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Effects of management intensity, function and vegetation on the biodiversity in urban ponds

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ABSTRACT

Ponds are important elements of green areas in cities that help counteract the negative consequences of urbanization, by providing important habitats for biodiversity in cities and being essential nodes in the overall landscape-scale habitat network. However, there is relatively little knowledge about the impacts of pond management intensity, function and environmental variables on urban pond biodiversity. In this study we addressed this gap by investigating which factors were correlated with the level of biodiversity in urban ponds, indicated by species richness of aquatic insects, in Stockholm, Sweden. Our study did not confirm any direct link between the perceived intensity of management or function of ponds and overall biodiversity. However, it seems that management can influence particular groups of species indirectly, since we found that *Trichoptera* richness (Caddisflies) was highest at intermediate management intensity. We suggest that this is caused by management of vegetation, as the amount of floating and emergent vegetation was significantly correlated with both the overall species richness and the richness of *Trichoptera* (Caddisflies). This relationship was non-linear, since ponds with an intermediate coverage of vegetation had the highest richness. Interestingly, the amount of vegetation in the pond was significantly affected by pond function and pond management. The overall species richness and richness of *Trichoptera* were also positively correlated with pond size. Since we found that the pattern of relations between species richness and environmental variables differed between the insect groups we suggest that it will be difficult to provide overall design and management recommendations for ponds in urban green areas. Therefore, it is recommended that to provide high aquatic diversity of species in urban areas one should aim at promoting high diversity of different types of ponds with differing management and environmental factors that shape them.

1. Introduction

The on-going global trend of urbanisation has important consequences for biodiversity, leading to the increasing fragmentation of natural environments and habitat loss (McDonald et al., 2008; Miller and Hobbs 2002). Even though some plant and animal species are able to inhabit urban areas, most are sensitive to the effects of urbanisation (Ghert and Chelsvig, 2003; Riley et al., 2005). However, detrimental influences of urban environments can be alleviated by the presence of green areas that may offer important habitats for biodiversity in cities and provide essential nodes in the overall landscape-scale habitat network (Angold et al., 2006; McKinney 2006). In some cases, these areas provide habitats for species that are decreasing elsewhere (e.g. Carrier and Beebee, 2003).

Ponds in cities are often classified as "green-space areas", because they are usually located within parks or other urban green zones and constitute important components of these areas (Harrison et al., 1995). Growing recognition of the importance of ponds and other small water bodies for maintaining biodiversity in cities (Colding et al., 2009; Fuyuki et al., 2014; Hassall and Anderson 2015) extends to conservation programmes for endangered species (Vermonden et al., 2009). Hassall and Anderson (2015) revealed that urban storm water management ponds can provide a similar level of biodiversity as urban wetlands in Ottawa, Canada.

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Similarly, Le Viol et al. (2009) investigated the diversity of macroinvertebrates in highway storm water ponds and concluded that these ponds supported communities at least as rich and diverse as pond communities in the surrounding landscape. Goertzen and Suhling (2013) showed that urban ponds may have great value for biodiversity but are threatened by various urban disturbances and lack of suitable design.

During the past decade, there has been an increase in the number of studies examining biodiversity of urban ponds (e.g. Gledhill et al., 2005; Hamer et al., 2012; Noble and Hassal, 2014). The majority of studies have focused on how ecological and land-cover variables or water chemistry affects the biodiversity of ponds (e.g. Biggs et al., 2005; Lamy, 2013; Leibold, 1999; Oertli et al., 2002). There remains, however, a lack of knowledge on how the intended primary function and management of ponds affects biodiversity in cities (Biggs et al., 2005; Hassall et al., 2011). Ponds in cities can have different primary functions, e.g., their purpose can be treatment of urban run-off, maintenance of natural value for biodiversity, or delivery of aesthetic experiences to citizens (Hassal, 2014). Each function may involve specific kinds of management activities that influence pond vegetation. In one of the few studies on pond management and biodiversity, Noble and Hassal (2014) identified a possible conflict between human and wildlife interest in pond management in Bradford District, UK, because many ponds were managed primarily for their aesthetic function such that removal of vegetation to keep the pond “neat” for people potentially decreased their ecological value. However, there is little research on how environmental factors and function and management of ponds are correlated and how this influences biodiversity of urban ponds.

In many countries the potential of urban ponds to deliver both biodiversity and ecosystem services such as water purification, flood control, or aesthetic experiences has been recognized. Many ponds and small water bodies have been filled during the past few decades, but there is now an increasing trend to restore and even to create new water bodies in many European cities (e.g., Åstebøl et al., 2004; Segaran et al., 2014; Starkl et al., 2013; Gledhill and James, 2012). In Sweden, ponds have been increasingly included in urban landscape planning since 1990, both for the purpose of water runoff management and to increase aesthetic appeal of nearby housing areas (Personal communication with representatives of municipalities in the city of Stockholm, 2014). Recently, the focus in Sweden has been to increase the value of these ponds for biodiversity, and some projects to restore ponds with the intention of improving their ecological value have been initiated (e.g. Ohlin, 2013). However, to our knowledge there are no studies that have focused on the relationships between the intended primary function of a pond, how intensively a pond is managed, and pond biodiversity. We use the example of a large Swedish city, Stockholm, to investigate this relationship. Such knowledge is important because it can help with pond design and management to enhance their ecological value and thus provide biodiversity rich habitats in the urban matrix.

The purpose of our study was to investigate which factors are responsible for determining biodiversity in Stockholm urban ponds with a focus on management. We selected environmental variables that were shown to affect biodiversity in a pilot study as well as variables associated with pond function and management that were identified during interviews.

2. Methods

2.1. Study area and selection of ponds

Our study was conducted in the city of Stockholm, capital of Sweden. The city has ca. 900 000 residents (Stockholm Stad, 2013) and is characterised by many waterways and high coverage of green areas (Andersson et al., 2009). Stockholm is a city where environmental considerations are high on the political agenda. It was the first city to be awarded the European Green Capital by the EU Commission in 2010, because of its holistic vision to combine economic growth with sustainable development (European Commission, 2010).

In the present study, we considered 43 ponds in central Stockholm covering seven municipalities (Fig. 1). We define ponds as natural or man-made water bodies having an area between 1 m^2 and 2 ha and holding water for at least 4 months of the year (Pond Conservation, 2002; Biggs et al., 2005). Ponds were selected from maps and by using information from municipal officials. We focused on densely populated areas in the city. We divided Stockholm into 1×1 km squares and only considered squares where >75% of the area was occupied by build-up land uses as defined in Terrain map (Terrängkartan^{TM}) of the Swedish mapping, cadastral and land registration authority (Lantmäteriet). Therefore, we excluded ponds located in golf courses, most often situated outside the populated areas, even if they have shown a great potential for fostering biodiversity in urban areas (Colding et al., 2009).

