UNIVERSIDADE FEDERAL DO RIO GRANDE DO SUL

# CIDADES COMO LABORATÓRIOS DE EVOLUÇÃO: UMA BROMÉLIA COMO MODELO DE ESTUDO DA ADAPTAÇÃO DE PLANTAS AO AMBIENTE URBANO

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## Sumário

# Resumo

A urbanização traz uma mistura complexa de mudanças no ambiente, aumentando a pressão sobre a diversidade das espécies locais. No entanto, pouco se sabe ainda como os impactos ecológicos decorrentes da urbanização afetam a evolução das populações de plantas que vivem nas cidades. A proposta desta tese foi abordar diferentes aspectos da urbanização como causadores de mudanças em espécies e comunidades de plantas, assim como contribuir com estudos genéticos e evolutivos de uma espécie nativa de bromélia (*Tillandsia aeranthos* (Loisel.) L.B.Sm.) que ocorre em ambientes urbanos no sul do Brasil. O Capítulo III apresenta uma revisão sistemática global que integra várias áreas da Ecologia Urbana para sumarizar como as espécies e comunidades de plantas estão respondendo a urbanização e quais os principais agentes dessas respostas (*Urban Drivers*). Após avaliar 171 estudos publicados nos últimos onze anos, foi revelado que a maioria deles relatou consequências negativas da urbanização. Os *Urban Drivers* mais citados como responsáveis pelas mudanças nas espécies e comunidades vegetais foram "mudança da cobertura do solo" e "invasão biótica". Dentre as consequências negativas mais relatadas, temos redução de riqueza de espécies vegetais, menor disponibilidade de polinizadores, homogeneização do ambiente, invasão de espécies não nativas, alterações nos tempos dos eventos fenológicos e dificuldades para o crescimento das plantas. No entanto, foi avaliado que algumas espécies apresentam evidências de adaptação a essas adversidades por meio de sua plasticidade fenotípica e também pela evolução de características que tornam os indivíduos resilientes ao ambiente urbano. O Capítulo IV, trata dos efeitos da urbanização na diversidade genética, estrutura genética e adaptação local em *T. aeranthos*. Seu metabolismo CAM é vantajoso para colonizar com sucesso as áreas urbanas, geralmente mais secas, tornando esta espécie um excelente modelo para o estudo de como as plantas podem se adaptar a estes ambientes. Para testar se haveria diferenças nos padrões genéticos entre populações que ocorrem em ambientes urbanos e não-urbanos, utilizamos uma amostragem na forma de transectos, distribuídos ao longo de um gradiente de urbanização. A fecundação cruzada obrigatória permitiu que esta espécie mantivesse altos níveis de diversidade genética, sem diminuição na diversidade genética com o aumento da urbanização. Em concordância com este resultado, a fragmentação causada pela urbanização não foi suficiente para separar as populações urbanas e não-urbanas através das análises de estrutura genética. A falta de

correlação entre os *outlier loci* detectados através do escaneamento genômico e o índice de urbanização encontrado em nosso estudo corrobora com os demais índices de diversidade genética, não sendo afetados pela urbanização. Portanto, pode-se supor que a dispersão pode ocorrer por pólen ou sementes entre populações urbanas e não-urbanas, sendo estes os principais determinantes da manutenção da diversidade genética nas populações de *T. aeranthos*. Por outro lado, o Capítulo V apresenta um estudo sobre a produção e a viabilidade de sementes de *T. aeranthos*, como forma de comparar um dos aspectos de seu sucesso reprodutivo em ambientes urbanos e não-urbanos. A produção de sementes não apresentou diferença estatisticamente significativa em relação ao gradiente de urbanização, porém a taxa de germinação diminuiu com o aumento da urbanização. Como o desenvolvimento da semente depende estritamente da fertilização cruzada bem-sucedida, é provável que tenha ocorrido uma falha no desenvolvimento da semente posterior à fertilização. Os resultados obtidos na presente tese contribuirão com os estudos avaliando os efeitos da urbanização em espécies vegetais, especialmente em ambientes tropicais, que ainda são menos representados em escala global.

# Abstract

Urbanization brings complex changes to the environment, increasing pressures on local species diversity. However, little is known about how the ecological impacts resulted from urbanization affect the evolution of plant populations living in cities. The purpose of this thesis was to address different aspects of urbanization causing changes in plant species and communities, as well as to contribute to genetic and evolutionary studies of a native species of bromeliad (*Tillandsia aeranthos* (Loisel.) L.B.Sm.) that occurs in urban environments in southern Brazil. The Chapter III presents a global systematic review that integrates several areas of Urban Ecology, summarizing how plant species and communities are responding to urbanization and which are the main agents of these responses (*Urban Drivers*). After evaluating 171 studies published in the last eleven years, we found that most of them reported negative consequences of urbanization. The *Urban Drivers* most cited as responsible for changes in plant species and communities were "land cover change" and "biotic invasion". Among the most reported negative consequences, we have reduced plant species richness, lower availability of pollinators, environment homogenization, invasion of non-native species, changes in the timing of phenological events and difficulties for plant growth. However, it was evaluated that some species show evidence of adaptation to these adversities through their phenotypic plasticity and also through the evolution of characteristics that make individuals resilient to the urban environment. In Chapter IV we described the effects of urbanization on genetic diversity, genetic structure and local adaptation in *T. aeranthos*. Its CAM metabolism is advantageous for successfully colonizing the generally drier urban areas, making this species an excellent model for studying how plants can adapt to these environments. To test whether there would be differences in genetic patterns between populations that occur in urban and non-urban environments, we used sampling in the form of transects, distributed along an urbanization gradient. Obligatory cross-fertilization has allowed this species to maintain high levels of genetic diversity, with no decrease in genetic diversity with increasing urbanization. In agreement with this result, the fragmentation caused by urbanization was not enough to separate urban and non-urban populations through genetic structure analyses. The lack of correlation between the *outlier loci* detected through genomic scan and the urbanization index in our study corroborates the other genetic diversity indices not being affected by urbanization. Therefore, it can be assumed that dispersion can occur by pollen or seeds between urban and non-urban populations, which are the main determinants of the maintenance of genetic diversity in *T. aeranthos* populations. On the other hand, Chapter V presents a study on the production and viability of *T. aeranthos* seeds, as a way of comparing one of the aspects of its reproductive success in urban and non-urban environments. Seed production showed no statistically significant difference in relation to the urbanization gradient, but the germination rate decreased with increasing urbanization. As seed development is strictly dependent on successful cross-fertilization, it is likely that was a failure in the post-fertilization development of the seed. The results obtained in this thesis will contribute to studies evaluating the effects of urbanization on plant species, especially in tropical environments, which are still less represented on a global scale.

# *Capítulo I*

# *Introdução geral*

# Introdução geral

## A urbanização e as ameaças à biodiversidade

A atividade humana está alterando permanentemente o planeta Terra. As mudanças geoquímicas, climáticas e biológicas em escala global causadas pelos humanos levou os cientistas a propor uma nova época, o Antropoceno, convenientemente colocado com seu início em meados do século XX (Waters *et al.*, 2016).

Uma das maiores particularidades do Antropoceno é o rápido aumento da urbanização em todo planeta e as consequências dessa urbanização para o meio ambiente. A proporção da superfície global que é classificada como urbana é atualmente estimada em aproximadamente 3% e esta proporção continuará aumentando. Em 2018, 55% da população mundial residia em áreas urbanas, em comparação com apenas 30% em 1950, refletindo uma mudança na distribuição espacial das populações das áreas rurais para as áreas urbanas. Espera-se que o crescimento da população urbana continue, de modo que a população urbana mundial atingirá cinco bilhões em 2028 e seis bilhões em 2041 (Parris, 2016; United Nations, 2018).

Embora as áreas urbanas ocupem apenas 3% da superfície global, o crescimento urbano impõe grandes desafios à conservação da biodiversidade. Em comparação com outras atividades humanas que causam a perda de habitats, a expansão da urbanização é a mais duradoura, pois já induziu a uma grande transformação de paisagens, produzindo algumas das maiores taxas de extinção local e frequentemente eliminando a grande maioria das espécies nativas locais (McKinney, 2002). Portanto, em uma era de expansão urbana global, entender como a urbanização afeta a biodiversidade e como as espécies se adaptam aos ambientes urbanos são fundamentais para a conservação da biodiversidade (McKinney, 2002; Aronson *et al.*, 2014).

Os processos biofísicos associados à construção das cidades são frequentemente semelhantes, fazendo com que o habitat urbano imponha condições e desafios semelhantes para a persistência de espécies não humanas. Parris (2016) classificou esses processos biofísicos como primários (p. ex. remoção da vegetação existente; construção de prédios, estradas e outras infraestruturas urbanas; substituição de superfícies permeáveis por impermeáveis; poluição) e secundários (p. ex. perda de habitat, fragmentação e isolamento;

mudanças climáticas; poluição do ar, água e solo), sendo os processos secundários consequências ecológicas dos primários. Esses processos biofísicos da urbanização podem ter efeitos sobre espécies, populações e comunidades inteiras, alterando a quantidade, qualidade, arranjo temporal e espacial de recursos dos quais microrganismos, fungos, plantas e animais dependem para sobreviver, como abrigo, ninhos, alimentos, água, luz solar e nutrientes (Parris, 2016).

A disciplina de Ecologia Urbana surgiu a partir da necessidade de compreender toda a complexidade das relações entre a comunidade biológica e o ambiente urbano. A Ecologia Urbana pode ser definida como a ecologia de todos os organismos – incluindo humanos – em ambientes urbanos, bem como ambientes que são impactados pela construção, expansão e a operação das cidades (Douglas *et al.*, 2011; Parris, 2016). O amadurecimento dessa disciplina nos últimos anos está permitindo avaliar como as mudanças ambientais em decorrência da urbanização afetam a ecologia populacional das espécies, a estrutura da comunidade e os processos ecossistêmicos (Donihue & Lambert, 2015).

Apesar do aumento considerável no número de estudos nesta área, o impacto da urbanização em plantas ainda é pouco compreendido. A maioria dos estudos recentes em Ecologia Urbana concentra-se em animais, particularmente mamíferos, artrópodes e pássaros, com menos estudos avaliando os efeitos da urbanização em plantas (Aronson *et al.*, 2014; Johnson, Thompson, & Saini, 2015; Rivkin *et al.*, 2019).

Sem dúvida, os centros urbanos, são uma das principais fontes de emissões de gases de efeito estufa e, portanto, também são responsáveis pelas mudanças climáticas globais. O exemplo melhor documentado de modificação climática antropogênica é o efeito da ilha de calor urbana (*Urban Heat Island* - UHI): as cidades tendem a ter temperaturas mais altas do ar e da superfície do que seus arredores rurais. Diversas características dos ambientes urbanos podem causar a formação da UHI. Estas incluem, por exemplo, o aumento de superfícies impermeáveis (gerando redução do albedo – poder de reflexão de uma superfície) e a redução de áreas cobertas por vegetação e água (causando redução da perda de calor devido ao resfriamento evaporativo) (Grimm *et al.*, 2008).

Estudos apontam algumas das mais severas consequências para as espécies e comunidades de plantas devido às mudanças climáticas, como redução na riqueza de espécies devido à filtragem de espécies que não são adaptadas a habitats mais quentes (Albrecht & Haider, 2013; Chen *et al.*, 2014; Concepción *et al.*, 2017; Yu *et al.*, 2021) e aumento do estresse hídrico devido ao aumento dos níveis de dióxido de carbono e ozônio nas áreas urbanas (Calfapietra, Peñuelas, & Niinemets, 2015; Lahr *et al.*, 2015; Lahr, Dunn, & Frank, 2018; Zipper *et al.*, 2017; Barradas & Esperon-Rodriguez, 2021). Porém, um dos efeitos mais pronunciados e documentados são as mudanças na fenologia das plantas, o que pode ser utilizado como um bioindicador para a detecção de mudanças climáticas (Jochner & Menzel, 2015). Muitos estudos envolvendo análises fenológicas utilizando tanto abordagens espaciais (comparação entre ambiente urbano e rural) como temporais (comparações ao longo de décadas/séculos de urbanização) detectaram mudanças nos padrões fenológicos em plantas, como avanços nos períodos de desdobramento, coloração e queda das folhas, floração e amadurecimento dos frutos (Neil, Landrum, & Wu, 2010; Jeong *et al.*, 2011; van Vliet *et al.*, 2014; Massetti, Petralli, & Orlandini, 2015; Comber & Brunsdon, 2015; He *et al.*, 2016; Davis, Major, & Taylor, 2016; Yakub & Tiffin, 2017; Li *et al.*, 2017; Qiu, Song, & Li, 2017; Lee, 2017; Lindh, McGahan, & Bluhm, 2018; Oliveira *et al.*, 2019; Su *et al.*, 2019; Cochard *et al.*, 2019; Kondratyeva *et al.*, 2020; Fisogni *et al.*, 2020; Fitchett & Raik, 2021). Uma dessincronização entre a épocas de floração e os polinizadores pode levar as espécies a um isolamento reprodutivo, especialmente em plantas que têm uma curta duração da floração (Jochner & Menzel, 2015). Consequentemente, os vários efeitos das mudanças climáticas sobre as espécies provavelmente modificarão a rede de interações ao nível de comunidade. A resposta de algumas espécies às mudanças climáticas pode constituir um impacto indireto nas espécies que delas dependem (Bellard *et al.*, 2012).

A poluição do ar, da água e dos solos causada pela urbanização também afeta negativamente as plantas, podendo trazer impactos no seu crescimento e desenvolvimento (Huang *et al.*, 2013; Vujić *et al.*, 2015; Bouraoui *et al.*, 2019; King & Hovick, 2020; Turmukhametova & Shadrina, 2020; Su *et al.*, 2021); reduções na concentração de clorofila e de taxas fotossintéticas (Molnár *et al.*, 2018; Xiao *et al.*, 2021) e interrupção das trocas gasosas estomáticas pela deposição de material particulado nas folhas (Singh *et al.*, 2020; Singh, 2021).

A contínua expansão das cidades altera as paisagens de vegetação nativa e agrícola transformando-as em áreas cobertas por superfícies impermeáveis. Em muitos casos, o habitat que resta é fragmentado em pequenas áreas ou manchas, que tendem a ser isoladas umas das outras por uma paisagem inóspita formada pela infraestrutura urbana (Parris, 2016). Quando os habitats são fragmentados em pequenas áreas, eles são suscetíveis a mudanças ao longo do tempo. Pequenas manchas de habitat são vulneráveis a efeitos de borda – mudanças nas condições físicas e biológicas que ocorrem na fronteira de dois ecossistemas ou tipos de habitat. Em habitats terrestres, estes podem incluir mudanças microclimáticas (por exemplo, aumento da radiação, temperatura e vento, diminuição da umidade), invasão de espécies exóticas, aumento da pressão de herbívoros e predadores, influxo de poluentes, mudanças na ciclagem de nutrientes, mudanças na dinâmica populacional das espécies e aumento da perturbação por humanos (Grimm *et al.*, 2008; Williams *et al.*, 2009; Hahs & McDonnell, 2013). Essas mudanças podem levar à redução da frequência e abundância de espécies de plantas nativas e aumento da prevalência da flora exótica introduzida, levando à extinção local de muitas espécies de plantas nativas das cidades e a criação de novas comunidades de plantas urbanas (Hahs *et al.*, 2009; Williams, Hahs, & Vesk, 2015). Do mesmo modo, espécies que sobrevivem à perda de habitat podem ser altamente suscetíveis à extinção local, já que a fragmentação do habitat deve selecionar um conjunto previsível de espécies que carregam características relacionadas à persistência de metapopulações ou persistência apesar do baixo tamanho populacional. Por exemplo, espécies com capacidade de dispersão limitada e baixa produção de sementes são mais vulneráveis à extinção por falhas na recolonização de habitats. Da mesma forma, espécies que são fortemente dependentes de outros organismos numa relação mutualista (por exemplo, polinizadores especializados para reprodução), correm maior risco de extinção pela fragmentação (Williams *et al.*, 2009).

Em suma, as cidades são ecossistemas, nos quais muitos organismos interagem e prosperam. Como em qualquer ecossistema, os sistemas urbanos são caracterizados por seus conjuntos de espécies, bem como por seus ambientes químicos e físicos, cujos componentes interagem para governar os processos ecológicos, como ciclagem de nutrientes e fluxo de energia (Grimm *et al.*, 2008; Donihue & Lambert, 2015). No entanto, os ecossistemas urbanos são caracterizados pelos fortes impactos antropogênicos. A contínua expansão da urbanização é inevitável em todo o planeta, e manter a diversidade e a função das comunidades biológicas assim como seus serviços ecossistêmicos associados, dentro e perto das cidades é uma tarefa muito difícil (Grimm *et al.*, 2008). Compreender como os fatores abióticos, bióticos e antropogênicos moldam as comunidades locais podem levar a práticas que garantem o futuro de cidades sustentáveis (Aronson *et al.*, 2016).

*Dinâmicas evolutivas em ambientes urbanos*

A urbanização vem com uma mistura complexa de mudanças no ambiente que se repetem consistentemente nas cidades, aumentando a pressão sobre a diversidade das espécies locais remanescentes (Johnson & Munshi-South, 2017). Os estudos em Ecologia Urbana fornecem cada vez mais evidências de como essas mudanças ambientais afetam a ecologia das espécies, a estrutura da comunidade e os processos ecossistêmicos. No entanto, sabemos muito menos sobre como os impactos ecológicos da urbanização afetam a evolução das populações de organismos que vivem nas cidades (Donihue & Lambert, 2015; Johnson & Munshi-South, 2017), e como essa evolução pode influenciar os processos e padrões ecológicos por meio de dinâmicas ecoevolutivas (Alberti, 2015). Uma das questões cruciais no debate sobre os efeitos ecológicos em decorrência da urbanização é se as espécies serão ou não capazes de se adaptar rápido o suficiente para acompanhar o ritmo acelerado dessas mudanças. Mesmo quando as espécies são capazes de persistir no ambiente a curto prazo, a redução prolongada no tamanho da população ameaça a sua persistência a longo prazo por meio de uma série de efeitos demográficos e genéticos previsíveis para as populações (Lande, 1988).

Por exemplo, a fragmentação e degradação dos habitats naturais em decorrência da urbanização frequentemente reduz o tamanho e aumenta o isolamento das populações nativas (Haddad *et al.*, 2015). Isto ocorre porque a fragmentação reduz o tamanho das manchas de habitat e aumenta o isolamento espacial dessas manchas, com uma matriz circundante inóspita. À medida que a extensão da matriz cresce, o tamanho dos fragmentos reduz, assim como a dispersão entre eles (Cheptou *et al.*, 2017). Espera-se que esses processos ecológicos aumentem a estocasticidade nas frequências alélicas por meio da deriva genética e dos efeitos fundadores e diminuam a dispersão e o movimento de alelos (ou seja, fluxo gênico) em paisagens urbanas (Johnson & Munshi-South, 2017). Mudanças nos fatores bióticos e abióticos geradas pela urbanização também podem alterar a seleção natural e impulsionar a evolução adaptativa (Donihue & Lambert, 2015). As primeiras evidências de evolução em resposta à urbanização incluem algumas das primeiras demonstrações de adaptação contemporânea na natureza. Por exemplo, o aumento da frequência da forma escura da mariposa *Biston betularia* Linnaeus (*peppered moth*) em resposta à poluição industrial elevada em torno das cidades do início do século XIX a meados do século XX

(Kettlewell, 1955). Apesar desse exemplo clássico, os estudos focados na evolução em ambientes urbanos só ganharam força recentemente (Rivkin *et al.*, 2019).

As consequências evolutivas das alterações ambientais nos sistemas urbanos têm sido amplamente estudadas do ponto de vista da genética de populações, com ênfase particular em como a fragmentação do habitat resulta em mudanças estocásticas mais pronunciadas nas frequências alélicas. A deriva pode diminuir a diversidade genética, reduzindo o potencial evolutivo das populações ao lidar com novos ambientes (Frankham, Ballou, & Briscoe, 2010). É esperado que este processo aumente em muitas espécies à medida que a extensão da urbanização aumenta. Por exemplo, habitats fragmentados levaram à redução da diversidade genética dentro das populações e maior diferenciação genética entre as populações urbanas de *Linaria vulgaris* Mill. (Bartlewicz *et al.*, 2015), em comparação com populações em habitats não urbanizados. Por outro lado, a urbanização não levou a uma perda da diversidade genética, e algumas medidas de urbanização até foram correlacionadas ao aumento da diversidade genética em populações de *Trifolium repens* L. (Johnson *et al.*, 2018). Embora o resultado do aumento dos efeitos da deriva (e a possível consequência na redução da diversidade dentro das populações) tenha sido identificado em várias cidades em outros organismos (Rivkin *et al.*, 2019), poucos estudos examinaram em detalhes quais características do ambiente urbano (por exemplo estradas, edifícios) afetam a demografia populacional em plantas. Padrões esperados de diversidade e estrutura genética baseados nos estudos realizados em habitats fragmentados (não-urbanos) oferecem previsões de como as plantas responderiam à urbanização. Porém, a forma rápida como ocorre a alteração da paisagem pela atividade humana fornecerá novos conhecimentos em relação à evolução das espécies nas cidades. Até recentemente, a adaptação à fragmentação antropogênica era considerada improvável, pois os processos evolutivos eram considerados ineficientes em escalas de tempo de algumas décadas. No entanto, estudos recentes demonstraram que a adaptação de traços de história de vida pode ocorrer em escalas de tempo curtos, com mudanças evolutivas observáveis em até apenas duas gerações (Kinnison & Hendry, 2001; Bodbyl Roels & Kelly, 2011).

Um efeito importante da fragmentação é o aumento da variabilidade nas condições abióticas e na estocasticidade demográfica que as pequenas populações estão sujeitas, gerando vários níveis de seleção nas populações. Os efeitos demográficos e genéticos populacionais alteram a forma como as espécies respondem à seleção (Cheptou *et al.*, 2017).

Ao reduzir o fluxo gênico entre manchas, a fragmentação pode melhorar a adaptação local de uma espécie na paisagem presente nas cidades. Por exemplo, pode ocorrer seleção diversificadora (seleção disruptiva) entre manchas causada por fatores abióticos (por exemplo, deterioração do habitat, efeitos de borda) ou fatores bióticos (por exemplo, perda de interação com organismos mutualistas). Consequentemente, a adaptação local e a deriva podem gerar populações localmente diferenciadas, mesmo em escalas de tempo curtas (Cheptou *et al.*, 2017). Por outro lado, a fragmentação pode selecionar espécies que possuem uma maior amplitude de nicho ecológico (ou seja, uma estratégia mais generalista), que permitam às espécies lidar com a heterogeneidade do habitat. Organismos que podem permanecer com sucesso nos habitats de borda e até mesmo na matriz terão mais habitats disponíveis para colonização e serão menos isolados (Cheptou *et al.*, 2017). Espécies que são especialistas podem sofrer mais com as flutuações de habitat do que espécies generalistas. Consequentemente, serão selecionadas estratégias de história de vida que aumentem a aptidão média em toda a paisagem (Comins, Hamilton, & May, 1980). De fato, uma maior frequência de plantas com traços generalistas (como dispersão pelo vento, baixa estatura e presença de sementes pequenas) nos ambientes urbanos foi encontrada em estudos de riqueza filogenética das comunidades de plantas urbanas (Knapp, Winter, & Klotz, 2017; Lopez, Urban, & White, 2018; Silva-Junior *et al.*, 2018; Cui *et al.*, 2019; El-Barougy *et al.*, 2021). Já espécies especialistas e aquelas com baixa dispersão são mais afetadas negativamente pela urbanização, confirmando que a fragmentação pode exercer tal seleção (Huang *et al.*, 2013; Mccune & Vellend, 2013; Everingham, Hemmings, & Moles, 2019; Oliveira *et al.*, 2020; Yang *et al.*, 2021).

O efeito final da fragmentação na dinâmica populacional de uma determinada espécie depende do grau em que a dispersão entre fragmentos é impedida. Uma alta dispersão pode manter uma população panmítica mesmo em paisagens severamente fragmentadas. Já uma dispersão menos efetiva resultará em dinâmicas metapopulacionais, onde as populações funcionam de forma independente, e as manchas experimentam extinção regular e subsequente recolonização. Uma fragmentação mais severa deixará manchas isoladas, como ilhas, onde as extinções locais não serão mais contrabalançadas pela colonização (Cheptou *et al.*, 2017). A probabilidade da dispersão ocorrer, por sua vez, depende da extensão espacial da fragmentação (distância entre manchas), da facilidade com que os indivíduos podem se dispersar pela matriz (permeabilidade da matriz) e da capacidade de dispersão da espécie.

Assim, espécies diferentes responderão à mesma paisagem de forma diferente (Hanski, 1991; Ricketts, 2001; Cheptou *et al.*, 2017).

Quando os tamanhos efetivos da população dos fragmentos são pequenos, pode haver seleção para melhores estratégias de dispersão entre os fragmentos. Em pequenas populações muitas vezes os cruzamentos se darão inevitavelmente entre aparentados, resultando em uma diminuição da aptidão devido à depressão endogâmica (Ricketts, 2001). Uma das consequências pode ser o aumento da seleção a favor de maior dispersão entre manchas como forma de evitar a depressão endogâmica. Desta forma, bons dispersores terão acesso a mais manchas e, portanto, a recursos mais abundantes (Cheptou *et al.*, 2017). Por outro lado, espera-se que a fragmentação também selecione contra a dispersão, pois a matriz hostil entre os fragmentos aumenta o risco de o organismo não encontrar um habitat adequado para seu estabelecimento (Bonte *et al.*, 2012). Por exemplo, uma das primeiras demonstrações dos efeitos de ambientes urbanos fragmentados na seleção de plantas foi realizada em *Crepis sancta* (L.) Babc. (Cheptou *et al.*, 2008). Esta planta apresenta variação hereditária para a proporção de sementes dispersantes e não-dispersantes, e experimentos mostraram que a fragmentação do habitat urbano impõe a seleção para a produção de mais sementes não-dispersivas, já que estas são mais propensas a pousar em substratos adequados próximos à planta-mãe, para germinação em habitats urbanos. Logo, a fragmentação nos ambientes urbanos pode impor seleção às populações de plantas, o que pode resultar em adaptação se as populações tiverem a variação genética adaptativa necessária para responder (Rivkin *et al.*, 2019). Entretanto, exemplos tão claros dos efeitos de ambientes urbanos na evolução por seleção natural existem em poucos sistemas, representando uma lacuna para a compreensão da evolução em ambientes urbanos.

Como as alterações físicas e demográficas da fragmentação afetam as espécies de maneiras diferentes, eles também podem interromper as interações entre as espécies, causando uma cascata de efeitos no nível da comunidade local. Estudos inspirados na biogeografia de ilhas mostram que espécies são perdidas em pequenos fragmentos, especialmente espécies que preferem habitats específicos (Laurance *et al.*, 2002). Essas extinções locais afetam as interações mutualísticas e antagônicas das espécies. Por exemplo, declínios de polinizadores, dispersores de sementes e predadores insetívoros em fragmentos da Floresta Amazônica podem diminuir o sucesso reprodutivo das plantas enquanto aumentam a herbivoria (Laurance *et al.*, 2002). De fato, a diminuição das populações e/ou atividade de polinizadores é um dos efeitos melhor estudados quando se trata de fragmentação, e descobriu-se que isso impede a reprodução de plantas que vivem nas cidades e que são dependentes de polinizadores (Aguilar *et al.*, 2006; Lhotte, Affre, & Saatkamp, 2014; Leong, Kremen, & Roderick, 2014; Irwin, Warren, & Adler, 2018; Oliveira *et al.*, 2019; Hou *et al.*, 2019). As plantas também podem possuir adaptações que ajudem a compensar os efeitos demográficos da fragmentação. Pequenas populações de plantas podem enfrentar uma escassez de parceiros, o que pode favorecer sistemas de cruzamento menos exigentes. De fato, a autocompatibilidade e a autofertilização em plantas são frequentemente associadas a populações pequenas (Baker, 1955), permitindo que essas espécies tenham sucesso reprodutivo em ambientes urbanos com menor frequência ou até mesmo na ausência de polinizadores (Ushimaru, Kobayashi, & Dohzono, 2014).

As características fenotípicas determinam a capacidade de uma espécie colonizar e estabelecer uma população em ambientes urbanos (Rivkin *et al.*, 2019). Entretanto, atualmente, não se sabe se os traços relacionados ao estabelecimento e proliferação de populações em áreas urbanas são resultado de características ancestrais, adaptação recente, plasticidade fenotípica (incluindo mudanças epigenéticas) ou uma combinação desses atributos. Múltiplos processos bióticos e abióticos determinam o subconjunto de espécies presentes nas cidades (Williams *et al.*, 2009; Aronson *et al.*, 2016). Além disso, embora a adaptação rápida nas cidades tenha sido encontrada em algumas espécies de plantas e animais (Johnson & Munshi-South, 2017), não está claro se isso é um fenômeno comum. Desvendar a importância relativa da adaptação versus plasticidade para o sucesso de uma espécie em uma determinada cidade contribuirá para entender como e por que algumas espécies, mas não outras, são capazes de colonizar e persistir em ambientes urbanos (Rivkin *et al.*, 2019). Para descobrir se as diferenças nas características entre populações urbanas e não-urbanas representam adaptações genéticas em vez de plasticidade não hereditária, métodos genômicos podem ser usados para examinar assinaturas de rápida adaptação à urbanização (por exemplo, escaneamentos genômicos: Harris & Munshi-South, 2017; Theodorou et al., 2018). Análises de expressão gênica também podem ser usadas para identificar divergências adaptativas e os fundamentos genéticos das respostas à urbanização (Harris, O’Neill, & Munshi-South, 2015). Juntas, essas abordagens esclarecerão a importância relativa e as interações entre características ancestrais, adaptação rápida e plasticidade fenotípica na determinação do sucesso ecológico das populações urbanas (Rivkin *et al.*, 2019).

As cidades alteram drasticamente o ambiente abiótico e biótico, o que pode influenciar a seleção natural nas populações urbanas. Apesar dessa simples previsão, os agentes que afetam a aptidão podem ser difíceis de discernir, em parte porque as mudanças ambientais nas cidades variam de simples a complexas. A urbanização inclui mudanças ambientais ao longo de múltiplas dimensões interativas, como a quantidade de superfície impermeável, temperatura, poluição (solo, ar, água, luz e som), disponibilidade de recursos, bem como a abundância e diversidade de concorrentes, predadores e mutualistas. Essa multidimensionalidade pode aumentar drasticamente a complexidade dos ambientes aos quais as populações devem se adaptar e pode tornar desafiadora a descoberta de agentes específicos de seleção. Identificar generalidades sobre a força, a forma e os agentes de seleção em ambientes urbanos permitiriam uma compreensão mecanicista dos processos adaptativos associados à urbanização (Rivkin *et al.*, 2019). Compreender as aplicações ecológicas e ecossistêmicas mais amplas da evolução urbana para espécies nativas representa uma importante prioridade para pesquisas futuras em ecologia evolutiva urbana. A integração dessas pesquisas pode ajudar a projetar estratégias que facilitem a adaptação das espécies à urbanização, melhorar a previsão de declínios populacionais e desenvolver planos de conservação e manejo apropriados (Alberti, 2015). A identificação dos impulsionadores da evolução nas cidades e a história evolutiva das espécies que as habitam pode ser usada para construir novas comunidades resilientes às pressões urbanas e melhorar a conservação da biodiversidade global.

### *A família Bromeliaceae*

Bromeliaceae (3782 espécies, Gouda *et al*., atualização contínua) é uma das maiores famílias de plantas com flores encontradas quase exclusivamente nos Neotrópicos (Stevens, 2013). Esta família é reconhecida como um dos melhores exemplos de radiação adaptativa – existe uma enorme diversidade de formas de vida, desde formas terrestres com raízes funcionais até epífitas completamente independentes do seu substrato para a nutrição (Benzing, 2000). Uma série de inovações que exigem a integração de um conjunto de características bioquímicas, anatômicas, fisiológicas e do organismo, permitiu a colonização e especiação bem-sucedida das bromélias em ambientes novos e sob estresse (Benzing, 2000; Crayn *et al.*, 2004, 2015; Givnish *et al.*, 2007, 2011, 2014; Males, 2016; Palma-Silva *et al.*, 2016).

As bromélias são plantas típicas do Novo Mundo, com distribuição geográfica que vai desde os estados da Virgínia, Texas e Califórnia, nos Estados Unidos (limite norte), ao norte da Patagônia, na Argentina (limite sul). A única exceção é *Pitcairnia feliciana* (A. Chev.) Harms & Mildbr distribuída no oeste da África, na região da Guiné, o que parece ter sido um evento de recente dispersão a longa distância (Givnish *et al.*, 2004). O Brasil apresenta uma grande diversidade de bromélias com 56 gêneros (24 endêmicos) e 1.386 espécies (1.186 endêmicas) (Flora e Funga do Brasil, 2022). Especificamente, a Mata Atlântica apresenta uma grande riqueza de espécies (Martinelli *et al.*, 2008), sendo as epífitas dentre as Angiospermas com o maior número de registros neste bioma (37,6%) (Ramos *et al.*, 2019).

Tradicionalmente, Bromeliaceae estava dividida em três subfamílias: Pitcairnioideae, Tillandsioideae e Bromelioideae (Smith & Downs, 1974; 1977; 1979). No entanto, análises filogenéticas realizadas com marcadores plastidiais indicaram o reconhecimento de oito subfamílias (Givnish *et al.*, 2007, 2011): Brocchinioideae, Lindmanioideae, Tillandsioideae, Hechtioideae, Navioideae, Pitcairnioideae, Puyoideae e Bromelioideae. Estima-se que a as bromélias surgiram no Escudo das Guianas há cerca de 100 milhões de anos atrás (Ma), durante o Período Cretáceo, com as subfamílias existentes começando a divergir há apenas 19 Ma (Givnish *et al.*, 2011). Os mesmos autores sugerem que as bromélias começaram a ocupar os Andes, a Amazônia, a América Central, o Caribe e o Escudo brasileiro entre 15 e 10 Ma.

Esta família é constituída por plantas terrestres, rupícolas e epífitas, herbáceas, variando de plantas delicadas e de pequeno porte, como *Tillandsia recurvata* (L.) L., com alguns centímetros de comprimento, até plantas de grande porte, como *Puya raimondii* Harms, encontrada nos Andes, que chega a atingir mais de 10 metros de altura (Smith & Downs, 1977). As bromélias apresentam grande variabilidade de formas, sendo em geral plantas bem características e ornamentais. Segundo Rizzini (1997) e Benzing (2000) os diferentes habitats e, especialmente, a natureza do substrato influenciam no aspecto da planta, que pode variar amplamente em tamanho e coloração das folhas, assim como na morfologia das flores. A evolução de uma ampla diversidade de cores e formas de estruturas florais atraíram os mais variados polinizadores, como insetos, morcegos e beija-flores (Benzing, 2000). Os beija-flores podem ser considerados os principais agentes polinizadores da família (Benzing, 2000; Dias *et al.*, 2014). Em um estudo de polinização com beija-flores, realizado em regiões da Mata Atlântica, as bromélias representaram 36% da flora ornitófila e 33% das flores mais ricas em néctar (Buzato, Sazima, & Sazima, 2000). Embora a maioria das bromélias seja polinizada por vertebrados, principalmente beija-flores e morcegos, as abelhas estão entre os visitantes mais frequentes de algumas espécies de corola curta com características ornitófilas. No entanto, poucos estudos identificaram insetos como polinizadores eficazes de bromélias (Kamke *et al.*, 2011). Essas plantas podem possuir frutos carnosos que são consumidos e dispersados por aves, e o tanque formado pelas suas folhas, em grande parte das espécies, permite o armazenamento de água e nutrientes, servindo para sua própria nutrição e fornecendo um microambiente favorável que proporciona alimento, abrigo e local de reprodução para diversos animais associados (Benzing, 2000; Fleming & Muchhala, 2008; Dias *et al.*, 2014).

Apesar de serem plantas quase exclusivamente herbáceas, é notável sua diversidade ecológica, visto que as espécies são encontradas em diferentes tipos de substratos, bem como nos mais variados habitats (Fischer & Araujo, 1995; Benzing, 2000), constituindo um sistema de estudo favorável às abordagens ecológicas e evolutivas.

### *Tillandsioideae, o gênero Tillandsia e o epifitismo*

A Subfamília Tillandsioideae Harms é a maior das oito subfamílias atualmente estabelecidas para a família Bromeliaceae, apresentando a maior distribuição geográfica dentre elas (Givnish *et al.*, 2007, 2011). As duas principais origens do epifitismo em bromélias se deram nesta subfamília, com o soerguimento do norte dos Andes ca. 15 Ma e em Bromelioideae (epífitas com tanque), com o soerguimento das Serras do Mar e da Mantiqueira na região da Mata Atlântica durante o Plioceno-Pleistoceno (Givnish *et al.*, 2014).

Espécies atmosféricas do gênero *Tillandsia* são epífitas que não formam tanque e que dependem quase exclusivamente de tricomas foliares absorventes para absorção de água e nutrientes (Benzing, 2000). As raízes dessas epífitas são frequentemente reduzidas, servindo apenas para fixação no substrato (Pittendrigh, 1948; Benzing, 1990). Os tricomas abundantes tornam suas folhas secas altamente refletivas, possivelmente reduzindo a perda de água e a fotoinibição (Pierce *et al.*, 2001). Conjuntos densos de tricomas desse gênero são altamente eficientes na captura de umidade quando combinados com suas folhas pequenas e estreitas que capturam eficientemente gotículas finas da atmosfera (Martorell & Ezcurra, 2007). As bromélias atmosféricas toleram melhor a dessecação do que as espécies de tanque, mas têm taxas mais baixas de fotossíntese por unidade de massa foliar (Benzing & Burt, 1970; Benzing & Renfrow, 1974; Reyes-García, Griffiths, & Rincón, 2008), e quase todas têm fotossíntese CAM (Crayn *et al.*, 2004).

As epífitas em geral são um grupo composto por mais de 31.000 espécies pertencentes a 84 famílias de plantas, representando uma importante proporção da flora mundial (Zotz *et al.*, 2021). São plantas que utilizam outras plantas como substrato para o crescimento, sem extrair delas nenhum recurso hídrico ou nutriente (Benzing, 1990). As epífitas vasculares germinam e crescem sobre forófitos (árvores-suporte) adquirindo uma distribuição espacial horizontal (entre as espécies de forófitos) e vertical (dentro dos forófitos, desde a base do fuste até a copa). A maioria das epífitas é angiosperma, sendo Orchidaceae e Bromeliaceae as famílias com maior número de representantes desse grupo de plantas (Zotz, 2013). A maior diversidade de epífitas é geralmente encontrada em habitats tropicais úmidos, como florestas nubladas e florestas tropicais (Gentry & Dodson, 1987; Benzing, 1990; Nieder, Prosperí, & Michaloud, 2001).

A maioria dos estudos sobre epífitas estão concentrados na morfologia, ecofisiologia e taxonomia dessas plantas (Benzing, 1978, 1990; Zotz, 2004; Wanek & Zotz, 2011). No entanto, mesmo com o grande número de angiospermas epífitas, pouco se sabe sobre a ecologia populacional deste grupo (Mondragón-Chaparro *et al.*, 2006; Matallana *et al.*, 2010; Zotz *et al.*, 2010). Como as epífitas estão sujeitas a pressões seletivas semelhantes, dado o habitat específico que ocupam, é esperado que elas compartilhem semelhanças em suas características demográficas, respostas das populações, padrões de mortalidade, sobrevivência e reprodução (Mondragon, Valverde, & Hernandez-Apolinar, 2015). Recentemente, biólogos da conservação começaram a se concentrar na ecologia populacional de epífitas, já que este grupo de plantas possui uma grande sensibilidade às mudanças climáticas globais, e tem grande potencial como ferramenta de avaliação para o estabelecimento do estado de conservação de florestas tropicais (Hsu *et al.*, 2012).

A estratégia de dispersão mais difundida entre as epífitas é a anemocoria, com a produção de grandes quantidades de sementes pequenas e leves, aumentando assim a probabilidade de que pelo menos algumas delas cheguem a locais seguros (Madison, 1977; Hughes *et al.*, 1994). Em populações de plantas epífitas, a dispersão de sementes pode acontecer de duas formas: as sementes permanecem próximas à sua fonte (ou seja, no mesmo forófito) ou deixam seu forófito original. Neste último caso, as espécies confrontam o processo altamente arriscado de cruzar a matriz de habitat onde os forófitos estão imersos e, eventualmente, encontrar um local seguro para germinação e crescimento (Mondragon *et al.*, 2015). Apesar do alto risco envolvido neste processo, o fato de tantas epífitas apresentarem adaptações de dispersão de sementes a longa distância torna evidente que suas vantagens evolutivas superam seus custos (Horvitz & Schemske, 1986). Provavelmente a maior vantagem para as epífitas dispersarem suas sementes a longas distâncias seja a maior possibilidade de colonização de novas manchas de habitat que podem ser efêmeras e/ou livres de competição (Mondragon *et al.*, 2015). Entretanto, observações de campo sugerem que a maioria das sementes tendem a cair perto da planta mãe, enquanto a dispersão de sementes a longa distância é relativamente rara (Mondragón-Chaparro *et al.*, 2006; Paggi *et al.*, 2010; Chaves *et al.*, 2021). Eventos de dispersão de sementes a longa distância são importantes para o aumento do fluxo gênico e para a expansão das áreas de distribuição das espécies através da fundação de novas populações.

As epífitas são atualmente consideradas um dos grupos de plantas mais ameaçadas. Isso se deve em parte à sua dependência da presença/disponibilidade de árvores hospedeiras e ao fato de que a maioria das epífitas satisfaz suas necessidades de nutrientes e água diretamente da atmosfera (Benzing, 1998; Zotz *et al.*, 2010; Obregon *et al.*, 2011), o que as torna vulneráveis a potenciais mudanças atmosféricas. Mudanças climáticas globais como aumento da temperatura e diminuição da precipitação já são percebidas atualmente, além do aumento da concentração atmosférica de $CO_2$ (Murphy *et al.*, 2004; Stainforth *et al.*, 2005). Estima-se que essas mudanças, principalmente a redução da umidade atmosférica, afetarão drasticamente o desempenho de epífitas e, portanto, sua dinâmica populacional (Benzing, 1990; Zotz *et al.*, 2010; Müller, Albach, & Zotz, 2018). Embora muitas epífitas apresentem metabolismo CAM, temperaturas alguns graus acima do ótimo noturno resultam em dessecação do tecido ou aumentam os déficits de saturação nas massas de ar circundantes, o que reduz a condutância estomática e, portanto, a disponibilidade de $CO_2$ para a fotossíntese

(Benzing, 1998). Em adição a isso, as mudanças no uso da terra e perda de habitats através do desmatamento podem diminuir a abundância e a diversidade de epífitas devido ao impacto das atividades humanas nas florestas em que habitam (Hietz, 1997; Wolf, 2005; KÖster *et al.*, 2009; Köster, Nieder, & Barthlott, 2011; Adhikari, Fischer, & Fischer, 2012; Chaves *et al.*, 2021). A mudança na diversidade e composição de espécies epífitas após um distúrbio antropogênico tem sido associada a dois processos: restrições na dispersão e alteração das condições microclimáticas no habitat remanescente (Mondragon *et al.*, 2015).

A maioria das epífitas apresenta sistemas de reprodução mistos, que em muitos casos incluem mecanismos que favorecem a fecundação cruzada. Essa variação nos sistemas de reprodução reflete na diversidade genética, no entanto, relativamente pouco foi revelado sobre a estrutura genética desse grupo de plantas em habitats fragmentados, e menos ainda em ecossistemas urbanos (Izuddin & Webb, 2015; Quail *et al.*, 2022). Além disso, os padrões de dispersão de sementes das epífitas permitem que este grupo colonize novos habitats, sendo capazes de atingir altos níveis de abundância em paisagens urbanas (Fenster & Martén-Rodríguez, 2007; Chaves *et al.*, 2021). Como resultado, as epífitas são um grupo ideal para o estudo da conectividade das plantas em paisagens urbanas e para ajudar na compreensão dos mecanismos evolutivos envolvidos nas respostas das plantas frente à urbanização.

# *Capítulo II*

# *Objetivos*

# Objetivos

## *Objetivo geral*

Abordar diferentes aspectos da urbanização como causadores de mudanças em espécies e comunidades de plantas, assim como contribuir com estudos genéticos e evolutivos de uma espécie nativa de bromélia (*Tillandsia aeranthos*) que ocorre em ambientes urbanos no sul do Brasil.

## *Objetivos específicos*

a. Revisar na literatura os aspectos da urbanização que geram respostas em espécies e comunidades de plantas em escala global;
b. Caracterizar a diversidade e a estrutura genética de *T. aeranthos* usando marcadores moleculares do tipo AFLP, ao longo de dois gradientes de urbanização, utilizando uma amostragem na forma de transecto;
c. Testar se há diferenciação alélica em populações amostradas ao longo dos transectos e se esta está correlacionada com o gradiente de urbanização;
d. Detectar *outlier loci* ao longo dos transectos que possam estar potencialmente ligados à divergência adaptativa relacionada com as mudanças causadas pela urbanização;
e. Buscar indícios dos efeitos da urbanização no sucesso reprodutivo, através da avaliação da produção e na viabilidade de sementes de populações de *T. aeranthos* ao longo de um gradiente de urbanização.

# *Capítulo III*

# *Urbanization driving changes in plant species and communities— A global view*

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# Urbanization driving changes in plant species and communities – A global view

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ABSTRACT

The urban landscape is becoming the environment most familiar to most of humanity, which has consequences for society, the environment, and biodiversity. Here, we present a systematic review of current research integrating several areas of urban ecology to evaluate how plants are responding to urbanization. We found that most studies were conducted in Europe and North America. The most represented biome was that of Temperate Broadleaf and Mixed Forests. The majority of studies reported negative consequences of urbanization. The Urban Drivers most commonly cited as responsible for changes in plant species and communities were Land Cover Change and Biotic Invasion. In general, urbanization reduces the richness of plant species and pollinators' availability. It also homogenizes the environment, promotes the invasion of non-native species, causes changes in phenological events, and presents difficulties for plant growth. However, some species show evidence of adaptation to these adversities through their phenotypic plasticity and the evolution of traits that make individuals resilient to the urban environment. Although it has been established that urbanization imposes novel selective pressures resulting in unique adaptations to city life, patterns of changes in biodiversity in response to urbanization can vary by region, biomes, city history, and the taxa studied. The Urban Drivers responsible for floristic responses work simultaneously by modifying the environment. Therefore, it is challenging to predict response patterns of plants under urban growth. It is necessary to outline strategies for continuous monitoring to document the progress of species in perpetuating themselves in the face of urbanization.

## 1. Introduction

Urbanization is a complex socio-economic process with numerous consequences for different levels of society, the environment, and biodiversity (United Nations, 2018). The urban landscape is becoming the most familiar environment for most of humanity, despite covering a small proportion of the Earth's surface (Niemelä, 2011). More than half of the world's people currently live in urban areas, with 55 % of the world's population residing in urban areas in 2018, compared with 30 % in 1950, reflecting a shift in the spatial distribution of populations from rural to urban areas (Parris, 2016; United Nations, 2018). Urban population growth is expected to continue, such that the world's urban population will reach five billion in 2028 and six billion in 2041 (United Nations, 2018).

Urbanization leads to primary (e.g., removal of existing vegetation and construction of urban infrastructure) and secondary (e.g.,

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habitat loss, fragmentation and isolation, climatic changes, pollution of air, water, and soil) processes that represent many challenges to the persistence of non-human species (Parris, 2016). Cities have distinctive characteristics due to the changes that happen to the land in urban areas. All ecosystems are affected by the same broad suite of state factors (Chapin et al., 2002), such as climate, substrate, resident organisms, relief, and the system's history. The urban ecosystem is no different. However, urban biotic communities are also highly influenced by anthropogenic impacts. Natural areas, agriculture, and forests are primarily replaced by predominantly built-up surfaces, changing cities' environment, which significantly impacts the physical environment, ecosystem processes, and ecology of organisms that live in cities (Grimm et al., 2008; McDonnell and MacGregor-Fors, 2016).

Biological introductions and human disturbance, combined with primary and secondary urbanization processes, impact species' births, deaths, immigration, and emigration rates. These combined factors lead to population growth or decline, which in turn may further alter intra- and interspecific interactions in different groups of organisms (Parris, 2016). Understanding the relative roles of abiotic, biotic, and anthropogenic filters that shape local communities provides a deeper understanding of community assembly, leading to practices that ensure the future of sustainable and biodiverse cities (Aronson et al., 2016).

The impacts of urbanization on biodiversity and ecosystems, especially on a global scale, are insufficiently understood. Previous work has established that urban areas impose novel selective pressures resulting in unique adaptations to city life (Niemelä, 2011). Although generalizations about the effect of urban areas on biodiversity are often made, actual patterns can vary by region, biomes, and city history (Müller et al., 2013; Niemelä, 2011). Furthermore, understanding which species are likely to benefit and which are likely to be detrimentally impacted will be important for biodiversity conservation and ecosystem services in an increasingly urbanized world (Williams et al., 2015). To improve our understanding of ecology in urban ecosystems, there is a need for comparative research worldwide (McPhearson et al., 2016).

