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SYNTHESIS & INTEGRATION

Review: Toward management of urban ponds for freshwater biodiversity

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Abstract. Many cities around the world are expanding and this trend in urbanization is expected to sharply increase over coming decades. At the same time, the integration of green and blue spaces is widely promoted in urban development, potentially offering numerous benefits for biodiversity. This is particularly relevant for urban waterbodies, a type of ecosystem present in most cities. However, site managers often lack the knowledge base to promote biodiversity in these waterbodies, which are generally created to provide other ecosystem services. To address this, our review presents guidelines for promoting biodiversity in urban ponds. We found a total of 516 publications indexed in ISI Web of Sciences related to this topic, of which 279 were retained for the purposes of our review. The biodiversity of urban ponds, measured by species richness, appears to be generally lower than in rural ponds; however, urban ponds often support threatened species. Furthermore, if well managed, urban ponds have the potential to support a much greater biodiversity than they currently do. Indeed, this review shows that a range of urban factors can impair or promote pond biodiversity, including many that can easily be controlled by site managers. Local factors include design (surface area, pond depth, banks and margins, shade, shoreline irregularity), water quality (conductivity, nutrients, heavy metals), and hydroperiod and biotic characteristics (stands of vegetation, fish, invasive species). Important regional factors include several indicators of urbanization (roads, buildings, density of population, impervious surfaces, car traffic), and the presence of other wetlands or green spaces in the surrounding landscape. We considered each of these factors and their potential impact on freshwater biodiversity. Taking into account the management measures listed in the publications reviewed, we have proposed a framework for the management of urban ponds, with guidelines to promote biodiversity and other ecosystem services, and to avoid ecosystem disservices or the creation of ecological traps. At the city scale, the biodiversity of a pondscape benefits from a high diversity of pond types, differing in their environmental characteristics and management.

Key words: aquatic ecosystems; flora and fauna; literature review; management; small waterbodies; species richness; urban wetlands.

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Introduction

Biodiversity is in crisis worldwide, with a particularly sharp decline in freshwaters (WWF 2016). At the same time, urbanization, one of the major factors of degradation, continues worldwide and is likely to have further significant impacts on natural habitats. Between 2000 and 2030, built-up areas are likely to nearly triple in surface area (Seto et al. 2012). To address this, local and international urban development policies promote the inclusion of green (vegetation) or blue (water) areas in the urban matrix. If these policies are successful, future cities are likely to support urban biodiversity linked to these blue and green networks. This urban biodiversity will often remain partly connected to rural biodiversity (directly or through corridors or stepping stones) and can contribute to the conservation of global biodiversity (Parris 2016), particularly because cities can act as hotspots for some threatened species (Ives 2016). The blue network includes streams, canals, rivers, ponds, wetlands, reservoirs, and lakes. Networks of small waterbodies, like ponds (waterbodies with a surface area up to 5 ha; Oertli et al. 2005), are acknowledged to support a large part of regional biodiversity (Williams et al. 2004, Angélibert et al. 2006, Davies et al. 2008). Therefore, ponds in urban areas could make an important contribution to freshwater conservation, although little is known of their role as refuges (Chester and Robson 2013), or about how to maximize conservation management in these waterbodies (Hassall et al. 2016). Some type of ponds, if not properly managed or if polluted, can nevertheless also act as ecological traps that increase the extinction risk of some populations (Hale et al. 2015, Sievers et al. 2018*c*).

Ponds are often numerous in the urban matrix, but are rarely of natural origin. Most are constructed by people (Hassall 2014, Oertli 2018) and their primary function is to provide specific services such as water purification and flow regulation (e.g., stormwater ponds), sediment trapping, aesthetic value (parks and garden ponds), environmental education, leisure activities such as boating and fishing, or irrigation. They are therefore generally managed to maximize these services rather than as habitats for freshwater biodiversity. They are also embedded in an urban matrix which is largely hostile to the movement of many species, and so, they are often biologically isolated from other freshwaters (Hassall et al. 2016). Urban ponds are very diverse in their design and situation in the landscape, and generally differ from natural or rural ponds in different respects (e.g., surface area, depth, artificial structures, water quality, exotic species).

In many cities, we expect an increase in the number of ponds, particularly linked to climate change adaptation. In the past 20 yr, for example, the number of stormwater ponds has increased fivefold in Melbourne, Australia (Hale et al. 2015). These changes highlight the great potential for the network of blue spaces in the urban matrix to contribute to freshwater biodiversity conservation. To turn this potential into a real contribution, there are still some issues that need to be addressed. On the one hand, urban ponds need to be suitable for providing their targeted services, and on the other hand, they should also offer high-quality habitats for the biodiversity. Many questions remain about how to balance this trade-off. Are the driving factors of biodiversity in urban areas the same as in the rural landscape? How do urban ponds behave? Are the factors recognized as key for natural ponds, for example, morphometric parameters (e.g., surface area), landscape factors (e.g., connectivity), and water quality (e.g., eutrophication; Oertli et al. 2010) the same for ponds in urban landscapes? Or are other factors more important? It can be expected that, for example, water pollution, pond isolation, and management practices (to achieve various ecosystem services) would be particularly important for the value of ponds as habitat for biodiversity in urban areas.

A global review of the scientific literature is needed to highlight the specificities of the urban environment and provide evidence-based guidelines to support the management of urban ponds for biodiversity conservation. Existing review on aquatic biodiversity in cities has made valuable contributions, but these have targeted specific taxonomic groups such as dragonflies (Villalobos-Jimenez et al. 2016) and amphibians (Hamer and McDonnell 2008), and they have not disentangled ponds from other types of waterbodies (like running waters). Hassall (2014) and Hassall et al. (2016) addressed the topic of urban pond ecology and biodiversity, but focused on a limited geographical frame (northwest Europe). There has been a recent increase in publications on the biodiversity of urban ponds (Fig. 1), with 75% of these coming from countries outside northwest Europe. This highlights the need for a global review addressing the potential of urban ponds to support freshwater biodiversity, the factors driving this potential, and the integration of conservation measures into urban pond management.

Independently of the type of landscape, a range of management guidelines for promoting the biodiversity of ponds have been proposed (Biggs et al. 1994, Williams et al. 1999, Semlitsch 2000, Williams et al. 2008, Chester and Robson 2013), and these have some relevance to urban ponds. Nevertheless, urban ponds often face different types of pressure compared to ponds in other landscapes. This review aims to identify management actions for promoting biodiversity in the urban landscape, based on studies conducted in various cities. To do this, we address the following questions:

1. Do urban ponds support ecological communities similar to those of non-urban ponds (e.g., with respect to species richness and community composition)?
2. Do urban ponds contribute to the conservation of freshwater biodiversity, and if so, for which taxonomic groups?
3. What are the main environmental factors affecting the biodiversity of ponds in cities?
4. Does urban pond biodiversity bring any disservices to humans in cities?
5. What types of management strategies should be used to optimize urban ponds as habitat for biodiversity?

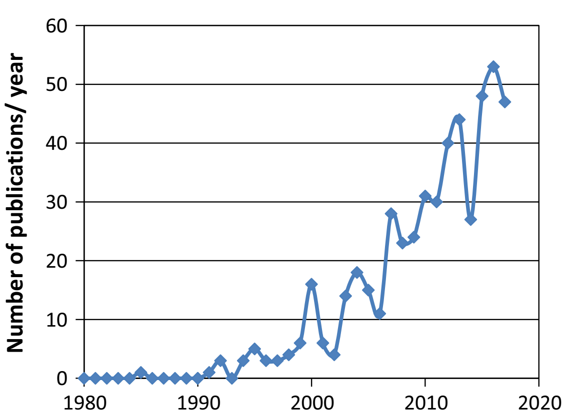

Fig. 1. Temporal increase in the number of publications dealing with urban ponds and their biodiversity, from 1980 to 2017.

Methods

We produced a database of publications to include in this review through four consecutive steps.

Step 1

Selection of peer-reviewed publications indexed in ISI Web of Science 15 January 2018 (databases: SCI-EXPANDED, SSCI, A&HCI, CPCI-S, CPCI-SSH, BKCI-S, BKCI-SSH, ESCI, CCR-EXPANDED, IC) that included topics related to the investigation, either in the title, the abstract, the keywords, or in the ISI's "keywords plus" field. The words searched were as follows: "TOPIC: ((periurban* or suburban* or urban* or city or cities) and (wetland* or pond*)), *AND* TOPIC: (flora or plants or macrophyt* or vertebrate* or invertebrat* or mammal* or fish* or bird* or insect* or amphibia* or frogs or macroinvertebrate* or zoobenthos or benthos or alga* or crustace* or dragonfl* or odonat* or reptilia* or mollusc* or phytoplankton or beetle* or coleopter* or zooplankton or butterfl* or lepidopter* or turtle* or fung* or biodiversity), *NOT* TOPIC: ((marine* or coastal*))." This first step produced 2428 references.

Step 2

We exported the reference list produced in step 1 and manually screened the titles (and if necessary the abstract). Publications that did not include any measure of biodiversity were discarded, as were also purely ecotoxicological or pollution studies. We also checked that the retained investigations concerned ponds, and publications investigating only larger waterbodies (lakes; area >5 ha) or large wetlands, as well as running waters, were discarded. Other outliers were also removed (e.g., studies realized entirely or mostly in non-urban environments, social studies). This second step produced 516 references and represented the core database of publications published by the end of 2017 on the topic of urban pond biodiversity.

Step 3

Screening of the abstracts (and if necessary, the main texts) to assess the relevance of the results to the review objectives and their statistical robustness. The discarded publications were (1) case studies with no relevance to this review (e.g., simple biodiversity inventories), (2) studies with speculative conclusions not supported by the results section, (3) studies without sufficient replication (e.g., studies of a single pond), with pseudo-replication or with non-comparable sets of ponds. The screening process in this third step left 280 references.

Step 4

A final screening to assess remaining 280 publications (the same screening as in step 2, but conducted on the abstract and potentially the text) discarded 31 references, but also highlighted 30 additional relevant references. This step resulted in a final list of 279 references, many of which are cited in the present review.

Description of Research to Date on Urban Pond Biodiversity

We used the 516 references produced by the two first steps of the literature screening to produce the descriptive statistics shown here.

Year of publication

Most (94%) of the 516 publications were published after the year 2000 (Fig. 1). This date saw the beginning of a sharp increase in the number of publications, with a rate increasing from about 10 publications/year in 2000 to 50 publications/year in 2017.

Geographic source of the publications

The continent of origin of the researchers who published these 516 publications (Fig. 2) was mainly North America (38%) and Europe (32%). The main countries of origin were the United States (32%), Australia (9%), UK (6%), Canada (5%), and China (4%). This geographical pattern underlines a strong imbalance and shows the low level of information available from Africa, South and Central America, and Asia, three continents that are undergoing rapid urban expansion (Parris 2016).

