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Stormwater management network effectiveness and implications for urban watershed function: A critical review

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Abstract
Deleterious effects of urban stormwater are widely recognized. In several countries, regulations have been put into place to improve the conditions of receiving water bodies, but planning and engineering of stormwater control is typically carried out at smaller scales. Quantifying cumulative effectiveness of many stormwater control measures on a watershed scale is critical to understanding how small-scale practices translate to urban river health. We review 100 empirical and modelling studies of stormwater management effectiveness at the watershed scale in diverse physiographic settings. Effects of networks with stormwater control measures (SCMs) that promote infiltration and harvest have been more intensively studied than have detention-based SCM networks. Studies of peak flows and flow volumes are common, whereas baseflow, groundwater recharge, and evapotranspiration have received comparatively little attention. Export of nutrients and suspended sediments have been the primary water quality focus in the United States, whereas metals, particularly those associated with sediments, have received greater attention in Europe and Australia. Often, quantifying cumulative effects of stormwater management is complicated by needing to separate its signal from the signal of urbanization itself, innate watershed characteristics that lead to a range of hydrologic and water quality responses, and the varying functions of multiple types of SCMs. Biases in geographic distribution of study areas, and size and impervious surface cover of watersheds studied also limit our understanding of responses. We propose hysteretic trajectories for how watershed function responds to increasing imperviousness and stormwater management. Even where impervious area is treated with SCMs, watershed function may not be restored to its predevelopment condition because of the lack of treatment of all stormwater generated from impervious surfaces; non-additive effects of individual SCMs; and persistence of urban effects beyond impervious surfaces. In most cases, pollutant load decreases largely result from run-off reductions rather than lowered solute or particulate concentrations. Understanding interactions between natural and built landscapes, including stormwater management strategies, is critical for successfully managing detrimental impacts of stormwater at the watershed scale.

KEYWORDS
best management practices, cumulative effects, green infrastructure, low impact development, stormwater control measures, stormwater management, urban catchments, urban hydrology

1 | INTRODUCTION

Stormwater run-off is a dramatic hydrologic manifestation of many changes that result from urbanization. Stormwater run-off is water that cannot infiltrate or be evapotranspired because impervious rooftops and pavements limit access to soil and plants and quickly convey run-off into pipes and channels. Effectively managing stormwater is a challenge faced by cities around the globe and is increasingly difficult as urban populations grow (Grimm et al., 2008). Increased precipitation intensity occurring in many regions as a result of climate change

(Westra, Alexander, & Zwiers, 2013) further exacerbates the challenge of stormwater management. Impacts of stormwater run-off from impervious surfaces are pervasive throughout urban areas. Stormwater run-off degrades the integrity of urban aquatic ecosystems, alters hydrologic regimes, elevates nutrient and contaminant concentrations, and harms aquatic plant and animal communities (Schueler, Fraley-McNeal, & Cappiella, 2009; Shuster, Morrison, & Webb, 2008). In the United States, urban stormwater run-off is the third largest source of water quality degradation in bays and estuaries and the sixth largest source of degradation for streams, as reported in national geospatial datasets (ATTAINS) (https://ofmpub.epa.gov/waters10/attains_nation_cy.control), and at least $US19.2 billion of capital investments are needed for improved stormwater management to meet water quality requirements (United States Environmental Protection Agency, 2016). The need to improve stormwater management practices for mitigating the impacts of urbanization has been gaining traction across the globe (CEC, 2000; Hamel, Daly, & Fletcher, 2013; Jia et al., 2015).

Although the effects of stormwater run-off are often recognized and regulated in receiving water bodies at the watershed scale, practices to manage stormwater are generally designed for smaller drainage areas (Roy et al., 2008). In this review, we will use stormwater control measures (SCMs) to describe engineered stormwater management practices, following current U.S. conventions, though the terminology in use varies regionally (Fletcher et al., 2015). SCM networks will be used to refer to the aggregation of SCMs within a watershed. Given significant ongoing and impending investment in stormwater management intended to improve watershed-scale conditions, there is an urgent need for information on cumulative effects of SCM networks, to inform decision-making about trade-offs across different stormwater management strategies.

At the site scale, developers can often select from an array of SCM types, and therefore, at the watershed scale, SCM networks can include multiple types of SCM designs. Some SCMs are designed primarily for water quantity function, whereas others have water quality improvement as a primary or coequal goal. The choice of SCM type can be constrained by regulations for water quantity, quality, or both (Balascio & Lucas, 2009). The first generation of SCMs were typically designed to detain stormwater, slowly releasing it into receiving waterways, to decrease peak flows, with limited regard to other components of the water balance (Burns, Fletcher, Walsh, Ladson, & Hatt, 2012; National Research Council, 2009). These detention-based SCMs (e.g., ponds and wetlands) are typically centralized, located within or proximal to surface waterways. Increased residence times in such SCMs improved capture of suspended sediments and associated pollutants through sedimentation and biological uptake of nutrients (Hathaway & Hunt, 2010). Since 2000, SCMs that enhance infiltration, evapotranspiration, and water capture for reuse have become increasingly popular and are encouraged by agencies such as the United States Environmental Protection Agency (EPA, 2007). Placement of these SCMs is decentralized in upland areas, to treat stormwater run-off closer to the source (Petrucci, Rioust, Deroubaix, & Tassin, 2013). This approach can include a single SCM or SCMs arranged in a sequence, or treatment train, to provide redundancy in hydrologic treatment and to maximize pollutant removal processes. Examples of these types of SCM include green roofs, bioretention, infiltration basins, and cisterns. Decentralized SCMs are becoming an increasingly common component of new development, whereas retrofitting existing developed areas with either centralized or decentralized SCMs remains challenging due to space constraints and social resistance (Shuster et al., 2008; Turner, Jarden, & Jefferson, 2016). Increased used of decentralized SCMs means that understanding the cumulative impacts of multiple SCMs is now important even at the scale of a single neighbourhood or development.

In this review, we examine the state of knowledge about effectiveness of stormwater management at the watershed scale and assess its implications for urban hydrologic function. Our first objective is to examine what empirical and modelling investigations have elucidated about cumulative effects of stormwater management strategies on hydrology and water quality (nutrients and sediment) and what challenges are faced in such studies. In this review, we first summarize the challenges to understanding cumulative effects of SCM networks, the approaches used to study cumulative effects, and the results of existing studies. Second, we propose a hypothesis of hysteresis behaviour in watershed hydrology and water quality as arising from urbanization and stormwater management. Last, we discuss management implications and research opportunities highlighted by existing studies and hysteresis hypothesis.

The spatial scale of our review encompasses watersheds in which multiple SCMs are designed to influence the hydrology of a receiving water body, or in which there is a mixture of run-off treated by SCMs and untreated stormwater run-off. Our review specifically examines watershed-scale effectiveness rather than single SCM input–output studies, which have been previously reviewed (Ahiablame, Engel, & Chaubey, 2012; Hatt, Fletcher, & Deletic, 2009; Hunt, Jarrett, Smith, & Sharkey, 2006; Koch, Febria, Gevrey, Wainger, & Palmer, 2014; Vogel & Moore, 2016). Our geographic scope encompasses North America, Europe, East Asia, and Australia because of availability of existing studies and congruence of approaches. Urban drainage issues are important in developing regions as well, but engineered solutions may need to be adapted in ways that are appropriate for the social and geographical contexts (Parkinson, Tayler, & Mark, 2007; Silveira, 2002).

2 | CHALLENGES TO UNDERSTANDING CUMULATIVE EFFECTS OF SCM NETWORKS

Cumulative effects of SCM networks are inherently difficult to quantify at a watershed scale. These difficulties result from needing to separate the stormwater management signal from the signal of urbanization itself, innate watershed characteristics that lead to a range of hydrologic and water quality responses, and the varying functions of multiple types of SCMs.

Hydrological responses of a watershed to urbanization itself (e.g., the initial stressor) vary on both a local and regional scale and are not necessarily predictable. The direction of some hydrological metrics (e.g., peak flow) responds consistently to urbanization but exhibit large ranges in magnitude of response at similar levels of urbanization due to physiographic differences (Hopkins et al., 2015). Other hydrological responses, such as baseflow, have shown contrasting responses to urbanization because of differences in how regions are urbanized over time, including the extent of vegetation removal or irrigation, as well as the age of stormwater and other infrastructure (Bhaskar, Beesley et al., 2016; O'Driscoll, Clinton, Jefferson, Manda, & McMillan, 2010; Price, 2011). Response to urbanization and to SCM networks can also be influenced by a watershed's capacitance, where capacitance is defined as how well the environmental setting facilitates stormwater run-off entering long subsurface flowpaths or evapotranspiring (Miles & Band, 2015). High watershed capacitance is associated with high soil infiltration rates, gentle topography, and deeper groundwater tables. High capacitance watersheds may be able to recover hydrologic and water quality functions more effectively as a result of stormwater management, than when it is implemented in low capacitance watersheds.

In terms of water quality, increased pollutant concentrations are well correlated with urbanization and population density (Hatt, Fletcher, Walsh, & Taylor, 2004; Peierls, Caraco, Pace, & Cole, 1991). Although pollutant retention and removal is observed in studies at the individual SCM scale, it remains elusive at the watershed scale with reductions in mass export highly variable and often attributed to hydrologic rather than biogeochemical drivers. Multiple factors likely influence this variability. First, biophysical processes in SCMs vary by constituent, creating diverging patterns in effectiveness attributed to SCM size, type, age, and location (Koch et al., 2014; Liu et al., 2017; Pennington, Kaplowitz, & Witter, 2003). For example, Winston, Page, and Hunt (2013) and Line and White (2015) observed decreased total phosphorus (TP) attributed to sedimentation of particulate fractions in bioretention areas, whereas Duan, Newcomer-Johnson, Mayer, and Kaushal (2016) found that particulate P was retained only during high flows and subsequently released during low flows. Second, watershed storage and release of pollutants is also related to historical land use (Chen, Hu, Guo, & Dahlgren, 2015; Van Meter & Basu, 2015). Accumulation of nutrients in soils over decades of fertilizer application from agriculture can lead to time lags between implementation of mitigation practices and measurable differences in water quality (Hamilton, 2012). Third, this variability can be attributed to human actions within urban landscapes, such as fertilizer application rates onto residential lawns as a function of age of development (Law, Band, & Grove, 2004; Zhou, Troy, & Grove, 2008). Together, this results in complex spatial and temporal patterns of water quality that are often difficult to disentangle.

SCM networks may need to treat a certain critical fraction of the watershed for their effect to be detected. If only a small fraction of impervious surfaces are draining to SCMs, we would not expect this effect to be evident on a watershed scale, as the cumulative effect would still be dominated by untreated impervious areas (Li, Fletcher, Duncan, & Burns, 2017). Moreover, SCMs are often not designed to completely mitigate all run-off; for example, in the United States, many state regulations require a fraction of run-off to be regulated for water quality treatment (MDE, 2009). Additionally, urbanization profoundly, and often irreversibly, changes various components of the pervious urban landscape, including soil bulk density and vegetative cover (Gregory, Dukes, Jones, & Miller, 2006; Line & White, 2015). These changes can affect the ability of pervious areas to infiltrate rainfall, and under certain circumstances, pervious areas may actually behave as sources of stormwater run-off (Lim, 2016).

Finally, different SCM design functions can produce contrasting effects on hydrology and water quality. In terms of hydrology, three dominant SCM functions are detention, infiltration, and harvest for evapotranspiration or reuse (Askarizadeh et al., 2015), with designs that incorporate water retention effectively supplying water to the other functions. Individual SCMs can include all three of these functions to varying degrees, and an SCM network can incorporate multiple types of SCMs with different mixes of these functions (Askarizadeh et al., 2015). In terms of water quality, dynamic biological, physical, and chemical processes that occur within individual SCMs act in often inconsistent ways: transforming inorganic nutrients to organic forms (Gold, Thompson, & Piehler, 2017), temporarily storing particulate bound metals through filtration and sedimentation (Davis, Shokouhian, Sharma, Minami, & Winogradoff, 2003), or removing nitrate through denitrification (Bettez & Groffman, 2012; Collins et al., 2010; Dietz & Clausen, 2006). SCMs designed to capture and store run-off in detention basins may increase sedimentation but do little to remove dissolved nutrients. Detention ponds may retain water and increase residence time for nutrient transformations during smaller events, but in larger events or events with wet antecedent conditions, residence times may be low with little effect on either hydrology and water quality (Jefferson, Bell, Clinton, & McMillan, 2015; Loperfido, Noe, Jarnagin, & Hogan, 2014). These examples highlight the complexity of function within a single SCM, which gets amplified as multiple SCMs of different designs are considered for treatment of many constituents.

3 | RESULTS OF EXISTING STUDIES

3.1 | Scope

Empirical and modelling studies of SCM effects have largely been concentrated in the eastern and midwestern United States, Europe, and Australia (Table 1; Figure 1). Modelling studies cover a larger range in watershed sizes (0.001 to 666 km^2) than empirical studies (0.006 to 202 km^2; Figure 2), because they are not constrained by sizes of actual watersheds with stormwater management. Percent impervious area in watersheds studied have ranged from 3.8% to 85% (Figure 2). In a few cases, all impervious area is “treated” (drains to an SCM) (e.g., Fanelli, Prestegaard, & Palmer, 2017; Hogan, Jarnagin, Loperfido, & Van Ness, 2014). However, other studies have focused on watersheds with implementation of SCMs for only a fraction of the impervious area (e.g., Bell, McMillan, Clinton, & Jefferson, 2016; Jarden, Jefferson, & Grieser, 2016). Not all studies report the same metrics for intensity of SCM implementation, making it difficult to compare (e.g., effective impervious area). The broad literature of studies that focused on differing urban development intensities or styles (e.g., compact vs. dispersed), without explicit attention to structural SCM practices (e.g., Pyke et al., 2011), was not considered within the scope of our review.

TABLE 1 Locations where studies of cumulative effects of SCM networks have been performed

Site	Location	Study types	Empirical study design	Model used	SCM function	Watershed sizes (km^2)	Impervious area (%)	Source
1	Annapolis, MD, USA	E	G		D, I	0.05–0.6	1–65	Fanelli et al. (2017)
2	AsanTangjung New Town, South Korea	M		SWMM	H, I	1.8	Not given	Lee et al. (2012)
3	Athens, GA, USA	M		SCS, StormNet Builder	H	2.4	54	Carter and Jackson (2007)
4	Auckland, New Zealand	M		MUSIC	D, H, I	0.83	27	Elliott, Trowsdale, and Wadhwa (2009)
5	Baltimore, MD, USA	EM	P	GSSHA, SCS TR-20	D, I, WQ	0.001–14	19–61	McCuen, (1974), Gilroy and McCuen (2009), Meierdiercks, Smith, Baeck, and Miller (2010), Newcomer-Johnson, Kaushal, Mayer, and Grese (2014), Smith, Smith, Baeck, and Miller (2015), Duan et al. (2016)
6	Baltimore-Washington, MD-DC, USA	E	P		D, I, WQ	0.5–34	14–53	Pennino, McDonald, and Jaffe (2016)
7	Beijing, China	M		SWMM, BMPDSS, process-based spreadsheet model	D, H, I	0.03–29.5	30–69.8	Jia, Lu, Yu, and Chen (2012), Liu, Bralts, and Engel (2015), Xing et al. (2016)
8	Bloomington, IL, USA	M		PCSWMM	D, H, I	88	27–50	Ahiablame and Shakya (2016)
9	Blacksburg, VA, USA	M		Bentley SewerGEMS	D	1.54	6.2	James and Dymond (2012)
10	Boston, MA, USA	EM	G	WinSLAMM	I	0.08–1	Not given	Hurley and Forman (2011), Thomas and Vogel (2012)
11	Brisbane, Australia	M		SWMM, MUSIC	H	1.4–27.9	5–70	Fletcher, Mitchell, Deletic, Ladson, and Se (2007), Aryal, Ashbolt, and Mcintosh (2016)
12	Chapel Hill, NC, USA	E	P, LT		D, I	0.03	24	Line and White (2015)
13	Charlotte, NC, USA	EM	G	MUSIC	D	.015–33	4–43	Gagrani, Diemer, Karl, and Allan (2014), Jefferson et al. (2015), Bell, McMillan, Clinton, and Jefferson (2016, 2017)
14	Cincinnati, OH, USA	EM	BACI	RECARGA	D, H, I	0.28–2	13–20	Shuster, Gehring, and Gerken (2007), Shuster and Rhea (2013), Roy et al. (2014)

(Continues)

TABLE 1 (Continued)

Site	Location	Study types	Empirical study design	Model used	SCM function	Watershed sizes (km^2)	Impervious area (%)	Source
15	Clarksburg, MD, USA	EM	P, LT	Monte Carlo spreadsheet model	D, I, WQ	1.1	30	Hogan et al., (2014), Loperfido et al. (2014), Rhea, Jarnagin, Hogan, Loperfido, and Shuster (2015), Bhaskar, Hogan, and Archfield (2016) Hopkins, Loperfido, Craig, Noe, and Hogan (2017), Sparkman, Hogan, Hopkins, & Loperfido (2017)
16	Clay County, FL, USA	M		HEC-HMS	D	2.5	13	Williams and Wise (2006)
17	College Station, TX, USA	M		HEC-HMS/SWMM	D, H, I	3	41	Damodaram et al. (2010)
18	Copenhagen, Denmark	M		MIKE URBAN CS/MOUSE	I	3	Not given	Roldin et al. (2012)
19	Cross Plains, WI, USA	E	P		D, I	0.77	36	Selbig and Bannerman (2008)
20	Dallas/Ft. Worth, TX, USA	M		SWAT	D, H, I	370	Not given	Giacomoni, Gomez, & Berglund (2014)
21	Espoo, Finland	M		SWMM	D, H, I	0.12	38.7	Guan, Sillanpää, and Koivusalo (2015)
22	Eugene, OR, USA	M		SWAT	I	28–270	2.2–9.7	Wu, Bolte, Hulse, and Johnson (2015)
23	Genoa, Italy	M		SWMM	I	0.06	60	Palla and Gnecco (2015)
24	Greenville, SC	E	G		D	0.31–7.26	Not given	Hur et al. (2008)
25	Guangzhou, China	M		SUSTAIN	H, I	0.3	59	Jia et al. (2015)
26	Gwangju, South Korea	M		SWMM	H, I	0.01	85	Baek et al. (2015)
27	Gwinnett County, GA, USA	E	P, LT		D	3–24	12–52 (EIA)	Aulenbach, Joiner, and Painter (2017)
28	Houston, Texas, USA	E	P		I	89	32	Yang and Li (2013)
29	Indianapolis, IN, USA	M		L-THIA-LID	D,H, I, WQ	51	45	Ahiablame, Engel, and Chaubey (2013), Liu, Chen, and Peng (2015)
30	Iowa City, IA, USA	M		SCS CN and UNSAT-H	Other	8.4	50	Holman-Dodds, Bradley, and Potter (2003)
31	Jacksonville, NC, USA	E	BACI		D	0.7–8.35	1.2–28	Gold et al. (2017)
32	Kansas City, KC USA	M		SUSTAIN	I	0.4	40	Lee et al. (2012)
33	King County, WA, USA	M		Based on HSPF	D	14.2	6–29 (EIA)	Booth and Jackson (1997)
34	Kitchener, Ontario, Canada	M		GAWSER	H, I	5.75	Not given	Zimmer, Heathcote, Whiteley, and Schroter (2007)

(Continues)

TABLE 1 (Continued)

Site	Location	Study types	Empirical study design	Model used	SCM function	Watershed sizes (km^2)	Impervious area (%)	Source
35	Lafayette, IN, USA	M		L-THIA-LID	H, I	0.03–1.1	Not given	Wright, Liu, Carroll, Ahiablame, and Engel (2016)
36	Melbourne, Australia	EM	BACI, P	MUSIC, RORB	D, H, I	4.5–40	13.5–70	Fletcher et al. (2007), Burns, Fletcher, Hatt, Anthony, and Walsh (2010), Hamel and Fletcher (2014), Burns et al. (2016)
37	Michigan City, IN, USA	M		L-THIA-LID		153.2	Not given	Liu, Theller, Pijanowski, and Engel (2016)
38	Middleton, WI, USA	EM	P	Infiltration patch spreadsheet model	D, I, WQ	0.33 (15–63)	17–31 (Not given)	Brander, Owen, and Potter (2004), Gebert, Rose, and Garn (2012)
39	Montgomery County, MD, USA	M		SCS TR-20	D	0.09	52	McCuen (1979)
40	Muskegon, MI, USA	M		PLOAD	D,H, I	137	Not given	Steinman, Isely, and Thompson (2015)
41	Nanjing, China	M		SCS CN	H, I	0.58	73.8	Zhang, Guo, and Hu (2016)
42	Nassau County, NY, USA	EM	P	Unspecified groundwater model	I	683	Not given	Ku, Hagelin, and Buxton (1992)
43	New Castle, new South Wales, Australia	EM	Other	WUFS	I	0.37	24	Lucas and Coombes (2009)
44	New Mexico, USA	M		MODFLOW-Surfact, UNSAT-H	I	0.34	31	Stephens, Miller, Moore, Umstot, and Salvato (2012)
45	New York City, NY, USA	M		SWMM	H, I	124	1.7–43.7	Zahmatkesh, Burian, Karamouz, Tavakol-Davani, and Goharian (2015)
46	Oklahoma City, USA	M		Topographic index	D,H, I	666	40	Martin-Mikle, de Beurs, Julian, and Mayer (2015)
47	Palermo, Italy	M		Integrated urban drain-infiltration BMP-storage tank model	WQ	0.128	68	Freni, Mannina, and Viviani (2010)
48	Paris, France	M		SWMM	D,H, I	0.023–5.5	30	Petrucci et al. (2012), Petrucci et al. (2013), Versini, Jouve, Ramier, Berthier, and De Gouvello (2016)
49	Parma, OH, USA	EM	BACI	SWMM	H, I	0.11	55.5	Jarden et al. (2016), Avellaneda, Jefferson, Grieser, and Bush (2017)
50	Perth, Austalia	EM	G	MODHMS, MIKE URBAN	I	51–112	24 – Na	Appleyard (1995), Barron, Barr, and Donn (2013), Locatelli et al. (2017)
51	Philadelphia, PA, USA	M		HEC-HMS, DYNFLOW	D, I	54	17	Emerson, Welty, and Traver (2005), Maimone, O'Rourke, Knighton, and Thomas (2011)

(Continues)

TABLE 1 (Continued)

Site	Location	Study types	Empirical study design	Model used	SCM function	Watershed sizes (km^2)	Impervious area (%)	Source
52	Phoenix, AZ, USA	E	P		D	1.2–5.6	49–69	Hale, Turnbull, Earl, Childers, and Grimm (2015)
53	Raleigh, NC, USA	EM	P	SWMM	D, I	0.02–21.5	30–84 (DCIA)	Pomeroy, Roesner, Coleman, and Rankin (2008), Wilson, Hunt, Winston, and Smith (2014)
54	Recklinghausen, North Rhine-Westphalia, Germany	M		GwNeu, HYDRUS-2D, SPRING	I	11.5	63–83	Göbel et al. (2004)
55	Richmond, VA, USA	M		SWMM	D, I	0.07	81	Lucas and Sample (2015)
56	St. Louis, MO, USA	M		SWMM	H, I	93	40	Di Vittorio and Ahiablame (2015)
57	Salt Lake City, UT, USA	M		SWMM	D	0.11	46	Feng, Burian, and Pomeroy (2016)
58	San Diego, CA, USA	M		SWMM	H	31	50	Walsh, Pomeroy, and Burian (2014)
59	Singapore, Singapore	M		MIKE URBAN	D, I	161	66	Trinh and Chui (2013)
60	Syracuse, NY, USA	M		MODFLOW	I	0.08	43	Endreny and Collins (2009)
61	Trier, Rhineland-Palatinate, Germany	E	LT		D, I	0.4	30	Kessler, Meyer, Seeling, Tressel, and Krein (2012)
62	Taoyuan County, Taiwan	M		SUSTAIN	I	88	Not given	Chen, Sheng, Chang, Kang, and Lin (2014)
63	Waterford, CT, USA	EM	P, LT	SWMM	D, H, I	0.02	22–33	Hood, Clausen, and Warner (2007), Dietz and Clausen (2008), Bedan and Clausen (2009), Rosa, Clausen, and Dietz (2015)
64	Wilmington, NC, USA	E	BACI		D, I, WQ	0.003–0.005	60	Winston, Page et al. (2013), Page, Winston, Mayes, Perrin, and Hunt (2014)
65	Woburn, MA, USA	M		Distributed rainfall–run-off model	I	25	Not given	Perez-Pedini et al. (2005)

Note. Study type indicates empirical (E), modelling (M), or both (EM). For empirical studies, study designs were categorized as comparisons over time: Long term (LT), before-after-control-impact (BACI) or comparisons over space: Paired (P), gradient (G). SCM function is categorized as detention (D), harvest (H), infiltration (I), and explicitly designed for water quality (WQ).

SWMM = stormwater management model.

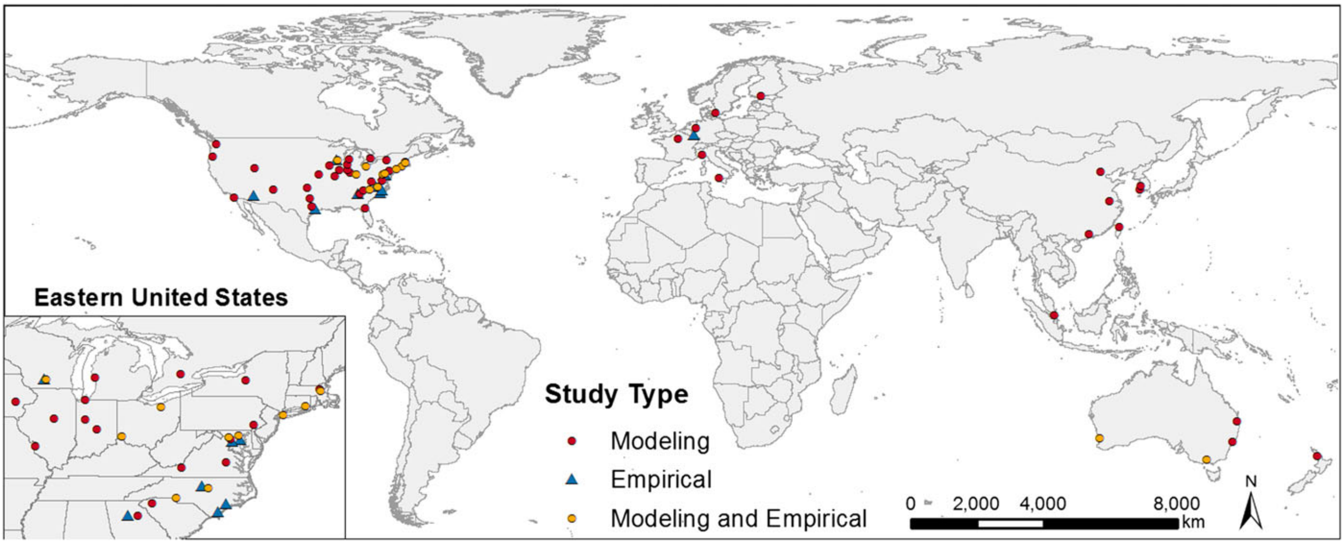

FIGURE 1 Location of empirical and modelling studies included in this review

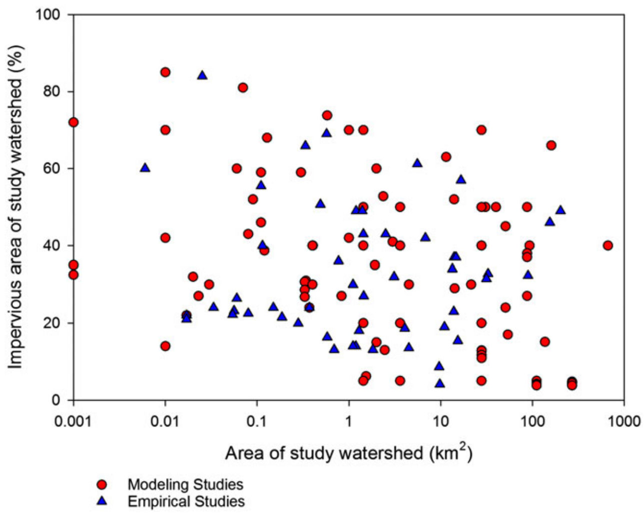

FIGURE 2 Total impervious area (%) versus logarithmic-scale watershed area (km^2) for watersheds where the effectiveness of cumulative stormwater management was studied (see Table 1)

3.2 | Different approaches used to study effects of SCM networks

On the basis of our literature review, both empirical and modelling approaches have been used to investigate the cumulative effectiveness of SCM networks at a watershed scale, although we found modelling studies to be more widely used. Modelling approaches have been used in 41 study locations, whereas 10 locations have been the focus of purely empirical studies, and 13 locations have had both empirical and modelling approaches deployed (Table 1). Empirical approaches are crucial to observe what actual effect SCMs are having on watershed function. Yet observational monitoring can only be used where SCMs have been installed at sufficient density such that their cumulative effects can be detected. A complementary approach is to use numerical experiments, in which modelling simulations are used to vary SCM density, type, or location, to expand the range of SCM, flow, and watershed conditions where we can gain information on cumulative SCM effectiveness, match sources with mitigation, and define thresholds in flow and water quality response.

