Title
Looking at Remedies for the Urban Stream Syndrome from a new Perspective: Novel Approaches for Restoring Preurban Hydrology and Water Quality in Human-Impacted Catchments

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Looking at Remedies for the Urban Stream Syndrome from a new Perspective:
Novel Approaches for Restoring Preurban Hydrology and Water Quality in
Human-Impacted Catchments

DISSERTATION

submitted in partial satisfaction of the requirements
for the degree of

DOCTOR OF PHILOSOPHY

in Civil Engineering

by

Asal Askarizadeh Bardsiri

Dissertation Committee:
Professor Stanley Grant, Chair
Professor David Feldman
Professor Brett Sanders

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DEDICATION

To

My father, Mansour Askarizadeh Bardsiri,
Who is gone but not forgotten,
For raising me to be a dedicated, ambitious, and inspired woman

My mother, Dorreh Reyhany Esfehany,
For raising me with her unconditional love and support

My soulmate, Milad Ranjkesh Khorasani,
For giving meaning to my life, and a new dimension to my existence

And my sister, Ghazal Askarizadeh Bardsiri,
For always brining joy and happiness into my life

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ABSTRACT OF THE DISSERTATION

Looking at Remedies for the Urban Stream Syndrome from a new Perspective:
Novel Approaches for Restoring Preurban Hydrology and Water Quality in
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By

Asal Askarizadeh Bardsiri

Doctor of Philosophy in Civil Engineering

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Professor Stanley Grant, Chair

Urbanization adversely impacts stream water quality and ecosystem services. It also disrupts the water and sediment budgets of streams. Novel technologies and out-of-the-box approaches are required to address the impaired stream health and function in human-impacted catchments. In this thesis, I look at the degraded condition of urban streams from both hydrological and water quality perspectives, and propose methods for returning the impaired urban streams condition close to the preurban state. From the hydrological standpoint, I explore the role of low impact development technologies in restoring natural water balance over urbanized catchments. I demonstrate that over annual time scales, the volumes of stormwater that should be infiltrated and harvested can be estimated from a catchment scale water balance given local climate conditions and preurban land cover. I conclude that for all but the wettest regions of the world, a much larger volume of stormwater runoff should be harvested than infiltrated to maintain stream hydrology in a preurban state. From the water quality perspective, I investigate the negative impacts of excess in-stream

nitrate concentration on human health and the ecosystem; and develop a catchment scale modeling framework for its management. This modeling framework advances the state-of-the-art by taking into account the natural ability of streams in treatment of nitrate (by biotic assimilation and denitrification). My model builds on a seminal study [Mulholland et al., 2008, Nature, 452, 202-205] that found in-stream treatment of nitrate declines non-linearly with increasing nitrate concentration in a stream. I explore the implications of this result for nitrate management in an urbanizing watershed in southeastern Australia. To this end, I couple the correlation for in-stream processing of nitrate with a stream network model of the Jacksons Creek watershed (Victoria, Australia). By exploring various scenarios for nitrate loading rate in the effluent of a recycled water plant within the catchment, stream network model predicts that as nitrate loading from a sewage treatment plant increases (or decreases), Jacksons Creek responds by reducing (or increasing) in-stream nitrate removal. Thus, the non-linear nature of in-stream treatment may reinforce socio-ecological feedback loops that drive urban streams into healthy or degraded states.

Chapter 1

Introduction

1.1. Problem Statement

The urban stream syndrome represents a constellation of characteristics that are frequently observed in urban streams, including impaired ecosystem structure and function, and changes in stream hydrology and morphology that can put at risk native habitats and adjacent properties (e.g., by increasing flood or erosion risks), and contribute to poor water quality [Walsh et al., 2005; Meyer et al., 2005; Wenger et al., 2009]. Many of the symptoms of the urban stream syndrome can be traced back to changes in catchment hydrology associated with increasing catchment imperviousness (e.g., by replacing grasslands or forests with roofs, parking lots, and roads) and installing formal drainage systems (e.g., storm drainage systems intended to prevent flooding by moving water from the urban landscape quickly to streams).

Curing the urban stream syndrome requires addressing the root causes of the disease, although it is a matter of debate precisely how this should be done, or even if it is feasible in many urban catchments. Even if all of the symptoms of the urban stream syndrome cannot be addressed in a given catchment, it should still be possible to reduce the impacts of some of the more significant stressors through smart watershed management.

For example, in a typical natural catchment, over an annual time-scale, the total volume of water that escapes the catchment through evapotranspiration and annual streamflow is equal to the mean annual rainfall. However, urbanization perturbs this natural water balance by redistributing mean annual rainfall between evapotranspiration and streamflow, and altering how water is delivered to the stream; from subsurface flow paths in the preurban state to a mixture of subsurface and overland flow. Employment of Low Impact Development (LID) technologies are novel alternative for restoring the lost infiltration and evapotranspiration in urban catchments by infiltrating and harvesting stormwater runoff.

The other example of smart watershed management approach is in-streams nitrate concentrations management through thoughtful assessment of all point and non-point sources of pollution, and consideration of natural treatment functions (or ecosystem services) that streams themselves may provide. At its core, the nitrate problem stems from the way humans alter natural landscapes for agricultural and urban land uses. These modifications often provide pathways through which bioavailable nitrogen flows from its point of application (e.g., as fertilizer on a garden or agricultural field) to streams and receiving waters. These pathways can be categorized as “non-point source” and “point-source”, depending on whether the location at which the nitrate is discharged to a stream can be readily identified (e.g., the end of a pipe, point source) or not (non-point source).

From a management perspective, watersheds that are undergoing a major transformation (e.g., from agricultural to urban land use) are particularly interesting, because the associated "terraforming" of the landscape opens up many possibilities for innovative management of point- and non-point sources of nitrate.

1.2. Research Objectives and Thesis Organization

In this thesis, I look at the degraded urban stream condition problem from both hydrological and water quality standpoints and propose remedy approaches. From the hydrological perspective, in **Chapter 2** [Askarizadeh et al., 2015], I describe how LID technologies can be used, together with simple hydrological water budgets, to mimic preurban stream hydrology and potentially reverse the urban stream syndrome. While there are many different forms of LID, in this chapter I focused on distributed technologies (such as rain gardens, green roofs, and bioswales) that capture urban stormwater runoff where it is generated (at the house-to-neighborhood scale) and then either: (1) infiltrate the runoff to support shallow groundwater and base flow in nearby streams; or (2) harvest the runoff, by which we mean use the captured stormwater runoff for any purpose that keeps it out of the stream. Examples of (2) are technologies that return the stormwater to the atmosphere (e.g., through evapotranspiration, as in a green roofs or irrigation of ornamental gardens) or utilize the storm water for in-home "fit for purpose" activities, such as flushing toilets, laundry, or even showering. In this chapter I present a hydrological framework for calculating the volume of stormwater runoff that should be infiltrated (LID_I) or harvested (LID_H) depending on local climate and preurban land-cover. In particular, the main conclusions of our chapter include: (1) the optimal ratio of infiltrated to harvested water (LID_I/LID_H) for any given urban catchment depends on local mean annual rainfall and preurban fraction of forest, not effective imperviousness, (2) in most regions of the world returning stream hydrology to a preurban state will require that more stormwater is harvested than infiltrated to compensate for lost evapotranspiration (3) the LID_I/LID_H ratio can guide the selection of appropriate LID technologies for urban watersheds, and (4) the successful use of LID to "cure" the urban stream syndrome is contingent on many path-dependent processes such as the ecological condition of streams and so-called cognitive lock-in, in which a watershed's history (both hydrological and social-ecological) impacts public perception of streams and their associated riparian zones, thereby determining the trajectory of future stream management. The supplementary information of this chapter is provided in **Appendix A**.

From the water quality perspective, I focus on catchment scale in-stream nitrate management. Relative to nitrate, two in-stream ecosystem services are of particular interest: the permanent removal of nitrogen from a stream by denitrification and assimilation by stream flora and fauna. In **Chapter 3,** I set out to conduct a series of modeling studies to explore the result of various nitrate point source management scenarios on natural in-stream nitrate removal efficiency by developing a catchment scale stream network model for an urbanizing catchment in southeast Australia. My modeling framework is built on results of a previously published seminal study of nitrate in-stream treatment [Mulholland et al., 2008]. They reported that nitrate removal efficiency declines non-linearly with increasing stream nitrate concentration. I incorporated this non-linear power law correlation into a catchment scale stream network model and studied the impact of changes in the way nitrate point sources are managed on stream nitrate concentration and watershed health. The major goal this chapter is to develop a decision support tool that can empower managers with the science they need to better control point source nitrate pollution. This framework is also to advance current state-of-the-art of catchment scale in-stream nitrate fate and transport modeling by considering the natural ability of streams in removing in-stream nitrate. The supplementary information of this chapter is provided in **Appendix B.**

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Chapter 2

From Rain Tanks to Catchments: Use of Low Impact Development to Address Hydrologic Symptoms of the Urban Stream Syndrome [1]

Abstract

Catchment urbanization perturbs the water and sediment budgets of streams, degrades stream health and function, and causes a constellation of flow, water quality, and ecological symptoms collectively known as the urban stream syndrome. Low impact development technologies address the hydrologic symptoms of the urban stream syndrome by mimicking natural flow paths and restoring a natural water balance. Over annual time scales, the volumes of stormwater that should be infiltrated and harvested can be estimated from a catchment scale water balance given local climate conditions and preurban land cover. For all but the wettest regions of the world, a much larger volume of stormwater runoff should be harvested than infiltrated to maintain stream hydrology in a preurban state. Efforts to prevent or reverse hydrologic symptoms associated with the

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urban stream syndrome will therefore require: (1) selecting the right mix of LID technologies that provide regionally tailored ratios of stormwater harvesting and infiltration; (2) integrating these LID technologies into next-generation drainage systems; (3) maximizing potential co-benefits including water supply augmentation, flood protection, improved water quality, and urban amenities; and (4) long-term hydrologic monitoring to evaluate the efficacy of LID interventions.

2.1. Introduction

Catchment urbanization is associated with a reduction in stream health, a condition known as the urban stream syndrome [Walsh et al., 2005; Meyer, Paul, and Taulbee, 2005; Wenger et al., 2009]. Marked symptoms of the urban stream syndrome include altered streamflow, morphology, water quality, and ecosystem structure and function (**Figure 2.1.A**). Although underlying causes of the urban stream syndrome will vary among catchments, its hydrologic symptoms are generally associated with replacing grassland and/or forests with impervious surfaces such as roads, parking lots, roofs, and sidewalks; building drainage and flood control infrastructure to convey rapidly stormwater runoff to streams (so-called formal drainage systems); and altering catchment water budgets (e.g., through water imports and exports) (**Figure 2.1.B**) [Walsh et al., 2005; Leopold, 1968; Burns et al., 2012; Townsend-Small et al., 2013; Groffman et al., 2005; Hughes et al., 2014].

Increasing catchment imperviousness generally reduces infiltration and evapotranspiration of rainfall, whereas formal drainages increase the hydraulic connectivity between catchments and streams [Dunne, 1978; Gordon et al., 2013; Liu et al., 2008; Walsh and Kunapo, 2009]. These two modifications have opposing effects on streamflow during wet and dry weather. During wet weather, the volume of stormwater delivered to a stream increases, the lag time between rainfall and storm flow gets shorter, and peak flow rate increases [Brown, 1988; Rose and Peters, 2001; Miller and et al., 2014]. During dry weather, streamflow decreases due to reduced infiltration over interannual time scales [Hamel, Daly, and Fletcher, 2013; Walsh, Fletcher, and Burns, 2012], although there are exceptions to this rule. Water importation can increase dry weather streamflow by increasing [Townsend-Small et al., 2013]: perennial discharge of wastewater effluent and nuisance runoff; and/or groundwater seepage into streams from leaks in subterranean drinking water supply and sewage collection pipelines. Management of surface water impoundments (e.g., dams and reservoirs) can also increase dry weather streamflow [Hopkins et al., 2015]. All of these catchment modifications, in addition to altering stream hydrology, degrade streamwater quality by raising stream temperature, changing the balance of nutrients, carbon, and oxygen in a stream, and facilitating the mobilization and transport of fine sediments, chemical pollutants, and human pathogens and their indicators [Walsh et al., 2005; Meyer, Paul, and Taulbee, 2005; Wenger et al., 2009; Miller et al., 2010; Reeves et al., 2004; Grant, Litton-Mueller, and Ahn, 2011; Rippy et al., 2014; Surbeck et al., 2010; Welty et al., 2009; Booth and Jackson, 1997]. Changes in water quality and hydrology (both symptoms of catchment urbanization) affect stream morphology, stability, ecology, and chemistry [Surbeck et al., 2010; Welty et al., 2009; Booth and Jackson, 1997; Hession, 2001; Poff, Bledsoe, and Cuhaciyan, 2006; Nilsson et al., 2003; Ometo et al., 2000; Morse, Huryn, and Cronan, 2003].

Catchment urbanization is commonly quantified using two metrics: total imperviousness and effective imperviousness [Walsh et al., 2005; Meyer, Paul, and Taulbee, 2005; Wenger et al., 2009; Welty et al., 2009; Booth and Jackson, 1997]. Total imperviousness is the fraction of catchment area covered with constructed impervious surfaces such as asphalt and roofs. Effective imperviousness represents the impervious fraction of the catchment area with hydraulic connection to a stream through a formal drainage system. Compared to total imperviousness, effective imperviousness is a better predictor of streamwater quality, ecological health, and channel form [Hatt et al., 2004; Taylor et al., 2004; Vietz et al., 2014]. Total imperviousness does not take into account whether flow from an impervious surface is conveyed directly to a stream, or instead drains to adjacent pervious areas where opportunities for filtration, infiltration, and flow attenuation are provided. The ecological condition of streams typically exhibits a wedge-shaped dependence on total imperviousness: streams in catchments with low total imperviousness exhibit a range of ecological conditions (from degraded to healthy) that narrows with increasing total imperviousness due to reduction in the maximum attainable stream health [Walsh et al., 2005; Meyer, Paul, and Taulbee, 2005; Wenger et al., 2009]. Effective imperviousness exhibits a less variable negative correlation with stream ecological condition, water quality, and channel form [Walsh and Kunapo, 2009].

The negative correlation between effective imperviousness and stream health raises the question: can hydrologic symptoms of the urban stream syndrome be prevented and/or reversed through urban forms that keep effective imperviousness low? Effective imperviousness can be kept low as an urban community develops (or reduced through retrofits of an already developed catchment) using technologies that intercept runoff from impervious surfaces at a variety of scales [Fletcher et al., 2008; Fletcher, Andrieu, and Hamel, 2013]. The intercepted runoff can be infiltrated to support groundwater (e.g., with unlined biofilters and permeable pavement), exported to the atmosphere by evapotranspiration (e.g., using green roofs, rain gardens, vegetated swales, wetlands, and urban forests), redirected from storm sewer systems to pervious surfaces (e.g., with downspout disconnection), and/or exported through the sanitary sewer system to downstream receiving waters (e.g., using rainwater tanks for toilet flushing) (**Figure 2.1.C**, see also **Table A.1** in **Appendix A**). These environmentally sensitive stormwater management systems go by a variety of names, including green infrastructure and low impact development (LID) technologies in the U.S., Water Sensitive Urban Design in Australia and Canada, and Sustainable Urban Drainage Systems in England [Fletcher et al., 2015]. In this review, we adopt the term LID technologies.

Acquiring and maintaining public support for LID technologies requires demonstrating that they are effective at minimizing flood risk and the negative impacts of urbanization on human and ecosystem health [Walsh et al., 2005, Brown et al., 2015; Poff et al., 2010]. In this review, we explore: (1) the variety of modeling approaches available for supporting LID selection and evaluation; (2) technologies available for stormwater infiltration and harvesting; and (3) implementation challenges including maintenance, climate change, path dependence, and site-specific constraints. A number of review articles have been written on LID technologies and their use for mitigating hydrologic, water quality, and ecological symptoms of the urban stream syndrome [Dietz, 2007; Rowe, 2011; Scholz and Grabowiecki, 2007; Roy-Poirier, Champagne, and Filion, 2010; Davis et al., 2009; Hamel and Fletcher, 2014; Burns et al., 2015]. However, these tend not to consider simultaneously the international scope of the problem, its potential solutions, and policy and technological barriers to practical implementation. Our review adopts a multidisciplinary (hydrology, engineering, social science, and ecology), multiscale (from individual LID types to whole catchments), and binational (U.S. and Australia) perspective.

Figure 2.1. Symptoms, causes, and cures of hydrologic perturbations associated with the urban stream syndrome. (A) Symptoms include: (1) altered streamflow (base flow, peak flow, annual runoff volume, flow variability); (2) altered stream morphology (stream width, depth, complexity, and disconnection from the riparian zone, hyporheic zone, and

flood plain); (3) impaired water and sediment quality (trash, nutrients, dissolved oxygen, toxicants, suspended solids, temperature); and (4) shifts in biological composition (loss of native species, reduction in sensitive species, increase in tolerant species, increase in invasive species) and loss of ecosystem services (organic matter retention and processing, nutrient removal, primary production, and respiration). (B) Causes include: (1) replacing grassland and/or forests with impervious surfaces such as roads, parking lots, roofs, and sidewalks; (2) building stormwater drainage and flood control infrastructure to convey rapidly stormwater runoff to streams (formal drainage systems); (3) reducing stream complexity by burying, straightening, and concrete-lining streams; and (4) altering overall water and sediment budgets through water importation, the construction of debris dams, and surface water impoundments. (C) Examples of LID technologies that can potentially address the hydrological challenges associated with the urban stream syndrome include unlined technologies that infiltrate stormwater runoff (e.g., unlined biofilters and permeable pavement) and technologies that harvest and export stormwater runoff from the catchment (e.g., green roofs and rainwater tanks used for irrigation or indoor toilet flushing). Top row includes images of urban creeks and drains in Orange County, California (from left to right: San Diego Creek, Costa Mesa Channel, Fullerton Creek, and a drain in the City of Irvine). Middle row includes two streetscapes and a buried stream in Orange County California, and Parker Dam at the start of the Colorado Aqueduct on the California−Nevada border. Bottom row includes an unlined biofilter in Latvia; permeable pavement in Westminster, California; green roof on a public building in Houston, Texas; and a Cistern in Melbourne (Australia).