2.2. Selection of variables and data collection

We defined biodiversity as species number, and we measured it as richness of aquatic insects, in the taxa dragonflies (*Odonata*), aquatic beetles (*Coleoptera*), aquatic true bugs (*Hemiptera*) and caddisflies (*Trichoptera*). Although former studies have shown that also other taxonomic groups significantly contribute to the biodiversity of urban ponds (Hassall and Anderson, 2015; Hill et al., 2015), the taxa selected for the purpose of this study provide an accurate proxy of general biodiversity, because these invertebrates represent different functional groups and their biodiversity is correlated with biodiversity of plants, vertebrates and other invertebrate groups (Hassall et al., 2011; Oertli et al., 2010). As species richness was significantly correlated with Shannon index for species ($p < 0.000$, $r = 0.82$) and species abundance ($p = 0.005$, $r = 0.42$) we did not include these variables in the analysis, because adding variables decreases statistical power of the models.

We collected information on environmental variables that were previously found to be correlated with aquatic insect diversity. In a pilot study conducted using 26 ponds in north and central parts of Stockholm (Andersson, 2014) that investigated the effects of environmental variables on the aquatic insect diversity, it was found that species richness is significantly correlated with the amount of aquatic vegetation (both floating and emergent) in ponds and their distance to the nearest building. We therefore included these variables. Similar effects of vegetation on species richness in ponds have been found in other studies (Biggs et al., 2005; Goertzen and Suhling, 2013; Hassall et al., 2011). However, we did not sample submerged vegetation. In addition, we added pond size as a potential variable that could affect biodiversity, because it has been shown to be important in studies of rural ponds (Biggs et al., 2005; Oertli et al., 2002), even if studies considering pond size have yielded conflicting results (Hassall et al., 2011).

Vegetation cover of the ponds (i.e. floating plants and plants growing in the ponds) was estimated by eye in August 2013 and 2014 in tenths and described in percentage ranging from a total cover of no vegetation (0%) at all to full cover (100%). Vegetation was recorded into two separate categories; floating leaf vegetation and emergent vegetation. Pond size and distance to nearest building was estimated with the software ArcGIS 9 and the Terrängkartan^{TM} map from Lantmäteriet.

Information on the ponds' main function, age and perceived intensity of management was collected through interviews. First, an open-ended interview (Kvale, 1996) lasting about 2.5 h was

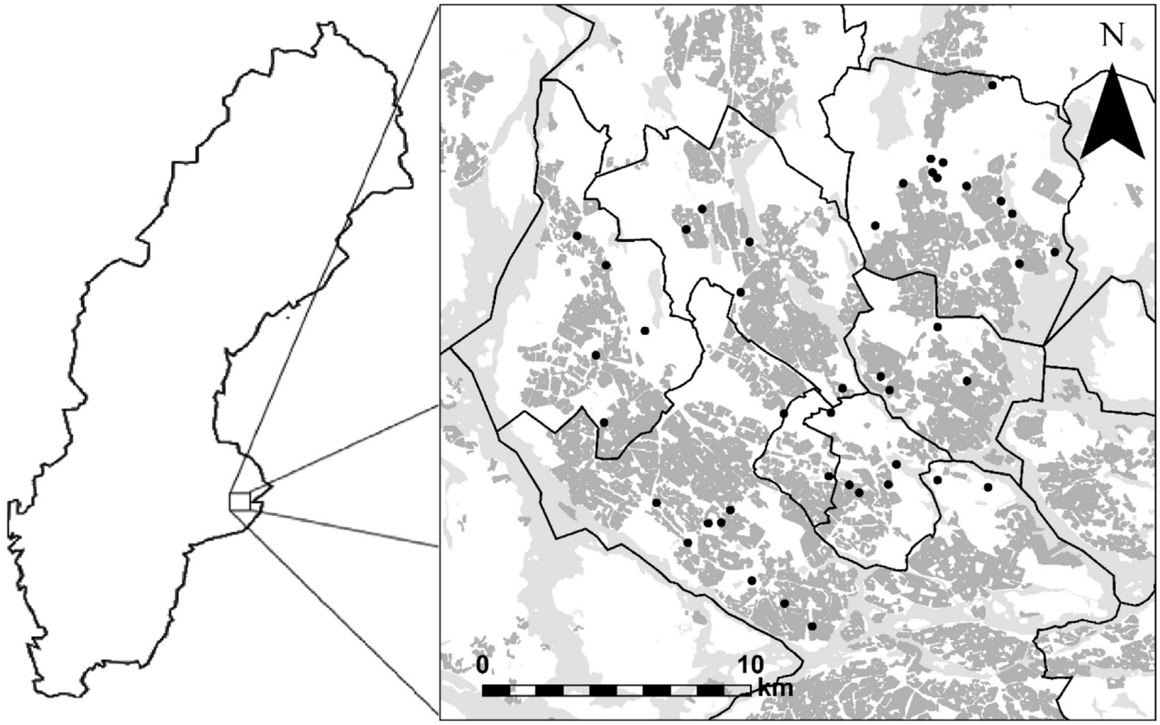

Fig. 1. Location of the 43 ponds in the centre of Stockholm metropolitan area. Dark grey shading indicates build-up areas, while light grey shading is water. The dark lines indicate municipal borders.

conducted with a representative of the centrally located municipality in Stockholm (Stockholm City) to get initial insight into pond management in Swedish cities in general and to get contact information for managers of particular ponds. Then, semi-structured interviews (Kvale, 1996) were conducted with representatives of seven municipalities in Stockholm: Danderyd, Järfalla, Sollentuna, Solna, Stockholm City, Sundbyberg and Täby. Four ponds did not belong to any municipality, but to another type of unit (the Swedish Royal Court, Bergius Botanic Garden, the Stockholm Water Company and a private horse riding stable) and thus representatives of these units were interviewed. Altogether, 18 people were interviewed, either face-to-face or via telephone. Additional information was obtained in individual cases via e-mail. Each interview lasted from 15 min to 1.5 h, depending on how many ponds the particular person could provide information about. During the interviews, information was gathered on ponds' main function (run-off, amenity and nature pond), age class (young, middle-aged, old) and perceived management intensity (none, moderate, strong). A description of categories for each of these factors is given in Table 1.

To estimate richness of aquatic insects we sampled them in spring and early summer (May–June) with a bottom scoop net with a diameter of 20 cm and a mesh size of 1.5 mm. Twenty one ponds were sampled in 2013, while the remaining 22 in 2014. Six samples were taken in each pond at a depth of 20–30 cm. The net was swept along the bottom in opposite directions (left to right) eight times on a 1 m stretch, which constituted one sample. By using six samples we covered all types of representative microhabitats along the shoreline, e.g. soft bottom, hard bottom with and without vegetation. The sampling strategy was derived from the guidelines by the SEPA (2006).

We sampled the aquatic life stages i.e. larvae in *Odonata* and *Trichoptera* and larvae and adults in *Coleoptera* and *Hemiptera*. All insects were determined to order at the pond site and were preserved in 70% ethanol, stored in labelled plastic tubes and brought back to the laboratory for species determination. All other species were rereleased back to their respective ponds. Specimens that could not be determined to species level were still included in the final analysis and set to family or genus-level and hence regarded as separate taxa. In most cases these specimens were early instar larvae. Larvae of *Coenagrion puella* and *C. pulchellum* are not

Table 1
Definition of the categorical data classes collected in the study.