We found that empirical studies used several different designs to examine the effects of SCM networks and can be categorized into two main study designs: (a) comparisons over time as SCMs are added to a watershed and (b) comparisons over space between two or more watersheds with varying levels of SCM implementation (Table 1). Each of these two main designs then has several subtypes of approaches, and some sites have had both comparisons over time and over space.

Studies over time can be longer term in duration as SCMs are implemented in a watershed. We defined long-term studies as those with monitoring periods of 4 years or longer. Most long-term study locations in the review (4 out of 5 locations) paired the watershed undergoing SCM implementation with one that has not undergone any changes to distinguish the effect of SCM implementation from climatic variability over longer monitoring periods. If the comparison between watersheds undergoing SCM implementation with an urban watershed with limited SCMs is established before SCMs are added, this approach is called before-after-control-impact. If an additional undeveloped watershed is included for comparison, it is called before-after-control-reference-impact experimental design. Most SCM implementations studied over time are in urbanizing landscapes, although a few studies of intensive retrofit efforts have been conducted (Jarden et al., 2016; Shuster & Rhea, 2013; Walsh, Fletcher, Bos, & Imberger, 2015). Detecting the effects of SCM implementation over time is difficult because other changes occurring in watersheds may mask the effect of SCMs (e.g., changes to vegetative, topographic, and soil characteristics; import and export of water; street repair). The second design type is more common than observing changes over time directly and substitutes space for time, where two or more watersheds with differing levels of SCM implementation are compared. These are often synoptic studies that are short in duration and occur after SCMs are implemented (e.g., 1–2 years; Meierdiercks et al., 2010; Hale et al., 2015; Fanelli et al., 2017). Comparisons of two watersheds (a paired approach) were more common (13 locations) than comparisons across a gradient of SCM implementation (four locations). Space for time substitutions have similar potential difficulties as any paired watershed study, where other factors that differ between watersheds could affect hydrologic or water quality response. However, they are useful in cases where long term or before-after-control-impact designs are not feasible. Ways to analyse hydrologic data to isolate the effect of SCMs in monitoring studies of various experimental designs are discussed in Li et al. (2017).

Most modelling studies explore effectiveness of SCM networks by combining different types and layouts across the landscape (Elliott & Trowsdale, 2007; Hamel et al., 2013). A common practice is to simulate predevelopment or current development conditions (baseline) and then sequentially add SCMs until a certain stormwater management goal is met. A variety of model types has been used that simulate hydrologic flowpaths and pollutant removal efficiency with a range of numerical approaches (e.g., process-based vs. stochastic). For example, the stormwater management model (Rossman & Huber, 2016) represents each SCM by a combination of vertical layers, with Manning's equation-based overland flow, a Green–Ampt infiltration model, soil properties, and underdrain characteristics. Watershed-scale models that use the Soil Conservation Service (SCS) curve number (United States Department of Agriculture, 1986) approach (e.g., LTHIA-LID) simulate SCM function by modifying the curve number to lengthen travel time and increase initial abstraction. Hydrological modelling has been also used to optimize type, location, size, and cost of SCMs (Baek et al., 2015; Endreny & Collins, 2009; Gilroy & McCuen, 2009; Liu et al., 2016; Xing et al., 2016). More complex numerical models (e.g., HYDRUS and FEFLOW), based on Richards' unsaturated flow model coupled with soil hydraulic functions, are able to describe subsurface flow; however, the amount of input data and computational effort exceed that of approaches described above. Pollutant removal by SCMs is most commonly simulated using data-driven removal percentages (e.g., LTHIA-LID) or first-order decay (e.g., MUSIC and SUSTAIN). Limitations of modelling studies are generally in the physical description of surface and subsurface interactions, and sewer-storm drainage system infiltration and exfiltration (Salvadore, Bronders, & Batelaan, 2015), as well as in the high treatment intensity implemented and the lack of confounding factors modelled (Li et al., 2017).

3.3 | Detention-based SCM networks

A majority (36 of 65) of the studied locations included detention-based SCM networks (Table 2), though often in conjunction with harvest- and infiltration-based SCMs. Detention-based SCMs were the sole focus of work in 10 locations. Most of these studies focused on aspects of peak flow and run-off volume, including water yield and run-off ratios. Despite the prevalence of detention SCM networks constructed over the last 30–40 years, some aspects of hydrologic response to these systems remain relatively understudied. For example, few of these studies quantified the effects of detention storage on recession coefficients and baseflow. Studies in arid and semi-arid regions are particularly scarce.

Empirical studies of detention-based SCM networks have shown mixed results. Several studies were able to detect a clear effect of detention-based SCMs on some hydrologic metrics. For example, detention basins were found to substantially increase streamflow for several days after storms in North Carolina, United States, reflecting their designed slow release of stored water (Jefferson et al., 2015). In an arid context where urbanization has decreased flooding from predevelopment Phoenix conditions, Hale et al. (2015) found that retention basins decreased run-off volumes. Other studies reported mixed results. In Baltimore, Maryland, United States, a watershed with a high density of SCMs was found to have lower annual run-off compared to an urban watershed with little stormwater management, but annual run-off was still higher than in a forested watershed (Meierdiercks et al., 2010). Bell et al. (2016) found that total impervious area, rather than SCM implementation, was the best predictor of peak flows and run-off ratio at the event scale. However, SCM implementation mitigation was a significant predictor over annual timescales.

Modelling approaches have also been used to examine cumulative effectiveness of detention-based SCM networks. Booth and Jackson (1997) found that detention basins in the Pacific Northwest, United States, were able to reduce peak flows, but that flow volume and duration were not able to be controlled by detention basins. Near Baltimore, Maryland, United States, Smith et al. (2015) modelled the detention basin network of the Dead Run watershed and found that detention basins reduced peak flows by a median of 11%, whereas an earlier study of stream gage data suggested that the basins may have lowered water yield by increasing evaporation (Nelson, Smith, & Miller, 2006). However, other modelling studies have found that detention basins may increase peak flows where changes in flow timing leads to synchronization from different parts of the watershed (Emerson et al., 2005; McCuen, 1974, 1979). The non-additive nature of SCMs means that observing effectiveness on a small scale does not mean this same effectiveness will translate to larger watershed scales. For example, McCuen (1979) found that peak flows were reduced for small storms at small drainage areas, but peak flows increased further downstream.

3.4 | Infiltration- and harvest-based SCM networks

Partly in response to lack of evidence that detention basins alone can ameliorate alterations to urban hydrologic function, SCM networks that focus on infiltration and water harvest (via evapotranspiration and reuse) are coming into wider use (Table 1). Perhaps because these technologies are newer, more hydrologically complex, and distributed throughout upland areas, there are more study locations (52 of 65) focused on cumulative effects of infiltration- and harvest-based SCM networks than of detention-based networks (Table 1). At half (26) of these locations, detention-based SCMs have been included in the studied watersheds.

Several empirical studies show reductions in peak flow from infiltration and harvest SCM networks (Bedan & Clausen, 2009; Jarden et al., 2016; Wilson et al., 2014), with one study documenting an order of magnitude decrease in median peak flow (Wilson et al., 2014). Run-off thresholds, or the minimum rainfall depth required to generate run-off, were higher in watersheds implemented with infiltration and harvest focused SCM networks than in watersheds with direct conveyance or detention-based SCMs (Fanelli et al., 2017; Hood et al., 2007; Loperfido et al., 2014). However, in watersheds implemented with the same SCM design, run-off thresholds decreased with greater

TABLE 2 Empirical and modelling studies that reported the cumulative effects of SCMs on hydrologic metrics: Peak flows, flow volume (at time scales shorter than annual), annual water yield (also called annual total streamflow volume), run-off ratio, time to peak, duration of stormflow, baseflow, recharge, groundwater elevation, evapotranspiration, and other

Site	Source	Peak flow	Flow volume	Annual water yield	Run-off ratio	Time to peak	Duration of stormflow	Baseflow	Recharge	Groundwater elevation	Evapotranspiration	Other
1	Fanelli et al. (2017)					x	x	x				x
2	Lee et al. (2012)	x										
3	Carter and Jackson (2007)	x	x									x
4	Elliott et al. (2009)	x						x				x
5	McCuen (1974)	x										
5	Gilroy and McCuen (2009)	x	x									x
5	Meierdiercks et al. (2010)	x	x	x	x							x
5	Newcomer-Johnson et al. (2014)		x									x
5	Smith et al. (2015)	x	x			x						x
5	Duan et al. (2016)		x									x
6	Pennino et al. (2016)						x	x				x
7	Jia et al. (2012)	x	x									
7	Liu, Bralts and Engel (2015)		x		x							
7	Xing et al. (2016)	x	x									x
8	Ahiablame and Shakya (2016)			x								
9	James and Dymond (2012)	x	x			x						
10	Thomas and Vogel (2012)									x		
11	Fletcher et al. (2007)	x	x					x				x
11	Aryal et al. (2016)	x	x				x					x
12	Line and White (2015)	x			x							
13	Gagrani et al. (2014)	x	x		x							
13	Jefferson et al. (2015)	x	x									x
13	Bell et al. (2016)	x		x	x							x
14	Shuster et al. (2007)		x						x		x	x
14	Shuster and Rhea (2013)	x	x			x	x					x
15	Hogan et al. (2014)			x				x				x
15	Loperfido et al. (2014)	x	x		x			x				x
15	Rhea et al. (2015)											x
15	Bhaskar, Hogan, Archfield (2016)		x	x				x			x	x
15	Hopkins et al. (2017)	x	x			x	x					x
16	Williams and Wise (2006)	x	x			x	x	x				x
17	Damodaram et al. (2010)	x	x									
18	Roldin et al. (2012)											x
19	Selbig and Bannerman (2008)	x	x	x	x	x						x

