Urban stream temperature patterns: Spatial and temporal heterogeneity in the Philadelphia region, Pennsylvania, USA

Accepted: Hydrological Processes January 2021

Running title: Urban stream temperature variability

Sarah Beganskas[1*] and Laura Toran[1]

[1]Earth and Environmental Science Department, Temple University, Philadelphia, PA 19122
[*]Corresponding author, sarah.beganskas@temple.edu

KEYWORDS

Stream temperature, urbanization, stormwater runoff, land cover, thermal mitigation, wastewater treatment plants (WWTPs), Wissahickon Creek, Cobbs Creek

ABSTRACT

Stream temperature is a critical water quality parameter that is not fully understood, particularly in urban areas. This study explores drivers contributing to stream temperature variability within an urban system, at 21 sites within the Philadelphia region, Pennsylvania, USA. A comprehensive set of temperature metrics were evaluated, including temperature sensitivity, daily maximum temperatures, time >20°C, and temperature surges during storms. Wastewater treatment plants (WWTPs) were the strongest driver of downstream temperature variability along 32 km in Wissahickon Creek. WWTP effluent temperature controlled local (1–3 km downstream) temperatures year-round, but the impacts varied seasonally: during winter, local warming of 2–7°C was consistently observed, while local cooling up to 1°C occurred during summer. Summer cooling and winter warming were detected up to 12 km downstream of a WWTP. Comparing effects from different WWTPs provided guidelines for mitigating their thermal impact; WWTPs that discharged into larger streams, had cooler effluent, or had lower discharge had less effect on stream temperatures. Comparing thermal regimes in four urban headwater streams, sites with more local riparian canopy had cooler maximum temperatures by up to 1.5°C, had lower temperature sensitivity, and spent less time at high temperatures, although mean temperatures were unaffected. Watershed-scale impervious area was associated with increased surge frequency and magnitude at headwater sites, but most storms did not result in a surge and most surges had a low magnitude. These results suggest that maintaining or restoring riparian canopy in urban settings will have a larger impact on stream temperatures than

stormwater management that treats impervious area. Mitigation efforts may be most impactful at urban headwater sites, which are particularly vulnerable to stream temperature disruptions. It is vital that stream temperature impacts are considered when planning stormwater management or stream restoration projects, and the appropriate metrics need to be considered when assessing impacts.

1 INTRODUCTION

Stream temperature is an important yet complex water quality parameter (Caissie, 2006). Fluctuations in stream temperature affect the physiology, distribution, and behavior of fish (Brewitt & Danner, 2014; DeWeber & Wagner, 2015; Whitledge, Rabeni, Annis, & Sowa, 2006) and macroinvertebrates (Sponseller, Benfield, & Valett, 2001; Stewart, Close, Cook, & Davies, 2013) living in streams. Water temperature is also an important control on in-stream nutrient metabolism and respiration (Acuña, Wolf, Uehlinger, & Tockner, 2008; Jabiol et al., 2020), with far-reaching water quality and stream health implications. Understanding stream thermal regimes is an ongoing field of research with many unanswered questions (Webb, Hannah, Moore, Brown, & Nobilis, 2008).

There are relatively few studies on stream temperature in urban settings compared with forested regions. Many interacting factors affect urban stream temperatures as part of the widely-studied urban stream syndrome, a consistently-observed response to urbanization which includes flashier streams, less diverse biota, and increased pollutant concentrations (Walsh et al., 2005). Thermal components of urban stream syndrome include increased solar radiation from reduced riparian shading (Booth, Kraseski, & Jackson, 2014; Roy, Faust, Freeman, & Meyer, 2005), increased air temperatures from the urban heat island effect (Pagliaro & Knouft, 2020), influx of heated runoff from impervious surfaces during storms (Nelson & Palmer, 2007), wastewater treatment plant (WWTP) effluent (Hamdhani, Eppehimer, & Bogan, 2020), and disconnection from groundwater due to channelization (Anderson, Anderson, Thaxton, & Babyak, 2010). Understanding spatial and temporal variability in urban stream temperatures has important implications for stream ecology and management (Wenger et al., 2009).

Many studies show that urban watersheds have different thermal regimes when compared with forested watersheds. Urban streams tend to have warmer mean and maximum temperatures and more frequent, intense pulses of heat during storms than forested streams (Arora, Toffolon, Tockner, & Venohr, 2018; Fanelli, Prestegaard, & Palmer, 2019; Somers et al., 2013; Walsh et al., 2005). However, there is substantial variability in streams' response to urbanization (Booth, Roy, Smith, & Capps, 2016). Regional variability in climate, geology, land cover, infrastructure design, and other factors may impact an individual stream's response to urbanization (Hale, Scoggins, Smucker, & Suchy, 2016; Parr, Smucker, Bentsen, & Neale, 2016). Some studies find that watershed-scale urban land cover is critical for determining stream temperature response (Rice, Anderson, & Thaxton, 2011; Singh & Chang, 2014), while others emphasize the importance of local riparian vegetation loss (Sponseller et al., 2001; Sun, Yearsley, Voisin, & Lettenmaier, 2015).

Because stream temperature varies hourly, daily, and seasonally, the thermal regime at one site is difficult to represent simply for comparison with other sites in assessing spatial variability. Many metrics are used to evaluate stream temperature (Steel et al., 2017), including mean, minimum, or maximum temperature (on daily, weekly, monthly, or annual scales); daily temperature range; time spent above a threshold temperature; annual temperature sensitivity; and the magnitude, frequency, and duration of pulses of heat ("surges") during storms. Metrics vary in terms of the length and resolution of temperature data required and their relevance to stream biota. For instance, mean temperatures do not include information about daily thermal ranges, although both factors independently affect stream biota (Wehrly, Wiley, & Seelbach, 2003). Similarly, temperature ranges do not indicate how much time was spent above a temperature threshold, even though exposure time is an important factor in determining thermal tolerance (Beitinger, Bennett, & McCauley, 2004). Many stream temperature studies focus on a limited set of metrics.

In this study, we comprehensively analyze one year of stream temperature data from 21 in-stream locations and 3 WWTPs within the Philadelphia region using many temperature metrics. Rather than comparing urban watersheds with forested watersheds, we aim to explore stream temperature variability within an urban system. We use these data to evaluate the following hypotheses:

1. WWTP effluent will cause local warming 1–3 km downstream of the treatment plants relative to upstream (Hamdhani et al., 2020). Downstream temperature increases will be largest in winter, when the difference between effluent and receiving stream temperature is largest (Kinouchi, Yagi, & Miyamoto, 2007).
2. Urban subwatersheds with more riparian canopy (Sweeney & Newbold, 2014) and less impervious area (Somers et al., 2013) will have cooler stream temperatures. Moving downstream, sites with more riparian canopy will have cooler temperatures (Coats & Jackson, 2020).
3. Surges of heat during storms will have a significant impact on stream temperatures and subwatersheds with more impervious area will have more frequent, larger surges (Nelson & Palmer, 2007; Somers et al., 2013).

Evaluating whether these hypotheses hold true in the Philadelphia region will contribute to an assessment of the relative impact of several urban drivers of stream temperature variability. Putting these drivers in context with appropriate metrics will provide guidance for stream temperature mitigation in other urban settings.

2 METHODS

2.1 Study sites

This study focuses on 21 sites within Wissahickon Creek and Naylors Run watersheds in the Philadelphia region, Pennsylvania, USA (**Figure 1A**). Wissahickon Creek discharges into the Schuylkill River and runs through Montgomery and Philadelphia Counties. Including its tributaries, the Wissahickon Creek watershed has a stream length of 185 km (USGS, 2019), and drains 165 km^2. Naylors Run is a tributary to Cobbs Creek within the Darby Creek watershed in Delaware County. Naylors Run comprises 8.8 km of stream length within a drainage area of 11.8 km^2. The Wissahickon watershed was divided into sub-watersheds for analysis: Upper Wissahickon (35.1 km^2), Middle Wissahickon (34.7 km^2), and Lower Wissahickon (54.4 km^2) comprise the main stem of Wissahickon Creek, while Sandy Run (32.3 km^2) and Paper Mill Run (6 km^2) are tributaries.

Land cover varies throughout these urban watersheds (**Figure 1B**). A riparian buffer has been established and protected along much of Wissahickon Creek and the lower third of the main stem runs through wooded Fairmont Park, Philadelphia. As a result, main-stem Wissahickon Creek and Sandy Run have "inverted" watersheds in which percent tree canopy increases and impervious area decreases moving downstream (**Figure S1**). Like many urban streams, Naylors Run, Wissahickon Creek, and its tributaries are classified as having impaired water quality (PADEP, 2018) and are characterized by low species diversity and tolerant benthic macroinvertebrate and fish species (DCVA, 2002; PWD, 2007). Both Wissahickon Creek and Naylors Run are classified as warm-water-fish habitat, although Wissahickon Creek is also trout-stocked (EPA, 2018).

Four WWTPs are located within the Wissahickon watershed. Upper Gwynedd WWTP discharges into the Upper Wissahickon, Ambler WWTP discharges into the Middle Wissahickon, and Abington and Upper Dublin WWTPs discharge into Sandy Run. Effluent from Abington, Upper Gwynedd, and Ambler WWTPs comprises a considerable fraction (up to 75%) of baseflow, while the Upper Dublin WWTP releases effluent at a lower rate. Discharge from Abington and Ambler WWTPs generally falls between 0.1 and 0.15 m^3/s, while discharge from Upper Gwynedd is usually 0.06–0.1 m^3/s (**Figure S2**). Temperature data were also collected at or near the outlet of Abington, Upper Gwynedd, and Ambler WWTPs to represent effluent conditions. Site names around the WWTPs include approximate distance upstream or downstream of the treatment plants (e.g., USUG2 is upstream [US] of Upper Gwynedd [UG] by ~2 km [2] and DSAb3 is downstream [DS] of Abington [Ab] by ~3 km [3]). Paper Mill Run, Lower Wissahickon, and Naylors Run have no WWTPs, although Lower Wissahickon sites are located downstream of WWTPs farther upstream.

Six headwater sites were identified that are upstream of WWTPs and have drainage areas <16 km^2 (**Figures 1A, S3**): Drexel Gardens Park (DGP, Naylors Run), Naylors Run Park (NRP, Naylors Run), Upstream Upper Gwynedd ~2 km (USUG2, Upper Wissahickon), Upstream Upper Gwynedd (USUG, Upper Wissahickon), Upstream Abington ~2 km (USAb2, Sandy Run), and Paper Mill Run (PMR). To roughly compare the relative baseflow at these headwater sites, 1–2 discharge measurements were taken at each site using a Sontek Handheld ADV

FlowTracker. Measurements were taken during the summer after several days with no rain to ensure baseflow conditions were represented. Direct comparison between sites is not possible as measurements were conducted on different days; however, the values provide the order of stream size. Baseflow measurements were <0.01 m^3/s at USUG2 and USAb2, ~0.01 m^3/s at DGP, and 0.02–0.06 m^3/s at NRP, PMR, and USUG.

2.2 Regional air temperature and precipitation data

Hourly air temperature and precipitation data were gathered from 22 weather stations (**Figure 1B**). We operate one weather station and data from the other 21 stations were downloaded from the Weather Underground Sensor Network (https://www.wunderground.com/wundermap), a network of personal and official weather stations (Audia et al., 2020). These crowd-sourced climate data have extensive spatial coverage, although the data may vary in quality (Hammerberg et al., 2018). The 21 Weather Underground stations selected for this study were screened for quality by comparison with data from the station we operate; the added spatial coverage allowed us to incorporate regional variability in air temperatures and precipitation. Weather station data were used to create a unique weather record for each subwatershed (**Figure 1A**) by averaging data from 4–7 nearby stations on an hourly time scale. Using multiple stations for each subwatershed ensured better representation of conditions throughout, particularly for precipitation, and reduced dependence on any one record. **Figure S4** shows weekly weather station data for each subwatershed. Hourly precipitation records were used to determine daily total precipitation, daily maximum hourly precipitation intensity, and the hour of maximum intensity for each day.

2.3 Stream temperature

One year of 15-min temperature data were collected during 2017–2018 at 21 in-stream sites and 3 WWTPs (**Figure 1A**) using water level (Onset HOBO U20, 0.44°C accuracy, 0.1°C resolution) and/or turbidity loggers (YSI 600OMS V2, 0.15°C accuracy, 0.01°C resolution). These loggers were placed in flowing water along the bank and shielded from incoming solar radiation in corrugated plastic tubing. Data were collected in all Wissahickon subwatersheds from 5/1/17 to 4/30/18 and in Naylors Run from 8/1/17 to 7/31/18; data gaps are noted in **Table 1**. Note that Naylors Run data are offset by three months relative to the other subwatersheds. Weather during

May–July 2017 compared to 2018 was similar enough to allow reasonable comparisons of annual stream temperature metrics between sites, but monthly comparisons for May, June, and July are not considered due to the timing discrepancy (**Figure S5**).