2.2. Catchment Scale Urban Water Balance

Case for Volume over Peak Flow Rate. In many countries, stormwater regulations place limits on the peak flow rate or high flow duration allowed to enter a stream from individual properties [Booth and Jackson, 1997]. To comply with these regulations, property owners typically install stormwater detention ponds that capture and slowly release runoff from large storms [Guo, 2001]. There are a number of well-documented problems with this approach, including [Booth and Jackson, 1997; Petrucci et al., 2013; Petrucci et al., 2014; Emerson, Welty, and Traver, 2005]: (1) the simultaneous release of stormwater from many properties within the catchment can cause downstream peak flows to exceed predevelopment conditions and erode downstream channels, even if the peak flows from individual properties remain within regulatory limits; (2) reduced infiltration associated with impervious surfaces cuts off the primary means by which water is normally supplied to a stream (through subsurface flow paths and resupply of shallow groundwater), and detention ponds do not typically address the problem; and (3) the superposition of post storm flows from multiple detention basins in a catchment distorts downstream dry weather flow regimes. Although a number of stream "sustainability" metrics have been proposed [Reichold et al., 2010; Giacomoni, Zechman, and Brumbelow, 2012], controlling (and ideally eliminating) the volume of stormwater runoff flowing to a stream through formal drainage systems is a prerequisite for maintaining and restoring the preurban flow regime (for reasons that will be detailed in the following sections [Walsh, Fletcher, and Burns, 2012; Petrucci et al., 2013, Petrucci et al., 2014, Emerson, Welty, and Traver, 2005]).

Impact of Urbanization on Catchment-Scale Water Budgets. Drawing on analogies with environmental flow management, Walsh et al., 2012 proposed a catchment-scale water balance (or "bucket") model to estimate the volume of water that should be infiltrated and harvested to maintain stream hydrology as close as possible to its preurban state. **Equation (1)** represents an annual water balance for a typical natural catchment assuming: the volume of water associated with soil moisture and shallow groundwater does not change appreciably over annual and longer time scales; and all water that infiltrates into the catchment eventually flows to the stream through subsurface routes (i.e., the infiltrated water is not lost from the catchment by deep seepage) [Zhang, Dawes, and, Walker, 2001; Zhang, Dawes, and Walker, 1999].

$$MAR = ET + S \tag{1}$$

Variables appearing in this equation include the mean annual rainfall depth in the catchment (*MAR*, volume of rainfall per catchment area per year), evapotranspiration depth (*ET*, volume of water returned to the atmosphere per catchment area per year), and annual streamflow depth (*S*, volume of water flowing in a stream per catchment area per year). The units of "depth per year" can be interpreted as the depth of water that would be obtained if the annual water volume associated with each term in **equation (1)** was evenly distributed over the catchment area.

Over annual time scales, subsurface flow constitutes the majority of streamflow in most natural catchments, including during storm events [Bhaskar and Welty, 2015; Burns et al., 2013; Booth, 1991; Buttle, 1994]. In this context, subsurface flow (sometimes referred to as "old water") is defined as rainfall that infiltrates and flows to a stream through shallow groundwater or the vadose zone as interflow and throughflow. By contrast, the contribution of overland flow (technically, Horton Overland Flow) to annual streamflow is generally small in natural catchments [Bhaskar and Welty, 2015; Burns et al., 2013; Booth, 1991; Buttle, 1994]. Neglecting overland flow, the annual water balance for a natural catchment can be approximated by **equation (2)** where S_{sub}^{pu} represents the contribution of subsurface flow to preurban streamflow (note the superscript "*pu*" refers to "preurban").

$$MAR = ET^{pu} + S_{sub}^{pu} \tag{2}$$

Urbanization perturbs this water balance in a number of ways by (1) redistributing *MAR* between *ET* and *S*, generally decreasing *ET* (except in regions where significant water importation occurs, see below) and increasing S; and (2) altering how water is delivered to the stream, from subsurface flow paths in the preurban state ($S = S_{sub}^{pu}$) to a mixture of subsurface flow (S_{sub}^{u}) and overland flow from effective imperviousness (S_{EI}) in the urban state: $S = S_{sub}^{u} + S_{EI}$ (note the superscript "u" refers to "urban"). Thus, **equation (3)** represents an annual water balance for an urbanized catchment (**Figure 2.2.A**).

$$MAR = ET^{u} + S_{sub}^{u} + S_{EI} \tag{3}$$

Values for S_{sub}^{u} and S_{EI} can be calculated from the mean annual rainfall (MAR), the fraction of the total catchment area that is covered with effective imperviousness (f_{EI}), and the stream coefficients for undeveloped (C_S) and effective impervious (C_{EI}) areas:

$$S_{sub}^{u} = MAR \times C_S \times (1 - f_{EI}) \tag{4a}$$

$$S_{EI} = MAR \times C_{EI} \times f_{EI} \tag{4b}$$

To illustrate the effect of urbanization on catchment water balance, evapotranspiration, subsurface flow, and overland flow are plotted against effective imperviousness in **Figure 2.2.B**. To generate this plot, we adopted stream coefficient and impervious runoff coefficient values of C_S= 0.3 and C_{EI}= 0.8, respectively; a region-specific procedure for calculating these coefficients is described later. As illustrated in the **Figure 2.2**, the Walsh bucket model predicts that urbanization is associated with a decline in evapotranspiration (because forests and/or grassland is replaced with impervious surface, denoted as the gray region in **Figure 2.2.B**), a decline in subsurface flow to streams (because resupply of the shallow groundwater by infiltration is reduced with increasing imperviousness, blue region), and an increase in the volume of overland flow entering the stream on an annual basis from effective imperviousness (red region).

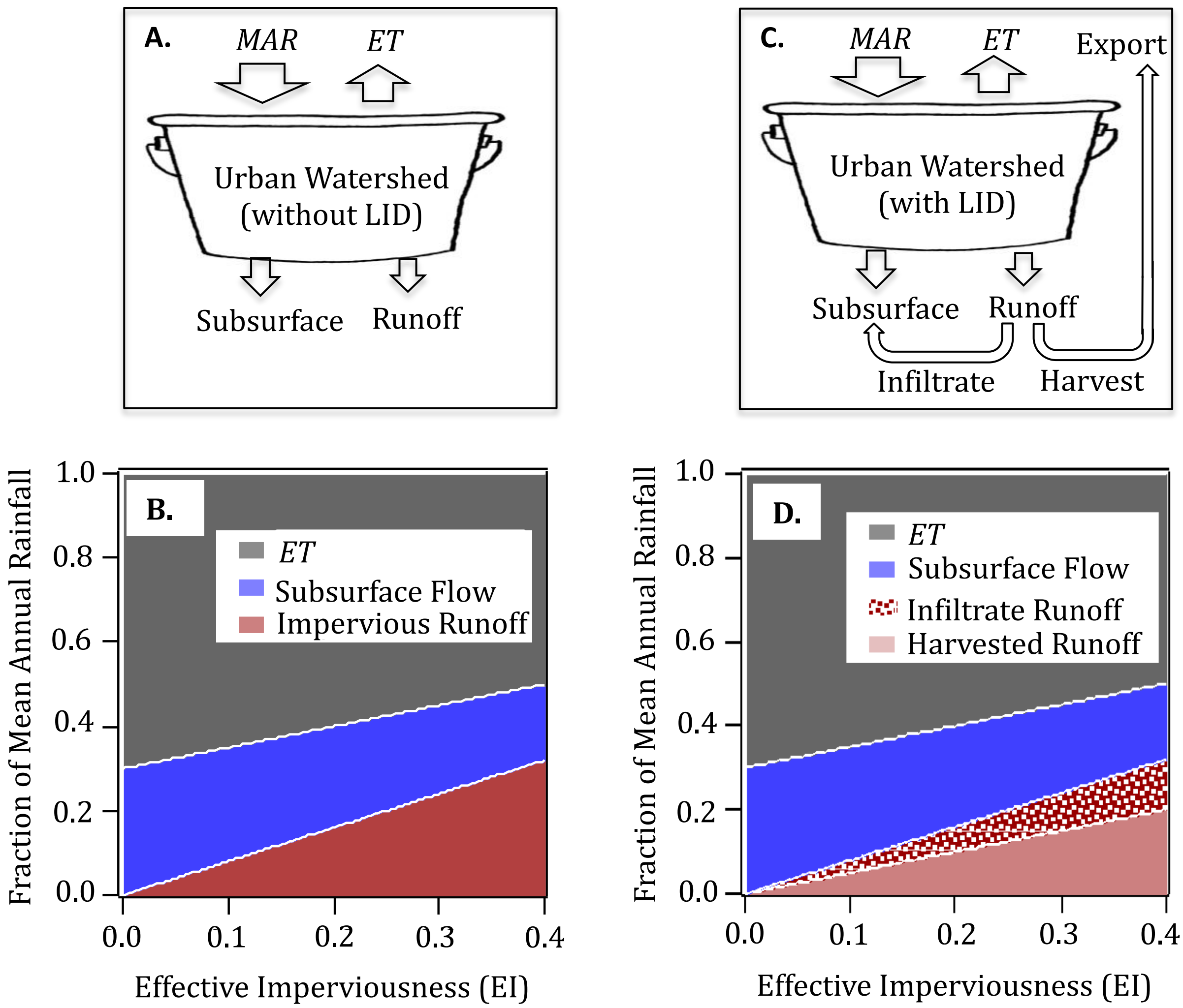

Figure 2.2. Catchment-scale water balance (or "bucket model") for calculating the volume of stormwater runoff that should be infiltrated and harvested. (A) Simplified form of the steady-state annual water budget for a catchment in which LID technologies are not implemented. Mean annual rainfall (*MAR*) is partitioned between evapotranspiration (*ET*), streamflow associated with subsurface infiltration (S_{sub}^{pu}), and streamflow associated with storm water runoff from connected imperviousness (S_{EI}). (B) Influence that urbanization (represented by effective imperviousness, f_{EI}) has on the distribution of *MAR* between *ET*, subsurface flow S_{sub}^{u}, and impervious runoff S_{EI}. These curves were generated using rearranged versions of **equation 3, 4a**, and **4b.** (C) LID technologies can mitigate the effects of effective imperviousness on catchment water balance by capturing impervious runoff for infiltration (LID_I, to support subsurface flow to the stream) and harvesting and exporting impervious runoff from the catchment (LID_H, to compensate for the decline in evapotranspiration frequently associated with urbanization). (D) By infiltrating and

harvesting stormwater runoff in the right proportions (determined by **equations (6a)** and **(6b)**), it is theoretically possible to maintain annual streamflow at preurban levels as effective imperviousness rises. Note that, technically speaking, if all runoff from effective imperviousness is harvested or infiltrated, then by definition effective imperviousness is zero. Thus, the horizontal axis in panel D should be regarded as the effective imperviousness that would have resulted if LID technologies had not been implemented. Curves in this panel were generated using rearranged versions of **equations (5), (6a),** and **(6b).** In all cases, the following stream and impervious runoff coefficients were assumed: C_S= 0.3, C_{EI}= 0.8.

Maintaining Preurban Hydrology through Infiltration and Harvesting. Two categories of LID technologies can be deployed to support preurban streamflow as a catchment develops. The first type, infiltration-based LID technologies, transfer stormwater runoff to the subsurface where it can recharge groundwater supplies and provide base flow for local streams. The second type, harvest-based LID technologies, capture the remaining runoff (i.e., the stormwater not infiltrated) and use it for any purpose that keeps it out of the stream (e.g., irrigation of ornamental plants and toilet flushing) [Grant et al., 2012]. In theory, preurban streamflow can be maintained if the right number and mix of these two LID types are deployed; namely, enough infiltration- and harvest-based LID technologies to exactly compensate for the infiltration and evapotranspiration lost by replacing forests and grassland with impervious surfaces. Applying these concepts to the catchment water balance described above, we arrive at **equation (5)**, where LID_I and LID_H denote the annual stormwater runoff depths that should be infiltrated and harvested, respectively (**Figure 2.2.C**):

$$MAR = (ET^{u} + LID_H) + (S_{sub}^{u} + LID_I) \tag{5}$$

The first term in parentheses equals the preurban evapotranspiration (ET^{pu}), whereas the second term in parentheses equals the preurban subsurface flow to the stream

(S_{sub}^{pu}). The volumes of runoff that should be infiltrated and harvested depend on the fraction of the catchment area covered with effective imperviousness f_{EI}:

$$LID_I = MAR \times C_S \times f_{EI} \tag{6a}$$

$$LID_H = MAR \times (C_{EI} - C_S) \times f_{EI}, \qquad C_{EI} > C_S \tag{6b}$$

Returning to the example presented above, subsurface flow to the stream is maintained at preurban levels (30% of mean annual rainfall), provided that a portion of stormwater runoff is captured and infiltrated as dictated by **equation (6a)**; i.e., the sum of the blue and brown stippled regions equals 30% across the entire range of f_{EI} in **Figure 2.2.D**. The portion of stormwater runoff not infiltrated, **equation (6b)**, should be harvested and kept out of the stream (light burgundy color, **Figure 2.2.D**). In this hypothetical example, the hydrology of the local stream is unchanged as the catchment urbanizes because: (1) subsurface flow to the stream is maintained at predevelopment levels, and (2) no stormwater runoff flows overland to the stream via effective imperviousness.

Tailoring Infiltration and Harvesting to Specific Regions. An interesting and previously overlooked consequence of the Walsh bucket model is that, for a given set of values for C_S and C_{EI}, the relative proportion of runoff volume that should be infiltrated and harvested is constant; i.e., their ratio does not depend on the fraction of the catchment area covered by effective imperviousness:

$$\frac{LID_I}{LID_H} = \frac{1}{(C_{EI}/C_S - 1)}, \quad C_{EI} > C_S \tag{7}$$

In the hypothetical example presented above, we arbitrarily selected values for C_S and C_{EI}. Region-specific stream coefficients and impervious runoff coefficients can be estimated from previously published correlations. For example, the impervious runoff coefficient can be estimated from an empirical correlation proposed by Walsh et al., 2012 based on runoff data collected in and around Melbourne (Australia):

$$C_{EI} = 0.230 + 0.206 \log_{10}(MAR) \quad (8)$$

Because this correlation is for impervious surfaces (as opposed to natural landscapes), it will likely apply to cities other than Melbourne (although this is an obvious target for future research). The stream coefficient C_S can be estimated from a correlation developed by Zhang et al., 1999 and 2001 based on streamflow measurements from 250 catchments worldwide. Zhang's correlation depends on the fraction f_F of the preurban catchment area covered with forest, together with evapotranspiration depths for forests (ET_F) and herbaceous plants and soil moisture (ET_H):

$$C_S = 1 - ET/MAR \quad (9a)$$

$$ET = f_F ET_f + (1 - f_F) ET_H \quad (9b)$$

$$ET_F = \frac{1+2(1410/MAR)}{1+2(1410/MAR)+MAR/1410} \quad (9c)$$

$$ET_H = \frac{1+0.5(1100/MAR)}{1+0.5(1100/MAR)+MAR/1100} \quad (9d)$$

After substituting these correlations into **equation (7)**, we find the ratio LID_I/LID_H required to maintain preurban streamflow depends on only two variables: the mean annual rainfall MAR and the fraction of the preurban catchment area covered with forest f_F(**Figure 2.3**). The thick black curve in the figure denotes combinations of MAR and f_F for which equal volumes of stormwater runoff should be infiltrated and harvested; i.e., $\log_{10}(LID_I/LID_H)=0$. For most of the climate and preurban states encapsulated in the figure, considerably more stormwater should be harvested than infiltrated (i.e., most of the plot is occupied by regions to the left of the thick black curve). This result calls for an emphasis on LID technologies that harvest stormwater over a wide range of climates.

Another interesting implication of **equation (7)** is that cities with very different climates and geographical locations can have similar infiltration-to-harvest ratios, as illustrated in **Figure 2.3** for two hypothetical cities with an infiltration-to-harvest ratio of 30%. The first city (point labeled C_1) is located in a relatively dry climate (MAR = 575 [mm year^{-1}]) and was mostly unforested prior to urbanization (f_F = 0.3). The second city (point labeled C_2) is in a wetter climate (MAR = 1050 [mm year^{-1}]) and was mostly forested prior to urbanization (f_F= 0.9). Pasadena (California) and Baltimore (Maryland) are two U.S. cities that meet the criteria for C_1 and C_2, respectively.

In practice, some fraction of water volume infiltrated by LID will be exported from the catchment, for example, to the atmosphere by evapotranspiration and/or to deep aquifers by seepage. Thus, the ratio LID_I/LID_H needed to restore catchment water balance may be larger than predicted by **equation (7)**, because some portion of infiltrated stormwater is automatically exported from the catchment before it reaches the stream (LID technologies are discussed in **Section 2.3**).

