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RESEARCH ARTICLE

Influence of temperature, precipitation, and cloud cover on diel dissolved oxygen ranges among headwater streams with variable watershed size and land use attributes

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Abstract

Dissolved oxygen (DO) concentrations in streams are driven by multiple, interacting biotic and abiotic processes. While DO variability largely reflect cyclic patterns of respiration and photosynthesis coupled to diel cycles, physical processes such as floods that disturb biofilms and variation in temperature disrupt such cycles. In urban settings, DO cycles are typically greatly altered by elevating nutrient concentrations and reducing light-shielding riparian vegetation. We analyzed diel variations in DO from sensors distributed throughout six headwater streams to quantify (1) diel DO range patterns among watersheds of varying size and urbanization intensity, (2) the conditions that lead to abrupt declines in diel DO ranges, and (3) the amount of time needed for diel DO ranges to recover post-disturbance. In very small streams, disruptions to diel DO ranges appear to occur following severe fluctuations in atmospheric temperatures while precipitation events were primarily related to diel DO disruptions in larger streams. Precipitation events ≥ 1.5 cm over a 1-day period or ≥ 2.5 over a 2-day period consistently resulted in abrupt depressions of diel DO variations. While we primarily analyzed abiotic variables, we acknowledge that photosynthetic activity producing DO was also an important variable as shown by an analysis of how cloud cover influenced DO variations. Recovery of diel DO ranges to pre-disturbance conditions varied among sites, with the smallest watershed site reaching 50% pre-disturbance ranges in an average of 4.5 days and the largest and most urban sites reaching the same threshold over an average of 2.1 days. Urban sites typically exhibited greater diel DO ranges but did not exhibit lower precipitation thresholds for resetting diel DO ranges. DO ranges were more likely to be disrupted by precipitation events when water temperatures were cooler, which suggested different impacts of hydrologic controls on DO variations across seasons. Our findings suggest that streams consistently possess discharge thresholds that, if exceeded, lead to abrupt declines in the magnitude of the diel change in DO, but urban streams may show greater variation in diel DO concentrations with implications for fish habitat, redox-sensitive microbial processes, and contaminant transport and transformation.

Keywords Dissolved oxygen · Diel variability · Urban streams

Introduction

Dissolved oxygen (DO) concentrations represent an ecologically crucial and highly dynamic environmental parameter in flowing waters (Bernhardt et al. 2018). Most aquatic macro-organisms require DO to fulfill cellular metabolism and consequently diminish DO while photosynthetic organisms generate DO during periods of daylight, resulting in ecosystem-wide diel cycling of concentrations (Odum 1956, Mullholland et al. 2005, Wilding et al. 2012). Although the relative balance of both processes varies significantly among settings, streams are more likely to be net heterotrophic compared to other aquatic ecosystems (Hoellein et al. 2013), possibly due to high rates of respiration during

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daylight hours (Hotchkiss and Hall 2014). However, DO in streams can also vary directly or indirectly with other highly dynamic variables, such as daily and seasonal temperature changes that alter DO solubility (Williams and Boorman 2012; Rajwa-Kuligiewicz et al. 2015), flood events that scour benthic periphyton and therefore reduce both photosynthesis and respiration (O'Connor et al. 2012), and eutrophication that elevates DO production via enhancing photosynthesis (e.g., Dodds 2007; Kaushal et al. 2014a, b; Smith and Kaushal 2015; Reisinger et al. 2017). Physical reaeration of DO at the surface also varies as a function of temporally dynamic variables such as water DO deficit, turbulence, velocity, and depth (Gromiec 1989; Moog and Jirka 1999; Izagirre et al. 2007). Because diel variations in DO concentrations are simultaneously impacted by a highly heterogeneous suite of physical, chemical, and biological variables, identifying dominant drivers of diel DO ranges within a system is difficult.

Hydrologic variability can play an important role during precipitation events, and the role of precipitation events in impacting diel variations/oscillations may differ seasonally by temperature. Many organisms exhibit sensitivity to low DO episodes (Davis 1975), the frequency of which are expected to increase with warming temperatures (Ficklin et al. 2013). Previous work has shown that there have been long-term rising stream and river temperatures in regions of the United States (Kaushal et al. 2010) and understanding how temperature and hydrologic variability influence diel DO oscillations is critical for managing urban water quality (Blaszczak et al. 2018). Additionally, changes in DO concentrations can significantly affect important biogeochemical processes such as nitrification, denitrification, and phosphorus release from sediments (Kemp and Dodds 2002, Veraart and Scheffer 2011, Rosamond et al. 2012, Duan et al. 2016). The role of DO variations can also be important in evaluating the effects of ecosystem degradation and restoration on the nitrogen uptake functions of urban streams (Pennino et al. 2014; Reisinger et al. 2019).

Anthropogenic environmental stressors that affect fundamental physicochemical properties of streams may also alter how DO concentrations temporally vary, particularly in urban settings. Whole-ecosystem metabolism in streams, defined as the balance of respiration and photosynthesis, represents a dominant driver of DO concentrations that can be radically altered by anthropogenic activity. Excess nutrient inputs from urban infrastructure can promote primary productivity in biofilms during diel daylight periods and cause heightened diel DO concentrations, especially when temperatures are high (Halliday et al. 2015; Hasenmueller et al. 2017). Reduced canopy cover, characteristic of many streams altered by land use change, can also significantly enhance DO production by photosynthesis (Hoellein et al. 2013). However, in some settings urbanization may suppress diel DO fluctuations due to elevated scouring event frequencies that remove biofilm and consequently diminish both respiration and photosynthesis rates (Blaszczak et al. 2019; Wang et al. 2003). Urban streams with low slope gradients may be acutely vulnerable to episodic low DO conditions due to the limited potential for physical reaeration (Wilding et al. 2012; Blaszczak et al. 2019). Changes to riparian canopy cover and associated effects on stream temperatures can also significantly alter DO concentrations by shading periphyton and moderating photosynthesis, with consequences on nitrification and denitrification (Sudduth et al. 2011; Hornbach et al. 2015).

Data from long-term or high frequency monitoring using autonomous sensors can help identify the most important agents regulating DO concentrations. Recent advances in autonomous sensor technology have lowered the cost and increased the ease of conducting such monitoring, which has enabled investigations of DO dynamics in streams that were previously not possible (Rode et al. 2016). For example, high-frequency records derived from automated sensors have helped to highlight complications in methods to assess DO in streams for water quality directive compliance (Skeffington et al. 2015) and assess the efficacy of water purchases for sustaining conditions suitable for aquatic life in streams subject to withdrawals (Null et al. 2017). Although advances in sensor technology have furthered our understanding of many chemical parameters in streams, the highly temporally dynamic nature of DO renders automated sensors particularly useful for such inquiries (Loperfido et al. 2009).

We employed automated DO sensors in six temperate, headwater streams varying in watershed size and urban land cover to investigate hydrologic and thermal factors influencing variability in diel DO ranges. Our intent was to quantify the frequency and intensity of low DO concentration events in urbanizing streams, thus we opted to broaden the spatial scale and number of sites for our study at the expense of more intensive monitoring at a single site that may comprehensively quantify ecosystem-scale metabolic rates. Early in our monitoring effort, we noted instances of abrupt declines in diel DO ranges following long periods of relatively large fluctuations between day and night, resulting in declines in diel range magnitude of up to 4 mg L^{-1}. We focused on predicting instances of such phenomena using precipitation and temperature data with implications for water quality, nutrient flux, and aquatic habitat and ecosystem integrity. Potential relationships between DO diel ranges and metrics associated with precipitation and temperature were explored at multiple temporal scales and hypothesized that the amplitude of diel variations of DO would increase with watershed size. Additionally, we hypothesized that DO diel ranges in small streams would be more regulated by temperature than larger and more urban streams, where DO would be more strongly influenced by hydrologic disturbances.

