From headwaters to receiving waters: river dynamics in an increasingly urban world

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Abstract

Rivers historically played, and continue to play, a fundamental role in supporting humanity through provisioning numerous resources and services. Through time, research has aimed at understanding rivers through hydrological, geomorphological, and energic lenses to determine how such river dynamics combine to structure aquatic ecological communities. This research has led to the development of various conceptual models to describe river dynamics and ecological community structure. However, as many urban regions are often built around water sources, rivers are heavily altered as hydrological, geomorphological, and energic dynamics changed as land use intensified and human populations increased. Such changes ultimately altered the structure of ecological communities, and the services provided by rivers in urban regions. Here, we review and synthesize natural river concepts and urban river concepts, while emphasizing the importance of considering more natural river dynamics as a guide for river restoration in urban...
Novelly, we connect river dynamics to their terminal receiving waters, as changing dynamics in rivers can ultimately alter receiving water dynamics. Moreover, rather than focus solely on the main river channel and terminal water body, we consider river dynamics and urban impacts to river associated wetlands, riparian zones, and hyporheic zones. Through linking river dynamics from headwaters to receiving waters, we synthesize and extend historical river concepts with more modern understanding of urban river dynamics across numerous aquatic zones. In this work, we highlight broad implications of urbanization and restoration for both academic research and applied management. Finally, we emphasize the potential of urban rivers in facilitating connections to nature for urban residents. With the importance of urban blue space recognized in the recently agreed upon Kunming-Montreal Global Biodiversity Framework, increasing access to, and ecological integrity in, urban blue spaces will require understanding river energy dynamics across the entire river course, from headwaters to receiving waters.

Keywords: urbanization, urban rivers, receiving waters, river continuum concept, urban stream syndrome, riparian zone, river restoration, nature-based solutions

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Introduction

There is a long history of river research aimed at understanding how geomorphology and hydrology impact the physical form of rivers and, in turn, structure riverine ecological communities. Predictable patterns in species distributions and functional roles are commonly
observed along the length of a river and change in response to the river’s physical structure (e.g., river slope, width, depth, substrate). The diverse forms of rivers worldwide have spurred decades of research developing conceptual models linking river form to energy, carbon flow, and biological community feedbacks (Vannote et al. 1980, Newbold et al. 1981, Ward and Stanford 1983, 1995a, Junk et al. 1989, Humphries et al. 2014, Raymond et al. 2016). More recently, conceptual models were extended to urban rivers (Walsh et al. 2005b), since urbanization is recognized as altering the dynamics of rivers. Our goal is to synthesize current concepts of the functioning of natural rivers (i.e., those without significant local anthropogenic impacts) and contrast these with those of urban rivers to inform mitigation and management strategies in our increasingly impervious world. We then consider the divergent characteristics of high-order natural and urban river systems as they relate to influences on large receiving waters because of their important implication to water resources and coastal ecology.

The River Continuum Concept (RCC; Vannote et al. 1980) is one of the most foundational papers on river structure. It conceptualizes the length of a river, from headwaters to the mouth, as a continuous gradient in physical condition (Vannote et al. 1980). These gradients elicit predictable biotic responses with downstream ecological communities being strongly influenced by upstream processes. The RCC is linked to the Nutrient Spiralling Concept initially proposed by Webster (1975). The Nutrient Spiralling Concept posits nutrients enter different abiotic and biotic compartments transforming as they flow continuously downstream, resembling a pattern of multiple spirals travelling unidirectionally (Newbold et al. 1981, Webster 2007). The ‘tightness’ of spirals (i.e., the number of spirals over a fixed distance) is associated with the ability of a stream to utilize nutrients. Vannote et al. (1980) indicated the number of spirals should be maximized under the RCC since communities evolve to capitalize on upstream
inefficiencies in processing of organic matter so minimal energy is lost along the river continuum. The idea of rivers as a longitudinal continuum of physical and biological patterns led to numerous studies empirically testing the main tenets of the RCC, either in agreement with or refuting some of its components (e.g., Winterbourn et al. 1981, Minshall et al. 1983, Doretto et al. 2020).

Advances on river concept developments since the RCC was proposed often included cases where the RCC failed to explain patterns in productivity, niche space, and biodiversity. Such gaps fueled the development of new concepts either extending the RCC to a greater diversity of river forms (e.g., Sedell et al. 1989, Ward and Stanford 1995) or conceptualizing the river form using different geomorphological factors (e.g., Tockner et al. 2000, Junk 2001). For example, Ward and Stanford (1983) recognized few lotic systems of any significant size are free flowing over their entire watercourse and they proposed the Serial Discontinuity Concept (SDC) to account for effects of impoundments (e.g., dams and reservoirs) on the RCC predictions. Junk et al. (1989) introduced the Flood Pulse Concept (FPC) to better account for relationships between the environment and biota in larger rivers with extensive floodplains promoting lateral exchange of energy, nutrients, and organisms between the floodplain and main river. The Riverine Productivity Model (RPM; Thorp and Delong 1994) stressed the importance of instream primary production and riparian litterfall as sources of energy in nearshore areas of large rivers with constricted channels and minimal floodplains. More recently, research has focused on combining river concepts into a unifying conceptual model (e.g. River Wave Model (RWM); Humphries et al. 2014), and incorporating the episodic nature of major hydrological events due to precipitation and snowmelt, into predicting organic matter dynamics (e.g. Pulse-Shunt Concept (PSC); Raymond et al. 2016). The evolution of river concepts over the last forty years since the RCC
was proposed has provided grounds for expanding our understanding of natural systems in a rapidly changing world (see Allen et al. 2020, Doretto et al. 2020 for reviews of river concepts.)

Understanding energy and material flow in more natural rivers is crucial for predicting how increasing anthropogenic pressure through urbanization will alter the functioning of these systems and the ecosystem services they provide. The Urban Stream Syndrome (Walsh et al. 2005b) outlines a suite of impacts causing ecological degradation common amongst urban rivers.

Rivers draining urbanized watersheds show increased hydrological response rates to precipitation and snowmelt events, altered channel stability and morphology, increased average temperature, elevated concentrations of contaminants and nutrients, and reduced species richness with dominance of disturbance-tolerant species (Walsh et al. 2005b). Kaushal and Belt (2012) linked the Urban Stream Syndrome to the RCC through the Urban Watershed Continuum (UWC). They indicated the concepts developed for natural and forested river ecosystems fell short in predicting flow of water, energy, and materials in the headwaters of urban rivers, so they drew heavily upon observations from the Baltimore Long-term Ecological Research site to develop the UWC (Kaushal and Belt 2012). To conceptualize the flow and transformation of energy in a river, they focused primarily on urban infrastructure development that increases the connectivity of the river with its watershed (i.e., drainage, sewers, and transportation networks; “artificially extended riparian zones”). Kaushal and Belt (2012) developed their model from foundational river concepts, they introduced four dimensions through which these flow paths and transformations occur: longitudinally down the river, latitudinally across the riparian zone, vertically from ground water through the water column, and through time. Although this model incorporates the overarching concepts from earlier these foundational river concepts, we feel further examination of the contrasts in physical structure, nutrient dynamics, an energy flow between natural and
urban rivers can be used to both guide research and management, and to recognize the implications of land use change on the critical role of rivers in nourishing receiving waters. The aim of this synthesis is to further integrate the Urban Stream Syndrome and the Urban Watershed Continuum with natural river concepts outlined above, and to include a strong emphasis on the physical form and kinetic energy dynamics within temperate river systems. Here, we postulate urbanization alters energy transformation/dissipation processes within the river and alters or removes key aspects of the physical environment that profoundly affect ecological communities and the overall functioning of urban rivers. Our synthesis uniquely incorporates impacts to the receiving water bodies which cascade from changes in urban river function. Some of these ideas may be alluded to in other river concepts but remain largely undeveloped. To achieve these goals, we outline the characteristics of a hypothetical natural river having elements common among northeastern North American temperate streams/rivers and found in many north temperate regions globally. We describe how the process of urbanization alters this hypothetical natural stream in terms of structure and functioning (1) within the river channel and (2) outside of the river channel, and (3) biological diversity across river zones. These alterations in river structure and function ultimately impact the materials transported to the receiving waters as rivers play a critical role in nourishing their receiving waters (e.g., a large lake or nearshore ocean), this connection to receiving waters being often overlooked when considering river ecosystems. Finally, we discuss specific mitigation strategies that restore more natural structure and functioning to urban river systems, and their receiving waters.
1. Characterizing a river

1A. What is a ‘natural’ river?

The term ‘natural’ within the context of this paper is defined as watersheds located outside of urban areas, and includes those streams with forestry or agricultural legacy effects (Pijanowski et al. 2007). To create our hypothetical natural river ecosystem, we used the basic conceptual approach outlined by Vannote et al. (1980) in their foundational paper on the River Continuum Concept (RCC) highlighting differences according to longitudinal sections of the river. We discuss spatial differences in rivers using headwaters (orders 1-3), mid reaches (4-6), lower reaches (>6), and terminal receiving waters. Each of these sections, as defined by their stream order (i.e., their size), has its own set of physical, chemical, and biological characteristics dictating energy and material input, nutrient cycling and transformation, and subsequent transport downstream. Receiving waters are strongly influenced by tributaries, with greatest influence on physical and biological conditions on the shoreline adjacent to river mouths which then can extend more broadly over the coastline and water body. Characteristics of the receiving waters may also influence upstream conditions and processes (e.g., nutrient subsidies via migratory fishes). Additionally, we incorporate concepts that evolved from the RCC allowing its core concepts to be extended to larger river systems and floodplains (Junk et al. 1989, Ward and Stanford 1995a, 1995b). We refer to the nutrient spiralling concept as it provides a useful analogy for conceptualizing the various compartments material and energy flow through and an understanding of the characteristics of the suspended load that ends up in receiving waters (Newbold et al. 1981, Webster 2007).

As is typical with temperate rivers, our hypothetical headwaters have higher elevational gradients and smaller catchment size than lower reaches, are constrained by bedrock, have a
forested riparian zone and dense canopy cover, and are fed primarily by groundwater (Ward and Stanford 1995b). Along the course of the river system, from headwaters to mid reaches, the elevational gradient declines, catchment area increases, the river becomes larger, the canopy opens, and there is less influence from ground water resulting in greater temperature variation. The alluvial channels characteristic of mid and lower reaches result in a mosaic of laterally migrating river channels through the floodplain over time (Ward and Stanford 1995b).

Floodplain vegetation in mid and lower reaches is characteristic of forested wetland species tolerant of periodic flooding and able to buffer the effects of intense flood events (Patch and Busch 1984). Our hypothetical river ecosystem will enter the receiving waters through a 7th order reach into a coastal embayment surrounded by wetlands. The receiving waters resemble a large freshwater lake with extensive wetland features along the mainland. Many of these characteristics will be comparable for marine receiving waters, although we highlight differences where appropriate. The nearshore zone is defined by water circulation patterns that produce strong shore parallel currents year-round through mechanisms that change with the season, although marine systems will also experience predictable tidal influences (Makarewicz et al. 2012, Yurista et al. 2016). Figure 1a depicts the various characteristics described of a natural river system.

1B. How has urbanization progressed in watersheds?

