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Development of a holistic understanding of current technical, environmental and social/institutional knowledge with regard to urban stormwater research

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Preface

This report *Urban stormwater research: an evidence synthesis* presents the results of one of the two funded projects within the call *synthesis analysis on urban stormwater runoff* from 2022. The research results from this call aim to produce a knowledge base that will contribute to policy development in sustainable stormwater management.

The project has been financed with the environmental research grants from the Swedish Environmental Protection Agency (SEPA) to support the knowledge needs of SEPA and the Swedish Agency for Marine and Water Management.

This report is written by Lian Lundy (Luleå University of Technology), Helene Österlund (Luleå University of Technology), Hanna Fors (Swedish University of Agricultural Sciences), Alexandra Müller (Luleå University of Technology), Snezana Gavric (Luleå University of Technology), Thomas B. Randrup (Swedish University of Agricultural Sciences), and Maria Viklander (Luleå University of Technology).

The authors are responsible for the content of the report.

Stockholm, February 2025

Johan Bogren
Acting Director, Sustainable Development Department

Förord

Denna rapport med titeln: *Urban stormwater research: an evidence synthesis* presenterar resultaten av ett av två beviljade syntesprojekt inom utlysningen *Synteser om dagvatten* från 2022. Forskningsresultaten från denna utlysning syftar till att ta fram kunskapsunderlag som ska bidra till policyutveckling inom hållbar dagvattenhantering.

Projektet har finansierats med medel från Naturvårdsverkets miljöforskningsanslag till stöd för Naturvårdsverkets och Havs- och vattenmyndighetens kunskapsbehov.

Denna rapport är författad av: Lian Lundy (Luleå Tekniska Universitet), Helene Österlund (Luleå tekniska universitet), Hanna Fors (Sveriges lantbruksuniversitet), Alexandra Müller (Luleå tekniska universitet), Snezana Gavric (Luleå tekniska universitet), Thomas B. Randrup (Sveriges lantbruksuniversitet) och Maria Viklander (Luleå tekniska universitet).

Författarna ansvarar för rapportens innehåll.

Stockholm, februari 2025

Johan Bogren
Tf. Avdelningschef för Hållbarhetsavdelningen

Contents

Summary

This report is the final deliverable of the project 'Urban stormwater research: an evidence synthesis' funded by the Swedish Environmental Protection Agency and the Swedish Agency for Marine and Water Management. This report is accompanied by the first, second and third deliverables which focus on reviewing and synthesising the international peer-review literature on:

- impacts of stormwater runoff on receiving water recipients from a cold climate perspective (see Appendix A),
- design, treatment performance and maintenance requirements of sustainable stormwater management systems operating in a cold climate (see Appendix B)
- understanding co-benefits and overcoming barriers for increased societal acceptance of sustainable stormwater management (see Appendix C)

This report provides an overview of the key conclusions from each detailed appendix and maps their findings against the requirements and policy objectives set out in the 2024 EU Urban Wastewater Treatment Directive, the UN Sustainable Development Goals (2015) and the Swedish environmental goals (2000). This activity is undertaken to further clarify when, where and how sustainable stormwater management can contribute to policy requirements and – if well implemented – to delivering societal benefits, including sustainable human and environmental health and well-being objectives. Drawing on these assessments. this report concludes with a series of research and policy recommendations to support Sweden in its transition to the implementation of sustainable stormwater management as the default option.

1. Introduction

Sustainable stormwater management has received considerable attention in Sweden over the past 20 years (e.g. Lundy et al., 2021). The traditional approach of rapidly draining runoff to the closest water body is no longer fit for purpose as a component of either current or future city living. The use of pipes moves – rather than manages – stormwater quantity and results in the direct discharge of a cocktail of pollutants to receiving waters from sources including traffic, industry and building materials (e.g., Müller et al. 2020). Therefore, within city planning, stormwater management has evolved from the traditional focus on technical aspects to integrate a wider range of dimensions as the need to address social, ecological, and economic values were recognised (Stahre, 2004). For example, the implementation of water sensitive urban design (WSUD) approaches can require space for blue-green infrastructure (BGI) which can collide with numerous other urban infrastructure demands, such as those associated with transport, energy and information technology (Dall O', 2020). Further, the use of WSUD approaches in existing areas often involves both public and private land, thus requiring new governance approaches. In addition, legislation to improve existing stormwater management and related spatial planning is limited (Cettner et al., 2013; Qiao et al., 2018). Within a Swedish context, the Planning and Building Act (SFS 2010:900) only applies to new (as opposed to all) developments. A similar situation of limited scope existed with regard to the Swedish Water and Sanitation Act (SFS 2006:412). However, a recent amendment (Sveriges riksdag, 2022) required municipalities to integrate sustainable stormwater management into the planning of new developments/areas of major re-development by 2023, and to develop and commence sustainable stormwater management plans in priority existing developments by 2025. There are currently both major challenges and opportunities to support Sweden's transition to sustainable stormwater management as a core mechanism to deliver safe, flood-resilient urban environments (Cettner et al., 2014).

Within this context, this report summarises the peer-review literature related to the following topics:

- Cold climate-pertinent international research where studies have assessed the impacts of urban runoff (associated with both rainfall and snowmelt events) within surface and groundwater receiving waters (see Appendix A for full report)
- The design, implementation and long-term governance and management of stormwater treatment systems operating under cold climate conditions (see Appendix B for the full report)
- Understanding co-benefits and overcoming barriers for increased societal acceptance of sustainable stormwater management (see Appendix C for the full report).

Together this report and its accompanying appendices provide a concise evidence base on state-of-the art in stormwater research in relation to these three specific fields (see Chapter 3) For further information on the studies reviewed and review methodology, please refer to the Appendices). This report then maps key findings against the requirements and policy objectives set out in the 2024 EU Urban Wastewater Treatment Directive, the UN Sustainable Development Goals (2015) and the Swedish environmental goals (2000) (see Chapter 4). Based on these evaluations, this report concludes with a series of research and policy recommendations to support Sweden in its transition to a stormwater management approach as the default option (Chapter 5).

2. Methodology

2.1 Systematic review

For full review methodologies, please see Appendices A, B and C. However, in brief, the peer-review database SCOPUS was searched using the key words: stormwater or ‘storm water ‘or runoff to capture as many articles as possible within the targeted field in June 2023 (returned 133,504 hits). This longlist of articles was filtered using the term ‘urban’ (returned 42,124 hits). This common pool of articles was then searched using topic-specific key terms to identify a short-list of papers which were uploaded to the open-access systematic review software Rayyan (www.rayyan.ai) for screening of titles and abstracts. Following screening, a final sub-set of articles was reviewed and a short-list of papers identified for inclusion. The number of papers identified at each stage for each topic are as follows:

- Impacts on recipients: 1,014 articles screened; 202 articles manually reviewed; 77 articles included (see Appendix A)
- Design, treatment performance and maintenance requirements: 662 articles screened; 279 articles manually reviewed; 103 articles included (see Appendix B)
- Understanding co-benefits and overcoming barriers for increased societal acceptance of sustainable stormwater management: 597 articles screened; 193 articles manually reviewed; 61 articles included (see Appendix C)

2.2 Mapping of research findings to selected policy requirements and objectives

Discussions with the Expert Advisory Panel led to identification of the following three policy areas for review in relation to stormwater management linkages:

- EU Urban Wastewater Treatment Directive (2024)
- UN Sustainable Development Goals (2015)
- Swedish Environmental Goals (2000)

Drawing on the findings of Appendices A, B and C, each of the above policies were reviewed to clarify when, where and how sustainable stormwater management could contribute to policy requirements.

3. Overview of key research findings

3.1 Impacts of stormwater runoff in receiving water recipients from a cold climate perspective

3.1.1 Scope

Research relating to the impacts of stormwater runoff within surface water and groundwater recipients located in Nordic countries and those with a similar climate (defined on discussion with project advisory board as Canada and the Baltic countries) was reviewed (see Appendix A). Research studies were identified using a systematic approach and evaluated to address the following core question: how does stormwater runoff impact on the chemical quality, geomorphology and/or ecological status of surface waters and groundwaters? As the focus of this topic is impacts within receiving waters, studies which focused on the quality and impacts of runoff prior to its discharge to surface water were excluded. Use of agreed search terms (see Appendix A for full methodological details) led to the identification of 1 011 papers for title and abstract review from which 202 papers were selected for full review. Common reasons for excluding papers include studies were not undertaken within/referred to a target country climate, studies did not consider receiving water body impacts and/or did not refer to an urban context. Full review of the 202 papers led to the final selection of 56 papers in relation to address surface water impacts and 21 papers to address the groundwater impacts question. Findings were structured in relation to two categories:

- studies that directly address each question through the collection of field samples in the targeted receiving water compartment
- studies which use data from other compartments/studies to infer potential implications on targeted compartments

For full findings see Appendix A.

3.1.2 Key findings

CHEMICAL IMPACTS

The majority of research identified relates to surface water impacts from a chemical quality perspective, with a particular focus on chlorides. For example, studies included found a significant correlation between urban factors and the salinization of lake waters, identifying urban areas (i.e. road networks and stormwater drainage systems) as primary sources of chloride contamination (e.g. Bermarija et al., 2023; Radosavljevic et al., 2022; Meriano et al., 2009). Surface water chloride concentrations in excess of acute and chronic receiving water threshold concentrations have been widely reported (e.g. Perera et al., 2009), with associated impacts including chloride-driven stratification of water bodies. Under such conditions, data indicates that lower water layers can become anoxic which can also lead to the release of previously bound phosphorous (P) from sediments into the water column (Radosavljevic et al. (2022). A similar relationship between urbanisation and groundwater has been reported for chlorides, with groundwater chloride concentrations increasing in proximity to urban areas (Eyles et al., (2013). The dynamic relationship between stormwater, groundwater and surface water has also received some attention, whereby stormwater transfers chlorides directly (via piped stormwater discharges) and indirectly (infiltration to groundwater which recharges surface water bodies) to surface waters (Meriano et al., 2009). However, whilst stormwater runoff-driven chloride inputs to surface and groundwaters is a seasonal input (linked to winter maintenance activities), groundwater recharge of surface waters can be a year-round process (Lembcke et a., 2017). Hence, a seasonal episodic input can effectively – through contamination of groundwater base flows – become a year-round impact phenomenon.

Whilst the evidence base is strongest for chlorides, a diversity of other substances has been reported to demonstrate similar behaviour trends with elevated concentrations of e.g. metals, polyaromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), per- and polyfluoroalkyl substances (PFAS) in surface waters and surface water sediments in areas adjacent to/downstream of urbanised areas (e.g. Frogner-Kockum et al., 2020; Rentz et al. 2011). For example, studies have reported concentrations of several metals in receiving waters and sediments in excess of receiving water environmental quality standards (EQS)/threshold values at sites receiving stormwater discharges (Frogner-Kockum et al., 2020; Kuusisto-Hjort and Hjort, 2013). Similar trends have been reported for PAHs, with a single study identifying stormwater runoff as a key source of microplastics discharging to a lake (Rochman et al. , 2022) and that an accidental release of PFAS and sprinkler water that discharged via a stormwater drainage system continued to have impacts in receiving water almost 10 years after the original spill event (when sampling stopped (Awad et al. 2011).

ECOLOGICAL IMPACTS

In terms of ecological effects, the impacts of stormwater discharges have been considered in several surface water studies where a range of test species have been evaluated against a variety of endpoints – from bioaccumulation and behaviour change to gene expression and mortality (e.g. Gillis et al., 2012; Popick et al., 2022a; Waara and Färm, 2008). However, although a reasonable degree of confidence can be associated with the statement stormwater runoff negatively impacts the chemical quality of receiving waters, the likelihood that elevated pollution concentrations will translate into negative ecological impacts is less certain. For example, several detailed field and laboratory-based studies employing a variety of test species have reported changes in a range of endpoints e.g. changes to community composition and abundance (Valleau et al., 2022). The reviewed studies typically focused on different pollutants or pollutant groups, with impacts linked to the presence of metals (especially zinc (Zn) and copper (Cu)), chlorides, PAHs and PFAS. Results show that different species – and even life stages within a single species – can vary in their sensitivity to particular pollutants (Gillis, 2012; Gillis et al., 2014). Further, whilst stormwater runoff-related toxicity studies show there is often an impact, there are also studies where no ecological impacts have been reported, even at sites where runoff concentrations exceed EQS (Rochfort et al. 2000; Popick et al., 2022b; Gillis et al., 2022). A range of factors driving this differential response have been proposed, including water hardness (studies have shown increasing toxicity with decreasing water hardness levels), 'upstream ecologies' being acclimated to pollution sources further upstream), the potential for antagonistic/synergistic effects associated with the unknown profile of stormwater discharges and buffering impacts of natural geology. However, whilst such factors are suggested at a high-level, much work remains to be done in relation to understanding the impact and interactions of pollution mixtures on receiving water biota (at both acute and chronic levels) before the locations/scenarios where urban stormwater discharges will have a deleterious effect on receiving water ecology can be robustly predicted.

HYDROGEOMORPHOLOGICAL IMPACTS

Only three studies on the hydrogeomorphological impacts of stormwater runoff within receiving waters under a cold climate could be sourced (Ariano and Oswald. 2022; Van Duin and Garcia, 2006; Eyles et al., 2003). Whilst each study adopted a different approach and scale, all three concluded that urbanisation impacts the hydrological cycle, with increased stormwater runoff volumes and flow rates leading to negative receiving water impacts including erosion, and contamination of receiving waters and their sediments.

3.1.3 Strengths and weaknesses of the evidence base

A limitation of the reviewed studies is that only 28 of the 77 papers involved the physical collection of receiving water samples from study sites (26 papers related to surface waters and 2 papers related to groundwaters), with the remaining studies either drawing on historic data sets, inferring impacts from e.g. modelling studies or suggesting potential impacts based on stormwater runoff concentrations. In the absence of a standard method for sampling and evaluating stormwater, these studies utilise a wide range of sampling methods, analytical protocols, experimental designs and test species. Hence it is not possible to undertake a meta-analysis of the available data sets. To-date the majority of studies have focused on surface water impacts from a chemical quality perspective, with groundwater impacts a relatively neglected topic. Most of the groundwater studies identified refer to limitations in the availability and/ or quality of groundwater data sets. Given the importance of both surface water and groundwater as a source of drinking water in many areas, this is a topic that requires urgent attention. However, even in relation to surface water impacts (which has been the subject of relatively more research), although the occurrence of a range of stormwater pollutants has been evaluated, there is a lack of depth within the emerging research base under cold climate conditions i.e. no parameter has been the subject of ≥5 independent studies in this specific context. Whilst understanding of impacts of stormwater discharges on surface water ecology have received some attention, this has typically focused on acute impacts as soluble pollutant loads are typically short in duration (i.e. a per event basis) with relatively little research addressing repeated exposures (e.g. multiple urban stormwater discharges) at either acute or chronic concentrations. Further, whilst 'good ecological status' under the EU Water Framework Directive is defined in relation to chemical, ecological and hydrogeological components, reviewed studies typically focus on chemicals or biota or hydrogeomorphology; no single study addressed the receiving water impacts of stormwater discharges from all three perspectives.

3.2 Design, performance and maintenance requirements of stormwater treatment technologies

3.2.1 Scope

The international peer-review literature relating to design, treatment erformance and maintenance requirements of stormwater treatment technologies from a cold climate perspective within Nordic countries and those with a similar climate (defined on discussion with project advisory board as Canada and the Baltic countries) was reviewed (see Appendix B). Research studies were identified using a systematic approach and evaluated to address the following core research questions:

- RQ 2.1: What is the treatment performance for different types of facilities and target pollutants?
- RQ 2.2: Are there design criteria and recommendations affecting the treatment performance and are these sensitive to differences in climate?
- RQ 2.3: Which type of maintenance is needed for the different types of facilities?

Use of agreed search terms (see Appendix B for full methodological description) led to the identification of a long-list of 662 papers. The titles and abstracts were reviewed for relevance to the core questions, leading to the shortlisting of 279 papers for full review. Common reasons for excluding papers include studies were not about stormwater, stormwater quality or stormwater treatment, or they were not conducted in an urban context. Full review of the 279 papers led to the final selection of 33 papers in relation to addressing treatment performance, 51 papers to address the design criteria and sensitivity to climate and 26 papers to address the maintenance topic. For full findings see Appendix B.

3.2.2 Key findings

TREATMENT PERFORMANCE FOR DIFFERENT TYPES OF FACILITIES AND TARGET POLLUTANTS

Regarding treatment performance for different types of facilities and target pollutants, only a few field studies were identified. These included three studies on ponds and wetlands (Al-Rubaei et al., 2016; Al-Rubaei et al. 2017a; Farrel and Scheckenberger, 2003), six studies on biofilters (Pineau et al. 2021; Brodeur-Doucet et al. 2021; Spraakman et al. 2020; Géhéniau et al. 2015; Lange et al. 2022, Beryani et al. 2023), one study each of grassed swales (Bäckström et al. 2006) and permeable pavements (Drake et al. 2014), and three studies each of reactive filters (Jensen et al. 2011; Milovanovic et al. 2022; Hallberg et al. 2022) and treatment trains (Guesdon et al. 2016; Sønderup et al. 2015; Pineau et al. 2021). In addition, a laboratory study evaluating the potential to apply coagulation/flocculation was also reviewed (Nyström et al. 2020). Most of these studies investigated the removal of metals (mainly Cu, Zn and lead (Pb)), sediments (measured as TSS) and nutrients (total phosphorus and total nitrogen and sometimes their speciation) from stormwater (see Table 1). A few studies, but not for all types of facilities, have also investigated organic pollutants (e.g. PAHs, oils, microplastics, etc.). In general, most of the studies, independent of technology, reported removal efficiencies for total Cu, Zn, Pb, TSS and phosphorous of 50 % or better on an event basis. Lowest removal efficiencies were obtained for the single study on swales where the studied swale was not able to remove TSS or metals to a high extent, with negative removals often reported. In most types of facilities, nitrogen reduction was in comparison to other pollutants quite low, rarely above 50 % and more frequently negative removal rates were observed (i.e. increased concentration) such as in several of the tested biofilters.

Table 1. Mean concentration removal efficiency [%] of different types of facilities. Removals are given for main pollutant groups TSS, total metals (Zn, Cu, Pb) and nutrients (TP and TN).

Facility type/region (No of events)	Notes	TSS	TZn	TCu	TPb	TP	TN	Reference
Ponds and wetlands								
Sweden (n=13)		96	90	91	96	89	59	Al-Rubaei et al., (2016)
Sweden (n=13)		91 ± 7	92 ± 8	91 ± 9	83 ± 32	80 ± 13	45 ± 27	Al-Rubaei et al. (2017a)
Canada (n=12)	Wetland 2[1]	−1520–97 (71)	−335–99 (57)	−458–96 (60)	−272–95 (66)	−249–88 (55)	nd	Farrel and Scheckenberger (2003)
-“-	Wetland 3	−32–99 (76)	−178–98 (65)	−41–97 (61)	−28–98 (66)	−17–95 (49)	nd	-“-
Bioretention								
Canada (n=2–4)		63–98	71–89	−27–78	−68–90	43–100	−57–71	Pineau et al. (2021)
Canada (n=24)		84	nd	nd	nd	6	−5	Brodeur-Doucet et al. (2021)
Canada (n=13)		63	nd	nd	24	nd	nd	Spraakman et al. (2020)
Canada (n=17)		74.5	48.3	−14.1	54.3	−65.3	nd	Géhéniau et al. (2015)
Sweden (n=6)		> 78	94	81	> 76	nd	nd	Lange et al. (2022)
Sweden (n=11)	Biofilter	95 ± 3						Beryani et al. (2023)
-“-	Biofilter + Chalk	95 ± 3						-“-
-“-	Sand filter	35–90						-“-
Grassed swales								
Sweden (n=13)		−129–47	−35–40	−288–(−12)	−186–12	nd	nd	(Bäckström et al., 2006)
Permeable pavements								
Canada (n=7)		> 80	62–82	50–62	nd	9–82	34–45	Drake et al. (2014)
Reactive filters								
Denmark (n=25)	Dual porosity filtration (DPF)	91.5–98.9	70.0–87.3	50.6–61.6	88.1–97.9	73.3–78.0	nd	Jensen et al. (2011)
Sweden (n=7)	Zeolite		51–94	52–82				Milovanovic et al. (2022)
Sweden (n=24)	Sand		93	67	nd	nd	nd	Hallberg et al. (2022)
Treatment trains								
Canada (n=nd)	Detention basin + filter + wetland	nd	nd	nd	7–36	nd	nd	Guesdon et al. (2016)
Denmark (n=nd)	Pond + sand filter	77	nd	nd	nd	78	nd	Sønderup et al. (2015)
	Pond + crushed concrete filter	78	nd	nd	nd	80	nd	
Canada (n=4)	5 Bioretention cells + Pond	81–98	90–95	27–78	63–90	29–100	−71–71	Pineau et al. (2021)

[1] Range for sampled events and median in brackets, calculated excluding the event with high negative removals. Water quality grab samples were taken at the outlet of Wetland 2.

DESIGN CRITERIA AND RECOMMENDATIONS AFFECTING THE TREATMENT PERFORMANCE AND DEPENDENCE ON CLIMATE

Several factors and activities were described in the reviewed articles as important to consider when designing and constructing stormwater treatment systems in a cold climate. These include directly climate-related factors such as:

- low air temperature
- ice cover on ponds and snow on the ground
- frost in biofilter media and soils and freeze-thaw cycles
- significant snowmelt volumes in spring
- non-growing or short growing season and dormant plants.

In addition, activities related to the cold seasons include:

- usage of grit and de-icing salt for winter road maintenance
- increased asphalt wear due to studded tyres
- cold starts of car engines.

In the reviewed papers, design recommendations related to cold climate were only found for a limited number of technologies; mainly ponds and wetlands, biofilters and filter materials in general.

For ponds and wetlands, the role of plants has frequently been investigated (see Figure 1). Plants were shown to increase the removal of nitrogen from the water, and may also increase the uptake of metals and chlorides. On the contrary, low temperatures and high salt concentrations may affect the removal of (especially dissolved) metals negatively.

Figure 1. Plants identified by Lauki et al. (2022) to be tolerant to salt. From left to right: *Iris pseudacorus*[1], *Amsonia tabernaemontana*[2] and *Baptisia australis*[3]. (Figure B10)

The function of biofilters in cold climate have been studied to a large extent in laboratory studies. TSS and particle bound pollutants are mainly retained by physical filtration processes and are little affected by temperature, salt or any other factors related to cold climates. However, due to high concentrations of TSS

[1] Photo obtained from: https://www.flickr.com/photos/coanri/98621275
[2] Photo obtained from: https://commons.wikimedia.org/wiki/File:Amsonia_nadrenska_Amsonia_tabernaemontana.jpg
[3] Photo obtained from: https://commons.wikimedia.org/wiki/File:Baptisia_australis_kz04.jpg

in snowmelt, coarse and well-draining filter materials are promoted to prevent the filters from clogging. For dissolved pollutants the removal is more dependent on the design of the biofilter, e.g. inclusion of a saturated zone, soil amendments and contact time. Salt in general has a negative impact on the retention of metals. From the metal species usually studied (Zn, Cu, lead (Pb), cadmium (Cd)), Cu, Pb and to some extent Cd were the most sensitive to salt and may be desorbed from the particulate phase and pass the filter bed. Dissolved Cu was also sensitive to temperature with higher temperatures leading to increased release of Cu mainly attributed to increased degradation of organic matter and its association with Cu. Nitrogen removal can be improved by the incorporation of submerged saturated zones, and lower temperatures have been shown to have a positive effect on the removal of nitrogen as well as phosphorus. Longer dry periods may affect nitrogen removal negatively. For organic pollutants, in general, climatic effects are less investigated and – though it is assumed that cold temperature and high salt loads have a negative effect on their removal – there are too few studies to-date to support development of strong conclusions. Plants in biofilters may affect the treatment performance. Better performance has been shown for planted biofilters compared to (non-planted) sand filters for phenolics, PAHs and hydrocarbons and – to some extent – also for microplastics.

A wide range of filter materials have also been investigated in laboratory studies (batch and column tests) with the purpose of using the materials as amendments in biofilters, tree pits etc. to increase treatment efficiency. Materials investigated include biochar, charcoal, iron-treated sand, compost, peat, clay, granulated activated carbon, pine bark, olivine, crushed limestone, zeolite, shell sand and wood chips. In studies where the effect of salt and temperature were evaluated, most studies showed no effects on the adsorption capacity (limited to P, Cu, Zn, Pb and occasionally Cd, chromium (Cr), nickel (Ni)) when salt was added to the test water.

MAINTENANCE NEEDS FOR DIFFERENT TYPES OF FACILITIES

As identified in the reviewed studies (see Appendix B), proper inspection and maintenance of stormwater treatment facilities is necessary to ensure that they continue to fulfil their function over time. Despite this, several studies reported that this is often neglected. Recommended maintenance activities identified for different technologies were regular inspection and removal of trash and large debris for wet ponds and constructed wetlands as well as biofilters. These types of facilities also need regular removal of accumulated sediments, particularly the forebays which are designed to promote pre-sedimentation. Furthermore, for wetlands, ponds, biofilters and grassed swales, care for the vegetation including mowing, pruning, removal, weeding and replanting is needed. Removal or replacement of soil or filter media should be implemented for grassed swales, biofilters and infiltration trenches. Permeable pavements need regular removal of accumulated particles to maintain their infiltration capacity and prevent them from clogging. An overview of the identified maintenance activities is presented in Table 2.

Table 2. Recommended maintenance activities according to facility type. (Table B10)

Stormwater control measure	Inspection of inlet and/or outlet structures	Removal of trash and large debris	Removal of sediments	Care for vegetation (mowing, pruning, removal, weeding, replanting)	Removal and/or replacement of surface layer of soil or filter media	Removal of particles from surface layer of structure
Wet pond	X[1,2]	X[1,2,3]	Forebay and/or whole facility[1,4,5,6,7,8]	X[1,3,5]		
Constructed wetland	X[1]	X[1,3,7]	Forebay and/or whole facility[1,7,9,10]	X[1,3,5]		
Grassed swale/ vegetative filter strip				X[11,12,13]	X[5,13]	
Biofilter	X[1,14]	X[1,14]	Forebay[1,12,14]	X[1,14]	X[1,5,14,15,16]	
Infiltration trench					X[1,5,17]	
Permeable pavement						X[1,5,18]

[1] Blecken et al., 2017, [2]Starzec et al., 2005, [3]Badiou et al., 2019, [4] Gavric et al., 2022, [5]Langeveld et al., 2022, [6]Olding et al., 2004, [7]Suits et al., 2023, [8]Wik et al., 2008, [9]Han et al., 2014, [10]Mungasavalli and Viraraghavan, 2006, [11]Al-Rubaei et al., 2017b, [12]Ekka et al., 2021, [13]Mooselu et al., 2022, [14]Beryani et al., 2021, [15]Furén et al., 2023, [16]Kratky et al., 2017, [17]Bergman et al., 2011, [18]Drake et al., 2013.

3.2.3 Strengths and weaknesses of the evidence base

A limitation of the reviewed studies is that in total only 17 studies assessed the performance of full-scale facilities in the field, and that organic contaminants of emerging concern are very sparsely reported; focus was generally on TSS, metals (typically Cu, Pb and Zn), chloride/conductivity, and nutrients (P and N). Moreover, many facility types have not been assessed at all at full-scale under cold climate conditions. Therefore, the performance in cold climate of the evaluated technologies cannot be compared and the removal of specific pollutants cannot be confidently targeted. There was a wide range of different filter amendments tested in lab-scale studies, which to some extent also considering cold climate. Despite promising results for some of the amendments, they need to be evaluated in the field before any recommendations can be made. While a range of maintenance activities are suggested for BGIs in the reviewed literature, it is also indicated that there is a need for better understanding of maintenance needs which will vary because in general, BGI systems are unique and lack standardised designs.

3.3 Understanding co-benefits and overcoming barriers for increased societal acceptance of sustainable stormwater management

3.3.1 Scope

The international peer-reviewed research relating to understanding co-benefits and overcoming barriers for increased societal acceptance of sustainable stormwater management in countries with a cold climate and with governance structures relevant to Sweden was reviewed (see Appendix C). Research studies were identified using a systematic approach and evaluated to address the following research questions (RQs):

- RQ 3.1 What actors and stakeholders are involved in sustainable stormwater management?
- RQ 3.2 Which co-benefits are created through sustainable stormwater management according to different actors and stakeholders?
- RQ 3.3 What are the underlying aims and incentives for sustainable stormwater management?
- RQ 3.4 What barriers and challenges to sustainable stormwater management and what conflicts between its stakeholders exist? How can these be overcome and resolved for increased societal acceptance of sustainable stormwater management?

Use of agreed search terms (see Appendix C) led to the identification of 597 papers for title and abstract review from which 193 papers were selected for full review. Common reasons for excluding papers include no actor perspective included in the study; no or only marginal focus on sustainable stormwater management; studies did not refer to an urban context; primary study focus on water quality, water supply, water security or stormwater reuse; and studies focused on agriculture and agricultural runoff. Full review of the 193 papers led to the final selection of 61 papers. For full findings see Appendix C.

3.3.2 Key findings

While conventional stormwater management originally aimed to deal with flood mitigation only, sustainable stormwater management is performed to reach multiple aims and manage stormwater while providing a wide range of co-benefits to stakeholders, striving to create multifunctional blue-green solutions and spaces (Fletcher et al., 2015). Use of this approach has increased in recent decades, and involves a wide range of solutions, and thus also actors. While few of the reviewed articles describe processes of stakeholder participation specifically in the development of blue-green solutions, stakeholder participation in the development of public urban green spaces in general is more widely studied (see e.g. Fors et al., 2021). This calls for a shift in both practice and research towards viewing the implementation and management of blue-green solutions as part of the governance and management of urban green spaces, rather than as a separate issue. In this way, lessons learnt from studies on green space management in general can be applied to sustainable stormwater management.

LOCAL RESIDENTS ARE AN UNTAPPED RESOURCE

The lack of Swedish studies on participatory sustainable stormwater management in the reviewed articles suggests that local residents are an untapped resource for the long-term management of existing blue-green solutions in Sweden. Inspiration for increased stakeholder participation in Sweden in the future can be found in the reviewed studies from the USA, UK, Finland, Denmark and Belgium, where residents participate through e.g. Rain Check and other educational programmes (Cousins, 2018; Dhakal and Chevalier, 2016; Fitzgerald and Laufer, 2017) or by installing rain barrels and downspout planters to manage water on their property (Fitzgerald and Laufer, 2017; Lieberherr and Green, 2018). We also found several examples from other countries of local residents being involved in the management of blue-green solutions on public land. In Portland Oregon, USA, people volunteer as 'Green Street Stewards' and maintain public rain gardens as a supplement to regular municipal maintenance (City of Portland, 2024; Shandas, 2015). Similar programmes include Adopt-a-Street and Adopt-a-Drain (Dhakal and Chevalier, 2016). Long-term monitoring data on the performance of blue-green solutions is important to identify changes over time as a basis for strategic management decisions. Local residents can contribute to such data collection by engaging in citizen-based environmental monitoring. In Kajaani, Finland, local residents were involved in water quality monitoring, collecting data on sources of pollution and levels of contamination in runoff (Laatikainen et al., 2020; 2019).

SEVERAL FACTORS AFFECT VALUED AND PROVIDED CO-BENEFITS

Different stakeholders attach different values and benefits to blue-green solutions. These are important to understand in order to create socially inclusive blue-green solutions. Local residents tend to emphasise place-specific values related to the use and experience of the blue-green solution, while municipal managers emphasise general and holistic values such as which solution is best for the whole city, the environment or in the future (Miller and Montalto, 2019; Thodesen et al., 2023; Vierikko and Niemelä, 2016). While these findings provide some guidance on the aspects that need to be considered when implementing blue-green solutions, the general public is itself a diverse stakeholder group. Therefore, in order to take full account of local needs and perspectives, it may be beneficial for local authorities to involve different stakeholder groups, and especially marginalised groups, in the planning, design, construction and management of blue-green solutions. Several factors affect which co-benefits are provided from blue-green solutions and valued by people, including who, where and when you ask, since this varies between stakeholder groups, regions and over time. In addition, the co-benefits that are created and valued depend on the blue-green solution in question, with different blue-green solutions differing in their ability to provide specific ecosystem services (Elliott et al., 2020; Ossa-Moreno et al., 2017). It is also important to locate blue-green solutions in the right place, to deliver co-benefits in neighbourhoods where they are most needed (Heckert and Rosan, 2016). This means that the crucial first step is to define the objective for the site and identify what co-benefits are needed there. The next step is to select the right blue-green solution for the site, that is, one that can deliver the desired co-benefits, and tailor the choice of plants and materials, the design and the maintenance to maximise these co-benefits.

TAKING DIVERSE NEEDS AND PERSPECTIVES INTO ACCOUNT

Sustainable stormwater management is driven by the desire of municipalities to manage stormwater while creating multifunctional spaces with a range of co-benefits. If the many desired objectives of sustainable stormwater management are to be achieved, several different perspectives need to be considered simultaneously when deciding how to implement and manage blue-green solutions. Like the governance and management of public urban green spaces in general, the governance and management of blue-green solutions needs to be place and context specific to create blue-green solutions that both manage stormwater and provide co-benefits. This involves taking into account a range of aspects related to both the physical conditions of the site and the needs of local residents and other stakeholders. The fact that blue-green solutions, unlike underground pipes, are visible to local residents means that the appearance of the system needs to be a priority, as public acceptance and involvement can be affected by negative perceptions of implemented blue-green solutions.

OVERCOMING BARRIERS TO SUSTAINABLE STORMWATER MANAGEMENT

A number of barriers and conflicts that hinder the successful implementation and management of blue-green solutions were identified. The most common barriers described were regulatory barriers, organisational barriers, knowledge barriers and economic barriers. Other barriers mentioned multiple times include the influence of the physical environment, and land ownership and social barriers. To overcome the identified barriers, there is a need for changed stormwater regulations that approach stormwater as a 'resource' rather than a 'hazard' (Wilfong et al., 2023), and that are more adaptive and less reliant on quantitative outcomes to allow for more integrative, collaborative planning and decision-making. In practice, this implies viewing blue-green solutions as a resource through economic valuation tools that monetise their primary and co-benefits; seeing blue-green solutions as insurance i.e. well-functioning BGI could help cities reduce the costs of floods and extreme events and build resilient cities; and discursively redefining stormwater as a resource (e.g. as a water supply issue) in laws and regulations to legally recognise the benefits of stormwater (Cousins, 2018).

To transform society's relationship with stormwater and realise its value as a resource, new forms of public participation in sustainable stormwater management are needed. Adopting holistic approaches to sustainable stormwater management that enable communities to participate in stormwater decision making, better advocate for their goals and aspirations, and interact with robust local government institutions is essential for effective and equitable long-term stormwater management. Above all, there is a need to improve communication and collaboration between urban stormwater stakeholders and to overcome departmental silos within local authorities for a more holistic approach to stormwater implementation and management. In addition to communication and collaboration across municipal departments, effective stormwater governance requires experimentation and a strategy for organisational learning, and that these three elements of governance are built into the planning process for blue-green solutions (Fitzgerald and Laufer, 2017). Successful implementation of blue-green solutions often needs to be preceded by soft actions, such as regulatory changes, that lead to the societal and institutional changes necessary for increased implementation of blue-green solutions (Dobre et al., 2018). The shift towards more sustainable stormwater management requires visionary government leadership at local, regional and national levels, with authorities supporting the process as drivers, coordinators and capacity builders (Harrington and Hsu, 2018).

4. Mapping of research findings to selected policy requirements and objectives

4.1 The new Urban Wastewater Treatment Directive (2024)

The 1991 EU Urban Wastewater Treatment Directive (UWWTD) established requirements for the collection and treatment of urban wastewater, with the central aim of protecting receiving waters (EU, 1991). Whilst the definition of urban wastewater in the 1991 Directive includes 'runoff rain water', little attention was given to the impacts of runoff on receiving water quality, other than to state that pollution from storm water overflows (also commonly referred to as combined sewer overflows) should be limited and that extreme water quality values due to heavy rainfall should not be taken into consideration when assessing compliance with discharge water quality requirements. However, a range of drivers including findings of the EU freshwater policy 'fitness check' (EC, 2012), the European Green Deal with its zero pollution commitments (2019), increasing societal unacceptability of stormwater overflows and rising awareness of potential role of stormwater as an alternative water source led to recognition that the UWWTD (1991) was no longer fit for purpose. The Commission began work on a recast EU UWWTD in 2022, with a revised text adopted by the EU Parliament in April 2024 (European Parliament, 2024).

The recast Directive identifies urban runoff[4] as an important source of pollution that could be avoided and the development of integrated urban wastewater management (IUWM) plans (Article 5 and Annex V) as a mechanism to address what is (together with stormwater overflows) 'a sizeable source of pollution to receiving waters'. IUWM plans should be developed for drainage areas of agglomerations of ≥ 100 000 person equivalents (p.e.) by 2033, and for populations between 10 000 and 100 000 p.e. by 2039 where stormwater overflows or urban runoff pose a risk to the environment or public health. Specific requirements for urban runoff include the establishment of measures to reduce pollution from urban runoff through a combination of source control measures and nature-based solutions. Annex V

[4] In the recast UWWTD (2024) urban runoff is defined as 'precipitation from agglomerations collected by combined or separate sewers'. The use of the term 'storm water' is reserved for storm water overflows (defined as the 'discharge of untreated urban wastewater into receiving waters from combined sewers caused by precipitation or system failures'). Whilst references to and requirements established for stormwater overflows and urban runoff directly overlap on the majority (if not all) occasions, the terms refer to separate processes and should not be used interchangeably. Reference to and management requirements of the latter term (urban runoff) are the focus here as this term relates to the use of urban stormwater runoff as used within the current study.

details sets out the requirements of IUWM plans in more detail with key actions related to urban runoff including:

- a detailed description of the:
  - network of collecting systems including urban runoff storage
  - monitoring of separate systems where discharges of urban runoff are polluted (i.e. expected to pose a risk to the environment or public health)
- identification of relevant and feasible measures to:
  - avoid the entry of unpolluted rainwaters into collecting systems
  - minimise releases of polluted urban runoff into receiving waters with priority given to the use of nature-based solutions
- consider water reuse within IUWM planning.

In terms of monitoring requirements, Article 21 includes the need for representative monitoring of discharges of urban runoff from separate systems to estimate concentrations and loads of a range of conventional – and where relevant – micro-pollutants (including microplastics). Monitoring of urban runoff to verify compliance with new requirements is also identified (preamble 43), and as such information on measures to address urban runoff should be included in the data sets made available to the Commission for compliance verification (preamble 46). However, in terms of specific guidelines on how these requirements should be met, the Directive only states that implementing powers should be conferred on the Commission to establish methodologies to support e.g. the development of IUWM plans. Hence, there is currently uncertainty over specifically how identified urban runoff monitoring, mitigation and verification requirements should be met.

In relation to wider stormwater management-derived benefits, Annex VI sets out information that should be made available to the public e.g. percentage of wastewater treated, annual average concentrations released by treatment plants etc. However, this is a top-down, transactional type of data sharing only, with no further types of engagement apparently envisaged. The notion of community benefits is referred to on a single occasion e.g. "all citizens benefit equally from efficient collection and treatment of urban wastewater", this is specifically linked to the use of EU funding reserves only. Whilst the use of nature-based solutions as a sustainable stormwater management approach is given preference with the recast Directive, no guidance on their use as multi-functional spaces which may generate a range of co-benefits is identified. Further, public participation (seen as crucial in underpinning the integration of nature-based solutions within the urban fabric) is only referred to in relation to data-sharing under the Aarhus Convention. Hence opportunities to support the delivery of stormwater management co-benefits and their access by local communities – of which there is a growing evidence base – are not explicitly recognised within the new Directive.

In the absence of specific socio-environmental or technical guidelines and methodologies, the focus on actions to manage stormwater is primarily held within IUWM plans (Annex V). Links between key areas of the IUWM plans and the current 'urban stormwater in cold climates' evidence base is reviewed in the following section.

4.1.1 Establishment of IUWM plans

IUWMs are to be developed at the scale of the agglomeration for areas with a population of ≥ 100 000 people by 2033. The development of plans at a city scale – as opposed to at a river basin scale as required by the EU Water Framework Directive (EU WFD, 2000) will require further mapping activities to enable data sets generated and interpreted for WFD purposes to be translated to support city-scale IUWM plan needs. Plans are also to be developed for areas with a population between 10 000 and 100 000 people by 2039 under certain circumstances which include when urban runoff poses a risk to the environment or public health. Whilst there is evidence that urban runoff negatively impacts the chemical quality of receiving waters for a number of parameters, studies are typically carried out a site or sub-catchment level and hence the identification of such locations at a city scale would be challenging. Further, whilst a range of substances have been reported in excess of available receiving water standards, the number of independent studies per substance is limited (typically < 5 per substance). The evidence base is even less well established for ecological impacts. This is partially a function of multiple types of ecological tests and end points utilised, with scarce – if any – data on the impacts of urban runoff on human health available. Hence, evidencing this requirement using the current evidence base would be extremely challenging. In addition, the use of BGI necessitates a new approach to stormwater management as it requires increased collaboration across stakeholder groups. From this perspective IUWM planning can be a mechanism to work towards achieving a ‘One Water’ approach i.e. where all parts of the urban water cycle, including urban water supply, sanitation, stormwater and wastewater, are considered as an integrated system and managed holistically to achieve sustainable economic, social and environmental goals.

4.1.2 Monitoring of urban runoff

The recast Directive sets out the need for representative monitoring to estimate concentrations and loads of a range of conventional – and where relevant – micro-pollutants (including microplastics). In terms of methodological know-how, the evidence base contains a wide range of studies in relation to impact and performance monitoring that suggest the physical collection of representative urban runoff samples should not be a challenge from a sampling perspective. However, as discussed above, the identification of locations which are representative of activities at a city-scale (which itself is spatially and temporally dynamic) remains an open challenge. Further, the majority of expertise in sampling accrued to-date relates to conventional sampling (i.e. collection of time- or flow- proportional samples using automatic samplers) which then require to be analysed in a laboratory before data sharing can occur. The use of sensors to provide real time data on urban runoff quality is an open area of research but the challenges it poses e.g. intermittent flows and multiple organic and inorganic substances – have yet to be resolved. In addition, whilst not well-studied, there is a growing international literature on the role of local residents as citizen scientists in generating environmental data sets through participation in a range of voluntary activities . This concept has yet to gain traction within Sweden and hence the potential for local residents to contribute to urban runoff monitoring via citizen-based monitoring programmes is currently an untapped resource.

4.1.3 Mitigation of urban runoff

The UWWTD (2024) expresses a strong preference for the use of NBS to reduce runoff volumes generated by impermeable surfaces, avoid the entry of unpolluted rainwater into receiving systems and minimise the release of polluted urban runoff to receiving waters. Whilst full-scale field studies on the performance of NBS to mitigate urban runoff are limited (only 17 field studies were identified), removal efficiencies of a range of pollutants were > 50 % irrespective of treatment NBS type (i.e. > 50 % of pollutants which enter the system are retained by it), with the exception of nitrogen where lower (or negative) removal efficiencies were often reported. Hence, whilst there is an emerging evidence base that a range of BGI do reduce urban runoff pollutant loads discharging to receiving waters, the current evidence is insufficient to enable treatment options to be selected based on the removal of a target pollutant to a specific level. The new UWWTD does refer to derogations for treatment of urban wastewater in cold climate conditions (defined as an annual average temperature of < 6 °C). However, these apply to reducing the for secondary (biological) treatment and the implications of this (if any) for the use of urban runoff NBS is not clear.

4.1.4 Evidencing compliance with UWWTD (2024) requirements

The Directive identifies the need to include urban runoff monitoring data in the data sets made available to the Commission for compliance verification. In the absence of a standard method, urban runoff is currently sampled, analysed and reported using a variety of methodologies and approaches which is a barrier to both undertaking a meta-analysis of available data sets and systematic data-sharing. Limited comparable data and data-sharing is also a barrier to sustainable stormwater management in that it also impedes practitioners from comprehending effectiveness of BGI and improving practices. For example, standardising the approach to assessing BGI performance is recognised as essential for documenting, communicating and promoting their benefits and potentially integrating them into legislation.

4.2 The Swedish environmental objectives

Sweden has 16 environmental objectives adopted by the Parliament in 1999 and 2005 (Sveriges miljömål, 2024). The overall goal with these objectives is to solve the main environmental problems in Sweden within one generation without causing increased environmental pressure outside Sweden (i.e. “the generational goal”), and the 16 environmental objectives describe the desired state of the environment. In this section the synthesized findings of this report are mapped against the 16 environmental objectives to see how sustainable stormwater management can be linked and contribute to achieving the environmental objectives.

To reach the environmental objectives, milestones have been adopted to show the way and the pace, and a number of indicators that show how the environmental work is progressing. Sweden’s environmental objectives also form an important part of Sweden’s work on the UN’s global sustainable development goals in Agenda 2030, particularly those related to the biosphere. Of the 16 environmental objectives, only one is currently considered achievable and three are close, so there are urgent needs for improvements.

4.2.1 Links between sustainable stormwater management and the Swedish environmental objectives

For the milestone of sustainable urban development, stormwater management in the existing built environment is identified as an important arena for transition in society. Of the 16 environmental objectives, five are directly related to stormwater management in the existing built environment (Sveriges miljömål, 2024):

- Zero eutrophication
- Flourishing lakes and streams
- Good-quality groundwater
- Non-toxic environment
- A good built environment

In addition, four environmental objectives have been identified within this project where sustainable stormwater management, including BGI/nature-based solutions (NBS), can contribute to varying degrees:

- Clean air
- A balanced marine environment, flourishing coastal areas and archipelagos
- Thriving wetlands
- A rich diversity of plant and animal life

As shown by the review of cold climate design, performance and maintenance of stormwater treatment technologies (see Section 3.2 and Appendix B), stormwater treatment can contribute to reducing nutrient concentrations in stormwater; mainly phosphorus, and to a lesser extent also nitrogen, provided that the right treatment technology is used and that the facility is operated and maintained appropriately. The **Zero eutrophication** environmental goal is assessed, among other things, on the basis of indicators of nitrogen and phosphorus loads to the sea, the environmental status of eutrophication under the Marine Environment Ordinance and the status of nutrients under the Water Management Ordinance (Sveriges miljömål, 2024). Stormwater management can thus to some extent be a means of reaching the environmental goal of **Zero eutrophication.**

Many of the challenges that threaten the environmental objectives of **Flourishing lakes and streams** and **A Balanced marine environment, flourishing coastal areas and archipelagos** are not related to water quality, apart from the presence of environmental toxins and problems with eutrophication and acidification (Sveriges miljömål, 2024). These environmental objectives are followed up with the indicator Good status of water where chemical status is an important factor. For this, sustainable stormwater management can play a significant role as urban stormwater is known to contribute to diffuse pollution to receiving waters such as lakes and streams (see Section 3.1 and Appendix A). The studies identified and summarised in Section 3.2 (see also Appendix B) indicated good potential to especially reduce the particle-bound fraction of Cu, Zn and Pb in stormwater facilities in cold climate, such as ponds, constructed wetlands and biofilters. For other substances, such as organic pollutants, the evidence base is poorer. Sustainable stormwater management can also indirectly contribute to the environmental goal **of a Non-toxic environment**. However, it does not affect the sources of the pollutants, which is the main aim of this environmental objective, but only reduces the further spread of the pollutant.

For a major part of the groundwater bodies, the desired environmental quality is not reached (Sveriges miljömål, 2024). This is often due to historically already contaminated soils leaching pollutants into groundwater. One of the indicators for the environmental goal of a **Good-quality groundwater** is the establishment of water protection areas, partly with the aim of protecting drinking water sources. Sustainable stormwater management, where stormwater is appropriately treated, is then a prerequisite. Stormwater management in cold climates needs to pay particular attention to the use of de-icing salt during the winter season in order not to risk contaminating groundwater (see Section 3.1 and Appendix A).

The National Board of Housing, Building and Planning (Boverket) emphasises the potential of ecosystem services to contribute to the environmental objective of **A good built environment** (Boverket, 2024). Nature-based solutions, such as ponds, constructed wetlands and stormwater biofilters (see Section 3.2 and Appendix B) for stormwater management have the potential to contribute to human well-being in addition to managing stormwater, but this requires that the appearance and co-benefits of the facilities are also taken into account, which is highlighted in Section 3.3 (see also Appendix C). In addition, for example, biofilters with planted trees can contribute to the environmental objective of **Clean air** due to the air cleaning capabilities of trees see Appendix C). Constructed wetlands for stormwater management can also contribute to the environmental objective of **Thriving wetlands**, for which one of the indicators is Constructed or hydrologically restored wetlands. Furthermore, both constructed wetlands and other nature-based stormwater solutions can strengthen the work on the environmental goal **A rich diversity of plant and animal life** in these environments.

For the remaining seven environmental objectives, i.e., **Reduced climate impact, Natural acidification only, A protective ozone layer, A safe radiation environment, Sustainable forests, A Varied Agricultural Landscape** and **A Magnificent Mountain Landscape** (Sveriges miljömål, 2024), sustainable stormwater management, with a focus on cold climate and based on what has been covered within the framework of this project, has been assessed as having marginal impact.

4.3 The UN Sustainable Development Goals

The UN's Sustainable Development Goals (SDGs) are a set of 17 goals, with more specific targets for each, that aim to create a better and more sustainable future for all around the world by addressing urgent challenges such as poverty, inequality, climate change, environmental degradation, peace and justice. In the following section, we map our synthesised findings from Sections 3.1, 3.2 and 3.3 (see Appendices A, B and C for full details) against the SDGs to clarify links between sustainable stormwater management and policy objectives and to support stakeholders to identify links between sustainable stormwater management and multiple societal benefits.

4.3.1 Links between sustainable stormwater management and the UN SDGs

The SDGs wedding cake, developed by Johan Rockström at the Stockholm Resilience Centre in collaboration with Pavan Sukhdev, reorganises the 17 Sustainable Development Goals into three layers: Biosphere, Society and Economy. This model emphasises that the biosphere (which includes goals related to climate, ecosystems and natural resources) is the foundation on which societal goals (such as health, education and equality) and economic goals (such as innovation and growth) depend, highlighting the interconnectedness of ecological health and human well-being. It is also a way of moving away from the current sectoral approach where social, economic and environmental development are seen as separate parts (Stockholm Resilience Centre, 2016).

By implementing and managing blue-green solutions in urban areas, municipalities and other stakeholders can contribute significantly towards achieving multiple SDGs simultaneously, creating more sustainable and resilient cities that benefit both people and the environment. Figure 2 helps visualise which of the SDGs sustainable stormwater management makes a direct contribution to. Below, we describe these relationships in greater detail.

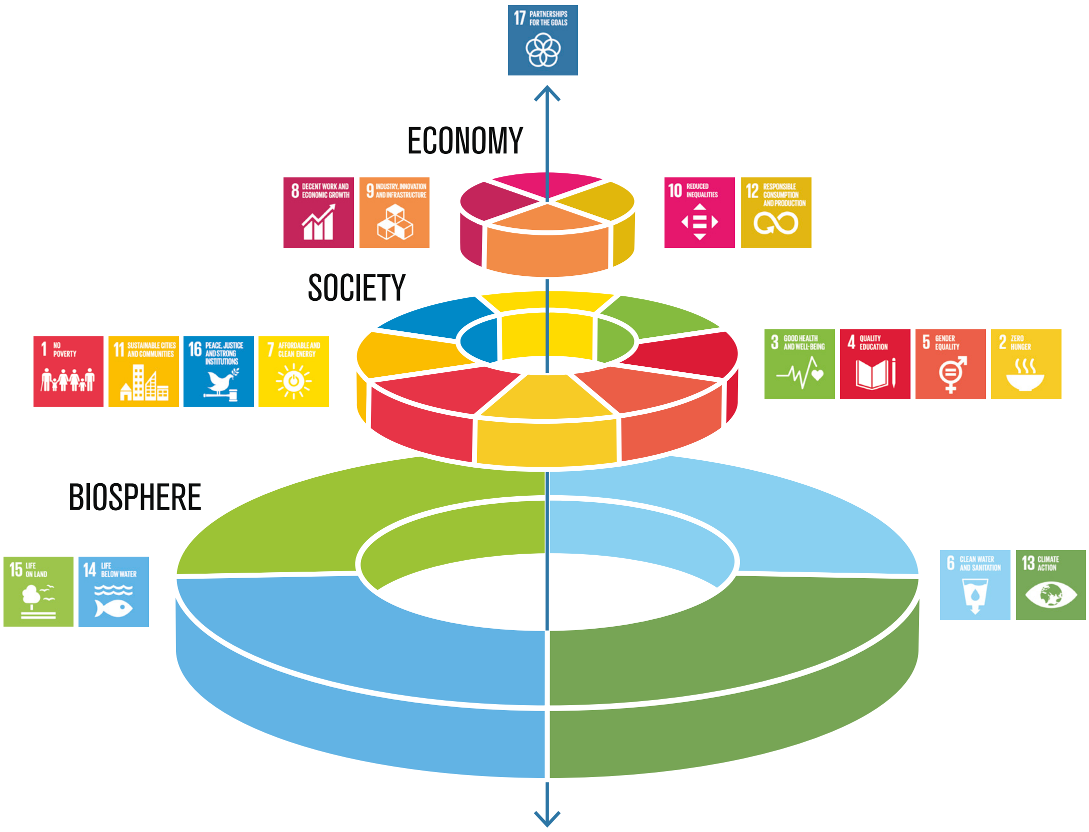

Figure 2. According to the SDGs wedding cake, the biosphere is the foundation of our society and economy and the basis of all the SDGs. Sustainable stormwater management makes a direct contribution to all of the four fundamental biosphere SDGs (SDGs 6, 13, 14, 15), to one of the society SDGs (SDG 11) and to two of the economy SDGs (SDGs 9, 12). Azote for Stockholm Resilience Centre, Stockholm University CC BY-ND 3.0.

4.3.2 Biosphere

Urban stormwater is known to contribute to diffuse pollution to receiving waters such as lakes and streams (see Section 3.1 and Appendix A). The studies identified and summarised in Section 3.2 (see also Appendix B) indicated good potential to especially reduce the particle-bound fraction of Cu, Zn and Pb in stormwater facilities in cold climate, such as ponds, constructed wetlands and biofilters. For other substances, such as organic pollutants, the evidence base is poorer. This means that sustainable stormwater management contributes to **SDG 6: Clean Water and Sanitation** by ensuring the protection and quality of freshwater resources, especially to **target 6.3**, which focuses on improving water quality by reducing pollution and minimising release of hazardous pollutants. If a large enough proportion of local residents were to manage stormwater locally on their property by installing rain barrels, this could contribute to **target 6.4**, aimed at increasing water-use efficiency across all sectors and ensuring sustainable withdrawals and supply of freshwater to address water scarcity. However, studies reviewed did not conclude how many rain barrels would be needed to make a difference at the city level. Catchment partnerships and other mechanisms that facilitates bridging the urban-rural divide in stormwater management or application of the 'One Water' approach, which means that all parts of the urban water cycle, including urban water supply, sanitation, stormwater and wastewater, are considered as an integrated system and managed holistically, rather than separately, could be a way to contribute to **target 6.5**, aimed at implementing integrated water resources management (see Appendix C for further details on such approaches for increased collaboration). If communication and collaboration between urban stormwater stakeholders is improved and departmental silos within local authorities overcome this could contribute to **target 6.b**, aimed at strengthening the participation of local communities in improving water and sanitation management.

Blue-green solutions contribute to **SDG 13: Climate Action** by helping to mitigate the impacts of climate change, mainly by reducing flood risks and to some extent by sequestering carbon through vegetation. Especially **target 13.1** is relevant here, as it is aimed at strengthening resilience and adaptive capacity to climate-related hazards and natural disasters.

Regarding SDG 14: Life Below Water, sustainable stormwater management can contribute to **target 14.1**, which focuses on reducing marine pollution from land-based activities, as stormwater runoff from urban areas often contain pollutants, making it important to avoid that the polluted water ends up in rivers, lakes and oceans and damages marine ecosystems.

SDG 15 Life on Land: Integrating green spaces and natural habitats within urban areas through blue-green solutions can support biodiversity conservation and enhance ecosystem services. It is important to involve stakeholders in the development of blue-green solutions to deliver the co-benefits they want and that are needed locally (see Appendix C to read more about co-benefits from blue-green solutions as perceived by different stakeholders).

4.3.3 Society

A clear advantage of blue-green solutions compared to underground pipes is the co-benefits that they deliver, which contribute to **Goal 11: Sustainable Cities and Communities,** which is about making cities and human settlements inclusive, safe, resilient and sustainable. When implemented and managed in a good way, blue-green solutions enhance the resilience of cities to climate change impacts by for example reducing urban heat island effects (aligning with **target 11.5**, which seeks to reduce the negative impacts of disasters, including water-related hazards such as flooding) and by improving air quality (aligning with **target 11.6** aimed at reducing environmental impact of cities). In order to contribute to **target 11.7** aimed at providing universal access to safe, inclusive and accessible, green and public spaces, in particular for women and children, older persons and persons with disabilities, there is a need to increase user participation in the development of multifunctional urban green spaces to provide spaces for recreation for a wide range of users. Spaces that both handle stormwater and have additional co-benefits cannot only be managed to mitigate flooding, but need to be adapted to the needs of various user groups, including traditionally marginalised ones. **Target 11.b** aims to increase the number of municipalities that adopt and implement local disaster risk reduction strategies. Sustainable stormwater management could play a vital role in such strategies. How to achieve sustainable stormwater management in practice through increased collaboration across stakeholder groups and departments is described in Section 3.3 (see Appendix C for full information).

4.3.4 Economy

Some types of blue-green solutions reduce the further spread of some pollutants (see Appendix B for details), thereby contributing to **Goal 12: Responsible Consumption and Production**, and especially **target 12.4** that aims to achieve the environmentally sound management of chemicals throughout their life cycle and significantly reduce their release to air, water and soil in order to minimise their adverse impacts on human health and the environment. Aging or inadequate stormwater infrastructure can fail during extreme weather events, causing economic and environmental damage. When locally adapted and implemented at a sufficiently large scale, adoption of blue-green solutions contributes to **Goal 9: Industry, Innovation, and Infrastructure** and more specifically to **target 9.1**, which focuses on developing quality, reliable, sustainable and resilient infrastructure to support economic development and human well-being.

4.3.5 Contributing to overcoming remaining challenges of Swedish SDG implementation

Sweden's Voluntary National Review 2021 is an assessment of progress made on Sweden's implementation of the 2030 Agenda (Government offices of Sweden, 2021). Sweden ranks highly in many international comparisons on SDG implementation, but consequences of the COVID-19 pandemic are hampering progress and exacerbating existing challenges. Therefore, some challenges remain, including overcoming the economic and social inequalities in Sweden, which are increasing; mental health challenges, violence and bullying of young people; and making consumption and production more sustainable and transition towards a circular economy (Government offices of Sweden, 2021).

Sustainable stormwater management can contribute to some of these remaining challenges of SDG implementation. According to Government offices of Sweden (2021), SDG delivery and accelerated actions require that implementation of the Agenda 2030 is guided by the Agenda's 'leave no one behind' principle, that child and youth perspectives are included, and that a whole-of-society approach is employed. A whole-of-society approach implies capitalising on the joint effort of actors involved in SDG implementation in Sweden, such as government agencies, municipalities, the research community, the civil society and the business community. Sustainable stormwater management can contribute to these ambitions if communication and collaboration between urban stormwater stakeholders is improved and departmental silos within local authorities overcome, forming a more socially inclusive, decentralised and holistic approach to stormwater implementation and management where the voice of young people and other traditionally marginalised groups are heard. Further Government offices of Sweden (2021) stresses the important role of municipalities and regions in the Agenda 2030 implementation, as SDGs are put in practice and innovations are developed and tested at the local level, which in turn contributes to implementation nationally and globally. This is also valid when it comes to sustainable stormwater management. In addition to communication and collaboration across municipal departments, effective stormwater governance in practice requires experimentation and a strategy for organisational learning, and that these three elements of governance are built into the planning process for blue-green solutions (see Appendix C).

As part of Sweden's ambition to remain a strong voice for the global implementation of the 2030 Agenda, Sweden aims to adapt to climate change, halt biodiversity loss and restore ecosystems (Government offices of Sweden, 2021). Blue-green solutions could potentially deliver such co-benefits. However, this will not happen automatically, as several factors influence which co-benefits are valued and provided (Figure 3). Local adaptation of blue-green solutions and participatory sustainable stormwater management processes are required to deliver valued co-benefits where they are needed the most.

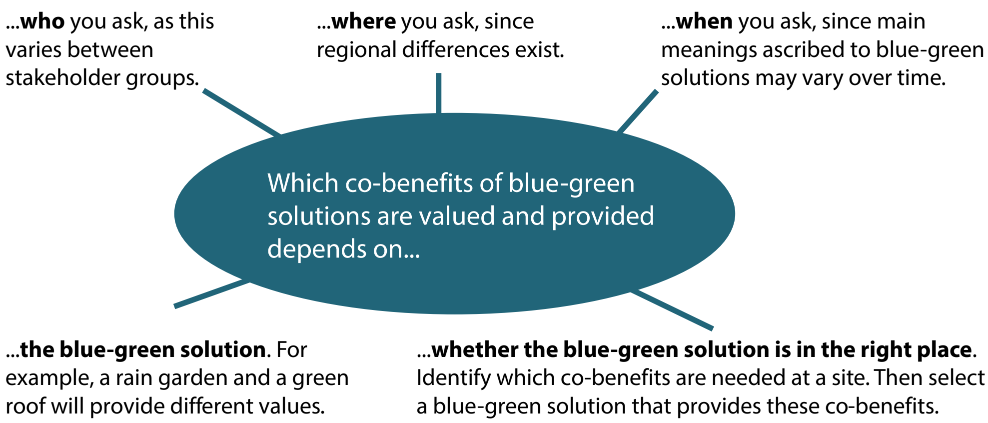

Figure 3. Factors affecting which co-benefits that are valued and provided.

5. Conclusions: knowledge gaps and recommendations for policy development and research

This report provides a high-level overview of existing knowledge on sustainable stormwater management from the Nordic countries and countries with a "Nordic climate". This knowledge is then mapped to key areas of policy to further clarify links between sustainable stormwater management and progress in achieving both receiving water protection and wider sustainability objectives. As a further contribution to supporting Sweden in its transition to a sustainable stormwater management approach, key findings and the mapping exercise results are used to identify a series of knowledge gaps (see below), together with research (see Section 5.2) and policy recommendations (see Section 5.3) which – if implemented – would enable these gaps to be closed.

5.1 Key knowledge gaps

In terms of receiving water impacts, a key finding is that a lack of data is limiting our ability to predict when and where stormwater will negatively impact receiving waters in cold climates. For example, whilst there is evidence that urban stormwater runoff negatively impacts the chemical quality of surface waters, this relates to a relatively limited number of studies i.e. ≤ 5 independent studies per pollutant type. Data related to urban runoff impacts on groundwater quality is scarce, and impacts of urban stormwater on surface and groundwater ecologies is also limited.

In terms of understanding BGI performance, there are in general few field studies on the performance of stormwater treatment in cold climates. Different technologies have been investigated to different extents; most studies focused on the removal of particles, metals (Cu, Zn and Pb) and nutrients (phosphorus and nitrogen) with limited information available on the removal of organic substances. Studies on design criteria and recommendations are to a large extent lacking for most treatment technologies (exceptions are ponds, wetlands and biofilters), as is field verification of the laboratory-based studies carried out to-date. Furthermore, there is limited data on how the maintenance needs of different facility types can be predicted and how performance evolves over the lifetime of a facility, and the extent to which maintenance practices can be used to renew functionality.

With regard to delivering a wider range of benefits, the multi-functional nature of BGI leads to a number of practical as well as organisational and governance related challenges. For traditional stormwater management, there are established methods for the calculation and measurement of water quality and quantity. However, there is a lack of corresponding norms and standards for blue-green solutions and standard methods for the measurement of their benefits and co-benefits. There are also a number of organisational and governance perspectives to consider, as many different priorities need to be set for multifunctional blue-green solutions. Different municipal departments and stakeholder groups have different values and institutional understandings, and the need for improved communication and collaboration between urban stormwater stakeholders is still hampered by a lack of collaboration across administrative boundaries. Furthermore, as blue-green solutions are local context dependent, the involvement of local residents is seen as a priority. However, the literature to-date remains vague on how to involve local people in sustainable stormwater management in a good way.

5.2 Recommendations for policy development

The following policy recommendations are made as a contribution towards supporting identified knowledge gaps to be addressed.

Environmental quality standards and monitoring

- Stormwater runoff is essentially an unintentional mix of substances and hence policy responses for tackling stormwater pollution should require and integrate the use of chemical and toxicity methodologies and data sets. This need is underscored by studies which have found no ecological impacts at sites where runoff concentrations exceed environmental quality standards and vice-versa.
- Embrace opportunities offered by developments in online sensing technologies to provide early warnings and rapid assessments of pollutants in receiving surface and ground waters. Whilst the use of sensors technologies underpins recent developments in drinking water security and wastewater treatment, the routine deployment of sensors in further urban water compartments e.g. stormwater is yet to be achieved. Whilst partly a function of the characteristics of stormwater (episodic and varying concentrations of multiple substances), policy support for stormwater sensors use would drive innovation with derived data sets facilitating the use of stormwater as an alternative water source.
- Support development of an integrated open-access platform for the systematic collection, assessment and sharing of urban water chemical and ecological data sets. This would facilitate the identification of key pollutant groups, their sources and inform the development of strategies to mitigate their impacts.

Operation, maintenance and management of facilities

- Stormwater quality varies on a site-by-site basis with respect to pollutants and their speciation, with treatment needs also differing as a function of receiving water status. Therefore, the desired function of a facility should be specified and prioritised at the outset with this knowledge used to inform system design and performance assessments.
- Many stormwater treatment facilities are already implemented but not centrally reported/documented. Development of a 'living inventory' of existing facilities is as a first step in assessing whether existing BGI need to be improved and adapted (retrofitted) to future needs due to e.g. climate change and/or densification of cities.

- Operation and maintenance are important for the long-term function of a facility, and it can also be costly. Development of user guides on how different facility types should be operated and maintained in a sustainable way would support practitioners in understanding and implementing maintenance needs.

Enhancing collaboration

- The desired, needed and actual co-benefits delivered vary between types of blue-green solutions, from place to place, over time and depending on which stakeholder group is asked. It is therefore important to specify and prioritise the desired functions and co-benefits of a facility, and work with stakeholders to design it to meet its purpose.
- There is a need to overcome departmental silos within local authorities for a more holistic approach to stormwater planning, design and management. Policies are needed that facilitate communication and collaboration between all urban stormwater stakeholders, both within the organisation and with stakeholders outside the local authority (e.g. private landowners, local residents, etc.).
- Local residents are an untapped resource in the long-term management of blue-green solutions, both in terms of managing stormwater on private land, which is a significant proportion of urban areas, and in participatory stormwater management on public land. Policies and regulations could be developed to support and encourage increased stakeholder participation through public education programmes on sustainable stormwater management. Other relevant opportunities to facilitate involvement include, for example, public stormwater management programmes such as "adopt a rain garden", incentive-based policies such as providing residents with rain barrels or financial reimbursement for installing green roofs or rain gardens on their property and citizen-based water quality monitoring programmes. As the general public is a diverse stakeholder group, there is also a need to involve different stakeholder groups, and especially marginalised groups, in all phases of development of blue-green solutions (planning, design, construction and management).

5.3 Recommendations for further research

The following research recommendations are made as a further contribution to developing the sustainable stormwater evidence base.

Environmental quality standards and monitoring

- Identify materials used in urban spaces that contribute to stormwater runoff pollution loads, quantify their release and mobilisation of conventional and emerging pollutants, and characterise how this varies with material age. Consider how this data could be used to raise awareness of the presence of substances in urban materials, inform future (re-)development plans to reduce their use and support development and implementation of IUWM plans.
- Develop/adapt and validate sensor technologies to enable the routine detection and quantification of a wide range of substances in urban water bodies. Generation of real time data sets would inform risk characterisation assessments and increase the reliability of urban water pollution models at a catchment scale which would again contribute to the development of IUWM plans.

- Development of standardised approaches to sampling, analysing and reporting stormwater quality and its impacts in receiving waters from ecological and chemical perspectives in relation to both short-term (acute) and chronic (long-term) exposures patterns. New requirements linked to the recast UWWTD (2024) provide an opportunity to establish standardised methodologies for the systematic collection of data sets to support modelling approaches and enable comparative assessments, and these opportunities should be taken.

Operation, maintenance and management of facilities

- Systematic collection data on treatment performance for a wider range of technologies (more than biofilters) and pollutants (e.g. organics), as well as pollutant speciation (e.g. the distribution between particulate, colloidal and dissolved phases) will enable the possibility to choose the type of facility and configuration that best suits the purpose, and demonstrate how to retrofit already existing facilities for improved treatment performance when space is limited.
- Investigate design criteria and recommendations affecting the treatment performance in cold climates for technologies in addition to ponds, wetlands and biofilters, and verify promising filter materials and amendments from laboratory studies in the field under environmental conditions. Some additional information would most likely be obtained through extended literature searches not limited to cold climate. However, a cold climate contributes to specific conditions that need to be considered to get an accurate overall picture.
- Long-term studies to document performance over the lifetime of a facility, validation of models for predicting changes in performance over time and the required frequency of maintenance and evaluation of the potential of sensors to monitor facilities and identify when maintenance is necessary.

Enhancing collaboration

- The most common challenges related to blue-green solutions are regulatory barriers, organisational barriers, knowledge barriers and economic barriers. To overcome identified barriers in practice, there is need for a more holistic, integrative, adaptive and interdisciplinary approach to research on sustainable stormwater management.
- Stakeholder participation in the development of public urban green spaces in general is more widely studied than in relation to blue-green solutions specifically. This calls for a shift in research towards viewing the implementation and management of blue-green solutions as part of the governance and management of urban green spaces, rather than as a separate issue.
- Effective stormwater governance requires experimentation and a strategy for organisational learning, and these elements need to be built into the planning process for blue-green solutions. There is a need for interdisciplinary research to better understand the drivers of transformative change towards sustainable stormwater management.
- The lack of Swedish studies on participatory sustainable stormwater management suggests that local residents are an untapped resource for the long-term management of existing blue-green solutions in Sweden. Future research should therefore focus on how to involve local residents in their development and long-term management.

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Appendix A

Stormwater impacts within receiving water recipients

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A1. Summary

This report reviews the international peer-review literature relating to impacts of stormwater runoff within receiving water recipients from a cold climate perspective.

The research studies included in this report were identified using a systematic approach and consider receiving water impacts from chemical, ecological and geomorphological perspectives. The majority of research identified relates to surface water impacts from a chemical quality perspective. However, the lack of a standard stormwater sampling and analysis methodology limits the ability to robustly compare findings between studies. The occurrence and impacts of chlorides (as a component of de-icing salts) is the most widely studied pollutant, with several studies identifying urban areas as its primary source and urban surface runoff as the pathway that connects sources with receiving water receptors. The occurrence of a range of further pollutants (e.g. metals, polyaromatic hydrocarbons and pesticides) in surface water and groundwater bodies located adjacent to urban areas is also reported. However, there is only a limited number of independent studies (i.e. ≤ five) for any single pollutant group let alone substance. Likewise, whilst the ecological impacts of stormwater discharges have been considered in several surface water studies, the use of different test species for a range of ecotoxicological endpoints – from bioaccumulation and behaviour change to gene expression and mortality – also provide a breadth but not depth of knowledge.

Despite the methodological and data limitations described above, it is possible to identify trends within the emerging evidence base as to the types and magnitudes of impacts that could be anticipated at sites receiving urban stormwater discharges. These include elevated surface water and sediment concentrations for a number of substances in proximity to urban areas (in comparison to control or ‘un-impacted’ background sites). However, although a reasonable degree of confidence can be associated with the statement stormwater runoff negatively impacts the chemical quality of receiving waters, the likelihood that elevated pollution concentrations will translate into negative ecological impacts is less certain due to complex interactions between pollutant properties and site-specific parameters (e.g. water hardness and pH) that has yet to be fully understood. Further, the balance of urban stormwater-driven receiving water impact research undertaken to date has focused on surface waters as opposed to groundwater bodies. Whilst presumably a reflection of the challenges of sampling collection, the urgent need to better understand the impacts of urban stormwater discharges on both urban surface water and groundwater quality is highlighted in light of their role as a source of drinking water in many areas.

A2. Introduction

This report reviewed the peer-review literature to address the following core questions:

- how does stormwater runoff impact on the chemical quality, geomorphology and/or ecological status of surface waters?
- how does stormwater runoff impact on the chemical quality, geomorphology and/or ecological status of groundwaters?

Use of a systematic approach (see Section A3) led to the identification of a long-list of 1011 papers. The titles and abstracts were reviewed for relevance to the core questions, leading to the shortlisting of 202 papers for full review. Common reasons for excluding papers include studies were not undertaken within/referred to a Nordic climate, did not consider receiving water body impacts and/or did not refer to an urban context. Full review of the 202 papers led to the final selection of 56 papers in relation to addressing surface water impacts and 21 papers to address the groundwater impacts question.

Synthesis findings are structured in relation to those that directly address each question (e.g. involve the collection of field samples in the targeted receiving water compartment) and those which use data from other compartments/studies to infer potential implications for the targeted compartment from chemical, (geo-)morphological and/or ecological perspectives. The report concludes with an evaluation of the strength of the evidence base for each core question addressed and identification of key knowledge gaps.

A3. Methodology

This review followed the PRISMA approach (Page et al., 2020), an established methodology to provide a transparent, complete and accurate account of how studies are identified and their characteristics reported within systematic reviews. The methodology involved two stages as follows:

- to facilitate alignment between the review activities (see also Appendices B and C), overarching keywords were identified and applied within a research database to identify a common 'longlist of papers'
- the common 'long list of papers' was further interrogated/refined using keywords relevant to the focus of each review topic (see also Appendices B and C).

A3.1 Common approach

Keywords for the common approach were: stormwater or "storm water" or runoff. These keywords were selected from an inclusivity perspective i.e. to capture as many articles as possible within the targeted field. These pre-defined keywords were entered into SCOPUS, a research publications database which provides access to > 90 million research documents including outputs from > 29,750 peer-reviewed journals (SCOPUS, 2023). The initial search was undertaken in June 2023 and returned 133,504 hits. This initial longlist was then filtered using the keyword 'urban' leading to the identification of 42,124 papers, with this set of articles providing a common data pool for each review to work with (Table A1). Papers were not shortlisted to focus on a particular time period.

Table A1. Key words used in the common approach to align activities within Appendices A, B and C

	Key words	Number of hits
Article title, abstract, keywords	Stormwater or "storm water" or runoff	133,504
Search within results	Urban	42,124

A3.2 Methodology to identify articles to address 'Impact on recipients' research questions

To identify relevant papers for this review, the 42,124 articles identified using the common approach were filtered using the keyword 'impact' as this term reflects the core focus this topic. Further filters were applied to extract documents which were in English, in the form of peer-reviewed research or review articles, and where at least one co-author had a cold climate country affiliation. Initially the geographic scope was limited to the Nordic countries and Canada, but this was extended following feedback from the project's expert advisory panel to additionally include the Baltic countries. Results of applying filters are reported for each step in Table A2.

Table A2. Overview of the results of applying identified filters to the long list of common papers from animapcts within receiving waters perspective

	Refining criteria and filters	Number of hits
Article title, abstract, key words	Impact	12,687
Language	Limit to English	12,125
Document type	Limit to peer-review research articles (9082) and review articles (554)	9,636
Geographic scope	Limit countries to cold climate and Baltic countries: Canada (542), Sweden (201), Denmark (138), Norway (101), Finland (78), Iceland (7) Estonia (14), Latvia (3), Lithuania (20) Greenland (2)	1,014

Figure A1 provides an overview of how the 1,014 articles reported in Table A2 were managed in relation to the two core research questions (pertaining to surface water and groundwater impacts) including reasons for exclusion. The initial screening of articles involved their export from SCOPUS and uploading into the open-access systematic review software Rayyan (www.rayyan.ai) which enables research teams to review the same set of articles and – through a blind reviewer mode – provides a mechanism for quality assurance by allowing more than one researcher to independently review a shortlist of articles and compare the consistency of decision-making. In this report two reviewers independently screened 20 abstracts and reached the same inclusion/exclusion decision on all papers providing confidence that decisions on inclusion/exclusion were being made consistently within the team undertaking this activity.

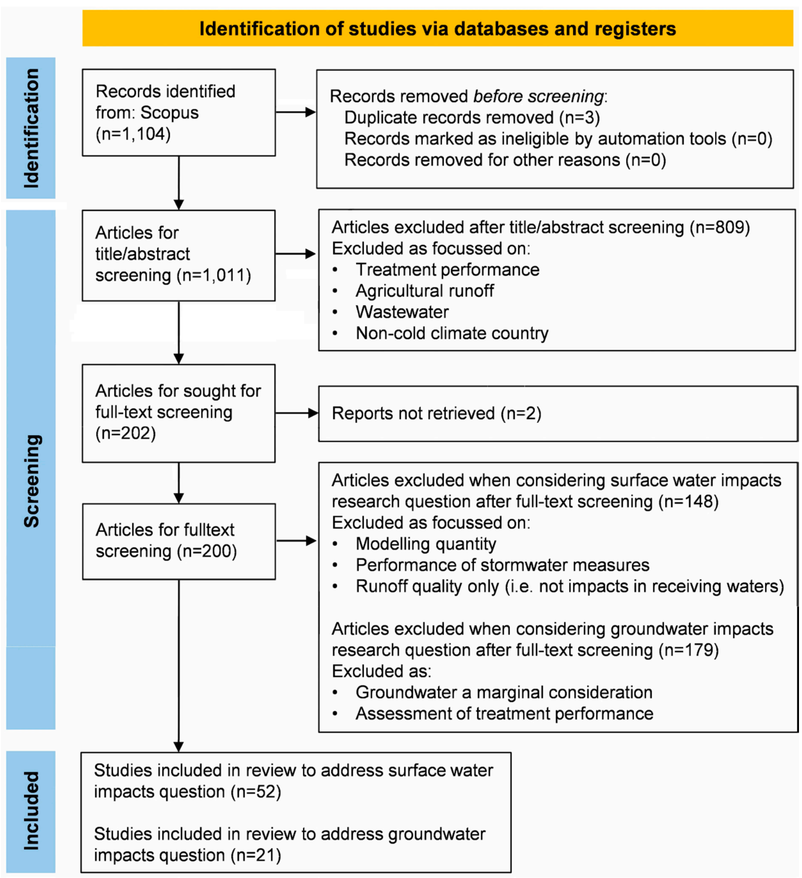

Figure A1. PRISMA 2020 flow diagram outlining the screening and filtering steps applied to the 1,014 papers extracted from SCOPUS.

A4. The impact of stormwater discharges on surface waters

This section explores the effects of direct stormwater discharges (i.e. the impact of stormwater without any form of treatment) on urban surface water bodies. The 200 papers extracted for full-text screening were reviewed and, given the focus of this review, preference was given to studies that evaluated changes in surface waters associated with the direct discharge of urban stormwater through the collection of receiving water samples and/or in-field receiving water quality measurements. In terms of identifying effects within urban surface water bodies, an effect is defined as a change in one or more of the following:

- chemical water quality (e.g. concentration of metals, organics, nutrients, physico-chemical parameters)
- biological components (e.g. occurrence of bacteria, antibiotic resistant bacteria/genes/fungi)
- ecological status (fish, invertebrates, plants etc; mortality or growth parameter; measure of fitness)
- hydrogeomorphology (reports of erosion occurring, change in receiving water velocity alters structure/nature of receiving water)

Figure A2 provides an overview of the results of applying this process. Of the 200 papers, just over 10 % (22 papers) refer to the impact of stormwater in surface water under field conditions. Whilst not involving the collection of surface water samples, a further sub-set of 30 papers which focus on e.g. stormwater runoff quality from chemical or ecological perspectives were also short-listed as they included discussions on potential impacts in surface waters.

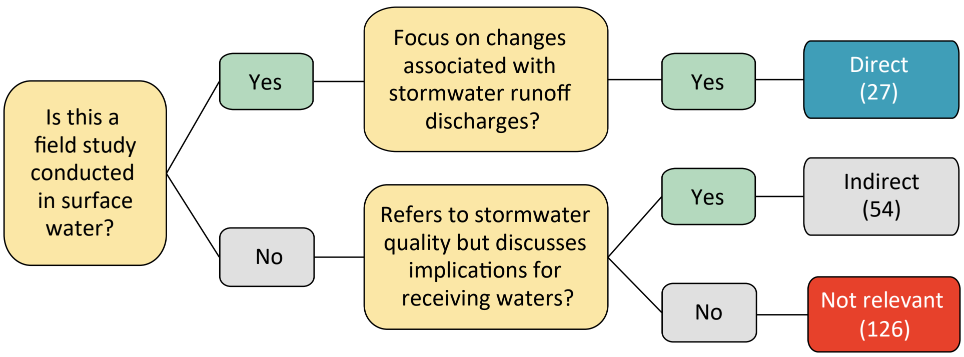

Figure A2. Flowchart of decision-making process used to select papers for section: impact in surface waters.

Using these selection criteria, 14 papers were categorized as “not relevant”. Key reasons for exclusion at this stage include:

- focused on modelling climate change, rainfall, and temperature patterns (predict potential changes in stormwater runoff volume in the future without addressing the impacts on receiving water bodies)
- related to the performance of stormwater management facilities (addressed in Appendix B)
- assessed stormwater quality (including relationship to land use activities) but did not discuss possible impacts in surface water bodies
- focused on agricultural – as opposed to urban – runoff.

A4.1 Directly relevant studies

Among the selected 22 studies identified as directly relevant, 66 % of the studies (14 papers) were conducted in Canada, with the remaining seven studies carried out in Northern European countries including Sweden, Finland, Norway and Denmark. In terms of focus, 12 papers examine chemical impacts of which two also address ecological impacts. Seven studies focus on ecological impacts, with a further three papers exploring hydrogeomorphological changes. An overview of the papers included in this section with investigated parameters presented in Tables A3, A4 and A5.

Table A3. Overview of parameters measured in studies evaluating impacts of stormwater discharges on surface waters in a cold climate

Metals	Organics	Nutrients	Electrical conductivity	De-icing salts	Temperature	Author and location
				X		Bermarija et al., 2023; Canada
		X		X		Radosavljevic et al., 2022; Canada
			X	X		Perera et al., 2009; Canada
					X	Kolath and Egemose, 2023; Denmark
		X				Long et al., 2014; Canada
	X					Rochman et al., 2022; Canada
	X					Awad et al., 2011; Canada*
X						Frogner-Kockum et al., 2020; Sweden **
X						Kuusisto-Hjort and Hjort, 2013; Finland
X	X					Rentz et al., 2011; Sweden
X						Blecken et al., 2012; Sweden
	X					Honkonen and Rantalainen, 2013; Finland

Key: * = study includes fish ecotoxicity tests; **studies include plant ecotoxicity tests.

Table A4. Overview of biota in studies evaluating impacts of stormwater discharges on surface waters in a cold climate

Fish	Invertebrates	Bacteria	Author and location
X			Awad et al., 2011, Canada
	X		Gillis, 2012; Canada
	X		Gillis et al., 2014; Canada
	X		Gillis et al., 2022; Canada
	X		Valleau et al., 2022; Canada
		X	St Laurent and Mazumder, 2014; Canada
X			Meland et al., 2010a; Norway

Table A5. Overview of hydrogeomorphological components included in studies evaluating impacts of stormwater discharges on surface waters in a cold climate

Hydrogeomorphology		Author and location
Flow	Geomorphology	
X		Ariano and Oswald, 2022; Canada
X	X	Van Duin and Garcia, 2006; Canada
X	X	Eyles et al., 2003; Canada

A4.1.1 Chemical impacts

Several studies have been conducted to explore the chemical impacts of stormwater runoff on receiving water bodies. These investigations have covered a range of parameters, including chlorides (generally from road salt applications), organic pollutants, with a specific focus on polyaromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), and per- and polyfluoroalkyl substances (PFAS), as well as metals including cadmium (Cd), chromium (Cr), lead (Pb), nickel (Ni), and zinc (Zn). The majority of these studies, particularly in Northern Europe – including Sweden, Denmark, and Finland – have involved the analysis of water and sediment samples, concentrating on changes in the receiving water bodies in terms of total concentrations and speciation. Research conducted in Canada has mainly focused on the impacts of road salt application.

PHYSICO-CHEMICAL PARAMETERS

Of the 14 selected papers that focus on chemical changes in water bodies, four specifically address physico-chemical changes, which include parameters such as chloride and temperature (see Table A6). All studies mentioned in this section were conducted on water samples.

Table A6. Overview of the physicochemical parameters effects and reference values referred to in the selected studies

Parameter	Type of effects reported	Environmental quality standard (EQS)/ reference values used	Author and location
Chloride	Increased Cl levels	20 mg/L (maximum background chloride concentration); 120 mg/L (chronic exposure guideline)	Bermarija et al., 2023; Canada
Chloride	Increased Cl levels; water stratification	Not provided	Radosavljevic et al., 2022; Canada
Chloride	Increased Cl levels	230 mg/L (chronic Cl) and 860 mg/L (acute Cl) set by the U.S. EPA	Perera et al., 2009; Canada
Temperature	Increase in stream temperature	Danish Environmental Protection Agency Standard: 21.5 °C (high and good condition streams) and 25 °C (moderate streams)	Kolath and Egemose, 2023; Denmark

All papers exploring the impact of road salt application were conducted in Canada. For instance, the study by Bermarija et al., (2023) investigates the effects of stormwater runoff on chloride concentrations in Canadian lakes, with a particular focus on the consequences of urban development and infrastructure. Utilizing geospatial analysis and statistical modelling, the research evaluates how land use, road density, and stormwater management infrastructure contribute to the increase in chloride levels in lakes. The data set included chloride measurements from 57 lakes sampled three times a year for five years. Results indicated that mean Cl concentrations in most of the lakes (i.e. 42 of the 57 sampled) exceeded the natural background concentration of 20 mg/L, with Cl concentrations in eight lakes exceeding the Canadian Council for Ministers of the Environment (CCME, 2011) chronic exposure Cl guideline of 120 mg/L (see Figure A3). The study reported a significant correlation between urban factors and the salinization of lake waters, identifying urban areas – characterized by extensive road networks and stormwater drainage systems – as primary sources of chloride contamination. This indicates that stormwater runoff, influenced by the application of road salts, plays a significant role in the increase of chloride levels within the examined lakes.

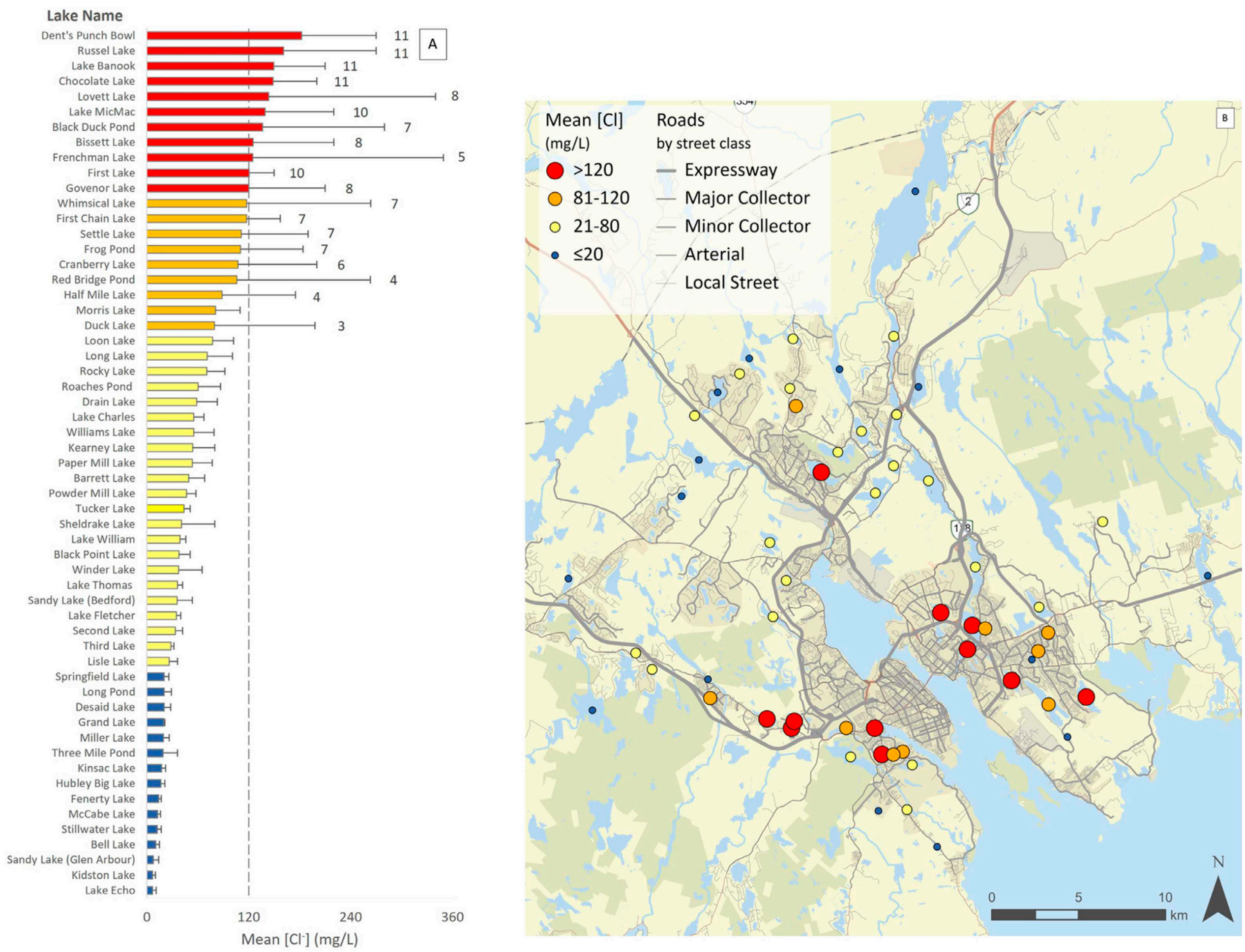

Figure A3. Average lake Cl concentrations from 2006–2011 (error bars show maximum Cl levels with subsequent number indicating number of guideline exceedances (excluding zero counts). The dashed line represents the CCME guideline for chronic exposure to protect aquatic life. The colours of the markers and bars indicate different chloride-impact risk categories. B: Locations of the lakes studied and their Cl levels (Bermarija et al., 2023). Published under Creative Commons Attribution (CC BY 4.0) license.

Radosavljevic et al. (2022) investigated the impact of urbanization on a lake in Ontario (Canada) on a range of water quality parameters include dissolved oxygen (DO), phosphorus (P), nitrogen (N), and chlorophyll-a (chl-a). Analysis of receiving water data collected from 1996 to 2018 indicated that urban development has led to increased chloride concentrations in the adjacent lake in a non-linear manner. For example, from 1996 to the early 2000s, a relatively small increase in chloride concentrations (from 43–50 mg/L) was observed. By 2008, chloride concentrations had increased to ~ 90 mg/L as the watershed's imperviousness rose from 28 % to 36 %. However, from 2008–2012, whilst the impervious surface area increased to 45 %, only a relatively modest rise in chloride concentrations to 100 mg/L was reported. From 2012 to 2018, the most rapid increase was observed with chloride concentrations increasing to a maximum of 175 mg/L (increase in imperviousness to 60 %). The findings implied that increase in chloride concentrations were enhancing water stratification (the separation of a water body into distinct horizontal layers by density and temperature). Such stratification leads to the development of anoxic conditions, as evidenced by changes in the average DO concentrations; from 10.2 mg/L in the epilimnion (upper layer) to 1.1 mg/L in the hypolimnion (the lower layer). From 2005, the hypolimnion was characterized by extended annual periods of DO deficient conditions, sometimes lasting throughout the entire year. Moreover, the average value of the anoxic factor (AF; a function of the number of anoxia and the hypolimnion and lake surface areas) for the period studied was 72 days per year, with exceedances of 80 days reported in 2005, 2008, and 2015. This statistically significant increase in AF suggests worsening anoxic conditions, which in turn implies an increase in internal phosphorus (P) loading (as previously bound P is released from sediments back into the water column) despite a stable or decreasing trend in external total phosphorus loads discharged to the lake (see Figure A4). Such P increases contribute to the lake's eutrophication, as determined through measurement of chlorophyll a (chl-a). The long-term average value of chl-a was 9.8 µgL−1 (range 0.2–41 µgL−1) indicating eutrophic conditions, indicating both the direct and indirect interactions between stormwater runoff and surface water quality.

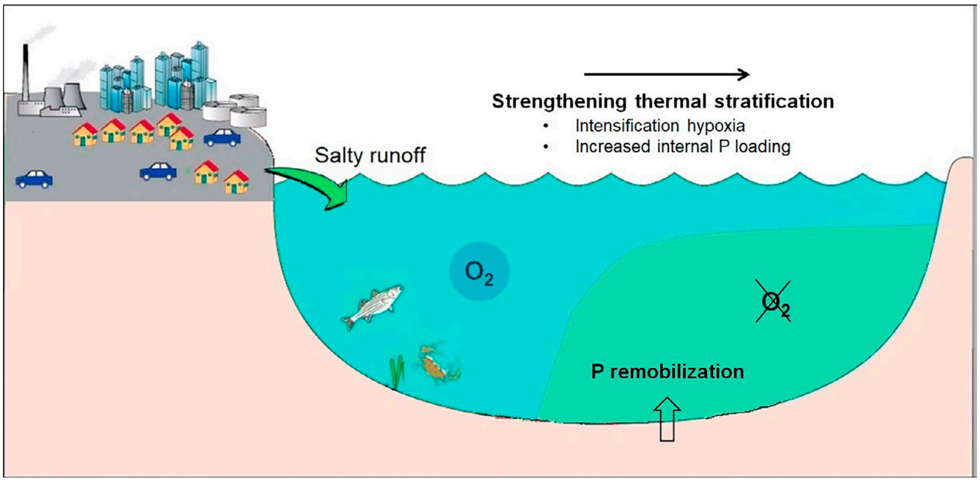

Figure A4. Schematic of the Radosavljevic et al. (2022) study illustrating intensified lake stratification and eutrophication in the lake due to the discharge of salty runoff. Published under Creative Commons Attribution (CC BY 4.0) license.

Extending the examination of road salt and runoff on water bodies, the study by Perera et al. (2009) explores the impact of road salt on stream chloride levels in urban watersheds in Toronto (Canada). Involving a combination of continuous monitoring for electrical conductivity (EC) and grab sampling for the chemical analysis of chloride across four major watersheds, the results established a bilinear relationship between EC and chloride. Probability of exceedance curves were developed for each monitoring station and used to evaluate the proportion of time stream chloride concentrations exceeded any given threshold value. The study drew on threshold concentration values set by the US EPA of 230 mg/L (chronic chloride concentrations) and 860 mg/L (acute chloride concentrations) with and an additional threshold value of 1,500 mg/L. The analysis indicated that the creek whose watershed was located within the city limits was likely to exceed the U.S. EPA's chronic threshold of 230 mg/L approximately 80 % of the time, with the acute threshold exceeded about 35 % of the time. In contrast, a river whose water shed included a less urbanized area only exceeded the chronic threshold 35 % of the time, with the study suggesting this was likely due to the less urbanized nature of its headwater areas. This supports the findings of Bermarija et al. (2023) and Radosavljevic et al. (2022), emphasizing the impact of stormwater runoff on receiving water bodies.

Changes in surface water body temperatures have also been associated with stormwater discharges. For example, Kolath and Egemose (2023) monitored the temperature of an urban stream to assess the impact of urbanization on water temperatures in a Danish catchment. By installing temperature and pressure loggers along the stream – from rural upstream areas, through the urban centre, to downstream sites – they evaluated the influence of urban stormwater discharges on the stream's thermal conditions. Their research shows significant increases in stream temperatures within urban areas in all months ranging from 0.3–1.9 °C, with increases most notable in the summer months (see Figure A5).

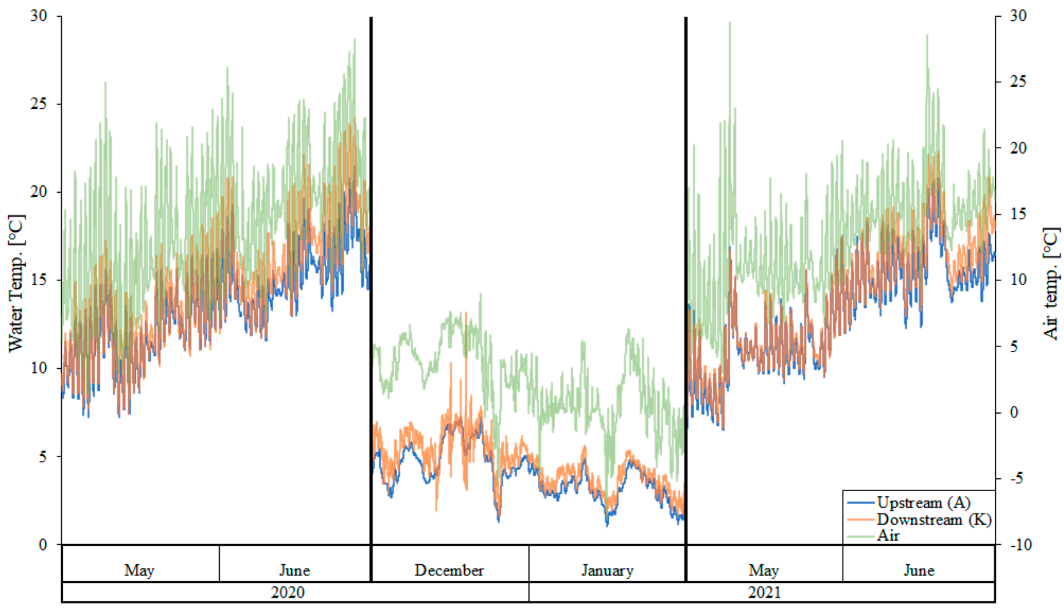

Figure A5. Water temperature fluctuations over a two year sampling period (Kolath and Egemose, 2023). Published under Creative Commons Attribution (CC BY 4.0) license.

NUTRIENTS

Among the studies reviewed, only Long et al., (2014) evaluated the precipitation-driven nutrient dynamics in the watersheds of Hamilton Harbour, Ontario (Canada), focusing on the impact of stormwater, snowmelt inputs, land use, and seasonality. The study involved collecting 87 24-hour level-weighted composite samples from July 2010 to May 2012 from four tributaries to Hamilton Harbour. Samples were analysed for phosphorus- and nitrogen-based nutrients, examining trends with catchment state, land use, and seasonality. Both rainfall and snowmelt events led to consistently higher P concentrations in receiving waters with a similar trend shown for N on a seasonal basis with the authors concluding that runoff has a strong influence on stream quality in both urban and agricultural contexts. Notably, while concentrations of P and many N species increased with flows, nitrate exhibited relatively little change in concentration across a range of flows, indicating that both runoff and groundwater are likely significant sources.

ORGANIC COMPOUNDS INCLUDING MICROPLASTICS

Four studies considered the impact of stormwater discharges on various organic compounds in surface water bodies and/or sediments (see Table A7). Whilst sediments are not the focus of this synthesis, they are included here due to the limited number of receiving water studies identified.

Table A7. Overview of the organic compounds and compartments in studies investigating the impact of stormwater on surface waters

Parameter	Media	Type of effects reported	EQS or reference values	Author and location
PFAS (PFOS)	Water and sediment	Decrease in aquatic PFOS concentrations, persisted in sediments	491 ng/L (pelagic organism chronic PNEC; Environment Canada)	Awad et al., (2011); Canada
PAHs	Sediment	Elevated PAHs concentrations on the top layer (0–2 cm) of the sediment	Swedish EPA sediment classification system	Rentz et al. (2011); Sweden
PAHs, PCBs	Sediment	Samples < 500m from shore exceed PAH threshold values. PCB exceed selected threshold and effects levels	Finnish environmental threshold values; US EPA effects ranges; OSPAR assessment criteria	Honkonen and Rantalainen (2013); Finland
Microplastics	Water	Stormwater identified as main contributor of microplastics	None provided	Rochman et al. (2022); Canada

Three studies focused on investigating the impact of stormwater runoff on receiving water bodies, specifically in terms of organic compounds such as PFAS, PAHs, PCBs and TPHs. For example, the study by Awad et al. (2011) evaluated the long-term impacts of an accidental release of 22 m^3 fire-fighting foam and 450 m^3 of sprinkler system water which entered storm drains and discharged to receiving waters at Toronto Airport (Canada) in 2000. Surface waters, sediments and fish sampling campaigns were undertaken in 2003, 2006, and 2009, with samples analysed for selected PFAS, with samples collected at three locations upstream and seven locations downstream of the 2000 spill outfall covering a 20 km stretch. The highest PFOS (perfluorooctanesulfonic acid) concentration were observed at the creek sampling site closest to the stormwater outfall that discharges from the airport, with concentrations of 690 ng/L and 290 ng/L reported, in 2003 and 2009, respectively (no data for 2006). Whilst a marked decrease in PFOS levels water samples was observed over time (from ~10^6 ng/L one day after the spill to 29–41 ng/L in 2009), sediment concentrations in the creek which directly received the foam discharge, remained consistent (i.e. 13 ng/g dw) a decade after the spill. Details of the impact on whole fish and fish liver (PFOS reported to induce changes in fish liver cells) are described in the ecological impact section (see Section A4.1.2).

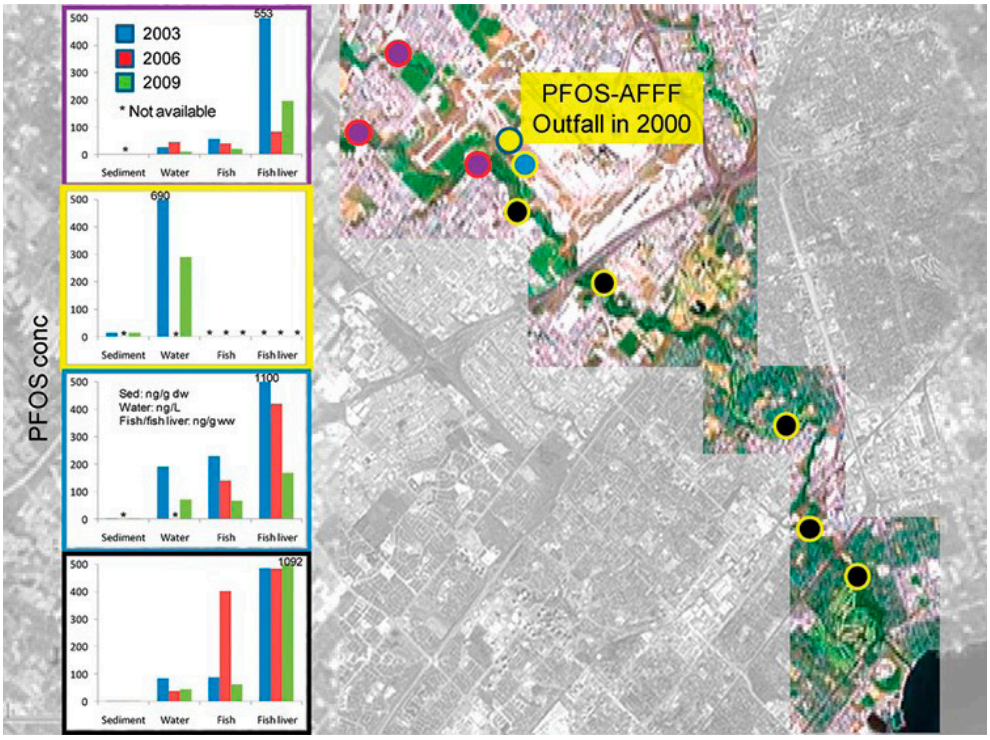

Figure A6. A map illustrating the sampling locations along the creek studied following the accidental release of PFCs, accompanied by graphs that summarize the spatial and temporal variations in PFOS concentrations. Reprinted with permission from Awad, E., Zhang, X., Bhavsar, S.P., Petro, S., Crozier, P.W., Reiner, E.J., Fletcher, R., Tittlemier, S.A., Braekevelt, E., 2011. Long-Term Environmental Fate of Perfluorinated Compounds after Accidental Release at Toronto Airport. Environ. Sci. Technol. 45, 8081–8089. https://doi.org/10.1021/es2001985. Copyright 2011, American Chemical Society.

Rentz et al. (2011) investigated the impact of stormwater runoff on the quality of a closed bay and the Lule River in Northern Sweden, with a particular focus on PAHs concentrations in different sediment layers. The study compared these concentrations with those of a reference site located at a greater distance from the shore. The study identified significantly elevated levels of several PAHs, including pyrene, phenanthrene, anthracene, fluoranthene, benzo(a)anthracene, chrysene, benzo(k)fluoranthene, and benzo(a)pyrene in the top sediment layer (0–2 cm). The total concentration of the US EPA 16 PAHs was 1,200 µg/kg dry mass (dm) for the closed bay in comparison to 93 µg/kg (dm) at the off shore reference site. In deeper sediment layers (14–16 cm), PAH concentrations decreased, with a 16 PAH sum total of 300 µg/kg (dm) reported in the bay in comparison to 250 µg/kg (dm) at the reference site. In comparison to Swedish Environmental Protection Agency sediment guideline values developed for 11 PAHs, sediments ranged in quality from highly contaminated (surficial sediments in the bay; maximum concentration of 1,097 µg/kg (dm)) to moderately contaminated (deeper sediment layers in both the bay and at the reference site). The study concluded stormwater runoff was as a key contributing source to the high level of contamination identified. Honkonen and Rantalainen (2013) also examined the effects of urbanisation on the occurrence and distribution of organic pollutants, PAHs and PCBs, in the sediments of a lake located near the city of Lahti (Finland). The study involved the collection of surface sediment samples from selected points throughout the lake, including four major stormwater outlets and 15 urban stormwater traps. Samples were analysed for the US EPA 16 PAHs and 28 PCBs. Results indicated sediments close to the urban shore typically reported higher pollution levels, with PAH 16 concentrations ranging from 3–16 mg/kg (dm) and PCB concentrations from 0.02–0.3 mg/kg (dm). The authors conclude that urban stormwater discharges significantly affect pollution levels, especially near stormwater discharge points. The study highlighted that all sediment samples within 500 meters of the shore exceeded Finnish threshold PAH levels. Elevated PCB concentrations were also observed, with seven PCBs exceeding selected Finnish, US EPA and OSPAR assessment criteria and effects levels.

Only one study explored the role of stormwater runoff in contributing to microplastic pollution in surface waters. The study by Rochman et al. (2022) on Lake Ontario (Canada) was part of a wider analysis across four North American regions, involving the characterization and comparison of microplastic concentrations in agricultural runoff, stormwater runoff, and treated wastewater effluents. Analysis focused on characterising microplastics in terms of shape, polymer type, and concentrations (by number). Concentrations varied between runoff and effluent types per site, with relative abundance between pathways also showing variation. Overall, average microplastic concentrations were highest in agricultural runoff at two sites, wastewater effluents at one site and in stormwater runoff at a fourth site emphasising the need for local monitoring to inform water resource managers' understanding of sources and pathways of MP at a local level and identifying appropriate mitigation measures, e.g. use of bioretention cells vs washing machine filters in relation to knowledge on contribution from stormwater vs wastewater effluents. In terms of receiving water body concentrations, a key finding was that pathway data (i.e. discharge type) did not always correlate with receiving water microplastic profiles. Discussions note that sampling surface water concentrations alone may lead to discrepancies as this does not account for microplastic fate once they enter the water body, concluding that water sampling downstream alone is likely not the best way to assess inputs from upstream pathways, recommending the inclusion of sediment samples within source identification studies.

METALS

Several studies have explored the effects of stormwater runoff on receiving water bodies with a focus on metal contamination, with all identified studies carried out in Sweden and Finland. The majority of these studies focused on analysing metal species in water and/or sediment samples. A summary of the studies discussed in this section is provided in Table A8.

Table A8. Overview of the stormwater runoff receiving water studies which investigated metals

Metals	Media	Type of effects reported	EQS/reference values	Author and location
Cd, Cu, Pb, Zn	Water and suspended sediments	Elevated Cu, Zn and suspended sediments during storm events in selected rivers	Swedish EPA EQS	Frogner-Kockum et al. (2020); Sweden
Cd, Cu, Pb, Zn	Water and suspended sediment	Elevated Cu and Zn in streams receiving runoff from the urban areas.	Reference values and Finland Ministry of Environment intervention levels	Kuusisto-Hjort and Hjort (2013); Finland
As, Cd, Co, Cr, Cu, Hg, Ni, Pb, Zn	Sediment and pore water	Elevated Cu, Pb and Zn concentrations in comparison to reference values	Swedish EPA Reference values	Rentz et al. (2011); Sweden
SiO_2, Al_2O_3, CaO, Fe_2O_3, MnO, Na_2O, P_2O_5,TiO_2, As, Cd, Co, Cr, Cu, Hg, Ni, Pb, S, V, Zn	Sediments	Elevated of Fe_2O_3, S, Cd, Co, Cr, Cu, Ni, Pb and Zn at sites receiving stormwater runoff	Swedish EPA background values Reference concentrations (Rentz et al., 2011)	Blecken et al. (2012); Sweden

For example, Frogner-Kockum et al. (2020) examined the concentrations of Cr, Cu, Pb, Zn in water and suspended sediments in three urban rivers (Gothenburg, Sweden), during various hydrological events including a spring flood, and wet and dry summer periods. Findings indicated that, particularly during the wet weather period, surface water concentrations of Cu and Zn exceeded respective EQSs. The study also measured suspended sediment concentrations at various locations under differing hydrological conditions, noting an increase in suspended sediment levels during wet weather events. This increase in suspended sediment concentrations implies a higher potential for particle-bound metal transport in urban watercourses during wet weather, further supported by the strong correlation observed between metal concentrations and suspended sediment levels, suggesting that runoff is a significant factor in contaminant transport. The study concluded that this relationship leads to immediate increases in contaminant levels within surface waters, which can persist beyond the initial rainfall event. These findings support earlier research by Kuusisto-Hjort and Hjort (2013) involving the analysis of basal and suspended sediments in 67 streams in the Helsinki metropolitan area (Finland) for several metals including Cd, Cu, Zn, and Pb. Findings indicated that the concentrations of these metals were significantly higher in suburban areas compared to the six background sites where land use was dominated by forest and fields. In particular, Zn and Cu concentrations exceeded the intervention levels (170 mg/kg for Zn and 50 mg/kg for Cu) for dredged material set by the Finland Ministry of the Environment, with elevated metal levels identified as a clear indicator of stormwater runoff contamination. A further study by Rentz et al. (2011) involving the analyses of sediment and sediment pore water samples for a range of metals, including Cd, Cu, Pb and Zn, also reached similar conclusions. This latter study involved the collection of samples at two locations: an enclosed bay receiving stormwater discharges and a reference site located further offshore as a site not directly affected by stormwater discharges. Results indicated that metal concentrations at the site directly impacted by stormwater runoff (i.e. Cu, Pb, Zn and Cd) exceeded those determined at the reference site. Finally, a study by Blecken et al. (2012) involved the collection of surficial sediment samples (0–2 cm) at sites close to three storm sewer outlets in Luleå, as well as from ditches and a downstream water body. Samples were analysed for a wide range of metals and determined values compared with background levels (see Table A8). This study also concluded that stormwater discharges had a significant impact on the levels of contaminants in the sediments. For example, Cr and Cu levels were substantially higher than background levels, with concentrations of Cd, Cu, Cr, Ni, Pb, and Zn exceeding those reported at the reference site.

A4.1.2 Ecological Impact

Five studies focused on analysing the effects of stormwater discharges on the ecological health of downstream water bodies though the collection and analysis of biota from the field (see Table A9 for an overview). Whilst not strictly stormwater runoff, given the limited number of articles two further 'aligned' studies were also included; Meland et al, (2010) assessed the ecological impacts of tunnel wash runoff and Awad et al. (2011) explored impact of an accidental spill and sprinkler water which entered the stormwater drainage system on fish species. Among these seven articles, six studies were carried out in Canada, with a single study in Norway, indicating the relatively limited field-derived evidence base (an overview of laboratory-based exposure studies is included in Section A4.2). These studies involved evaluating the impact on a range of different trophic groups including fish, filter feeders (e.g. mussels), aquatic invertebrates (*Cladocera*) and decomposers (bacteria) sourced from impacted water bodies.

Table A9. Overview of the studies investigating ecological impacts of stormwater discharges to receiving waters

Species	Type of test/assessment	Type of effects reported	Author
Fish *Notropis cornutus*	Analysis of whole fish and fish liver for PFAS	PFAS profile in fish samples collected from the closest location to the downstream of the spill site was dominated by PFOS.	Awad et al. (2011); Canada
Sea trout (*Salmo trutta L.*)	Length of fish collected upstream and downstream of a pond receiving tunnel wash runoff	Growth reduction in sea trout in the downstream compared with sea trout in the upstream	Meland et al., 2010a); Norway
Mussel *Lasmigona costata*	Metal concentrations in gills, reproductive status and immune function assessment	Elevated levels of metals in gills and declines in total mussel wet weight, shell length and lifespans	Gillis (2012); Canada
Mussel *Lasmigona costata*	Biomarkers of metal exposure, oxidative stress and general health	Elevated levels of metals in gills, signs of oxidative stress and cellular damage, possible metabolic disruptions and overall health degradation	Gillis (2014); Canada
Larval and juvenile stages of the mussel *Lampsilis fasciola*	Acute toxicity of NH_3, Cl, Cu, K	Winter runoff (salt applications) acutely toxic to mussels at early life stages in laboratory settings, absence of mussel population in one area implied the impact of salt-laden runoff	Gillis et al. (2022); Canada
Cladocera sub-fossils in dated sediment cores from six lakes	Paleolimnological analyses to assess the long-term effects of chloride	Distinct taxonomic shifts, including increases in relative abundances of *Chydorus brevilabris*, *Eurycercus spp.*, and the *Daphnia pulex* complex, and decreases in the relative abundance of *Bosmina spp.* and *Alona spp.*	Valleau et al. (2022); Canada
Bacteria (Faecal Coliform)	Statistical analysis	Higher precipitation events associated with increased faecal coliform levels in water bodies	St Laurent and Mazumder (2014); Canada

Two studies considered impacts on fish: Awad et al. (2011) and Meland et al. (2010a). The study by Awad et al. (2011) explored the ecological effects of an accidental spill of fire-fighting foam and sprinkler water which entered the stormwater drainage system at Toronto International Airport (Canada) and discharged into adjacent receiving waters. Surface waters, sediments and fish sampling campaigns were undertaken in 2003, 2006, and 2009, with samples collected at three locations upstream and seven locations downstream of the 2000 spill outfall covering a 20 km stretch analysed for selected PFAS (see Section A4.1.1. for discussion on chemical impacts). The study included the collection and analysis of the common shiner fish (*Notropis cornutus*) (both as whole fish and liver samples). Data sets from both creeks identified differences between fish and creek water PFAS concentrations. It was observed that short-chain PFAS and perfluorocarboxylic acids (PFCAs) were less common in fish compared to the water, indicating that different compounds have varying tendencies to bioaccumulate. Notably, PFOS emerged as the predominant compound in the fish collected near the spill site, showing significantly higher levels than in fish collected further downstream, suggesting continuous PFOS exposure. The profile of PFAS in fish livers mirrored that of the whole fish. PFOS emerged as the predominant compound, with its relative abundance increasing in fish sampled at progressively further downstream sites. The findings highlight the persistence of PFOS and other perfluorinated compounds in the environment, which can enter water bodies via stormwater management infrastructure and accumulate in the food chain.

A second fish study by Meland et al., (2010a) examined the effects of runoff from tunnel wash water on sea trout (*Salmo trutta L.*) in a stream receiving discharges downstream of a sedimentation pond designed to treat the tunnel wash water. The researchers collected sea trout samples both upstream and downstream from the pond's discharge point. These fish were measured, and their sizes were compared with historical data. The findings revealed that, despite passing through a sedimentation pond, the runoff from tunnel wash water remains significantly contaminated with a 21 % reduction in growth observed in the fish caught downstream of the pond outlet compared to those from upstream. This growth reduction is likely attributable to exposure to the polluted runoff from nearby tunnels and roads, especially since no such difference was observed prior to the construction of new tunnels and the sedimentation pond (see Figure A7).

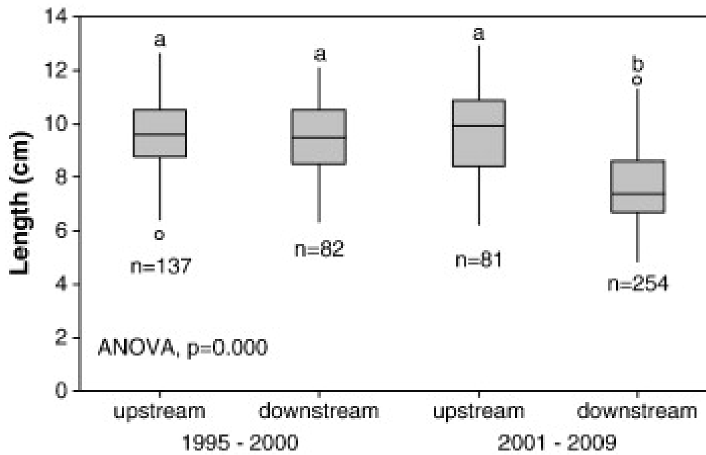

Figure A7. Box plot of fish length upstream and downstream of a pond which began receiving tunnel wash water runoff in 2000. Reprinted from Sci. Total Environ. vol. 408, Meland, S., Borgstrom, R., Heier, L.S., Rosseland, B.O., Lindholm, O., Salbu, B., Chemical and ecological effects of contaminated tunnel wash water runoff to a small Norwegian stream, pp 4107–4117, Copyright 2010, with permission from Elsevier.

Three studies focused on the impacts of urban stormwater discharges on various species and life stages of freshwater mussels (Gillis, 2012; Gillis et al., 2014; Gillis et al., 2022). The 2012 study investigated the cumulative impacts of stormwater runoff and municipal wastewater on adult freshwater mussels (*Lasmigona costata*) within the Grand River watershed (Canada). Focusing on four sites along a gradient of contamination (i.e. level of proximity to municipal wastewater treatment plants and stormwater runoff discharges), the study spanned three field seasons. During this time, mussels were collected and analysed for various health indicators including age, condition factor (total mussel wet weight (g)/shell length (mm)), reproductive status, tissue metal content, and immune function. Results identified impacts of stormwater runoff and municipal wastewater on the health of mussels, notably elevated levels of metals (e.g. Al, Co, Cr, Pb, Ni, and Zn) in the gills of mussels found downstream of urban areas. Furthermore, the mussels in areas affected by stormwater runoff and wastewater showed a decline in condition and lifespan, highlighting the adverse health effects of exposure to urban pollution. The study also observed variable patterns in immune function, specifically in phagocytosis activity, over the study period, indicating the complex nature of the relationship between environmental stressors and aquatic organisms' immune response.

In a further study, Gillis et al. (2014) focused on the effects of urban pollution on freshwater mussels (*Lasmigona costata*), through an evaluation of oxidative stress caused by exposure to contaminants commonly found in municipal wastewater and stormwater runoff. This research, conducted at the same locations as the Gillis et al., (2012), compared mussels from areas with differing levels of urban influence to understand the physiological impact of pollutants such as Pb, Cr, Zn, and silver (Ag). The findings indicated the bioaccumulation of metals in the gills of mussels from sites downstream of urban areas. Specifically, concentrations of Pb, Cr, Zn, and Ag were found to be up to twenty-two times higher than those in mussels from less impacted areas upstream. Moreover, these mussels exhibited signs of oxidative stress and cellular damage, highlighting the adverse effects of urban contaminants on receiving water fauna. A significant finding was the reduced antioxidant capacity in mussels from the most polluted sites, indicating their compromised ability to counteract oxidative stress. Additionally, general health indicators e.g. gill tissue protein levels, were negatively impacted, suggesting possible metabolic disruptions due to their exposure to urban derived pollutants. This study provides further evidence of the impacts stormwater runoff (in combination with wastewater effluents) can have on aquatic life.

A later study by Gillis et al. (2022) examined the effects of winter stormwater runoff (i.e. with elevated levels of road salts) on a different species of freshwater mussels (*Lampsilis fasciola*), focusing on the toxicity of runoff and meltwater on mussel populations under both field and laboratory conditions. Within the field, receiving waters and associated mussel populations located near bridges discharging runoff were analysed for a range of parameters. The findings indicated that runoff had elevated chloride, potassium, ammonia and Zn concentrations. Despite elevated concentrations, field mussel populations downstream of runoff input site showed no significant reduction in viability. However, mussels were absent from one area due to anoxic conditions (which can be driven by the presence of chloride-induced stratification within receiving waters), while a further population of mussels located further downstream from the bridge displayed a higher abundance and diversity of mussel species. Laboratory-based investigations included acute toxicity tests involving exposure of mussel larvae to various conditions, including undiluted and diluted runoff samples as well as sodium chloride solutions to determine the target species sensitivity to salt. Taken together, results suggest that salt-laden stormwater runoff can impact mussel distribution, abundance and diversity, but further research is required as other contributing factors e.g. agricultural impacts were also present. Valleau et al. (2022) took an alternative approach to evaluating the impacts of road runoff derived salts on receiving waters, analysing sediment cores from six lakes to track changes in subfossil Cladocera populations. The study focused on comparing the composition of Cladocera subfossils before and after road salting application began in the 1950s, to determine the ecological effects of chloride-laden runoff on these aquatic organisms. The findings showed significant changes in Cladocera communities in lakes exposed to road salt, especially those with higher chloride levels (32.8–90.9 mg/L), in contrast to a reference lake with chloride concentrations of < 1mg/L. Results indicated that prior to 1950, Cladocera communities were of a stable composition dominated by *Bosmina spp.* Post-commencement of road salting activities, there was a taxonomic shift in lakes impacted by road salt with an increase in the abundance of species such as *Chydorus brevilabris*, *Eurycercus spp.*, and *Daphnia pulex*. These changes were most evident in lakes with the highest chloride concentrations, implying a direct biological reaction to the rising chloride levels due to road-salt runoff.

Whilst not directly considering ecotoxicity, a study by St Laurent and Mazumder (2014) involved monitoring faecal coliforms (FC) concentrations in urban receiving waters under a range of hydro-meteorological conditions. The study analysed seasonal and inter-annual variations in hydro-meteorological factors to understand their influence on FC levels. Findings indicating the significant role of surface runoff in their transport from land to receiving waters, with larger precipitation events (in terms of rainfall depth) associated with increased FC levels in receiving waters. The study concluded that increased precipitation patterns (depth, frequency and intensity), as projected under many climate change scenarios, could lead to higher levels of faecal contamination in surface waters.

A4.1.3 Hydrogeomorphological impacts

Whilst evaluation of the chemical and ecological impacts of stormwater runoff has been the focus of several studies, research into the hydrogeomorphological effects has been relatively neglected with only three studies (all undertaken in Canada) identified. These include a study by Van Duin and Garcia, (2006) which investigated the impacts of urbanisation on a creek in Alberta (Canada). The study involved monitoring channel widths and depths at several points along a stretch from an upstream rural location, through an urbanised reach to locations downstream of the urban area. The study found that urbanisation leads to significant increases in runoff volumes and flow rates discharging to the creek, causing flooding, erosion and water quality problems downstream of urban areas. The study emphasises the importance of controlling runoff volume to prevent channel enlargement and mitigate downstream flooding, noting the use of runoff volume control measures could also significantly reduce contaminant loadings to urban water bodies. At a larger scale, a study by Ariano and Oswald, (2022) investigated the impact of varying levels of impervious surface cover and the connectivity of piped drainage systems on urban water hydrology. The study involved the analysis of 176 mesoscale watersheds to evaluate the effects of total impervious area (TIA) and a modified version of TIA that accounts for transfer through storm and combined sewer systems (referred to as sewer-corrected TIA) on urban catchment hydrologic responses. Findings indicated that TIA is as robust a predictor of hydrologic behaviour in watersheds as sewer-corrected TIA with the advantage that the former is much easier to measure. A further study applied a multi-disciplinary approach to understanding the impacts of urbanisation at a watershed scale (Frenchman's Bay, Canada), drawing on geology, geochemistry, sedimentology, hydrogeology, hydrology, geophysics and aquatic data sets to assess urban impacts (Eyles et al., 2003). The study concluded that urbanisation had significantly altered the hydrological cycle, leading to increased runoff, erosion, and contamination of receiving water bodies with the Highway 401 transportation corridor identified the single largest influence on receiving water quality in the watershed.

A4.2 Receiving water impacts suggested in aligned studies

Whilst not involving the collection of samples from surface waters receiving stormwater discharges or an in-field assessment of ecological impacts, 30 papers on aligned topics (e.g. studies on the quality of stormwater runoff, laboratory-based bioassays using real world or synthetic stormwater samples/sediments) included discussion on possible receiving water impacts. These studies are briefly reviewed here for completeness.

Several stormwater runoff quality studies (including three review articles; Mayer et al., 1999; Ramakrishna and Viraraghavan, 2005; St-Hilaire et al., 2016) discussed potential impacts in receiving waters related to a range of pollutants determined in stormwater runoff. These include nutrients (Brudler et al., 2019), metals (Lindfors et al., 2020; Marsalek et al., 1997; Stone and Marsalek, 1996), organic compounds (Allan et al., 2016; 1997; Müller et al., 2022), tire wear and road wear particles (Rødland et al., 2022), additives used in construction materials (Burkhardt et al., 2011) and pollutants related to road salt applications such as cyanide (Exall et al., 2011). An overview of receiving water impacts suggested within these aligned studies is summarised below.

The most commonly suggested impacts of urban stormwater discharges on receiving waters can be summarised as 'risk to aquatic life' (e.g. Allan et al., 2016; Brudler et al., 2019; Burkhardt et al., 2011; Exall et al., 2011; Lindfors et al., 2020; Marsalek et al., 1997; Mayer et al., 1999; Müller et al., 2022; Rødland et al., 2022; St-Hilaire et al., 2016; Stone and Marsalek, 1996). However, the specific nature of this impact e.g. type and magnitude is not described. Other types of receiving water impacts suggested include an increase in fine sediment loadings, changes in sediment physico-chemical conditions e.g. as result of wet-weather driven resuspension events, channel erosion, alterations to flow regime and changes in water temperature. Increases in receiving water chloride concentrations and associated changes in density gradients and salt-induced stratification are also suggested (Ramakrishna and Viraraghavan, 2005). Brudler et al. (2019) and Ramakrishna and Viraraghavan, (2005) also refer to the stimulation of algal growth which – on decomposition by microbial populations – can lead to a sharp reduction in dissolved oxygen concentrations and the suffocation of other aquatic biota.

In terms of ecological impacts, 12 studies involved a laboratory-based assessment of the exposure of a variety of receiving water species to field-collected sediments (Rochfort et al. 2000; Karlavičienė et al. 2009; Nakajima et al. 2005), runoff derived from various land uses and activities (Popick et al., 2022a; Waara and Färm, 2008; Meland et al., 2010b; Meland et al., 2010c; Bertling et al., 2006; Heijerick et al., 2002), and snow melt (Popick et al., 2022b). Alternative approaches were taken by Bunt and Jacobson (2021) and Sanzo and Hecnar (2006) who evaluated the impacts of synthetic stormwater (e.g. tap water dosed with dog faeces and chlorides at varying concentrations) respectively. A summary of test species, types of tests employed and effects observed are presented in Table A10, with results indicating that a range of impacts have been detected in relation to a variety of test species and life stages. However, not all stormwater discharges (even those that exceed selected EQS) have an effect on all species, and the reasons for this is not yet fully understood.

Table A10. Overview of laboratory-based ecotoxicity studies

Species	Type of test/assessment	Type of effects reported	Author and location
Cress *Lepidium sativum*	Sediment toxicity assessed via seed germination and root growth	Seed germination failure and root length inhibition	Karlavičienė et al. (2009); Lithuania
Bacteria *(V fischeri);* algae *Pseudokirchneriella subcapitata)*	Inhibition of bacterial luminescence; and algal growth inhibition test	Sediment extracts were toxic to both algae and bacteria.	Nakajima et al. (2005); Denmark
Benthic organisms *(e.g. Hyalella azteca,* and *Tubifex tubifex*	Toxicity test (acute or chronic not specified)	No biological effects	Rochfort et al. (2000); Canada
Bacteria (*Vibrio fischeri*); algae (*Raphidocelis subcapitata*);	Inhibition of bacterial luminescence; chronic toxicity (algae)	Inhibition of algal growth in 5 of 16 samples, no inhibition on bacterial luminescence reported.	Popick et al., 2022a; Canada
Bacteria (V fischeri), crustaceans (e.g. *Daphnia magna;* duckweed (*Lemna minor)*	65 samples from 15 storm events were assessed using acute tests	None of the stormwater samples were found to be toxic; growth stimulation reported in 5 *Lemna minor* tests	Waara and Färm, 2008; Sweden
Brown trout (*Salmo trutta L.*)	Fish exposed to tunnel wash water runoff	Accumulation of metals in fish gills, change in blood plasma ions and glucose	Meland et al., 2010b; Norway
Brown trout (*S. trutta L.*)	Fish exposed to four 24 h simulated runoff events	Elevated metals in gills and liver, increased activity of antioxidant defense system; higher blood glucose.	Meland et al., 2010c; Norway
Bacteria (*Alcaligenes eutrophus);* algae *(R. subcapitata)*	Bioavailability of zinc using bacteria and algal growth inhibition tests	Correlation between acute ecotoxic response and total zinc concentration (R = 0.86)	Bertling et al., 2006; Sweden
Bacteria (*A. eutrophus); algae (R. subcapitata)*	Bacteria and algal growth inhibition tests	Toxicity observed and associated with Zn^{2+}	Heijerick et al., 2002; Sweden
Algae (*R subcapitata*) and bacteria (*V. fischeri*)	Inhibition of bacterial luminescence; chronic toxicity (algae)	No significant toxicological trends were observed in either *R. subcapitata* or *V. fischeri*	Popick et al., 2022b; Canada
Creek chub (*Semotilus atromaculatus*)	96 h exposure to fresh and dried dog waste in stagnant and aerated waters	Increases in mortality, and behaviour changes with increasing concentrations	Bunt and Jacobson, 2021; Canada
Larval wood frogs (*Rana sylvatica*)	Acute and chronic toxicity tests	Reduced activity and weight, imapct on metamorphosis	Sanzo and Hecnar, 2006; Canada

Whilst laboratory-based, the studies by Rochfort et al. (2000), Karlavičienė et al. (2009) and Nakajima et al. (2005) involve the use of real world receiving sediments and are therefore described in further detail here. The study by Rochfort et al. (2000) assessed the impacts of combined sewer overflows (CSOs) and stormwater discharges using a benthic assessment technique which integrated sediment chemistry (metals, PAHs, nutrients, and particle size), sediment toxicity to four benthic species *(Hyalella azteca, Chironomus riparius, Hexagenia spp.* and *Tubifex tubifex)*, and the structure of benthic invertebrate communities identified in the field. The study involved the collection of sediments from ten sites (ponds and streams) which were classified according to their level of exposure to stormwater discharges and CSOs. The study's approach was based on the assumption that contaminant concentrations would indicate the degree of exposure to wet-weather discharges and that benthic biological effects would reflect the impact of this exposure. However, the findings challenged this assumption, as even sites with sediments exceeding provincial quality guidelines (e.g. see Reynoldson and Day, 1998) did not consistently show a biological response. Several explanations were proposed, including the possibility that contaminants were not bioavailable, biota was resistant to the concentrations found, or that effects of other confounding factors such as grain size, habitat type, or nutrient levels obscured the impact of contaminants. Furthermore, the study highlighted the importance of considering sediment heterogeneity, including variations in grain size and nutrient levels, which are known to influence toxicity and affect benthic community structure. The limited correlations found suggest a complex interaction between sediment quality and biological responses, emphasising the need for moving beyond simple contaminant concentration measurements to a more holistic understanding of environmental impact.

The study by Karlavičienė et al. (2009) investigated the toxicity of sediments collected from a small urban stream impacted by stormwater discharges through an assessment of the germination of seeds and root growth of the plant *Lepidium sativum* in comparison to an uncontaminated control. The findings showed a different range of toxicity levels from no toxicity in samples collected upstream of the urbanized area to increasing levels of toxicity with increasing levels of anthropogenic influence. The toxic effects were strongest at sampling sites receiving discharges from urbanised areas, with the exception of a single sediment sample collected within an urban area which displayed no toxic effects (believed to be function of relatively low levels of sediment deposition at that site). The final study by Nakajima et al. (2005) involved an evaluation of the bioavailability of PAHs in urban stream sediments through the use of *Vibrio fischeri* (acute Microtox test) and *Pseudokirchneriella subcapitata* (an algal growth inhabitation test). Results indicated that urban stream sediments were toxic to both species, with lower molecular weight PAHs suggested to show greater bioavailability in sediments in contrast to their higher molecular weight counterparts.

A5. The impact of stormwater discharges on groundwater

Of the 200 papers identified for inclusion in this review, 27 papers included the term groundwater in its title and/or abstract. Of the 27 papers short-listed for evaluation, four were not relevant (e.g. referred to treatment or groundwater only referred to in passing) and two papers (dated 1996 and 1986) could not be sourced in full. In terms of geographic location, the majority of articles (16 of the final 21 papers) were undertaken within a Canadian context, with only two located in Denmark and three in Sweden. The final paper is a review article which has an international remit. Figure A8 gives an overview of the focus of the 21 shortlisted papers, identifying that only two of the papers involve the collection of groundwater samples as part of the sampling programme (Eyles et al., 2013; Meriano et al., 2009). However, as both studies are located in the Lake Ontario watershed of Frenchman's Bay and involve the use of the same groundwater wells sampled over the same time period, it is likely that the underpinning data (whilst used in different applications in each publication) originate from a single sampling programme. This emphasises the limited evidence base related to the impact of urban stormwater discharges on groundwater quality or quantity. Seven articles (Bhavsar et al., 2008; Bradford and Gharabaghi et al., 2004; Earon et al., 2012; Indris et al., 2020; Lembecke et al., 2019; Oswald et al., 2023; Williams et al., 2000) draw upon groundwater data from various sources whilst the groundwater remaining papers using a variety of – primarily modelling based approaches – to infer impacts of urban stormwater on groundwater bodies.

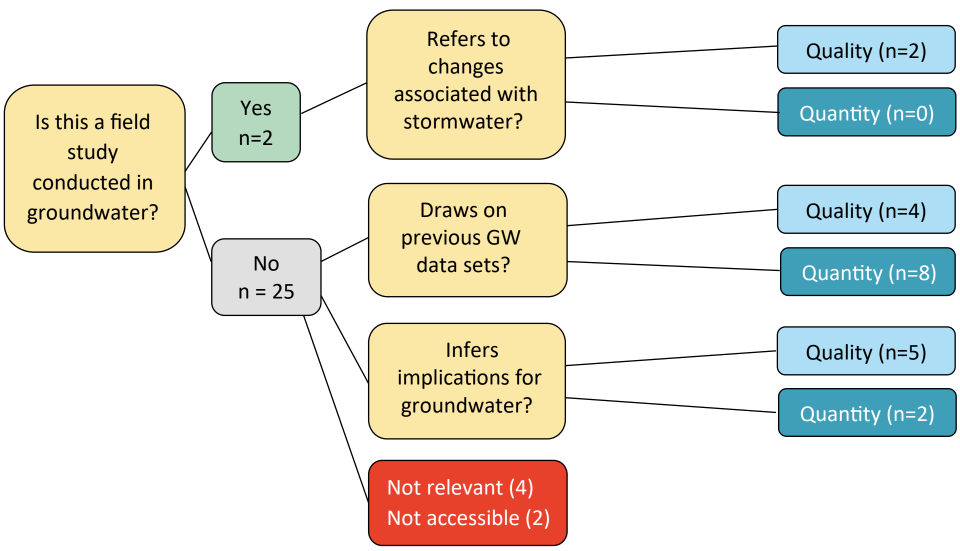

Figure A8. Overview of the focus of the 27 papers shortlisted for inclusion in the stormwater-groundwater synthesis section.

A5.1 Studies considering groundwater quality

Over a three year monitoring programme, Meriano et al., (2009) monitored nine groundwater sampling wells located upstream and downstream of a major highway (AADT 177,000) for a range of parameters. Surface water samples collected from a nearby creek were also analysed for electrical conductivity together with flow and precipitation data, which together supported the development of a chloride mass balance. Of the 1700 tonnes of road salt applied to the study sub-catchment in winter 2003–2004, approximately 50 % was predicted to accumulate in the shallow groundwater resulting in severe degradation of its water quality. Groundwater chloride concentrations peaked at 1664 mg/L in a sampling well downflow of the major highway, with mean concentrations at four of the five sampling wells located downflow of the highway exceeding the Ontario Provincial Water Quality Objective of 250 mg/L (see Figure A9). Whilst rainfall events recharge groundwater and hence provide a pathway for chlorides to enter groundwater, larger storm events can dilute chloride concentrations.

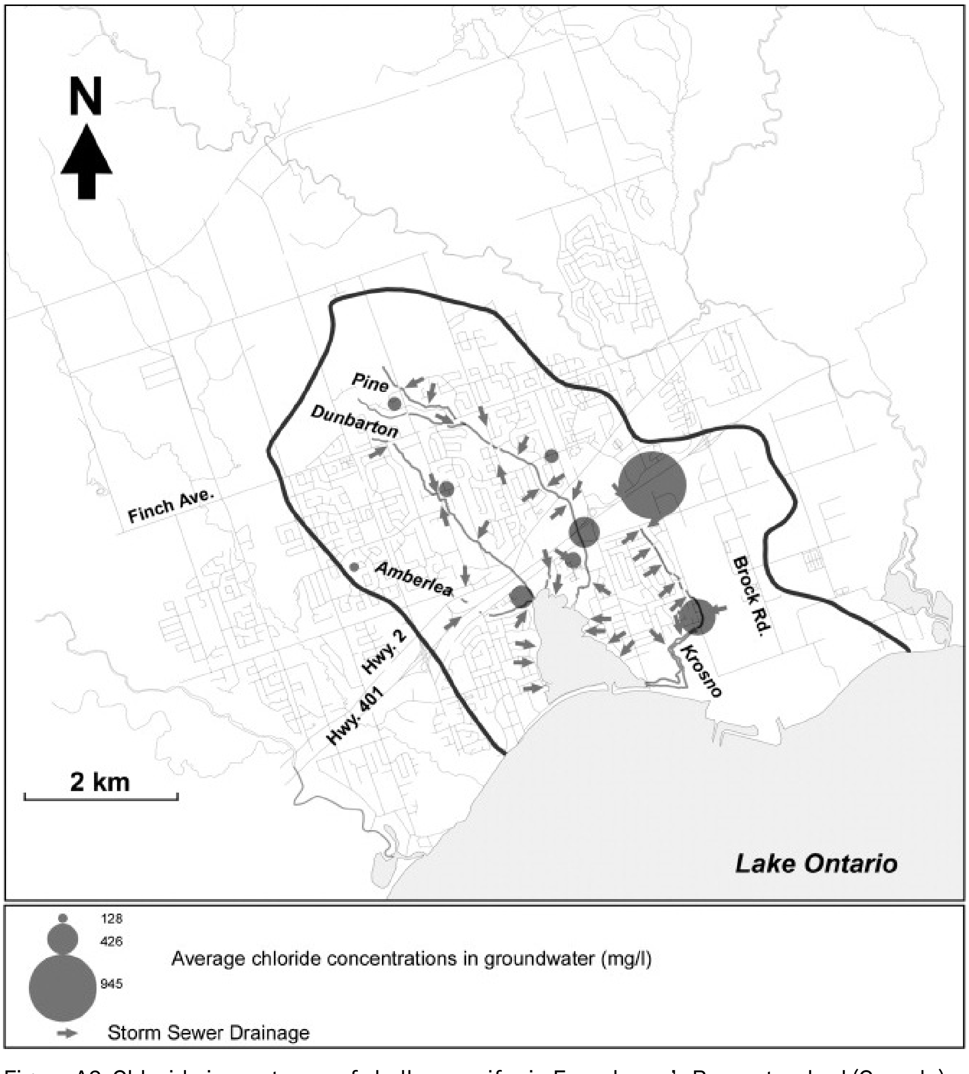

Figure A9. Chloride impact map of shallow aquifer in Frenchman's Bay watershed (Canada). Reprinted from Journal of Contaminant Hydrology, 107(1–2), Meriano, M., Eyles, N. and Howard, K.W.F., Hydrogeological impacts of road salt from Canada's busiest highway on a Lake Ontario watershed (Frenchman's Bay) and lagoon, City of Pickering', pp. 66–81, Copyright 2009, with permission from Elsevier.

However, this dilution effect is only temporary and elevated chloride levels were subsequently reported. The study concluded that – of the 50 % of chlorides entering groundwater within the study catchment – 70 % were estimated to leave via surface water recharge, indicating the dynamic relationship between stormwater runoff, groundwater and surface water bodies. A later study by Eyles et al., (2013) is located in the same catchment and involves collection of samples from the same groundwater wells as those utilised in Meriano et al., (2009). Whilst it's likely the two papers draw on data from the sampling campaign, analysis by Eyles et al., (2013) involves a wider assessment of the impacts of urbanisation on surface water and groundwater bodies at a catchment scale. This includes a comparison of groundwater quality upstream and downstream of Highway 401, reporting mean chloride levels in a groundwater well 10m down gradient of the highway as up to seven times higher than other groundwater samples. Eyles et al., (2013) also discuss the implications of 'brackish groundwater' recharge on surface water, concluding that as the movement of groundwater to surface water is a continuous process, the impact of road salts on surface waters is not only a winter season phenomenon. Further evidence of the sustained year round impact of winter salts on receiving surface waters as a result of groundwater contribution to stream baseflow is the reference to the identification of a small population of Atlantic blue crabs (a marine species) in a Canadian creek in the summer of 2011 (Lembcke et a., 2017).

An alternative approach to exploring road salt contamination of groundwater was taken by Williams et al., (1999), involving the collection of water samples from 23 freshwater springs with the authors arguing that springs – as direct outlets of groundwater – provide an accessible opportunity to monitor groundwater quality. The study site is the Grater Toronto Area (Canada) and the 23 springs – located in both rural and urban areas – were sampled on a monthly basis over a period of 12 months. