); iii) the countries where the study was carried out (Question 3); iv) registered non-native tree and/or shrub species (Question 4); v) the spatial scale of comparison between native *vs.* non-native species (Question 5) (for definitions and examples, see Supplementary Table 1); and, vi) the main biological metrics used to assess the effects of non-native species on leaf decomposition and on communities of aquatic microorganisms and aquatic invertebrates (Question 6) (for definitions and examples, see Supplementary Table 2). After extracting the variables, data analysis was performed and the results were described using descriptive statistics (e.g., absolute numbers, and/or percentage, and/or mean ± standard deviation). For Question 1, to assess the evolution of the number of papers published over time (years), we also constructed trendlines for each theme based on linear regression models. The analyses were performed with the Microsoft© Excel 365 software.

Table 1 Central questions that structured this scientometric, the data collected to answer each question, and the main results. See Table S1 for explanation and examples regarding each data considered in question 5, and Table S2 for explanation and examples regarding each data considered in question 6.

Questions	Data	Results
Q1: What is the number and time trend of such publications?	The publication year of each paper.	Fig. 2
Q2: Which journals have the highest number of publications?	Journal where the papers were published.	Fig. 3 Table S4
Q3: Which countries have the highest number of published studies carried out, in whole or in part, in their territory?	Countries where the experimental procedures for each paper were performed.	Fig. 4 Table S5 Table S8
Q4: Which non-native species are the most studied in the referred publications?	Non-native trees and/or shrubs recorded in the papers. This includes species selected especially for experiments or those present in the streams riparian vegetation associated or not with other non-native species and/or native species.	Fig. 5 Table S6 Table S7 Table S8
Q5: What are the spatial scales of comparation most used in those publications?	Spatial scales of comparison between native species *vs.* non-native species: 1) ‘leave *vs.* leave’; 2) ‘branches *vs.* branches’; 3) ‘stream stretch *vs.* stream stretch’; 4) ‘stream *vs.* stream’; 5) ‘others’.	Fig. 6
Q6: What are the most used biological metrics in said publications?	For *decomposition*: 1) ‘decomposition drive by microorganisms and invertebrates’; 2) ‘decomposition drive by microorganisms’ and, 3) ‘decomposition evaluated in microcosm/mesocosm’ For *microorganisms*: 1) ‘abundance or relative abundance’; 2) ‘richness’; 3) ‘diversity indexes’; 4) ‘composition’; 5) ‘reproductive activity’; 6) ‘biomass’; 7) ‘microbial respiration’; 8) ‘colony forming units’ and, 9) ‘size of colonies’. For *invertebrates*: 1) ‘total abundance and/or density’; 2) ‘total richness’; 3) ‘total biomass’; 4) ‘diversity indexes’; 5) ‘biological indexes’; 6) ‘composition’; 7) ‘functional feeding groups abundance and/or density’; 8) ‘functional feeding groups richness’; 9) ‘functional feeding groups biomass’; 10) ‘indicator taxa’; 11) ‘food preference and/or consumption’; 12) ‘development and/or growth’; 13) ‘mortality and/or survival’; 14) ‘construction and/or characteristics of shelters’; 15) ‘locomotion activity’; 16) ‘stable isotopes’; 17) ‘functional diversity’ and, 18) ‘others’.	Fig. 7

Results

Number of Publications

A total of 127 papers published between 1981 and 2020 met all the defined criteria and were included in this scientometrics (Fig. 1; Table S3). Of these, ~64% of the total (n = 81 papers) approached *decomposition* (2.0 ± 2.8 papers/year; mean ± standard deviation), ~ 24% (n = 31) approached *microorganisms* (0.8 ± 1.2 papers/year) and ~ 81 % (n = 103) approached *invertebrates* (2.5 ± 3.1 papers/year). Over 40 years, a significant increase in the number of published papers was observed, especially in the 2011-2020 decade. Proportionally, the greatest increase in published papers was detected between the decades of 2001-2010 and 2011-2020. In these decades, the annual average of published papers regarding *decomposition* was from 1.3 ± 1.1 papers/year to 5.2 ± 3.7 papers/year (4-fold increase), for *microorganisms* it was from 0.5 ± 0.5 papers/year to 1.9 ± 1.8 papers/year (3.8-fold increase), and for *invertebrates* it was from 2.3 ± 2.1 papers/year to 6.6 ± 3.2 papers/year (2.9-fold increase) (Fig. 2).

Fig. 1 Global distribution pattern (n = 152) of places where the studies included in this scientometrics (n = 127) were performed. We used a reference point (e.g., streams - whenever possible - parks, conservation areas or cities) for each study, except for those carried out in more than one country (where we insert two or more reference points). Developed using the QGIS 3.4 software (map creation) and Microsoft PowerPoint 365® (map formatation). Spatial data (i.e., countries boundaries) was obtained from the Natural Earth database (https://www.naturalearthdata.com/). See Table S3 for the complete list of studies included in this scientometrics.

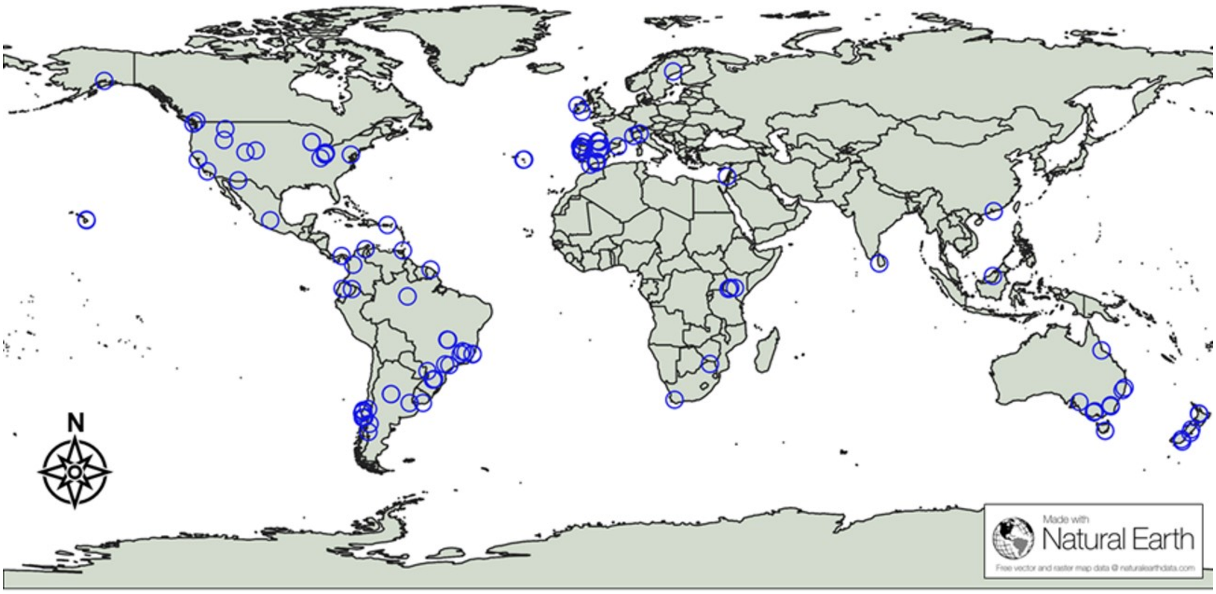
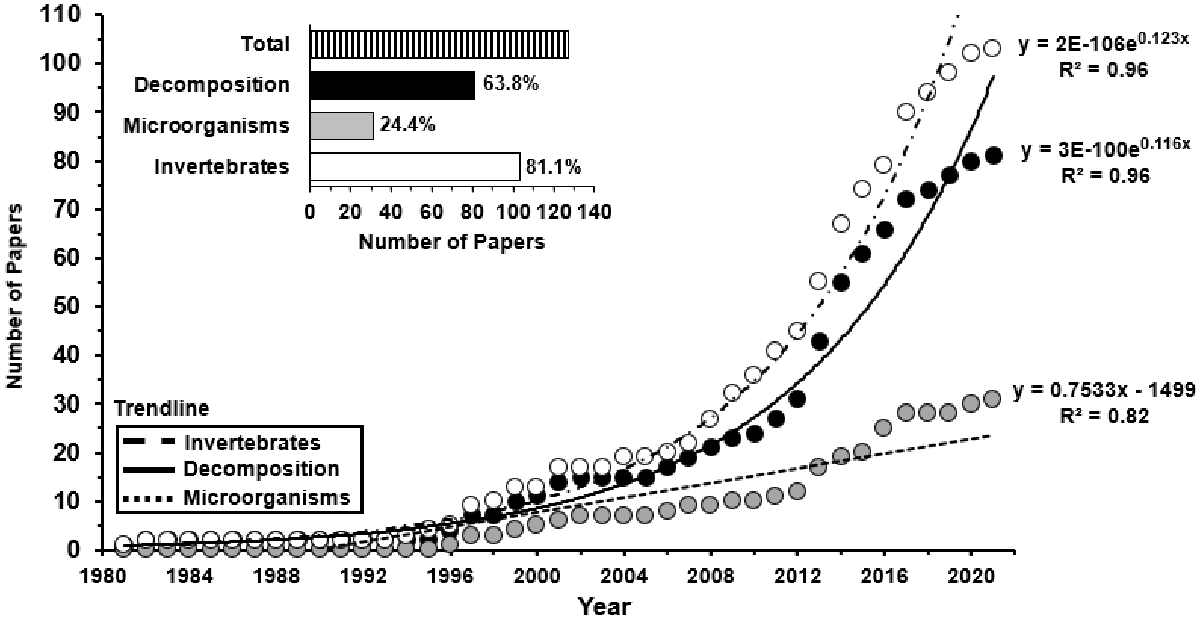


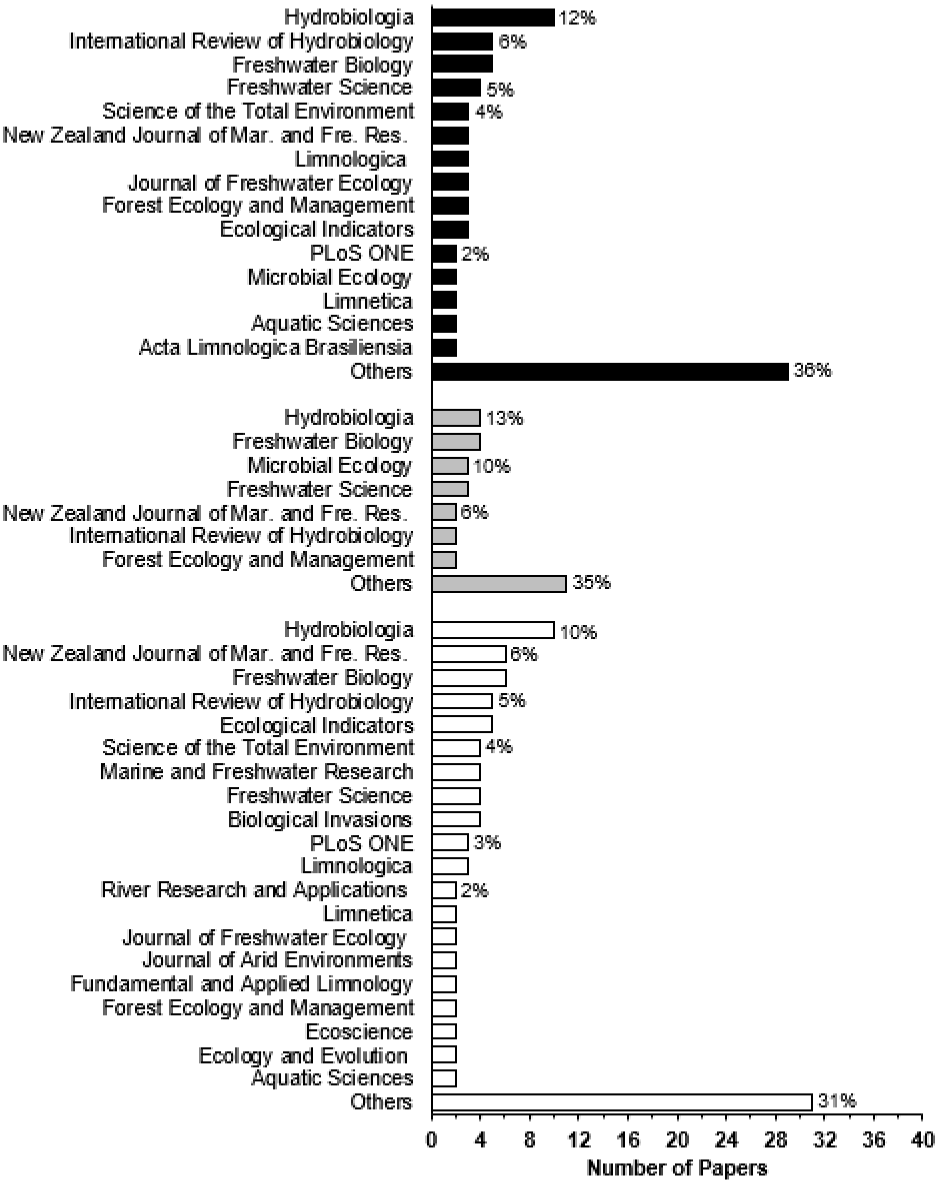
Fig. 2 Cumulative and absolut (in smaller size) number of papers over time approaching the themes *decomposition* (black circles), *microorganisms* (grey circles) and *invertebrates* (white circles) included in this scientometrics (1981-2020). For each, we inform the trendline (exponential for *decomposition* and *invertebrates* and linear for *microorganisms*), the model fit (R^2) and the trendline equation.



Journals

The papers included in this scientometrics were published in 57 journals. Of these, 77% (n = 44 journals) had papers approaching *decomposition*, ~32% (n = 18) had papers approaching *microorganisms*, and ~89% (n = 51) had papers approaching *invertebrates*. 'Hydrobiologia' (n = 12 papers), 'Freshwater Biology' (n = 8), 'New Zealand Journal of Marine and Freshwater Research' (n = 7), 'Ecological Indicators' (n = 6), 'International Review of Hydrobiology' (n = 6), 'Freshwater Science' (n = 6) and 'Limnologica' (n = 5) were the journals with the highest number of published papers (5 papers or more). In these journals, collectively, ~ 41% of the published papers approached *decomposition* (n = 33 papers), ~ 54% (n = 17) of the published papers approached *microorganisms* and ~ 39% (n = 38) of the published papers approached *invertebrates* (Fig. 3; Table S4).

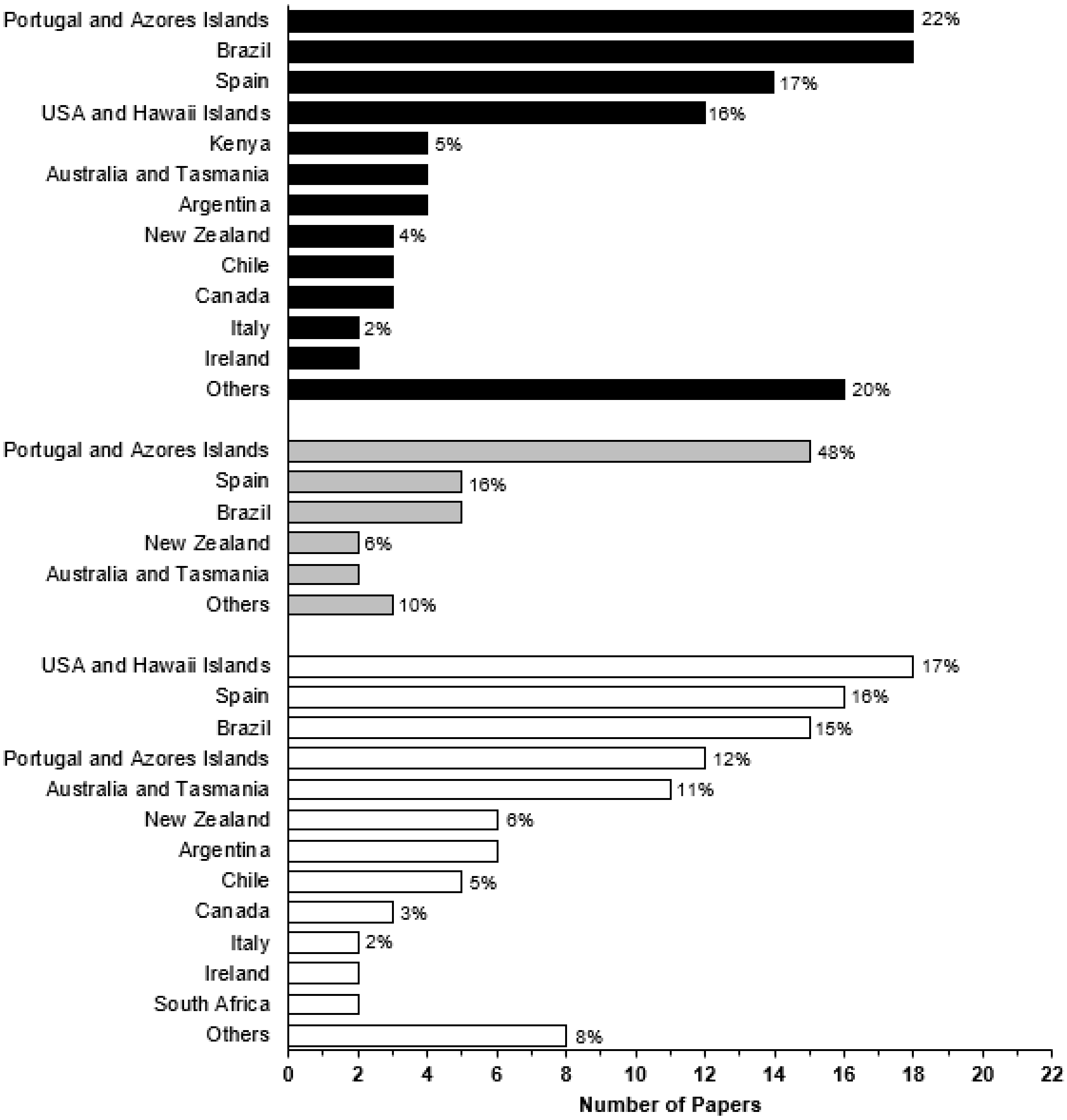
Fig. 3 Journals where two or more papers approaching the themes *decomposition* (black bars), *microorganisms* (grey bars) and *invertebrates* (white bars) were published. Journals with a single published paper were grouped under 'Others'. Next to the bars, we inform the representativity (in percentage) of each journal considering the total number of studies published for each theme. Equal bars have the same representativity. New Zealand Journal of Mar. and Fre. Res. = New Zealand Journal of Marine and Freshwater Research.