The discipline of urban ecology has provided increasing evidence of how environmental changes affect species' population ecology, community structure, and ecosystem processes (Donihue and Lambert, 2015). Specific trends of living organisms' responses to urban stressors have been predicted based on prior knowledge of ecology and evolution (Johnson et al., 2015). Most of the recent studies in urban evolutionary ecology have focused on animals, particularly mammals, arthropods, and birds, and fewer studies have examined the effects of urbanization on plant evolution (Aronson et al., 2014; Johnson et al., 2015; Rivkin et al., 2019). Previous reviews on the topic of urban ecology have specified issues regarding the possible responses of plants to the biotic and abiotic changes that occur in urban areas (e.g., Alberti et al., 2017; Aronson et al., 2016; Cadotte et al., 2017; Eckert et al., 2010; Harrison and Winfree, 2015; Jochner and Menzel, 2015; Johnson et al., 2015; Rai, 2016; Williams et al., 2015, 2009). However, how the impacts of urbanization drive the evolution of living organisms in cities remains poorly understood (Donihue and Lambert, 2015; Johnson and Munshi-South, 2017; Rivkin et al., 2019).

Most studies examine only a few populations in one city (Johnson and Munshi-South, 2017). However, cities worldwide have different development histories and urbanization varies in intensity. Moreover, cities are inserted in different biomes and climates that can lead to a variety of responses in local biodiversity. The different contexts in which cities have developed their histories provide an excellent opportunity to understand the repeatability and pace of evolution in response to human activity. Considering this, we still need to understand more deeply the effects that this inevitable increase in urbanization will bring to the environment to facilitate the establishment of resilient and sustainable urban ecosystems (Johnson and Munshi-South, 2017; Niemelä, 2011).

In this article, we do a comprehensive systematic review of what has been found in the last eleven years of publications in urban ecology, focusing on plant species and communities worldwide. Our goals were to (1) summarize how urbanization drives plant changes in species and communities worldwide, (2) to understand whether plants respond to urban environments via phenotypical plasticity or through adaptation, (3) and to analyze whether Urban Drivers generate different responses based on the biome and urban density in plant species and communities. We also aimed to integrate several areas that fall under urban ecology to evaluate species plant and communities' structure changes in response to Urban Drivers.

## 2. Methods

We undertook a broad systematic review of peer-reviewed articles reporting data about the effects of urbanization in two ecological levels (species and communities) in vascular plants worldwide. We focused on studies and observations made completely in the field, comparing the results obtained between urban and non-urban areas (e.g., natural, agricultural, and green areas). We also included studies reporting comparisons between different time points in the history of increasing urbanization in various cities. We did not include studies exclusively performed with urban environment simulations in laboratory/greenhouse or those that evaluated flora that did not occur spontaneously in the study area.

From June 2020 to January 2022, we searched the literature in the Web of Science (WoS) database. The related search keywords were "urban plants" AND "species richness" OR "ecology" OR "genetic" OR "evolution" OR "adaptation" OR "pollinators" OR "reproduction" OR "dispersion" OR "physiology." We considered the matches to our search terms within the topic, title, keywords, and abstract. Studies meeting the inclusion criteria were: i) published in a peer-reviewed journal, ii) written in English, iii) reporting observational or experimental field data about effects of urbanization in plants, iv) published between 2010 and 2021. We recorded the studies in a spreadsheet, and relevant information was extracted and allocated into categories to summarize the methodologies used in the articles and the obtained results.

We classified the studies' methods according to the designed approach: urban–non-urban gradient, urban vs. non-urban (spatial comparison between urban and non-urban habitats), or temporal (changes over years of urbanization). We also recorded locations globally, considering the 14 terrestrial biomes of the world according to Dinerstein et al. (2017) (updated version of the Terrestrial Ecoregions of the World from Olson et al., 2001). When the city of the study was provided, we classified them as "megacity" if they have more than 1 million inhabitants, in keeping with the United Nations' World Economic Situation and Prospects (United Nations, Department of Economic and Social Affairs, 2018), or as "small cities" if they have fewer than 1 million inhabitants. We also recorded the variables analyzed in each study. We categorized them into major areas to overview the main variables studied over the past eleven years. The variables allowed us to identify the following principal major areas reviewed: "community traits" (e.g., species richness and abundance, vegetation cover, demography, non-native species richness, herbivory, species conservation status), "physical environment" (e.g., land use, temperature, precipitation, evapotranspiration, soil properties, air and soil pollution, habitat type, percentage of impervious surfaces, carbon dioxide [$CO_2$] concentrations), "morphological traits" (e.g., diameter at breast height, leaf area, shape, number of leaves, size), "physio-chemical traits" (e.g., leaf content [carbon and nitrogen], biomass, photosynthesis, stomatal conductance, thermal and shade tolerance, gas exchange measurements), and "reproductive biology" (e.g., phenology [flowering], flower morphology, pollination, reproductive success, seed dispersion, germination, and fruiting).

The responses of plant species and communities to urbanization were categorized according to the following Urban Drivers: biotic invasion, climate, human activities, land cover change, and pollution (Alberti et al., 2017; Aronson et al., 2016; Piana et al., 2019; Pickett et al., 2011). Biotic invasion represents the introduction of non-native species in the habitat. Climate refers to climate changes in urban areas, especially warming. Human activities are how humans directly act on urban biodiversity (e.g., through gardening practices). Land cover change refers to fragmentation and land conversion, and pollution is attributed to environmental contamination through soil and air pollutants. These responses were classified as negative (urban habitats negatively affect fitness), neutral (when species' and communities' responses do not differ between urban/rural habitats), or positive (species and communities are successful in the urban environment, maintaining the ecological balance). If a study found several responses, we considered each one individually according to the abovementioned criteria.

## 3. Results

The number of studies evaluating plant responses to urbanization has increased over the past ten years (Fig. 1). We found 171 studies that fit our selection criteria. Most of the studies were carried out in temperate biomes (113), with Temperate Broadleaf and Mixed Forests most represented with 102 studies (Fig. 2). Most studies took place in Europe and North America (60 and 43, respectively). Forty studies were conducted in Asia, 14 in South America, eight in Africa, and only seven in Oceania (Fig. 2).

Most of the studies (89) used the urban–non-urban gradient to describe how urbanization drives changes in species and communities. Forty-two studies used a simple comparison between urban and non-urban habitats, and eighteen used the temporal approach to explain how urbanization is changing species and communities over the years. In addition, 22 studies used combinations of these approaches. Most of the studies (103) were conducted with comparisons in different plant communities, and 68 studies analyzed specific species. Thus, analyses involving general community traits are among the most studied. Further details on the methodologies used in the studies to determine plants' responses to urbanization are found in Supplementary Table 1.

The most commonly cited Urban Drivers responsible for changes in plants in the reviewed literature (Fig. 3) were land cover change (129) and biotic invasion (56). Pollution was the least cited (Fig. 3). Overall, the studies identified varied responses in plants, often showing the same species or community demonstrating beneficial, neutral, or harmful responses, depending on which major areas were analyzed (Supplementary Table 2). However, most studies (133) reported negative responses (Supplementary Table 2). Table 1 summarizes how plants are responding to urbanization.

## 4. Discussion

### 4.1. Urbanization and urban drivers

The responses of plants to urbanization can be complex and varied (Table 1; Supplementary Table 2). More than one Urban Driver may be responsible for some community and plant traits changes, or several different changes may result from one driver. The

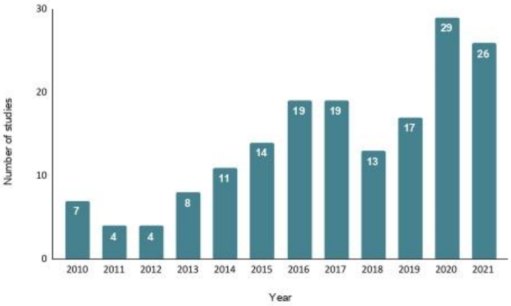


**Fig. 1.** Number of studies relating changes in plant species and communities in response to urbanization between 2010 and 2021.

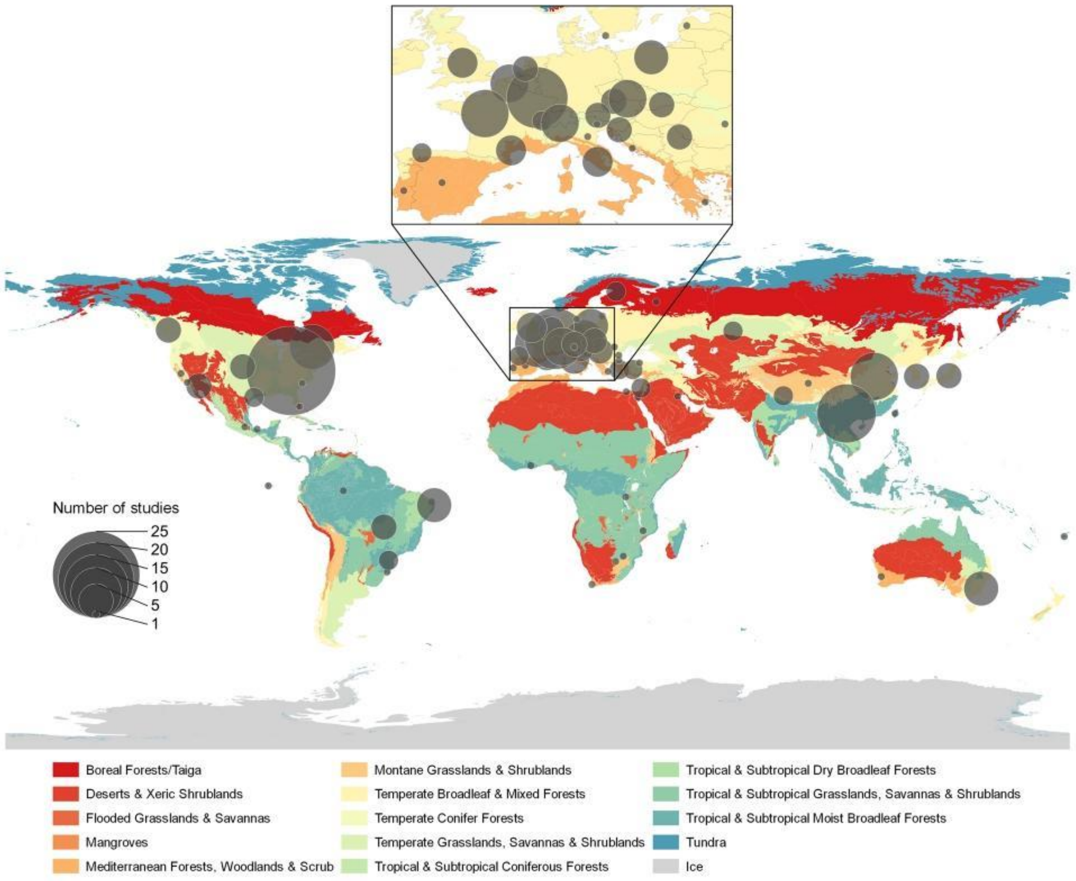


**Fig. 2.** Number of studies relating changes in plant species and communities in response to urbanization within their locations on biomes according to Dinerstein et al. (2017) (updated version of the Terrestrial Ecoregions of the World from Olson et al., 2001).

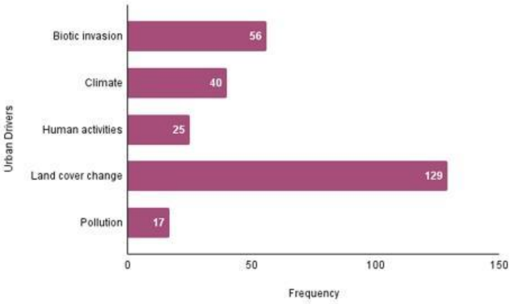


**Fig. 3.** The frequency at which Urban Drivers are held responsible for changes in plant species and communities based on the literature review from the last eleven years.

consequences are a network of responses connected to various drivers, making systematic cause-and-effect classification through frameworks in urban environments difficult (Aronson et al., 2016; Williams et al., 2015). We found that urbanization mainly negatively affects plants compared with neutral and positive responses, both in megacities and small cities. They result from a single driver or by combination among different Urban Drivers.

**Table 1**
Negative, neutral, and positive examples of responses of plants to six Urban Drivers. The data was compiled based on a literature review from the last eleven years of publications of the effects of urbanization on plant species and communities.

| Urban Drivers | Negative | Neutral | Positive |
|---|---|---|---|
| *Land cover change* | Lower plant cover; prevalence of abiotic pollination and dispersion; lower ability to reproduce sexually; reduction of dispersal ability between fragments; lower reproductive success; higher proportions of non-native, generalist, and disturbance-tolerant species | Species richness remains stable; reproduction modes did not change in response to urbanization | Urbanization of grasslands areas keep high nature conservation value through the favoring of insect-pollinated species; adaptations to attract more pollinators; reproduction modes changes in response to urbanization; biotic differentiation |
| *Climate* | Higher abundance of non-native species adapted to warmer conditions; phenological events start earlier; asynchrony between the phenology of plants and pollinators; decrease in species richness; changes in photosynthetic rates; lower water use efficiency | No changes in the timing of phenological events | Thriving of native thermophilous species; urban areas as a refuge for endangered species with affinities for hot and dry surfaces |
| *Pollution* | Decrease in species richness; delays in phenology; higher leaf nitrogen concentration; lower content of chlorophyll-a; associated arthropod biota affected through the reduction in palatability; higher weed coverage | Inconsistent changes in leaf morphology | More species with higher affinities to nutrient soil content |
| *Human activities* | Facilitation of the spread of non-native plant species by roads and railways; increase in ornamental non-native species in urban gardens, with lower functional richness; beneficial of human dispersed plants; higher abundance of species with lower conservation value; decrease in species richness | – | Human preferences for showy flowers may are good for pollinators in urban areas, favoring animal-pollinated plants |
| *Biotic Invasion* | Loss of species richness by the colonization of non-native species phylogenetically similar to the resident species; reduction in the diversity of native species; evolution of non-native species to compete with natives; pollinators preference for non-native species; species turnover; | Native species keeping fitness front invasion of non-native species | Increase of total plant species richness |

### *4.2. Urban Drivers and their effects on plant species and communities*

#### *4.2.1. Land cover change*

When it comes to urbanization, fragmentation due to land cover change is the central axis of biodiversity loss. A considerable amount of the reviewed literature described decreased species richness in plant communities related to land cover change (Abd El-Wahab, 2016; Albrecht and Haider, 2013; Anderson et al., 2020; de Araújo et al., 2021; Aronson et al., 2015; Bigirimana et al., 2011; Cochard et al., 2017; Deák et al., 2016; Dolan et al., 2011; Fekete et al., 2020; Glišić et al., 2021; Güler, 2020; Heneidy et al., 2021; Kalusová et al., 2019; Kolbe et al., 2016; Kummli et al., 2021; Lopez et al., 2018; Lowry et al., 2020; Melliger et al., 2018; Nero, 2019; Peng et al., 2019; Planchuelo et al., 2020a; Salgado et al., 2021; Silva-Junior et al., 2018; Tian et al., 2015; du Toit et al., 2020; Vakhlamova et al., 2014; Vallet et al., 2010a; Wang et al., 2020a; Wang et al., 2020b; Wu et al., 2013; Yan et al., 2019; Zhang et al., 2016) - Supplementary Table 2). The disturbance created by land cover changes, fragmenting landscapes from native vegetation to areas covered by impervious surfaces, homogenizes the physical environment. In consequence, the urbanization process creates similar environments that select identical traits and life histories in both native and non-native species, causing a reduction in species richness and shifts in the community's composition (species turnover) (Aronson et al., 2016; Brice et al., 2017, 2016; Dolan et al., 2017; Freitas et al., 2020; McKinney, 2006; Paquin et al., 2020).

Studies with functional-trait approaches have demonstrated specific features of plants selected by the urban environment leading to habitat homogenization since low functional-trait diversity in communities was detected in several cases (Anderson et al., 2020; Concepción et al., 2017; Melliger et al., 2018; Milanović et al., 2021; Nock et al., 2013; Ribeiro et al., 2019; Trentanovi et al., 2013). Some of the responses to habitat homogenization were a higher presence of generalist and disturbance-tolerant species (Everingham et al., 2019; Huang et al., 2013; Mccune and Vellend, 2013; Oliveira et al., 2020; Yang et al., 2021), ruderal species (Bigirimana et al., 2011; Wang et al., 2020a, 2020b; Zhang et al., 2020), and pioneers (Concepción et al., 2015; Nero, 2019; Pennington et al., 2010) as well as a decline in phylogenetic richness (Cui et al., 2019; El-Barougy et al., 2021; Knapp et al., 2017; Lopez et al., 2018; Silva-Junior et al., 2018). In these studies, we found a convergence of selected traits in urban habitats related to loss of animal-related characteristics (e.g., shifts to self- and wind-pollination and dispersal), fast growth rates (e.g., short stature and small seeds), long flowering periods, and demanding resource-use strategy (e.g., softer wood, larger leaves, lower leaf mass per area). Reductions in the number of functional groups of organisms decrease the efficiency by which whole communities capture biologically essential resources, which is acknowledged as critical for community stability and functioning (Cardinale et al., 2012). These findings raise that analyzing functional traits in plants is more potent in capturing species' response to environmental filtering driven by urbanization (Anderson et al., 2020; Dolan et al., 2017) beyond just tabulating changes in species richness.

At the same time, some studies revealed that native plant communities could be resilient to fragmentation, keeping their diversity (Fekete et al., 2020; Hahs and McDonnell, 2013; Landis and Leopold, 2014; Lechuga-Lago et al., 2017; Santos et al., 2016; Schwoertzig et al., 2016). The persistence of native species in urban areas shows us that we have alternatives to face the consequences of urbanization. The city's structures can provide favorable and heterogeneous environments for stress-adapted native species and even endangered species (Dallimer et al., 2012; Fekete et al., 2020; Heneidy et al., 2021; Paquin et al., 2020; Planchuelo et al., 2020b; Qian et al., 2020; Schmidt et al., 2014; Schmiedel et al., 2015; Sonti et al., 2021; Trammell et al., 2020). Investigating the functional traits and ecological strategies of native species adapted to fragmented environments can help establish conservation priorities, using local native adapted species for urban greening purposes. Also, forests and protected areas close to urban environments have been essential in maintaining and sending propagules of native species to fragmented areas (Blood et al., 2016; Davis et al., 2015; Gulezian and Nyberg, 2010).

Because of low habitat fragment connectivity in urban environments, insect-pollinated species can experience a lower quality of pollinator services in terms of decreased visitation rate and the lowered ability of pollinators to carry pollen between fragmented populations (Bartlewicz et al., 2015; Geslin et al., 2013), bringing different consequences for populations and communities. In most cases, both self-compatible and self-incompatible plants showed a decrease in reproductive success due to the reduction of the variety of pollinators, their low effectiveness, and low visit frequency, often reducing the seed set (Hou et al., 2019; Irwin et al., 2018; Leong et al., 2014; Lhotte et al., 2014; Oliveira et al., 2019). *Teucrium pseudochamaepitys*, an endangered species in Southern France, experienced low reproductive success, despite effective pollination in urban habitats (Lhotte et al., 2014). Lhotte et al. (2014) showed a lower diversity of pollinators, with *Apis mellifera* ensuring 97 % of pollination. They demonstrate that changes in land use threaten the short-term persistence of populations due to competition.

However, other studies suggested that pollinators may also be mobile between urban fragments (Filiz et al., 2015; Kobayashi et al., 2018; Van Rossum, 2010; Verboven et al., 2014). Verboven et al. (2014) analyzed the pollinator frequency in *Trifolium repens*, a species pollinated mainly by generalist bumblebees. Generalist species with large foraging ranges were previously found not to be strongly affected by urban land use (Bates et al., 2011). This finding reinforces that plant species that are not supported by these generalist pollinators are not the most suitable species for conservation efforts in cities.

Species that do not depend on pollinators for reproduction can maintain their reproductive success when performing selfing under conditions of lower pollinator availability (Ushimaru et al., 2014). The ability to tolerate fragmentation with traits that allow reproduction by selfing has also been identified by analyzing genetic diversity in *Viola grypoceras* in Kyoto, Japan (Toma et al., 2015). This study found no significant difference in allelic richness between the urban fragmented and suburban forests.

Genetic studies can be valuable for describing how plant populations have responded to habitat fragmentation due to land cover change. Molecular markers described limited gene flow between urban and rural populations (Bartlewicz et al., 2015; Emel et al., 2021; Wodkiewicz and Gruszczyńska, 2014) detected through high genetic structure between populations. On the other hand, there can be a failure to detect a genetic response to habitat fragmentation due to insufficient time having elapsed since fragmentation, as seen by Delnevo et al. (2021). Despite the current intense fragmentation, they found similar levels of genetic diversity across populations and a weak spatial genetic structure in *Conospermum undulatum* in Western Australia, highlighting the importance of considering the historical habitat dynamics when investigating fragmentation consequences in long-lived plants (Delnevo et al., 2021). These authors related that the currently fragmented populations of *C. undulatum* were part of a continuous population 70 years ago, resulting in the present weak genetic structure.

Knowing the reproductive system of a species and how populations respond to fragmentation through genetic studies is an effective way of evaluating the success of a species in the face of urbanization. Studying plant species with alternatives to biotic pollination and species pollinated by animals that can move between fragments is essential to establish which species are suitable for creating green areas. Above that, maintaining protected forest areas close to cities would be a more effective way to maintain more specific plant-pollinator relationships, maximizing species richness and ecosystem stability.

### *4.2.2. Climate*

Fragmentation only is often insufficient to explain plant species' distributions in urban landscapes (Williams et al., 2009). Fragmentation in urban habitats includes the introduction of materials that are not biologically active covering the soils in urban habitats, allowing additional drivers to be responsible, on a smaller or larger scale, for decreasing the number of species (Grimm et al., 2008; McKinney, 2002). These drivers include climate (which increases the phenomenon of the Urban Heat Island [UHI]) and soil and air pollution (Pickett et al., 2011). These particularities associated with urban areas significantly impact species in anthropogenic habitats (Williams et al., 2009). Indeed, increased temperatures in cities were related to reduced species richness by filtering out species not adapted to warmer habitats (Albrecht and Haider, 2013; Chen et al., 2014; Concepción et al., 2017; Yu et al., 2021). For example, Chen et al. (2014) demonstrated that the increase in mean annual temperature over the past half-century in Harbin, China, caused a decrease in temperate species and increased in tropical species over the years.

One of the best bioindicators for anthropogenic climate change is phenology. Studies on urban phenology helped to detect UHIs. There is a strong relationship between changes in the timing of natural events, such as leaf unfolding, flowering, fruit ripening, or leaf coloring and fall and changes in regional temperatures (Jochner and Menzel, 2015). Studies at a spatial scale detected advanced and more extended phenological events (Cochard et al., 2019; Davis et al., 2016; Fisogni et al., 2020; Kondratyeva et al., 2020; Massetti et al., 2015; Oliveira et al., 2019; Su et al., 2019; Yakub and Tiffin, 2017). Long-term studies that focused on more prolonged periods of

observations (Fitchett and Raik, 2021; Lee, 2017; Lindh et al., 2018; Neil et al., 2010; van Vliet et al., 2014; Xingyuan et al., 2016), as well as those that combined both temporal and spatial analyses also detected earlier phenology events (Comber and Brunsdon, 2015; Jeong et al., 2011; Li et al., 2017; Qiu et al., 2017). Analyzes involving a large time scale associated with data between urban and non-urban populations can be useful for comparing the period in which there were changes in phenology and the events of expansion of urbanization in cities.

In contrast, some studies did not find significant changes in phenology in urban gradients (Jochner et al., 2013, 2012; Zipper et al., 2016). The authors suggest that other specific characteristics of the regions where the cities are located, such as altitude, humidity and vegetation type, can influence climatic conditions, making it difficult to analyze and interpret urban phenology. Santangelo et al. (2020a) reported later phenology in *Trifolium repens*. However, they suggest that changes in the biotic community (e.g., insects) or other components of the abiotic environment (e.g., soil moisture) were responsible for this observed pattern. Other factors that interact with temperature changes can make it challenging to analyze results. Therefore, detailed investigations of the system studied are needed to fully understand the effect of urban environmental conditions on plant phenology (Jochner and Menzel, 2015).

Little is known regarding vegetation changes in phenology affecting plant-animal interactions. Changes in flowering phenology lead to a mismatch between the flowering period and pollinators' activity (Fisogni et al., 2020), leading to a possible reduction of pollination services in urban areas and a reduction in reproductive success. Plants may respond differently from pollinators to climatic conditions; therefore, it is suggested that phenological studies focusing on specific species should consider the behavior of pollinators and how they respond to climate change. On the other side, Davis et al. (2016) described increased flower productivity in *Angophora costata*, *Eucalyptus pilularis*, and *Corymbia gummifera* within urban landscapes due to more extended flowering periods, leading to a higher abundance of nectarivorous parrots in Sydney, Australia. More extended periods of phenological events in urban vegetation result in more food resources for associated animals in the form of nectar and pollen (Jochner and Menzel, 2015). This shows that cities can maintain species that have good productivity in warmer habitats, providing resources that attract local animals and increasing the productivity of the habitat.

In addition to higher temperatures attributable to UHI, climate change can lead to increased carbon dioxide and ozone levels in urban areas and greater drought stress experienced by plants (Barradas and Esperon-Rodriguez, 2021; Calfapietra et al., 2015). Zipper et al. (2017) demonstrated that UHIs significantly increase evapotranspiration demand associated with variability in impervious cover, with consistent relationships from year to year in Madison, United States. Lahr et al., (2018, 2015) demonstrated direct negative effects of temperature and water availability on photosynthesis.

Conversely, Searle et al. (2012) did not find changes in the photosynthetic capacity of foliage in *Quercus rubra* driven by urbanization in New York, United States. Climate change is a constant threat to biodiversity, increasing species' vulnerability. Knowing the vulnerabilities of species can help decision-makers plan conservation actions that minimize the effects of global warming on plant productivity.

### *4.2.3. Pollution*

High levels of soil and atmospheric pollution have potentially adverse impacts on the growth and development of plants (Rai, 2016). Different consequences were found when analyzing the effects of pollution in urban areas. Contaminated soils also have been described as the cause of reduced species richness and structural injuries (Bouraoui et al., 2019; Huang et al., 2013; King and Hovick, 2020). The high concentration of soil contaminants (Cadmium, Copper, and Zinc) and salts presented in the cited studies above are detrimental to plant development, causing a reduction in growth by accumulating in roots, stems, and leaves (Kabata-Pendias, 2004). Air pollution was associated with a decrease in the quality of seeds (Turmukhametova and Shadrina, 2020) and affected plant growth, seen through changes in the size and shape of the plant parts studied by Su et al. (2021) and Vujić et al. (2015). Jochner et al. (2015) found an association between delays in phenology and the presence of air pollutants. The authors indicate that this association might be attributable to other environmental conditions statistically correlated with pollution. Because they excluded the influence of air temperature in their analyses, other factors altered by pollution in urban areas, such as radiation and soil nutrients, may be related to the delay in phenology (Jochner et al., 2015).

Chlorophyll measurement is an important tool for evaluating the effect of air pollutants on plants, as it plays an essential role in plant metabolism. Reductions in chlorophyll concentration are usually associated with air pollution (Rai, 2016). Molnár et al. (2018) demonstrated lower chlorophyll-a content in *Celtis occidentalis* leaves, proving that this species is an effective indicator of anthropogenic emission of air pollutants. In the same line, Xiao et al. (2021) showed decreased photosynthetic rate and total chlorophyll contents in the canopies in Guangzhou, China. Still, herbs and shrubs responded with an increase to the same parameter (Xiao et al., 2021).

Roadside plantations are expected to combat air pollution since trees can act as a sink for $CO_2$ by fixing carbon during photosynthesis (Singh, 2021). The deposition of particulate matter on the leaves interrupted stomatal leaf exchanges by.

reduced $CO_2$ assimilation rate (photosynthesis) and the exchange of water (transpiration) in *Alstonia scholaris*, *Grevillea robusta*, and *Mangifera indica* (Singh, 2021; Singh et al., 2020). However, these species presented modulations in physiological and biochemical processes for acclimatizing or adapting to the altered environment, contributing to future species selection to combat air pollution in cities (Singh et al., 2020).

Pollution was the least cited urban driver among the studies evaluated. The rapid growth of cities has dramatically increased emissions of pollutants with heavy metals in the soil and toxic gases in the air, decreasing the quality of habitat in urban centers. More studies need to be carried out using pollution as one of the parameters of comparison between communities and plant species, since pollution is very present in cities, especially those in developing countries.

### *4.2.4. Biotic invasion*

Apart from the higher similarity in native species composition among cities, seen through the reduction of species richness, studies described a spread of non-native species in urban habitats (Bigirimana et al., 2011; Blouin et al., 2019; Knapp et al., 2017; Lososová et al., 2016; Oliveira et al., 2020). This process of replacing localized native species with increasingly widespread non-native species promotes biotic homogenization, beyond habitat homogenization. Non-native species may enrich local biodiversity, but global diversity is decreased by the extinction of local species that are lost to the global species pool (McKinney, 2006).

The invasion of non-native species into new regions provokes researchers' concern because of perceived threats to agricultural productivity, native biodiversity and ecosystem functioning, and human aesthetics and wellbeing. Cadotte et al. (2017) observed that the alterations of the physical environment in urban habitats offer unique opportunities for these invasions. They create a unique combination of environmental conditions that do not exist in unmodified landscapes. Here, we found that land cover change is one of the most significant opportunities for the gain of non-native species, as seen through their increase in proximity of urban centers (Atasoy et al., 2018; Blood et al., 2016; Chimaimba et al., 2020; Dallimer et al., 2012; Fornal-Pieniak et al., 2020; Greene and Blossey, 2014; Kalusová et al., 2019; Kühn et al., 2017; Landis and Leopold, 2014; Malkinson et al., 2018; Paquin et al., 2020; Park et al., 2021; Pellegrini et al., 2021; Pennington et al., 2010; Schmiedel et al., 2015; Tew et al., 2021; Tordoni et al., 2017; Vakhlamova et al., 2014; Yan et al., 2019), and with the urbanization process over the years (Celesti-Grapow and Ricotta, 2021; Kopel et al., 2015; Wirth et al., 2020; Yu et al., 2021). Furthermore, climate change is crucial for establishing non-native thermophilic species favored by warm conditions typical of urban habitats (Géron et al., 2021; Schmidt et al., 2014; Sołtysiak, 2020). Similarly, soil modification by higher quantities of nutrients and pollutants was related to an increase of non-native species (Grella et al., 2018; Lopez et al., 2018).

Cadotte et al. (2017) also infer that cities are islands of reduced competition or offer an escape from natural enemies. Moreover, other authors describe higher competitive skills in non-native species depending on their inserted context (Fukano et al., 2020). Ruderal species that use the competitive ruderal (CR) strategy allocate most of their nutrient supply to growth and reproduction to better compete for space and achieve wide distribution (Chen et al., 2014). Disturbed surfaces support the establishment of non-native species because other species have low competitive ability (Deák et al., 2016). Nitrophilic species have a more remarkable competitive ability due to the high presence of nitrogen in urban soils and thus can exclude less nitrophilic species (King and Hovick, 2020). Zeeman et al. (2017) demonstrated a significant shift in the composition of urban grasslands towards taller plants within non-native species, which is correlated with competitive vigor. In a temporal analysis, Cui et al. (2019) suggest that similar ecological niches between native and invasive species lead to a decrease in the competitive power of native species due to the lack of natural enemies for invasives in the local urban ecosystem.

Furthermore, Cadotte et al. (2017) observe that high immigration rates of non-native species are facilitated by humans (see Section 4.2.5). For example, *Carpobrotus edulis*, a succulent plant native to South Africa, has been intentionally introduced globally as an ornamental to prevent soil erosion in coastal areas in Galicia, Spain. The higher presence of *C. edulis* in urban coastal dunes increases soil impacts, allowing for the establishment of the native but ruderal, *Scolymus hispanicus*, which is not adapted to the typically limiting conditions of coastal dunes (Lechuga-Lago et al., 2017).

The performance of these non-native species in the novel habitats will define their persistence in becoming invasive species. They need to deal with a suite of biotic interactions beyond just competition. In particular, they must interact with consumers, pathogens, and mutualists (Cadotte et al., 2017); therefore, we should pay attention to possible changes in the communities' dynamics in urban habitats. For example, in Florida, *Ficus microcarpa* trees were planted as ornamentals as early as 1912. Today, adults are seed sources for fig-eating birds that disperse this invasive species and make it difficult to control the species' abundance (Caughlin et al., 2012). Non-native trees in urban habitats shelter fewer invertebrates, being a possible stronger driver of invertebrate declines in urban areas in Sweden (Jensen et al., 2021). The invasive shrub *Cytisus scoparius* (Scotch broom) is under natural selection by pollinators for increased flower size, but only in urban populations, in Lacey-Olympia, United States. This natural selection by pollinators could result in an invading species' adaptive evolution into a new pollination niche (Bode and Tong, 2018).

We can see the biotic invasion as an urban driver by itself, but it is strongly influenced by the other drivers described. The adverse conditions provided by the urban habitat are very favorable for the spread of non-native species and are one of the greatest threats to local biodiversity. This makes biotic invasion one of the priorities for continuously monitoring populations of exotic species already present in cities and efforts to reduce the spread of new exotic species.

### *4.2.5. Human activities*

Human-assisted "intentional" migration is a major conduit for non-native species that become invasive, and the impact of invaders on native biodiversity remains an area of intense research (Van Kleunen et al., 2015). People move among cities and introduce organisms for a variety of reasons. During the European colonization of Burundi and Brazil, for example, many European and tropical ornamental species of different biomes from local species were introduced for landscaping purposes (Bigirimana et al., 2011; Oliveira et al., 2020). In the contemporary world, the use of species for horticultural interest (Dolan et al., 2011; Greene and Blossey, 2014) and ornamental and gardening practices (Dolan et al., 2017; dos Santos et al., 2020; Gaggini et al., 2017; Löki et al., 2019; Ranta and Viljanen, 2011) are other forms of human activities that introduce exotic species. Human activities facilitate dispersion by roads and railways (Concepción et al., 2016; Gaggini et al., 2017; Rivkin et al., 2021; Skultety and Matthews, 2017; Štajerová et al., 2017; Vakhlamova et al., 2016). Vehicles are an important vector for plant dispersal; thus, rails, roads, rivers, or canals can play as corridors for the spread of plants, resulting in high propagule pressure of exotic species (Ricotta et al., 2014).

Environmental education works involving the dissemination of knowledge of local species should be implemented to prevent the entry of new exotic ornamental species. In addition, encouraging the use of native species for ornamentation is important to restore the productivity of the local ecosystem, increasing resources for the animal community associated with plant species.

### 4.3. Does urbanization lead to phenotypic plasticity or adaptation in plants?

The drivers that impact fitness in plant communities can be challenging to discern. Urbanization includes environmental change along multiple interacting dimensions that can dramatically increase the complexity of environments to which populations must adapt. Understanding how traits may diverge with urbanization in response to Urban Drivers is necessary to understand the evolution of organisms in urban environments (Rivkin et al., 2019; Santangelo et al., 2022). Here, we have demonstrated that the number of studies describing how Urban Drivers are responsible for changes in plant species and communities has increased in recent years.

Species have evolved phenotypic traits in their historic ranges that might confer an ecological advantage allowing for colonization and establishment in urban environments (Rivkin et al., 2019). For example, five of the six studies with graminoid communities showed positive responses to urban environments. These studies confirmed that this group of plants might be highly adapted to this habitat type. Graminoids have assured insect pollination and can persist in disturbed soils by producing small, durable seeds (Albrecht and Haider, 2013; Beal-Neves et al., 2020; Cochard et al., 2017). Other traits, such as the capacity to grow taller individuals and increase vegetative reproduction, facilitate access to more resources and allow a species to spread (Cochard et al., 2019; Zeeman et al., 2017). Grassland ecosystems are evolutionarily related to disturbance regimes that maintain the high species richness of these systems under appropriate intensity and frequency (Overbeck et al., 2005). Another group that can take advantage of urbanization's new conditions, as Duffy and Chown (2016) observe, are the $C_4$ plants. Because $C_4$ plants are organisms adapted to warm environments, through the increase of photosynthetic activity, it is possible that native plants of this group will not be harmed by the global phenomenon of Urban Heat Islands.

Environmental changes induced by urbanization often select plants with specific traits that enhance plant persistence, competition, or resource retention (Ilyas et al., 2021; Murray-Stoker and Johnson, 2021; Song et al., 2019; Vallet et al., 2010b). These authors reported that plants in the urban forests presented an acquisitive strategy through a higher specific leaf area. Higher leaf nitrogen was also related to urban plants (Nikula et al., 2010; Song et al., 2019). Vasconcellos et al. (2019) demonstrated through dendrochronological characteristics of *Ceiba speciosa* growing in urban areas of Rio de Janeiro, Brazil, that this species is plastic and stress-tolerant under polluted conditions. Pollutant levels are lower during the rainy season, and microclimatic conditions may influence radial growth relationships. The authors showed that urban trees have a more immediate growth response to rain than rural ones.

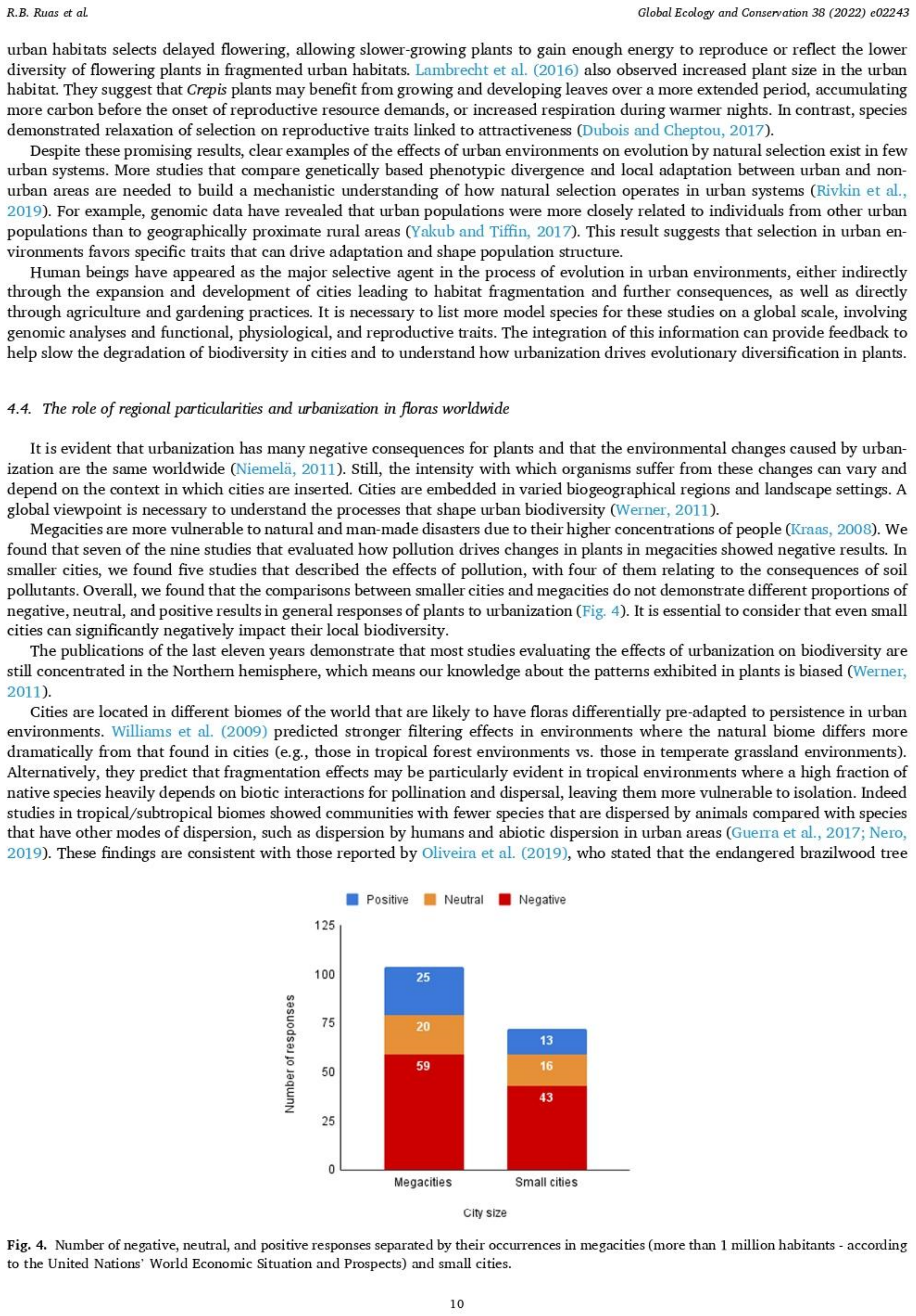
Little is known about the phenotypic changes induced by pollinator and mate limitations on reproductive traits in disturbed urban plant populations (Eckert et al., 2010). For example, *Commelina communis*, an annual species in the metropolitan area of Osaka-Kobe, Japan, showed increased reductions in traits promoting outcrossing under pollen-limited conditions induced by urbanization (Ushimaru et al., 2014). The authors argue that the observed floral diversity across the urbanization gradient may not be driven by urbanization itself but by urbanization-induced pollinator-limited conditions. Larger flowers were described by Santangelo et al. (2020a) in *Trifolium repens*, associated with near-complete turnover of pollinator morphological groups along the urbanization gradient in Toronto, Canada. Hou et al. (2019) described shifts in flower structures in *Gentiana dahurica*, which demonstrated plasticity in producing longer flowers as urbanization intensified. These shifts may have enhanced the floral display and pollinator attraction to ensure reproductive success. However, pollinator visits to the urban area were insufficient to ensure species' reproductive success. Irwin et al. (2018) showed that pollination was associated with the selection for larger floral display sizes in *Gelsemium sempervirens*. However, there is little difference in phenotypes between urban and non-urban sites. Together, these results provide important insights into the possible adaptive evolution of reproductive traits in response to pollinators' limitations in urban areas.

Rivkin et al. (2021) showed how urbanization has the potential to lead to novel selection pressures on ecological interactions by analyzing Darwin's finches and its essential food resource, *Tribulus cistoides*, in the Galápagos Islands. *T. cistoides* presents a mericarp defended by sharp spines with variations in their morphology. Rivkin et al. (2021) found that urbanization imposes phenotypic selection against small mericarps, potentially leading to the evolution of larger fruits.

A recent study has used the globally distributed plant white clover (*Trifolium repens*) as a model for examining evolutionary responses to urbanization (Santangelo et al., 2020b). This species exhibits a Mendelian polymorphism that produces hydrogen cyanide (HCN), an antiherbivore defense trait. This trait has been shown to exhibit adaptive evolution to winter temperature gradients (Santangelo et al., 2020b; Thompson et al., 2016), selecting against cyanogenesis due to reduced snow cover in cities. Thompson et al. (2016) showed no evidence that herbivory is likely to explain the observed urban-rural clines in cyanogenesis. Johnson et al. (2018) also demonstrated that populations did not exhibit changes in genetic diversity with increasing urbanization, indicating that genetic drift is unlikely to explain urban-rural clines in HCN frequency. Analyzing changes in the frequency of genotypes that produce HCN and the frequency of the alleles at genes that epistatically interact to produce HCN, Santangelo et al. (2020b) demonstrated that urban environments often select for similar phenotypes. However, different selective agents and targets underlie the evolutionary response in different cities.

Another model species that has received attention with descriptions of different patterns of natural selection is *Crepis sancta*, a common ruderal winter annual in southern France. The species can produce both dispersing and non-dispersing seeds (Imbert, 1999). Dubois and Cheptou (2017) demonstrated that fragmented urban populations of *C. sancta* that occur in small patches surrounded by an unsuitable matrix, such as asphalt and concrete, produce a higher fraction of non-dispersing seeds. This selection pattern had been observed previously (Cheptou et al., 2008) and is consistent with the higher cost of dispersing seeds in more fragmented habitats. Through neutral microsatellite markers, Dubois and Cheptou (2017) accessed the levels of genetic diversity among populations and traits related to plant size. They observed that the substantial quantitative variation among populations for size was independent of fragmentation and urbanization, reflecting the patterns of neutral genetic variation or the role of other selective agents.

Another interesting trait that may be passing through natural selection in *C. sancta* is the pattern of flowering phenology. Delayed phenology was observed by Dubois and Cheptou (2017) and Lambrecht et al. (2016). Both studies suggest that natural selection in



urban habitats selects delayed flowering, allowing slower-growing plants to gain enough energy to reproduce or reflect the lower diversity of flowering plants in fragmented urban habitats. Lambrecht et al. (2016) also observed increased plant size in the urban habitat. They suggest that *Crepis* plants may benefit from growing and developing leaves over a more extended period, accumulating more carbon before the onset of reproductive resource demands, or increased respiration during warmer nights. In contrast, species demonstrated relaxation of selection on reproductive traits linked to attractiveness (Dubois and Cheptou, 2017).

Despite these promising results, clear examples of the effects of urban environments on evolution by natural selection exist in few urban systems. More studies that compare genetically based phenotypic divergence and local adaptation between urban and non-urban areas are needed to build a mechanistic understanding of how natural selection operates in urban systems (Rivkin et al., 2019). For example, genomic data have revealed that urban populations were more closely related to individuals from other urban populations than to geographically proximate rural areas (Yakub and Tiffin, 2017). This result suggests that selection in urban environments favors specific traits that can drive adaptation and shape population structure.

Human beings have appeared as the major selective agent in the process of evolution in urban environments, either indirectly through the expansion and development of cities leading to habitat fragmentation and further consequences, as well as directly through agriculture and gardening practices. It is necessary to list more model species for these studies on a global scale, involving genomic analyses and functional, physiological, and reproductive traits. The integration of this information can provide feedback to help slow the degradation of biodiversity in cities and to understand how urbanization drives evolutionary diversification in plants.

### *4.4. The role of regional particularities and urbanization in floras worldwide*

It is evident that urbanization has many negative consequences for plants and that the environmental changes caused by urbanization are the same worldwide (Niemelä, 2011). Still, the intensity with which organisms suffer from these changes can vary and depend on the context in which cities are inserted. Cities are embedded in varied biogeographical regions and landscape settings. A global viewpoint is necessary to understand the processes that shape urban biodiversity (Werner, 2011).

Megacities are more vulnerable to natural and man-made disasters due to their higher concentrations of people (Kraas, 2008). We found that seven of the nine studies that evaluated how pollution drives changes in plants in megacities showed negative results. In smaller cities, we found five studies that described the effects of pollution, with four of them relating to the consequences of soil pollutants. Overall, we found that the comparisons between smaller cities and megacities do not demonstrate different proportions of negative, neutral, and positive results in general responses of plants to urbanization (Fig. 4). It is essential to consider that even small cities can significantly negatively impact their local biodiversity.

The publications of the last eleven years demonstrate that most studies evaluating the effects of urbanization on biodiversity are still concentrated in the Northern hemisphere, which means our knowledge about the patterns exhibited in plants is biased (Werner, 2011).

Cities are located in different biomes of the world that are likely to have floras differentially pre-adapted to persistence in urban environments. Williams et al. (2009) predicted stronger filtering effects in environments where the natural biome differs more dramatically from that found in cities (e.g., those in tropical forest environments vs. those in temperate grassland environments). Alternatively, they predict that fragmentation effects may be particularly evident in tropical environments where a high fraction of native species heavily depends on biotic interactions for pollination and dispersal, leaving them more vulnerable to isolation. Indeed studies in tropical/subtropical biomes showed communities with fewer species that are dispersed by animals compared with species that have other modes of dispersion, such as dispersion by humans and abiotic dispersion in urban areas (Guerra et al., 2017; Nero, 2019). These findings are consistent with those reported by Oliveira et al. (2019), who stated that the endangered brazilwood tree

**Fig. 4.** Number of negative, neutral, and positive responses separated by their occurrences in megacities (more than 1 million habitants - according to the United Nations' World Economic Situation and Prospects) and small cities.

(*Paubrasilia echinata*) demonstrated a reduction in reproductive success in the urban ecosystem in Recife, Brazil, indicating a depletion in the pollination process. Oliveira et al. (2019) also observed changes in the visiting behavior of an effective pollinator that behaved like an occasional pollinator in urban habitats. In contrast, Lobo et al. (2016) found a remarkable diversity and abundance of bee species in the inflorescences of *Tabebuia aurea* in the city of Brasília, Brazil. These authors attributed this result to the conservation of large parks and green zones in Brasília as part of the territorial planning of this recently founded (1960) city. Further studies in cities located in tropical and subtropical biomes are still needed to understand which Urban Drivers may be most important in the urban community assembly in these locations. Also, it is essential to consider the process of forming cities as a factor that will differentiate the consequences of urbanization on local biodiversity.

## 5. Conclusion

Research on how plants have responded to urbanization has increased in recent years, but many gaps remain to be filled. How each Urban Driver acts separately continues to be challenging to disentangle. This article contributes to the documentation of recent findings on the changes driven by urbanization at two ecological levels: plant species and communities. We found generally negative responses at both levels and a few cases of acclimatization and promising results demonstrating the potential adaptation of species to anthropogenic disturbances. The main weakness in this literature review was the lack of studies on biomes other than temperate, which results in limited knowledge of the patterns exhibited in plant species and communities. Another interesting finding was that even small cities could suffer from megacities' same consequences on their biodiversity. This reinforces the importance of conducting studies on a broad geographic scale and sampling from different cities to predict different forces among Urban Drivers in a variety of urban contexts. More studies are needed to elucidate whether there are different outcomes from the urban flora in other biomes.