Topics included in the publications

All 516 publications are related directly to the biodiversity of urban ponds, as this was the basic selection criterion. More specifically, the main topics of the publications, as expressed in the abstract, title, or keyword list, showed

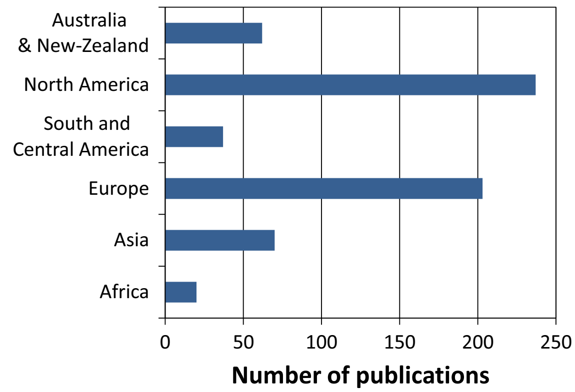

Fig. 2. Geographic location of the researchers who published the 516 publications selected (see section above). The total number of researchers is 629, because a single publication can have multiple co-authors of different geographic origin.

research trends related to (1) the assessment of the urban freshwater biodiversity, (2) the impact of urban driving factors on this biodiversity, and (3) the adaptation to or mitigation of urban impacts through management (Fig. 3). Biodiversity assessment was mainly investigated through measures of species richness and to a lesser extent of conservation value (threatened species presence). The main driving factors investigated to measure the impact on biodiversity were local factors, such as the water quality and the presence of aquatic vegetation, but also regional factors linked to the landscape structure (e.g., connectivity, fragmentation, pond isolation). One of the driving factors, which seemed here to be of lesser concern, was the urban heat island (and global warming). This is surprising given its importance in larger cities around the world. Adaptation to urbanization was expressed in the publications by the management measures undertaken on urban ponds for conserving or promoting biodiversity. The most cited word in the 516 publications was management (in one-third of the publications), and this reflects the fact that urban waterbodies have a strong relationship with humans. In this instance, management can include one or several ecosystem services (including provision of habitat for biodiversity). The large concern with management clearly also justifies the current review, which is specifically targeted at this important topic. This also separates urban waterbodies from natural waterbodies in rural environments, which are subject to less human intervention. The present review will therefore follow the logical flow issued from this classification of research topics (i.e., Fig. 3). Firstly, we present the review of the literature assessing freshwater biodiversity linked to urban ponds. Secondly, we consider each of the driving factors and their potential impacts on freshwater biodiversity, and also present management issues. And thirdly, we propose a framework for the management of urban ponds.

Criteria used to measure the level of urbanization

The type of measure describing the level of urbanization was very heterogeneous in the published literature. For example, in a selection of recent publications (Table 1), approaches differed according to the type of measurement or the spatial scale investigated. Measurements were linked to the presence of buildings or roads, the proportion of impervious surface, the proportion of urban areas (a category often available in local land-use databases) or human population density. The spatial scale was generally a buffer area of a given radius around the pond investigated (from 50 m to 10 km), sometimes with different radii investigated in a single study. Alternatively, the spatial scale investigated was the catchment or sub-catchment.

The results of a given study are without doubt influenced by the type of urbanization measure considered. This impedes the possibility of conducting meta-analyses with published studies. This shows that the development of a standardized measure of urbanization is needed.

Taxonomic group investigated

In the 516 reviewed publications, the taxonomic groups most cited (within the abstract, title, or keyword list) were amphibians (28%),

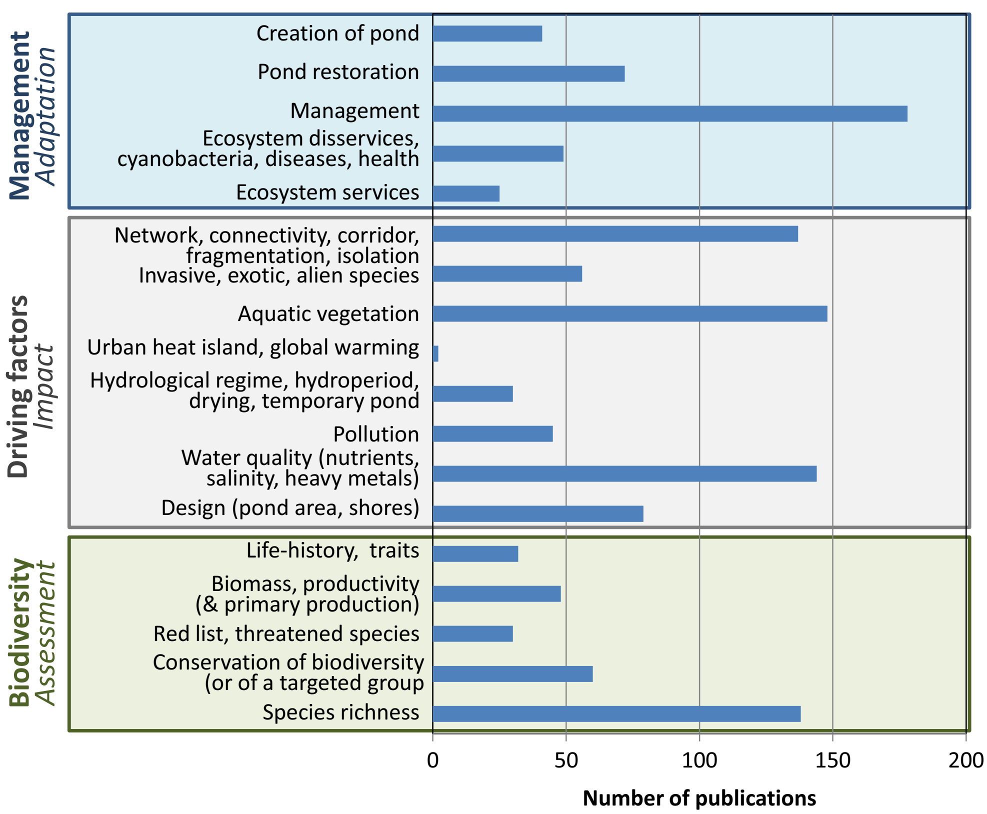

Fig. 3. Number of publications (from 516 reviewed publications) using words linked to certain research topics, within the abstract, title, or keywords. The topics are mainly related to assessment of the biodiversity, to impact from urban driving factors and to adaptation through management.

Table 1. Examples of criteria used for characterizing the level of urbanization around a pond in a selection of recent publications, showing the diversity of approaches.

Type of urbanization metric	Measure	Spatial scale of the measure	Example of studies
Presence of buildings	Percentage of built-up area, that is, percentage of area covered by buildings	50 m–3.2 km radii	Gianuca et al. (2018)
	Percentage built-up area, that is, surface area occupied by buildings, houses, and industrial infrastructure, with roads and parking lots excluded	3.2 km radius	Brans et al. (2017)
	Percentage built-up area, that is, surface area occupied by buildings	200, 500, 800 m radii	Blicharska et al. (2017)
	Built-up area	500 m radius	Holtmann et al. (2017)
	Areas with buildings (low + high rise buildings)	200 m radius	Heino et al. (2017)
	Percentage of buildings: commercial, residential, and parking lots	1 and 2 km radii	Zhang et al. (2016)
	Areas of low, medium, and high urban residential density (six per class), based on city classification	Surrounding landscape	Mimouni et al. (2015)
Presence of roads	Road length within buffer area	10, 100 m, and 1 km	Villasenor et al. (2017)
	Road density in a buffer area	300 m to 10 km	Marsh (2017)
	Road density and urban infrastructure	500 m radius	Roe et al. (2011)
Impervious surfaces	Impervious surfaces	50, 100, 250, 500 m, 1 km, and 2.5 km	Thornhill et al. (2017)
	Impervious surface cover in a buffer area	300 m to 10 km	Marsh (2017)
	% covered in impervious surfaces	catchment	Mackintosh et al. (2017)
	Percentage of impervious surfaces: pavement, driveways, footpaths, and other human-building sites.	1 km and 2 km radii	Zhang et al. (2016)
	Cover of impervious surfaces (buildings and roads)	500 m, 2 km, 5 km radii	Straka et al. (2016)
	Impervious cover (Ontario Geospatial Data)	0.2 km to 2.6 km radii, at 0.2-km intervals	Patenaude et al. (2015)
	Percentage of surface covered by artificial surfaces (FAO GLC-SHARE)	watershed	Castilla et al. (2015)
	Percentage of impervious surface	sub-watershed	Vincent and Kirkwood (2014)
Urban land use	Proportion of urban land use in a buffer	100 m, 200 m, 400 m, 800 m, 1.6 km radii	Le Gall et al. (2018)
	Proportion of urban land use in a buffer	1 km buffer	Hill et al. (2017)
	Type "Urban," from merged types from the Land cover Florida Natural Areas Inventory	2 km buffer	Faller and McCleery (2017)
	Proportion of urban land (Land Cover Circa 2000 dataset) in a buffer	1 km buffer	Hassall and Anderson (2015)
	Land cover (urban industrial, urban residential (including gardens)) from the South African National Land Cover dataset (NLCD)	100 m, 400 m, 1 km radii	Calder et al. (2015)
Distance to city center	Distance to city center	no limit	Pawlikiewicz and Jurasz (2017)
Human population	Number of residents living around ponds	200, 500, 800 m radii	Blicharska et al. (2017)
	Human population density in a buffer area	1 km radius	Hamer and Parris (2011)
Development	Development in a buffer area	300 m to 10 km	Marsh (2017)

fish (25%), plants (22%), and birds (17%; Fig. 4). Note that this does not differentiate between publications where the taxonomic group is the central topic, and those where it is only marginally investigated. Considering only the title gave a better assessment of when a taxonomic group was the focus of the investigation, in which case 17% of the publications included the keyword "amphibian" (or frog) in the title, and only 4% included the keyword "fish." Fish were often not investigated as a component of biodiversity but as a factor impairing biodiversity.

Biodiversity in Urban Ponds

Species richness in urban ponds compared with non-urban ponds

The impact of urbanization on pond species richness is often reported to be negative at the local scale (alpha diversity). Indeed, species richness was shown to be significantly lower in urban ponds than in non-urban ponds in several large cities in North America (Portland, Southeastern Wisconsin, Front Range Region Colorado, Iowa, Wisconsin, Chicago), Europe (Bradford, Lodz, West Midlands, Stockholm), and Australia (Melbourne, Canberra). The taxonomic groups considered were aquatic plants (Magee et al. 1999, Noble and Hassall 2015), zooplankton (Dodson et al. 2005, Pawlikiewicz and Jurasz 2017), macroinvertebrates (Johnson et al. 2013, Noble and Hassall 2015, Thornhill et al. 2017, Sievers et al. 2018*a*), amphibians (Knutson et al. 1999, Lofvenhaft et al. 2004, Hamer and Parris 2011, Johnson et al. 2013, Westgate et al. 2015, Sievers et al. 2018*a*), reptiles (Johnson et al. 2013), and wetland birds (Ward et al. 2010).

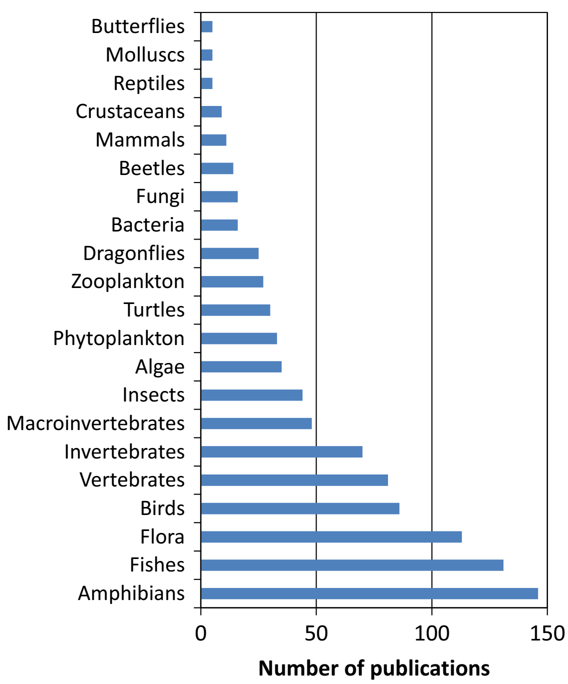

Fig. 4. Taxonomic groups cited in the abstract, title, or keyword listed in the 516 reviewed publications. Total n = 950 citations, as more than one taxonomic group was investigated in some publications.