Methods

Study sites

Our study was conducted in six perennial headwater streams with watersheds within and north of the city of Pittsburgh, Pennsylvania, USA (Fig. 1). All sites were located within the suburban fringe of Pittsburgh, although each site retains some secondary growth deciduous forest cover in the watersheds (Table 1). Sites consisted of small first-order to third order headwater streams with perennial flow. Each watershed constitutes part of the upper Ohio River and Mississippi River drainages. The climate in the study region is temperate humid with a mean annual temperature of 11.1 °C and 88.5 cm of precipitation (Arguez et al. 2010).

Sites spanned gradients of watershed urbanization, ranging from 0 to 85% urban (0–42.7% impervious surface cover) based on land use classification defined by the 2011 National Land Cover Database (Homer et al. 2015), which reports land cover on a 30 m pixel resolution. Multiple efforts to restore ecological functions to the most urbanized site, Nine Mile Run, have been implemented in attempt to

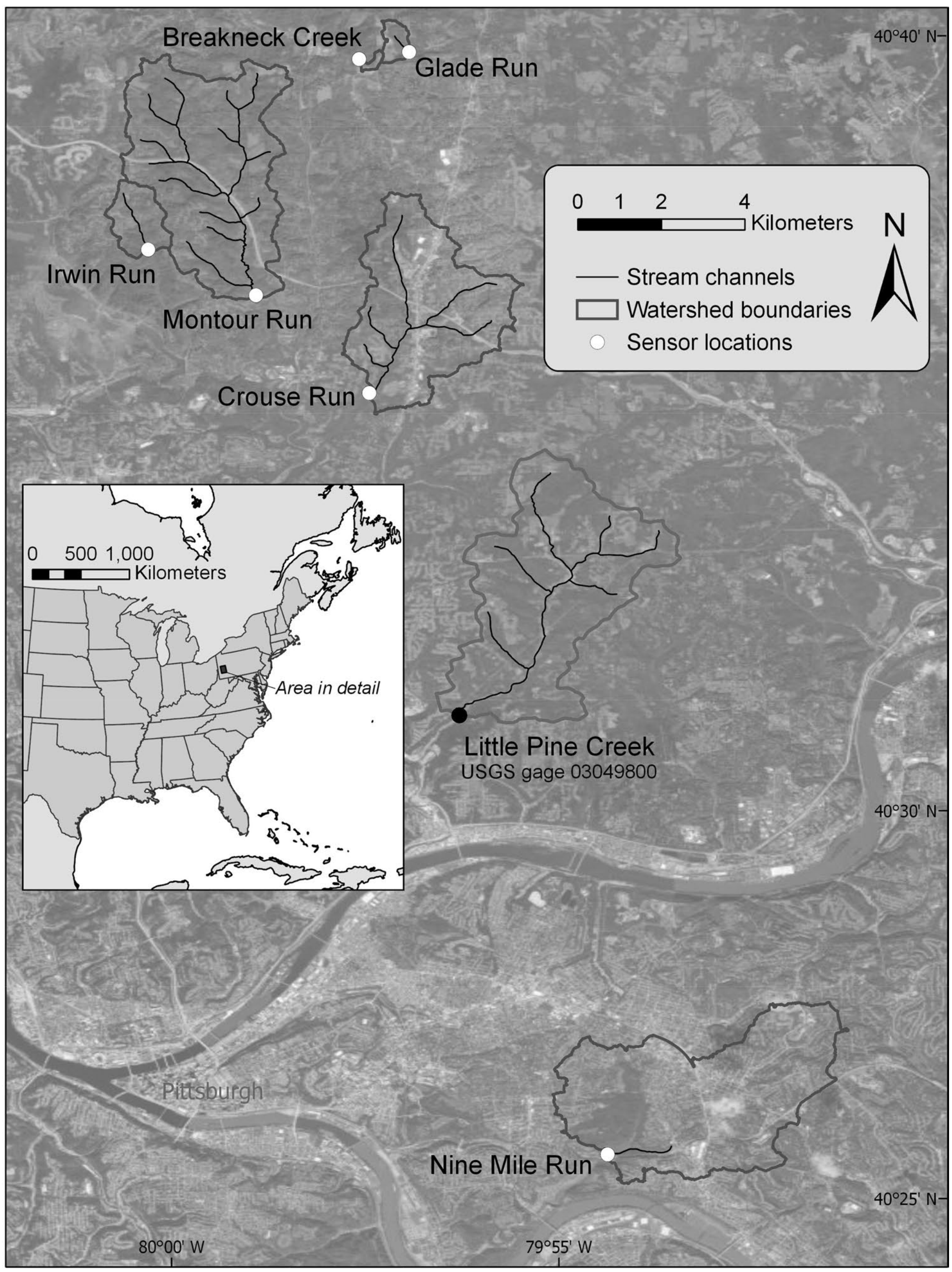

Fig. 1 Map of the study sites depicting watershed boundaries and channels with perennial flow. Channels do not include stream reaches buried by urban development, as is evident in the Nine Mile Run site, where the stream channel is daylighted only in the lowermost reach. Inset shows location of study region relative to eastern North America

Table 1 Study site watershed attributes

Site	Watershed size (km^2)	Strahler order	Slope (m m^{-1} × 10^{-2})	% Urbanization	% Pasture and row crop	% Forest	% Impervious surface cover
Breakneck Creek	0.1	1	4.1	48.6	19.5	31.9	12.1
Glade Run	0.7	1	2.1	2.4	49.2	48.3	0.0
Irwin Run	1.5	1	1.5	43.7	1.1	55.2	6.4
Crouse Run	9.7	2	2.0	71.3	0.6	28.0	21.5
Montour Run	12.5	3	0.6	29.2	6.2	64.2	4.5
Nine Mile Run	13.8	3*	0.1	84.7	0.1	15.2	42.7

*Estimate based on buried stream reaches (Bain et al. 2014)

reverse damage caused by near-complete channel burial (see Fig. 1) and outfalls of combined sewerage overflows (CSOs; Bain et al. 2014). Urban land cover at other sites consists primarily of roadways, parking lots, and rooftops without direct CSO outfalls, although septic tanks and leaking sewerage lines likely contribute nutrients and other contaminants to the streams. Most land cover in all study site watersheds not classified as urban was deciduous broadleaf or mixed coniferous/deciduous forest.

Sensor deployment and maintenance

We deployed miniDOT sensors (Precision Measurement Engineering, Inc., Vista, California, United States of America) at each site to monitor DO concentrations and water temperature. The sensor uses an optode to measure DO concentrations via fluorescence. Each unit was fit with copper anti-fouling plates and deployed in polyvinyl chloride pipes with perforations to allow flow across the sensing location. Deployment sites were selected in slow-moving habitat, either runs or pools, and the sensing elements were positioned about 10 cm deep during baseflow. Sensors were set to record DO and temperature every 15 min. Following the initial deployment, sensors were visited with variable frequency but no more than a period of 28 days to retrieve data and check on the status of the device. Sensors were occasionally buried by sediment or exposed to the atmosphere during visits, in which case the sensor housing was relocated to an area ≤ 5 m from the original position.

Data retrieved from the DO sensors were quality-controlled prior to all analyses. Periods where exposure to the atmosphere or burial by sediment were clearly distinguishable by abnormally high or low and consistent values, respectively. All such data points were removed from the record before subsequent inquiries. Active sensor deployment periods ranged from 393 (Nine Mile Run) to 620 (Glade Run) days between 2016 and 2018. Although the deployment period varied among sites, each site deployment period overlapped with cold and warm seasons during deployment. The manufacturer recommends recalibration after 500,000 readings, which was not exceeded during the study period for any site. However, the calibration status for all sensors was verified approximately midway through the study period by placing the sensors in water that was artificially saturated with oxygen via forced aeration. We retrieved a total of 278,476 DO readings from sensors among all sites over the study period. Temporal coverage was poorest in Nine Mile Run (393 days out data successfully retrieved out of 826 possible days) due to the tendency of high flows and associated sediment pulses that disrupted sensor performance at that site.

DO saturation (%) was also quantified for each quality-controlled reading. We applied the DO.saturation function from package rMR (Moulton 2018) using DO concentration, water temperature, and atmospheric pressure as arguments. Atmospheric pressure data applied in these estimates were from the National Oceanic Atmospheric Administration (NOAA) weather station at the Pittsburgh International Airport (NOAA 2020b), which collected observations hourly.