Urbanization tended to occur in a predictable sequence of events, with development often located at river mouths or junctions along major rivers as a result of establishment of transportation corridors (e.g. shipping ports) (Kostof and Castillo 2005). As river-mouths became ports and centers for trade and transport, dredging and hardening of shorelines occurred to accommodate larger ships and reduce erosion to shorelines and infrastructure (Hartig and Bennion 2017).
Destruction of wetlands accompanied dredging and hardening, with all three actions leading to degradation of nearshore habitats for organisms, loss of shoreline physical diversity and changing near-shore energy dynamics. As ports developed, and anthropogenic pressure expanded into rivers, clearing of vegetation upstream began as forestry and agriculture predominated (Pijanowski et al. 2007, Tayyebi et al. 2015). Reductions in vegetation cover in watersheds naturally increased erosion rates (Walsh et al. 2005b). Rivers were dammed with small structures to retain water for irrigation or to provide a source of power for mills, a practice dating back to at least the 1600s in North America (Walter and Merritts 2008). By the 1840s it is estimated there were more than 65,000 such mills in the Eastern United States alone (Walter and Merritts 2008). Even small dams and culverts alter the longitudinal characteristics of the river, introducing novel barriers to species dispersal and increasing habitat fragmentation (Perkin and Gido 2012, Fencl et al. 2015). Next, urban landscape replaced the natural landscape further with artificial surfaces and hard infrastructure including roads, sidewalks, and roofs. To protect infrastructure, further engineering was then required through stormwater drains and shoreline hardening to control river patterns and flood plain dynamics, which are areas periodically inundated with water during high flows (Junk et al. 1989). Additionally, streams and smaller rivers were historically placed into physical pipes underground (“stream burial”) to accommodate city growth and maintain safety of public drinking water (Buchholz and Younos 2007). This series of steps transformed the functions of a river and its receiving waters from an interconnected ecosystem capable of using and transforming allochthonous inputs into a disconnected system with a highly variable flow regime and low ability to effectively process material. Urban rivers tend to function as pipes connecting the landscape to the receiving waterbodies. These pipes rapidly ‘shunt’ materials and contaminated waters into rivers and downstream waterbodies during high discharge events (e.g.
Pulse-Shunt Concept; Raymond et al. 2016), which subsequently impact the ecological communities within rivers and their receiving waters. Additionally, accumulation of engineered materials like building materials and household products, and other diverse contaminants in receiving water port-lands, have further aggravated the ecological integrity of urban rivers. Figure 1b depicts the various characteristics of urbanized streams.

1C. How can restoration improve urban river functioning?

With increased recognition of urban pressures on hydrology and aquatic ecology has come an increased interest in restoring urban rivers and receiving waters to more natural and resilient systems (Bernhardt and Palmer 2007). A variety of management methods are available with a shared goal of either reducing the rate of stormwater discharge into rivers and receiving waters or increasing habitat by reintroducing more vegetation. These actions aid in slowing water and increasing infiltration into systems, with stormwater controls dissipating the elevated levels of energy entering urban rivers, allowing for establishment of a more natural flow regime.

Ultimately, in contrast to an artificial, fast-flowing, species-poor river with little heterogeneity and material processing, restoration efforts seek to re-establish functional patterns of material transport and processing to create habitat that supports more desirable ecological communities and enhances ecosystem services. Figure 1c depicts a model of urban river management integrating nature-based solutions and technological advancements.

Below, we first discuss natural river functioning, then examine urban development and its impacts on rivers and near-shore receiving waters, and finally highlight management actions related to mitigating the impacts discussed.
2. Within-stream dynamics: hydrology and production

2A. Natural river hydrology

The kinetic energy within a river is largely determined by elevational gradients increasing the rate of flow of water downslope (Vannote et al. 1980). In natural rivers, water velocity tends to be greatest in high elevational headwaters and lowest in low elevational floodplain reaches near the receiving waters (Vannote et al. 1980).

The high kinetic energy of headwater streams is considered an ‘erosive’ environment resulting in a smooth, V-shaped river channel (Gilvear and Bravard 1996). In many such regions, only boulders and bedrock can withstand the shear stress of the water and remain in-place (Lane 1957), whereas slightly smaller particles like cobble are deposited further downstream.

Transported and deposited along the streambed, these particles dissipate the kinetic energy of water by increasing roughness (Gilvear and Bravard 1996, Hynes 2001) and reduce further erosion of the streambank and streambed (Giller and Malmqvist 1998). Large objects like fallen trees, branches, and other organic matter sometimes accumulate to form debris dams, creating a quiet zone of greatly reduced flow on their downstream edges. This region of low flow/low energy creates an opportunity for smaller inorganic particles like gravel, sand, and silt, as well as organic matter to become embedded and retained in the headwater reaches (Richardson and Danehy 2007) creating habitat heterogeneity, supporting higher levels of biodiversity.

In the mid-reaches of the river, the gradient in elevation is reduced, resulting in a decline in the kinetic energy and velocity of water. This corresponds with an increase in the size and width of the river as more tributaries join the mainstem (Strahler 1957, Vannote et al. 1980). Slowing the flow of water allows deposition onto the substrate of smaller particles like gravel and coarse sand, while smaller particles like fine sands and silt will continue to be eroded and transported.
The widening of the river and deposition of larger particles introduces variation in depth and consequently variation in flow across the channel. Sinuosity of the river channel is created by fluvial action, where higher flows along deeper areas of the channel erode the riverbank forming the outer bend of the channel (Lancaster and Bras 2002). Along the inner slower flowing bend, deposition of particles results in shallow habitats (Lancaster and Bras 2002). These hydrogeologic processes create dynamic habitats with substantial heterogeneity in substrate, habitat, and food availability for biota.

In the lower reaches of the river, the low elevational gradient and slow-moving water creates a depositional environment where only fine particles (e.g. silt and clay) are eroded and transported by the river (Alekseevskiy et al. 2008). Close to the river mouth, there is a strong interaction between the river and the receiving waters. The standing body of water creates a pressure gradient force countering the gravity-driven flow conditions and causes the river flow to decelerate within a transitional region called the backwater zone (Lamb et al. 2012, Fernandes et al. 2016). During low and moderate flow conditions, the backwater zone is characterized by flow deceleration toward the receiving waters and very low sediment transport rates (Fernandes et al. 2016). Flow deceleration and the lateral spreading of the offshore river plume creates a highly depositional environment near the river mouth leading to sedimentation promoting the formation of deltas and deltaic wetlands (Batzer and Baldwin 2012, Lamb et al. 2012). Conversely, during infrequent high flow periods, flow within the backwater zone results in scouring of the riverbed and a deepening of the river channel near its mouth (Lane 1957, Lamb et al. 2012). This dynamic interplay between flow-driven erosional and depositional processes within the backwater zone gives rise to unique benthic habitat.
River mouths are mixing areas where water and associated materials from the lake and the river interact (Larson et al. 2013). This area represents a fundamental transition zone in the pathway of sediment, organic material, and nutrient transport from watersheds to receiving waters. In river mouths, the importance of fluvial processes is diminished with increasing importance of lentic processes and sediment transport is shaped by both upstream (e.g., water flow) and downstream (e.g., storm-surge, changes in lake level) effects (Fernandes et al. 2016). Water movement within a river is typically considered a unidirectional process (Vannote et al. 1980). However, receiving waters can move into river mouths during episodic storm surges and by currents generated from wind-induced thermocline oscillations (internal seiche; Bedford 1992). These processes result in the longitudinal movement of the mixing zone up into the lower reaches of the river, causing a reversal of flow, dilution of sediment, nutrient, and contaminant concentrations (Larson et al. 2013), and potentially major changes in water temperature and light regime. Under low and moderate river flow conditions, mixing between lake and river water may occur within the lower reaches, whereas under high flows, the mixing and deposition areas are pushed lakeward (Larson et al. 2013).

The fate of the constituents of a river plume when they enter the lake are dependent on multiple interacting factors (e.g. dissolved or particulate form of the constituent and its biological reactivity in the lake, the density of river water relative to lake water, lake circulation and thermal stratification) (Rueda and MacIntyre 2010). While mixing gradients may vary widely over time, several generalities can be expected. Within freshwater systems, temperature primarily determines the density of a water mass as opposed to salinity density gradients in marine systems or rivers that are heavily impacted by anthropogenic de-icing salt inputs (Bedford 1992, Rao and Schwab 2007). Thermal gradients are common in river mouths and can
be more pronounced during spring and fall conditions when river water heats or cools, respectively, more rapidly than lake water (Bedford 1992). When the river discharges into the nearshore of a lake with a steep depth gradient, less dense river plumes (e.g., springtime conditions) will remain in the surface layers, whereas more dense river plumes (fall conditions) will sink and begin mixing with lake water. This mixing causes a dilution of the interfacial fluid at the top of the river plume causing it to change density and form intrusions at multiple depths as it sinks and encounters depths of equal density (Cortés et al. 2014).

When the river sediment reaches the lake, the sinking rate of the particulate material is sorted based on particle size and density. The degree of sediment deposition is dependent on the energy regime at the river mouth with the larger particles deposited closest to the river mouth and finer less dense particles distributed further lakeward (Wright 1977). Variation in wind direction and speed, resulting in changes to wave energy and currents, may redistribute these sediment loads over time. Dissolved substances will remain suspended in the river plume, with immediate dispersal patterns determined by the dominating shoreline currents which tend to be parallel to the exposed shores of large lakes. Beyond the shoreline of the lake, the prevailing wind conditions and lake circulation patterns determine the spread of river plumes in large lakes (Rao and Schwab 2007). The mechanisms by which river plumes are trapped in the nearshore vary over the year as thermal structure, and consequently water circulation patterns, change with the seasons. Marine receiving waters are typically larger and contain higher wave action compared to freshwater receiving waters, river plume size in marine systems is substantial and size of internal waves created from river plumes compare to similar findings of tide-topography measurements.
2B. Natural river production

System productivity is the rate at which energy or matter flows from the resource pool through the food web over time, establishing the amount of biomass at each trophic level and the number of trophic levels within an ecosystem (Hairston, and Hairston, 1993). In aquatic ecosystems, the rate of primary production is controlled by availability of light and nutrients (principally phosphorus and nitrogen) and available habitat for producers (Hill et al. 2009). Light availability for aquatic ecosystems can be influenced by season, the amount of canopy cover, and water transparency due to the concentration of dissolved organic matter and amounts of planktonic organisms and suspended solids (Julian et al. 2008). While primary production establishes the available autochthonous material within the ecosystem, allochthonous resources are also a vital component to the overall ecosystem productivity. Microbial energy pathways consuming dissolved organic carbon (DOC) can be also important within the hyporheic zone. The allochthonous resources that enter aquatic ecosystems from adjacent ecosystems can be dependent on climate, geology, and land use patterns. The relative contributions of autochthonous and allochthonous resources to productivity changes along the river length to the receiving waters.

In headwater streams, the dense forest canopy cover shades and strongly reduces available light and in-stream photosynthesis. This reduction creates a heterotrophic environment where community respiration (R) exceeds in-stream primary production (P) (i.e. P/R < 1; Vannote et al. 1980). High in-stream habitat heterogeneity (e.g. fallen trees, branches, boulders, and gravel) promotes nutrient availability in-stream through tighter nutrient spiralling as material enters multiple biotic and abiotic compartments during the downstream transportation process (Newbold et al. 1981). Riparian detritus composed of larger sized particles fuels the food web in
headwater streams (Webster et al. 1999). The breakdown of larger particles (e.g. leaf litter) by a combination of physical, chemical, and biological processes (Webster et al. 1999) results in fine-particulate organic matter (FPOM), dissolved organic matter (DOM), and mineralized nutrients in headwater streams. Fine particulate matter will be quickly taken up by heterotrophic organisms or physical sorption processes (Vannote et al. 1980). This results in headwater reaches having a high proportion of coarse particulate organic matter relative to fine particulate organic matter (CPOM:FPOM) in the transported material.

