The Effect of Urbanization and Dam Removal on Stream Metabolism

by

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To my father Prof. Mohammednuman AbuHarbid.

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TABLE OF CONTENTS

CHAPTER

PAGE

1	INTROD	UCTION	1
	1.1	Motivation	1
	1.2	Background Information	5
	1.2.1	Primary Production in Streams	5
	1.2.2	Stream Metabolism	6
	1.2.3	Calculating Stream Metabolism from Diel Change in Dissolved	
		Oxygen	8
	1.2.3.1	Direct Measurements	8
	1.2.3.2	Calculating the Reaeration Coefficient	9
	1.2.3.3	Indirect Measurements	12
	1.2.4	Methods to quantify urbanization	13
	1.2.5	The Influence of Restoration Projects on Stream Metabolism	15
	1.2.5.1	Dam Removal Restoration Projects	20
	1.3	Objectives	22
	1.4	Dissertation Overview	26
2	THE EFF	FECT OF FLOOD EVENTS ON ECOSYSTEM METABOI	\mathbf{ISM}
	IN SUBU	JRBAN STREAMS	29
	2.1	Abstract	29
	2.2	Introduction	30
	2.3	Study Area	34
	2.4	Methods	39
	2.4.1	Whole-stream metabolism	39
	2.4.2	Defining flood events and quantifying stream flashiness \ldots .	41
	2.4.3	Metabolism resilience and resistance to storms	42
	2.4.4	Statistical analysis	43
	2.5	Results	45
	2.5.1	Whole-stream metabolism	45
	2.5.2	Flow patterns and metabolism signal	49
	2.5.3	Metabolism resistance and resilience	51
	2.5.4	Factors influencing metabolism in urban streams	57
	2.6	Discussion	60
	2.6.1	Metabolism resistance and resilience to storms	60
	2.7	Conclusion	64
	2.8	Acknowledgments	65

TABLE OF CONTENTS (Continued)

CHAPTER

PAGE

3	THE IN	MPACTS OF DAM REMOVAL ON STREAM ECOSYSTEM		
	META	BOLISM	66	
	3.1	Abstract	66	
	3.2	Introduction	67	
	3.3	Study Area	71	
	3.4	Methods	74	
	3.4.1	Whole-stream metabolism	74	
	3.4.2	Statistical analysis	75	
	3.5	Results	76	
	3.5.1	Temporal variation in stream metabolism following dam removal	76	
	3.5.2	Spatial variation in stream metabolism following dam removal	82	
	3.5.3	Net ecosystem production following dam removal	82	
	3.6	Discussion	84	
	3.6.1	The influence of dam removal on stream metabolism	84	
	3.6.2	Metabolism data confirms increased autotrophy with dam removal	86	
	3.7	Conclusion	87	
4	THE IN	VFLUENCE OF URBANIZATION ON STREAM METABOI	LISM	88
	4.1	Abstract	88	
	4.2	Introduction	89	
	4.3	Methods	93	
	4.3.1	Site selection	93	
	4.3.2	Urban land use gradient and flashiness indices	96	
	4.3.3	Whole-stream metabolism	100	
	4.3.4	Statistical analysis	101	
	4.4	Results	102	
	4.4.1	Urban land use gradient and flashiness indices	102	
	4.4.2	Whole-stream metabolism	103	
	4.4.3	The effect of urbanization indices on stream metabolism \ldots	107	
	4.5	Discussion	112	
	4.5.1	Urban and flashiness indices variations	112	
	4.5.2	Stream metabolism of urban streams	113	
	4.5.3	Continuous metabolism data confirm the predominance of heterotro	ophy	115
	4.6	Conclusion	117	
5	CONCI	LUSIONS	118	
	5.1	Results	118	
	5.2	Future Work	120	
	APPEN	NDICES	124	
	CITED	LITERATURE	150	

TABLE OF CONTENTS (Continued)

CHAPTE	\mathbf{R}										PAGE
	VITA .	 	158								

LIST OF TABLES

TABLE	$\mathbf{P}A$	GE
Ι	SITES NAMES, LOCATIONS, AREA, THE CLOSEST USGS STATION DRAINAGE AREA KM ² , SITE DISTANCE FROM USGS STATION KM, RELATIVE AMOUNT OF WATERSHED THAT IS DEVELOPED AREA, AND IMPERVIOUS SURFACE COVER (ISC), AND R-B FLASHINESS INDEX CALCULATED FROM THE ADJUSTED DISCHARGE DATA AT EACH SITE	38
II	MEAN (\pm STD. DEV.) DAILY METABOLISM (G O ₂ M ² D ⁻¹) FOR SUMMER PERIOD, FALL PERIOD, AND WHOLE STUDY PERIOD	47
III	AVERAGE FLOW ($Q_A VERAGE$) IN M ³ /S, MAXIMUM FLOW (Q_{MAX}) IN M ³ /S, THE NUMBER OF DAYS WHERE $Q > 2Q_{AVERAGE}$, AND THE FLASHINESS INDEX REPRESENTED BY MEAN AND (±) ST.D. FOR STORMS STUDIED AT EACH SITE	50
IV	RESISTANCE OF GPP AND ER TO FLOODS BY SITE REPRESENT AS THE MEAN (±) ST.D., RESULTS OF T-TEST TO COMPARE DIFFERENCE BETWEEN GPP AND ER RESISTANCE, RESILIENCE OF GPP AND ER TO FLOODS IN DAYS REPRESENTED AS THE MEAN (±) ST.D., AND RESULTS OF T-TEST TO COMPARE DIFFERENCE BETWEEN GPP AND ER RESILIENCE. FLOODS STIMULATION IS THE NUMBER OF TIMES FLOODS STIMULATED GPP OR ER TO STUDIED FLOODS TOTAL NUMBER OF (INCREAS AFTER THE FLOOD).	ED E SED 53
V	LOCATIONS, DISTANCE FROM DAM REMOVAL IN KM, AND DATE RANGES.	73
VI	AUTO-REGRESSIVE INTEGRATED MOVING AVERAGE (ARIMA) MODEL STRUCTURE FOR GROSS PRIMARY PRODUCTION (<i>GPP</i>) AND ECOSYSTEM RESPIRATION (<i>ER</i>) FOR ALL STATIONS PRE AND POST DAM REMOVAL. AS DESCRIBED IN THE METHODS, MODEL STRUCTURE IS REPRESENTED AS (P, D, Q). SE = STANDARD ERROR, θ IS THE MOVING AVERAGE COEFFICIENT, φ IS THE AUTO-REGRESSIVE COEFFICIENT. REFER TO TABLE 1 FOR STATION LOCATIONS	78

LIST OF TABLES (Continued)

TABLE

PAGE

VII	MEAN (+/- STANDARD DEVIATION) GROSS PRIMARY PRODUCTI	ON
	(GPP) ECOSYSTEM RESPIRATION (ER) , AND NET ECOSYSTEM	
	METABOLISM (G $O_2 M^{-2} D^{-1}$) FOR GOOD HOPE DAM STATIONS	
	SEPARATED ON FOUR TIME PERIODS, 2 MONTHS BEFORE,	
	1 MONTH BEFORE, 1 MONTH AFTER DAM REMOVAL, AND	
	2 MONTHS AFTER DAM REMOVAL. % CHANGE REFERS TO	
	THE DIFFERENCE BETWEEN MEAN GPP OR ER COMPARED	
	TO THE VALUES OF ONE MONTH BEFORE THE DAM REMOVAL.	80
VIII	LIST OF VARIABLES THAT WERE USED FOR THE CALCULATION	
	OF THE URBAN LAND USE GRADIENT INDEX. (NOTE: ONLY	
	BOLD VARIABLES WHICH CORRELATED WITH 2011-2015 POPULA	TION
	DENSITY AT A LEVEL ≥ 0.50 WERE USED IN THE CALCULATION	
	OF THE URBAN INTENSITY INDEX.)	97
IX	SITES LISTED ACCENDING FROM LOWER TO HIGHER URBAN	
171	LAND USE GRADIENT INDEX (ILLIG) INDEX CLOSEST USGS	
	STATION BASIN AREAS IN KM2 WATERSHED NAME ULUC	
	INDEX VALUES AND ELASHINESS INDEX VALUES	00
	INDEX VALUES, AND FLASHINESS INDEX VALUES	99
Х	MEAN (STD. DEV.) DAILY METABOLISM (G $O_2 M^{-2}D^{-1}$), FOR	
	THE WHOLE STUDY PERIOD. REFER TO TABLE 4.2 FOR	
	SITES NAMES.	104
XI	RESULTS OF THE TWO-PHASE LINEAR REGRESSION FOR	
	SITES WITH LOWER URBAN INTENSITY (URBAN LAND USE	
	GRADIENT INDEX (ULUG) < 23.88) AND SITES WITH HIGHER	
	URBAN INTENSITY (ULUG > 23.88)	111
XII	REAERATION COEFFICIENT EMPIRICAL EQUATIONS (HAIDER	
	ET AL., 2013)	125
XIII	RELATIONSHIP BETWEEN THE DISCHARGE (Q) AND THE	
	REAERATION COEFFICIENT (K) WHEN SIGNIFICANT MODEL	
	EXISTED.	128

LIST OF TABLES (Continued)

TABLE

PAGE

XIV	AUTO-REGRESSIVE INTEGRATED MOVING AVERAGE (ARIMA)	
	MODEL STRUCTURE FOR GROSS PRIMARY PRODUCTION	
	(GPP) AND ECOSYSTEM RESPIRATION (ER) FOR ALL SITES.	
	ACCORDING TO THE METHODS SECTION, THE MODEL STRUCT	ΓURE
	IS DESCRIBED AS (P, D, Q) . SE = STANDARD ERROR, θ IS	
	THE MOVING AVERAGE COEFFICIENT, ϕ IS THE AUTO-REGRES	SSIVE
	COEFFICIENT.	129
XV	METABOLISM REINVIGORATION	147
XVI	VARIABLES USED FOR CALCULATING THE URBAN LAND	
	USE INDEX AND SAMPLE CALCULATIONS FOR SITE1	149

LIST OF FIGURES

FIGURE		PAGE
1	Stream Functions Pyramid	3
2	Nighttime Regression Method for Estimating K and ER	11
3	Inverse Modeling Application	13
4	Conventional agricultural ditch with floodplain added	18
5	An example of pre and post dam removal restored site $\ldots \ldots \ldots$	22
6	Expected Effects of Urbanization on Stream Metabolism	23
7	Expected Effects of Dam Removal on Stream Metabolism $\ \ldots \ \ldots \ \ldots$	25
8	Predictions for resistance (i.e., magnitude of change) and resilience (i.e, speed of recovery) for gross primary production (GPP) and ecosystem respiration (ER) following a storm in an urban stream	34
9	Study site locations are in northeastern Illinois, USA, in the Chicago metropolitan region. WBBR = West Branch DuPage River at Butterfield Road, WBMG = West Branch DuPage River at McDowell Grove, WBWD = West Branch DuPage River at Warrenville Grove dam, EBHL = East Branch DuPage River at Hidden Lake, EBBR = East Branch DuPage River Butterfield Road, SCBR Salt Creek at Butterfield Road, and SCYR = Salt Creek at York Road	36
10	Mean (\pm Std. dev.) monthly <i>GPP</i> and <i>ER</i> results in g O ₂ m ⁻² d ⁻¹ for the seven sites.	48
11	Daily precipitation (P) , discharge (Q) , GPP , ER , and NEP in for WBMG 2013	51
12	(A) GPP resistance to storms vs relative change in discharge (in units of m ³ /s: m ³ /s) for all sites. (B) ER resistance to storms vs relative change in discharge (in units of m ³ /s: m ³ /s) for all sites	54

LIST OF FIGURES (Continued)

FIGURE

13	(A) Developed area vs site average GPP resistance to floods, (B) Developed area vs site average ER resistance to floods, (C) Impervious surface cover vs site average GPP resistance to floods, and (D) Impervious surface cover vs site average ER resistance to floods	56
14	Results of linear regression between GPP resistance (GPP_R) , ER resistance (ER_R) and storm flashiness index (RB)	57
15	<i>ER</i> vs <i>GPP</i> in all sites for non-storm days for: A) West Branch DuPage River, B) East Branch DuPage River, and C) Salt Creek. <i>ER</i> vs <i>GPP</i> in all sites for storm days for: D) West Branch DuPage River, E) East Branch DuPage River, and F) Salt Creek	59
16	Good Hope dam and study sites in Pennsylvania, USA. The reference station is near Lambs Gap Road, and is situated 4 km upstream of the dam, the upstream station on the Conodoguinet Creek and situated 35.1 m upstream of the dam, and the downstream station on the = Conodoguinet Creek and situated 38.6m downstream of the dam	72
17	Daily gross primary production (GPP) and ecosystem respiration (ER) at the upstream, downstream, and references sites.	81
18	GPP: ER ratio for the Good Hope dam 2 months before (A), 1 month before (B), 1 month after (C), and 2 months after the dam removal (D).	83
19	Expected outcome of the influence of urban intensity on gross primary production and ecosystem	93
20	The 18 Study Sites on Land Cover Map	95
21	The relationship between ULUG Index values and NEP to determine the threshold (i.e., ULUG index=23.88) value of the response trend	103
22	Mean daily GPP and ER in g O ₂ m ⁻² d ⁻¹ for the 18 sites color coded based on the ULUG index (i.e., sites with low urban intensity has ULUG index ≤ 23.88 , and sites with high urban intensity has ULUG index \geq 23.88	106
23	GPP versus ER for (A) sites with lower degree of urban intensity (ULUG index < 23.8) and (B) sites with higher degree of urban intensity (ULUG index > 23.8)	107

LIST OF FIGURES (Continued)

FIGURE

24	The relationship between: urban land use gradient index (ULUG) and GPP for (A) sites with low urban intensity (ULUG index < 23.8) and (B) sites with high urban intensity (ULUG index > 23.8), ULUG index and ER for (C) sites with ULUG index < 23.88 and (D) sites with ULUG index < 23.88 and (D) sites with ULUG index < 23.88 and (F) sites with ULUG index > 23.8	109
25	The relationship between: flashiness index (ULUG) and GPP for (A) sites with low urban intensity (ULUG index < 23.8) and (B) sites with high urban intensity (ULUG index > 23.8), FI index and ER for (C) sites with ULUG index < 23.88 and (D) sites with ULUG index > 23.8, and FI index and NEP for (E) sites with ULUG index < 23.88 and (F) sites with ULUG index > 23.8	110
26	GPP and ER resistance to floods vs R-B index for each flood event for all sites using the four definitions for resistance.	127
27	A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP , C) ER , and D) NEP in g O ₂ $m^{-2} d^{-1}$ for WBBR 2009	130
28	A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP , C) ER , and D) NEP in g O ₂ $m^{-2} d^{-1}$ for WBBR 2012, see Table1 for sites abbreviations.	131
29	A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP , C) ER , and D) NEP in g O ₂ $m^{-2} d^{-1}$ for WBBR 2013, see Table1 for sites abbreviations	132
30	A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP , C) ER , and D) NEP in g O ₂ $m^{-2} d^{-1}$ for WBMG 2009, see Table1 for sites abbreviations.	133
31	A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP , C) ER , and D) NEP in g O ₂ $m^{-2} d^{-1}$ for WBMG 2013, see Table1 for sites abbreviations	134
32	A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP , C) ER , and D) NEP in g O ₂ $m^{-2} d^{-1}$ for WBWD 2012, see Table1 for sites abbreviations.	135

LIST OF FIGURES (Continued)

FIGURE

33	A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP , C) ER , and D) NEP in g O ₂ $m^{-2} d^{-1}$ for WBWD 2013, see Table1 for sites abbreviations.	136
34	A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP, C) ER, and D) NEP in g O ₂ $m^{-2} d^{-1}$ for EBBR 2009, see Table1 for sites abbreviations.	137
35	A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP , C) ER , and D) NEP in g O ₂ $m^{-2} d^{-1}$ for EBHL 2010, see Table1 for sites abbreviations.	138
36	A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP, C) ER, and D) NEP in g O ₂ $m^{-2} d^{-1}$ for EBHL 2013, see Table1 for sites abbreviations.	139
37	A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP, C) ER, and D) NEP in g O ₂ $m^{-2} d^{-1}$ for SCYR 2009, see Table1 for sites abbreviations.	140
38	A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP, C) ER, and D) NEP in g O ₂ $m^{-2} d^{-1}$ for SCYR 2010, see Table1 for sites abbreviations.	141
39	A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP, C) ER, and D) NEP in g O ₂ $m^{-2} d^{-1}$ for SCBR 2010, see Table1 for sites abbreviations.	142
40	A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP, C) ER, and D) NEP in g O ₂ $m^{-2} d^{-1}$ for SCBR 2011, see Table1 for sites abbreviations.	143
41	A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP, C) ER, and D) NEP in g O ₂ $m^{-2} d^{-1}$ for SCBR 2013, see Table1 for sites abbreviations.	144
42	GPP and ER Reinvigoration at EBHL \ldots	146

SUMMARY

The main goal of this dissertation is to explore the influence of urbanization and dam removal on stream function represented by stream metabolism (i.e., gross primary production (GPP), ecosystem respiration (ER) and net ecosystem production (NEP)). Moreover, the research examines dam removal as a means of restoring stream metabolism. In a study of seven sites around the Chicago region during summer and fall of 2009-2013, results showed that stream GPP and ER decreased during storms, but on average ER show significantly greater resistance than GPP. After floods, both ER and GPP recovered to pre-flood levels within approximately 1-10 days, with no significant difference between GPP and ER resilience to floods. Results from this study demonstrate that GPP in urban streams is more susceptible to disturbance than the ER. Low production as a result of continued flood events can result in low oxygen levels in water and therefore can affect organisms that prefer specific DO levels.

This dissertation demonstrates that ecosystem metabolism changes significantly following dam removal. More specifically, dam removal increased GPP and ER levels at the upstream and downstream dam removal sites, immediately after removal of the dam. Away from the studied dam, a reference site showed only a small seasonal decline in GPP and ER across the same months. Metabolism in the upstream and downstream sections of the restored rivers shows similar GPP and ER rates and similar rates to a reference site following dam removal. Moreover, dam removal moves the streams towards autotrophy by increasing GPP more than

SUMMARY (Continued)

ER. Overall, I concluded that dam removal can restore aquatic ecosystems by enhancing metabolism rates even in non-production seasons (i.e., winter). The study suggests that dam removal should be considered as a preferable means of restoration for areas with low GPP and ER. Moreover, the dam removal timing can also play an important role on the behavior of metabolism response to dam removal (e.g., if the dam was removed in spring where algae bloom, GPP can increase to much higher levels). A future research should investigate multiple dam removal case studies to address the influence of seasonality on dam removal.

Finally, in a study of 50 watersheds across a gradient in urban intensity (18 sites located on 13 watersheds in the Midwestern US during summer/fall periods.), urbanization was quantified using the urban land use gradient index (ULUG) which is derived from infrastructure, land cover, and population variables, in addition to the flashiness index which is derived from normalized variation of daily flow regime. I showed that both GPP and ER decreased sharply (in absolute value) above 23.8 ULUG. Moreover, results concluded that urbanization is a major controlling factor on stream metabolism as it increases heterotrophy by reducing GPP more than ER. This study suggests that metabolism restoration projects should be targeted to high urban intensities. A future study can include much larger number of sites with different site characteristics (i.e., organic matter availability, canopy cover).

CHAPTER 1

INTRODUCTION

1.1 Motivation

Human activities cause multiple physical, chemical, and biological changes in urban streams, known as the "urban stream syndrome" (Walsh et al., 2005; Meyer et al., 2005). Physical changes in urban stream syndrome include altering channel geomorphology through bank incision and bed mobilization, which reduces geomorphic features (Doyle et al., 2000; Vietz et al., 2016). Chemical changes include increasing contaminants and nutrient levels transported from the adjacent landscape (Hatt et al., 2004; Bernhardt et al., 2008; Sudduth et al., 2011). Biological changes include reducing biodiversity by decreasing intolerant and increasing tolerant biotic species, and altering ecosystem processes (e.g. ecosystem productivity, leaf decomposition, and nutrient cycling) (Paul and Meyer, 2001).

Urban streams are characterized by "flashy hydrology" which is a primary reason for urban stream syndrome (Walsh et al., 2005). Flashy hydrology is associated with the occurrence of more frequent high flows due to increased impervious surface cover of the surrounding watershed that alters the quantity and quality of water delivered to urban streams (Walsh et al., 2005; Kaushal et al., 2008; Vietz et al., 2016). High flows are a well-known controlling factor on stream ecosystem structure and function (Poff et al., 1997; Reisinger et al., 2017). Stream functions are defined as: "The physical, chemical, and biological processes that occur in ecosystems" (Clean Water Act 33 CFR 332.2; 40 CFR 230.92). Stream functions can be divided into five levels (see Figure Figure 1): hydrology (transport of water from the watershed to the channel), hydraulics (water transport through the channel or the floodplain), geomorphology (transport of wood and sediment to create channel bed shape), physicochemical (the processing of nutrients and organic matter), and biology (biodiversity and aquatic and riparian life) (StreamMechanics, 2014). Although it is challenging to quantify relationships between flashy hydrology and stream function (Sudduth et al., 2011); there is a great interest in assessing these relationships with the hope of finding successful methods for improving stream functions in urban streams (Bunn and Davies, 2000; Young et al., 2008; Bunn et al., 2010). The proposed work seeks to quantify and evaluate stream function in urban streams with particular focus on the stream metabolism and the effectiveness of stream restoration projects.