Additional variables exploring DO variation

Our goal was to determine how parameters associated with watershed size, weather, and seasonality affect diel patterns of DO in streams of the site network. Daily DO minima and maxima were calculated for each date with > 90 successfully recorded DO readings within a 24-h period to estimate diel ranges. We also determined if minima and maxima occurred during daylight or nighttime for each date and site by recording when each metric was observed using sunrise/sunset phenological data for Pittsburgh provided by the United States Naval Observatory (USNO 2020).

Preliminary analyses of our records revealed many instances when DO diel ranges abruptly declined relative to prior dates and remained low for multiple days (Fig. 2a). Such phenomena, hereafter referred to as diel range reset events (DRREs), were detected in each site, often occurred on the same date in multiple sites, and were found to be not associated with sensor maintenance visits. Because DRREs represented dramatic changes (up to 4 mg L^{-1} between

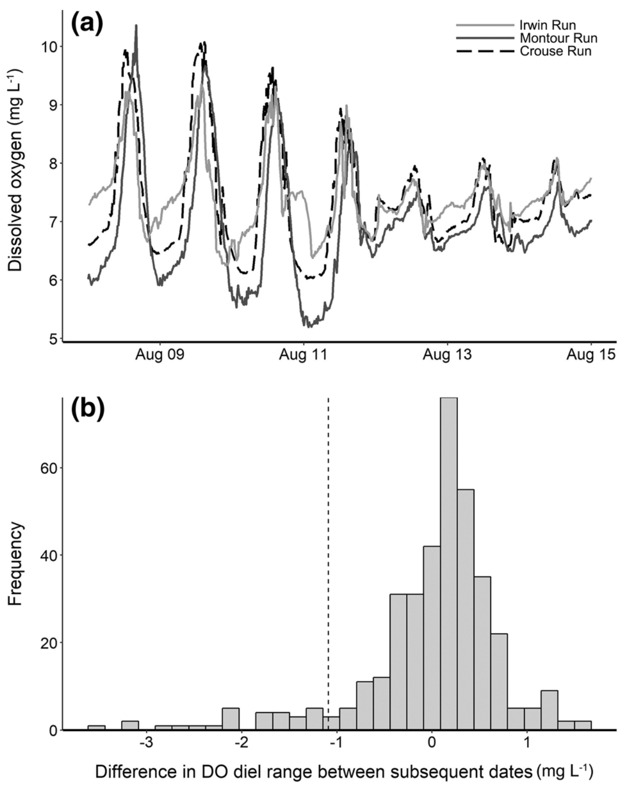

Fig. 2 **a** An example of a dissolved oxygen (DO) diel range reset event (DRRE) occurring in three study sites on 12 August 2017. **b** Distribution of differences in diel DO ranges in subsequent days for Crouse Run. The dashed line shows 1.5 standard deviations away from the mean; differences below this value were considered DRREs

subsequent days) in diel ranges with potential implications for stream biota and biogeochemical processes, we aimed to identify any meteorological or seasonal conditions that lead to an elevated probability of DRRE occurrence. At each site we identified dates when DRREs occurred by flagging when the difference in diel DO ranges between subsequent dates was more than 1.5 standard deviations below the mean (Fig. 2b). We chose this threshold because dates below this metric consistently represented a left skew with respect to the otherwise normal distribution of diel DO ranges (Fig. 2b).

We used several metrics that quantified meteorological conditions and ambient DO concentrations at multiple temporal scales in attempt to identify phenomena associated with DRRE occurrence. For each date on record, the mean water temperature and mean DO, for the preceding 2-, 5-, and 10-day periods along with total precipitation for each date plus the three preceding intervals were calculated. We quantified precipitation metrics for use as a proxy of hydrologic dynamics, as our sites were not monitored for discharge. Precipitation data were summarized daily from the Three Rivers Wet Weather rain gage network, which consists of heated precipitation gages that record precipitation on 5-min intervals throughout the metropolitan region (TRWW 2019). For each site, data from the closest rain gage in the network (≤ 5 km) was retrieved to account for differences in precipitation among watersheds. We estimated the diel and 2-day ranges of atmospheric temperature for each date using data from the Acmetonia Lock three weather station operated by NOAA (NOAA 2020a), located no more than 10 km from any site. Finally, each date on record was categorized as winter, spring, summer, or autumn based on astronomical calendar dates associated with solstices and equinoxes for the northern hemisphere. Any 2-, 5-, or 10-day metric associated with missing data in the corresponding site-specific DO record was removed prior to further analyses.

To assess how DO ranges in study sites were affected by possible light limitation of photosynthetic activity, we calculated mean daily cloud cover using hourly recorded data reported in oktas from the NOAA weather station at the Pittsburgh International Airport (NOAA 2020b). Mean cloud cover was assessed during daylight hours only using the sunrise/sunset phenological data from the USNO described above.

Our network of sensors lacked equipment to record discharge data and used precipitation as a proxy for hydrologic change. We therefore linked hydrologic responses to precipitation events for streams in our study region by assessing data from United States Geologic Survey (USGS) discharge monitoring site on Little Pine Creek (USGS gage 03,049,800). Little Pine Creek is a 14.9 km^2 watershed with moderate urban cover (4.8% impervious surface cover) within our stream network (Fig. 1) that has been analytically compared as a semi-rural control site to Nine Mile Run previously (Bain et al. 2014). Distinct flood events were identified from hydrologic records reported at 15-min intervals from calendar years 2016 through 2018 by flagging continuous periods where discharge exceeded three times the monthly median flow (Olden and Poff 2003). Precipitation data from the TRWW gage at Shaler (TRWW 2019) were linked to Little Pine Creek discharge data to assess flood responses to precipitation as outlined below.

Statistical analyses

To identify which metrics were most associated with DRREs, we generated boosted regression tree models that predicted DRRE occurrence with meteorological, temperature, and ambient DO metrics. Boosted regression tree (BRT) models are agglomerations of simple regression trees that are iteratively tested using a machine learning approach to produce an additive model that can detect nonlinearities among variables (Elith et al. 2008, Elith and Leathwick 2017). Each site-specific model was fit using a Bernoulli distribution for the dependent variable (DRRE occurrence), an interaction depth of one, bag fraction of 0.5, learning rate of 0.001, and 1000 trees. Models were assessed using the fraction of relative influence in the model for each variable, with those ≥ 20.0% considered meaningfully important. The same methodology to model DRRE event probability was applied to DO saturation to determine if patterns using this parameter rather than DO concentrations deviated from one another.

We also quantified how quickly DO diel ranges approached pre-DRRE conditions following the event to compare the speed of recovery among sites. To do so, the number of days required to return to 50% of the pre-DRRE diel range was calculated for each DRRE event at each site. We opted for the 50% metric rather than quantify the time required until full recovery because this threshold was met more quickly and precipitation events following the DRRE would have distorted the recovery time estimate. Furthermore, post-DRRE periods where precipitation totals exceeded 5 cm were removed from analyses. Recovery rates in units of days until 50% pre-DRRE recovery were compared among sites using a generalized linear model (GLM) with a Poisson error distribution. Mean water temperature during the recovery period was also included as a covariate in the GLM model to determine if recovery rates varied among seasons.

Cloud cover was assessed as a proxy for photosynthetic activity using linear models predicting DO diel range from mean diel cloud cover during daylight hours at each site. Because primary productivity has been shown to be limited by low temperatures in streams similar to those in our study area (Francoeur et al. 1999), only dates where mean water temperatures recorded in DO sensors exceeded 10 °C were included in these analyses. Although primary production can occur in temperatures below this threshold, we elected for a conservative metric to ensure that we limit analyses to periods where photosynthetic activity was very unlikely to be limited by cold conditions.

