Landscape and Urban Planning 176 (2018) 22–29

Contents lists available at ScienceDirect

Landscape and Urban Planning

journal homepage: www.elsevier.com/locate/landurbplan

Research Paper

Urban hot-tubs: Local urbanization has profound effects on average and extreme temperatures in ponds

Kristien I. Brans*,[1], Jessie M.T. Engelen[1], Caroline Souffreau, Luc De Meester

Laboratory of Aquatic Ecology, Evolution and Conservation, KU Leuven, Ch. Deberiotstraat 32, B-3000 Leuven, Belgium

ARTICLE INFO

Keywords:
Ponds
Urban blue space
Urban heat island effect
Temperature
Growing season
Anthropogenic warming

ABSTRACT

While urbanization-driven warming (urban heat island effect, UHI) has been extensively studied and demonstrated for air temperature, UHI effects on water temperature of ponds are unknown. We investigated (1) whether the UHI impacts man-made urban ponds and tested whether urban ponds have higher mean, maximum and minimum water temperatures and lower daily water temperature fluctuations than rural ponds, (2) whether this is related to time of the day (day versus night), season, and urbanization scale (3200 versus 50 m radius around the pond), and (3) whether the approximated length of growing season is prolonged in urban ponds. Temperature loggers were placed in 30 ponds in Northern Belgium, spanning a broad range of urbanization. We found strong evidence of urban-driven warming. Mainly local urbanization (50 m radius) drove temperature differences throughout the year and even more so in spring and summer, with mean summer temperatures being up to 3.04 °C higher in urban compared to rural ponds, and maximum summer temperatures on average up to 3.69 °C higher. Strikingly, daily temperature fluctuated around 2 °C more in locally urban ponds compared to rural ponds in summer. Length of the growing season estimates show prolongation with up to 45 days in locally urban compared to rural systems, mainly due to an earlier start. Generally, our results show that UHIs impact water temperature of ponds. This warming can have profound consequences for biota inhabiting these systems, and should therefore be considered in future urban planning to reduce deterioration of these habitats and improve their socio-ecological value.

1. Introduction

More than half of the world's population currently lives in densely concentrated urban cores (UN, 2014). These urban cores generate complex environments, as they integrate a multitude of physical, chemical and climatic gradients (Grimm et al., 2008; Kaye, Groffman, Grimm, Baker, & Pouyat, 2006). One consequence of urbanization is a microclimate shift, known as the urban heat island effect (UHI, Arnfield, 2003; Oke, 1973; Ward, Lauf, Kleinschmit, & Endlicher, 2016), which involves an increased ambient temperature in city centers compared with rural areas and the urban fringe. This UHI effect (Bornstein, 1968; Oke, 1973; Voogt & Oke, 2003) is the result of increased land cover by buildings, roads, and other artificial surfaces linked to urbanization and industrialization (Rizwan, Dennis, & Liu, 2008; Stewart & Oke, 2012). Specific street geometric characteristics, the proportion of artificial surfaces with low albedo values and high warmth emissivity, anthropogenic heat production, and city size influence the strength of the UHI effect and have been used to predict its intensity (Hart & Sailor, 2009; Oke, 1982; Voogt & Oke, 2003; Ward et al., 2016).

While the UHI effect has been extensively studied and shown for air temperatures (Hart & Sailor, 2009; Ward et al., 2016), so far effects on water temperature have been largely overlooked. Recent evidence supports the warming of streams and rivers due to urbanization (Hester & Bauman, 2013; Kaushal, Likens, Jaworski, et al., 2010). Surrounded by high levels of built-up area and impervious surfaces, rivers and streams in urban areas often receive warm water inflows, especially in summer and during heat waves, causing thermal pollution (Herb, Janke, Mohseni, & Stefan, 2008; Hester & Bauman, 2013; Lieb & Carline, 2000). For standing waters such as ponds and shallow lakes, however, no data are available on the UHI effect on water temperature nor on the potential mechanisms behind it. We expect that higher ambient air temperatures in cities, less shading, and thus higher solar irradiation, will indeed impact the water temperature of ponds in urban areas. Although the high specific heat capacity of water reduces the speed with which temperatures rise in aquatic systems, high thermal inertia also slows down cooling during the night and might thus increase the UHI effect during sufficiently long-lasting warm periods and