Figure 1. Stream Functions Pyramid. (StreamMechanics, 2014)

Many restoration projects have been implemented to improve urban stream function and reduce flashy hydrology. Restoration projects include stream bank stabilization, channel reconfiguration, dam removal, and stream corridor restoration (Speed et al., 2016). Restoration projects seek to recover native aquatic organisms and return streams to pre-impacted conditions in order to improve overall stream health (Violin et al., 2011). Although these projects may enhance water quality and protect streams from bed erosion (Hobson Creek 2008), some projects further degraded stream function (Biggs et al., 1998; Bednarek, 2001; Colangelo and Jones, 2005; McKie et al., 2006). While many studies address the response of stream function to several examples of restoration projects (e.g., floodplain restoration, natural channel design, and riparian vegetation) (Hornbach et al., 2015; Roley et al., 2014; Sudduth et al., 2011), less research focused on the recovery of stream function following dam removal (Zhang, 2012), an objective of the proposed work.

Stream metabolism is an integrative metric of stream ecosystem processes (Hoellein et al. 2013) consisting of gross primary production (GPP), ecosystem respiration (ER) and net ecosystem production (NEP). GPP measures the amount of organic matter formed within the ecosystem. ER is the process of consuming energy by both autotrophs (producers) and heterotrophs (consumers) using allochthonous (internal) or autochthonous (external) energy sources. NEP is the difference between GPP and ER (Mulholland et al., 2001; Roberts et al., 2007a). Low levels of GPP and high levels of ER (negative NEP) at a stream indicates that a stream depends on external energy sources (from upstream or the terrestrial area). Therefore, NEP is considered an index of the importance of internal relative to external organic carbon input. Stream metabolism integrates all aerobic (oxygen requiring) organisms including autotrophs and heterotrophs at all habitat levels including planktonic, benthic and hyporheic zones (the zone where surface and groundwater interact). Hence, stream metabolism is an important metric for understanding ecosystem function including stream controls on food web and energy fluxes, ability to remove access nutrients, and response to human activities (Odum, 1956; Marcarelli et al., 2011; Hotchkiss et al., 2015).

1.2 Background Information

1.2.1 Primary Production in Streams

Primary production is the process of producing energy in the form of organic compounds (e.g., sugars or glucose) from inorganic compounds (e.g., water and C). The process of synthesizing organic carbon (carbon fixation) is called gross primary production (*GPP*), and this process typically takes place through photosynthesis (Equation Equation 1.1). The process of converting organic carbon to inorganic carbon (carbon mineralization) is called ecosystem respiration (*ER*) (Equation Equation 1.2) (Cullen, 2001).

$$CO_2 + H_2O + \text{light} \xrightarrow{\text{GPP}} CH_2O + O_2$$
 (1.1)

$$(CH_2O)_n + O_2 \xrightarrow{\text{ER}} CO_2 + H_2O + \text{heat}$$
 (1.2)

Primary production is the basis of biogeochemical cycling and ecological processes. Heterotrophic organisms (consumers) transfer the produced energy up the food web by consuming fixed carbon. GPP quantifies the biomass produced through photosynthesis in units of grams of CO_2 per unit area per unit time interval. Since CO_2 is found in many chemical forms in water that can exchange quickly, tracking the evolution of O_2 is a simpler method for measuring GPP(Cullen, 2001), and therefore GPP and ER are often expressed in units of grams of O_2 per unit area per unit time.

1.2.2 Stream Metabolism

Spatiotemporal variability in GPP and ER is driven by both proximal and distal factors (Bernot et al., 2010). Proximal factors influencing GPP include light availability, stream size, terrestrial input, and flow disturbance (Allan and Castillo, 2007). Proximal factors influencing ER include temperature, and labile organic matter (organic matter that breaks down quickly and is rich of nutrients) availability (Bott et al. 1985; Hill et al. 2000). Distal factors influencing both GPP and ER include climate, geology, riparian vegetation and land use (Bernot et al., 2010). In an inter-regional study to compare the influence of land use on stream metabolism, Bernot et al. (2010) calculated stream metabolism in 9 streams, 8 regions, with three land use categories: native vegetation, residential, and agricultural land uses. Results showed that GPP and ER differed significantly among regions and land use categories; forested streams had lower mean GPP compared to open canopy vegetated sites and 30% lower mean GPPthan agricultural and urban sites. All regions in their study had negative NEP (heterotrophic stream). The authors concluded that human activities and altering land use result in reducing regional changes in GPP.

The ratio of gross primary production to ecosystem respiration (GPP/ER) can be used to show the importance of internal versus external energy sources. When GPP/ER < 1, most of the energy supplied to the food web in the stream comes from outside the stream channel (allochthonous energy sources). While when GPP/ER > 1, more energy supporting the food web is produced within the stream channel (autochthonous energy sources). Stream ecosystems with very low GPP/ER are heterotrophic streams, while stream ecosystems with GPP/ER approaches 1 are autotrophic streams (Allan and Castillo, 2007). It is expected for streams to have a metabolic transition from hetrotrophy to autotrophy with distance from small streams to larger rivers, and this transition value is expected to occur when GPP/ER = 1. However, in a study along the Little Tennessee River in North Carolina, (McTammany et al., 2003) found that the GPP/ER increases with distance downstream and the transition value from heterotrophy to autotrophy was about GPP/ER = 0.78. Uehlinger et al. (1998) showed a strong effect of flow disturbances on GPP/ER. Immediately after storm, a decline in GPP/ER occurred as storms scoured algae and disturbed ecosystem production more than its respiration (Webster and Meyer, 1997; Uehlinger and Naegeli, 1998).

Previous studies found different responses of stream function to urbanization. Many studies showed an increase of GPP relative to ER in urban streams compared to forested streams (Iwata et al., 2007; Bernot et al., 2010). In contrast, other studies show no consistent relationship between urbanization and stream metabolism (Meyer et al., 2005; Von Schiller et al., 2008; Sudduth et al., 2011). Urbanization might enhance GPP and ER through the increase of light availability (Young et al., 2008), loss of canopy cover (Bunn et al., 1999; Fellows et al., 2006), and the increase of eutrophication (Rao et al., 1979; Izagirre et al., 2008). Therefore, predicting the effect of urbanization on stream metabolism is difficult because of the multiple factors that enhance or reduce metabolism metrics (Meyer et al., 2005).

1.2.3 Calculating Stream Metabolism from Diel Change in Dissolved Oxygen

1.2.3.1 Direct Measurements

Levels of dissolved oxygen (DO) in streams can be used as a proxy to quantify ecosystem metabolism. With the development and affordability of advanced DO sensors, open-channel estimations of DO have become the standard method for calculating stream metabolism (Hall and Hotchkiss, 2017). Odum (1956) developed the "whole-stream open channel method", which assumes that three main processes are responsible for altering DO signal in waters: GPP happens during day time, ER occurs during day and night, and that the diffusion of DO between air and water depends on the saturation level and the reaeration coefficient (K)(Izagirre et al., 2008).

Following (O'Connor and Di Toro, 1970), stream metabolism metrics can be calculated based on the 1-dimensional mass balance advection-diffusion-reaction equation of DO which is based on conservation laws, i.e a box model (Equation Equation 1.3).

$$\frac{\partial DO}{\partial t} = -U\frac{\partial DO}{\partial x} + K_x \frac{\partial^2 DO}{\partial x^2} + \underbrace{GPP(t) - ER(t)}_{NEP} + K(DO_s - DO)$$
(1.3)

where U is stream velocity, x is the distance, K is the reaeration rate, and DO_s is the DO concentration at saturation (The maximum DO that will dissolve in water at a given temperature and pressure under equilibrium). Further simplifications to the DO mass balance

in order to estimate stream metabolism are discussed below.

Assuming a uniform distribution of all biota with distance x, DO concentration will not vary spatially and equation (Equation 1.3) can be re-written as

$$\frac{\partial DO}{\partial t} = -U \frac{\partial DO}{\partial x} + K_x \frac{\partial^2 DO}{\partial x^2} + \underbrace{GPP(t) - ER(t)}_{NEP} + K(DO_s - DO)$$
(1.4)

1.2.3.2 Calculating the Reaeration Coefficient

While the whole-stream metabolism method remains the commonly used method for calculating stream metabolism, several different approaches are available to calculate K, a critically important parameter.(Izagirre et al., 2007; Riley and Dodds, 2012). Three main methodologies are used to calculate K: tracer methods, direct measurement of K from the DO diel curve, and empirical equations. Tracer methods, considered the most accurate for small turbulent streams (Marzolf et al., 1994), are conducted by constant-rate injection of a conservative tracer (e.g., dye) and a tracer gas at a specific site, then sampling water at multiple sites downstream to determine the decrease in tracer concentration (Rathbun et al., 1978). Unfortunately, tracer methods are expensive and therefore researchers have sought alternative methods to calculate K (Izagirre et al., 2007).

The main alternative to tracer methods is the direct measurements of K from the diel change in DO using the nighttime regression method. According to the nighttime regression method by (Hornberger and Kelly, 1975), at night, only ER and K are assumed to cause changes in DO, and therefore equation (Equation 1.4) can be re-written as the following:

$$-\frac{\partial DO}{\partial t} = \underline{GPP(t)}^{\bullet} - \underline{CR(t)} + K \underbrace{(DO_s - DO)}_{DO_{definit}}$$
(1.5)

By plotting the nighttime decrease in DO concentrations per unit time versus the oxygen saturation deficit, ER is the y-intercept and K would be the slope of regression (Figure Figure 2). The nighttime regression method fails in situations with high production rates or high K values when change oxygen concentrations vary little at night. To fill the gaps caused by these situations, K can be related to discharge, however this approach must be done for each site separately.



Figure 2. Nighttime Regression Method for Estimating K and ER (Izagirre et al., 2007).

A simple method to use equation (Equation 1.5) to calculate stream metabolism metrics is described in the following steps:

- 1. Calculate DO_s from empirical equations (See section Table XII).
- 2. Calculate the reaeration coefficient K and night time ER using the night time regression method.
- 3. Calculate GPP using equation (Equation 1.5).
- 4. Calculate NEP from the difference between GPP and ER.

The second alternative to the tracer methods is calculating K using empirical equations (see table A.1 in the appendices). Reaeration coefficient empirical equations are based on hydraulic variables such as: channel slope, Froude number, and water velocity (Genereux and Hemond, 1992) since stream velocity is related to aeration by stream rapids. The accuracy of empirical equations can be limited for stream conditions that are not consistent with that of the prototype stream. This is particularly the case for low order, oligotrophic and high gradient streams, which may produce error ranging from 40% to 125% (Demars et al., 2015).

1.2.3.3 Indirect Measurements

Inverse modeling involves two main steps: identifying a suitable model of DO dynamics (see equation Equation 1.6 as an example) and determining the best fit values for GPP, ERand sometimes K to represent the original DO measurements. Several models can be used for inverse modeling of stream metabolism. Hall and Hotchkiss (2017) used Bayesian parameter estimation or the maximum likelihood estimation (MLE) from Markov Chain Monte Carlo sampling (MCMC) to calculate GPP and ER as a function of DO according to the following equation, which is based on a forward Euler numerical solution for equation (Equation 1.4):

$$DO_{i} = DO_{(i-\Delta t)} + \left(\frac{GPP * PAR_{t}}{z \sum PAR}\right) - \frac{ER}{z}\Delta t + K(DO_{sat,(i-\Delta t)} - DO_{i-\Delta t})\Delta t$$
(1.6)

the parameter PAR (photosynthesis active radiation) represents the relative amount of light during day time, and z is water depth which is used to convert the units of GPP and ER from $gO_2L^{-1}d^{-1}$ to the units of $gO_2m^{-2}d^{-1}$). Figure (Figure 3) shows an example an inverse model of stream metabolism using the method of Hall et al. (2017).



Figure 3. Inverse Modeling Application, (Hall and Hotchkiss, 2017)

1.2.4 Methods to quantify urbanization

Urbanization can be quantified using the "Urban land-use gradient index" (ULUG) first proposed by (McMahon and Cuffney, 2000). This index defines the degree of urban intensity from low to high based on 53 basin variables. These variables include information on basin infrastructure, land cover, socioeconomic factors, and population density. The selected variables were then normalized for the basin area (i.e., variables with unit area were normalized to percentages of basin area, variables with unit length were normalized to kilometer per square kilometer, and variables with unit of count were normalized to counts/100 km²). The variables ranged from 0 to 1 based on range standardizing and the final value for the ULUG ranged from 0-100. The original purpose of the ULUG is to allow for sites selection that represent urban gradient for studies related to the influence of urbanization gradient on and in-stream water quality. This index was also used by (Coles et al., 2004) to determine the influence of urbanization on physical, chemical, and biological characteristics of New England coastal streams. Despite the detailed information about basin characteristics, the ULUG index ignored the percentage of impervious surface. This percentage can play a major role on defining the quantity and quality of water entering the adjacent streams and therefore, ecosystem function (McMahon and Cuffney, 2000).

"Flashy hydrology" is a common characteristic of urban streams which is associated with frequent high flood events because of increased impervious surface cover that changes quantity and quality of water delivered to urban streams (Walsh et al., 2005; Kaushal et al., 2008; Vietz et al., 2016). High discharge events are often a key controlling factor on stream ecosystems (Poff et al., 1997; Reisinger et al., 2017). Therefore, to quantify the effect of urbanization on stream metabolism, it is important to quantify variables that are related to stream flashiness. (Baker et al., 2004) proposed an index called "flashiness index" to represent change in flow regime based on mean daily mean flow data. Flashiness index is calculated by first calculating the pathlength of flow oscillations by summing the absolute values of changes in the mean daily flow. Then, the flashiness index is determined by dividing the pathlength by the total discharge for the entire time interval (equation Equation 1.7). (Baker et al., 2004) calculated the flashiness index for 515 Midwestern cropland streams for 27 years. Mean flashiness index for the 27 years ranged from 0 to 1.2.

$$R - B_{\text{index}} = \frac{\sum_{i=1}^{n} Q_i - Q_{i-1}}{\sum_{i=1}^{n} Q_i}$$
(1.7)

where Qi is the average daily flow in units of m³/s during day I and n is the number of days. The numerator on the right-hand side of equation 6 is analogous to the mathematical concept of the 'total variation' of a function and the denominator provides an appropriate normalization factor.

To quantify the effect of urbanization on stream metabolism, I will use the two indices (i.e., ULUG and flashiness index) as a representation to urbanization. Relationships between urbanization indices and metabolism metrics (i.e., GPP, ER, and NEP) will be created to investigate the effect of urbanization on stream metabolism.

1.2.5 The Influence of Restoration Projects on Stream Metabolism

Stream restoration projects aim to restore the ecological structure and function of river systems (Arango et al., 2015). Several stream restoration projects have been implemented to improve habitat diversity and water quality by diversification of in-stream habitat and channel modifications (Bernhardt et al., 2005). Although these projects often alter channel morphology and improve habitat heterogeneity, they may also result in no significant improvements (Bernhardt and Palmer, 2011) or further degradation of the stream ecosystem (Biggs et al., 1998; Bednarek, 2001; Colangelo and Jones, 2005; McKie et al., 2006). In general, very few studies assess the effectiveness of restoration projects in improving stream function (Violin et al., 2011). Furthermore, most studies focused on agricultural streams (Giling et al., 2013; Roley et al., 2014) and very few focused on urban streams. Moreover, metabolism calculations in urban streams are typically estimated over short-term periods (i.e., days to months). (Sudduth et al., 2011; Violin et al., 2011; Arango et al., 2015). Larger studies are needed since stream metabolism might change with seasons and across sites with different characteristics (i.e., availability of organic matter, change in canopy cover, temperature) (Uehlinger 2000, Bernote et al. 2010).

(Giling et al., 2013) assessed the effectiveness of small scale riparian replanting projects for improving stream metabolism in four degraded agricultural streams in southeastern Australia. Two-station whole-stream metabolism was calculated and compared between treated and untreated sites. Metabolism calculations were estimated once in early summer and for a longer period (6-16 days) in late autumn for the sites. Overall, the replanted reaches had higher ER and lower NEP values compared to the unplanted sites due to the higher input of organic matter from the new plants, and slightly lower GPP rates due to the small amount of new canopy cover compared to the available light intensity in the area. The strength of this study is the clear analysis of the spatial variation in stream metabolism. However, the study relies on short term calculations of stream metabolism, which may not capture the temporal variation of metabolism response to the restoration project especially in the summer period.

(Roley et al., 2014) used 5-years of continuous metabolism calculations at an agricultural stream to evaluate the effectiveness of flood plain restoration project in northern Indiana, USA. The project involved implementing 2-stage ditches (see figure Figure 4 for a conventional agricultural ditch with flood plain added) to improve channel stability during storms. GPP, ER, and NEP were estimated at upstream (unaltered) and downstream (restored) sites for one year before and four years after the restoration project to examine the effect of the restoration project on stream metabolism metrics. Results showed that floodplain restoration did not affect metabolism metrics during base flow, but during storms, GPP increased post-restoration due to increased stream width through the floodplain. GPP was more resilient to storms post-restoration than pre-restoration. The restoration had no effect on stream metabolism metabolism metrics.



Figure 4. Conventional agricultural ditch with floodplain added, (Roley et al., 2014)

For urban streams, (Violin et al., 2011) studied the effectiveness of natural channel design (NCD) restoration projects in urban stream at North Carolina. The project regraded and reshaped the channel, installing root wads to stabilize the banks, creating riffles by adding coarse to the bed material, and revegetating riparian areas. The study compared the biological and physical characteristics of four stations in a degraded urban reach, four sites in a restored urban reach, and four sites in a forested reach. The goal of this study was to evaluate the ability of NCD projects in restoring physical and biological structure of urban streams. The results showed that no clear differences between restored sites and degraded urban sites were different from forested sites. The authors explained the reason for this similarity between degraded and restored reaches is a time lag between the restoration project and community recovery and their sampling was implemented before sufficient period for habitat recovery. Therefore, (Violin et al., 2011) concluded that reach-scale restoration projects were not successful in improving habitat in urban streams, and they suggested that the time of the restoration project is an important metrics in assessing the effectiveness of the restoration projects

(Sudduth et al., 2011) compared stream metabolism across twelve different urban streams in North Carolina. They focused on the effectiveness of NCD restoration projects in urban areas. Four sites were selected in NCD restored urban streams, four were in non-restored urban streams, and four sites were located in minimally impacted forested streams. The study showed that the restoration project did not affect stream depth, habitat, or particle size in comparison to degraded sites. (Sudduth et al., 2011) found that temperature was the primarily controlling factor on both GPP and ER, this is because of the increased temperature due to the removal of riparian vegetation during implementing NCD projects. GPP, ER and NEP were measured once in summer and once in winter for this study. Furthermore, metabolism measurements were made during base flows, but storm flows can have a strong influence on stream metabolism, especially in areas with flashy hydrology (Beaulieu et al., 2013). Despite numerous studies, there is a lack of a comprehensive, long-term study that investigates stream ecosystem metabolism recovery period after storms in urban streams.

1.2.5.1 Dam Removal Restoration Projects

Dams provide recreation (in the form of new lakes), supply water, flood control and provide electricity (Heinz Center for Science, 2002). Due to concerns about lack of maintenance, age and risk of failure, many dams are considered for removal (Bednarek, 2001; Roberts et al., 2007a). Dam removal restores the original flow regime and thus reconnects riparian vegetation. However, removing dams may also increase the flow of sediment and contaminants from upstream (Hart et al., 2002; Gangloff, 2013). Physical and chemical impacts of dam removal can cause changes to several river characteristics: river morphology, flooding, sediment transport, erosion, wetlands, water quality (ICF, 2005).

In the past three decades, about 1,100 dams were removed in the United States and abroad (American Rivers, 2016). However, very few dam removal projects were monitored (USGS Science Database, 2016). The reason for the few number of dam removal monitoring studies is because of the long permitting processes or limited monitoring funding (Magilligan et al., 2016). Most monitoring studies investigate the effect of removing the dam on channel adjustment, sediment transport, or fish and habitat passage (Bushaw-Newton et al., 2002; Ashley et al., 2006; Whitener, 2013). Only one study examined the effect of dam removal on stream metabolism (Zhang, 2012).

Potential effects of dam removal on streams differ spatially and temporally, and can be directly related to metabolism parameters including: returning the natural flow regime, returning
natural temperature variation, increasing DO concentration, changing algal biomass and restoring natural riparian vegetation (Figure Figure 5) (Bushaw-Newton et al., 2002). The lack of continuous DO and temperature data before and after dam removal represents a substantial impediment to analyze the effect on dam removal on stream metabolism using whole-stream open channel method. The few existing studies monitor DO only one time each season, and for only one year, which does not allow investigation of the temporal factors changing DOconcentrations (e.g. seasonal variation, and changes due to flood events) (Wieferich, 2011). Zhang et al. (2012) predicted the influence of dam removal by modeling pre-removal DOmeasurements. They found that GPP at the upstream reach decreased by 30% following dam removal due to the combined sewer overflow influence. SAs part of the proposed work, I will provide the first investigation (to my knowledge) of the impact of dam removal on stream metabolism on continuous, multiyear time scale (see chapter 3).



Figure 5. An example of pre and post dam removal restored site, (Cooper, 2013)

1.3 Objectives

The research seeks to quantify the effect of urban land use and dam removal on stream metabolism. The overall goals of this project are to develop an understanding of the complex relationship between urbanization, flashy hydrology, and stream metabolism. The research seeks to answer the following questions:

1. What is the effect of urban land use intensity on stream ecosystem metabolism?

- 2. Do storm events in urban streams disturb stream metabolism metrics? If so, how quickly do they recover?
- 3. Do dam removal projects affect stream metabolism?

Urban land use produces a variety of effects on stream metabolism metrics (Figure Figure 6). Urban land use can increase light availability by reducing canopy cover and therefore increases GPP. Similarly, ER is expected to increase with water temperature and delivery of nutrients. Storm events often decrease GPP due to cloud cover (reduction in light availability) and scouring of algae (Figure Figure 6).