There are, however, many exceptions, and species richness in urban ponds was found to be greater for some other taxonomic groups. For example, bird abundance and species richness were greater at urban versus rural wetlands in Rhode Island (USA; McKinney et al. 2011). For Cladocera (zooplankton), smaller species were more diverse in more urbanized ponds in contrast to larger-bodied species, which were more diverse in less urbanized systems (Gianuca et al. 2018). Macroinvertebrate taxa tolerant of environmental pressures can also dominate in urban ponds, and this is the case with Oligochaeta or Chironomidae, which are often very numerous in terms of both abundance and species richness (Bishop et al. 2000, Wood et al. 2001, Lunde and Resh 2012, Mackintosh et al. 2015, Hill et al. 2017). This suggests that different taxonomic groups respond differently to the main urban drivers of biodiversity, as observed in a study of the species richness of different insect classes in ponds in Stockholm (Blicharska et al. 2017). Similarly, at the species level, responses to a set of environmental variables can be species-specific, such that two species can respond in opposite ways to the same factor, as shown for aquatic plants (Ehrenfeld 2008) and amphibians (Hamer and Parris 2011). Within many taxonomic groups, most species may suffer from urban conditions while others can tolerate them or even benefit from them. Another exception to the generally negative association between species richness and urbanization is illustrated by studies of aquatic plant communities. Indeed, floristic species richness can be greater in urbanized environments, although this is the result of deliberate introductions of both native and non-native species (e.g., Oertli et al. 2018). These non-native species can represent more than 50% of the species present, as observed in Portland, Oregon (Magee et al. 1999).

Contribution of urban ponds to the conservation of freshwater biodiversity

In many cases, urban ponds support fewer species than in rural ponds (see previous section), but these urban ponds can nevertheless provide a habitat for numerous species. Several studies also show that these urban species pools include threatened species of conservation concern at the regional or national scale. Examples include urban wetlands in north-central Florida which supported the round-tailed muskrat (*Neofiber alleni*), a wetland obligate rodent and near-endemic in Florida which is considered of conservation concern (Faller and McCleery 2017). In another study, urban parks in California supported populations of the damselfly *Ischnura gemina* (Hannon and Hafernik 2007), vulnerable on the IUCN Red List. Urban waterbodies in North America, if properly managed, may serve as refuges for turtle populations such as those of the western pond turtle *Emys marmorata*. This species is declining throughout its range (Spinks et al. 2003) and is classified as vulnerable on the IUCN Red List. Overall, 12 dragonfly species of conservation concern from Central Europe occur in cities (Goertzen and Suhling 2015). Furthermore, amphibians—which as a group are particularly threatened worldwide—are often well represented in urban waterbodies, as illustrated by investigations conducted in Melbourne, Australia (Hamer and Parris 2011), Edmonton, Canada (Scheffers and Paszkowski 2013), Potchefstroom, South Africa (Kruger et al. 2015), and Münster, Germany (Holtmann et al. 2017).

Urban ponds often support lower species richness than non-urban ponds (Hill et al. 2016, Thornhill et al. 2017). However, urban ponds also have a role to play in the conservation of freshwater biodiversity, and this contribution could increase if the quality of urban habitats was enhanced. We will demonstrate in the following sections that there are many ways to manage urban ponds to promote their quality and biodiversity, and so enhance their conservation value.

Furthermore, as the high value of ponds for biodiversity conservation is linked to the complementary of pond types (presenting different ecological niches) at the landscape (pondscape) scale (Oertli et al. 2002, Williams et al. 2004, Hill 2018), then urban ponds, if diversified and presenting a high local environmental heterogeneity, can collectively present the same biodiversity as non-urban ponds. This has, for example, been demonstrated with macroinvertebrate communities in the UK (Hill et al. 2016).

Impact of Urbanization on the Biodiversity of Urban Ponds and Implications for Management

The urban cocktail of driving factors

Many environmental factors are linked to urbanization (Parris 2016), and it is often the combination of these factors (and potentially their interactions) that drives urban biodiversity. These could be considered as the urban cocktail of environmental factors, which may include both positive and negative factors. Several urban cocktails composed of local and/or regional factors have been identified to date (Table 2). Local factors are linked to pond design (surface area, pond depth, type of margins, shade, shoreline irregularity), water chemistry and hydrology (conductivity, nutrients, heavy metals, hydroperiod), or to the characteristics of biological communities (presence of aquatic vegetation, fishes, invasive species). The regional factors include different indicators of urbanization (roads, buildings, density of the human population, impervious surfaces, car traffic) and the presence of other wetlands, green open spaces, or vegetation in the surrounding landscape (e.g., forest patches). In synthesis, compared to natural or rural ponds, urban ponds are often smaller, shallower, younger, with a more regular shoreline, they include artificial structures (bottom or margin), and they are located within a built environment (Fig. 5). Water quality is often poor due to pollution, and urban ponds commonly support exotic species (including plants, fish, and ducks). We address each of these different factors separately in this review.

Design: pond area

Habitat surface area is a key factor in ecology, driving the species richness of ecosystems (Mac Arthur and Wilson 1967). This is also relevant for

Table 2. Examples of urban cocktails (sets of environmental factors) linked to the biodiversity of urban ponds.

Urban cocktail	Taxonomic group and measured metric	Ponds studied	Geographic location	Reference
Water conductivity (−); Proportion of urban land-use in a buffer area (within 1 km) (−); Presence of nearby wetlands (+)	Macroinvertebrates: family-level richness	30 urban waterbodies (20 stormwater and 10 other wetlands)	Ottawa, Canada	Hassall and Anderson (2015)
Urban land use in a buffer area (within 100 m) (−); Engineered edges (−); Shading (−); Nutrient-enrichment (−); Macrophyte stands and floating vegetation (+)	Macroinvertebrates: species richness, conservation value	30 ponds in a gradient of urbanization	West Midlands, UK	Thornhill et al. (2017)
Percentage of vegetation cover (+); Presence of stocked fish for recreational angling (−)	Macroinvertebrates: conservation value	60 old industrial mill ponds within the urban environment	UK	Wood et al. (2001)
Proportion of urban land use in a buffer area (within 1 km) (−); Road density (-); Introduced fish species (−)	Vertebrates (amphibians, turtles, snakes), macroinvertebrates: species richness and diversity	201 wetlands (ponds) from urban, agricultural and grassland areas	Front Range region, Colorado (USA)	Johnson et al. (2013)
Density of human residents in a buffer area (within 1000 m) (−); Water conductivity (−); Proportion of green open space within 1000 m of the pond (+)	Amphibians: species richness	65 urban ponds (from parks and garden)	Greater Melbourne, Australia	Hamer and Parris (2011)
Urban areas in a buffer area (within 100 m up to 1 km) (−); Road surfaces (total length; within 100 m up to 1 km of the pond) (−); Traffic measurements (mean number of vehicles per hour) (−)	European tree frog (*Hyla arborea*): presence/absence	76 ponds in a gradient of urbanization	Western Switzerland	Pellet et al. (2004*a*)
Total nitrogen concentrations (+); Aquatic vegetation (+); Nature of terrestrial habitat (+ or −); Area of wetlands in a 100-m buffer belt (+)	Amphibians: species presence/absence	75 urban wetlands (stormwater, natural upland, and river valley)	Edmonton, Canada	Scheffers and Paszkowski (2013)
Extent of vegetation in the riparian zone (+); Extent of vegetation in the wider landscape (+)	Amphibians: species presence/absence	320 wetlands in a gradient of urbanization	Canberra, Australia	Westgate et al. (2015)
Percentage of impervious surface in a buffer area (−); Distance to nearest forest patch (−); Pond depth (+); Hydroperiod length (+)	Amphibians: species presence/absence	100 ponds, wetlands, and swales	Gresham, OR, USA	Guderyahn et al. (2016)
Water conductivity (−); Heavy metal pollution (−)	Eastern long-necked turtle (*Chelodina longicollis*): relative abundance	55 wetlands across an urban–rural gradient	Melbourne, Australia	Stokeld et al. (2014)
Number of wetlands in a buffer area (+); Perimeter that was vegetated (+); Surface area (+); Distance to nearest wetland (+); Public accessibility (+); Shoreline irregularity (+)	Waterbirds: community structure, abundance, density	53 waterbodies	Southeastern suburbs of Melbourne, Australia	Murray et al. (2013)

Note: Impact is indicated as being either positive (+) or negative (−). These examples have been chosen to encompass a range of taxonomic groups and geographical regions.

pond species richness, although contrasting patterns or responses have been reported according to the taxonomic group considered (Oertli et al. 2002). Several case studies, conducted in different cities and with different taxonomic groups, report a relationship between pond surface area and freshwater biodiversity (see details in Appendix S1). The relationship tends to be positive for microcrustaceans (Cladocera; Pinel-Alloul and Mimouni 2013, Mimouni et al. 2015), macroinvertebrates (Hill et al. 2015), specifically aquatic insects (Blicharska et al. 2016), dragonflies (Jeanmougin et al. 2014), amphibians (Parris 2006), and insectivorous bats (Straka et al. 2016).

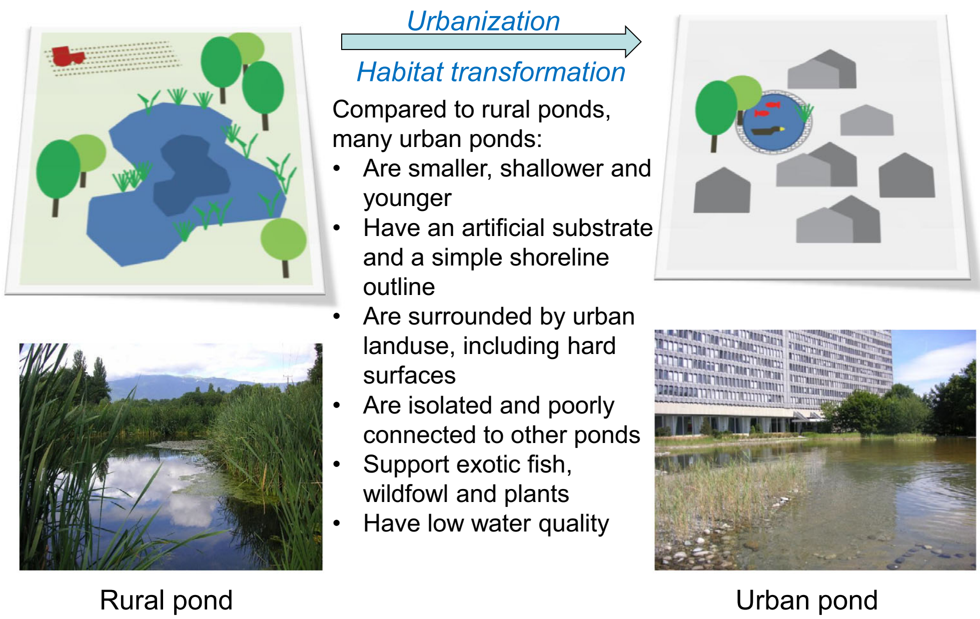

Fig. 5. Schematic representation of a "typical" urban pond compared to a rural pond. Urban pond management is targeted to enhance various services useful to society, but rarely for the provision of habitat for native biodiversity (Oertli and Ilg 2014).