To link flood responses in Little Pine Creek to precipitation metrics we employed to assess DRREs, we calculated the maximum observed discharge for 208 flood events that occurred between 2016 and 2018 and linked this metric to 1- and 2-day precipitation totals associated with each flood event. Thresholds of precipitation events that triggered significant flood responses were identified using changepoint regression (Fong et al. 2017b) on ln-transformed flood magnitude and precipitation data. Estimated thresholds were back-transformed to identify 1- and 2-day precipitation totals that were likely to initiate moderate to severe flood events.

All statistical models were developed using R version 3.5.2 (R Development Core Team 2018). Package gbm (Greenwell et al. 2019) was used to develop BRT models and package chngpt (Fong et al. 2017a) was used to calculated precipitation thresholds that triggered hydrologic responses.

Results

DO ranges exhibited clearly disparate patterns among sites (Fig. 3). Although daily means varied mostly between 14 mg L^{-1} during winter and 8 mg L^{-1} during summer in all sites, diel variability for both DO concentration and saturation was lowest in the site with the smallest watershed (Breakneck Creek) and greatest in the largest and most urbanized watershed (Nine Mile Run; Figs. 3, 4). Diel DO ranges in Nine Mile Run were higher on average and more variable than all other sites (Fig. 4), with some days ranging < 2 mg L^{-1} and others exceeding 10 mg L^{-1}. DO saturation shifted along the size gradient, with saturation rarely exceeding 100% in the two smallest sites (Breakneck Creek and Glade Run) while supersaturation was commonly observed throughout most seasons in all other sites (Fig. 3).

Patterns in diel DO minima and maxima timing also appeared to vary among sites based on a watershed size gradient (Fig. 5). Although DO minima were most likely observed during dark hours and maxima during daylight hours in all sites, DO maxima were observed during dark hours in the smallest sites (Glade Run and Breakneck Creek) during over a quarter of the daily records, suggesting that temperature fluctuations were sometimes more capable of controlling diel patterns than biological activity. In larger sites, the diel DO minima and maxima patterns suggested biological controls on DO, with nearly all maxima observed during daylight hours and minima during dark hours. However, diel minima were observed in the largest and most urban stream (Nine Mile Run) during daylight hours during nearly a quarter of the record.

BRT models proved capable of predicting DRRE occurrence. Explained deviance among models ranged from a low of 0.23 (Glade Run) to a high of 0.38 (Nine Mile Run); cross-validation correlation coefficients ranged from 0.47 to 0.63 (Table 2). The most influential controls on DRRE occurrence as identified by the BRTs varied among the six sites. The most consistent and strongest predictors of DRRE occurrence were metrics associated with precipitation (Table 2, Fig. 6), which accounted for nearly all relative influence in the Montour Run BRT model and dominated relative influence in most other site models. Partial dependency plots (Fig. 7), which illustrate the standardized marginal effect of individual parameters on the response variable (in this case, the likelihood of a DRRE event occurring), revealed that DRRE events were most likely to occur at colder water temperatures, higher levels of precipitation, and dates with greater ranges of air temperatures. Trends shown in Fig. 8 also illustrate the nature of relationship (i.e., linear or other) between the predictor variables and response and nonlinear threshold

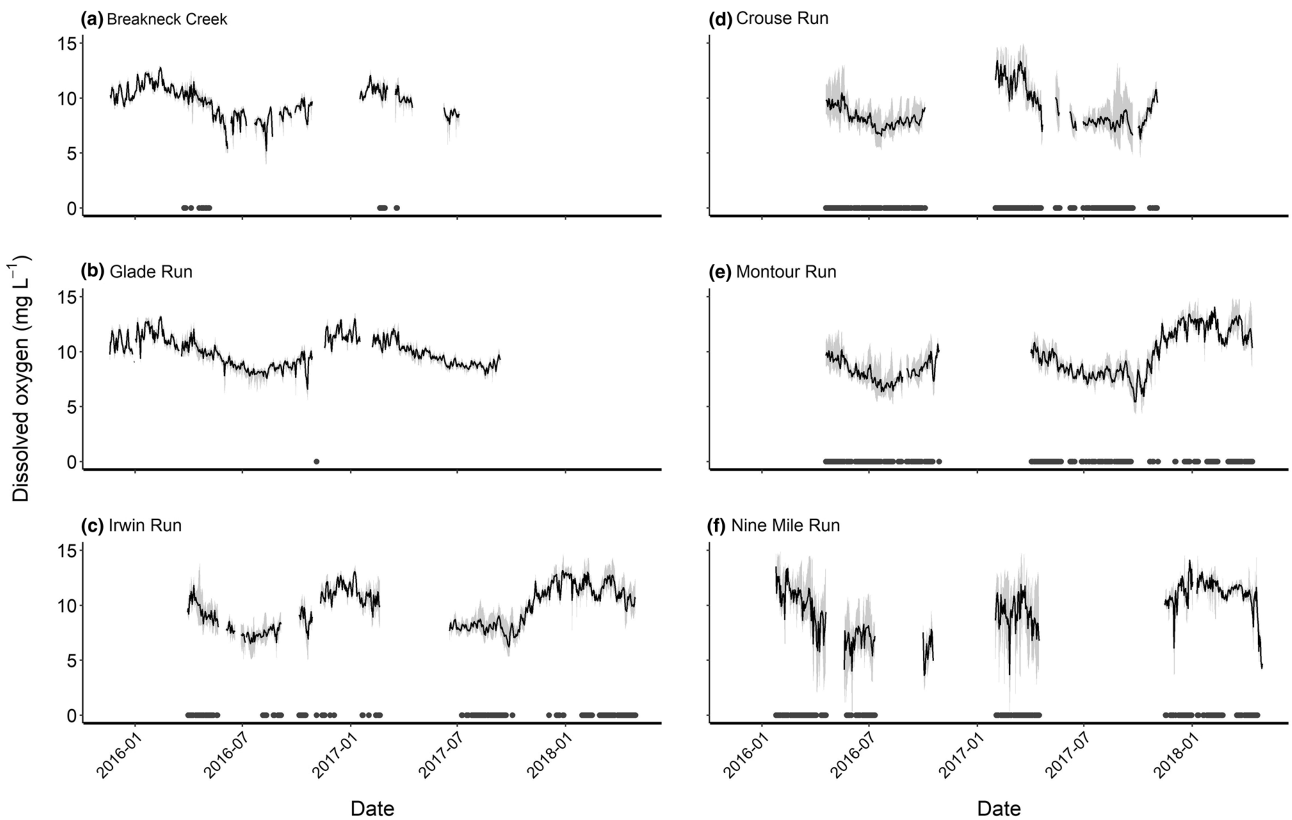

Fig. 3 Patterns in dissolved oxygen (DO) in each site over the course of the study period. The dark line represents the daily average and gray bands illustrate diel ranges. Filled circles above the x-axis represent dates in which DO saturation exceeded 100%. Gaps in the record reflect instances when sensors failed, or data were deemed unreliable

effects appear to be associated with precipitation and air temperature ranges. For example, a DRRE would be expected if 1-day precipitation exceeds 2 cm and/or if 2-day precipitation exceeds about 3 cm in all but one sites. In the two smallest watershed sites with lowest urban cover (Glade and Irwin Runs), air temperature variability showed relative strong influence, with diel temperature ranges exceeding 15 °C leading to a likely DRRE event.

Analyses of DRRE patterns for models with DO quantified as concentration versus saturation were largely congruent. In all sites except Glade Run, DRRE events for DO concentration and saturation co-occurred ≥ 50% of the time (Supplementary Table S1). Consequently, models using DO saturation to quantify DRRE largely mirrored those generated from DO concentration (Supplementary Figures S1 and S2).

Recovery of diel DO ranges to pre-DRRE levels varied significantly among sites ($F_{5,150}=2.9$, $p=0.0161$, Fig. 8a) and increased along a temperature gradient ($F_{1,150}=7.8$, $p=0.0060$, Fig. 8b). Recovery times decreased consistently along a watershed size gradient, with 50% pre-DRRE diel ranges reached after a mean of 4.5 days in the smallest site (Breakneck Creek) and means of 2.1 in the two largest sites: Montour and Nine Mile Runs. Warmer water temperatures tended to result in longer recovery times (GLM coefficient for water temperature ± 1 standard error, °C = 0.103 ± 0.026).