Figure 6. Expected Effects of Urbanization on Stream Metabolism

Removing dams increases downstream water velocity and sediment transport. Sites downstream of the dam removal are expected to experience reductions in GPP due to decreased light through turbidity and sediment load, in addition, ER is expected to increase due to nutrient supply from the dam impoundment or from upstream point source discharges (i.e., WWTP). Sites upstream of the dam removal are expected to experience reduction in GPP rates through increased flow velocity and scouring of algae, while ER is expected to increase due to the higher supply of nutrients. Dam removal may result in channel incision upstream and stream aggradation downstream (Chaplin et al., 2005; Ashley et al., 2006; Whitener, 2013). However, the stream may eventually return to an equilibrium where the stream bed will become stable, allowing for regrowth of algae and metabolism to improve. (Figure Figure 7).



Figure 7. Expected Effects of Dam Removal on Stream Metabolism

This dissertation objectives are:

• Assess the influence of storm events on stream metabolism in urban streams.

I calculated stream metabolism from measurements of dissolved oxygen at seven sites at a highly urbanized area in the Chicago region for the fall/summer periods between 2009-2014. In addition, I calculated metabolism resistance and resilience to storms following (Uehlinger, 2000).

- Analyze the effectiveness of dam removal restoration projects on urban stream function. I calculated stream metabolism before and after dam removal at the Good Hope dam (Pennsylvania). Metabolism metrics were calculated at multiple stations around the dam to determine the spatial variation in the effect of dam removal on stream metabolism.
- Analyze the influence of urban land use intensity on stream metabolism. Urbanization was quantified by calculating the Urban Land Use Gradient and flashiness indices for 18 sites along Indiana, Illinois, Wisconsin, Missouri, and Kansas. Then, those indices were related to *GPP*, *ER* and *NEP*. Depending on the value of ULUG index, the sites were grouped into two groups. *GPP* : *ER* ratio was compared between low to moderate urban intensity and high urban intensity.

1.4 Dissertation Overview

The remainder of this dissertation is composed of three self-contained research papers that have been or will be submitted to peer reviewed journals followed by conclusions. Consequently, there is some redundancy particularly with regard to the introductory material in each chapter.

Chapter 2 discusses the influence of flood events on metabolism of urban streams. It studies seven sites along three watersheds in DuPage Illinois: West Branch DuPage River, East Branch DuPage River, and Salt Creek over summer and fall of 2009-2014. Stream metabolism is calculated from hourly measurements of dissolved oxygen (DO), temperature, and discharge (Q) using the open channel whole-stream metabolism method. The effect of flood events is determined by calculating GPP and ER resistance and resilience. The results of this study show that flood events decreased GPP significantly more than ER and they both recovered within 1-10 days. This chapter shows that flood events sustain heterotrophic more than autotrophic conditions.

Chapter 3 examines the spatial and temporal effects of dam removal restoration project on stream metabolism. It focusses on the Good Hope dam on the Condonguinet Creek PA. Stream metabolism (i.e., GPP, ER and NEP) was calculated at three stations two months before and two months after the dam removal. The results of this chapter shows that the removal of a low head dam improved metabolism rates relatively rapidly, returning GPP and ER to reference conditions in scale and seasonality, and may thereby potentially facilitate recovery of other ecosystem parameters

Chapter 4 is an analysis to quantify the influence of urbanization on stream metabolism. I calculated stream metabolism from the open channel whole-stream metabolism method using the USGS StreamMetabolizer tool. I calculated stream metabolism in 18 sites located on 13 watersheds around Wisconsin, Illinois, Indiana, Missouri, and Kansas states, USA for six months in summer/fall periods. Metabolism results were compared to urbanization variables (i.e., flashiness index and urban land use gradient index) by grouping sites into low to intermediate and high urbanized sites. The findings of this chapter shows that urbanization is a major controlling factor on stream metabolism, and both GPP and ER declined sharply above 23.8 ULUG index. Overall, the results of this chapter indicated that increased urban intensity moves streams towards further heterotrophy by reducing GPP more than ER.

Finally, Chapter 5 presents the main conclusions, and discusses the state-of-the-art in metabolism of urban streams and prospects for future research.

CHAPTER 2

THE EFFECT OF FLOOD EVENTS ON ECOSYSTEM METABOLISM IN SUBURBAN STREAMS

2.1 Abstract

Urban and suburban streams are characterized by rapid changes in flow during flood events, which can affect ecosystem function and stream metabolism (i.e., gross primary production (GPP), ecosystem respiration (ER) and net ecosystem production (NEP)). We calculated stream metabolism from hourly measurements of dissolved oxygen (DO), temperature, and flow (Q) at seven sites around the Chicago region during summer and fall of 2009-2013. We examined the effect of flood events by calculating metrics that characterized GPP and ER for resistance (i.e., magnitude of change) and resilience (i.e., speed of recovery). Diel patterns in DO and GPP were observed during base flow conditions and flood events. Streams showed net heterotrophy at most of the sites, as GPP ranged from 0.98 - 6.61 g O₂ m⁻², ER ranged from -3.62 to -19.53 g O₂ m⁻², and NEP ranged from -16.84 to 1.06 g O₂ m⁻². Following floods, both GPP and ER decreased but ER exhibited a significantly higher resistance (i.e., changed less) than GPP. After floods, both ER and GPP recovered to pre-flood levels within approximately 1-10 days and there was no significant difference between ER and GPP resilience. Overall, our study indicates that large flow events sustain heterotrophic conditions more than autotrophic conditions. We conclude that flood events significantly affect stream metabolism and flood flashiness is a primary controlling factor on metabolism reduction. Our results will be valuable for stormwater management and stream restoration projects targeting ecosystem function.

2.2 Introduction

Human activities cause multiple physical, chemical, and biological changes in urban streams, Known as the "urban stream syndrome" (Walsh et al., 2005; Meyer et al., 2005). Physical changes in urban stream syndrome include altering stream geomorphology through channel reconfiguration, bank incision, and bed mobilization, which reduce structural complexity (Doyle et al., 2000; Vietz et al., 2016). Chemical changes include increasing contaminants and nutrient levels transported from the adjacent landscape (Hatt et al., 2004; Bernhardt et al., 2008; Sudduth et al., 2011). Biological changes include reducing biodiversity by decreasing intolerant and increasing tolerant biotic species, and altering ecosystem processes (e.g. ecosystem productivity, leaf decomposition, and nutrient cycling) (Paul and Meyer, 2001).

Urban streams are characterized by "flashy hydrology" which is a primary reason for urban stream syndrome (Walsh et al., 2005). Flashy hydrology is associated with more frequent high discharge events due to the increase of impervious surface cover that alters quantity and quality of water delivered to urban streams (Walsh et al., 2005; Kaushal et al., 2008; Vietz et al., 2016). High discharge events are often a key controlling factor on stream ecosystems (Poff et al., 1997; Reisinger et al., 2017). Alterations to the timing and magnitude may affect organisms with life history events connected to flooding patterns prior to watershed development (Poff et al., 1997).

Stream metabolism is an integrative metric of stream ecosystem processes (Hoellein et al., 2013). Stream metabolism consists of gross primary production (GPP), ecosystem respiration (ER) and net ecosystem production (NEP). GPP is the amount of organic matter formed within the ecosystem, ER is the process of consuming energy by both autotrophs and heterotrophs using allochthonous or autochthonous energy sources, and NEP is the difference between GPP and ER (Mulholland et al., 2001; Roberts et al., 2007a). Thus, NEP is considered an index of the importance of internal relative to external organic carbon input. Stream metabolism integrates all aerobic organisms including autotrophs and heterotrophs in addition to all habitat types including planktonic, benthic and hyporheic zones. Hence, stream metabolism is an important metrics for understanding ecosystem function including stream controls on food web and energy fluxes, nutrient cycling, and stream response to human activities (Odum, 1956; Marcarelli et al., 2011; Hotchkiss et al., 2015).

Previous studies show different responses of stream metabolism to urbanization. Many studies show increases in GPP relative to ER in urban streams compared to forested streams (Iwata et al., 2007; Bernot et al., 2010). In contrast, others showed no consistent relationship between urbanization and stream metabolism (Meyer et al., 2005; Von Schiller et al., 2008; Sudduth et al., 2011). (Izagirre et al., 2008) found that GPP increases with the increase of nutrient loading and that ER increases with the increase of biological oxygen (O_2) demand (BOD). Urbanization might enhance GPP and ER through the increase in light (Bunn et al., 1999; Young et al., 2008), due to loss of canopy cover (Bunn et al., 1999; Fellows et al., 2006), and eutrophication (Rao et al., 1979; Izagirre et al., 2008). However, urbanization might disturb GPP and ER by scouring algae during storms and reducing benthic storage of organic matter (Meyer et al., 2005). Therefore, predicting the effect of urbanization on stream metabolism is difficult because of the different factors that enhance or reduce metabolism metrics.

Flooding can affect both GPP and ER (Fisher et al., 1982), although the rate of recovery often differs between the two metrics. GPP recovers according to the rates of regrowth for primary producers. However, ER may recover more quickly because heterotrophic communities are not typically scoured from surfaces during floods in the same way as autotrophs but are instead transported along with organic matter (Uehlinger 2000). Stream metabolism in natural, undeveloped areas is expected to be more resistant (i.e., less change before and after disturbance) and resilient (i.e., faster recovery after disturbance) to floods than flashy, urbanized streams (Reisinger et al. 2017). Reisinger et al. (2017) quantified the response of stream metabolism to flood events in urban steams in Baltimore, Maryland across a range of flood magnitudes (from 0.77 to 2.75 years). In that study, GPP was less resistant to flood events than ER (during floods, GPP and ER declined 80% and 66% on average, respectively). However, GPP and ER showed equivalent resilience of 4-18 days. Previous studies on the response of stream metabolism to urbanization and flashy hydrology were limited to few number of flood events per site (Reisinger et al., 2017). Moreover, very few studies (Larsen and Harvey 2017) investigated the resistance and resilience of urban stream metabolism across multiple years or quantified stream flashiness and its influence on stream metabolism.

In this study, we calculated stream metabolism in three urban streams located in DuPage County, Illinois, USA in summer-fall months between 2009 and 2014. Moreover, we examined the response of stream metabolism to storms by calculating metabolism resistance and resilience to storms for all the available storms in all the sites. We hypothesized that GPP would have less resistance and resilience than ER, based on the scouring action of storms on primary producer communities. (Figure Figure 8).



Figure 8. Predictions for resistance (i.e., magnitude of change) and resilience (i.e., speed of recovery) for gross primary production (GPP) and ecosystem respiration (ER) following a storm in an urban stream.

2.3 Study Area

Our study focused on urbanized watersheds in the Chicago Metropolitan Area. Sites were selected based on the availability of DO, temperature, and flow data. Suitable sites include

the three main tributaries of the Des Plaines River: Salt Creek, East Branch DuPage River, and West Branch DuPage River (EPA, 2004; EPA, 2008) (Figure 9). We studied two sites at Salt Creek: Butterfield Road (SCBR, 2.5 km downstream of USGS 05531300) and at York Road (SCYR, 3.7 km upstream of USGS 05531500). We studied two sites on the East Branch: Hidden Lake (EBHL, 0.7 km downstream of USGS 05540160) and Butterfield Road (EBBR, in the same location of USGS 05540160). Finally, we investigated three sites on the West Branch: Butterfield Road (WBBR, 1.27 km upstream of USGS 05540095), McDowell Grove (WBMG, 3 km downstream of USGS 05540095), and at the Warrenville Grove dam (WBWD, 0.68 km upstream of USGS 05540095). Although WBBR and WBWD are close, WBWD is located in a wetland and immediately upstream of a dam that was removed in 2011, which may have affected the metabolism pattern (2, Table 1). Watershed areas ranged from 209 to 393 km² with a combined area of approximately 600 km² in Cook and DuPage Counties. The developed areas within the watershed are primarily residential (Figure 9), and annual mean flows range from 3.1 to 4.4 m³/s based on data for 1946-2004 (Illinois 2004). Impervious surface cover ranged from 32.4% to 39.4%. Percent of developed area ranged from 80.6% to 95.4% (Figure 9, Table I).

Wastewater treatment plants that discharge effluent into these streams include 11 publicly-owned treatment works (POTWs) and 6 combined sewer overflows (CSOs) at Salt Creek, 11 POTWs and 1 CSO at the East Branch DuPage River, and 7 POTWs at the West Branch DuPage River. The three watersheds are characterized by extensive impervious surface cover and flooding from combined sewers and overland flow (Illinois 2004).



Figure 9. Study site locations are in northeastern Illinois, USA, in the Chicago metropolitan region. WBBR = West Branch DuPage River at Butterfield Road, WBMG = West Branch DuPage River at McDowell Grove, WBWD = West Branch DuPage River at Warrenville Grove dam, EBHL = East Branch DuPage River at Hidden Lake, EBBR = East Branch DuPage River Butterfield Road, SCBR Salt Creek at Butterfield Road, and SCYR = Salt

Continuous hourly data of *DO* and temperature were provided by the DuPage River and Salt Creek Workgroup (DRSCW). Probes were the HACH DS5X equipped with a luminescent dissolved oxygen probe. Sensors were calibrated every 4 weeks. Data were screened for low (<300 micro siemens) conductivity, which indicates the sonde was exposed to air, and data not meeting guidelines were removed. Flow data at each site were adjusted from the closest USGS station using the drainage area weighted discharge approach, with drainage area collected from USGS StreamStats (https://water.usgs.gov/osw/streamstats/). Water depth was calculated using Manning's equation with channel characteristics (i.e., channel cross section, slope, Manning's roughness) obtained from the DuPage River Storm Water Division. Seven suitable sites were selected during summer and fall months between 2009 and 2013 (Figure 9, Table I).

Station	Watershed	USGS station	Drainage area [km ²]	Distance from USGS gauge [km]	Developed area (%)	ISC area (%)	R-B Index
WBBR	West Branch - Butterfield Road	05540095	232.9	1.27 upstream	82.3	32.8	0.39
WBMG	West Branch - McDowell Grove	05540095	270.2	0.68 upstream	80.6	32.4	0.39
WBWD	West Branch - Warrenville Grove dam	05540095	234.2	3 downstream	81.8	32.6	0.39
EBHL	East Branch - Hidden Lake	05540160	80.2	0.7 downstream	91.6	37.2	0.33
EBBR	East Branch - Butterfield Road	05540160	65.4	0	95.4	39	0.33
SCYR	Salt Creek - York Road	05531500	295.5	3.7 upstream	93.8	39.4	0.29
SCBR	Salt Creek - Butterfield Road	05531300	258.0	2.5 downstream	93.6	39.3	0.27

TABLE I

SITES NAMES, LOCATIONS, AREA , THE CLOSEST USGS STATION, DRAINAGE AREA KM², SITE DISTANCE FROM USGS STATION KM, RELATIVE AMOUNT OF WATERSHED THAT IS DEVELOPED AREA, AND IMPERVIOUS SURFACE COVER (ISC), AND R-B FLASHINESS INDEX CALCULATED FROM THE ADJUSTED DISCHARGE DATA AT EACH SITE.

2.4 Methods

2.4.1 Whole-stream metabolism

Stream metabolism can be calculated based on the *DO* mass balance equation following (O'Connor and Di Toro, 1970)

$$\frac{\partial DO}{\partial t} = -U \frac{\partial DO}{\partial x} + \underbrace{GPP(t) - ER(t)}_{NEP} + K \underbrace{(DO_s - DO)}_{DO_{\text{deficit}}}$$
(2.1)

where U is the stream velocity, x is the distance, K is the reaeration rate, and DO_s is the DO concentration at saturation. Assuming a uniform distribution of plants with distance x, DO concentration will not vary spatially and equation (Equation 3.1) can be re-written as

$$\frac{\partial DO}{\partial t} = \underbrace{GPP(t) - ER(t)}_{NEP} + K \underbrace{(DO_s - DO)}_{DO_{\text{deficit}}}$$
(2.2)

Calculations of whole-stream metabolism were performed using RIVERMET (Izagirre et al., 2007). The software follows the (Odum, 1956) method for a single station open-channel calculation. The main input for this software are continuous data of DO(mg/L), temperature (Kelvin), discharge (m^3/s) , water depth (m), and the times of sunrise and sunset.

RIVERMET first calculates the DO_s following the American Public Health Association (APHA) (1992) using the following equation

$$\ln(DO_s) = -139.34411 + (1.575701 * \frac{10^5}{T}) - (6.642308 * \frac{10^7}{T^2}) + (1.243800 * \frac{10^{10}}{T^3}) - (8.621949 * \frac{10^{11}}{T^4})$$
(2.3)

We use the nighttime regression method to calculate K and ER by analyzing the nighttime drop in DO (Hornberger and Kelly, 1975). GPP occurs only during daylight, so ER and reaeration are the only factors that alter DO concentration at night. Examining the previous equation with GPP set to zero at night, the DO mass balance equation forms linear equation (e.g., y = ax + b) for the nighttime decrease in DO per unit time $\left(\frac{\partial DO}{\partial t}\right)$ in terms of the DOdeficit (i.e., $DO_s - DO$) times K (the slope) and ER (the y-intercept). Thus, we calculated a daily value of K from a linear regression to the nighttime decrease in DO per hour $\left(\frac{\partial DO}{\partial t}\right)$ and DO deficit (i.e., $DO_s - DO$). For days where the nighttime regression method failed (i.e., the relationship between $\left(\frac{\partial DO}{\partial t}\right)$ and DO deficit is insignificant because DO varied little at night, or K was very large), we obtained K using an empirical relationship between the observed Qand K (for each site).

The night-time regression method calculates K at 20 °C. To correct K for variations in stream temperature, the following equation was used, where θ is 1.0241 (Thyssen, 1982):

$$K_T = K_{20^{\circ}C}(\theta^{T-20}) \tag{2.4}$$

ER was corrected for temperature using the following equation, where θ is 1.07 (Erlandsen and Thyssen, 1983):

$$ER_T = ER_{20^{\circ}C}(\theta^{T-20}) \tag{2.5}$$

Finally, GPP was calculated based on the hourly time step of the input data once the other terms are known. Daily GPP was calculated by adding all hourly GPP during daylight.

Daily NEP was calculated from the difference between daily ER and daily GPP. RIVERMET reports GPP, ER and NEP results in units of $(mg \ O_2 \ L^{-1} \ d^{-1})$. Thus, we multiplied the metabolism metrics by the water depth, inferred from the nearest USGS station, in order to convert volume-based units $(mg \ O_2 \ L^{-1} \ d^{-1})$, used in RIVERMET, to areal units of $g \ O_2 \ L^{-1} \ d^{-1}$.

The relationship between GPP and ER was determined by calculating the ratio GPP/ER; where if GPP/ER is > 1, then the stream was autotrophic and if GPP/ER is < 1, then the stream was heterotrophic. We defined $GPP_{sumnmer}$ or ER_{summer} as the average daily GPP or ER for June, July, and August. Similarly, we defined GPP_{fall} or ER_{fall} as the average daily GPP or ER for September, and October.

2.4.2 Defining flood events and quantifying stream flashiness

Flood events were defined as local maxima in the daily flow data using the 'findpeaks' function in the Signal Processing Toolbox of MATLAB, which returns all local maxima (i.e., peaks) exceeding a threshold selected as twice the average flow for each season. Due to the variation in stream flow during seasons, we used the 'findpeaks' function to detect the floods relative to the average seasonal flow (i.e., summer and fall) for each site separately.

We calculated the flashiness index for each flood events by modifying the approach of Baker et al. (2004). Flood flashiness index is calculated by normalizing the variation in the average daily flow according to the following equation:

$$R - B_j = \frac{\sum_{i=j-2}^{n=j+4} |Q_i - Q_{i-1}|}{Q_{\text{week mean}}}$$
(2.6)

were j is the flood event, Q_i is the average daily flow in units of m³/s during day i and n is the number of days which was selected based on a visual analysis of each flood event. The numerator on the right-hand side of equation Equation 2.6 is analogous to the mathematical concept of the 'total variation' of a function and the denominator provides an appropriate normalization factor.

2.4.3 Metabolism resilience and resistance to storms

Multiple definition for metabolism resistance to storms were tried in this study (See Resistance definitions in the appendix section .1) and based on the results, we defined resistance (R) as the ratio between pre-flood and at- flood values modified from (Uehlinger 2006), which was written as

$$R = \frac{X_{\text{at-storm}}}{X_{\text{pre-storm}}}$$
(2.7)

where X represents either GPP or ER. We defined the pre-flood values as as the last measurement before the flood. Post-flood values were set as the first measurement after flood. Situations where no data before or after the flood were excluded from the analysis.

We quantified metabolism resilience to storms (i.e., speed of recovery) as the number of days for metabolism metrics to return to pre-storm levels (Uehlinger, 2000). To conduct the resilience analysis, local minima, i.e., GPP_{min} and ER_{min} at most of the flood events, were found using the MATLAB 'findpeaks' function in the Signal Processing Toolbox. Linear recovery trends (with slopes; s) beginning at GPP_{min} and ER_{min} were then fit to the daily post-flood GPPand ER data. We defined pre-flood values as the last measurement before the flood. Finally, recovery periods were estimated from the time between the post-flood minimum and the preflood value according to the following equation

$$RP = (X_{\text{mean}} - X_{\text{min}})/s \tag{2.8}$$

where X represents either GPP or ER. Flood events were excluded from the analysis when GPP or ER values did not recover to the pre- flood value (i.e., the mean value of GPP or ER) after several weeks. The resilience analysis was restricted to flood events with adequate data availability for both pre- and post-flood conditions as some gaps in the DO or flow data exist.

2.4.4 Statistical analysis

To assess the differences in metabolism results among sites and to account for the correlation in GPP and ER between nearby sites, we used an "ARIMA" (auto-regressive integrated moving average) model (Box and Jenkins 1970). ARIMA models are composed of three sets of parameters: autoregressive parameters (AR, p), which check for relationships between the variable of interest at time (t) and the same variable at previous time step, stationarity parameters (I, D), which is used when there is no constant mean and variance in the response variable, and moving average parameters (MA, q), which check for relationships between the response variable at time t and the residual of the previous time step. If the response variable is not stationary, the data need to be differenced using 1st or 2nd order differencing, D=1or 2, respectively (Cryer and Chan 2008). To ensure that we accounted for autocorrelation, we plotted the residuals as a timeseries using the autocorrelation function and the partial autocorrelation function that detects the existence of autocorrelation (Zuur et al. 2009). The best model fit was selected based on the Akaike's Information Criterion AIC (Akaike 1973). Including a seasonal component resulted in poorer model fits.