An increase in pond surface area has also been linked to a greater number of individuals (e.g., waterbirds; Murray et al. 2013), increased reproductive success (e.g., amphibians; Fuyuki et al. 2014), and changes in species assemblages (e.g., waterbirds or macroinvertebrates; Sanderson et al. 2005, Murray et al. 2013). Some species are also linked to a given pond-size range (e.g., amphibians, waterbirds, insectivorous bats; Hamer et al. 2012, Murray et al. 2013, Straka et al. 2016).

However, there are also cases where no significant habitat surface area effect has been recorded. This was the case with the composition of assemblages (e.g., birds; McKinney et al. 2011) or species behavior (wetland passerines movement between sites; Calder et al. 2015). In contrast, in Dortmund (Germany), a decrease in dragonfly species richness was observed with increased pond surface area (Goertzen and Suhling 2013). In this particular example, the larger ponds had poor-quality, homogenous habitat due to impairment by dense waterfowl populations.

Small ponds can be particularly abundant in some urban areas (e.g., residential areas), and in the UK, the number of garden ponds has been estimated at 2.5–3.5 million with a mean size of 1 m^2 (Davies et al. 2009). Despite having a limited faunal diversity (Hill and Wood 2014), garden ponds provide a haven for some species of specific conservation interest (e.g., the common frog *Rana temporaria* and common toad *Bufo bufo* in the UK; Davies et al. 2009). The dense networks of garden ponds contribute to regional connectivity, and these ponds act as stepping stones, refuge areas, and breeding sites.

Pond area is also often closely related to the hydroperiod, with larger ponds that tend to have a longer hydroperiod which may benefit some species but be detrimental to others (see *Hydroperiod*).

Implications for management.—These results underline that for the same urban pond type, a larger area is generally valuable for promoting biodiversity. Large ponds (e.g., >0.5 ha) have the potential to support greater species richness (alpha diversity) for several taxonomic groups (although not all). However, assemblage composition may change with an increase in surface area, and this suggests that in order to promote biodiversity at the city scale (gamma diversity), it is important to have a diversity of pond sizes.

The pondscape approach (Boothby 1997, Hill 2018) is particularly relevant here, and there is a need to diversify pond sizes within a pond network. Larger ponds are relatively rare in some cities (McKinney and Charpentier 2009) where it is therefore important to promote their creation to restore the range of pond surface area in the landscape. This is nevertheless not always the case, and large ponds can be well represented in cities when they bring an ecosystem service useful for the citizen (e.g., water purification, flood protection, leisure).

Design: pond margins

The margins of a natural waterbody are often varied, including slopes of different angles, some densely colonized by helophytes (e.g., reeds, cattails, sedges) and with large drawdown zones. The pond margin therefore provides a diverse range of microhabitats used by the fauna for resting, reproduction, sheltering from predators or unfavorable weather conditions, feeding, and migration between aquatic and terrestrial habitats. As a result, the pond margin supports a large part of a pond's biodiversity and constitutes a key element of pond design (Williams et al. 1997).

In urban environments, the pond shore is often reduced to a very narrow zone, poorly vegetated, and sometimes replaced by an artificial concrete substrate. Inevitably, this has negative consequences for biodiversity, as shown in various studies. For example, a positive relationship between frog occurrence and shallow margins was reported for urban wetlands in Edmonton (Canada; Scheffers and Paszkowski 2013), and for urban ponds in Greater Melbourne (Australia; Hamer et al. 2012). The slope of pond margins is clearly linked to the presence of emergent and fringing vegetation that itself positively impacts biodiversity. We develop this topic further in the *Aquatic vegetation* below.

Artificial structures (such as concrete or stone walls) are also often a major component of pond margins and can potentially hinder the completion of the life cycle of amphibious species (invertebrates or vertebrates). A vertical wall surrounding a pond, compared with a gently sloping bank, can impair the quality of the habitat and lead to a decrease in the number of amphibian species present (Parris 2006). This is because ground-dwelling frogs that cannot climb vertical surfaces are excluded. Concrete walls will also affect the emergence of some dragonfly species, because the transition from the aquatic larval to the aerial adult stage requires emergent plants (Corbet 1999). Aquatic beetles (like Dytiscidae) also need a muddy substrate for oviposition and pupation.

A few investigations have, however, shown that steep slopes at the margin of ponds can benefit some elements of biodiversity. Ponds with such feature were associated with higher amphibian species richness in the city of Potchefstroom (South Africa; Kruger et al. 2015). Similarly, the occurrence of one species out of nine was positively associated with increased depth at the margin of urban ponds in Greater Melbourne (Hamer et al. 2012). Such contrasting results highlight that, at the pondscape scale, a diversity of margin characteristics should be encouraged. Ponds with steep banks can play a role in supporting and enhancing the regional species richness, although they should not predominate in the pondscape.

*Implications for management.—*In general, shallow pond margins favor the development of aquatic vegetation that in turn provides a high diversity of habitats for fauna. This feature of the pond design should be promoted so that it increases and becomes the norm in the urban pondscape. Concrete substrates and pond walls often hinder biodiversity and, generally, should be avoided. At a pondscape scale, each pond type can, however, make a useful contribution to regional biodiversity.

Water quality

*Pollution of urban ponds.—*In natural landscapes, ponds are relatively untouched by pollution and can remain in a relatively pristine state. Airborne pollution can nevertheless be a source of pollution in certain regions, leading to acidification or nutrient enrichment. When agriculture predominates in a rural landscape, ponds can be subjected to many types of pollution (Brönmark and Hansson 2002). The prevalent source of pollution tends to be nutrient input, leading to pond eutrophication, but pesticides or airborne pollution are also significant. Pond biodiversity is therefore often under pressure from multiple pollution sources.

In urban environments, only few ponds can escape water pollution. This problem can be particularly acute due to surface runoff that can bring suspended solids, nutrients, heavy metals, pesticides, polychlorinated biphenyls, polycyclic aromatic hydrocarbons, endocrine disrupting chemicals, salts, bacteria, and many other pollutants. This topic is largely covered by literature dealing with streams (Paul and Meyer 2001) or specifically with stormwater ponds (Collins et al. 2010), and so will not be reviewed further here. Ponds are also subjected to chemical treatments intended to control unwanted species (mollusks, mosquitoes, cyanobacteria) or improve water clarity. All these urban pollutants have a potential impact on aquatic biodiversity. They can totally exclude sensitive species and impair the fitness of the remaining species (activity, physiology, life history). This will be explored further in the following sections, with an emphasis on water conductivity (as an indicator of urban pollution) and on nutrient inputs.

Some ponds in the urban matrix can, however, have surprisingly good water quality, in some cases even better than in intensively managed rural landscapes. This includes, for example, ponds in which the water is renewed regularly (often with tap water) and also those which have periodical drying and dredging. Such urban management practices are likely to negatively impact freshwater biodiversity, but little information is available on this subject.

*Conductivity and salts.—*Urban pond water can naturally have high electrical conductivity (up to 3000 μS/cm) due to the underlying geology, as, for example, urban ponds in Vienna (Austria; Schagerl et al. 2011). Conductivity is, however, often an indicator of broader pollution, reflecting road-salt inputs (Brand et al. 2010) or heavy metal pollution (Stokeld et al. 2014). Conductivity is often highly correlated with the level of urbanization in the catchment (Hassall and Anderson 2015) and with several other parameters (pH, water quality, salt content, type of urbanization). Existing studies have rarely been able to disentangle the impact of conductivity from these other correlated variables. Pond species richness is often lower when conductivity is high, and this chemical parameter is most likely itself a negative driver of biodiversity. A lower species richness linked to high conductivity has been observed for diatom species (phytoplankton) in urban ponds in Austria (Schagerl et al. 2011), macroinvertebrate families in wetlands and stormwater ponds in Ottawa (Canada; Hassall and Anderson 2015), amphibian species in urban or suburban ponds in Melbourne (southeastern Australia; Hamer and Parris 2011, Hamer 2016), and in wetlands in the Front Range region of Colorado (USA; Johnson et al. 2013). High water conductivity is also associated with a lower probability of the presence of some threatened species, including *Hyla arborea* in Western Switzerland (Pellet et al. 2004*b*). The salinization of urban waterbodies is an emerging problem and is forecasted to increase in the future, especially in regions with cold climate. For a review of the impact of salinization on freshwater organisms, see Castillo et al. (2018). However, all species are not equally sensitive to water conductivity. The impact of conductivity can sometimes be observed on species abundance rather than on presence/absence. This was the case for a freshwater turtle, the eastern long-necked turtle (*Chelodina longicollis*), which had lower abundances in urban wetlands with higher conductivity in Melbourne, Australia, although conductivity did not impact occurrence (Stokeld et al. 2014). Lower abundance was also observed for amphibians in ponds with higher water conductivity in boreal Alberta (Canada; Browne et al. 2009). In contrast, other groups can be more abundant in ponds with high conductivity, for example, mosquitoes in wetlands and mesocosms in Columbus, Ohio (USA; Yadav et al. 2012).

The impact of conductivity on organisms is often linked to specific ions that are the main cause of high conductivity readings. Key compounds include salt (NaCl) but also heavy metals and other substances. Road salts at relatively low concentrations have toxic effects on amphibians (Sanzo and Hecnar 2006, Jones et al. 2017). A microcosm investigation with an amphibian species (*Hyla versicolor*) showed that survival of embryos is negatively correlated with water conductivity (Brand et al. 2010). Toxicity is likely related to loss of osmoregulatory control as a result of NaCl exposure (Jones et al. 2015). The negative impacts of salinity on biodiversity have been demonstrated for several other taxonomic groups, and microinvertebrates appear to be particularly sensitive to salinity (Castillo et al. 2018). Cascade effects across an entire trophic chain have also been reported: When zooplankton are negatively impacted, algae or phytoplankton growth can be stimulated (Van Meter et al. 2011*b*, Jones et al. 2017). The food web structure can be deeply altered: For example, high algal abundance can promote tadpole presence (Van Meter et al. 2011*a*). Road deicing salts are not, however, toxic for all amphibian species (Gallagher et al. 2014). Moderate salinity can also indirectly benefit amphibians by reducing the prevalence of chytrid infection (Heard et al. 2014).

Nutrients.—Small waterbodies have naturally higher concentrations of nutrients in their water and sediment than larger systems (e.g., lakes), and their productivity levels are therefore often mesotrophic to eutrophic, or even hypertrophic. However, they may still support a rich biodiversity adapted to these conditions (Rosset et al. 2014). An excess of allochthonous inputs of nutrients can lead to a deterioration of habitat conditions which may only support a selection of resistant species (e.g., tolerant of anoxic conditions). If all ponds in a network suffer from high nutrient inputs, then regional biodiversity will be negatively impacted. Such situations can occur in both urban and non-urban areas. An excessive input of nutrients is often linked to the use of fertilizers in the pond catchment (e.g., from lawns in parks or private gardens), wastewater (often from domestic misconnections or illegal discharges), or animal waste (including pets). High nitrogen inputs are also attributed to combustion-derived N aerosols or NOx associated with transportation.

The concentrations of nutrients in urban ponds can be similar to those of ponds from other types of land uses (Johnson et al. 2013, Vincent and Kirkwood 2014). Running waters are often lower in phosphorus and nitrogen content in urban areas compared to agricultural areas (Paul and Meyer 2001), and urban ponds can also be expected to be nutrient-poor compared to ponds in agricultural areas. Some types of urban ponds are, however, particularly rich in nutrients, such as stormwater ponds that are created in urban areas specifically for trapping nutrients, ponds that are over-stocked with fish, or ponds that support large duck populations, often as a result of over-feeding.