Countries

A total of 29 countries (including their respective overseas territories) were registered. Of these, ~97% of the total (n = 28 countries) had papers approaching *decomposition*, ~28% (n = 8) had papers approaching *microorganisms*, and ~69% (n = 20) had papers approaching *invertebrates*. Despite this, only 5 countries (~17% of the listed countries) were indexed in ~76% of published papers: Brazil (n = 22 papers), USA/Hawaii Islands (n = 21), Portugal/Azores Islands (n = 21), Spain (n = 20) and Australia/Tasmania (n = 13). Together, these countries appeared in ~81% (n = 66 papers) of papers approached *decomposition*, ~90% (n = 28) of papers approached *microorganisms*, and ~70% (n = 72) of papers approached *invertebrates* (Fig. 4; Table S5; Table S8).

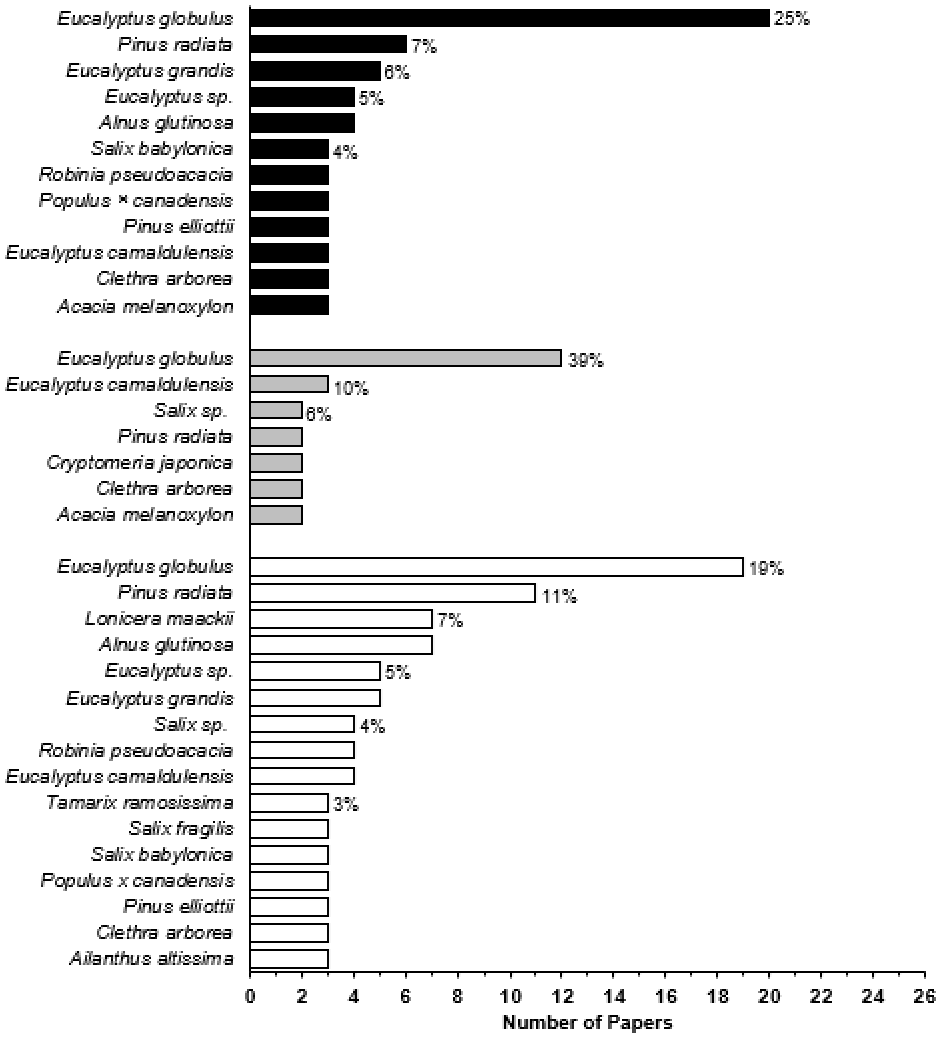
Fig. 4 Countries where two or more papers approaching the themes *decomposition* (black bars), *microorganisms* (grey bars) and *invertebrates* (white bars) carried out their research. Countries with a single published paper were grouped under 'Others'. Next to the bars, we informe the representativity (in percentage) of each country considering the total number of papers published for each theme. Equal bars have the same representativity. Note that some papers carried out their research in more than one country and, therefore, the sum of the representativity of each theme exceeds 100%.



Non-native Species

A total of 89 non-native trees and/or shrubs were recorded in this scientometrics. Of these, ~80% of the total (n = 71 species) of non-native species were listed in papers approaching *decomposition*, ~36% (n = 32) in papers approaching *microorganisms*, and ~93% (n = 83) in papers approaching *invertebrates*. Despite this, only 6 non-native species (~7% of the total) were predominant in ~50% of the published papers: *Eucalyptus globulus* (n = 28 papers), *Pinus radiata* (n = 12), *Alnus glutinosa* (n = 8), *Eucalyptus grandis* (n = 7), *Lonicera maackii* (n = 7) and, *Eucalyptus camaldulensis* (n = 5). Together, these non-native species were recorded in ~49% (n = 40 papers) of papers approached *decomposition*, ~58% (n = 18) of papers approached *microorganisms*, and ~51% (n = 53) of papers approached *invertebrates* (Fig. 5; Table S6; Table S7; Table S8).

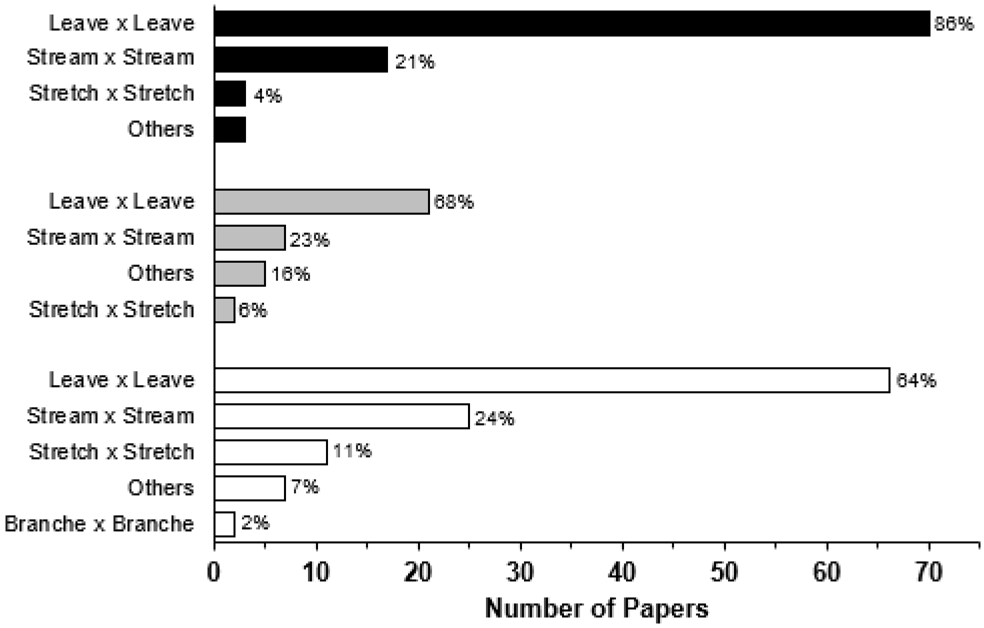
Fig. 5 Non-native trees and/or shrubs recorded in the papers approaching the themes *decomposition* (black bars), *microorganisms* (grey bars) and *invertebrates* (white bars). Due the great number of non-native species, we presented just the data regarding non-native trees and/or shrubs recorded in two or more papers (for *microorganisms*) and in three or more papers (for *decomposition* and *invertebrates*). Next to the bars, we informe the representativity (in percentage) of each species considering the total number of papers published for each theme. Equal bars have the same representativity.



Spacial Scales of Comparation

The spatial scale of comparison 'leave *vs.* leave' was the most representative, appearing in ~86% (n = 70 papers) of papers approaching *decomposition*, in ~68% (n = 21) of papers approaching *microorganisms*, and ~64% (n = 66) of papers approaching *invertebrates*. The spatial scale of comparison 'stream *vs.* stream' was the second most representative, being listed in ~ 21% (n = 17 papers) of papers approaching *decomposition*, in ~ 23% (n = 7) of papers approaching *microorganisms* and ~ 24% (n = 25) of papers approaching *invertebrates*. The other scales, individually, were recorded in less than 20% of the papers for each theme approached (Fig. 6).

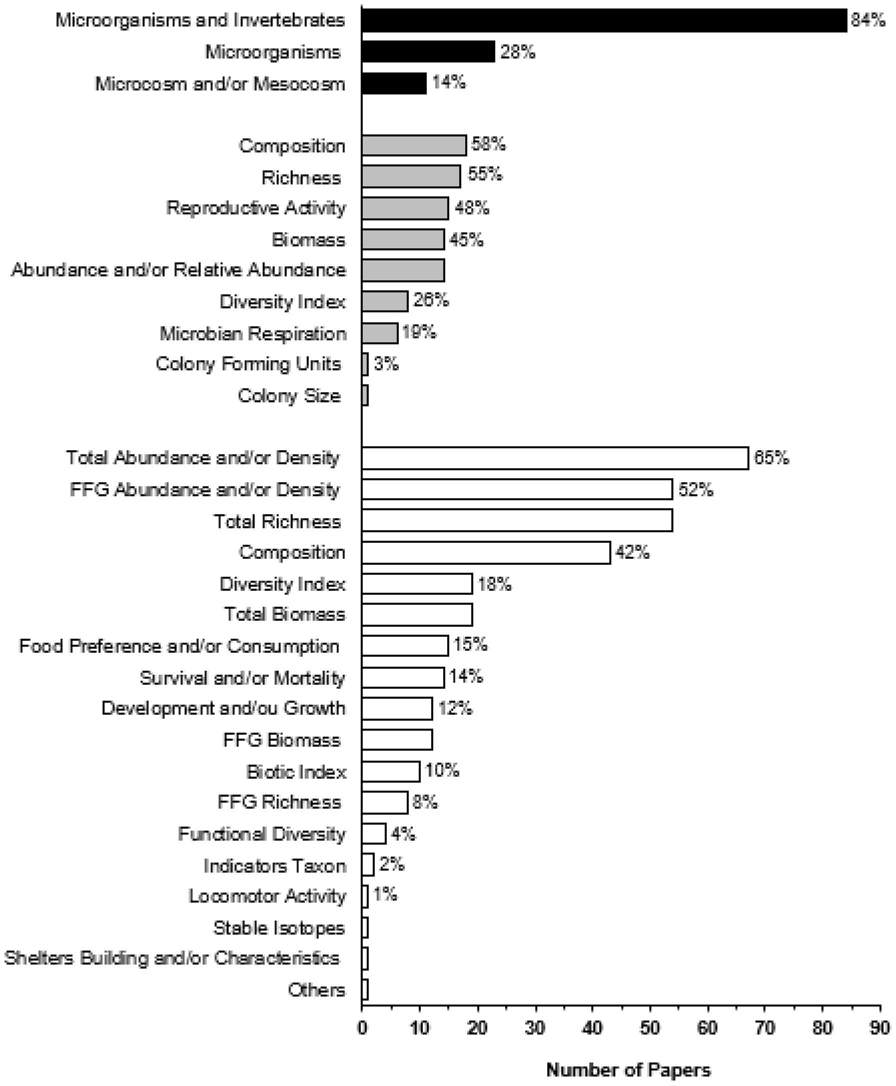
Fig. 6 Spatial scales of comparison recorded in the papers approaching the themes *decomposition* (black bars), *microorganisms* (grey bars) and *invertebrates* (white bars). Next to the bars, we informe the representativity (in percentage) of each spatial scale of comparison considering the total number of papers published for each theme. Equal bars have the same representativity. Note that in some papers were considered more than one spatial scale of comparison and, therefore, the sum of the representativity of each theme exceeds 100%.



Biological Metrics

Considering the papers approaching *decomposition*, the most representative biological metrics were 'leaf decomposition driven by microorganisms and invertebrates' (in ~84% of the total papers; n = 68 papers) and, in second place, 'leaf decomposition driven by microorganisms' (~ 28%; n = 23). Leaf decomposition with microcosm/mesocom experiments carried out in studies appeared in less than 15% of the evaluated papers. For papers approaching *microorganisms*, the most representative biological metrics were, in this order, 'composition' (in ~58% of the total of papers; n = 18 papers), ‘richness’ (~ 55%; n = 17), ‘reproductive activity’ (~ 48%; n = 15), ‘biomass’ and ‘abundance’ (both with ~ 45%; n = 14). The other biological metrics were individually listed in less than 30% of the papers. For papers approaching *invertebrates*, the most representative biological metrics were, in this order, ‘total abundance and/or density’ (in ~65% of the total of papers; n = 67 papers), ‘total richness’ (~52%; n = 54), ‘abundance and/or density of functional feeding groups’ (~ 52%; n = 54) and ‘composition’ (~ 42%; n = 54). The other biological metrics were recorded, separately, in less than 20% of the papers (Fig. 7).

Fig. 7 Main biological metrics recorded in the papers approaching the themes *decomposition* (black bars), *microorganisms* (grey bars) and *invertebrates* (white bars). Next to the bars, we informe the representativity (in percentage) of each biological metric considering the total number of papers published for each theme. Equal bars have the same representativity. Note that some papers were considered more than one biological metric and, therefore, the sum of the representativity of each theme exceeds 100%. FFG = Functional Feeding Groups.



Discussion

Number of Publications

° The number of papers increased over time, specially between 2011-2020

Our results showed an increase in the number of papers over the years, especially during the 2011-2020 decade. This may be related, to a certain extent, to the popularization of internet access and the modernization of the submission, review and publication process (becoming more agile), in conjunction with the access to publications (that became wider). This finding is also partially discussed by other scientometric studies (see Frehse et al., 2016; Brasil et al., 2020). Associated with this, and possibly being the main cause of this increase in the number of papers, is the debate and the growing interest of ecologists in understanding the effects of biological invasions on the structure, composition and functioning of ecosystems (see Ehrenfeld, 2010; Strayer, 2012; Sutherland et al., 2013; Gallardo et al., 2016 and Pišek et al., 2020). Corroborating our results, other scientometric studies approaching biological invasions also demonstrate that the increase in the number of publications in the last two decades appears to be a general trend (see Frehse et al., 2016; Pinto et al., 2020; Santos et al., 2021).

The comparatively smaller number of papers approaching microorganisms is noteworthy. In them, effects on aquatic hyphomycetes were more studied than on other microorganisms (e.g., bacteria). This is possibly related to the fact that aquatic hyphomycetes are considered to be the main decomposing microorganisms of freshwater aquatic ecosystems (Abelho, 2001; Marks, 2019). On the other hand, both aquatic hyphomycetes and aquatic invertebrates play a key role in leaf decomposition in streams (Abelho, 2001; Graça, 2001; Marks, 2019). Thus, we could expect a relatively equitable number of papers approaching microorganisms and invertebrates, which clearly did not occurred. This may be related to different factors. First, the aquatic invertebrates have been known and studied for the longest time: for exemple, studies evaluating the role of aquatic invertebrates on allochthonous processing leaf litter from streams were published as early as the 1930s (see Tank et al., 2010). As a result, a few decades later, aquatic invertebrates were already being used as biological indicators of the streams environmental and ecosystem integrity (e.g., some biological indexes, such as BMWP and HBI, exist since the 1970s) (see Buss et al., 2015; Herman & Nejadhashemi, 2015). Aquatic hyphomycetes, although formally described in the 1940s (by Ingold in 1942), began to be studied extensively within stream ecology especially from the 1970s onwards (see Bärlocher, 2016). Secondly, knowledge of aquatic hyphomycetes in tropical and subtropical streams, although increasing recently, is still smaller when compared to temperate streams (Graça et al., 2015; Duarte et al., 2016; Graça et al., 2016) which possibly means that fewer researchers are studying this topic, and, thus, fewer papers tend to be produced. Thirdly, the methodology required for studying aquatic hyphomycetes tends to be more complex than for aquatic invertebrates (e.g., compare the methodology described for each group in Gonçalves et al., 2012; Menéndez et al., 2013; Rezende et al., 2014 and Seena et al., 2017), which may reflect, consequently, in a smaller number of publications.

Journals

° *The papers were published in journals with scope mostly related to i) freshwater ecosystems; ii) aquatic ecosystems in general; and, iii) ecological research with emphasis on ecological indicators.*

The papers included in this scientometrics were published in a total of 57 journals. The journals with the highest number of papers (n = ≥ 5), tend to publish research mostly related to freshwater ecosystems (‘Freshwater Biology’, ‘Freswater Science’ and ‘Limnologica’), aquatic ecosystems in general (‘Hydrobiologia’, ‘New Zealand Journal of Marine and Freshwater Research’ and ‘International Review of Hydrobiology’) and environmental research with an emphasis on ecological indicators (‘Ecological Indicators’). Only five papers came from journals that publish research exclusively related to biological invasions (‘Biological Invasions’ and ‘Invasive Plant Science and Management’). This finding was proportionately smaller than in other scientometric studies approaching biological invasions (e.g., Barbosa et al., 2012; Frehse et al., 2016). In this scientometrics, the most of the journals with the highest number of papers usually had high academic impact, especially in the disciplinary categories 'Marine and Freshwater Biology', with an Impact Factor between 11 and 53/116 in 2020 (Journal Citation Report – Clarivate, 2021), and 'Aquatic Sciences', located on the first (Q1) and second (Q2) quartiles in 2020 (where Q1 = high and Q4 = low academic impact) (Scientific Journal Rankings; Scimago Journal and Country Rank, 2021).