We also used simple regressions to compare metabolism metrics with watershed impervious surface cover and developed area. The influence of stream flashiness on metabolism resistance and resilience to floods was assessed using linear regressions between GPP and ER resistance to floods vs. R - B for all flood events for all sites combined and for each site individually. To determine the influence of flood events on the relationship between GPP and ER, we compared regression lines between GPP and ER for flood days and for non-flood days. ARIMA and linear regressions were computed with MATLAB and the Statistics Toolbox (Release 2016b, The MathWorks Inc., Illinois).

2.5 Results

2.5.1 Whole-stream metabolism

The night-time regression method was successful 95% of the nights at WBBR (West Branch at Butterfield Road) during 2013 (the highest rate of success), and 22% of the nights at SCBR (Salt Creek at Butterfield Road) during 2009 (the lowest success rate). Occasionally, the nighttime regression method yielded poor results during nights with very small diel change in DO. Regressions between K and Q yielded significant relationship at all the sites (see Table XIII in the appendix). Dates where nighttime regression failed were filled with the empirical relationship between K and Q.

The best-fit ARIMA model for GPP had the same structure at most of the sites (1,1,1) except for WBMG and SCBR (Table XIV in the appendix) indicating that those two sites are influenced by different physicochemical factors. Among the rest of the sites, SCYR had a different intercept indicating a significantly lower GPP compared to the sites with the same model structure. The best-fit ARIMA model for ER had the same structure (1,1,1) at all the sites however, SCBR had a different intercept than the rest of the sites indicating a different magnitude of ER (Table XIV in the appendix).

Mean daily GPP ranged from 0.98 $gO_2m^{-2}d^{-1}$ at SCBR to 6.61 $gO_2m^{-2}d^{-1}$ at WBWD (Figure 22, Table X). ER had a wider range of values than GPP (from -3.7 $gO_2m^{-2}d^{-1}$ at EBHL to -19.53 $gO_2m^{-2}d^{-1}$ at SCBR) (Figure 22, Table X). Mean daily NEP ranged from -16.84 $gO_2m^{-2}d^{-1}$ at SCBR to 1.06 $gO_2m^{-2}d^{-1}$ at WBMG (Figure 22, Table X). During summer, metabolic activity was high in all the sites (Table 2), where mean daily summer GPP ranged from 1.13 $gO_2m^{-2}d^{-1}$ at SCBR to 8.51 $gO_2m^{-2}d^{-1}$ at WBWD and mean daily summer ER ranged from -4.7 $gO_2m^{-2}d^{-1}$ at EBHL to -22.28 $gO_2m^{-2}d^{-1}$ at SCBR. Metabolic activity was lower during the fall season, GPP ranged from 0.47 $gO_2m^{-2}d^{-1}$ at SCBR to 5.08 $gO_2m^{-2}d^{-1}$ at WBMG. ER ranged from -2.38 $gO_2m^{-2}d^{-1}$ at EBHL to -15.71 $gO_2m^{-2}d^{-1}$ at SCBR (Figure 22, Table X).

Sites -	Summer period					Fall period			Whole study period			
	GPP	ER	NEP	n	GPP	ER	NEP	n	GPP	ER	NEP	n
WBBR	5.63 ± 3.77	$\textbf{-8.16} \pm \textbf{7.29}$	-2.2 ± 6.97	199	3.25 ± 1.5	-4.41 ± 2.71	$\textbf{-0.86} \pm 2.1$	94	5.08 ± 3.41	-7.23 ± 6.57	$\textbf{-1.83}\pm6.09$	293
WBMG	5.15 ± 2.64	$\textbf{-4.46} \pm 2.11$	1.05 ± 2.78	151	5.08 ± 3.18	$\textbf{-4.48} \pm \textbf{2.89}$	1.09 ± 3.01	44	5.14 ± 2.77	$\textbf{-4.46} \pm 2.3$	1.06 ± 2.82	199
WBWD	$\textbf{8.51} \pm \textbf{3.76}$	$\textbf{-9.34} \pm 5.49$	$\textbf{-0.23} \pm \textbf{4.56}$	88	3.61 ± 2.85	$\textbf{-9.75} \pm 5.64$	$\textbf{-5.86} \pm 7.45$	67	$\textbf{6.61} \pm \textbf{4.18}$	$\textbf{-9.5} \pm 5.54$	$\textbf{-2.41} \pm \textbf{6.45}$	196
EBBR	2.68 ± 1.4	$\textbf{-4.07} \pm 1.22$	-1.23 ± 1	201	$\textbf{0.93} \pm \textbf{0.44}$	$\textbf{-2.38} \pm 0.42$	$\textbf{-1.37} \pm 0.55$	166	$\textbf{2.21} \pm \textbf{1.44}$	$\textbf{-3.62}\pm1.3$	$\textbf{-1.27}\pm0.9$	117
EBHL	6.47 ± 4.77	$\textbf{-10.56} \pm \textbf{9.62}$	$\textbf{-3.74} \pm \textbf{9.63}$	87	2.44 ± 1.42	$\textbf{-5.02} \pm \textbf{2.56}$	$\textbf{-2.38} \pm 2.18$	33	$\textbf{4.97} \pm \textbf{4.34}$	$\textbf{-8.5}\pm\textbf{8.22}$	$\textbf{-3.24} \pm 7.77$	358
SCBR	3.32 ± 2.63	$\textbf{-22.28} \pm 15.24$	$\textbf{-18.84} \pm \textbf{14.93}$	190	1.55 ± 1.1	-15.71 ± 12.51	$\textbf{-14.07} \pm 12.46$	96	2.59 ± 2.3	$\textbf{-19.53} \pm \textbf{14.49}$	$\textbf{-16.84} \pm \textbf{14.1}$	248
SCYR	1.13 ± 1.01	$\textbf{-9.91} \pm \textbf{8.96}$	$\textbf{-8.71} \pm \textbf{8.45}$	106	0.47 ± 0.22	$\textbf{-3.92} \pm 1.72$	$\textbf{-3.41} \pm 1.73$	223	$\textbf{0.98} \pm \textbf{0.94}$	$\textbf{-8.54} \pm \textbf{8.3}$	$\textbf{-7.49} \pm 7.79$	275

TABLE II

MEAN (± STD. DEV.) DAILY METABOLISM (G ${\rm O}_2~{\rm M}^2{\rm D}^{-1})$ FOR SUMMER PERIOD, FALL PERIOD, AND WHOLE STUDY PERIOD

Stream metabolism was net heterotrophic (i.e., GPP/ER < 1) for most of the observations in 2 out of the 3 streams. Results showed net autotrophy did not occur on any dates for SCYR (0 of 248 d) but did occur at 26% of the metabolism results at WBBR (76 of 293 d). In contrast, WBMG was autotrophic for 74% of the measurements (138 of 199) and WBWD was autotrophic for 48% of the measurements (93 of 196) (Table Figure 22).



Figure 10. Mean (\pm Std. dev.) monthly *GPP* and *ER* results in g O₂ m⁻² d⁻¹ for the seven

sites.

2.5.2 Flow patterns and metabolism signal

All the sites were characterized by frequent flow peaks (Figure 11). Seasonally-averaged flow Q ranged from 1.18 m³/s at EBBR to 6.94 m³/s at SCBR. The number of flood days (i.e., $Q > 2Q_{average}$) ranged from 9 at EBBR to 33 at EBHL. The maximum daily flow ranged from 6.09 m³/s at EBBR, which is about 105% greater than its average daily Q, to 82.84 m³/s at SCBR which is about 181% greater than its average daily Q (Table III). Mean R-B index values for studied flood events ranged from 0.25 at SCBR to 0.48 at EBHL (Table III).

Sites	Qaverage	Q _{max}	Flood days	R-B index	
Siles	m ³ /s	m³/s	(Q>2Qaverage)	$(m^3 s^{-1}/m^3 s^{-1})$	
WBBR	12.15	27.52	27	$\textbf{0.44} \pm \textbf{0.18}$	
WBMG	2.67	25.06	17	0.34 ± 0.14	
WBWD	1.92	10.52	17	0.32 ± 0.13	
EBBR	1.18	6.09	9	0.42 ± 0.07	
EBHL	1.47	14.75	33	$\textbf{0.48} \pm \textbf{0.13}$	
SCBR	3.93	31.11	42	0.25 ± 0.09	
SCYR	6.94	82.84	31	0.35 ± 0.13	

TABLE III

AVERAGE FLOW ($Q_A VERAGE$) IN M³/S, MAXIMUM FLOW (Q_{MAX}) IN M³/S, THE NUMBER OF DAYS WHERE $Q > 2Q_{AVERAGE}$, AND THE FLASHINESS INDEX REPRESENTED BY MEAN AND (±) ST.D. FOR STORMS STUDIED AT EACH SITE



Figure 11. Daily precipitation (P), discharge (Q), GPP, ER, and NEP in for WBMG 2013. Refer to Appendix A.3 for similar figures for all sites and all years.

2.5.3 Metabolism resistance and resilience

In general, our results showed that flood events decreased GPP more than ER. ER resistance to floods was significantly higher than GPP resistance when all sites were considered

together (paired t-test: p = 0.014, n = 65; Table IV). *GPP* resistance ranged from 0.01 at SCBR to 3.3 at WBMG, while *ER* resistance ranged from 0.59 at WBBR to 4.76 at EBHL. The number of flood events used to calculate metabolism resistance for both GPP and ER ranged from 4 at SCBR and EBBR to 15 at WBBR. Flood events frequently simulated *ER*; more than 80% of the floods increased *ER* levels (in absolute value) following the flood event compared to pre-flood values. On the other hand, less than 40% of the flood events increased *GPP* levels following the flood events compared to pre-flood values (Table IV).

Unlike the patterns of resistance, ER resilience was not significantly different from GPP resilience when all data were combined or when sites considered separately (paired t-test, p= 0.451; Table IV). GPP recovery period ranged from 0.92 d at SCYR to 9.54 d at EBHL, and ER recovery period ranged from 1.58 d at SCBR to 5.35 d at EBBR (Table IV).

Sites	Resistance				flood stimulation			
	GPP	ER	р	GPP	ER	р	GPP	ER
WBBR	0.94 ± 0.44	1.49± 0.81	0.021	3.83 ± 3.37	1.68 ± 1.02	0.262	7/15	12/15
WBMG	$1.25{\pm}~0.97$	1.68 ± 0.84	0.159	6.65 ± 6.25	$\textbf{3.67} \pm \textbf{5.08}$	0.829	5/10	7/10
WBWD	$0.99{\pm}~0.69$	1.8 ± 0.97	0.042	$\textbf{4.49} \pm \textbf{3.05}$	$\textbf{1.97} \pm \textbf{1.29}$	0.557	4/12	10/13
EBBR	$0.75{\pm}0.19$	1.1 ± 0.23	0.015	1.95 ± 2.02	5.35 ± 6.2	0.563	0/4	3/4
EBHL	$0.89{\pm}~0.43$	2.43 ± 1.44	0.004	9.54 ± 2.34	$\textbf{2.58} \pm \textbf{15.99}$	0.383	3/13	11/15
SCBR	2.2 ± 2.52	$2.02{\pm}~1.37$	0.884	2.29 ± 0.2	1.58 ± 1.09	0.387	5/7	6/8
SCYR	2.2 ± 2.52	$2.02{\pm}~1.37$	0.582	0.92*	2.62*	-	2/4	4/4
All sites	1.23 ± 1.41	1.8 ± 1.06	0.014	$\textbf{4.01} \pm \textbf{3.51}$	$\textbf{3.88} \pm \textbf{7.95}$	0.151	26/65	53/65

*no error bars indicate that only one flood event was analyzed

TABLE IV

RESISTANCE OF GPP AND ER TO FLOODS BY SITE REPRESENTED AS THE MEAN (±) ST.D., RESULTS OF T-TEST TO COMPARE DIFFERENCE BETWEEN GPP AND ER RESISTANCE, RESILIENCE OF GPP AND ER TO FLOODS IN DAYS REPRESENTED AS THE MEAN (±) ST.D., AND RESULTS OF T-TEST TO COMPARE DIFFERENCE BETWEEN GPP AND ER RESILIENCE. FLOODS STIMULATION IS THE NUMBER OF TIMES FLOODS STIMULATED GPP OR ER TO STUDIED FLOODS TOTAL NUMBER OF (INCREASED AFTER THE FLOOD).



Figure 12. (A) GPP resistance to storms vs relative change in discharge (in units of m³/s: m³/s) for all sites. (B) ER resistance to storms vs relative change in discharge (in units of m³/s: m³/s) for all sites.

When all sites were considered together, there was a notable relationship between metabolism and R-B index (Figure 12). GPP resistance decreased significantly with the increase of R-B index of flood ($R^2=0.017$, p= 0.035; Figure 12A; Figure 14). On the other hand, ERresistance increased significantly with the increase in R-B index of each flood event ($R^2=0.14$, p; 0.01; Figure 12B; Figure 14). GPP showed no relationship to watershed developed land area and impervious surface cover (GPP and percent developed area: $R^2=0.006$, p=0.804; GPPand percent of impervious surface: $R^2=0.003$, p=0.913; Figure 13A, B). The same pattern was found for ER (ER and percent developed area: $R^2=0.006$, p=0.873; ER and percent impervious surface: $R^2=0.001$, p=0.763; Figure 13C, D).



Figure 13. (A) Developed area vs site average GPP resistance to floods, (B) Developed areavs site average ER resistance to floods, (C) Impervious surface cover vs site average GPPresistance to floods, and (D) Impervious surface cover vs site average ER resistance to floods.
Sites	Linear regression (R-B and GPP_R)	р	r ²	Linear regression (R-B and ER_R)	р	r ²	n
WBBR	$GPP_{R} = -0.31RB + 1.07$	0.651	0.016	$ER_{R} = 0.96RB + 1.10$	0.455	0.043	15
WBMG	$GPP_R = 2.738RB + 0.31$	0.246	0.164	$ER_{R} = 4.61RB + 0.09$	<0.01	0.634	10
WBWD	GPP _R = -1.38RB +1.39	0.431	0.068	ER _R = 5.07RB+0.18	<0.01	0.537	13
EBBR	$GPP_R = -1.41RB + 1.33$	0.418	0.339	ER _R = -0.85RB +1.45	0.697	0.095	5
EBHL	$GPP_{R} = 1.33RB + 0.35$	0.331	0.136	$ER_{R} = 5.81RB - 0.34$	0.113	0.213	15
SCBR	$GPP_{R} = -19.3RB + 6.76$	0.081	0.487	ER_{R} = -0.36 RB + 2.07	0.971	0.001	8
SCYR	$GPP_{R} = -9.88RB + 5.10$	0.186	0.660	ER_{R} = -8.32RB +4.41	0.102	0.801	4
All sites	$GPP_{R} = -2.43 RB + 2.13$	0.035	0.068	$ER_{R} = 2.64RB + 0.81$	<0.01	0.143	70

Figure 14. Results of linear regression between GPP resistance (GPP_R) , ER resistance

 (ER_R) and storm flashiness index (RB).

2.5.4 Factors influencing metabolism in urban streams

To investigate the influence of flood events on the relative magnitude and relationship between stream metabolism metrics, we plotted ER versus GPP for each watershed separately. The slope of the relationship between ER and GPP on flood and non-flood days indicates a change between net autotrophic or heterotrophic conditions. At all sites, flood events reduced GPP more than ER as depicted by the shallower slope of the regression between ER and GPP(Figure 15). Moreover, at some measurements, flood events increased ER levels (Figure 15D, E, and F). As a result, flood events moved all sites toward further heterotrophy. The East Branch DuPage River showed the greatest transition towards heterotrophy as the slope of the regression line between ER and GPP decreased by 90.8% for flood days (slope = 0.103) compared to non-flood days (Figure 15 E; slope = 1.09). Likewise, the regression slope between ER and GPP decreased by 70.7% for West Branch DuPage River (Figure 15 D) and 14.2% for the (Figure 15 F).



Figure 15. *ER* vs *GPP* in all sites for non-storm days for: A) West Branch DuPage River, B) East Branch DuPage River, and C) Salt Creek. *ER* vs *GPP* in all sites for storm days for: D)

West Branch DuPage River, E) East Branch DuPage River, and F) Salt Creek.

2.6 Discussion

2.6.1 Metabolism resistance and resilience to storms

Flashy hydrology can control metabolism, and primary producers can respond to flood events that occur on a regular basis (Poff et al., 1997). However, with the multitude of stressors that affect urban stream ecosystems (Walsh et al., 2005), determining the influence of sporadic flood events can complicate the interpretation of long-term metabolism datasets (Reisinger et al., 2017). Our findings suggest that larger flood events depress GPP more than smaller ones which is consistent with findings in the literature (Uehlinger 2000, Roberts et al. 2007). This might be due to increase the scour effect of flood events with the increase of flood magnitude (Uehlinger 2000). Moreover, larger floods might bring more turbidity (Lawler et al. 2006) that can block light from reaching the benthic communities and therefore, disturb GPP (Bernot et al. 2010).

Resistance of GPP (a reduction of 20-99%) and ER (a reduction of 3-60%), differed from previous research in urban, alpine and forested streams. In a study of an urban stream, Reisinger et al. (2017) found higher reduction of ER (N= 11floods compared to our dataset (N=65 floods). The discrepancy may be attributed in part because of floods from Hurricane Sandy (500 m^3/s) in the analysis which reduced both metabolism components (i.e., GPP and ER) to a greater degree. In the same study, no significant difference was found between GPPand ER resistance, which was inconsistent with our findings. One reason for this difference could be due tight coupling of GPP and ER in streams from this region, and ER can be carbon limited (Larsen et al. 2017). At our sites, we expect less coupling due to high organic matter input and storage (i.e., WWTP, combined sewers, and sedimentation) at our urban streams, which may help explain the high *ER* resistance to floods compared to Reisinger et al (2017). *GPP* at our sites showed less of a change (i.e., more resistant) than Uehlinger (2000) in an alpine river. We attribute this difference to the different definition of resistance between our and their study. Uehlinger (2000) defined the resistance as metabolism value one day after the flood divided by the value one day before the flood while our definition of resistance as metabolism value at the day of the storm divided by the value one day before the storm. The selected approach in our study was more suitable to our urban sites as metabolism metrics tended to recover very rapidly (sometimes in less than a day). Therefore, we suggest that factors which drive GPP and ER coupling, as well as calculation of metabolism resistance can affect patterns when comparing among sites and studies.

Contrary to our predictions, we found no relationship between resistance of GPP and ER with factors including discharge and watershed land use (i.e., developed area, impervious surface cover). One probable reason for the non-significant relationships is the limited variability in urban development (urban development in our studied sites ranged from 80 to 100%) and impervious surface cover (percent of impervious cover in out studied sites ranged from 32 to 40%) across our study sites. Future studies might benefit from including wider range of urban development and additional factors that quantify urbanization.

Resilience of GPP (i.e., 0.92 -9.54 days) and ER (i.e., 1.58 -5.35 days), is similar to some of the previous research in urban streams. Reisinger et al. (2017) showed GPP resilience of 4-18 days and ER resilience of 2-9 days in urban streams of the Baltimore region. In addition, similar recovery period (i.e. 1-2 weeks) was reported for algal biomass in a naturally flashy desert stream (Fisher et al. 1982). Larsen et al. (2017) found longer recovery periods of ER (i.e., up to 30 days) in Chesapeake Bay tributaries, attributed to the lack carbon to fuel heterotrophic ER, and tight coupling of GPP with ER following disturbance. Collectively, the emerging syntheses suggest that flood disturbances can play a major role in altering ecosystem function across stream ecosystem types and biomes and can have a stronger influence on autotrophic processes (including autotrophic contribution to ER), and heterotrophic processes, where carbon is limiting to heterotrophic respiration.

Our findings suggest that larger flood events depress GPP more than smaller ones which is consistent with findings in the literature (Uehlinger 2000, Roberts et al. 2007). This might be due to increasing the scour effects of flood events with increasing flood magnitude (Uehlinger 2000). Moreover, larger floods might bring more turbidity (Lawler et al. 2006) that can block light from reaching the benthic communities and therefore, disturb GPP (Bernot et al. 2010).

In contrast to GPP, floods reduced ER less than GPP indicating that flood events negatively affect autotrophs more than heterotrophs (Beaulieu et al 2013). Our results were consistent with previous studies (Uehlinger 2006, Roberts et al. 2007, Roley et al. 2014) which shows that flood events can often stimulate ER. We found similar results to Roely et al (2014) where about 80% of the studied floods stimulated ER while i40% of the studied floods stimulated GPP. Increased flood frequency can increase the release of organic matter which elevates ERrates (Kaplan and Bott 1982, Uehlinger 2006). Our results imply that storm flashiness is a primary driver of metabolism response to flood events; we found that ER resistance increased significantly with the increase in R-B index. We suggest that other measurements of flood events (e.g., flood frequency, recurrence interval) and environmental conditions (e.g., leaf fall, temperature, light) may be needed to further enhance our understanding metabolism response to flood disturbances in urban streams.

Results for resilience and resistance patterns partially matched our predictions (Figure 8) with higher ER resistance and relative to GPP and no significant difference in resilience. Different resistance of GPP and ER might be because autotrophs can be scoured off surfaces, and heterotrophs can be moved along with their organic substrates, and therefore heterotrophic communities could be more resistant. However, similar resilience between GPP and ER in the studied suburban streams might be due to the low canopy cover and availability of organic matter and nutrients that could fuel both autotrophs and heterotrophs very rapidly. Similar resilience and different resistance between GPP and ER was found at a previous study of urban streams (Reisinger et al. 2017). Additional research of resistance and resilience across wider number of urban streams is warranted.