(Continues)

TABLE 2 (Continued)

Site	Source	Peak flow	Flow volume	Annual water yield	Run-off ratio	Time to peak	Duration of stormflow	Baseflow	Recharge	Groundwater elevation	Evapotranspiration	Other
20	Giacomoni et al. (2014)	x					x					
21	Guan et al. (2015)	x	x									x
22	Wu et al. (2015)	x		x			x	x				x
23	Palla and Gnecco (2015)	x	x									x
24	Hur et al. (2008)	x	x		x							
25	Jia et al. (2015)	x		x								
27	Aulenbach et al. (2017)	x	x									x
28	Yang and Li (2013)				x							
29	Ahiablame et al. (2013)			x								
29	Liu, Chen and Peng (2015)		x									x
30	Holman-Dodds et al. (2003)	x	x	x	x				x		x	x
32	Lee et al. (2012)			x								
33	Booth and Jackson (1997)	x					x					
34	Zimmer et al. (2007)	x	x	x			x	x			x	x
35	Wright et al. (2016)			x								x
36	Fletcher et al. (2007)	x	x					x				x
36	Burns et al. (2010)	x	x				x					x
36	Hamel and Fletcher (2014)		x		x	x		x				x
36	Burns et al. (2016)				x							
38	Brander et al. (2004)		x									x
38	Gebert et al. (2012)	x		x				x				
39	McCuen (1979)	x	x			x	x					x
41	Zhang et al. (2016)		x					x				
42	Ku et al. (1992)				x				x	x		
43	Lucas and Coombes (2009)	x	x							x		x
44	Stephens et al. (2012)								x	x	x	
45	Zahmatkesh et al. (2015)	x		x								
48	Petrucci et al. (2012)	x				x						x
48	Petrucci et al. (2013)	x	x									
48	Versini et al. (2016)	x	x									x
49	Jarden et al. (2016)	x	x			x						x
49	Avellaneda et al. (2017)	x		x								
50	Appleyard (1995)								x			x
50	Barron et al. (2013)		x	x	x		x		x	x	x	x

(Continues)

TABLE 2 (Continued)

Site	Source	Peak flow	Flow volume	Annual water yield	Run-off ratio	Time to peak	Duration of stormflow	Baseflow	Recharge	Groundwater elevation	Evapotranspiration	Other
50	Locatelli et al. (2017)									x		
51	Emerson et al. (2005)	x										x
51	Maimone et al. (2011)								x	x		
52	Hale et al. (2015)	x	x		x		x					x
53	Pomeroy et al. (2008)	x					x					x
53	Wilson et al. (2014)	x	x		x							x
54	Göbel et al. (2004)		x		x				x	x	x	x
55	Lucas and Sample (2015)	x	x									x
56	Di Vittorio and Ahiablame (2015)			x								
57	Feng et al. (2016)			x							x	
58	Walsh et al. (2014)	x		x								
59	Trinh and Chui (2013)	x		x		x		x	x			
60	Endreny and Collins (2009)									x		
61	Kessler et al. (2012)	x	x	x	x			x		x	x	x
63	Hood et al. (2007)	x	x		x	x	x					x
63	Dietz and Clausen (2008)		x	x								
63	Bedan and Clausen (2009)	x	x	x								x
63	Rosa et al. (2015)	x	x		x							x
64	Page, Winston, Mayes, Perrin, and Hunt (2015)				x							
64	Winston, Page et al. (2013)	x	x		x	x						x
65	Perez-Pedini et al. (2005)	x										

Note. SCM = stormwater control measure.

impervious cover, suggesting a decreased SCM benefit in watersheds with high impervious cover (Fanelli et al., 2017). Studies investigating run-off volume after implementation of harvest and infiltration SCMs found little change (Dietz & Clausen, 2008; Line & White, 2015; Selbig & Bannerman, 2008), or small but statistically significantly decreased run-off volume (Shuster & Rhea, 2013). Water yield (mean annual streamflow) was found to be lower in low impact development (LID) watersheds, with infiltration- and harvest-based SCM networks, compared to direct conveyance or detention-based SCM watersheds (Bedan & Clausen, 2009; Bhaskar, Hogan, Archfield, 2016; Hogan et al., 2014), but still higher than forested watersheds (Bhaskar, Hogan, Archfield, 2016; Hogan et al., 2014). Where discharge measurements were taken only within the storm sewer network, infiltration and harvest SCMs were shown to decrease run-off volumes (Avellaneda et al., 2017; Jarden et al., 2016), but the measurement location would not account for longer, deeper flowpaths that contribute to baseflow.

In modelling studies, peak flow and volume are consistently found to decrease with harvest and infiltration SCMs across climates (Avellaneda et al., 2017; Brander et al., 2004; Feng et al., 2016; Gilroy & McCuen, 2009; Holman-Dodds et al., 2003; Perez-Pedini et al., 2005). However, in some situations, peak flows could be exacerbated with wet antecedent conditions and use of infiltration SCMs (Williams & Wise, 2006). Hydrological effects of SCMs increased linearly with effective impervious area reduction, as modelled by scenarios implementing different densities of green roofs and permeable pavement (Palla & Gnecco, 2015). Furthermore, simulation results revealed that a minimum of 5% effective impervious area reduction was required for an SCM network to provide any noticeable hydrologic benefit (e.g., a 3% reduction in peak flow but no changes in run-off volume), which is equivalent to replacing 16% of the watershed's parking and road surfaces with permeable pavement (Palla & Gnecco, 2015). However, the implementation of infiltration-based SCMs on at least 11% of effective impervious areas (equivalent to 16% roads and 20% roofs) were required to reduce peak flows by 10% and run-off volumes by 5%. In San Diego, California, United States, surface run-off volume reduction increases linearly with the percentage of impervious area targeted by a rainwater harvesting scenario (Walsh et al., 2014). Also, available storage capacity and hydrological performance showed a linear increasing relationship in that study. Other studies have revealed non-additive effects of SCM networks. Numerical simulations have shown greater run-off volume reduction when SCMs were implemented near a watershed's outlet (Di Vittorio & Ahiablame, 2015). Perez-Pedini et al. (2005) argued that factors such as soil infiltration properties, land use, watershed network connectivity, upstream contributing area, and distance to stream channel are factors that influence the ability of SCMs to control run-off; however, these relationships are complex and unlikely to be explained by most urban hydrological models.

Infiltration- and harvest-based SCM networks have also been shown, via models, to affect multiple water balance components. For example, when green roofs and bioretention cells were modelled in an urban catchment in Salt Lake City, Utah, United States, run-off volume was reduced, and actual evapotranspiration was increased for an average weather year and when compared to baseline developed condition (Feng et al., 2016). Using a calibrated and validated stormwater management model for both development and stormwater treatment conditions, Avellaneda et al. (2017) quantified a reduction of surface run-off and an increase in infiltration for a catchment equipped with bioretention cells, rain gardens, and rain barrels. Although the combination of all types of SCMs led to larger changes in water balance components than any components individually, bioretention cells outperformed the cumulative effects of rain gardens and rain barrels.

Some studies have explicitly focused on changes to groundwater recharge and baseflow as affected by infiltration-based SCM networks. In an empirical study with infiltration SCMs, baseflow and total streamflow increased significantly during development, as vegetative cover decreased, compared to a detention-based urban watershed and a forested watershed (Bhaskar, Hogan, Archfield, 2016). In the same watershed, streamflow recessions were more gradual after urbanization with infiltration-based SCMs compared to during urbanization. In another small watershed, groundwater levels were found not to significantly increase over time as urbanization with infiltration facilities took place, although stormflow became better controlled (Kessler et al., 2012). Although an infiltration-based SCM in Annapolis, Maryland, United States, did intercept run-off for small rainfall events, baseflow in the stream was significantly lower than in forested reference streams; there was no difference in baseflow between the watershed implemented with SCMs and an adjacent urban watershed with no SCM implementation (Fanelli et al., 2017). Infiltration SCMs implemented in Boston were found to raise water tables in a small but significant way (Thomas & Vogel, 2012). Modelling simulations showed that increased infiltration SCMs could lead to greater recharge (Endreny & Collins, 2009; Göbel et al., 2004; Holman-Dodds et al., 2003; Maimone et al., 2011), although results were mixed on the magnitude of these changes on a watershed scale for baseflow (Hamel & Fletcher, 2014) and evapotranspiration (Holman-Dodds et al., 2003). A few studies simulated interactions between greater infiltration and combined sewer systems (Endreny & Collins, 2009; Maimone et al., 2011; Roldin et al., 2012). In Perth (Western Australia), implementation of extensive stormwater infiltration led to a rise in groundwater levels, which could potentially cause seepage above terrain; an increase in baseflow; and higher localized evapotranspiration rates due to the rise in groundwater levels in non-urban areas (Barron et al., 2013; Locatelli et al., 2017).