Countries and Non-Native Species

° *Brazil, USA/Hawaii Islands, Spain, Portugal/Azores Islands and Australia/Tasmania were the most studied countries.*

° *Eucalyptus globulus*, *Pinus radiata, Alnus glutinosa, Eucalyptus grandis*, *Lonicera maackii* and *Eucalyptus camaldulensis were the most studied non-native species.*

The countries and non-native species with the highest number of records were interrelated. The most registered non-native species seem to have a wide occurrence in said countries (including the riparian zone of streams) which may explain the high number of studies found. Therefore, we chose to discuss the main trends associated with countries along with the non-native species. A total of 29 countries were registered in our study. Most studies were carried out in a single country (e.g., Gonçalves et al., 2012; Menéndez et al., 2013), although some were carried out in two countries (e.g., García et al., 2014; Ferreira et al., 2015) or more (e.g., Boyero et al., 2015; Ferreira et al., 2019). Brazil, USA/Hawaii Islands, Spain, Portugal/Azores Islands and Australia/Tasmania concentrated the largest number of studies. In Brazil, Portugal/Azores Islands and Spain, most studies evaluated non-native species that generally establish themselves through monoculture forestry, while in the USA/ Hawaii Islands and Australia/Tasmania, most studies evaluated non-native species established through biological invasion. Altogether, 89 non-native species were recorded, however, over half of the papers were related just to *Eucalyptus globulus, Pinus radiata, Alnus glutinosa, Eucalyptus grandis, Lonicera maackii* and *Eucalyptus camaldulensis.*

In Brazil (e.g., Gonçalves et al., 2012; Tonin et al., 2014), Spain (e.g., Martínez et al., 2013b; Ferreira et al., 2015) and Portugal/Azores Islands (e.g., Raposeiro et al., 2014; Ferreira et al., 2015) the most recorded non-native species were *Eucalyptus* and *Pinus*. *Eucalyptus* and *Pinus* are traditionally cultivated in tree monocultures at large plantations, both in Brazil (Valduga et al., 2016) and in the Iberian Peninsula (Pra et al., 2019). *Eucalyptus* and *Pinus* monocultures are planted in several countries – including Brazil, Portugal and Spain – where, in general, they are cultivated due to economic interests (e.g., wood production) (Chudy & Cubbage, 2020). In the USA/Hawaii Islands, the most common non-native species were *Lonicera maackii* (e.g., McNeish et al., 2015; McNeish et al., 2017) and *Tamarix* (e.g., Going & Dudley, 2008; Moody & Sabo, 2013). Both the *Lonicera maackii* (see McNeish & McEwan, 2016 and Little et al., 2021) and the *Tamarix* species (see Shafroth et al., 2005 and Morisette et al., 2006) are considered invasive and highly endemic in riparian zones at several parts of american territory. In Australia/Tasmania the most registered non-native species were *Alnus glutinosa* (e.g., Ratnarajah & Barmuta, 2009; Boyero et al., 2012) and *Salix* (e.g., McInerney et al., 2016; McInerney & Ress, 2017). *Alnus glutinosa* is considered a non-native species with high invasive potential in several countries (see Keet et al., 2020). In Australia, however, although it can occur in riparian vegetation of streams (McKie & Cranston, 2001), there is not much information about its invasive potential. The high number of studies considering *Alnus* *glutinosa* in Australia/Tasmania seems to be, curiously, associated with the characteristics of its leaves and branches, which are attractive to the colonization and/or consumption of aquatic invertebrates (e.g., Ratnarajah & Barmuta, 1999; McKie & Cranston, 2001). *Salix* species, however, have a high invasive potential in Australia (Cremer, 2003) and high occurrence in riparian zones of streams in this country (Read & Barmuta, 1999; McInerney et al., 2016).

Spacial Scales of Comparation

° *'Leave vs. leave' and 'stream vs. stream' were the most studied spacial scales of comparation.*

For the spatial scales of comparison, 'leave *vs.* leave' and then 'stream *vs.* stream' were the most representative. Most of the papers we evaluated used one (e.g., Tonin et al., 2014; Biasi et al., 2020) or two spatial scales of comparison (e.g., Martínez et al., 2013b; Ferreira et al., 2015). The direct comparison between leaves of native and non-native species under similar conditions (e.g., the same stream) allowed us to evaluate mainly the effect of leaf characteristics on leaf decomposition and communities of aquatic microorganisms and aquatic invertebrates (König et al., 2014; Medina-Villar et al., 2015b; Kuglerová et al., 2017). Nevertheless, this spatial scale of comparison disregards the effects of the quantity and diversity of leaf litter resources available in the streams. Studies comparing native streams *vs.* non-native streams (e.g., Martínez et al., 2013b; Menéndez et al., 2013), or native stretches of water *vs.* non-native stretches of water in a same stream (e.g., McNeish et al., 2015; McNeish et al., 2017), directly or indirectly consider the quantity and diversity of leaf litter resources available in streams and, with this, can assess how this affects the leaf decomposition, as well as the communities of aquatic microorganisms and aquatic invertebrates. In this context, the use of different spatial scales of comparison can be a useful tool in the evaluation of both the effect on leaf characteristics, and on the quantity and availability of native and non-native leaf litter resources in the leaf decomposition, communities of aquatic microorganisms and communities aquatic invertebrates in streams.

Biological Metrics

° *i) Leaf decomposition mediated by microorganisms and invertebrates; ii) structure, composition, reproductive activity and biomass of microorganisms communities; and, iii) structure, composition and trophic structure of invertebrates communities were, respectively, the most studied biological metrics.*

For *decomposition*, 'leaf decomposition mediated by microorganisms and invertebrates' was the most representative biological metric. This seems to indicate a greater interest in studies that overview the leaf decomposition process in streams, while issues associated exclusively with aquatic microorganisms or with microcosm/mesocosm experiments are still proportionately less studied.

For *microorganisms*, 'composition', 'richness', 'reproductive activity', 'biomass' and 'abundance' were the most representative indicators and, in general, referred to aquatic hyphomycetes. Traditionally, these biological metrics are used to assess the aquatic hyphomycete community in streams (e.g., Chauvet et al., 1997; Bärlocher & Graça, 2002; Lecerf & Chauvet, 2008; Menéndez et al., 2013; Sales et al., 2015; Seena et al., 2017; Biasi et al., 2020). Ergosterol analysis and determination of sporulation rates are used, respectively, as proxies for fungal biomass and reproductive activity (Krauss et al., 2011; Canhoto et al., 2016). Richness and abundance, however, are structural characteristics that allows us to assess both the number and the relative importance of each species, while the community structure allows us to assess the existing species. In our study, the structure and composition of the aquatic hyphomycete community was assessed predominantly using traditional taxonomic identification methods, based on the identification and counting of conidia (e.g., Menéndez et al., 2013; Seena et al., 2017). Taxonomic identification of aquatic hyphomycetes based on molecular and genetic methods appears to be an increasing trend, particularly in the last two decades (Bärlocher, 2016). In line with this trend, in our research, two studies used molecular methods to determine the diversity of microorganisms, especially fungi and bacteria (i.e., operational taxonomic units – OTU) (e.g., Fernandes et al., 2013; McInerney et al., 2016).

For *invertebrates*, 'total abundance and/or density', 'richness', 'composition', as well as the 'abundance and/or density of functional feeding groups' were the most recorded biological indicators. These biological metrics are used to assess ecosystem integrity in streams either directly (e.g., richness) or indirectly (e.g., composing different biotic indices such as BMWP) (see Herman & Nejadhashemi, 2015). On that account, our results corroborate Brasil et al. (2020), that found the same biological metrics (except abundance and/or density of functional feeding groups) as the most recorded indicators in a database containing 355 publications. Considering the functional feeding groups, the replacement of native riparian vegetation by non-native species, in many cases, can change the quality of leaf litter resources available in streams. This tends to affect more intensely the abundance/density of shredders (e.g., Martínez et al., 2013b; Ferreira et al., 2015) which may explain the high number of papers that evaluated the abundance and/or density of functional feeding groups.

Most studies, except those approaching *decomposition*, evaluated different biological metrics at the same time. In many cases, some biological indicators tend to be more sensitive to the effects of non-native species than others. Many studies concerning *microorganisms* (e.g., Menéndez et al., 2013; Ferreira et al., 2017; Gonçalves et al., 2017) and *invertebrates* (e.g., Gonçalves et al., 2012; Martínez et al., 2013b; König et al., 2014; Tonin et al., 2014; Ferreira et al., 2015) demonstrate, collectively, significant changes in some biological metrics and no alterations in others. Thus, in a broader ecological context, the use of different biological metrics can be an interesting strategy to obtain a more comprehensive view of the effects of non-native species on leaf decomposition and, especially, on the communities of aquatic microorganisms and aquatic invertebrates.

Conclusions

In summary, in this study we observed that: i) the number of publications increased over time, especially during the 2011-2020 decade, and the *invertebrates* was proportionally more studied than *decomposition* and *microorganisms*; ii) about 40% of the papers were published of only seven journals (~12% of the total), whose scopes were related to aquatic ecosystems (especially freshwater) and ecological indicators - journals specialized in biological invasions presented, proportionally, few publications; iii) about 76% of the papers refer to studies carried out in the territory of only five countries (*Brazil, USA/Hawaii Islands, Spain, Portugal/Azores Islands and Australia/Tasmania,* ~17% of the total); iv) about 50% of the papers were related to only six non-native species (*Eucalyptus globulus*, *Pinus radiata, Alnus glutinosa, Eucalyptus grandis*, *Lonicera maackii* and *Eucalyptus camaldulensis*, ~7% of the total) and, of these, tree non-native species traditionally cultivated into monoculture forests for commercial exploitation (*Eucalyptus* spp. and *Pinus* sp.) were proportionally more studied; v) about 86% of the papers used as a spatial scale of comparison the direct comparison between leaves from native species vs. leaves from non-native species and; vi) for *decomposition*, 'leaf decomposition mediated by microorganisms and invertebrates' as the most representative metric, occurring in ~84% of the papers. For *microorganisms*, 'composition', 'richness', 'reproductive activity', 'biomass' and 'abundance' were the most representative metrics, occurring in at least 45% of the papers. For *invertebrates*, 'abundance and/or total density', 'total richness', 'abundance and/or density of functional feeding groups', 'richness of functional feeding groups' and 'composition' were the most representative metrics, occurring in at least 42% of the papers. We believe that the determination of these trends can help future studies to formulate new questions about ecological aspects still little explored in the evaluation of the effects of non-native tree and/or shrub species on leaf decomposition and the communities of microorganisms and invertebrates in streams.

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Ethics Declarations

Authors Contributions: LEF, LUH and MMP conceived the study design; LEF carried out the search of the interest publications and data extraction. LEF performed the data analysis and conducted the writing of the manuscript with the help of LUH and MMP. All authors critically reviewed the manuscript and agreed to its submission.

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Declaration of interests: The authors declare having no conflicts of interest.

Data Availability: The data used to generate the results in this paper are not available.

Material and/or Code Availability: Not applicable. This study did not use additional materials or codes.

Ethics Approval: Not applicable. This study did not carry out experiments involving humans or animals.

Consent to participate: Not applicable. This study did not carry out experiments involving humans.

Consent for publication: Not applicable. This study did not carry out experiments involving humans.

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Vannote, R. L., Minshall, G. W., Cummins, K. W., Sedell, J. R., & Cushing, C. E. (1980). The River *Continuum* Concept. *Canadian Journal of Fisheries and Aquatic Sciences*, *37*(1), 130-137. https://doi.org/10.1139/f80-017

Wallace, J. B., & Webster, J. R. (1996). The role of macroinvertebrates in stream ecosystem function. *Annual Review of Entomology*, *41*(1), 115-139. https://doi.org/10.1146/annurev.en.41.010196.000555

Webster, J. R., & Benfield, E. F. (1986). Vascular plant breakdown in freshwater ecosystems. *Annual Review of Ecology and Systematics*, *17*(1), 567-594. https://doi.org/10.1146/annurev.es.17.110186.003031

Supplementary Material List

Table S1. Description of the main spatial scales of comparison considered in the papers included in this scientometric for *decomposition*, *microorganisms,* and *invertebrates*. Between brackets we inform examples of papers. See Table S3 for the complete list of papers included in this scientometric.

Spatial scales of comparation	**Explanation**
'Leave *vs*. Leave'	Comparison between leaves from native species *vs.* leaves from non-native species [as in 2, 8, 46, 77, 86 and 122]
'Branche *vs*. Branche'	Comparison between branches from native species *vs.* branches from non-native species [as in 81 and 82]
'Stretch *vs*. Stretch'	Comparison, in the same stream, between stretches with riparian vegetation composed mostly of native species *vs.* stretches with riparian vegetation composed mostly of non-native species [as in 61, 65, 72, 79, 84 and 85]
'Stream *vs*. Stream'	Comparison between streams with riparian vegetation composed mostly of native species *vs.* streams with riparian vegetation composed mostly of non-native species [as in 1, 38, 75, 87, 89 and 99]
'Others'	When no have comparison between native *vs.* non-native species. Evaluation of the effect of litterfall or litterfall extracts from only non-native species [as in 6, 11, 18, 19, 27 and 34]

Table S2. Description of the main metrics considered for *decomposition*, *microorganisms* and *invertebrates* in the papers included in this scientometric. Between brackets we indicate examples of papers. See Table S3 for the complete list of papers included in this scientometric.

Metrics	**Explanation**
Decomposition	
'Leaf decomposition driven by microorganisms and invertebrates'	Leaf decomposition driven together by microorganisms and invertebrates in streams. In general, in studies using coarse mesh litter bags [as in 36, 49, 59, 86 and 122] or leaf peaks [as in 51, 70, 71 and 99]
'Leaf decomposition driven by microorganisms'	Leaf decomposition driven by microorganisms in streams. In general, in studies using fine mesh litter bags [as in 07, 08, 13, 36 and 86]
'Leaf decomposition evaluated in microcosm/mesocosm'	Leaf decomposition evaluated in studies carried out in microcosm/mesocosm (e.g., artificial tanks, Erlenmeyers, aquariums and others) [as in 12, 25, 28, 44 and 60]
Microrganisms	
'Abundance'	Estimates of the abundance (absolute and/or relative) of aquatic hyphomycetes based on conidia identification and counts [as in 23, 32, 44, 87 and 117]
'Biomass'	Estimates of the quantities of ergosterol and ATP produced, alone or together, by aquatic hyphomycetes and/or bacteria [as in 34, 46, 47, 86 and 117]
'Reproductive Activity'	Estimates of sporulation rates and/or conidia production by aquatic hyphomycetes [as in 7, 8, 23, 44 and 77]
'Richness'	Number of taxa identified based on conidia identification and count [as in 6, 8, 23, 87 and 117] or molecular methods [as in 31 and 79].
'Composition'	Estimates of the community composition variation based, specially, on taxonomic abundance of aquatic hyphomycetes [as in 6, 8, 44, 87 and 117]
'Diversity Index'	Estimates of diversity indexes (e.g., Shannon and Simpson diversity index and Evenness index) [as in 7, 23, 77, 87 and 117]
'Microbian Respiration'	Estimation of microbial respiration (from aquatic hyphomycetes and bacteria together) [as in 15, 19, 40, 94 and 101]
'Colony Size'	Estimation of the size of fungal colonies, in general, in studies carried out in microcosm/mesocosm [as in 17]
'Colony Forming Units'	Estimation of colony forming units of bacteria and fungi together and/or separately [as in 113]
Invertebrates	
'Total Abundance and/or Density'	Estimates of abundance (absolute and/or relative) and/or density of the invertebrate community [as in 1, 59, 76, 87 and 118]
'Total Richness'	Estimates of richness (absolute and/or rarefied) of the invertebrate community [as in 34, 47, 59, 89 and 117]
'Total Biomass'	Estimates of biomass of the invertebrate community [as in 21, 46, 63, 109 and 117]
'Functional Feeding Groups Abundance and/or 'Density'	Estimates of abundance (absolute and/or relative) and/or density of one or more functional feeding groups of invertebrates [as in 1, 24, 45, 47 and 62]
'Functional Feeding Groups Richness'	Estimates of richness (absolute and/or rarefied) of one or more functional feeding groups of invertebrates [as in 21, 33, 60, 87 and 89]
'Functional Feeding Groups Biomass'	Estimates of biomass of one or more functional feeding groups of invertebrates [as in 63, 65, 77, 111 and 122]
'Composition'	Estimates of variation in community composition based, specially, on taxonomic abundance and/or functional traits of invertebrates [as in 1, 59, 87, 99 and 120]
'Diversity Index'	Estimates of diversity indexes (e.g., Shannon and Simpson diversity index and Evenness index) [as in 38, 75, 79, 117 and 118]
'Biotic Index'	Estimates of biotic indexes (e.g., BMWP, HBI, QBR, IBF) [as in 24, 38, 95 and 124]
'Functional Diversity'	Estimates of different functional traits of the invertebrate community [as in 14, 62, 85 and 91]
'Indicators Taxon'	Estimates of indicator taxa using, in general, specific statistical tests [as in 74 and 99]
'Stable Isotopes'	Estimates of invertebrate diet based on stable isotope analysis [as in 54]
'Food Preference and/or Consumption'	Estimates of food preference and/or consumption of invertebrates, in general, in studies carried out in microcosm/mesocosm [as in 5, 22, 58, 64 and 93]
'Survival and/or Mortality'	Estimates of invertebrate survival and/or mortality, in general, in studies carried out in microcosm/mesocosm [as in 11, 19, 58 and 93]
'Development and/or Growth'	Estimates of the development and/or growth of invertebrates, in general, in studies carried out in microcosm/mesocosm [as in 5, 18, 43, 58, 93]
'Locomotor Activity'	Estimates of locomotion activity of invertebrates, in general, in studies carried out in microcosm/mesocosm [as in 2]
'Shelters Building and/or Characteristics'	Estimates of the construction process and/or characteristics of invertebrate shelters, in general, in studies carried out in microcosm/mesocosm [as in 26]
'Others'	Estimates of the quantity of proteins and lipids in the body biomass of invertebrates, in general, in studies carried out in microcosm/mesocosm [as in 64]

Table S3. List of papers included in this scientometric. For each paper, we established a number for easy identification, the complete reference, the theme approached (D = *Decomposition*; M = *Microorganisms* and I = *Invertebrates*) and the first database where each paper was obtained (WS = Web of Science; Sc = Scopus; GS = Google Scholar and PL = Personal Library).