Photosynthetic activity likely contributed significantly to diel DO ranges in all sites, including the smallest sites with closed canopies. DO diel ranges on dates where mean water temperatures exceeded 10 °C significantly declined as a function of mean diel cloud cover in all sites (Fig. 9) though the variance accounted for differed among sites, with R^2 values in linear models ranging from 0.01 (Nine Mile Run) to 0.14 (Glade Run).

The assessment of hydrologic events in Little Pine Creek from 2016 to 2018 suggested that the thresholds of precipitation identified in BRT models correspond to significantly elevated discharge. Discharge records included 176 high flow events over the period monitored with a mean (± 1 standard error) duration of 0.58 ± 0.06 days. Changepoint analyses detected a threshold of 1.0 cm (95% confidence range 0.1–2.8 cm) over a 1-day period as enough precipitation to elevate maximum flood discharge and a threshold of 0.4 cm (95% confidence range 0.1–3.9 cm) over a 2-day period (Fig. 10). Such thresholds correspond to those indicated by BRT models of DRRE occurrence that reported

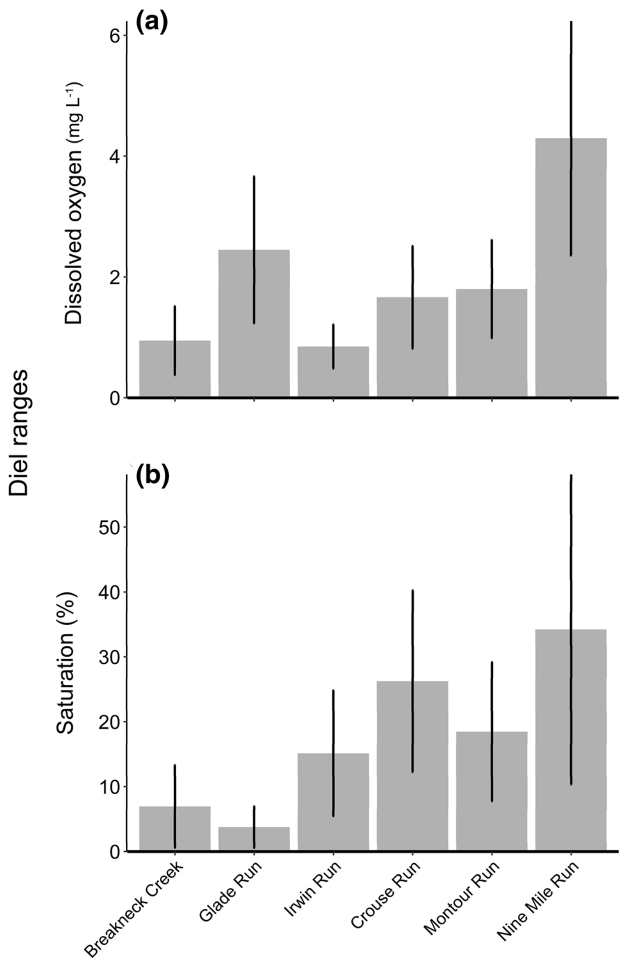

Fig. 4 Mean diel dissolved oxygen (DO) ranges among the six study sites. Values illustrate mean (± 1 standard deviations) diel ranges of **a** DO concentrations and **b** DO saturation. Sites are ordered by watershed size

1- and 2-day precipitation totals as important factors (compare thresholds in Fig. 10 to partial dependencies in Fig. 7).

Discussion

Most of our knowledge regarding dissolved oxygen patterns has been from stream metabolism studies in forested and agricultural watersheds (Young and Huryn 1999; Mulholland et al. 2001; Gücker et al. 2009), but there has been growing work analyzing DO oscillations in urban streams (Bernot et al. 2010, Pennino et al. 2014, Leford et al. 2017). Our findings suggest that precipitation events may significantly diminish the amplitude of diel DO oscillations associated with respiration and photosynthesis in both forested and urban streams. Although our watershed sample size precludes the ability to directly test the role of watershed size and urbanization, patterns in our data suggest that as streams expand in size, the likelihood of diel maxima occurring during daylight hours escalates (Mejia et al. 2018); this likely reflects greater shifts between respiration and primary productivity in larger systems with more periphyton (Bott et al. 1985; Kaushal et al. 2014a, b; Hotchkiss et al. 2015). More urbanized streams tend to exhibit greater diel DO ranges, potentially due to increased respiration and primary production driven by eutrophication, as higher levels of labile organic matter in urban streams can support microbial respiration (e.g., He et al. 2011, Klose et al. 2012, Miskewitz and Uchrin 2013, Kaushal et al. 2014a, b, Smith and Kaushal 2015). Among all sites, levels of precipitation capable of elevating discharge typically reduce the magnitude of diel DO oscillations. Such events may occur as floods disrupt respiration and especially primary production (Uehlinger 2000; Roberts et al. 2007; Reisinger et al. 2019) and/or because floods alter reaeration of DO at the surface (Wilcock 1988).

Precipitation events influence diel DO variations

The behavior of DO concentrations in our stream sites suggests that thresholds of precipitation are capable of routinely resetting diel DO variations. High 1- and/or 2-day precipitation events, which likely corresponded to elevated discharge, were significantly associated with DO DRREs in each site as diel DO ranges were consistently found to be depressed for multiple days following these events. One potential mechanism for such patterns is that elevated discharge increases scouring and reduces microbial biomass, thereby lowering the magnitude of DO ranges in subsequent days (Uehlinger 2000, Reisinger et al. 2018, Blaszczak et al. 2019). Other studies have found that primary productivity can be greatly reduced during precipitation events if elevated flows increase turbidity (Young and Huryn 1996; Griffiths et al. 2013) or decrease respiration rates by flushing organic matter (Acuña et al. 2004, 2007). However, precipitation events and increased stream discharge have also been shown to mobilize sediments and reduce primary productivity rates by displacing periphyton (O'Connor et al. 2012; Blaszczak et al. 2019).

Precipitation events and periods of elevated discharge could also alter reaeration, further complicating the interacting processes that structure diel DO dynamics in streams. Increased water velocity and turbulence, both of which increase during flood periods, typically enhance reaeration (Thyssen and Erlansen 1987; Raymond et al. 2012; Haider et al. 2013). However, relationships between DO reaeration coefficients and discharge can be highly site-specific (Aristegi et al. 2009; Demars et al. 2015) or altogether absent (Melching and Flores 1999). Both positive and negative relationships between reaeration coefficients and flow rates have been observed even within networks of streams with similar watershed characteristics

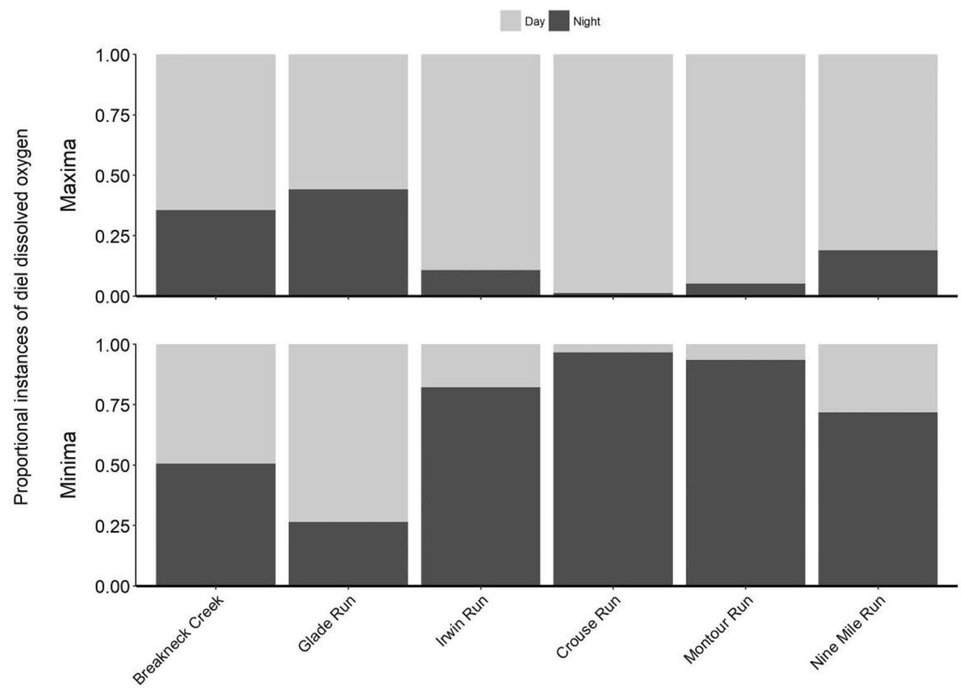

Fig. 5 Proportional summaries of diel dissolved oxygen (DO) minima and maxima with respect to whether each metric occurred during daylight or nighttime hours. The proportions reflect the entire record of each study site. Sites are ordered by watershed size