For each site, we used Python (v. 2.7) to calculate daily minimum temperature, maximum temperature, temperature range, mean temperature, and total time above 20°C and 25°C. These daily metrics were averaged or summed into weekly, monthly, and annual records. Temperature sensitivity at each site was calculated as the slope of a linear regression between weekly mean stream and air temperatures (Kelleher et al., 2012). The strength (r^2) of this linear regression and the difference between mean monthly stream and air temperatures in the warmest (July) and coolest (January) months were also determined (Beaufort, Moatar, Sauquet, Loicq, & Hannah, 2020).

Stream temperature surges were identified as instances when temperature increased by ≥2°C in 30 min during a precipitation event (Nelson & Palmer, 2007). We identified surges in stream temperature records and used hourly precipitation data to confirm when a storm was the likely cause. We focus on surges during May–September, as storms during these months are most likely to deliver substantial heat input to streams (Herb, Janke, Mohseni, & Stefan, 2008). Surge frequency was calculated as the percentage of rainy days on which a surge was observed; this approach accounted for differences in precipitation patterns between sites. Surge magnitude represented the relative intensity of the surge within the thermal regime at each site. The maximum temperature during each surge was compared to the average daily maximum temperature at that site during that month (not including days with temperature surges), and surge magnitude was defined as the number of standard deviations (SDs) above the mean maximum temperature. This approach does not rely on pre-surge stream temperature, which varies with season and time of day. Surge duration was defined as the time it took for stream temperatures to return to within 2°C of pre-surge values (Nelson & Palmer, 2007).

Defining criteria to evaluate when a downstream site was no longer impacted by WWTP effluent temperatures was challenging. For local-scale impacts (≤3 km), we considered stream temperatures "recovered" if mean weekly temperatures were within 0.5°C of temperatures

upstream of the WWTP. Outside of local-scale WWTP impact, it was uncommon to see mean weekly temperatures increase by more than 0.5°C between consecutive downstream sites. On the other hand, sites farther downstream have higher baseflow due to tributary input, and we might not expect temperatures there to return to upstream temperatures, particularly for WWTPs that discharge into headwater streams. Therefore, we considered farther downstream sites to be "recovered" if stream temperatures were similar to the Mouth of Wissahickon Creek, even if these temperatures remained elevated relative to upstream temperatures. This approach accounts for downstream warming (**Figure S6**), as the Mouth is the farthest downstream site and is least affected by WWTP input. These interpretations assume that the Mouth is fully disconnected from WWTP thermal input and other factors that may significantly impact its temperature relative to upstream sites, including localized groundwater discharge. As it is impossible to know what temperatures in Wissahickon Creek would be without WWTPs, these interpretations are inherently uncertain.

2.4 Land cover

Land cover data for the region were obtained at 1-m resolution with these categories: barren, tree canopy, mown grass, roads, structures, and other impervious surfaces (UVM Spatial Analysis Lab, 2016). For each in-stream site, we delineated the area draining into that point and clipped the drainage area to include the region up to 50 m from the stream channel and no more than 200 m upstream, representing a local riparian buffer (Sponseller et al., 2001; Sweeney & Newbold, 2014). We calculated the percent of each land cover type at local riparian and watershed scales; we also aggregated roads, structures, and other impervious surfaces into a single category representing all impervious area.

Land cover analyses focused on the six headwater sites (see Section 2.1) to avoid confounding impacts from WWTPs. At local riparian and watershed scales, we evaluated the strength and statistical significance of correlations between each land cover type and an abbreviated set of stream temperature metrics: mean monthly maximum and mean temperatures, temperature sensitivity, and total time above 20°C. For the monthly metrics, we excluded May, June, and July to avoid interannual differences (see Section 2.3). We evaluated both Pearson and Spearman correlation; as results were similar, we present only Pearson results.

3 RESULTS

3.1 Downstream temperature patterns & WWTPs

3.1.1 Stream temperature characterization

Analysis of downstream trends focused on 32 km along main-stem Wissahickon Creek. Moving downstream, mean annual temperature at the Mouth was 0.5°C warmer than at the most upstream site (USUG2), resulting in low average rates of downstream warming (~0.015°C/km; **Figure 2A**). Downstream warming rates varied throughout the year; on a weekly scale USUG2 was up to 2.2°C warmer than the Mouth (**Figure S6**). There was no apparent correlation between mean annual stream temperature and distance downstream or drainage area (r^2 = 0.02 and r^2 = 0.03, respectively).

Average daily stream temperature ranges showed more variability between sites than monthly mean temperatures (**Figure 2B**). Mean annual daily temperature ranges decreased moving downstream from >4°C at the Wissahickon (USUG2) and Sandy Run (USAb2) headwaters to 1.7°C at the Mouth. Increasing minimum temperatures contributed about twice as much to this downstream trend as decreasing maxima. The average minimum at the Mouth was ~1.5°C warmer than at headwater sites USUG2 and USAb2, while the average maximum was ~0.8°C cooler than USUG2 and USAb2 (**Figure S7**). Drainage area and annual mean range had a modest inverse correlation (r^2 = 0.51, p = 0.15).

Stream temperatures regularly exceeded 20°C at all sites in this study during May through October (**Figure 3A**); the six headwater sites (USUG2, USUG, USAb2, PMR, DGP, and NRP) spent substantially less time above 20°C than downstream sites (**Table 2**). While most sites exceeded 25°C at some point, such high stream temperatures were not sustained; in July, sites spent at most ~6 hr/d above 25°C (**Figure 3B**). Headwater sites showed variability, with Paper Mill Run (PMR) exceeding 25°C most often.

The slope of stream–air temperature relationships (temperature sensitivity) at all sites ranged 0.52–0.81 and averaged 0.73; all sites had r^2 >0.9 between mean weekly air and stream temperatures (**Table 2**). At all but four sites, July stream temperatures were cooler than air

temperatures, with about half the sites ~1°C cooler. Below-freezing air temperatures resulted in average January stream temperatures at least 3°C warmer than air at all sites. There was not a clear relationship between drainage area and temperature sensitivity ($r^2 = 0.14$).

3.1.2 Effect of WWTP effluent on stream temperatures

WWTPs had significant local-scale impact on stream temperatures; sites ~1 and ~3 km downstream of the three WWTPs had different thermal regimes than sites 0–1 km upstream. ~1 km downstream of the WWTPs, mean annual temperatures were higher (**Figure 2A**) and temperature sensitivity lower (**Table 2, Figure S8**) than upstream. The magnitude of the impact varied between WWTPs. For example, Upper Gwynedd and Abington WWTPs were associated with large local increases in time spent over 20°C and decreases in daily mean temperature range during most months, while Ambler was not (**Figures 2B,3A**).

WWTP impact on local downstream temperatures varied throughout the year (**Figure 4**). In cooler months, there was significant local downstream warming relative to upstream; from November through January, mean monthly temperatures ~1 km downstream were at least 6°C, 3°C, and 1°C warmer than upstream of Abington, Upper Gwynedd, and Ambler WWTPs, respectively. In warmer months, local downstream impacts were more modest. During June and July, ~1 km downstream of Ambler and Abington WWTPs mean temperatures were 0.4–0.9°C cooler, maximum temperatures were 0.8–1.4°C cooler, and time >25°C was much lower than upstream (**Figures 2A, S7A, 3**).

The persistence of WWTP impact farther downstream was different for each WWTP and varied throughout the year. We evaluated trends in the difference between weekly mean stream temperatures upstream of the WWTP and temperatures at increasing downstream distances (**Figure 5**). At sites 3.2 km (DSAb3) and 6.2 km (SR-Mouth) downstream from Abington WWTP, weekly mean stream temperatures were 0.5–3.7°C warmer than upstream values from October through April and ~0.5°C cooler than upstream values from mid-June to mid-July. 3.0 km downstream of Upper Gwynedd WWTP (DSUG3), weekly mean stream temperatures were 0.6–2.6°C warmer than upstream values throughout the year. By 9.2 km downstream of Upper Gwynedd (USAmb), stream temperatures were close to upstream values in September–April but

remained elevated in May–August. 1.8 km downstream of Ambler WWTP (DSAmb3), weekly mean stream temperatures were within 0.5°C of upstream values except from October–January, when downstream temperatures were up to 1.9°C warmer than upstream. Below the confluence with Sandy Run, 2.8 km (FtWash) and 8.0 km (USPMR) downstream of Ambler WWTP, temperatures remained elevated.

3.2 Subwatershed comparisons: Land cover, storms, and stream temperature

3.2.1 Relationships between land cover and stream temperatures

The local riparian scale had much more land cover variation among the six headwater sites than the watershed scale (**Figure 6**). Local riparian impervious area did not correlate strongly with percent tree canopy or mown grass and there was no relationship between local- and watershed-scale land cover (**Figure S9**).

Sites with a greater fraction of local riparian canopy tended to spend less time >20°C (**Figure 7A**), with each additional percent canopy reducing annual time >20°C by 1.5 min/yr (0.025 hr/yr) on average. Similarly, during warm months, sites with more canopy had lower maximum temperatures (**Figure 7C**). In August, site PMR had an average maximum temperature of 24.3°C with 17% riparian canopy, while site USUG had an average maximum of 22.9°C with 97% riparian canopy. The slope of the relationship between percent canopy and monthly maximum temperature was consistent from August to October, with each additional percent canopy associated with a ~0.013°C reduction in maximum temperatures, although the relationship was weaker in October. During cooler months, there was no relationship between percent canopy and maximum temperatures (**Figure S10**). Sites with more local riparian canopy tended to have lower temperature sensitivity (**Figure 7E**), although this relationship was weaker than for monthly mean maximum temperatures and time >20°C. There were no significant relationships between percent canopy and monthly mean temperatures. At the local riparian scale, percent canopy and percent mown grass had a strong inverse correlation (r^2 = 0.89, **Figure S9A**), thus mown grass results showed similar but inverse relationships with stream temperatures. In contrast, impervious area in the local riparian zone was not clearly associated with stream temperature patterns, including time >20°C, maximum temperatures during warm months, and

temperature sensitivity (**Figure 7B,D,F**); results were similarly unclear for impervious area at the watershed scale.

Two pairs of headwater sites were compared to evaluate downstream changes in land cover and stream temperatures; in both pairs, the downstream site had more riparian canopy than the upstream site. In Upper Wissahickon, moving downstream from USUG2 (13% riparian canopy) to USUG (97% riparian canopy) was associated with cooling on 84% of warm-weather days (May–September): daily maximum temperatures on average were 0.8°C lower, and daily mean on average 0.3°C lower, at USUG compared to USUG2 (**Figure 8**). Notably, during the warmest month of the year (July), downstream cooling was observed at USUG on only 50% of days. In Naylors Run, DGP (upstream, 55% riparian canopy) had daily maximum temperatures on average 0.6°C warmer, and daily mean temperatures on average 0.15°C warmer, than NRP (downstream, 82% riparian canopy); DGP was warmer than NRP on 95% of warm-weather days. Temperature sensitivity did not change moving downstream from USUG2 to USUG or from DGP to NRP (**Table 2**).

3.2.2 Stream temperature surges during storms

Temperature surges (changes of ≥2°C in 30 min) were observed at all headwater sites with drainage areas ≤10.1 km^2 (**Figures 9, S11**), including two sites along Naylors Run (DGP and NRP) and one each in Upper Wissahickon (USUG2), Paper Mill Run (PMR), and Sandy Run (USAb2). In Naylors Run, surges occurred on 32% of rainy days, while surges were less frequent in the other subwatersheds (**Table 3**); surge magnitudes were also greater in Naylors Run compared to the other sites (**Figure S12**). Surge frequency and magnitude correlated well with percent impervious area at the watershed scale, but not at the local scale (**Figure 10**). Across all sites, surge duration ranged 0.5–4.25 hours with a median of 2.75 hours (**Table S1**). Surges at Naylors Run sites DGP and NRP had similar magnitudes (*t*-test $p = 0.16$), but duration was significantly higher ($p = 0.002$) at the downstream site, NRP (median 3.1 hours), compared to DGP upstream (median 2 hours). Days with stream temperature surges did not stand out in terms of total precipitation or precipitation intensity; many days with similar regional precipitation characteristics did not have a surge (**Figures S11,S13**).

4 DISCUSSION

4.1 WWTPs are a strong driver of downstream temperature patterns

As expected, WWTPs contributed significant warming to downstream sites along Wissahickon Creek and Sandy Run. WWTPs' influence on observed downstream changes in stream temperatures was more significant than the impacts of varying stream size or riparian shading detected across these sites. At a local scale (1–3 km downstream), WWTPs affected mean, minimum, and maximum temperatures on daily, weekly, and annual time scales (**Figures 2, S7**); they weakened the relationship between stream temperature and air temperature (**Table 2, Figure S8**); and they affected how much time sites spent at the warmest temperatures (**Figure 3**). In urban settings, WWTP impact on stream temperature cannot be overlooked (Hamdhani et al., 2020).

The effect of each WWTP on downstream temperatures depended on effluent temperature and discharge compared to the stream. Abington WWTP discharges into the smallest stream (USAb2 baseflow <0.01 m^3/s), while Ambler WWTP discharges into a larger stream that includes effluent from Upper Gwynedd WWTP. The combination of warm effluent, high effluent discharge (**Figure S2**), and low baseflow upstream caused Abington WWTP to contribute more substantial stream temperature increases and temperature sensitivity decreases (**Figures 4, S8**) than the other WWTPs. In contrast, the combination of relatively cool effluent and high baseflow upstream caused Ambler WWTP to have a more modest impact on stream thermal regimes.