The balance between GPP and ER is important for the carbon cycle in streams, which can change due to anthropogenic activities (i.e., wastewater effluent, fertilizer) and hydrologic patterns (Kaushal et al. 2014). Carbon can enter the streams during flood events from upstream or from the terrestrial area resulting in higher resistance of ER relative to GPP. The results of our study showed significant changes to metabolic activity as a response to even minor flood events. Therefore, additional research is still needed to show the effect of flood events on stream metabolism across a wider temporal and spatial ranges. In addition, future studies can focus on the influence of flashy hydrology on the coupling between GPP and ER across different stream characteristics.

2.7 Conclusion

Urban streams are characterized by flashy hydrology in addition to other physicochemical changes. Flood events caused a reduction in GPP more than ER, and both GPP and ER recovered to pre-flood levels within 1 to 10 days. Overall, our study showed that flood events in suburban streams sustain heterotrophic more than autotrophic conditions. We conclude that although ER might be expected to have higher resistance and higher resilience than GPP in urban streams, other site characteristics (i.e., availability of organic matter, and light) can determine the site-specific response of metabolism to flood events. Future research might include stream metabolism resistance and resilience calculations at larger number of sites with different site characteristics (i.e., stream bed material, organic matter input, canopy cover) to examine the patterns over larger geographic regions and gradients of urban intensity

2.8 Acknowledgments

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CHAPTER 3

THE IMPACTS OF DAM REMOVAL ON STREAM ECOSYSTEM METABOLISM

3.1 Abstract

Dam removal projects in the United States have increased due to the biological and physiochemical effects of dams on stream ecosystems, as well as concerns about dam age and maintenance. The influence of dam removal on ecosystem processes such as ecosystem metabolism (i.e., gross primary production (GPP), ecosystem respiration (ER) and net ecosystem production (NEP)) is rarely studied. Dam removal may influence metabolism at the upstream and downstream reaches via changes in release of fine sediment downstream, changes in flow regime and water velocity, riparian vegetation, light availability, and alterations to the assemblage of streambed substrata. We calculated stream metabolism before and after the removal of the Good Hope low-head dam in Conodoguinet Creek urban watershed in Pennsylvania. We focussed the analysis on three different stations: upstream of the dam (35.1 m), downstream of the dam (38.6 m) and at a reference station where hydrology was unaffected by the dam (4 km upstream). Using dissolved oxygen (DO) data, we calculated daily stream metabolism for two months before and two months after the dam removal at all three locations. We found that dam removal increased GPP and ER levels at the upstream and downstream dam removal sites, immediately after removal of the dam. The reference site showed only a small seasonal and

gradual decline in GPP and ER across the same months. In addition, metabolism in the upstream and downstream sections of the restored rivers showed similar GPP and ER rates and similar rates to a reference site following dam removal in the long term. Moreover, dam removal moves the streams' metabolism signal towards autotrophy by increasing GPP more than ER. Overall, we found that removal of a low head dam enhancing metabolism rates relatively rapidly, returning GPP and ER to reference conditions in scale and seasonality, and may thereby potentially facilitate recovery of other ecosystem parameters.

3.2 Introduction

Dams provide many important ecosystem services, including recreation, water supply, flood control, and hydroelectricity (Graf, 2002). Due to concerns about lack of maintenance, age and risk of failure, many dams are considered for removal (Born et al., 1998; Bednarek, 2001; Roberts et al., 2007b; O'Connor et al., 2015). Dam removal can cause changes to several river characteristics including river morphology, flooding, sediment transport, erosion, wetlands, and water quality (ICF, 2005). Dam removal helps restore the original flow regime, by removing the pool conditions in upstream section and enhancing sediment transport and hydrologic variability to the downstream section. Reconnection of the previously dammed stream into a single flowing reach has many consequences for biota. For example, dam removal reconnects riparian vegetation by allowing water to flood more often on the riparian zone and therefore, creating small ponds and wetlands that benefit biota (Ouellet and Dodson 1985, Dadswell 1996, Bednarek et al., 2001). Dam removal can also increase biodiversity and population densities of native species. For example, the removal of the Dead Lake dam on Chilopa River increased flow fluctuations and expanded the number of fish species from 34 to 61 (Bednarek et al., 2001). However, removing dams may increase the flow of contaminants from upstream (Hart et al., 2002; Gangloff, 2013).

In the last three decades, more than 1,100 dams were removed in the United States and abroad (American Rivers, 2016), however, only about 10% dam removal projects were monitored for ecological responses (USGS Science Database, 2016). Limitations in funding and the long permitting process were responsible for the lack of monitoring efforts (Magilligan et al., 2016). For the few monitored dam removal projects, most studies investigated the effects of channel adjustment, sediment transport, water quality or fish passage (Bushaw-Newton et al., 2002; Ashley et al., 2006; Whitener, 2013) and their results varied depending on physical characteristics of the rivers. After the removal of Manatauny Creek Dam in Pennsylvania, water quality did not change much due to the infrequent temperature stratification and short residence time of less than 2 hours (Bushaw-Newton et al., 2002). In contrast, the removal of Rock Dale Dam in Koshkonong Creek Wisconsin caused adverse effects on downstream water quality due to the release of phosphorous-rich sediment from the impoundment area (Doyle et al., 2003). On the other hand, the removal of Edwards Dam in Kennebee River Maine improved water quality in the impoundment area and allowed the area to meet state's water quality standards (from class C before dam removal to class B) (ICF Consulting 2005). Very few studies addressed the effects of dam removal on the functional properties of streams including stream metabolism (Finger et al., 2007; Zhang et al., 2014).

Ecosystem metabolism is a metric that represents key transformations of carbon in aquatic environments (Hoellein et al., 2013). Metabolism consists of gross primary production (GPP), ecosystem respiration (ER) and net ecosystem production (NEP). GPP measures the organic matter created within the ecosystem, ER measures the consumption of organic matter by autotrophs and heterotrophs from allochthonous or autochthonous organic matter sources, and NEP is the difference between production and respiration (Mulholland et al., 2001; Roberts et al., 2007a), which indicates the importance of internal relative to external energy sources (Allan and Castillo, 2007). Stream metabolism is an important metric for understanding ecosystem function, including food webs, nutrient cycling, energy fluxes, and responses to human activities (Odum, 1956; Marcarelli et al., 2011; Hotchkiss et al., 2015).

Assessing the influence of dam removal on stream ecosystem requires continuous measurements of DO, temperature and discharge. The limited availability of continuous datasets before and after dam removal has inhibited research to just a few published studies based on modeling predictions. For example, Zhang et al. (2014) modeled the effect of dam removal on stream productivity using only pre-dam removal data. They found that dam removal could decrease GPP at the upstream reach by 30% due to the effluent of combined sewer overflows. Finger et al. (2007) estimated that GPP decreased by 12% downstream of a hypothetical dam removal. Finally, there are no assessments of dam removal on ecosystem function covering more than one season. Ecosystem metabolism is strongly affected by environmental factors that change within seasons including changes in day length, temperature, and stream flow, so continuous monitoring of DO and temperature is needed, before and after dam removal, and across distinct seasonal changes.

In addition to continuous measurements, the study of how dam removal affects stream metabolism will require a careful conceptual framework to explain the mechanisms driving changes GPP and ER, before and after dam removal. Dam removal affects stream metabolism by restoring natural flow regimes, water depths, temperature variations, sedimentation, turbidity, turbulence, and riparian vegetation (Bushaw-Newton et al. 2002). Dam removal can affect the heterotrophic and autotrophic organisms that affect ecosystem metabolism, as well as the physiochemical conditions that influence DO concentrations. At upstream sites, we expect negative NEP before dam removal, as the reservoir above the dam will increase sedimentation and heterotrophic activity. Following dam removal, we expect an increase in GPP at the upstream reach due to 1) reduced sediment and water depth, 2) greater light penetration to the benthos, and 3) faster flowing water which could allow benthic autotrophs to flourish. Downstream of the dam and prior to dam removal, we expect the reach will be autotrophic, due swiftly flowing water with reduced influence of scouring floods and reduced sediment delivery. After dam removal, we expect the downstream reach to become more balanced between autotrophic and heterotrophic activity. Overall, we expect stream metabolism at the downstream and upstream reach will be more similar to each other after dam removal

compared to beforehand. In addition, we expect GPP and ER at upstream and downstream sites to become more similar to the reference reach after the dam is removed, and the river becomes a single free-flowing ecosystem.

3.3 Study Area

We used the USGS Dam Removal Database (www.sciencebase.gov, accessed on June 2017) to locate sites with the necessary data available to calculate the effects of dam removal on ecosystem metabolism. The database indicated only 25 studies with continuous *DO* data before and after dam removal. Only a single dam removal project, the removal of a low-head dam in Conodoguinet Creek, an urban water shed in Good Hope, PA, USA, provided data of sufficient length, coverage, and quality for calculations of stream metabolism. Our work focuses on understanding stream metabolism at this site.

The Good Hope Mill dam was a 1.8-meter-high concrete dam that was constructed prior to 1918 (i.e., no exact date was determined; Chapin et al. 2005) to supply power to the Good Hope Mill. The dam has an impoundment of 2.25 km in length and a mean annual flow of 17.21 m³/s (USGS station 01570000). The dam was removed during 3 days period starting November 2, 2001 (Chaplin et al., 2005).



Figure 16. Good Hope dam and study sites in Pennsylvania, USA. The reference station is near Lambs Gap Road, and is situated 4 km upstream of the dam, the upstream station on the Conodoguinet Creek and situated 35.1 m upstream of the dam, and the downstream station on the = Conodoguinet Creek and situated 38.6m downstream of the dam.

We calculated stream metabolism at three different monitoring stations close to the dam. The upstream station was 35.1 m upstream of the dam, the downstream station was 38.6 m downstream of the dam, and the reference station was situated 4 km upstream of the dam, outside of the area of influence of this dam on stream hydrology and geomorphology (Figure 20, Table V). For the three monitoring stations, we used dissolved oxygen (DO) observations and temperature (T) which were gathered from a monitoring study by Chaplin et al. (2005).

To account for change in Q between the USGS gauge and the study sites, we used the drainage area weighted discharge approach. We calculated timeseries water depth using the Manning's equation with the adjusted Q and channel characteristics (i.e., cross section, slope, stream bed material) at each location which was gathered from Chaplin et al (2005) study.

Station	Distance	USGS	Data before dam	Data after dam
Sution	Distuice	station	removal	removal
Reference	4 km upstream	01570000	Sep-Oct 2001	Nov-Dec 2001
Upstream	35.1 m upstream	01570000	Sep-Oct 2001	Nov-Dec 2001
Downstream	38.6 m downstream	01570000	Sep-Oct 2001	Nov-Dec 2001

TABLE V

LOCATIONS, DISTANCE FROM DAM REMOVAL IN KM, AND DATE RANGES.

3.4 Methods

3.4.1 Whole-stream metabolism

The *DO* mass balance equation (Eq. Equation 3.1) was used to calculate stream metabolism (O'Connor and Di Toro, 1970)

$$\frac{\partial DO}{\partial t} = -U \frac{\partial DO}{\partial x} + \underbrace{GPP(t) - ER(t)}_{NEP} + K \underbrace{(DO_s - DO)}_{DO_{\text{deficit.}}}$$
(3.1)

where U is the stream velocity, x is the distance, K is the reaeration rate, and DO_s is the DO concentration at saturation. Assuming a uniform distribution of plants with distance x, DO concentration will not vary spatially and equation (Equation 3.1) can be re-written as

$$\frac{\partial DO}{\partial t} = \underbrace{GPP(t) - ER(t)}_{NEP} + K \underbrace{(DO_s - DO)}_{DO_{\text{deficit}}}$$
(3.2)

Stream metabolism calculations were completed using the "stream Metabolizer" model (Appling et al. 2017). This model uses a Bayesian approach to estimates unknown metabolism metrics (i.e., GPP and ER) from measured input data including DO, DO_s , day length, and water depth (Hobbs and Hooten 2015, Hall et al. 2016). We obtained both DO and temperature data from the U.S. Geological Survey Pennsylvania Water Science Center. We calculated DOsaturation (DO_s) based on temperature data (T) using an empirical equation derived by the American Public Health Association (APHA) (1992), which is given by:

$$\ln(DO_s) = -139.34411 + (1.575701 * \frac{10^5}{T}) - (6.642308 * \frac{10^7}{T^2}) + (1.243800 * \frac{10^{10}}{T^3}) - (8.621949 * \frac{10^{11}}{T^4})$$
(3.3)

A relationship between USGS field data of discharge and water depth was created and used to calculate daily water depth. Since GPP is only active during daylight, the daytime length was determined from photosynthetically active radiation (PAR) data which was estimated using the following function:

$$P(t) = amp * (\sin(\omega t + phase)) * (amp * \sin(\omega t + phase) > 0)$$
(3.4)

We set the amp to 1600 μ mol m⁻² s⁻¹, $\omega = 2\pi/24$, and *phase* to $-\pi/2$. We initialized the streamMetabolizer model with the model's generic prior probability distribution functions for *GPP* and *ER* (8–4 and 10–5 g O₂ m⁻² d⁻¹, respectively) and use initial estimates of daily reaeration (log K) of 1.79–1 d⁻¹ following Tassone et al. (2017).

3.4.2 Statistical analysis

We used an auto-regressive integrated moving average (ARIMA) model to compare spatial and temporal variations in stream metabolism before and after dam removal (Box and Jenkins 1970). ARIMA models requires three set of parameters: autoregressive parameters (AR, p) which test for relationships between the variable of interest at time (t) and the same variable at previous time step, moving average coefficients (MA, q) which test for relationships between the response variable at time t and the residual of the previous time step, and stationarity parameters (I) which is used if the mean and variance of the response variable are not constant. Non-stationarity requires detrending data by differencing it using 1st or 2nd order differencing (Cryer and Chan 2008). To test if autocorrelation was accounted for, we plotted the residuals as a timeseries using the autocorrelation function and the partial autocorrelation function that detects the existence of autocorrelation (Zuur et al. 2009). Including a seasonal component resulted in poorer model fits. The best model fit was selected based on the Akaik's Information Criterion AIC (Akaik 1973). We used ARIMA analysis for before and after the dam removal for each site separately. If dam removal influenced metabolism metrics, model structure or coefficients would be different before and after dam removal.

We used simple regression to compare the relationship between GPP and ER at all stations for each month before and after the dam removal. ARIMA and linear regressions were computed with MATLAB and the Statistics Toolbox (Release 2016b, The MathWorks Inc., Illinois).

3.5 Results

3.5.1 Temporal variation in stream metabolism following dam removal

There were significant differences in ecosystem metabolism before and after dam removal at the upstream and downstream stations, but no significant changes at the reference station (Table VII, Table VI, and Figure 17). At both the upstream and downstream stations, ARIMA showed the best-fit model for GPP changed from (0,1,1) to (1,1,1) indicating that both stations faced a change in physicochemical factors influencing stream metabolism (Table VI). Similarly, at the same stations, the best-fit model for ER changed from (1,1,1) to (0,1,1) indicating a physicochemical change influenced ER at both sites similarly considering the similarity of model coefficient (Table VI). On the other hand, neither the model structure nor the coefficient changed significantly at the reference station suggesting no effect of the dam removal on stream metabolism at the reference station (Table VI).

Site	Response variable		Pre-dam removal				Post dam removal			
		ARIMA model structure	Component	Coefficient	SE	ARIMA model structure	Component	Coefficient	SE	
Reference	GPP	(1,1,1)	θ	-0.178	0.137	(1,1,1)	θ	-0.571	0.139	
			φ	0.234	0.113		φ	0.635	0.113	
	ER	(0,1,1)	φ	0.115	0.139	(0,1,1)	φ	0.301	0.134	
Upstream	GPP	(0,1,1)	φ	0.187	0.249	(1,1,1)	θ	-0.718	0.331	
							φ	0.833	0.275	
	ER	(1,1,1)	θ	0.841	0.116	(0,1,1)	φ	0.215	0.135	
			φ	-0.552	0.203					
Downstre am	GPP	(0,1,1)	φ	0.079	0.165	(1,1,1)	θ	-0.623	0.175	
							φ	0.991	0.143	
	ER	(1,1,1)	θ	0.491	0.107	(0,1,1)	φ	0.115	0.138	
			φ	-0.834	0.175					

TABLE VI

AUTO-REGRESSIVE INTEGRATED MOVING AVERAGE (ARIMA) MODEL STRUCTURE FOR GROSS PRIMARY PRODUCTION (*GPP*) AND ECOSYSTEM RESPIRATION (*ER*) FOR ALL STATIONS PRE AND POST DAM REMOVAL. AS DESCRIBED IN THE METHODS, MODEL STRUCTURE IS REPRESENTED AS (P, D, Q). SE = STANDARD ERROR, θ IS THE MOVING AVERAGE COEFFICIENT, φ IS THE AUTO-REGRESSIVE COEFFICIENT. REFER TO TABLE 1 FOR STATION LOCATIONS. Upstream of the dam, mean GPP increased significantly after dam removal (Table VII, Figure 17, an increase of 1467%). Likewise, upstream ER increased significantly following dam removal (Table VII, Figure 17, an increase of 702%). Downstream of the dam, GPP increased significantly after dam removal (Table VII, Figure 17, an increase of 1978%). Similarly, downstream ER increased significantly by 661% after dam removal (Table VII, Figure 17). At the reference station, GPP and ER showed a gradual decrease across the 4 months of measurement (Table VII, Figure 17).

sites	2 months pre-dam removal						
	GPP	% Change	ER	% Change	NEP	n	
Reference	9.78 ± 2.66		$\textbf{-8.53}\pm1.95$		1.25 ± 2.63	25	
Upstream	1.77 ± 0.64		$\textbf{-1.66} \pm \textbf{0.73}$		$\textbf{0.08} \pm \textbf{0.58}$	25	
Downstream	1.58 ± 0.64		$\textbf{-1.48} \pm \textbf{0.76}$		$\textbf{0.09}\pm\textbf{0.52}$	25	
		1-month pre-dam removal					
Reference	7.17 ± 1.56		-5.81 ± 2.42		1.35 ± 2.45	31	
Upstream	$\textbf{0.28}\pm\textbf{0.22}$		$\textbf{-0.54} \pm 0.32$		$\textbf{-0.27} \pm 0.43$	17	
Downstream	$\textbf{0.19} \pm \textbf{0.11}$		$\textbf{-0.56} \pm \textbf{0.27}$		$\textbf{-0.37} \pm 0.29$	31	
	1-month post-dam removal						
Reference	4.86 ± 2.62	↓ 32.2 %	$\textbf{-3.57}\pm2.08$	↓ 39 %	1.29 ± 1.86	25	
Upstream	$\textbf{4.39} \pm \textbf{1.51}$	↑ 1467 %	-4.33 ± 1.41	↑ 702 %	$\textbf{0.07} \pm 1.74$	25	
Downstream	$\textbf{3.95} \pm \textbf{1.36}$	↑ 1978 %	$\textbf{-4.26} \pm 1.75$	↑ 661 %	$\textbf{-0.31} \pm \textbf{2.25}$	25	
		2 months post-dam removal					
Reference	2.35 ± 0.68	↓ 67.22 %	$\textbf{-0.72} \pm \textbf{0.99}$	↓ 88 %	1.63 ± 1.28	21	
Upstream	$\textbf{5.99} \pm \textbf{1.70}$	↑ 2039 %	$\textbf{-4.29} \pm 1.09$	↑ 694 %	1.69 ± 1.19	30	
Downstream	$\textbf{4.82} \pm \textbf{1.74}$	↑ 2436 %	$\textbf{-5.49} \pm 1.63$	↑880 %	$\textbf{-0.67} \pm 2.15$	30	

TABLE VII

MEAN (+/- STANDARD DEVIATION) GROSS PRIMARY PRODUCTION (GPP) ECOSYSTEM RESPIRATION (ER), AND NET ECOSYSTEM METABOLISM (G O₂ M⁻² D⁻¹) FOR GOOD HOPE DAM STATIONS SEPARATED ON FOUR TIME PERIODS, 2 MONTHS BEFORE, 1 MONTH BEFORE, 1 MONTH AFTER DAM REMOVAL, AND 2 MONTHS AFTER DAM REMOVAL. % CHANGE REFERS TO THE DIFFERENCE BETWEEN MEAN GPP OR ER COMPARED TO THE VALUES OF ONE MONTH BEFORE THE DAM REMOVAL.



Figure 17. Daily gross primary production (GPP) and ecosystem respiration (ER) at the upstream, downstream, and references sites.

3.5.2 Spatial variation in stream metabolism following dam removal

Prior to dam removal, the upstream and downstream sites had similar ARIMA model structure and coefficients for GPP(0,1,1) indicating similar physicochemical factors influencing GPP at both stations, but each different from the reference station. In the same manner, the upstream and downstream stations had similar structure for ER(1,1,1) and were different from the reference station (Table VI). After dam removal, all stations had similar structure of ER(0,1,1) and GPP(1,1,1) indicating some homogeneity in channel physiochemical characteristics (i.e., the distribution of organic matter and production across the stations) (Table VI).

3.5.3 Net ecosystem production following dam removal

Dam removal enhanced GPP to a greater degree than ER at both the upstream and downstream sites (Figure 18 B, C). For example, one month following dam removal, average daily GPP at the upstream site increased to 8.7 g O₂ m⁻² d⁻¹, compared to 1.2 g O₂ m⁻² d⁻¹ just prior to pre-dam removal (Table 2; Fig Figure 18 B, C). However, ER at the upstream site changed to 8.2 g O₂ m⁻² d⁻¹ compared to 3.2 g O₂ m⁻² d⁻¹ before dam removal (Figure 18). The same patterns occurred for the downstream site (Table 2; Figure 18 B, C). Overall, dam removal increased positive NEP by increasing GPP more than ER (Figure 18). The transition towards autotrophy was clear for the upstream and downstream sites, while there were no changes at the reference station (Figure 18 B, C). The relative number of days where GPP : ER is greater than 1 (i.e., positive NEP) increased from 38% to 62% one month following the dam removal and to 81% two months following the dam removal (Figure 18 B, C, D). Finally, dam removal resulted in ecosystem metabolism patterns that were more similar amount the 3 study locations. Before dam removal, the reference station had separate cluster of metabolism results from the upstream and downstream sites (Fig Figure 18. A, B). Following dam removal, GPP and ER of the three stations became more similar one and two months following dam removal (Figure 18 B, C, D).