In mid-reaches, an increase in stream width reduces the degree of canopy cover over the stream, allowing more light to reach the stream surface and penetrate to the bottom. This increases photosynthesis in mid-reaches resulting in elevated levels of primary production compared with the headwaters (Vannote et al. 1980). Macrophytes, macroalgae, and periphyton colonize the river channel where there is suitable substrate available for attachment (Doretto et al. 2020). The reduced stream canopy cover decreases the relative importance of leaf litter input, shifting the system toward a more autotrophic state (i.e. P/R >1; Vannote et al. 1980, Doretto et al. 2020).

During seasonal flooding events, flood pulses can promote high levels of aquatic productivity by increasing the river-terrestrial ecosystem connectivity (Junk et al. 1989, Ward and Stanford 1995b) through large-scale exchanges of organic (e.g., bacteria, phytoplankton, zooplankton, benthos, detritus, and DOM) and inorganic (e.g. sediments) material between the floodplain and the river channel (Cuffney 1988). Food webs within mid-reaches receive CPOM from upstream ‘leakage’ (i.e. inefficiencies in upstream processing), FPOM from in-stream autotrophic sources, and CPOM and FPOM through lateral transport from the floodplain during wet period (Vannote et al. 1980, Ward and Stanford 1995b, Doretto et al. 2020).
Vannote et al. (1980) postulated primary production in lower reaches would be limited by depth and turbidity in the main channel resulting in a heterotrophic environment (P/R < 1). However, the littoral zone of large rivers (Riverine Productivity Model) and their floodplain wetlands (Flood Pulse Concept) provide substantial habitat for autochthonous and allochthonous primary production (i.e. P/R > 1; Junk et al. 1989, Thorp and Delong 1994). The predictability of floodplain inundation promotes the proliferation of certain macrophytes that are adapted to seasonal inundation and variable water levels. These macrophytes maintain high rates of primary production throughout the lower reaches of the river (Hamilton 2009) and provide habitat for periphyton. Additionally, floodplain lakes (e.g., oxbow lakes) can be rich in phytoplankton during the dry phase and provide a substantial flux of allochthonous primary production to the river during seasonal flushing (Hamilton 2009). Attached algae within the littoral zone of large rivers can be an integral part of ecosystem-level primary production, as aquatic consumers gain greater nutritional value of algal cells compared with the structural-rich tissues of vascular plants (Thorp and Delong 1994, Hamilton 2009). The littoral zone, depositional areas where shallow bars form, and side channels are important slow-flowing habitats for capturing and processing refractory CPOM within lower reaches (Thorp and Delong 1994). Thus, autochthonous and allochthonous production of FPOM within lower reaches can be high, and there can be substantial transport of FPOM and CPOM to the receiving waters.

At the receiving waters, the trophic status of a lake is indicative of its overall ecosystem productive capacity and the status is influenced by climate, lake morphometry and nutrient supply (Dillon and Rigler 1975). Relative to rivers, larger temperate lakes are generally more dependent on autochthonous resources as energy and nutrients from within the ecosystem support primary production (Hiriart-Baer et al. 2008, Brett et al. 2017). Consequently, large lakes
tend to have lower levels of suspended materials and dissolved nutrients relative to that of the incoming river water (Larson et al. 2016). This results in primary production of large lakes being strongly limited by nutrients, compared with light and disturbance (due to canopy cover and flow driven turbulence and high turbidity) in river environments (Larson et al. 2013). Nearshore zones of large lakes tend to be areas of high productivity in contrast to offshore zones (Frost and Culver 2001, Makarewicz et al. 2012) due to a combination of physical structures and the allochthonous materials derived from both terrestrial and river inputs. While lakes receive a substantial amount of energy and nutrients from the rivers, some of this energy and nutrients is relocated back into rivers through the movement of organisms such as migratory salmonids, catostomids, and additional fish species (Flecker et al. 2010). Additionally, wetlands along receiving waters can retain, process, and recirculate nutrients and energy, and can represent substantial resource subsidies to nearshore ecosystems.

The mixing of river and lake waters at river mouths results in the formation of river plumes that are enriched with nutrients, DOC, and suspended solids relative to the lake (Larson et al. 2013). Dissolved and biologically available forms of nutrients within positively buoyant river plumes entering the surface mixed layer of the lake will be immediately available for phytoplankton or bacterial growth. In contrast, negatively buoyant river plumes can either form metalimnetic intrusions above the compensation depth (depth where photosynthesis = respiration) or they can sink below the compensation depth interacting with the lakebed. Intrusions entering below the compensation depth can result in nutrients and labile organic matter being absorbed to sediment particles and buried or used by bacteria (Cortés et al. 2014).

River plumes are generally higher in biologically available forms of phosphorus such as soluble reactive phosphorus than lake water, resulting in relatively higher rates of primary productivity.
and plankton biomass in the mixing plumes (Johengen et al. 2008, Makarewicz et al. 2012). Heavier particulate matter settles from the river plumes and provides a food resource for benthic invertebrates localized near the depositional areas adjacent to the river mouth (Smith and Simpkins 2018). Phosphorous entering the lake ecosystem may be bound to particles and not readily available to support plant growth. Resuspension events and lake currents redistribute the phosphorus away from the river mouth until it is broken down and becomes biologically available. Additionally, river plumes transport phytoplankton and other non-nektonic biota originating within the river and its floodplain to the receiving waters, and upon death and decomposition, nutrients become available to lake algal and bacterial assemblages (Kreis et al. 1983) and in other cases seed in-lake populations contributing to productivity within the lake plume. Longshore transport of river plumes may be extensive with mixing areas extending for kilometers away. Nearshore surveys of Lake Ontario during the summer stratified season have shown nutrient-enriched river plumes extending from the river mouth 6 km downwind along the shoreline after precipitation events and elevated river discharge (Howell and Benoit 2021). The nearshore waters during the thermal bar regime, the post-winter period when shoreline water temperature exceeds 4°C while the broader lake 4°C isotherm remains near the shoreline, can be enriched in allochthonous materials derived from tributary discharges during spring runoff and this can be associated with increased levels of phytoplankton biomass in the nearshore (Basterretxea et al. 2018).

Wind-induced sediment resuspension events can redistribute buried river-derived sediments in the water column which can have effects on the biogeochemistry and trophic processes of nearshore ecosystems (Johengen et al. 2008). Episodic resuspension events during storms can result in substantial amounts of sediment being brought up into the water column, releasing
nutrients regenerated within the sediments (Johengen et al. 2008). These resuspension events can stimulate high levels of heterotrophic production along the shoreline (Cotner et al. 2000). Whereas phytoplankton productivity generally increases with sediment-derived nutrient enrichment except for under the most extreme events that cause light limitation from a highly turbid environment (Cotner et al. 2000). The influence of river discharge on lake productivity plays out over diverse spatial and temporal scales, as well as across the biological elements of the lake ecosystem from microorganisms to fish (Larson et al. 2013). Although the most obvious visually, the stimulation of phytoplankton productivity over the plume-mixing areas is likely less important over the longer term than other effects. Instead, the accumulative loading of dissolved nutrients, including the breakdown of these particulate-bound forms dispersed over the lake, will contribute most to system productivity and, in general, sustain the long-term productivity of the system (e.g., Great Lakes; Dolan and Chapra 2012). Benthic autotrophs and invertebrates intercept nutrients and particulate organic material, respectively, distributed to the lakebed over the mixing plumes which influences productivity, species selection and trophic interactions over shorelines adjacent to river mouths (Makarewicz et al. 2012).

The effects on productivity of the discharge to the receiving waters are intensified in the lower reaches of rivers (Larson et al. 2013). Higher relative levels of autotrophic productivity may result in less dissolved phosphorus, however, more of the particulate-bound phosphorus is likely readily broken down, favoring benthic algae in the mixing areas and dispersal of bio-available phosphorus beyond the immediate mixing plume (Chomicki et al. 2016). Similarly, less DOC is likely to be refractory, with a more liable component associated with autotrophic production and microbial activity in the lower river (Dila and Biddanda 2015). Phytoplankton and other micro-
organisms adapted to the conditions in the lower river may jump start growth responses in the lake over the mixing areas and facilitate further biotic responses within the plume, as is the case in riverine estuaries in marine systems (Broadley et al. 2022). In other words, there may be a confluence of populations that under natural conditions facilitate more effective use of the nutrient resources discharged by river into the lake, a novel type of nutrient spiral.

2C. How urbanization changes hydrology and production

Urbanization results in increased amounts of impervious surface cover regardless of the urbanizing location within the watershed (Paul and Meyer 2008). Impervious surface, in the context of urbanization, is an artificial surface covered with materials preventing infiltration of water into soil (Arnold and Gibbons 1996, Scalenghe and Marsan 2009). Reduced infiltration within river watersheds results in more precipitation entering the nearest tributary through overland flow or underground stormwater pipes (Dunne and Leopold 1978, Arnold and Gibbons 1996), increasing the volume and velocity of stormwater and thus increasing flood risk (Leopold 1968). This leads to a flashy hydrograph with rapidly rising and falling limbs, typically with more extreme responses in both directions than prior to urbanization (Arnold and Gibbons 1996, Walsh et al. 2005b). The high velocity and increased kinetic energy of the water during high discharge events in urbanized rivers causes rapid transport (i.e. a “shunt”; Raymond et al. 2016) of water and entrained materials to downstream reaches, and greater erosive forces contributing to large-scale erosion of the riverbank (Arnold and Gibbons 1996, Chin 2006). Urban river systems during high-flow periods represent an erosive environment typical of more natural headwater streams. Eroded sediment is transported downstream until the water velocity declines, and sediment is deposited from larger to smaller particles (Booth and Jackson 1997). With increased velocity of water and the strength of the Pulse-Shunt of energy downstream (Raymond
et al. 2016), increases in sediment size are anticipated at the river-mouth relative to more natural systems where larger particles would be distributed further upstream. However, the influence of the amount and particle size distributions of particulate material loaded to urban rivers from sources atypical of natural rivers such as stormwater and dirt washed from roads, makes particle size distribution downstream difficult to predict. The predictability of particle-size distributions in lower river reaches and at river mouths of urban rivers is also reduced because of the stronger and more frequent episodic high-energy transport events. This predictability may be further reduced with projected changes in climate and hydrological dynamics in North Temperate regions (Nelson et al. 2009), which suggest increased magnitudes of extreme events and increased frequency of the event size currently considered to represent extremes.

Large pulses of water and associated energy (i.e. increased hydrograph amplitude) are expected within natural systems under the Flood Pulse Concept (Junk et al. 1989), and urbanization causes more extreme and more frequent loading events due to the overall increase in discharge within the system. Extreme discharge events also increase the concentrations and loads of suspended solids, which are thus more broadly distributed throughout the year (rather than in spring high flows). The seasonality of river flow is dampened with urbanization, as high flows occur more frequently rather than in natural north-temperate systems which often see high flows occur mostly in spring and fall (Valtanen et al. 2014). Moreover, changes in proportion of urban snow, rain, and freeze-thaw events in winter due to temperature differences (e.g. due to impacts from urban heat island effects and climate change), will lead to higher levels of runoff in the wintertime, with less snowpack remaining for spring high flow events (Ho and Gough 2006), further disrupting the natural seasonality of hydrological regimes and biological cycles of organisms. Interestingly, in arid systems, an opposite effect is found whereby arid urban river
systems become less flashy likely due to more constant discharge from sewage treatment plants and storage of waters in engineered retention basins (McPhillips et al. 2019).