Reference	**Theme**			**Database**			
	D	**M**	**I**	**WS**	**Sc**	**GS**	**PL**
1. Abelho, M., & Graça, M. A. S. (1996). Effects of eucalyptus afforestation on leaf litter dynamics and macroinvertebrate community structure of streams in Central Portugal. *Hydrobiologia, 324*(3), 195-204. https://doi.org/10.1007/BF00016391	**x**	**-**	**x**	**-**	**-**	**-**	**x**
2. Alonso, A., de Aldana, B. V., Castro-Díez, P., Medina-Villar, S., & Pérez-Corona, M. E. (2020). Effects of leaf litter extracts from four tree species on aquatic invertebrates: an ecotoxicological risk assessment approach. *Aquatic Ecology, 54*(4), 1155-1168. https://doi.org/10.1007/s10452-020-09800-x	**-**	**-**	**x**	**x**	**-**	**-**	**-**
3. Amuok, E. (2020). Litter processing of Exotic and Indigenous leaves in Njoro River, Kenya. *Egerton Journal of Science and Technology, 17*(1-139), 132-147.	**x**	**-**	**x**	**-**	**-**	**x**	**-**
4. Bailey, J. K., Schweitzer, J. A., & Whitham, T. G. (2001). Salt cedar negatively affects biodiversity of aquatic macroinvertebrates. *Wetlands, 21*(3), 442-447. https://doi.org/10.1672/0277-5212(2001)021[0442:SCNABO]2.0.CO;2	**x**	**-**	**x**	**-**	**x**	**-**	**-**
5. Balibrea, A., Ferreira, V., Gonçalves, V., & Raposeiro, P. M. (2017). Consumption, growth and survival of the endemic stream shredder *Limnephilus atlanticus* (Trichoptera, Limnephilidae) fed with distinct leaf species. *Limnologica, 64*, 31-37. https://doi.org/10.1016/j.limno.2017.04.002	**-**	**-**	**x**	**-**	**-**	**x**	**-**
6. Balibrea, A., Ferreira, V., Balibrea, C., Gonçalves, V., & Raposeiro, P. M. (2020). Contribution of macroinvertebrate shredders and aquatic hyphomycetes to litter decomposition in remote insular streams. *Hydrobiologia, 847*(10), 2337-2355. https://doi.org/10.1007/s10750-020-04259-1	**x**	**x**	**x**	**-**	**-**	**x**	**-**
7. Bärlocher, F., & Graça, M. A. S. (2002). Exotic riparian vegetation lowers fungal diversity but not leaf decomposition in Portuguese streams. *Freshwater Biology, 47*(6), 1123-1135. https://doi.org/10.1046/j.1365-2427.2002.00836.x	**x**	**x**	**-**	**x**	**x**	**-**	**-**
8. Biasi, C., Fontana, L. E., Restello, R. M., & Hepp, L. U. (2020). Effect of invasive *Hovenia dulcis* on microbial decomposition and diversity of hyphomycetes in Atlantic forest streams. *Fungal Ecology, 44*, 100890. https://doi.org/10.1016/j.funeco.2019.100890	**x**	**x**	**-**	**x**	**x**	**-**	**-**
9. Blanco, J. F., & Gutiérrez-Isaza, N. (2014). Leaf litter mass loss rates and associated fauna of tree species commonly used in neotropical riparian reforestation. *Acta Biológica Colombiana, 19*(1), 91-100. https://doi.org/10.15446/abc.v19n1.38155	**x**	**-**	**x**	**-**	**x**	**-**	**-**
10. Bo, T., Cammarata, M., López-Rodríguez, M. J., de Figueroa, J. M. T., Baltieri, M., Varese, P., & Fenoglio, S. (2014). The influence of water quality and macroinvertebrate colonization on the breakdown process of native and exotic leaf types in sub-alpine stream. *Journal of Freshwater Ecology, 29*(2), 159-169. https://doi.org/10.1080/02705060.2013.879538	**x**	**-**	**x**	**x**	**x**	**-**	**-**

Table S3. Continuation...

Reference	Theme			Database			
	D	M	I	WS	Sc	GS	PL
11. Borth, E. B., Custer, K. W., & McEwan, R. W. (2018). Lethal effects of leaf leachate from the non-native invasive shrub Amur honeysuckle (*Lonicera maackii*) on a model aquatic organism (*Hyalella azteca*). *Écoscience, 25*(2), 189-197. https://doi.org/10.1080/11956860.2018.1426261	-	-	x	x	-	-	-
12. Boyero, L., Barmuta, L. A., Ratnarajah, L., Schmidt, K., & Pearson, R. G. (2012). Effects of exotic riparian vegetation on leaf breakdown by shredders: a tropical–temperate comparison. *Freshwater Science, 31*(2), 296-303. https://doi.org/10.1899/11-103.1	x	-	x	x	-	-	-
13. Boyero, L., Pearson, R. G., Gessner, M. O., Dudgeon, D., Ramírez, A., Yule, C. M., Callisto, M., Pringle, C. M., Encalada, A. C., Arunachalam, M., Mathooko, J., Helson, J. E., Rincón, J., Bruder, A., Cornejo, A., Flecker, A. S., Mathuriau, C., M'Erimba, C., Gonçalves, J. F., Moretti, M., & Jinggut, T. (2015). Leaf-litter breakdown in tropical streams: is variability the norm?. *Freshwater Science, 34*(2), 759-769. https://doi.org/10.1086/681093	x	-	-	-	-	x	-
14. Cabrini, R., Canobbio, S., Sartori, L., Fornaroli, R., & Mezzanotte, V. (2013). Leaf packs in impaired streams: the influence of leaf type and environmental gradients on breakdown rate and invertebrate assemblage composition. *Water, Air, & Soil Pollution*, *224*(10), 1-13. https://doi.org/10.1007/s11270-013-1697-8	x	-	x	-	-	x	-
15. Campos, D., Alves, A., Lemos, M. F., Correia, A., Soares, A. M., & Pestana, J. L. (2014). Effects of cadmium and resource quality on freshwater detritus processing chains: a microcosm approach with two insect species. *Ecotoxicology, 23*(5), 830-839. https://doi.org/10.1007/s10646-014-1223-9	x	x	x	-	-	x	-
16. Canhoto, C., & Graça, M. A. S. (1996). Decomposition of *Eucalyptus globulus* leaves and three native leaf species (*Alnus glutinosa, Castanea sativa* and *Quercus faginea*) in a Portuguese low order stream. *Hydrobiologia, 333*(2), 79-85. https://doi.org/10.1007/BF00017570	x	x	-	x	x	-	-
17. Canhoto, C., & Graça, M. A. S. (1999). Leaf barriers to fungal colonization and shredders (*Tipula lateralis*) consumption of decomposing *Eucalyptus globulus*. *Microbial Ecology, 37*(3), 163-172. https://doi.org/10.1007/s002489900140	-	x	x	-	-	-	x
18. Canhoto, C., & Laranjeira, C. (2007). Leachates of *Eucalyptus globulus* in intermittent streams affect water parameters and invertebrates. *International Review of Hydrobiology*, *92*(2), 173-182. https://doi.org/10.1002/iroh.200510956	-	-	x	-	-	x	-
19. Canhoto, C., Calapez, R., Gonçalves, A. L., & Moreira-Santos, M. (2013). Effects of *Eucalyptus* leachates and oxygen on leaf-litter processing by fungi and stream invertebrates. *Freshwater Science, 32*(2), 411-424. https://doi.org/10.1899/12-062.1	x	x	x	-	-	x	-
20. Casas, J. J., & Gessner, M. O. (1999). Leaf litter breakdown in a Mediterranean stream characterised by travertine precipitation. *Freshwater Biology, 41*(4), 781-793. https://doi.org/10.1046/j.1365-2427.1999.00417.x	x	-	-	-	-	x	-

Table S3. Continuation...

Reference	**Theme**			**Database**			
	D	**M**	**I**	**WS**	**Sc**	**GS**	**PL**
21. Casas, J. J., Larrañaga, A., Menéndez, M., Pozo, J., Basaguren, A., Martínez, A., Perez, J., Gonzalez, J. M., Molla, S., Casado, C., Descals, E., Roblas, N., Lopez-Gonzalez, J. A. & Valenzuela, J. L. (2013). Leaf litter decomposition of native and introduced tree species of contrasting quality in headwater streams: how does the regional setting matter? *Science of Total Environmental 458–460*, 197–208. https://doi.org/10.1016/j.scitotenv.2013.04.004	x	-	x	x	-	-	-
22. Casotti, C. G., Kiffer, W. P., & Moretti, M. S. (2014). Leaf traits induce the feeding preference of a shredder of the genus *Triplectides* Kolenati, 1859 (Trichoptera) in an Atlantic Forest stream, Brazil: a test with native and exotic leaves. *Aquatic Insects, 36*(1), 43-52. https://doi.org/10.1080/01650424.2014.1001399	-	-	x	x	-	-	-
23. Chauvet, E., Fabre, E., Elosegui, A., & Pozo, J. (1997). The impact of eucalypt on the leaf-associated aquatic hyphomycetes in Spanish streams. *Canadian Journal of Botany, 75*(6), 880-887. https://doi.org/10.1139/b97-097	-	x		x	x	-	-
24. Chellaiah, D., & Yule, C. M. (2018). Riparian buffers mitigate impacts of oil palm plantations on aquatic macroinvertebrate community structure in tropical streams of Borneo. *Ecological Indicators, 95*, 53-62. https://doi.org/10.1016/j.ecolind.2018.07.025	-	-	x	x	x	-	-
25. Correa-Araneda, F., Boyero, L., Figueroa, R., Sánchez, C., Abdala, R., Ruiz-García, A., & Graça, M. A. S. (2015). Joint effects of climate warming and exotic litter (*Eucalyptus globulus* Labill.) on stream detritivore fitness and litter breakdown. *Aquatic Sciences, 77*(2), 197-205. https://doi.org/10.1007/s00027-014-0379-y	x	-	x	x	x	-	-
26. Correa-Araneda, F., Basaguren, A., Abdala-Díaz, R. T., Tonin, A. M., & Boyero, L. (2017). Resource-allocation tradeoffs in caddisflies facing multiple stressors. *Ecology and Evolution, 7*(14), 5103-5110. https://doi.org/10.1002/ece3.3094	-	-	x	-	-	-	x
27. Custer, K. W., Borth, E. B., Mahoney, S. D., & McEwan, R. W. (2017). Lethal and sublethal effects of novel terrestrial subsidies from an invasive shrub (*Lonicera maackii*) on stream macroinvertebrates. *Freshwater Science, 36*(4), 750-759. https://doi.org/10.1086/694895	-	-	x	x	-	-	-
28. Da Silva, D. J., Valduga, A. T., Molozzi, J., Fornel, R., Restello, R. M., & Hepp, L. U. (2018). Leaching of carbon from native and non-native leaf litter of subtropical riparian forests. *Journal of Limnology, 77*(2), 247-254. https://doi.org/10.4081/jlimnol.2018.1662	x	-	-	x	x	-	-
29. Davies, J. N., & Boulton, A. J. (2009). Great house, poor food: effects of exotic leaf litter on shredder densities and caddisfly growth in 6 subtropical Australian streams. *Journal of the North American Benthological Society, 28*(2), 491-503. https://doi.org/10.1899/07-073.1	-	-	x	x	-	-	-
30. Fargen, C., Emery, S. M., & Carreiro, M. M. (2015). Influence of *Lonicera maackii* invasion on leaf litter decomposition and macroinvertebrate communities in an urban stream. *Natural Areas Journal, 35*(3), 392-403. https://doi.org/10.3375/043.035.0303	x	-	x	x	x	-	-

Table S3. Continuation...

Reference	Theme			Database			
	D	M	I	WS	Sc	GS	PL
31. Fernandes, I., Duarte, S., Cássio, F., & Pascoal, C. (2013). Effects of riparian plant diversity loss on aquatic microbial decomposers become more pronounced with increasing time. *Microbial Ecology, 66*(4), 763-772. https://doi.org/10.1007/s00248-013-0278-0	x	x	-	-	-	x	-
32. Ferreira, V., Elosegi, A., Gulis, V., Pozo, J., & Graça, M. A. S. (2006). *Eucalyptus* plantations affect fungal communities associated with leaf-litter decomposition in Iberian streams. *Archiv fur Hydrobiologie, 166*(4), 467-490. https://doi.org/10.1127/0003-9136/2006/0166-0467	x	x	-	-	-	x	-
33. Ferreira, V., Larrañaga, A., Gulis, V., Basaguren, A., Elosegi, A., Graça, M. A. S., & Pozo, J. (2015). The effects of eucalypt plantations on plant litter decomposition and macroinvertebrate communities in Iberian streams. *Forest Ecology and Management, 335*, 129-138. https://doi.org/10.1016/j.foreco.2014.09.013	x	-	x	x	x	-	-
34. Ferreira, V., Raposeiro, P. M., Pereira, A., Cruz, A. M., Costa, A. C., Graça, M. A. S., & Gonçalves, V. (2016). Leaf litter decomposition in remote oceanic island streams is driven by microbes and depends on litter quality and environmental conditions. *Freshwater Biology, 61*(5), 783-799. https://doi.org/10.1111/fwb.12749	x	x	x	-	-	x	-
35. Ferreira, V., Faustino, H., Raposeiro, P. M., & Gonçalves, V. (2017). Replacement of native forests by conifer plantations affects fungal decomposer community structure but not litter decomposition in Atlantic island streams. *Forest Ecology and Management, 389*, 323-330. https://doi.org/10.1016/j.foreco.2017.01.004	x	x	-	-	-	x	-
36. Ferreira, V., Boyero, L., Calvo, C., Correa, F., Figueroa, R., Gonçalves, J. F., Goyenola, G., Graça, M. A. S., Hepp, L. U., Kariuki, S., López-Rodríguez, A., Mazzeo, N., M'Erimba, C., Monroy, S., Peil, A., Pozo, J., Rezende, R. & Teixeira-de-Mello, F. (2019). A global assessment of the effects of *Eucalyptus* plantations on stream ecosystem functioning. *Ecosystems, 22*(3), 629-642. https://doi.org/10.1007/s10021-018-0292-7	x	-	-	-	-	x	-
37. Fierro, P., Quilodrán, L., Bertrán, C., Arismendi, I., Tapia, J., Pena-Cortés, F., Hauensteine, E., Arriagadaa, R., Fernández, E. & Vargas-Chacoff, L. (2016). Rainbow trout diets and macroinvertebrates assemblages responses from watersheds dominated by native and exotic plantations. *Ecological Indicators, 60*, 655-667. https://doi.org/10.1016/j.ecolind.2015.08.018	-	-	x	x	x	-	-
38. Fierro, P., Bertrán, C., Tapia, J., Hauenstein, E., Peña-Cortés, F., Vergara, C., Cerna, C. & Vargas-Chacoff, L. (2017). Effects of local land-use on riparian vegetation, water quality, and the functional organization of macroinvertebrate assemblages. *Science of the Total Environment, 609*, 724-734. https://doi.org/10.1016/j.scitotenv.2017.07.197	-	-	x	x	x	-	-
39. Freund, J. G., Thobaben, E., Barkowski, N., & Reijo, C. (2013). Rapid in-stream decomposition of leaves of common buckthorn (*Rhamnus cathartica*), an invasive tree species. *Journal of Freshwater Ecology, 28*(3), 355-363. https://doi.org/10.1080/02705060.2013.770802	x	-	-	-	-	x	-
40. Gama, M., Gonçalves, A. L., Ferreira, V., Graça, M. A. S., & Canhoto, C. (2007). Decomposition of fire exposed *Eucalyptus* leaves in a Portuguese lowland stream. *International Review of Hydrobiology, 92*(3), 229-241. https://doi.org/10.1002/iroh.200610916	x	x	-	-	-	x	-

Table S3. Continuation...