Table 2 Boosted regression tree model performance and predictor variable relative influence values for models predicting diel range reset event (DRRE) occurrence

	Breakneck Creek	Glade Run	Irwin Run	Crouse Run	Montour Run	Nine Mile Run
Model performance						
Deviance explained	0.29	0.23	0.3	0.38	0.33	0.38
CV correlation	0.48	0.63	0.52	0.57	0.55	0.47
Relative influence (%)						
1-day precipitation	–	4.4	**28.7**	**42.9**	**45.3**	**38.6**
2-day precipitation	**48.7**	17.8	8.9	**22.4**	–	–
5-day precipitation	11.3	9.1	–	–	16.3	–
10-day precipitation	–	9.4	18.7	–	13.8	–
1-day air temperature range	–	**30.3**	**31.2**	–	–	–
2-day air temperature range	–	15.5	–	–	–	–
2-day DO concentration	–	–	–	–	–	–
5-day DO concentration	**30.5**	–	–	–	–	–
10-day DO concentration	9.6	–	–	–	9.2	**26.2**
2-day water temperature	–	13.5	–	–	–	**23.9**
5-day water temperature	–	–	–	–	–	–
10-day water temperature	–	–	12.5	**24.6**	9.5	–
Season	–	–	–	10	–	–

Models included only predictor variables that exceeded 5% relative influence in initial models. Variables deemed highly influential (exceeding 20% of relative influence) are highlighted in bold and their partial dependency plots are illustrated in Fig. 6

(Izagirre et al. 2007; Aristegi et al. 2009). While elevated turbulence would increase reaeration during high flow events, such an effect could be counteracted by decreasing reach travel time (Wallin et al. 2011). Therefore, identifying the role of reaeration in shaping DO oscillations under different discharge levels may require direct measurement of the process.

We cannot directly compare the relative importance of altered reaeration rates versus disruptions to metabolic rates in shaping DRREs using our findings because we did

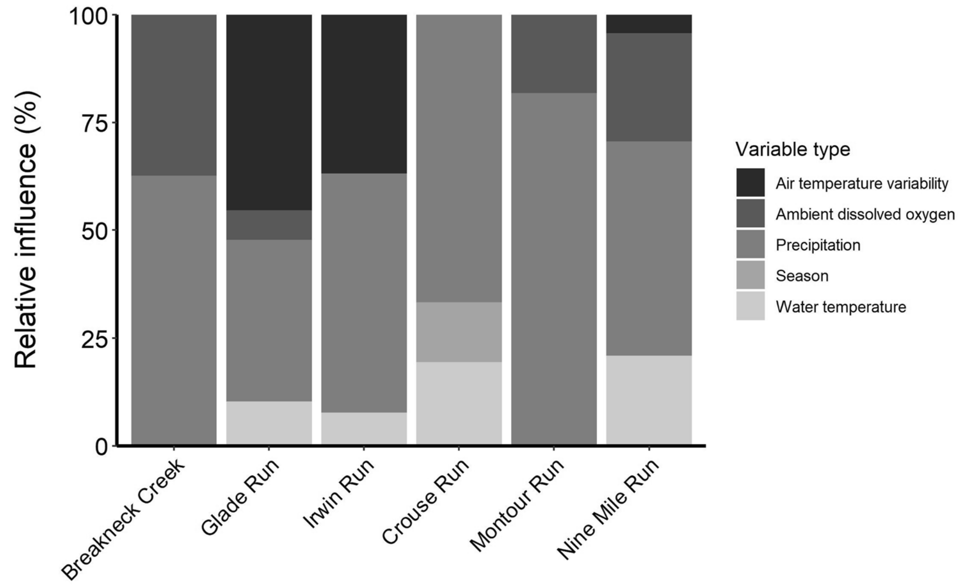

Fig. 6 Relative influences of variable types on dissolved oxygen (DO) diel range reset event (DRRE) probability in boosted regression tree models for each site

not measure reaeration or biofilm density. However, several attributes of our results suggest that biofilm sloughing may have been relatively more important. The median duration of flood periods in Little Pine Creek, defined as discharge exceeding three times the monthly median flow, was 0.24 days while median DRRE recovery periods (defined as the amount of time transpired before DO diel ranges reached 50% of their pre-DRRE state) were ~ 2 days or more in most sites. Furthermore, DRRE recovery times were longest (~ 4 days) in our smallest site, where channel slope was steepest and flood discharge periods following a precipitation event would have been shortest. Moderate flood events have proven to be capable of causing mass periphyton sloughing events (Biggs et al. 1989; Biggs et al. 1999) that are capable of causing disruptions to metabolic processes during post-flood periods (Reisinger et al. 2018, Blaszczak et al. 2019; Uehlinger 2000). Nevertheless, we cannot rule out flood-driven changes to reaeration rates as the mechanistic cause behind DRRE events and the two mechanisms we outline above are not mutually exclusive.

Though our findings focus on the amplitudes of diel DO oscillations and not metabolic rates, the consistent association of DRREs with precipitation events in all sites suggests that discharge pulses are consistently capable of disrupting biological activity and/or reaeration across a range of watershed settings. We propose that streams exhibit different thresholds of discharge influenced by air temperatures and seasonality that, if exceeded, cause whole-ecosystem reductions in biological activity that can persist for days following the event based on watershed size and urbanization. Diel DO ranges in our network streams appear to consistently respond, if 1-day precipitation exceeds ~ 2 cm and/or if 2-day precipitation exceeds ~ 2.5 cm (Fig. 7). Such thresholds are likely to prove system-specific due to the hydrologic template and periphyton community present, as periphyton species possess differential resistance to high flow events (Graba et al. 2013; Tornés and Sabater 2010). Nonetheless, identifying these thresholds is important because diel variations of DO can be linked to changes in redox potentials, mobilization and transformation of chemical cocktails in urban streams during hydrologic events (Kaushal et al. 2018), and can be indicative of abrupt declines in whole stream nitrogen uptake functions (Reisinger et al. 2019).

Comparison of DO concentrations and oscillations in urban versus forested watersheds

Our findings also illustrate how controls on DO concentrations in urban streams differ from those in less impacted watersheds. Although urban streams exhibit a tendency to frequently lose periphyton biomass due to flood-driven scour (e.g., Murdock et al. 2004; He et al. 2011; Smith and Kaushal 2015; Reisinger et al. 2017), observed thresholds of precipitation that led to DRREs did not appear to be substantially lower in our urbanized sites. Our most urbanized site, Nine Mile Run, does flood in response to lower precipitation thresholds relative to nearby streams with watersheds lacking urban land cover (Bain et al. 2014) and would therefore be presumed to exhibit DRREs with lower precipitation. Consequently, periphyton communities in urban streams may shift such that metabolic processes may become more resilient to flood events (Reisinger et al. 2017). Furthermore, DO regimes in urban streams can greatly vary among geomorphic settings, from consistently near-hypoxic to near-saturation and may be particularly sensitive to channel