While there is higher likelihood for flooding and erosion during wet weather events, lower baseflow often occurs due to reduced groundwater contributions from reduced infiltration, leading to high inter-annual variability in discharge both over shorter periods of time and annually (Dunne and Leopold 1978, Arnold and Gibbons 1996). The hyporheic zone, a subsurface layer of porous substrate that involves a mixture of surface water and ground water, can be influenced by flood pulses and change in substrate as these factors change infiltration rates, and water is either processed through the hyporheic zone and into groundwater or released back into surface water (Krause et al. 2011). High river discharge and velocity produces an erosional environment and low flows create a strongly depositional environment in urbanized rivers (Arnold and Gibbons 1996, Booth et al. 2002). The increased magnitude of events and increased frequency in the changes within this erosional/depositional paradigm in urban rivers changes river morphology, resulting in wider and shallower channels with consequential impacts on water temperature and production (Chin 2006, Colosimo and Wilcock 2007). To combat increased flooding and erosional issues within urbanized areas, engineering practices have been used such as channelization, dams, armouring of the riverbanks (e.g. gabions) and the riverbed itself (concrete riverbed) (Figure 2) (Stein et al. 2013). These engineering ‘solutions’ transform meandering, braided, and branching rivers into straighter, wider, deeper, and shorter channels with greater elevational gradient and water flow velocity and capacity (Arnold et al. 1982, Brooker 1985). As the meandering, braiding, and branching nature of rivers is altered, the natural migration of channels caused by erosion/sedimentation dynamics is reduced and river channels become more fixed in their shape. As discussed in Ward and Stanford (1995b) through their
extension of the serial discontinuity concept to floodplains, reducing these natural meandering
dynamics through flow regulation and channelization leads to less connectivity between rivers
and their original floodplains. While these controls may aid the flood response, they do not
contribute to maintaining, let alone restoring ecosystem functions to the pre-urbanized condition
(Booth et al. 2002). These factors result in greater volumes of water moving downstream at
much more rapid rates than pre-channelized conditions. Consequently, the ‘localized’ effects of
urbanization are shunted downstream at an even faster rate in channelized rivers, exacerbating
disturbances (e.g. increased frequency of flooding) in lower reaches (Brooker 1985). The
shunting of water downstream ‘loosens’ nutrient spirals, reducing nutrient cycling capacity
within the system (Figure 1b).

Altered river hydrology leads to increases in unprocessed materials (e.g. detritus) entering
receiving water bodies (Kaushal and Belt 2012, Raymond et al. 2016). Increased overland flow
and erosion of the riverbank and riverbed increase soil-leached nutrients, particle-bound
nutrients, and various pollutants entering the receiving waters (Walsh et al. 2005b).

Unpredictable discharge events move dissolved and particulate nutrients into the receiving
waters, potentially at various times throughout the year decoupled from seasonal cycles of river
discharge and production in the receiving waters (Johengen et al. 2008, Larson et al. 2013).

Combined with loss of riparian vegetation (discussed below), these materials are not biologically
captured and processed to the degree as they would in a natural river. Yet, under low flow
conditions, the increased levels of nutrients and DOC may lead to periods of elevated utilization
of dissolved material by algae and other microorganisms for growth, resulting in temporary
accumulation of biomass in the river with its subsequent dispersal downstream under higher
discharge. Dissolved nutrients (nitrogen and phosphorus), particle-bound nutrients, and
unprocessed CPOM enter receiving water bodies more frequently and at higher concentrations (Larson et al. 2013). While influx of riparian vegetation is commonly reduced, increased inputs entering the system can originate further away from the stream due to urban drain systems and stormwater outlets (artificially creating an enhanced riparian zone effect), potentially leading to increased inputs of variable quality (Khamis et al. 2018). Not only is an increased amount of material transferred, but the diversity of the constituents may increase because of novel types of nutrient forms and organic material leached from roads, gardens, hardened surface and cross connections between sanitary and storm sewer systems, all being delivered by storm water systems from further distances in watersheds (Walsh et al. 2005a, Burns et al. 2016).

With these changes to receiving water body inputs, we expect to see potentially diverse outcomes in the lake. Beyond the broader extent of the mixing zone, the increased loading and deposition of materials near the river mouth, may lead to an expansion of the benthic enrichment zone as more readily observed in embayments of lakes receiving urban river discharge where assemblages of invertebrate tolerant of organic pollution proliferate (Brinkhurst 1970, Breneman et al. 2000). With an expansion of the benthic enrichment zone, the river mouth is expected to see increased abundance and productivity of benthic invertebrates in the longshore mixing area (Larson et al. 2013). Alteration in the seasonal pattern of discharge from natural rivers results in atypical period of elevated nutrient loading during the summer, potentially stimulating phytoplankton and benthic algal production at a time of heightened nutrient stress. Conversely, large sediment plumes during the growing season may suppress benthic and water column productivity through light-limitation and in the case of benthic biota by smothering of the lakebed with silt (Howell and Dove 2017). In concert, increased loading and periods of high turbidity events during the growing season likely result in overall heightened but more erratic
patterns of productivity with boom-and-bust cycles, further disrupting natural cycles and trophic responses. Less certain yet arguably as quantitatively relevant is the significance of the flux of particulate material laden with organic substrate and nutrients during the typical high discharge seasons of fall to spring which due to light and temperature regimes are less biologically reactive in the receiving waters than during the growing season. Urban rivers are anticipated to load larger quantities of sedimenting suspended solids spaced more broadly over this period than natural rivers (MacKenzie et al. 2022). In the absence of inter-season nearshore retention, lake focusing is expected to distribute this material to the offshore lake irrespective of river type. However, suspended material retained on the nearshore lakebed will accumulate to a higher degree and be more processed at the time of start of the growing season in the case of the urban rivers.

3. How erosion management can be used to re-naturalize river hydrology

3A. Riparian vegetation – slowing influx of urban runoff

Erosion of riverbanks can impact infrastructure like roads, buildings, and trails and can pose a significant human risk (Bernier et al. 2021). Erosion-mitigation techniques employed in urban settings include the armoring of riverbanks with concrete channels, gabions, or large concrete blocks to divert and dissipate the force of water (Chin and Gregory 2009). While civil engineering approaches dominated in the last century, bioengineering solutions have been used as far back as ancient civilizations (Rauch et al. 2022). Bioengineering approaches provide a more natural solution to erosion control, as vegetation can be used to stabilize banks and reinforce soil (Moreau et al. 2022), providing the potential for vegetation to grow, spread spatially, and regenerate over time, thereby reducing overall mitigation costs. Increased amounts and diversity of native riverbank vegetation will directly increase biodiversity, and indirectly
increase biodiversity by providing habitat for other riparian species (Zhang et al. 2022). Through establishing more vegetation in riverbanks and riparian zones, local stabilization is possible, however catchment-scale effects are also important to consider. Watershed catchment areas, and smaller upstream and downstream segments, should be considered if local projects are to be successful (Rey et al. 2019). For example, planting more vegetation catchment wide would have a positive effect, but the small positive effect of specific segment of riparian zones may be overwhelmed by catchment-wide urban impacts (Rey et al. 2019).

3B. Wetlands and shoreline strengthening

Other than vegetation, approaches to dissipate energy and reduce erosion can include introduction of riprap areas, particularly where stormwater discharge enters rivers with high velocity (Hughes et al. 2014). Riprap areas and boulders can reduce the pulse-shunt of water through the system and have numerous positive benefits for benthic and hyporheic zones as water will better infiltrate (Lawrence et al. 2013). The development of artificial wetlands (freshwater) or marshes (saltwater) at the river mouth where possible (e.g., the Toronto Portlands), will allow for energy reduction as river water enters the receiving waters (Figure 3). This would duplicate the historic role of river-mouth wetlands in river ecosystems. While erosion is a natural process, the desire to re-naturalize urban rivers and receiving waters shorelines must be considered within the context of a cityscape, wherein bioengineering solutions are considered with risk to infrastructure and people. This makes some projects more feasible than others (e.g., stormwater ponds may not be possible in an already heavily urbanized area versus increased riparian zones catchment wide).
4. Watershed zones: riparian and flood plain, hyporheic, wetlands, receiving waters

4A. Natural riparian and flood plain

Riparian zones are vegetated areas of terrestrial land tracing the length of rivers and streams where the strongest interaction between aquatic and terrestrial components occurs (Vannote et al. 1980). The riparian zone and floodplain can provide many essential hydrogeomorphic and biological functions that help to structure lotic ecosystems along the continuum from headwaters to receiving waters.

The floodplain area in the headwaters is usually quite small or absent, due to the incised V-shaped river channel (Wetzel 2001). Riparian vegetation along these higher elevational reaches is integral for stabilization of the stream bank where hillslope erosional processes and landslips are prevalent (Steiger et al. 2005). The root network of vegetation holds the bank substrate in place, while above-ground biomass slows the overbank flow of water and entrained particles (Gregory et al. 1991). Vegetated riparian zones contribute to stabilizing headwater streams by moderating summer low flows and ground-water recharge and increasing water-storage capacity of the soils (Buttle 2011). Overhanging riparian vegetation also reduces relative evapotranspiration through shading headwaters, creating cooler microclimates, and regulating solar energy inputs (Trimmel et al. 2018). Combined with the morphology of the stream, large water-holding capacity of the soil, and dominance of ground-water supply, the riparian zone reduces variation in discharge in headwater streams and results in a slower hydrological response with peak flows (Richardson and Danehy 2007, Buttle 2011).

The riparian zone in mid-reaches of the river encompasses a wider area than in the headwaters. There is often a defined floodplain where the meandering reaches slowly migrate laterally through erosional and depositional processes (Ward and Stanford 1995a). Groundwater in mid-
reaches contributes less to stream flow compared to surface flow, and combined with increased
catchment area, this results in greater variation in discharge and temperature. With increasing
catchment area there is greater potential for storms to cause high flow events leading to periodic
inundation of the floodplain (Junk et al. 1989). This lateral buffer zone dissipates the longitudinal
flow of energy, increasing the connectivity of the river with the adjacent terrestrial area (Ward
and Stanford 1995a). Peak flows are higher in mid-reaches than in headwaters and there is a
perceptible rising and falling limb of the hydrograph, although the hydrologic response time can
be long (i.e., broad rising and falling limbs).

The riparian zone is widest in lower reaches as the stream gradient decreases from upstream to
downstream, floodplains become larger, and catchment size and discharge volume increase
(Hamilton 2009). Natural floodplains in lower reaches are densely vegetated with wetland plant
species reducing the current velocity and promoting sediment accumulation during wet periods.
Continued reduction in the relative contribution of groundwater in lower reaches results in
increased potential for greater seasonal variation in flow and water temperature (Vannote et al.
1980). The hydrologic response time continues to slow given increased catchment size because it
takes longer for water from headwaters to reach the mouth following storm events, or with very
large river systems there is an integration and smoothing of effects from different storm events
across the catchment. Thus, peak flows are lower than in mid-reaches and the hydrograph
resembles a broadly shaped bell curve.