In urban areas, the existing literature rarely reported nutrients to be the main driver of pond species richness. Nutrients can, however, affect community composition in urban ponds, as shown for algal assemblages in Vienna (Austria; Schagerl et al. 2011), macroinvertebrates in the West Midlands (UK; Thornhill et al. 2017), and amphibians in Edmonton (Canada; Scheffers and Paszkowski 2013). In this last example, the occurrence of one of the amphibian species (the wood frog *Lithobates sylvaticus*) was positively associated with high levels of nitrogen. High levels of nutrients (and eutrophication) are also known to be a key driver of cyanobacterial presence and blooms in urban ponds (Peretyatko et al. 2010, Vincent and Kirkwood 2014, Waajen et al. 2014, Castilla et al. 2015, Lurling et al. 2017). This is a particularly deleterious impact of eutrophication in urban areas, as some cyanobacteria produce toxins which can be harmful to people and their pets. Furthermore, urban ponds are also often stocked with carp, and large populations can lead to excessive bioturbation and phosphorus release from the sediments, which then favors cyanobacteria blooms (Waajen et al. 2014). In these cases, eutrophication will potentially raise health concerns and drive pond management in urban areas toward measures to reduce the risk of harm.

Other pollutants.—Evidence about the impact of urban pollutants on biodiversity abounds. Various pollutants from urban ponds have been shown to be present in organisms or even to bioaccumulate. This was, for example, the case with heavy metals in aquatic plants (Bonanno 2011), Gammaridae (Lieb and Carline 2000), fish (Campbell 1994), and amphibians (Priyadarshani et al. 2015), with pesticides in amphibian (Smalling et al. 2013) and damselfly (Van Praet et al. 2014), and with polycyclic aromatic hydrocarbons in adult damselflies (Heintzman et al. 2015).

These urban pollutants impact the biology of species in a variety of ways. For example, heavy metals affect the survival, behavior, and immune system of amphibians (Priyadarshani et al. 2015, Sievers et al. 2018*b*) and increased levels of heavy metal pollution in ponds negatively influence the activity of some species of bats (Straka et al. 2016). Estrogen contamination linked to urbanization is also suspected of modifying sex ratios in amphibian populations (Lambert et al. 2015). Domestic wastewater contamination in urban ponds may be a contributor to intersex in wild amphibians (Smits et al. 2014). Chemicals used to manage gardens and ponds are known to influence amphibian immune function: Even low pesticide doses result in reduced antibody production in leopard frogs (*Rana pipiens*; Albert et al. 2007). Urban pollutants can also act as a filter for species assemblages. For example, some Chironomidae (dipteran) species were negatively associated with sediment pollution in Australian urban wetlands, when other species were positively associated with this pollution (Carew et al. 2007). Bat species richness decreased with increasing levels of heavy metal pollution in urban wetlands in Melbourne (Australia; Straka et al. 2016), highlighting the pollution sensitivity of some species.

*Implications for management.—*Evidence from this review shows that nutrients are often not the main problem for urban biodiversity. Nevertheless, nutrient pollution can contribute to the degradation of some pond types (such as stormwater ponds). Negative impacts on biodiversity are more often caused by the other pollutants present in urban ponds. Therefore, the presence of these pollutants should be one of the main concerns during an ecological assessment of an urban pond. Management measures to address pollution may not necessarily be required, depending on the pond type and on the associated targeted ecosystem services. For example, some ponds are created specifically for trapping pollutants, and therefore, the presence of pollutants in these ponds is inevitable and demonstrates they function adequately. However, management actions to discourage wildlife from using highly polluted ponds may be appropriate in some cases, particularly where threatened species are present (Sievers et al. 2018*c*). The usual management measures to reduce pollution can be undertaken (e.g., management of the water source and of the catchment area) at ponds that are managed specifically for their aesthetic values and their biodiversity. Management should also consider both the waterbody and pondscape scale, with the main objective as the diversification of pond types within a pond network.

Hydroperiod

In natural or rural waterbodies, hydroperiod (i.e., duration and frequency of inundation) is a particularly important factor driving the composition of biotic assemblages. The biotic communities of temporary waterbodies often support fewer species than those of permanent ponds, but include specialized species and many threatened species (Collinson et al. 1995, Nicolet et al. 2004). Hydroperiod is particularly important for amphibians (Semlitsch 2000, Snodgrass et al. 2000), as a factor per se, but also to limit predator presence (e.g., fish; Chester and Robson 2013, Hamer and Parris 2013).

The importance of hydroperiod for biodiversity has also been widely demonstrated in urban systems. Because the hydroperiod influences amphibian occurrence, the majority (86%) of the publications reviewed on "hydroperiod" was focused on this group and included urban ponds from different regions: Baltimore County, Maryland (USA; Gallagher et al. 2014), southeastern New Hampshire (USA; Veysey et al. 2011), central Pennsylvania (USA; Rubbo and Kiesecker 2005), and southern Australia (Hamer and Parris 2013). Temporary habitats are free of predatory fishes and can therefore benefit certain amphibian species (Hamer and Parris 2013). Note, however, that permanent waterbodies are not detrimental to amphibian populations when they are free of predatory fish (Westgate et al. 2015). The amphibians most affected by urbanization are those associated with short hydroperiods (Pillsbury and Miller 2008), because temporary ponds are often rare in urban landscapes.

Water depth is also a crucial factor for wetland vegetation, because most emergent plants grow where the water depth in less than 60–80 cm. In addition, hydroperiod characteristics have a significant impact on the survival of plant species, as, for example, illustrated by a study on the plant communities of stormwater wetlands in Brisbane, Australia (Greenway et al. 2007). For macroinvertebrates, hydroperiod is also an important factor affecting assemblages, as shown in a study of urban ponds in Milnrow, UK (Sanderson et al. 2005), and for aquatic bugs and beetles in ponds in Cape Town, South Africa (Legnouo et al. 2014).

Human actions can modify the hydroperiod of urban ponds, making inundation more unpredictable in terms of both frequency and intensity, with subsequent impacts on biodiversity. Moderate hydrological perturbation may not have much consequence on biotic assemblages, but drying of the pond margin (also potentially the reed belt) or even the whole pond will have profound impact. Some, often smaller, ponds (e.g., garden or park ponds) are totally emptied for cleaning and for the removal of organic matter. These measures can help maintain good water quality for the short term but can also remove most life from the pond (adults, larvae, and propagules), requiring recolonization by flora and fauna. While potential colonists often originate from propagules present in the urban pondscape, they also arrive via deliberate introductions (e.g., plants, fish) by pond owners. Amphibians are particularly impacted by the alteration of a pond hydrological cycle, because the completion of larval metamorphosis may be impaired if the pond dries too early (if the hydroperiod is shortened) or because of increased predation (if the hydroperiod is lengthened or connections made with lakes, rivers, or canals which support fish; Semlitsch 2000).

*Implications for management.—*Urbanization tends to modify pond hydrology and favors permanent waterbodies (Rubbo and Kiesecker 2005, Hamer and Parris 2013, Urban and Roehm 2018). These more permanent ponds support higher overall animal diversity but exclude temporary-pond specialists. Conserving the full assemblage of pond species in urban areas will require protecting and creating temporary ponds (Hamer and Parris 2013, Urban and Roehm 2018). However, climate warming is forecasted to cause a reduction in the hydroperiod of some ponds (Wilson et al. 2013) and could lead to an increase in the abundance of temporary ponds in cities. At the pondscape scale, management should also promote a range of natural hydroperiods to support targeted biodiversity and avoid artificial hydroperiods that are harmful to this biodiversity.

Urban heat island

The impact of the urban heat island effect on the temperature regime of urban ponds is obvious. For example, urban ponds tended to be warmer in a set of 201 ponds investigated in Front Range region of Colorado (USA; Johnson et al. 2013) and increased temperatures may also reduce the hydroperiod of ponds (Wilson et al. 2013). Warmer water will exclude many cold-adapted or stenothermic species and will favor eurythermic species, in the same way that it is already occurring in natural landscapes (Rosset and Oertli 2011). The impact on species richness can in some cases be positive, including the colonization by species coming from warmer areas exceeding the number of species excluded. The species living in urban ponds will therefore tend to present species traits linked to warmer temperature than the species living in ponds from the surrounding rural landscape. Evidence of the effect of temperature increase has been presented for terrestrial urban ecosystems (Piano 2017), but still remains to be studied for urban ponds. Warmer temperatures in urban ponds can also have an impact at the species level. Species can present warm-adapted populations or even genetic adaptations to warm temperatures. This has been demonstrated in urban ponds of the Flanders region (Belgium), respectively, for the Cladocera (Crustacea) *Daphnia magna* (Brans et al. 2017) and the damselfly *Coenagrion puella* (Odonata; Tuzun et al. 2017).

*Implications for management.—*The management at the local scale (pond) can promote ponds in shaded (and cooler) areas, for example, near buildings or trees.

Aquatic vegetation

The presence of aquatic macrophytes (emergent, submerged, floating) is a well-known factor that tends to increase biodiversity in natural ponds (Biggs et al. 1994). It is therefore not surprising that the same positive relationship has been reported in most studies of urban ponds. However, the plant communities of urban ponds often differ from those of non-urban ponds in terms of species composition, abundance, spatial organization, and temporal dynamics. Site managers tend to promote a selected type of vegetation, motivated by aesthetic concerns (gardens, parks) or by the functional service targeted (e.g., water treatment; Dhote and Dixit 2009). Plants in urban ponds are often highly managed, and urban ponds partially or totally constructed of concrete present a lower potential for the development of rooted aquatic plants than those with a more natural substrate. In addition, most of the margins of urban ponds are free of vegetation and lack the dense bed of emergent macrophytes that usually characterizes natural ponds. Finally, for several types of urban ponds, management measures include mowing or removing aquatic macrophytes to maintain various pond functions (e.g., drainage, water purification, prevention of eutrophication, aesthetic value, leisure). For these reasons, large macrophyte beds are often missing from urban ponds, and there is less vegetation to provide habitat for animals (in terms of species diversity and extent, and duration throughout the year) than in rural or natural ponds, leading to lower biodiversity value.

The presence of vegetation is a key factor driving the presence of amphibians in urban ponds, although not for all species. This was shown in ponds from several cities, including Shanghai (China; Zhang et al. 2016), Melbourne (Hamer et al. 2012) and Canberra (Westgate et al. 2015; Australia), Portland (Oregon, USA; Holzer 2014), and Edmonton (Canada; Scheffers and Paszkowski 2013). Turtles are also dependent on the presence of vegetation. For example, in southeastern New Hampshire, abundance of the common aquatic turtle (*Chrysemys picta*) was greater in ponds with extensive stands of marginal vegetation than in ponds lacking these features (Marchand and Litvaitis 2004). Many wetland-dependent bird species are linked to habitat structure and the extent of emergent vegetation at a pond. This was shown for bird communities in wetlands in the urbanized regions of Chicago (USA; Ward et al. 2010) and Melbourne (Australia; Murray et al. 2013).

Invertebrates also respond to the presence of vegetation in urban ponds. Caddisflies (Trichoptera) are particularly linked to aquatic vegetation: This group was more species rich at intermediate coverage of vegetation in urban ponds in Stockholm, Sweden (Blicharska et al. 2016). Dragonfly (Odonata) diversity was positively linked with the coverage of submerged macrophytes in ponds in Paris, France (Jeanmougin et al. 2014), and with the diversity of aquatic and terrestrial vegetation in Dortmund, Germany (Goertzen and Suhling 2013). Macroinvertebrate assemblages of high conservation value were more likely to be found in ponds with complex macrophyte stands and floating-leaved vegetation in urban ponds from the West Midlands of the UK (Thornhill et al. 2017).