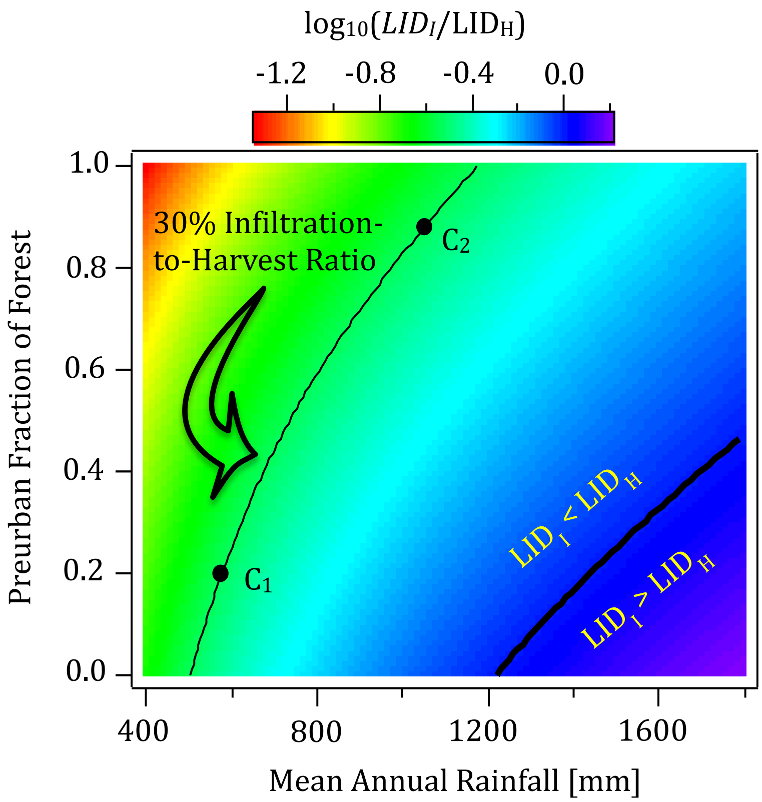

Figure 2.3. Relative volumes of runoff that should be infiltrated and harvested (LID_I/LID_H) to maintain a preurban flow regime in catchment streams, plotted as a function of mean annual rainfall (*MAR*) and the fraction of the preurban catchment covered with forest (f_F). Color denotes logarithmically transformed values of the ratio LID_I/LID_H calculated by combining **equations (7), (8)**, and **(9a)–(9d)**. Most of the plot area is located to the left of the thick black curve (which corresponds to combinations of *MAR* and f_F where the infiltration and harvest volumes are equal, $\log_{10}(LID_I/LID_H)=0$), implying that more stormwater should be harvested than infiltrated across most climates and preurban forest covers. The thin black curve corresponds to all values of *MAR* and f_F where the required infiltration volume is 30% of the required harvest volume. The dots on the curve (labeled C_1 and C_2) represent two cities with very different climates and preurban land covers but the same required infiltration-to-harvest ratio (see main text)

Strengths and Limitations of the Walsh Bucket Model. The strength of the catchment-scale water balance model presented above is its simplicity and the fact that it can be readily applied to various regions around the world; however, the model entails a number of assumptions that may not be satisfied in practice.

First, the catchment water balance **equation (1)** may not apply in all cases. For example, the importation of water to Los Angeles has caused dry weather flow in the region's urban impacted rivers to increase 250% or more over the past 50 years; summer flow in the iconic Los Angeles River has increased approximately 500% over that period of time.6 In other regions, the withdrawal of water from urban streams, together with sewer infiltration and inflow (I&I), can significantly alter a catchment's water balance. The Ipswich River in Massachusetts has gone dry for extended periods due to municipal water withdrawal [Zimmerman et al., 2010]. In metropolitan catchments surrounding Baltimore, Maryland, I&I can exceed annual streamflow [Bhaskar and Welty, 2012].

Second, in some urban catchments, subsurface water (i.e., "old water") is still a dominant source of storm flow in urban impacted rivers [Bhaskar and Welty, 2015]. Although the underlying mechanism for this observation is not well understood, a possible implication is that urbanization may induce excess storm flow in urban rivers via two mechanisms: (1) by increasing effective imperviousness (as assumed in the Walsh bucket model); and (2) by altering the rate at which old water is delivered to a stream during storms (e.g., by accelerating the transfer of rainfall to the subsurface through leaky storm and/or sanitary sewer systems). In urban areas where the second process applies, reducing effective imperviousness alone may not control the volume of water delivered to a stream during storms.

Third, the Walsh bucket model does not take into account regional physiography and geology that can influence both patterns of urbanization as well as intrastorm stream responses (e.g., the effects of urbanization on stream flashiness tends to be buffered in catchments with permeable soils, level slopes, and high lake density) [Hopkins et al., 2015].

In principle, the first limitation can be addressed by adding terms to the catchment water balance **equation (1)** that account for regional variations in the import and export of water over annual time scales. Addressing the second and third limitations, on the other hand, may require more sophisticated (spatially and temporally explicit) models that capture the influence of surface and subsurface storage and local hydrogeology on intrastorm, as well as interstorm, streamflow variability (see modeling tools in **Section 2.4**). Next we turn our attention to commonly adopted LID technologies, and discuss their utility in light of the catchment water balance model described above.

2.3. LID Technologies for Maintaining or Restoring Preurban Hydrology

The Walsh bucket model presented above suggests that LID technologies have the potential to remedy hydrologic symptoms associated with the urban stream syndrome. Translating theory to practice will require a diverse set of LID technologies tailored to (1) capture all stormwater runoff before it enters the stream; and (2) infiltrate and/or harvest the captured runoff in the proper proportions. In practice, many different factors go into the selection of LID technologies (e.g., flood protection, operation and maintenance costs, site-specific constraints, and human and ecosystem co-benefits) [Facility for Advancing Water Biofiltration, 2009; Walsh et al., 2015]. Here we take the position that the first-order concern in LID technology selection should be maintaining (or restoring) preurban flow regimes, with secondary consideration given to other constraints and benefits. Accordingly, in this section we classify several popular LID technologies relative to the three end points that underpin the Walsh bucket model presented in **Section 2.2**: the percent of runoff volume harvested, infiltrated, or left as overland flow (represented by vertices of the ternary diagram in **Figure 2.4**; see also **Table A.1** in **Appendix A**). Given our focus on restoring a preurban flow balance, we opted not to discuss technologies that work only by storage and attenuation, despite their utility for mitigating peak storm flows [Guo, 2001; Loperfido et al., 2014] (see beginning of **Section 2.2**).

Infiltration Technologies. Examples of infiltrative systems include infiltration trenches [Harrington, 1989; Charlesworth, Harker, and Rickard, 1989] and permeable pavement [Brattebo and Booth, 2003; Shuster et al., 2005] (represented in **Figure 2.4** by a teal arrow, gray arrow, gray dashed box, and brown arrow). Infiltration trenches and permeable pavement without under-drains (i.e., drains that collect some fraction of the outflow from a system) infiltrate the highest percentage of runoff (60–100% runoff removed) [Hirschman, Collins, and Schueler, 2008]. Permeable pavement with under-drains infiltrate less runoff because a fraction of outflow is piped to the storm sewer system (25–66% runoff removed [Hirschman, Collins, and Schueler, 2008], gray arrow, **Figure 2.4**). Rerouting this piped fraction to a storage facility can transform permeable pavement with under-drains from infiltration to hybrid systems (i.e., technologies that both infiltrate and harvest, dashed gray box, **Figure 2.4**), assuming that the captured water is used for irrigation (evapotranspiration) or in-house activities (e.g., toilet flushing) that transfer the water to the sanitary sewer system [Beecham and Chowdhury, 2012; Fletcher et al., 2013]. Although treated stormwater is rarely used for domestic purposes in the U.S., such systems are actively being trialed in Southeast Australia (see **Section 2.4**) [Low et al., 2015].

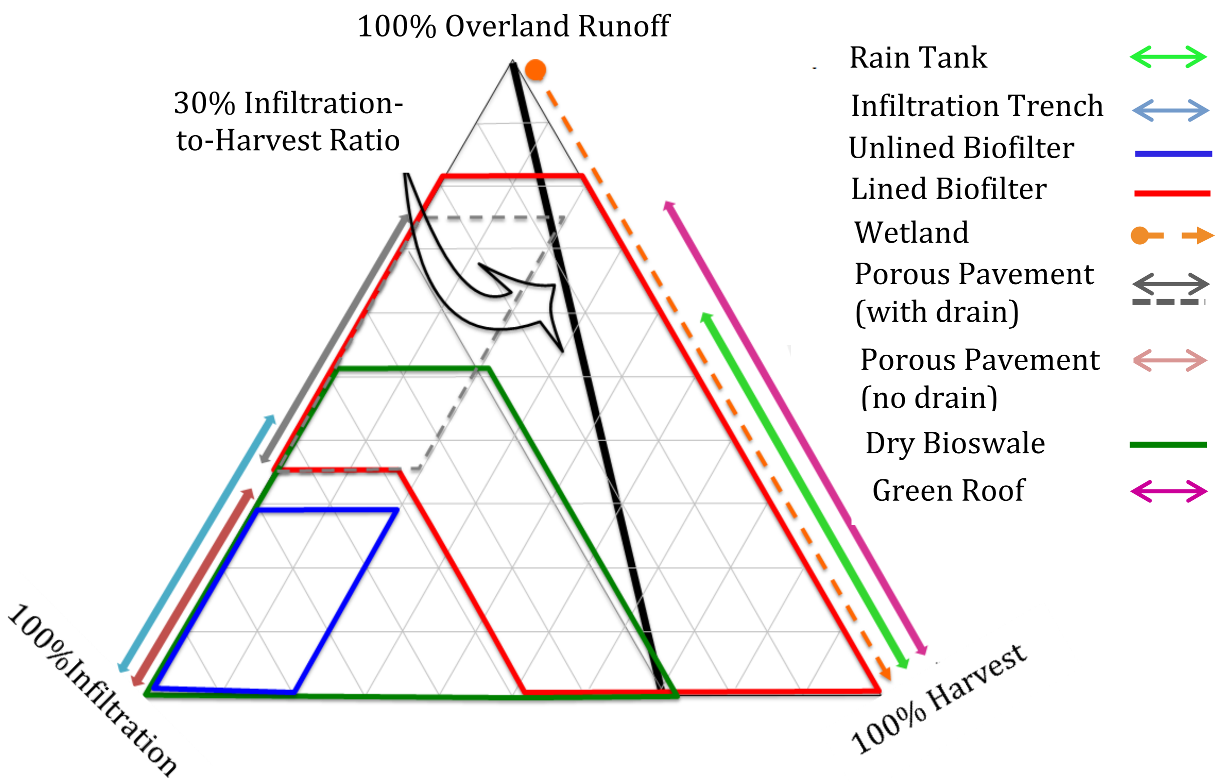

Figure 2.4. Ternary representation of field and laboratory data on the performance of popular LID technologies relative to percentage of runoff volume infiltrated (lower left vertex), harvested (lower right vertex), and allowed to flow to the stream through connected imperviousness (top vertex). The designation "with drain" refers to systems in which treated effluent can be routed to storage facilities for nonpotable uses, such as garden irrigation and toilet flushing. The designation "without drain" refers to systems in which treated effluent leaches directly into the subsurface. Arrows along the side of the ternary diagram denote systems that are used primarily for infiltration (left leg of the triangle) or for harvesting (right leg of the triangle). Polygons indicate hybrid systems that can be "tuned" to provide specific infiltration-to-harvest ratios. Solid colored lines reflect observed performance, whereas colored dashed lines denote theoretical performance (i.e., the performance is possible but not documented). The thick black line with a blue halo marks the location of hybrid systems that achieve a 30% infiltration-to-harvest ratio (corresponding to the black curve in **Figure 2.3**, see text). Data used to generate this figure are discussed in the main text.

Harvesting Technologies. Examples of harvest-based LID include green roofs [Berndtsson, 2010; Nicholson et al., 2009; Wong, 2006], rainwater tanks [Coombes and Kuczera, 2003; Kahinda, Taigbenu, and Boroto, 2007], and wetlands [Persson, Somes, and

Wong, 1999; Rousseau et al., 2008] (shown as a pink arrow, green arrow, and orange dashed arrow, respectively, **Figure 2.4**). A broad range of harvest efficiencies have been noted for green roofs (23−100% runoff removed) [Hirschman, Collins, and Schueler, 2008; Ahiablame, Engel, and Chaubey, 2012]. Green roofs export runoff mostly in the form of evapotranspiration, with the soil/media matrix dominating export in the winter (low harvest: ~34% runoff removed) and the "green" component contributing to export in the summer (high harvest: ~67% runoff removed) [Berndtsson, 2010]. Rainwater tanks harvest between 35 and 90% of runoff on average 72 depending on the ratio of tank size to roof area, storm frequency and duration, the number of acceptable rainwater uses (e.g., toilet flushing, clothes washing, hot water supply, or garden irrigation), and building occupancy. Human use of rainwater is expected to be higher in multistory residential and office buildings than in commercial/industrial buildings, given the greater number of inhabitants per unit area of imperviousness [Burns et al., 2010]. Although wetlands typically export relatively small volumes of runoff in the form of evapotranspiration (0−3% runoff removed [Burns et al., 2005; Hirschman, Collins, and Schueler, 2008]), outflow can be tapped for human use, substantially increasing the overall percentage of runoff harvested. Upward of 50−100% harvest has been reported for wetland systems in South Australia and New South Wales, resulting in potable water savings of $120,000 to $663,120 per year (in 2006 AUD) [Hatt, Deletic, and Fletcher, 2006].

Hybrid Technologies. LID technologies that both harvest and infiltrate stormwater runoff, or "hybrid technologies", appear as polygons in **Figure 2.4**. Examples of hybrid technologies include unlined biofilters (no under-drain, blue polygon), partially or completely lined biofilters (with under-drain, red polygon) [Dietz and Clausen, 2005; Le

Coustumer et al., 2009], and dry bioswales (unlined with an under-drain, green polygon). The term "dry bioswales" refers to swales that are intended to dry out between storms. Two configurations for a household biofilter (lined with an underdrain versus unlined with no underdrain) are illustrated in **Figure 2.5**.

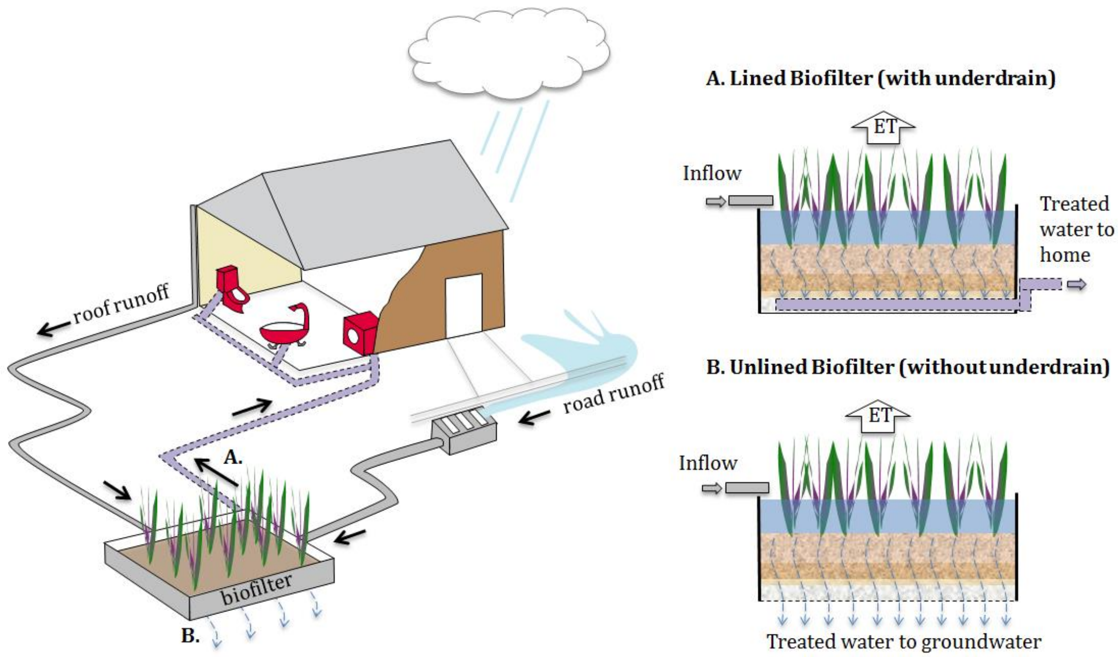

Figure 2.5. Biofilters are a hybrid LID technologies that can be tuned to achieve different levels of stormwater harvest and infiltration. In the example illustrated here a biofilter is configured to receive both roof and road runoff. In a harvest configuration, treated water from the biofilter can provide nonpotable water to the home for toilet flushing, laundry, and hot water supply (lined biofilter with underdrain, A). In an infiltration configuration, thebiofilter supports groundwater recharge and stream baseflow (unlined biofilter without underdrain, B). In both configurations, a portion of the water processed by the biofilter is lost to the atmosphere through evapotranspiration (*ET*), another form of harvesting. Colored layers in the biofilters (upper and lower right panels) delineate ponding zone (blue), filter media (brown), transition layer (light brown), and gravel layer (gray). Adapted from Figure 2 of Grant et al., 2013 and Grant et al., 2012.

To date, few studies have quantified the percent runoff harvested through evapotranspiration for hybrid systems. Values as low as 2–3% runoff removal have been reported for unlined biofilters; however, these percentages may be low because a substantial portion of infiltrated runoff passes into upper soil layers where additional (unquantified) evapotranspiration may occur [Hamel et al., 2011; Burns et al., 2012]. Higher evapotranspiration values (>19% runoff removed) have been reported in lined biofilters [Hamel et al., 2011]. Thus, a tentative range for percent runoff harvested via evapotranspiration across biofilters (lined and unlined) is 2−19%. Unlined biofilters are primarily infiltration systems, with evapotranspiration constituting their primary contribution to harvest (total runoff removed ranging from 73 to 99%; evapotranspiration, 2−19%; and infiltration, 71−97%) [Loperfido et al., 2014; Burns et al., 2010; Hatt, Deletic, and Fletcher, 2006]. In contrast, lined biofilters are often used to treat stormwater prior to discharge to a storm sewer system; the treated effluent can also be captured and stored for subsequent human use, increasing harvest potential (total runoff removed ranging from 20 to 100%; evapotranspiration, 2−19%; human use, 0−80%; and infiltration, 1−63%) [Hirschman, Collins, and Schueler, 2008; Hamel et al., 2011; Li et al., 2009]. Dry bioswales are effective for harvesting and infiltrating runoff, with near 100% runoff removal achieved over a broad combination of infiltration and harvesting percentages (total runoff removed ranging from 46 to 100%; evapotranspiration, 2−19%; human use, 0−54%; and infiltration, 27−96%) [Hirschman, Collins, and Schueler, 2008; Hamel et al., 2011; Li et al., 2009]. The effectiveness of dry bioswales for harvesting runoff can be attributed to their relatively large surface area to catchment area ratio, compared to other hybrid systems [Hirschman, Collins, and Schueler, 2008].