Variable			
Main function	*Urban run-off pond* Main aim is to slow down the run-off (may also mean protection from flooding) and/or clean the water	*Nature pond* Main aim is to enhance biodiversity	*Amenity pond* Main aim is to increase attractiveness of the area for people (including value for recreation)
Age class	*Young* Up to 7 years old	*Middle-aged* From 8–25 years old	*Old* Over 25 years old
Perceived management intensity	*None* No management is conducted	*Moderate* Relatively low intensity management, e.g. removal of some portion of vegetation from the pond each few years or regular management around the pond such as mowing or grazing, but not in the pond	*Strong* Frequent and intensive management, e.g. removal of large part of vegetation each year or every two years, or small removal of vegetation each year and larger interference each 4–5 years.

Table 2
Summary statistics for environmental variables (min, max, mean and median values).

Variable [Name]	Min	Max	Mean	Median
Pond size (m^2) [PondSize]	13	17,219	1905.2	753.0
Distance to nearest building (m) [Distance Build]	12	747	147.8	86.7
Proportion covered by floating vegetation (%) [FloatingVeg]	0	90	19.8	10
Proportion covered by emergent vegetation (%) [EmergentVeg]	0	90	36.3	30

Table 3
Summary statistics for function, age and perceived management data (number and% of ponds in each category).

Variable				Sum
Main function [Function]	Run-off2558%	Nature 9 21%	Amenity921%	43100%
Pond age [Age]	Young 16 37%	Middle-aged1944%	Old 8 19%	43100%
Perceived management intensity [Management]	None 16 37%	Moderate 16 37%	Strong 11 26%	43100%

distinguishable and were therefore regarded as the same species in the analysis. The same applies to three cases among the *Trichoptera* where larvae could not be distinguished between two species. These were i) *Limnephilus affinis and* L. *incisus,* ii) *Limnephilus luridus and* L. *ignavus and* iii) *Oligotricha stricta and O. lapponica.*

2.3. Data analysis

We used multivariate General Linear Models (GLM) to explain the relationships between explanatory variables and overall species richness (all invertebrate groups together) and richness of each of the separate groups. In the initial analyses we included all explanatory variables and then we used a backwards model selection for selecting the models that best explained the relationships between the response variable (overall species richness or richness of particular species groups) and the explanatory variables. Categorical explanatory variables (Function, Age and Management) were used as fixed factors in the models, while the continuous explanatory variables (PondSize, DistanceBuild, EmergentVeg and FloatingVeg) were covariates. We did not account for the interactions between the explanatory variables due to small size of some categories of the categorical variables. For variables EmergentVeg and FloatingVeg we also included a quadratic term, in addition to the linear function, as the relationship between species richness and vegetation cover seemed to be non-linear (see Results section). We set a threshold value of $p = 0.15$ for removing the variables from the model in finding the best model (see Bendel and Afifi, 1977 for explanation of the choice of this threshold). In short, using a p-value of 0.15 reduces the probability of performing a type I statistical error. Visual inspection of our response variable showed that it did not deviate from normality.

When the multivariate GLM models produced significant effects we explored the relationships between individual variables further using either linear or quadratic regression models.

As vegetation had the strongest effect on species richness we also investigated if there was an effect of management and pond function on the amount of vegetation. This was done using GLM models that had pond management or pond function as categorical variables and vegetation as response variables.

3. Results

3.1. Overall species richness

Overall species richness in the ponds varied from 1 to 22, and the mean number of species per pond was 9.74. The most frequent species were *Coleoptera* and *Odonata* species (Online resource 1). The most species rich were ponds designed to enhance biodiversity, followed by run-off and amenity ponds (Online resource 2). Size of ponds ranged from 13 to slightly over 17 000 m^2 and they were located within the distance of 12–747 m from the nearest building, while vegetation cover ranged from 0 to 90% (Table 2). Most of the ponds were either middle-aged or young with none or moderate management and as much as 58% of them were run-off ponds (Table 3).

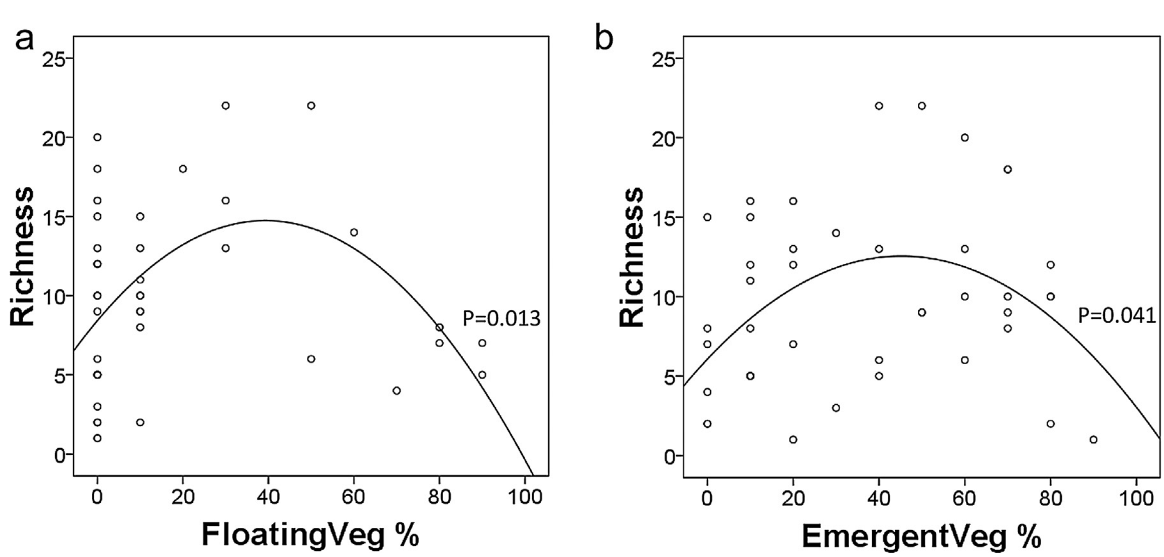

Fig. 2. Relationship between species richness and amount of floating (a) and emerging (b) vegetation. The curves shows a quadratic regression fit to the data points.

Table 4
Result of GLM model for the overall species richness.