Figure 18. GPP : ER ratio for the Good Hope dam 2 months before (A), 1 month before (B),

1 month after (C), and 2 months after the dam removal (D). Percent values indicate the percentage of measurements that are heterotrophic (i.e., GPP : ER < 1) and the percentage of measurements that are autotrophic (i.e., GPP : ER > 1).

3.6 Discussion

3.6.1 The influence of dam removal on stream metabolism

Dam removal has become an effective means of restoring river degraded aquatic communities (Gottens et al., 2009; Ellsworth et al., 2012; Cooper et al., 2013). Dam removal has many effects on stream habitat structure and function, including exposing riffles and pools that allows for new habitats, increasing the number of native fluvial species that are adapted to flowing water, and improved water quality (Bednarek et al. 2001, EPA 2016). This study is the first empirical assessment of dam removal on ecosystem metabolism and adds to the growing body of literature on dam removal impacts to water quality by showing relatively rapid recovery of metabolism to reference conditions.

The response of stream metabolism to dam removal was consistent with our predictions for the upstream station. One month following the dam removal, GPP increased at the upstream site suggesting that the decrease in water level and reduction in sediment following dam removal might have increased light reaching the benthic communities and therefore, enhanced GPP (Whitener, 2013; Granata et al., 2008; Magilligan et al., 2016). Similarly, ER levels increased significantly at the upstream site, which is also consistent with the expected outcomes (Figure 17, Figure 18). One probable reason for the increase in ER levels might be due to the high temperature (Risley et al 2012), the transport of organic matter from upstream, and the increase of autochthonous energy sources (i.e., GPP) which might elevate autotrophic respiration (Bernot et al. 2010).

Contrary to our predictions, the metabolism patterns at the downstream site were similar to the upstream site (i.e., increased GPP and ER). One probable reason for this similarity is the proximity of the downstream site to the dam removal (38.6 m). With this close proximity, any effect of dam removal on downstream turbidity might have occurred very rapidly and may not have a sustained effect on GPP and ER. We did not measure suspended sediments directly, so can only infer this process from the pattern in metabolism. The presence of dam might cause an overgrowth of algae which can eventually prevent light from reaching the benthic communities (Creek et al. 2007). Therefore, removing the dam can cause faster moving water which might flourished some of the algae and enhanced production at the benthic community.

At the reference station (4 km US), both GPP and ER began decrease in September and continued through November (when the dam was removed) and December, following our general expectations for seasonal drivers of ecosystem metabolism (Roberts et al. 2007, Hoellein et al. 2007, Roley et al. 2014). As the dataset covered early fall to winter, the lower temperatures during the study period likely decreases ER, and lower light availability decreases GPP during the winter. Chaplin et al. (2005) found similar results with no major changes in water quality or chemistry at the reference station after the removal of the Good Hope dam. These reference data also supported the inferences we generated regarding the effect of the dam removal as a major event in affecting patterns in ecosystem metabolism at the dam-adjacent study sites. We recommend that future assessments of dam removal take advantage of long-term (i.e., years) pre- and post-sampling at the upstream and downstream sites, as well as a nearby reference condition to account for environmental drivers that affect metabolism over seasonal time scales.

The season in when dam removal occurs can have different effects on stream metabolism and water quality. The Good Hope dam was removed in the fall/winter period where both GPP and ER tend to decrease over time. If the removal was done during summer season, stream metabolism might have responded differently. GPP might increase to high levels with the increase in light intensity, and similarly ER might reach higher values. The pattern might affect stream organisms that prefer specific levels of GPP or ER. For example, Santucci et al. (2005) found that low-head dam removal further degraded the stream in warm waters by degrading stream habitat and fragmenting the river landscape. Therefore, we suggest decision makers consider the best timing of dam removal depending on the goal of the stream restoration.

3.6.2 Metabolism data confirms increased autotrophy with dam removal

Flowing waters are often heterotrophic, as allochthonous carbon is critical to support food webs and nutrient transformations, especially in forested headwater streams (Mulholland et al. 2001, Battin et al. 2008, Rosenfeld and Mackay 1987). Urban streams are characterized by increased autotrophy (Bernot et al. 2010) (Figure 18A). In this study, dam removal increased the autotrophic measurements by reducing ER more than GPP (Figure 18B) especially at the sites closer to the dam. Our results showed dam removal supported a shift from negative NEP towards positive NEP specially at the upstream site. We suggest that new geomorphic characteristics following dam removal (i.e., increased coarse sediment and decreased fine sediment) increased dominance of inorganic substrata, reducing organic matter standing stock (Roley et al. 2014) and therefore, having reduced ER compared to GPP.

3.7 Conclusion

The alteration in stream hydrology and geomorphology through river restoration can influence stream ecosystem metabolism. Dam removal increased both GPP and ER at the upstream and downstream locations, by changing the stream from two separated habitats into a single habitat that sustained primary producers and likely flushed organic matter from the previously impounded area. Overall, we illustrated that dam removal can affect stream metabolism and therefore, management practices should consider the linkage between stream metabolism and dam removal in optimization restoration efforts. Future dam removal studies should consider monitoring DO levels as a mean to calculate stream metabolism. Finally, additional research is needed with wider number of dam removal studies to draw more certain conclusions on dam removal and stream metabolism.

CHAPTER 4

THE INFLUENCE OF URBANIZATION ON STREAM METABOLISM

4.1 Abstract

Urbanization is known to alter ecosystem metabolism in streams, but how stream metabolism changes in response to increasing urban intensity is poorly understood. In this study, we investigated the influence of urbanization on stream metabolism using the open channel whole-stream metabolism method. We calculated stream metabolism at 18 sites located on 13 watersheds in the Midwestern US during periods covering summer and fall. We related metabolism metrics results to urbanization variables (i.e., urban land use gradient index (ULUG) and flashiness index) and found a threshold of 23.8 ULUG, where net ecosystem production (NEP) and gross primary production (GPP) changed from positively to negatively correlated with urban intensity. Results showed that both GPP and ER did not significantly change (in absolute value) above 23.8 ULUG. Moreover, we found that stream respiration increased with the increase in stream flashiness despite the degree of urbanization. Over all, we found that increased urban intensity confirms increased heterotrophy by reducing GPP more than ER. This study suggests that to improve urban stream metabolism, restoration projects should be targeted to high urban intensities. Future research can include a national-wide urban stream metabolism and ULUG calculations to confirm the threshold.

4.2 Introduction

Urban streams face multiple physical, chemical, and biological stressors, collectively called the "urban stream syndrome" (Meyer et al., 2005; Walsh et al., 2005). Physical changes include altered stream geomorphology through bank incision, channel reconfiguration, and bed mobilization (Doyle et al., 2000; Vietz et al., 2016). Chemical changes include increased contaminant and nutrient loading from adjacent landscapes (Hatt et al., 2004; Bernhardt et al., 2008; Sudduth et al., 2011), addition of nutrients such as nitrogen and phosporus (Correll, 1998), and a complex suite of synthetic chemicals from point and non-point sources (Koplin et al. 2002). Biological changes include altered ecosystem processes (e.g. nutrient cycling, ecosystem productivity, and leaf decomposition) and decline of intolerant biotic species (Paul and Meyer, 2001).

Stream metabolism is a metric of stream ecosystem processes (Hoellein et al., 2013) consisting of gross primary production (GPP), ecosystem respiration (ER) and net ecosystem production (NEP). GPP is the amount of organic matter created within the ecosystem, ER is the amount of energy consumed by both autotrophs and heterotrophs using allochthonous or autochthonous sources, and NEP is the difference between production and respiration (Mulholland et al., 2001; Roberts et al., 2007a). Hence, NEP measures the importance of internal relative to external organic carbon input (Allan and Castillo, 2007). Stream metabolism integrates all habitat types including planktonic, benthic, and hyporheic zones in addition to all aerobic organisms including autotrophs and heterotrophs. Therefore, stream metabolism is important in understanding ecosystem function including stream controls on food webs and energy fluxes, nutrient cycling, and stream response to human activities (Odum, 1956; Marcarelli et al., 2011; Hotchkiss et al., 2015).

Urbanization alters stream metabolism, but how stream metabolism changes in response to increasing urban intensity is poorly understood (Meyer et al., 2005; Von Schiller et al., 2008; Sudduth et al., 2011; Iwata et al., 2007; Bernot et al., 2010). Many studies showed an increase in GPP relative to ER in urban streams compared to forested streams through the increase in light intensity (Bunn et al., 1999; Fellows et al., 2006; Iwata et al., 2007; Bernot et al., 2010), and eutrophication (Rao et al., 1979; Izagirre et al., 2008). However, hydrologic changes from urbanization might reduce GPP and ER as increased frequency and intensity of flooding scours algae and reduces benthic storage of organic matter (Meyer et al., 2005). Other studies showed no consistent relationship between urbanization (based on percent of developed area and impervious surface cover) and stream metabolism (Meyer et al., 2005; Von Schiller et al., 2008; Sudduth et al., 2011). The lack of clear relationships between stream metabolism and urbanization are stem from the limited time scales of calculations for stream metabolism, which can vary days, seasons, and years (Uehlinger et al. 2000, Roberts et al 2007, Izagirre et al. 2008, Qasem et al. 2018 – under review). In addition to the need for continuous, long-term metabolism measurements, predicting the effects of urbanization on stream metabolism also requires a comprehensive approach to quantifying urbanization.

(McMahon and Cuffney, 2000) quantified urbanization via the "urban land-use gradient index" (ULUG). The authors defined ULUG index as a measure of degree of urban intensity from low (ULUG=0) to high (ULUG=100) based on basin-wide variables. These variables include: infrastructure (e.g., road intensity, point sources.), land cover, and socioeconomics (e.g., population, housing, income). (McMahon and Cuffney, 2000) designed the ULUG metric to investigate the relationship between urbanization and stream water quality. (Coles et al., 2004) used the ULUG index to determine the influence of urbanization on chemical, biological, and physical characteristics of New England coastal streams. Coles et al. (2004) found that the relationship between urban intensity and many variables of interest no longer changed after a threshold of 35 ULUG index indicating that most response of water quality variables occur at low to moderate urban intensities. The ULUG metric has not yet been applied to ecosystem metabolism in streams.

In addition to an integrative metric of land-use such as ULUG, research on drivers of ecosystem metabolism in streams can benefit from an index which quantifies patterns in hydrology. Urban streams experience "flashy hydrology", the occurrence of rapid-onset high discharge events due to impervious surface cover (Walsh et al., 2005; Kaushal et al., 2008; Vietz et al., 2016). High discharge events are often a key controlling factor on stream ecosystems and metabolism (Poff et al. 1997, Reisinger et al. 2017, Qasem et al. 2018 – under review). (Baker et al., 2004) proposed the "flashiness index" to characterize flow regimes based on variations in mean daily flow data. In their study of 515 Midwestern streams for the period from 1975

through 2001, Baker et al. (2004) found that urban streams showed characteristically high values of flashiness index (range = 0.651-0.712), whereas forested streams showed low values of flashiness index (range= 0.04-0.14). In addition, stream flashiness decreased with increasing watershed area despite the land use. Previous studies have not used the flashiness index to quantify the role of hydrology on ecosystem metabolism in streams which span a wide land-use gradient from rural to urban conditions.

The goal of this study was to assess the influence of urban intensity on seasonally-averaged stream metabolism across multiple sites. We calculated stream metabolism at 18 sites located in 13 watersheds around Wisconsin, Illinois, Indiana, Missouri, and Kansas, USA for six months during summer and fall, when the light availability (primary controlling factor on GPP) is the highest and when ER peaks (Hart 2013). We related metabolism metrics (e.g., GPP, ER, and NEP) to urbanization variables (i.e., ULUG and flashiness index). At low to moderate ULUG index, we predicted an increase in GPP with the ULUG index due to the increased light availability and nutrient supply (Figure 19). Similarly, we expected an increase in ER due to the increase in GPP and the transport of labile organic matter in fall (Bernot et al. 2010) (Figure 19). At the highest ULUG index sites, we expected metabolism to be reduced due to factors such as scour, contaminant loading from wastewater treatment plants, and a reduction in allochthonous organic matter due to the disconnection between stream and riparian vegetation (Figure 19).


Figure 19. Expected outcome of the influence of urban intensity on gross primary production and ecosystem

4.3 Methods

4.3.1 Site selection

We analyzed 18 stream reaches in 13 watersheds (Figure 20). Sites were selected based on the following criteria: located in an urban area (based on National Land Cover Database 2011 (NLCD 2011) and Google Earth aerial image), and all data were available for metabolism calculations (i.e., dissolved oxygen (DO), discharge (Q), and temperature (T)). Dissolved oxygen and temperature data were provided by the Metropolitan Water Reclamation District of Greater Chicago (MWRD), the DuPage River Salt Creek Workgroup (DRSCW), and the US Geological Survey (USGS).

For sites with no Q data, we calculated the drainage-area-weighted discharge from the closest USGS station. To consider variability in site selection, we made sure sites did not share the same USGS gauge. Water depth (z) was calculated from USGS field data of width and cross-sectional area assuming a rectangular cross-section. A relationship between discharge and water depth was created at each site to calculate daily water depth. For sites with no Q data, water depth was calculated using the Manning's equation and channel characteristics (i.e., Manning's roughness, slope, cross section) which was provided by the DuPage River Storm Water Division and MWRD.

Basin areas were calculated from the StreamStats USGS Watershed Delineation online tool. For the few sites that the tool was not available (i.e., sites located in Missouri and Kansas: 4, 6, 11, 12, and 14.), we manually delineated watershed boundaries using the Hydraulic Unit Code 12 (HUC12) watershed data and national topography map in conjunction with geographic information systems (GIS) program. Drainage areas ranged from 8 to 331 km² (Figure 20; Table IX), and the smaller and larger watersheds were evenly distributed across the urban land use gradient (Table IX). One period of six months of continuous summer-fall data was selected for each site. The reason for selecting this data range is that some of the sites were only limited to summer-fall data.



Figure 20. The 18 Study Sites on Land Cover Map

4.3.2 Urban land use gradient and flashiness indices

To derive the ULUG included three main categories of variables: infrastructure, land cover, and population (Table VIII; McMahon et al., (2000)). For the infrastructure variables, we obtained the number of toxic release sites from the U.S Environmental Protection Agency (EPA) Toxic Release Inventory for 2016, the number of point source discharges from the EPA Discharge Monitoring Report (DMR) Pollutant Loading Tool for 2016, the road density data from USGS National Transportation 2016 Dataset, and the number of dams at each basin from the United States Army Corps National Inventory of Maps. In addition, percentages of each land cover category were calculated using GIS according to The National Land Cover Database (NLCD) 2011. Population data were obtained from the United States census 2011-2015 American Community Survey 5-Year Estimates.

To select the most important variables for the ULUG index calculations, we followed the approach of (Coles et al., 2004) who selected the variable based on strong correlation with total population. We selected variables that correlated with the 2011-2015 population density ($|rho| \ge 0.5$) across the watersheds (Table VIII). Of the initial 20 variables, only 10 variables were used to calculate the ULUG index.

Variable	Spearman rank correlation with 2011 2015 population density
Infrastructure variable	S
Number of Toxic release inventory sites /1km2 of basin area	0.58
Roads density in basin [road length(km)/basin area(km2)]	0.83
Number of point source discharges/1km of basin area	0.61
Number of dams/1km of basin area	0.57
Land cover variables	
Percentage of drainage area in open water	0.03
Percentage of drainage area in developed open space	0.64
Percentage of drainage area in low-intensity residential	0.81
Percentage of drainage area in Medium-intensity residential	0.58
Percentage of drainage area in high-intensity residential	0.52
Percentage of drainage area in barren land	0.32
Percentage of drainage area in deciduous forest	0.71
Percentage of drainage area in evergreen forest	0.47
Percentage of drainage area in mixed forest	0.03
Percentage of drainage area in shrub	0.43
Percentage of drainage area in herbaceous	0.48
Percentage of drainage area in pasture/hay	0.74
Percentage of drainage area in cultivated crops	0.36
Percentage of drainage area in woody wetlands	0.67
Percentage of drainage area in emergent herbaceous wetlands	0.46

TABLE VIII

LIST OF VARIABLES THAT WERE USED FOR THE CALCULATION OF THE URBAN

LAND USE GRADIENT INDEX. (NOTE: ONLY BOLD VARIABLES WHICH

CORRELATED WITH 2011-2015 POPULATION DENSITY AT A LEVEL ≥ 0.50 WERE

USED IN THE CALCULATION OF THE URBAN INTENSITY INDEX)

Following McMahon (2000), original variables were transformed to range from 0-1 using the following equation:

$$Y = \frac{x - x_{min}}{x_{max} - x} \tag{4.1}$$

Where Y is the transformed variable and x is the original variable. The mean of the transformed variables was calculated for each site and transformed (in the same manner of equation Equation 4.2) and multiplied by 100 to give a range of 0 to 100 ULUG index (see Table XVI in the appendix for an example of ULUG index calculations).

To calculate the flashiness index (Baker et al. 2004) for the sites included in the current study, we applied the following equation:

$$FI = \frac{\sum_{i=1}^{n} |Q_i - Q_{i-1}|}{\sum_{i=1}^{n} Q_i}$$
(4.2)

where Q_i is the average daily flow in units of m³/s during day *i* and *n* is the number of days.

Site	USGS station	Basin area [km²]	Watershed	ULUG	Flashiness index
1	05435943	20.3	Badger Mill Creek	0	0.31
2	04087050	8	Little Menomonee River	5.33	0.51
3	04085108	44.8	East River	13.1	0.56
4	06893390	64.17	Indiana Harbor Canal	16.62	0.77
5	04095090	331	Portage-Buns Waterway	18.43	0.22
6	06893970	8.4	Spring Branch Creek	18.52	0.91
7	05540095	90.7	West Branch DuPage River	20.22	0.43
8	05531044	51.9	Salt Creek	21.27	0.09
9	05536105	113	North Branch DuPage River	23.88	0.49
10	05540160	26.6	East Branch DuPage River	29.02	0.31
11	06893300	26.6	East Fork Little Blue River	31.11	0.79
12	06893820	98.2	Little Blue River	31.52	0.61
13	04087119	10.3	Honey Creek	31.77	1.27
14	06893890	34.4	East Fork Little Blue River	36.84	0.18
15	40871488	11.34	Wilson Park Creek	51.52	1.33
16	05531300	91.5	Salt Creek	67.06	0.36
17	04092750	33.16	Indiana Harbor Canal	71.1	0.61
18	05531500	115	Salt Creek	100	0.38

TABLE IX

SITES LISTED ACCENDING FROM LOWER TO HIGHER URBAN LAND USE GRADIENT INDEX (ULUG) INDEX, CLOSEST USGS STATION, BASIN AREAS IN KM2, WATERSHED NAME, ULUG INDEX VALUES, AND FLASHINESS INDEX VALUES.

4.3.3 Whole-stream metabolism

We calculated stream metabolism using the DO mass balance equation

$$\frac{\partial DO}{\partial t} = -U \frac{\partial DO}{\partial x} + \underbrace{GPP(t) - ER(t)}_{NEP} + K \underbrace{(DO_s - DO)}_{DO_{\text{deficit.}}}$$
(4.3)

where U is the stream velocity, x is the distance, K is the reaeration rate, and DO_s is the DO concentration at saturation. Assuming no significant changes in DO concentration in space, equation (4.3) can be re-written as

$$\frac{\partial DO}{\partial t} = \underbrace{GPP(t) - ER(t)}_{NEP} + K \underbrace{(DO_s - DO)}_{DO_{\text{deficit}}}$$
(4.4)

Stream metabolism calculations were completed using the "streamMetabolizer" model based on a Bayesian approach (Appling et al. 2017), which determines unknown metabolism metrics (i.e., GPP and ER) from known input data including DO, DO_s , day length, and water depth (Hobbs and Hooten 2015, Hall et al. 2016) that results in high Bayesian posterior probability which is the normalized product of the prior probability for parameter values and the likelihood that the parameters produce the observed O_2 values.

We estimated DO_s based on temperature data (T) following the American Public Health Association (APHA) empirical equation (1992), which is given by:

$$\ln(DO_s) = -139.34411 + (1.575701 * \frac{10^5}{T}) - (6.642308 * \frac{10^7}{T^2}) + (1.243800 * \frac{10^{10}}{T^3}) - (8.621949 * \frac{10^{11}}{T^4})$$
(4.5)

GPP only occurs during daytime, hence the daytime length was calculated from photosynthetically active radiation (PAR) data which was estimated using the following function:

$$P(t) = amp * (\sin(\omega t + phase)) \cdot * (amp * \sin(\omega t + phase) > 0)$$

$$(4.6)$$

where amp to 1600 μ mol m⁻² s⁻¹, $\omega = 2\pi/24$, and phase to $-\pi/2$.

We used the streamMetabolizer model's generic prior probability distribution functions as initial values for *GPP* and *ER* (i.e., 8 ± 4 and 10 ± 5 g O₂ m⁻² d⁻¹, respectively). We followed Tassone et al. (2017) for the initial estimates of daily reaeration (log K) of 1.79 ± 1 d⁻¹.