* Corresponding author.
E-mail address: kristien.brans@kuleuven.be (K.I. Brans).
[1] These authors contributed equally to this work.

https://doi.org/10.1016/j.landurbplan.2018.03.013
Received 28 May 2017; Received in revised form 23 January 2018; Accepted 15 March 2018
0169-2046/

heat waves. Because of these slow cooling rates, we expect daily (i.e. measured over 24 h periods) temperature fluctuations to be relatively stable in urban ponds during summer, causing a relatively narrow daily temperature range. Larger sized cities (high regional levels of urbanization) can induce a prolonged heat island effect in autumn and winter by trapping heat on a regional scale, thereby creating a warmer microclimate within the city (Schwarz, 2010). Therefore during autumn and winter regional urbanization could induce temperature differences, albeit of a smaller intensity, in ponds situated in large urban areas compared to ponds in large rural areas.

Ponds, pools, and small lakes provide an essential habitat for aquatic biodiversity as they often have well-developed aquatic vegetation supporting multiple organism groups such as amphibians and insects (Hassall, 2014; Hill, Biggs, Thornhill, et al., 2016; Williams et al., 2004). In urbanized areas, ponds are often created by humans for their esthetic and recreational value (e.g. garden ponds, city park ponds) or with a functional purpose (e.g. stormwater wet ponds which capture stormwater runoff from impervious surfaces to mitigate flooding and improve water quality in urban areas), while the biodiversity value of these systems is secondary. Yet, recent studies emphasize the importance of urban ponds for aquatic biodiversity and the fact that biodiversity should be taken into account in pond design (e.g. Gianuca, Pantel, & De Meester, 2016; Hassall, 2014; Hill et al., 2016). Temperature is one of the most important drivers of physiological processes and phenology in nature (Angilletta, 2009), and higher water temperatures due to the UHI effect might therefore have a strong impact on aquatic biota (Brans, Jansen et al., 2017). The early onset and extended duration of the growing season in urban areas (Zhang, Friedl, Schaaf, Strahler, & Schneider, 2004; Zipper et al., 2016) triggers early flowering in plants and causes phenological shifts in many ectotherms (Kaiser, Merckx, & Van Dyck, 2016; Neil & Wu, 2006). In aquatic systems, the impact of the UHI effect will depend on its effects on average and maximum temperatures (e.g. during heat waves), seasonality, and amplitudes of temperature fluctuations. As a consequence of these changes, the length of the growing season for aquatic organisms might be profoundly impacted by the UHI, as already observed in streams (Nelson & Palmer, 2007).

In this study, we tested whether the UHI also affects the water temperature of urban ponds. We monitored water temperature in 30 ponds situated over a broad range of urbanization using temperature loggers during a 12-month period. We hypothesized (1) that urban ponds exhibit significantly higher temperatures than rural ponds, (2) that the difference in mean, maximum, and minimum temperature between urban and rural systems strongly depends on the scale at which urbanization is assessed (local versus regional), the time of day (day versus night), and the season, (3) that daily variation in temperatures is more buffered in ponds in larger urban areas (regional scale) compared to rural ponds, and (4) that the approximated growing season starts earlier and ends later in urban compared to rural ponds.

2. Materials and methods

2.1. Study design and data acquisition

2.1.1. Study area, urbanization levels, and temperature monitoring

We installed 30 temperature loggers (HOBO TidbiT v2 Water Temperature Logger UTBI-001) in 30 ponds along a well-defined urbanization gradient (Fig. 1; Table S1 in Supplementary Information). All selected ponds were shallow (< 3 m maximum depth), with a focus on small- to intermediate-sized (all studied ponds < 2 ha, according to Hassall, 2014; all ponds except two < 0.5 ha) man-made city ponds, park ponds, garden ponds and ponds in (semi-)natural areas (e.g. ponds created as part of nature conservation activities or old farmland ponds), not connected to other water-related infrastructure. Percentage built-up area (BA) was derived from the Large Reference Database (LRD; available via the Flemish Institute for Geographic Information, FIGA) and was used as a proxy for the level of urbanization (Table S1, see also Brans, Govaert, et al., 2017; Gianuca et al., 2016). LRD is an object-oriented, vector-based map with a scale of 1/250. The accuracy of the location of buildings varies between 5 cm (when measured *in situ*) and 20 cm (when based on aerial pictures). This metric excludes roads and parking lots. Therefore, a BA of 10% already indicates a high amount of urbanization. Ponds were categorized according to percentage BA around the pond into being urban (> 10% BA), semi-urban (5–10% BA), and rural (< 5% BA) ponds at both a local (50 m radius around the pond, excluding the pond itself) and regional (3200 m radius) scale (in line with Piano, De Wolf, Bona, et al., 2017). This entails that rural ponds at the local scale are purely defined based on percentage BA within a 50 m radius, so that this category groups ponds in nature reserves, farmland ponds, park and garden ponds in cities provided that locally the percentage of built-up area is low. The number of ponds in the local and regional urbanization categories was balanced (Table S1). Temperature loggers were attached to a floating device in order to keep the loggers at a depth of 15 cm below water surface (construction information see Fig. S1 in Supplementary Information). Temperature was logged at 15 min intervals (96 measurements/day) for a period of one year (November 2014–November 2015, 367 days). Data output from each logger was obtained using the HOBO digital software (HOBOware, ONSET). Example temperature profiles for the region of Ghent and Leuven are given in Fig. 2.

2.1.2. Temperature response variables

Based on the raw data from the temperature loggers we calculated eight temperature regime proxies for all ponds: mean, minimum and maximum daily temperature, daily temperature range, daytime mean temperature, daytime temperature range, nighttime mean temperature, and nighttime temperature range. Daily refers to the 24 h cycle, thus combining daytime and nighttime observations. Daily temperature range (DTR) was calculated as the difference in minimum and maximum daily temperatures. Time intervals for calculating daytime and nighttime temperature means and ranges were based on average sunset and sunrise times for each month (see Table S2).

Additionally, three parameters related to growing season were calculated: start, end, and length of the growing season. Length of the growing season (LGS) was approximated as the number of days the mean daily water temperature exceeded a certain threshold value (8, 10, 12, 14, 16, 18, or 20 °C); these temperatures span a range corresponding to ecologically relevant thermal stimuli for organisms known from literature (Jobling, 1981; Schwartz & Hebert, 1987; Ward & Stanford, 1982). Start of the growing season was the first of three consecutive days at which water reached a temperature higher than the respective threshold value. End of the growing season was considered as the first day at which water temperature no longer exceeded the threshold value.

2.2. Data analysis

From the initial set of 30 temperature loggers, six loggers were removed by a third party and another six loggers were removed from analyses due to logging errors (accidental measuring of air or sediment temperatures or improper functioning of the device). Raw data of the remaining 18 temperature loggers were checked for inconsistencies. Three ponds showed obvious errors for longer time periods (perhaps because they were temporarily exposed to the air or because of software problems); these time periods were treated as missing values (Mechelen: n = 129 days, Rotselaar 1: n = 90 days, Rotselaar 2: n = 212 days). Mean daily temperature values that exceeded 3 standard deviations from mean annual temperatures were excluded because such extreme values are indicative of temperature measurements of loggers exposed to the air (DeWeber & Wagner, 2014). This resulted in the removal of 1 to 14 daily temperature observations in a total of 16 ponds.