We predicted that WWTP impacts on stream temperature would be local, affecting downstream sites up to 3 km from the WWTP. During spring and fall, WWTP warming impacts were local as predicted, recovering to temperatures similar to upstream or the Wissahickon Mouth by 3 km downstream (**Figure 5**). However, during winter months, elevated temperatures downstream of Upper Gwynedd WWTP persisted between 3 and 9.7 km downstream. Downstream of Abington and Ambler WWTPs, winter and summer stream temperature changes persisted at least 7.8 and 12.1 km downstream, respectively, and recovered before reaching the Mouth, 11.3 km farther downstream. These results demonstrate that during both winter and summer, WWTPs can influence stream temperatures more than 10 km downstream, a more persistent impact than we hypothesized.

The potential for WWTP to impact stream health varied throughout the year. As predicted, the greatest magnitude of warming occurred in winter months. Macroinvertebrates, which experience year-round disturbance from environmental factors including stream temperature (Helms, Schoonover, & Feminella, 2009), may be negatively impacted by winter stream warming. During the warmest weeks of the year, Abington and Ambler WWTPs caused up to 1.5°C of downstream cooling (**Figure 4**). As trout are particularly sensitive to high temperatures during summer and fall (Kovach et al., 2016), cooling during the warmest time of year may be a net benefit for these fish.

WWTPs were not the only factor to contribute to warming along Wissahickon Creek. Downstream warming with increasing stream size was observed along 32 km of Wissahickon Creek, though the rate (0.015°C/km) was low compared to many rivers, which exhibit stronger warming or variable downstream thermal patterns (Fullerton et al., 2015). The low rate of observed downstream warming may be a result of urbanized (artificially warm) headwaters (Rice et al., 2011) or the relatively short stream length incorporated in this study. Others (e.g., Arora et al., 2018) have documented seasonal variability in downstream thermal patterns, including summertime downstream warming and wintertime downstream cooling, as was observed in this study (**Figure S6**). The lack of correlation between mean annual stream temperature and distance downstream or drainage area in Wissahickon Creek suggests that local factors (WWTPs and shading) may partially obscure the impact of stream size on temperatures. However, sites with larger drainage areas had lower temperature ranges (**Figure 2B**), presumably due to thermal buffering of larger streamflow volumes. Moving downstream, minimum temperatures increased faster than maximum temperatures decreased; nighttime stream cooling may depend more directly on buffering capacity and stream size, while daytime maximum temperatures are also sensitive to shading.

Temperature sensitivity did not show strong downstream trends outside of WWTP influence; sensitivities at headwater sites (0.76, 0.78) fell into a similar range as the most downstream sites (0.73, 0.79). Smaller-order streams are expected to have lower temperature sensitivity, with values <0.5 indicating significant groundwater input (Arora et al., 2018; Chang & Psaris, 2013).

The impacts of urbanization on temperature sensitivity are less clear: Urbanization can increase temperature sensitivity due to decreased shading and disrupted groundwater connectivity (Erickson & Stefan, 2000; Kelleher et al., 2012) or decrease temperature sensitivity due to stormwater inputs or other artificial heat sources (Gu et al., 2015; Rice et al., 2011). In this study, WWTP effluent reduced temperature sensitivity at downstream sites, as expected. Upstream of WWTPs, temperature sensitivities were high for an urban setting, suggesting that decreased shading and/or disrupted groundwater connectivity may play a role. Quantifying groundwater inputs is beyond the scope of this study; stream temperatures that were cooler than air in July could indicate groundwater buffering, but high temperature sensitivity values at those sites suggests that buffering is not substantial (**Table 2**).

Data from this study suggest that warm-water fish living in Wissahickon Creek and Naylors Run do not see negative impacts due to sustained high temperatures. Warm-water fish, such as bass, have optimal growth at mean daily stream temperatures of ~22°C and start to see stress above ~27°C (Whitledge et al., 2006). Common macroinvertebrates have a wide range of thermal limits, some in the range of 20–25°C (Stewart et al., 2013), a range of temperatures often observed in this study (**Figure 3**). In the northeastern U.S., higher stream temperatures are a strong predictive factor for lower trout populations, with a low probability of finding trout where sustained summer stream temperatures exceed 20°C (DeWeber & Wagner, 2015); thus high summer temperatures throughout Wissahickon Creek (**Figure 3**) may stress stocked trout populations.

A comparison of the relative impact of multiple WWTPs in this study suggests that reducing WWTP effluent temperature, reducing WWTP effluent discharge, and/or having WWTPs discharge into larger streams will reduce their impact on downstream temperatures. While WWTPs contributed substantial warming much of the year that could harm stream ecosystems, two WWTPs caused cooling for up to 12 km downstream during the warmest weeks of the year (**Figure 5**). WWTPs may also benefit stream ecosystems by increasing baseflow discharge in urban settings that are disconnected from groundwater (Bischel et al., 2013).

4.2 Local riparian canopy, but not impervious surfaces, influenced headwater stream temperatures

We predicted that more local riparian canopy would lead to cooler temperatures, a result documented in other studies (Moore, Spittlehouse, & Story, 2005; Rice et al., 2011), and tested this hypothesis across six sites that ranged 13–97% local canopy cover. More local riparian canopy was associated with less time spent >20°C and lower warm-weather maximum temperatures (**Figure 7**), but no change in mean temperatures. The relationship between canopy and maximum temperatures weakened in October and disappeared in November (**Figure S10**), consistent with the timing of leaf fall in this region. More local riparian canopy was also associated with lower temperature sensitivities (**Figure 7E**), consistent with others' findings (Kelleher et al., 2012). Overall, large differences in local riparian canopy cover between urban subwatersheds resulted in more modest stream temperature differences than expected; urban sites with full riparian canopy may still have significantly warmer temperatures than forested sites (Somers et al., 2013). While well-vegetated riparian buffers in an otherwise urban setting may contribute to cooler stream temperatures, there is likely a limit to their impact.

When streams move from open to shaded reaches, significant cooling is often observed (Coats & Jackson, 2020), but not always (Nebgen & Herrman, 2019). Several factors, including watershed size and channel alteration, may affect a stream's thermal response to increased canopy moving downstream (O'Briain, Shephard, Matson, Gordon, & Kelly, 2020). As expected, downstream cooling was observed in this study between two pairs of headwater sites when the downstream site had more local riparian canopy. Furthermore, downstream cooling was more significant for the pair of sites with a bigger disparity in canopy cover (**Figure 8**). We observed lower mean and maximum temperatures at downstream, shadier sites, but maximum temperatures were more sensitive to shading than mean temperatures, similar to other studies (Groom, Dent, Madsen, & Fleuret, 2011; Johnson, 2004). Persistent WWTP influence prevented the evaluation of downstream canopy impacts at other pairs of sites.

Contrary to expectations, impervious area at the local or watershed scale was not directly correlated with time >20°C, warm-weather maximum temperatures, or temperature sensitivity in headwater streams (**Figure 7**). The six sites in this study fell within a narrow range of 30–47%

impervious surfaces in their watersheds (**Figure 6B**), far above minimum levels that have been found to affect stream ecology (Cuffney, Brightbill, May, & Waite, 2010), and differences within this range may be difficult to detect. The impact of urbanization on stream temperatures is not uniform (Booth et al., 2016). For example, Hassett et al. (2018) observed clear differences in stream temperature along an urban gradient of 47 watersheds but noted that the thermal regimes in the five most forested and five most urbanized watersheds overlapped so that some individual forested and urban watersheds were indistinguishable. Local factors, including infrastructure age and historical land cover (Parr et al., 2016), play an important role in determining an individual watershed's response to urbanization. Thus, stream temperature differences across a narrow urban gradient may be inconsistent and difficult to detect.

Factors contributing uncertainty to this land cover analysis included stream size variation, land cover configuration, groundwater discharge into streams, and impacts from farther upstream. The headwater sites varied in size, with baseflow varying from <0.01 m^3/s to 0.06 m^3/s. Variable, diffuse groundwater discharge among these sites is an independent factor that may affect stream temperature regimes and is not accounted for in this study. Land cover configuration, as distinct from land cover composition, also influences stream water quality (Ding et al., 2016; Liu et al., 2020). While the present study was limited to six sites, larger datasets may clarify land cover trends that are not apparent in smaller studies (Booth et al., 2014); incorporating additional sites, where possible, would reduce reliance on any one site and would allow fuller exploration of land cover–stream temperature relationships across different scales.

4.3 Watershed-scale impervious surfaces contributed to stream temperature surges only during some storms

As expected, subwatersheds with more impervious area had greater surge frequency and magnitude (**Figure 10, Table 3**). Local-scale impervious area did not correlate with surge frequency or magnitude, emphasizing that watershed-scale processes are important for determining urban stream response to storm events (Somers et al., 2013). These trends were largely driven by Naylors Run, which had more impervious area (47%) than the other subwatersheds (30–42%). This modest difference in impervious area was associated with major differences in surge frequency (32% compared to <10%) and surge magnitude (>1.3 SDs

compared to ≤0.8 SDs). It is likely that other characteristics of Naylors Run, in addition to its impervious area, contributed to its more frequent surges. Land cover configuration could play an effect; much of Naylors Run contains higher-density urban housing, while the other headwater sites are in comparatively lower-density suburban areas. As surges propagated downstream in Naylors Run from DGP to NRP, surge magnitude stayed constant but surge duration increased, similar to observations in other urban streams (Nelson & Palmer, 2007).

It is uncertain whether temperature surges had a meaningful impact on stream health. Surges were only observed at sites with relatively small drainage areas, emphasizing the disproportional impact that storms may have on small, first-order streams (Rice et al., 2011; Wu, Thompson, Kolka, Franz, & Stewart, 2013). While WWTP effluent may have buffered downstream sites from experiencing surges, the largest site upstream of a WWTP (USUG) had no recorded surges. Surges were a source of heat to headwater streams during some storms, although they did not elevate stream temperatures outside the range of temperatures observed on hot, clear days. In Sandy Run, Paper Mill Run, and Upper Wissahickon, the five days with the highest maximum temperatures had no rainfall and surge magnitudes were low. In Naylors Run, surges had higher magnitudes but still accounted for fewer than half of the top ten daily maximum temperatures at NRP and DGP. At all sites, surges were infrequent (1–9 surges/yr) and dissipated in a matter of hours (**Table 3**). Many studies report relationships between higher mean temperatures and reduced stream biota health or abundance at the scale of days, weeks, or months (Dallas & Ross-Gillespie, 2015; DeWeber & Wagner, 2015); however, it is unclear whether brief, infrequent pulses of heat affect stream biota.

In this study, 68–94% of rainy days did not cause a temperature surge. Days with surges did not differ from days without surges in terms of regional precipitation characteristics (**Figures S11, S13**), which suggests that surges are the result of finer-scale heterogeneity in precipitation rather than occurring exclusively during the largest storms. Croghan, Van Loon, Sadler, Bradley, and Hannah (2019) found that only the finest-scale precipitation data (5-min, 1-km^2 radar data) generated useful correlations with temperature surges, while regional data did not. In the absence of such high-resolution data, it is difficult to effectively characterize or predict an individual storm's impact on stream temperature (Berne, Delrieu, Creutin, & Obled, 2004).

Results from this study suggest that mitigation efforts focused on impervious area and storm flows will have minimal impact on stream temperature outside of some select storms. Many studies document water quality and stream ecology impacts of stormwater management practices (Zhang & Chui, 2019); few studies address their impact on stream temperatures, and fewer still focus on temperature patterns beyond surges during select storms (Cockerill, Anderson, Harris, & Straka, 2017). A recent study of 11 headwater streams found that stormwater control measures reduced surge magnitudes but had no mitigating impact on maximum temperatures (Fanelli et al., 2019). Furthermore, some stream restoration projects reduce riparian canopy, leading to higher stream temperatures (Sudduth, Hassett, Cada, & Bernhardt, 2011). It is critical that stream temperature impacts are considered as part of stormwater management and stream restoration efforts (Cockerill & Anderson, 2014).

5 CONCLUSIONS and IMPLICATIONS

Stream temperature is a critical water quality parameter that is not fully understood, particularly in urban systems. This study explored drivers contributing to stream temperature variability at 21 sites in the Philadelphia region. Downstream changes were evaluated along Wissahickon Creek and headwater sites in four urban subwatersheds were compared. A comprehensive set of temperature metrics were evaluated at each urban site. Each driver impacted some stream temperature metrics more than others, demonstrating the importance of considering temperature metrics that are appropriate to the driver or process in question.

WWTPs were the strongest driver of downstream temperature variability in this study, causing consistent downstream warming of 2–7°C during cooler months and affecting nearly all stream temperature metrics considered. WWTP effluent temperature controlled local (1–3 km downstream) temperatures year-round. Notably, effluent from two WWTPs reduced downstream summer temperatures, potentially benefitting stream health. Summer cooling and winter warming from WWTP effluent persisted up to 12 km downstream. WWTPs that discharged into larger streams, had cooler effluent, and/or had lower discharge had less harmful effects on stream temperatures, providing guidelines for mitigating WWTP impact.