Matching LID Technologies to Storm Water Management Goals. According to **Figure 2.3**, the volume of stormwater that should be harvested far exceeds the volume that should be infiltrated for most climates and preurban forest cover. Thus, in many locales, the emphasis should be on harvest-based LID technologies. This may prove challenging in practice, because distributed harvest systems that capture stormwater runoff at its source (e.g., rainwater tanks and green roofs) only treat one form of impervious area (rooftops) leaving runoff from other, potentially much more extensive imperviousness (e.g., roads parking lots and residential driveways) untreated [Fletcher et al., 2008]. Although regional (or end-of-catchment) LID such as wetlands can be employed to harvest the remainder, this approach is at the expense of water quality in reaches upstream of regional facilities [Burns et al., 2012]. Alternatively, runoff from roads and driveways can be captured and harvested using distributed hybrid systems (e.g., lined biofilters, dry bioswales, and permeable pavement with underdrains) configured to provide nonpotable water for human use (configuration "A" in **Figure 2.5**).

At the parcel scale, LID technologies (or combinations of LID technologies) can be selected to match catchment-scale goals for the volume of runoff to be infiltrated and harvested. For the two hypothetical cities described in **Section 2.2** (see points C_1 and C_2, **Figure 2.3**), the required infiltration-to-harvest ratio is 30%, which translates to a straight line in **Figure 2.4** (see thick black line with blue halo). In practice, this infiltration-to-harvest ratio can be achieved by selecting hybrid technologies that cross or enclose the line (e.g., lined biofilters "tuned" to achieve the 30% target) and/or by a combination of infiltration and harvest technologies designed to operate toward the harvesting end of the spectrum (e.g., treatment trains consisting of large rain tanks that overflow to unlined biofilters) [Burns et al., 2012].

2.4. Optimizing LID Selection at the Catchment Scale

Modeling Tools. A number of modeling tools are available for optimizing the selection and siting of LID technologies so as to minimize flood risk, maximize human and ecosystem cobenefits, and stay within capital, maintenance, and operation costs [Elliott and Trowsdale, 2007; Jayasooriya et al., 2014]. These optimization schemes have several elements in common, including: (1) a spatially explicit (e.g., GIS-based) platform that includes information on the informal and formal drainage for a site and candidate locations for LID technologies; (2) a rainfall-runoff model that routes stormwater through the catchment; (3) an objective function that quantifies hydrologic performance (e.g., relative to stormwater harvest and infiltration targets, see **Section 2.2**) and costs of candidate LID configurations; and (4) an algorithm that identifies optimal solutions (e.g., by minimizing one or more objective functions) [Damodaram and Zechman, 2013; Yeh and Labadie, 1997; Perez-Pedini et al., 2005; Petrucci, 2013; Reichold, 2010] or finds the greatest unit improvement in stormwater control per unit incremental cost [Shoemaker et al., 2009; Giacomoni, 2015; Yazdi and Salehi, 2014]. Examples include software packages developed by university researchers [Damodaram and Zechman, 2013; Zhang, 2009; Zhen et al., 2004] the Model for Urban Storm water Improvement Conceptualization (MUSIC) [eWater, 2009], and the U.S. Environmental Protection Agency's System for Urban Storm water Treatment and Integration (SUSTAIN) [Shoemaker et al., 2009].

Rainfall/runoff models can also be used to explore how a particular stormwater management strategy might impact receiving water quality. An example is the U.S. Environmental Protection Agency's study of the Illinois River (a multijurisdictional tributary of the Arkansas River in the states of Arkansas and Oklahoma) in which a catchment model based on Hydrologic Simulation Program Fortran (HSPF) was calibrated for nutrients and the output linked to a hydrodynamic and water quality model for Lake Tenkiller [Michael Baker Inc., 2013]. EPA used the resulting HSPF model to identify a set of stormwater management scenarios that met total maximum daily load targets for the lake.

Recent advances in uncertainty quantification can be exploited to improve the utility of stormwater modeling tools. An example is the DREAM and AMALGAM statistical toolboxes [Michael Baker Inc., 2013; Vrugt and Robinson, 2007; Vrugt et al., 2008, 2009, and 2011] that quantify model parameter and predictive uncertainty using Markov chain Monte Carlo simulation. DREAM has been widely used for model-data synthesis, hypothesis testing, and analysis of model malfunctioning in various time series applications. AMALGAM uses a multiple objective approach to produce a suite of equally acceptable (Pareto optimal) solutions from which stakeholders can select the option best suited to their collective needs [Laloy and Vrugt, 2012]. Importantly, both statistical packages take into account all forms of uncertainty, from model formulation error to data noise and bias, and thus reveal both what is known and what is not known about a system. Such information can assist managers and stakeholders by clarifying how much confidence can be placed in model predictions, and by identifying areas where targeted investment (e.g., in data collection or model development) would significantly improve model predictions.

Two unknowns that contribute to model uncertainty include: (1) long-term maintenance of LID technologies (will their hydraulic performance degrade over time?) and (2) changing climate (how will LID form and function change under future climate scenarios?). With the exception of rain tanks and wetlands, all of the LID technologies summarized in **Figure 2.4** include a step in which the captured stormwater is filtered through a granular media. As a consequence these systems are vulnerable to clogging (reduction in permeability over time) due to a variety of influent and filter-specific physical, chemical, and biological processes [Hatt, Deletic, and Fletcher, 2008; Kandra, McCarthy, and Deletic, 2015]. Because clogging reduces the volume of stormwater that can be harvested or infiltrated (and potentially effects pollutant removal [Rippy, 2015]), sustained hydraulic performance requires routine inspection, cleaning, and replacement of the filter media. In a recent comparison of biofilters in Melbourne (Victoria) and Los Angeles (California), Ambrose and Winfrey, 2015 noted that larger systems tend to be maintained by the government agency responsible for their construction. On the other hand, the responsibility for maintaining smaller distributed systems is often transferred to land owners with uncertain results. If hydraulic performance of these systems degrades over time, model simulations premised on as-built permeability will overestimate stormwater volumes that can be harvested and infiltrated postconstruction, and potentially pose a flood risk. Confounding this maintenance issue is the fact that stormwater management systems, in general, are sized based on the idea that historical climate is a good predictor of future climate [Booth and Jackson, 1997] an assumption that is violated under climate change [Brown, 2010]. Climate change also has implications for the "green" component of many LID systems [Ambrose and Winfrey, 2015; Levin and Mehring, 2015]. In the end, both challenges (uncertain maintenance and uncertain climate) are probably best addressed by using uncertainty quantification where possible (e.g., with DREAM and

AMALGAM, see above), factoring in redundancy, and designing smart (perhaps modular) LID systems that can readily adapt to a changing world [Brown, 2010; Hering et al., 2013].

Practical Constraints. One of the primary outcomes of the catchment water balance described in **Section 2.2** is that, for most areas of the world, restoring catchment water balance will require a focus on harvest-based LID technologies. A win-win example is using harvested rainwater and road runoff for in-home activities (e.g., for toilet flushing, laundry, and hot water supply, configuration A, **Figure 2.5**), thereby protecting streams and reducing potable water consumption [Grant et al., 2012]. However, in the U.S. a number of institutional barriers presently limit the indoor use of nonpotable water. These include [Roy et al., 2008; Kloss, 2008; Garrison et al., 2011]: (1) low uniform water prices that create an environment where consumers and developers have little incentive to invest in schemes to reduce potable water consumption, although this is changing in the southwestern U.S.; (2) plumbing codes that do not explicitly address rainwater use or inadvertently prohibit it by requiring that downspouts be connected to the storm sewer collection system; (3) a patchwork of local, state, and federal regulations with various and conflicting treatment standards; (4) prohibitions against indoor use of non-potable water in some locales that prevent local water utilities from sponsoring such schemes; (5) different interpretations of who owns stormwater runoff, with some states (e.g., Colorado) prohibiting residential capture and reuse of stormwater on the premise that all rainfall has been already allocated to downstream users; and (6) resistance from drinking water providers over concerns that wide-scale adoption of rainwater and stormwater harvesting may endanger public health, or lead to revenue loss.

Although public health concerns are often cited as a barrier to the adoption of harvested rainwater and stormwater for nonpotable uses in the U.S., the scientific evidence (and practical experience) generally do not support that contention. Public health concerns stem from the fact that both sources of water can harbor microorganisms that cause human disease [Ahmed, Gardner, and Toze, 2011; Grebel et al., 2013]. Human infection depends on multiple factors-including pathogen type and load, the mode of exposure, and susceptibility--that are best assessed through epidemiological studies and/or a Quantitative Microbial Risk Assessment (QMRA) framework that includes hazard identification, exposure assessment, dose−response assessment, and risk characterization [Lim and Jiang, 2013; Lim, Hamilton, and Jiang, 2015].

An epidemiological study of children in rural South Australia found that drinking roof harvested rainwater posed no more risk of gastroenteritis than drinking water from a reticulated supply [Hayworth et al., 2006]. However, concerns have been raised about the study's sensitivity (ability to detect an effect against background rates of infection) given that only 1016 people participated [Ahmed, Gardner, and Toze; 2011]. QMRA studies, which have been advocated as a more sensitive alternative to epidemiological investigations [Ahmed, Gardner, and Toze, 2011], indicate that minimally treated stormwater and rainwater may be acceptable for certain in-home uses, such as toilet flushing [Ahmed, Gardner, and Toze, 2011; Lim et al., 2015]. Rainwater also appears acceptable for garden irrigation and showering [Ahmed, Gardner, and Toze, 2011; Lim and Jiang, 2013]. However, the suitability of stormwater runoff (e.g., from parking lots or roads) for these purposes is less well understood [Lim et al., 2015]. Across the board, proper design and maintenance of collection systems as well as appropriate disinfection measures such as UV disinfection and chlorination are necessary to achieve public health targets for in-home use [Ahmed, Gardner, and Toze, 2011]. Currently, more than 2 million Australians use roof-harvested rainwater for potable or nonpotable supply [Ahmed, Gardner, and Toze, 2011]. The State of Victoria now requires new homes to have a rainwater tank for garden watering and in-home uses such as toilet flushing (although solar hot water heating can be installed as an alternative, suggesting that this instrument has a broad focus on "sustainability", rather than a specific focus on water management) [Low et al., 2015]. Australia's ongoing experiment with rainwater tanks (and more recently biofilters) should provide a wealth of data and experience with which health officials around the world can objectively evaluate the risks and benefits for in-home use.

Site-specific constraints may also impede infiltration schemes. For example, the City of Irvine (California, U.S.) discourages stormwater infiltration at certain locations due to low soil permeability, locally perched shallow groundwater, and concern that groundwater contaminants (such as selenium) may be mobilized into local streams or the deep aquifer used for potable supply [Daniel B. Stephens and Associates, Inc, 2013]. This concern is shared by the Orange County Water District (which manages the local groundwater basin that supplies drinking water to more than 2 million residents) and the Orange County Healthcare Agency (which manages public health for the county), and is enshrined in County regulatory statutes [Orange County Code]. Thus, for this particular region of Southern California, infiltration may be feasible in only a few locations and under fairly strict control; for example, at large centralized facilities strategically placed to facilitate runoff treatment and recharge to deep groundwater aquifers [Reilly, Horne, and Miller, 1999].

2.5. Evaluating LID Efficacy

Once LID technologies have been selected and implemented, ongoing monitoring programs are needed to ensure goals are being met. A number of recent reviews summarize field data and modeling approaches for evaluating the effects of land-use and land-cover change (in general) and LID interventions (in particular) on catchment-scale hydrologic budgets and streamflow [Ahiablame, Engel, and Chaubey, 2012; O'Driscoll et al., 2010; DeFries and Eshleman, 2004; Bosch and Hewlett, 1982; Peel, 2009; Sahin and Hall, 1996, Zégre et al., 2010; Hall et al., 2014; Machiwal and Jha, 2009]. Generally, the available techniques can be classified into three types: (1) modeling approaches; (2) timeseries analyses; and (3) paired catchments. Modeling approaches simulate the influence of land-cover change on the rainfall-runoff relationship, potentially revealing a causal link between the former and latter while controlling for climate variability. This approach is particularly useful when the goal is to evaluate "what if" scenarios (e.g., evaluating how the storm hydrograph might change in response to various LID interventions, see discussion of modeling tools in **Section 2.4**) [Miguel et al., 2013; Walsh, Pomeroy, and Burian, 2014], and in cases where long-term rainfall-runoff records are not available. Alternatively, when the goal is a post de facto evaluation of an LID intervention, time series analysis can be conducted on rainfall and hydrograph data, provided quality data are available both before and after the intervention. A variety of time series tools are available including graphical methods [Wang et al., 2007; Simmons and Reynolds, 1982; Zhang et al., 2014], autoregressive models [Farahmand and Fleming, 2007; Yang and Bowling, 2014], linear and curvilinear regression models [Zhang et al., 2006; Buttle, 1994; Wang et al., 2006], multiple linear regression models [Jiongxin, 2005; Little et al., 2009; Xu, 2013], trend identification tools [Li, Feng, and Wei, 2013; Hejazi and Moglen, 2008; Peterson, Nieber, and Kanivetsky, 2011; Shao, Li, and Xu, 2010; López-Moreno et al., 2006], and change point analysis [Perreault et al., 2000]. Interpretation of time series data can be complicated by climate variability over the time of observation [Farahmand and Fleming, 2007; Beighley and Moglen, 2002; Lørup, Refsgaard, and Mazvimavi, 1998; Doyle and Barros, 2011; Getnet, Hengsdijk, and Ittersum, 2014].

The gold standard for assessing the hydrologic impact of landuse change is paired (or triplicated) catchment studies, in which the catchment of interest is paired with a control catchment (and a reference catchment, in the case of a triplicate design) of similar climate and physiography [Brown et al., 2005; Watson et al., 2001]. There is a long history of using paired catchment studies to assess the impact of vegetation change on catchment hydrology [Zhang, Dawes, and Walker, 2001; Zhang, Dawes, and Walker, 1999; Hibbert, 1967], but the technique has been applied only recently to assess the impacts of LID interventions on stream health. Such studies collectively demonstrate that adopting LID technologies for stormwater management (over conventional centralized retention and detention basins) markedly improves the hydraulic performance of streams, as measured by higher baseflow, lower peak discharge and runoff volumes during moderate storms, increased lag times, and retention of smaller more frequent precipitation events [Loperfido, 2014; Hood, Clausen, and Warner, 2007; Selbig and Bannerman, 2008; Bedan and Clausen, 2009; Shuster and Rhea, 2013]. These field results are generally supported by modeling studies, although centralized stormwater control measures may perform better than distributed LID systems for controlling peak discharge from large storms [Williams and Wise, 2006; Damodaram et al., 2010], a problem that could presumably be overcome by proper LID technology placement and design. Not surprisingly, none of the urban stormwater management approaches perform as well as non-urbanized (reference) catchments [Loperfido et al., 2014]. Thus, it can be argued that the best approach for protecting stream health is to place strict limits on urban development within a catchment. Short of this goal, however, distributed LID technologies should be used for managing stormwater runoff [Loperfido et al., 2014; Booth and Jackson, 1997].

The next frontier is paired catchment studies that evaluate how LID interventions simultaneously influence the hydrologic, water quality, and ecological response of streams. One example is Little Stringybark Creek in Melbourne (Australia). In collaboration with a local water utility, researchers developed a financial incentive scheme to encourage homeowners to install rainwater tanks and unlined biofilters, and worked with the local municipality to install larger neighborhood-scale infiltration and harvesting systems [Walsh, Fletcher, and Ladson, 2005; Fletcher et al., 2011; Fletcher, 2015; Bos and Brown, 2015]. To determine if these retrofits are impacting flow, water quality, and ecology in Little Stringybark Creek, researchers are employing a "before/after control reference impact" (BACRI) study, consisting of the study catchment (where LID technologies are implemented), two urban control catchments (with similar levels of effective imperviousness, but where LID technologies are not implemented), and two non-urbanized reference catchments representing natural conditions [Walsh, Fletcher, and Ladson, 2005]. Although such experiments are ambitious and challenging [Walsh and Fletcher, 2015], they are a rigorous field test for how well LID technologies insulate streams from catchment urbanization. The project has already generated important lessons in relation to community engagement [Brown et al., 2015; Bos and Brown, 2015], institutional aspects

[Burns, Wallis, and Matic, 2015], and the performance of LID technologies in flood reduction [Burns et al., 2015]. There are some early signs that the retrofit may be improving water quality in the creek [Walsh et al., 2015].

Regardless of which approach is adopted (modeling, time series, or paired catchment), appropriate statistical methods should be used to link LID intervention to changes in stream performance, after taking into account instrument accuracy and precision [Changnon and Demissie, 1996]. A critical consideration is the predicted change of the response variable (e.g., baseflow or peak discharge) relative to extraneous sources of variation and noise. For example, if modeling studies suggest that baseflow will increase by 1 to 2 [L s^{-1}], then flow measurements must have precision less than half this value [Fletcher and Deletic, 2008].

2.6. Context- and Path-Dependence of the Urban Stream Syndrome

In this final section, we describe social, environmental, and ecological factors that may make the urban stream syndrome context and path dependent. By this we mean that the hydrologic, water quality, and ecological state of a stream depends not only on the extent of LID intervention (as measured, for example, by the volume of stormwater harvested and infiltrated) but also on the environmental context and historical path by which the catchment arrived at its current state.

Cognitive Lock-in. Cognitive lock-in is one form of path dependence that can arise from positive feedback between the societal perception, management, and the physical and biological condition of a stream; it tends to vary within communities depending on their state(s) of economic development [Walsh et al., 2005; Grimm et al., 2000; Ferguson et al., 2013]. The term "cognitive lock-in" originates from the field of social psychology, where it has been applied to understanding consumer habits and choices with respect to a product or service [Johnson, Bellman, and Lohse, 2003; Murray and Haubl, 2007]. The idea is that repeated consumption or use of a product results in a (cognitive) switching cost that increases the probability that a consumer will continue to choose that product or service over alternatives. As applied here, cognitive lock-in can affect stream health in positive or negative ways (**Figure 2.6**). If a community perceives their stream is a threat (e.g., due to the damage it might cause by flooding), local managers may be pressured to enact policies that degrade a stream's aesthetic and ecological value (e.g., through installation of formal drainage with high effective imperviousness, and stream burial), unintentionally reinforcing negative perceptions of the stream as a drain (red loop in the figure). Conversely, if a stream is perceived as a valuable asset, local managers may respond by enacting policies that protect the stream from urbanization, reinforcing positive perceptions of the stream as an asset through increased property value and the provision of green space and other ecosystem services (green loop). Examples of cognitive lock-in abound in stormwater management [Walsh et al., 2005; Petrucci, 2013], and its manifestations are evident in urban centers as diverse as Los Angeles, Paris, Moscow, and Melbourne [Petrucci, 2013; Pahl-Wostl, Gupta, and Petry, 2008; Deverell and Hise, 2005; Kibel, and Rivertown, 2007]. A common pattern is that, as cities industrialize, prevailing public values call for harnessing and restraint of urban rivers for flood control and property development (favoring the red loop), while postindustrial development leads to demand for restoration of recreational, aesthetic, cultural heritage, and ecological values (favoring the green loop).