4.3.4 Statistical analysis

To determine differences among metabolism results across sites, we used one-way ANOVA for GPP, ER, followed by Tukey's multiple comparison test. To determine the relationship between each metabolism metric (i.e., GPP, ER, and NEP) and urbanization metric (i.e., ULUG and flashiness index), we used the two-phase linear regression model of Atanasov & Stoimenova (2017). This model detects if there is a threshold in the data after which the linear trend changes form. The ANOVA and linear regressions analyses were completed using MATLAB and the Statistics Toolbox (Release 2016b, The MathWorks Inc., Illinois).

4.4 Results

4.4.1 Urban land use gradient and flashiness indices

Sites were numbered according to the value of the ULUG index, with site 1 as the least urbanized site (ULUG index = 0) and site 18 as the most urbanized site (18) (Figure 20, Table IX). Flashiness index ranged from 0.09 at site 6 to 1.33 at site 15 (Table IX) and was unrelated to basin area (p=0.09; r^2 =0.224). The two-phase linear-regression results of *NEP* versus ULUG index showed a break point (i.e., threshold) at a value of 23.9 ULUG index (Figure 21). Based on this result, we divided the sites into two groups: sites with ULUG index < 23.9 sites with ULUG index > 23.9. The first group included the sites 1 to site 8, and the second group included the sites 10 to 18. There was no difference in the flashiness index of the two groups (1-way ANOVA; F= 0.15, p =0.701; Table IX).



Figure 21. The relationship between ULUG Index values and NEP to determine the threshold (i.e., ULUG index=23.88) value of the response trend.

4.4.2 Whole-stream metabolism

Metabolism rates differed significantly among sites (1-way ANOVA; GPP: F= 251.54, p < 0.01; ER: F= 176.8, p < 0.01). Mean daily GPP ranged from 0.21 g O₂ m⁻²d⁻¹ at Site 11 to 18.71 g O₂ m⁻²d⁻¹ at Site 9. Mean daily ER ranged from -0.89 g O₂ m⁻²d⁻¹ at Site 11 to -12.83 g O₂ m⁻²d⁻¹ at Site 7. Mean daily NEP ranged from -9.10 g O₂ m⁻²d⁻¹ at Site 18 to 5.93 g O₂ m⁻²d⁻¹ at Site 9 (Table X, Figure 22).

Site #	GPP	ER	NEP	n
1	3.38 ± 1.14	-6.17 ± 2.41	$\textbf{-2.78} \pm \textbf{2.51}$	183
2	0.55 ± 0.27	-2.83 ± 1.74	-2.27 ± 1.81	183
3	1.08 ± 1.21	$\textbf{-7.68} \pm 2.79$	$\textbf{-6.59} \pm \textbf{3.32}$	183
4	3.78 ± 2.66	$\textbf{-9.36} \pm \textbf{2.82}$	$\textbf{-5.58} \pm \textbf{3.62}$	183
5	3.80 ± 2.39	-12.21 ± 2.61	-8.41 ± 3.48	182
6	0.62 ± 0.43	$\textbf{-2.80} \pm \textbf{2.14}$	$\textbf{-2.19} \pm \textbf{2.34}$	182
7	10.25 ± 5.73	$\textbf{-12.83} \pm \textbf{3.09}$	-2.57 ± 5.44	115
8	14.80 ± 7.36	$\textbf{-11.10} \pm \textbf{8.29}$	$\textbf{3.69} \pm \textbf{5.33}$	108
9	18.71 ± 9.22	$\textbf{-12.76} \pm \textbf{9.91}$	5.93 ± 6.43	103
10	1.67 ± 1.05	$\textbf{-2.96} \pm 1.91$	-1.28 ± 2.18	125
11	0.21 ± 0.11	$\textbf{-0.89} \pm 1.02$	$\textbf{-0.68} \pm 1.04$	183
12	1.11 ± 0.78	-3.46 ± 2.52	$\textbf{-2.34} \pm \textbf{2.93}$	183
13	2.61 ± 1.29	-6.16 ± 2.25	-3.56 ± 2.21	182
14	1.16 ± 0.54	-8.63 ± 3.13	-7.47 ± 2.88	183
15	13.93 ± 6.94	$\textbf{-11.47} \pm \textbf{1.81}$	2.45 ± 7.17	183
16	6.32 ± 10.53	-9.22 ± 4.37	$\textbf{-2.99} \pm \textbf{14.44}$	127
17	4.42 ± 2.27	$\textbf{-12.08} \pm 1.93$	-7.66 ± 2.08	183
18	1.15 ± 1.27	-10.25 ± 3.21	$\textbf{-9.10} \pm \textbf{3.50}$	129

TABLE X

MEAN (STD. DEV.) DAILY METABOLISM (G ${\rm O}_2~{\rm M}^{-2}{\rm D}^{-1}),$ FOR THE WHOLE STUDY PERIOD. REFER TO TABLE 4.2 FOR SITES NAMES.

GPP at sites 1-9, corresponding to a lower degree of urbanization, was significantly higher than *GPP* at sites 10-18, corresponding to a higher degree of urbanization (1-way ANOVA; *GPP*: F = 6.83, p = 0.005; Figure 22; Table X). Similarly, *ER* at sites 1-9 (i.e., lower degree of urbanization) was significantly higher than *ER* at sites 10-18 (i.e., higher degree of urbanization)

(1-way ANOVA; *ER*: F= 3.73, p = 0.045; Figure 22; Table X). Accordingly, *NEP* at sites 1-9 (i.e., lower urbanization) was significantly higher than *NEP* of sites 10-18 (i.e., higher urbanization) (1-way ANOVA; *NEP*: F= 5.26, p =0.021; Figure 22; Table X). Sites with higher degree of urbanization were net heterotrophic (*GPP/ER* < 1) for most of the measurements (~ 94.9%), while sites with lower degree of urbanization was net autotrophic for ~ 10.4% of the measurements (Figure 23).



Figure 22. Mean daily GPP and ER in g $O_2 m^{-2} d^{-1}$ for the 18 sites color coded based on the ULUG index (i.e., sites with low urban intensity has ULUG index ≤ 23.88 , and sites with high urban intensity has ULUG index ≥ 23.88



Figure 23. GPP versus ER for (A) sites with ULUG index < 23.8 and (B) sites with ULUG index > 23.8

4.4.3 The effect of urbanization indices on stream metabolism

For the sites with ULUG < 23.8 (i.e., lower urbanization), GPP was positively related to the ULUG index (p=0.051; r² =0.390; Figure 24A; Table XI). On the other hand, sites with ULUG > 23.8 GPP showed no significant relationship with the ULUG index (p=0.581; $r^2=0.039$; Figure 24B; Table XI). Similarly, *ER* increased significantly with ULUG index for sites <23.8(p=0.041; r²=0.461; Figure 24C; Table XI) and had no significant relationship for sites with ULUG index >23.88 (p=0.228; r²=0.175; Figure 24D; Table XI). As a result, no significant relationship existed between *NEP* and ULUG index for sites with ULUG <23.8 (p=0.371; r²=0.115; Figure 24E; Table XI), while *NEP* decreased significantly with ULUG index >23.8 (p=0.035; r²=0.455; Figure 24F; Table XI). No significant relationships existed between stream metabolism metrics (i.e., *GPP*, *ER* and *NEP*) and the flashiness index (Figure 25A; Table XI).





(A) sites with low urban intensity (ULUG index < 23.8) and (B) sites with high urban intensity (ULUG index > 23.8), ULUG index and ER for (C) sites with ULUG index < 23.88and (D) sites with ULUG index > 23.8, and ULUG index and NEP for (E) sites with ULUG

index < 23.88 and (F) sites with ULUG index > 23.8



Figure 25. The relationship between: flashiness index (ULUG) and *GPP* for (A) sites with low urban intensity (ULUG index < 23.8) and (B) sites with high urban intensity (ULUG index > 23.8), FI index and *ER* for (C) sites with ULUG index < 23.88 and (D) sites with ULUG index > 23.8, and FI index and *NEP* for (E) sites with ULUG index < 23.88 and (F) sites with ULUG index > 23.8

	Urban intensity	Variables	Linear Regression	р	r^2	df
		GPP and ULUG index	<i>GPP</i> = 0.527 ULUG - 1.638	0.050	0.390	9
Low		ER and ULUG index	<i>ER</i> = 0.393 ULUG + 3.121	0.041	0.461	9
		NEP and ULUG index	<i>NEP</i> = 0.2 ULUG - 5.3635	0.371	0.115	9
	Low	GPP and flashiness index	<i>GPP</i> = -11.267 FI + 11.759	0.239	0.190	9
		ER and flashiness index	<i>ER</i> = -9.098 FI + 13.447	0.157	0.263	9
		NEP and flashiness index	<i>NEP</i> = -3.454 FI - 0.668	0.621	0.041	9
		Flashiness index and ULUG index	FI = 0.001 ULUG + 0.454	0.913	0.001	9
		GPP and ER	<i>GPP</i> = 0.867 <i>ER</i> - 1.495	<0.001	0.455	2052
_		GPP and ER GPP and ULUG index	<i>GPP</i> = 0.867 <i>ER</i> - 1.495 <i>GPP</i> = -0.0512 ULUG + 7.14	<0.001 0.581	0.455 0.039	2052 10
_		GPP and ER GPP and ULUG index ER and ULUG index	$GPP = 0.867 \ ER - 1.495$ $GPP = -0.0512 \ ULUG + 7.14$ $ER = 0.080 \ ULUG + 4.525$	<0.001 0.581 0.228	0.455 0.039 0.175	2052 10 10
_		GPP and ER GPP and ULUG index ER and ULUG index NEP and ULUG index	<i>GPP</i> = 0.867 <i>ER</i> - 1.495 <i>GPP</i> = -0.0512 ULUG + 7.14 <i>ER</i> = 0.080 ULUG + 4.525 <i>NEP</i> = -0.149 ULUG + 3.488	<0.001 0.581 0.228 0.035	0.455 0.039 0.175 0.455	2052 10 10 10
_	High	GPP and ER GPP and ULUG index ER and ULUG index NEP and ULUG index GPP and flashiness index	$GPP = 0.867 \ ER - 1.495$ $GPP = -0.0512 \ ULUG + 7.14$ $ER = 0.080 \ ULUG + 4.525$ $NEP = -0.149 \ ULUG + 3.488$ $GPP = 5.082 \ FI + 1.483$	<0.001 0.581 0.228 0.035 0.381	0.455 0.039 0.175 0.455 0.091	2052 10 10 10 10
_	High	GPP and ER GPP and ULUG index ER and ULUG index NEP and ULUG index GPP and flashiness index ER and flashiness index	GPP = 0.867 ER - 1.495 $GPP = -0.0512 ULUG + 7.14$ $ER = 0.080 ULUG + 4.525$ $NEP = -0.149 ULUG + 3.488$ $GPP = 5.082 FI + 1.483$ $ER = 0.873 FI - 8.874$	<0.001 0.581 0.228 0.035 0.381 0.482	0.455 0.039 0.175 0.455 0.091 0.005	2052 10 10 10 10 10 10
_	High	GPP and ER GPP and ULUG index ER and ULUG index NEP and ULUG index GPP and flashiness index ER and flashiness index NEP and flashiness index	GPP = 0.867 ER - 1.495 $GPP = -0.0512 ULUG + 7.14$ $ER = 0.080 ULUG + 4.525$ $NEP = -0.149 ULUG + 3.488$ $GPP = 5.082 FI + 1.483$ $ER = 0.873 FI - 8.874$ $NEP = 5.821 ULUG - 7.247$	<0.001 0.581 0.228 0.035 0.381 0.482 0.233	0.455 0.039 0.175 0.455 0.091 0.005 0.171	2052 10 10 10 10 10 10
_	High	GPP and ERGPP and ULUG indexER and ULUG indexNEP and ULUG indexGPP and flashiness indexER and flashiness indexNEP and flashiness indexFlashiness index and ULUG index	GPP = 0.867 ER - 1.495 $GPP = -0.0512 ULUG + 7.14$ $ER = 0.080 ULUG + 4.525$ $NEP = -0.149 ULUG + 3.488$ $GPP = 5.082 FI + 1.483$ $ER = 0.873 FI - 8.874$ $NEP = 5.821 ULUG - 7.247$ $FI = -0.003 FI + 0.757$	<0.001 0.581 0.228 0.035 0.381 0.482 0.233 0.648	0.455 0.039 0.175 0.455 0.091 0.005 0.171 0.021	2052 10 10 10 10 10 10 10

TABLE XI

RESULTS OF THE TWO-PHASE LINEAR REGRESSION FOR SITES WITH LOWER URBAN INTENSITY (URBAN LAND USE GRADIENT INDEX (ULUG) < 23.88) AND SITES WITH HIGHER URBAN INTENSITY (ULUG > 23.88)

4.5 Discussion

4.5.1 Urban and flashiness indices variations

To our knowledge, the present analysis is the first attempt to examine the influence of urbanization on stream metabolism by quantifying urbanization using multiple watershed characteristics (e.g., ULUG). We investigated changes in stream metabolism for 18 sites across a gradient of urbanization characterized by the ULUG index (McMahon and Cuffney, 2000). We found a threshold of 23.8 ULUG index where the response of studied variables (i.e., stream flashiness and NEP) changed from a positive to a negative relationship with the ULUG index. This was lower than the threshold of 35 ULUG index detected by (Coles et al., 2004), who studied the influence of urbanization on water quality. The reason for the different threshold between our study and Coles et al. (2004) can be due to the variability in sites. Similar to Coles et al. (2004), we found limited changes to variables of interest (e.g., GPP and ER) in response to increasing in urban intensity above this threshold Indicating that the highest change in metabolism in response to increased urban intensity occurs at low to moderate urban intensity levels.

Similar to the findings in the previous literature (Morley and Karr, 2002; Coles et al., 2004), the relationship between urban intensity and stream flashiness was not significant. It is possible that the existence of dams, wetlands, or ponds have reduced the stream flashiness near the study sites. Moreover, we found no significant difference between flashiness index of sites low urban intensity (i.e., ULUG index < 23.8) and sites of high urban intensity (i.e., ULUG index >23.8) indicating that some other physical attributes of the landscape (i.e., green infrastructure) impacted stream flashiness that were not accounted for using the ULUG index. Therefore, a suggested future study might be exploring other factors influencing stream flashiness and add it to the definition of ULUG index.

We compared our results to previously published flashiness index values. McMahon et al. (2000) calculated the flashiness index for 515 Midwestern streams from 1975 through 2001. Out of the 515 studied streams, we compared our flashiness index values to sites with drainage area of less than 500 km². Similar to their findings, we found no significant difference between our values for flashiness index and their values for non-urban streams (1-way ANOVA; F=0.03; p=0.863). The similarity between flashiness index of our study sites (urban land use) and non-urban land use indicates that the flashiness index might not be a good representation of urbanization.

4.5.2 Stream metabolism of urban streams

The results of our metabolism calculations showed that all sites were biologically active and metabolism rates (i.e., GPP, ER, and NEP) were within the published values of previous literature in urban streams (e.g., Meyer et al. 2005, Von Schiller et al. 2008, Bernot et al. 2010, Sudduth et al. 2011). Our findings were consistent with the predicted outcome of the response of stream metabolism to urbanization in (Figure 19). Lower levels of GPP at higher urban intensity compared to lower intensity might be due to the reduction in canopy cover due to the removal of trees and therefore, increasing light intensity to reach the benthic communities which can enhance GPP levels (Bernot et al. 2010) and consistent with the findings of Fellow et al. (2006) and Young et al. (2008). Overall, our results showed that the high urban intensity was associated with a low levels of in-stream GPP, consistent with the findings of previous literature (e.g., (Kennen, 1999), (May et al., 1999), (Morley and Karr, 2002), Coles et al. 2004).

The existence of point source discharges can input nutrients (Hatt et al. 2004, Sudduth et al. 2011, Bernhardt et al. 2008) that might enhance GPP levels in urban streams (Izagirre et al. 2008). Similarly, low urban intensities were associated with increased ER levels which can also be due to the input of nutrients (Hoellein et al. 2017) and BOD from point source discharges (Izagirre et al. 2008). Coles et al. (2004) showed that the increase in urban intensity can cause increases in water temperature due to the heat from roads, parking lots and heat islands from cities (Galli et al. 1991, LeBlanc 1997, Paul and Meyer 2001). Therefore, an increase in ER levels is expected to occur as urban intensity increases since temperature is one of the main factors enhancing ER (Bott et al. 1985; Hill et al. 2000). Moreover, industrial land use can also increase ER by increasing exposure to labile organic matter (Paul 1999, Izagirre et al 2008).

On the other hand, highest urban intensities were associated with low GPP and ER not differing. One probable reason can be due to the discharge of contaminants that can reduce metabolism levels (Hatt et al. 2004, Sudduth et al. 2011, Bernhardt et al. 2008). Another reason can be to the reduction in light intensity due to the built environment and with the existence of higher flows that can scour the algae and therefore, reducing GPP levels, which is consistent with the findings of Meyer et al. (2005).

As a result of GPP and ER pattern, NEP tends to increase with the increase in urban intensity up to the threshold of 23.8 ULUG. This suggests that low to moderate levels in urban intensity can enhance GPP more than ER. On the other hand, at higher levels of urban intensity above a threshold of 23.8 ULUG, urbanization reduces GPP at a higher rate and therefore, results in decreasing overall NEP. Unlike the results of Coles et al. (2004), stream metabolism changed the most at higher levels of urban intensity (> 23.8 ULUG index) where the slope of the regression between NEP and ULUG index was steeper.

Unexpectedly, our results showed no significant relationship between the increase in flashiness and the reduction in GPP; however, a more significant relationship existed between increased flashiness and the reduction in ER. One probable reason might be that storm events can import organic matter which can fuel ER as previous studies showed increased ER resistance with the increase in storm flashiness due (Qasem et al. 2018 under review). According to the response of GPP and ER to stream flashiness, NEP tends to decrease with the increase in stream flashiness by decreasing ER more than GPP.

4.5.3 Continuous metabolism data confirm the predominance of heterotrophy

Flowing waters are often heterotrophic, since allochthonous carbon is critical to support food webs and nutrient transformations (Mulholland et al. 2001, Battin et al. 2008, Rosenfeld and Mackay 1987). Similar to the findings of Uehlinger (2006), sites with moderate urban intensity (ULUG index close to 23.8) reported some autotrophy. Moderate level of urban intensity might be associated with the highest-nutrient, most open-canopy, coarser-bed stream therefore, an increase in stream autotrophy can be expected. On the other hand, and similar to previous studies (e.g., Roley et al. 2014), highly urbanized sites are rarely autotrophic (NEP > 1) which might be due to the higher input of contaminants (Hatt et al. 2004, Sudduth et al. 2011, Bernhardt et al. 2008) and the reduction in light availability compared to moderate urban intensity sites..

The significant relationship between GPP and ER at both lower and higher urban intensity groups suggests that these streams depend on primary production as an important carbon source (i.e., autochthonous energy source), which is expected as a result of the disconnection between streams and terrestrial areas due to the built environment (Bernot et al. 2010). Moreover, the urban stream syndrome suggests that with the increase in urbanization, GPP is expected to increase due to the increase in light availability and ER is expected to decrease due to the reduction in organic matter (Meyer et al. 2005, Izagirre et al. 2008, Bernhardt et al. 2008, Hatt et al. 2004), causing a shift from allochthonous to a more autochthonous system. Our results showed, however, that this might happen at only low to intermediate urban intensity, but not at higher urbanization intensities where a reduction in GPP and an increase in ER might occur.

4.6 Conclusion

Human impacts in urban streams can alter stream metabolism, but how stream metabolism respond to the increase in urban intensity is poorly understood. In this study we analyzed the influence of increased urban intensity on metabolism metrics (i.e., GPP, ER and NEP). We calculated stream metabolism at 18 sites across a gradient of urban intensity represented by the urban land use gradient index ULUG (McMahon et al. 2000) and the flashiness index FI (Baker et al. 2004). We found a threshold of 23.8 ULUG index where the response of GPP and ER no longer exists following this threshold and NEP response changed from positive to negative. Moreover, we found that stream respiration increased with the increase in stream flashiness despite the degree of urbanization. Over all, we found that increased urban intensity confirms increased heterotrophy by reducing GPP more than ER.

CHAPTER 5

CONCLUSIONS

5.1 Results

This dissertation provides insights on the effects of many aspects of urbanization (i.e., the degree of urban intensity, flashy hydrology, and dam removal restoration in urban areas) on stream metabolism. In Chapter 2, I calculated stream metabolism at seven sites around West Branch DuPage River, East Branch DuPage River, and Salt Creek, DuPage, IL. In addition, I quantified the effects of flood events on metabolism by calculating metabolism resistance and resilience to floods. The main finding of Chapter 2 is that flood events are a main controlling factor on stream metabolism (i.e., GPP and ER). I also found that urban streams are characterized by high metabolic activity despite the many stressors they face. My results supported previous studies and showed that ER was more resistant to flood events than GPP, and sometimes flood events increase ER levels by importing organic matter from upstream. In addition, results showed significant reduction in GPP and ER recovered rapidly to pre-flood levels suggesting that metabolism is less resistant more resilient to flood events in urban streams. Overall, the findings of this chapter showed that increased flashiness in urban streams sustain heterotrophic conditions more than autotrophic conditions.

In Chapter 3, I analyzed the influence of dam removal on stream metabolism by calculating GPP and ER before and after the removal of the Good Hope, a low-head dam in Conodoguinet Creek urban watershed in Pennsylvania. I analyzed three sites: upstream of the dam (35.1 m), downstream of the dam (38.6 m) and at a reference station where hydrology was unaffected by the dam (4 km upstream). I found that dam removal increased GPP and ER levels at the upstream and downstream dam removal sites, immediately after removal of the dam. GPP likely increased due to increased light availability through lowering water level at the upstream site, and similar behavior at the site right downstream of the dam can be due to the lotic conditions that could remove the overgrowth of algae in the settled water that could block light from reaching the benthic communities. Increased ER levels could be due to increased temperature as a typical effect of dam removal (Risley et al 2012) in addition to importing organic matter from upstream locations. Simultaneously, the reference site showed only a small seasonal and gradual decline in GPP and ER across the same months. In addition, metabolism in the upstream and downstream sections of the restored rivers showed similar GPP and ERrates to each other and to a reference site following dam removal. Moreover, dam removal moves the streams towards autotrophy by increasing GPP more than ER. Overall, we found that dam removal can restore aquatic ecosystems by enhancing metabolism rates even in non-production seasons (i.e., winter).