The practice of removing or mowing vegetation in urban ponds is assumed to be one of the factors leading to low aquatic plant diversity (Noble and Hassall 2015) and also impacts insect species richness (Blicharska et al. 2016). The management of marginal and aquatic vegetation is often coupled with the removal of fine sediments by dredging, a management practice frequently associated with angling ponds which tend to have reduced macroinvertebrate diversity (Wood et al. 2001).

*Implications for management.—*The presence of structured vegetation in ponds, including large beds of submerged, floating-leaved, and emergent macrophytes and a shoreline well vegetated by helophytes, is a key factor for sustaining biodiversity. Management practices promoting biodiversity can easily enhance these conditions, for example, by adjusting pond design and mowing regimes. Actions to enhance vegetation can be some of the easiest and most effective management measures to support biodiversity. At the pondscape scale, having several ponds with a range of macrophyte coverages and structural complexities is likely to provide the greatest opportunity for urban pond diversity.

Non-native invasive species

Ponds, like other ecosystems, are being colonized by an increasing number of non-native species. Several of these species can establish large populations, disperse successfully at the regional scale, and become invasive. Urban ecosystems, including urban ponds (Oertli et al. 2018), constitute hotspots of non-native species introduction. In cities, intentional introduction is the main pathway for the colonization of ponds by non-native species. For example, garden ponds are planted with non-native plants or stocked with non-native fish. The aquarium and ornamental plant trade are responsible for many releases of species in the environment (Padilla and Williams 2004).

Plant communities in urban ponds include a large proportion of non-native species, including invasive species (Magee et al. 1999, Ehrenfeld 2008, Oertli et al. 2018), and this has consequences for native biodiversity. These non-native plant species can trigger a cascade of altered species interactions (Mackay et al. 2016). Such ecosystem changes can even lead to the changes of the transmission dynamics of vector-borne pathogens that imperil human health, such as West Nile virus in mosquitoes (Mackay et al. 2016). Non-native vegetation can also negatively impact vertebrate presence. For example, non-native vegetation was negatively associated with occupancy for several amphibian species in wetlands from an urbanized landscape in Gresham, Oregon (Guderyahn et al. 2016).

Introduced fish are the faunal group most likely to be recorded in urban wetlands (Johnson et al. 2013); this topic is developed in the next section *Fishes*. Mollusks and crustaceans are among the most frequent freshwater macroinvertebrate invaders, and the pet trade is considered to be one of the main pathways for new introductions (Patoka et al. 2017). These two groups of invertebrates are therefore broadly distributed in urban ponds. For example, the New Zealand mud snail *(Potamopyrgus antipodarum)* has colonized many stormwater ponds in Eastern Scotland (UK; Briers 2014). However, the impact of these species on the functioning of urban ponds or on native biodiversity requires further investigation.

Non-native reptile or amphibian species can also be present in urban ponds. Non-native turtles are often deliberately introduced to urban ponds, where they may compete with native species, especially for basking sites (Spinks et al. 2003). Urban wetlands are also more likely to support non-native bullfrogs (*Lithobates catesbeianus*; Johnson et al. 2013), a species which tends to reduce native amphibian diversity (Kiesecker et al. 2001, Rowe and Garcia 2014). For example, the leopard frog (*Lithobates pipiens*) decline in Colorado (USA) is linked to an increase in urban development and colonization by non-native bullfrogs (Johnson et al. 2011).

Implication for management.—Invasive non-native species can be a threat to the biodiversity of urban ponds. Preventing the introduction of such species within a region is widely promoted as being a more cost-effective and environmentally desirable strategy than actions undertaken after establishment (Leung et al. 2002). These measures rely on social interaction, including good communication with stakeholders (e.g., managers, the general public) and also the use of the relevant legislative frameworks (i.e., regulation on species trade). Early detection and eradication are two other complementary management strategies, which should be linked to the level of risk posed by a particular invasive species.

Fishes

Native fishes can be present in ponds, and even some species that are of conservation concern (Copp et al. 2008). Nevertheless, in most urban ponds, the presence of fish is linked to introductions, mostly of non-native species, often at high densities. The number of ornamental varieties of fish in ponds was found to increase as distance of a pond from the nearest road decreased (Copp et al. 2005), highlighting a human-driven pathway of pond colonization. Mosquitofish (*Gambusia* spp.) are also often introduced by managers or pond owners with the intention of controlling mosquitoes. These species tend to represent a large proportion of the fish communities in urban ponds and exert a high pressure on amphibian populations, as demonstrated, for example, in urban wetlands in the Willamette Valley, Oregon (USA; Pearl et al. 2005). Other examples include urban ponds in Australia, where tadpoles have been shown to suffer high rates of predation by the invasive mosquitofish (*Gambusia holbrooki)* in Sydney (Remon et al. 2016) and Melbourne (Hamer and Parris 2013). Mosquitofish can also affect the composition of the zooplankton community through selectively feeding on larger zooplankters (Pyke 2008). Other fish species, such as aquarium and garden-pond species, can also have an impact on pond biodiversity. For example, the pumpkinseed sunfish (*Lepomis gibbosus*) is widely distributed in Europe and occurs especially in urban waters. In the Netherlands, urban ponds populated by this fish supported much lower macroinvertebrate abundance than ponds without (van Kleef et al. 2008). Urban ponds are also often stocked with high densities of large non-native carp, leading to major changes in biodiversity. Indeed, large population of carp can prevent the growth of submerged macrophytes, directly (through herbivory) or indirectly (through increasing turbidity), and also prey on zooplankton (especially larger individuals), thus shifting pond ecosystems toward a turbid, phytoplankton-dominated state (Scheffer et al. 1993).

*Implications for management.—*As for invasive non-native species, management strategies need to be targeted at prevention, in particular through communication with stakeholders and private pond owners and through an appropriate legislative framework. If necessary, eradication can also be an appropriate solution (e.g., net fishing or electrofishing, or pond draining and refilling). At the pondscape scale, it is nevertheless possible to maintain some fish-stocked ponds, as their particular species assemblage can bring a contribution to regional diversity.

Landscape-scale factors

For natural or rural ponds, the landscape-scale environmental factors are of central importance for pond biodiversity (Cottenie et al. 2003, Jeffries 2005, Hill 2018) for several reasons. Firstly, the biodiversity of a given pond is strongly linked to the presence of other ponds in the landscape, and together, these ponds constitute the pond network (pondscape). Indeed, many species have metapopulations in ponds across the landscape. In addition to pond density and location in the landscape, the surrounding land use determines the capacity of species to move from one pond to the other. Secondly, land use in the pond catchment has a direct influence on water quality. Thirdly, the landscape around a pond provides habitats for the terrestrial stages of amphibious species (e.g., amphibians, many insects). In urban areas, the importance of the landscape scale is expected to be magnified because the environment is often hostile to species movement, polluted, and lacking key resources for amphibious species. All these factors can reduce the value of pond habitats and affect aquatic biodiversity.

*Scale of investigation.—*Due to the importance of landscape factors for urban pond ecology, these have been included in most studies researching the impact of urbanization on pond biodiversity. However, the extent of the investigated geographical area has varied substantially between studies (see Table 1 for an overview). For example, the smallest radius (50 m) had the greatest impact on the zooplankton community in urban ponds in Belgium, among 7 radii investigated up to 3.2 km (Gianuca et al. 2018). Much larger radii (800 m–1.8 km) were found to influence vegetation and benthic macroinvertebrate assemblages in urban ponds in Ottawa (Patenaude et al. 2015). Other scales shown to be relevant to urban pond ecosystems include 100 m for macroinvertebrates in West Midlands, UK (Thornhill et al. 2017), 100–300 m for wetland birds in eastern Massachusetts, USA (Tavernia and Reed 2010), and 500 m for submerged and floating-leaved macrophytes in Hyogo, Japan (Akasaka et al. 2010). Differences in the scale of influence have also been reported for the same taxonomic group (e.g., amphibians): 200 m in Gresham, Oregon, USA (Guderyahn et al. 2016), 300–1000 m in the Eastern and Central USA (Marsh 2017), and 1 km in southeastern Australia (Villasenor et al. 2017).

These differences are undoubtedly linked to the taxonomic groups investigated, but also to the type of matrix and the urban form. Measures of urbanization were also very heterogeneous (Table 1), and this is likely to have affected the results. Furthermore, the same category of urbanization (e.g., buildings, roads) can have very different effects on species dispersal depending on the city considered: For example, building height can be very heterogeneous and so is the intensity of vehicle traffic. In consequence, we recommend that future studies include several landscape scales, with buffers between 50 m and 2 km around urban ponds. The type of urban matrix should also be carefully described, in particular the elements that may affect the dispersal of individual organisms or their propagules (e.g., building heights, vehicle traffic, corridors, stepping stones).

*Landscape urbanization.—*The number of plant and animal species inhabiting a pond is generally impacted by the land use in the buffer area around urban ponds, and in particular the amount of land covered by buildings and/or impervious surfaces. This has been shown, for example, with aquatic insects in Stockholm, Sweden (Heino et al. 2017), dragonflies in Paris, France (Jeanmougin et al. 2014), frogs in central Iowa, USA (Pillsbury and Miller 2008), and amphibians, aquatic reptiles, aquatic insects, mollusks, and crayfish in the Front Range region of Colorado, USA (Johnson et al. 2013). There is an extensive literature on this topic for amphibians, which was partly reviewed by Hamer and McDonnell (2008), but other taxonomic groups have clearly been less thoroughly investigated. The decrease in species richness at urban ponds is linked to the filtering of the regional species pool, with the exclusion of many species sensitive to urbanization. This was, for example, demonstrated by a study of the endangered European tree frog (*Hyla arborea* L.) which was excluded from the most densely urbanized areas in northeastern Germany (Fischer 2015) and in Western Switzerland (Pellet et al. 2004*a*). Similarly for ponds in Gresham, Oregon (USA), urbanization of the area around ponds was negatively correlated with site occupancy for all amphibian species (Guderyahn et al. 2016). In some locations, individual species do not become locally extinct but their abundance is significantly reduced. This was the case for most anuran species in urbanized ponds around the cities of Ottawa and Gatineau (Canada; Gagne and Fahrig 2007) and in central Iowa (USA; Pillsbury and Miller 2008).

The urbanized matrix frequently also includes vegetated areas (grasslands, shrubs, forests). If these green areas are within relatively short distances of ponds (i.e., within 50–1000 m), they can provide habitats for the terrestrial stages of amphibious species. Green areas can have a positive or negative effect on water and sediment quality, by filtering stormwater or acting as a source of nutrients and/or pesticides. Forested areas are known to be important for amphibian communities (Simon et al. 2009), as demonstrated in urban ponds in Portland (Oregon, USA; Holzer 2014). Distance to the nearest forest patch was negatively correlated with site occupancy for all amphibian species in ponds in Gresham, Oregon (Guderyahn et al. 2016). Green spaces, covered by lawns, meadows, shrubs, and trees (e.g., in parks or gardens), can also benefit biodiversity. For example, the diversity of terrestrial vegetation was positively linked to the diversity of dragonflies in urban ponds in Dortmund, Germany (Goertzen and Suhling 2013). Similarly, amphibian species richness increased substantially in urban ponds surrounded by a high proportion of green open space in Melbourne (Australia; Hamer and Parris 2011). Maintaining connectivity between ponds and greenspaces in suburban areas is also important for semi-aquatic turtles, as shown in the Charlotte Metropolitan area, North Carolina (USA; Guzy et al. 2013). However, some types of green spaces can impair the biodiversity value of ponds. The proportion of the pond catchment covered by intensively managed lawn was negatively correlated with zooplankton richness, macrophyte abundance, molluskan presence, and amphibian presence in stormwater ponds in Madison, Michigan (USA), which was probably linked to the use of fertilizer and pesticides (Dodson 2008).