Urbanization Thresholds. Path dependence can also play a role in observed urbanization thresholds. Urbanization thresholds are defined as a critical level of urban intensity (e.g., as measured by effective imperviousness, road density, or the metropolitan area national urban intensity index, MA-NUII183) at which symptoms of the urban stream syndrome begin to manifest if the catchment is urbanizing, or disappear if an already urbanized catchment is being retrofitted with LID technologies. Most evidence for the existence of urbanization thresholds comes from comparing metrics of stream health (hydrology, water quality, and/or ecology) across two or more nearby catchments with different levels of imperviousness (i.e., paired catchment studies, see **Section 2.5**). For example, Walsh et al., 2009 found that stream health (as measured by hydrologic indicators, water quality, and biodiversity) was good in two catchments with low effective imperviousness (<1%), but poor in two nearby catchments with elevated effective imperviousness (5 and 22%).

Effective imperviousness thresholds of up to 10% have been associated with significant degradation in one or more stream metrics [Booth and Jackson, 1997]. As noted by Hopkins et al., 2015 this particular threshold may reflect the tendency of urban communities to transition from mostly informal drainages below 10% to mostly formal drainages above 10% imperviousness (although their measure of imperviousness is a satellite product that may not equate to effective imperviousness). Collectively, such studies suggest that preventing the urban stream syndrome requires keeping effective imperviousness well below 10% and perhaps below 1%, although there is considerable study-to–study variability depending on climate, physiography, geology, land-use, and stream history [Arnold and Gibbons, 1996; Jacobson, 2011; King et al., 2011; Roy et al., 2014; Utz, Eshleman, and Hilderbrand, 2011].

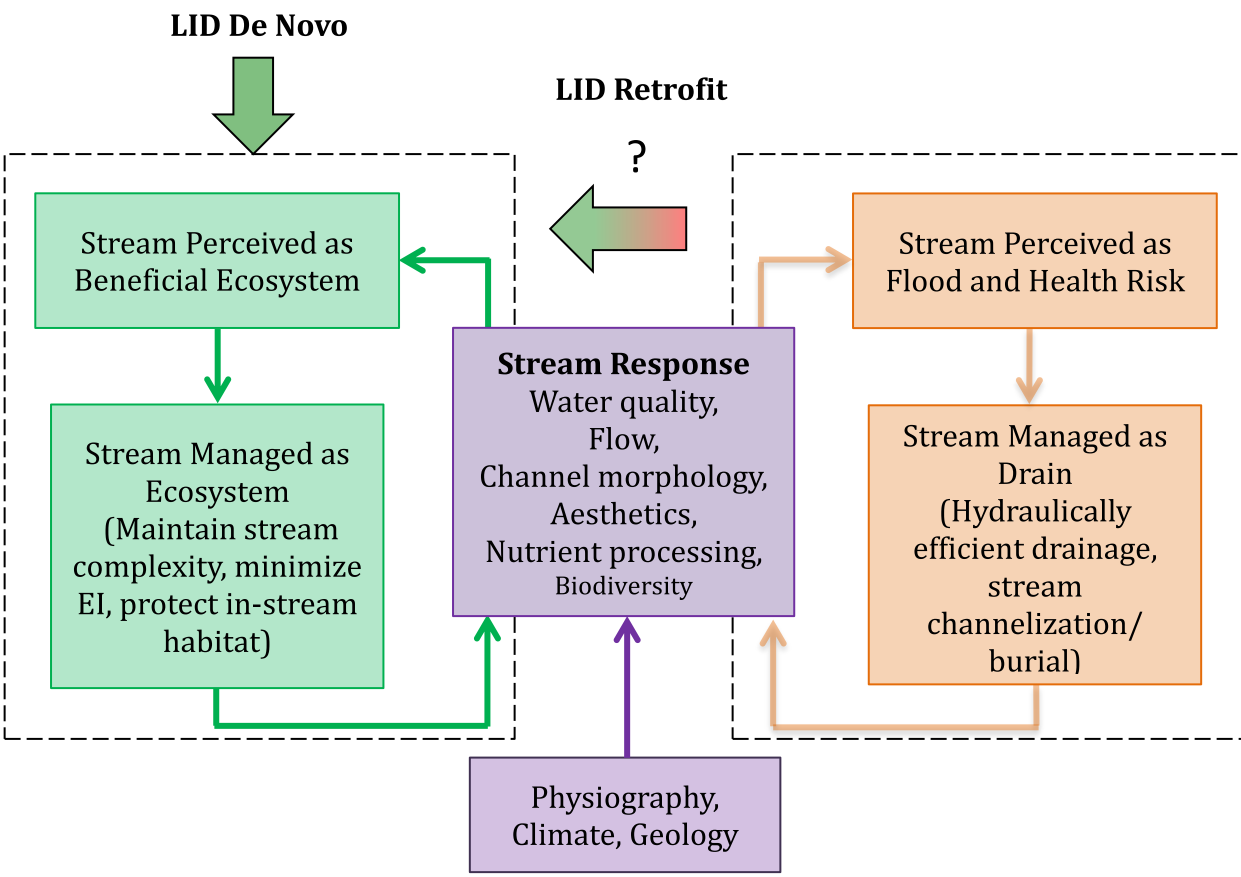

Figure 2.6. Social–ecological feedback loops can lead to "cognitive lock-in" in which streams are maintained in either a degraded state (because they are perceived primarily as storm drains, right loop) or healthy state (because they are perceived as ecologically valuable assets, left loop). The left loop may be more likely to occur if LID technologies are incorporated into an urban space as a city develops ("LID De Novo"). Retrofitting an already developed area with LID technologies may or may not trigger a transition from the right loop to the left loop ("LID Retrofit") (see main text). Adapted from Figure 3 in Walsh et al., 2005 and Figure 3 in Grimm et al., 2000.

In some streams urbanization thresholds may not be observed [1997]. As part of the U.S. Geological Survey's National Water Quality Assessment (NAWQA) Program, Cuffney et al., 2010 evaluated the impact of urbanization on in-stream invertebrate assemblages (a measure of stream ecosystem structure and function) across urban-to-rural gradients in nine metropolitan areas of the U.S. They found that invertebrate assemblages were strongly related to urban intensity (MA-NUII), but only when the urban development occurred within forests or grassland. A much weaker (or nonexistent) correlation was observed in areas where agriculture or grazing predominated, presumably because those streams were already degraded. Importantly, in forests and grassland there was no urbanization threshold below which ecosystem assemblages were resistant to urbanization. Even small impervious fractions were associated with "significant assemblage degradation and were not protective" [Cuffney et al., 2010].

That imperviousness thresholds are not always present is not surprising, given that effective imperviousness is only one of many stressors that can negatively impact urban stream health. For example, salinization has an enormous ecological toll on streams worldwide [Williams, 2001]. Although road runoff clearly contributes to the problem (particularly in northern climates where salt is used for deicing roads [Kaushal et al., 2005; Kelly et al., 2008]), there are other sources of salt that would not be eliminated by reducing effective imperviousness alone (e.g., irrigation return flows). Other examples of urban stream stressors include loss of riparian habitat and tree canopy, impoundments that alter flow regimes and elevate temperatures, point source discharges of nutrients, heavy metals, and contaminants of emerging concern, to name a few [Walsh et al., 2005; Meyer, Paul, and Taulbee, 2005; Wenger et al., 2009]. Thus, reducing effective imperviousness may be a necessary, but not sufficient, condition for curing the urban stream syndrome in some catchments.

For all of the reasons stated above, it is difficult to predict the imperviousness threshold (if one exists) at which stream conditions will markedly improve as an urbanized catchment undergoes an LID retrofit. Shuster and Rhea, 2013 reported a small but significant improvement in the hydrological condition of a small suburban creek (Shepherd Creek, Cincinnati, Ohio) after installing 165 rain barrels and 81 unlined biofilters in the 1.8 km^2 catchment (reducing effective imperviousness by approximately 1%, mostly from roofs). However, a follow-up study of the same field site reported little change in water quality and ecology of the stream compared to a control stream in the nearby catchment [Roy et al., 2014]. The authors suggest a number of possible explanations for the lack of a water quality and ecological response, most notably that, despite the relatively large investment in LID retrofits, effective imperviousness in the catchment was not reduced to levels where improvements in stream health would be expected (after retrofits, the effective imperviousness in the Shepherd Creek catchment was still above 10%). The authors concluded that, "additional research is needed to define the minimum effect threshold and restoration trajectory for retrofitting catchments to improve the health of stream ecosystems" [Roy et al., 2014]. Ongoing retrofits in the Little Stringy Bark Creek project (see **Section 2.5**), which will reduce effective imperviousness below 1%, may eventually shed light on this important issue.

Although it is fair to say that LID technologies are not a cure for all symptoms of the urban stream syndrome in all catchments, they do address critical hydrologic and geomorphic symptoms of the disease while providing myriad co-benefits and subsidiary ecosystem services, including water quality improvement, flood protection, green space, recreation and aesthetic value, wildlife habitat and corridors, carbon sequestration, pollination services, urban heat island cooling, and a much needed supply of non-potable ("fit-for-purpose") water in drought prone areas such as Southeast Australia and Southwest U.S. [Walsh et al., 2005; Low et al., 2015; Grant et al., 2012; Wong, 2006; Aghakouchak et al., 2014; Endreny, 2008; Coutts et al., 2013].

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Chapter 3

Ecosystems on the Edge: In-Stream Treatment and Watershed Nitrate Management

Abstract

Streams provide an array of ecosystem services, including nitrate removal by biotic assimilation and denitrification. When measured across 72 U.S. streams and over timescales of hours-to-days, Mulholland et al., 2008 reported that nitrate removal efficiency declines non-linearly with increasing nitrate concentration. Here we explore the implications of this result for nitrate management in a rapidly urbanizing watershed (Jacksons Creek) in southeastern Australia. Remarkably, we find that Mulholland's relationship applies to Jacksons Creek and over timescales (months-to- years) appropriate for the monitoring and regulation of nitrate pollution at the watershed scale. Taking Mulholland et al.'s result into account, a stream network model predicts that as nitrate loading from a sewage treatment plant increases (or decreases), Jacksons Creek responds by reducing (or increasing) in-stream nitrate removal. Thus, the non- linear nature of in-

stream treatment may reinforce socio-ecological feedback loops that drive urban streams into healthy or degraded states.

3.1. Introduction

Over the past century anthropogenic activities more than doubled the loading of nitrate to terrestrial landscapes [Lassaletta et al., 2009; Kaushal et al., 2008]. Human inputs of bioavailable nitrogen to freshwater ecosystems now exceed the planetary boundary past which earth's critical systems could become inhospitable to humans [Steffen et al., 2015]. According to the U.S. National Academy of Engineering, solving this problem is a grand challenge facing engineers in the 21st century [National Academy of Engineering (http://www.engineeringchallenges.org/challenges/nitrogen.aspx)].

The ecological consequences of anthropogenic nitrate loading are mitigated, to some degree, by in-stream treatment; i.e., the ability of streams to remove bioavailable nitrate by conversion to dinitrogen gas (denitrification) and plant or microbial uptake (assimilation) [Stream Solute Workshop, 1990]. In a seminal study of in-stream treatment, Mulholland et al., 2008 reported that nitrate removal efficiency declines non-linearly with increasing stream nitrate concentration. A number of potential mechanisms could account for this observation, including the transition of denitrification kinetics from first- order (at low nitrate concentration) to zero-order (at high nitrite concentration) [Azizian et al., 2015]. In this chapter we set out to answer the following question: does the non-linear nature of in-stream treatment set the stage for "ecological surprises" [Gordon et al., 2008] in which small changes in the way nitrate point sources are managed disproportionately impact stream nitrate concentration and watershed health?

3.2. Stream Network Model

To answer the above question we prepared a 4th order stream network model (resolution 30 m) for a rapidly urbanizing region of the Jacksons Creek and Riddells Creek watersheds in Southeast Australia (**Figure 3.1**). The stream network model, which has 338 reaches (median length 500 m), was used to simulate the steady-state loading, transport, and biotic assimilation and denitrification of stream nitrate (see **Figures B.1** and **B.2** for details in **Appendix B**).

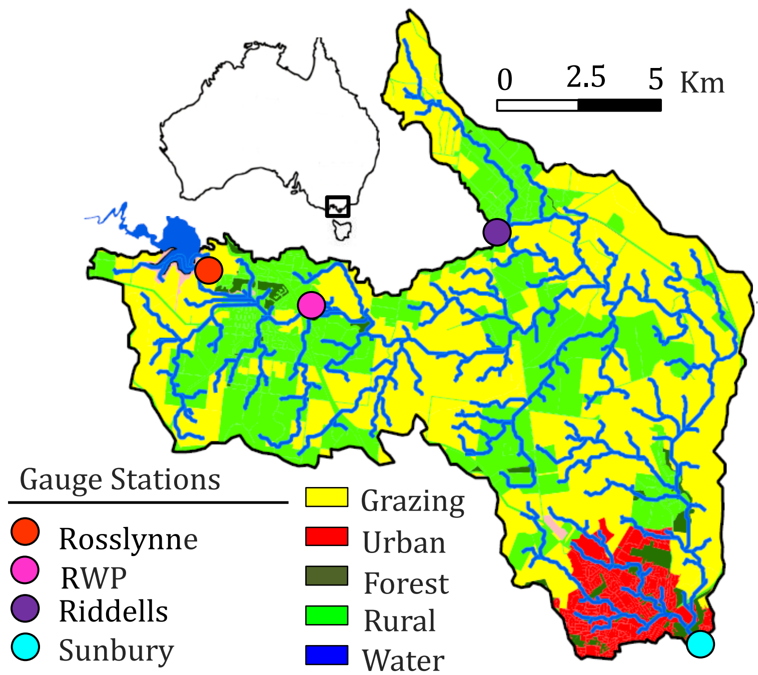

Figure 3.1. The 160 km^2 study area includes portions of two watersheds (Jacksons Creek (139 km^2) and Riddells Creek (21 km^2)) located northwest of Melbourne, a rapidly growing city of over 4 million people in Southeast Australia. Land use is mostly rural and grazing land, but over the next 50 years will likely transform as Melbourne's regional transportation corridors expand into the watershed, spurring additional residential and commercial development and transforming the region into a bedroom community for metropolitan Melbourne [Townsend et al., 2015; Feldman et al., 2015]. The current ecological condition of streams in the study area is classified as poor by the Victorian Environmental Protection Authority, in part because nitrate concentrations frequently exceed Australian and New Zealand guidelines at multiple monitoring stations along

Jacksons and Riddells Creek [Townsend et al., 2015]. Furthermore, nitrate exported from Jacksons Creek may impact Port Phillip Bay, an ecologically important estuary at risk for nitrate-limited algal blooms and eutrophication [Department of Natural Resources and Environment, 2011].

3.2.1. Gisborne Recycled Water Plant (RWP)

Anthropogenic sources of nitrate in the study area include point sources (primarily recycled water from the Gisborne Recycled Water Plant, hereafter referred to as the "RWP") and non-point sources (primarily contaminated groundwater and runoff from grazing and urban areas). This letter focuses on nitrate loading from the RWP, which is located approximately 4 km downstream of the Rosslynne Reservoir on Jacksons Creek (**Figure 3.1**). The RWP is of interest for several reasons. First, population growth in the catchment is expected to increase the volume of recycled water produced by the RWP more than 20% over the next five years alone (from ~1.4 [ML day^{-1}] in 2016 to ~1.7 [ML day^{-1}] in 2021) (personal communication, Western Water). Second, the RWP presently produces more recycled water than there is demand (e.g., for agricultural and open space irrigation) and therefore the excess volume must be released to Jacksons Creek [Townsend et al., 2015; Feldman et al., 2015]. Third, several reaches of Jacksons Creek temporarily ceased flowing during the Millennium Drought (which lasted for 12 years, from 1997 to 2009) [Low et al., 2015; Grant et al., 2013; AghaKouchak et al., 2014] causing significant stress to sentinel species (e.g., platypus). It has been argued that releasing more recycled water to the creek would ensure environmental flows during periods of water scarcity [Townsend et al., 2015]. Fourth, although the RWP was upgraded to tertiary treatment in 2008 (which reduced nitrate concentrations in the effluent more than tenfold) there is continuing regulatory pressure to decrease nitrate loading [GHD Corporation, (2015); Department of

Natural Resources and Environment, 2002]. Finally, these issues, while specific to Jacksons Creek, are symptomatic of many watersheds around the world facing agricultural-to-urban land-use transformations and water quality and quantity stressors [Feldman et al., 2015].

3.2.2. Water Budget for Jacksons Creek

Water flows into the stream network model at two upstream boundaries (labeled “Rosslynne”, “Riddells” in **Figure 3.1**) and the RWP. Water flows out of the stream network in the City of Sunbury (labeled “Sunbury” in the figure). Over the 25 year period for which overlapping data are available (1990 through 2014), annual discharge at these stations exhibits substantial inter-annual variability (**Figure B.3.B**). Flows were generally lower during the Millennium Drought (1997 to 2009) than before or after. In any given year, flow generally decreased in order: Sunbury > Riddells > Rosslynne > RWP. Documented extractions (e.g., agricultural diversions) are orders of magnitude below inflows from Rosslynne, Riddells, or the RWP (**Figure B.3**).

Annual yield coefficients (Y, volume of water produced by the watershed per unit area per time, units of [m s^{-1}]) were calculated by performing a water volume balance over the stream network (**Figure 3.2**, see **Appendix B** for details). For most of the Millennium Drought the yield ($Y < 1 \times 10^{-9}$ [m s^{-1}]) was at least five times lower than before or after ($Y = 5 \times 10^{-9}$ to 8×10^{-9} [m s^{-1}]). Indeed, during one of the worst years of the Millennium Drought (2003) the yield was negative ($Y = -2 \times 10^{-10}$ [m s^{-1}]), implying that more water entered the stream network than flowed out at Sunbury, perhaps due to enhanced evapotranspiration, undocumented water extractions, and/or surface water lost to groundwater.