Reference	Theme			Database			
	D	M	I	WS	Sc	GS	PL
41. Gantes, P., Marano, A. V., & Rigacci, L. (2011). Changes in the decomposition process associated with the invasion of *Gleditsia triacanthos* (honey locust) in pampean streams (Buenos Aires, Argentina). *Journal of Freshwater Ecology, 26*(4), 481-494. https://doi.org/10.1080/02705060.2011.578397	x	x	-	-	-	-	x
42. García, L., Pardo, I., & Richardson, J. S. (2014). A cross-continental comparison of stream invertebrate community assembly to assess convergence in forested headwater streams. *Aquatic Sciences, 76*(1), 29-40. https://doi.org/10.1007/s00027-013-0308-5	x	-	x	-	-	x	-
43. Going, B. M., & Dudley, T. L. (2008). Invasive riparian plant litter alters aquatic insect growth. *Biological Invasions, 10*(7), 1041-1051. https://doi.org/10.1007/s10530-007-9182-1	-	-	x	x	x	-	-
44. Gomes, P. P., Medeiros, A. O., & Gonçalves, J. F. (2016). The replacement of native plants by exotic species may affect the colonization and reproduction of aquatic hyphomycetes. *Limnologica, 59*, 124-130. https://doi.org/10.1016/j.limno.2016.05.005	x	x	-	x	x	-	-
45. Gonçalves, J. F., Rezende, R. S., França, J., & Callisto, M. (2012a). Invertebrate colonisation during leaf processing of native, exotic and artificial detritus in a tropical stream. *Marine and Freshwater Research, 63*(5), 428-439. http://dx.doi.org/10.1071/MF11172	x	-	x	-	-	x	-
46. Gonçalves, J. F., Rezende, R. S., Martins, N. M., & Gregório, R. S. (2012b). Leaf breakdown in an Atlantic Rain Forest stream. *Austral Ecology, 37*(7), 807-815. http://dx.doi.org/10.1111/j.1442-9993.2011.02341.x	x	x	x	x	x	-	-
47. Gonçalves, J. F., Couceiro, S. R. M., Rezende, R. S., Martins, R. T., Ottoni-Boldrini, B. M. P., Campos, C. M., Silva, J. O., & Hamada, N. (2017). Factors controlling leaf litter breakdown in Amazonian streams. *Hydrobiologia, 792*(1), 195-207. http://dx.doi.org/10.1007/s10750-016-3056-4	x	x	x	-	-	x	-
48. Harding, J. S., & Winterbourn, M. J. (1995). Effects of contrasting land use on physico-chemical conditions and benthic assemblages of streams in a Canterbury (South Island, New Zealand) river system. *New Zealand Journal of Marine and Freshwater Research, 29*(4), 479-492. https://doi.org/10.1080/00288330.1995.9516681	-	-	x	-	x	-	-
49. Hepp, L. U., Biasi, C., Milesi, S. V., Veiga, F. O., & Restello, R. M. (2008). Chironomidae (Diptera) larvae associated to *Eucalyptus globulus* and *Eugenia uniflora* leaf litter in a subtropical stream (Rio Grande do Sul, Brazil). *Acta Limnologica Brasiliensia, 20*(4), 345-350.	x	-	x	-	-	x	-
50. Hepp, L. U., Delanora, R., & Trevisan, A. (2009). Compostos secundários durante a decomposição foliar de espécies arbóreas em um riacho do sul do Brasil. *Acta Botanica Brasilica, 23*, 407-413. https://doi.org/10.1590/S0102-33062009000200012	x	-	-	-	-	-	x

Table S3. Continuation...

Reference	Theme			Database			
	D	**M**	**I**	**WS**	**Sc**	**GS**	**PL**
51. Herbst, G., & Reice, S. R. (1982). Comparative leaf litter decomposition in temporary and permanent streams in semi-arid regions of Israel. *Journal of Arid Environments, 5*(4), 305-318. https://doi.org/10.1016/S0140-1963(18)31612-4	x	-	x	-	-	x	-
52. Hicks, B. J., & Laboyrie, J. L. (1999). Preliminary estimates of mass-loss rates, changes in stable isotope composition, and invertebrate colonisation of evergreen and deciduous leaves in a Waikato, New Zealand, stream. *New Zealand Journal of Marine and Freshwater Research*, *33,* 221-232 https://doi.org/10.1080/00288330.1999.9516872	x	-	x	x	-	-	-
53. Hladyz, S., Gessner, M. O., Giller, P. S., Pozo, J., & Woodward, G. U. Y. (2009). Resource quality and stoichiometric constraints on stream ecosystem functioning. *Freshwater Biology, 54*(5), 957-970. https://doi.org/10.1111/j.1365-2427.2008.02138.x	x	x	x	x	x	-	-
54. Hladyz, S., Åbjörnsson, K., Giller, P. S., & Woodward, G. (2011). Impacts of an aggressive riparian invader on community structure and ecosystem functioning in stream food webs. *Journal of Applied Ecology, 48*(2), 443-452. https://doi.org/10.1111/j.1365-2664.2010.01924.x	x	-	x	x	x	-	-
55. Jayawardana, J. M. C. K., & Westbrooke, M. (2010). Potential effects of riparian vegetation changes on functional organisation of macroinvertebrates in central Victorian streams. *The Victorian Naturalist, 127*(2), 36-48.	-	-	x	-	x	-	-
56. Jayawardana, J. M. C. K. (2011). Littoral macroinvertebrates in relation to native and exotic riparian vegetation in streams of central Victoria, Australia. *International Journal of Energy Environment and Economics, 19*(6), 635-351.	-	-	x	-	x	-	-
57. Kennedy, K. T. M., & El-Sabaawi, R. W. (2018). Decay patterns of invasive plants and plastic trash in urban streams. *Urban Ecosystems, 21*(5), 817-830. https://doi.org/10.1007/s11252-018-0771-9	x	-	x	-		x	-
58. Kiffer, W. P., Mendes, F., Casotti, C. G., Costa, L. C., & Moretti, M. S. (2018). Exotic *Eucalyptus* leaves are preferred over tougher native species but affect the growth and survival of shredders in an Atlantic Forest stream (Brazil). *PLoS One, 13*(1), e0190743. https://doi.org/10.1371/journal.pone.0190743	-	-	x	-		x	-
59. König, R., Hepp, L. U., & Santos, S. (2014). Colonisation of low-and high-quality detritus by benthic macroinvertebrates during leaf breakdown in a subtropical stream. *Limnologica, 45*, 61-68. https://doi.org/10.1016/j.limno.2013.11.001	x	-	x	x	x	-	-
60. Kuglerová, L., García, L., Pardo, I., Mottiar, Y., & Richardson, J. S. (2017). Does leaf litter from invasive plants contribute the same support of a stream ecosystem function as native vegetation?. *Ecosphere, 8*(4), e01779. https://doi.org/10.1002/ecs2.1779	x	-	x	x	x	-	-

Table S3. Continuation...

Reference	Theme			Database			
	D	M	I	WS	Sc	GS	PL
61. Laćan, I., Resh, V. H., & McBride, J. R. (2010). Similar breakdown rates and benthic macroinvertebrate assemblages on native and *Eucalyptus globulus* leaf litter in Californian streams. *Freshwater Biology, 55*(4), 739-752. https://doi.org/10.1111/j.1365-2427.2009.02312.x	x	-	x	-		x	-
62. Larrañaga, A., Basaguren, A., Elosegi, A., & Pozo, J. (2009). Impacts of *Eucalyptus globulus* plantations on Atlantic streams: changes in invertebrate density and shredder traits. *Fundamental and Applied Limnology, 175*(2), 151. https://doi.org/10.1127/1863-9135/2009/0175-0151	-	-	x	-		x	-
63. Larrañaga, S., Larrañaga, A., Basaguren, A., Elosegi, A., & Pozo, J. (2014a). Effects of exotic eucalypt plantations on organic matter processing in Iberian streams. *International Review of Hydrobiology, 99*(5), 363-372. https://doi.org/10.1002/iroh.201301665	x	-	x	x	x	-	-
64. Larrañaga, A., Basaguren, A., & Pozo, J. (2014b). Resource quality controls detritivore consumption, growth, survival and body condition recovery of reproducing females. *Marine and Freshwater Research, 65*(10), 910-917. https://doi.org/10.1071/MF13165	-	-	x	-		x	-
65. Lester, P. J., Mitchell, S. F., & Scott, D. (1994). Effects of riparian willow trees (*Salix fragilis*) on macroinvertebrate densities in two small Central Otago, New Zealand, streams. *New Zealand Journal of Marine and Freshwater Research*, *28*(3), 267-276. https://doi.org/10.1080/00288330.1994.9516614	-	-	x	-	x	-	-
66. Lewis, S. E., Freund, J. G., & Beaver, M. (2017). Consumption of native green ash and nonnative common buckthorn leaves by the amphipod *Gammarus pseudolimnaeus*. *The American Midland Naturalist, 177*(1), 100-111. https://doi.org/10.1674/0003-0031-177.1.100	-	-	x	-		x	-
67. Li, A. O., & Dudgeon, D. (2008). The effects of leaf litter characteristics on feeding and fitness of a tropical stream shredder, *Anisocentropus maculatus* (Trichoptera: Calamoceratidae). *Marine and Freshwater Research, 59*(10), 897-901. https://doi.org/10.1071/MF08120	-	-	x	-	-	x	-
68. Lidman, J., Jonsson, M., Burrows, R. M., Bundschuh, M., & Sponseller, R. A. (2017). Composition of riparian litter input regulates organic matter decomposition: Implications for headwater stream functioning in a managed forest landscape. *Ecology and Evolution, 7*(4), 1068-1077. https://doi.org/10.1002/ece3.2726	x	-	x	x	x	-	-
69. Little, M. N., Custer, K. W., Borth, E. B., Chapman, J. I., Kukla, M. J., Kuminecz, C., Maloney, M. E., Woods, M. J., McEwan, R. W. (2021). The influence of riparian invasion by the terrestrial shrub *Lonicera maackii* on aquatic macroinvertebrates in temperate forest headwater streams. *Biological Invasions, 23*(1), 25-35. https://doi.org/10.1007/s10530-020-02349-8	-	-	x	x	-	-	-
70. López, E. S., Felpeto, N., & Pardo, I. (1997). Comparisons of methods to study the processing of *Alnus glutinosa* and *Eucalyptus globulus* leaves in a forested headwater stream. *Limnetica, 13*(2), 13-18.	x	-	x	-	-	x	-

Table S3. Continuation...

Reference	Theme			Database			
	D	M	I	WS	Sc	GS	PL
71. MacKenzie, R. A., Wiegner, T. N., Kinslow, F., Cormier, N., & Strauch, A. M. (2013). Leaf-litter inputs from an invasive nitrogen-fixing tree influence organic-matter dynamics and nitrogen inputs in a Hawaiian river. *Freshwater Science, 32*(3), 1036-1052. https://doi.org/10.1899/12-152.1	x	x	x	-	-	x	-
72. Magoba, R. N., & Samways, M. J. (2010). Recovery of benthic macroinvertebrate and adult dragonfly assemblages in response to large scale removal of riparian invasive alien trees. *Journal of Insect Conservation, 14*(6), 627-636. https://doi.org/10.1007/s10841-010-9291-5	-	-	x	x	x	-	-
73. Mancilla, G., Valdovinos, C., Azocar, M., Jorquera, P., & Figueroa, R. (2009). Replacement effect of riparian native vegetation on benthic macroinvertebrates community in temperate climate streams, Central Chile. *Hidrobiologica, 19*(3), 193-203.	-	-	x	x	x	-	-
74. Márquez, J. A., Cibils, L., Principe, R. E., & Albariño, R. J. (2015). Stream macroinvertebrate communities change with grassland afforestation in central Argentina. *Limnologica, 53*, 17-25. https://doi.org/10.1016/j.limno.2015.05.002	-	-	x	x	-	-	-
75. Márquez, J. A., Principe, R. E., Martina, L. C., & Albariño, R. J. (2017). Pine needle litter acts as habitat but not as food source for stream invertebrates. *International Review of Hydrobiology, 102*(1-2), 29-37. https://doi.org/10.1002/iroh.201601856	x	-	x	x	-	-	-
76. Martínez, A., Larrañaga, A., Pérez, J., Basaguren, A., & Pozo, J. (2013a). Leaf-litter quality effects on stream ecosystem functioning: a comparison among five species. *Fundamental and Applied Limnology, 183*, 239-248. https://doi.org/10.1127/1863-9135/2013/0514	x	-	x	x	x	-	-
77. Martínez, A., Larrañaga, A., Pérez, J., Descals, E., Basaguren, A., & Pozo, J. (2013b). Effects of pine plantations on structural and functional attributes of forested streams. *Forest Ecology and Management, 310*, 147-155. https://doi.org/10.1016/j.foreco.2013.08.024	x	x	x	-	-	x	-
78. Masese, F. O., Kitaka, N., Kipkemboi, J., Gettel, G. M., Irvine, K., & McClain, M. E. (2014). Litter processing and shredder distribution as indicators of riparian and catchment influences on ecological health of tropical streams. *Ecological Indicators, 46*, 23-37. https://doi.org/10.1016/j.ecolind.2014.05.032	x	-	-	x	x	-	-
79. McInerney, P. J., Rees, G. N., Gawne, B., Suter, P., Watson, G., & Stoffels, R. J. (2016). Invasive willows drive instream community structure. *Freshwater Biology, 61*(9), 1379-1391. https://doi.org/10.1111/fwb.12778	-	x	x	x	-	-	-
80. McInerney, P. J., & Rees, G. N. (2017). Co-invasion hypothesis explains microbial community structure changes in upland streams affected by riparian invader. *Freshwater Science, 36*(2), 297-306. https://doi.org/10.1086/692068	-	x	-	-	x	-	-

Table S3. Continuation...

Reference	Theme			Database			
	D	**M**	**I**	**WS**	**Sc**	**GS**	**PL**
81. McKie, B. G. L., & Cranston, P. S. (1998). Keystone coleopterans? Colonization by wood-feeding elmids of experimentally immersed woods in south-eastern Australia. *Marine and Freshwater Research, 49*(1), 79-88. https://doi.org/10.1071/MF97086	**-**	**-**	**x**	**x**	**-**	**-**	**-**
82. McKie, B., & Cranston, P. S. (2001). Colonisation of experimentally immersed wood in south eastern Australia: responses of feeding groups to changes in riparian vegetation. *Hydrobiologia, 452*(1), 1-14. https://doi.org/10.1023/A:1011974813551	**-**	**-**	**x**	**x**	**x**	**-**	**-**
83. McNeish, R. E., Benbow, M. E., & McEwan, R. W. (2012). Riparian forest invasion by a terrestrial shrub (*Lonicera maackii*) impacts aquatic biota and organic matter processing in headwater streams. *Biological Invasions, 14*(9), 1881-1893. https://doi.org/10.1007/s10530-012-0199-8	**x**	**-**	**x**	**x**	**-**	**-**	**-**
84. McNeish, R. E., Moore, E. M., Benbow, M. E., & McEwan, R. W. (2015). Removal of the invasive shrub, *Lonicera maackii*, from riparian forests influences headwater stream biota and ecosystem function. *River Research and Applications, 31*(9), 1131-1139. https://doi.org/10.1002/rra.2808	**-**	**-**	**x**	**x**	**x**	**-**	**-**
85. McNeish, R. E., Benbow, M. E., & McEwan, R. W. (2017). Removal of the invasive shrub, *Lonicera maackii* (Amur honeysuckle), from a headwater stream riparian zone shifts taxonomic and functional composition of the aquatic biota. *Invasive Plant Science and Management, 10*(3), 232-246. https://doi.org/10.1017/inp.2017.22	**-**	**-**	**x**	**x**	**x**	**-**	**-**
86. Medina-Villar, S., Alonso, F. Á., De Aldana, B. R. V., Corona, M. E. P, & Castro-Díez, M. P. (2015). Decomposition and biological colonization of native and exotic leaf litter in a stream. *Limnetica, 34* (2), 293-310. http://hdl.handle.net/10017/35685	**x**	**x**	**x**	**x**	**x**	**-**	**-**
87. Menéndez, M., Descals, E., Riera, T., & Moya, O. (2013). Do non-native *Platanus hybrida* riparian plantations affect leaf litter decomposition in streams?. *Hydrobiologia, 716*(1), 5-20. https://doi.org/10.1007/s10750-013-1539-0	**x**	**x**	**x**	**x**	**x**	**-**	**-**
88. Miserendino, M. L., & Pizzolon, L. A. (2004). Interactive effects of basin features and land-use change on macroinvertebrate communities of headwater streams in the Patagonian Andes. *River Research and Applications, 20*(8), 967-983. https://doi.org/10.1002/rra.798	**-**	**-**	**x**	**x**	**-**	**-**	**-**
89. Miserendino, M. L., & Masi, C. I. (2010). The effects of land use on environmental features and functional organization of macroinvertebrate communities in Patagonian low order streams. *Ecological Indicators, 10*(2), 311-319. https://doi.org/10.1016/j.ecolind.2009.06.008	**-**	**-**	**x**	**x**	**x**	**-**	**-**
90. Moline, A. B., & Poff, N. L. (2008). Growth of an invertebrate shredder on native (*Populus*) and non-native (*Tamarix, Elaeagnus*) leaf litter. *Freshwater Biology, 53*(5), 1012-1020. https://doi.org/10.1111/j.1365-2427.2008.01960.x	**-**	**-**	**x**	**x**	**x**	**-**	**-**

Table S3. Continuation...