4B. Natural hyporheic zone

The hyporheic zone is less commonly acknowledged in riverine dynamics compared to riparian
zones. It plays a key role in each stream segment, as it is an intermediary among surface water,
groundwater, and the floodplain, and exchanges of water and materials from these three zones
occurs within the hyporheic zone. It extends into the stream banks, and its characteristics are determined by sediments and water flow (Boulton et al. 2010). This zone, sometimes extending laterally considerable distances, is important for biogeochemical processing, temperature regulation, microbial activity, and as a refuge for benthic organisms (Krause et al. 2011). The hyporheic zone supports its own community of organisms, referred to as hyporheos (Krause et al. 2011) and it is important for benthic organisms as well as fish which spawn in redds (Boulton et al. 2010). Infiltration into the hyporheic zone slows energy transfer longitudinally and transfers it more vertically into groundwater (Krause et al. 2011).

4C. Natural wetlands and receiving waters

Wetlands are important in both rivers and receiving waters. Regardless of where wetlands are located, wetland plants provide resistance to flow and reduce current velocity, reduce bottom shear stress and bed erosion, contribute to wave attenuation, and provide an area for floodwater energy to dissipate (Acreman and Holden 2013). Wetlands regulate peak flow and baseflow in many regions (Ameli and Creed 2019) and have been compared to sponges as they retain water in flood events and slow its transfer downstream. In a river system, wetlands can be along various reaches, with some wetlands being dependent on rainwater, groundwater, and river overflow as part of the floodplain system (Albert et al. 2005). At the receiving waters, riverine wetlands form as ‘delta’ wetlands or ‘drowned river mouth’ wetlands (Uzarski 2009). In these systems, river water and lake water mix, creating what some refer to as freshwater estuaries (Herdendorf 1990). ‘Delta’ wetlands form where large deposits of fine sediment occur (Uzarski 2009). Along with sediment retention, wetlands retain POM, which is stored and used by organisms (Yarwood 2018). Wetland systems provide protection to rivers as wetlands reduce upstream erosion through limiting extensive backflowing of lake water into rivers, thereby
providing a buffer against wave action during storm events (Batzer and Baldwin 2012). Although many wetlands are found at the interface with the receiving waters and along the nearshore, wetlands can be located anywhere along a river course, and are sustained by periodic flooding of rivers, groundwater inputs, and surface runoff (Wohl et al. 2021).

4D. How urbanization changes riparian and flood plain zones

Urban rivers have reduced vegetation within their riparian zone, which in turn negatively impacts a wide variety of stream ecosystem functions (Kingsford 2015). Reduced riparian vegetation causes urban rivers to become less efficient at storing water essential for maintaining baseflow during dry periods while also becoming more susceptible to high flow events. The rate of evapotranspiration, particularly the evaporation component, will be greater in urban rivers relative to more natural rivers as the system will have a reduced infiltration rate due to the increase proportion of impervious surfaces present in urban landscapes (Li et al. 2021). The structural integrity of the riparian zone is reduced in the absence of riparian vegetation resulting in higher rates of sedimentation in aquatic environments. Additionally, loss of riparian vegetation results in the broader loss of aquatic and terrestrial biodiversity, critical nutrient processing, and uptake of harmful compounds by riparian vegetation essential for healthy riverine biota (Groffman et al. 2003, 2005). Riparian vegetation is a vital energy source for stream biota through detrital inputs (e.g. leaves, tree branches) and supplies suitable habitat for terrestrial invertebrates consumed by fishes, including economically valuable salmonids (Baxter et al. 2005). Differences in the type of allochthonous resources between streams from forested catchments and urbanized streams (i.e., CPOM vs. FPOM) will lead to distinct changes in stream biota composition. Without riparian vegetation, the amount of large woody debris within rivers decreases leading to a loss of suitable habitat for many invertebrates and fishes. Loss of this
debris within urban rivers reduces retention of additional allochthonous materials within the ecosystem (Lassettre and Kondolf 2012). While riparian zone loss decreases allochthonous inputs to the river, primary producers thrive due to the increased temperature and light availability from canopy cover loss (Roberts and Bilby 2009). This transition from allochthonous to autochthonous energy sources can result in a shift in the local biota, particularly in headwater reaches that are often heavily reliant on resources from adjacent ecosystems (Vannote et al. 1980). An alternative source of allochthonous materials in urban rivers can be stormwater catchments that can help offset losses of resources originating from the riparian zone, however, the type and quantity of material will affect the composition of riverine biota. While the degree of riparian zone loss may vary from river to river, riparian vegetation loss in urban rivers generally alters flow regimes, which impacts river functioning.

4E. How urbanization changes hyporheic zones

Hyporheic zones in urban regions play an important role in biogeochemical processing and transport of nutrients, and pollution, into groundwater (Lawrence et al. 2013). Sedimentation and fine particles can essentially clog the hyporheic zone, as diverse sediments are needed for a healthy functioning zone to allow for proper flow (Krause et al. 2011). Such blockages impact habitat availability and processing rates (Krause et al. 2011). Additionally, the hyporheic zone aids in temperature regulation, thus blockages impede this essential function of the zone (Krause et al. 2011). As DOC dynamics change in urban hyporheic zones, microbial energy pathways may also change, particularly in the hyporheic zone where the light can be limited. Loading of nutrients, chloride, and other contaminants to groundwater from leaky infrastructure and infiltration (Kaushal and Belt 2012, Kaushal et al. 2014), and subsequent recharge to rivers may also serve to spread the flux into the river more broadly over time than related to the above-
ground loading episodes, such as the winter application of de-icing salts to roads (Eyles et al. 2013). Additionally, hyporheic zones and groundwater recharge zones may be reduced in spatial extent as increased impervious cover reduces water infiltration rates (Shuster et al. 2005). Hyporheic zones are understudied, but potentially crucial zones for regulation of urban streams.

4F. How urbanization changes wetlands and receiving waters

Wetland habitats, if present at the river mouth, have been and continue to be lost early during the urbanization process, as urban centers are commonly located at the lower reaches of natural rivers (Faulkner 2004). Historically, many of these wetlands were infilled to create more usable space or deepened through dredging and turned into vertical shorelines (i.e., concrete walls) for harbours and to reduce erosion. Those wetlands not directly altered by these activities face increased variation, magnitude, and frequency of flow events and current velocities, and they may be covered with larger sediment and see increased mixing rates. Existing wetlands may see variable conditions, with some potentially drying out or changing their submergent to emergent vegetation ratio as water levels change. Receiving-water photic zones will become reduced due to increased sediment loading. This change in light availability can cause declines in biological diversity and production in receiving waters and wetlands as benthic productivity declines and light availability for submerged vegetation is reduced (Jia et al. 2020). Increased flow events throughout the year ultimately lead to unpredictable patterns for organisms (rather than the more predictable seasonal flow patterns), potentially impacting reproduction and early life stages of organisms, and leading to selection for more stress tolerant species (Walsh et al. 2005b). The ecological character of the river to lake and interface changes when wetlands are replaced with hardened shoreline and channelized river mouths as often the case in urban areas. The loss of wetland habitat results in reduced biological diversity in the lower river and less movement of
organism between the river and the adjacent nearshore of the lake and consequently lower biodiversity in the nearshore of the lake as well. Changes in timing and rates of loading of nutrients, organic material and sediments loading are expected to select more strongly for species tolerant of disturbance, higher nutrient availability, and organic enrichment in the absence of the moderating effects on material flux of wetlands. Periodic development of anoxia in organic-rich lower river channels in delta areas historically occupied by wetlands, but which are now used for contemporary shipping and transport, is expected to exert strong selection pressure on a diversity of biota in the lower water column. Further, nearshore regions of receiving waters may experience increased frequency and magnitudes of algal blooms and cyanobacteria, potentially impacting water quality (e.g., taste and odor, biotoxins like microcystin) and ecosystem services (reference).

5. How restoration strategies can improve stormwater management

Stormwater control measures are used to divert and reduce the rate of stormwater discharge and energy into a receiving waterbody, typically a river. Stormwater control measures can include infrastructure like stormwater management ponds (both permanent or wet ponds and temporary or dry stormwater management ponds) which hold water temporarily for gradual release into nearby waterbodies (Vogel and Moore 2016). Other approaches to stormwater control include installation of permeable pavement, swales, and rain gardens (Figure 4) (Shannon 2013).

Stormwater ponds are surface-level, open-air ponds which can be designed to provide a variety of benefits to both humans (e.g., greenspace) and wildlife (new aquatic ecosystems, bird breeding habitats), beyond their designed engineering solution. Permeable pavement is made of specific materials which allow for some degree of water infiltration (Imran et al. 2013). Swales are vegetated ditches which promote infiltration rather than runoff to stormwater drains (Revitt et
Rain gardens are patches of vegetation in depressions on the landscape (Dietz and Clausen 2005) (Figure 3). These engineering features act to mimic the effect of soil and groundwater uptake, as water is diverted first to another destination before reaching a river, thus dampening flood effects, increasing groundwater contributions, and in some designs allowing for complete control over release into local waterbodies. In many areas, water released from stormwater systems is slowed through placement of riprap and large rocks (Hughes et al. 2014). Such rocks allow for more turbulent flow patterns, that reduce the velocity and increase the aeration of discharged waters (Hughes et al. 2014) prior to entering urban waterways.

5A. Stormwater ponds

Stormwater ponds reduce stormwater discharge into rivers through the retention of runoff during high flow events. While retaining stormwater, they also accumulate sediments, nutrients, and contaminants (Flanagan et al. 2021). As a result, ponds fill up with sediment, reducing their available storage volume, and require periodic maintenance to maintain their function (e.g. dredging) (Erickson et al. 2018). Ponds can be designed with specific riparian and emergent vegetation known for uptake of nutrients, and vegetative islands can provide both nutrient uptake capacity, flood protection, and habitat for organisms (Jefferson et al. 2017). However, unless this vegetation is removed periodically, these stored nutrients will be released back into the pond waters. Stormwater ponds can act as novel aquatic habitat for fishes (potentially even supporting urban fisheries; Adams et al. 1984) and aquatic biota, steppingstones for migratory birds, and metacommunities for amphibians. The recreational value of stormwater ponds can be developed through development of pathways or trails, informational signs, and easy access from local neighborhoods (Prudencio and Null 2018), providing diverse ecosystem services. These ponds can be directly connected to receiving rivers or can be designed for gradual infiltration into the

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ground, becoming “dry” ponds that may be used for playing fields or other temporary uses. There are diverse designs for stormwater ponds, including having permanent ponds that have water discharged from the surface or from deeper in the pond to reduce thermal loading (Van Seters et al. 2019). Although these ponds may be relatively shallow (1-4m depth), they may have highly elevated chloride levels due to road salt runoff (Semadeni-Davies 2006) and be stratified due to their thermal or salinity gradients which can then lead to anoxia in deeper waters (Song et al. 2013), releasing phosphorus from sediments (Stajkowski et al. 2023). Ponds may represent sources of multiple stressors into streams through their discharge of warmer waters that have elevated chloride and low oxygen. As a series of these ponds may discharge into a relatively short length (e.g. hundreds of meters) of small tributaries, there is the potential for cumulative effects of these multiple stressors, yet these issues and their impacts are effectively unexplored. While stormwater ponds increase ecosystem services, they also increase the concept of serial discontinuity as lentic ponds are connected to free-flowing lotic rivers (Ward and Stanford 1983).