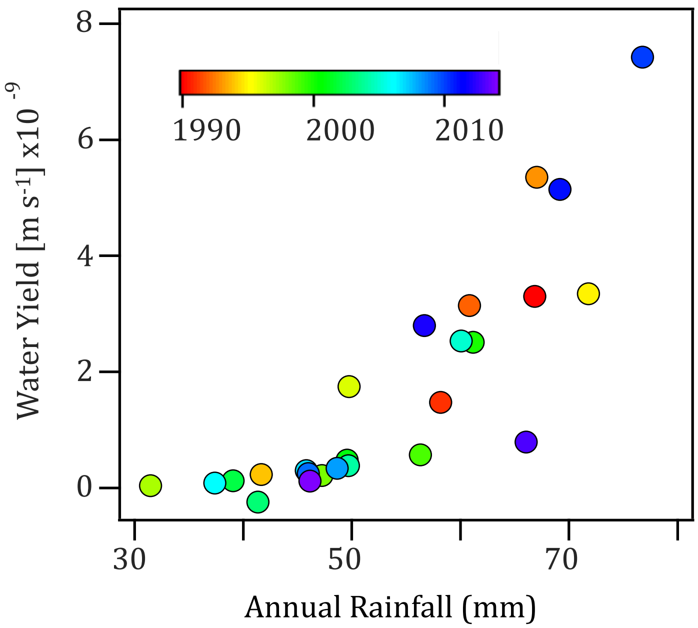

Figure 3.2. Annual yield coefficient calculated from balancing stream inflows and outflows over the study area, plotted as a function of annual rainfall.

3.2.3. Nitrate Budget for Jacksons Creek

Closing the nitrate budget for Jacksons Creek requires: (1) nitrate concentrations at all stations where water flows into (Rosslynne, Riddells, the RWP) or out of (Sunbury) the stream network; (2) the flux of nitrate from all non-point sources in the watershed ($F_{nitrate}$, mass of non-point source nitrate entering the creek per watershed area per time, units of [kg m^{-2} s^{-1}]); and (3) in-stream treatment as quantified by the nitrate uptake velocity (ϑ_f, the flux of nitrate into the streambed normalized by in-stream nitrate concentration, units [m s^{-1}]). These are discussed in turn.

Nitrate Inflows and Outflows. As is the case for many watersheds around the world, historical measurements of nitrate in Jacksons Creek are discontinuous, non-overlapping, and have variable sampling frequency. Over the 35 years for which nitrate data are available (1978 to 2012), the RWP and Rosslynne, Riddells, and Sunbury stations were sampled only eight times on the same day, and there are multiple years when one or more were not sampled. Nitrate measurements were most common at Rosslynne (N = 207, 1978-2012) followed by Sunbury (N = 205, 1990-2009), the RWP (N = 83, 2003-2014), and Riddells (N = 46, 1978-1990) (**Figure B.6**). In the modeling studies described below we focused on a non-drought period (1990 to 1996). Over this six-year period of time, monthly nitrate data are available for Sunbury and a Multiple Linear Regression model was used to fill data gaps at Rosslynne and Riddells (see **Appendix B** for details). The RWP effluent concentration was set equal to the median value of all measurements prior to the tertiary upgrade in 2008 (17.7 [mg L^{-1}], 95% CI of 0.28 to 30 [mg L^{-1}], N = 52).

Non-Point Source Nitrate Flux and In-stream Treatment. In general, non-point source nitrate flux ($F_{nitrate}$) can originate from nitrate-contaminated groundwater in gaining streams [Spalding and Exner, 1993; Wakida and Lerner, 2005], agricultural return flows [Goolsby, 1992; McIsaac et al., 2001], and urban runoff [Ackerman and Schiff, 2003; Howarth et al., 2002]. Although nitrate flux is often estimated from land-use patterns [Tong and Chen, 2002; Howarth et al., 1996], such an approach is complicated at our study site by the uncertain (and likely climate dependent) contributions of different nitrate sources (e.g., leachate from an old landfill site, surface runoff, groundwater) [Townsend et al., 2015]. Additional uncertainties apply to the in-stream treatment of nitrate, as quantified by the nitrate uptake velocity (ϑ_f). While we could simply adopt Mulholland's non-linear correlation for ϑ_f, it is not clear that this correlation (based on stable isotope nitrate seeding studies conducted in 72 U.S. streams over timescales of hours-to-days) applies to southeastern Australia over the much longer (months-to-years) timescales pertinent to watershed management.

Given the uncertainties raised above, we conducted a model optimization study with the goal of obtaining Jackson Creek specific values for $F_{nitrate}$ and ϑ_f. This required invoking two simplifications: (1) ϑ_f is a fixed constant (i.e., does not vary by reach or with time); and (2) $F_{nitrate}$ does not vary by reach, but does vary with time in proportion to the yield coefficient:

$$F_{nitrate}(t) = C_{NPS-NO_3^-} \times Y(t) \tag{1}$$

Equation (1) implies that all non-point source water flowing into Jacksons Creek has a fixed nitrate concentration, $C_{NPS-NO_3^-}$[mg-N L^{-1}]. With these two assumptions our stream network model has only two unknown quantities: ϑ_f and $C_{NPS-NO_3^-}$. Because these two unknowns can take on values spanning many orders of magnitude [Mulholland et al., 2008], they were expressed in the stream network model as power-law functions with unknown exponents d and e: $C_{NPS-NO_3^-} = 10^d$ [mg-N L^{-1}] and $\vartheta_f = 10^e$ [m s^{-1}]. Estimates for d and e were obtained by optimizing agreement between monthly model predicted and measured stream nitrate concentration at the Sunbury station from 1990 to 1996 (N=72). Adopting Bayesian inference (using Markov chain Monte Carlo simulation implemented in the statistical package DREAM [Vrugt et al., 2009]) with uniform priors ($d \in [-4,2]$ and $e \in [-8,-3]$) and a Gaussian maximum likelihood function, we obtained the following optimal values for the two unknowns (**Figure B.4**): $\vartheta_f = 5.62 \times 10^{-6}$ (4.36×10^{-6}, 8.12×10^{-6}) [m s^{-1}] and $C_{NPS-NO_3^-} = 0.50$ (0.144, 0.809) [mg-N L^{-1}] (parentheticals represent 95% credible intervals). The optimal value of $C_{NPS-NO_3^-}$ (and by inference $F_{nitrate}$, see **equation (1)**) is consistent with grazing (<1 [mg-N L^{-1}]) [Huertos et al., 2001], the dominant land-use for the period over which the model was optimized (1990 to 1996). To compare the optimal ϑ_f to Mulholland's correlation, we first calculated from our model stream nitrate concentrations ($C_{S-NO_3^-}$) for all 338 reaches at a monthly time step from 1990 to 1996 (N = 24336), using the optimal values of ϑ_f and $C_{NPS-NO_3^-}$ described above. Remarkably, our Jackson Creek specific values for ϑ_f and $C_{S-NO_3^-}$ closely align with Mulholland's nitrate uptake velocity measurements and power-law correlation (compare red triangle, circles, and solid black line, **Figure 3.3**). These results suggest that Mulholland's correlation applies to the Jacksons Creek watershed, and over timescales (months-to-years) appropriate for watershed management.

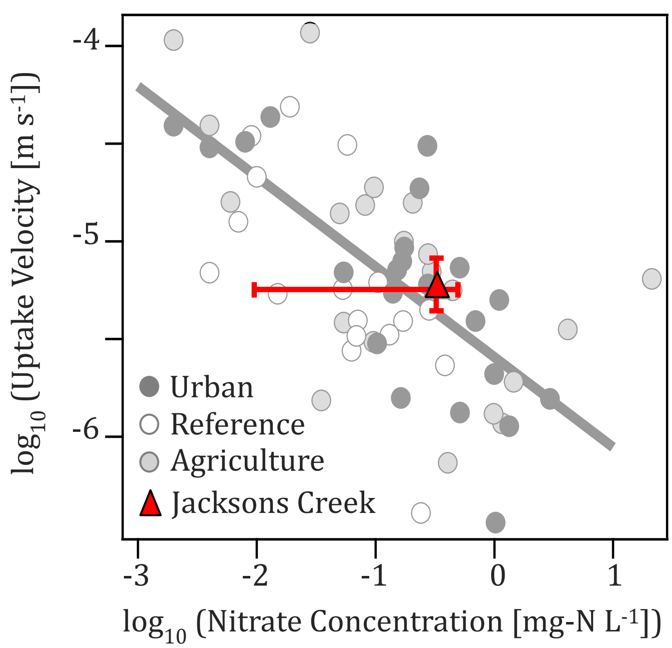

Figure 3.3 Correlation between nitrate uptake velocity and in-stream nitrate concentration reported by Mulholland et al. [Mulholland et al., 2008], (black line represents a non-linear (power-law) fit to uptake velocities measured in 72 streams across the U.S., indicated by

grey circles) compared with optimal values of the nitrate uptake velocity and in-stream nitrate concentrations estimated by fitting our stream network model to flow and nitrate monitoring data in the Jacksons Creek and Riddells Creek watersheds (red triangle, vertical bars represent the 95% credibility limits for nitrate uptake velocity, horizontal bars represent the 95% confidence limits for all in-stream nitrate concentrations predicted by the optimal stream network model, see main text).

3.3. RWP Effluent Management and Ecosystem Function

To determine how nitrate loading from the RWP might affect nitrate removal by in-stream treatment we: (1) coupled our stream network model to Mulholland's correlation for nitrate uptake velocity (see **Appendix B** for details), and (2) simulated the fraction of total nitrate load at Sunbury removed by in-stream treatment, adopting optimal values for ϑ_f and $C_{NPS-NO_3^-}$ (described above) and median values (calculated over the time span 1990 to 1996) of all other watershed parameters (**Figure 3.4**). For these calculations we defined the percent nitrate load removed by in-stream treatment as follows: $100 \times (L_{\vartheta_f=0} - L_{\vartheta_f\neq 0})/L_{\vartheta_f=0}$, where $L_{\vartheta_f=0}$ and $L_{\vartheta_f\neq 0}$ represent the simulated nitrate load at Sunbury when in-stream nitrate treatment is turned off and turned on, respectively. The stream network model was forced with a broad range of RWP operating conditions, including effluent discharge ranging from 10^{-2} to 10^{2} [ML day^{-1}] and effluent nitrate concentration ranging from 10^{-2} to 10^{2} [mg-N L^{-1}]. The upper range of effluent nitrate concentration might occur during a treatment plant failure or a sewage overflow event.

Our model predicts that in-stream treatment of nitrate improved (from 70% to 90%) after the RWP's tertiary treatment upgrade in 2008 (two red circles, **Figure 3.4**). Because in-stream treatment is an ecosystem service that can be monetized [Beseres et al., 2013; Randall et al., 2013; Chen et al., 2005], the additional nitrate removed by the stream may have offset capital and operating costs associated with the plant upgrade. The model also diagnoses scenarios that could lead to catastrophic loss of in-stream treatment, or ecocide (defined here as < 10% in-stream nitrate treatment). Our model predicts that ecocide occurs for all combinations of effluent discharge (Q_{eff}, [ML day^{-1}]) and nitrate concentration ($C_{eff-NO_3^-}$, [mg-N L^{-1}]) that satisfy the inequality: $Q_{eff} \times C_{eff-NO_3^-} > 1000$. While the current RWP operating condition is "on the edge" of a steep reduction in in-stream treatment (see red circle with black center, **Figure 3.4**) it is far from ecocide ($Q_{eff} \times C_{eff-NO_3^-} = 1.2$).

Four RWP management scenarios are of particular interest given the challenges currently facing Jacksons Creek (arrows in **Figure 3.4**). Increasing the volume of RWP effluent discharged to Jacksons Creek tenfold to 10 [ML day^{-1}] (e.g., as a result of long-term population growth in the sewershed, coupled with a limited market for recycled water and/or interest in protecting the habitat of sentinel species in the face of water scarcity), reduces in-stream treatment by 26% if RWP effluent nitrate concentration remains at the current level (1.5 [mg-N L^{-1}]) (scenario (a)) or 66% if RWP effluent concentration increases to the pre-upgrade level (17.7 [mg-N L^{-1}]) (scenario (b)). On the other hand, decreasing the volume of RWP effluent discharged to Jacksons Creek ten-fold to 0.1 [ML day^{-1}] (e.g., through more aggressive implementation of recycled water reuse programs), increases in-stream treatment by 7% if RWP effluent nitrate concentrations remains at the current level (scenario (c)) or 6% if RWP effluent concentration increases to the pre-upgrade level (scenario (d)).

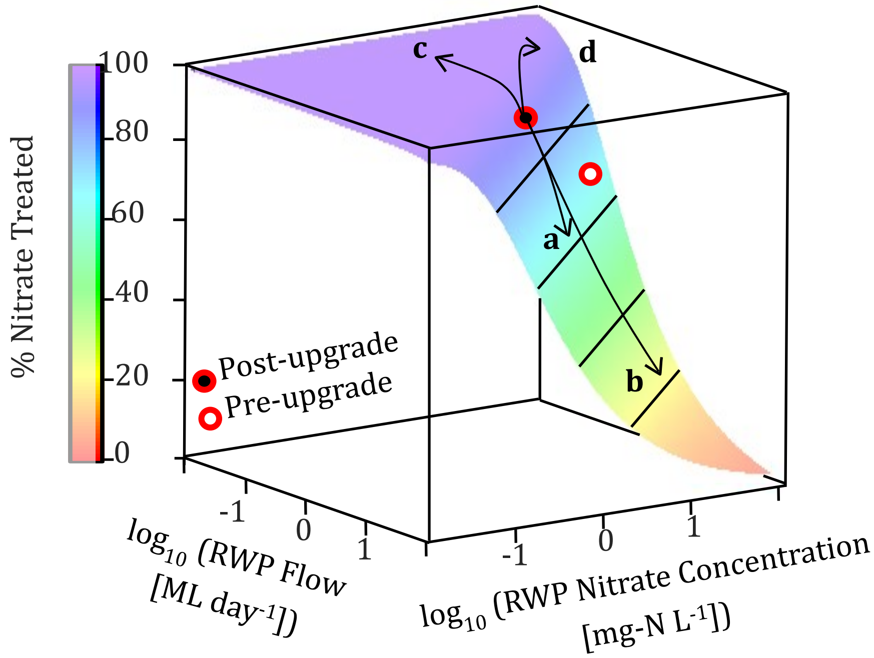

Figure 3.4. Percent of total nitrate load at Sunbury removed by in-stream treatment (vertical axis, contours, and color), plotted as a function of a hypothetical range of nitrate concentration (bottom right axis) and volumetric discharge (bottom left axis) from the Gisborne Recycled Water Plant (RWP). The red circles represent RWP operating conditions prior (red circle with white center) and after (red circle with black center) the RWP was upgraded from secondary to tertiary treatment standards in 2008. Arrows show trajectories (on the curved surface) of four management scenarios: (a) increasing the effluent discharge from Q_{eff} =1 to 10 [ML day^{-1}] while maintaining tertiary treatment standards; (b) increasing the effluent discharge from Q_{eff}=1 to 10 [ML day^{-1}] while relaxing standards to secondary-treatment; (c) decreasing the effluent discharge from Q_{eff}=1 to 0.1 [ML day^{-1}] while maintaining tertiary treatment standards; and (d) decreasing the effluent discharge from Q_{eff} =1 to 0.1 [ML day^{-1}] while relaxing standards to secondary-treatment. For these simulations we assumed effluent concentrations in secondary-treated and tertiary-treated effluent of 17.7 [mg-N L^{-1}] and 1.5 [mg-N L^{-1}], respectively (equal to the median values of effluent data reported by Western Water before and after the RWP upgrade, respectively).

These results reveal that in-stream treatment acts to amplify the positive or negative effects of point source (and presumably non-point source) management decisions. When nitrate loading to a stream is increased, the stream responds by reducing in-stream treatment, and disproportionately more nitrate is exported from the watershed to sensitive receiving waters, in this case the Maribyrnong River and Port Phillip Bay [Feldman et al., 2015]. Conversely, if nitrate loading is reduced, the stream responds by increasing in-stream treatment, and disproportionately less nitrate is exported.

Others have noted that positive socio-ecological feedback loops can "trap" urban streams in a state of low or high ecological integrity [Askarizadeh et al., 2015; Walsh et al., 2005; Grimm et al., 2000]. This can occur, for example, when a community's perception of a stream (e.g., as a valuable resource or a conduit for waste) influences management decisions that, in turn, determine ecological outcomes and reinforce prevailing perceptions. The non-linear nature of in-stream treatment may help drive urban streams into one of the two feedback loops, by decreasing the stability of intermediate states.

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Chapter 4

Conclusions and Future Research Step

In this thesis I have contributed to the scholarly discussion of natural treatment systems, including low impact development (LID) technologies designed to capture and treat stormwater runoff as well as urban streams that provide critical ecosystem services, including nitrate removal by denitrification and assimilation. These two studies are complementary, in that they address two critical aspects of urban sustainability including water quantity and water quality.

With respect to water quantity my focus was on understanding how placement and selection of LID technologies might be optimized to restore hydrologic water balance in urban watersheds, as well as provide a new water resources for water-limited urban areas. Specifically, I found that:

- The optimal ratio of infiltrated to harvested water for any given urban catchment depends on local mean annual rainfall and preurban fraction of forest, not effective imperviousness.
- In most regions of the world returning stream hydrology to a preurban state will require that more stormwater is harvested than infiltrated to compensate for lost evapotranspiration.

- The successful use of LID to cure the urban stream syndrome is contingent on the ecological condition of streams and public perception of streams.