Variables in the selected model	F	P	Models'R^2
Function	2.411	0.104	0.270
FloatingVeg	5.213	0.029	
$FloatingVeg^2$	4.237	0.047	
EmergentVeg	3.223	0.081	
$EmergentVeg^2$	2.338	0.135	
PondSize	5.198	0.029	

During the backward selection the variables age and perceived management intensity were excluded from the models as they did not show any significant effect on the species richness. In the selected model that best explained overall pond richness, floating vegetation and pond size had most significant effect on overall richness (Table 4). Visual inspection on the influence of floating vegetation on species richness suggested a humped shaped relationship with richness, with the highest richness at intermediate vegetation cover (Fig. 2a). A regression model using quadratic floating vegetation cover as a quadratic response variable showed a significant effect of floating vegetation cover ($P = 0.013$; $R^2 = 0.195$; Fig. 2a). There was a trend for a significant relationship between emergent vegetation and species richness (Table 4), and regression model using quadratic emergent vegetation cover as a response variable showed a significant effect of floating vegetation cover on species richness ($P = 0.041$; $R^2 = 0.148$; Fig. 2b). In contrast when inspecting the relationship between richness and pond size using a linear regression, the linear effect of size on richness disappeared ($P = 0.137$; $R^2 = 0.053$; Fig. 3). Pond function had no significant effect on overall species richness (Table 4).

Table 5
Result of GLM model for the *Trichoptera* richness.

Variables in the selected model	F	P	Models'R^2
Management	4.007	0.028	0.547
Age	2.627	0.087	
FloatingVeg	10.489	0.003	
$FloatingVeg^2$	7.406	0.010	
EmergentVeg	9.299	0.004	
$EmergentVeg^2$	7.054	0.012	
PondSize	4.709	0.037	

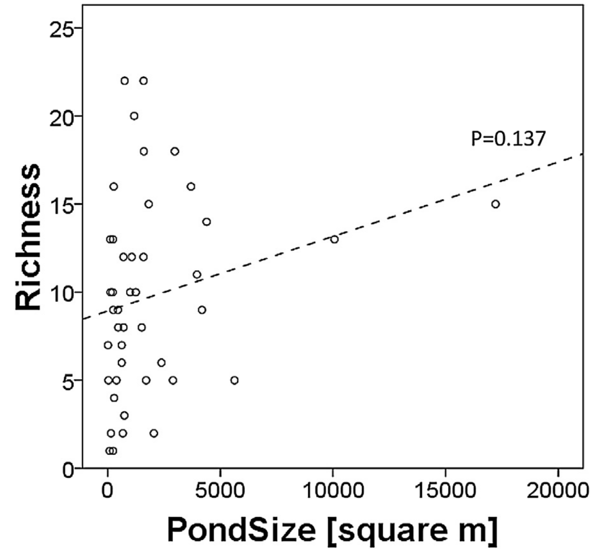

Fig. 3. The relationship between species richness and pond size. The line shows the best fit from a linear regression model, but it should be noted that it was non-significant. The line was added to facilitate visual interpretation of the full model, which showed a significant relationship.

3.2. *Trichoptera* richness

Trichoptera richness was affected significantly by floating and emergent vegetation (Table 5). Visual inspection of the individual influence of these variables on richness suggests a humped shaped relationship between richness and vegetation (Fig. 4). We therefore ran two quadratic regressions using quadratic floating vegetation cover and quadratic emergent vegetation cover as response variables respectively. These two models showed a significant relationship between vegetation cover and richness ($P < 0.001$; $R^2 = 0.374$ for floating and $P = 0.008$; $R^2 = 0.216$ for emergent vegetation; Fig. 4a and b). The richness of *Trichoptera* was also affected significantly by management and pond size (Table 5). Ponds with a moderate management intensity showed the highest richness (Fig. 5). When inspecting the relationship between richness and pond size using a linear regression, the linear effect of size on richness disappeared ($P = 0.148$; $R^2 = 0.050$; Fig. 6).

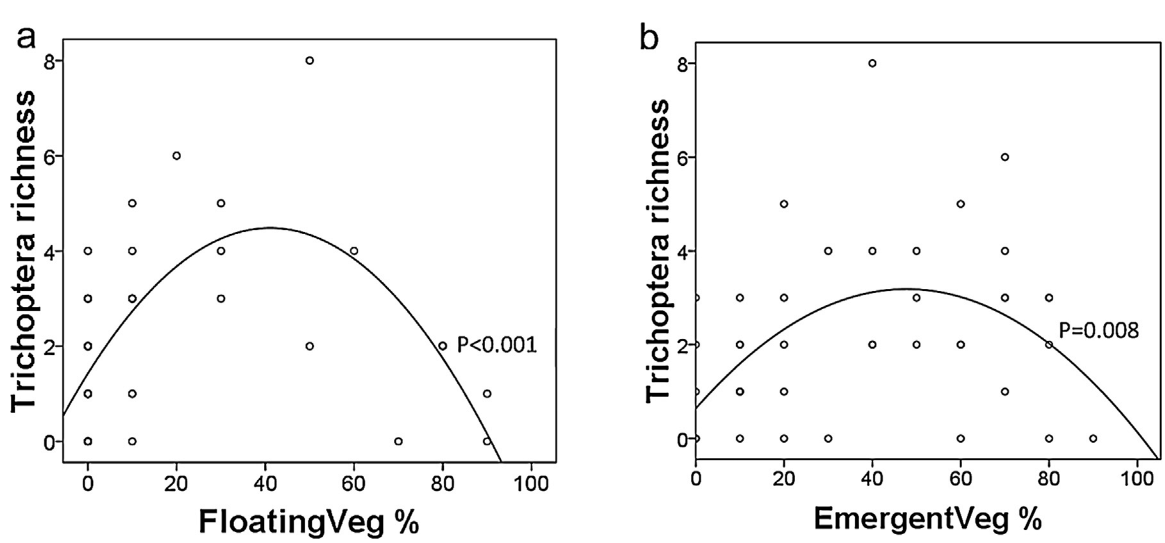

Fig. 4. Relationship between *Trichoptera* richness and amount of floating (a) and emergent (b) vegetation. The curves shows a quadratic regression fit to the data points.

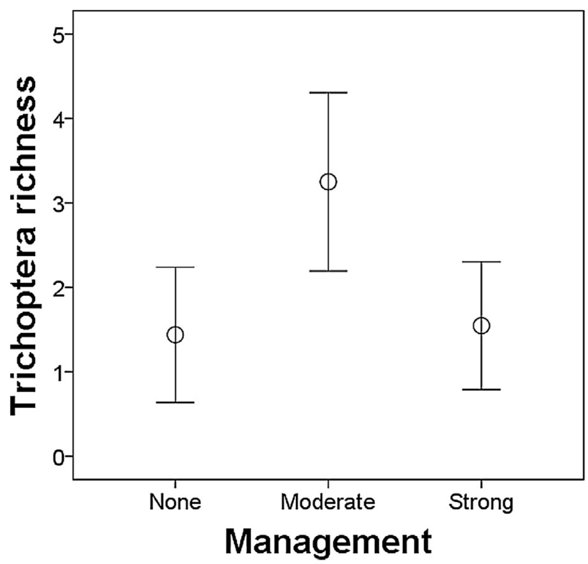

Fig. 5. *Trichoptera* richness in ponds with different management intensity. The bars represent the 95% confidence interval.