Consequence of landscape urbanization: fragmentation and pond isolation.—One of the main impacts of urbanization in the area around ponds is landscape fragmentation. This leads to the division of pond networks into smaller networks, and sometimes ultimately to the complete isolation of certain ponds. A pond network can support many species that act as metapopulations, thus requiring the frequent exchange of individuals or propagules between ponds for the persistence of metapopulations over time. Therefore, any decrease in the efficiency of dispersal between ponds can threaten these species. Fragmentation is obviously species-specific, differing in its impact on species with active terrestrial dispersal (e.g., amphibians), active aerial dispersal (e.g., dragonflies), passive aerial dispersal (plant seeds), or passive dispersal through a carrier (e.g., microcrustaceans or mollusks transported by birds or mammals). Within these four broad categories, distinction can also be made regarding a species' ability for dispersal. For example, strong flyers (e.g., anisopteran dragonflies, birds) move much longer distances than poor flyers (e.g., caddisflies), and so, critical dispersal distances will be very different according to the species considered. Without barriers, dispersal distances can range from some tens of meters (e.g., midges or caddisflies, with windless conditions) to several kilometers (e.g., waterbirds, anisopteran dragonflies). Barriers such as large roads, long and high buildings and rivers can significantly decrease the distance that a particular species can disperse across the urban landscape.

Habitat fragmentation and isolation as a result of urbanization are some of the main threats to amphibian populations (Hamer and McDonnell 2008), and consequently, there is an extensive body of research on this topic. The impact of fragmentation on amphibian populations has been demonstrated at the species level with the use of genetic tools, for example, for the growling grass frog (*Litoria raniformis)* in the urbanizing landscape of southern Australia (Hale et al. 2013, Keely et al. 2015), and the common frog *(Rana temporaria*) in urban sites from Brighton (UK; Hitchings and Beebee 1997). Fragmentation also affects amphibian species richness. In Melbourne (Australia), the most-isolated pond in a study was predicted to support only 12–19% of the amphibian species of the least-isolated pond (Parris 2006). Habitat fragmentation resulting from dispersal barriers (roads) was also reported in the Front Range region of Colorado (USA), including a negative relationship with amphibian species richness, aquatic reptiles, aquatic insects, mollusks, and crayfish (Johnson et al. 2013). The spatial effect of fragmentation was also demonstrated for aquatic macroinvertebrates in Milnrow, UK (Sanderson et al. 2005), and for zooplankton pond metacommunities in Columbia and Baltimore, Maryland (USA; Sokol et al. 2015). Within some taxonomic groups, small-sized species are expected to dominate urban communities, as dispersal limitation increases with increasing body size in zooplankton (De Bie 2012). For example, small cladoceran species dominated assemblages of more urbanized ponds in Belgium, whereas large-bodied, strong competitors prevailed in less urbanized systems (Gianuca et al. 2018).

In addition to the creation of barriers to dispersal, urbanization leads to habitat destruction and therefore to a reduction in pond density (e.g., by pond infilling). This also contributes to pond isolation and impacts metapopulations. A reduction in wetland density decreases the probability that populations will be rescued from extinction by nearby source populations: Local populations cannot be considered independent of source-sink processes that connect wetlands at the landscape or regional level (Semlitsch 2000). A large weight of evidence demonstrates the importance of these processes. For example, a highly significant correlation was observed between pond density and species richness of invertebrates and macrophytes in the Borough of Halton (northwest England; Gledhill et al. 2008). Macrophyte richness was correlated with the abundance of wetlands within 500 m of ponds in western Japan (Akasaka et al. 2010). The extent of wetlands in the surrounding landscape also had positive effects on aquatic vegetation cover and on the richness of benthic invertebrates in Eastern Ontario (Canada; Patenaude et al. 2015). Riparian corridors can partly mitigate the impact of fragmentation. There was evidence of the positive effect of aquatic connectivity on the occurrence of the striped marsh frog (*Limnodynastes peronei)* in the urbanized southeastern Australia, which emphasizes the importance of riparian corridors in urban settings (Hamer et al. 2012).

*Implications for management.—*Management is often much easier at smaller spatial scales than at the landscape scale, because it is linked to urban planning strategies. Management activities should first target the area immediately surrounding the pond, up to a radius of 2000 m, where good-quality terrestrial habitats (green spaces, forests, other aquatic habitats) should be encouraged, and where dispersal processes should be enhanced (by, e.g., reducing barriers to species movement, enhancing corridors and stepping stones). Management should then also target the pondscape, at the scale of the whole city, and aim to increase the density of ponds in the network (i.e., creating new, high-quality ponds) and promote species dispersal between ponds.

Disservices Provided by the Biodiversity of Urban Ponds

To date, the disservices potentially provided by the biodiversity of urban ponds have not been widely investigated, although these should be taken into account in developing management strategies promoting the biodiversity of urban areas. Evidence shows that there are several disservices associated with pond biodiversity in the urban environment and that these can sometimes lead to negative attitudes toward ponds, potentially leading to loss through infilling.

One of the most acute of these disservices is to provide a breeding habitat for biting insects such as mosquitoes. Some pond design features, such as shallow water and emergent vegetation (Knight et al. 2003), can increase the abundance of undesirable biting insects, which can also act disease vectors. In certain parts of the world, living in a city near a pond is considered a health risk. For example, in some African regions, there is increased risk of infection with *Plasmodium falciparum* (Matthys et al. 2006). This is also the case in South America, with the presence of malaria in several urban areas (Brochero et al. 2005).

Toxin-producing algae (cyanobacteria) are another potential disservice at urban ponds. Hypertrophic ponds can support noxious cyanobacterial blooms which present a problem for human health, particularly where bathing is allowed (e.g., in cities in Belgium and the Netherlands; Peretyatko et al. 2012, Waajen et al. 2014). Cyanobacteria are also frequently present in stormwater ponds (Vincent and Kirkwood 2014). Fish populations (native or non-native species) are managed in some urban ponds to reduce the risk of cyanobacteria blooms. Biomanipulation involving fish removal or the introduction of piscivorous fishes can help reduce the risk of eutrophication from overstocking. This is due to cascading effects down the food chain: Lowering predation pressure on zooplankton increases herbivore pressure on phytoplankton, which in turn favors a clear-water state with macrophytes, free of cyanobacteria blooms (Peretyatko et al. 2012).

Invasive non-native species (particularly plants and fish) are very common in the urban environment, as highlighted previously (see *Non-native species*). The presence of these species is mostly linked to deliberate introduction by humans. The negative impact of non-native species relates essentially to those that are invasive and threaten native biodiversity. Abundant populations of non-native invasive species in the urban environment can potentially be a risk to biodiversity if these species have a reservoir of propagules that can disperse toward surrounding landscapes, as demonstrated for terrestrial plants (von der Lippe and Kowarik 2008). Numerous urban ponds are hydrologically isolated, but connections (even those that are transitory) to stream networks can markedly enhance the probability of dispersal. Indeed, for freshwater exotic species, the main pathway of dispersal from the original point of introduction to the wider environment is determined by hydrological connectivity (Lodge et al. 1998).

Ranaviruses, linked to mass amphibian die-offs in North America, Europe, and Asia, are associated with urbanization in Britain (North et al. 2015). This is because urban areas can be a reservoir from which propagules spread to the natural or rural environment. Some water birds can also cause a nuisance when their population density is too high. For example, dramatic population increases of the native white ibis (*Threskiornis molucca*) in urban areas in south-western Australia have resulted in their classification as a nuisance species (Martin et al. 2012). The croaking of amphibians (frogs) can be noisy and affect the well-being of citizens living near ponds. Although some municipalities receive complaints linked to these issues, we did not find any studies investigating this topic or providing detailed information. It is also important to remember that the pond itself can act as a disservice for biodiversity, when a reduced availability of high-quality habitats turn them it into an ecological trap (Hale et al. 2015).

*Implications for management.—*These disservices show that ponds in the urban environment are closely linked with humans, and so, social considerations are of prime importance for urban pond management. A pond needs the support of the local community to persist or be created in the urban landscape. Management therefore needs to integrate this constraint, and aim to reduce disservices that cause major problems. Clearly, good communication with the local community and relevant authorities is also of prime importance.

A Management Framework for Optimizing the Biodiversity of Urban Ponds

A particularity of urban ponds, linked to their close relationship with human activities, is that most are actively managed. This management generally aims to secure and optimize one (or several) ecosystem services. These services include aesthetic value, water purification, flood control, the production of fish or leisure activities (e.g., bathing, boating, fishing). Management less frequently targets the provision of habitat for biodiversity. The most popular urban pond management practices include hydroperiod modification by managing water levels or drying out the entire pond, dredging, mowing marginal aquatic vegetation, removing submerged or floating-leaved vegetation, feeding of aquatic birds or fish, the introduction of non-native species (e.g., plants, fishes, turtles), and the use of chemical products (e.g., for algal control). These management measures are generally conducted without consideration for biodiversity. Management practices could influence most of the environmental factors governing pond biodiversity (Fig. 6). The previous sections reviewed in some detail the impact of these different environmental factors on biodiversity, with the implications for pond management to promote biodiversity, while some of the publications reviewed proposed management guidelines to enhance biodiversity (Appendix S2). Here, we identify some global trends and propose a framework for the management of urban ponds to optimize their biodiversity. The framework includes several distinct modules (each relating to an environmental factor), with general guidelines proposed for each (Table 3). These are compatible with the provision of ecosystem services by urban ponds, and specific guidelines can be chosen to support the provision of a specific service. The general objective of this framework is to add the biodiversity habitat service to the other services targeted by management.

The global framework can be summarized by these key points.

1. The pondscape should include a broad diversity of pond types, with varied environmental characteristics (e.g., pond age, surface area, depth, primary productivity, shade) and managed by a range of practices (e.g., hydrology, vegetation removal, fish introduction).
2. Pond quality should be high at the local scale in order to, firstly, provide a range of habitats for biodiversity and, secondly, to avoid the creation of ecological traps. A high-quality pond is characterized by: good water quality (no excessive nutrient inputs, low concentrations of pollutants), the presence of stands of aquatic vegetation (submerged, floating-leaved, and emergent plants), banks with gentle slopes (and no vertical walls), and the absence of invasive non-native species.
3. The quality of the habitat in the area surrounding the pond should also be considered. Within several hundred meters of the pond, the habitat should include green spaces (without intensively managed lawns) and a low cover of impervious surfaces. Wooded areas should also be present, where possible.
4. The density of high-quality ponds should be high in the urban matrix, and species should be able to disperse easily between them.

Conclusions and Future Directions

Conclusions

The urban pond is a particular type of pond, different from a natural or rural pond (Fig. 5). It supports diverse biodiversity although often lower native species richness than ponds from more rural landscapes. The community composition also differs from that of natural ponds and may include non-native species. However, the urban pond community can support species of conservation concern or flagship species. Species adapted to the particularities of the urban environment are also often present. For all these reasons, urban pond communities need to be promoted and protected, and included in biodiversity-conservation strategies. This is also true for social reasons (e.g., maintaining the link between humans and nature), although this topic was not specifically reviewed here.