5B. Rain gardens and swales

Rain gardens are small patches of shrubs, grasses, and herbaceous vegetation used to slow the flow of surface water towards waterbodies and divert runoff to ground water through infiltration (Figure 3). Unique from larger engineered systems, rain gardens can be easily incorporated into residential areas (e.g., front lawn, back lawn, parkland), maintained by residents and managers alike. Rain gardens can be effective at retaining water, allowing slow infiltration into the soil or slowing its release (Shuster et al. 2017). Rain gardens can effectively remove physical contaminants such as litter and debris which are known to be associated with various types of vegetation (Schreyers et al. 2021). Swales are depressions in land, functionally ditches,
facilitating water infiltration rather than water flowing into storm sewer systems (Revitt et al. 2017). They can be placed between sidewalks and roadways, as well as in-between road lanes, they can also support biodiversity and can be considered small green spaces. While effective at increasing water infiltration, accumulated physical debris must be removed from rain gardens and swales by residents or entities such as municipalities as debris may eventually blow (or flow) away or build up to unsightly levels.

5C. Permeable pavement

Permeable pavement is a type of porous pavement which allows infiltration, leading to less stormwater runoff and reducing the temperatures of surface runoff (Figure 3) (Imran et al. 2013). While more costly than traditional pavement, permeable pavement creates cooler surfaces as there is more air exchange with soil which has a cooling effect compared to traditional impermeable pavement. Permeable pavement is a suitable alternative to traditional pavement for parking lots, sidewalks, driveways, however, are not currently as reliable for use in highways or busy roads as it degrades under higher forces such as frequent high-speed braking.

5D. Wetland reintroduction

Reintroduction and restoration of wetlands is increasingly considered as a stormwater control measure (Vietz et al. 2016). This strategy has effectively been used in the Niagara River, where the artificial establishment of historic wetlands aided in freshwater mussel conservation (Benshoff and Filipski 2021). Vegetation mats have also been successfully introduced into the canal of the Chicago River to create a floating wetland system (Peterson et al. 2021).

Constructed wetlands, whether floating or otherwise, can have strong influences on stormwater, a high capacity for nutrient retention, and provide important ecological habitat (Kennedy and
Overall, the main goal of storm water control mechanisms is to encourage reduction in velocity of water, allow for more infiltration, and reduce impacts such as high temperatures and pollution loads.

6. Biological responses to river dynamics

6A. Natural river biodiversity: from headwaters to receiving waters

Organisms respond to environmental filters on physiological and ecological traits, resulting in different taxa persisting under different sets of environmental conditions (Smith and Powell 1971, Townsend and Hildrew 1994). Variation in energy inputs, riparian conditions, and production scale up to differences in ecological communities within stream reaches.

In headwaters, densely vegetated riparian zones create a unique microclimate, with low light levels and reduced daily fluctuations in temperature, humidity, and wind (Richardson and Danehy 2007). Empirical observations of invertebrate communities in headwaters of quasi-natural streams match the predictions laid out by the River Continuum Concept. Generally, forested areas and riparian vegetation result in substantial inputs of CPOM to the river. Regions of low flow around large stationary objects (e.g., fallen branches, boulders) are prime areas for colonization by benthic organisms, either on the objects within the viscous laminar sublayer, between the particles in the low-flow quiet zone, or within the interstitial space of the substrate.

Thus, greater substrate heterogeneity results in greater habitat and food availability to benthic organisms (Giller and Malmqvist 1998). Inputs from riparian zones and forested areas provide material for shredders and collectors to consume (Vannote et al. 1980). Aquatic insects, rather than molluscs and crustaceans, are expected to be dominant in the upper reaches (Vannote et al. 1980), as these feeding groups convert CPOM into FPOM. Headwaters support rich small-bodied invertivore communities (Vannote et al. 1980) which capitalize on benthic and riparian
organisms. Fish communities from groundwater-dominated headwaters tend to contain species preferring cold and cool water, whereas stream reaches primarily relying on surface water runoff contain fish species with warmer thermal preferences (Chu et al. 2008). The patchy and more isolated nature of headwater sites relative to the entire stream network leads to more different communities between headwater streams (beta diversity) within a watershed (Edge et al. 2017), and this beta diversity decreases in downstream river segments with greater connectivity (Brown and Swan 2010, Finn et al. 2011). Seasonal fish movement may result in changes in fish community composition or variation in composition of life stages of fish species due to adult migration and spawning, and residence of young within river systems due to potential reduced predation on sensitive life stages further upriver (Nunn et al. 2010).

In mid-reaches, decreased levels of leaf litter and increased autochthonous primary production support communities of benthic invertebrates co-dominated by collectors and grazers. Rooted vascular plants increase in their relative contribution to primary production in mid-reach segments than upper (Vannote et al. 1980). Piscivores and invertivores are co-dominant in these streams (RCC), due in part to the shift in invertebrate community (Vannote et al. 1980). Mid-reaches are considered to support higher levels of species richness of different taxonomic groups than headwaters as habitat patches are more connected and both warmwater and coolwater species can take advantage of more variable thermal regimes (Minshall et al. 1985). The riparian zone and floodplain are exposed to periodic hydrogeomorphic disturbances that tend to ‘reset’ communities in a predictable and regular manner maintaining higher levels of biodiversity (Steiger et al. 2005). Seasonal growth of periphytic algae is expected to increase as light reaching the riverbed increases (Sofi et al. 2023), particularly in areas where substrate is more stable,
resulting in a diversified food source and acting to modulate levels of dissolved nutrient passing through the river during the growing season.

In lower reaches, invertebrate communities are expected to be dominated by collectors and predators, with collectors dominating due to higher levels of detritus and bacteria in the lower reaches (Vannote et al. 1980). Fish species richness is highest in the lower reaches, however, beta diversity is lower within lower reaches as habitat patches are more connected relative to headwaters (Brown and Swan 2010). Warm water species are more frequently found in lower reaches and piscivores are more common components of the fish community compared to headwaters (Vannote et al. 1980). Species richness differs seasonally due to migrating species and transient lacustrine species entering from receiving waterbodies, in addition to the resident species. In many natural systems, riparian wetlands are commonly found near receiving waterbodies and support high levels of biodiversity, particularly of invertebrates (Patch and Busch 1984). The tendency towards broader and shallower river dimensions further increases areal extent of habitat for periphytic algae and with the increased presence of macrophytes providing opportunity for epiphytic assemblages of algae suited for optimally accessing incident light.

The receiving waters are a key, but often overlooked, component of the river network, and vice versa, as its habitat conditions differ considerably from that within the river. The transitional area at the river mouth creates high habitat diversity as lake- and river-based processes interact to produce substantial heterogeneity in substrate size, water flow, and refugia. This heterogeneity in the physical environment can support high levels of biological diversity within river-mouth ecosystems (Richardson et al. 2021). River inflows provide a time-varying influx of nutrients and resources, such as organic matter and drifting invertebrates (Wipfli and Gregovich 2002),
into the receiving waters on which many species capitalize. The intersection between rivers and lakes also acts as a thermal refuge for species throughout multiple seasons. For example, stream inflows may be cooler in the summer relative to shallow areas of the lake, which can be beneficial for cool- and cold-water species such as brook trout (Curry et al. 1997), or warmer during spring (e.g., Chomicki et al. 2016). Additionally, tributaries, including the wetlands, may serve as spawning and nursery habitats (Jude and Pappas 1992), and provide juvenile fish access to additional resources, and a refuge from large predators residing in receiving waters. Accessible tributaries and spawning habitat are vital components for migratory species, some of which generally reside in the receiving waters for most of their lives. Spawning runs of salmonids are known to transfer large amounts of nutrients from receiving waters to headwater systems through bodies of spent adult salmon and nutrient-rich eggs (Naiman et al. 2002, Jonsson and Jonsson 2003), and similar contributions may occur through large-scale spawning runs of catostomids and other species from lakes to rivers. The strong connection between rivers and the receiving waters can have a profound influence on the biological diversity of both habitat types. As many lake species capitalize on river-lake mouth dynamics at some point in their life cycle, river-lake mouth interfaces tend to be of higher diversity than the broader lake community, in part due to the transient nature of some lacustrine species into rivers.

Urbanization impacts biological diversity from headwaters to the river-receiving water interface. Changes in hydrological regime, habitat availability, and introduced non-native species have profound impacts throughout river systems (Walsh et al. 2005b). These changes can lead to both local and regional controls on community diversity as both local abiotic conditions and regional dispersal capability through river networks are impacted (Brown and Swan 2010). We focus next
on both local and regional conditions through examining urban pollution (local control) and species dispersal barriers (regional control).

6B. How urbanization changes river biodiversity

6BI. How urban pollutants change biodiversity

Urban rivers are subject to pollution from multiple point and non-point sources. Urban pollution represents one of the leading causes of water quality declines in rivers (EPA 1988). Some forms of urban pollution include nutrients, chemical contaminants, thermal, and anthropogenic debris (e.g., plastic) (Kaushal and Belt 2012, Hobbie et al. 2017, Windsor et al. 2019a). Pollutants can function as selective physiological filters on biological communities, leading to biological communities potentially diverging from those described in the river continuum concept and more natural river systems.

Urbanized rivers experience nutrient enrichment from multiple sources including stormwater combined sewer overflows (Schliemann et al. 2021), wastewater treatment plants (Gibson and Meyer 2007), and fertilizer runoff (e.g., parks, gold course, private lawns; Hobbie et al. 2017). As urbanized rivers often have a history of agricultural practices within the watershed (Wolman 1967), they may have legacy effects of nutrient enrichment. As with other systems experiencing nutrient enrichment, harmful algal blooms (e.g. cyanobacteria with microcystin and other biotoxins) and hypoxia can occur (Schindler and Hecky 2009). Nutrient loading impacts trophic levels differently, whereby primary consumers such as zooplankton may benefit from an algal bloom while secondary consumers such as fish may decline due to reduced oxygen levels (Watson et al. 2016). Increased nutrients impact receiving-water dynamics by expanding the zone of biological productivity beyond the mixing zone of the river mouth, further exacerbating nearshore and basin wide potential for eutrophication (Flo et al. 2011, Sorokovikova et al. 2012,
Howell and Benoit 2021). Differential impacts amongst trophic levels resulting from increased primary productivity in receiving waters typically result in loss of species diversity, reduced efficiency of energy transfer across trophic levels, and increased water quality concerns due to proliferation of organisms adapted to eutrophic conditions (Smith and Schindler 2009). Contaminated river plumes entering receiving waters, both oceanic and freshwater, are concerning for recreational swimming and surfing, with surf zones sometimes found to have fecal counts far above government standards near urban river outlets after storms (Ahn et al. 2005).