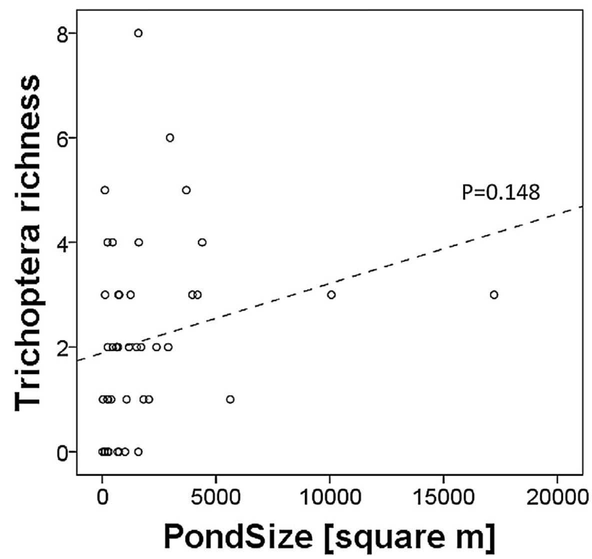

Fig. 6. The relationship between *Trichoptera* richness and pond size. The line shows the best fit from a linear regression model, but it should be noted that it was non-significant. The line was added to facilitate visual interpretation of the full model, which showed a significant relationship.

Table 6
Result of GLM model for the *Odonata* richness.

Variables in the selected model	F	P	Models'R^2
FloatingVeg	3.129	0.085	0.208
$FloatingVeg^2$	2.785	0.104	
EmergentVeg	3.193	0.082	
$EmergentVeg^2$	2.539	0.120	
DistanceBuild	2.916	0.096	
PondSize	2.380	0.132	

3.3. Odonata *richness*

There were no strong effects of the variables on *Odonata* richness (Table 6). However there was a trend for a significant effect of vegetation and distance to buildings. Visual inspection of the effect of these variables taken individually, suggests that both floating and emergent vegetation showed a humped shaped relationship with richness showing the highest richness at intermediate vegetation cover ($P = 0.019$; $R^2 = 0.179$ and $P = 0.064$; $R^2 = 0.128$, respectively; quadratic regression models; Fig. 7a and b). Ponds with a greater distance to buildings tended to have higher richness, however this relationship was not significant ($P = 0.149$; $R^2 = 0.050$ linear regression; Fig. 8).

3.4. Coleoptera *and* Hemiptera *richness*

Neither *Coleoptera* nor *Hemiptera* richness was affected significantly by any of the explanatory variables we included, and the full models had a very low R^2 values (Tables 7 and 8).

3.5. *Vegetation in the ponds*

The amount of emergent, but not floating vegetation was significantly affected by management intensity ($P = 0.005$ and $P = 0.254$, Fig. 9a and b), with the lowest amount of vegetation at the strongest management. The amount of emergent vegetation was also significantly affected by the pond main function ($P = 0.005$), with run-off ponds having most vegetation, followed by nature and amenity ponds (Fig. 9c). The relationship between pond function and the

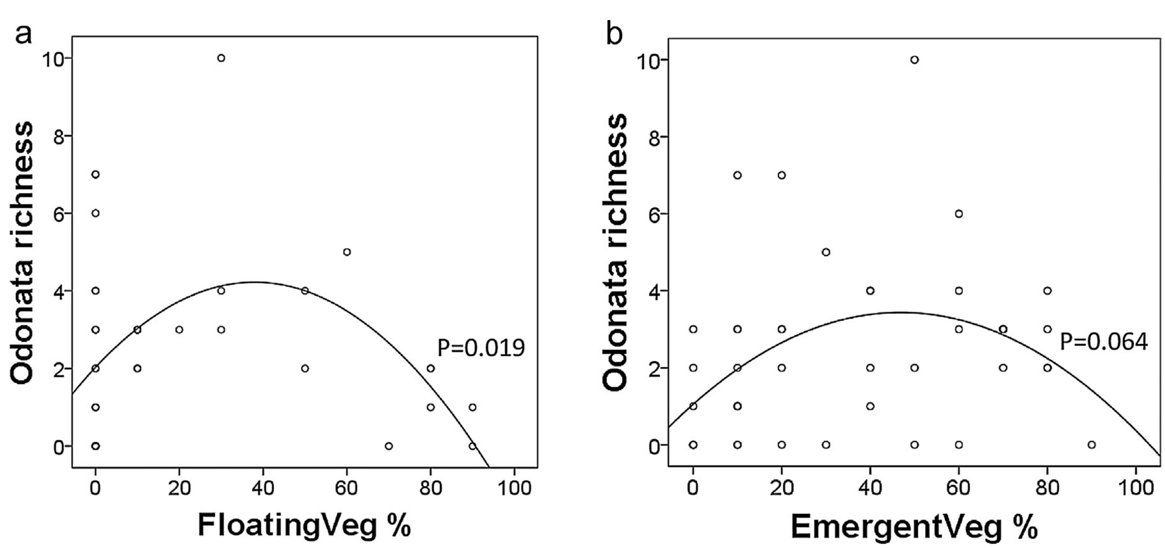

Fig. 7. Relationship between *Odonata* richness and amount of floating (a) and emerging (b) vegetation. The curves shows a quadratic regression fit to the data points.

Table 7
Result of GLM model for the *Coleoptera* richness.

Variables in the selected model	F	P	Models'R^2
Management	0.056	0.946	0.161
Function	1.943	0.161	
Age	0.161	0.852	
FloatingVeg	0.025	0.877	
FloatingVeg^2	0.007	0.932	
EmergentVeg	0.569	0.457	
EmergentVeg^2	0.595	0.446	
DistanceBuild	0.003	0.954	
PondSize	0.540	0.568	

Table 8
Result of GLM model for the *Hemiptera* richness.

Variables in the selected model	F	P	Models'R^2
Management	1.853	0.174	−0.051
Function	1.307	2.86	
Age	0.304	0.740	
FloatingVeg	0.835	0.368	
FloatingVeg^2	0.586	0.450	
EmergentVeg	0.172	0.681	
EmergentVeg^2	0.089	0.768	
DistanceBuild	0.022	0.884	
PondSize	0.667	0.421	

amount of floating vegetation was close to significant (P = 0.068), with amenity ponds having largest cover of floating vegetation (Fig. 9d).

4. Discussion

We did not find strong evidence of the importance of perceived management intensity for the biodiversity, but we found that one of our studied insect orders was affected. However, the interpretation of our "management intensity" variable, which was obtained through a generalization of information from interviews, needs to be interpreted with caution. The measure we used can be seen as a humble indication of management intensity, but does not account for concrete management measures and approaches. Therefore, the future research needs to address urban pond management in a more comprehensive and quantitative way, measuring the actual management, including the amount of vegetation removed, the amount of working hours spent, etc., over longer time periods, and comparing particular management strategies. Nevertheless, our results suggest that management can affect richness indirectly since management often includes the removal of vegetation. The GLM models in our study have revealed that overall richness and richness of *Trichoptera* was associated with the amount of vegetation (especially floating vegetation), and that ponds with an intermediate coverage of vegetation had the highest richness. This was not shown in the GLM for other taxonomic groups, however, for *Odonata*, the linear regression showed significant effect of the floating vegetation.