It is increasingly necessary to identify which factors put biodiversity at risk, applying the methodologies needed in each area of knowledge to explore how this process can be more fully mitigated. Urban environments are of intrinsic ecological interest, and cities are open laboratories where evolutionary changes occur in real-time. More studies that integrate different aspects of plant communities and species, such as reproductive, physiological, and genetic traits, and monitoring climatic conditions and pollution within cities are vital to establishing priorities for conservation. The establishment of more model species for these large-scale studies is a robust alternative to understand better the processes and patterns of plant responses to urbanization. More broadly, it is essential to outline strategies for continuous monitoring and documenting the progress of species in perpetuating themselves in this new type of environment.

## Data Availability

Data will be made available on request.

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### *Declaration of competing interest*

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

## Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.gecco.2022.e02243.

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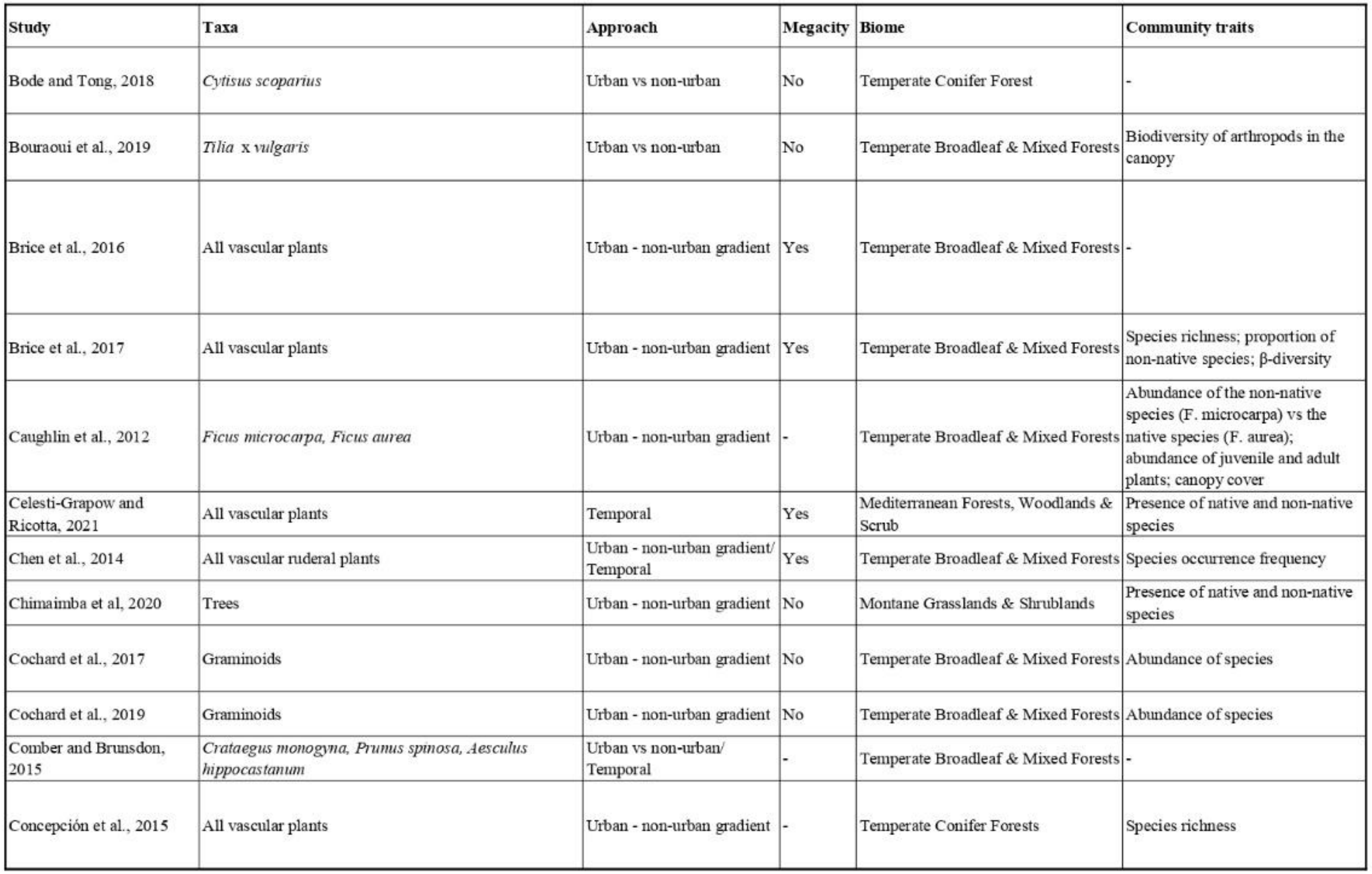
**Supplementary Table 1.** Methodologies used in the studies to determine plants' responses to urbanization.

| Study | Title | DOI | Continent | Country | City/Region |
|---|---|---|---|---|---|
| Abd El-Wahab, 2016 | Plant assemblage and diversity variation with human disturbances in coastal habitats of the western Arabian Gulf. | https://doi.org/10.1007/s40333-016-0084-x | Asia | Kuwait | Southern coastal area of Kuwait |
| Albrecht and Haider, 2013 | Species diversity and life history traits in calcareous grasslands vary along an urbanization gradient. | https://doi.org/10.1007/s10531-013-0437-0 | Europe | Germany | Munich |
| Anderson et al., 2020 | Post-apartheid ecologies in the City of Cape Town: An examination of plant functional traits in relation to urban gradients. | https://doi.org/10.1016/j.landurbplan.2019.103662 | Africa | South Africa | Cape Town |
| Aronson et al., 2015 | Urbanization promotes non-native woody species and diverse plant assemblages in the New York metropolitan region. | https://doi.org/10.1007/s11252-014-0382-z | North America | United States | New York |
| Atasoy et al., 2018 | Evaluating the distribution of invasive woody vegetation around riparian corridors in relation to land use. | https://doi.org/10.1007/s11252-017-0729-3 | North America | United States | Alabama |
| Barradas and Esperon-Rodriguez, 2021 | Ecophysiological Vulnerability to Climate Change in Mexico City's Urban Forest | https://doi.org/10.3389/fevo.2021.732250 | North America | Mexico | Mexico City |
| Bartlewicz et al., 2015 | Population genetic diversity of the clonal self-incompatible herbaceous plant Linaria vulgaris along an urbanization gradient. | https://doi.org/10.1111/bij.12602 | Europe | Belgium | Leuven |
| Beal-Neves et al., 2020 | The Influence of Urbanization and Fire Disturbance on Plant-floral Visitor Mutualistic Networks. | https://doi.org/10.3390/d12040141 | South America | Brazil | Porto Alegre |
| Bigirimana et al., 2011 | Alien plant species dominate the vegetation in a city of Sub-Saharan Africa. | https://doi.org/10.1016/j.landurbplan.2010.12.012 | Africa | Burundi | Bujumbura |
| Blood et al., 2016 | How Do Urban Forests Compare? Tree Diversity in Urban and Periurban Forests of the Southeastern US. | https://doi.org/10.3390/f7060120 | North America | United States | Southeastern USA |
| Blouin et al., 2019 | Increase in non-native species richness leads to biotic homogenization in vacant lots of a highly urbanized landscape. | https://doi.org/10.1007/s11252-019-00863-9 | North America | Canada | Montréal and Quebec |

| Study | Taxa | Approach | Megacity | Biome | Community traits |
|---|---|---|---|---|---|
| Abd El-Wahab, 2016 | All vascular plants | Urban - non-urban gradient | - | Deserts & Xeric Shrublands | Species richness and abundance; β-diversity; conservation status; canopy cover, height and diameters |
| Albrecht and Haider, 2013 | Graminoids | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | Total number of species; evenness; numbers of endangered and non-native species; canopy height |
| Anderson et al., 2020 | All vascular plants | Urban - non-urban gradient | Yes | Mediterranean Forests, Woodlands & Scrub | Species richness; cover contribution; total cover |
| Aronson et al., 2015 | Woody plants | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | Species richness; proportion of non-native species; β-diversity |
| Atasoy et al., 2018 | *Ligustrum sinense, Elaeagnus pungens, Triadica sebifera* | Urban - non-urban gradient | - | Temperate Broadleaf & Mixed Forests | Forest community form; cover by strata; canopy cover; species total cover; growth stage |
| Barradas and Esperon-Rodriguez, 2021 | *Acacia longifólia, Acer negundo, Alnus acuminata, Buddleja cordata, Celtis occidentalis, Dodonaea viscosa, Eucalyptus camaldulensis, Erythrina amerciana, Fraxinus uhdei, Ligustrum lucidum, Liquidambar styraciflua, Populus alba, Populus deltoides, Quercus rugosa, Ulmus parvifolia* | Urban - non-urban gradient | Yes | Tropical & Subtropical Coniferous Forests; Deserts & Xeric Shrublands | - |
| Bartlewicz et al., 2015 | *Linaria vulgaris* | Urban - non-urban gradient | No | Temperate Broadleaf & Mixed Forests | - |
| Beal-Neves et al., 2020 | Graminoids | Urban vs non-urban | Yes | Tropical & Subtropical Moist Broadleaf Forests | Londo's scale cover value |
| Bigirimana et al., 2011 | All vascular plants | Urban - non-urban gradient | No | Tropical & Subtropical Grasslands, Savannas & Shrublands | Species abundances; life forms; ecological groups; presence of non-native species |
| Blood et al., 2016 | Woody plants | Urban vs non-urban | - | Temperate Broadleaf & Mixed Forests; Temperate Grasslands, Savannas & Shrublands | Species richness; presence of non-native species |
| Blouin et al., 2019 | All vascular plants | Urban - non-urban gradient | Yes/ No | Temperate Broadleaf & Mixed Forests | Species richness; life form; presence of non-native species; β-diversity |

| Study | Physical environment | Morphological traits | Physio-chemical traits | Reproductive biology | Other traits |
|---|---|---|---|---|---|
| Abd El-Wahab, 2016 | - | - | - | - | - |
| Albrecht and Haider, 2013 | Soil N, P; Soil pH; temperature; soil and vegetation disturbance; proportion of sealed area; habitat size; habitat type | Specific leaf area | Phytomass; Leaf persistence | Start and duration of flowering; Pollination; Seed longevity, mass, shape, dispersal; vegetative dispersal | - |
| Anderson et al., 2020 | - | Leaf type; rooting depth | - | Pollination mechanism; seed dispersal mechanism | - |
| Aronson et al., 2015 | Proportion of urban land cover | - | - | - | - |
| Atasoy et al., 2018 | Surrounding land cover; visual indication of flooding | Diameter at breast height | - | - | - |
| Barradas and Esperon-Rodriguez, 2021 | Temperature; photosynthetically active radiation; vapor pressure deficit; water availability | - | Stomatal conductance, leaf water potential | - | - |
| Bartlewicz et al., 2015 | Percentage of impervious surfaces | - | - | Clonal diversity; seed set | DNA Molecular markers (SSR); genetic variation within and among populations |
| Beal-Neves et al., 2020 | Land cover classes; Time since fire | - | - | Frequency of floral visitors | - |
| Bigirimana et al., 2011 | Habitat type; degree of trampling by man and/or cattle; presence of shade; existence of a permanent source of humidity; soil type | - | - | - | - |
| Blood et al., 2016 | Land use | Height tree; crown width; crown light exposure; diameter at breast height | - | - | - |
| Blouin et al., 2019 | Local level of urbanization | Height | Light requirement; thermal tolerance; wetland occurence | Seed dispersal; pollination vectors; seed weight and lenght; vegetative reproduction | - |

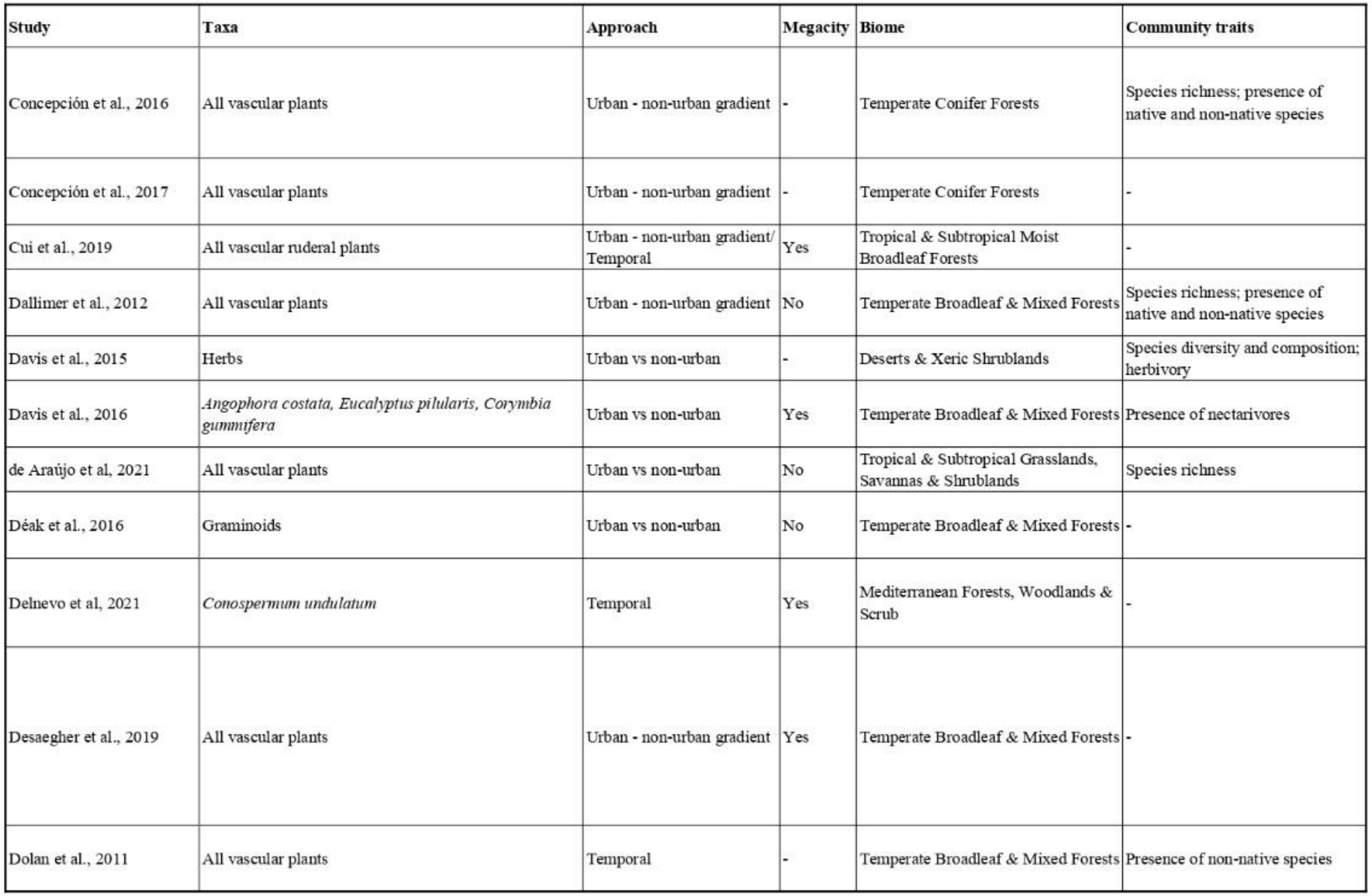
| Study | Title | DOI | Continent | Country | City/Region |
|---|---|---|---|---|---|
| Bode and Tong, 2018 | Pollinators exert positive selection on flower size on urban, but not on rural Scotch broom (Cytisus scoparius L. Link). | https://doi.org/10.1093/jpe/rtx024 | North America | United States | Lacey-Olympia |
| Bouraoui et al., 2019 | Deicing Salt Pollution Affects the Foliar Traits and Arthropods' Biodiversity of Lime Trees in Riga's Street Greeneries. | https://doi.org/10.3389/fevo.2019.00282 | Europe | Latvia | Riga |
| Brice et al., 2016 | Environmental filtering and spatial processes in urban riparian forests | https://doi.org/10.1111/jvs.12425 | North America | Canada | Montréal |
| Brice et al., 2017 | Does urbanization lead to taxonomic and functional homogenization in riparian forests? | https://doi.org/10.1111/ddi.12565 | North America | Canada | Montréal |
| Caughlin et al., 2012 | Urbanized landscapes favored by fig-eating birds increase invasive but not native juvenile strangler fig abundance. | https://doi.org/10.1890/11-1694.1 | North America | United States | South Florida |
| Celesti-Grapow and Ricotta, 2021 | Plant invasion as an emerging challenge for the conservation of heritage sites: the spread of ornamental trees on ancient monuments in Rome, Italy. | https://doi.org/10.1007/s10530-020-02429-9 | Europe | Italy | Rome |
| Chen et al., 2014 | Dynamics of ruderal species diversity under the rapid urbanization over the past half century in Harbin, Northeast China. | https://doi.org/10.1007/s11252-013-0338-8 | Asia | China | Harbin |
| Chimaimba et al, 2020 | Urban tree species composition and diversity in Zomba city, Malawi: Does land use type matter? | https://doi.org/10.1016/j.ufug.2020.126781 | Africa | Malawi | Zomba |
| Cochard et al., 2017 | Grassland plant species occurring in extensively managed road verges are filtered by urban environments | https://doi.org/10.1080/17550874.2017.1350764 | Europe | France | Nantes, Angers and La Roche-sur-Yon |
| Cochard et al., 2019 | Intraspecific trait variation in grassland plant communities along urban-rural gradients. | https://doi.org/10.1007/s11252-019-0827-5 | Europe | France | Nantes and Angers |
| Comber and Brunsdon, 2015 | A spatial analysis of plant phenophase changes and the impact of increases in urban land use. | https://doi.org/10.1002/joc.4030 | Europe | United Kingdom | - |
| Concepción et al., 2015 | Impacts of urbanisation on biodiversity: the role of species mobility, degree of specialisation and spatial scale | https://doi.org/10.1111/oik.02166 | Europe | Switzerland | Central Plateau of Switzerland |



| Study | Taxa | Approach | Megacity | Biome | Community traits |
|---|---|---|---|---|---|
| Bode and Tong, 2018 | *Cytisus scoparius* | Urban vs non-urban | No | Temperate Conifer Forest | - |
| Bouraoui et al., 2019 | *Tilia* x *vulgaris* | Urban vs non-urban | No | Temperate Broadleaf & Mixed Forests | Biodiversity of arthropods in the canopy |
| Brice et al., 2016 | All vascular plants | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | - |
| Brice et al., 2017 | All vascular plants | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | Species richness; proportion of non-native species; β-diversity |
| Caughlin et al., 2012 | *Ficus microcarpa, Ficus aurea* | Urban - non-urban gradient | - | Temperate Broadleaf & Mixed Forests | Abundance of the non-native species (F. microcarpa) vs the native species (F. aurea); abundance of juvenile and adult plants; canopy cover |
| Celesti-Grapow and Ricotta, 2021 | All vascular plants | Temporal | Yes | Mediterranean Forests, Woodlands & Scrub | Presence of native and non-native species |
| Chen et al., 2014 | All vascular ruderal plants | Urban - non-urban gradient/ Temporal | Yes | Temperate Broadleaf & Mixed Forests | Species occurrence frequency |
| Chimaimba et al, 2020 | Trees | Urban - non-urban gradient | No | Montane Grasslands & Shrublands | Presence of native and non-native species |
| Cochard et al., 2017 | Graminoids | Urban - non-urban gradient | No | Temperate Broadleaf & Mixed Forests | Abundance of species |
| Cochard et al., 2019 | Graminoids | Urban - non-urban gradient | No | Temperate Broadleaf & Mixed Forests | Abundance of species |
| Comber and Brunsdon, 2015 | *Crataegus monogyna, Prunus spinosa, Aesculus hippocastanum* | Urban vs non-urban/ Temporal | - | Temperate Broadleaf & Mixed Forests | - |
| Concepción et al., 2015 | All vascular plants | Urban - non-urban gradient | - | Temperate Conifer Forests | Species richness |

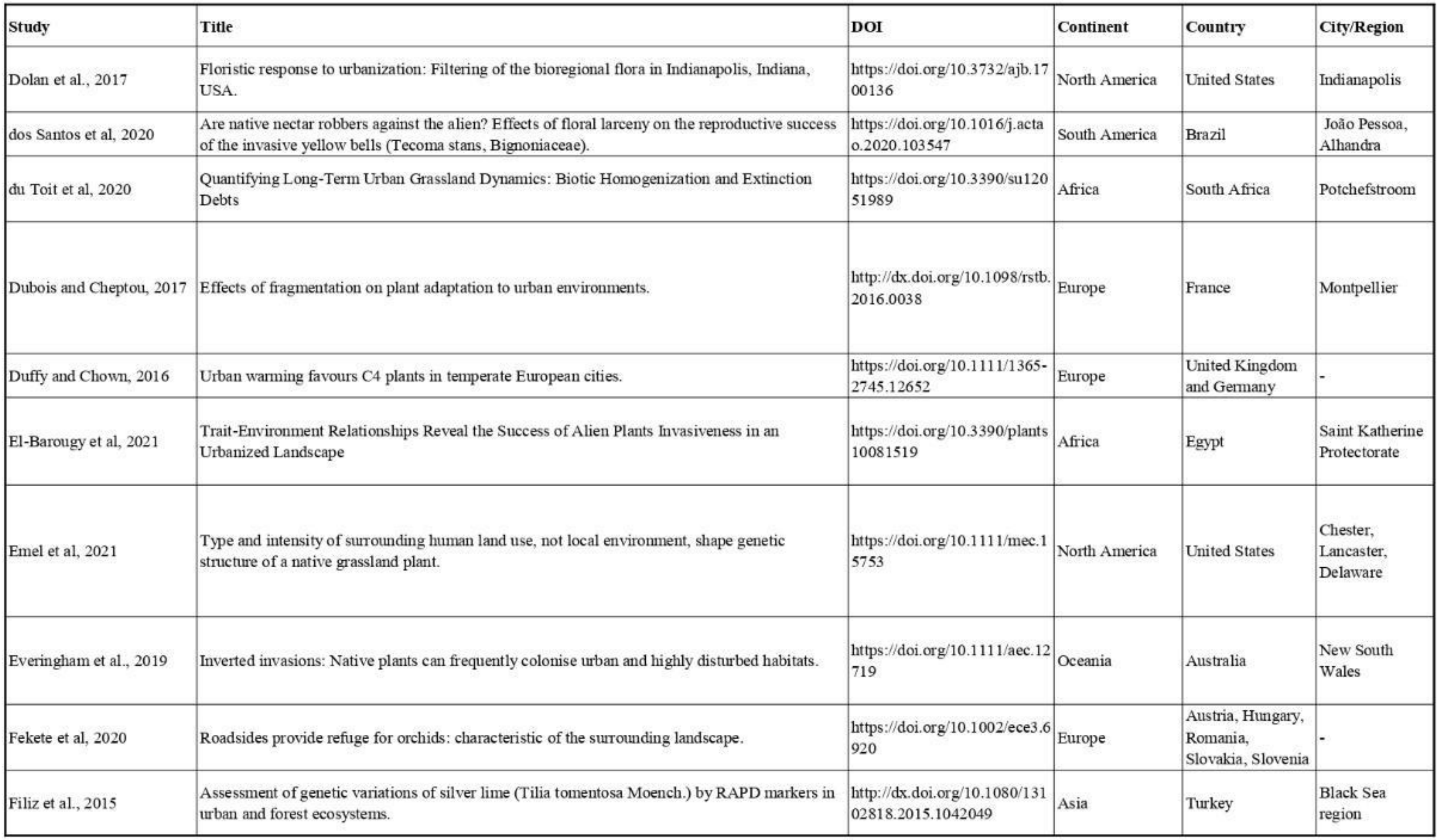
| Study | Physical environment | Morphological traits | Physio-chemical traits | Reproductive biology | Other traits |
|---|---|---|---|---|---|
| Bode and Tong, 2018 | - | - | - | Pollinators frequency; proportion of flowers tripped; flower size; pollinator selection on floral size | - |
| Bouraoui et al., 2019 | Soil properties | - | Salt accumulation in the foliage; chlorophyll fluorescence; structural injury in the foliage | - | - |
| Brice et al., 2016 | Forest area; seasonal flood intensity; time since the last major disturbance; historical agricultural and human land use; proportion of impervious surfaces; human influence index | Diameter at breast height; height | Shade tolerance; wetness index | Seed weight; seed bank; vegetative reproduction; dispersal vector | - |
| Brice et al., 2017 | - | Height | Shade tolerance; wetness index | Seed weight; seed bank; reproduction; diaspore morphology; seed buoyancy | - |
| Caughlin et al., 2012 | - | Diameter at breast height | - | Seed dispersors | - |
| Celesti-Grapow and Ricotta, 2021 | - | Plant form | - | Dispersal strategies | - |
| Chen et al., 2014 | Climate variation and land cover changes along the years | - | - | - | - |
| Chimaimba et al, 2020 | Land cover type | Diameter at breast height | - | - | - |
| Cochard et al., 2017 | Land use | - | - | Pollination syndrome; dispersal mode; speed of a falling diaspore; seed bank longevity; seed mass | - |
| Cochard et al., 2019 | Soil analysis; temperature | Specific leaf area; maximum height | - | Phenology (flowering) | - |
| Comber and Brunsdon, 2015 | Proportions of land use | - | - | Phenology (flowering) | - |
| Concepción et al., 2015 | Proportion of sealed area; climate; topography; variables related to other land-uses; landscape heterogeneity | - | Degree of specialisation | Dispersal modes | - |

| Study | Title | DOI | Continent | Country | City/Region |
|---|---|---|---|---|---|
| Concepción et al., 2016 | Impacts of urban sprawl on species richness of plants, butterflies, gastropods and birds: not only built-up area matters. | https://doi.org/10.1007/s11252-015-0474-4 | Europe | Switzerland | Central Plateau of Switzerland |
| Concepción et al., 2017 | Contrasting trait assembly patterns in plant and bird communities along environmental and human-induced land-use gradients. | https://doi.org/10.1111/ecog.02121 | Europe | Switzerland | Central Plateau of Switzerland |
| Cui et al., 2019 | Phylogenetic and functional structures of plant communities along a spatiotemporal urbanization gradient: Effects of colonization and extinction. | https://doi.org/10.1111/jvs.12724 | Asia | China | Shanghai/ Harbin |
| Dallimer et al., 2012 | Contrasting patterns in species richness of birds, butterflies and plants along riparian corridors in an urban landscape. | https://doi.org/10.1111/j.1472-4642.2012.00891.x | Europe | United Kingdom | Sheffield |
| Davis et al., 2015 | Top-down vs. bottom-up regulation of herbaceous primary production and composition in an arid, urbanizing ecosystem. | https://doi.org/10.1016/j.jaridenv.2015.01.018 | North America | United States | Central Arizona-Phoenix |
| Davis et al., 2016 | Do trees flower longer in the city? A comparison of flowering eucalyptus trees in streets, remnants and continuous forest and their association with nectarivorous birds. | https://doi.org/10.1007/s11252-015-0515-z | Oceania | Australia | Sydney |
| de Araújo et al, 2021 | Plant diversity drives responses of gall-inducing insects to anthropization in Neotropical savannas | https://doi.org/10.1007/s42965-021-00148-3 | South America | Brazil | Montes Claros |
| Déak et al., 2016 | Grassland vegetation in urban habitats - testing ecological theories. | https://doi.org/10.14471/2016.36.017 | Europe | Hungary | Debrecen |
| Delnevo et al, 2021 | Genetic and ecological consequences of recent habitat fragmentation in a narrow endemic plant species within an urban context | https://doi.org/10.1007/s10531-021-02256-x | Oceania | Australia | Western Australia, Perth |
| Desaegher et al., 2019 | How does urbanization affect the reproductive characteristics and ecological affinities of street plant communities? | https://doi.org/10.1002/ece3.5539 | Europe | France | Paris |
| Dolan et al., 2011 | Documenting effects of urbanization on flora using herbarium records. | https://doi.org/10.1111/j.1365-2745.2011.01820.x | North America | United States | Marion County |

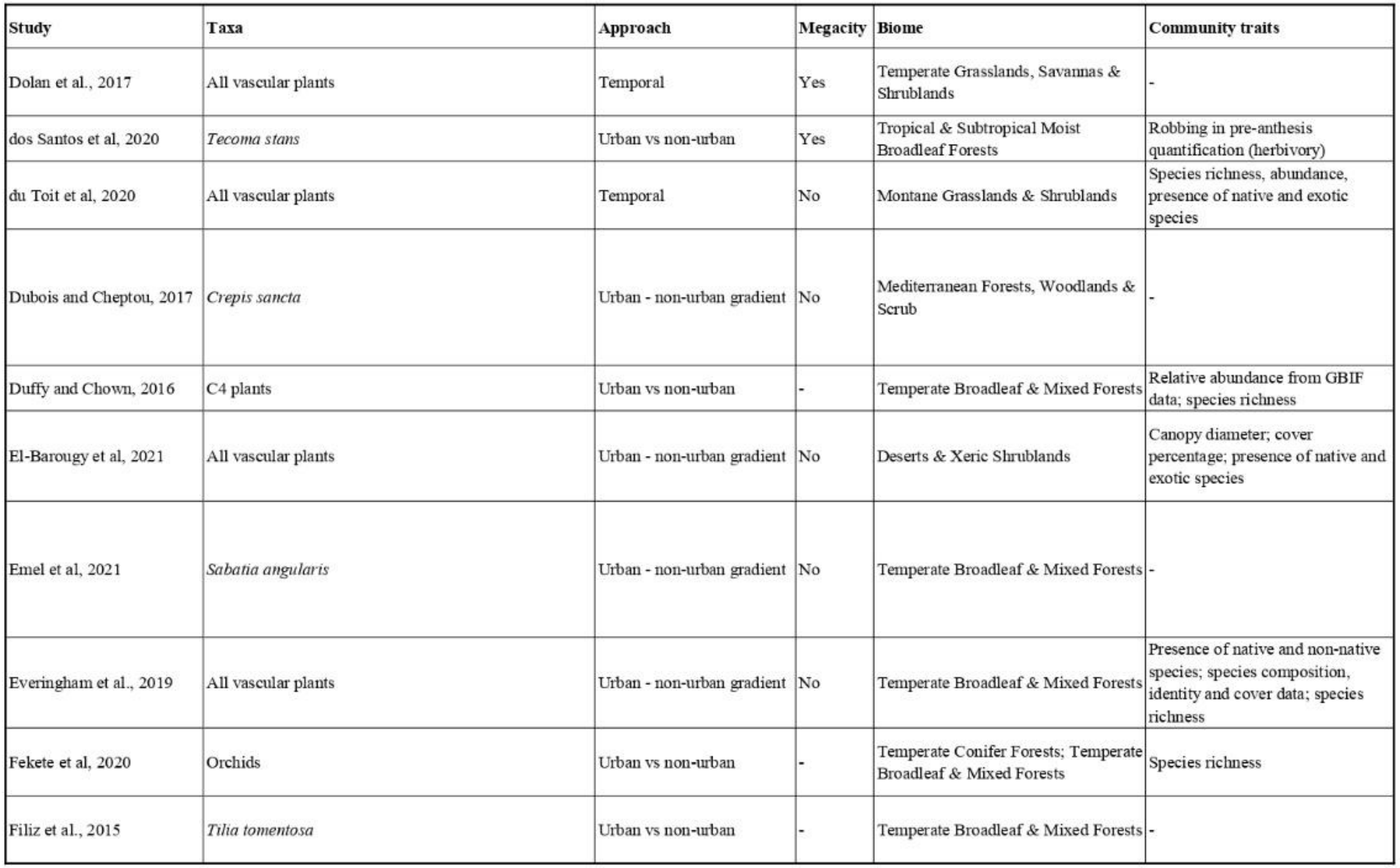


| Study | Taxa | Approach | Megacity | Biome | Community traits |
|---|---|---|---|---|---|
| Concepción et al., 2016 | All vascular plants | Urban - non-urban gradient | - | Temperate Conifer Forests | Species richness; presence of native and non-native species |
| Concepción et al., 2017 | All vascular plants | Urban - non-urban gradient | - | Temperate Conifer Forests | - |
| Cui et al., 2019 | All vascular ruderal plants | Urban - non-urban gradient/ Temporal | Yes | Tropical & Subtropical Moist Broadleaf Forests | - |
| Dallimer et al., 2012 | All vascular plants | Urban - non-urban gradient | No | Temperate Broadleaf & Mixed Forests | Species richness; presence of native and non-native species |
| Davis et al., 2015 | Herbs | Urban vs non-urban | - | Deserts & Xeric Shrublands | Species diversity and composition; herbivory |
| Davis et al., 2016 | *Angophora costata, Eucalyptus pilularis, Corymbia gummifera* | Urban vs non-urban | Yes | Temperate Broadleaf & Mixed Forests | Presence of nectarivores |
| de Araújo et al, 2021 | All vascular plants | Urban vs non-urban | No | Tropical & Subtropical Grasslands, Savannas & Shrublands | Species richness |
| Déak et al., 2016 | Graminoids | Urban vs non-urban | No | Temperate Broadleaf & Mixed Forests | - |
| Delnevo et al, 2021 | *Conospermum undulatum* | Temporal | Yes | Mediterranean Forests, Woodlands & Scrub | - |
| Desaegher et al., 2019 | All vascular plants | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | - |
| Dolan et al., 2011 | All vascular plants | Temporal | - | Temperate Broadleaf & Mixed Forests | Presence of non-native species |

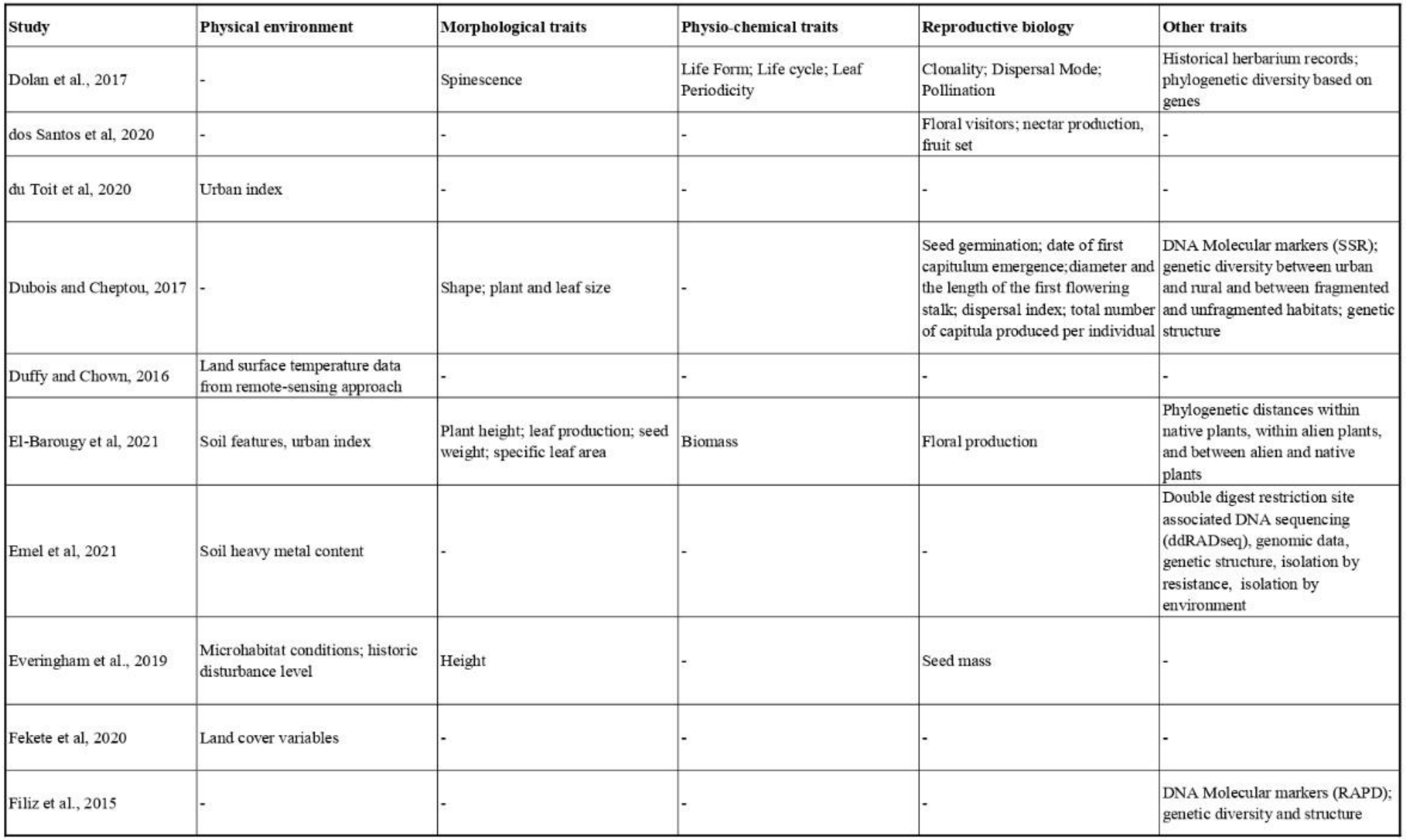
| Study | Physical environment | Morphological traits | Physio-chemical traits | Reproductive biology | Other traits |
|---|---|---|---|---|---|
| Concepción et al., 2016 | Urban area; average age of urban area; human population density; spatial dispersion of urban areas; climatic, topographic, and land use variables | - | Degree of specialisation; life cycle | Dispersal modes | - |
| Concepción et al., 2017 | Climate; resource availability; disturbance; land use; landscape heterogeneity; climate; topography | Height | Life history; Leaf Dry Matter Content | Dispersal modes; duration of the flowering period | - |
| Cui et al., 2019 | - | Height; vegetation form | Life cycle | Fruiting time | Phylogenetic diversity |
| Dallimer et al., 2012 | Land cover; topography; sealed surface; human population density; habitat diversity | - | - | - | - |
| Davis et al., 2015 | Soil inorganic nitrogen availability | - | Biomass | - | - |
| Davis et al., 2016 | - | Diameter at breast height | - | Phenology (flowering period); number of flowers produced per tree | - |
| de Araújo et al, 2021 | - | - | - | - | - |
| Déak et al., 2016 | Ellenberg ecological indicator values for temperature, water, and nitrogen | - | Behaviour type | - | - |
| Delnevo et al, 2021 | - | - | - | - | DNA Molecular markers (SSR); genetic diversity; genetic structure; spatial analysis of genetic differentiation |
| Desaegher et al., 2019 | Land cover; proportion of impervious surfaces; Ellenberg indicator values for light, temperature, continentality, atmospheric moisture, soil moisture, soil reaction, soil nutrient content, and soil salt content | - | Affinity indices for soil granulometry and for soil organic matter | Pollinator morphological typology; floral morphotype; level of entomophily, dispersal vector, beginning and end of flowering; vegetative reproduction; self-compatibility; autogamy; presence or absence of tubular corolla and nectar spur | - |
| Dolan et al., 2011 | Wetland classification | - | Physiognomic group; species' fidelity to high-quality habitats and tolerance of disturbance | - | Historical herbarium records |



| Study | Title | DOI | Continent | Country | City/Region |
|---|---|---|---|---|---|
| Dolan et al., 2017 | Floristic response to urbanization: Filtering of the bioregional flora in Indianapolis, Indiana, USA. | https://doi.org/10.3732/ajb.1700136 | North America | United States | Indianapolis |
| dos Santos et al, 2020 | Are native nectar robbers against the alien? Effects of floral larceny on the reproductive success of the invasive yellow bells (Tecoma stans, Bignoniaceae). | https://doi.org/10.1016/j.actao.2020.103547 | South America | Brazil | João Pessoa, Alhandra |
| du Toit et al, 2020 | Quantifying Long-Term Urban Grassland Dynamics: Biotic Homogenization and Extinction Debts | https://doi.org/10.3390/su12051989 | Africa | South Africa | Potchefstroom |
| Dubois and Cheptou, 2017 | Effects of fragmentation on plant adaptation to urban environments. | http://dx.doi.org/10.1098/rstb.2016.0038 | Europe | France | Montpellier |
| Duffy and Chown, 2016 | Urban warming favours C4 plants in temperate European cities. | https://doi.org/10.1111/1365-2745.12652 | Europe | United Kingdom and Germany | - |
| El-Barougy et al, 2021 | Trait-Environment Relationships Reveal the Success of Alien Plants Invasiveness in an Urbanized Landscape | https://doi.org/10.3390/plants10081519 | Africa | Egypt | Saint Katherine Protectorate |
| Emel et al, 2021 | Type and intensity of surrounding human land use, not local environment, shape genetic structure of a native grassland plant. | https://doi.org/10.1111/mec.15753 | North America | United States | Chester, Lancaster, Delaware |
| Everingham et al., 2019 | Inverted invasions: Native plants can frequently colonise urban and highly disturbed habitats. | https://doi.org/10.1111/aec.12719 | Oceania | Australia | New South Wales |
| Fekete et al, 2020 | Roadsides provide refuge for orchids: characteristic of the surrounding landscape. | https://doi.org/10.1002/ece3.6920 | Europe | Austria, Hungary, Romania, Slovakia, Slovenia | - |
| Filiz et al., 2015 | Assessment of genetic variations of silver lime (Tilia tomentosa Moench.) by RAPD markers in urban and forest ecosystems. | http://dx.doi.org/10.1080/13102818.2015.1042049 | Asia | Turkey | Black Sea region |

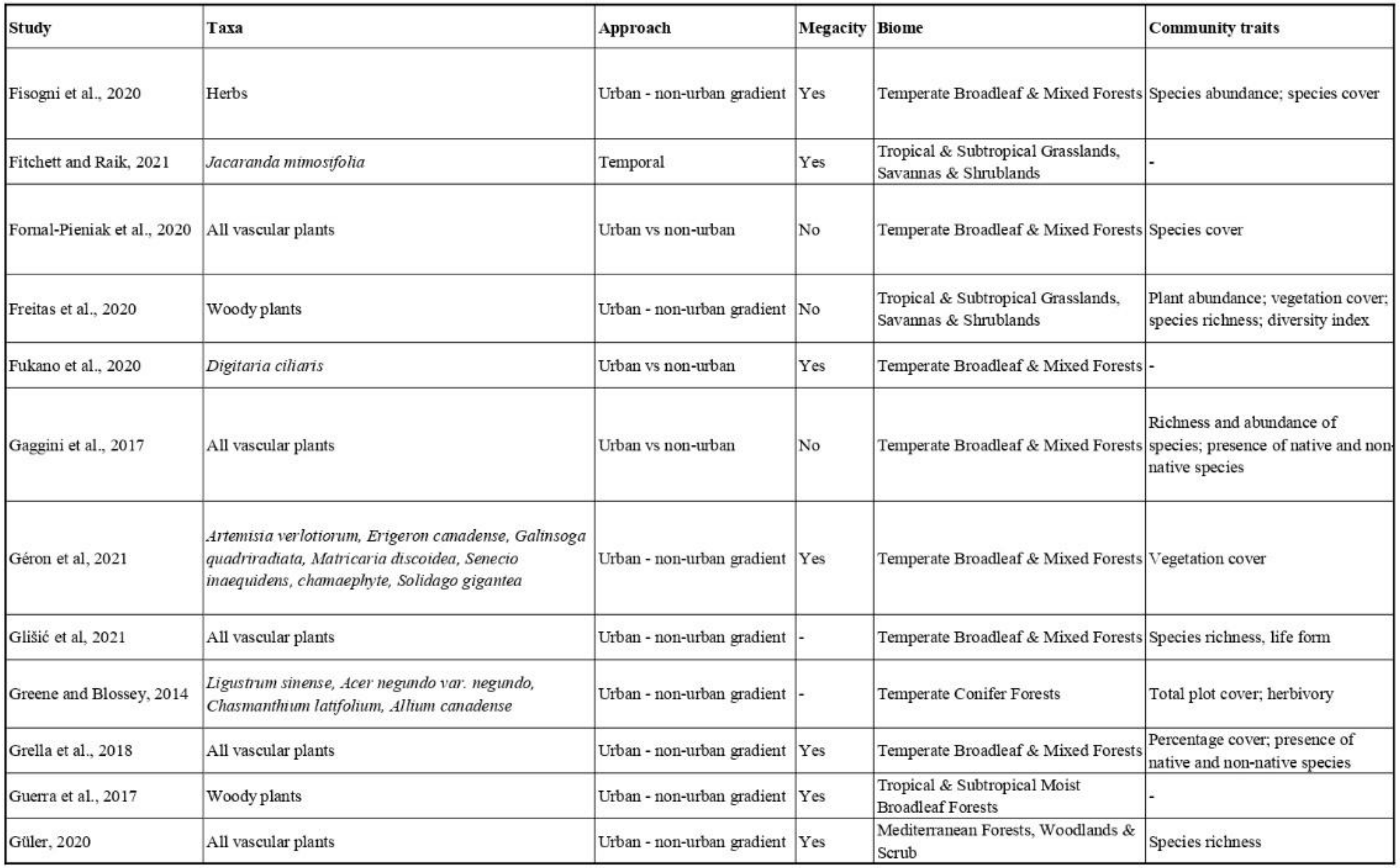


| Study | Taxa | Approach | Megacity | Biome | Community traits |
|---|---|---|---|---|---|
| Dolan et al., 2017 | All vascular plants | Temporal | Yes | Temperate Grasslands, Savannas & Shrublands | - |
| dos Santos et al, 2020 | *Tecoma stans* | Urban vs non-urban | Yes | Tropical & Subtropical Moist Broadleaf Forests | Robbing in pre-anthesis quantification (herbivory) |
| du Toit et al, 2020 | All vascular plants | Temporal | No | Montane Grasslands & Shrublands | Species richness, abundance, presence of native and exotic species |
| Dubois and Cheptou, 2017 | *Crepis sancta* | Urban - non-urban gradient | No | Mediterranean Forests, Woodlands & Scrub | - |
| Duffy and Chown, 2016 | C4 plants | Urban vs non-urban | - | Temperate Broadleaf & Mixed Forests | Relative abundance from GBIF data; species richness |
| El-Barougy et al, 2021 | All vascular plants | Urban - non-urban gradient | No | Deserts & Xeric Shrublands | Canopy diameter; cover percentage; presence of native and exotic species |
| Emel et al, 2021 | *Sabatia angularis* | Urban - non-urban gradient | No | Temperate Broadleaf & Mixed Forests | - |
| Everingham et al., 2019 | All vascular plants | Urban - non-urban gradient | No | Temperate Broadleaf & Mixed Forests | Presence of native and non-native species; species composition, identity and cover data; species richness |
| Fekete et al, 2020 | Orchids | Urban vs non-urban | - | Temperate Conifer Forests; Temperate Broadleaf & Mixed Forests | Species richness |
| Filiz et al., 2015 | *Tilia tomentosa* | Urban vs non-urban | - | Temperate Broadleaf & Mixed Forests | - |