Chemical contaminants such as heavy metals (Beasley and Kneale 2002), pesticides (Meftaul et al. 2020), persistent organic contaminants (Windsor et al. 2019b), pharmaceuticals (Huerta et al. 2018), and major ions (e.g., chloride), are found at elevated levels in urban rivers (Kaushal et al. 2020). Metals and persistent organic pollutants, associate with organic materials and may be resuspended from the sediment under turbulent conditions (De Miguel et al. 2005). In a system prone to increased flash flooding, impacts of particle-associated contaminants may be exacerbated. Additionally, reports have shown urban areas having increasingly high salt signatures, as north temperate regions commonly use road de-icing salts during the winter season (Kaushal et al. 2005, Dugan et al. 2017). Secondary salinization resulting from de-icing is now a year-round concern, as summertime concentrations surpass federal guidelines for protection of aquatic life (Lawson and Jackson 2021). Salt concentrations impact density of water as more saline, denser river waters flowing into receiving waters may change the dynamics of river plumes in receiving waters. Thus, salt is not only a chemical contaminant harmful to organisms, but also has the capacity to change physical patterns in water stratification and circulation (Findlay and Kelly 2011, Hintz and Relyea 2019).
Thermal pollution signatures can be found in urban waters as vegetation decreases and urban heat island effects increase, thus increasing temperature of stormwater runoff (Herb et al. 2008). Impacts to river temperature are further compounded by reduction in groundwater upwelling due to impervious surfaces which reduce infiltration and recharging of groundwater. Temperature is a fundamental component limiting aquatic organisms physiologically and biogeographically (Jackson and Harvey 1989, Jackson et al. 2001), and temperature tolerances can be surpassed in urban regions. This reduces habitat availability for some riverine species, as warmer water selects for warm-water species and reduces the ability of cool- and cold-water communities to persist. However, earlier introduction of warmer temperatures in the spring may ‘jump start’ biological activity, expanding growing seasons, which can have negative or positive outcomes. The growth of key species may be positive, but growth of algae can produce negative outcomes for rivers and receiving water nearshore. Effluent associated with structures such as wastewater treatment or electrical generating plants can support novel biological communities near these sites, including invasive species as these areas may have both elevated water temperatures (Castañeda et al. 2018) but also less seasonal variation in temperature, thereby facilitating species having different temperature tolerances than native communities in the area. Receiving waters may see thermal loading potentially causing thermal plumes as heat energy mixes with receiving waters, similar to those recognized from power stations (Ingleton and McMinn 2012). Conversely, releases of water from sewage treatment plants may have a cooling effect in rivers during summer months, moderating the seasonal variability in temperature within a localized area. In the short term relevant to episodic events, the depth at which the river plume intrudes into the lake will dictate how these contaminants enter the lake food web. Entrained pollutants (e.g., nutrients, hydrophilic contaminants) in the surface waters can enter the lake food web.
through phytoplankton uptake and zooplankton grazing (Pearce et al. 2021), whereas pollutants in dense sinking plumes can enter the food web through the benthic community. Once in the food web, some pollutants can bio-magnify through trophic transfer and disperse further into the lake due to organismal movement (Blais et al. 2007).

Anthropogenic debris, most commonly plastic but also include brick, stone, glass, lumber, and metal, are commonly found in urban environments and can alter the physical and chemical conditions of a river (Effland and Pouyat 1997). These materials create novel habitats, increase habitat heterogeneity, and supply substrate for organisms (McCormick et al. 2016), but may also alter the chemical environment. For example, plastics are associated with a suite of chemicals ranging from chemical additives during manufacturing to contaminants adsorbed from the environment (Rochman et al. 2019). These contaminants may alter individual organisms or entire communities (McCormick et al. 2016, Bucci et al. 2020). Other anthropogenic materials such as concrete can alter water chemistry and may have downstream impacts to biological communities (Purdy and Wright 2019).

Diverse suites of urban pollutants can be considered as ‘cocktails of contaminants’ (Kaushal et al. 2020). As urban waters flow into receiving waters, urban pollutants in particulate form will be most strongly deposited and buried in the sediment near the river mouth whereas dissolved pollutants can be transported along the perimeter of the lake through strong longshore currents that persist year-round. Pollutants can become concentrated in bays and in areas that promote the formation of eddies and then disperse offshore primarily through wind-induced upwelling and downwelling events (e.g., during summer stratification and thermal bar periods) (Jirka and Volker 2005, Rao and Schwab 2007). Increased deposition of organic particulate matter at the river mouth is expected to increase loadings of particulate-bound hydrophobic contaminants near
the river mouth while dissolved contaminants should be carried farther away from the river mouth. Elevated levels of major ions (e.g., Ca, Mg, Cl) is a characteristic feature of urban shoreline adjacent to rivers stemming from river loading. As conservative tracers, their shoreline distribution in the water column depicts a zone where direct exposure to substances collected and transferred by the river is likely occurring in the lake, illustrating the connectivity between the pollution impacts in the river and the receiving water (Howell et al. 2012). Urban pollutants can have far-reaching effects depending on factors such as the biological transformation potential of the pollutant, the pollutant residence time, the physiological mechanism of harm specific pollutants can cause at the organismal level, and the ultimate resulting ecological disruptions.

6BII. How urbanization increases barriers to species dispersal

Damming river sections to slow flow, the incorporation of culverts to allow flow through road networks, and the burying of streams underground has consequences for movement of nutrients and energy, and for the dispersal and life cycles of organisms (Figure 1b). The action of damming can be linked to the Serial Discontinuity Concept developed by Ward and Stanford (1983). The Serial Discontinuity Concept describes the process by which free-flowing rivers are turned into series of lentic and lotic regions, leading to disruption in flow and energy (Ellis and Jones 2013). Organisms such as riverine fish and mussels may become increasingly restricted in their ability to move in systems with such barriers. Dam design will selectively impede movement by restricting access to only fish able to jump over even very small dams or all species in the case of larger dams unless structures are incorporated to facilitate upstream movements (e.g. fish ladders) (Katopodis et al. 2001). Although some fish species can pass through certain barriers, there can be a significant additional energetic cost associated with the passage of the barrier (Noonan et al. 2012). Burying streams underground and moving them into
pipes poses a significant barrier to dispersal, as organisms may not be able to move through buried stream sections or may avoid such dark environments due to unknown risks, and resources may not be available due to lack of primary productivity in buried sections (Hintz et al. 2022). Dams, improper culvert design, and stream burial can lead to decreased species diversity within sites or change to include lentic species, and it may lead to increased beta diversity in metacommunities as sites become inaccessible and regional species pools are restricted through limited dispersal (i.e. sites become more dissimilar due to differential species losses) (Edge et al. 2017).

Barriers reduce the accessibility of spawning habitat for migratory species which can lead to profound impacts on the entire ecosystem through the absence of key resource subsidies for stream biota and limiting the available habitat for juvenile fish (Jones and Mackereth 2016). In the case of aerial dispersers, such as some aquatic insect species, river dams and culverts may not restrict their movements, but nonetheless may be affected due to changes in transport of CPOM/CFOM. Built infrastructure, however, may impede dispersal between river sections as buildings and gray infrastructure can directly block aerial dispersal attempts or provide confusing cues regarding water availability (e.g. light reflected from asphalt and buildings providing comparable cues to light reflected from water surfaces, Fraleigh et al. 2021). The ubiquity of these engineered elements in urban watersheds impacts the overall functioning of the river, from nutrient and energy cycling to large organisms like fish.

While barriers to dispersal are generally considered negative for native species, they can be important in deterring the spread of invasive non-native species (Pratt et al. 2009). The globalization of trade, travel, and recreation associated with urban regions increases the potential for non-native species to invade ecosystems (Padayachee et al. 2017). Invaders can hitchhike
into new systems through mechanisms like ballast water or recreational bait containers (Padayachee et al. 2017). Species can also be introduced through intentional release of unwanted specimens like fishes and crayfishes, e.g. goldfish in many areas (Maceda-Veiga et al. 2019; Figure 2). Without barriers to movement, invasives can then disperse throughout entire systems. There are many ecological mechanisms (e.g., enemy release, outcompeting natives, transferring novel pathogens) by which non-native species can establish once they arrive in an urban region. Invaders can, in turn, have profound impacts on systems, such as the filtering of water by zebra mussels leading to changes in water quality (Horgan and Mills 1997) and even changing physical habitat. Many invasions potentially lead to further decreases in native species due to competition or direct predation on native biota such as in the case of sea lamprey (Docker and Potter 2019). Invasive species themselves represent a form of contaminant with one major difference, they increase over time and space rather than degrade or decrease in number over time. Essentially, they are a self-propagating contaminant if no containment (e.g., barriers) or eradication plan is put into place.

6D. How remediation and restoration can improve ecological conditions

6DI. Pollution remediation

Several strategies have been employed to improve the pollution status of urban rivers. While policies like the Clean Water Act (United States of America) emphasize point source pollution, approaches to urban pollutant remediation must be multipronged to stop pollutants at the source, consider non-point source pollutants, address novel contaminants (e.g., plastic), and remediate existing pollutants. Physical pollution remediation is the process by which physical properties of the river are altered to improve ecosystem structure and functioning. Updated remediation strategies implementing water diversion and retention strategies are quickly becoming a common
remediation strategy in rivers. These strategies include gross pollutant traps, sedimentation ponds, shallow ponds, and stormwater ponds, all of which share a goal of slowing the flow of water to allow sediment to settle and to trap nutrients like phosphorus. This strategy is based on the principle that many pollutants are associated with smaller particles (e.g., suspended solids, sediment) in streams. By removing these small particles from the river ecosystems, pollution is reduced. Where thermal pollution is of concern, bottom-draw stormwater ponds may help reduce warm water from flowing directly into rivers, although these deeper waters may contain higher concentrations of certain pollutants (e.g., chloride, and sediment-based pollutants) and have lower concentrations of dissolved oxygen as described earlier. Thermal pollution can be reduced through increased use of riparian vegetation which provide shade for rivers (Anbumozhi et al. 2005, Bowler et al. 2012).

Physical remediation strategies include aeration and sediment dredging in rivers. Aeration improves water quality by increasing oxygen levels (Burden et al. 2008), which stimulates microorganismal growth and aids in restoring regular functioning (Lamping et al. 2005). While effective in some cases (Lamping et al. 2005), aeration has recently caused concern as it can potentially create microbial aerosols, particularly with untreated sewage in urban areas (Dueker and O’Mullan 2014). Sediment dredging is another practice used to remove contaminated sediment from a contaminated water body (Bridges et al. 2010). Sediment dredging has been proposed as a remediation strategy in some superfund sites within the USA, such as the Hudson River (Dueker and O’Mullan 2014). However, other studies suggest sediment dredging may not be the most effective means of remediation due to difficulties related to measuring efficacy, and remaining uncertainties around contaminated sediment transport and resuspension (Gustavson et al. 2008). Instead of dredging, efforts have moved towards capping the contaminated sediment,
introducing amendments (e.g., organic carbon), and building a new ecosystem on top of the
capped sediment (Gustavson et al. 2008). Toronto’s “Portland Flood Protection Project” uses this
approach, where new parkland and aquatic habitat is being created in a previously industrial area
with a variety of approaches incorporated to deal with contaminated soils (Donnelly 2022). To
reduce plastic pollution and other litter in aquatic ecosystems, trash capturing devices in a variety
of shapes and designs are now being implemented (Helinski et al. 2021). In addition to removing
debris, these devices have the potential to also remove other sorbed chemical contaminants from
the environments where they are deployed.

This description of pollution remediation strategies is not extensive; the variety of strategies
continues to grow and change as our understanding of pollutants in urban rivers shifts. The field
of pollution remediation is constantly evolving, and novel solutions can be used to remediate
existing pollution, while policy change can aid in reducing sources of pollution.