3.5 | Cumulative effects of SCMs on water quality

Results from the majority of studies suggest that implementation of SCMs reduces mass export of dissolved (e.g., soluble reactive phosphorus [SRP] and nitrate) and particulate (e.g., total suspended solids and total phosphorus) pollutants and that the primary mechanism underlying these patterns is hydrological rather than biogeochemical (Table 3). Monitoring SCM implementation over time at the watershed scale, for both detention- and infiltration-based networks, shows that pollutant load reductions are frequently tied to reductions in run-off generation (Ahiablame et al., 2013; Dietz & Clausen, 2008; Hale et al., 2015; Selbig & Bannerman, 2008; Steinman et al., 2015). For example, Bedan and Clausen (2009) observed significant reductions to peak discharge and total flow volume in an LID watershed, which translated to load reductions for nitrate (NO_3^-) and total Kjedahl

TABLE 3 Empirical (E) and modelling (M) studies that reported the cumulative effects of SCMs on water quality

				Load				Event mean concentration				
Site	Source	Study type	Annual/event scale	Diss. Nutrients	Part. Nutrients	TSS	Metals, bacteria, and other	Diss. Nutrients	Part. Nutrients	TSS	Metals, bacteria, and other	Baseflow concentration
4	Elliot et al. (2009)	M	A			x				x		x
5	Newcomer-Johnson et al. (2014)	E	E	x								x
5	Duan et al. (2016)	E	E	x	x		x					x
6	Pennino et al. (2016)	E/M	A	x	x							
10	Hurley and Forman (2011)	M	A		x							
11/36	Fletcher et al. (2007)	M	A		x	x						
12	Line and White (2015)	E	A	x	x	x						
13	Gagrani et al. (2013)	M	A/E		x	x			x	x		
13	Bell et al. (2017)	E	E					x			x	x
14	Roy et al. (2014)	E	A									x
15	Hopkins et al. (2017)	E	E					x	x	x		x
15	Sparkman et al. (2017)	M	A		x	x						
19	Selbig and Bannerman (2008)	E	A		x	x						
24	Hur et al. (2008)	E	E			x				x		
26	Baek et al. (2015)	M	E							x		
28	Yang and Li (2013)	E	A									x
29	Ahiablame et al. (2013)	M	A									
29	Liu, Chen and Peng (2015)	M	A	x	x	x	x					
31	Gold et al. (2017)	E	A/E					x		x	x	x
32	Lee et al. (2012)	M	A			x						
36	Burns et al. (2016)	E	A		x	x			x	x		
37	Liu et al. (2016)	M	A	x	x	x						
38	Gebert et al. (2012)	E	A		x	x			x	x		
40	Steinman et al. (2015)	M	A		x	x						
46	Martin-Mikle et al. (2015)	M	A		x	x						
47	Freni et al. (2010)	M	E			x						
52	Hale et al. (2015)	E	E	x	x	x	x	x	x	x	x	

(Continues)

TABLE 3 (Continued)

Site	Source	Study type	Annual/event scale	Load				Event mean concentration				Baseflow concentration
				Diss. Nutrients	Part. Nutrients	TSS	Metals, bacteria, and other	Diss. Nutrients	Part. Nutrients	TSS	Metals, bacteria, and other	
53	Wilson et al. (2014)	E	E	×	×	×		×	×	×		
62	Chen et al. (2014)	M	A		×	×	×					
63	Dietz and Clausen (2008)	E	A	×	×							
63	Bedan and Clausen (2009)	E	A	×	×	×	×	×	×	×	×	
63	Rosa et al. (2015)	M	A/E		×							
64	Winston, Page et al. (2013)	E	E	×	×	×	×	×	×	×	×	
64	Page et al. (2015)	E	E	×	×	×	×	×	×	×	×	

Note. Pollutant loads and event mean concentrations (EMC) are separated into dissolved and particulate nutrients, total suspended sediments (TSS) and metals, bacteria, and others (e.g., chlorophyll *a*, biochemical oxygen demand, and organic contaminants).

nitrogen, compared to a watershed with no SCMs. However, the effect was not consistent among solutes, as they observed increased SRP loads and total suspended sediments (TSS) loads. Similarly, a study in Raleigh, North Carolina, United States, observed 11-fold decreases in peak discharge in a watershed with distributed, infiltration-based SCMs compared to a watershed with centralized, dry detention; however, no differences were detected in event mean concentrations (EMCs) (Wilson et al., 2014). In arid urban watersheds, nutrient and dissolved organic carbon fluxes decreased with retention basin density and increased with imperviousness (Hale et al., 2015). However, these patterns were not observed in concentrations that support the finding across studies of hydrology driving water quality. This suggests that reductions in run-off volume may have a larger impact on pollutant loads than treatment within the SCM itself at the catchment scale. However, more work is needed to understand the impacts of run-off storage on treatment processes and delayed release on instream concentrations (Bell et al., 2017; Jefferson et al., 2015).

The TP export was the most variable water quality metric, with both increases (Selbig & Bannerman, 2008) and decreases in export (Bedan & Clausen, 2009; Dietz & Clausen, 2008; Line & White, 2015; Wilson et al., 2014) reported in empirical studies. For example, retention of TP at the watershed scale was primarily attributed to high density of SCMs (such as bioretention areas and ponds) that are highly retentive of sediment and associated pollutants (Davis, 2007). Although stormwater ponds are often considered effective sinks of TSS and attached pollutants (Hogan & Walbridge, 2007), in some cases, they may also generate solids through algal production (Gold et al., 2017) or be modified with floating islands to enhance removal of nutrients and metals (Borne, Fassman, & Tanner, 2013; Winston, Hunt et al., 2013). Monitoring studies of the effectiveness of individual SCMs revealed considerable variability among SCM types and pollutants. Physical and biological processes that drive pollutant retention can be enhanced in SCMs, but variability in design affects residence times and rates of retention and removal processes (Reisinger, Groffman, & Rosi-Marshall, 2016; Zhu, Dillard, & Grimm, 2005). Although the potential exists for these structures to achieve even greater water quality improvements than predicted by run-off reductions alone, demonstration of this at the watershed scale is still lacking.

Modelling studies typically conduct simulations over annual or multi-year timescales to account for the effects of climate variability and across stormwater implementation scenarios. For example, Gagrani et al. (2014) used the MUSIC model to show significant reductions in total nitrogen, TP, and TSS loads in simulations with ponds and bioretention areas compared to piped drainage. When distributed rain gardens at the household level were added to the simulation, little further reductions were observed. Models have also been used to optimize pollutant removal and cost (Liu, Chen & Peng, 2015) and to determine most acceptable areas for siting of SCMs based on biophysical and societal constraints (Lee et al., 2012; Martin-Mikle et al., 2015; Steinman et al., 2015).

Localized conditions and temporal variability at the small watershed scale can have significant effects on both run-off generation and SCM function. During construction, low infiltration capacity through compaction of soils can lead to decreased functionality of infiltration SCMs (Line & White, 2015), and increased fertilization to establish vegetation can lead to increased dissolved nutrient export (Bedan & Clausen, 2009; Line & White, 2015). Over time, short- and long-term changes in vegetation in SCMs, riparian zones, and other forested urban areas can be a sink for nitrogen through uptake and denitrification (Bettez & Groffman, 2012) but a source of phosphorus via leaf litter decomposition (Bratt et al., 2017; Selbig, 2016). These studies highlight the importance of matching stormwater controls with pollutant sources. For example, rain gardens in residential areas may improve hydrologic response, but they often treat rooftop run-off, which has low pollutant concentrations compared to run-off from lawns and roads (Roy et al., 2014). Similar to hydrologic controls, type of development within the watershed and position with the stream network may influence changes in water quality. For example, in older urban watersheds, Bell et al. (2017) showed that SCM outflow concentrations of dissolved nutrients were lower than instream, implying that SCMs have potential to decrease stream concentrations if sufficient run-off is captured and treated.

Temporal variability is best seen in effectiveness of SCMs during large storms, which can have a disproportionate effect on annual pollutant export (Bell et al., 2016; Shields et al., 2008). In a 5-year paired watershed study in Wisconsin, United States, precipitation from small, frequent events generated run-off from a direct conveyance watershed, whereas run-off was retained in the LID watershed (infiltration SCMs and grassed conveyance), and large storms during either winter months with frozen soils or saturated conditions led to increases in export (Selbig & Bannerman, 2008).

3.6 | Common confounding factors

Two broad categories of confounding factors, built and natural environment, were identified as potential drivers of variability in hydrologic and biogeochemical response to watershed-scale SCMs. Confounding factors associated with the built environment included presence, density, function, and connectivity of impervious cover and SCMs as well as SCM arrangement (centralized vs. distributed), age of development, and past land use. Variability in these factors complicated interpretation of hydrologic and biogeochemical results. Confounding factors associated with the natural environment included differences in tree canopy cover, grass cover, fertilizer application rates, fall leaf off, local geology and soils, and heterogeneity in precipitation amount and intensity among study sites. These confounding factors were most often acknowledged in studies using a paired watershed design. Differences in canopy cover among study watersheds were cited as a natural factor influencing annual water yield (Bell et al., 2016; Bhaskar, Hogan, Archfield, 2016; Loperfido et al., 2014). Differences in grass cover and fertilizer application rates were cited as the primary explanation for variability in TP export, particularly if the distributed SCM watershed exported more TP than the direct conveyance watershed (Bedan & Clausen, 2009). Local geology and soils were referenced as factors associated with hydrologic trends related to groundwater recharge flowpaths and infiltration rates (Brander et al., 2004; Holman-Dodds et al., 2003).