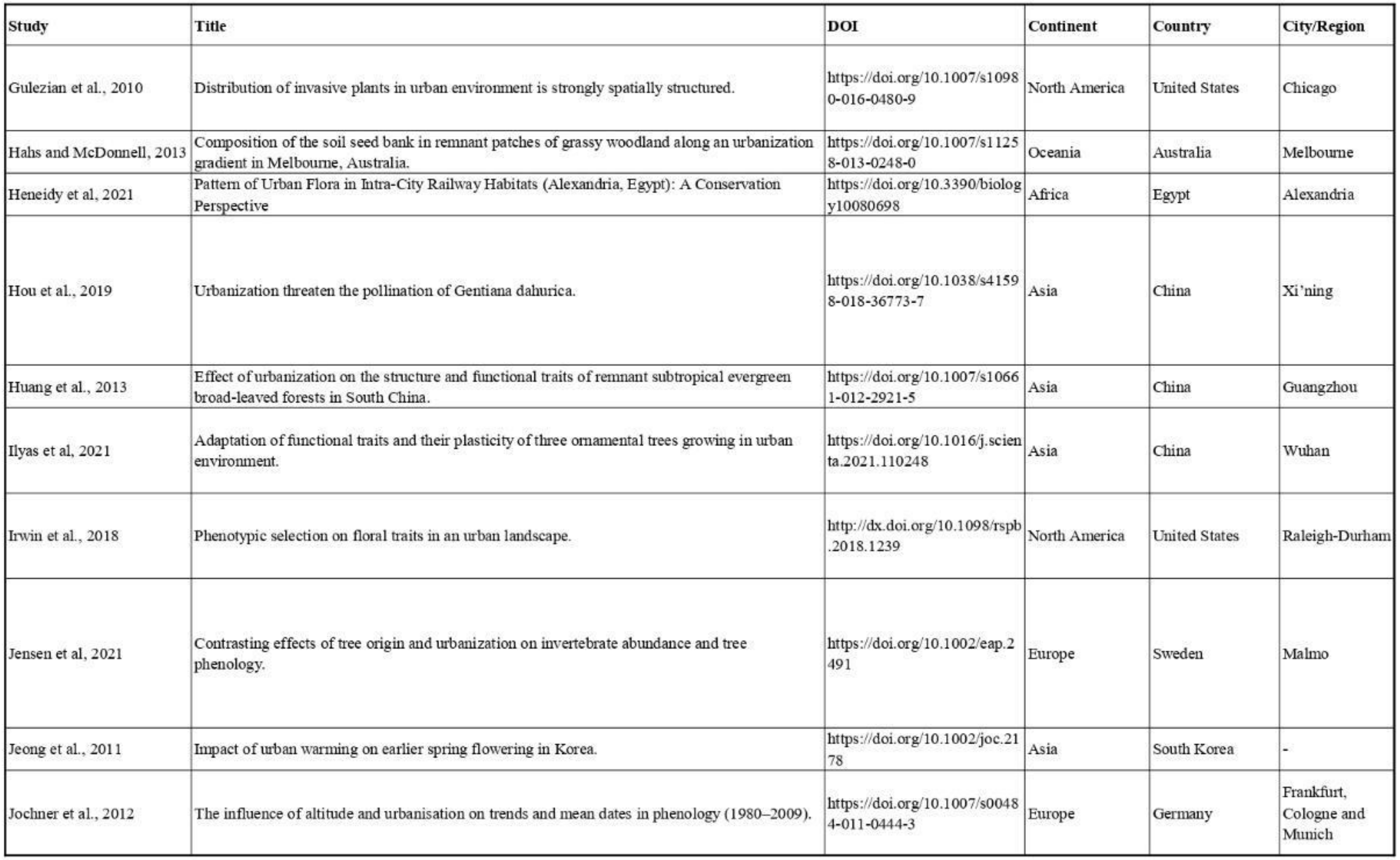
| Study | Physical environment | Morphological traits | Physio-chemical traits | Reproductive biology | Other traits |
|---|---|---|---|---|---|
| Dolan et al., 2017 | - | Spinescence | Life Form; Life cycle; Leaf Periodicity | Clonality; Dispersal Mode; Pollination | Historical herbarium records; phylogenetic diversity based on genes |
| dos Santos et al, 2020 | - | - | - | Floral visitors; nectar production, fruit set | - |
| du Toit et al, 2020 | Urban index | - | - | - | - |
| Dubois and Cheptou, 2017 | - | Shape; plant and leaf size | - | Seed germination; date of first capitulum emergence;diameter and the length of the first flowering stalk; dispersal index; total number of capitula produced per individual | DNA Molecular markers (SSR); genetic diversity between urban and rural and between fragmented and unfragmented habitats; genetic structure |
| Duffy and Chown, 2016 | Land surface temperature data from remote-sensing approach | - | - | - | - |
| El-Barougy et al, 2021 | Soil features, urban index | Plant height; leaf production; seed weight; specific leaf area | Biomass | Floral production | Phylogenetic distances within native plants, within alien plants, and between alien and native plants |
| Emel et al, 2021 | Soil heavy metal content | - | - | - | Double digest restriction site associated DNA sequencing (ddRADseq), genomic data, genetic structure, isolation by resistance, isolation by environment |
| Everingham et al., 2019 | Microhabitat conditions; historic disturbance level | Height | - | Seed mass | - |
| Fekete et al, 2020 | Land cover variables | - | - | - | - |
| Filiz et al., 2015 | - | - | - | - | DNA Molecular markers (RAPD); genetic diversity and structure |

| Study | Title | DOI | Continent | Country | City/Region |
|---|---|---|---|---|---|
| Fisogni et al., 2020 | Urbanization drives an early spring for plants but not for pollinators | https://doi.org/10.1111/oik.07274 | Europe | France | Lille |
| Fitchett and Raik, 2021 | Phenological advance of blossoming over the past century in one of the world's largest urban forests, Gauteng City-Region, South Africa. | https://doi.org/10.1016/j.ufug.2021.127238 | Africa | South Africa | Gauteng City-Region |
| Fornal-Pieniak et al., 2020 | Vascular flora of urban forests in a medium-sized city in Poland: comparison with nature reserves in the city's surrounding. | https://doi.org/10.1080/01426397.2020.1850662 | Europe | Poland | Tarnów |
| Freitas et al., 2020 | Urbanization alters the composition, but not the diversity and structure, of Neotropical savanna woody plant communities. | https://doi.org/10.1007/s12224-020-09366-4 | South America | Brazil | Montes Claros |
| Fukano et al., 2020 | Contemporary adaptive divergence of plant competitive traits in urban and rural populations and its implication for weed management. | https://doi.org/10.1111/1365-2745.13472 | Asia | Japan | Tokyo |
| Gaggini et al., 2017 | Settlements as a source for the spread of non-native plants into Central European suburban forests. | http://dx.doi.org/10.1016/j.actao.2016.12.008 | Europe | Switzerland | Basel/ Lugano |
| Géron et al, 2021 | Shielded environments reduce stress in alien Asteraceae species during hot and dry summers along urban-to-rural gradients | https://doi.org/10.1002/ece3.7872 | Europe | Belgium | Flanders, Brussels Capital Region, and the north of Wallonia |
| Glišić et al, 2021 | Influence of Habitat Types on Diversity and Species Composition of Urban Flora—A Case Study in Serbia | https://doi.org/10.3390/plants10122572 | Europe | Serbia | - |
| Greene and Blossey, 2014 | Patterns of Privet: Urbanizing Watersheds, Invasive Ligustrum sinense, and Performance of Native Plant Species in Piedmont Floodplain Forests. | https://doi.org/10.1007/s10021-014-9774-4 | North America | United States | Piedmont ecoregion |
| Grella et al., 2018 | Invasive weeds in urban riparian zones: the influence of catchment imperviousness and soil chemistry across an urbanization gradient. | https://doi.org/10.1007/s11252-018-0736-z | Oceania | Australia | Sydney |
| Guerra et al., 2017 | Urban or rural areas: which types of surrounding land use induce stronger edge effects on the functional traits of tropical forests plants? | https://doi.org/10.1111/avsc.12315 | South America | Brazil | Recife |
| Güler, 2020 | Plant species diversity and vegetation in urban grasslands depending on disturbance levels. | https://doi.org/10.2478/s11756-020-00484-0 | Asia | Turkey | Izmir |

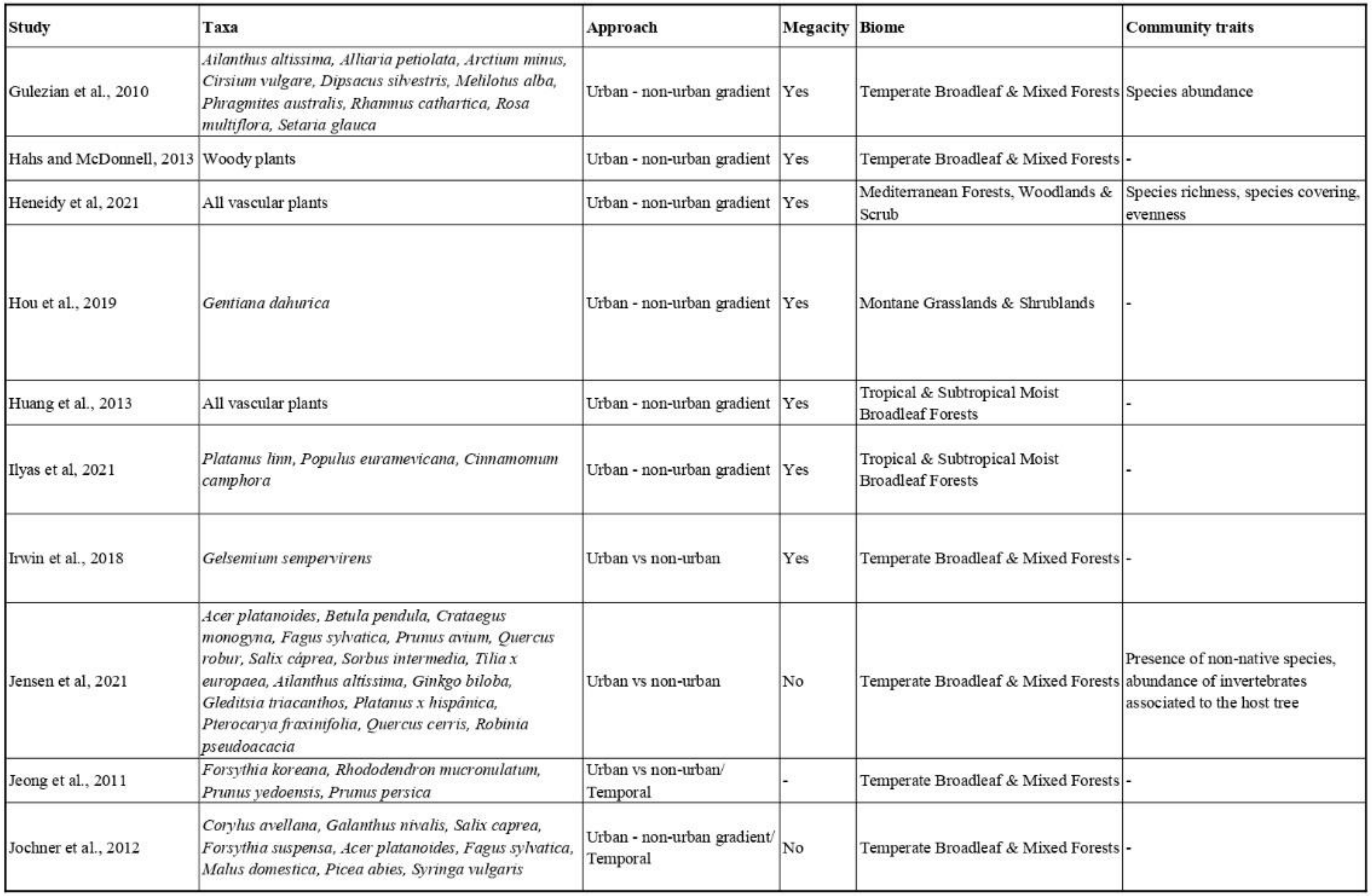


| Study | Taxa | Approach | Megacity | Biome | Community traits |
|---|---|---|---|---|---|
| Fisogni et al., 2020 | Herbs | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | Species abundance; species cover |
| Fitchett and Raik, 2021 | *Jacaranda mimosifolia* | Temporal | Yes | Tropical & Subtropical Grasslands, Savannas & Shrublands | - |
| Fornal-Pieniak et al., 2020 | All vascular plants | Urban vs non-urban | No | Temperate Broadleaf & Mixed Forests | Species cover |
| Freitas et al., 2020 | Woody plants | Urban - non-urban gradient | No | Tropical & Subtropical Grasslands, Savannas & Shrublands | Plant abundance; vegetation cover; species richness; diversity index |
| Fukano et al., 2020 | *Digitaria ciliaris* | Urban vs non-urban | Yes | Temperate Broadleaf & Mixed Forests | - |
| Gaggini et al., 2017 | All vascular plants | Urban vs non-urban | No | Temperate Broadleaf & Mixed Forests | Richness and abundance of species; presence of native and non-native species |
| Géron et al, 2021 | *Artemisia verlotiorum, Erigeron canadense, Galinsoga quadriradiata, Matricaria discoidea, Senecio inaequidens, chamaephyte, Solidago gigantea* | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | Vegetation cover |
| Glišić et al, 2021 | All vascular plants | Urban - non-urban gradient | - | Temperate Broadleaf & Mixed Forests | Species richness, life form |
| Greene and Blossey, 2014 | *Ligustrum sinense, Acer negundo var. negundo, Chasmanthium latifolium, Allium canadense* | Urban - non-urban gradient | - | Temperate Conifer Forests | Total plot cover; herbivory |
| Grella et al., 2018 | All vascular plants | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | Percentage cover; presence of native and non-native species |
| Guerra et al., 2017 | Woody plants | Urban - non-urban gradient | Yes | Tropical & Subtropical Moist Broadleaf Forests | - |
| Güler, 2020 | All vascular plants | Urban - non-urban gradient | Yes | Mediterranean Forests, Woodlands & Scrub | Species richness |

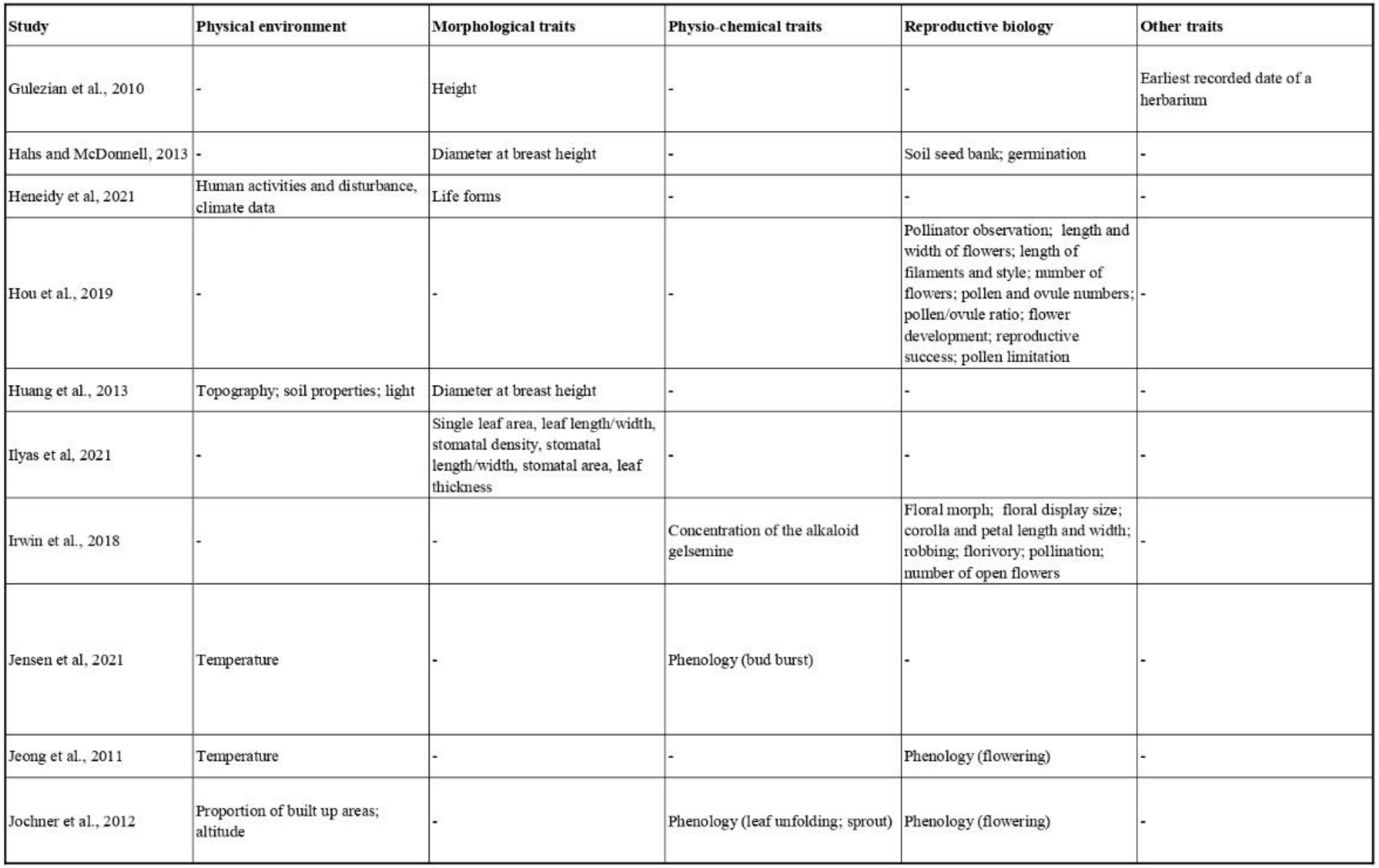
| Study | Physical environment | Morphological traits | Physio-chemical traits | Reproductive biology | Other traits |
|---|---|---|---|---|---|
| Fisogni et al., 2020 | Land cover | - | - | Phenology (flowering); flight period of wild pollinators; abundance and diversity of pollinators | - |
| Fitchett and Raik, 2021 | Temperature | - | - | Phenology (flowering) | - |
| Fornal-Pieniak et al., 2020 | indicator values of vascular plants for light, temperature, continentality, soil moisture, trophy, soil acidity and | Diameter at breast height | - | Seed dispersal | - |
| Freitas et al., 2020 | - | Diameter at breast height | - | - | - |
| Fukano et al., 2020 | - | Height and width; number of shoots and total number of nodes | Competition; dry weight | - | - |
| Gaggini et al., 2017 | Soil characteristics; percentage area covered by forests, agriculture land, traffic infrastructure, built-up area and ornamental gardens | - | - | - | - |
| Géron et al, 2021 | Urban index, sky view factor (SVF) | Height, internode space, specific leaf area | Chlorophyll content, anthocyanins, and flavonols indices | - | - |
| Glišić et al, 2021 | Urban index, climatic characteristics | - | - | - | - |
| Greene and Blossey, 2014 | Land use; soil infiltration, soil nutrients, and ground water table depth | Height | - | Seed germination | - |
| Grella et al., 2018 | Soil properties | - | - | - | - |
| Guerra et al., 2017 | - | Height; circumference | Light requirements; regeneration strategy | Seed size, dispersal syndrome | - |
| Güler, 2020 | - | - | - | - | - |



| Study | Title | DOI | Continent | Country | City/Region |
|---|---|---|---|---|---|
| Gulezian et al., 2010 | Distribution of invasive plants in urban environment is strongly spatially structured. | https://doi.org/10.1007/s10980-016-0480-9 | North America | United States | Chicago |
| Hahs and McDonnell, 2013 | Composition of the soil seed bank in remnant patches of grassy woodland along an urbanization gradient in Melbourne, Australia. | https://doi.org/10.1007/s11258-013-0248-0 | Oceania | Australia | Melbourne |
| Heneidy et al, 2021 | Pattern of Urban Flora in Intra-City Railway Habitats (Alexandria, Egypt): A Conservation Perspective | https://doi.org/10.3390/biology10080698 | Africa | Egypt | Alexandria |
| Hou et al., 2019 | Urbanization threaten the pollination of Gentiana dahurica. | https://doi.org/10.1038/s41598-018-36773-7 | Asia | China | Xi'ning |
| Huang et al., 2013 | Effect of urbanization on the structure and functional traits of remnant subtropical evergreen broad-leaved forests in South China. | https://doi.org/10.1007/s11066 1-012-2921-5 | Asia | China | Guangzhou |
| Ilyas et al, 2021 | Adaptation of functional traits and their plasticity of three ornamental trees growing in urban environment. | https://doi.org/10.1016/j.scienta.2021.110248 | Asia | China | Wuhan |
| Irwin et al., 2018 | Phenotypic selection on floral traits in an urban landscape. | http://dx.doi.org/10.1098/rspb.2018.1239 | North America | United States | Raleigh-Durham |
| Jensen et al, 2021 | Contrasting effects of tree origin and urbanization on invertebrate abundance and tree phenology. | https://doi.org/10.1002/eap.2491 | Europe | Sweden | Malmo |
| Jeong et al., 2011 | Impact of urban warming on earlier spring flowering in Korea. | https://doi.org/10.1002/joc.2178 | Asia | South Korea | - |
| Jochner et al., 2012 | The influence of altitude and urbanisation on trends and mean dates in phenology (1980–2009). | https://doi.org/10.1007/s00484-011-0444-3 | Europe | Germany | Frankfurt, Cologne and Munich |



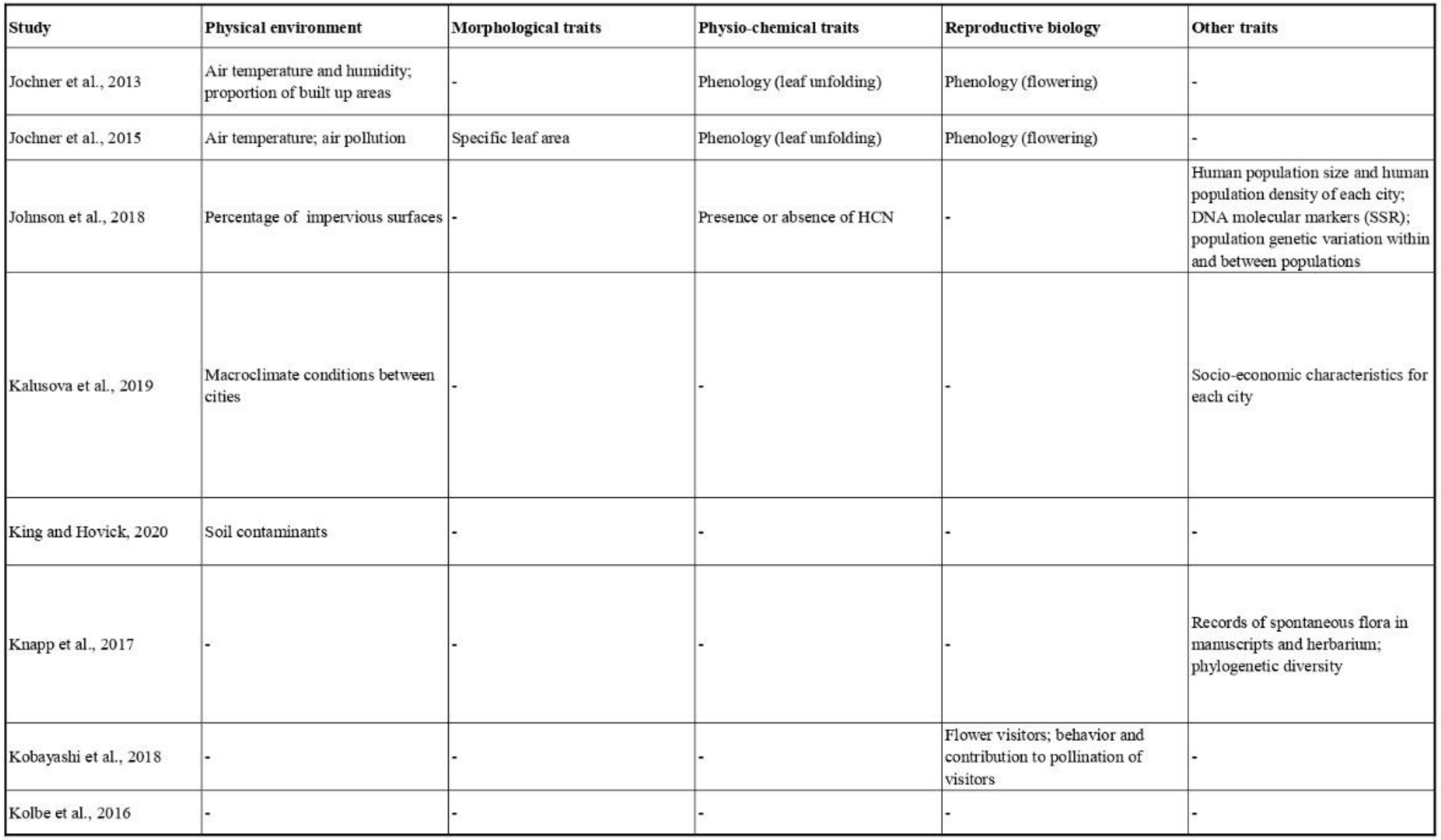
| Study | Taxa | Approach | Megacity | Biome | Community traits |
|---|---|---|---|---|---|
| Gulezian et al., 2010 | *Ailanthus altissima, Alliaria petiolata, Arctium minus, Cirsium vulgare, Dipsacus silvestris, Melilotus alba, Phragmites australis, Rhamnus cathartica, Rosa multiflora, Setaria glauca* | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | Species abundance |
| Hahs and McDonnell, 2013 | Woody plants | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | - |
| Heneidy et al, 2021 | All vascular plants | Urban - non-urban gradient | Yes | Mediterranean Forests, Woodlands & Scrub | Species richness, species covering, evenness |
| Hou et al., 2019 | *Gentiana dahurica* | Urban - non-urban gradient | Yes | Montane Grasslands & Shrublands | - |
| Huang et al., 2013 | All vascular plants | Urban - non-urban gradient | Yes | Tropical & Subtropical Moist Broadleaf Forests | - |
| Ilyas et al, 2021 | *Platanus linn, Populus euramevicana, Cinnamomum camphora* | Urban - non-urban gradient | Yes | Tropical & Subtropical Moist Broadleaf Forests | - |
| Irwin et al., 2018 | *Gelsemium sempervirens* | Urban vs non-urban | Yes | Temperate Broadleaf & Mixed Forests | - |
| Jensen et al, 2021 | *Acer platanoides, Betula pendula, Crataegus monogyna, Fagus sylvatica, Prunus avium, Quercus robur, Salix cáprea, Sorbus intermedia, Tilia x europaea, Ailanthus altíssima, Ginkgo biloba, Gleditsia triacanthos, Platanus x hispânica, Pterocarya fraxinifolia, Quercus cerris, Robinia pseudoacacia* | Urban vs non-urban | No | Temperate Broadleaf & Mixed Forests | Presence of non-native species, abundance of invertebrates associated to the host tree |
| Jeong et al., 2011 | *Forsythia koreana, Rhododendron mucronulatum, Prunus yedoensis, Prunus persica* | Urban vs non-urban/ Temporal | - | Temperate Broadleaf & Mixed Forests | - |
| Jochner et al., 2012 | *Corylus avellana, Galanthus nivalis, Salix caprea, Forsythia suspensa, Acer platanoides, Fagus sylvatica, Malus domestica, Picea abies, Syringa vulgaris* | Urban - non-urban gradient/ Temporal | No | Temperate Broadleaf & Mixed Forests | - |



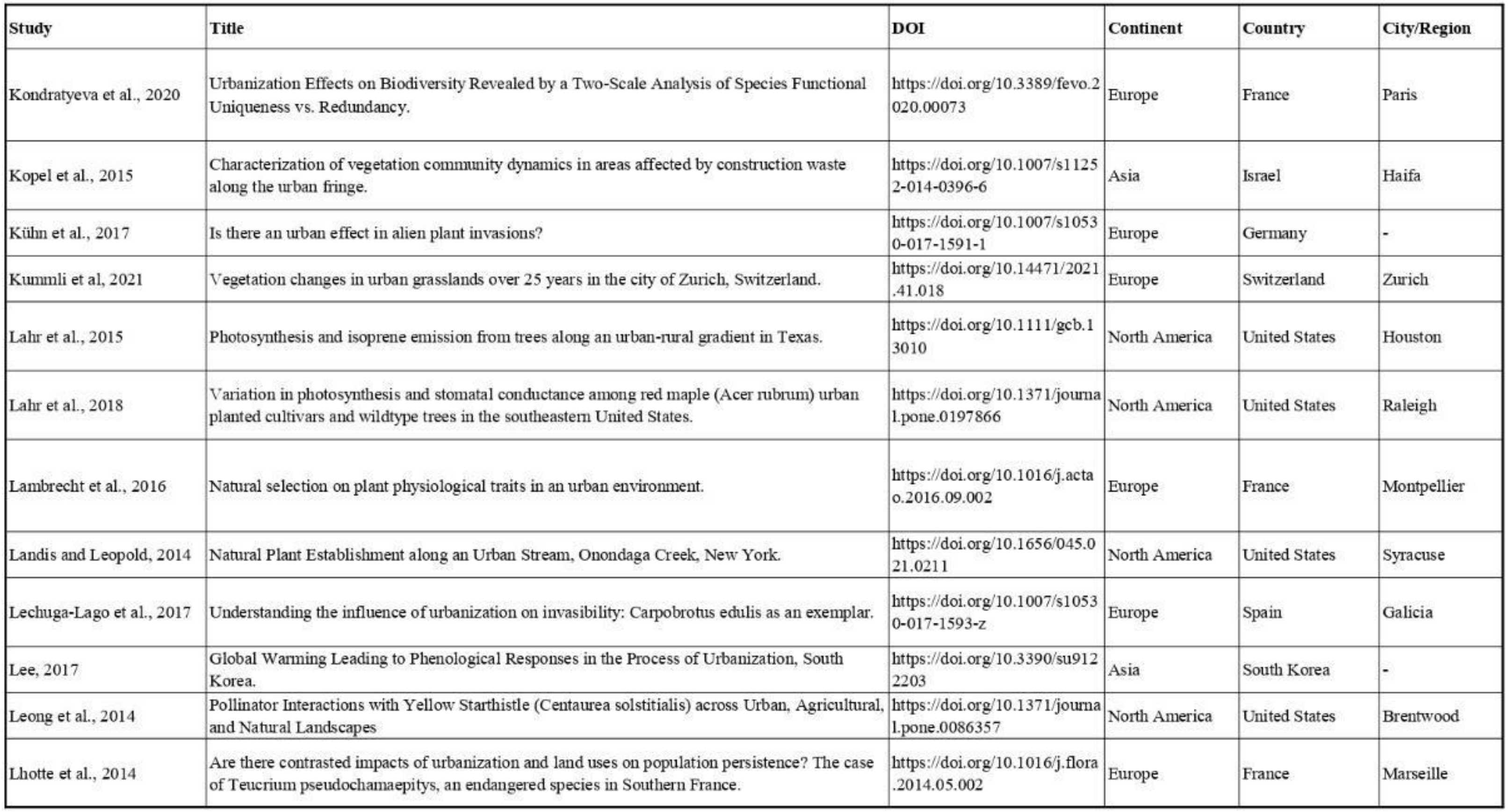
| Study | Physical environment | Morphological traits | Physio-chemical traits | Reproductive biology | Other traits |
|---|---|---|---|---|---|
| Gulezian et al., 2010 | - | Height | - | - | Earliest recorded date of a herbarium |
| Hahs and McDonnell, 2013 | - | Diameter at breast height | - | Soil seed bank; germination | - |
| Heneidy et al, 2021 | Human activities and disturbance, climate data | Life forms | - | - | - |
| Hou et al., 2019 | - | - | - | Pollinator observation; length and width of flowers; length of filaments and style; number of flowers; pollen and ovule numbers; pollen/ovule ratio; flower development; reproductive success; pollen limitation | - |
| Huang et al., 2013 | Topography; soil properties; light | Diameter at breast height | - | - | - |
| Ilyas et al, 2021 | - | Single leaf area, leaf length/width, stomatal density, stomatal length/width, stomatal area, leaf thickness | - | - | - |
| Irwin et al., 2018 | - | - | Concentration of the alkaloid gelsemine | Floral morph; floral display size; corolla and petal length and width; robbing; florivory; pollination; number of open flowers | - |
| Jensen et al, 2021 | Temperature | - | Phenology (bud burst) | - | - |
| Jeong et al., 2011 | Temperature | - | - | Phenology (flowering) | - |
| Jochner et al., 2012 | Proportion of built up areas; altitude | - | Phenology (leaf unfolding; sprout) | Phenology (flowering) | - |

| Study | Title | DOI | Continent | Country | City/Region |
|---|---|---|---|---|---|
| Jochner et al., 2013 | Using phenology to assess urban heat islands in tropical and temperate regions. | https://doi.org/10.1002/joc.3651 | South America/ Europe | Brazil/ Germany | Campinas/ Munich |
| Jochner et al., 2015 | The effects of short- and long-term air pollutants on plant phenology and leaf characteristics. | https://doi.org/10.1016/j.envpol.2015.07.040 | Europe | Germany | Munich |
| Johnson et al., 2018 | Contrasting the effects of natural selection, genetic drift and gene flow on urban evolution in white clover (Trifolium repens). | http://dx.doi.org/10.1098/rspb.2018.1019 | North America | Canada | Ontario |
| Kalusova et al., 2019 | Similar responses of native and alien floras in European cities to climate. | https://doi.org/10.1111/jbi.13591 | Europe | Austria, Belgium, Bulgaria, Croatia, Czech Republic, France, Germany, Greece, Hungary, Italy, Netherlands, Poland, Portugal, Romania, Serbia, Slovakia, Slovenia, Spain, Switzerland | Western, Central and Southern Europe |
| King and Hovick, 2020 | Wetland plant community variation across replicate urban to rural gradients: non-native species as both drivers and passengers in systems impacted by anthropogenic land-use. | https://doi.org/10.1007/s11252-020-01012-3 | North America | United States | Ohio |
| Knapp et al., 2017 | Increasing species richness but decreasing phylogenetic richness and divergence over a 320-year period of urbanization. | https://doi.org/10.1111/1365-2664.12826 | Europe | Germany | Halle |
| Kobayashi et al., 2018 | Comparison of Visitors and Pollinators of *Mucuna macrocarpa* between Urban and Forest Environments. | https://doi.org/10.3106/ms2018-0029 | Asia | Japan/ Taiwan | Okinawa-jima Island |
| Kolbe et al., 2016 | Effects of natural and anthropogenic environmental influences on tree community composition and structure in forests along an urban-wildland gradient in southwestern Ohio. | https://doi.org/10.1007/s11252-016-0531-7 | North America | United States | Southwestern Ohio |

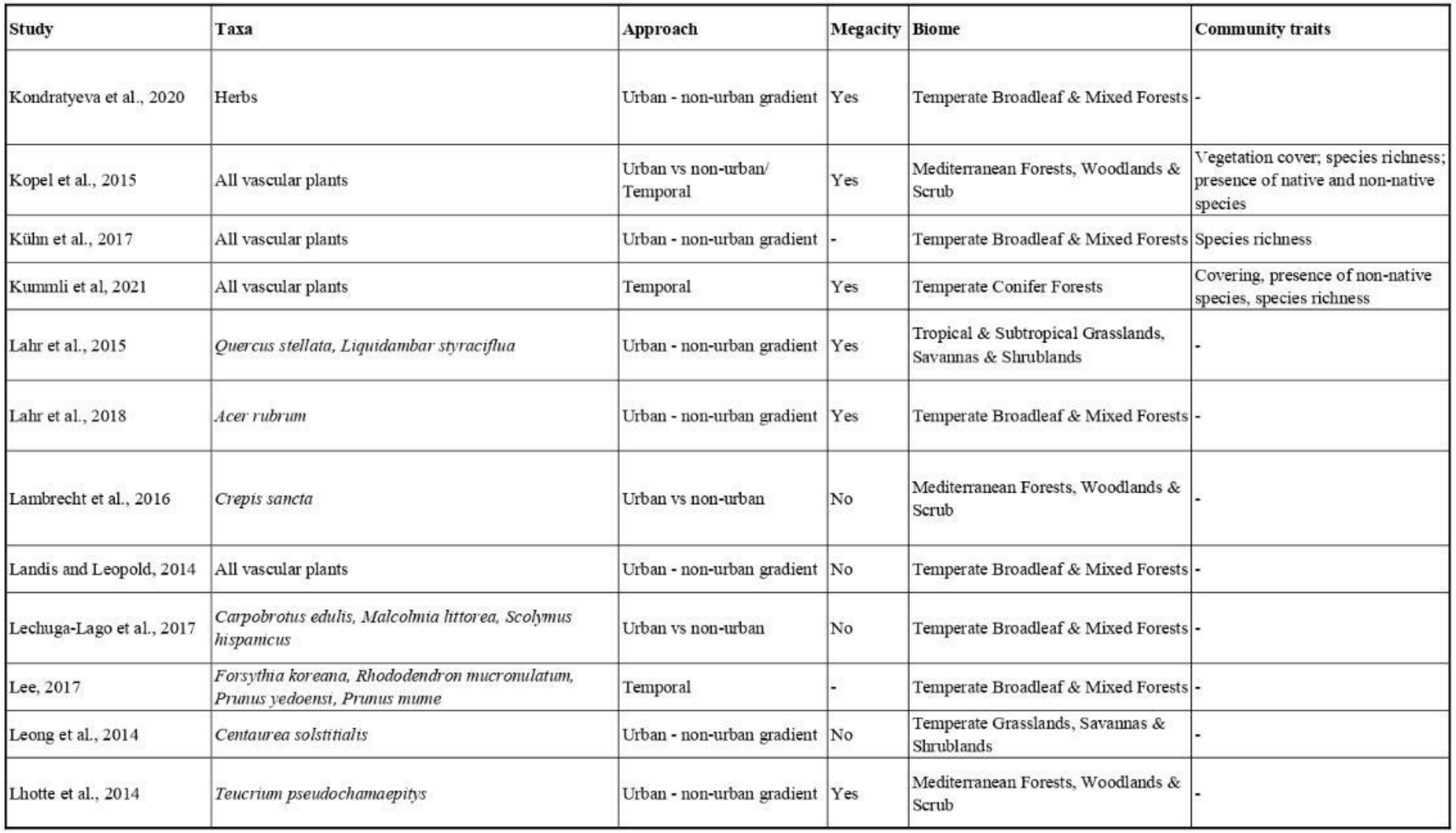
| Study | Taxa | Approach | Megacity | Biome | Community traits |
|---|---|---|---|---|---|
| Jochner et al., 2013 | *Tabebuia chrysotricha, Caesalpinia peltophoroides, Tipuana tipu, Betula pendula* | Urban - non-urban gradient | Yes | Tropical & Subtropical Moist Broadleaf Forests; Temperate Broadleaf & Mixed Forests | - |
| Jochner et al., 2015 | *Betula pendula, Corylus avellana, Aesculus hippocastanum* | Urban vs non-urban | Yes | Temperate Broadleaf & Mixed Forests | - |
| Johnson et al., 2018 | *Trifolium repens* | Urban - non-urban gradient | - | Temperate Broadleaf & Mixed Forests | Density of populations |
| Kalusova et al., 2019 | All vascular plants | Urban - non-urban gradient | - | Mediterranean Forests, Woodlands & Scrub; Temperate Broadleaf & Mixed Forests; Temperate Conifer Forests; Temperate Grasslands, Savannas & Shrublands | Presence of native and non-native species; species richness |
| King and Hovick, 2020 | Wetland herbs | Urban - non-urban gradient | - | Temperate Broadleaf & Mixed Forests | Species abundance; species richness; abundance of native and non-native species |
| Knapp et al., 2017 | All vascular plants | Temporal | No | Temperate Broadleaf & Mixed Forests | Presence of native and non-native species; extirpated species; species richness; native immigrants; non-native immigrants; today's threatened species; potential future non-native immigrants |
| Kobayashi et al., 2018 | *Mucuna macrocarpa* | Urban vs non-urban | - | Tropical & Subtropical Moist Broadleaf Forests | - |
| Kolbe et al., 2016 | Woody plants | Urban - non-urban gradient | - | Temperate Broadleaf & Mixed Forests | Canopy cover; species richness, diversity and abundance; evenness |



| Study | Physical environment | Morphological traits | Physio-chemical traits | Reproductive biology | Other traits |
|---|---|---|---|---|---|
| Jochner et al., 2013 | Air temperature and humidity; proportion of built up areas | - | Phenology (leaf unfolding) | Phenology (flowering) | - |
| Jochner et al., 2015 | Air temperature; air pollution | Specific leaf area | Phenology (leaf unfolding) | Phenology (flowering) | - |
| Johnson et al., 2018 | Percentage of impervious surfaces | - | Presence or absence of HCN | - | Human population size and human population density of each city;<br>DNA molecular markers (SSR);<br>population genetic variation within and between populations |
| Kalusova et al., 2019 | Macroclimate conditions between cities | - | - | - | Socio-economic characteristics for each city |
| King and Hovick, 2020 | Soil contaminants | - | - | - | - |
| Knapp et al., 2017 | - | - | - | - | Records of spontaneous flora in manuscripts and herbarium;<br>phylogenetic diversity |
| Kobayashi et al., 2018 | - | - | - | Flower visitors; behavior and contribution to pollination of visitors | - |
| Kolbe et al., 2016 | - | - | - | - | - |



| Study | Title | DOI | Continent | Country | City/Region |
|---|---|---|---|---|---|
| Kondratyeva et al., 2020 | Urbanization Effects on Biodiversity Revealed by a Two-Scale Analysis of Species Functional Uniqueness vs. Redundancy. | https://doi.org/10.3389/fevo.2020.00073 | Europe | France | Paris |
| Kopel et al., 2015 | Characterization of vegetation community dynamics in areas affected by construction waste along the urban fringe. | https://doi.org/10.1007/s11252-014-0396-6 | Asia | Israel | Haifa |
| Kühn et al., 2017 | Is there an urban effect in alien plant invasions? | https://doi.org/10.1007/s10530-017-1591-1 | Europe | Germany | - |
| Kummli et al, 2021 | Vegetation changes in urban grasslands over 25 years in the city of Zurich, Switzerland. | https://doi.org/10.14471/2021.41.018 | Europe | Switzerland | Zurich |
| Lahr et al., 2015 | Photosynthesis and isoprene emission from trees along an urban-rural gradient in Texas. | https://doi.org/10.1111/gcb.13010 | North America | United States | Houston |
| Lahr et al., 2018 | Variation in photosynthesis and stomatal conductance among red maple (Acer rubrum) urban planted cultivars and wildtype trees in the southeastern United States. | https://doi.org/10.1371/journal.pone.0197866 | North America | United States | Raleigh |
| Lambrecht et al., 2016 | Natural selection on plant physiological traits in an urban environment. | https://doi.org/10.1016/j.actao.2016.09.002 | Europe | France | Montpellier |
| Landis and Leopold, 2014 | Natural Plant Establishment along an Urban Stream, Onondaga Creek, New York. | https://doi.org/10.1656/045.021.0211 | North America | United States | Syracuse |
| Lechuga-Lago et al., 2017 | Understanding the influence of urbanization on invasibility: Carpobrotus edulis as an exemplar. | https://doi.org/10.1007/s10530-017-1593-z | Europe | Spain | Galicia |
| Lee, 2017 | Global Warming Leading to Phenological Responses in the Process of Urbanization, South Korea. | https://doi.org/10.3390/su9122203 | Asia | South Korea | - |
| Leong et al., 2014 | Pollinator Interactions with Yellow Starthistle (Centaurea solstitialis) across Urban, Agricultural, and Natural Landscapes | https://doi.org/10.1371/journal.pone.0086357 | North America | United States | Brentwood |
| Lhotte et al., 2014 | Are there contrasted impacts of urbanization and land uses on population persistence? The case of Teucrium pseudochamaepitys, an endangered species in Southern France. | https://doi.org/10.1016/j.flora.2014.05.002 | Europe | France | Marseille |



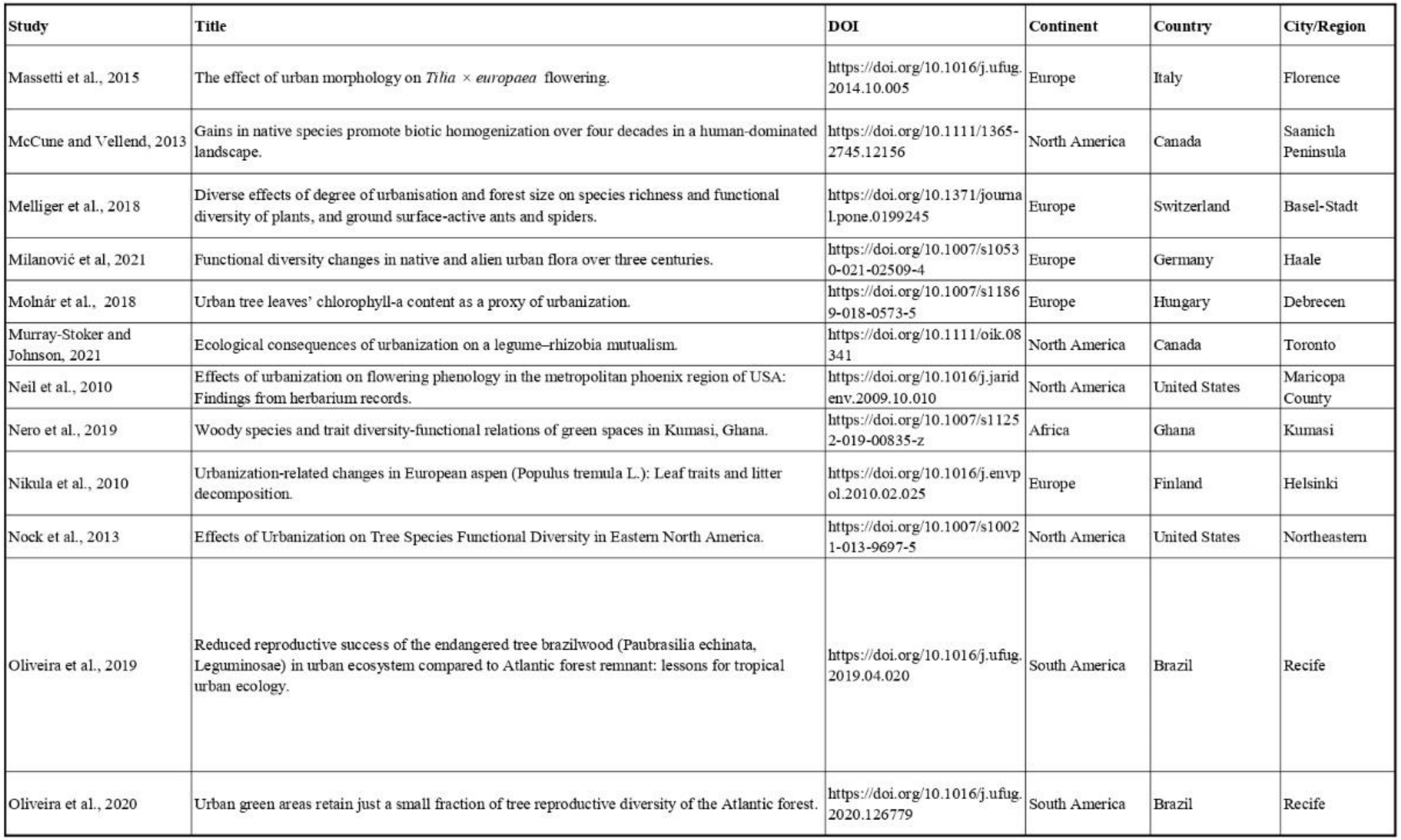
| Study | Taxa | Approach | Megacity | Biome | Community traits |
|---|---|---|---|---|---|
| Kondratyeva et al., 2020 | Herbs | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | - |
| Kopel et al., 2015 | All vascular plants | Urban vs non-urban/ Temporal | Yes | Mediterranean Forests, Woodlands & Scrub | Vegetation cover; species richness; presence of native and non-native species |
| Kühn et al., 2017 | All vascular plants | Urban - non-urban gradient | - | Temperate Broadleaf & Mixed Forests | Species richness |
| Kummli et al, 2021 | All vascular plants | Temporal | Yes | Temperate Conifer Forests | Covering, presence of non-native species, species richness |
| Lahr et al., 2015 | *Quercus stellata, Liquidambar styraciflua* | Urban - non-urban gradient | Yes | Tropical & Subtropical Grasslands, Savannas & Shrublands | - |
| Lahr et al., 2018 | *Acer rubrum* | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | - |
| Lambrecht et al., 2016 | *Crepis sancta* | Urban vs non-urban | No | Mediterranean Forests, Woodlands & Scrub | - |
| Landis and Leopold, 2014 | All vascular plants | Urban - non-urban gradient | No | Temperate Broadleaf & Mixed Forests | - |
| Lechuga-Lago et al., 2017 | *Carpobrotus edulis, Malcolmia littorea, Scolymus hispanicus* | Urban vs non-urban | No | Temperate Broadleaf & Mixed Forests | - |
| Lee, 2017 | *Forsythia koreana, Rhododendron mucronulatum, Prunus yedoensi, Prunus mume* | Temporal | - | Temperate Broadleaf & Mixed Forests | - |
| Leong et al., 2014 | *Centaurea solstitialis* | Urban - non-urban gradient | No | Temperate Grasslands, Savannas & Shrublands | - |
| Lhotte et al., 2014 | *Teucrium pseudochamaepitys* | Urban - non-urban gradient | Yes | Mediterranean Forests, Woodlands & Scrub | - |

| Study | Physical environment | Morphological traits | Physio-chemical traits | Reproductive biology | Other traits |
|---|---|---|---|---|---|
| Kondratyeva et al., 2020 | Land cover | Leaf size; specific leaf area; plant height | Leaf dry matter content | Life span; pollen vector, seed dispersal mode; type of reproduction; phenology (flowering); seed weight | - |
| Kopel et al., 2015 | - | - | - | - | - |
| Kühn et al., 2017 | Climate; geology; land cover; stems | - | - | - | - |
| Kummli et al, 2021 | - | - | - | - | - |
| Lahr et al., 2015 | Temperature; CO2 concentration | - | Gas exchange measurements; basal isoprene emission; rates of photosynthesis | - | - |
| Lahr et al., 2018 | Temperature; CO2 concentration | - | Gas exchange measurements; rates of photosynthesis and stomatal conductance | - | - |
| Lambrecht et al., 2016 | Temperature; soil moisture; climate | Number of leaves; diameter of the rosette | Rates of photosynthesis; stomatal conductance; leaf nitrogen content; water-use efficiency; Phenology (senescence) | - | - |
| Landis and Leopold, 2014 | Urban cover | - | - | Seedbank in soil | - |
| Lechuga-Lago et al., 2017 | Soil pH, conductivity, water content, nutrients; soil enzymatic activities | - | - | Germination performances of the invasive vs the native species | - |
| Lee, 2017 | Climate; precipitation | - | - | Phenology (budding and flowering) | - |
| Leong et al., 2014 | - | - | - | Bee visitation; seed set | - |
| Lhotte et al., 2014 | - | - | - | Pollinator activities; number of flowers, fruits, mature seeds; reproductive success | - |