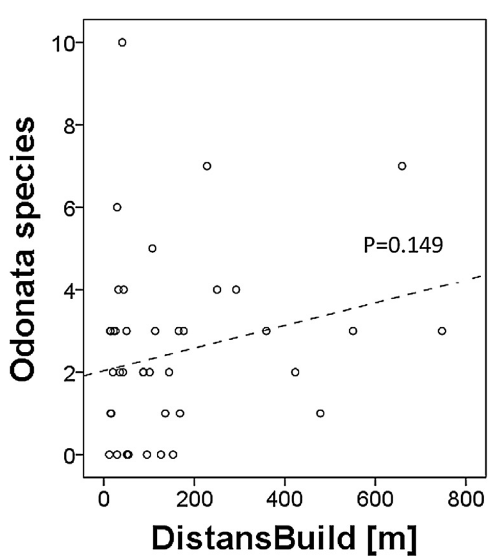

Fig. 8. The relationship between *Odonata* richness and distance to buildings. The line shows the best fit from a linear regression model, but it should be noted that it was non-significant. The line was added to facilitate visual interpretation of the full model, which showed a significant relationship.

In the case of general species richness, there was more pronounced effect of vegetation on richness when considering individual variables (vegetation cover) in linear model, compared to GLM. The differences between results of linear regression and GLM can be linked to the differences in complexity of these two types of models. In the GLM model multicollinearity is present whereas that is not the case in the simple regression model. Nevertheless, the relationship between richness of at least some of the investigated groups and vegetation is in line with findings of other authors. For example, in the study by Goertzen and Suhling (2013), both terrestrial and aquatic vegetation were the major determinants of diversity of dragonflies of urban ponds. The same pattern with respect to vegetation has also been found in rural waters with regard to dragonflies (e.g. Hinden et al., 2005; Remsburg and Turner, 2009) and for several different groups of macro-invertebrates (Hill et al., 2015). The greatest richness of macro-invertebrates in waters with intermediate level of vegetation was explained by previous studies investigating pond microhabitats available for different taxonomic groups. For example, Bazzanti et al. (2010) have shown that lower macro-invertebrate richness (i.e., reduced faunal diversification and abundance) in unvegetated zones was explained by the presence of fine sediments, higher levels of nutrients, and relatively low oxygen content. On the other hand, the authors emphasized the importance of vegetation in providing stability of sediments, better oxygenation and diversity of habitats and availability of food resources that increased species richness.

Considering both our findings and findings of the previous studies, we agree with Goertzen and Suhling (2013), who concluded that "the most prominent determinant of diversity at urban ponds is the same as in natural environments". Our pilot study (Andersson, 2014) showed a relationship between vegetation cover and species richness. However, our results should be considered with caution, because submerged vegetation was not sampled. Low to intermediate densities of submerged macrophytes probably increase species richness because more structure and thus habitat is available for aquatic invertebrates (St Pierre and Kovalenko, 2014). But high densities of submerged vegetation might result in negative effects such as an anaerobic conditions as a cause of the decomposition of this organic material.

We found that the relationship between richness and vegetation cover was non-linear and showed the highest richness at intermediate vegetation cover. We suggest that this humped shaped relationship may be caused by a combination of habitat heterogeneity and productivity (Huston, 2014; Rosenzweig, 1995). When productivity increases, more plant species can colonize, which causes an increase in spatial heterogeneity, availability of microhabitats, and resource availability for plant eating insects and, in turn, their insect predators (Bazzanti et al., 2010). However, with even higher productivity plant richness goes down (Huston, 2014; Rosenzweig, 1995), as the community becomes dominated by a few competitive plant species that provide less habitat heterogeneity and a less diverse food resource. This finding has potential

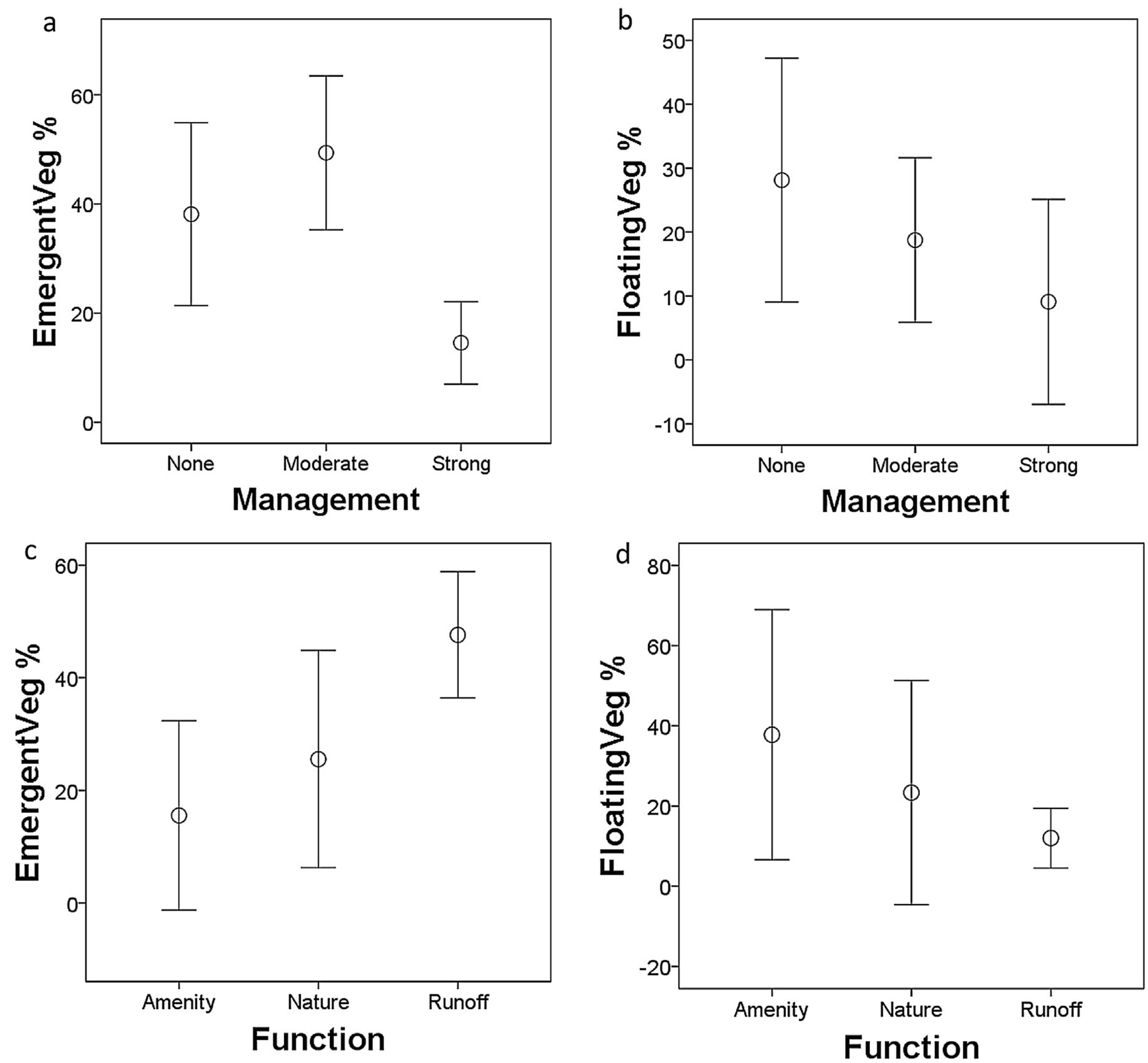

Fig. 9. Relation between management intensity and amount of emergent vegetation (a); management intensity and floating vegetation (b); pond function and emergent vegetation (c); pond function and floating vegetation (d). The bars represent the 95% confidence interval.