In Chapter 4, I calculated stream metabolism across 18 sites located on 13 watersheds around Wisconsin, Illinois, Indiana, Missouri, and Kansas states, USA in for six months in summer/fall periods. Moreover, I related metabolism results to urbanization variables (i.e., flashiness index and ULUG index). A significant decrease in GPP and ER was observed in highly urbanized sites compared to low and intermediate sites. Moreover, we found that urbanization results in further heterotrophy by reducing GPP more than ER, particularly at highly urbanized sites. A threshold of about 28.8 ULUG index was observed were the the relationship changed from increasing GPP and ER with ULUG to decreasing GPP and ERwith ULUG. Following this threshold the relationship became weaker showing that the stream metabolism changed the most at low to moderate urban intensity levels. Overall, I found that stream metabolism calculations in urban streams indicated that urbanization is a major controlling factor in altering stream metabolism, and furthermore that ULUG index is effective in defining a gradient in urbanization.

5.2 Future Work

There is an ongoing need for understanding stream metabolism to help decision makers on the best practices of stream restoration in urban areas. Decades of measurements of stream metabolism has improved our knowledge of ecosystem function. However, these measurements has also highlighted the complexity of factors influencing stream metabolism in urban streams. Some research paths, discussed below, are suggested to enable research in ecosystem metabolism.

1. Continuous long-term monitoring.

One of the main limitations for this dissertation was data availability. For stream metabolism calculations, continuous measurements of DO, temperature, discharge and water depth

are required. The limited availability of such data was an obstacle in my calculations and limited the number of sites included in both dam removal and urban intensity analysis. Long-term monitoring can facilitate a good platform for comparing stream metabolism across regions and seasons, and it can provide more robust conclusions about stream metabolism in urban streams. Therefore, I recommend continuous monitoring of DO and temperature when conducting a dam removal project. In addition, increasing the number of USGS sites that collects DO and temperature data especially in urban environments. These data can be used to address the response of stream metabolism to other factors in urban streams including: wastewater effluent and climate change on a global scale. Moreover, continuous monitoring associated with dam removal can help decision makers in evaluate the effectiveness of their restoration approaches.

2. Multiple approaches and methods.

No single method of stream metabolism calculations is adequate for all situations. In this dissertation, only the single station whole stream metabolism was used for all metabolism calculations. Although the two station approach is limited to data availability between two stations, it may be beneficial to assess the performance of the two approaches in urban stream metabolism calculations. Moreover, the reaeration coefficient is an important parameter in calculating the whole stream metabolism. I recommend the Bayesian approach as a very promising technique in reducing the uncertainty of reaeration calculations however, a worth study might analyze the difference between Bayesian, nighttime regression

method and reaeration empirical equations.

3. Larger-scale metabolism calculations.

Stream metabolism in urban streams faces multiple stressors which makes it challenging to draw clear conclusions on its response to hydrologic disturbances, like high flow events. Although I analyzed the influence of urban intensity on stream metabolism on multiple sites, a larger number of sites and longer-term metabolism calculations are warranted. For example, future studies might conduct metabolism calculations across the entire USA and compare urban stream metabolism across multiple regions and climates. Although the results of ULUG index is effective in quantifying urbanization, a larger number of sites are needed to confirm the results. Another interesting study might calculate stream metabolism across all USGS sites that has sufficient data, then relate those results to ULUG index of each site in order to guide restoration management in urban streams.

4. Different land use comparison.

Despite the complex factors facing stream metabolism in urban areas, some research showed that there is similarity between metabolism response to urban and agricultural land use. These sites are characterized by reduced biodiversity as some species prefer specific levels of metabolic activities. Future research can include a comparison between metabolic activity in urban versus agricultural land use across multiple sites for multiple seasons and years. Moreover, a comparison of the response of stream metabolism to flow disturbances (i.e., flood events) across the two land-use types is warranted.

5. Metabolism response to climate change.

The response of stream metabolism to climate change (i.e., increased water temperature and altered flow regimes) remains another largely unexplored topic. Furthermore, the response of larger rivers to increased water temperature and altered flow regimes might be different than small streams. A good approach to address the influence of climate change is to conduct a multi-region comparison studies. Another consequence of climate change is intensification of the hydrologic cycle. This can be directly related to stream metabolism as I showed that storm events are a major controlling factor on stream metabolism. More intense storm events can result in further disturbing metabolic communities which leads to disturbing the entire food web. Furthermore, increased draught might limit metabolic activity. Therefore, a suggested future study might be investigating the response of stream metabolism in changing environments over many years or even decades calculations of stream metabolism. APPENDICES

Eq.	Author	Abbre	e vK	System
1	O'Connor and Dobbins (1958)	OD	$K = 3.93 \frac{U^{0.5}}{H^{1.5}}$	Conceptual
2	Churchil et al (1962)	CH	$K = \frac{5.026U}{H^{1.67}}$	Large Rivers
3	Owens el al. (1964)	OW	$K = 5.32 \frac{U^{0.67}}{H^{1.85}}$	Small and Large Rivers
4	Langbein and Durum (1967)	LD	$K = 5.134 \frac{U}{H^{1.33}}$	Large Rivers
5	Bennett and Rathburn (1972)	BR	$K = 5.5773 \frac{U^{0.607}}{H^{1.689}}$	Large and small rivers
6	Bansal (1973)	BA	$K = 4.1528 \frac{U^{0.6}}{H^{1.4}}$	Medium to large rivers
7	Baecheler and Lazo (1999)	BL	$K = 1.923 \frac{U^{1.325}}{H^{2.006}}$	Mountains rivers
8	Jha et al. (2001)	$_{\rm JH}$	$K = 5.792 \frac{\sqrt{U}}{H^{0.25}}$	River
9	Isaacs and Gaudy (1968)	IG	$K = 4.7531 \frac{U}{H^{1.5}}$	Recirculating cylindrical flume
10	Eloubaldy (1969)	\mathbf{EL}	$K = 4.05 \frac{U}{H^{1.5}}$	Recirculating
11	Isaacs et al. (1969)	IS	$K = 3.6 \frac{U}{H^{1.5}}$	Recirculating cylindrical
12	Negulescu and Rojanski (1969)	NR	$K = 10.9 [\frac{U}{H^{1.5}}]^{0.85}$	Recirculating
13	Padden and Gloyna (1972)	\mathbf{PG}	$K = 4.54 \left[\frac{U}{H^{1.5}}\right]^{0.703}$	Recirculating
14	Krenkel and Orlob (1962)	KO	$K = 173(SU)^{0.404}H^{-0.66}$	Recirculating
15	Cadwallader and McDonnel (1969)	CM	$K = 186(SU)^{0.5}H^{-1}$	-
16 17	Tsiovoglou and Neal (1976) Grant (1976)	TN	K = 3170S K = 22700 SU	Streams Small Streams
19	The second stal (1087)	TH	$K = 2270050$ $K = 2784 U^{0.734} S^{0.93}$	Small streams
10	Smooth (1088)	SM	$K = 6764 \frac{-1000}{H^{0.42}}$ $K = 542 \text{c}^{0.6236} \text{c}^{10.5325} \text{c}^{1-0.7258}$	Small streams
20	Mogg and Jirka (1998)	MI	K = 5435 C $HK_{\pi} = 1740 U^{0.46} S^{0.79} H^{0.74}$	-
21	Melching and Flores (1999)	MF	$K = 596(US)^{0.528}Q^{-0.136}$	Large rivers
22	Tackston and Krenkel (1969)	TK	$K = 0.000125(1 + F^{0.5})\frac{u^*}{U}$	and streams Large rivers
23	Eloubaldy (1969)	EL	$K = 154(\frac{u^*}{H})$	Recirculating
24	Lau (1972)	LA	$K = 2506.7 \frac{U}{H} (\frac{u^*}{U})^3$	flume Large rivers
25	Parkhurst and Pomerov (1972)	PP	$K = 23.04 \frac{(1+0.17F^2)(SU)^{0.375}}{U}$	Streams, Rivers
26	Alonso et al. (1975)	AL	$K = 123 \frac{u^*}{H}$	Recirculating
27	Thyssen and Jeppesen (1980)	TJ	$K = 23000 \frac{U^{0.76} (1+F)^{2.66} S^{1.13}}{H^{0.60}}$	Small streams
28	Takston and Dawson (2001)	TD	$K = 0.000025(1 + 9F^{0.25})\frac{u^*}{H}$	Large rivers
29	Gualtieri and Gualtieri (2004)	GG	$K = [(D_m)^{2/3} (\frac{gS}{2vR_{g-t}})^{1/3}]/H$	Flume
U is	s the mean stream velocity, m/s	u^* : flu	id shear velocity, m/s	
H:	water depth, m	g: acce	eleration due to gravity, m/s^2	
Q:	discharge, m^{\prime}/s	F: Fro	bude number, dimensionless m^2/c	
R_{g}	$_{-t}$: gas-transfer Reynolds number, dimensionless	v: kine	etic velocity of fluid, m^2/s	

TABLE XII

REAERATION COEFFICIENT EMPIRICAL EQUATIONS (HAIDER ET AL., 2013)

.1 Definitions of metabolism resistance to storms

Multiple resistance definitions were tried in this analysis of GPP and ER resistance to floods. We used the following definitions:

$$R = \frac{x_{\text{one day after the flood}}}{x_{\text{one day before the flood}}} \tag{1}$$

$$\mathbf{R} = \frac{x_{\text{at the day of the flood}}}{x_{\text{one day before the flood}}} \tag{.2}$$

$$\mathbf{R} = \frac{x_{\text{at the day of the flood}}}{x_{\text{mean seasonal value}}} \tag{.3}$$

$$R = \frac{x_{\text{one day after the flood}}}{x_{\text{mean seasonal value}}}$$
(.4)

Where R is the resistance and x is GPP or ER. Overall, all the approaches showed that GPP resistance tend to decrease with the increase of R-B index and ER resistance tend to decrease with the increase of R-B index. However, only the second definition showed significant relationships. The other three approaches showed results that were not consistent with the visual observation of metabolism metrics calculations results (e.g., GPP has significantly higher resistance than ER, GPP resistance increased with the increase of R-B index at most of the sites). Based on the results illustrated in Figure 26, we decided on using the second definition with the resistance equivalent to the value at the day of the flood divided by the value one day after the flood.



Figure 26. GPP and ER resistance to floods vs R-B index for each flood event for all sites using the four definitions for resistance.

Station	Year	Days	95% Significance	99% Significance	Model	r2
	2009	129	74	56	K=0.0002Q4-0.0056Q3+0.0534Q2-0.1729Q+0.3282	0.12
WBBR	2012	113	74	59	K=-0.0149Q2+0.1125Q+0.0082	0.56
	2013	116	113	110	K = 0.0178Q + 0.1359	0.33
WDMC	2009	98	84	76	K= 0.5175Q-0.515	0.29
WBMG	2013	115	109	103	$K = 0.2191 * Q^{-0.193}$	0.05
WBWD	2012	83	73	60	K = 0.0209Q + 0.0667	0.51
	2013	120	93	82	K=0.1152Q^0.359	0.2
	2010	128	45	24	$K = 0.0172Q^2 + 0.0032Q + 0.1044$	0.58
EBHL	2011	122	54	41	$K = 0.1851Q^{-0.9}$	0.43
	2013	124	77	58	K = 0.3425*Q^-0.319□	0.17
EBBR	2009	126	21	13	$K = 0.0882Q^{0.2585}$	0.26
	2009	45	10	2	K= 0.0189Q+ 0.0158	1
SCBR	2010	115	20	11	$K = 0.6156e^{-0.275Q}$	0.49
	2011	108	28	20	$K = 0.2799 Q^{\wedge 0.212}$	0.05
	2013	128	79	63	$K = 0.3656e^{-0.293Q}$	0.16
SCVP	2009	164	63	32	K = 0.0239Q + 0.1901	0.29
SUIK	2010	128	40	17	K = 0.0309 * Q + 0.131	0.19

TABLE XIII

RELATIONSHIP BETWEEN THE DISCHARGE $\left(Q\right)$ AND THE REAERATION

COEFFICIENT (K) WHEN SIGNIFICANT MODEL EXISTED.
	Response								
	GPP				ER				
Site	ARIMA model structure	Component	Coefficient	SE	ARIMA model structure	Component	Coefficient	SE	
WBBR	(1,1,1)	θ	-0.867	0.045	(1,1,1)	θ	-0.922	0.035	
		φ	0.471	0.039		φ	0.496	0.063	
WBMG	(4,1,1)	θ	-0.999	0.022	(1,1,1)	θ	-0.915	0.052	
		φ	0.683	0.084		φ	0.298	0.085	
		φ	-0.055	0.098					
		φ	0.141	0.079					
		φ	-0.088	0.070					
WBWD	(1,1,1)	θ	-0.845	0.043	(1,1,1)	θ	-0.865	0.062	
		φ	0.483	0.077		φ	0.407	0.101	
EBHL	(1,1,1)	θ	-0.644	0.041	(1,1,1)	θ	-0.943	0.023	
		φ	0.023	0.038		φ	0.398	0.035	
EBBR	(1,1,1)	θ	0.776	0.147	(1,1,1)	θ	-0.766	0.093	
		φ	-0.889	0.115		φ	0.329	0.131	
SCBR	(5,1,1)	θ	-0.921	0.062	(1,1,1)	θ	-1.000	0.013	
		φ	0.349	0.059		φ	0.531	0.041	
		φ	-0.065	0.061					
		φ	0.114	0.050					
		φ	-0.016	0.063					
		φ	-0.061	0.079					
SCYR	(1,1,1)	θ	-0.628	0.055	(1,1,1)	θ	-0.963	0.021	
		φ	0.078	0.065		φ	0.672	0.051	

TABLE XIV

[AUTO-REGRESSIVE INTEGRATED MOVING AVERAGE (ARIMA) MODEL STRUCTURE FOR GROSS PRIMARY PRODUCTION (*GPP*) AND ECOSYSTEM RESPIRATION (*ER*) FOR ALL SITES. ACCORDING TO THE METHODS SECTION, THE MODEL STRUCTURE IS DESCRIBED AS (*P*, *D*, *Q*). SE = STANDARD ERROR, θ IS THE MOVING AVERAGE COEFFICIENT, ϕ IS THE AUTO-REGRESSIVE COEFFICIENT.



Figure 27. A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP, C) ER, and D) NEP in g O₂ $m^{-2} d^{-1}$ for WBBR 2009



Figure 28. A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP, C) ER, and D) NEP in g O₂ $m^{-2} d^{-1}$ for WBBR 2012, see Table1 for sites abbreviations.



Figure 29. A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP, C) ER, and D) NEP in g O₂ $m^{-2} d^{-1}$ for WBBR 2013, see Table1 for sites abbreviations.



Figure 30. A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP, C) ER, and D) NEP in g O₂ $m^{-2} d^{-1}$ for WBMG 2009, see Table1 for sites abbreviations.



Figure 31. A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP, C) ER, and D) NEP in g O₂ $m^{-2} d^{-1}$ for WBMG 2013, see Table1 for sites abbreviations.



Figure 32. A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP, C) ER, and D) NEP in g O₂ $m^{-2} d^{-1}$ for WBWD 2012, see Table1 for sites abbreviations.



Figure 33. A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP, C) ER, and D) NEP in g O₂ $m^{-2} d^{-1}$ for WBWD 2013, see Table1 for sites abbreviations.



Figure 34. A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP, C) ER, and D) NEP in g O₂ $m^{-2} d^{-1}$ for EBBR 2009, see Table1 for sites abbreviations.



Figure 35. A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP, C) ER, and D) NEP in g O₂ $m^{-2} d^{-1}$ for EBHL 2010, see Table1 for sites abbreviations.



Figure 36. A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP, C) ER, and D) NEP in g O₂ $m^{-2} d^{-1}$ for EBHL 2013, see Table1 for sites abbreviations.



Figure 37. A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP, C) ER, and D) NEP in g O₂ $m^{-2} d^{-1}$ for SCYR 2009, see Table1 for sites abbreviations.



Figure 38. A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP, C) ER, and D) NEP in g O₂ $m^{-2} d^{-1}$ for SCYR 2010, see Table1 for sites abbreviations.



Figure 39. A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP, C) ER, and D) NEP in g O₂ $m^{-2} d^{-1}$ for SCBR 2010, see Table1 for sites abbreviations.



Figure 40. A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP, C) ER, and D) NEP in g O₂ $m^{-2} d^{-1}$ for SCBR 2011, see Table1 for sites abbreviations.



Figure 41. A) Daily precipitation (P) in inches (the bar graph), flow (Q) in m^3/s , B) GPP, C) ER, and D) NEP in g O₂ $m^{-2} d^{-1}$ for SCBR 2013, see Table1 for sites abbreviations.

.2 Metablolism Reinvigoration

In many situations during summer period, GPP or ER peak to values double or triple the mean value of the entire period which we called 'reinvigoration'. As it clear in the figures of Appendix B, reinvigoration occurred in late June of 2009 at site WBBR (Figure A.1), in early July of 2012 at site WBBR (Figure A.2), in late June of 2009 at site EBBR (Figure A.8), in early July of 2010 at site EBHL (Figure A.9), and in mid-July of 2013 at site SCBR (Figure A.15). In such situations, the minimum GPP or ER value was not associated with the peak discharge. Based on visual analysis of metabolism results, metabolism reinvigoration is not associated with neither discharge nor precipitation.



Figure 42. GPP and ER Reinvigoration at EBHL

We estimated metabolism reinvigoration (r) as the percent change between GPP or ER peaks (peaks = 2* mean) and the mean value of the entire data for each site separately as given in the following equation:

$$\mathbf{r} = \frac{X_{\text{peak}}}{X_{\text{mean}}} \tag{.5}$$

where X represents GPP or ER.

TABLE XV

GPP AND ER REINVIGORATION REPRESENTED AS THE MEAN (\pm) ST.D.

Sitor	Reinvigoration					
Sites	GPP	ER				
WBBR	2.32 ± 0.23	2.37 ± 0.36				
WBMG	2.80 ± 0.88	3.16 ± 0.60				
WBWD	2.23 ± 0.29	2.20^{*}				
EBHL	3.88 ± 1.40	2.83 ± 0.94				
EBBR	2.80 ± 0.78	2.08^{*}				
SCBR	3.69 ± 1.59	2.64 ± 0.52				
SCYR	2.78 ± 0.80	3.41 ± 0.96				

*No error bars indicates that only one storm event was analyzed.

GPP reinvigorated to about 223% higher than the mean GPP at WBWD, to 388% at EBHL. ER reinvigorated to about 220% higher than the mean value at WBWD to 341% at SCYR site. This phenomenon occurred during the summer period at all sites. One possible reason for this increase in GPP is the increase in light availability in addition to washing

the overgrowth of algae during summer small storm events, which allowed the light to get to the production communities at the stream bed. ER maybe increased due to the increased temperature or supply of organic matter from the adjacent landscape.

	Site# 1							
#	Variable	Value	Units	Adjusment	Maximum	Minimum	Adjusment	
	Infrastructure variables							
1	Number of Toxic release inventory sites /1km2 of basin area		count	0.181159	1.3379872	0	0.135397	
2	Roads density in basin [road length(km)/basin area(km2)]	326.21	length	11.8192	17.28879	2.374599198	0.633263	
3	Number of point source discharges/1km of basin area	0	count	0	0.0618047	0	0	
4	Number of dams/1km of basin area	0	count	0	0.09997	0	0	
	Land-cover variables							
5	Percentage of drainage area in open water	0.01215	area	0.044022	8.4844547	0	0.005189	
6	Percentage of drainage area in developed open space	2.991	area	10.83696	112373.78	8.243164631	2.31E-05	
7	Percentage of drainage area in low-intensity residential	12.598	area	45.64493	209.18881	8.647294589	0.184489	
8	Percentage of drainage area in Medium-intensity residential	9.34	area	33.84058	94.816131	0.741482966	0.351839	
9	Percentage of drainage area in high-intensity residential	2.591	area	9.387681	46.157293	0.085170341	0.201912	
10	Percentage of drainage area in barren land	0	area	0	0.5107621	0	0	
11	Percentage of drainage area in deciduous forest	0.0356	area	0.128986	17.064881	0	0.007559	
12	Percentage of drainage area in evergreen forest	0	area	0	0.2475375	0	0	
13	Percentage of drainage area in mixed forest	0.011	area	0.039855	3.3250927	0	0.011986	
14	Percentage of drainage area in shrub	0	area	0	4.3598571	0	0	
15	Percentage of drainage area in herbaceous	0.0178	area	0.064493	6.4335232	0.005960265	0.009106	
16	Percentage of drainage area in pasture/hay	0.0139	area	0.050362	33.199162	0	0.001517	
17	Percentage of drainage area in cultivated crops	0	area	0	429.17768	0	0	
18	Percentage of drainage area in woody wetlands	0.0054	area	0.019565	8.8136642	0.019565217	0	
19	Percentage of drainage area in emergent herbaceous wetlands	0	area	0	1.3949971	0	0	
	Population							
20	Total Population	132899		48.15181	48.151812	4.79237664	1	
						Average Value of all metrics	0.127114	
						Maximum value of all sites	0.548825	
						Minimum value of all sites	0.101837	
						Index adjusted to 0-100 range	5.654947	

TABLE XVI

VARIABLES USED FOR CALCULATING THE URBAN LAND USE INDEX AND

SAMPLE CALCULATIONS FOR SITE1

CITED LITERATURE

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