4 | HYSTERESIS RESULTING FROM URBANIZATION AND STORMWATER MANAGEMENT

We find that although the goal of stormwater management is broadly to restore predevelopment hydrology and water quality, in practice, this goal is rarely achieved. Instead, stormwater management shifts the system onto a new trajectory, which may not return it to the initial predevelopment state, a system behaviour often referred to as hysteresis. We posit that trajectories of watersheds undergoing development and subsequent SCM mitigation could be conceptualized with hysteresis loops for metrics describing water quantity or quality. In the hysteresis loops, the relationship between unmitigated impervious area and each hydrologic and water quality metric is shaped by watershed capacitance, local climate, and non-stormwater processes of urbanization (e.g., vegetation changes; water import). In Figures 3, 4, the brown line represents the expected trajectory of each metric if there are no SCMs installed and all run-off is delivered to the stream untreated. On the basis of the studies described above (e.g., Bell et al., 2016, 2017; Palla & Gnecco, 2015; Roy et al., 2014), treating a small portion of the impervious area often fails to produce detectible water quantity or quality changes for particular metrics. Purple lines in Figures 3, 4 represent the installation of SCMs in the watershed, with the length of the line representing the degree of SCM installation necessary to trigger a shift in the water quantity or quality response. The degree of hysteresis is a function of types of SCMs, their design standards, how well the SCMs perform relative to those standards, and the SCM network arrangement. In Figure 3, the three blue lines show how the trajectory and shape of the hysteresis loop vary depending on SCM function (i.e., detention, infiltration, and harvest). From a pollutant reduction perspective, this will also vary by pollutant type (e.g., dissolved, particulate, biologically reactive, and conservative), as shown in Figure 4. Although hysteresis can be most easily visualized for a watershed that urbanizes without stormwater management and then has an SCM network retrofit into the landscape (e.g., Roy et al., 2014), the concept can also be applied to new developments where SCM networks are built at the time of urbanization. In that case, we cannot measure the whole hysteresis loop but simply see residual between starting (pre-urbanization) conditions (brown line at 0% impervious cover) and ending conditions as a combined result of urbanization and stormwater management (blue-green line at the final untreated impervious cover).

A simple example of a hysteresis loop as a result of stormwater management is 2-year peak flows in streams (Figure 3A). As watersheds are urbanized, impervious surfaces increase peak flows (e.g., Leopold, 1968; brown line), and SCM networks decrease them (e.g., Holman-Dodds et al., 2003; blue line). However, limited storage capacities and overflow bypass systems of SCMs impart limitations in peak flow reduction beyond a design storm size (Water Environment Federation and American Society of Civil Engineers, 2012). For example, infiltration-based SCMs in the United States are often designed to capture the first inch of run-off for water quality purposes. Additional run-off beyond the SCM storage capacity bypasses it and is transported directly to the stream. As a result, if a fraction of impervious surface in a watershed is treated by SCMs, peak flows decrease

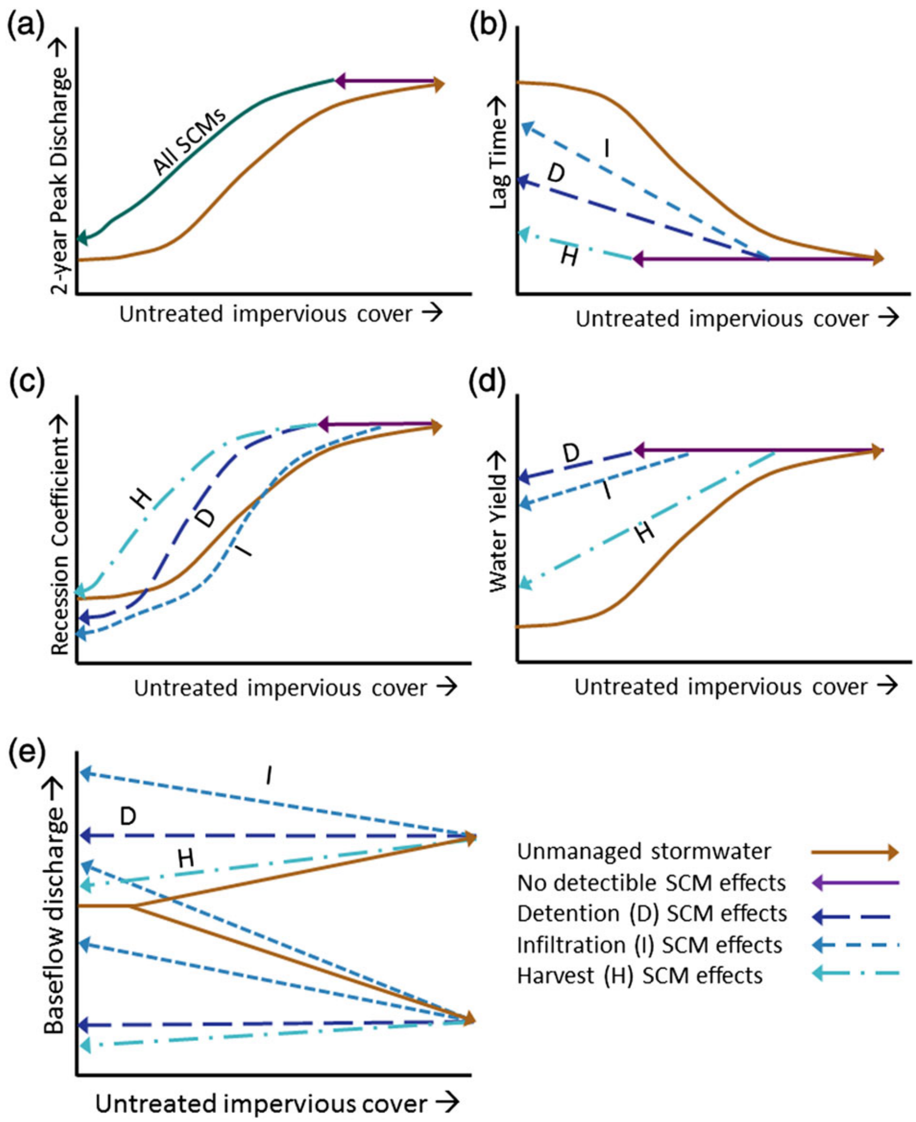

FIGURE 3 Conceptual model of urbanization and cumulative stormwater management effects on hydrologic response. The brown arrow represents the effects of urbanization processes where stormwater is uncontrolled. The purple arrow represents stormwater control measure (SCM) networks that treat too little impervious area for their effects to be detected. The blue arrows represent detectible effects of SCM implementation. Long dashed lines indicate the effects of detention SCMs, short dashed lines indicate the effects of infiltration SCMs, and dashed-dotted lines indicate the effects of harvest SCMs. The designed focus on peak flow mitigation across all SCM types means that their effect on peak flows is broadly similar, so is not broken out in (A). (A) Two-year peak discharge versus untreated impervious cover. (B) Lag time versus untreated impervious cover. (C) Water yield (total annual discharge) versus untreated impervious cover. (D) Discharge recession coefficient (where higher coefficients indicate faster hydrograph recessions) versus untreated impervious cover. (E) Baseflow discharge versus untreated impervious cover

but usually not by the extent they would if the treated impervious surface did not exist at all. Even if 100% of impervious area is treated, peak flows may remain higher than predevelopment flows. This concept is presented in Figure 3A where the *y*-intercept of blue line is greater than the brown line.

We hypothesize that the shape of the hysteresis loop depends on the functions supported by the SCMs. For example, lag times (e.g., between peak precipitation and peak discharge) tend to decrease with urbanization (Figure 3B), whereas recession coefficients increase (i.e., falling limb returns to baseflow more rapidly), because of higher drainage efficiency of engineered flowpaths (e.g., Leopold, 1968) (Figure 3C). Detention SCMs increase lag times by having a storage capacity that must be filled before release to the stream is maximized, and decrease recession coefficients by supplementing flow on the falling limb through slow release of stored water (e.g., McCuen, 1979). Infiltration-based SCM networks can lead to even greater delays between precipitation and arrival of SCM-treated stormwater at the stream by diverting water into slow subsurface flowpaths (e.g., Fanelli et al., 2017; Hood et al., 2007), resulting in more gradual recessions (e.g., Bhaskar, Hogan, Archfield, 2016). Increased retention time is the primary mechanism for sediment retention and therefore reduces export of sediment and associated pollutants (e.g., Hogan & Walbridge, 2007) but can also lead to increased production of algae in nutrient rich systems (e.g., Gold et al., 2017), thereby increasing particulate and dissolved organic matter (Figure 4). Conversely, harvest-based SCM networks would not be expected to have a large effect on lag times or recession behaviour until substantial impervious surface is treated, because their primary effect is to remove water from

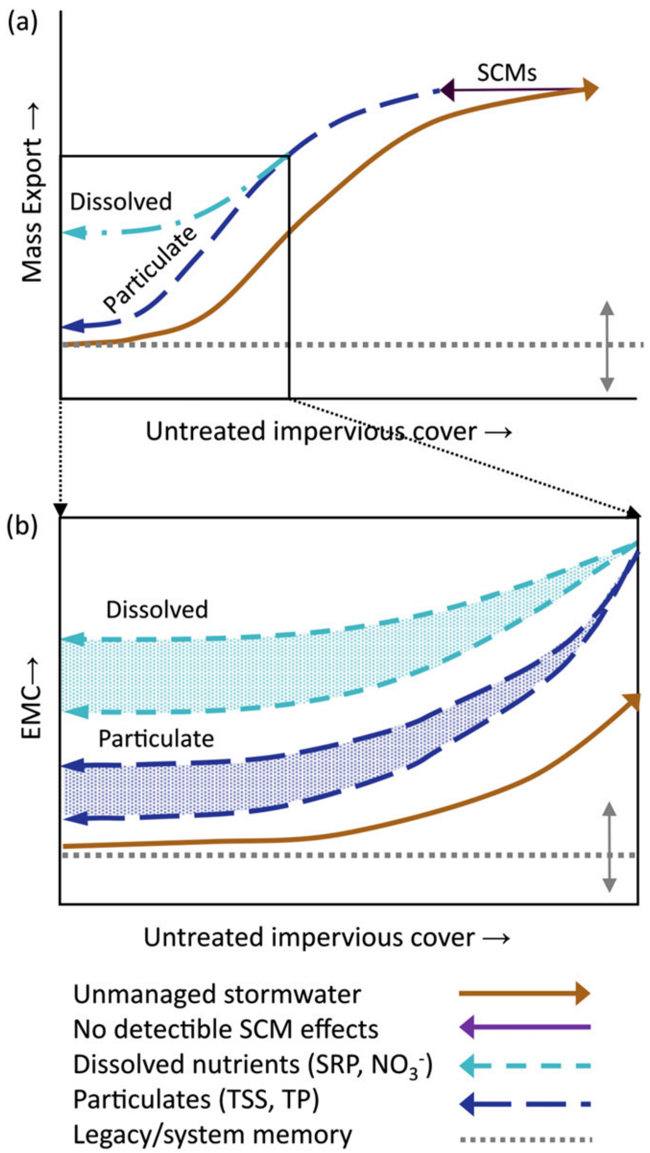

FIGURE 4 Conceptual model of urbanization and cumulative stormwater management effects on water quality response. The brown arrow represents the effects of urbanization processes where stormwater is uncontrolled. The purple arrow represents stormwater control measure (SCM) networks that treat too little impervious area for their effects to be detected. The teal and blue arrows represent detectible effects of SCM implementation. The short dashed lines show expected patterns for dissolved pollutants; long dashed lines show particulate pollutants. (A) Mass export versus untreated impervious cover. (B) Event mean concentration (EMC) versus untreated impervious cover, zoomed in on the portion of the curve where observable changes are expected

reaching the stream entirely, rather than redistribute it in time (Figure 3B,C). High intensity rainfall and wet antecedent conditions will shorten lag times, making conditions with an SCM network similar to those without treated stormwater (Hood et al., 2007).