implication for pond design and management, as it suggests that maintenance of intermediate levels of vegetation has potential to improve overall species diversity. When designing urban ponds, however, one should consider the primary purpose of an individual pond (Hill et al., 2015). As different taxonomic groups have various requirements, "one fit for all" management strategy would benefit only some groups. Based on our results, keeping intermediate levels of vegetation would support *Trichoptera* and overall species richness, while *Odonata*, *Hemiptera* and *Coleoptera* may require other management approaches. More research is needed to investigate how particular management solutions influence particular species, especially when one considers the importance of microhabitats on biodiversity (Bazzanti et al., 2010). Gledhill et al. (2005) suggested that management of urban ponds can be ineffective in maintaining high biodiversity, particularly in case of aquatic plants requiring more complex management strategies. Support for this from our study is the link between species richness in urban ponds and their vegetation, which can indirectly be influenced by management intensity. Noble and Hassal (2014) claimed that proper management aiming at increasing aquatic plants increases the ecological quality of ponds. In contrast, our study shows that the vegetation-richness relationship is not necessarily linear, as richness increases up to a certain point of vegetation cover and then decreases. There is need for research that would investigate this relationship in more detail with regard to concrete management measures to explore its particular nuances that could be useful in planning the proper management.

Because the primary function of a pond affected the amount of vegetation, with run-off ponds characterized by the highest amount of emergent and lowest amount of floating vegetation, one could also expect a significant relationship between primary function and species richness (influenced by vegetation). Noble and Hassal (2014) suggested that ponds maintained for amenity value may be less species rich, due to decreased amount of vegetation, because the pond managers tend to keep the ponds "neat" for esthetic purposes. For example, Nassauer (2004) found that clean, mowed wetlands are perceived as more attractive by people. To mitigate or avoid potential conflict between human and wildlife interests, Noble and Hassal (2014) promoted informing the public about the importance of aquatic vegetation in ponds to facilitate changes in perceptions concerning pond attractiveness. They also suggested that managers working with ponds should be trained about the ecological requirements of different species (e.g. about the need for planting aquatic vegetation that has poor dispersal rate). However, in our study we found no evidence for a direct significant link between pond function and richness, only an indirect effect, whereas management affected vegetation and vegetation affected richness.

In our models pond size showed a positive relationship with overall species richness and richness of *Trichoptera*. We note, however, that this relationship disappeared when linear regression models were used to investigate relationship between richness and pond size. The main reason for these different results is that the multivariate GLMs take into account all variables. Nevertheless, the result from the full model is in accordance with some other pond studies (e.g. Kadoya et al., 2004; Oertli et al., 2002). We suggest that larger ponds have more habitats, which support more species (Rosenzweig, 1995). A larger effective area also results in a higher population size which reduces extinction rates (Melbourne and Hastings, 2008). Although Goertzen and Suhling (2013) found a negative relationship between pond size and richness of *Odonata* in urban ponds, while Oertli et al., (2010) found a positive relationship for the same group. Furthermore, Oertli et al. (2010) showed that the relationship species richness and pond size was very weak for other taxonomic groups. In our study, the relationship between pond size and species richness also held only for some groups of species, and we found high and low species richness among the small ponds we sampled. Taking into account this wide span of values, we agree with Oertli et al. (2002) that conservation efforts should be applied to ponds of different sizes in order to cover the full taxonomic breadth of the insect fauna.

We did not find significant effects of our selected variables on Odonata, Coleoptera and Hemiptera, but we acknowledge that they might be influenced by other factors that were not measured in our study. Such differences make it difficult to provide overall recommendations for pond design and management although our results suggest maintenance of intermediate vegetation cover in ponds for some taxonomic groups. In addition, while vegetation cover is an important variable to improve overall species richness, the results of this study show that different environmental variables are important drivers of richness in different macro-invertebrate orders. Because of that, individual management strategies need to be developed to target either particular taxa or overall richness. Hassall et al. (2011) recommended in such cases to promote high diversity of different type of ponds (and thus with differing management and environmental factors that shape them) to promote high species diversity at landscape scale. Our results suggest that when planning ponds in green areas in cities, one should consider creating ponds of different sizes and both ponds with intermediate vegetation cover and ponds with other levels of vegetation to support species with other requirements.

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References

Åstebøl, S.U., Hvidved-Jacobsen, T., Simonsen, Ø., 2004. Sustainable stormwater management at Fornebu—from an airport to an industrial and residential area of the city of Oslo, Norway. Sci. Total Environ. 334–335, 239–249.

Andersson, E., et al., 2009. Patterns and scale relations among urbanization measures in Stockholm, Sweden. Landsc. Ecol. 24, 1331–1339.

Andersson, J., 2014. Aquatic Insect Community Structure in Urban Pond s: Effects of Environmental Variables. Degree Project in Biology, Master of Science. Biology Education Centre and Department of Ecology and Genetics. Uppsala University, Sweden.

Angold, P.G., Sadler, J.P., Hill, M.O., Pullin, A., Rushton, S., Austin, K., Small, E., Wood, B., Wadsworth, R., Sanderson, R., Thompson, K., 2006. Biodiversity in urban habitat patches. Sci. Total Environ. 360, 196–204.

Bazzanti, M., Coccia, C., Dowgiallo, M.G., 2010. Micro distribution of macroinvertebrates in a temporary pond of Central Italy: taxonomic and functional analyses. Limnologica 40, 291–299.

Bendel, R.B., Afifi, A.A., 1977. Comparison of stopping rules in forward 'Stepwise' regression. J. Am. Stat. Assoc. 72, 46–53.

Biggs, J., Williams, P., Whitfield, M., Nicolet, P., Weatherby, A., 2005. 15 years of pond assessment in Britain: results and lessons learned from the work of pond conservation. Aquat. Conserv. Mar. Freshw. Ecosyst. 15, 671–693.

Carrier, J.-A., Beebee, T.J.C., 2003. Recent, substantial, and unexplained declines of the common toad Bufo bufo in lowland England. Biol. Conserv. 111, 395–399.

Colding, et al., 2009. Golf courses and wetland fauna. Ecol. Appl. 19, 1481–1491.

European Commission, 2010. http://ec.europa.eu/environment/europeangreencapital/winning-cities/2010-stockholm (accessed on 13.03.15).