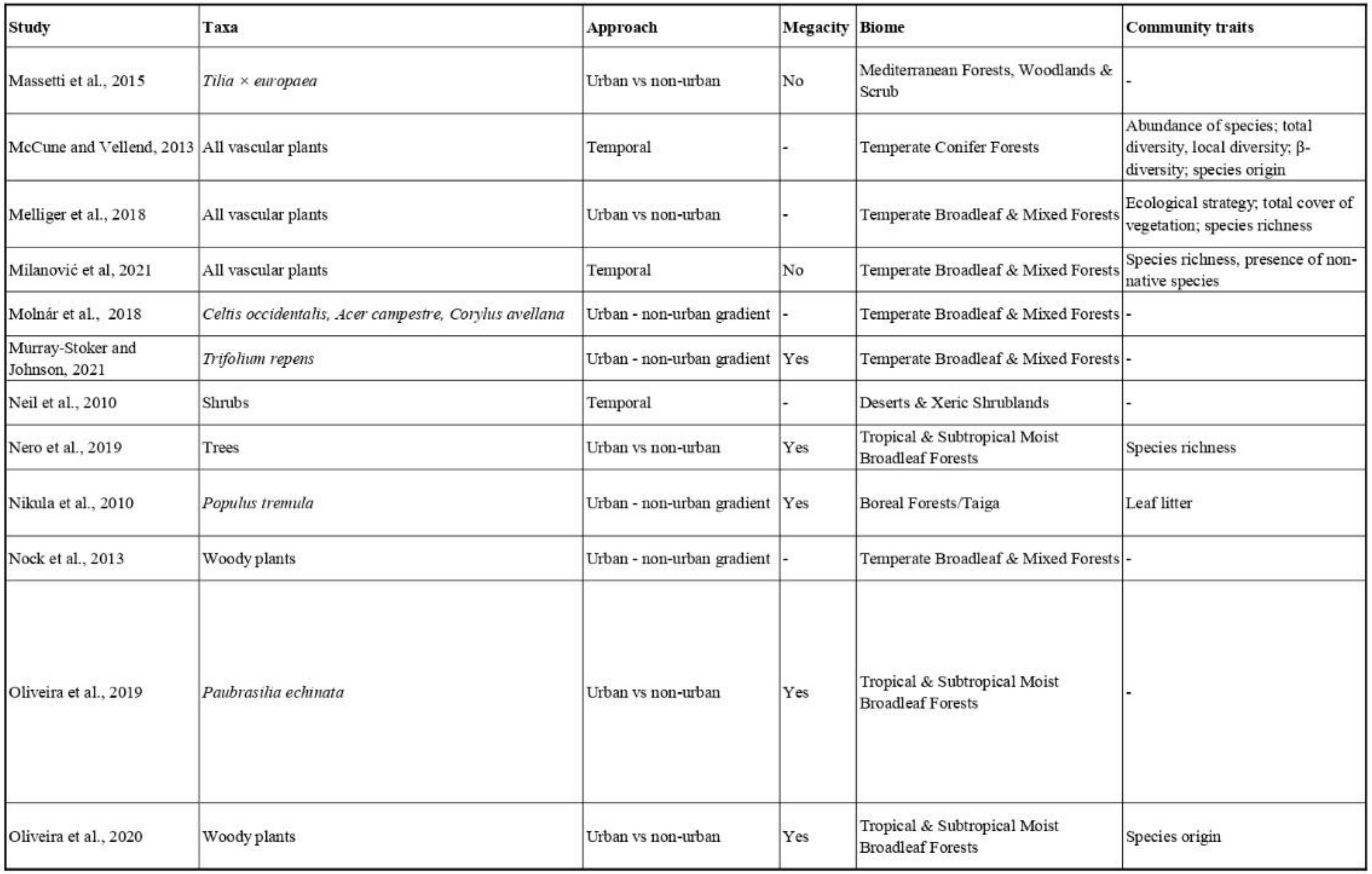
| Study | Title | DOI | Continent | Country | City/Region |
|---|---|---|---|---|---|
| Li et al., 2017 | Response of vegetation phenology to urbanization in the conterminous United States. | https://doi.org/10.1111/gcb.13562 | North America | United States | - |
| Lindh et al., 2018 | Changes in urban plant phenology in the Pacific Northwest from 1959 to 2016: anthropogenic warming and natural oscillation. | https://doi.org/10.1007/s00484-018-1567-6 | North America | United States | Salem |
| Lobo et al., 2016 | Visitation rate of pollinators and nectar robbers to the flowers and inflorescences of Tabebuia aurea (Bignoniaceae): effects of floral display size and habitat fragmentation. | https://doi.org/10.1111/boj.12435 | South America | Brazil | Brasilia |
| Löki et al., 2019 | Predictors of conservation value of Turkish cemeteries: A case study using orchids. | https://doi.org/10.1016/j.landurbplan.2019.02.016 | Asia | Turkey | - |
| Lopez et al., 2018 | Testing the effects of four urbanization filters on forest plant taxonomic, functional, and phylogenetic diversity. | https://doi.org/10.1002/eap.1812 | North America | United States | North Carolina |
| Lososová et al., 2016 | Biotic homogenization of urban floras by alien species: the role of species turnover and richness differences. | https://doi.org/10.1111/jvs.12381 | Europe | Austria, Belgium, Czech Republic, Germany, Hungary, Netherlands, Poland, Slovakia, Slovenia, Switzerland | - |
| Lowry et al, 2020 | Spatial patterns of presence, abundance, and richness of invasive woody plants in relation to urbanization in a tropical island setting. | https://doi.org/10.1016/j.ufug.2019.126516 | Oceania | Fiji | Suva |
| Malkinson et al., 2018 | From rural-urban gradients to patch – matrix frameworks: Plant diversity patterns in urban landscapes. | https://doi.org/10.1016/j.landurbplan.2017.09.021 | Asia | Israel | Haifa |

| Study | Taxa | Approach | Megacity | Biome | Community traits |
|---|---|---|---|---|---|
| Li et al., 2017 | All vascular plants | Urban - non-urban gradient/ Temporal | - | Temperate Conifer Forests; Deserts & Xeric Shrublands; Mediterranean Forests, Woodlands & Scrub; Temperate Grasslands, Savannas & Shrublands; Temperate Broadleaf & Mixed Forests; Tropical & Subtropical Grasslands, Savannas & Shrublands; Flooded Grasslands & Savannas | - |
| Lindh et al., 2018 | All vascular plants | Temporal | No | Temperate Broadleaf & Mixed Forests | - |
| Lobo et al., 2016 | *Tabebuia aurea* | Urban vs non-urban | Yes | Tropical & Subtropical Grasslands, Savannas & Shrublands | - |
| Löki et al., 2019 | Orchids | Urban - non-urban gradient/ Temporal | - | Mediterranean Forests, Woodlands & Scrub; Temperate Broadleaf & Mixed Forests; Temperate Conifer Forests | Species richness; abundance; probability of finding threatened taxa; tree cover; proportion of native trees |
| Lopez et al., 2018 | Herbs | Urban - non-urban gradient | - | Temperate Broadleaf & Mixed Forests | Species abundance |
| Lososová et al., 2016 | All vascular plants | Urban - non-urban gradient | - | Temperate Broadleaf & Mixed Forests; Temperate Conifer Forests | Presence on native and non-native species; species turnover; species richness |
| Lowry et al, 2020 | *Citharexylum spinosum, Psidium guajava, Schefflera actinophylla, Spathodea campanulata, Clerodendrum chinense, Clidemia hirta, Lantana camara, Leucaena leucocephala, Piper aduncum, Solanum torvum, Coccinia grandis, Ipomoea cairica, Merremia peltata, Mikania micrantha* | Urban - non-urban gradient | No | Tropical & Subtropical Moist Broadleaf Forests | Species richness, abundance |
| Malkinson et al., 2018 | All vascular plants | Urban - non-urban gradient | Yes | Mediterranean Forests, Woodlands & Scrub | Presence on native and non-native species |

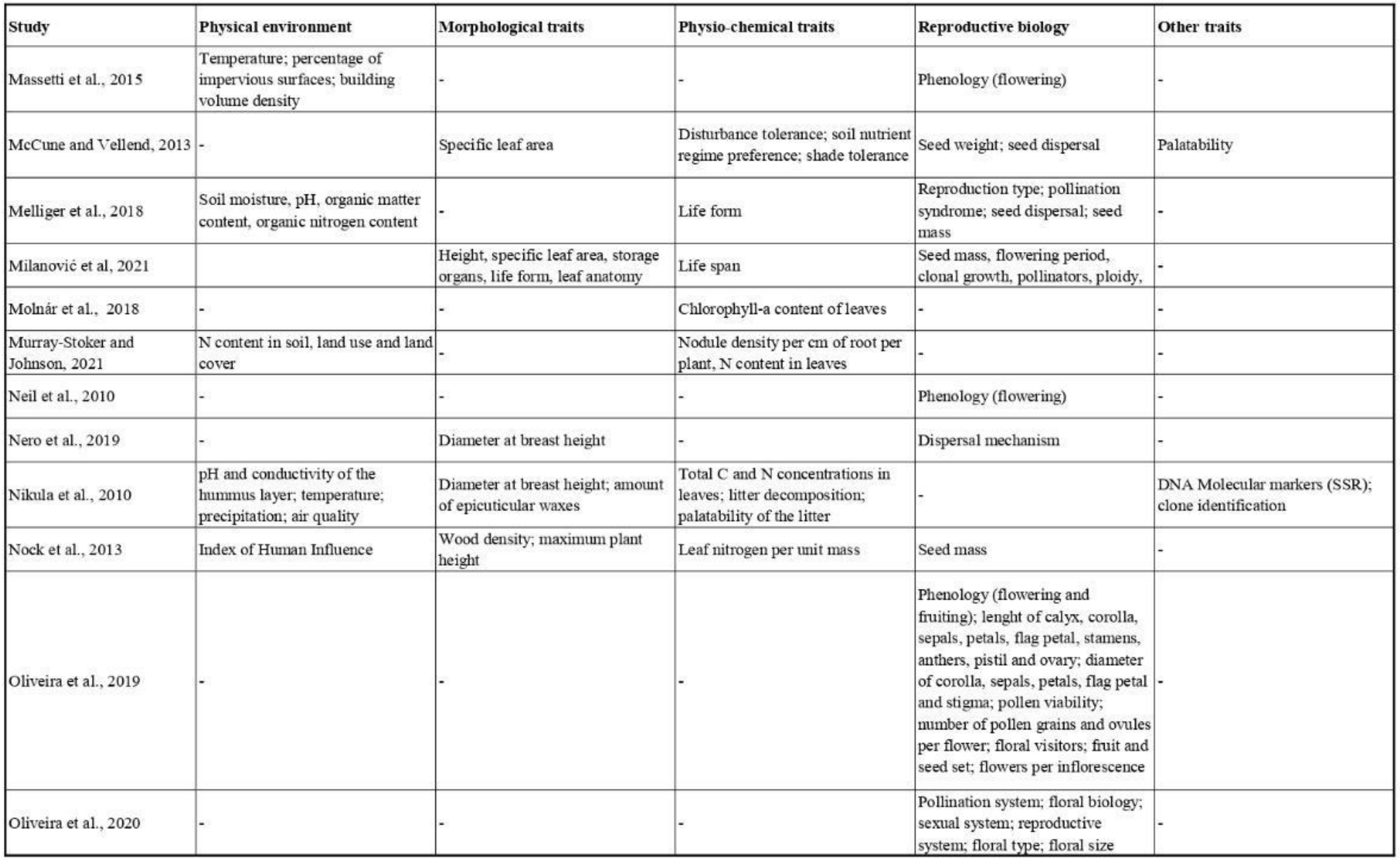
| Study | Physical environment | Morphological traits | Physio-chemical traits | Reproductive biology | Other traits |
|---|---|---|---|---|---|
| Li et al., 2017 | Land use | - | Phenology (greenness) | - | Remote sensing |
| Lindh et al., 2018 | Temperature | - | Phenology (Start leaf; defoliation) | Phenology (flowering) | - |
| Lobo et al., 2016 | - | - | - | Floral visitors; flower size; nectar production | - |
| Lőki et al., 2019 | Latitude, longitude, altitude, area of the cemetery; settlement type | - | - | - | Age, density and type of graves in cemeteries |
| Lopez et al., 2018 | Wetland indicator status; area of forest cover; cover of ornamental plant; UHI; urban stream morphology; additions of soil nutrients; pollutants | Specific leaf area | Leaf type (evergreen vs. deciduous); leaf nitrogen content | Dispersal mode; seed mass | Phylogenetic diversity |
| Lososová et al., 2016 | - | - | - | - | - |
| Lowry et al, 2020 | - | - | - | - | - |
| Malkinson et al., 2018 | Habitat type; percentage of sealed surface | - | - | - | - |



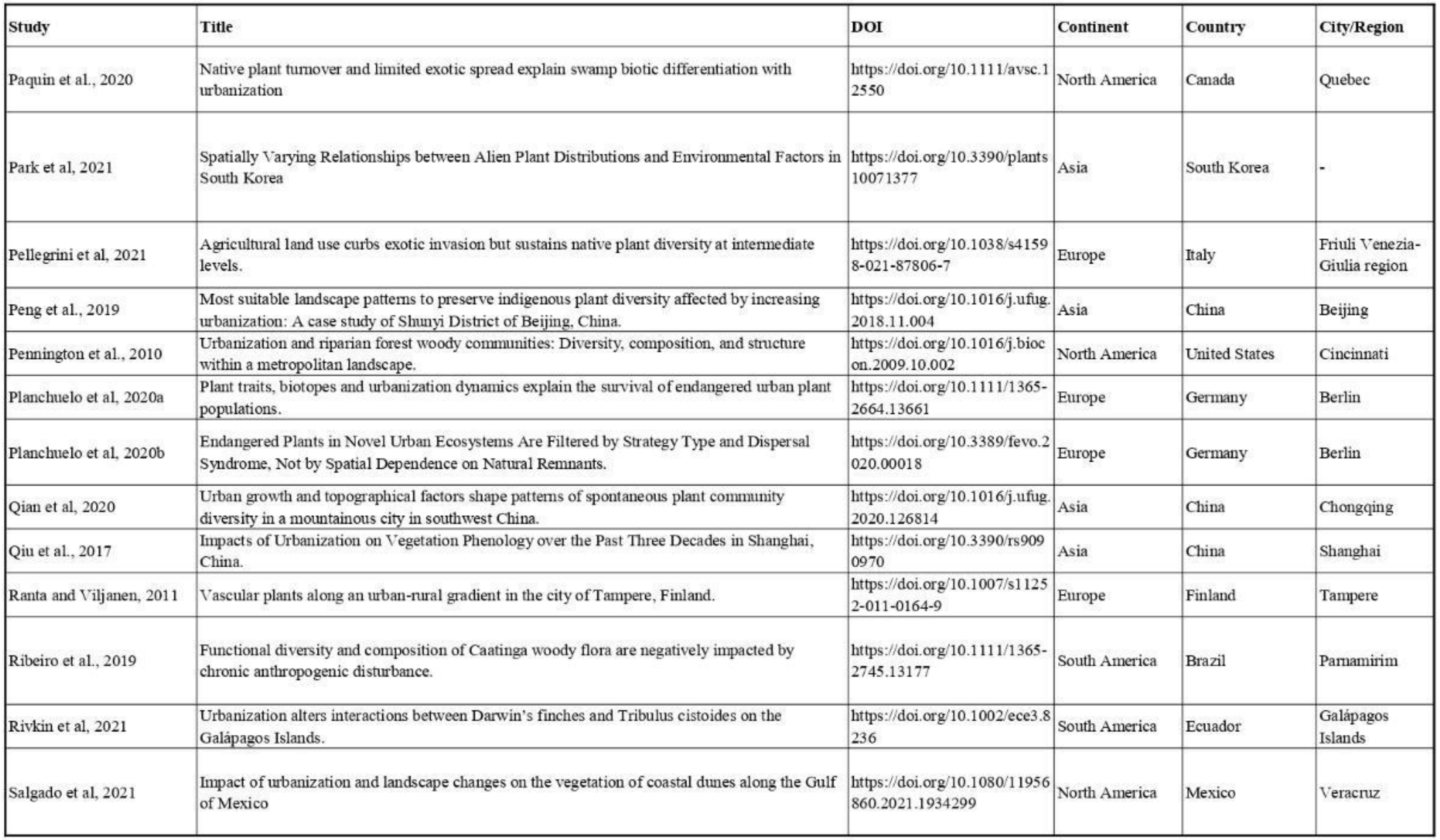
| Study | Title | DOI | Continent | Country | City/Region |
|---|---|---|---|---|---|
| Massetti et al., 2015 | The effect of urban morphology on *Tilia × europaea* flowering. | https://doi.org/10.1016/j.ufug.2014.10.005 | Europe | Italy | Florence |
| McCune and Vellend, 2013 | Gains in native species promote biotic homogenization over four decades in a human-dominated landscape. | https://doi.org/10.1111/1365-2745.12156 | North America | Canada | Saanich Peninsula |
| Melliger et al., 2018 | Diverse effects of degree of urbanisation and forest size on species richness and functional diversity of plants, and ground surface-active ants and spiders. | https://doi.org/10.1371/journal.pone.0199245 | Europe | Switzerland | Basel-Stadt |
| Milanović et al, 2021 | Functional diversity changes in native and alien urban flora over three centuries. | https://doi.org/10.1007/s10530-021-02509-4 | Europe | Germany | Haale |
| Molnár et al., 2018 | Urban tree leaves' chlorophyll-a content as a proxy of urbanization. | https://doi.org/10.1007/s11869-018-0573-5 | Europe | Hungary | Debrecen |
| Murray-Stoker and Johnson, 2021 | Ecological consequences of urbanization on a legume–rhizobia mutualism. | https://doi.org/10.1111/oik.08341 | North America | Canada | Toronto |
| Neil et al., 2010 | Effects of urbanization on flowering phenology in the metropolitan phoenix region of USA: Findings from herbarium records. | https://doi.org/10.1016/j.jaridenv.2009.10.010 | North America | United States | Maricopa County |
| Nero et al., 2019 | Woody species and trait diversity-functional relations of green spaces in Kumasi, Ghana. | https://doi.org/10.1007/s11252-019-00835-z | Africa | Ghana | Kumasi |
| Nikula et al., 2010 | Urbanization-related changes in European aspen (Populus tremula L.): Leaf traits and litter decomposition. | https://doi.org/10.1016/j.envpol.2010.02.025 | Europe | Finland | Helsinki |
| Nock et al., 2013 | Effects of Urbanization on Tree Species Functional Diversity in Eastern North America. | https://doi.org/10.1007/s10021-013-9697-5 | North America | United States | Northeastern |
| Oliveira et al., 2019 | Reduced reproductive success of the endangered tree brazilwood (Paubrasilia echinata, Leguminosae) in urban ecosystem compared to Atlantic forest remnant: lessons for tropical urban ecology. | https://doi.org/10.1016/j.ufug.2019.04.020 | South America | Brazil | Recife |
| Oliveira et al., 2020 | Urban green areas retain just a small fraction of tree reproductive diversity of the Atlantic forest. | https://doi.org/10.1016/j.ufug.2020.126779 | South America | Brazil | Recife |



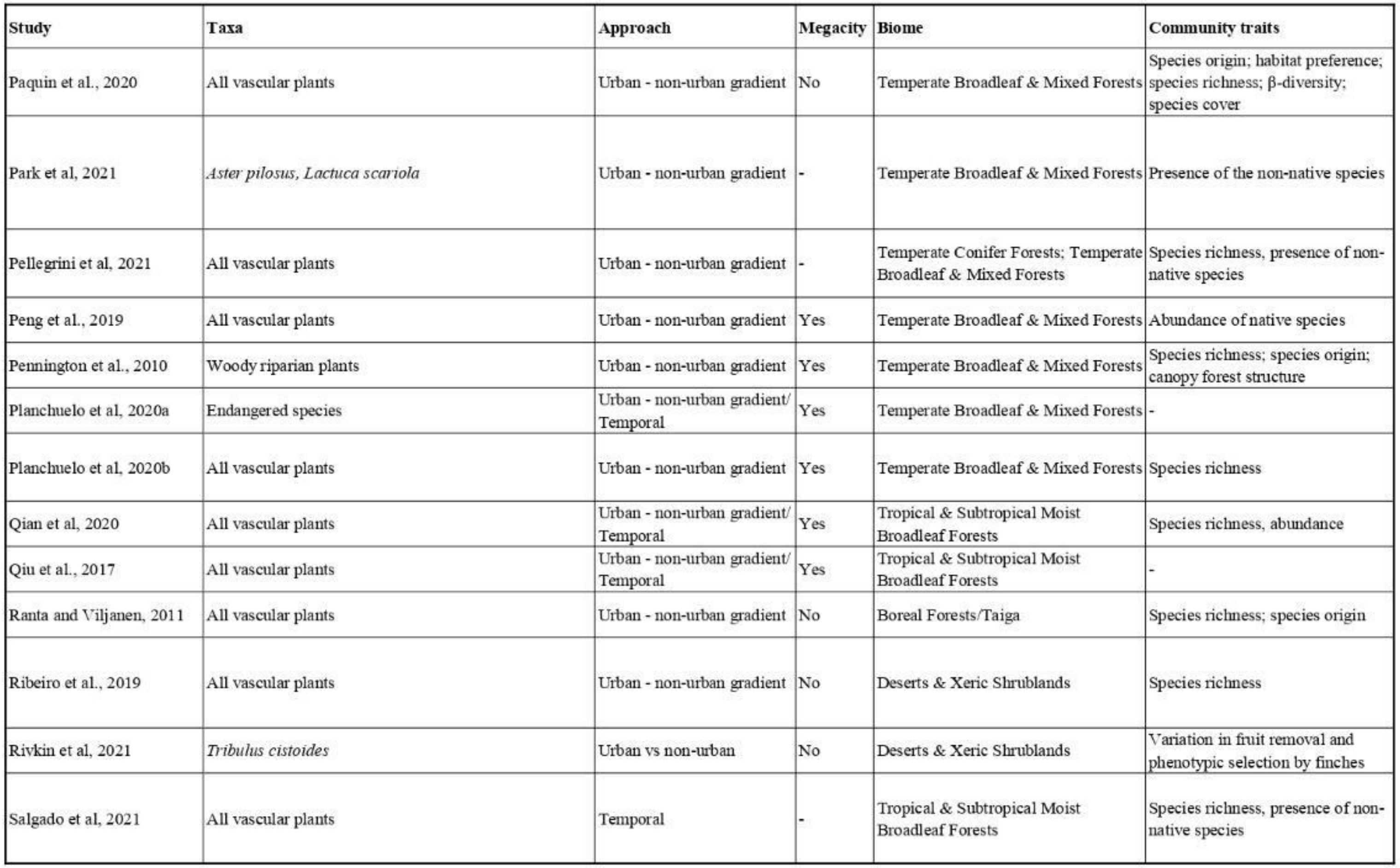
| Study | Taxa | Approach | Megacity | Biome | Community traits |
|---|---|---|---|---|---|
| Massetti et al., 2015 | *Tilia × europaea* | Urban vs non-urban | No | Mediterranean Forests, Woodlands & Scrub | - |
| McCune and Vellend, 2013 | All vascular plants | Temporal | - | Temperate Conifer Forests | Abundance of species; total diversity, local diversity; β-diversity; species origin |
| Melliger et al., 2018 | All vascular plants | Urban vs non-urban | - | Temperate Broadleaf & Mixed Forests | Ecological strategy; total cover of vegetation; species richness |
| Milanović et al, 2021 | All vascular plants | Temporal | No | Temperate Broadleaf & Mixed Forests | Species richness, presence of non-native species |
| Molnár et al., 2018 | *Celtis occidentalis, Acer campestre, Corylus avellana* | Urban - non-urban gradient | - | Temperate Broadleaf & Mixed Forests | - |
| Murray-Stoker and Johnson, 2021 | *Trifolium repens* | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | - |
| Neil et al., 2010 | Shrubs | Temporal | - | Deserts & Xeric Shrublands | - |
| Nero et al., 2019 | Trees | Urban vs non-urban | Yes | Tropical & Subtropical Moist Broadleaf Forests | Species richness |
| Nikula et al., 2010 | *Populus tremula* | Urban - non-urban gradient | Yes | Boreal Forests/Taiga | Leaf litter |
| Nock et al., 2013 | Woody plants | Urban - non-urban gradient | - | Temperate Broadleaf & Mixed Forests | - |
| Oliveira et al., 2019 | *Paubrasilia echinata* | Urban vs non-urban | Yes | Tropical & Subtropical Moist Broadleaf Forests | - |
| Oliveira et al., 2020 | Woody plants | Urban vs non-urban | Yes | Tropical & Subtropical Moist Broadleaf Forests | Species origin |



| Study | Physical environment | Morphological traits | Physio-chemical traits | Reproductive biology | Other traits |
|---|---|---|---|---|---|
| Massetti et al., 2015 | Temperature; percentage of impervious surfaces; building volume density | - | - | Phenology (flowering) | - |
| McCune and Vellend, 2013 | - | Specific leaf area | Disturbance tolerance; soil nutrient regime preference; shade tolerance | Seed weight; seed dispersal | Palatability |
| Melliger et al., 2018 | Soil moisture, pH, organic matter content, organic nitrogen content | - | Life form | Reproduction type; pollination syndrome; seed dispersal; seed mass | - |
| Milanović et al, 2021 |  | Height, specific leaf area, storage organs, life form, leaf anatomy | Life span | Seed mass, flowering period, clonal growth, pollinators, ploidy, | - |
| Molnár et al., 2018 | - | - | Chlorophyll-a content of leaves | - | - |
| Murray-Stoker and Johnson, 2021 | N content in soil, land use and land cover | - | Nodule density per cm of root per plant, N content in leaves | - | - |
| Neil et al., 2010 | - | - | - | Phenology (flowering) | - |
| Nero et al., 2019 | - | Diameter at breast height | - | Dispersal mechanism | - |
| Nikula et al., 2010 | pH and conductivity of the hummus layer; temperature; precipitation; air quality | Diameter at breast height; amount of epicuticular waxes | Total C and N concentrations in leaves; litter decomposition; palatability of the litter | - | DNA Molecular markers (SSR); clone identification |
| Nock et al., 2013 | Index of Human Influence | Wood density; maximum plant height | Leaf nitrogen per unit mass | Seed mass | - |
| Oliveira et al., 2019 | - | - | - | Phenology (flowering and fruiting); lenght of calyx, corolla, sepals, petals, flag petal, stamens, anthers, pistil and ovary; diameter of corolla, sepals, petals, flag petal and stigma; pollen viability; number of pollen grains and ovules per flower; floral visitors; fruit and seed set; flowers per inflorescence | - |
| Oliveira et al., 2020 | - | - | - | Pollination system; floral biology; sexual system; reproductive system; floral type; floral size | - |

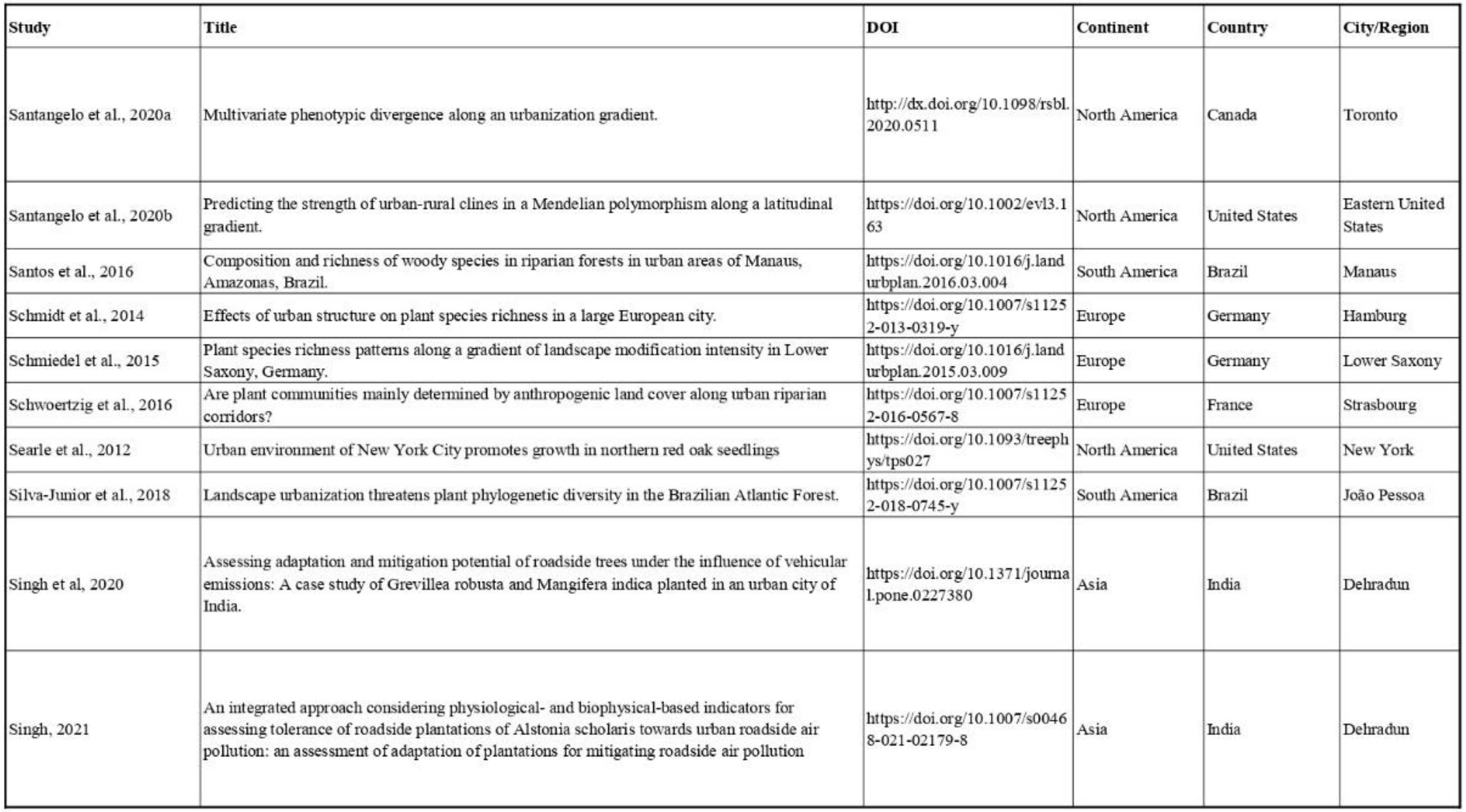


| Study | Title | DOI | Continent | Country | City/Region |
|---|---|---|---|---|---|
| Paquin et al., 2020 | Native plant turnover and limited exotic spread explain swamp biotic differentiation with urbanization | https://doi.org/10.1111/avsc.12550 | North America | Canada | Quebec |
| Park et al, 2021 | Spatially Varying Relationships between Alien Plant Distributions and Environmental Factors in South Korea | https://doi.org/10.3390/plants10071377 | Asia | South Korea | - |
| Pellegrini et al, 2021 | Agricultural land use curbs exotic invasion but sustains native plant diversity at intermediate levels. | https://doi.org/10.1038/s41598-021-87806-7 | Europe | Italy | Friuli Venezia-Giulia region |
| Peng et al., 2019 | Most suitable landscape patterns to preserve indigenous plant diversity affected by increasing urbanization: A case study of Shunyi District of Beijing, China. | https://doi.org/10.1016/j.ufug.2018.11.004 | Asia | China | Beijing |
| Pennington et al., 2010 | Urbanization and riparian forest woody communities: Diversity, composition, and structure within a metropolitan landscape. | https://doi.org/10.1016/j.biocon.2009.10.002 | North America | United States | Cincinnati |
| Planchuelo et al, 2020a | Plant traits, biotopes and urbanization dynamics explain the survival of endangered urban plant populations. | https://doi.org/10.1111/1365-2664.13661 | Europe | Germany | Berlin |
| Planchuelo et al, 2020b | Endangered Plants in Novel Urban Ecosystems Are Filtered by Strategy Type and Dispersal Syndrome, Not by Spatial Dependence on Natural Remnants. | https://doi.org/10.3389/fevo.2020.00018 | Europe | Germany | Berlin |
| Qian et al, 2020 | Urban growth and topographical factors shape patterns of spontaneous plant community diversity in a mountainous city in southwest China. | https://doi.org/10.1016/j.ufug.2020.126814 | Asia | China | Chongqing |
| Qiu et al., 2017 | Impacts of Urbanization on Vegetation Phenology over the Past Three Decades in Shanghai, China. | https://doi.org/10.3390/rs9090970 | Asia | China | Shanghai |
| Ranta and Viljanen, 2011 | Vascular plants along an urban-rural gradient in the city of Tampere, Finland. | https://doi.org/10.1007/s11252-011-0164-9 | Europe | Finland | Tampere |
| Ribeiro et al., 2019 | Functional diversity and composition of Caatinga woody flora are negatively impacted by chronic anthropogenic disturbance. | https://doi.org/10.1111/1365-2745.13177 | South America | Brazil | Parnamirim |
| Rivkin et al, 2021 | Urbanization alters interactions between Darwin's finches and Tribulus cistoides on the Galápagos Islands. | https://doi.org/10.1002/ece3.8236 | South America | Ecuador | Galápagos Islands |
| Salgado et al, 2021 | Impact of urbanization and landscape changes on the vegetation of coastal dunes along the Gulf of Mexico | https://doi.org/10.1080/11956860.2021.1934299 | North America | Mexico | Veracruz |

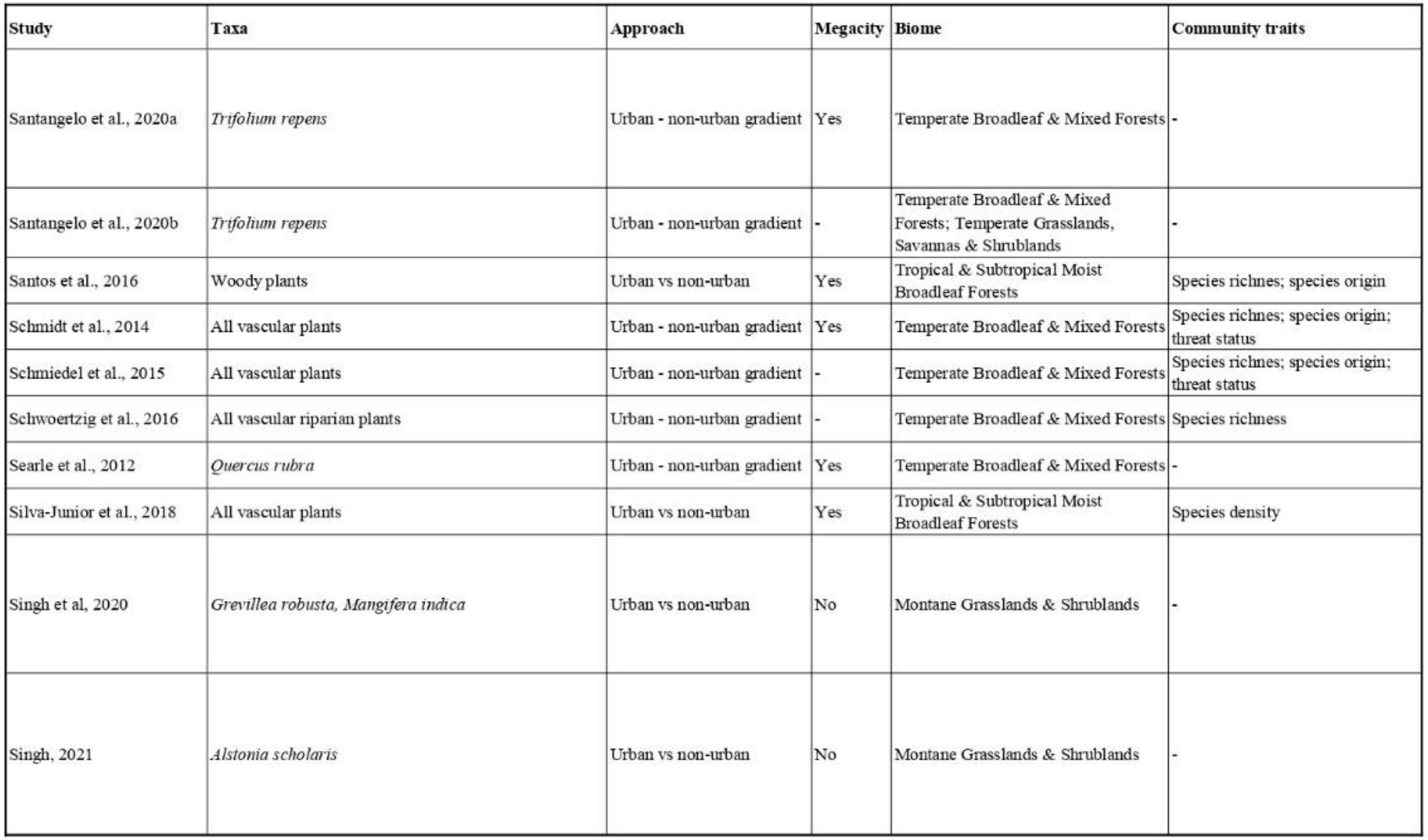


| Study | Taxa | Approach | Megacity | Biome | Community traits |
|---|---|---|---|---|---|
| Paquin et al., 2020 | All vascular plants | Urban - non-urban gradient | No | Temperate Broadleaf & Mixed Forests | Species origin; habitat preference; species richness; β-diversity; species cover |
| Park et al, 2021 | *Aster pilosus, Lactuca scariola* | Urban - non-urban gradient | - | Temperate Broadleaf & Mixed Forests | Presence of the non-native species |
| Pellegrini et al, 2021 | All vascular plants | Urban - non-urban gradient | - | Temperate Conifer Forests; Temperate Broadleaf & Mixed Forests | Species richness, presence of non-native species |
| Peng et al., 2019 | All vascular plants | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | Abundance of native species |
| Pennington et al., 2010 | Woody riparian plants | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | Species richness; species origin; canopy forest structure |
| Planchuelo et al, 2020a | Endangered species | Urban - non-urban gradient/ Temporal | Yes | Temperate Broadleaf & Mixed Forests | - |
| Planchuelo et al, 2020b | All vascular plants | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | Species richness |
| Qian et al, 2020 | All vascular plants | Urban - non-urban gradient/ Temporal | Yes | Tropical & Subtropical Moist Broadleaf Forests | Species richness, abundance |
| Qiu et al., 2017 | All vascular plants | Urban - non-urban gradient/ Temporal | Yes | Tropical & Subtropical Moist Broadleaf Forests | - |
| Ranta and Viljanen, 2011 | All vascular plants | Urban - non-urban gradient | No | Boreal Forests/Taiga | Species richness; species origin |
| Ribeiro et al., 2019 | All vascular plants | Urban - non-urban gradient | No | Deserts & Xeric Shrublands | Species richness |
| Rivkin et al, 2021 | *Tribulus cistoides* | Urban vs non-urban | No | Deserts & Xeric Shrublands | Variation in fruit removal and phenotypic selection by finches |
| Salgado et al, 2021 | All vascular plants | Temporal | - | Tropical & Subtropical Moist Broadleaf Forests | Species richness, presence of non-native species |

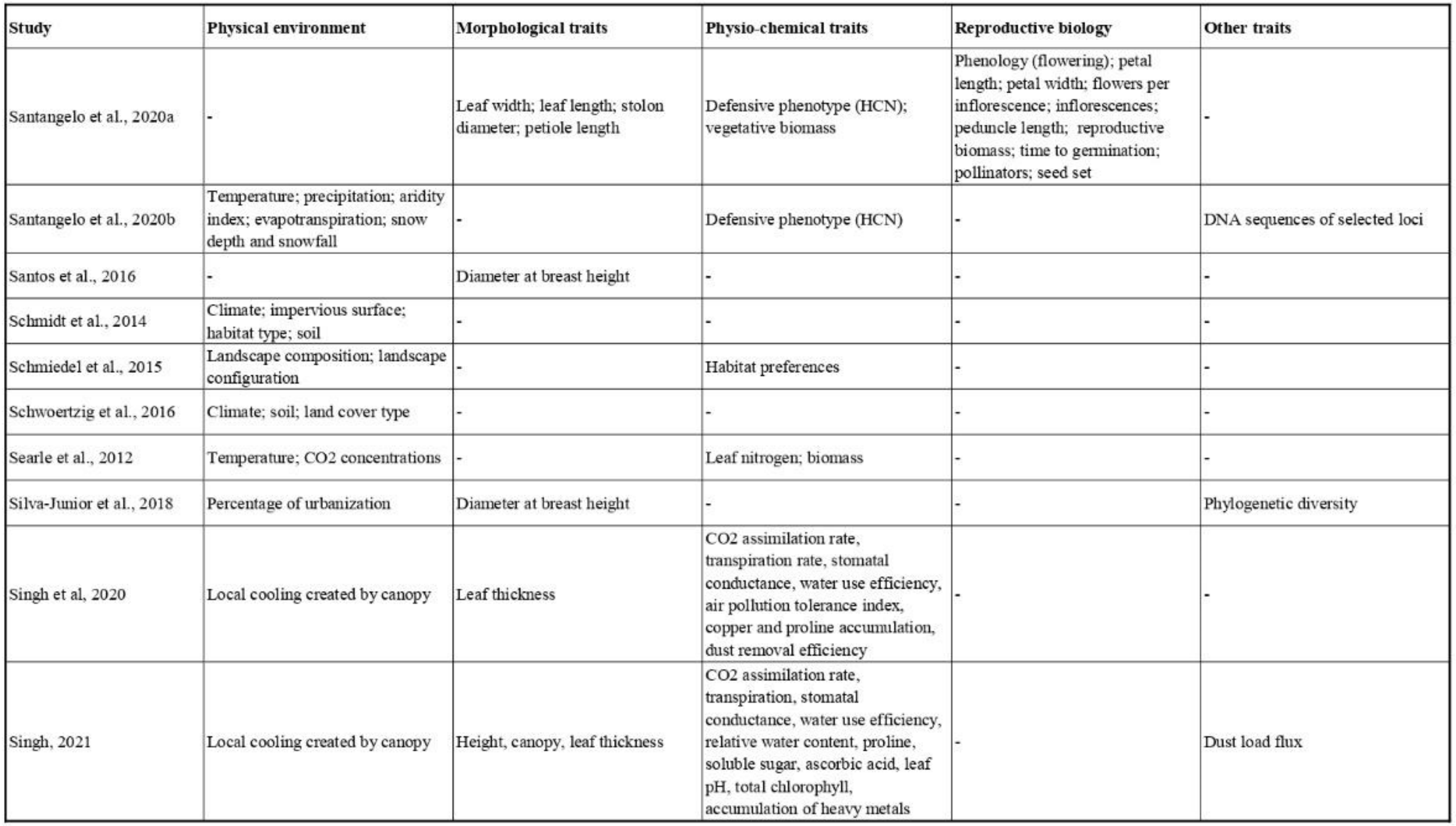
| Study | Physical environment | Morphological traits | Physio-chemical traits | Reproductive biology | Other traits |
|---|---|---|---|---|---|
| Paquin et al., 2020 | Land cover; Soil features | - | - | - | - |
| Park et al, 2021 | Anthropogenic activity, land-use variables, annual mean air temperature, annual temperature range, annual precipitation, topographic properties | - | - | - | - |
| Pellegrini et al, 2021 | Elevation, Monthly mean temperature, Mean rainfall, land use | - | - | - | - |
| Peng et al., 2019 | Complexity, fragmentation, diversity, and urbanization | - | - | - | - |
| Pennington et al., 2010 | Land-cover data | Diameter at breast height | Species moisture requirement | - | - |
| Planchuelo et al, 2020a | Land use | Specific leaf area, plant height | - | - | - |
| Planchuelo et al, 2020b | Land use | - | - | Dispersal syndrome, seed mass, lenght, width, numer and terminal velocity | - |
| Qian et al, 2020 | Land use, topography | - | - | - | - |
| Qiu et al., 2017 | - | - | Phenology (growing season) | - | Remote sensing |
| Ranta and Viljanen, 2011 | Land use | - | - | - | Human population |
| Ribeiro et al., 2019 | Chronic antropogenic disturbance | Diameter at breast height; Specific leaf area; mass per area; dry matter content and thickness; wood density | Deciduousness | Dispersion | - |
| Rivkin et al, 2021 | - | - | - | Seed removal by finches | - |
| Salgado et al, 2021 | Urban index | - | Functional traits: burial resistance, dispersal by ocean water, salinity resistance and type of below-ground structures | - | - |



| Study | Title | DOI | Continent | Country | City/Region |
|---|---|---|---|---|---|
| Santangelo et al., 2020a | Multivariate phenotypic divergence along an urbanization gradient. | http://dx.doi.org/10.1098/rsbl.2020.0511 | North America | Canada | Toronto |
| Santangelo et al., 2020b | Predicting the strength of urban-rural clines in a Mendelian polymorphism along a latitudinal gradient. | https://doi.org/10.1002/evl3.163 | North America | United States | Eastern United States |
| Santos et al., 2016 | Composition and richness of woody species in riparian forests in urban areas of Manaus, Amazonas, Brazil. | https://doi.org/10.1016/j.landurbplan.2016.03.004 | South America | Brazil | Manaus |
| Schmidt et al., 2014 | Effects of urban structure on plant species richness in a large European city. | https://doi.org/10.1007/s11252-013-0319-y | Europe | Germany | Hamburg |
| Schmiedel et al., 2015 | Plant species richness patterns along a gradient of landscape modification intensity in Lower Saxony, Germany. | https://doi.org/10.1016/j.landurbplan.2015.03.009 | Europe | Germany | Lower Saxony |
| Schwoertzig et al., 2016 | Are plant communities mainly determined by anthropogenic land cover along urban riparian corridors? | https://doi.org/10.1007/s11252-016-0567-8 | Europe | France | Strasbourg |
| Searle et al., 2012 | Urban environment of New York City promotes growth in northern red oak seedlings | https://doi.org/10.1093/treephys/tps027 | North America | United States | New York |
| Silva-Junior et al., 2018 | Landscape urbanization threatens plant phylogenetic diversity in the Brazilian Atlantic Forest. | https://doi.org/10.1007/s11252-018-0745-y | South America | Brazil | João Pessoa |
| Singh et al, 2020 | Assessing adaptation and mitigation potential of roadside trees under the influence of vehicular emissions: A case study of Grevillea robusta and Mangifera indica planted in an urban city of India. | https://doi.org/10.1371/journal.pone.0227380 | Asia | India | Dehradun |
| Singh, 2021 | An integrated approach considering physiological- and biophysical-based indicators for assessing tolerance of roadside plantations of Alstonia scholaris towards urban roadside air pollution: an assessment of adaptation of plantations for mitigating roadside air pollution | https://doi.org/10.1007/s00468-021-02179-8 | Asia | India | Dehradun |



| Study | Taxa | Approach | Megacity | Biome | Community traits |
|---|---|---|---|---|---|
| Santangelo et al., 2020a | *Trifolium repens* | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | - |
| Santangelo et al., 2020b | *Trifolium repens* | Urban - non-urban gradient | - | Temperate Broadleaf & Mixed Forests; Temperate Grasslands, Savannas & Shrublands | - |
| Santos et al., 2016 | Woody plants | Urban vs non-urban | Yes | Tropical & Subtropical Moist Broadleaf Forests | Species richnes; species origin |
| Schmidt et al., 2014 | All vascular plants | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | Species richnes; species origin; threat status |
| Schmiedel et al., 2015 | All vascular plants | Urban - non-urban gradient | - | Temperate Broadleaf & Mixed Forests | Species richnes; species origin; threat status |
| Schwoertzig et al., 2016 | All vascular riparian plants | Urban - non-urban gradient | - | Temperate Broadleaf & Mixed Forests | Species richness |
| Searle et al., 2012 | *Quercus rubra* | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | - |
| Silva-Junior et al., 2018 | All vascular plants | Urban vs non-urban | Yes | Tropical & Subtropical Moist Broadleaf Forests | Species density |
| Singh et al, 2020 | *Grevillea robusta, Mangifera indica* | Urban vs non-urban | No | Montane Grasslands & Shrublands | - |
| Singh, 2021 | *Alstonia scholaris* | Urban vs non-urban | No | Montane Grasslands & Shrublands | - |