6DII. Removing barriers to species dispersal

Removing dams and daylighting streams are management strategies currently in use to remove
barriers to species dispersal and improve functioning of urban streams. The removal of dams and
barriers in watersheds can have a profound positive impact on the entire ecosystem from nutrient
processing to the spawning success of migratory species (Stanley and Doyle 2003). Many dam
removal projects result in the improvement of both abiotic and biotic conditions (Burroughs et al.
2010, Hansen and Hayes 2012). An unobstructed flow regime ensures nutrients and energy are
distributed more evenly throughout the watershed rather than being highly variable among
sections of the watershed as described by the Serial Discontinuity Concept (Ward and Stanford
1983). Unobstructed flows are more like natural systems as described by the River Continuum
Concept, the River Wave Concept, and the Pulse-Shunt Concept (Vannote et al. 1980, Kibler et
Removing dams can also reduce the scouring effect, commonly seen in urban rivers, and establish more natural riffle-pool sequences (Bednarek 2001, Kibler et al. 2011). In contrast, removing dams may increase immediate scouring during storm events as dams may help function in storing water and slowing current velocity (Chiu et al. 2013). For biota, the removal of barriers leads to increased species movement and habitat accessibility throughout the watershed which can lead to increased biotic diversity. Fish passages (e.g., fish ladders) and assisted movement of migratory species improve migration success rate, however, the ability to navigate to spawning habitat is increased further by removal of these barriers. However, removing barriers may also increase the susceptibility of rivers to upstream colonization of invasive/non-native species, such as sea lamprey and round goby in the Great Lakes (Milt et al. 2018). Dam removal may also release contaminated sediments downstream (Stanley and Doyle 2003).

The negative effects of culverts, a common barrier to fish movement, can be mitigated through proper design and implementation (Favaro et al. 2014). Culvert design must consider heights and diameters of culverts to maintain connectivity and appropriate water depth to aid in species movements. For example, counter-sunk and bottomless culverts do not impede fish passage whereas culverts with a high slope will act as a barrier to fish movement (Price et al. 2010). While there are many positive effects of barrier removal and mitigation in watersheds, these actions can also result in negative short-term effects on the ecosystem, such as increased sediment load (Poff and Hart 2002), which can require further mitigation.

Stream burial is now a less common practice due to increased recognition of the ecological importance of riverscapes and watersheds, and numerous failures of the underground systems to accommodate unusually high precipitation events (Kaushal et al. 2015). Historically, many
streams were buried underground in pipes to manage waste and contaminated waters in an “out of sight, out of mind” style approach. These underground streams shunt water from upstream into receiving waters and are devoid of most non-microbial productivity other than that which it receives from upstream, like the Pulse-Shunt concept of Raymond et al. (2016). To mitigate buried streams, “stream daylighting” projects seek to re-expose streams to daylight and establish more aquatic habitat and aquatic organism passage (Wild et al. 2011). Daylighting streams reduces the incidences of costly blockages which can occur in piped systems (Wild et al. 2011). Successful daylighting projects are noted since the late 1900s. For example, the daylighting of 8.4 km of the Cheonggyecheon river in Seoul, South Korea led to economic benefits to the area, and later became a tourist attraction (Khirfan et al. 2020). Evidence suggests daylighting can change ecological communities, with upstream communities being of the utmost importance to support establishment of new communities (Neale and Moffett 2016). For example, Neale and Moffett (2016) found over 40 new macroinvertebrate taxa shortly after the daylighting of a New Zealand stream. While daylighting is a promising avenue for management, it may be difficult to facilitate if buildings and infrastructure must be moved to accommodate the floodplain. Overall, daylighting streams can bring hydrological, ecological, and economic benefits to surrounding areas as more natural flows can be established, ecological barriers are reduced for dispersal, property values increase with access to nature/streams, and maintenance costs associated with stormwater infrastructure decrease (Wild et al. 2011).

These management strategies have been effectively used in urban rivers to improve river functioning. However, it is important to establish realistic goals on an individual basis, as some watersheds may return to approximate pre-barrier/burial ecological conditions whereas others may suffer from permanent changes to watershed conditions (Hansen and Hayes 2012).
Discussion

Understanding natural systems is fundamental to conceptualizing how changes to watersheds will affect rivers and receiving water dynamics. Our synthesis describes how river systems are governed by energy dynamics, how patterns within rivers impact their receiving waters, how urbanization alters rivers, and how different restoration strategies can work to mitigate these impacts. First, we extended our current understanding of natural river concepts by considering receiving waters and discussed how urbanization can change both rivers and their receiving waters. Next, we discussed both how management actions seek to restore natural riverscape functioning and limits of restoration. Throughout, we highlighted the impacts of urbanization in receiving waters, a crucial area generally overlooked in the literature considering flowing waters.

We emphasize the importance of considering natural river dynamics when re-developing urban streams, as management strategies can use natural systems as a guide while taking advantage of newer technologies. In natural rivers, soil and vegetation manage infiltration and dispersion of energy to maintain predictable patterns and support biodiversity and river functioning (NRC 2002). Used in tandem, natural solutions and new technologies and techniques need to become increasingly important as frequency of extreme hydrological events increases due to climate change (Kõiv-Vainik et al. 2022). For example, technological advances allowing for almost continuous monitoring physical and biological monitoring aquatic systems can be leveraged to increase data collection and facilitate more robust analyses (Allan et al. 2018, Cooke et al. 2022, Besson et al. 2022). Management strategies can be approached through different scales, and positive impacts can stretch beyond a single aspect of a river’s reach. For example, increased vegetation in the riparian zone can reduce floods and increase infiltration while also increasing habitat for biota throughout the entire watershed. Vegetation can also be planted in stormwater...
ponds, and greenspace surrounding ponds can be enhanced to support biodiversity and for recreational activities, thus increasing overall ecosystem services. Various restoration actions and their benefits are highlighted in Table 1, similar exercises can be used in the first steps of restoration planning, with context specific tradeoffs and cost-benefit maximization considered on both short and long-time scales. Urban environmental management is ultimately governed by policy and planning. Policies may be context-specific, as watersheds are unique in their geophysical characteristics and geopolitical histories which both influence management decisions. As more knowledge is gained through the ongoing era of restoration, adaptive management frameworks will increase in importance (Smith et al. 2016). Such management plans must be based on the best available data and consider both historical conditions and the goal of future conditions on specific time scales.

While restoration management often involves scientific techniques (e.g., engineering, ecology), it can also involve increasing public ecological awareness. Research shows enjoyment of nature is correlated with pro-environmental behavior (Liu et al. 2022). Therefore, management strategies should include planning and construction of trails, boat launches, parks, and educational or event spaces near urban rivers. Local environmental groups can then be encouraged to use natural spaces to gather and recruit new members. Examples of local environmental groups include Friends of the Rouge (Toronto), Friends of the Chicago River (Chicago), and the Hudson River Foundation (New York). These groups promote positive environmental action, a sense of community, and often have strong voices in city planning initiatives. Recreational spaces can be used to increase environmental education through the planning of accessible educational signage and welcoming spaces for individuals to explore their relationship with urban waters (Zingraff-Hamed et al. 2021). Similarly, public art displays, and
placement of management structures, such as trash capture devices, in highly visible areas could provide further opportunities for education, enhancing awareness, and even employment (Figure 3). Trash capture devices for example, can be used to inform policy, limit point sources, or create targeted campaigns because data is collected on the amount and types of litter collected within the trap (Helinski et al. 2021). When planning any of these strategies, careful attention must be given to reducing barriers to accessing nature through inclusion of designs cognizant of physical and socio-economic barriers. River management can be influenced by local social engagement. If more naturalized rivers are to receive greater public support compared to more degraded areas, communication with the public about the many benefits of more ecologically functional urban rivers is key to strengthen community support for river restoration (Guimarães et al. 2021).

By synthesizing previous river concepts and novel management techniques, we provide a watershed scale view of urban rivers from headwaters to receiving waters. Our work stresses additional consideration of urban receiving waters and the fate of urban river water as it reaches and disperses into larger receiving waters (typically where many large cities began and continue to exist), potentially altering the entirety of the terminal waterbody. With urbanization continuing to progress at unprecedented rates (McDonald et al. 2020), focus must shift to the large potential of urban rivers, their impacts on receiving water bodies, and how negative impacts can be managed as cities continue to rapidly expand.

**Conclusion**

1. Rivers are governed by energy dynamics which further determine ecological dynamics. Urbanization fundamentally changes energy and ecological dynamics of river ecosystems.

2. Rivers do not exist in isolation, and changes in rivers can alter receiving water dynamics.
3. River concepts can be extended to urban rivers, and further to urban restoration dynamics, to better understand how river dynamics from their headwaters to their receiving waters.

4. Understanding how river energy and ecological dynamics change with urbanization and restoration is important as restoration is occurring at the same time as further urbanization as the human population becomes more urban.

5. Extending river concepts to understand changes in river dynamics will aid in urban environmental planning and provide a framework for expectations under different conditions.

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Figure 1: Characteristics of natural (A), urbanized (B), and restored streams (C). In the natural stream, woody debris and wetland vegetation provide aquatic habitat, two tributaries flow into the main channel, there are multiple nutrient spirals (red), and there is sufficient groundwater and surface exchange through the hyporheic zone. In the urbanized stream, woody debris and wetlands are reduced, tributaries are buried underground, there is reduced nutrient spiraling, and pollutants enter the river as runoff (yellow) or through groundwater. Additionally, impervious surface is increased in the downstream reaches, and forest upstream is converted to farmland. In the restored stream, natural elements are re-introduced into the stream to improve functioning. For example, woody debris is added to the riverbed, tributaries are ‘daylit’, stormwater flows through a pond system before entering the main river channel, and wetland vegetation is introduced at the mouth. These changes improve nutrient spiraling and exchange between groundwater and surface water.
Figure 2: Distribution of nutrient spiraling and contaminant fate in A) a natural river-mouth and B) an urbanized river-mouth. In a natural river mouth, (A) there are tighter (more processing) nutrient spirals (red), and a more heterogeneous size distribution of substrate. Diverse invertebrates and algae are found in the heterogeneous substrate. In an urbanized river, pollutants (yellow) can create stratification in the water column, and the substrate is concrete channelized. These changes reduce the tightness of nutrient spirals, reducing processing along the river and lead to a biotically homogeneous riverbed that lacks habitat heterogeneity, and is contaminated with pollutants.
Figure 3: Examples of urban stream features: roadway culvert (A), trash pollution (B), channelization of river (C), and invasive goldfish in a stormwater management pond (D). Photo sources: Lauren Lawson (2019)
Figure 4: Panel of urban stormwater and pollution management techniques: permeable pavement/parking (Panel A, Source: Rachel Giles), restored urban wetland (Panel B, Source: Great Ecology [https://greatecology.com/portfolio-item/corktown-common-waterfront-park-ecolgical-design/]), trash capture device (Panel C, Source: University of Toronto Trash Team), and rain garden (Panel D, Toronto and Region Conservation Authority).
Table 1: Categories of services different restoration techniques provide. Green shading indicates whether the technique can be considered a hard or soft engineering approach, with hard engineering indicating artificial man-made solutions, and soft indicating more nature-based solutions involving more vegetation. *Costs are associated with investments of costs over time, not initial cost of project. For example, we recognize wetland creation is a high-cost project initially, however, long-term costs associated with maintenance may be relatively lower than approaches like trash capture devices which need to be serviced or management continuously through time.

<table>
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<tr>
<th>Technique</th>
<th>Water Storage</th>
<th>Habitat</th>
<th>Recreation</th>
<th>Education</th>
<th>Ecological Connectivity</th>
<th>Pollutant Storage</th>
<th>Nutrient Processing</th>
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