With respect to water quality, my focus was on understanding how stream networks in rapidly urbanizing areas function as natural treatments, and specifically how management of point (and by implication non-point) sources of nitrate within the watershed can have unintended consequences for the ecosystem services delivered by streams, specifically in this case nitrate removal by assimilation and denitrification. Specifically, I found:

- Rates of in-stream treatment inferred from flow and nitrate measurements in Jacksons creek are in quantitative agreement with Mulholland et al., predictions, despite significant differences in the timescales of observation (hours-to-days, months-to-years in our case), and location (72 streams in the U.S., southeastern Australia in our Study).
- The non-linear nature of in-stream treatment of nitrate tends to magnify the impact of decisions made by watershed managers, by ramping up in-stream treatment when nitrate loading is reduced, or by ramping down in-stream when nitrate loading is increased.
- A major implication of this last result is that in-stream treatment may help drive through positive socio-ecological feedback loops, urban streams into healthy or degraded states.

- Finally, exploring impact of catchment scale low impact development technologies employment in conjunction with nitrate in-stream treatment in nitrate management is an interesting topic for future research.

Appendix A

Supplemental Information for Chapter 2

Table A.1. Summary of popular LID technologies discussed in the text

LID Technology	Description	Endpoint	Case Study	Design Guidelines
Infiltration trench	Vertical infiltration systems with granular media characterized by high hydraulic conductivity that facilitates storm water infiltration. Typically unvegetated, but some systems are planted.	Infiltration	Flood mitigation using infiltration trenches and rainwater tanks in Melbourne, AU [1].	MDE, 2000 [2]
Wetland /Pond	Storm water collection basins that provide retention and biological treatment of storm water. These systems are often vegetated with native wetland plants, and store, but do not infiltrate storm water.	Harvesting	Many AU systems, including the Monash SE stormwater harvesting system and the Royal Park wetland [3]	MDE, 2000[2]
Porous pavement (with/ without drain)	Vertical infiltration systems capped with porous material such as concrete, pavers, or asphalt. Storm water infiltrates into a drainage layer and is either piped (to storm drains or a storage facility) or allowed to percolate into the surrounding soil.	Infiltration (harvest possible in systems with drains)	Long-term effectiveness of permeable pavement in Renton, Washington, U.S [4].	Caltrans, 2014 [5]
Dry Bioswale	Shallow ditches that convey (horizontally) and vertically infiltrate storm water. These systems are vegetated (often with grasses and	Hybrid	Evaluates the performance of a bioswale with engineered soil for treating and infiltrating runoff	MDE, 2000 [2]

	native plants). They do not pond water for a long period of time – water flows in, through, and out.		in a parking lot in California, U.S.[6]	
Biofilter (with/ without drain)	Vertical infiltration systems that receive runoff from impervious surfaces and are landscaped with perennial flowers and native vegetation. These systems can be fitted with under drains that route water to the storm sewer or to storage facilities. Biofilters that are lined and have drains are suited for harvest whilst systems that are unlined and without drains promote infiltration. Unlined biofilters are sometimes referred to as "raingardens".	Hybrid	Evaluates the hydrologic and pollutant removal performance of three field-scale biofiltration systems in AU [7]	FAWB, 2009 ; MDE, 2000[2] PGCo, 2007 [9]
Rain tank	Storm water storage tanks (ranging in size from small barrels to cisterns) that receive runoff from roofs.	Harvest	Evaluates the efficacy of rainwater tanks for preventing storm sewer overflows in France [10]	MPMSAA, 2008[11]
Green roof	Roofs that capture and infiltrate storm water to support rooftop vegetation. Green roofs are planted with a range of plant types (grasses to trees).	Harvest	Compares runoff quantity and quality from a greenroof and a control roof in Connecticut, U.S [12]	City of Toronto, 2013[13]

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Appendix B

Supplemental Information for Chapter 3

Text B.1. Defining the Stream Network Model

The nodes and reaches of the stream network model for Jacksons Creek were defined as follows.

(1) The watershed areas for Jacksons Creek (300.3 km^2) and Riddells Creek (108 km^2) were delineated using the 30 m ASTER GDEM (Advanced Spaceborne Thermal Emission and Reflection Radiometer Global Digital Elevation Model, retrievable through NASA JPL database) implemented in ArcGIS (v. 10.2, ESRI, Redland, California).

(2) The upper and lower boundaries of the stream network model were selected as follows:

- The upper boundary on Jacksons Creek was gauge station 230206 at the Rosslynne Reservoir outlet (hereafter referred to as the "Rosslynne" boundary);
- The upper boundary on Riddells Creek was gauge station 230204 ("Riddells");
- The lower boundary on Jacksons Creek was gauge station 230202 in the City of Sunbury ("Sunbury") (**Figure B.1**).

Several considerations went into the selection of these model boundaries. First, historical data on stream flow and stream nitrate concentration—data needed for simulating the flow of water and nitrate through the stream network model—are available at Rosslynne, Riddells, and Sunbury. Second, from a management perspective we are most interested in nitrate load management over the portion of the catchment located downstream of the Rosslynne Reservoir (the upstream boundary on Jacksons Creek) and upstream of the Sunbury Recycled Water Plant (the downstream boundary on Jacksons Creek). The upstream boundary on Riddells Creek was chosen because historical flow and nitrate data are available for this site, and it is at roughly the same elevation (370 m) as the Rosslynne Reservoir outlet on Jacksons Creek (473 m); by comparison, the peak ridgeline elevation at the top of the catchment is 824 m.

(3) Sub-watersheds within the study area were delineated using the ArcHydro toolbox within ArcGIS (**Figure B.1**) using the ASTER GDEM and a stream delineation threshold of 700 (0.3% of the total number of pixels in the GDEM). This threshold value was chosen so as to maximize agreement between the stream network model generated from the GDEM and a published 1:25,000 stream map of the study area:

(https://www.data.vic.gov.au/data/dataset). The resulting 4th order stream network had 338 reaches and nodes (**Figure B.1.B**).

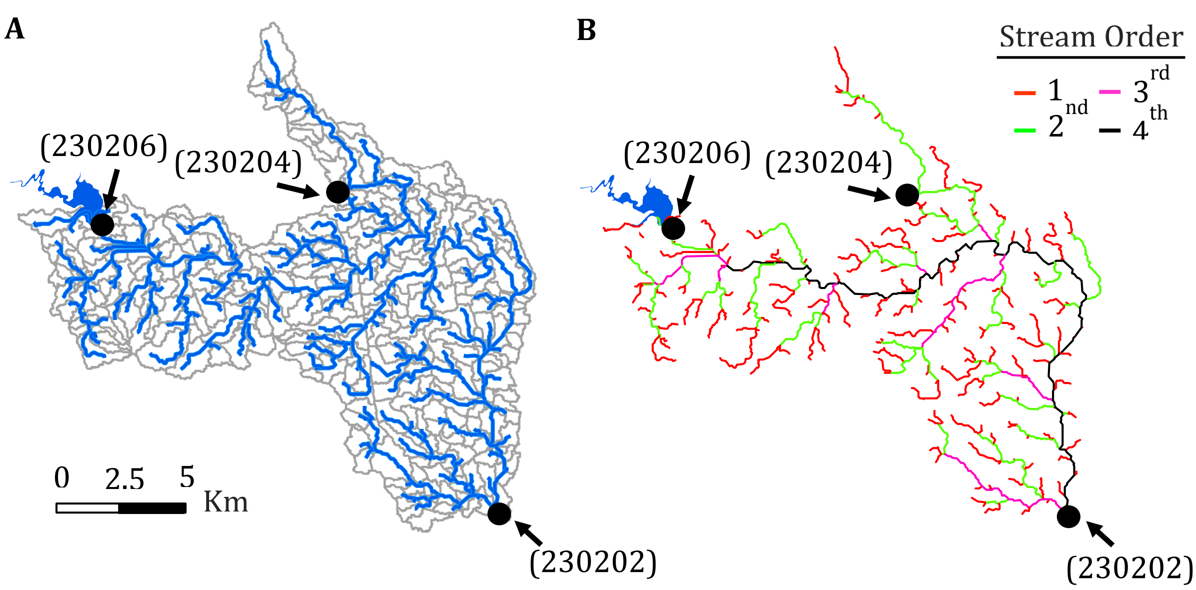

Figure B.1. (A) The boundary of our study area within the Jacksons Creek and Riddells Creek watersheds, together with delineated sub-watersheds and the stream network model generated using ArcGIS. (B) The Strahler stream order of reaches within the stream network model.

Text B.2. Water and Nitrate Budgets

A schematic illustration of the water and nitrate budgets for any reach of the stream network is shown in **Figure B.2.** Mathematical details of the water and nitrate budgets are discussed below.

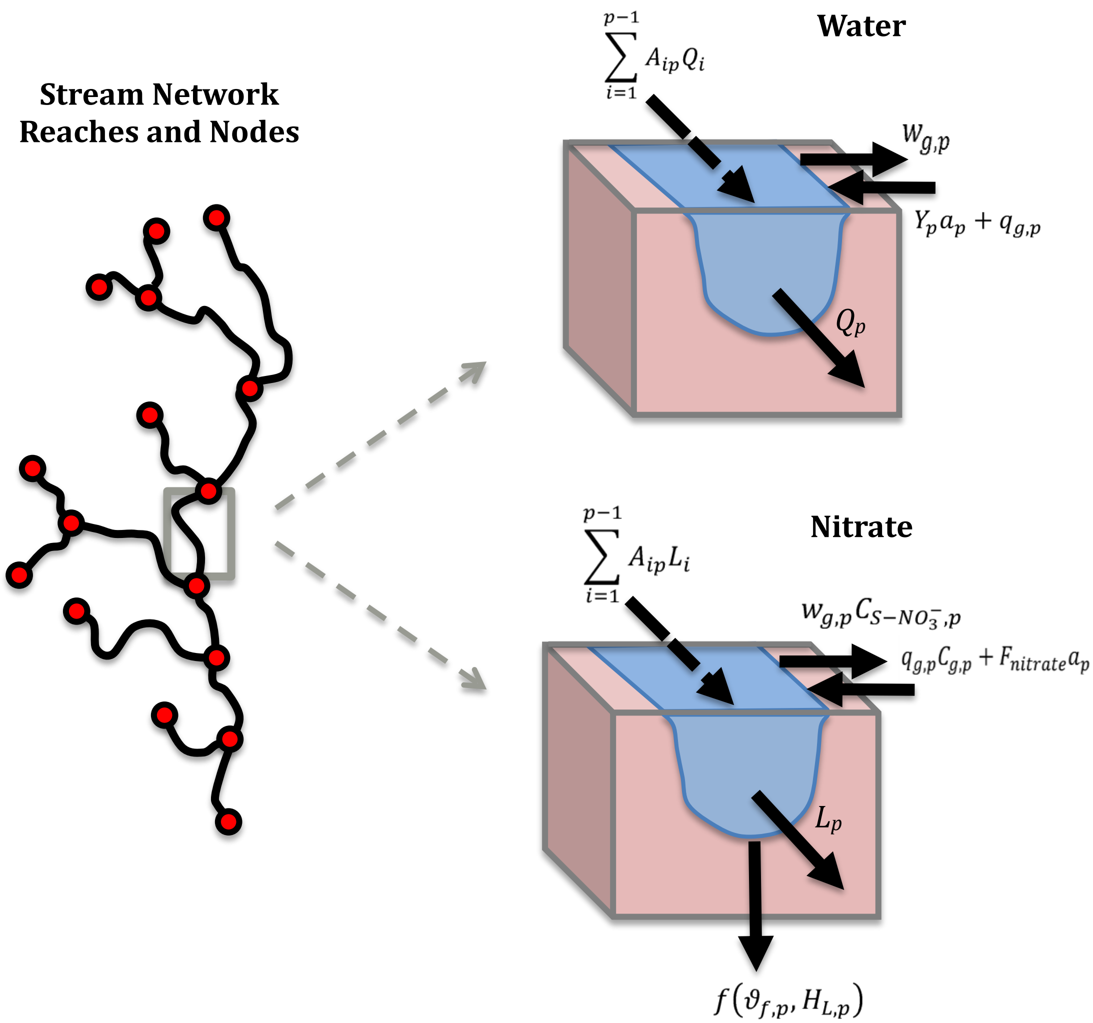

Figure B.2. Water volume (top box) and nitrate load (bottom box) budgets for a single reach

Water Budget. The discharge of water from the pth reach, Q_p[m^3 s^{-1}], is calculated as follows:

$$Q_p = \sum_{i=1}^{p-1} A_{ip} Q_i + Y_p a_p + q_{g,p} - w_{g,p} \quad \text{(B1)}$$

Terms on the right hand side of **equation (B1)** represent (from left to right): (1) the sum of all discharge coming from upstream reaches with a direct connection to the *p*th reach; (2) water draining from the sub-watershed to the *p*th reach; (3) gauged flow discharged to the *p*th reach (e.g., from reservoir releases or wastewater effluent discharges); and (4) gauged withdrawals from the *p*th reach (e.g., for irrigation).

Variables appearing in **equation (B1)** are defined as follows:

A_{ip} = adjacency matrix for the stream network, which indicates if the *p*th reach is directly connected to the *i*th reach (=1) or not (=0) [-]

Y_p = yield coefficient for the sub-watershed draining to the *p*th reach, which physically represents the flux of water volume from the sub-watershed [m s^{-1}]

a_p = area of the sub-watershed draining to the *p*th reach [m^2]

$q_{g,p}$ = inflow of water into the *p*th reach from point source discharges [m^3 s^{-1}]

$w_{g,p}$ = gauged withdrawal of water from the *p*th reach [m^3 s^{-1}]

Routing nitrate through the network. The load of nitrate passing out of the *p*th reach, L_p [mg-N s^{-1}], is calculated as follows:

$$L_p = \left(\sum_{i=1}^{p-1} A_{ip} L_i + q_{g,p} C_{g,p}\right) e^{-\vartheta_{f,p}/H_{L,p}} + F_{nitrate} a_p \left(H_{L,p}/\vartheta_{f,p}\right)\left(1 - e^{-\vartheta_{f,p}/H_{L,p}}\right) - w_{g,p} C_{S-NO_3^-,p} \quad \text{(B2)}$$

Terms on the right hand side of **equation (B2)** represent (from left to right): (1) loading of nitrate to the *p*th reach from all connected upstream reaches and point source discharges within the reach (the exponential accounts for nitrate removal by assimilation and denitrification over the reach); (2) non-point source inputs of nitrate into the *p*th reach and their removal by assimilation and denitrification over the reach; and (3) the removal of nitrate mass by withdrawals within the *p*th reach. **Equation (B2)** can be derived by performing a steady-state nitrate mass balance over the *p*th reach. Variables not already defined include:

$C_{g,p}$ = nitrate concentration associated with gauged flow into the *p*th reach [mg-N L^{-1}]

$F_{nitrate}$ = flux of non-point source nitrate from the *p*th sub-watershed [mg-N m^{-2} s^{-1}]

$\vartheta_{f,p}$= nitrate uptake velocity in the *p*th reach, which physically represents the flux of nitrate into the stream bed by biotic processes (denitrification and assimilation) normalized by the average nitrate concentration in the reach [m s^{-1}]

$H_{L,p}$ = hydraulic loading rate for the *p*th reach [m s^{-1}]

$C_{S-NO_3^-,p}$= average stream nitrate concentration in the *p*th reach [mg-N L^{-1}]

If the nitrate uptake velocity is much smaller than the hydraulic loading rate, **equation (B2)** simplifies as follows:

$$L_p \approx \left(\sum_{i=1}^{p-1} A_{ip} L_i + q_{g,p} C_{g,p}\right) e^{-\vartheta_{f,p}/H_{L,p}} + F_{nitrate} a_p - w_{g,p} C_{S-NO_3^-,p}\ ,\ \vartheta_{f,p} \ll H_{L,p} \qquad \text{(B3)}$$

This simplified form of the nitrate budget was adopted for the stream network model described in Mulholland et al., 2008. For the model simulations described in **Chapter 3**, the more general form of the mass balance (**equation (B2)**) was adopted.

Text B.3. Parameter Estimation

Adjacency Matrix. The Adjacency matrix ($A_{i,p}$) represents the topology of the stream network. To populate the adjacency matrix (which consists of 338 rows and columns, matching the number of reaches in our model) we implemented a computer code in MATLAB that: (a) assigned a unique number to each reach/node (corresponding to the row or column number in the Adjacency matrix); and (b) systematically evaluated whether the pth reach/node was connected to any ith reach/node (matrix entry =1) or not (matrix entry =0).

Subwatershed Area. The subwatershed area associated with each reach, a_{p}, was calculated in ArcGIS.

Reach Length. The reach length l_p for the pth reach was calculated in ArcGIS.

Hydraulic Loading Rate. The hydraulic loading rate for the pth reach is the ratio of the volumetric flow rate and the wetted surface area (estimated from the product of wetted perimeter P_p and the reach length l_{p}):

$$H_{L,p} = \frac{Q_p}{P_p l_p} \tag{B4}$$

The wetted perimeter for the pth reach was estimated from stream discharge using the power-law correlation developed by Leopold and Maddock, 1953 where a and b are empirical constants:

$$P_p = aQ_p^b \tag{B5}$$

We adopted values of the empirical constants proposed by Lacey,1930 (a=4.8 and b=0.5), for which the units of Q_p and P_p must be [m^3 s^{-1}] and [m], respectively.

Gauged inflow and Outflow at Model Boundaries. Gauged discharge at inflow (Rosslynne, Riddells) and outflow (Sunbury) boundaries were obtained from the State of Victoria Department of Environment, Land, Water, and Planning (DELWP) online data repository (http://data.water.vic.gov.au/monitoring.htm). The corresponding site IDs are

230216 (Rosslynne), 230204 (Riddells), and 230202 (Sunbury) (**Figure B.3.A**). Historical discharge of treated sewage effluent from the Gisborne RWP post 1996 (the RWP was not operating prior to this year) were obtained from directly from Western Water (the RWP owner) and licensed water withdrawals (e.g., for irrigation) were obtained from a previously published report [Townsend et al., 2015]. Because documented withdrawals are orders of magnitude lower than inflow to the stream network (**Figure B.3.B**), we set the withdrawal terms in **equations (B1)** and **(B2)** to zero: $w_{g,p} = 0$.