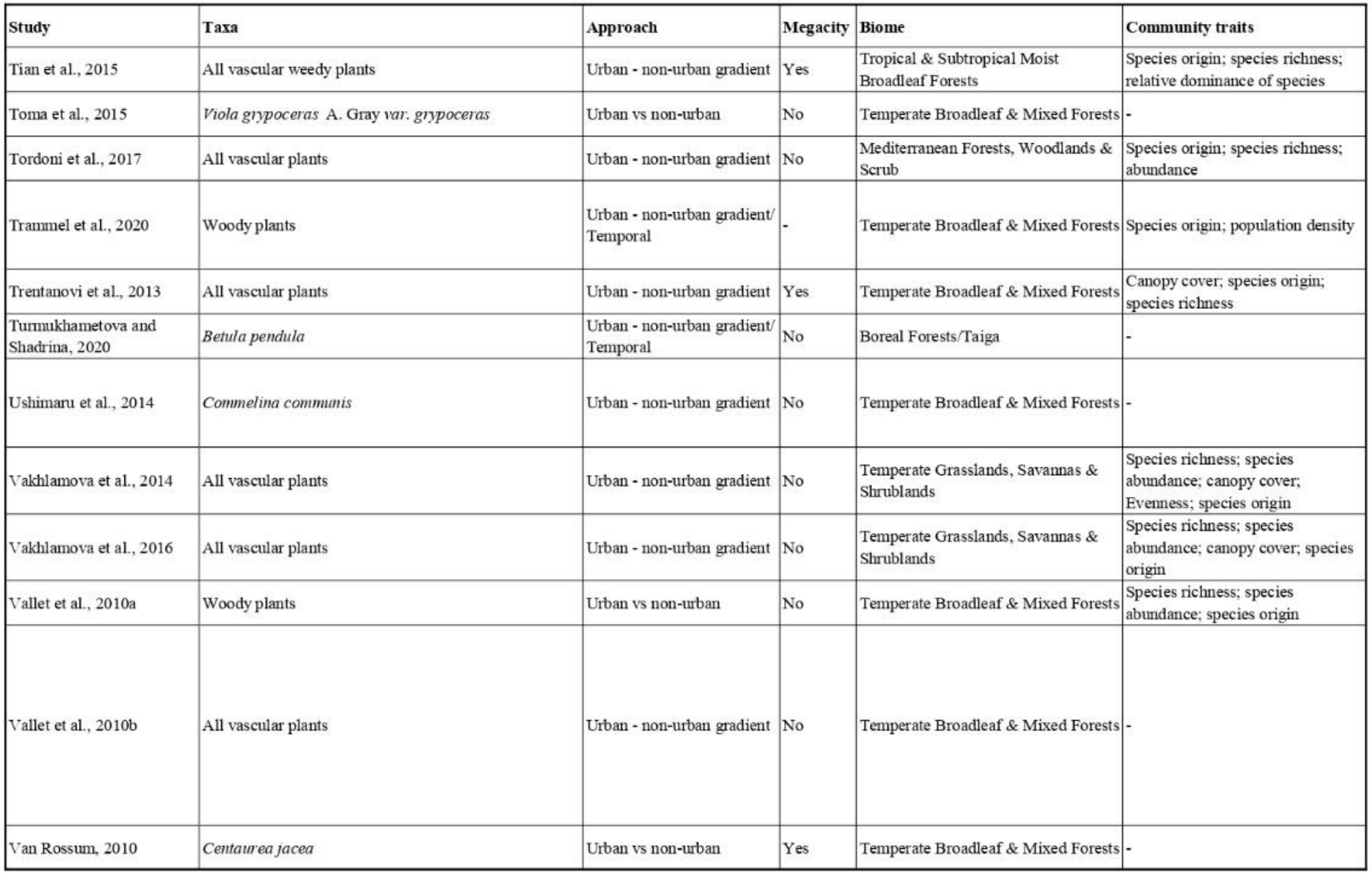
| Study | Physical environment | Morphological traits | Physio-chemical traits | Reproductive biology | Other traits |
|---|---|---|---|---|---|
| Santangelo et al., 2020a | - | Leaf width; leaf length; stolon diameter; petiole length | Defensive phenotype (HCN); vegetative biomass | Phenology (flowering); petal length; petal width; flowers per inflorescence; inflorescences; peduncle length; reproductive biomass; time to germination; pollinators; seed set | - |
| Santangelo et al., 2020b | Temperature; precipitation; aridity index; evapotranspiration; snow depth and snowfall | - | Defensive phenotype (HCN) | - | DNA sequences of selected loci |
| Santos et al., 2016 | - | Diameter at breast height | - | - | - |
| Schmidt et al., 2014 | Climate; impervious surface; habitat type; soil | - | - | - | - |
| Schmiedel et al., 2015 | Landscape composition; landscape configuration | - | Habitat preferences | - | - |
| Schwoertzig et al., 2016 | Climate; soil; land cover type | - | - | - | - |
| Searle et al., 2012 | Temperature; CO2 concentrations | - | Leaf nitrogen; biomass | - | - |
| Silva-Junior et al., 2018 | Percentage of urbanization | Diameter at breast height | - | - | Phylogenetic diversity |
| Singh et al, 2020 | Local cooling created by canopy | Leaf thickness | CO2 assimilation rate, transpiration rate, stomatal conductance, water use efficiency, air pollution tolerance index, copper and proline accumulation, dust removal efficiency | - | - |
| Singh, 2021 | Local cooling created by canopy | Height, canopy, leaf thickness | CO2 assimilation rate, transpiration, stomatal conductance, water use efficiency, relative water content, proline, soluble sugar, ascorbic acid, leaf pH, total chlorophyll, accumulation of heavy metals | - | Dust load flux |

| Study | Title | DOI | Continent | Country | City/Region |
|---|---|---|---|---|---|
| Skultety and Matthews, 2017 | Urbanization and roads drive non-native plant invasion in the Chicago Metropolitan region. | https://doi.org/10.1007/s10530-017-1464-7 | North America | United States | Chicago |
| Sołtysiak, 2020 | Does the urban heat island determine the distribution of Fallopia taxa in cities? – preliminary study from Wrocław (Central Europe). | https://doi.org/10.12775/EQ.2020.010 | Europe | Poland | Wrocław |
| Song et al., 2019 | Changes in plant functional traits and their relationships with environmental factors along an urban-rural gradient in Guangzhou, China. | https://doi.org/10.1016/j.ecolind.2019.105558 | Asia | China | Guangzhou |
| Sonti et al, 2021 | Chlorophyll fluorescence parameters, leaf traits and foliar chemistry of white oak and red maple trees in urban forest patches | https://doi.org/10.1093/treephys/tpaa121 | North America | United States | New York,<br>Philadelphia,<br>Baltimore |
| Štajerová et al., 2017 | Distribution of invasive plants in urban environment is strongly spatially structured. | https://doi.org/10.1007/s10980-016-0480-9 | Europe | Czech Republic | Hradec Kralove |
| Su et al, 2021 | Leaf Functional Traits Vary in Urban Environments: Influences of Leaf Age, Land-Use Type, and Urban–Rural Gradient. | https://doi.org/10.3389/fevo.2021.681959 | Asia | China | Beijing |
| Su et al., 2019 | Leaf and male cone phenophases of Chinese pine (Pinus tabulaeformis Carr.) along a rural-urban gradient in Beijing, China. | https://doi.org/10.1016/j.ufug.2019.05.009 | Asia | China | Beijing |
| Tew et al, 2021 | Quantifying nectar production by flowering plants in urban and rural landscapes. | https://doi.org/10.1111/1365-2745.13598 | Europe | United Kingdom | - |
| Thompson et al., 2016 | Urbanization drives the evolution of parallel clines in plant populations. | http://dx.doi.org/10.1098/rspb.2016.2180 | North America | Canada | Toronto |

| Study | Taxa | Approach | Megacity | Biome | Community traits |
|---|---|---|---|---|---|
| Skultety and Matthews, 2017 | *Alliaria petiolata, Aster subulatus, Atriplex patula, Bolboschoenus maritimus, Dipsacus fullonum, Dipsacus laciniatus, Hemerocallis fulva, Hordeum jubatum, Lysimachia nummularia, Lythrum salicaria, Phragmites australis, Puccinellia distans, Solanum dulcamara, Solidago sempervirens, Spergularia species, Viburnum recognitum* | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | Propagule Sources |
| Sołtysiak, 2020 | *Fallopia japonica, Fallopia sachalinensis, Fallopia × bohemica* | Urban - non-urban gradient/ Temporal | No | Temperate Broadleaf & Mixed Forests | Presence of the non-native species |
| Song et al., 2019 | *Schima superba, Castanopsis fissa, Acronychia pedunculata, Diplospora dubia, Ardisia quinquegona, Diospyros morrisiana, Castanopsis chinensis, Psychotria rubra, Memecylon nigrescens, Syzygium hancei, Pygeum topengii, Cinnamomum burmanni, Cryptocarya concinna, Endospermum chinense, Aidia pycnantha, Microdesmis caseariifolia, Machilus suaveolens, Machilus chinensis, Engelhardtia fenzlii, Alleizettella leucocarpa, Neolitsea phanerophlebia, Symplocos adenopus, Michelia skinneriana, Daphniphyllum subverticillatum, Manglietia moto, Neolitsea chuii* | Urban - non-urban gradient | Yes | Tropical & Subtropical Moist Broadleaf Forests | Relative frequency, density and dominance of species |
| Sonti et al, 2021 | *Quercus alba, Acer rubrum* | Urban vs non-urban | Yes | Temperate Broadleaf & Mixed Forests | - |
| Štajerová et al., 2017 | *Epilobium adenocaulon, Galeobdolon argentatum, Mahonia aquifolium, Matricaria discoidea, Oenothera biennis, Physocarpus opulifolius, Rhus typhina, Rumex thyrsiflorus, Syringa vulgaris* | Urban - non-urban gradient | No | Temperate Broadleaf & Mixed Forests | - |
| Su et al, 2021 | *Pinus tabuliformis* | Urban - non-urban gradient/ Temporal | Yes | Temperate Broadleaf & Mixed Forests | - |
| Su et al., 2019 | *Pinus tabulaeformis* | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | - |
| Tew et al, 2021 | Community of specific flowering species | Urban - non-urban gradient/ Temporal | - | Temperate Broadleaf & Mixed Forests | - |
| Thompson et al., 2016 | *Trifolium repens* | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | Herbivory |

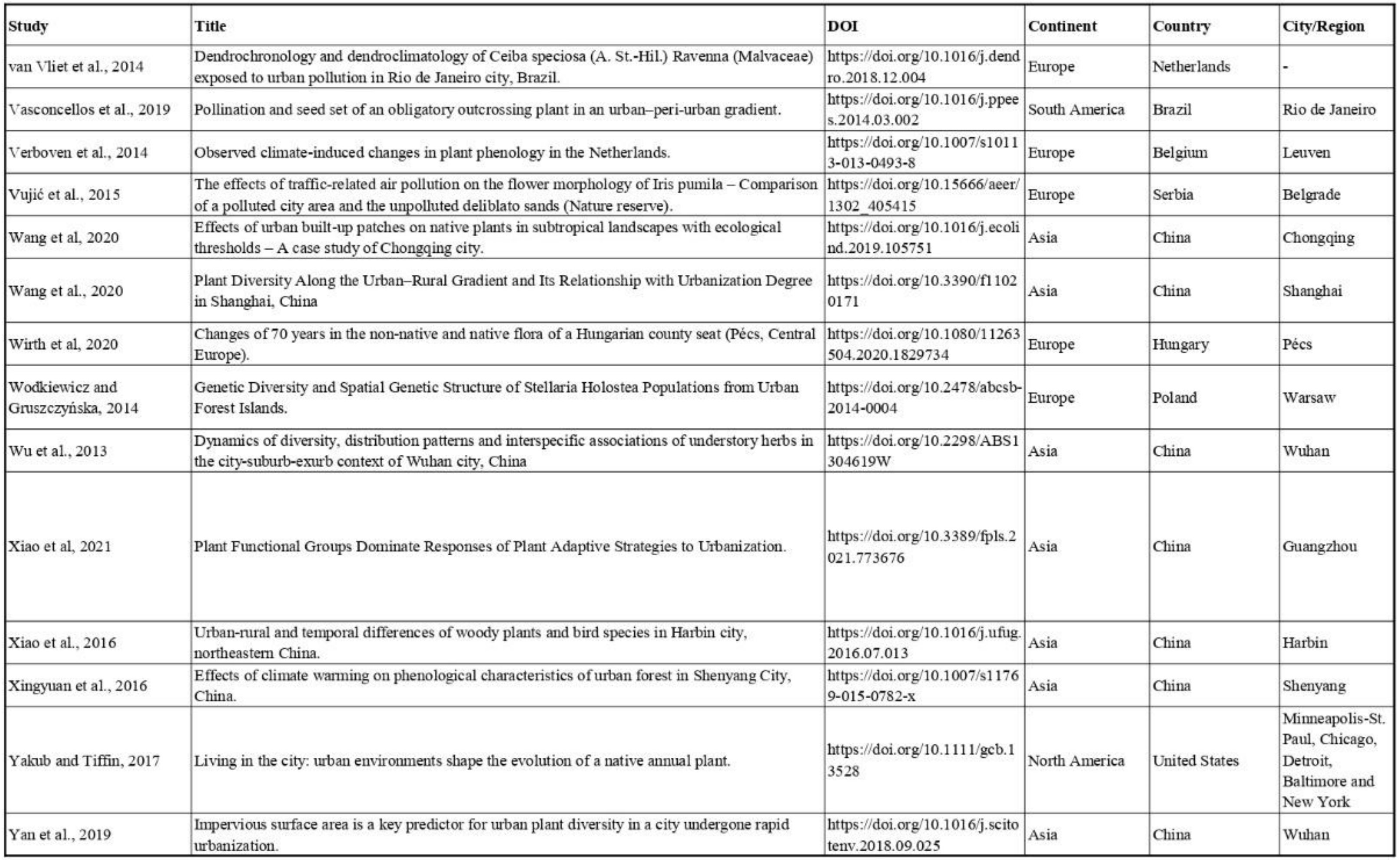
| Study | Physical environment | Morphological traits | Physio-chemical traits | Reproductive biology | Other traits |
|---|---|---|---|---|---|
| Skultety and Matthews, 2017 | Land cover; road type | - | - | - | - |
| Sołtysiak, 2020 | Temperature | - | - | - | - |
| Song et al., 2019 | Soil properties | Diameter at breast height; leaf area | Leaf carbon, nitrogen and phosphorus; photosynthetic rate | - | - |
| Sonti et al, 2021 | - | Diameter at breast height, stomatal density | Chlorophyll content, foliar nutrients | - | - |
| Štajerová et al., 2017 | Land use type; habitat richness | - | - | - | - |
| Su et al, 2021 | - | Stomatal density, leaf width, leaf lenght, specific leaf area | Leaf N, P and K concentration | - | - |
| Su et al., 2019 | Impervious cover; temperature | Length of needles | Phenology (leaf and male cones) | - | - |
| Tew et al, 2021 | - | - | - | Floral abundance, nectar sugar production | - |
| Thompson et al., 2016 | Ground temperature; historical weather data | - | Defensive phenotype (HCN); biomass | Reproductive fitness | - |

| Study | Title | DOI | Continent | Country | City/Region |
|---|---|---|---|---|---|
| Tian et al., 2015 | Distribution patterns and traits of weed communities along an urban–rural gradient under rapid urbanization in Shanghai, China. | https://doi.org/10.1111/wbm.12062 | Asia | China | Shanghai |
| Toma et al., 2015 | Factors affecting the genetic diversity of a perennial herb Viola grypoceras A. Gray var. grypoceras in urban fragmented forests. | https://doi.org/10.1007/s10980-015-0197-1 | Asia | Japan | Kyoto |
| Tordoni et al., 2017 | Diversity patterns of alien and native plant species in Trieste port area: exploring the role of urban habitats in biodiversity conservation. | https://doi.org/10.1007/s11252-017-0667-0 | Europe | Italy | Trieste |
| Trammel et al., 2020 | Temperate deciduous forests embedded across developed landscapes: Younger forests harbour invasive plants and urban forests maintain native plants. | https://doi.org/10.1111/1365-2745.13400 | North America | United States | Northern Delaware and southeastern Pennsylvania |
| Trentanovi et al., 2013 | Biotic homogenization at the community scale: disentangling the roles of urbanization and plant invasion. | https://doi.org/10.1111/ddi.12028 | Europe | Germany | Berlin |
| Turmukhametova and Shadrina, 2020 | Changes in the Fluctuating Asymmetry of the Leaf and Reproductive Capacity of Betula pendula Roth Reflect Pessimization of Anthropogenically Transformed Environment | https://doi.org/10.3390/sym12121970 | Europe | Russia | Yoshkar-Ola |
| Ushimaru et al., 2014 | Data from: Does urbanization promote floral diversification? Implications from changes in herkogamy with pollinator availability in an urban-rural area. | https://doi.org/10.5061/dryad.pd775 | Asia | Japan | Ikoma |
| Vakhlamova et al., 2014 | Changes in plant diversity along an urban–rural gradient in an expanding city in Kazakhstan, Western Siberia. | https://doi.org/10.1016/j.landurbplan.2014.08.014 | Asia | Kazakhstan | Pavlodar |
| Vakhlamova et al., 2016 | Effects of road type and urbanization on the diversity and abundance of alien species in roadside verges in Western Siberia. | https://doi.org/10.1007/s11258-016-0565-1 | Asia | Kazakhstan | Pavlodar |
| Vallet et al., 2010a | The effects of urban or rural landscape context and distance from the edge on native woodland plant communities. | https://doi.org/10.1007/s10531-010-9901-2 | Europe | France | Angers and Nantes |
| Vallet et al., 2010b | Using biological traits to assess how urbanization filters plant species of small woodlands. | https://doi.org/10.1111/j.1654-109X.2010.01087.x | Europe | France | Angers, Nantes and Rennes |
| Van Rossum, 2010 | Reproductive success and pollen dispersal in urban populations of an insect-pollinated hay-meadow herb | https://doi.org/10.1016/j.ppees.2009.08.002 | Europe | Belgium | Brussels |

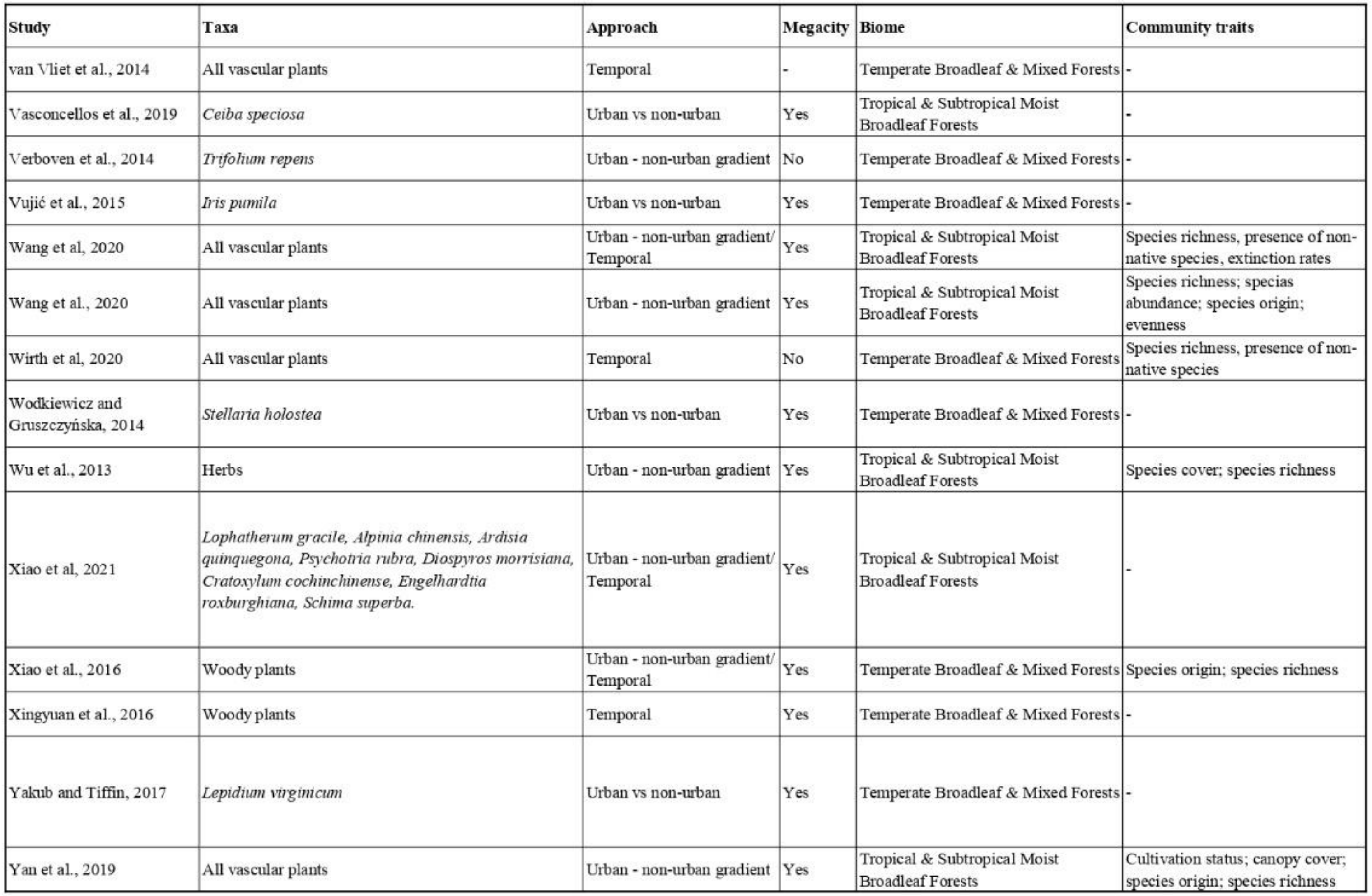


| Study | Taxa | Approach | Megacity | Biome | Community traits |
|---|---|---|---|---|---|
| Tian et al., 2015 | All vascular weedy plants | Urban - non-urban gradient | Yes | Tropical & Subtropical Moist Broadleaf Forests | Species origin; species richness; relative dominance of species |
| Toma et al., 2015 | *Viola grypoceras* A. Gray *var. grypoceras* | Urban vs non-urban | No | Temperate Broadleaf & Mixed Forests | - |
| Tordoni et al., 2017 | All vascular plants | Urban - non-urban gradient | No | Mediterranean Forests, Woodlands & Scrub | Species origin; species richness; abundance |
| Trammel et al., 2020 | Woody plants | Urban - non-urban gradient/ Temporal | - | Temperate Broadleaf & Mixed Forests | Species origin; population density |
| Trentanovi et al., 2013 | All vascular plants | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | Canopy cover; species origin; species richness |
| Turmukhametova and Shadrina, 2020 | *Betula pendula* | Urban - non-urban gradient/ Temporal | No | Boreal Forests/Taiga | - |
| Ushimaru et al., 2014 | *Commelina communis* | Urban - non-urban gradient | No | Temperate Broadleaf & Mixed Forests | - |
| Vakhlamova et al., 2014 | All vascular plants | Urban - non-urban gradient | No | Temperate Grasslands, Savannas & Shrublands | Species richness; species abundance; canopy cover; Evenness; species origin |
| Vakhlamova et al., 2016 | All vascular plants | Urban - non-urban gradient | No | Temperate Grasslands, Savannas & Shrublands | Species richness; species abundance; canopy cover; species origin |
| Vallet et al., 2010a | Woody plants | Urban vs non-urban | No | Temperate Broadleaf & Mixed Forests | Species richness; species abundance; species origin |
| Vallet et al., 2010b | All vascular plants | Urban - non-urban gradient | No | Temperate Broadleaf & Mixed Forests | - |
| Van Rossum, 2010 | *Centaurea jacea* | Urban vs non-urban | Yes | Temperate Broadleaf & Mixed Forests | - |

| Study | Physical environment | Morphological traits | Physio-chemical traits | Reproductive biology | Other traits |
|---|---|---|---|---|---|
| Tian et al., 2015 | Land use type; light intensity; soil properties | Height | Life form | - | - |
| Toma et al., 2015 | Area of urban forests; distance between urban forests | - | - | - | DNA Molecular markers (SSR); genetic diversity |
| Tordoni et al., 2017 | - | - | - | - | - |
| Trammel et al., 2020 | Percentage of impervious surface | Diameter at breast height | - | - | Temporal aerial images |
| Trentanovi et al., 2013 | Land use type | - | - | - | - |
| Turmukhametova and Shadrina, 2020 | Average score of anthropogenic impact | - | - | Seed quality; infructescence length, width, ratio length/thickness | - |
| Ushimaru et al., 2014 | Land use type | - | - | Pollinator frequency; flower abundance; reproductive success; heights of the stigma and anther; lenght of petal; P:O ratio | - |
| Vakhlamova et al., 2014 | Land use type | - | - | - | - |
| Vakhlamova et al., 2016 | - | - | - | Seed bank; dispersal type | - |
| Vallet et al., 2010a | Ellenberg indicator values for light, soil pH and nitrogen | - | - | - | - |
| Vallet et al., 2010b | Ellenberg values for light, soil humidity, soil basicity, soil fertility | Canopy height; leaf position; specific leaf area; shoot growth form | Leaf persistence; Plant life span | Aerial vegetative multiplication; underground vegetative multiplication; first flowering; length of flowering period; start month of seed shedding period; length of seed shedding; seed number; mean weight of germinule; seed bank longevity; pollination; fruit dispersal | - |
| Van Rossum, 2010 | - | - | - | Floral display; number of mature seeds; pollen dispersal patterns | - |



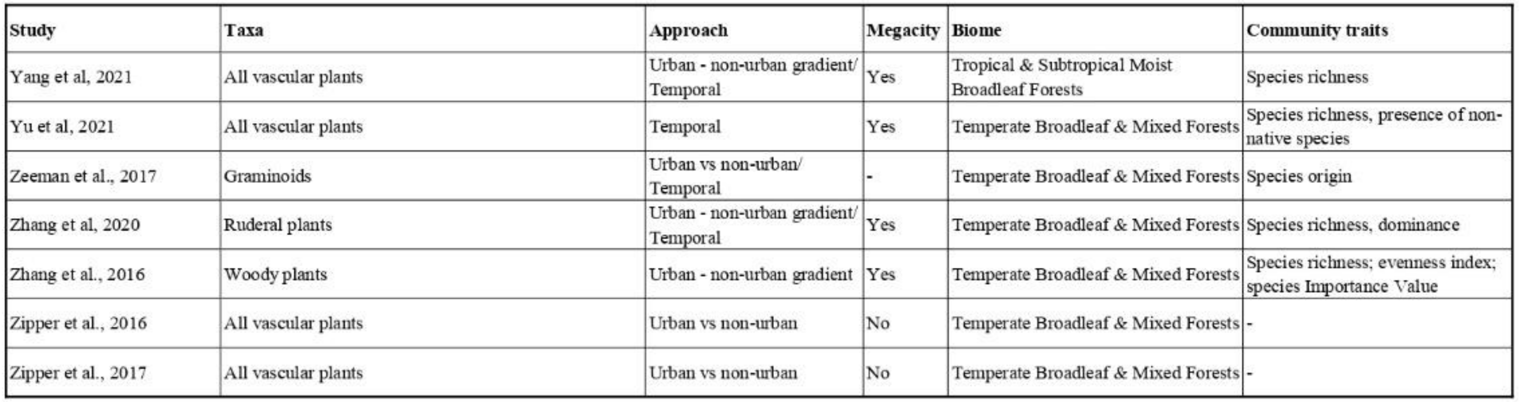
| Study | Title | DOI | Continent | Country | City/Region |
|---|---|---|---|---|---|
| van Vliet et al., 2014 | Dendrochronology and dendroclimatology of Ceiba speciosa (A. St.-Hil.) Ravenna (Malvaceae) exposed to urban pollution in Rio de Janeiro city, Brazil. | https://doi.org/10.1016/j.dendro.2018.12.004 | Europe | Netherlands | - |
| Vasconcellos et al., 2019 | Pollination and seed set of an obligatory outcrossing plant in an urban–peri-urban gradient. | https://doi.org/10.1016/j.ppees.2014.03.002 | South America | Brazil | Rio de Janeiro |
| Verboven et al., 2014 | Observed climate-induced changes in plant phenology in the Netherlands. | https://doi.org/10.1007/s10113-013-0493-8 | Europe | Belgium | Leuven |
| Vujić et al., 2015 | The effects of traffic-related air pollution on the flower morphology of Iris pumila – Comparison of a polluted city area and the unpolluted deliblato sands (Nature reserve). | https://doi.org/10.15666/aeer/1302_405415 | Europe | Serbia | Belgrade |
| Wang et al, 2020 | Effects of urban built-up patches on native plants in subtropical landscapes with ecological thresholds – A case study of Chongqing city. | https://doi.org/10.1016/j.ecolind.2019.105751 | Asia | China | Chongqing |
| Wang et al., 2020 | Plant Diversity Along the Urban–Rural Gradient and Its Relationship with Urbanization Degree in Shanghai, China | https://doi.org/10.3390/f11020171 | Asia | China | Shanghai |
| Wirth et al, 2020 | Changes of 70 years in the non-native and native flora of a Hungarian county seat (Pécs, Central Europe). | https://doi.org/10.1080/11263504.2020.1829734 | Europe | Hungary | Pécs |
| Wodkiewicz and Gruszczyńska, 2014 | Genetic Diversity and Spatial Genetic Structure of Stellaria Holostea Populations from Urban Forest Islands. | https://doi.org/10.2478/abcsb-2014-0004 | Europe | Poland | Warsaw |
| Wu et al., 2013 | Dynamics of diversity, distribution patterns and interspecific associations of understory herbs in the city-suburb-exurb context of Wuhan city, China | https://doi.org/10.2298/ABS1304619W | Asia | China | Wuhan |
| Xiao et al, 2021 | Plant Functional Groups Dominate Responses of Plant Adaptive Strategies to Urbanization. | https://doi.org/10.3389/fpls.2021.773676 | Asia | China | Guangzhou |
| Xiao et al., 2016 | Urban-rural and temporal differences of woody plants and bird species in Harbin city, northeastern China. | https://doi.org/10.1016/j.ufug.2016.07.013 | Asia | China | Harbin |
| Xingyuan et al., 2016 | Effects of climate warming on phenological characteristics of urban forest in Shenyang City, China. | https://doi.org/10.1007/s11769-015-0782-x | Asia | China | Shenyang |
| Yakub and Tiffin, 2017 | Living in the city: urban environments shape the evolution of a native annual plant. | https://doi.org/10.1111/gcb.13528 | North America | United States | Minneapolis-St. Paul, Chicago, Detroit, Baltimore and New York |
| Yan et al., 2019 | Impervious surface area is a key predictor for urban plant diversity in a city undergone rapid urbanization. | https://doi.org/10.1016/j.scitotenv.2018.09.025 | Asia | China | Wuhan |



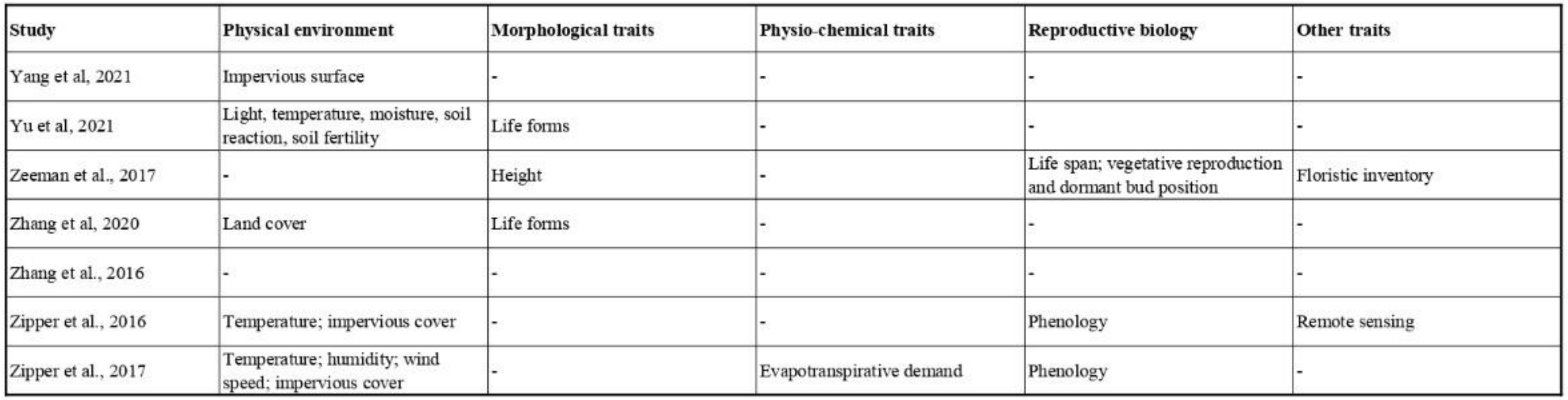
| Study | Taxa | Approach | Megacity | Biome | Community traits |
|---|---|---|---|---|---|
| van Vliet et al., 2014 | All vascular plants | Temporal | - | Temperate Broadleaf & Mixed Forests | - |
| Vasconcellos et al., 2019 | *Ceiba speciosa* | Urban vs non-urban | Yes | Tropical & Subtropical Moist Broadleaf Forests | - |
| Verboven et al., 2014 | *Trifolium repens* | Urban - non-urban gradient | No | Temperate Broadleaf & Mixed Forests | - |
| Vujić et al., 2015 | *Iris pumila* | Urban vs non-urban | Yes | Temperate Broadleaf & Mixed Forests | - |
| Wang et al, 2020 | All vascular plants | Urban - non-urban gradient/ Temporal | Yes | Tropical & Subtropical Moist Broadleaf Forests | Species richness, presence of non-native species, extinction rates |
| Wang et al., 2020 | All vascular plants | Urban - non-urban gradient | Yes | Tropical & Subtropical Moist Broadleaf Forests | Species richness; specias abundance; species origin; evenness |
| Wirth et al, 2020 | All vascular plants | Temporal | No | Temperate Broadleaf & Mixed Forests | Species richness, presence of non-native species |
| Wodkiewicz and Gruszczyńska, 2014 | *Stellaria holostea* | Urban vs non-urban | Yes | Temperate Broadleaf & Mixed Forests | - |
| Wu et al., 2013 | Herbs | Urban - non-urban gradient | Yes | Tropical & Subtropical Moist Broadleaf Forests | Species cover; species richness |
| Xiao et al, 2021 | *Lophatherum gracile, Alpinia chinensis, Ardisia quinquegona, Psychotria rubra, Diospyros morrisiana, Cratoxylum cochinchinense, Engelhardtia roxburghiana, Schima superba.* | Urban - non-urban gradient/ Temporal | Yes | Tropical & Subtropical Moist Broadleaf Forests | - |
| Xiao et al., 2016 | Woody plants | Urban - non-urban gradient/ Temporal | Yes | Temperate Broadleaf & Mixed Forests | Species origin; species richness |
| Xingyuan et al., 2016 | Woody plants | Temporal | Yes | Temperate Broadleaf & Mixed Forests | - |
| Yakub and Tiffin, 2017 | *Lepidium virginicum* | Urban vs non-urban | Yes | Temperate Broadleaf & Mixed Forests | - |
| Yan et al., 2019 | All vascular plants | Urban - non-urban gradient | Yes | Tropical & Subtropical Moist Broadleaf Forests | Cultivation status; canopy cover; species origin; species richness |

| Study | Physical environment | Morphological traits | Physio-chemical traits | Reproductive biology | Other traits |
|---|---|---|---|---|---|
| van Vliet et al., 2014 | Temperature | - | Phenology (leaf unfolding; autumn colouring; leaf fall) | Phenology (flowering; fruits ripe) | - |
| Vasconcellos et al., 2019 | Temperature; precipitation | Growth rings | - | - | Ages and rates of growth of the species |
| Verboven et al., 2014 | Land use type | - | - | Flower visitation rates; seed set | - |
| Vujić et al., 2015 | - | - | - | Length, width, centroid size and shape variation of petals | - |
| Wang et al, 2020 | Land use | Life forms | - | - | - |
| Wang et al., 2020 | Land use | - | - | - | - |
| Wirth et al, 2020 | - | Life forms | - | Dispersal type | - |
| Wodkiewicz and Gruszczyńska, 2014 | - | - | - | - | Allozymes; spatial genetic structure; genetic variability; clonal diversity |
| Wu et al., 2013 | - | Height | - | - | - |
| Xiao et al, 2021 | - | Leaf thickness and surface area | C, N and P content in leaves, concentrations of heavy metals, concentrations of starch in the leaves, stomatal conductance, photosynthetic rate, chlorophyll content, Rubisco content, dry biomass | - | - |
| Xiao et al., 2016 | - | - | - | - | - |
| Xingyuan et al., 2016 | Temperature | - | Phenology (germination) | - | - |
| Yakub and Tiffin, 2017 | - | Rosette height; rosette diameter; height and number of branches at the time of senescence | Phenology (germination; bolting) | Phenology (flowering); number of fruits | Genotype-by-sequencing; molecular divergence among populations; structuring of genomic variation |
| Yan et al., 2019 | Percentage of total impervious surface area | - | - | - | - |

| Study | Title | DOI | Continent | Country | City/Region |
|---|---|---|---|---|---|
| Yang et al, 2021 | Impacts of the remnant sizes, forest types, and landscape patterns of surrounding areas on woody plant diversity of urban remnant forest patches. | https://doi.org/10.1007/s11252-020-01040-z | Asia | China | Guiyang |
| Yu et al, 2021 | Dynamic change of composition and functions of flora adapting to rapid urbanization: a case study of Hangzhou, China | https://doi.org/10.15666/aeer/1905_40074023 | Asia | China | Hangzhou |
| Zeeman et al., 2017 | Biotic homogenization in an increasingly urbanized temperate grassland ecosystem. | https://doi.org/10.1111/jvs.12507 | Oceania | Australia | Victoria |
| Zhang et al, 2020 | The Diversity Distribution Pattern of Ruderal Community under the Rapid Urbanization in Hangzhou, East China | https://doi.org/10.3390/d12030116 | Asia | China | Hangzhou |
| Zhang et al., 2016 | Effects of forest type and urbanization on species composition and diversity of urban forest in Changchun, Northeast China. | https://doi.org/10.1007/s11252-015-0473-5 | Asia | China | Changchun |
| Zipper et al., 2016 | Urban heat island impacts on plant phenology: intra-urban variability and response to land cover. | https://doi.org/10.1088/1748-9326/11/5/054023 | North America | United States | Madison |
| Zipper et al., 2017 | Urban heat island-induced increases in evapotranspirative demand. | https://doi.org/10.1002/2016GL072190 | North America | United States | Madison |



| Study | Taxa | Approach | Megacity | Biome | Community traits |
|---|---|---|---|---|---|
| Yang et al, 2021 | All vascular plants | Urban - non-urban gradient/ Temporal | Yes | Tropical & Subtropical Moist Broadleaf Forests | Species richness |
| Yu et al, 2021 | All vascular plants | Temporal | Yes | Temperate Broadleaf & Mixed Forests | Species richness, presence of non-native species |
| Zeeman et al., 2017 | Graminoids | Urban vs non-urban/ Temporal | - | Temperate Broadleaf & Mixed Forests | Species origin |
| Zhang et al, 2020 | Ruderal plants | Urban - non-urban gradient/ Temporal | Yes | Temperate Broadleaf & Mixed Forests | Species richness, dominance |
| Zhang et al., 2016 | Woody plants | Urban - non-urban gradient | Yes | Temperate Broadleaf & Mixed Forests | Species richness; evenness index; species Importance Value |
| Zipper et al., 2016 | All vascular plants | Urban vs non-urban | No | Temperate Broadleaf & Mixed Forests | - |
| Zipper et al., 2017 | All vascular plants | Urban vs non-urban | No | Temperate Broadleaf & Mixed Forests | - |



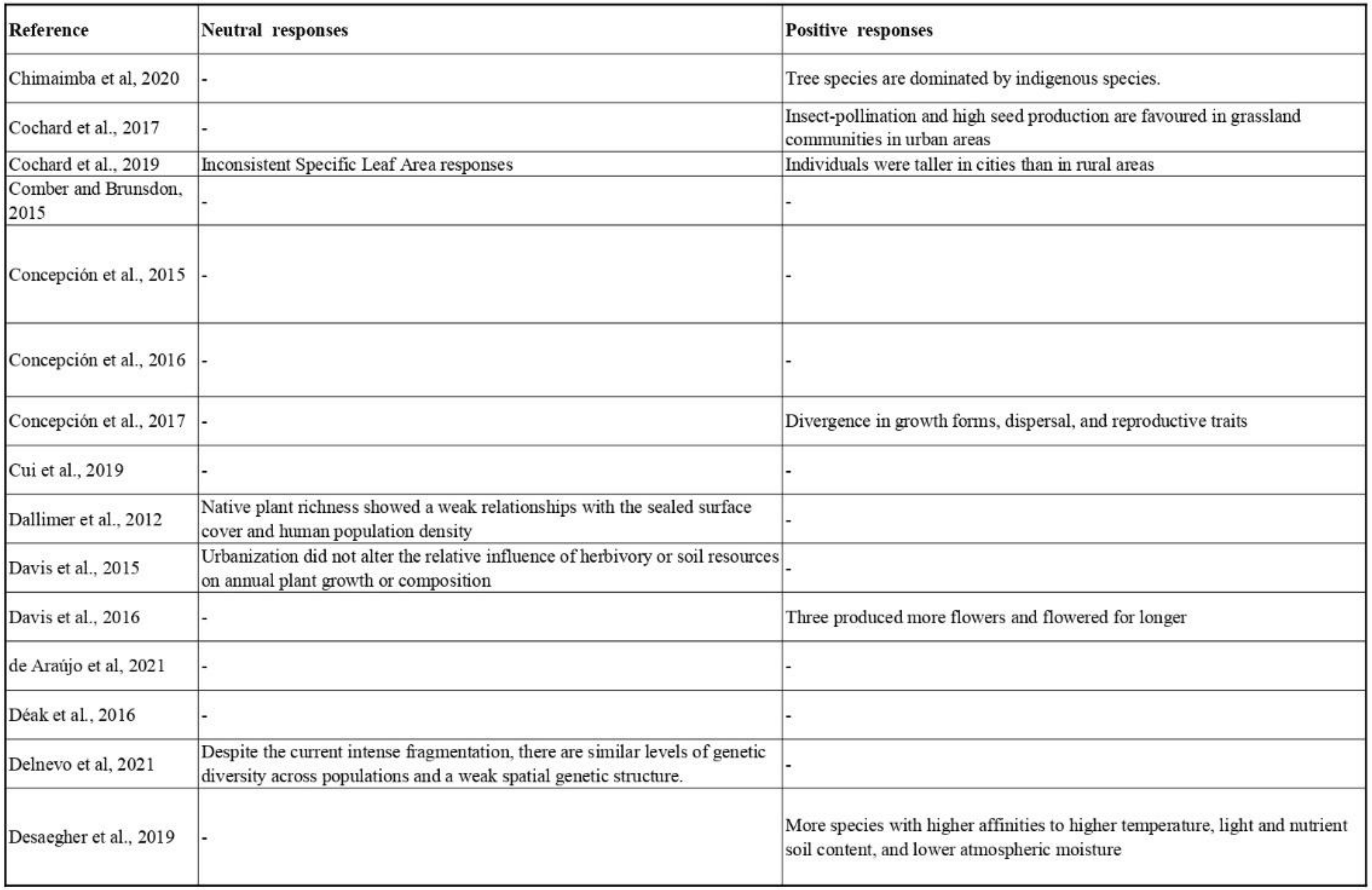
| Study | Physical environment | Morphological traits | Physio-chemical traits | Reproductive biology | Other traits |
|---|---|---|---|---|---|
| Yang et al, 2021 | Impervious surface | - | - | - | - |
| Yu et al, 2021 | Light, temperature, moisture, soil reaction, soil fertility | Life forms | - | - | - |
| Zeeman et al., 2017 | - | Height | - | Life span; vegetative reproduction and dormant bud position | Floristic inventory |
| Zhang et al, 2020 | Land cover | Life forms | - | - | - |
| Zhang et al., 2016 | - | - | - | - | - |
| Zipper et al., 2016 | Temperature; impervious cover | - | - | Phenology | Remote sensing |
| Zipper et al., 2017 | Temperature; humidity; wind speed; impervious cover | - | Evapotranspirative demand | Phenology | - |

**Supplementary Table 2.** Plants' responses to urbanization according to the analyzed studies.

| Reference | Taxa | Urban drivers | Negative responses |
|---|---|---|---|
| Abd El-Wahab, 2016 | All vascular plants | LC | Lower species richness; lower plant cover; lower species diversity |
| Albrecht and Haider, 2013 | Graminoids | CL/ LC | Lower native species richness; lower numbers of endangered species; earlier flowering; disfavoring of windpollinated species |
| Anderson et al., 2020 | All vascular plants | HA/ LC/ UH | Lower species richness; lower plant cover; lower plant trait diversity; grass domination; less species that reward animals through pollination and seed dispersal; poorer communities have limited green space and lower plant and trait diversity |
| Aronson et al., 2015 | Woody plants | BI/ LC | Lower native species richness; increase of non-native species |
| Atasoy et al., 2018 | *Ligustrum sinense, Elaeagnus pungens, Triadica sebifera* | BI/ LC | Cover of two of the invasive species tended to increase |
| Barradas and Esperon-Rodriguez, 2021 | *Acacia longifólia, Acer negundo, Alnus acuminata, Buddleja cordata, Celtis occidentalis, Dodonaea viscosa, Eucalyptus camaldulensis, Erythrina amerciana, Fraxinus uhdei, Ligustrum lucidum, Liquidambar styraciflua, Populus alba, Populus deltoides, Quercus rugosa, Ulmus parvifolia* | CL | Higher temperatures and vapor pressure deficit limited the stomatal function of all species |
| Bartlewicz et al., 2015 | *Linaria vulgaris* | LC | Decrease of the clonal diversity; less ability to reproduce sexually |
| Beal-Neves et al., 2020 | Graminoids | HA | - |
| Bigirimana et al., 2011 | All vascular plants | BI/ HA/ LC | Lower native species richness; ruderal plants with a high abundance of introduced species |
| Blood et al., 2016 | Woody plants | BI | Increase species richness by greater numbers of non-natives |
| Blouin et al., 2019 | All vascular plants | BI/ LC | Biotic homogenization occurr resulted from non-native species invasion |
| Bode and Tong, 2018 | *Cytisus scoparius* | BI | Pollinators select increased flower size, but only in urban populations |
| Bouraoui et al., 2019 | *Tilia* x *vulgaris* | PO | De-icing salt accumulation increases leaf injury, cascading up through the trophic chain and negatively affecting also the associated arthropod biota |
| Brice et al., 2016 | All vascular plants | LC | Land transformation act indirectly on hydrological disturbances altering functional composition of riparian plant communities |
| Brice et al., 2017 | All vascular plants | LC | Urbanization induced taxonomic and functional differentiation through its effect on species loss and turnover |
| Caughlin et al., 2012 | *Ficus microcarpa, Ficus aurea* | BI | Seed-dispersing birds favours juvenile invasive fig abundance |
| Celesti-Grapow and Ricotta, 2021 | All vascular plants | BI | The diversity of the native flora has steadily decreased, while there has been na increase in non-native, larger and more damaging species. |
| Chen et al., 2014 | All vascular ruderal plants | BI/ CL | Increase of tropical originated species; decrease in species richness due to urbanization and climate change along the years |

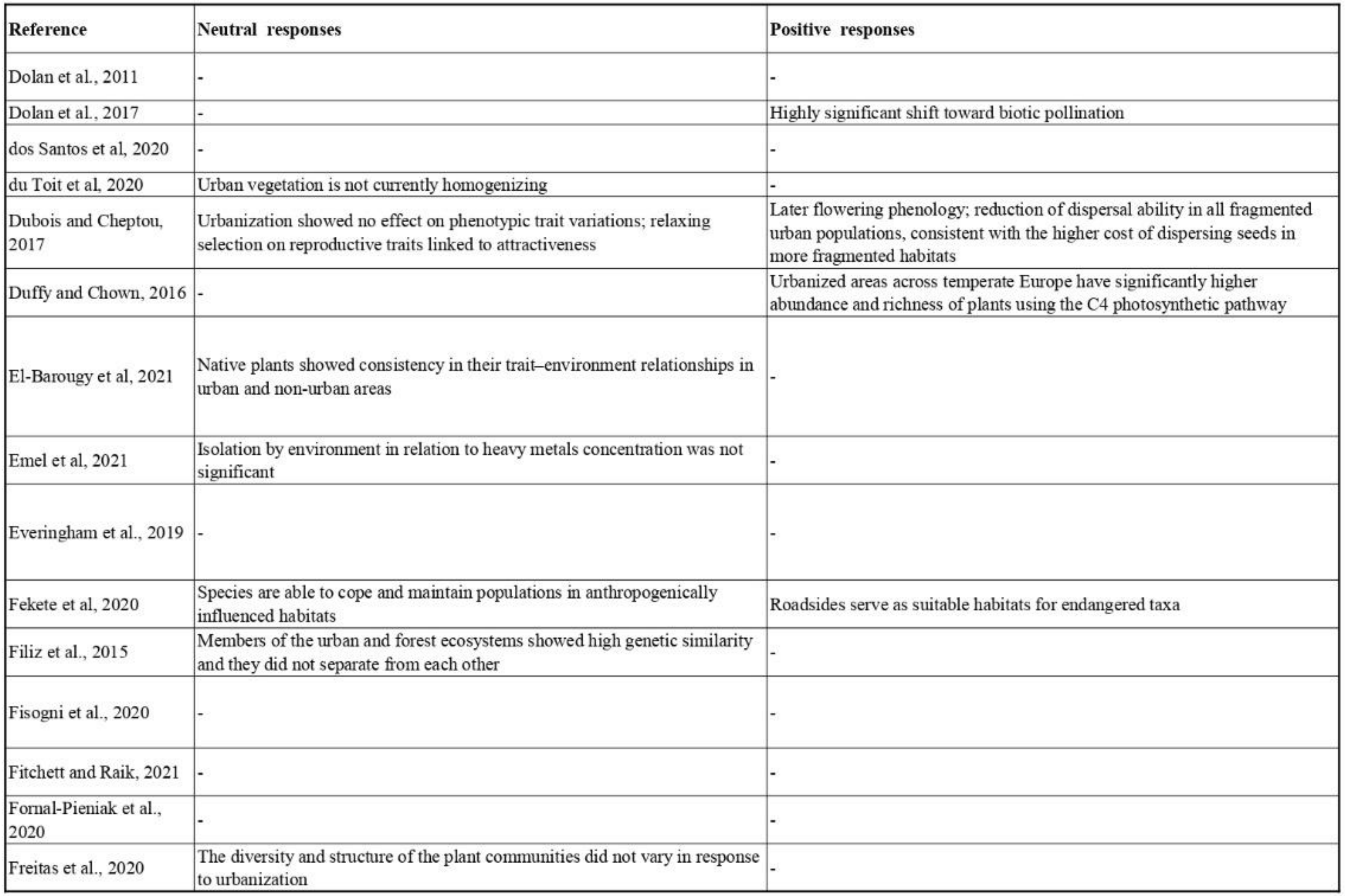
| Reference | Neutral responses | Positive responses |
|---|---|---|
| Abd El-Wahab, 2016 | - | - |
| Albrecht and Haider, 2013 | - | Favoring of plants with insect pollination; more plants with small and durable seeds; some of the endangered species exclusively grew at the urban sites |
| Anderson et al., 2020 | - | - |
| Aronson et al., 2015 | Urbanization did not advance the biotic homogenization | - |
| Atasoy et al., 2018 | - | - |
| Barradas and Esperon-Rodriguez, 2021 | - | - |
| Bartlewicz et al., 2015 | - | - |
| Beal-Neves et al., 2020 | - | Higher species richness; higher floral visitor richness; higher network asymmetry |
| Bigirimana et al., 2011 | - | - |
| Blood et al., 2016 | Maintenance of native species richness | - |
| Blouin et al., 2019 | - | - |
| Bode and Tong, 2018 | - | - |
| Bouraoui et al., 2019 | - | - |
| Brice et al., 2016 | - | - |
| Brice et al., 2017 | - | - |
| Caughlin et al., 2012 | - | - |
| Celesti-Grapow and Ricotta, 2021 | - | - |
| Chen et al., 2014 | - | - |

| **Reference** | **Taxa** | **Urban drivers** | **Negative responses** |
|---|---|---|---|
| Chimaimba et al, 2020 | Trees | BI/ HA /LC | Residential areas, roads and institutional land have high proportion of exotic trees |
| Cochard et al., 2017 | Graminoids | LC | Lower species richness |
| Cochard et al., 2019 | Graminoids | CL/ PO | Early flowering for almost all urban populations of each species |
| Comber and Brunsdon, 2015 | *Crataegus monogyna, Prunus spinosa, Aesculus hippocastanum* | CL/ LC | First flowering date is getting earlier |
| Concepción et al., 2015 | All vascular plants | LC | Urbanization favours highly mobile species with narrow habitat ranges; the positive response of specialist plants to urbanisation was most likely driven by species within this group that prefer eutrophic habitats, such as early successional species |
| Concepción et al., 2016 | All vascular plants | BI/ HA/ LC | Increased proliferating non-native and ruderal plants; species richness of plants inhabiting eutrophic places, annual, highly dispersive, wind- and human dispersed plants are benefitted |
| Concepción et al., 2017 | All vascular plants | CL/ HA/ LC | Convergence in vegetative plant traits (plant height and leaf dry matter content) |
| Cui et al., 2019 | All vascular ruderal plants | BI/ LC | Phylogenetic clustering of ruderal species with increasing spatial and temporal urbanization gradients |
| Dallimer et al., 2012 | All vascular plants | BI/ LC | Neophyte richness increased closer to the urban core along all rivers |
| Davis et al., 2015 | Herbs | LC | - |
| Davis et al., 2016 | *Angophora costata, Eucalyptus pilularis, Corymbia gummifera* | CL | - |
| de Araújo et al, 2021 | All vascular plants | LC | Plant species richness decreased in anthropized plots resulting in responses in species richness and composition of gall-inducing insects |
| Déak et al., 2016 | Graminoids | BI/ LC | Lower species richness; higher ratio of weeds, disturbance-tolerants, drought-tolerant; large proportion of alien and cosmopolitan species |
| Delnevo et al, 2021 | Conospermum undulatum | LC | - |
| Desaegher et al., 2019 | All vascular plants | CL/ LC/ PO | Proportion of autogamous, self-compatible, and nonentomophilous species was significantly higher in urban plant communities; insect-pollinated species with long and narrow tubular corollas were disadvantaged in urban areas |