Reference	Theme			Database			
	D	M	I	WS	Sc	GS	PL
91. Monroy, S., Martínez, A., López-Rojo, N., Pérez-Calpe, A. V., Basaguren, A., & Pozo, J. (2017). Structural and functional recovery of macroinvertebrate communities and leaf litter decomposition after a marked drought: Does vegetation type matter?. *Science of the Total Environment, 599*, 1241-1250. https://doi.org/10.1016/j.scitotenv.2017.05.093	x	-	x	-	-	x	-
92. Moody, E. K., & Sabo, J. L. (2013). Crayfish impact desert river ecosystem function and litter-dwelling invertebrate communities through association with novel detrital resources. *PLoS One, 8*(5), e63274. https://doi.org/10.1371/journal.pone.0063274	x	-	x	x	x	-	-
93. Moretti, M. S., Becker, B., Kiffer, W. P., da Penha, L. O., & Callisto, M. (2020). *Eucalyptus* leaves are preferred to cerrado native species but do not constitute a better food resource to stream shredders. *Journal of Arid Environments, 181*, 104221. https://doi.org/10.1016/j.jaridenv.2020.104221	-	-	x	-	-	x	-
94. Parkyn, S. M., & Winterbourn, M. J. (1997). Leaf breakdown and colonisation by invertebrates in a headwater stream: comparisons of native and introduced tree species. *New Zealand Journal of Marine and Freshwater Research, 31*(3), 301-312. https://doi.org/10.1080/00288330.1997.9516768	x	x	x	-	-	x	-
95. Peralta-Maraver, I., López-Rodríguez, M. J., Fenoglio, S., Bo, T., Luzón-Ortega, J. M., & de Figueroa, J. M. T. (2011). Macroinvertebrate colonization of two different tree species leaf packs (native *vs.* introduced) in a Mediterranean stream. *Journal of Freshwater Ecology, 26*(4), 495-505. https://doi.org/10.1080/02705060.2011.595554	-	-	x	x	x	-	-
96. Pereira, A., Geraldes, P., Lima-Fernandes, E., Fernandes, I., Cássio, F., & Pascoal, C. (2016). Structural and functional measures of leaf-associated invertebrates and fungi as predictors of stream eutrophication. *Ecological Indicators, 69*, 648-656. https://doi.org/10.1016/j.ecolind.2016.05.017	x	x	x	-	-	x	-
97. Pereira, A., & Ferreira, V. (2021). Invasion of native riparian forests by *Acacia* species affects in-stream litter decomposition and associated microbial decomposers. *Microbial Ecology, 81*(1), 14-25. https://doi.org/10.1007/s00248-020-01552-3	x	x	-	-	-	x	-
98. Pidgeon, R. W. J., & Cairns, S. C. (1981). Decomposition and colonisation by invertebrates of native and exotic leaf material in a small stream in New England (Australia). *Hydrobiologia, 77*(2), 113-127. https://doi.org/10.1007/BF00008869	x	-	x	-	x	-	-
99. Principe, R. E., Márquez, J. A., Martina, L. C., Jobbágy, E. G., & Albariño, R. J. (2015). Pine afforestation changes more strongly community structure than ecosystem functioning in grassland mountain streams. *Ecological Indicators, 57*, 366-375. https://doi.org/10.1016/j.ecolind.2015.04.033	x	-	x	x	-	-	-
100. Quinn, J. M., Cooper, A. B., Davies-Colley, R. J., Rutherford, J. C., & Williamson, R. B. (1997). Land use effects on habitat, water quality, periphyton, and benthic invertebrates in Waikato, New Zealand, hill-country streams. *New Zealand Journal of Marine and Freshwater Research, 31*(5), 579-597. https://doi.org/10.1080/00288330.1997.9516791	-	-	x	-	-	x	-

Table S3. Continuation...

Reference	Theme			Database			
	D	M	I	WS	Sc	GS	PL
101. Quinn, J. M., Burrell, G. P., & Parkyn, S. M. (2000). Influences of leaf toughness and nitrogen content on in-stream processing and nutrient uptake by litter in a Waikato, New Zealand, pasture stream and streamside channels. *New Zealand Journal of Marine and Freshwater Research, 34*(2), 253-271. https://doi.org/10.1080/00288330.2000.9516931	x	x	-	x	x	-	-
102. Raposeiro, P. M., Martins, G. M., Moniz, I., Cunha, A., Costa, A. C., & Gonçalves, V. (2014). Leaf litter decomposition in remote oceanic islands: The role of macroinvertebrates *vs.* microbial decomposition of native *vs.* exotic plant species. *Limnologica, 45*, 80-87. https://doi.org/10.1016/j.limno.2013.10.006	x	-	-	x	-	-	-
103. Ratnarajah, L., & Barmuta, L. A. (2009). The effects of leaf toughness on feeding preference by two Tasmanian shredders. *Hydrobiologia, 636*(1), 173-178. https://doi.org/10.1007/s10750-009-9946-y	-	-	x	-	-	-	x
104. Read, M. G., & Barmuta, L. A. (1999). Comparisons of benthic communities adjacent to riparian native eucalypt and introduced willow vegetation. *Freshwater Biology, 42*(2), 359-374. https://doi.org/10.1046/j.1365-2427.1999.444474.x	-	-	x	x	-	-	-
105. Reinhart, K. O., & VandeVoort, R. (2006). Effect of native and exotic leaf litter on macroinvertebrate communities and decomposition in a western Montana stream. *Diversity and Distributions, 12*(6), 776-781. https://doi.org/10.1111/j.1472-4642.2006.00252.x	x	-	x	x	x	-	-
106. Remor, M. B., Santos, C., Sampaio, S. C., & Sgarbi, L. F. (2013). The effects caused by the replacement of native riparian forest with *Eucalyptus* sp. on the benthic macro-invertebrate community. *Journal of Food, Agriculture & Environment, 11(2)*, 1444 - 1448.	x	-	x	x	x	-	-
107. Rezende, R. S., Petrucio, M. M., & Gonçalves, J. F. (2014). The effects of spatial scale on breakdown of leaves in a tropical watershed. *PLoS One, 9*(5), e97072. https://doi.org/10.1371/journal.pone.0097072	x	x	x	-	-	x	-
108. Rezende, R. S., Biasi, C., Hepp, L. U., Petrucio, M. M., & Gonçalves, J. F. (2019). Effects of leaf litter traits on alpha and beta diversities of invertebrate assemblages in a tropical watershed. *Ecología Austral, 29*(3), 365-379. https://doi.org/10.25260/EA.19.29.3.0.750	x	-	x	-	-	x	-
109. Riedl, H. L., Clements, W. H., & Pejchar, L. (2019). An introduced plant is associated with declines in terrestrial arthropods, but no change in stream invertebrates. *Canadian Journal of Fisheries and Aquatic Sciences, 76*(8), 1314-1325. https://doi.org/10.1139/cjfas-2018-0098	-	-	x	x	x	-	-
110. Roberts, M., Strauch, A. M., Wiegner, T., & Mackenzie, R. A. (2016). Leaf Litter Breakdown of Native and Exotic Tree Species in Two Hawaiian Streams that Differ in Flow. *Pacific Science, 70*(2), 209-222. https://doi.org/10.2984/70.2.7	x	-	-	-	x	-	-

Table S3. Continuation...

Reference	Theme			Database			
	D	M	I	WS	Sc	GS	PL
111. Roon, D. A., Wipfli, M. S., & Wurtz, T. L. (2014). Effects of invasive European bird cherry (*Prunus padus*) on leaf litter processing by aquatic invertebrate shredder communities in urban Alaskan streams. *Hydrobiologia, 736*(1), 17-30. https://doi.org/10.1007/s10750-014-1881-x	x	-	x	x	-	-	-
112. Royer, T. V., Monaghan, M. T., & Minshall, G. W. (1999). Processing of native and exotic leaf litter in two Idaho (USA) streams. *Hydrobiologia, 400*, 123-128. https://doi.org/10.1023/A:1003703130930	x	-	-	x	x	-	-
113. Sampaio, A., Cortes, R., & Leão, C. (2001). Invertebrate and microbial colonisation in native and exotic leaf litter species in a mountain stream. *International Review of Hydrobiology: A Journal Covering all Aspects of Limnology and Marine Biology, 86*(4-5), 527-540. https://doi.org/10.1002/1522-2632(200107)86:4/5<527::AID-IROH527>3.0.CO;2-D	x	x	x	-	-	x	-
114. Samways, M. J., Sharratt, N. J., & Simaika, J. P. (2011). Effect of alien riparian vegetation and its removal on a highly endemic river macroinvertebrate community. *Biological Invasions, 13*(6), 1305-1324. https://doi.org/10.1007/s10530-010-9891-8	-	-	x	x	x	-	-
115. Santos, G. R., Otto, M. S. G., Passos, J. R. D. S., Onofre, F. F., Rodrigues, V. A., de Paula, F. R., & Ferraz, S. F. D. B. (2019). Changes in decomposition rate and litterfall in riparian zones with different basal area of exotic *Eucalyptus* in south-eastern Brazil. *Southern Forests: a Journal of Forest Science, 81*(4), 285-295. https://doi.org/10.2989/20702620.2019.1633503	x	-	-	x	x	-	-
116. Schulze, D. J., & Walker, K. F. (1997). Riparian eucalypts and willows and their significance for aquatic invertebrates in the River Murray, South Australia. *Regulated Rivers: Research & Management: An International Journal Devoted to River Research and Management, 13*(6), 557-577. https://doi.org/10.1002/(SICI)1099-1646(199711/12)13:6<557::AID-RRR485>3.0.CO;2-Q	x	-	x	x	-	-	-
117. Seena, S., Carvalho, F., Cássio, F., & Pascoal, C. (2017). Does the developmental stage and composition of riparian forest stand affect ecosystem functioning in streams?. *Science of the Total Environment, 609*, 1500-1511. https://doi.org/10.1016/j.scitotenv.2017.07.252	x	x	x	-	-	x	-
118. Serra, M. N., Albariño, R., & Villanueva, V. D. (2013). Invasive *Salix fragilis* alters benthic invertebrate communities and litter decomposition in northern Patagonian streams. *Hydrobiologia, 701*(1), 173-188. https://doi.org/10.1007/s10750-012-1270-2	x	-	x	x	x	-	-
119. Swan, C. M., Healey, B., & Richardson, D. C. (2008). The role of native riparian tree species in decomposition of invasive tree of heaven (*Ailanthus altissima*) leaf litter in an urban stream. *Écoscience, 15*(1), 27-35. https://doi.org/10.2980/1195-6860(2008)15[27:TRONRT]2.0.CO;2	x	-	x	x	x	-	-
120. Thompson, R. M., & Townsend, C. R. (2004). Land-use influences on New Zealand stream communities: Effects on species composition, functional organisation, and food-web structure. *New Zealand Journal of Marine and Freshwater Research, 38*(4), 595-608. https://doi.org/10.1080/00288330.2004.9517265	-	-	x	x	x	-	-

Table S3. Continuation...

Reference	Theme			Database			
	D	**M**	**I**	**WS**	**Sc**	**GS**	**PL**
121. Tonello, G., Loureiro, R. C., Krause, P., Silva, C., Ongaratto, R. M., Sepp, S., Restello, R. M., & Hepp, L. U. (2014). Colonização de invertebrados durante a decomposição de diferentes detritos vegetais em um riacho subtropical. *Revista Brasileira de Biociências, 12*(2), 98.	x	-	x	-	-	-	x
122. Tonin, A. M., Hepp, L. U., Restello, R. M., & Gonçalves, J. F. (2014). Understanding of colonization and breakdown of leaves by invertebrates in a tropical stream is enhanced by using biomass as well as count data. *Hydrobiologia, 740*(1), 79-88. https://doi.org/10.1007/s10750-014-1939-9	x	-	x	-	-	x	-
123. Trevisan, A., & Hepp, L. U. (2007). Dinâmica de componentes químicos vegetais e fauna associada ao processo de decomposição de espécies arbóreas em um riacho do norte do Rio Grande do Sul, Brasil. *Neotropical Biology and Conservation, 2*(1), 55-60.	x	-	x	-	-	x	-
124. Urdanigo, J. P., Ponce, M. D., Cajas, C. T., Fonseca, C. S., Benitez, R. Y., Albán, K. A., Chúez, N. G., & Mancera-Rodríguez, N. J. (2019). Diversity of aquatic macroinvertebrates along creeks with different riparian cover in Murocomba Protector Forest, Ecuador. *Revista de Biología Tropical, 67*(4), 861-878. http://dx.doi.org/10.15517/rbt.v67i4.35190	-	-	x	x	-	-	-
125. Valdovinos, C. (2001). Riparian leaf litter processing by benthic macroinvertebrates in a woodland stream of central Chile. *Revista Chilena de Historia Natural, 74*(2), 445-453.	x	-	x	x	-	-	-
126. Vaz, A. A., Vaz, A. A., Pelizari, G. P., Gomes, D. B., & Smith, W. S. (2019). Colonização de folhas por invertebrados aquáticos em um riacho tropical: há diferenças entre espécies nativas e *Eucalyptus grandis* (Hill ex Maiden) nas épocas chuvosa e seca?. *Biotemas, 32*(3), 51-64. http://dx.doi.org/10.5007/2175-7925.2019v32n3p51	-	-	x	-	-	x	-
127. Walpola, H., Leichtfried, M., Amarasinghe, M., & Füreder, L. (2011). Leaf litter decomposition of three riparian tree species and associated macroinvertebrates of Eswathu Oya, a low order tropical stream in Sri Lanka. *International Review of Hydrobiology, 96*(1), 90-104. http://dx.doi.org/10.1002/iroh.201011248	x	-	x	x	x	-	-

Table S4 Total number of papers published in each journal (T = Total) and considering the different themes approach (D = *Decomposition;* M = *Microorganisms*; I = *Invertebrates*). *The papers can approach more than one theme, therefore, total number of papers per journal does not necessarily correspond to the sum of the number of papers of each theme.

Journals	**T***	**Themes**		
		D	**M**	**I**
1. Acta Biologica Colombiana	**1**	1	0	1
2. Acta Limnologica Brasiliensia	**2**	2	0	1
3. Aquatic Ecology	**1**	0	0	1
4. Aquatic Insects	**1**	0	0	1
5. Aquatic Sciences	**2**	2	0	2
6. Archiv Fur Hydrobiologie	**1**	1	1	0
7. Austral Ecology	**1**	1	1	1
8. Biological Invasions	**4**	1	0	4
9. Biotemas	**1**	0	0	1
10. Canadian Journal of Botany-Revue Canadienne de Botanique	**1**	0	1	0
11. Canadian Journal of Fisheries and Aquatic Sciences	**1**	0	0	1
12. Diversity and Distributions	**1**	1	0	1
13. Ecología Austral	**1**	1	0	1
14. Ecological Indicators	**6**	3	1	5
15. Ecology and Evolution	**2**	1	0	2
16. Ecoscience	**2**	1	0	2
17. Ecosphere	**1**	1	0	1
18. Ecosystems	**1**	1	0	0
19. Ecotoxicology	**1**	1	1	1
20. Egerton Journal of Science and Technology	**1**	1	0	1
21. Forest Ecology and Management	**3**	3	2	2
22. Freshwater Biology	**8**	5	4	6
23. Freshwater Science	**6**	4	3	4
24. Fundamental and Applied Limnology	**2**	1	0	2
25. Fungal Ecology	**1**	1	1	0
26. Hidrobiologica	**1**	0	0	1
27. Hydrobiologia	**12**	10	4	10
28. International Journal of Energy, Environment and Economics	**1**	0	0	1
29. International Review of Hydrobiology	**6**	5	2	5
30. Invasive Plant Science and Management	**1**	0	0	1
31. Journal of Applied Ecology	**1**	1	0	1
32. Journal of Arid Environments	**2**	1	0	2
33. Journal of Food Agriculture & Environment	**1**	1	0	1
34. Journal of Freshwater Ecology	**4**	3	1	2
35. Journal of Insect Conservation	**1**	0	0	1
36. Journal of Limnology	**1**	1	0	0
37. Journal of the North American Benthological Society	**1**	0	0	1
38. Limnetica	**2**	2	1	2
39. Limnologica	**5**	3	1	3
40. Marine and Freshwater Research	**4**	1	0	4

Table S4 Continuation...

Journals	T*	Themes		
		D	M	I
41. Microbial Ecology	**3**	2	3	1
42. Natural Areas Journal	**1**	1	0	1
43. Neotropical Biology and Conservation	**1**	1	0	1
44. New Zealand Journal of Marine and Freshwater Research	**7**	3	2	6
45. Pacific Science	**1**	1	0	0
46. PLoS ONE	**3**	2	1	3
47. Regulated Rivers-Research & Management	**1**	1	0	1
48. Revista Brasileira de Biociências	**1**	1	0	1
49. Revista Chilena de Historia Natural	**1**	1	0	1
50. Revista de Biologia Tropical	**1**	0	0	1
51. River Research and Applications	**2**	0	0	2
52. Science of the Total Environment	**4**	3	1	4
53. Southern Forests	**1**	1	0	0
54. The American Midland Naturalist	**1**	0	0	1
55. Urban Ecosystems	**1**	1	0	1
56. Victorian Naturalist	**1**	0	0	1
57. Water, Air, & Soil Pollution	**1**	1	0	1
58. Wetlands	**1**	1	0	1
Total Number of Papers	**127**	81	31	103

Table S5 Total number of papers published in each country (T = Total), considering the different themes approached (D = *Decomposition*; M = *Microorganisms*; I = *Invertebrates*). *The papers can approach more than one theme, therefore, the total number of papers per country does not necessarily correspond to the sum of the number of papers for each theme.

Countries	**T***	**Themes**		
		D	**M**	**I**
1. Argentina	**7**	4	1	6
2. Australia/Tasmania	**13**	4	2	11
3. Brazil	**22**	18	5	15
4. Canada	**3**	3	0	3
5. Chile	**6**	3	0	5
6. Colombia	**1**	1	0	1
7. Costa Rica	**1**	1	0	0
8. Ecuador	**2**	1	0	1
9. French Guiana	**1**	1	0	0
10. Hong Kong	**2**	1	0	1
11. India	**1**	1	0	0
12. Ireland	**2**	2	1	2
13. Israel	**1**	1	0	1
14. Italy	**2**	2	0	2
15. Kenya	**4**	4	0	1
16. Malaysia and Borneo	**2**	1	0	1
17. Mexico	**1**	1	0	0
18. New Zealand	**7**	3	2	6
19. Panama	**1**	1	0	0
20. Portugal/Azores Islands	**21**	18	15	12
21. Puerto Rico	**1**	1	0	0
22. South Africa	**2**	0	0	2
23. Spain	**20**	14	5	16
24. Sri Lanka	**1**	1	0	1
25. Sweden	**1**	1	0	1
26. Trinidad	**1**	1	0	0
27. Uruguay	**1**	1	0	0
28. USA/Hawaii Islands	**21**	12	1	18
29. Venezuela	**1**	1	0	0

Table S6 Non-native trees and/or shrubs species listed in this scientometric. *Scientific names of species are presented as they appear in the papers, and therefore may not accurately reflect current nomenclature. UN = undetermined.