Fuyuki, A., Yamaura, Y., Nakajima, Y., Ishiyama, N., Akasaka, T., Nakamura, F., 2014. Pond area and distance from continuous forests affect amphibian egg distributions in urban green spaces: a case study in Sapporo, Japan. Urban For. Urban Green. 13, 397–402.

Ghert, S.D., Chelsvig, J.E., 2003. Bat activity in an urban landscape: patterns at the landscape and microhabitat scale. Ecol. Appl. 13, 939–950.

Gledhill, D.G., James, P., 2012. Socio-economic variables as indicators of pond conservation in an urban landscape. Urban Ecosyst. 15, 849–861.

Gledhill, D.G., James, P., Davies, D.H., 2005. Urban pond: a landscape of multiple meanings. In: Paper Presented at the 5th International Postgraduate Research Conference in The Built and Human Environment, University of Salford.

Goertzen, D., Suhling, F., 2013. Promoting dragonfly diversity in cities: major determinants and implications for urban pond design. J. Insect Conserv. 17, 299–399.

Hamer, A.J., Smith, P.J., McDonnel, M.J., 2012. The importance of habitat design and aquatic connectivity in amphibian use of urban stormwater retention ponds. Urban Ecosyst. 15, 451–471.

Harrison, C., Burgess, J., Millward, A., Dawe, G., 1995. Accessible Natural Greenspace in Towns and Cities: A Review of Appropriate Size and Distance Criteria English Nature Research Report No. 153. English Nature, Peterborough.

Hassal, C., 2014. The ecology and biodiversity of urban ponds. Wiley Interdiscip. Rev.: Water 1, 187–206.

Hassall, C., Anderson, S., 2015. Stormwater ponds can contain comparable biodiversity to unmanaged wetlands in urban areas. Hydrobiologia 745, 137–149.

Hassall, C., Hollinshead, J., Hull, A., 2011. Environmental correlates of plant and invertebrate species richness in ponds. Biodivers. Conserv. 20, 3189–3222.

Hill, M.J., Mathers, K.L., Wood, P.J., 2015. The aquatic macroinvertebrate biodiversity of urban ponds in a medium-sized European town (Loughborough, UK). Hydrobiologia 766, 225–238.

Hinden, H., Oertli, B., Menetrey, N., Sager, L., Lachavanne, J.-B., 2005. Alpine pond biodiversity: what are the related environmental variables? Aquat. Conserv. 15, 613–624.

Huston, M.A., 2014. Disturbance, productivity, and species diversity: empiricism vs logic in ecological theory. Ecology 95, 2382–2396.

Kadoya, T., Suda, S., Washitani, I., 2004. Dragonfly species richness on man-made ponds: effects of pond size and pond age on newly established assemblages. Ecol. Res. 19, 461–467.

Kvale, S., 1996. InterViews – An Introduction to Qualitative Research Interviewing Thousand Oaks. SAGE Publications, California.

Lamy, T., 2013. Variation in habitat connectivity generates positive correlations between species and genetic diversity in a metacommunity. Mol. Ecol. 22, 4445–4456.

Le Viol, I., Mocq, J., Julliard, R., Kerbiriou, C., 2009. The contribution of motorway stormwater retention ponds to the biodiversity of aquatic macroinvertebrates. Biol. Conserv. 142, 3163–3171.

Leibold, M.A., 1999. Biodiverity and nutrient enrichment in pond plankton communities. Evol. Ecol. Res. 1, 73–95.

McDonald, R.I., Kareiva, P., Forman, R.T.T., 2008. The implications of current and future urbanization for global protected areas and biodiversity conservation. Biol. Conserv. 141, 1695–1703.

McKinney, M.L., 2006. Effects of urbanisation on species richness: a review of plants and animals. Urban Ecosyst. 11, 161–176.

Melbourne, B.A., Hastings, A., 2008. Extinction risk depends strongly on factors contributing to stochasticity. Nature 454, 100–103.

Miller, J.R., Hobbs, R.J., 2002. Conservation where people live and work. Conserv. Biol. 16, 330–337.

Nassauer, J., 2004. Monitoring the success of metropolitan wetland restorations: cultural sustainability and ecological function. Wetlands 24, 756–765.

Noble, A., Hassal, C., 2014. Poor ecological quality of urban ponds in northern England: causes and consequences. Urban Ecosyst. 18, 649–662.

Oertli, B., Joye, D.A., Castella, E., Juge, R., Cambin, D., Lachavanne, J.B., 2002. Does size matter? The relationship between pond area and biodiversity. Biol. Conserv. 104, 59–70.

Oertli, B., Céréghino, R., Biggs, J., Declerck, S., Hull, A.P., Miracle, M.R., 2010. Pond Conservation in Europe. Developments in Hydrobiology, vol. 210. Spinger Verlag.

Ohlin, V., 2013. Groddjur i Stockholm, projektrapportering. Lokala Naturvårdsatsningen. Stockholm Stad, Stockholm.

Pond Conservation, 2002. A Guide to Monitor the Ecological Quality of Ponds and Canals Using PSYM. Pond Conservation Trust. Oxford Brookes University.

Remsburg, A.J., Turner, M.G., 2009. Aquatic and terrestrial drivers of dragonfly (Odonata) assemblages within and among north-temperate lakes. J. North Am. Benthol. Soc. 28 (1), 44–56.
Riley, S.P.D., Busteed, G.T., Kats, L.B., Vandergon, T.L., Lee, L.F.S., Dagit, R.G., Kerby, J.L., Fisher, R.N., Sauvajot, R.M., 2005. Effects of urbanization on the distribution and abundance of amphibians and invasive species in Southern California streams. Conserv. Biol. 19, 1894–1907.
Rosenzweig, M.L., 1995. Species Diversity in Space and Time. Cambridge University Press, Cambridge.
SEPA (Swedish Environmental Protection Agency), 2006. Bottenfauna I sjöars litoral och vattendrag. SS–EN 27 828.
Segaran, R.R., Lewis, M., Ostendorf, B., 2014. Stormwater quality improvement potential of an urbanised catchment using water sensitive retrofits into public parks. Urban For. Urban Green. 13, 315–324.
St Pierre, J.I., Kovalenko, K.E., 2014. Effect of habitat complexity attributes on species richness. Ecosphere 5 (2), art22.
Starkl, M., Amerasinghe, P., Essl, L., Jampani, M., Kumar, D., Asolekar, S.R., 2013. Potential of natural treatment technologies for wastewater management in India. J. Water Sanit. Hyg. Dev. 3, 500–511.
Stockholm Stad, 2013. http://www.stockholm.se/OmStockholm/Fakta-och-kartor (accesses 02.06.15).
Vermonden, K., Leuven, R.S.E.W., van der Velde, G., van Katwijk, M.M., Roelofs, J.G.M., Jan Hendriks, A., 2009. Urban drainage systems: an undervalued habitat for aquatic macroinvertebrates. Biol. Conserv. 142, 1105–1115.