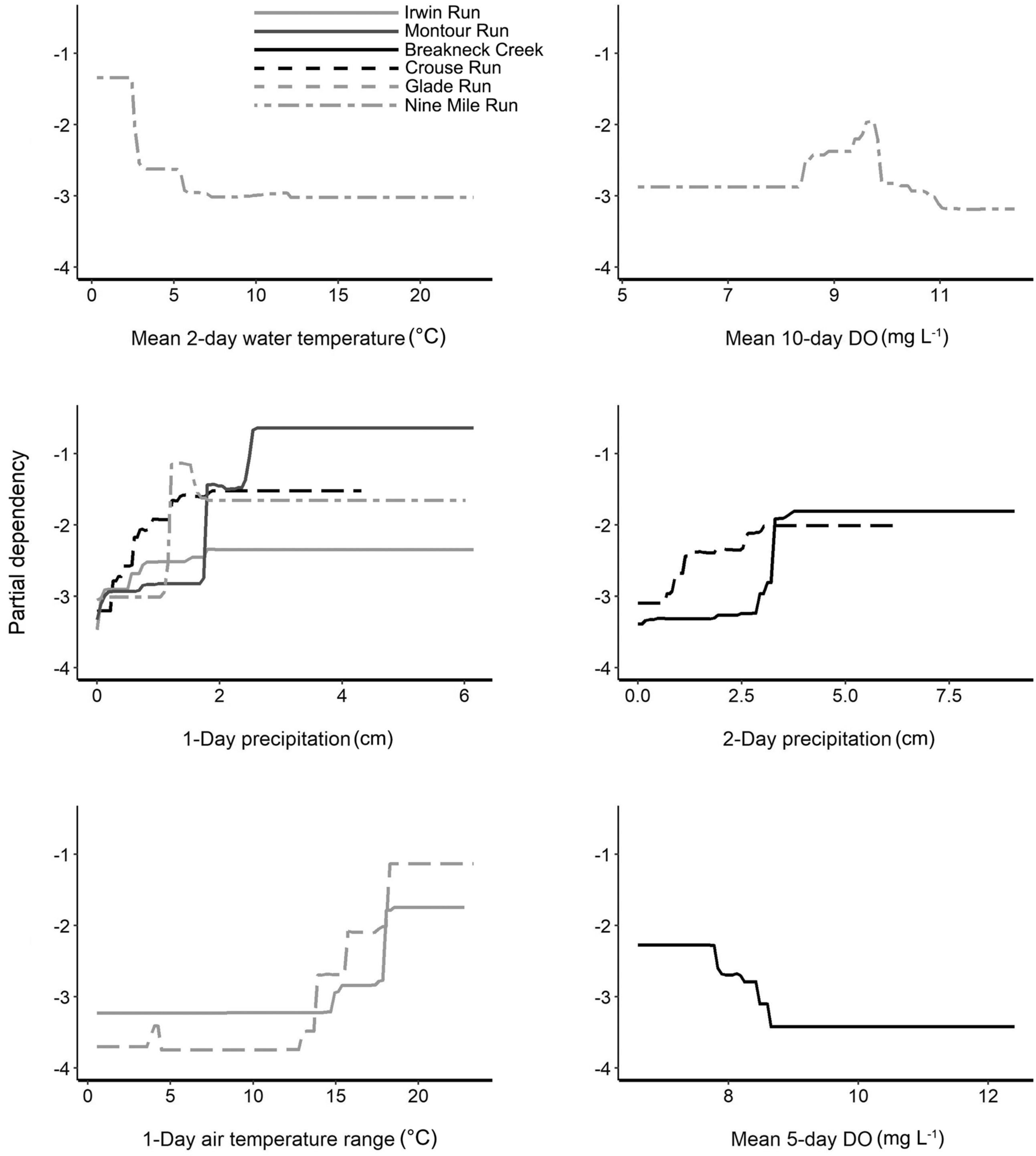

Fig. 7 Site-specific partial dependency plots of variables with high relative influence on diel DO range reset events. Lines represent the marginal effect of each site/parameter combination on the likelihood of a diel range reset event (DRRE) occurring, with larger values indicating higher relative likelihood (see Elith and Leathwick 2017). Only variables with ≥ 20% relative influence are shown

slope (Blaszczak et al. 2019). Diel DO ranges in our most urbanized site, Nine Mile Run, were found to recover faster than all other sites, suggesting that the periphyton community possesses resistance to flood events or recovers faster due to elevated nutrient concentrations delivered by combined sewerage overflows.

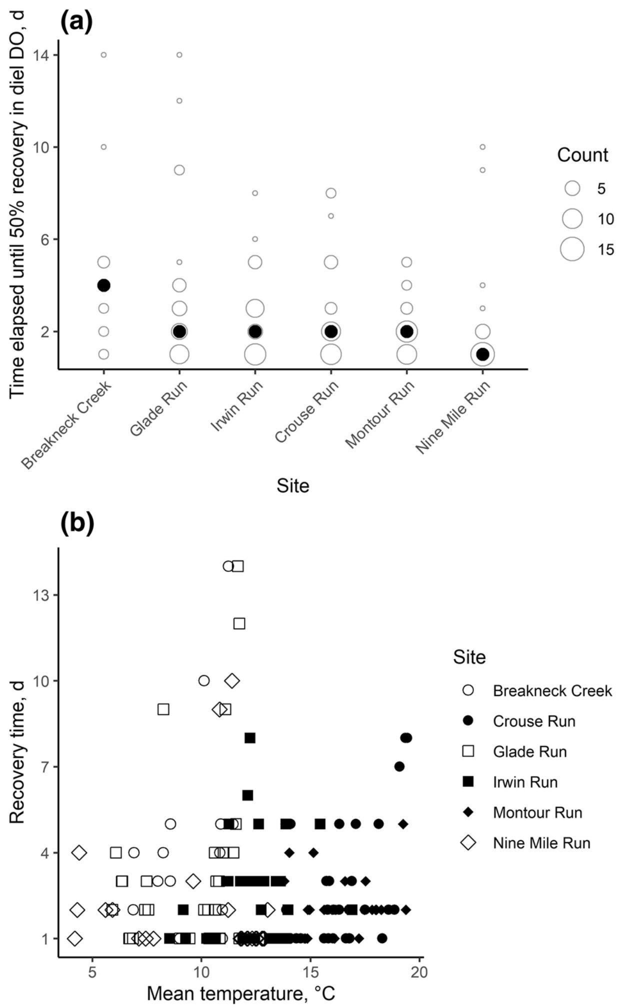

Fig. 8 **a** Time elapsed **d** until the recovery of diel dissolved oxygen (DO) range following a diel range reset event (DRRE) reached 50% of the diel DO range observed during the day prior to the corresponding DRRE. Filled points represent medians and open points are observations with sizes scaled by the count of observations. **b** Recovery of diel DO ranges following a DRRE along a temperature gradient. Temperatures on the x-axis correspond to the mean observed water temperature during the recovery period

In our larger and urban sites, Nine Mile and Crouse Runs, DRREs were unexpectedly more associated with cooler temperatures and/or when antecedent mean ambient DO concentrations were relatively high. Warmer temperatures typically elevate periphyton productivity, resulting in greater diel DO ranges particularly in urbanized streams (Wassenaar et al. 2010; Klose et al. 2012), which would suggest that conditions inherent during summer would more likely lead to DRREs. However, because our data are limited to six sites, such concepts remain speculative.

Other potential factors that can influence stream DO oscillations

Other findings from this effort need to be considered in the context of the study limitations. Continuous data on additional parameters, such as photosynthetically active radiation (PAR), precise field-derived estimates of channel slopes, reaeration coefficients, and salinity and discharge measurements would have enabled the capacity for far greater inquiry related to hydrologic metrics and metabolic processes than we were able to conduct with a single DO sensor per site (Grace et al. 2015). Because our initial intent was to quantify the frequency and intensity of low DO concentration events in urbanizing streams, we opted to broaden the spatial scale and number of sites for our study at the expense of more intensive monitoring at a single site. Emerging networks of open-access integrated sensor networks coupled with routine biological surveys, such as those maintained by the National Ecological Observatory Network (NEON; Goodman et al. 2015) and the Australian Supersite/TERN network (Karan et al. 2016), will increase the ability to investigate relationships among dynamic environmental attributes in streams to test hypotheses related to our results. Additionally, open-access data from networks designed to characterize stream metabolic signatures (StreamPULSE; Bernhardt et al. 2018) and a key metanalysis with a similar objective (Appling et al. 2018) further enhance the ability to conduct such inquiries. The temporal coverage of our data were incomplete due to events that disrupted sensor performance, particularly in Nine Mile Run, which limited the extent to which we could quantify patterns related to seasonality. Most studies deploying in situ sensors in streams confront this challenge, especially in urban systems where flood frequencies are elevated. Despite our data gaps, we were able to assess data from all seasons in each site. Furthermore, the disjoint temporal extents among sites may ensure that the relationships we report do not idiosyncratically reflect our study period.