The environmental factors characterizing the local and landscape scales and driving the biodiversity of urban ponds (Fig. 6) are partly the same as those important for pond biodiversity in other types of landscapes. However, some factors are specific to the urban environment or are exacerbated in this type of environment (e.g., hydraulic functioning, pollutants, pond isolation, non-native species). These factors are intimately linked to human activities and, as a result, urban ponds tend to be more actively managed than other pond types. Furthermore, the relative importance of these factors and the way that they are expressed is clearly different in the urban environment than in other types of landscapes. The interaction between factors is also unique, and in this respect, there are still many gaps in our knowledge. Another particularity of urban ponds, specifically linked to their proximity to human activities, is that they are often managed to optimize a particular ecosystem service, regardless of the impact on the provision of habitat for biodiversity.

The specificity of urban ponds and of the urban pondscape requires a management strategy that is urban-specific. We propose here a management framework based on the review of scientific studies. The guidelines presented here should assist managers in promoting an additional service: the provision of habitat for biodiversity. The framework stresses the need to

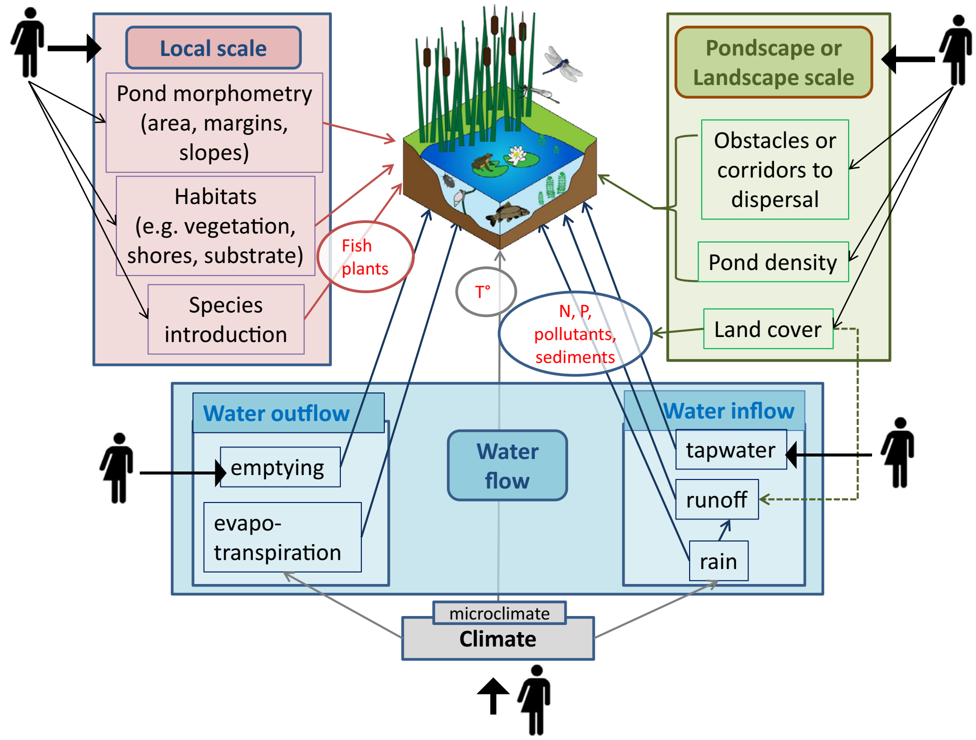

Fig. 6. Schematic overview of the main drivers of pond biodiversity in the urban landscape, underlining their strong relationship with management. Management can be focused on various services that ponds can provide (including habitat for biodiversity).

adopt a pondscape approach, and rather than focusing on a single, optimal pond type, managers should aim to achieve a wide diversity of pond types at the landscape scale. This is particularly important as different species can respond differently to the same driving factor. Only by managing the pond resource collectively can species richness be increased in a pondscape.

Future directions

Biodiversity studies often focus on one taxonomic group. This was the case for more than 90% of the publications selected for this review. There was also a bias toward some groups, particularly amphibians (Fig. 4). Evidence shows that, for pond biodiversity, no single taxonomic group is a good surrogate for other groups (Ilg and Oertli 2017), and therefore, we should aim to take a multi-taxon approach (Bolpagni et al. 2019). Geographically, the majority of the research originates from North America, Europe, and Australia (Fig. 2). Research should now be expanded to continents which are under particularly intense urbanization pressures (Africa, South America, Asia).

In the context of global climate change, some factors may be of crucial importance in future and, therefore, need additional research attention. Relatively little is known about the relationship between hydroperiod and biodiversity in ponds in urban areas, for example. This lack of knowledge could hinder effective management of urban freshwater biodiversity because hydroperiod is one of the environmental factors most influenced by human activities. In many cities, the characteristic of hydrological cycles is expected to change in future, while the heat island effect is likely to be exacerbated by climate change. Urban ponds already tend to have warmer water and support species adapted to these conditions. Urban pond communities could therefore act as source populations for colonizing ponds outside urban areas and potentially contribute to climate change adaptation. Changing

Table 3. A management framework for optimizing the biodiversity of urban ponds, comprised of several distinct modules (one for each environmental factor), with general guidelines proposed for the pond and the pondscape scales.

Environmental factor	Pond scale		Pondscape scale (city scale)	
	Guideline	Rationale	Guideline	Rationale
Surface area	Increase pond surface area.	Larger areas support more species for several taxonomic groups (but not all) Large ponds can be rare in some cities	All classes of pond surface area need to be represented in the pondscape, including larger ponds. Missing (or underrepresented) types of ponds should be created	Each pond size can support some unique species and assemblages. A wider range of pond size at the landscape scale can support greater regional species richness. However, several small ponds support more species than a single large pond of the same surface area
Bank slopes	Create a gently sloping bank. Avoid concrete walls	Gently sloping banks are favorable to aquatic plants and promote a diverse range of habitats for fauna	The pondscape should be dominated by ponds with a gently sloping bank, but other pond types can also add some value in the pondscape. Missing (or underrepresented) types of ponds should be created	Each pond type can support some unique species
Nutrients (N and P)	No action (but see the pondscape scale)	Nutrients are often not a central problem for urban biodiversity	Ponds with varying nutrient status should all be represented in the pondscape. Missing (or underrepresented) types of ponds should be created. This most often concerns oligotrophic or mesotrophic ponds	Each pond trophic status may support some unique species. A wider range of pond types with different trophic statuses support greater regional species richness
Other urban pollutants	Their concentration in the water should be assessed (and monitored). Management measures will depend on the ecosystem services targeted. If necessary, the usual management to reduce pollution can be undertaken (e.g., management of the water source and of the catchment area)	Negative impacts on biodiversity are often reported for urban pollutants present in ponds	Reduce the risk of ponds becoming ecological traps in the network. If ponds become ecological traps, then promote their isolation from other ponds in the landscape and act to discourage wildlife from using them as habitat	Some type of polluted ponds (as those intended to filter pollutants from stormwater) can have their place in a network comprising also good-quality ponds. Their number should not dominate the network and they should not constitute ecological traps
Hydroperiod	Promote natural hydroperiods required by biodiversity. Avoid artificial hydroperiods (in terms of frequency and duration) that are harmful to biodiversity	Artificial drying, at the wrong time of the year or lasting too long, can be harmful to a significant component of pond biodiversity. Long periods of drying of the shore are also detrimental	The network should include both ephemeral and permanent ponds. Missing (or underrepresented) types of ponds should be created. This will often concern temporary ponds	Urbanization tends to favor permanent ponds that can support higher overall diversity, but exclude ephemeral-pond specialists. Conserving the full assemblage of urban pond species will often require protecting and creating temporary ponds

(Table 3. Continued.)

Environmental factor	Pond scale		Pondscape scale (city scale)	
	Guideline	Rationale	Guideline	Rationale
Aquatic vegetation	Management practices should aim to preserve and favor the presence of large beds of macrophytes (submerged, floating-leaved, emergent). Adjustment of the pond design (e.g., including gentle bank angles) can promote this type of vegetation	The presence of structured vegetation in ponds, including large beds of submerged, floating-leaved, and emergent macrophytes, offers multiple habitats for fauna	Most ponds in a network should be vegetated. However, some ponds without vegetation have their place in (and contribute to) a pondscape	Conserving the full assemblage of urban pond species requires the presence of all pond types in the pondscape
Non-native invasive species Fishes	Monitor biodiversity for the early detection of non-native invasive species Assess the risk linked to non-native invasive species Eradicate non-native invasive species presenting a risk at the local or regional scale	Some non-native invasive species present a risk for biodiversity, at the pond scale or at the pondscape scale	Same measures as for the pond scale	Some non-native invasive species present a risk for biodiversity at the pondscape scale (and also for the surrounding landscapes)
Pond buffer area	In the pond buffer area (up to a 2000-m radius, but especially near the pond), good-quality habitats (green spaces, forests, other wetlands) should be encouraged Enhance the opportunities for dispersal (reduce barriers to movement, promote corridors and stepping stones)	Many amphibious species use terrestrial habitats to complete their life cycle To maintain viable metapopulations, many species need to disperse safely between ponds	At the city scale, increase the density of ponds in a network (through the creation of new, high-quality ponds) The movement of biodiversity between ponds should be enhanced (reduce barriers to species movement, promote corridors and stepping stones)	Many species need to disperse between ponds to maintain viable metapopulations A dense network of high-quality ponds is required for this process to be successful
Disservices provided by the biodiversity of urban ponds	Identify disservices and their impact on citizens Manage to reduce problematic disservices Social engagement and communication are of prime importance	To be accepted by the community in the urban environment, a pond should provide few disservices	The disservices that can potentially impact the pondscape need to be identified	One disservice can potentially spread from a single pond to the entire network, and potentially also to the surrounding landscape

Note: A summary of the rationale is also included (for more details, see the sections corresponding to each environmental factor).

climate can also be related indirectly to changes in water chemistry (e.g., increased salt content), and therefore, its impact on biodiversity needs to be investigated.

The presence of non-native species, including invasive species, is a characteristic of urban ponds. The occurrence of these species is still increasing (Hussner et al. 2014) and therefore represents a growing concern. The impacts of non-native species on urban pond biodiversity need to be better understood at the pond scale, but also specifically in terms of the ecosystem services that motivate pond creation. At the landscape scale, species dispersal across urban landscapes requires further investigation, including their potential interaction with surrounding rural landscapes.

The pondscape scale, integrating both pond network and landscape, is currently recognized as the most effective approach for conserving and promoting pond biodiversity (Hill 2018). However, its importance for freshwater biodiversity conservation needs to be better documented, for all types of landscape. This is particularly relevant for the urban landscape, where many elements of the (mostly human-made) landscape matrix are unique, as is their interrelation with biodiversity. Key questions include the following: What is the optimal pond density for sustaining a rich biotic community at the pond and pondscape scales? With this objective in mind, complementary pond types need to be investigated in different urban pondscapes, including the identification of high-quality ponds and potential ecological traps. Potential links between the urban and the surrounding rural pondscape also need to be assessed. Is the urban pondscape independent of the rural pondscape, or does it need continuous inflow of organisms and propagules from rural areas? Conversely, is there an outflow of organisms and propagules from urban pondscapes toward rural pondscapes? In particular, studies of barriers to dispersal, corridors, and stepping stones are required, and genetic tools will be useful for this purpose. As noted above, most metapopulation studies in urban pondscapes have been conducted on amphibians and now need to be extended to other taxonomic groups. Finally, many disparate measures have been used to describe the urban environment around a pond; we recommend that researchers develop a more standardized way to characterize urban ponds to allow for ease of comparison and meta-analyses of multiple datasets.

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Supporting Information

Additional Supporting Information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/ecs2.2810/full