A different pattern emerges when considering seasonal to annual water yields (Figure 3D) and event scale run-off ratios (not shown), which tend to increase with imperviousness. Detention-based SCM networks have limited capacity to decrease volumes delivered to receiving water bodies (e.g., Bell et al., 2016), although sufficiently large surface areas might slightly decrease water yields through enhanced evaporation (e.g., Nelson et al., 2006). Harvest-based SCM networks decrease run-off ratios at the event timescale and can decrease seasonal to annual water yield if evapotranspiration or water reuse is significant enough (Askarizadeh et al., 2015). Infiltration-based SCMs usually include some component of harvest via enhanced evapotranspiration, so they fall along an intermediate pathway but cannot alone return water yield to predevelopment conditions (e.g., Askarizadeh et al., 2015; Bhaskar, Hogan, Archfield, 2016; Hogan et al., 2014).

Although we can make general predictions of effects of different SCM network types on baseflow (Figure 3E), groundwater recharge, and evapotranspiration (not shown), it is currently challenging to predict how these processes will respond to urbanization for a given watershed or region, thus creating considerable uncertainty as to the starting point to measure SCM effects (Bhaskar, Beesley et al., 2016). If that conundrum can be sidestepped, then we expect detention-based SCM networks will not change baseflow or groundwater recharge, though they could increase evaporation (as with water yield). Infiltration-based SCMs will increase groundwater recharge and baseflow, whereas harvest-based SCMs will increase evapotranspiration. For SCM networks that involve both infiltration and harvest (e.g., bioretention cells), effects on baseflow, groundwater recharge, and evapotranspiration will depend on relative importance of the two processes (e.g., Hamel & Fletcher, 2014; Holman-Dodds et al., 2003).

Because mass export is a function of both volume and concentration, we expect that mass will mirror run-off and that loads will generally decrease as run-off is retained in the watershed (Figure 4A). We also expect that hysteresis patterns will vary for dissolved and particulate fractions because pollutant retention mechanisms are different. Many SCM design standards, particularly in the United States, focus on capturing the first inch of run-off. This is based on the concept of a "first flush" effect in which early run-off has higher concentrations of pollutants. This effect is strongest for particulates (e.g., TSS) and less for dissolved nutrients (NO_3^-, SRP) (e.g., Hogan & Walbridge, 2007); however, significant loading can occur later in the event (e.g., Hathaway, Tucker, Spooner, & Hunt, 2012). Because export later in the event is not well controlled with current SCM designs, we anticipate a lag in load reductions behind flow reductions. We also expect a higher baseline due to long-term storage and release of reactive solutes in SCMs and urban soils, inputs from septic and combined sewers, and continued bank erosion from degraded urban channels (Hopkins et al., 2017; Paul & Meyer, 2001).

Although the number of empirical studies is limited, a consistent finding was that little change was observed in EMCs at the watershed scale. Performance of individual SCMs shows potential for decreasing EMCs (Bell et al., 2017), but we anticipate that such decreases will not be observable until critical thresholds are reached (Figure 4B). The shape of the EMC response curve will depend upon biophysical processes within SCMs, the design type of SCM, storm size, and antecedent conditions, although considerably more data are needed to test these relationships. For example, as multiple storms occur in succession, the first event may flush pollutants from storage zones in the landscape (groundwater, riparian soils, and SCMs), followed by dilution in subsequent storms as sources are depleted. Additionally, SCMs designed for improved water quality (e.g., detention ponds with wetland vegetation) would cause different response than SCMs designed for flood control or water capture (Koch et al., 2014).

5 | MANAGEMENT IMPLICATIONS AND NEXT STEPS FOR RESEARCH

Hydrology of urban watersheds with managed stormwater differs from predevelopment conditions for several reasons. First, despite “treating” an impervious area by design, stormwater management is typically not designed with sufficient storage to mitigate all run-off from that area. Second, SCM effectiveness at the site scale is not simply additive to cumulative effectiveness of SCM networks at the watershed scale because of differential time lags, SCM interaction, inconsistent water quality mechanisms, and spatial arrangement. Last, impacts of urbanization beyond impervious surfaces continue to alter hydrology. Conceptually, it is possible to return to a predevelopment hydrologic condition (Askarizadeh et al., 2015) without the residuals illustrated in the above hysteresis curves if SCM networks perfectly mitigate all effects of urbanization.

Given designs of individual SCMs, an SCM network aiming to achieve predevelopment hydrologic function may have to include redundant SCMs to ensure no run-off is generated. Micro-scale source control SCMs (e.g., permeable pavement sidewalks and driveways, downspouts with dry wells, and streetside swales) that treat run-off directly where it is generated could be effective at treating the full volume of stormwater generated across a wide range of hydrologic conditions. Redundant and micro-scale SCM networks could be cost prohibitive in many areas, especially where stormwater control is retrofitted into existing urban areas. Such strategies may be easier to implement in new development, where SCM networks are explicitly included in the initial development. The concept of watershed capacitance should be expanded to include the ease with which an effective SCM network can be implemented, given available space, existing infrastructure, and natural landscape characteristics.

Regulatory frameworks or economic incentive structures would likely need to change to encourage widespread adoption of stormwater management strategies that emphasize redundant SCMs or micro-scale source control for all surfaces, as may be needed to attain predevelopment hydrologic and water quality conditions. Intensification of precipitation extremes in a globally changing climate may spur such stormwater management strategies as current approaches become insufficient to protect communities and infrastructure from flooding or minimize environmental degradation.

Review of existing literature on cumulative effects of stormwater management suggests several key areas where research is needed. Hysteresis curves (Figures 3–4) are based on available data and understanding of processes driving hydrologic and water quality response but ultimately serve as a suite of testable hypotheses for future research across regions with varying watershed capacitance. Because it is rare to be able to track a watershed along the full hysteresis pathway illustrated, creative approaches to modelling and empirical studies are needed to disentangle the magnitude and causes of hysteresis and residuals between predevelopment and managed stormwater conditions. The preponderance of studies in temperate and humid environments also means that a different set of hysteresis curves may be needed for tropical and arid or semi-arid regions, where existing literature is much sparser.

One obvious, but difficult, next step is that we need more empirical studies on cumulative SCM effectiveness, particularly for watersheds that are large (>20 km^2) and have a high intensity of urbanization and SCM treatment. At these larger scales, commonly used modelling approaches tend to simplify reality to such an extent that important nuances may be missed, because including all processes would result in complex models, with high data needs and computing requirements. For empirical studies, it is difficult to identify where large, highly urban and highly treated watersheds exist, as larger watersheds tend to have low SCM treatment intensity (Bell et al., 2016). However, one of many institutional barriers to regional implementation of stormwater management is uncertainty in regional-scale performance of SCM networks, which can only be addressed by a larger number of regional-scale studies in a variety of settings (Roy et al., 2008).

This work identifies the need for a common set of response metrics reported across studies, enabling more robust meta-analyses. Across 89 studies of hydrologic response, there were 28 metrics reported, and across 34 studies of water quality, 17 different pollutants were reported at event and annual scales. Even where common metrics were used, reference conditions to which metrics could be compared were variable (undeveloped, direct conveyance, or detention SCMs), making quantitative comparisons difficult. Simultaneously, it is important to note that much more is known about peak flows, flow volumes, and water yields than other aspects of the hydrograph (e.g., recession) or water balance components (e.g., evapotranspiration and groundwater recharge), yet these hydrologic functions play important roles in influencing biogeochemical processes and ecosystem function. The paucity of empirical studies on water quality response highlights additional challenges including sample collection and analysis cost, logistical constraints, and highly variable responses. Advances in sensor technology (Rode et al., 2016) have great potential to rapidly advance our understanding of temporal dynamics, but these are currently often cost prohibitive.

Hydrological and water quality models that incorporate SCM effectiveness have great potential to enhance understanding of hysteresis effects shown in Figures 3 and 4. However, SCM model parameters are typically based on observational monitoring studies of single SCMs. As these are aggregated to the watershed scale, it is important to consider interacting effects of SCMs in series (e.g., treatment trains), age of practices, and effectiveness across a range of storm sizes and antecedent conditions. These issues are particularly important as hydrologic intensification is expected to increase.

This synthesis of 100 studies reveals broad patterns of SCM network effectiveness and supports current practices that emphasize infiltration and harvest SCMs rather than detention-based SCM networks. However, examining these studies revealed gaps in our understanding of how SCM networks can be implemented to restore hydrologic function in diverse climatic and urban settings at a variety of scales. Our findings also emphasize that instream improvements in water quality are often the result of run-off reduction, rather than biophysical processes happening within individual SCMs. Despite the body of work reviewed here, it is clear that there is significant need for research to fill these gaps and inform design and management of SCM networks and enhance protection of communities and aquatic ecosystems.

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