Implications and conclusion

Our work highlights the importance of accounting for watershed context when considering the factors regulating DO oscillations. Streams draining very small watersheds may not always exhibit the typical diel patterns in DO cycling driven by biological processes, particularly when atmospheric temperatures shift rapidly in a short period. Additionally, moderate to high precipitation events appear to be capable of disrupting the extent to which biological processes and/or reaeration control DO. Climate change models forecast greater variability in temperature fluctuations and/or extremes (Horton et al. 2015) and more frequent moderate to severe flood events (Ban et al. 2015, Mallakpour

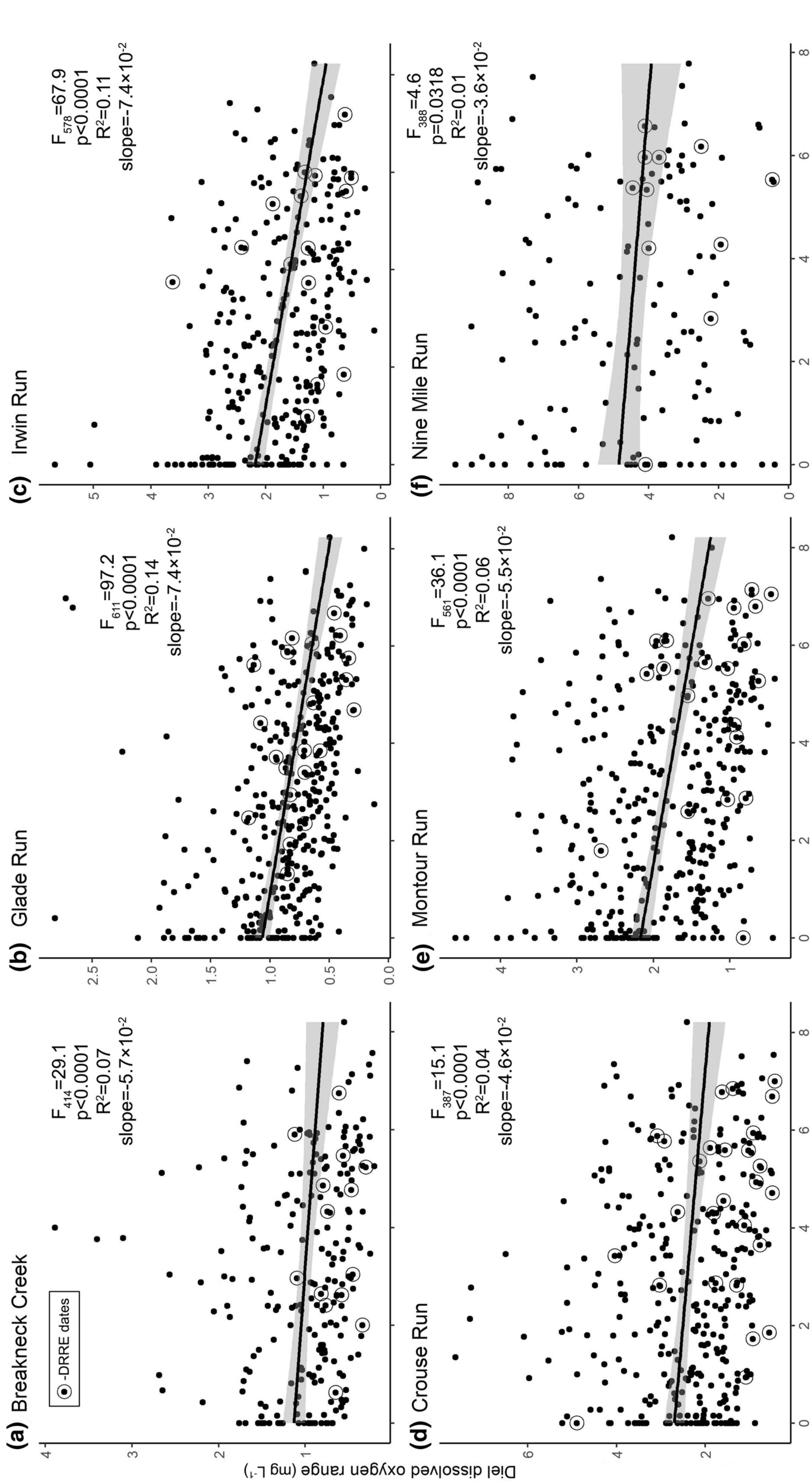

Fig. 9 Relationship between mean cloud cover and diel dissolved oxygen (DO) ranges observed on each sampling date in each site. Linear regression model metrics are included for each site and lines depict best fits and associated 95% confidence intervals. Dates corresponding to diel range reset event (DRRE) occurrences are delineated with enlarged, hollow points

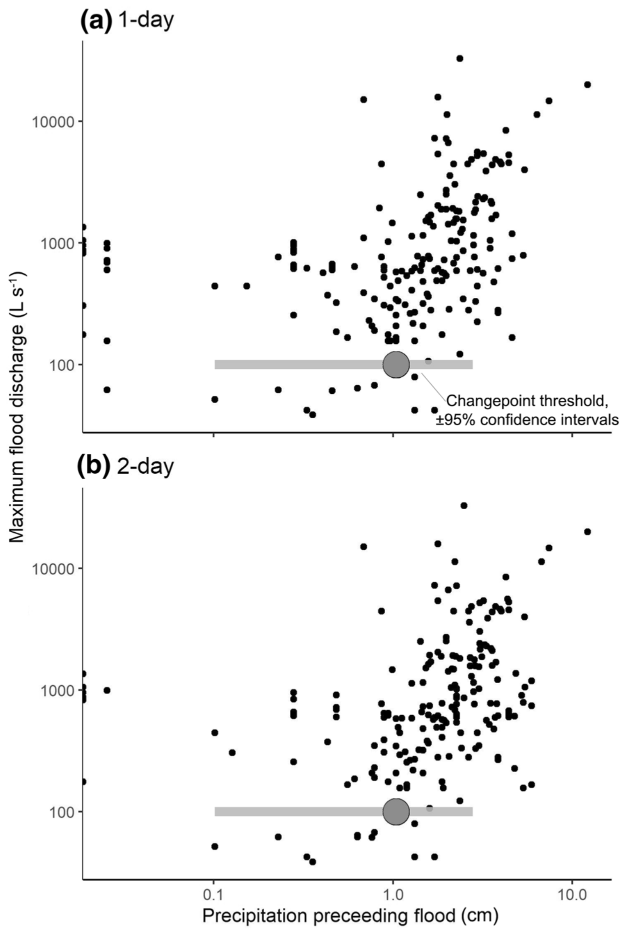

Fig. 10 Flood magnitude in Little Pine Creek along gradients of 1- and 2-day precipitation totals prior to the initial onset of each flood. Changepoint thresholds (±95% confidence intervals) estimating where the relationship significantly shifts are depicted with enlarged gray points

and Villarini 2015) in many regions with climates like those from our study area. There has been increased variability in streamflow in the Mid-Atlantic U.S. over the past century (Kaushal et al. 2014a, b) and warming stream and river temperatures (Kaushal et al. 2010). Such long-term changes to climate attributes suggest that the nature of DO regimes in streams may also be evolving, with potential consequences for critical processes related to the parameter such as nutrient cycling and greenhouse gas fluxes (Harrison et al. 2005; Rosamond et al. 2012).

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