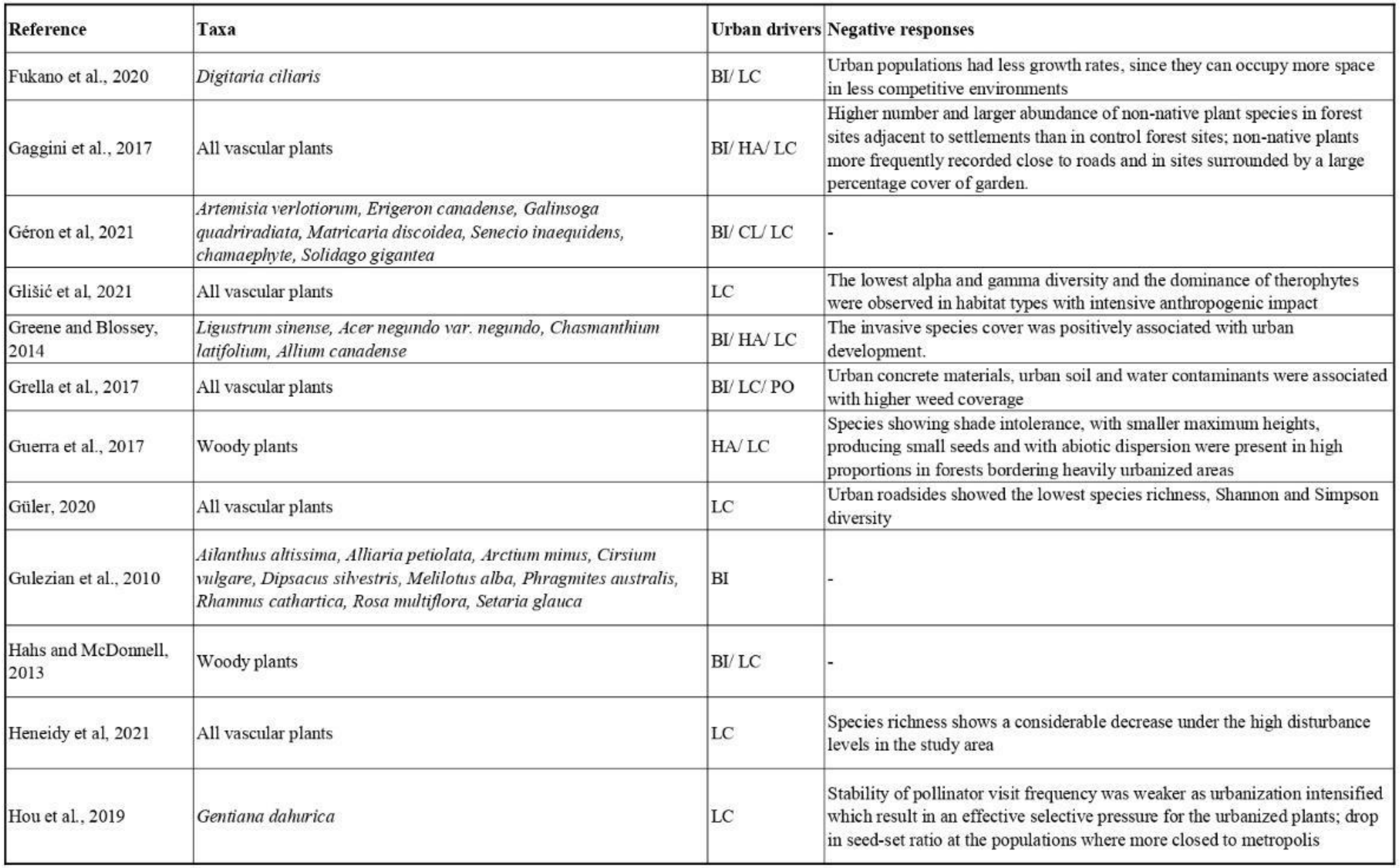


| Reference | Neutral responses | Positive responses |
|---|---|---|
| Chimaimba et al, 2020 | - | Tree species are dominated by indigenous species. |
| Cochard et al., 2017 | - | Insect-pollination and high seed production are favoured in grassland communities in urban areas |
| Cochard et al., 2019 | Inconsistent Specific Leaf Area responses | Individuals were taller in cities than in rural areas |
| Comber and Brunsdon, 2015 | - | - |
| Concepción et al., 2015 | - | - |
| Concepción et al., 2016 | - | - |
| Concepción et al., 2017 | - | Divergence in growth forms, dispersal, and reproductive traits |
| Cui et al., 2019 | - | - |
| Dallimer et al., 2012 | Native plant richness showed a weak relationships with the sealed surface cover and human population density | - |
| Davis et al., 2015 | Urbanization did not alter the relative influence of herbivory or soil resources on annual plant growth or composition | - |
| Davis et al., 2016 | - | Three produced more flowers and flowered for longer |
| de Araújo et al, 2021 | - | - |
| Déak et al., 2016 | - | - |
| Delnevo et al, 2021 | Despite the current intense fragmentation, there are similar levels of genetic diversity across populations and a weak spatial genetic structure. | - |
| Desaegher et al., 2019 | - | More species with higher affinities to higher temperature, light and nutrient soil content, and lower atmospheric moisture |

| Reference | Taxa | Urban drivers | Negative responses |
|---|---|---|---|
| Dolan et al., 2011 | All vascular plants | BI/ HA/ LC | Significant turnover in species presence; lower native species richness; increase in non-native plants |
| Dolan et al., 2017 | All vascular plants | HA/ LC | Natives species are different between the current and historical floras |
| dos Santos et al, 2020 | Tecoma stans | HA | Urban plants faced up to higher resource theft (nectar and pollen) and lower fruit set |
| du Toit et al, 2020 | All vascular plants | LC | Indigenous forb species richness is declining significantly |
| Dubois and Cheptou, 2017 | *Crepis sancta* | LC | - |
| Duffy and Chown, 2016 | C4 plants | CL | - |
| El-Barougy et al, 2021 | All vascular plants | BI/ LC | Significant relationships of alien species traits with human-made pressures and soil resources in urban areas; functional and phylogenetic distances declined between the aliens and natives in urban areas, indicating biotic homogenization with increasing urbanization |
| Emel et al, 2021 | Sabatia angularis | LC | High-intensity urbanization presents a major barrier between populations, resisting gene flow 10–200 times more than natural lands |
| Everingham et al., 2019 | All vascular plants | HA/ LC | Native species were disproportionately likely be generalist species; native species colonising urban habitats tended to have traits associated with a fast life-history, including short stature and small seeds; high number of native plant species are colonising anthropogenic habitats |
| Fekete et al, 2020 | Orchids | LC | There is a significant negative impact of agricultural and urban land covers on orchid diversity and abundance in general. |
| Filiz et al., 2015 | *Tilia tomentosa* | LC | - |
| Fisogni et al., 2020 | Herbs | LC | Plant flowering showed advancement of the flowering peak in sites at high urbanization; pollinator communities did not show any clear shift of their flight phenology |
| Fitchett and Raik, 2021 | *Jacaranda mimosifolia* | CL | An advance of flowering in 2.1 days per decade is calculated for the period 1927–2019 |
| Fornal-Pieniak et al., 2020 | All vascular plants | BI/ LC | More alien plant species were growing in the urban forests |
| Freitas et al., 2020 | Woody plants | LC | Species turnover |



| Reference | Neutral responses | Positive responses |
|---|---|---|
| Dolan et al., 2011 | - | - |
| Dolan et al., 2017 | - | Highly significant shift toward biotic pollination |
| dos Santos et al, 2020 | - | - |
| du Toit et al, 2020 | Urban vegetation is not currently homogenizing | - |
| Dubois and Cheptou, 2017 | Urbanization showed no effect on phenotypic trait variations; relaxing selection on reproductive traits linked to attractiveness | Later flowering phenology; reduction of dispersal ability in all fragmented urban populations, consistent with the higher cost of dispersing seeds in more fragmented habitats |
| Duffy and Chown, 2016 | - | Urbanized areas across temperate Europe have significantly higher abundance and richness of plants using the C4 photosynthetic pathway |
| El-Barougy et al, 2021 | Native plants showed consistency in their trait–environment relationships in urban and non-urban areas | - |
| Emel et al, 2021 | Isolation by environment in relation to heavy metals concentration was not significant | - |
| Everingham et al., 2019 | - | - |
| Fekete et al, 2020 | Species are able to cope and maintain populations in anthropogenically influenced habitats | Roadsides serve as suitable habitats for endangered taxa |
| Filiz et al., 2015 | Members of the urban and forest ecosystems showed high genetic similarity and they did not separate from each other | - |
| Fisogni et al., 2020 | - | - |
| Fitchett and Raik, 2021 | - | - |
| Fornal-Pieniak et al., 2020 | - | - |
| Freitas et al., 2020 | The diversity and structure of the plant communities did not vary in response to urbanization | - |

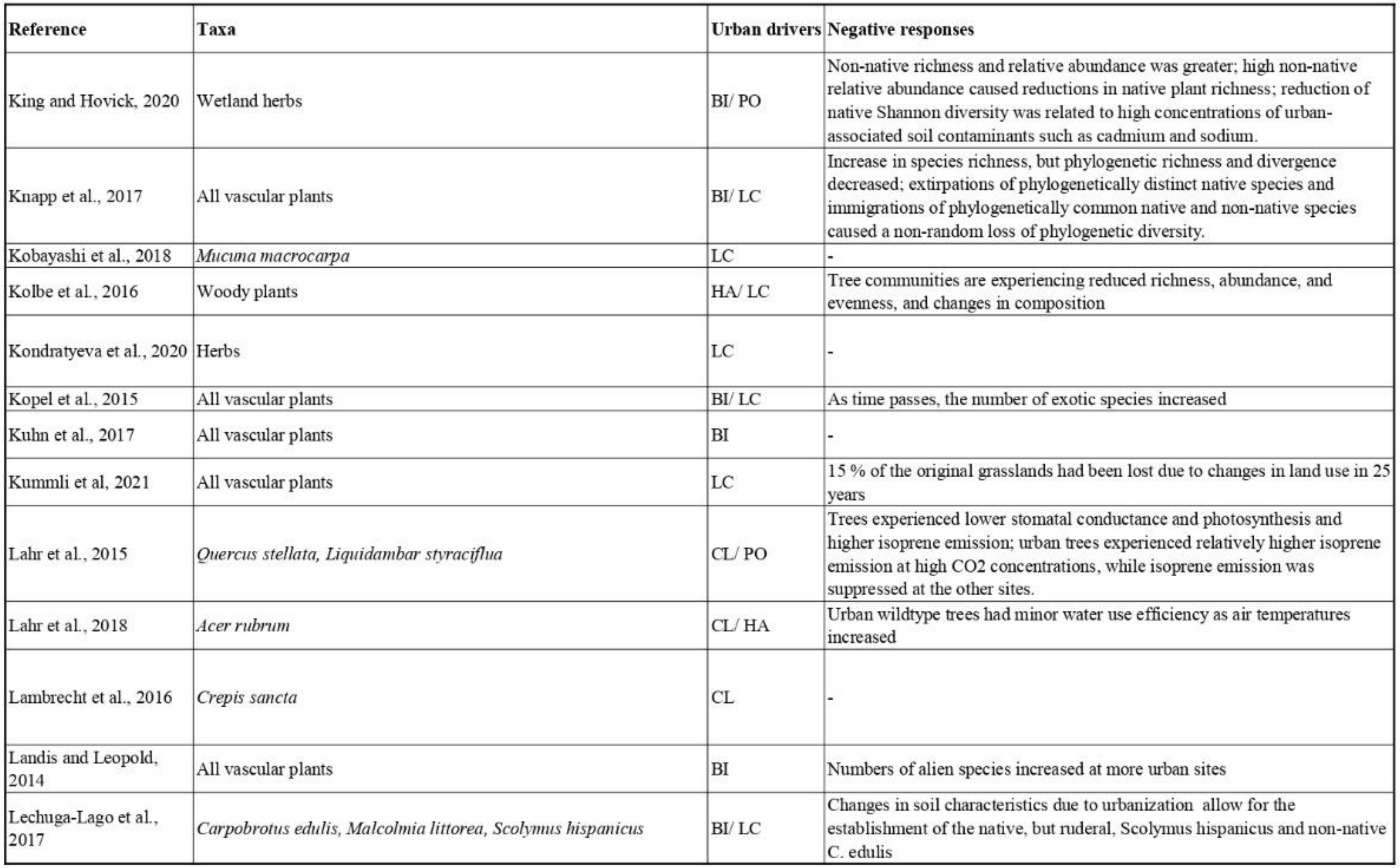


| **Reference** | **Taxa** | **Urban drivers** | **Negative responses** |
|---|---|---|---|
| Fukano et al., 2020 | *Digitaria ciliaris* | BI/ LC | Urban populations had less growth rates, since they can occupy more space in less competitive environments |
| Gaggini et al., 2017 | All vascular plants | BI/ HA/ LC | Higher number and larger abundance of non-native plant species in forest sites adjacent to settlements than in control forest sites; non-native plants more frequently recorded close to roads and in sites surrounded by a large percentage cover of garden. |
| Géron et al, 2021 | *Artemisia verlotiorum, Erigeron canadense, Galinsoga quadriradiata, Matricaria discoidea, Senecio inaequidens, chamaephyte, Solidago gigantea* | BI/ CL/ LC | - |
| Glišić et al, 2021 | All vascular plants | LC | The lowest alpha and gamma diversity and the dominance of therophytes were observed in habitat types with intensive anthropogenic impact |
| Greene and Blossey, 2014 | *Ligustrum sinense, Acer negundo var. negundo, Chasmanthium latifolium, Allium canadense* | BI/ HA/ LC | The invasive species cover was positively associated with urban development. |
| Grella et al., 2017 | All vascular plants | BI/ LC/ PO | Urban concrete materials, urban soil and water contaminants were associated with higher weed coverage |
| Guerra et al., 2017 | Woody plants | HA/ LC | Species showing shade intolerance, with smaller maximum heights, producing small seeds and with abiotic dispersion were present in high proportions in forests bordering heavily urbanized areas |
| Güler, 2020 | All vascular plants | LC | Urban roadsides showed the lowest species richness, Shannon and Simpson diversity |
| Gulezian et al., 2010 | *Ailanthus altissima, Alliaria petiolata, Arctium minus, Cirsium vulgare, Dipsacus silvestris, Melilotus alba, Phragmites australis, Rhamnus cathartica, Rosa multiflora, Setaria glauca* | BI | - |
| Hahs and McDonnell, 2013 | Woody plants | BI/ LC | - |
| Heneidy et al, 2021 | All vascular plants | LC | Species richness shows a considerable decrease under the high disturbance levels in the study area |
| Hou et al., 2019 | *Gentiana dahurica* | LC | Stability of pollinator visit frequency was weaker as urbanization intensified which result in an effective selective pressure for the urbanized plants; drop in seed-set ratio at the populations where more closed to metropolis |

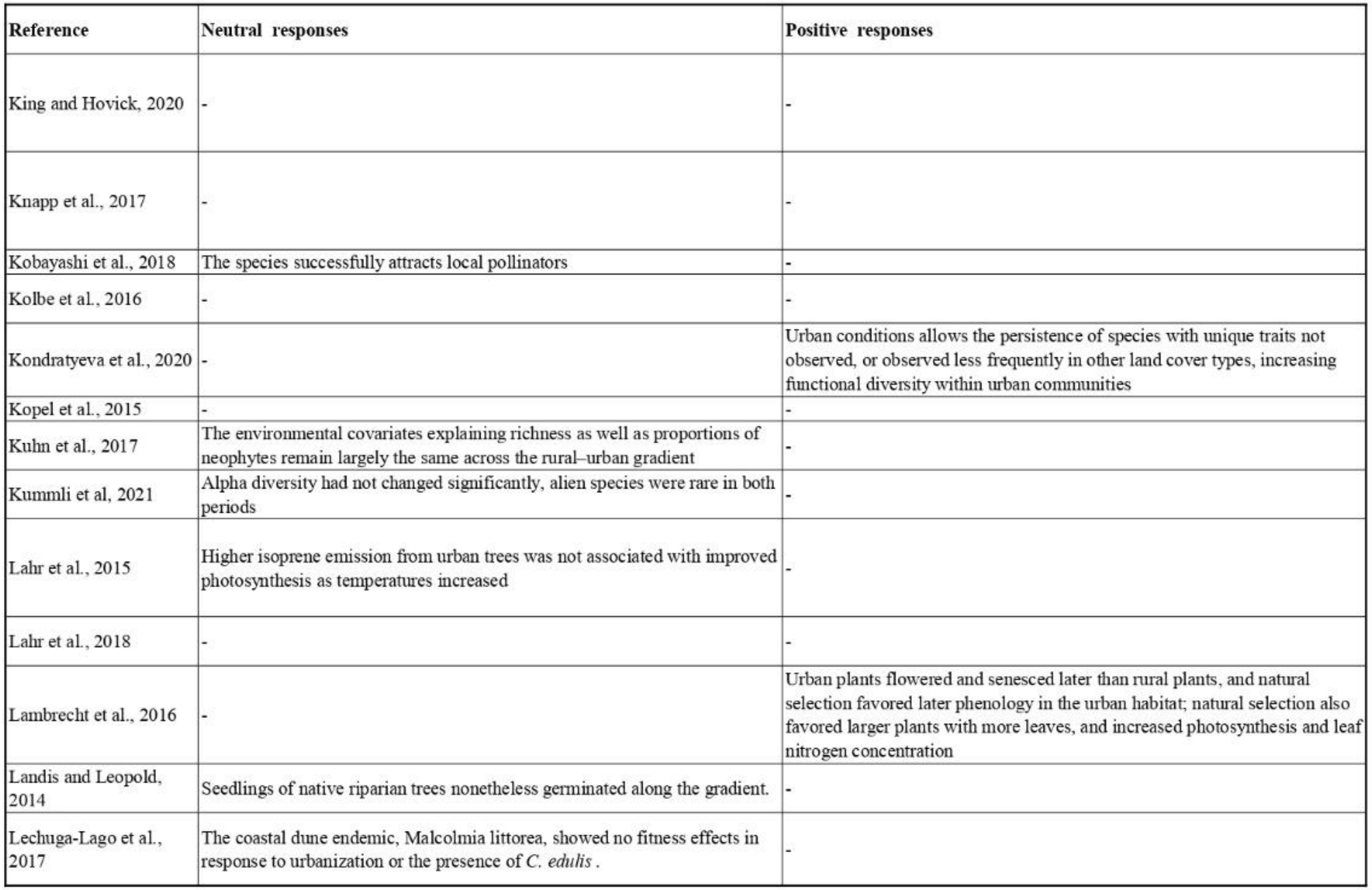
| Reference | Neutral responses | Positive responses |
|---|---|---|
| Fukano et al., 2020 | - | - |
| Gaggini et al., 2017 | - | - |
| Géron et al, 2021 | Alien plant phenotypic responses across all species did not strongly correlate with urbanity at local and landscape scale | - |
| Glišić et al, 2021 | - | - |
| Greene and Blossey, 2014 | - | - |
| Grella et al., 2017 | - | - |
| Guerra et al., 2017 | - | - |
| Güler, 2020 | Few alien species were found with only slightly increasing abundances due to disturbances. Considerable biotic homogenisation did not occur. | - |
| Gulezian et al., 2010 | - | Abundance of invasive species was negatively related to percent impervious surface, and presence was negatively related to local income |
| Hahs and McDonnell, 2013 | Species richness per plot of emergent seedlings was best explained by average annual rainfall, rather than the degree of urbanization in the surrounding landscape | Indigenous plant species can coexist with non-indigenous monocot species within urban areas |
| Heneidy et al, 2021 | - | Species richness shows a remarkable increase under moderate disturbance, coinciding with the increase in the numbers of both alien and weed species richness |
| Hou et al., 2019 | Urbanization did not change the pollination mode of the species | As urbanization intensified, plant borne more flowers and the flower size became “longer”; more pollen numbers, higher P/O ratio and longer floral duration at more urbanization-intensified populations |

| Reference | Taxa | Urban drivers | Negative responses |
|---|---|---|---|
| Huang et al., 2013 | All vascular plants | LC/ PO | Native herb diversity was lower in urban forests; soil heavy metal concentrations were negatively correlated with herb diversity; pioneer species and stress-tolerant species can survive and vigorously establish their population dominance in the urban environment |
| Ilyas et al, 2021 | *Platanus linn, Populus euramevicana, Cinnamomum camphora* | PO | - |
| Irwin et al., 2018 | *Gelsemium sempervirens* | LC | Urban landscape had increased florivory and decreased pollen receipt |
| Jensen et al, 2021 | *Acer platanoides, Betula pendula, Crataegus monogyna, Fagus sylvatica, Prunus avium, Quercus robur, Salix cáprea, Sorbus intermedia, Tilia x europaea, Ailanthus altíssima, Ginkgo biloba, Gleditsia triacanthos, Platanus x hispânica, Pterocarya fraxinifolia, Quercus cerris, Robinia pseudoacacia* | BI/ CL | In the urban environment, nonnative trees hosted significantly fewer invertebrates compared to native trees. Furthermore, the nonnative trees had a delayed phenology compared to native species |
| Jeong et al., 2011 | *Forsythia koreana, Rhododendron mucronulatum, Prunus yedoensis, Prunus persica* | CL | Urban warming has led to an advance in the timing of first-flowering of several days to weeks during recent decades, while the intrinsic physiology of plants to sense thermal energy has not been changed |
| Jochner et al., 2012 | *Corylus avellana, Galanthus nivalis, Salix caprea, Forsythia suspensa, Acer platanoides, Fagus sylvatica, Malus domestica, Picea abies, Syringa vulgaris* | CL/ LC | - |
| Jochner et al., 2013 | *Tabebuia chrysotricha, Caesalpinia peltophoroides, Tipuana tipu, Betula pendula* | CL/ LC | Phenological onset dates of Betula pendula were able to describe local temperature variations in Munich; phenology of Tipuana was sufficiently sensitive to detect urban heat island effects in Campinas |
| Jochner et al., 2015 | *Betula pendula, Corylus avellana, Aesculus hippocastanum* | CL/ LC/ PO | Increased ozone, NO2, NOx and PM levels were significantly related to delays in phenology. |
| Johnson et al., 2018 | *Trifolium repens* | LC/ UH | - |
| Kalusova et al., 2019 | All vascular plants | BI/ LC | The proportion of alien species increased in highly urbanized habitats; native species richness decreased with increasing disturbance |

| **Reference** | **Neutral responses** | **Positive responses** |
|---|---|---|
| Huang et al., 2013 | - | - |
| Ilyas et al, 2021 | - | The trees at different locations (urban - exurban) indicate a highly significant increase in functional traits analysed, proving that plasticity has a major role in plant adaptation to environmental changes |
| Irwin et al., 2018 | Pollination was associated with selection for larger floral display size, but there is little difference in phenotypes between urban and non-urban sites; the lack of strong difference between urban and non-urban sites in reproduction suggests no extreme maladaptation to the urban landscape in the year of study. | - |
| Jensen et al, 2021 | - | - |
| Jeong et al., 2011 | - | - |
| Jochner et al., 2012 | Trends in phenology did not change significantly with higher altitudes or urbanised areas. | - |
| Jochner et al., 2013 | There were no significant correlations with flowering phenophases of Tabebuia and Caesalpinia; | - |
| Jochner et al., 2015 | Measurements of foliar characteristics of birch were not suitable for bio-monitoring pollution | - |
| Johnson et al., 2018 | Populations did not exhibit changes in genetic diversity with increasing urbanization; populations frequently exhibited isolation-by-distance and extensive gene flow along most urban–rural transects | HCN frequency increased for every kilometre from an urban centre - urbanization repeatedly drives parallel evolution of an ecologically important trait across many cities that vary in size |
| Kalusova et al., 2019 | - | - |

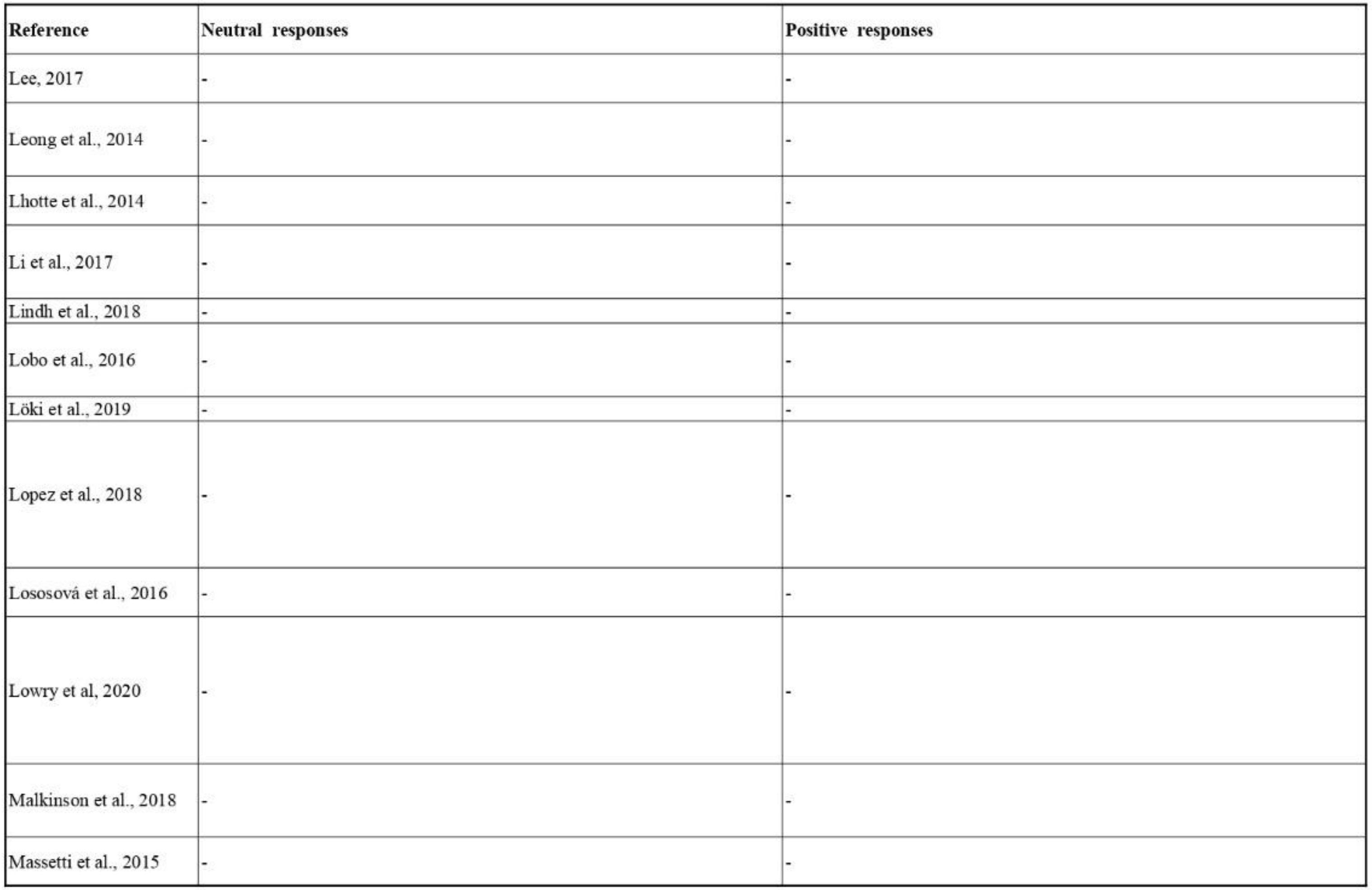


| Reference | Taxa | Urban drivers | Negative responses |
|---|---|---|---|
| King and Hovick, 2020 | Wetland herbs | BI/ PO | Non-native richness and relative abundance was greater; high non-native relative abundance caused reductions in native plant richness; reduction of native Shannon diversity was related to high concentrations of urban-associated soil contaminants such as cadmium and sodium. |
| Knapp et al., 2017 | All vascular plants | BI/ LC | Increase in species richness, but phylogenetic richness and divergence decreased; extirpations of phylogenetically distinct native species and immigrations of phylogenetically common native and non-native species caused a non-random loss of phylogenetic diversity. |
| Kobayashi et al., 2018 | *Mucuna macrocarpa* | LC | - |
| Kolbe et al., 2016 | Woody plants | HA/ LC | Tree communities are experiencing reduced richness, abundance, and evenness, and changes in composition |
| Kondratyeva et al., 2020 | Herbs | LC | - |
| Kopel et al., 2015 | All vascular plants | BI/ LC | As time passes, the number of exotic species increased |
| Kuhn et al., 2017 | All vascular plants | BI | - |
| Kummli et al, 2021 | All vascular plants | LC | 15 % of the original grasslands had been lost due to changes in land use in 25 years |
| Lahr et al., 2015 | *Quercus stellata, Liquidambar styraciflua* | CL/ PO | Trees experienced lower stomatal conductance and photosynthesis and higher isoprene emission; urban trees experienced relatively higher isoprene emission at high CO2 concentrations, while isoprene emission was suppressed at the other sites. |
| Lahr et al., 2018 | *Acer rubrum* | CL/ HA | Urban wildtype trees had minor water use efficiency as air temperatures increased |
| Lambrecht et al., 2016 | *Crepis sancta* | CL | - |
| Landis and Leopold, 2014 | All vascular plants | BI | Numbers of alien species increased at more urban sites |
| Lechuga-Lago et al., 2017 | *Carpobrotus edulis, Malcolmia littorea, Scolymus hispanicus* | BI/ LC | Changes in soil characteristics due to urbanization allow for the establishment of the native, but ruderal, Scolymus hispanicus and non-native C. edulis |



| Reference | Neutral responses | Positive responses |
|---|---|---|
| King and Hovick, 2020 | - | - |
| Knapp et al., 2017 | - | - |
| Kobayashi et al., 2018 | The species successfully attracts local pollinators | - |
| Kolbe et al., 2016 | - | - |
| Kondratyeva et al., 2020 | - | Urban conditions allows the persistence of species with unique traits not observed, or observed less frequently in other land cover types, increasing functional diversity within urban communities |
| Kopel et al., 2015 | - | - |
| Kuhn et al., 2017 | The environmental covariates explaining richness as well as proportions of neophytes remain largely the same across the rural–urban gradient | - |
| Kummli et al, 2021 | Alpha diversity had not changed significantly, alien species were rare in both periods | - |
| Lahr et al., 2015 | Higher isoprene emission from urban trees was not associated with improved photosynthesis as temperatures increased | - |
| Lahr et al., 2018 | - | - |
| Lambrecht et al., 2016 | - | Urban plants flowered and senesced later than rural plants, and natural selection favored later phenology in the urban habitat; natural selection also favored larger plants with more leaves, and increased photosynthesis and leaf nitrogen concentration |
| Landis and Leopold, 2014 | Seedlings of native riparian trees nonetheless germinated along the gradient. | - |
| Lechuga-Lago et al., 2017 | The coastal dune endemic, Malcolmia littorea, showed no fitness effects in response to urbanization or the presence of *C. edulis* . | - |

| Reference | Taxa | Urban drivers | Negative responses |
|---|---|---|---|
| Lee, 2017 | *Forsythia koreana, Rhododendron mucronulatum, Prunus yedoensi, Prunus mume* | CL | The species have advancements in phenology due changes in temperature |
| Leong et al., 2014 | *Centaurea solstitialis* | LC | Bee visitation was highest in urban and agricultural land use contexts, but in contrast, seed set rates in these human-altered landscapes were lower than in natural sites |
| Lhotte et al., 2014 | *Teucrium pseudochamaepitys* | LC | Reproductive success is low in each site despite an effective insect pollination; diversity of pollinators is lower in urban sites |
| Li et al., 2017 | All vascular plants | CL | Phenology cycle in urban areas starts earlier and ends later resulting in a longer growing season length when compared to the respective surrounding urban areas |
| Lindh et al., 2018 | All vascular plants | CL | In general, phenological events starts earlier in recent years. |
| Lobo et al., 2016 | *Tabebuia aurea* | LC | Increase in floral displays among remnant individuals, probably reflecting an absence of lateral competition, were not sufficient to maintain high visitation rates to the flowers |
| Löki et al., 2019 | Orchids | HA | Conservation value was smaller in urban cemeteries |
| Lopez et al., 2018 | Herbs | BI/ HA/ LC/ PO | Native species, and species richness overall, declined in sites with less surrounding forest cover; decrease in taxonomic, phylogenetic, and functional richness in sites with higher ornamental species cover; Soil zinc influenced the composition, but not diversity, of plant communities, with more non-native plant species found in sites with higher zinc concentrations. |
| Lososová et al., 2016 | All vascular plants | BI | Changes in β-diversity of urban plant communities induced by the establishment of alien species |
| Lowry et al, 2020 | *Citharexylum spinosum, Psidium guajava, Schefflera actinophylla, Spathodea campanulata, Clerodendrum chinense, Clidemia hirta, Lantana camara, Leucaena leucocephala, Piper aduncum, Solanum torvum, Coccinia grandis, Ipomoea cairica, Merremia peltata, Mikania micrantha* | LC | Richness of species and overall abundance significantly lower in the urban sector |
| Malkinson et al., 2018 | All vascular plants | BI | Rare and native species are negatively associated with degree of urbanization; non-native species are positively associated with urbanization |
| Massetti et al., 2015 | *Tilia × europaea* | CL/ LC | Flowering occurred first in the areas with a higher percentage of impervious surfaces |

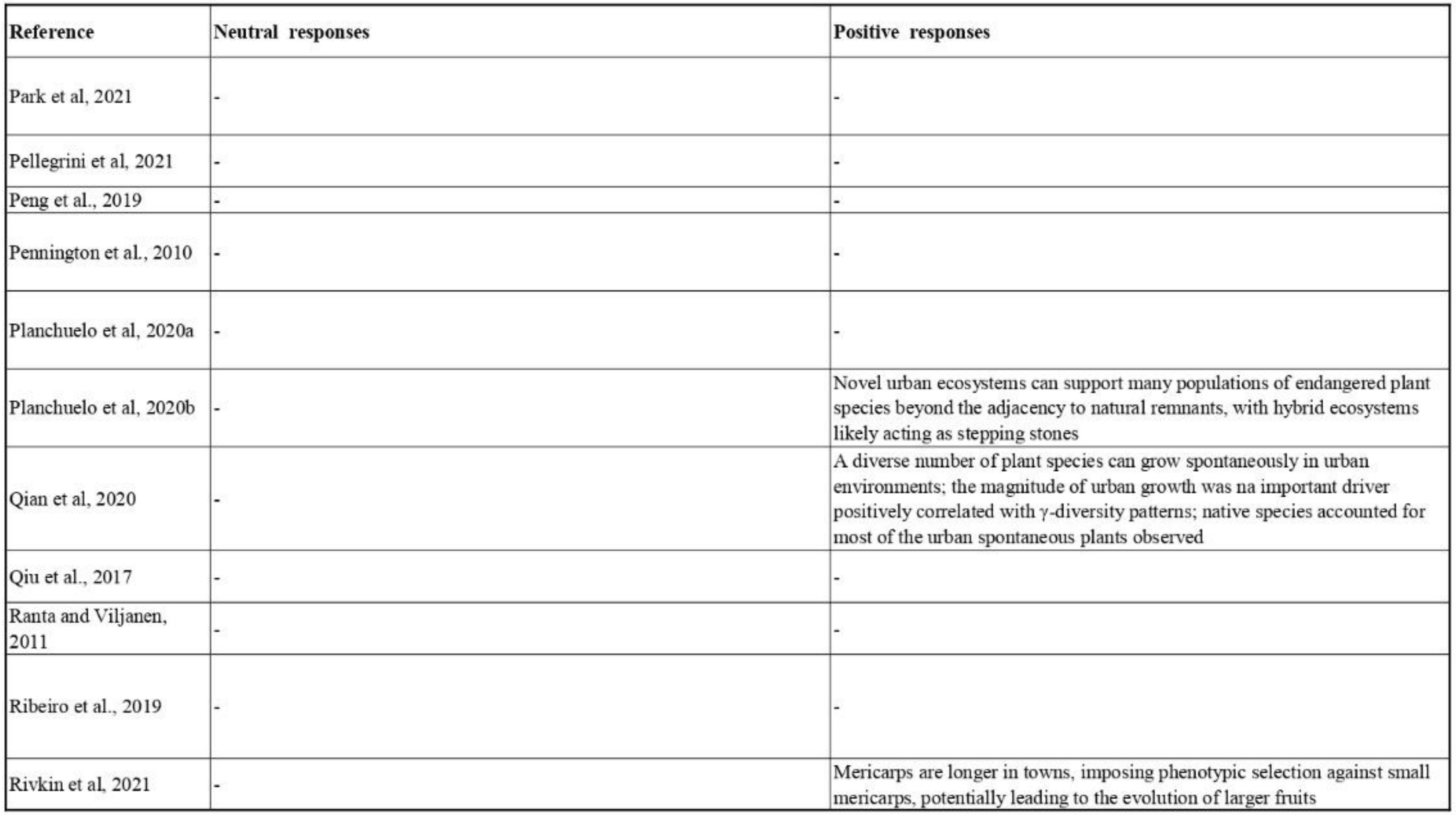


| Reference | Neutral responses | Positive responses |
|---|---|---|
| Lee, 2017 | - | - |
| Leong et al., 2014 | - | - |
| Lhotte et al., 2014 | - | - |
| Li et al., 2017 | - | - |
| Lindh et al., 2018 | - | - |
| Lobo et al., 2016 | - | - |
| Löki et al., 2019 | - | - |
| Lopez et al., 2018 | - | - |
| Lososová et al., 2016 | - | - |
| Lowry et al, 2020 | - | - |
| Malkinson et al., 2018 | - | - |
| Massetti et al., 2015 | - | - |

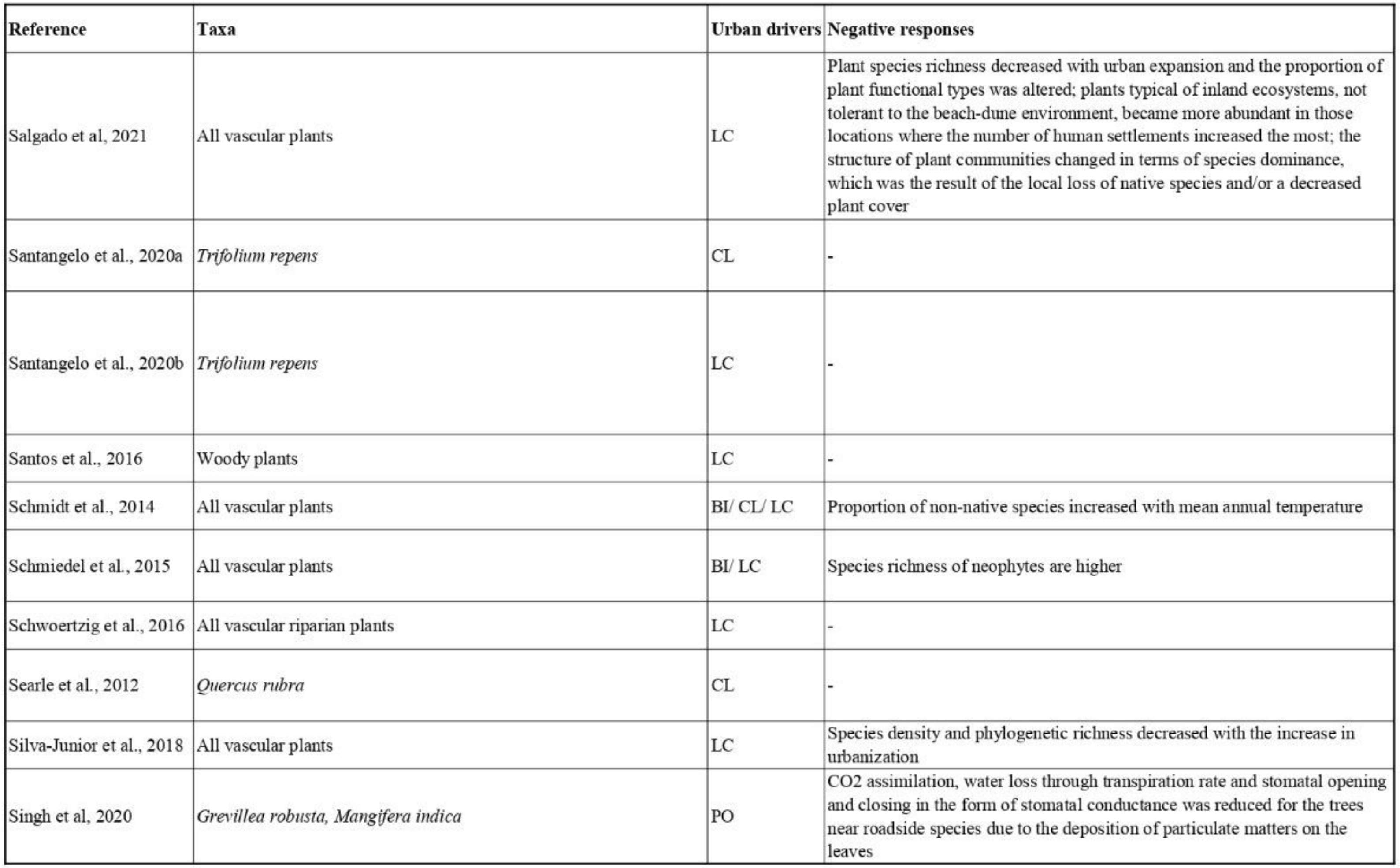
| Reference | Taxa | Urban drivers | Negative responses |
|---|---|---|---|
| McCune and Vellend, 2013 | All vascular plants | BI/ LC | Most successful species tended to be exotic; biotic homogenization was correlated with the colonization of common, disturbance-tolerant natives |
| Melliger et al., 2018 | All vascular plants | LC | Species richness decreased with degree of urbanisation; functional dispersion of plants decreased with increasing degrees of urbanization |
| Milanović et al, 2021 | All vascular plants | LC | Functional diversity significantly increased for neophytes and invasive species compared to native species, Functional diversity of multiple functional traits combined decreased over time. |
| Molnár et al., 2018 | *Celtis occidentalis, Acer campestre, Corylus avellana* | PO | Chlorophyll-a content of C. occidentalis leaves was the lowest in the urban area |
| Murray-Stoker and Johnson, 2021 | *Trifolium repens* | LC | - |
| Neil et al., 2010 | Shrubs | LC | Urbanization may have a significant effect on the flowering phenology of a small but substantial proportion of plants |
| Nero, 2019 | Trees | LC | Native species richness was lowest in the core urban area; pioneers and anthropochory dispersed species were the most abundant in urban landscape |
| Nikula et al., 2010 | *Populus tremula* | CL/ LC | - |
| Nock et al., 2013 | Woody plants | LC | Tree density is often low and results in low diversity at local scales |
| Oliveira et al., 2019 | *Paubrasilia echinata* | LC | Brazilwood trees in urban areas had smaller inflorescences, fewer flowers/inflorescences, larger flowers, less pollen/flower, lower richness of floral visitors and lower fruit and seed set when compared to natural ecosystem. |
| Oliveira et al., 2020 | Woody plants | BI/ HA/ LC | Small portion of tree species inhabiting urban green areas is native; squares and parks were impoverished in number of species and individuals with specialized pollination systems; urban green areas were dominated by generalist-pollinated trees with reproductive traits associated with easily accessible floral resources and an increase of hermaphrodites; squares and parks had similarly low functional diversity of reproductive traits |
| Paquin et al., 2020 | All vascular plants | BI/ LC | The richness of exotics increased with urbanization intensity |

| Reference | Neutral responses | Positive responses |
|---|---|---|
| McCune and Vellend, 2013 | - | - |
| Melliger et al., 2018 | - | - |
| Milanović et al, 2021 | Overall functinal diversity remained constant despite species turnover | - |
| Molnár et al., 2018 | - | - |
| Murray-Stoker and Johnson, 2021 | - | White clover has lower investment in rhizobia with greater N availability in the soil in urban areas |
| Neil et al., 2010 | - | - |
| Nero, 2019 | - | - |
| Nikula et al., 2010 | Specific leaf area showed no differences between habitats | Urbanization resulted in foliar acclimatization, exhibiting a higher amount of epicuticular waxes and N concentration, and a lower C:N ratio; litter decay was faster at the urban site and for urban litter |
| Nock et al., 2013 | - | - |
| Oliveira et al., 2019 | - | - |
| Oliveira et al., 2020 | - | - |
| Paquin et al., 2020 | The richness of natives remained stable with urbanization | Biotic differentiation, rather than homogenization, characterized the response of swamp plant communities to urbanization. |

| Reference | Taxa | Urban drivers | Negative responses |
|---|---|---|---|
| Park et al, 2021 | Aster pilosus, Lactuca scariola | BI/ CL/ LC | The invasiveness of the two alien plants was strongly associated with anthropogenic effects such as human population density, residential area, and road density |
| Pellegrini et al, 2021 | All vascular plants | BI/ CL/ LC | The success of plant invasion at the landscape scale is mainly due to the spread of urban land use |
| Peng et al., 2019 | All vascular plants | LC | Urbanization intensity negatively influenced plant diversity |
| Pennington et al., 2010 | Woody riparian plants | BI/ LC | Exotic canopy species and native early-successional species increased with the level of urbanization while native canopy and understory species declined. |
| Planchuelo et al, 2020a | Endangered species | LC | More than one–third of populations went extinct during the observation period, population survival was inversely correlated to the increase in impervious surfaces |
| Planchuelo et al, 2020b | All vascular plants | LC | Endangered plant species in novel ecosystems are filtered for ruderal strategy type and wind dispersal syndrome |
| Qian et al, 2020 | All vascular plants | LC | - |
| Qiu et al., 2017 | All vascular plants | LC | Vegetation phenology in the urban area is significantly different from its rural surroundings, with growing seasons during longer |
| Ranta and Viljanen, 2011 | All vascular plants | BI/ HA/ LC | Non-natives showed the highest values in suburban areas |
| Ribeiro et al., 2019 | All vascular plants | LC | CAD effects were stronger in adult communities by negatively affecting functional richness, dispersion and their effect sizes; CAD also altered the functional composition of leaf mass per area, woody density and leaf area of adult assemblages |
| Rivkin et al, 2021 | *Tribulus cistoides* | HA /LC | Predation was higher in towns than natural habitats |