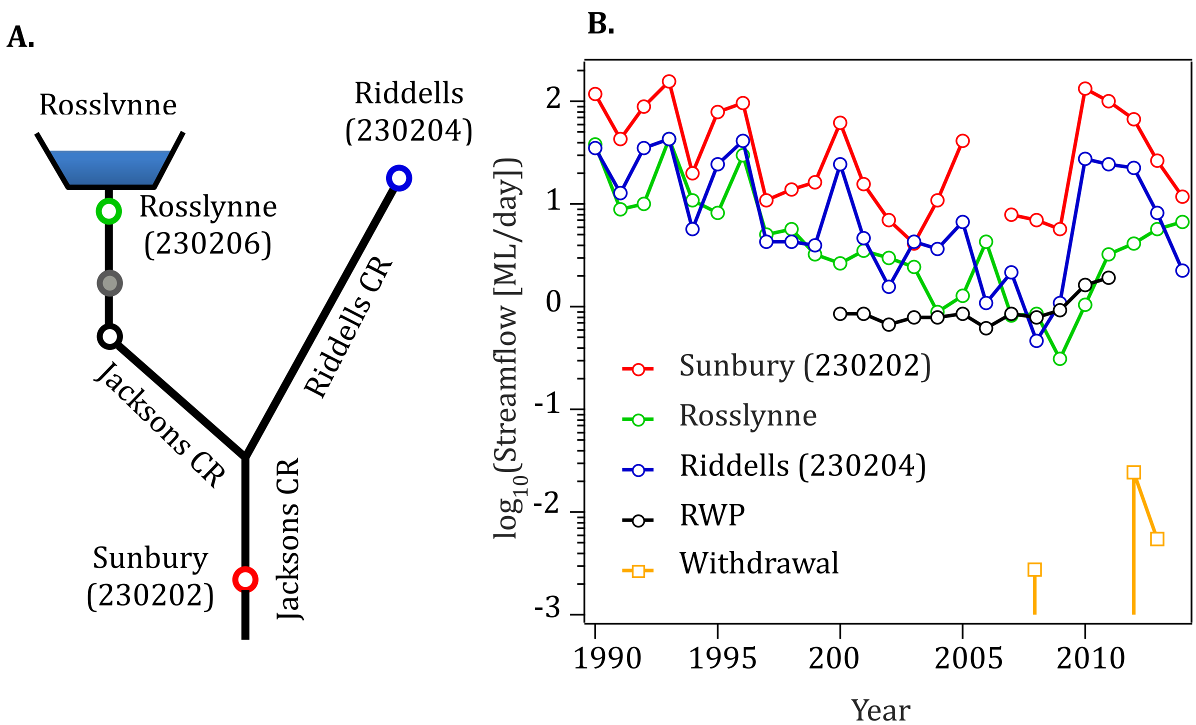

Figure B.3. (A) Schematic diagram showing the relative locations of Rosslynne Reservoir, Gisborne RWP, and stream network model boundaries, including gauge stations 230206 ("Rosslynne"), 230204 ("Riddells"), and 230202 ("Sunbury"). Also shown is the gauge station (203233) used for validating the network model stream flow predictions. (B) Water discharge recorded at the boundary stations and Gisborne RWP; also shown are documented withdrawals from Jacksons and Riddells Creek.

Nitrate Uptake Velocity and Non-Point Source Nitrate Flux. Estimates for v_f were obtained by: (1) optimizing the stream network model so as to maximize agreement between predictions and measurements of nitrate concentrations at the Sunbury (see main **Chapter 3**); or (2) coupling the stream network model to Mulholland's correlation (see **Section A.5** below). $F_{nitrate}$ was assumed constant across all sub-watersheds, but varied with time in proportion to the watershed yield coefficient: $F_{nitrate}(t) = C_{NPS-NO_3^-} \times Y(t)$ (same as equation (1) in **Chapter 3**). The proportionality constant $C_{NPS-NO_3^-}$ represents the average non-point-source nitrate concentration associated with the water flowing from the sub-catchment to a reach. An estimate of $C_{NPS-NO_3^-}$ specific to our study area was obtained by Bayesian inference (see **Figure B.4** and **Chapter 3** for details).

Watershed Yield. A monthly or annual (depending on context) watershed yield coefficient was calculated and applied to all sub-watersheds as follows:

$$Y_p = \frac{Q_{Sunbury} - Q_{Rosslynne} - Q_{Riddells} - Q_{RWP}}{A_W} \quad \text{(B6)}$$

Variables appearing here include average volumetric flow rates measured at Sunbury (Q_{Sunbury}), Rosslynne ($Q_{\text{Rosslynne}}$), Riddells (Q_{Riddells}), and RWP (Q_{RWP}) stations, and the total area of the watershed encompassed by the stream network model (A_{w} = 160 km^2).

Text B.4. Validation of the Stream Network Model

Given measured inflows at the upstream boundary stations (Rosslynne and Riddells) and measured discharge from the RWP, our flow network model (**equation (B1)**) simulates stream discharge in all 338 reaches. To validate the flow model, we compared model predicted and historical flow measurements at gauge station 203233 on Jacksons Creek, located upstream of the RWP discharge (see **Figure B.3.A**). Importantly, data from this gauge station were not used to calibrate or drive the stream network model, so comparison of model-predicted and measured discharge at 203233 represents a true test of the model's predictive power. Model predictions closely track monthly average flow measurements at 203233 for the period 1994 through 2001 (**Figure B.5.A**). The model is biased somewhat low during winter months (June through August, **Figure B.5.B**) but otherwise falls close to the 1:1 line (Nash-Sutcliffe Efficiency index is 0.69); the Nash-Sutcliff Efficiency index ranges from $-\infty$ to 1, where 1 is a perfect model fit, 0 is no better than the mean, and < 0 is worse than the mean.

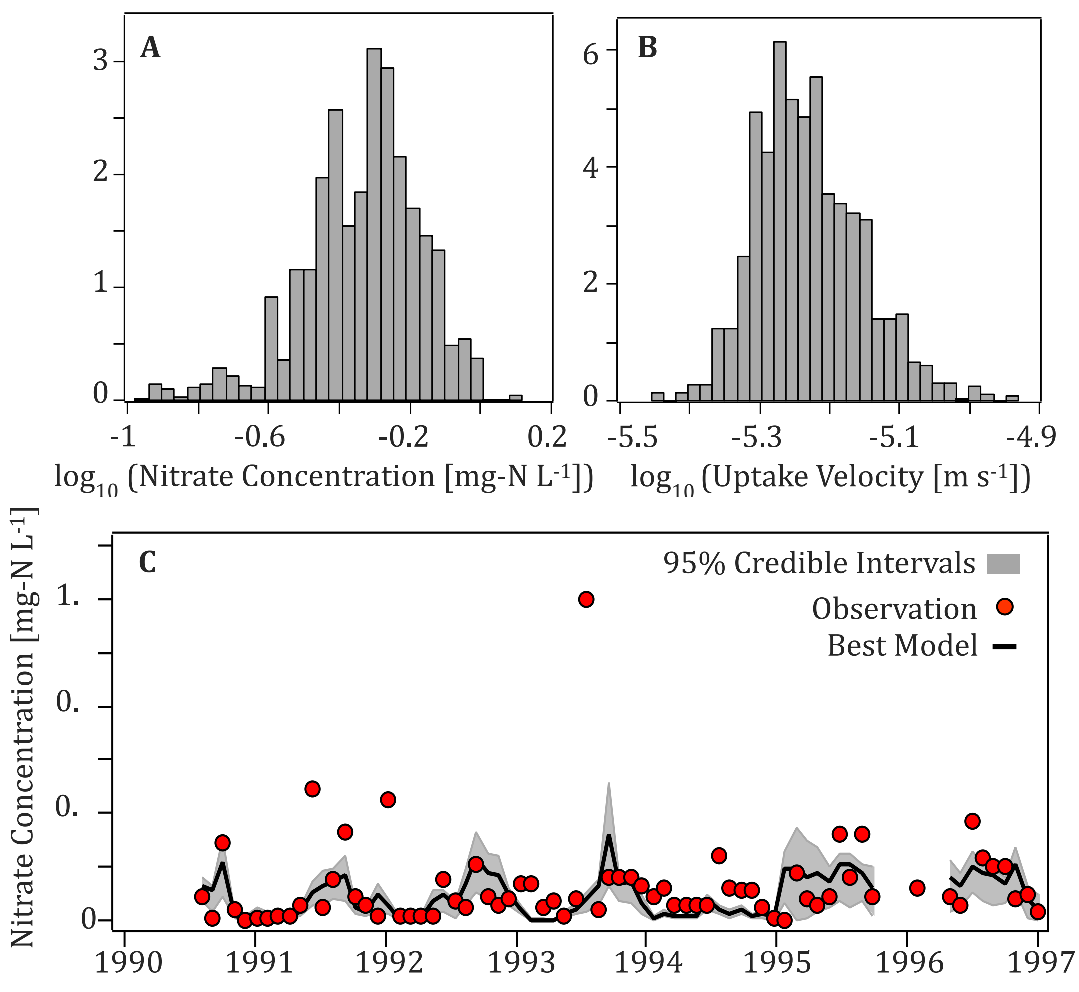

Figure B.4. Posterior distributions for non-point source nitrate concentration (A) and nitrate uptake velocity (B). (C) Time series of measured nitrate concentrations at Sunbury and model predictions using the optimal values from panels (A) and (B).

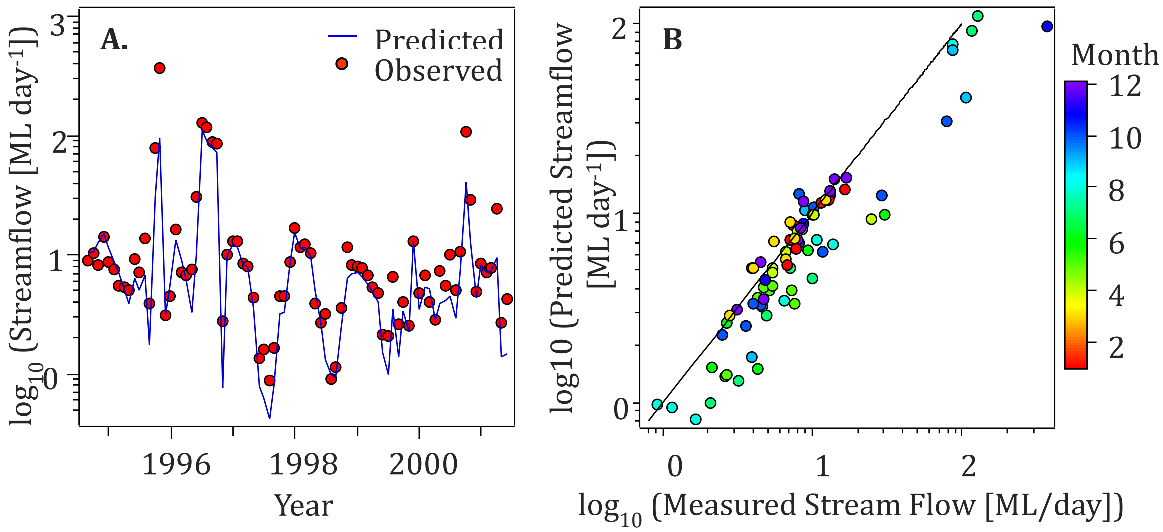

Figure B.5. Validation of stream discharge predicted by the stream network model at gauge station 203233, including a comparison of model-predicted and measured timeseries (A) and 1:1 plot (B).

Text B.5. Coupling Mulholland's Correlation and the Stream Network Model

In general, the nitrate uptake velocity will vary throughout the stream network and also with time, reflecting the local nitrate concentrations present in each reach. To prepare **Figure 3.1.D** in **Chapter 3** we computed the nitrate uptake velocity in the *p*th reach ($\vartheta_{\mathrm{f,p}}$) based on the average stream nitrate concentration in the *p*th reach ($C_{S-NO_3^-,p}$) using Mulholland's power-law correlation:

$$\log \vartheta_{f,p} = -0.462 \times \log[C_{S-NO_3^-,p}] - 2.206 \qquad \text{(B7)}$$

Following Mulholland et al., 2008 we estimated the average stream nitrate concentration from the quotient of nitrate load and water discharge leaving the *p*th reach:

$$C_{S-NO_3^-,p} = \frac{\sum_{i=1}^{p-1} A_{ip}L_p + q_{g,p}C_{g,p} + F_{nitrate}a_p}{Q_p} \tag{B8}$$

Text B.6. Addressing Data Gaps in Nitrate Measurements

To address the sporadic nature of nitrate sampling in the Jacksons Creek watershed (see **Figure B.6** and **Chapter 3**), we had several choices. We could compute a median value based on all nitrate measurements collected at a particular station, and then assigned this median value to the variable $C_{g,p}$ in **equation (B2)**. However, by adopting this approach we might miss important temporal patterns (e.g., seasonal variations) in the loading of nitrate to the stream network. Instead, we prepared Multiple Linear Regression (MLR) models, using log (base 10) transformed nitrate concentration as the dependent variable and the following candidate predictor variables and their pair-wise interaction terms: (1) 24h antecedent precipitation, (2) binary rain/no-rain, (3) maximum daily air temperature, (4) binary higher/lower than mean air temperature, and (5) season (Fall, Winter, Spring, Summer). The variance inflation factor (VIF) was calculated for all variables in each model to check for multi-collinearity; any variables with VIF > 5 were excluded from the analysis. Functional marginality was invoked so that the model could not contain pairwise interaction terms without also including both parent variables.

For Rosslynne, the "best" MLR model (evaluated using Akaike Information Criterion corrected for small sample size) included maximum daily temperature and Fall (15% variance explained). For Riddles the best MLR model included maximum daily temperature, Fall, and a Fall-temperature interaction term (50% variance explained, see **Figure B.7**). Although the variance explained by the MLR model for Rosslynne is low (15%, possibly due to the uneven sampling frequency at this site) both MLR models are significantly better than the alternative "null model"; i.e., using the median nitrate value alone ($p<0.05$)

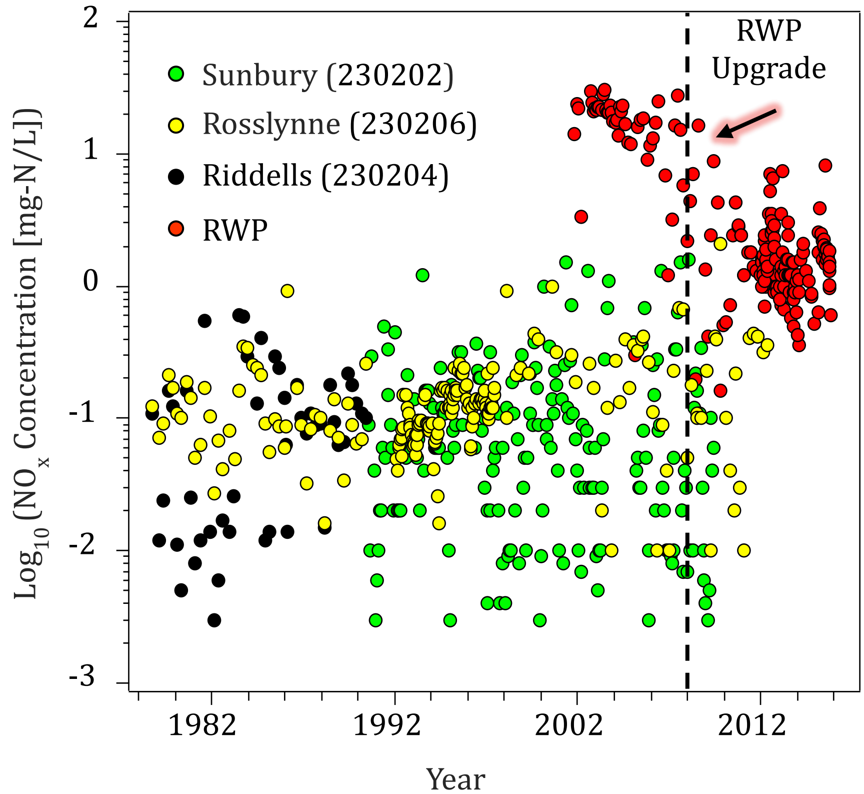

Figure B.6. A compilation of nitrate monitoring data at the model boundaries and in the RWP effluent.

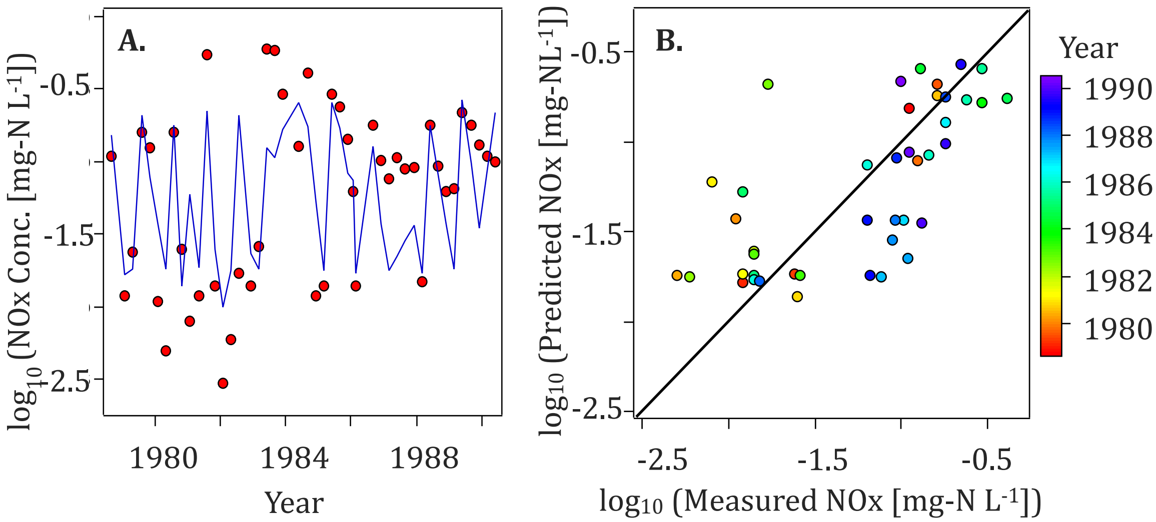

Figure B.7. Multiple Linear Regression (MLR) model fit to nitrate concentrations measured at the Riddells station. Shown here are a comparison of model-predicted and measured nitrate concentrations presented in time series (**A**) and 1:1 (**B**) formats.

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