Differences in land cover also affected stream temperatures. Comparing four urban subwatersheds, sites with more riparian canopy had cooler maximum temperatures by up to 1.5°C and spent less time at high temperatures, although mean temperatures were unaffected. Moving downstream from less to more riparian canopy, the effect was similar. Watershed-scale impervious area was associated with increased surge frequency and magnitude but did not affect temperature patterns outside of some storms. Surges occurred infrequently and briefly, so it is unclear how much they affect stream health; furthermore, most storms did not result in a surge. Maintaining or restoring riparian canopy in urban settings will have a larger impact on stream temperatures than stormwater management that treats impervious area. It is vital that stream temperature impacts are considered when planning stormwater management or stream restoration projects.

Mitigation efforts may be most impactful at urban headwater sites, which are particularly vulnerable to stream temperature disruptions. In this study, headwater sites had greater daily temperature fluctuations, were more likely to experience surges during storms, and were the most impacted by WWTP effluent and land cover. Mitigating rising stream temperatures in urban settings is a site-specific challenge: just as stream responses to urbanization vary, the management tools that most effectively reduce stream temperatures in urban settings may also vary.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are openly available in CUASHI HydroShare at https://doi.org/10.4211/hs.3f298a462cfb4e2c983abd085f0770b6.

ACKNOWLEDGMENTS

The William Penn Foundation (Grant # 39-18) provided funding for this study. Dr. Sarah Ledford, Chelsea Kanaley, and Dr. Rob Ryan helped collect the temperature data used. Chris Kobol helped with ArcGIS land cover analyses. This manuscript was much improved thanks to thorough, thoughtful feedback from four reviewers.

Table 1. Site descriptions, drainage areas, and temperature data availability.

Site ID	Subwatershed	Location description	Distance downstream (km) †	Drainage area (km^2)	Temperature data	Data gap
USUG2	Upper Wissahickon	2.4 km upstream of Upper Gwynedd WWTP	5.8	10.1	05/01/17–04/30/18	–
USUG		0.1 km upstream of Upper Gwynedd WWTP	8.1	15.6	05/01/17–04/30/18	–
DSUG1		1.4 km downstream of Upper Gwynedd WWTP	9.6	17.4	05/01/17–04/30/18	–
DSUG2		2.1 km downstream of Upper Gwynedd WWTP	10.2	18.1	05/01/17–11/28/17	–
DSUG3		3.0 km downstream of Upper Gwynedd WWTP	11.1	22.7	05/01/17–04/30/18	–
USAmb	Middle Wissahickon	0.6 km upstream of Ambler WWTP	17.8	60.3	05/01/17–04/30/18	–
DSAmb1		0.7 km downstream of Ambler WWTP	19.1	67.1	05/01/17–04/30/18	–
DSAmb2		1.3 km downstream of Ambler WWTP	19.7	68.0	05/01/17–11/28/17	–
DSAmb3		1.8 km downstream of Ambler WWTP	20.2	69.7	05/01/17–04/30/18	–
FtWash	Lower Wissahickon	USGS gauge at Fort Washington	21.2	102.5	05/01/17–04/30/18	–
USPMR		0.6 km upstream of Paper Mill Run confluence	26.4	126.4	05/01/17–04/24/18	07/08/17–08/10/17
Mouth		USGS gauge at Wissahickon Creek mouth	37.7	165.0	05/01/17–04/30/18	–
USAb2	Sandy Run	1.1 km upstream of Abington WWTP	1.8	6.1	05/01/17–04/30/18	–
DSAb1		1.1 km downstream of Abington WWTP	4.1	10.5	05/01/17–04/30/18	–
DSAb2		2.0 km downstream of Abington WWTP	4.9	11.7	05/01/17–11/23/17	–
DSAb3		3.2 km downstream of Abington WWTP	6.1	12.6	05/01/17–04/30/18	12/23/17–01/22/18
SR-USM		1.7 km upstream of the mouth of Sandy Run	8.2	30.5	05/01/17–04/24/18	–
SR-Mouth		0.9 km upstream of the mouth of Sandy Run	9.1	31.9	05/01/17–04/24/18	–
PMR	Paper Mill Run	0.3 km upstream of the mouth of Paper Mill Run	3.3	5.9	05/01/17–04/24/18	07/08/17–08/10/17
DGP	Naylors Run	In Drexel Gardens Park	3.0	7.2	08/01/17–07/31/18	–
NRP		In Naylors Run Park	4.3	8.4	08/01/17–07/31/18	–

† Distance downstream is measured from channel initiation as defined by the USGS NHD dataset (USGS, 2019)

Table 2. Relationships between air and stream temperatures and time spent at high temperatures at all sites and WWTPs. Solid horizontal lines delineate subwatersheds; shading indicates WWTPs.

Subwatershed	Site ID	Time >20°C (days)	Time >25°C (days)	Temp. sensitivity (slope)	r^2	July $T_{air} - T_{stream}$	January $T_{air} - T_{stream}$
Upper Wissahickon	USUG2	85.3	7.0	0.76	0.95	1.2	–3.0
	USUG	81.2	4.3	0.75	0.97	1.3	–3.2
	UG WWTP	168.9	33.9	0.44	0.80	–0.5	–13.6
	DSUG1	115.9	12.7	0.69	0.94	0.0	–6.6
	DSUG3	105.2	9.0	0.75	0.97	0.1	–4.6
Middle Wissahickon	USAmb	98.6	10.6	0.79	0.96	0.0	–3.1
	Amb WWTP	103.1	0.1	0.51	0.87	2.3	–9.4
	DSAmb1	99.4	5.1	0.73	0.96	0.4	–4.5
	DSAmb3	98.5	5.1	0.74	0.97	0.5	–4.2
Lower Wissahickon	FtWash	98.4	4.4	0.74	0.97	0.5	–4.1
	USPMR	92.2	1.7	0.73	0.97	0.8	–4.2
	Mouth	100.2	8.8	0.79	0.96	–0.1	–3.3
Sandy Run	USAb2	93.3	8.2	0.78	0.96	0.5	–3.2
	Ab WWTP	149.1	0.01	0.39	0.79	0.9	–13.1
	DSAb1	109.6	0.5	0.52	0.91	1.4	–9.7
	DSAb3	94.8	7.9	0.68	0.98	0.9	–8.5
	SR-USM	94.0	1.5	0.70	0.98	1.2	–4.5
	SR-Mouth	94.1	1.5	0.70	0.97	1.1	–4.7
Paper Mill Run	PMR	100.3	17.9	0.81	0.95	–0.6	–2.9
Naylors Run	DGP	85.8	2.2	0.71	0.98	1.7	–3.6
	NRP	82.4	0.8	0.71	0.98	1.9	–3.7

Table 3. Surge characteristics and impervious area at each headwater site.

		IMPERVIOUS AREA		SURGE CHARACTERISTICS		
Subwatershed	**Site**	**Watershed (%)**	**Local riparian (%)**	**Surge frequency † (%)**	**Average surge magnitude ‡ (SDs)**	**Average surge duration § (hr)**
Upper Wissahickon	USUG2	36.8	0.2	8.3	0.7	0.5
Upper Wissahickon	USUG	34.1	0.1	0	–	–
Paper Mill Run	PMR	29.8	5.0	5.9	0.1	3
Sandy Run	USAb2	41.8	29.0	7.4	0.8	2.5
Naylors Run	DGP	47.1	27.5	32	1.6	2
Naylors Run	NRP	47.3	3.5	32	2.0	2.9

† Percent of rainy days with a surge (May–September)
‡ Standard deviations (SDs) above the mean maximum temperature
§ Time for stream temperatures to return to within 2°C of pre-surge values (Nelson & Palmer, 2007)

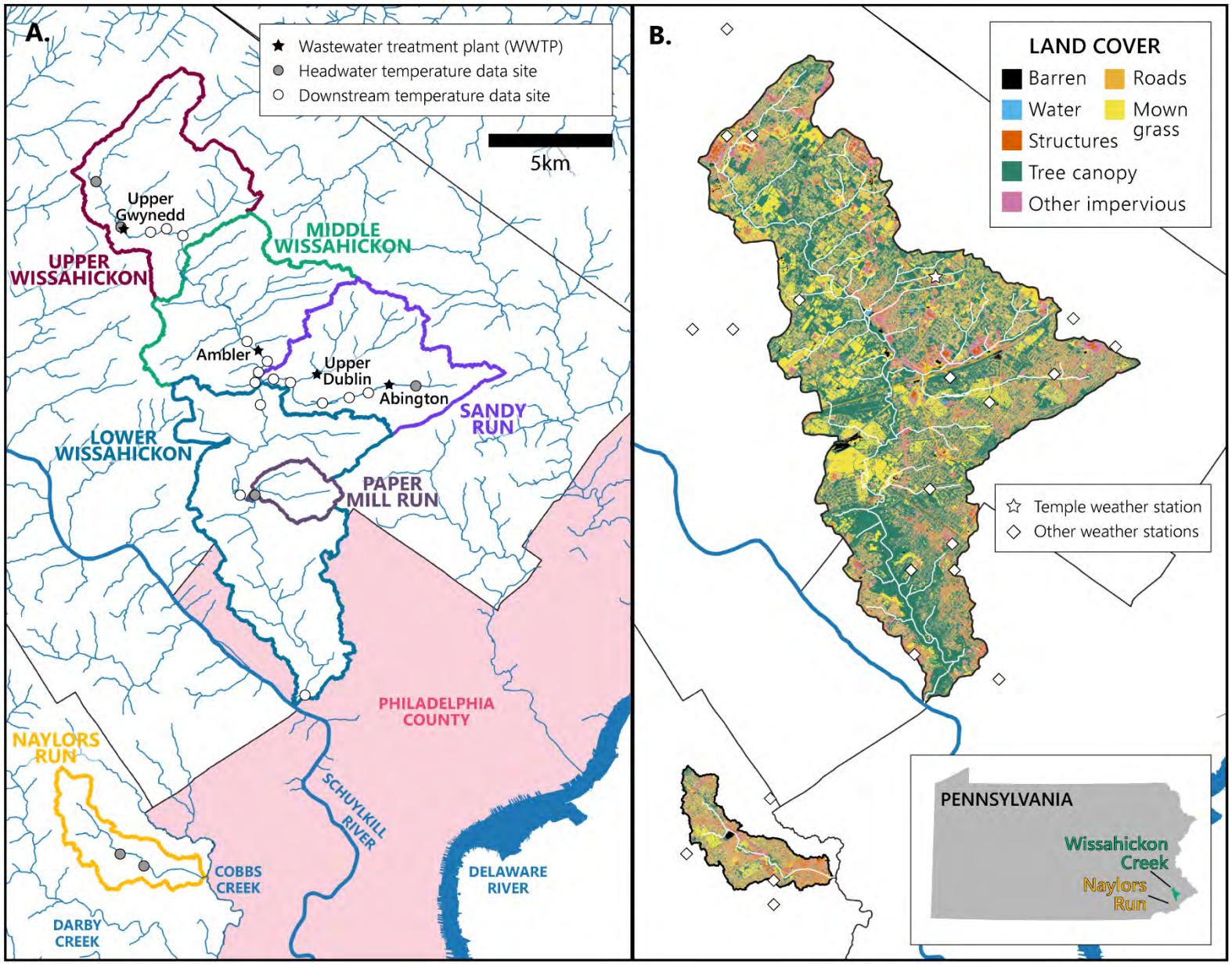
A.
Wastewater treatment plant (WWTP)
Headwater temperature data site
Downstream temperature data site
5km
Upper Gwynedd
UPPER WISSAHICKON
MIDDLE WISSAHICKON
Ambler
Upper Dublin
Abington
SANDY RUN
LOWER WISSAHICKON
PAPER MILL RUN
PHILADELPHIA COUNTY
NAYLORS RUN
SCHUYLKILL RIVER
COBBS CREEK
DELAWARE RIVER
DARBY CREEK
B.
LAND COVER
Barren
Roads
Water
Mown grass
Structures
Tree canopy
Other impervious
Temple weather station
Other weather stations
PENNSYLVANIA
Wissahickon Creek
Naylors Run

Figure 1 (*previous page)*. This study focuses on 21 in-stream sites across an urban region. A. Area map of Philadelphia with Naylors Run and Wissahickon Creek watersheds; Wissahickon Creek is divided into subwatersheds. Temperature monitoring sites (*circles*) and wastewater treatment plants (WWTPs, *black stars*) are shown. B. Land cover (UVM Spatial Analysis Lab, 2016) and all weather stations used in this study. *Inset* shows location within Pennsylvania, USA.

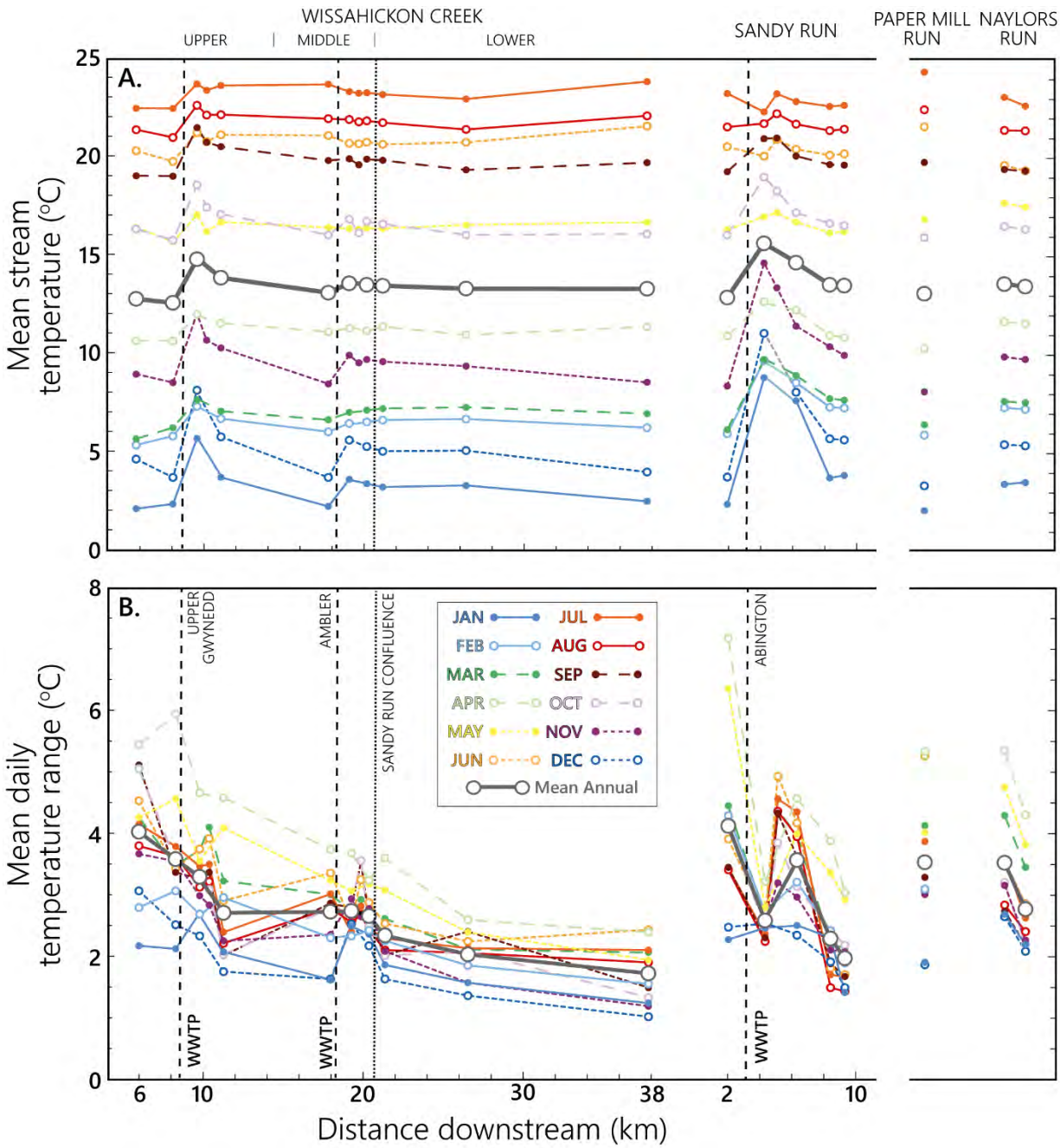

Figure 2. Downstream trends in monthly mean stream temperatures and daily temperature ranges show the local-scale impact of WWTPs year-round. Mean monthly (*colors*) and annual (*gray*) stream temperatures (A) and daily temperature ranges (B) are shown in downstream order.

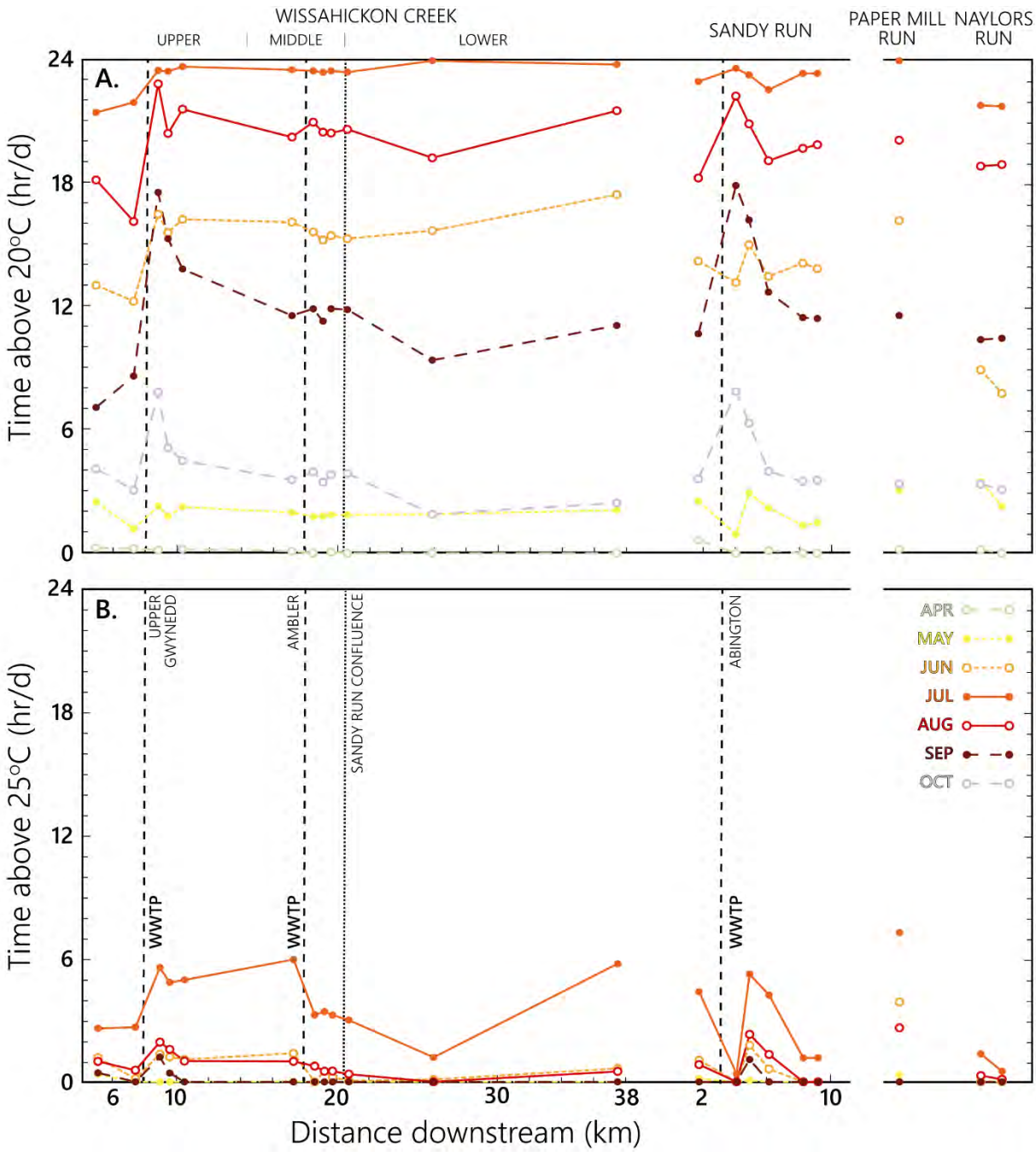

Figure 3. Sites ~1 km downstream of Upper Gwynedd and Abington WWTPs spent more time above 20°C than upstream sites, while WWTP impact on time above 25°C varied. Monthly time spent above 20°C (A) and 25°C (B) is shown in downstream order for April through October. Stream temperatures rarely exceeded 20°C during the months not shown.

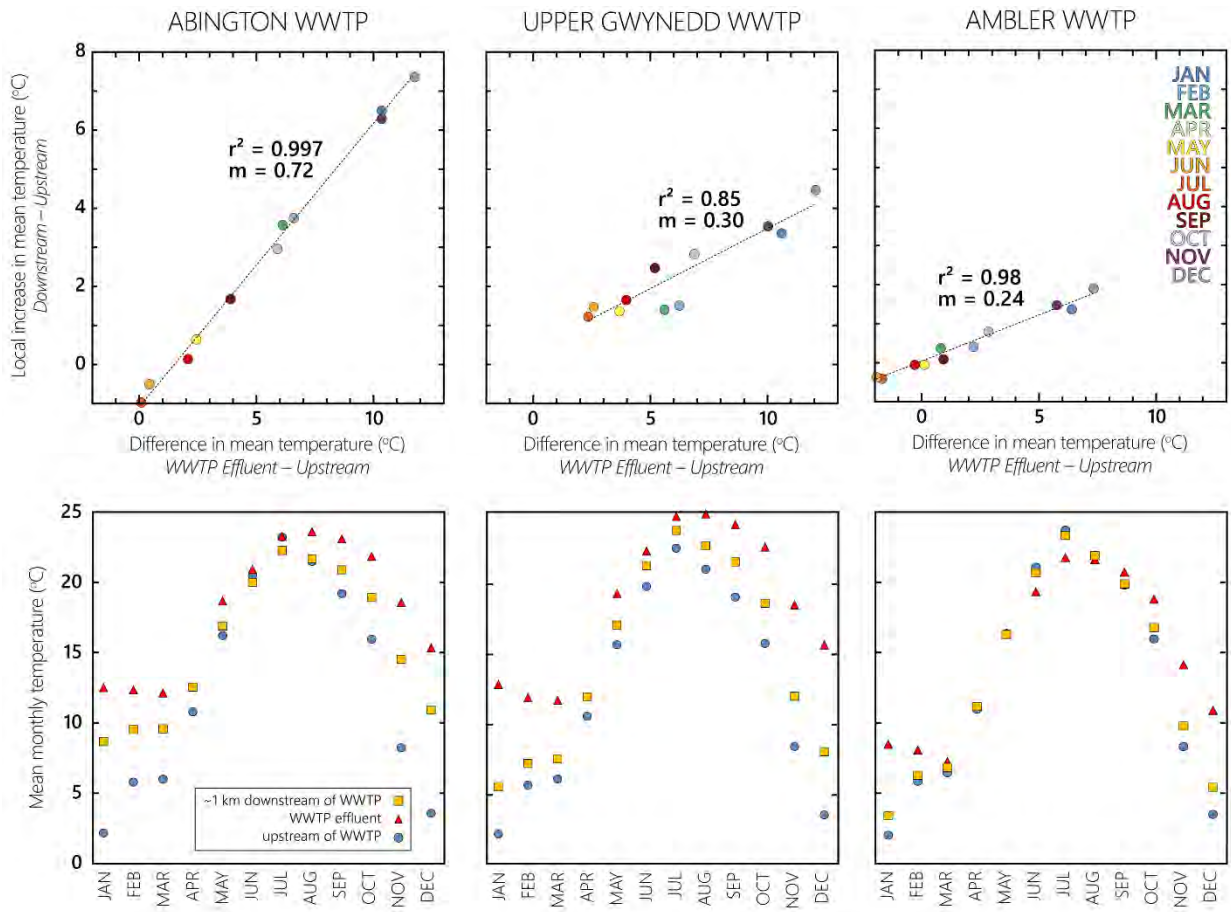

Figure 4. Three WWTPs had varying impact on stream temperatures ~1 km downstream. *Top row.* Increase in mean stream temperature ~1 km downstream relative to upstream is plotted against the difference between WWTP effluent and upstream temperature for Abington, Upper Gwynedd, and Ambler WWTPs. *Bottom row.* Monthly average downstream (*yellow squares*), upstream (*blue circles*), and effluent (*red triangles*) temperatures at Abington, Upper Gwynedd, and Ambler WWTPs.

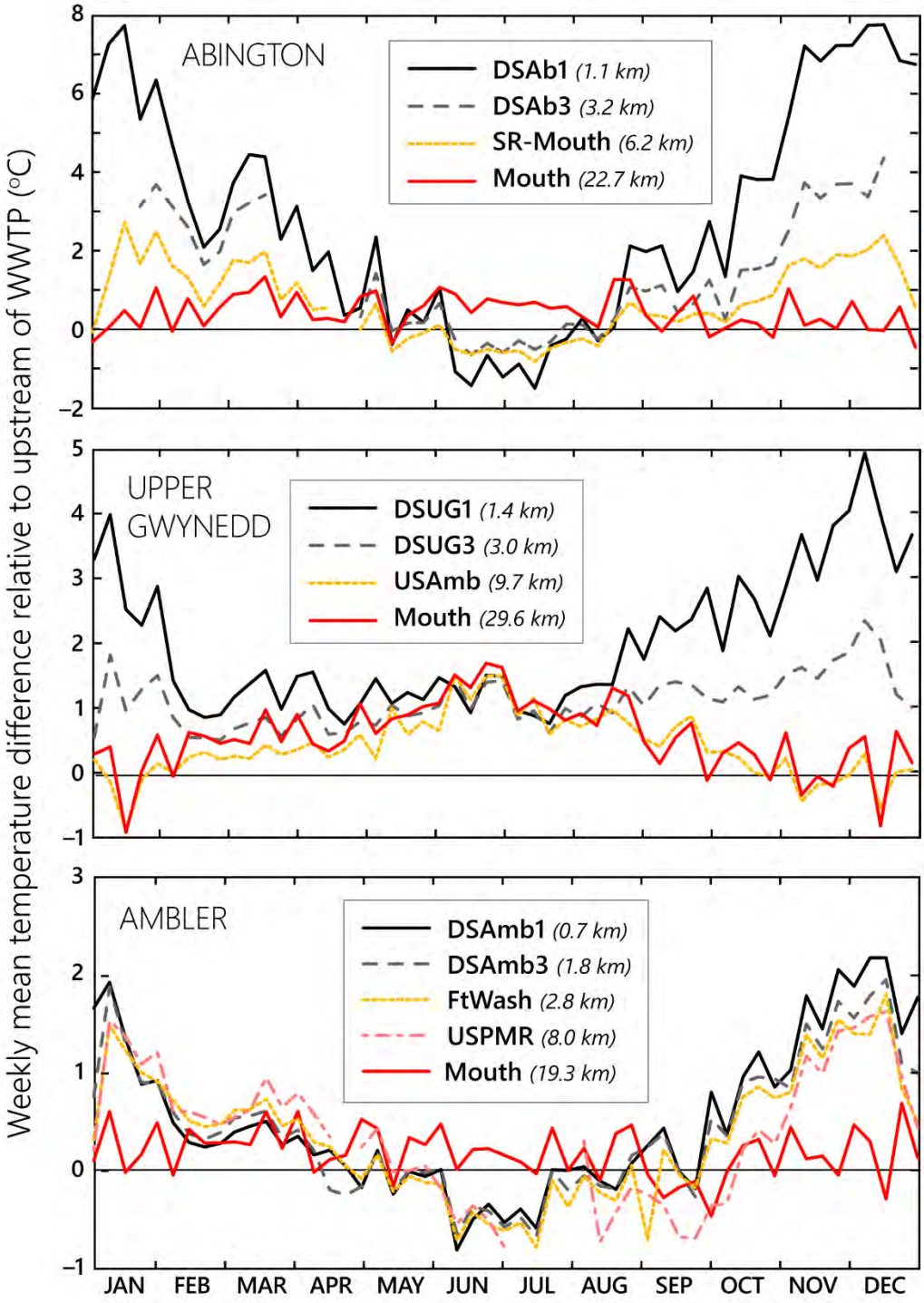

Figure 5. Effluent from Abington and Ambler WWTPs influenced temperatures up to 12 km downstream during summer and winter. Weekly mean temperature differences relative to upstream of the WWTPs are shown for sites downstream of Abington (*top*), Upper Gwynedd (*middle*), and Ambler (*bottom*) WWTPs. Distances downstream from the WWTP are indicated in parentheses. The influence of Abington and Ambler WWTPs extends beyond Fort Washington (FtWash), where Sandy Run flows into Wissahickon Creek.

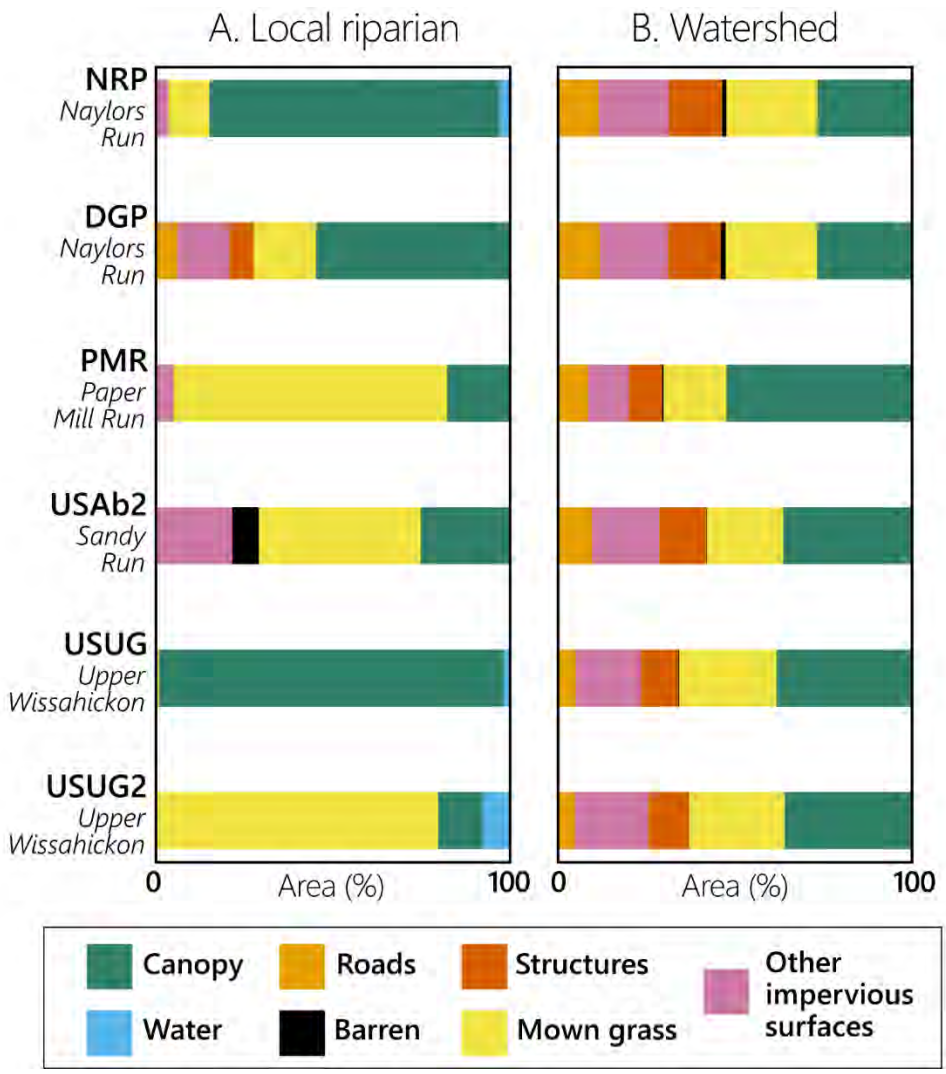

Figure 6. Headwater sites' land cover varied at the local riparian scale but was similar at the watershed scale. Bars are subdivided by color to show the percent area within each land cover category at the local riparian (A) and watershed (B) scales.

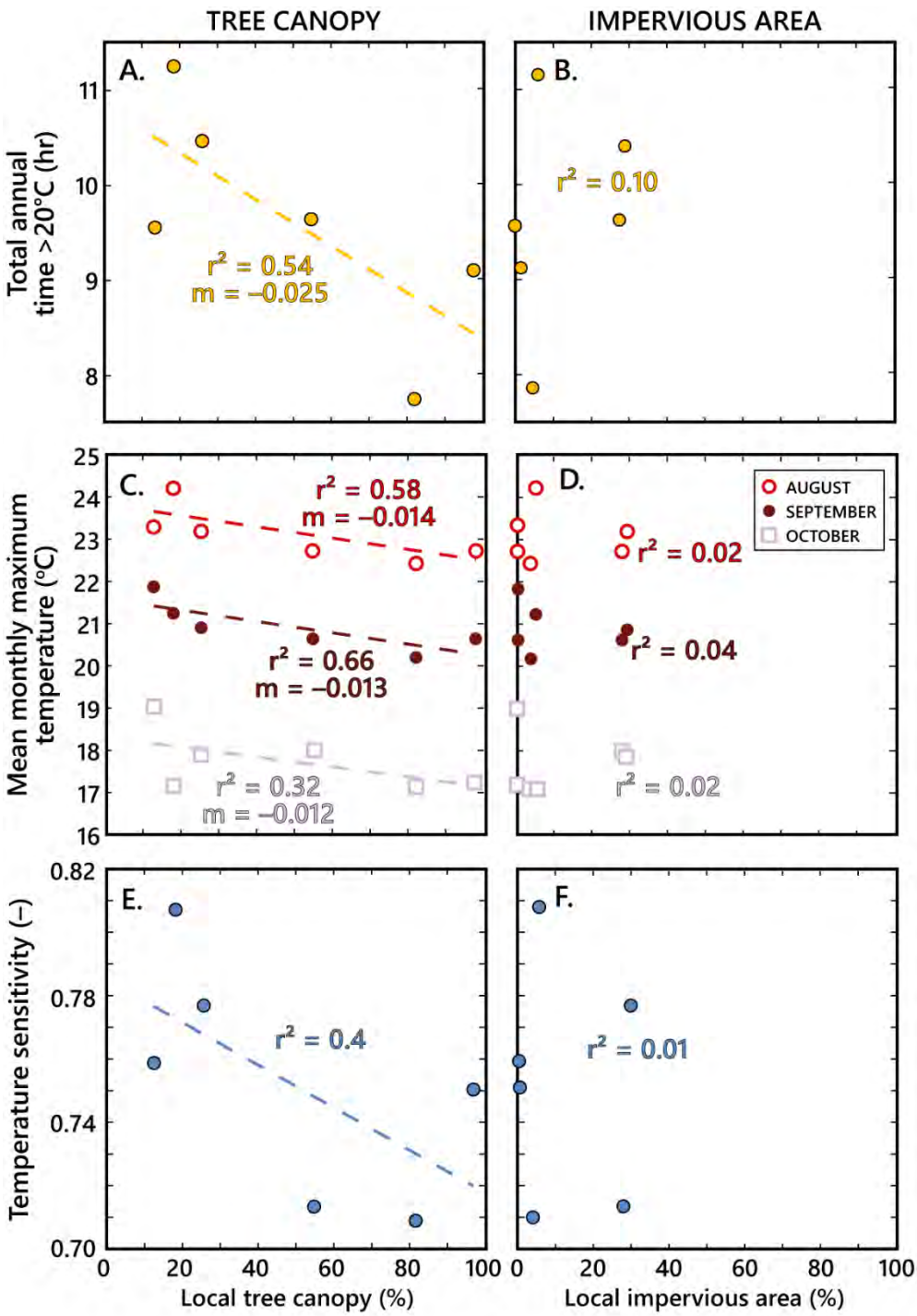

Figure 7. Local riparian tree canopy affected warm-weather stream temperatures, while impervious area did not have a clear impact. Total annual time >20°C (*top row*), mean monthly maximum temperature during August–October (*middle row*), and annual temperature sensitivity (*bottom row*) plotted against percent local riparian tree canopy (*left column*) or impervious area (*right column*).

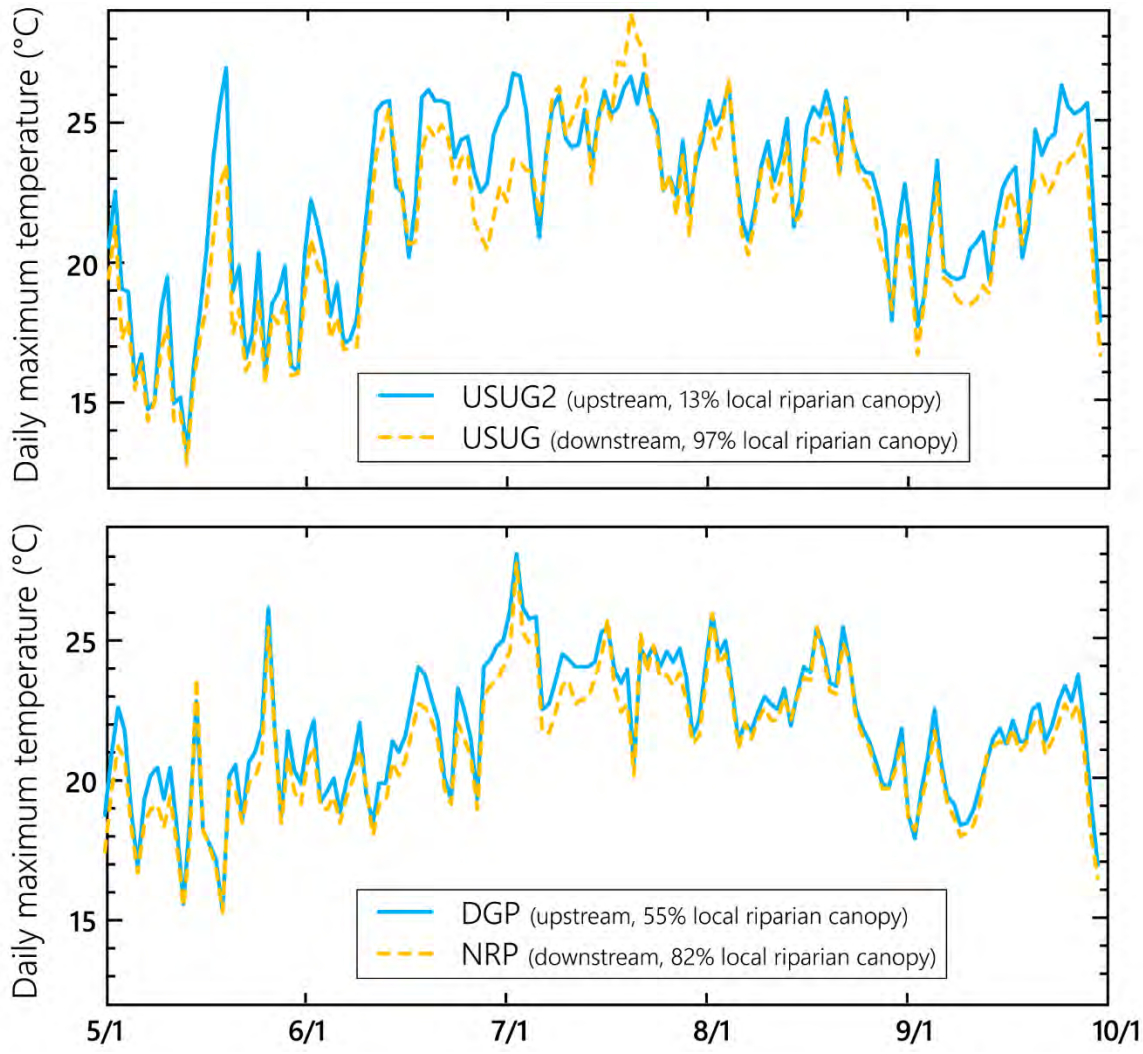

Figure 8. Increasing riparian canopy modestly cooled headwater stream temperatures in Upper Wissahickon and Naylors Run. Daily maximum temperatures are shown for pairs of headwater sites in Upper Wissahickon (*top*) and Naylors Run (*bottom*) from May through September. Upstream sites (*solid blue lines*) had less riparian canopy and (usually) higher maximum temperatures than downstream sites (*dotted yellow lines*).

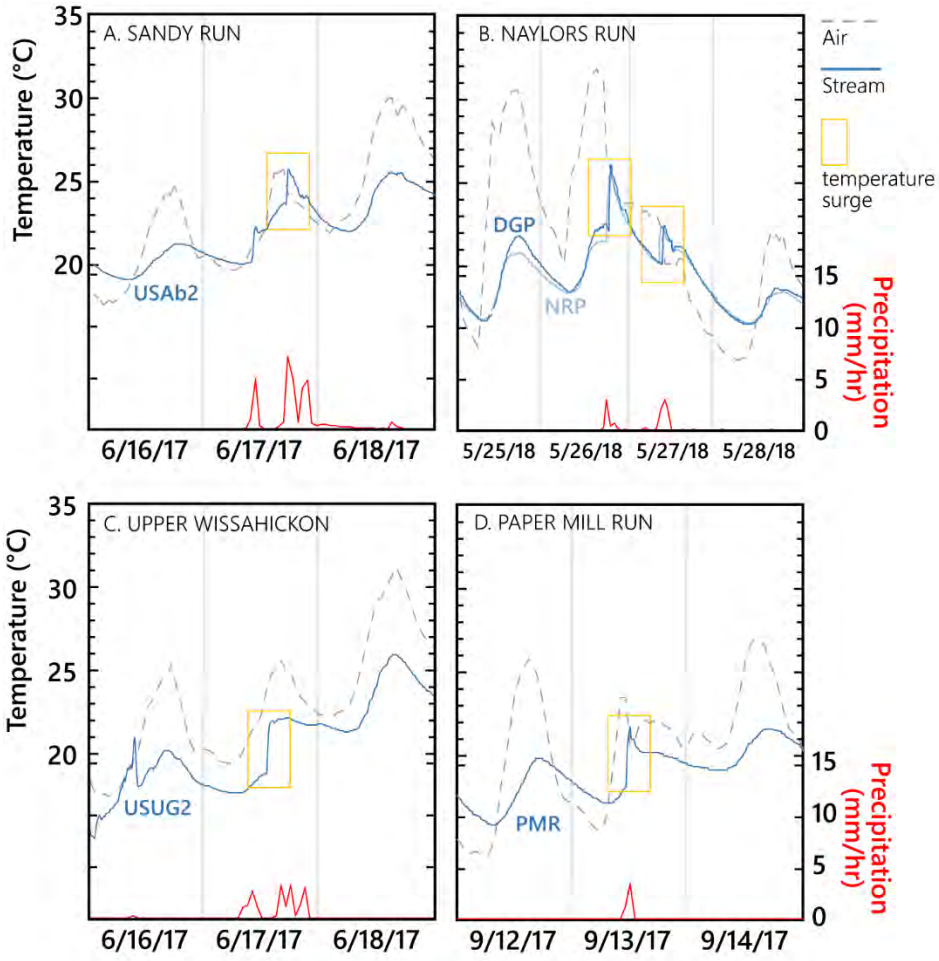

Figure 9. Stream temperature surges were observed during storms at headwater sites in four subwatersheds. Several days (divided by *vertical solid gray lines*) of hourly air temperature (*dashed gray line*), 15-min stream temperature (*solid blue line*), and hourly precipitation (*solid red line*) are shown for Sandy Run (A), Naylors Run (B), Upper Wissahickon (C), and Paper Mill Run (D). Stream temperature surges are indicated with yellow boxes. See **Figure S11** for similar plots of all observed surges not included here.

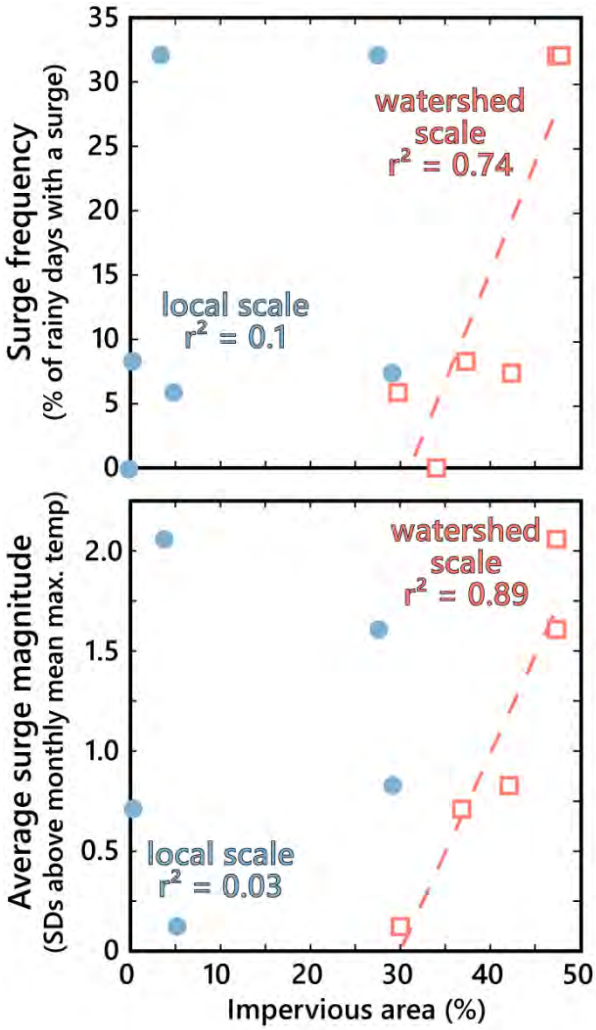

Figure 10. Watershed-scale impervious area was associated with greater surge frequency and magnitude, while local-scale impervious area was not. Surge frequency (*top*) and average surge magnitude (*bottom*) at each site are plotted against percent impervious area at watershed (*open red squares*) and local riparian (*solid blue circles*) scales.

REFERENCES

Acuña, V., Wolf, A., Uehlinger, U., & Tockner, K. (2008). Temperature dependence of stream benthic respiration in an Alpine river network under global warming. *Freshwater Biology*, *53*(10), 2076–2088. https://doi.org/10.1111/j.1365-2427.2008.02028.x

Anderson, W. P., Anderson, J. L., Thaxton, C. S., & Babyak, C. M. (2010). Changes in stream temperatures in response to restoration of groundwater discharge and solar heating in a culverted, urban stream. *Journal of Hydrology*, *393*(3–4), 309–320. https://doi.org/10.1016/j.jhydrol.2010.08.030

Arora, R., Toffolon, M., Tockner, K., & Venohr, M. (2018). Thermal discontinuities along a lowland river: The importance of urban areas and lakes. *Journal of Hydrology*, *564*, 811–823. https://doi.org/10.1016/j.jhydrol.2018.05.066

Audia, E., Woller-Skar, M. M., & Locher, A. (2020). Crowd-sourced data link land use and soil moisture to temperature and relative humidity in southwest Michigan (USA). *Theoretical and Applied Climatology*. https://doi.org/10.1007/s00704-020-03429-4

Beaufort, A., Moatar, F., Sauquet, E., Loicq, P., & Hannah, D. M. (2020). Influence of landscape and hydrological factors on stream–air temperature relationships at regional scale. *Hydrological Processes*, *34*(3), 583–597. https://doi.org/10.1002/hyp.13608

Beitinger, T. L., Bennett, W. A., & McCauley, R. W. (2004). Temperature tolerances of North American freshwater fishes exposed to dynamic changes in temperature. *Journal of Fish Biology*, *58*, 237–275. https://doi.org/:10.1111/j.1095-8649.2004.00453.x

Berne, A., Delrieu, G., Creutin, J.-D., & Obled, C. (2004). Temporal and spatial resolution of rainfall measurements required for urban hydrology. *Journal of Hydrology*, *299*(3–4), 166–179. https://doi.org/10.1016/j.jhydrol.2004.08.002

Bischel, H. N., Lawrence, J. E., Halaburka, B. J., Plumlee, M. H., Bawazir, A. S., King, J. P., McCray, J. E., Resh, V. H., & Luthy, R. G. (2013). Renewing urban streams with recycled water for streamflow augmentation: Hydrologic, water quality, and ecosystem services management. *Environmental Engineering Science*, *30*(8), 455–479. https://doi.org/10.1089/ees.2012.0201

Booth, D. B., Kraseski, K. A., & Jackson, C. R. (2014). Local-scale and watershed-scale determinants of summertime urban stream temperatures. *Hydrological Processes*, *28*(4), 2427–2438. https://doi.org/10.1002/hyp.9810

Booth, D. B., Roy, A. H., Smith, B., & Capps, K. A. (2016). Global perspectives on the urban stream syndrome. *Freshwater Science*, *35*(1), 412–420. https://doi.org/10.1086/684940

Brewitt, K. S., & Danner, E. M. (2014). Spatio-temporal temperature variation influences juvenile steelhead (Oncorhynchus mykiss) use of thermal refuges. *Ecosphere*, *5*(7), 92. https://doi.org/10.1890/ES14-00036.1

Caissie, D. (2006). The thermal regime of rivers: A review. *Freshwater Biology*, *51*(8), 1389–1406. https://doi.org/10.1111/j.1365-2427.2006.01597.x

Chang, H., & Psaris, M. (2013). Local landscape predictors of maximum stream temperature and thermal sensitivity in the Columbia River Basin, USA. *Science of The Total Environment*, *461–462*, 587–600. https://doi.org/10.1016/j.scitotenv.2013.05.033

Coats, W. A., & Jackson, C. R. (2020). Riparian canopy openings on mountain streams: Landscape controls upon temperature increases within openings and cooling downstream. *Hydrological Processes*, *34*(8), 1966–1980. https://doi.org/10.1002/hyp.13706

Cockerill, K., & Anderson, W. P. (2014). Creating false images: Stream restoration in an urban setting. *Journal of the American Water Resources Association*, *50*(2), 468–482. https://doi.org/10.1111/jawr.12131

Cockerill, K., Anderson, W. P., Harris, F. C., & Straka, K. (2017). Hot, salty water: A confluence of issues in managing stormwater runoff for urban streams. *Journal of the American Water Resources Association*, *53*(3), 707–724. https://doi.org/10.1111/1752-1688.12528

Croghan, D., Van Loon, A. F., Sadler, J. P., Bradley, C., & Hannah, D. M. (2019). Prediction of river temperature surges is dependent on precipitation method. *Hydrological Processes*, *33*(1), 144–159. https://doi.org/10.1002/hyp.13317

Cuffney, T. F., Brightbill, R. A., May, J. T., & Waite, I. R. (2010). Responses of benthic macroinvertebrates to environmental changes associated with urbanization in nine metropolitan areas. *Ecological Applications*, *20*(5), 19.

Dallas, H., & Ross-Gillespie, V. (2015). Review: Sublethal effects of temperature on freshwater organisms, with special reference to aquatic insects. *Water SA*, *41*(5), 712–726. https://doi.org/10.4314/wsa.v41i5.15

DCVA. (2002). *Darby Creek Watershed River Conservation Plan* (p. 339). Darby Creek Valley Association. http://archive.phillywatersheds.org/doc/Darby_Cobbs_RCP.pdf

DeWeber, J. T., & Wagner, T. (2015). Predicting brook trout occurrence in stream reaches throughout their native range in the eastern United States. *Transactions of the American Fisheries Society*, *144*(1), 11–24. https://doi.org/10.1080/00028487.2014.963256

Ding, J., Jiang, Y., Liu, Q., Hou, Z., Liao, J., Fu, L., & Peng, Q. (2016). Influences of the land use pattern on water quality in low-order streams of the Dongjiang River basin, China: A multi-scale analysis. *Science of The Total Environment*, *551–552*, 205–216. https://doi.org/10.1016/j.scitotenv.2016.01.162

EPA. (2018). *Water Quality Standards for Pennsylvania* (Chapter 93; p. 162). United States Environmental Protection Agency. https://www.epa.gov/sites/production/files/2014-12/documents/pawqs-chapter93.pdf

Erickson, T. R., & Stefan, H. G. (2000). Linear air/water temperature correlations for streams during open water periods. *Journal of Hydrologic Engineering*, *5*(3), 317–321. https://doi.org/10.1061/(ASCE)1084-0699(2000)5:3(317)

Fanelli, R. M., Prestegaard, K. L., & Palmer, M. A. (2019). Urban legacies: Aquatic stressors and low aquatic biodiversity persist despite implementation of regenerative stormwater

conveyance systems. *Freshwater Science*, *38*(4), 818–833. https://doi.org/10.1086/706072

Fullerton, A. H., Torgersen, C. E., Lawler, J. J., Faux, R. N., Steel, E. A., Beechie, T. J., Ebersole, J. L., & Leibowitz, S. G. (2015). Rethinking the longitudinal stream temperature paradigm: Region-wide comparison of thermal infrared imagery reveals unexpected complexity of river temperatures. *Hydrological Processes*, *29*(22), 4719–4737. https://doi.org/10.1002/hyp.10506

Groom, J. D., Dent, L., Madsen, L. J., & Fleuret, J. (2011). Response of western Oregon (USA) stream temperatures to contemporary forest management. *Forest Ecology and Management*, *262*(8), 1618–1629. https://doi.org/10.1016/j.foreco.2011.07.012

Gu, C., Anderson, W. P., Colby, J. D., & Coffey, C. L. (2015). Air-stream temperature correlation in forested and urban headwater streams in the Southern Appalachians. *Hydrological Processes*, *29*(6), 1110–1118. https://doi.org/10.1002/hyp.10225

Hale, R. L., Scoggins, M., Smucker, N. J., & Suchy, A. (2016). Effects of climate on the expression of the urban stream syndrome. *Freshwater Science*, *35*(1), 421–428. https://doi.org/10.1086/684594

Hamdhani, H., Eppehimer, D. E., & Bogan, M. T. (2020). Release of treated effluent into streams: A global review of ecological impacts with a consideration of its potential use for environmental flows. *Freshwater Biology*, *00*, 1–14. https://doi.org/10.1111/fwb.13519

Hammerberg, K., Brousse, O., Martilli, A., & Mahdavi, A. (2018). Implications of employing detailed urban canopy parameters for mesoscale climate modelling: A comparison between WUDAPT and GIS databases over Vienna, Austria. *International Journal of Climatology*, *38*, e1241–e1257. https://doi.org/10.1002/joc.5447

Hassett, B. A., Sudduth, E. B., Somers, K. A., Urban, D. L., Violin, C. R., Wang, S.-Y., Wright, J. P., Cory, R. M., & Bernhardt, E. S. (2018). Pulling apart the urbanization axis: Patterns of physiochemical degradation and biological response across stream ecosystems. *Freshwater Science*, *37*(3), 653–672. https://doi.org/10.1086/699387

Helms, B. S., Schoonover, J. E., & Feminella, J. W. (2009). Seasonal variability of landuse impacts on macroinvertebrate assemblages in streams of western Georgia, USA. *Journal of the North American Benthological Society*, *28*(4), 991–1006. https://doi.org/10.1899/08-162.1

Herb, W. R., Janke, B., Mohseni, O., & Stefan, H. G. (2008). Thermal pollution of streams by runoff from paved surfaces. *Hydrological Processes*, *22*(7), 987–999. https://doi.org/10.1002/hyp.6986

Jabiol, J., Gossiaux, A., Lecerf, A., Rota, T., Guérold, F., Danger, M., Poupin, P., Gilbert, F., & Chauvet, E. (2020). Variable temperature effects between heterotrophic stream processes and organisms. *Freshwater Biology*, *00*, 1–12. https://doi.org/10.1111/fwb.13520

Johnson, S. L. (2004). Factors influencing stream temperatures in small streams: Substrate effects and a shading experiment. *Canadian Journal of Fisheries and Aquatic Sciences*, *61*(6), 913–923. https://doi.org/10.1139/f04-040

Kelleher, C., Wagener, T., Gooseff, M., McGlynn, B., McGuire, K., & Marshall, L. (2012). Investigating controls on the thermal sensitivity of Pennsylvania streams. *Hydrological Processes*, *26*(5), 771–785. https://doi.org/10.1002/hyp.8186

Kinouchi, T., Yagi, H., & Miyamoto, M. (2007). Increase in stream temperature related to anthropogenic heat input from urban wastewater. *Journal of Hydrology*, *335*(1–2), 78–88. https://doi.org/10.1016/j.jhydrol.2006.11.002

Kovach, R. P., Muhlfeld, C. C., Al-Chokhachy, R., Dunham, J. B., Letcher, B. H., & Kershner, J. L. (2016). Impacts of climatic variation on trout: A global synthesis and path forward. *Reviews in Fish Biology and Fisheries*, *26*(2), 135–151. https://doi.org/10.1007/s11160-015-9414-x

Liu, J., Liu, X., Wang, Y., Li, Y., Jiang, Y., Fu, Y., & Wu, J. (2020). Landscape composition or configuration: Which contributes more to catchment hydrological flows and variations? *Landscape Ecology*, *35*(7), 1531–1551. https://doi.org/10.1007/s10980-020-01035-3

Moore, D. R., Spittlehouse, D. L., & Story, A. (2005). Riparian microclimate and stream temperature response to forest harvesting: A review. *Journal of the American Water Resources Association*, *41*(4), 813–834. https://doi.org/10.1111/j.1752-1688.2005.tb03772.x

Nebgen, E. L., & Herrman, K. S. (2019). Effects of shading on stream ecosystem metabolism and water temperature in an agriculturally influenced stream in central Wisconsin, USA. *Ecological Engineering*, *126*, 16–24. https://doi.org/10.1016/j.ecoleng.2018.10.023

Nelson, K. C., & Palmer, M. A. (2007). Stream temperature surges under urbanization and climate change: Data, models, and responses. *Journal of the American Water Resources Association*, *43*(2), 440–452. https://doi.org/10.1111/j.1752-1688.2007.00034.x

O'Briain, R., Shephard, S., Matson, R., Gordon, P., & Kelly, F. L. (2020). The efficacy of riparian tree cover as a climate change adaptation tool is affected by hydromorphological alterations. *Hydrological Processes*, *34*, 2433–2449. https://doi.org/10.1002/hyp.13739

PADEP. (2018). *Pennsylvania Integrated Water Quality Monitoring and Assessment Report*. Pennsylvania Department of Environmental Protection. https://www.depgis.state.pa.us/2018_integrated_report/index.html.

Pagliaro, M. D., & Knouft, J. H. (2020). Differential effects of the urban heat island on thermal responses of freshwater fishes from unmanaged and managed systems. *Science of The Total Environment*, *723*, 138084. https://doi.org/10.1016/j.scitotenv.2020.138084

Parr, T. B., Smucker, N. J., Bentsen, C. N., & Neale, M. W. (2016). Potential roles of past, present, and future urbanization characteristics in producing varied stream responses. *Freshwater Science*, *35*(1), 436–443. https://doi.org/10.1086/685030

PWD. (2007). *Wissahickon Creek Watershed Comprehensive Characterization Report*. Philadelphia Water Department. http://archive.phillywatersheds.org/doc/Wissahickon_CCR.pdf

Rice, J. S., Anderson, Jr., W. P., & Thaxton, C. S. (2011). Urbanization influences on stream temperature behavior within low-discharge headwater streams. *Hydrological Research Letters*, *5*, 27–31. https://doi.org/10.3178/hrl.5.27

Roy, A. H., Faust, C. L., Freeman, M. C., & Meyer, J. L. (2005). Reach-scale effects of riparian forest cover on urban stream ecosystems. *Canadian Journal of Fisheries and Aquatic Sciences*, *62*, 2312–2329. https://doi.org/10.1139/f05-135

Singh, S., & Chang, H. (2014). Effects of land cover change on water quality in urban streams at two spatial scales. *International Journal of Geospatial and Environmental Research*, *1*(1), 23.

Somers, K. A., Bernhardt, E. S., Grace, J. B., Hassett, B. A., Sudduth, E. B., Wang, S., & Urban, D. L. (2013). Streams in the urban heat island: Spatial and temporal variability in temperature. *Freshwater Science*, *32*(1), 309–326. https://doi.org/10.1899/12-046.1

Sponseller, R. A., Benfield, E. F., & Valett, H. M. (2001). Relationships between land use, spatial scale and stream macroinvertebrate communities. *Freshwater Biology*, *46*(10), 1409–1424. https://doi.org/10.1046/j.1365-2427.2001.00758.x

Steel, E. A., Beechie, T. J., Torgersen, C. E., & Fullerton, A. H. (2017). Envisioning, quantifying, and managing thermal regimes on river networks. *BioScience*, *67*(6), 506–522. https://doi.org/10.1093/biosci/bix047

Stewart, B. A., Close, P. G., Cook, P. A., & Davies, P. M. (2013). Upper thermal tolerances of key taxonomic groups of stream invertebrates. *Hydrobiologia*, *718*(1), 131–140. https://doi.org/10.1007/s10750-013-1611-9

Sudduth, E. B., Hassett, B. A., Cada, P., & Bernhardt, E. S. (2011). Testing the Field of Dreams Hypothesis: Functional responses to urbanization and restoration in stream ecosystems. *Ecological Applications*, *21*(6), 1972–1988. https://doi.org/10.1890/10-0653.1

Sun, N., Yearsley, J., Voisin, N., & Lettenmaier, D. P. (2015). A spatially distributed model for the assessment of land use impacts on stream temperature in small urban watersheds. *Hydrological Processes*, *29*(10), 2331–2345. https://doi.org/10.1002/hyp.10363

Sweeney, B. W., & Newbold, J. D. (2014). Streamside forest buffer width needed to protect stream water quality, habitat, and organisms: A literature review. *Journal of the American Water Resources Association*, *50*(3), 560–584. https://doi.org/10.1111/jawr.12203

USGS. (2019). *National Hydrography Dataset*. United States Geological Survey. https://www.usgs.gov/core-science-systems/ngp/national-hydrography/access-national-hydrography-products

UVM Spatial Analysis Lab. (2016). *High-Resolution Land Cover, Commonwealth of Pennsylvania, Chesapeake Bay Watershed and Delaware River Basin.* University of Vermont. https://www.pasda.psu.edu/uci/DataSummary.aspx?dataset=3193

Walsh, C. J., Roy, A. H., Feminella, J. W., Cottingham, P. D., Groffman, P. M., & Morgan, R. P. (2005). The urban stream syndrome: Current knowledge and the search for a cure. *Journal of the North American Benthological Society*, *24*(3), 706–723. https://doi.org/10.1899/04-028.1

Webb, B. W., Hannah, D. M., Moore, R. D., Brown, L. E., & Nobilis, F. (2008). Recent advances in stream and river temperature research. *Hydrological Processes*, *22*(7), 902–918. https://doi.org/10.1002/hyp.6994

Wehrly, K. E., Wiley, M. J., & Seelbach, P. W. (2003). Classifying regional variation in thermal regime based on stream fish community patterns. *Transactions of the American Fisheries Society*, *132*, 18–38. https://doi.org/10.1577/1548-8659(2003)132<0018:CRVITR>2.0.CO;2

Wenger, S. J., Roy, A. H., Jackson, C. R., Bernhardt, E. S., Carter, T. L., Filoso, S., Gibson, C. A., Hession, W. C., Kaushal, S. S., Martí, E., Meyer, J. L., Palmer, M. A., Paul, M. J., Purcell, A. H., Ramírez, A., Rosemond, A. D., Schofield, K. A., Sudduth, E. B., & Walsh, C. J. (2009). Twenty-six key research questions in urban stream ecology: An assessment of the state of the science. *Journal of the North American Benthological Society*, *28*(4), 1080–1098. https://doi.org/10.1899/08-186.1

Whitledge, G. W., Rabeni, C. F., Annis, G., & Sowa, S. P. (2006). Riparian shading and groundwater enhance growth potential for smallmouth bass in Ozark streams. *Ecological Applications*, *16*(4), 1461–1473. https://doi.org/10.1890/1051-0761(2006)016[1461:RSAGEG]2.0.CO;2

Wu, J. Y., Thompson, J. R., Kolka, R. K., Franz, K. J., & Stewart, T. W. (2013). Using the Storm Water Management Model to predict urban headwater stream hydrological response to climate and land cover change. *Hydrology and Earth System Sciences*, *17*(12), 4743–4758. https://doi.org/10.5194/hess-17-4743-2013

Zhang, K., & Chui, T. F. M. (2019). Linking hydrological and bioecological benefits of green infrastructures across spatial scales – A literature review. *Science of The Total Environment*, *646*, 1219–1231. https://doi.org/10.1016/j.scitotenv.2018.07.355