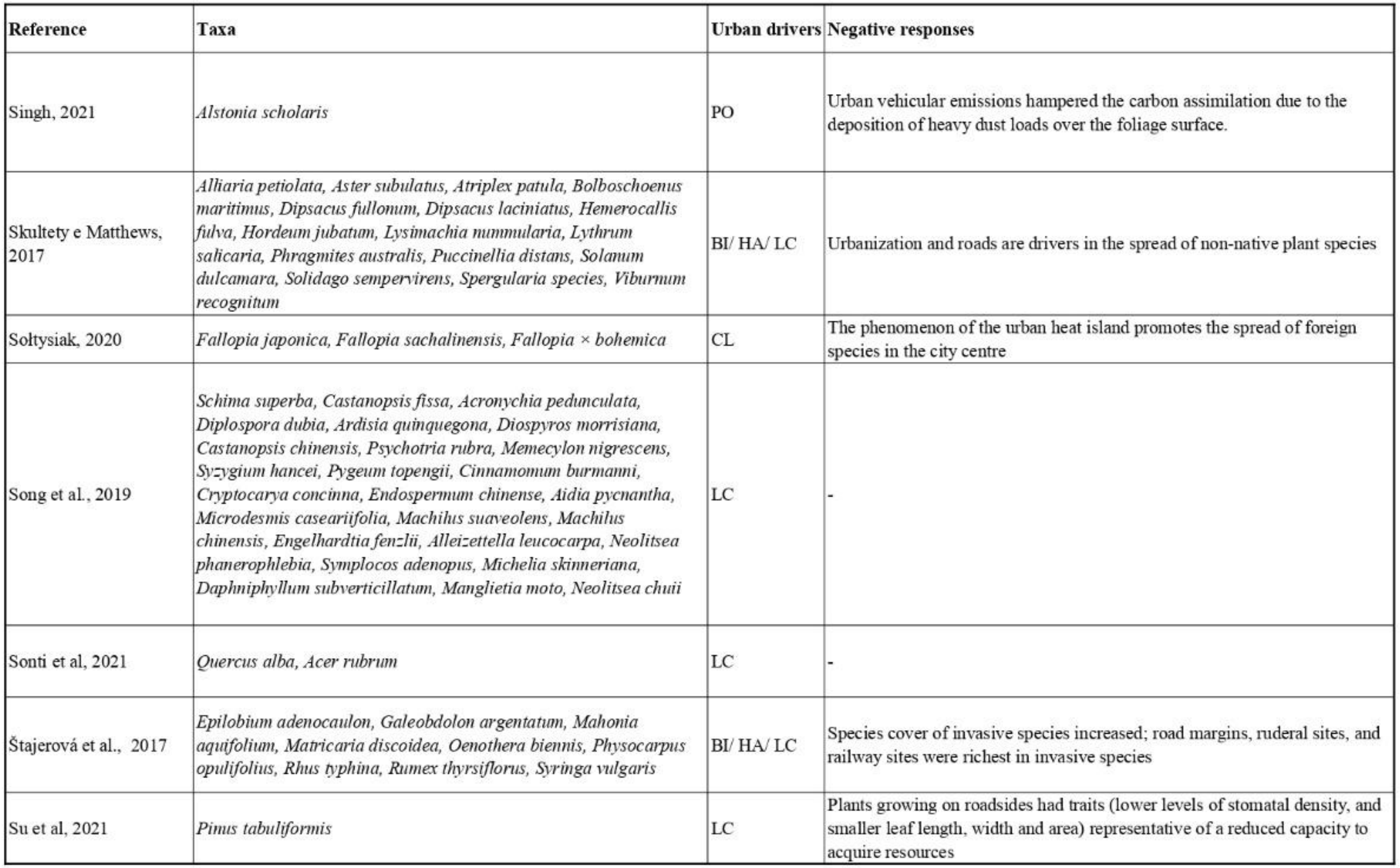


| Reference | Neutral responses | Positive responses |
|---|---|---|
| Park et al, 2021 | - | - |
| Pellegrini et al, 2021 | - | - |
| Peng et al., 2019 | - | - |
| Pennington et al., 2010 | - | - |
| Planchuelo et al, 2020a | - | - |
| Planchuelo et al, 2020b | - | Novel urban ecosystems can support many populations of endangered plant species beyond the adjacency to natural remnants, with hybrid ecosystems likely acting as stepping stones |
| Qian et al, 2020 | - | A diverse number of plant species can grow spontaneously in urban environments; the magnitude of urban growth was na important driver positively correlated with γ-diversity patterns; native species accounted for most of the urban spontaneous plants observed |
| Qiu et al., 2017 | - | - |
| Ranta and Viljanen, 2011 | - | - |
| Ribeiro et al., 2019 | - | - |
| Rivkin et al, 2021 | - | Mericarps are longer in towns, imposing phenotypic selection against small mericarps, potentially leading to the evolution of larger fruits |

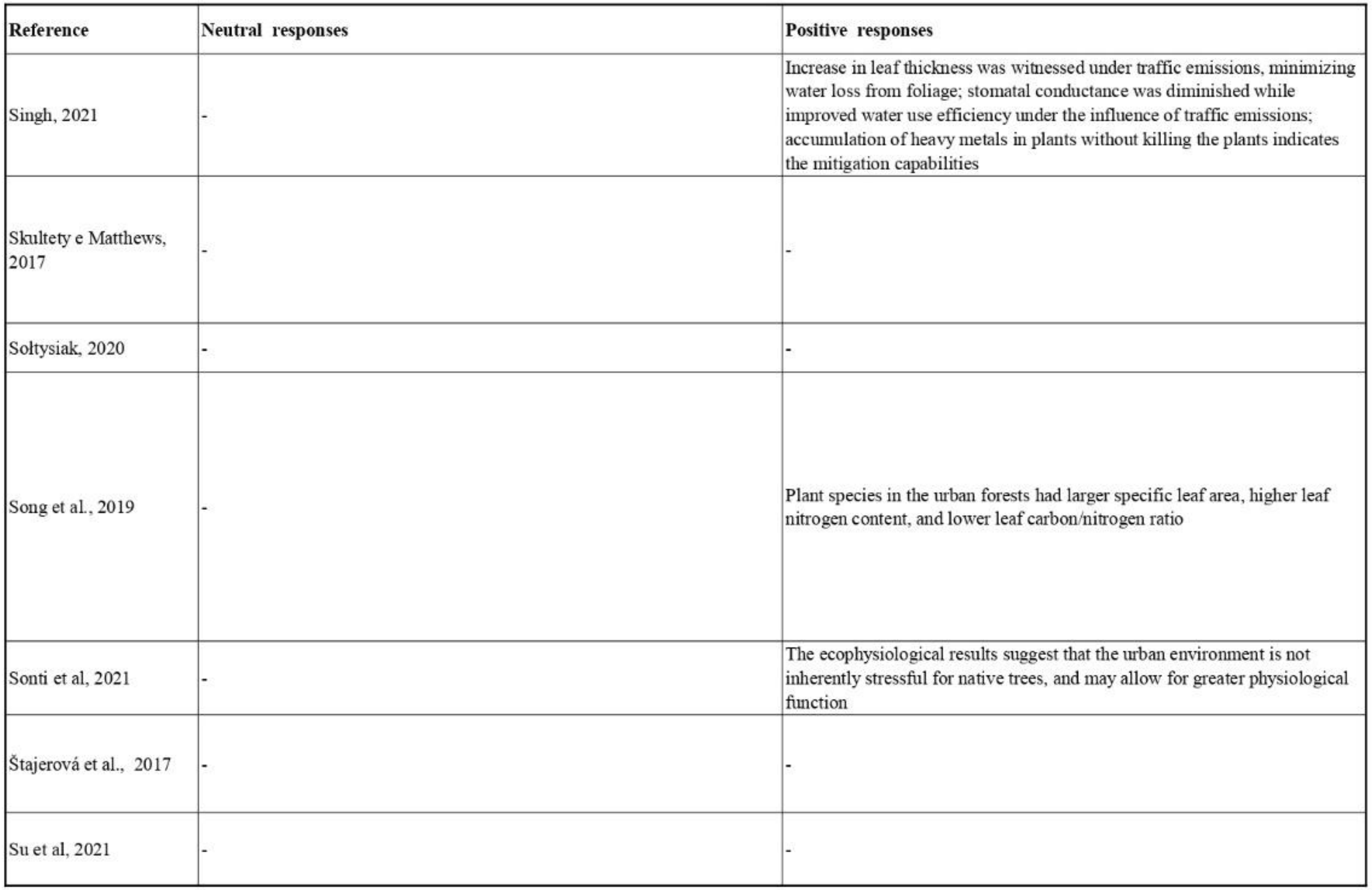


| Reference | Taxa | Urban drivers | Negative responses |
|---|---|---|---|
| Salgado et al, 2021 | All vascular plants | LC | Plant species richness decreased with urban expansion and the proportion of plant functional types was altered; plants typical of inland ecosystems, not tolerant to the beach-dune environment, became more abundant in those locations where the number of human settlements increased the most; the structure of plant communities changed in terms of species dominance, which was the result of the local loss of native species and/or a decreased plant cover |
| Santangelo et al., 2020a | *Trifolium repens* | CL | - |
| Santangelo et al., 2020b | *Trifolium repens* | LC | - |
| Santos et al., 2016 | Woody plants | LC | - |
| Schmidt et al., 2014 | All vascular plants | BI/ CL/ LC | Proportion of non-native species increased with mean annual temperature |
| Schmiedel et al., 2015 | All vascular plants | BI/ LC | Species richness of neophytes are higher |
| Schwoertzig et al., 2016 | All vascular riparian plants | LC | - |
| Searle et al., 2012 | *Quercus rubra* | CL | - |
| Silva-Junior et al., 2018 | All vascular plants | LC | Species density and phylogenetic richness decreased with the increase in urbanization |
| Singh et al, 2020 | *Grevillea robusta, Mangifera indica* | PO | CO2 assimilation, water loss through transpiration rate and stomatal opening and closing in the form of stomatal conductance was reduced for the trees near roadside species due to the deposition of particulate matters on the leaves |

| **Reference** | **Neutral responses** | **Positive responses** |
|---|---|---|
| Salgado et al, 2021 | Invasive species were relatively scarce and did not increase with land cover change. | - |
| Santangelo et al., 2020a | Clines in HCN production did not evolve in cities with the lowest temperatures and greatest snowfall | Strongest clines occurred in the warmest cities where snow and frost are rare, suggesting that alternative selective agents are maintaining clines in warmer cities |
| Santangelo et al., 2020b | - | Families from urban sites had evolved later phenology and germination, larger flowers, thinner stolons, reduced cyanogenesis, greater biomass and greater seed set. Pollinator observations revealed near-complete turnover of pollinator morphological groups along the urbanization gradient, which may explain some of the observed divergences in floral traits and phenology. |
| Santos et al., 2016 | Absence of significant differences in species richness and diversity between urban and rural basins | - |
| Schmidt et al., 2014 | Species richness differed only slightly between the urbanization zones and increased with habitat diversity | High proportions of endangered species were found in urban areas |
| Schmiedel et al., 2015 | - | In urban areas, species richness of neophytes and high-nutrient indicators are bigger; number of native, threatened and even low-nutrient species is high in urban areas |
| Schwoertzig et al., 2016 | Similar compositional patterns were found within the most urbanized sites with the establishment of ubiquitous species | - |
| Searle et al., 2012 | Increase in biomass in seedlings; urban seedlings allocated more growth to leaves; no difference in photosynthetic capacity of foliage per unit area. | - |
| Silva-Junior et al., 2018 | Phylogenetic divergence and structure did not respond to urbanization | - |
| Singh et al, 2020 | - | Water use efficiency of trees was enhanced at roadside trees compared to control sites; The leaf thickness of both the species has been found to be more in the trees growing along the roadside |

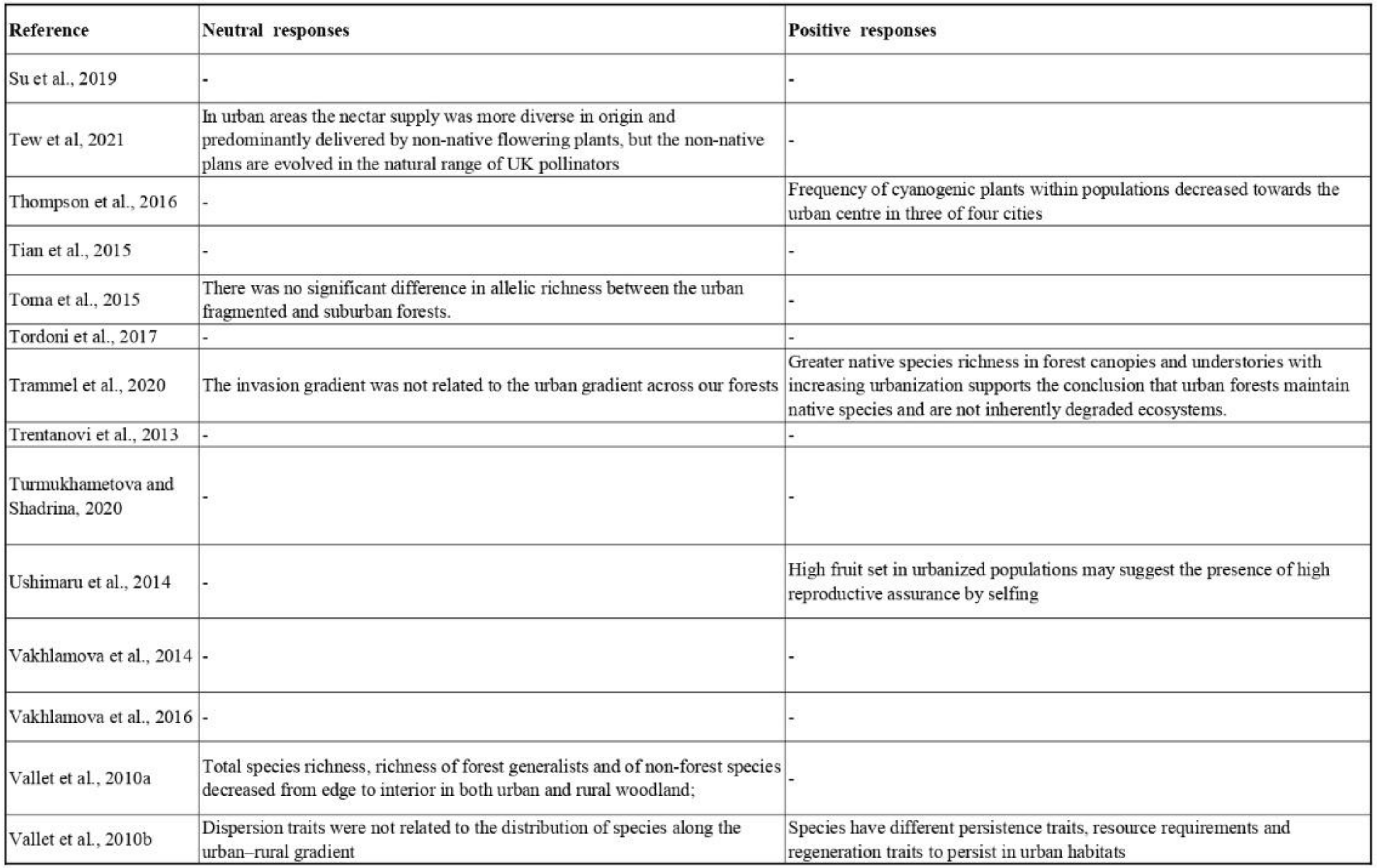


| Reference | Taxa | Urban drivers | Negative responses |
|---|---|---|---|
| Singh, 2021 | *Alstonia scholaris* | PO | Urban vehicular emissions hampered the carbon assimilation due to the deposition of heavy dust loads over the foliage surface. |
| Skultety e Matthews, 2017 | *Alliaria petiolata, Aster subulatus, Atriplex patula, Bolboschoenus maritimus, Dipsacus fullonum, Dipsacus laciniatus, Hemerocallis fulva, Hordeum jubatum, Lysimachia nummularia, Lythrum salicaria, Phragmites australis, Puccinellia distans, Solanum dulcamara, Solidago sempervirens, Spergularia species, Viburnum recognitum* | BI/ HA/ LC | Urbanization and roads are drivers in the spread of non-native plant species |
| Sołtysiak, 2020 | *Fallopia japonica, Fallopia sachalinensis, Fallopia × bohemica* | CL | The phenomenon of the urban heat island promotes the spread of foreign species in the city centre |
| Song et al., 2019 | *Schima superba, Castanopsis fissa, Acronychia pedunculata, Diplospora dubia, Ardisia quinquegona, Diospyros morrisiana, Castanopsis chinensis, Psychotria rubra, Memecylon nigrescens, Syzygium hancei, Pygeum topengii, Cinnamomum burmanni, Cryptocarya concinna, Endospermum chinense, Aidia pycnantha, Microdesmis caseariifolia, Machilus suaveolens, Machilus chinensis, Engelhardtia fenzlii, Alleizettella leucocarpa, Neolitsea phanerophlebia, Symplocos adenopus, Michelia skinneriana, Daphniphyllum subverticillatum, Manglietia moto, Neolitsea chuii* | LC | - |
| Sonti et al, 2021 | *Quercus alba, Acer rubrum* | LC | - |
| Štajerová et al., 2017 | *Epilobium adenocaulon, Galeobdolon argentatum, Mahonia aquifolium, Matricaria discoidea, Oenothera biennis, Physocarpus opulifolius, Rhus typhina, Rumex thyrsiflorus, Syringa vulgaris* | BI/ HA/ LC | Species cover of invasive species increased; road margins, ruderal sites, and railway sites were richest in invasive species |
| Su et al, 2021 | *Pinus tabuliformis* | LC | Plants growing on roadsides had traits (lower levels of stomatal density, and smaller leaf length, width and area) representative of a reduced capacity to acquire resources |



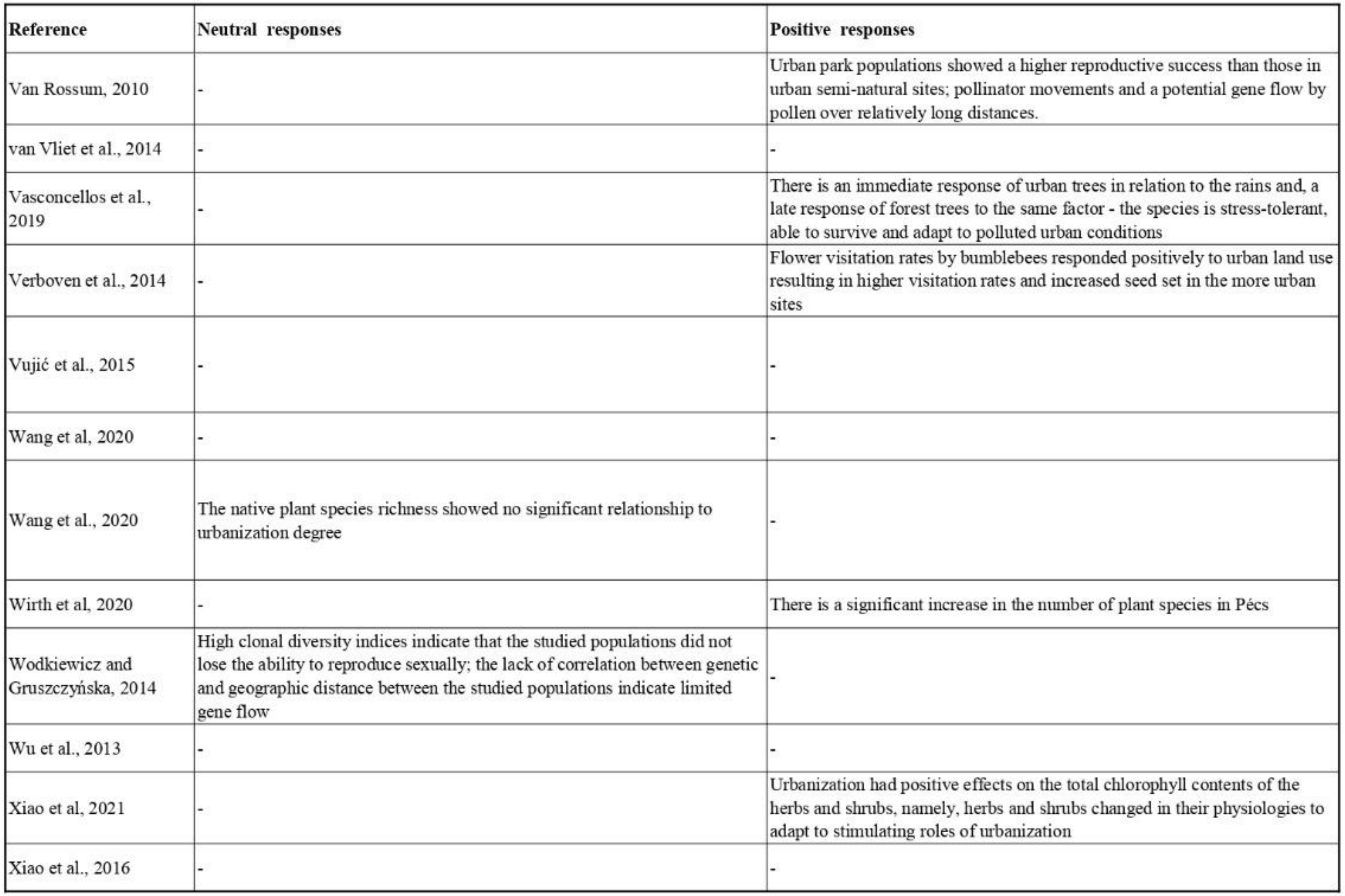
| Reference | Neutral responses | Positive responses |
|---|---|---|
| Singh, 2021 | - | Increase in leaf thickness was witnessed under traffic emissions, minimizing water loss from foliage; stomatal conductance was diminished while improved water use efficiency under the influence of traffic emissions; accumulation of heavy metals in plants without killing the plants indicates the mitigation capabilities |
| Skultety e Matthews, 2017 | - | - |
| Sołtysiak, 2020 | - | - |
| Song et al., 2019 | - | Plant species in the urban forests had larger specific leaf area, higher leaf nitrogen content, and lower leaf carbon/nitrogen ratio |
| Sonti et al, 2021 | - | The ecophysiological results suggest that the urban environment is not inherently stressful for native trees, and may allow for greater physiological function |
| Štajerová et al., 2017 | - | - |
| Su et al, 2021 | - | - |

| Reference | Taxa | Urban drivers | Negative responses |
|---|---|---|---|
| Su et al., 2019 | *Pinus tabulaeformis* | CL/ LC | The increased percentage of impervious cover could advance the timing of leaf and male cone phenophases. |
| Tew et al, 2021 | Community of specific flowering species | BI/ HA /LC | - |
| Thompson et al., 2016 | *Trifolium repens* | CL/ LC | - |
| Tian et al., 2015 | All vascular weedy plants | LC | The number of weed communities in urban areas was less than the number found in suburban and rural areas |
| Toma et al., 2015 | *Viola grypoceras* A. Gray *var. grypoceras* | LC | - |
| Tordoni et al., 2017 | All vascular plants | BI/ LC | Urban plots account for most of the alien species in the sampling |
| Trammel et al., 2020 | Woody plants | BI/ LC | - |
| Trentanovi et al., 2013 | All vascular plants | BI/ LC | Urbanity proved to homogenize the native species pool |
| Turmukhametova and Shadrina, 2020 | *Betula pendula* | PO | An increase in seed production and variety of fruit-producing organs in size and shape indicates poor quality of the environment and of the seeds themselves. Anthropogenic impact also impairs the quality of seeds, decreases their germination energy and germination capacity. |
| Ushimaru et al., 2014 | *Commelina communis* | LC | Pollinator and mate availability decreased significantly with developed land area; most urbanized populations suffered from significant pollinator-limited male and/or female reproductive success |
| Vakhlamova et al., 2014 | All vascular plants | BI/ LC | Plant diversity decreased with near to the city centre and was also influenced by the type of land use; the percentage of alien species increase |
| Vakhlamova et al., 2016 | All vascular plants | BI/ HA | More alien species were found in the road verges at the city edge than in the rural surroundings along both road types |
| Vallet et al., 2010a | Woody plants | LC | Urban woodland edges were not as rich in forest specialists as rural edges |
| Vallet et al., 2010b | All vascular plants | LC | - |



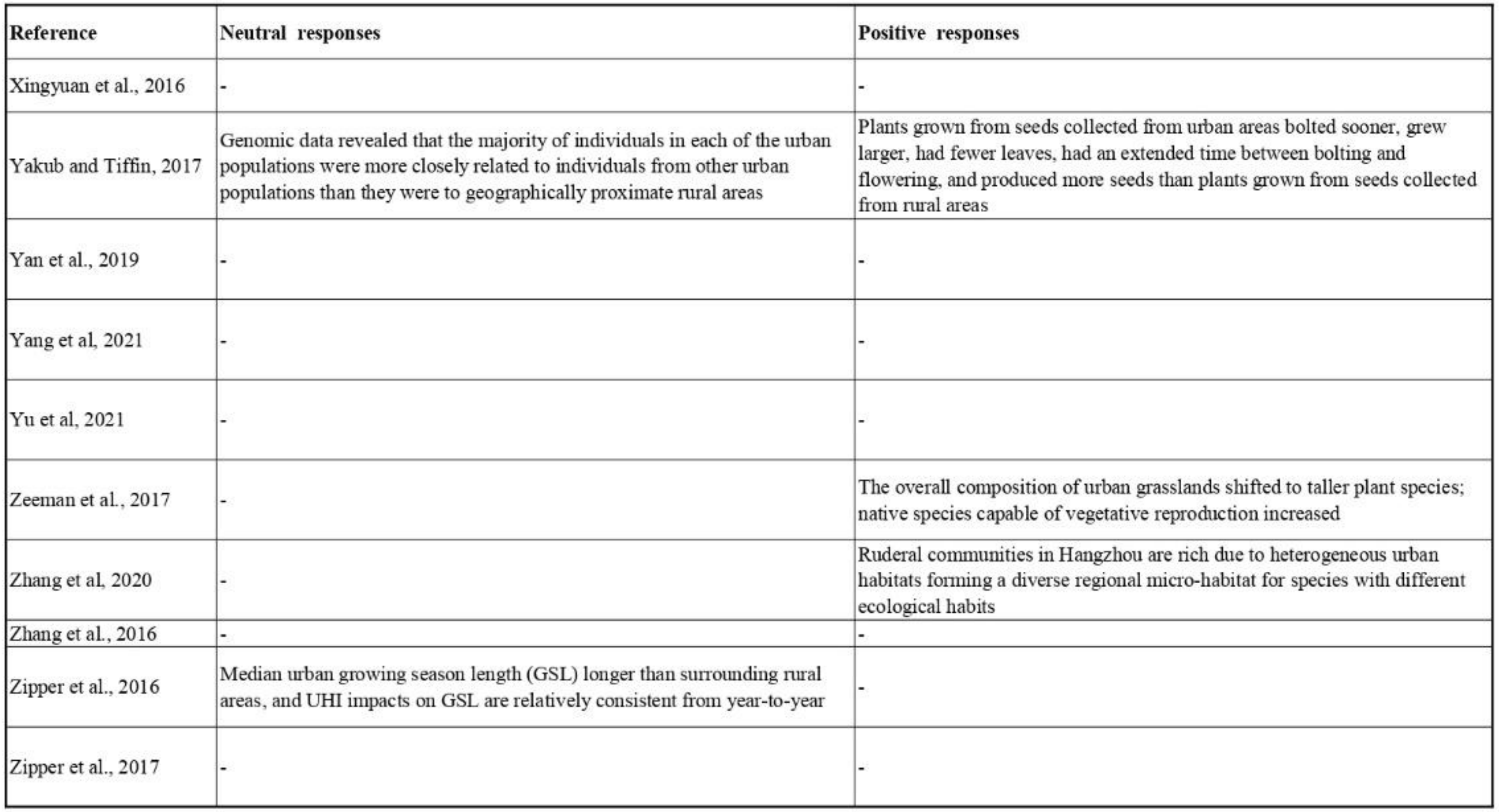
| Reference | Neutral responses | Positive responses |
|---|---|---|
| Su et al., 2019 | - | - |
| Tew et al, 2021 | In urban areas the nectar supply was more diverse in origin and predominantly delivered by non-native flowering plants, but the non-native plans are evolved in the natural range of UK pollinators | - |
| Thompson et al., 2016 | - | Frequency of cyanogenic plants within populations decreased towards the urban centre in three of four cities |
| Tian et al., 2015 | - | - |
| Toma et al., 2015 | There was no significant difference in allelic richness between the urban fragmented and suburban forests. | - |
| Tordoni et al., 2017 | - | - |
| Trammel et al., 2020 | The invasion gradient was not related to the urban gradient across our forests | Greater native species richness in forest canopies and understories with increasing urbanization supports the conclusion that urban forests maintain native species and are not inherently degraded ecosystems. |
| Trentanovi et al., 2013 | - | - |
| Turmukhametova and Shadrina, 2020 | - | - |
| Ushimaru et al., 2014 | - | High fruit set in urbanized populations may suggest the presence of high reproductive assurance by selfing |
| Vakhlamova et al., 2014 | - | - |
| Vakhlamova et al., 2016 | - | - |
| Vallet et al., 2010a | Total species richness, richness of forest generalists and of non-forest species decreased from edge to interior in both urban and rural woodland; | - |
| Vallet et al., 2010b | Dispersion traits were not related to the distribution of species along the urban–rural gradient | Species have different persistence traits, resource requirements and regeneration traits to persist in urban habitats |

| **Reference** | **Taxa** | **Urban drivers** | **Negative responses** |
|---|---|---|---|
| Van Rossum, 2010 | *Centaurea jacea* | LC | - |
| van Vliet et al., 2014 | All vascular plants | CL | Changes in climate explain variation in timing of phenological events from year to year |
| Vasconcellos et al., 2019 | *Ceiba speciosa* | CL/ PO | - |
| Verboven et al., 2014 | *Trifolium repens* | LC | - |
| Vujić et al., 2015 | *Iris pumila* | PO | Decrease of standard length and centroid size in the polluted environment was detected, as well as a change in the shape; the standards were shorter and wider in the polluted area when compared to those in the unpolluted environment. |
| Wang et al, 2020 | All vascular plants | LC | Extinction of native plants is directly related to the sizes and landscape configurations of urban construction lands |
| Wang et al., 2020 | All vascular plants | LC | The richness of annual herbs exhibited a negative relationship to urbanization degree; the richness of the all plants, woody plants and perennial herbs presented significant positive relationship with urbanization degree, maybe due to the increase of exotic and ruderal species |
| Wirth et al, 2020 | All vascular plants | BI/ LC | The number of woody species neophytes and short-lived alien plants increased |
| Wodkiewicz and Gruszczyńska, 2014 | *Stellaria holostea* | HA/ LC | - |
| Wu et al., 2013 | Herbs | LC | The composition, diversity indices, mean and total richness gradually decreased |
| Xiao et al, 2021 | *Lophatherum gracile, Alpinia chinensis, Ardisia quinquegona, Psychotria rubra, Diospyros morrisiana, Cratoxylum cochinchinense, Engelhardtia roxburghiana, Schima superba.* | CL/ PO | Negative correlations between heavy metal concentrations and maximum photosynthetic rates were all detected in the canopies |
| Xiao et al., 2016 | Woody plants | BI/ CL/ LC | Urbanization had the function of species conservations with sharp increases of alien species and tropical type plants. |



| Reference | Neutral responses | Positive responses |
|---|---|---|
| Van Rossum, 2010 | - | Urban park populations showed a higher reproductive success than those in urban semi-natural sites; pollinator movements and a potential gene flow by pollen over relatively long distances. |
| van Vliet et al., 2014 | - | - |
| Vasconcellos et al., 2019 | - | There is an immediate response of urban trees in relation to the rains and, a late response of forest trees to the same factor - the species is stress-tolerant, able to survive and adapt to polluted urban conditions |
| Verboven et al., 2014 | - | Flower visitation rates by bumblebees responded positively to urban land use resulting in higher visitation rates and increased seed set in the more urban sites |
| Vujić et al., 2015 | - | - |
| Wang et al, 2020 | - | - |
| Wang et al., 2020 | The native plant species richness showed no significant relationship to urbanization degree | - |
| Wirth et al, 2020 | - | There is a significant increase in the number of plant species in Pécs |
| Wodkiewicz and Gruszczyńska, 2014 | High clonal diversity indices indicate that the studied populations did not lose the ability to reproduce sexually; the lack of correlation between genetic and geographic distance between the studied populations indicate limited gene flow | - |
| Wu et al., 2013 | - | - |
| Xiao et al, 2021 | - | Urbanization had positive effects on the total chlorophyll contents of the herbs and shrubs, namely, herbs and shrubs changed in their physiologies to adapt to stimulating roles of urbanization |
| Xiao et al., 2016 | - | - |

| Reference | Taxa | Urban drivers | Negative responses |
|---|---|---|---|
| Xingyuan et al., 2016 | Woody plants | CL | Influenced by climate warming, the germination, leafing, and flowering phenologies of this urban forest advanced |
| Yakub and Tiffin, 2017 | *Lepidium virginicum* | LC | - |
| Yan et al., 2019 | All vascular plants | LC | An increase in percentage of total impervious surface area reduced plant diversity; the ratio of exotic to endemic plant species increased as percentage of total impervious surface area increased. |
| Yang et al, 2021 | All vascular plants | LC | The β2-diversity of adult trees was higher than that of sapling/seedlings. It is believed that species found in the sampling/seedling layer are more likely to be generalists and have high urban tolerance. |
| Yu et al, 2021 | All vascular plants | BI/ CL/ LC | Invasive and ornamental species increased; a significant increase in the number of thermophilous plant species was observed, which is closely related to global warming and the heat island effect in cities |
| Zeeman et al., 2017 | Graminoids | BI/ LC | The most urbanized sites had the highest number of non-native species in both the current and historical data sets; exotic species with an annual life span increased in frequency |
| Zhang et al, 2020 | Ruderal plants | LC | Ruderal diversity decreased from the rural region to central urban area; species with shorter life span and smaller size become more dominant in high urbanized areas |
| Zhang et al., 2016 | Woody plants | LC | Species richness and diversity were the lowest in the urban core |
| Zipper et al., 2016 | All vascular plants | CL/ LC | - |
| Zipper et al., 2017 | All vascular plants | CL/ LC | Plant water requirements are significantly higher in urban areas compared to rural areas, driven by increased air temperature with minimal effects of decreased air moisture content |



| Reference | Neutral responses | Positive responses |
|---|---|---|
| Xingyuan et al., 2016 | - | - |
| Yakub and Tiffin, 2017 | Genomic data revealed that the majority of individuals in each of the urban populations were more closely related to individuals from other urban populations than they were to geographically proximate rural areas | Plants grown from seeds collected from urban areas bolted sooner, grew larger, had fewer leaves, had an extended time between bolting and flowering, and produced more seeds than plants grown from seeds collected from rural areas |
| Yan et al., 2019 | - | - |
| Yang et al, 2021 | - | - |
| Yu et al, 2021 | - | - |
| Zeeman et al., 2017 | - | The overall composition of urban grasslands shifted to taller plant species; native species capable of vegetative reproduction increased |
| Zhang et al, 2020 | - | Ruderal communities in Hangzhou are rich due to heterogeneous urban habitats forming a diverse regional micro-habitat for species with different ecological habits |
| Zhang et al., 2016 | - | - |
| Zipper et al., 2016 | Median urban growing season length (GSL) longer than surrounding rural areas, and UHI impacts on GSL are relatively consistent from year-to-year | - |
| Zipper et al., 2017 | - | - |

# *Capítulo IV*

# *A bromeliad living in the city: a case of a native species resilient to urbanization in South Brazil*

**Artigo a ser submetido na revista American Journal of Botany**

# *Capítulo V*

# *Effects of urbanization on the production and seed viability of* Tillandsia aeranthos *(Bromeliaceae)*

**Artigo a ser submetido na revista Flora**

# *Capítulo VI*

# *Considerações finais*

# Considerações finais

Nesta tese foram apresentadas as consequências da crescente urbanização e como espécies de plantas estão respondendo à essas mudanças drásticas no ambiente. Apesar do aumento considerável no número de estudos na área da Ecologia Urbana, o impacto da urbanização em plantas ainda é pouco compreendido, especialmente em seus aspectos evolutivos. Revisões anteriores apontaram como aspectos específicos (por exemplo, traços de história de vida, fisiologia, biologia reprodutiva – Alberti et al. (2017); Aronson et al. (2016); Cadotte et al. (2017); Eckert et al. (2010); Harrison & Winfree (2015); Jochner & Menzel (2015); Johnson et al. (2015); Rai (2016); Williams et al. (2015)) de grupos de plantas podem ser prejudicados com a urbanização. Porém, uma compilação compreendendo vários desses aspectos avaliados em escala global, assim como quais fatores específicos levaram à diferentes respostas das plantas frente à urbanização, ainda não havia sido feita. A proposta desta tese foi abordar diferentes aspectos da urbanização como causadores de mudanças em espécies e comunidades de plantas, assim como contribuir com estudos genéticos e evolutivos de uma espécie nativa de bromélia (*Tillandsia aeranthos*) que ocorre em ambientes urbanos no sul do Brasil.

O **Capítulo III** consiste de um artigo de revisão sistemática global integrando várias áreas da Ecologia Urbana para sumarizar como as espécies e comunidades de plantas estão respondendo a urbanização e quais os principais agentes dessas respostas (aqui chamados de *Urban Drivers*). Após avaliar 171 estudos publicados nos últimos onze anos, foi revelado que a maioria dos estudos foi realizada na Europa e na América do Norte, sendo o bioma mais representado o de Florestas Temperadas e Florestas Mistas. Além disso, a maioria dos estudos relatou consequências negativas da urbanização. Os *Urban Drivers* mais citados como responsáveis pelas mudanças nas espécies e comunidades vegetais foram "mudança da cobertura do solo" (alteração das paisagens de vegetação nativa, transformando-as em áreas cobertas por superfícies impermeáveis) e "invasão biótica" (introdução de espécies não nativas no habitat). De forma resumida, os estudos analisados mostraram que a urbanização reduz a riqueza de espécies vegetais e a disponibilidade de polinizadores. Além disso, ela também homogeneíza o ambiente, promove a invasão de espécies não nativas, causa alterações nos tempos dos eventos fenológicos e impõe dificuldades para o crescimento das plantas. No entanto, foi avaliado que algumas espécies apresentam evidências de adaptação a essas adversidades por meio de sua plasticidade fenotípica e também pela evolução de características que tornam os indivíduos resilientes ao ambiente urbano. Embora tenha sido estabelecido que a urbanização impõe novas pressões seletivas resultando em adaptações únicas à vida da cidade, os padrões de mudanças na biodiversidade em resposta à urbanização podem variar por região, bioma, história da cidade e especialmente de acordo com a espécie ou grupo taxonômico. Os *Urban Drivers* responsáveis pelas respostas da vegetação atuam simultaneamente modificando o ambiente, portanto, é um desafio prever os padrões de resposta das plantas sob o crescimento rápido da urbanização.

Surge então uma grande necessidade de realizar mais estudos na área da Ecologia Urbana com espécies de plantas, principalmente em locais com uma menor representatividade de estudos, como os biomas tropicais e em países em desenvolvimento. Portanto, o **Capítulo IV** desta tese trata dos efeitos da urbanização na diversidade genética, estrutura genética e adaptação local em *T. aeranthos*, uma epífita auto incompatível com alta capacidade de colonizar habitats urbanos no sul do Brasil. Esta espécie, pertencente à família Bromeliaceae, possui capacidade de colonizar com sucesso uma ampla gama de habitats devido a inúmeras adaptações morfológicas e fisiológicas, como suculência foliar, metabolismo ácido das crassuláceas (CAM) e tricomas especializados de absorção de água (Benzing, 2000). Portanto, seu metabolismo CAM é vantajoso para colonizar com sucesso as áreas urbanas, geralmente mais secas, tornando esta espécie um excelente modelo para o estudo de como as plantas podem se adaptar a estes ambientes (Judith *et al.*, 2013).

Este estudo foi realizado utilizando uma amostragem na forma de dois transectos, distribuídos ao longo de um gradiente de urbanização, para testar se haveria diferenças nos padrões genéticos entre populações que ocorrem em ambientes urbanos e não-urbanos. Encontramos uma maior proporção de variação genética dentro das populações, confirmando o padrão esperado para uma espécie que é auto incompatível. Além disso, a fecundação cruzada obrigatória permitiu que esta espécie mantivesse altos níveis de diversidade genética, permitindo que suas populações tolerem uma maior variedade de regimes ambientais, como climas extremos, poluentes, degradação e fragmentação do habitat, que caracterizam o ambiente urbano. De fato, não encontramos uma diminuição na diversidade genética com o aumento da urbanização, como seria esperado se os efeitos da deriva genética fossem maiores nesses habitats fragmentados. Em concordância com este resultado, a fragmentação causada pela urbanização não foi suficiente para separar as populações urbanas e não-urbanas através das análises de estrutura genética. Em uma espécie com ampla capacidade de dispersão pelo vento, como *T. aeranthos*, a modificação humana na paisagem pode aumentar indiretamente as taxas de dispersão entre populações, removendo barreiras ao fluxo gênico. Por exemplo, estradas com alto tráfego de veículos podem ajudar na dispersão de sementes que se dispersam com o vento, diminuindo o impacto da fragmentação do habitat. A falta de correlação entre os *outlier loci* detectados através do escaneamento genômico e o índice de urbanização encontrado em nosso estudo corrobora com os demais índices de diversidade genética não sendo afetados pela urbanização. Portanto, pode-se supor que a dispersão pode ocorrer por pólen ou sementes entre populações urbanas e não-urbanas, sendo estes os principais determinantes da manutenção da diversidade genética nas populações de *T. aeranthos*.

Seguindo esta linha de raciocínio, no **Capítulo V** desta tese, foi apresentado um estudo sobre a produção e a viabilidade de sementes de *T. aeranthos*, como forma de comparar um dos aspectos de seu sucesso reprodutivo em ambientes urbanos e não-urbanos. A reprodução por sementes desempenha um papel fundamental na adaptação, migração, adaptação e persistência populacional das espécies vegetais (Fenner & Thompson, 2005). Espécies auto incompatíveis só produzem sementes através da fecundação cruzada, portanto elas são altamente dependentes de polinizadores para sua reprodução (Richards, 1997). Assim, espera-se que espécies de plantas auto incompatíveis sejam mais suscetíveis a mudanças nas comunidades de polinizadores (mudanças na abundância, composição e/ou comportamento de forrageamento) como resultado da fragmentação dos habitats urbanos. Consequentemente, seu sucesso reprodutivo deve ser mais afetado pela urbanização do que o sucesso reprodutivo de plantas auto compatíveis (Aguilar *et al.*, 2006).

A produção de sementes não apresentou diferença estatisticamente significativa em relação ao gradiente de urbanização. Por outro lado, a taxa de germinação das sementes variou de 0% a 69%, com diferença estatisticamente significativa em relação ao gradiente de urbanização. Como o desenvolvimento da semente depende estritamente da fertilização cruzada bem-sucedida, é provável que tenha ocorrido uma falha no desenvolvimento da semente posterior à fertilização. Novos estudos com observações mais detalhadas do sistema reprodutivo de *T. aeranthos* replicados em diferentes paisagens urbanizadas devem ser realizados para avaliar as consequências da urbanização para esta espécie.

A Ecologia Urbana fornece cada vez mais evidências de como as mudanças ambientais afetam a ecologia populacional das espécies, a estrutura da comunidade e os processos ecossistêmicos (Rivkin *et al.*, 2019), porém sabemos muito menos sobre os impactos ecológicos da urbanização na evolução das populações de plantas que vivem nas cidades. Desta forma, os resultados obtidos na presente tese contribuirão com os estudos avaliando os efeitos da urbanização em espécies vegetais, especialmente em ambientes tropicais, que ainda são menos representados em escala global. Estudar a ecologia e a evolução das plantas em áreas urbanas nos leva a uma melhor compreensão sobre problemas relacionados à conservação e a estabilidade ambiental em um planeta cada vez mais urbanizado. Logo, estudos ecológicos e evolutivos em áreas urbanas se tornam cada vez mais urgentes para a conservação da biodiversidade.

# *Capítulo VII*

# *Referências bibliográficas*

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# Apêndices

## Apêndice 1

### Artigo publicado durante o período do doutorado fora do escopo da tese

**Ruas RB**, Paggi GM, Aguiar-Melo C, Hirsch LD, Bered F. 2020. Strong genetic structure in *Dyckia excelsa* (Bromeliaceae), an endangered species found on ironstone outcrops in Pantanal, Brazil. **Botanical Journal of Linnean Society**, https://doi.org/10.1093/botlinnean/boz099

## Apêndice 2

### Prêmio de produtividade do PPGBM-UFRGS

3º Lugar na categoria Área Vegetal/Microrganismo – Nível Doutorado – ano de 2019

UFRGS
UNIVERSIDADE FEDERAL
DO RIO GRANDE DO SUL

# CERTIFICADO

PPGBM
Programa de Pós-Graduação
em Genética e Biologia Molecular
da UFRGS

Certificamos que **Renata de Barros Ruas** recebeu o 3º lugar no prêmio de Aluna com Maior Produtividade no ano de 2019 (Área Vegetal/Microrganismo - Nível Doutorado), concedido pelo Programa de Pós-Graduação em Genética e Biologia Molecular (PPGBM) da Universidade Federal do Rio Grande do Sul (UFRGS).

Porto Alegre, 06 de janeiro de 2020.

Prof. Dr. Thales Renato Ochotorena de Freitas
Coordenador PPGBM-UFRGS

Elmo Jurandir Antunes Cardoso
Coord. Administrativo PPGBM-UFRGS

## Apêndice 3

### Fotos de *Tillandsia aeranthos*

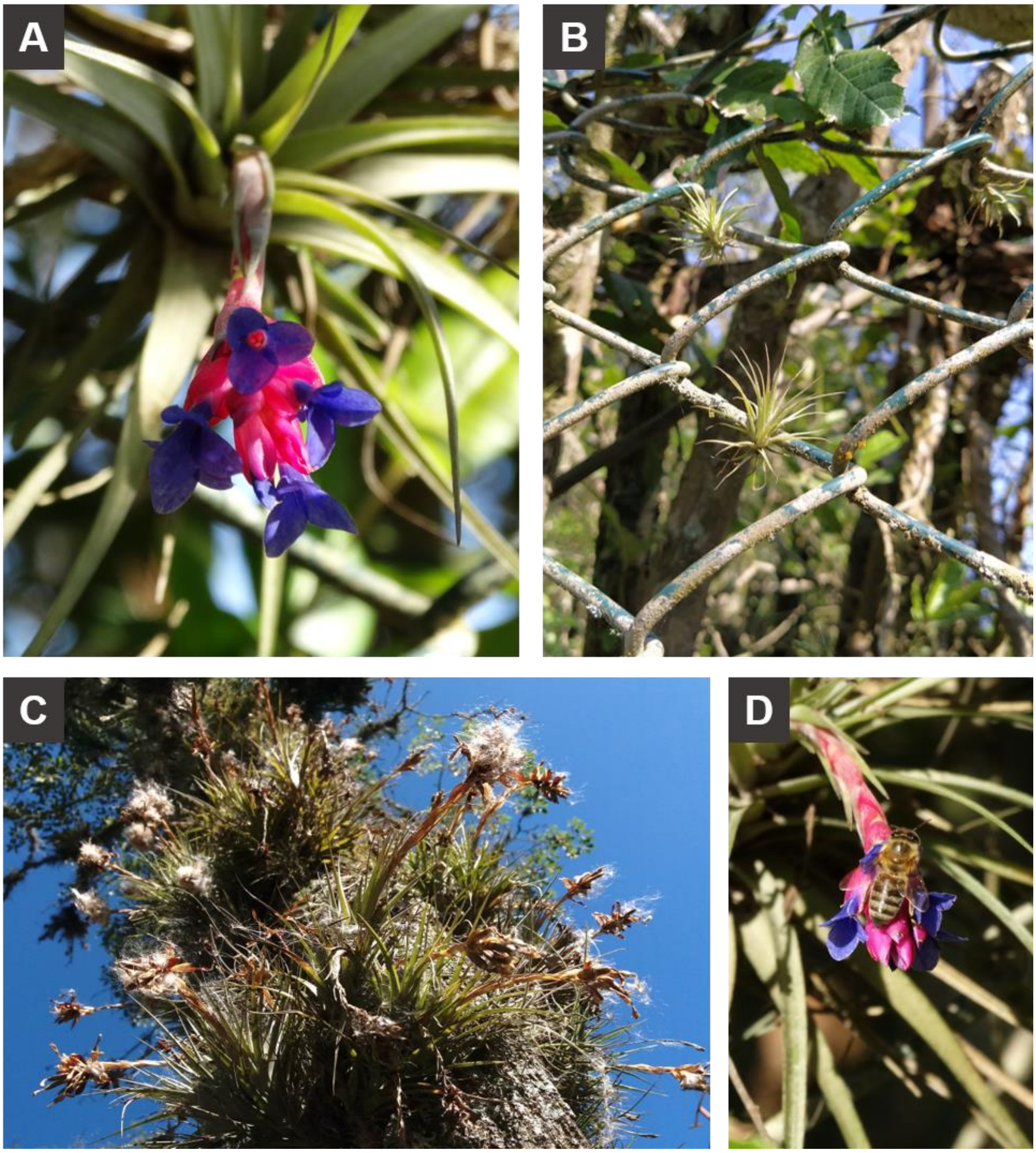

**Figura 1.** A – Detalhe da inflorescência; B – Indivíduos jovens estabelecidos em substrato artificial; C – Aglomerados de *T. aeranthos* em época de dispersão de sementes; D – Visitante floral avistado em observações preliminares de polinizadores.