Non-native Tree and/or Shrub Species *	**Growth Form**	**Source**$^{\text{Note}}$
1. *Acacia dealbata* Link	Tree	Lorenzi et al. (2018)
2. *Acacia longifolia* (Andrews) Willd.	Tree	Lorenzi et al. (2018)
3. *Acacia mearnsii* De Wild	Tree	Lorenzi et al. (2018)
4. *Acacia melanoxylon* R. Br.	Tree	Lorenzi et al. (2018)
5. *Acer platanoides* L.	Tree	Trees and Shrubs Online
6. *Acer pseudoplatanus* L.	Tree	Trees and Shrubs Online
7. *Acer pseudoplatanus 'Atropurpureum'* Spaethii	Tree	Trees and Shrubs Online
8. *Acer pseudoplatanus var. corstorphinense* Schwer.	Tree	Trees and Shrubs Online
9. *Aesculus hippocastaneum* L.	Tree	Trees and Shrubs Online
10. *Ailanthus altissima* (Mill.) Swingle	Tree	Trees and Shrubs Online
11. *Alnus glutinosa* (L.) Gaertn.	Tree	Trees and Shrubs Online
12. *Alstonia macrophylla* Wall. ex G. Don	Tree	Lorenzi et al. (2018)
13. *Ardisia elliptica* Thunb.	Shrub	Global Invasive Species Database
14. *Artocarpus altilis* (Parkinson) Fosberg	Tree	Instituto Hórus
15. *Betula pendula* Roth	Tree	Trees and Shrubs Online
16. *Buddleja davidii* Franch.	Shrub	Global Invasive Species Database
17. *Cassia fistula* L.	Tree	Lorenzi et al. (2018)
18. *Ceasalpinia decapetala* (Roth) Alston	Shrub	Others
19. *Cinnamomum camphora* Sieb.	Tree	Lorenzi et al. (2018)
20. *Clethra arborea* Aiton	Tree	Trees and Shrubs Online
21. *Cryptomeria japonica* (Thunb. ex L. f.) D. Don	Tree	Lorenzi et al. (2018)
22. *Elaeagnus angustifolia* L.	Tree/Shrub	Trees and Shrubs Online
23. *Elaeis guineensis* Jacq.	Tree	Global Invasive Species Database
24. *Eucalyptus camaldulensis* Dehn.	Tree	Lorenzi et al. (2018)
25. *Eucalyptus cloeziana* F. Muell.	Tree	Others
26. *Eucalyptus globulus* Labill.	Tree	Lorenzi et al. (2018)
27. *Eucalyptus gomphocephala* DC.	Tree	Others
28. *Eucalyptus grandis* W. Hill ex Maiden	Tree	Lorenzi et al. (2018)
29. *Eucalyptus nitens* (Deane & Maiden)	Tree	Trees and Shrubs Online
30. *Eucalyptus regnans* F. Muell.	Tree	Trees and Shrubs Online
31. *Eucalyptus rostrata* Cav.	Tree	Others
32. *Eucalyptus saligna* Smith	Tree	Lorenzi et al. (2018)
33. *Eucalyptus* sp.	Tree/Shrub	Trees and Shrubs Online
34. *Fagus sylvatica* L.	Tree	Trees and Shrubs Online
35. *Falcataria moluccana* (Miq.) Barneby	Tree	Lorenzi et al. (2018)
36. *Ficus elastica* Roxb. Ex Hornem	Tree	Lorenzi et al. (2018)
37. *Gleditsia triacanthos* L.	Tree	Lorenzi et al. (2018)
38. *Gmelina arborea* Roxb.	Tree	Lorenzi et al. (2018)
39. *Grevillea robusta* A. Cunn. ex. R. Br.	Tree	Lorenzi et al. (2018)
40. *Hevea brasiliensis* (Willd. ex Adr. de Juss.) Muell. Arg.	Tree	Instituto Hórus
41. *Hibiscus* sp.	UN	-
42. *Hovenia dulcis* Thunb.	Tree	Lorenzi et al. (2018)
43. *Juglans nigra* L.	Tree	Trees and Shrubs Online
44. *Lonicera maackii* (Rupr.) Herder	Shrub	Trees and Shrubs Online
45. *Maesa montana* DC.	Tree/Shrub	Trees and Shrubs Online
46. *Melaleuca quinquenervia* (Cav.) S.T. Blake.	Tree	Global Invasive Species Database
47. *Michelia champaca* L.	Tree	Trees and Shrubs Online
48. *Pinus contorta* Douglas ex Loudon	Tree	Trees and Shrubs Online

Table S6 Continuation...

Non-native Tree and/or Shrub Species *	Growth Form	Source[Note]
49. *Pinus elliottii* Engelm.	Tree	Lorenzi et al. (2018)
50. *Pinus lambertiana* Dougl.	Tree	Trees and Shrubs Online
51. *Pinus patula* Schiede ex Schltdl. & Cham.	Tree	Lorenzi et al. (2018)
52. *Pinus ponderosa* Dougl. ex Laws.	Tree	Lorenzi et al. (2018)
53. *Pinus radiata* D. Don	Tree	Lorenzi et al. (2018)
54. *Pinus* sp.	Tree	Trees and Shrubs Online
55. *Pittosporum undulatum* Vent.	Tree	Lorenzi et al. (2018)
56. *Platanus × acerifolia* Willd.	Tree	Trees and Shrubs Online
57. *Platanus hispanica* Mill. ex Münchh	Tree	Trees and Shrubs Online
58. *Platanus hybrida* Brot.	Tree	Lorenzi et al. (2018)
59. *Platanus orientalis* L.	Tree	Lorenzi et al. (2018)
60. *Populus × canadensis* Moench	Tree	Lorenzi et al. (2018)
61. *Prunus laurocerasus* L.	Shrub	Trees and Shrubs Online
62. *Prunus padus* L.	Tree	Trees and Shrubs Online
63. *Pseudotsuga menziesii* (Mirb.) Franco	Tree	Trees and Shrubs Online
64. *Psidium cattleianum* Sabine	Tree/Shrub	Global Invasive Species Database
65. *Pterocarpus indicus* Willd.	Tree	Others
66. *Quercus canariensis* Willd.	Tree	Trees and Shrubs Online
67. *Quercus coccinea* Münchh.	Tree	Lorenzi et al. (2018)
68. *Rhamnus cathartica* L.	Shrub	Trees and Shrubs Online
69. *Rhododendron irroratum* Franch.	Tree/Shrub	Trees and Shrubs Online
70. *Rhododendron ponticum* L.	Tree/Shrub	Trees and Shrubs Online
71. *Robinia neomexicana* Gray	Tree/Shrub	Trees and Shrubs Online
72. *Robinia pseudoacacia* L.	Tree	Lorenzi et al. (2018)
73. *Rubus armeniacus* Focke	Shrub	Others
74. *Rubus fruticosus* L.	Shrub	Instituto Hórus
75. *Salix babylonica* L.	Tree	Lorenzi et al. (2018)
76. *Salix fragilis* L.	Tree	Trees and Shrubs Online
77. *Salix* sp.	Tree/Shrub	Trees and Shrubs Online
78. *Schefflera actinophylla* (Endl.) Harms	Tree	Lorenzi et al. (2018)
79. *Schizolobium parahybum* (Vell.) SF Blake	Tree	Instituto Hórus
80. *Solanum mauritianum* Scop.	Tree/Shrub	Global Invasive Species Database
81. *Spathodea campanulata* P.(Beauv.)	Tree	Lorenzi et al. (2018)
82. *Tabebuia rosea* (Bertol.) DC.	Tree	Lorenzi et al. (2018)
83. *Tamarix ramosissima* Ledeb.	Tree/Shrub	Trees and Shrubs Online
84. *Tamarix* sp.	Tree/Shrub	Trees and Shrubs Online
85. *Tectona grandis* L.f.	Tree	Lorenzi et al. (2018)
86. *Terminalia catappa* L.	Tree	Lorenzi et al. (2018)
87. *Tilia cordata x europaea* L.	Tree	Trees and Shrubs Online
88. *Ulmus procera* Salisb.	Tree	Trees and Shrubs Online
89. *Viburnum macrocephalum* Fort.	Shrub	Trees and Shrubs Online

Note: References consulted to determine the most common form of growth for each species:
- ***Trees and Shrubs Online*** – Database online (https://treesandshrubsonline.org/)
- ***Global Invasive Species Database*** - Database online (http://www.iucngisd.org/gisd/)
- ***Instituto Hórus*** – Database online (http://bd.institutohorus.org.br/)
- ***Lorenzi et al. (2018)*** – Lorenzi, H.; Bacher, L. B.; Torres, M. A. V. (2018). *Árvores e Arvoretas Exóticas no Brasil: madeireiras, ornamentais e aromáticas*. Nova Odessa, São Paulo: Instituto Plantarum de Estudos da Flora, p. 464.
- ***Others*** – Specific papers for each non-native trees and/or shrubs species .

Table S7 Number of papers where each non-native tree and/or shrub species were recorded (T = Total), considering the different themes approached (D = *Decomposition*; M = *Microorganisms*; I = *Invertebrates*). *The papers can approach more than one theme, therefore, the total number of papers per species does not necessarily correspond to the sum of the number of papers for each theme. ** Scientific names of species are presented as they appear in the papers, and therefore may not accurately reflect current nomenclature. *** See Table S3 for the corresponding reference numbers.

Non-native Tree and/or Shrub Species**	**T***	**Theme**			**Reference*****
		D	**M**	**I**	
1. *Acacia dealbata* Link	**1**	1	1	0	97
2. *Acacia longifolia* (Andrews) Willd.	**1**	0	0	1	114
3. *Acacia mearnsii* De Wild	**2**	0	0	2	72,114
4. *Acacia melanoxylon* R. Br.	**4**	3	2	2	34, 73, 97, 102
5. *Acer platanoides* L.	**1**	1	0	1	105
6. *Acer pseudoplatanus* L.	**1**	1	1	1	53
7. *Acer pseudoplatanus 'Atropurpureum'* Spaethii	**1**	1	1	1	53
8. *Acer pseudoplatanus var. corstorphinense* Schwer.	**1**	1	1	1	53
9. *Aesculus hippocastaneum* L.	**1**	1	1	1	53
10. *Ailanthus altissima* (Mill.) Swingle	**3**	2	1	3	02, 86, 119
11. *Alnus glutinosa* (L.) Gaertn.	**8**	4	1	7	5, 6, 12, 13, 52, 81, 82, 103
12. *Alstonia macrophylla* Wall. ex G. Don	**1**	1	0	1	127
13. *Ardisia elliptica* Thunb.	**1**	1	0	1	12
14. *Artocarpus altilis* (Parkinson) Fosberg	**1**	1	0	1	9
15. *Betula pendula* Roth	**1**	1	0	1	52
16. *Buddleja davidii* Franch.	**1**	1	1	1	53
17. *Cassia fistula* L.	**1**	1	0	1	12
18. *Ceasalpinia decapetala* (Roth) Alston	**1**	0	0	1	72
19. *Cinnamomum camphora* Sieb.	**1**	0	0	1	29
20. *Clethra arborea* Aiton	**4**	3	2	3	5, 6, 34, 102
21. *Cryptomeria japonica* (Thunb. ex L. f.) D. Don	**1**	1	2	1	6, 35
22. *Elaeagnus angustifolia* L.	**3**	2	0	2	60, 90, 112
23. *Elaeis guineensis* Jacq.	**1**	0	0	1	24
24. *Eucalyptus camaldulensis* Dehn.	**5**	3	3	4	44, 46, 47, 93, 114
25. *Eucalyptus cloeziana* F. Muell.	**2**	2	1	2	107, 108
26. *Eucalyptus globulus* Labill.	**28**	20	12	20	1, 7, 15, 16, 17, 18, 19, 22, 23, 25, 26, 31, 32, 40, 42, 49, 58, 61, 62, 63, 64, 70, 78, 91, 96, 102, 113, 117
27. *Eucalyptus gomphocephala* DC.	**1**	0	0	1	72
28. *Eucalyptus grandis* W. Hill ex Maiden	**7**	6	0	5	28, 45, 50, 121, 122, 123, 126
29. *Eucalyptus nitens* (Deane & Maiden)	**1**	1	0	1	42
30. *Eucalyptus regnans* F. Muell.	**1**	1	1	0	101
31. *Eucalyptus rostrata* Cav.	**1**	1	0	1	51
32. *Eucalyptus saligna* Smith	**2**	2	0	1	3, 115
33. *Eucalyptus* sp.	**6**	4	0	5	9, 33, 36, 37, 38, 106
34. *Fagus sylvatica* L.	**1**	1	1	1	53
35. *Falcataria moluccana* (Miq.) Barneby	**2**	2	1	1	71, 110
36. *Ficus elastica* Roxb. Ex Hornem	**1**	1	0	1	9
37. *Gleditsia triacanthos* L.	**1**	1	1	0	41
38. *Gmelina arborea* Roxb.	**1**	0	0	1	124
39. *Grevillea robusta* A. Cunn. ex. R. Br.	**1**	1	0	1	3
40. *Hevea brasiliensis* (Willd. ex Adr. de Juss.) Muell. Arg.	**1**	1	0	1	127

Table S7 Continuation...

Non-native Tree and/or Shrub Species**	T*	Theme			Reference***
		D	M	I	
41. *Hibiscus* sp.	**1**	1	0	1	9
42. *Hovenia dulcis* Thunb.	**2**	2	1	1	8, 59
43. *Juglans nigra* L.	**1**	1	1	0	101
44. *Lonicera maackii* (Rupr.) Herder	**7**	2	0	7	11, 27, 30, 69, 83, 84, 85
45. *Maesa montana* DC.	**1**	1	0	1	12
46. *Melaleuca quinquenervia* (Cav.) S.T. Blake.	**1**	0	0	1	67
47. *Michelia champaca* L.	**1**	1	0	1	12
48. *Pinus contorta* Douglas ex Loudon	**1**	1	0	1	68
49. *Pinus elliottii* Engelm.	**4**	3	0	3	28, 74, 75, 99
50. *Pinus lambertiana* Dougl.	**1**	0	0	1	89
51. *Pinus patula* Schiede ex Schltdl. & Cham.	**1**	0	0	1	72
52. *Pinus ponderosa* Dougl. ex Laws.	**1**	0	0	1	89
53. *Pinus radiata* D. Don	**12**	6	2	11	21, 73, 76, 77, 81, 82, 89, 91, 100, 101, 120, 125
54. *Pinus* sp.	**2**	0	0	2	38, 48
55. *Pittosporum undulatum* Vent.	**2**	2	1	1	34, 102
56. *Platanus × acerifolia* Willd.	**1**	1	0	1	59
57. *Platanus hispanica* Mill. ex Münchh	**2**	2	0	2	21, 76
58. *Platanus hybrida* Brot.	**1**	1	1	1	87
59. *Platanus orientalis* L.	**1**	1	0	0	20
60. *Populus × canadensis* Moench	**4**	3	1	3	21, 76, 95, 101
61. *Prunus laurocerasus* L.	**2**	2	1	2	14, 53
62. *Prunus padus* L.	**1**	1	0	1	111
63. *Pseudotsuga menziesii* (Mirb.) Franco	**1**	0	0	1	89
64. *Psidium cattleianum* Sabine	**1**	1	0	0	110
65. *Pterocarpus indicus* Willd.	**1**	1	0	1	12
66. *Quercus canariensis* Willd.	**1**	1	0	1	12
67. *Quercus coccinea* Münchh.	**1**	1	1	1	94
68. *Rhamnus cathartica L.*	**2**	1	0	1	39, 66
69. *Rhododendron irroratum* Franch.	**1**	1	0	1	12
70. *Rhododendron ponticum* L.	**2**	2	1	2	53, 54
71. *Robinia neomexicana* Gray	**1**	0	0	1	109
72. *Robinia pseudoacacia* L.	**4**	3	1	4	2, 10, 14, 86
73. *Rubus armeniacus* Focke	**2**	2	0	2	57, 60
74. *Rubus fruticosus* L.	**1**	0	0	1	73
75. *Salix babylonica* L.	**3**	3	1	3	94, 98, 116
76. *Salix fragilis* L.	**3**	1	0	3	65, 88, 118
77. *Salix* sp.	**5**	0	2	4	55, 56, 79, 80, 104
78. *Schefflera actinophylla* (Endl.) Harms	**1**	1	0	1	9
79. *Schizolobium parahybum* (Vell.) SF Blake	**1**	0	0	1	124
80. *Solanum mauritianum* Scop.	**2**	1	0	1	72, 102
81. *Spathodea campanulata* P.(Beauv.)	**1**	1	0	0	110
82. *Tabebuia rosea* (Bertol.) DC.	**1**	1	0	1	12
83. *Tamarix ramosissima* Ledeb.	**3**	2	0	3	4, 43, 92
84. *Tamarix* sp.	**1**	0	0	1	90
85. *Tectona grandis* L.f.	**1**	1	0	1	9
86. *Terminalia catappa* L.	**1**	1	0	1	9
87. *Tilia cordata x europaea* L.	**1**	1	1	1	53
88. *Ulmus procera* Salisb.	**1**	1	1	1	94
89. *Viburnum macrocephalum* Fort.	**1**	1	0	1	12

Table S8 Number of occurrences of each non-native tree and/or shrub species in each country/region. Some geographically related and/or neighboring countries/regions were considered together. Ar = Argentina; Au/Ta = Australia/Tasmania; Br = Brazil; Ca = Canada; Cl = Chile; Co = Colombia; CR = Costa Rica; Ec = Ecuador; FG = French Guiana; HK = Hong Kong; In = India; Ir = Ireland; Is = Israel; It = Italy; Ke = Kenya; Ma/Bo = Malaysia/Borneu; Me = Mexico; NZ = New Zealand; Pa = Panama; Po/Az = Portugal/Azores Islands; PR = Porto Rico; SA = South Africa; Sp = Spain; SL = Sri Lanka; Sw = Sweden; Tr = Trinidad; Ur = Uruguay; US/Ha = United States of America/Hawaii Islands; Ve = Venezuela. * Scientific names of species are presented as they appear in the papers, and therefore may not accurately reflect current nomenclature. ** See Table S3 for the corresponding reference numbers.

Species*	**Ag**	**Au / Ta**	**Br**	**Ca**	**Cl**	**Co**	**CR**	**Ec**	**FG**	**HK**	**In**	**Ir**	**Is**	**It**	**Ke**	**Ma / Bo**	**Me**	**NZ**	**Pa**	**Po / Az**	**PR**	**SA**	**Sp**	**SL**	**Sw**	**Tr**	**Ur**	**US / Ha**	**Ve**	**References****
1. *Acacia dealbata*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	97
2. *Acacia longifolia*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	114
3. *Acacia mearnsii*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**2**	-	-	-	-	-	-	-	72, 114
4. *Acacia melanoxylon*	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**3**	-	-	-	-	-	-	-	-	-	34, 73, 97, 102
5. *Acer platanoides*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	105
6. *Acer pseudoplatanus*	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	53
7. *Acer pseudoplatanus 'Atropurpureum'*	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	53
8. *Acer pseudoplatanus var. corstorphinense*	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	53
9. *Aesculus hippocastaneum*	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	53
10. *Ailanthus altissima*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**2**	-	-	-	-	**1**	-	02, 86, 119
11. *Alnus glutinosa*	-	**5**	**1**	-	-	-	**1**	**1**	**1**	**1**	**1**	-	-	-	**1**	**1**	**1**	**1**	**1**	**2**	**1**	-	-	-	-	**1**	-	-	**1**	5, 6, 12, 13, 52, 81, 82, 103
12. *Alstonia macrophylla*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	127
13. *Ardisia elliptica*		**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	12
14. *Artocarpus altilis*	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	9

Table S8 Continuation…

Species*	Ag	Au / Ta	Br	Ca	Cl	Co	CR	Ec	FG	HK	In	Ir	Is	It	Ke	Ma / Bo	Me	NZ	Pa	Po / Az	PR	SA	Sp	SL	Sw	Tr	Ur	US / Ha	Ve	References**
15. *Betula pendula*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	52
16. *Buddleja davidii*	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	53
17. *Cassia fistula*	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	12
18. *Ceasalpinia decapetala*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	72
19. *Cinnamomum camphora*	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	29
20. *Clethra arborea*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**4**	-	-	-	-	-	-	-	-	-	5, 6, 34, 102
21. *Cryptomeria japonica*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**2**	-	-	-	-	-	-	-	-	-	6, 35
22. *Elaeagnus angustifolia*	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**2**	-	60, 90, 112
23. *Elaeis guineensis*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	24
24. *Eucalyptus camaldulensis*	-	-	**4**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	44, 46, 47, 93, 114
25. *Eucalyptus cloeziana*	-	-	**2**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	107, 108
26. *Eucalyptus globulus*	-	-	**3**	**1**	**1**	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	**14**	-	-	**9**	-	-	-	-	**1**	-	1, 7, 15, 16, 17, 18, 19, 22, 23, 25, 26, 31, 32, 40, 42, 49, 58, 61, 62, 63, 64, 70, 78, 91, 96, 102, 113, 117
27. *Eucalyptus gomphocephala*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	72
28. *Eucalyptus grandis*	-	-	**7**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	28, 45, 50, 121, 122, 123, 126
29. *Eucalyptus nitens*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	42
30. *Eucalyptus regnans*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	101

Table S8 Continuation…

Species*	Ag	Au / Ta	Br	Ca	Cl	Co	CR	Ec	FG	HK	In	Ir	Is	It	Ke	Ma / Bo	Me	NZ	Pa	Po / Az	PR	SA	Sp	SL	Sw	Tr	Ur	US / Ha	Ve	References**
31. *Eucalyptus rostrata*	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	51
32. *Eucalyptus saligna*	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	3, 115
33. *Eucalyptus* sp.	-	-	**3**	-	**3**	**1**	-	-	-	-	-	-	-	-	**1**	-	-	-	-	**2**	-	-	**2**	-	-	-	**1**	-	-	9, 33, 36, 37, 38, 106
34. *Fagus sylvatica*	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	53
35. *Falcataria moluccana*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**2**	-	71, 110
36. *Ficus elastica*	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	9
37. *Gleditsia triacanthos*	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	41
38. *Gmelina arborea*	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	124
39. *Grevillea robusta*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	3
40. *Hevea brasiliensis*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	127
41. *Hibiscus* sp.	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	9
42. *Hovenia dulcis*	-	-	**2**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	8, 59
43. *Juglans nigra*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	101
44. *Lonicera maackii*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**7**	-	11, 27, 30, 69, 83, 84, 85
45. *Maesa montana*	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	12
46. *Melaleuca quinquenervia*	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	67
47. *Michelia champaca*	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	12
48. *Pinus contorta*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	68

Table S8 Continuation…

Species*	Ag	Au / Ta	Br	Ca	Cl	Co	CR	Ec	FG	HK	In	Ir	Is	It	Ke	Ma / Bo	Me	NZ	Pa	Po / Az	PR	SA	Sp	SL	Sw	Tr	Ur	US / Ha	Ve	References**
49. *Pinus elliottii*	**3**	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	28, 74, 75, 99
50. *Pinus lambertiana*	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	89
51. *Pinus patula*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	72
52. *Pinus ponderosa*	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	89
53. *Pinus radiata*	**1**	**2**	-	-	**2**	-	-	-	-	-	-	-	-	-	-	-	-	**3**	-	-	-	-	**4**	-	-	-	-	-	-	21, 73, 76, 77, 81, 82, 89, 91, 100, 101, 120, 125
54. *Pinus* sp.	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	38, 48
55. *Pittosporum undulatum*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**2**	-	-	-	-	-	-	-	-	-	34, 102
56. *Platanus × acerifolia*	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	59
57. *Platanus hispanica*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**2**	-	-	-	-	-	-	21, 76
58. *Platanus hybrida*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	87
59. *Platanus orientalis*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	20
60. *Populus × canadenses*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	**3**	-	-	-	-	-	-	21, 76, 95, 101
61. *Prunus laurocerasus*	-	-	-	-	-	-	-	-	-	-	-	**1**	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	14, 53
62. *Prunus padus*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	111
63. *Pseudotsuga menziesii*	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	89
64. *Psidium cattleianum*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	110
65. *Pterocarpus indicus*	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-		-	12
66. *Quercus canariensis*	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	12

Table S8 Continuation…

Species*	Ag	Au / Ta	Br	Ca	Cl	Co	CR	Ec	FG	HK	In	Ir	Is	It	Ke	Ma / Bo	Me	NZ	Pa	Po / Az	PR	SA	Sp	SL	Sw	Tr	Ur	US / Ha	Ve	References**
67. *Quercus coccinea*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	94
68. *Rhamnus cathartica*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**2**	-	39, 66
69. *Rhododendron irroratum*	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	12
70. *Rhododendron ponticum*	-	-	-	-	-	-	-	-	-	-	-	**2**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-		-	53, 54
71. *Robinia neomexicana*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	109
72. *Robinia pseudoacacia*	-	-	-	-	-	-	-	-	-	-	-	-	-	**2**	-	-	-	-	-	-	-	-	**2**	-	-	-	-	-	-	2, 10, 14, 86
73. *Rubus armeniacus*	-	-	-	**2**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	57, 60
74. *Rubus fruticosus*	-	-	-		**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	73
75. *Salix babylonica*		**2**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	94, 98, 116
76. *Salix fragilis*	**1**		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	**1**	-	-	-	-	-	-	-	-		65, 88, 118
77. *Salix* sp.	-	**5**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	55, 56, 79, 80, 104
78. *Schefflera actinophylla*	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-		-	-	-	-	-	-	-	-	-	9
79. *Schizolobium parahybum*	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-		-	-	-	-	-	-	-	-	-	124
80. *Solanum mauritianum*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	**1**	-	-	-	-	-	-	-	72, 102
81. *Spathodea campanulata*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	110
82. *Tabebuia rosea*	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-		-	12
83. *Tamarix ramosissima*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**3**	-	4, 43, 92
84. *Tamarix* sp.	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	90
85. *Tectona grandis*	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	9
86. *Terminalia catappa*	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	9

Table S8 Continuation…

Species*	Ag	Au / Ta	Br	Ca	Cl	Co	CR	Ec	FG	HK	In	Ir	Is	It	Ke	Ma / Bo	Me	NZ	Pa	Po / Az	PR	SA	Sp	SL	Sw	Tr	Ur	US / Ha	Ve	References**
87. *Tilia cordata x europaea*	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	53
88. *Ulmus procera*	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**1**	-	-	-	-	-	-	-	-	-	-	-	94
89. *Viburnum macrocephalum*	-	**1**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	12

7. CONCLUSÃO GERAL

Nesta Tese, eu *avaliei a influência da presença de árvores e/ou arbustos não nativos e não nativos invasores na vegetação ripária em processos que compõem o funcionamento de riachos.* Ao desenvolver e concluir este trabalho acredito que, de fato, novas descobertas foram realizadas e que elas podem contribuir com o conhecimento ecológico existente sobre o funcionamento de riachos e sobre os efeitos de espécies não nativas, principalmente espécies não nativas invasoras, sobre o funcionamento destes ecossistemas. Parte dos achados desta Tese estiveram concentrados em experimentos realizados *in situ*, onde avaliei o efeito de uma espécie arbórea não nativa considerada invasora no sul do Brasil – *Hovenia dulcis* – sobre o funcionamento de riachos subtropicais de Mata Atlântica. Utilizando *H. dulcis* como espécie - modelo, acredito que estas descobertas podem contribuir para um melhor entendimento sobre a influência de espécies arbóreas não nativas invasoras sobre as fontes de energia (e.g., aporte de serrapilheira alóctone), ciclagem de nutrientes (e.g., decomposição foliar) e relações tróficas (e.g., comunidade de invertebrados aquáticos) em riachos subtropicais de Mata Atlântica.

O primeiro objetivo específico desta Tese foi *avaliar o efeito das características fenológicas de H. dulcis no padrão temporal e na quantidade de entrada de serapilheira alóctone por meio da contribuição relativa de folhas, galhos e partes reprodutivas de espécies nativas e de H. dulcis em riachos com e sem a presença de H. dulcis na vegetação ripária.* No Capítulo I, eu e os coautores deste estudo, descobrimos que, devido as suas características fenológicas, a presença de *H. dulcis* na vegetação ripária pode alterar o padrão temporal da entrada de serapilheira alóctone em riachos subtropicais. Isso ocorre pois *H. dulcis* possui algumas fenofases - especialmente a queda das folhas - que ocorreram mais cedo do que as da maioria das espécies arbóreas nativas. Com isso, no riacho onde *H. dulcis* estava presente na vegetação ripária, observamos elevada entrada de serapilheira alóctone durante o outono e não durante o inverno e a primavera como normalmente ocorre em formações florestais de Mata Atlântica compostas exclusivamente por espécies arbóreas nativas na região sul do Brasil. Além disso, os elevados aportes de serrapilheira alóctone oriundos de *H. dulcis* registrados no outono foram gerados, curiosamente, não apenas por grandes quantidades de folhas senescentes (como esperado), mas também por elevadas quantidades de galhos e material reprodutivo (especialmente pseudofrutos). É importante reiterar, no entanto, que esses resultados foram observados em um riacho onde a densidade estimada de indivíduos de *H. dulcis* na vegetação ripária era alta (neste estudo, ~ 650 ind.ha^{-1}) e, portanto, não é possível saber se as mudanças observadas neste estudo para a entrada de serrapilheira alóctone também ocorrem em riachos onde a densidade dessa espécie na vegetação ripária é menor. Numa perspectiva futura, novos estudos podem avaliar qual é a densidade de indivíduos de *H. dulcis* presentes na vegetação ripária necessária para que alterações nas características da entrada de serrapilheira alóctone em riachos sejam significativas. Outro aspecto que também pode ser explorado por estudos futuros é se, além da densidade de indivíduos, a idade e tamanho (e.g., área basal) dos indivíduos adultos de *H. dulcis* presentes na vegetação ripária afetam a produção de serrapilheira alóctone nos riachos.

O segundo objetivo específico desta Tese foi *avaliar o efeito das folhas de H. dulcis, disponíveis individualmente e/ou em conjunto com folhas de espécies nativas, na decomposição foliar e na estrutura e composição da comunidade de invertebrados aquáticos associados em riachos.* No Capítulo II, eu e os coautores deste estudo, descobrimos que as folhas de *H. dulcis* apresentam altas taxas de decomposição foliar o que, entretanto, não afetou (i.e., não acelerou) a decomposição foliar de outras espécies nativas associadas a elas. Além disso, a presença de folhas de *H. dulcis* não mostrou efeitos negativos evidentes na estrutura e nem alterou claramente a composição da comunidade de invertebrados associados. Apesar disso, é importante ressaltar que nossos resultados devem ser vistos com cautela. Num contexto ecológico mais amplo, as folhas de *H. dulcis* podem ser consideradas um recurso alóctone efêmero - em primeiro lugar, por sua rápida decomposição foliar e, em segundo lugar, por estarem disponíveis em grande quantidade nos riachos apenas durante três ou quatro meses do ano (transição verão-outono). Dessa forma, isso reforça a importância da presença de uma vegetação ripária diversificada para garantir, ao longo do ano, um abastecimento diversificado de serrapilheira alóctone aos riachos. Numa perspectiva futura, novos estudos podem avaliar se o processo de decomposição foliar e as comunidades de invertebrados aquáticos diferem de forma sazonal ao longo do ano (e.g., entre as estações) em riachos com densidades distintas de *H. dulcis* na vegetação ripária em virtude dos efeitos da variação de disponibilidade de serrapilheira alóctone oriunda de *H. dulcis* nos riachos durante as estações.

Finalmente, o terceiro e último objetivo desta Tese foi *avaliar as principais tendências de estudos abordando os efeitos de espécies arbóreas e/ou arbustivas não nativas e não nativas invasoras sobre a decomposição foliar e sobre as comunidades de microrganismos e invertebrados aquáticos em riachos.* No Capítulo III, eu e os coautores deste estudo, realizamos uma revisão sistematizada e cienciometria para quantificar o número de estudos sobre a temática, elencando os periódicos, os países, as espécies arbóreas e/ou arbustivas não nativas, as escalas espaciais de comparação e as métricas biológicas mais estudados. Foram considerados 127 artigos publicados entre 1981 e 2020 em 57 periódicos. O tema *decomposição* foi abordado em ~64% dos artigos, *microrganismos* em ~24% e *invertebrados* em ~81%. Entre 29 países registrados, Brasil, Estados Unidos, Portugal, Espanha e Austrália foram os mais estudados. Entre 89 espécies arbóreas/arbustivas não nativas registradas, *Eucalyptus globulus*, *Pinus radiata*, *Alnus glutinosa*, *Eucalyptus grandis*, *Lonicera maackii* e *Eucalyptus camaldulensis* foram as mais estudadas. Considerando as escalas espaciais de comparação, a maior parte dos estudos comparou folhas nativas *vs* folhas não nativas. Finalmente, para *decomposição*, 'decomposição foliar mediada por microrganismos e invertebrados' foi a métrica mais representativa; para *microrganismos*, medidas de 'composição', 'riqueza', 'atividade reprodutiva', 'biomassa' e 'abundância' foram as métricas biológicas mais registradas enquanto, para *invertebrados*, medidas de 'abundância e/ou densidade total', 'riqueza total', 'abundância e/ou densidade de grupos tróficos funcionais', 'riqueza de grupos tróficos funcionais' e 'composição' foram as métricas biológicas mais mencionadas nos artigos. De forma complementar, também avaliamos, especificamente, o número de estudos que testaram os efeitos de *H. dulcis* (modelo biológico de espécie arbórea não nativa utilizado nos Capítulos I e II desta Tese) sobre a decomposição foliar e sobre as comunidades de microrganismos e invertebrados aquáticos em riachos tanto em escala global quando para o Brasil. Ao todo, *H. dulcis* foi registrada em apenas dois estudos (1,5% do total), ambos realizados na região sul do Brasil. Além disso, no Brasil, *H. dulcis* foi a segunda espécie arbórea e/ou arbustiva não nativa mais estudada (atrás apenas de espécies de *Eucalyptus*), sendo registrada em cerca de ~ 9% do total dos estudos realizados neste país. Acredito que os achados deste capítulo, possam contribuir com novas pesquisas ao oferecer uma visão do conhecimento construído ao longo do tempo (tendências) e, ao mesmo tempo, demonstrar as lacunas e/ou os aspectos ecológicos menos estudados e que podem ser mais bem explorados em estudos futuros. Neste contexto, estudos futuros podem avaliar, no caso especialmente das comunidades de microrganismos e invertebrados aquáticos, quais são as métricas biológicas mais sensíveis (i.e., mais suscetíveis a alterações) a presença de espécies arbóreas e/ou arbustivas na vegetação ripária de riachos. Outro aspecto que pode ser investigado é quais são as espécies arbóreas e/ou arbustivas não nativas presentes na vegetação ripária mais associadas a alterações significativas nas métricas biológicas destas comunidades.

Por fim, acredito que esta Tese, especificamente os Capítulos I e II, possa contribuir com o conhecimento existente sobre os efeitos da presença de *H. dulcis* em ecossistemas da Mata Atlântica do sul do Brasil e, dessa forma, auxiliar na elaboração de estratégias de manejo e gestão da espécie. De fato, nossos achados demonstram que a presença de *H. dulcis* na vegetação ripária de riachos subtropicais de Mata Atlântica pode modificar alguns processos ecossistêmicos de pequenos riachos (e.g. decomposição foliar e algumas métricas estruturais das comunidades de invertebrados aquáticos). Apesar disso, efeitos ou mudanças ecológicas negativas ao funcionamento de riachos não foram claramente evidentes. Dessa forma, num contexto de manejo e gestão, nossos achados não devem ser vistos de forma isolada, mas sim como parte de um conjunto maior de informações oriunda de pesquisas que avaliaram os efeitos da presença de *H. dulcis* em ecossistemas aquáticos (e.g., riachos) e terrestres (e.g., remanescentes florestais) e sua relação com as respectivas comunidades biológicas nativas destes ecossistemas.

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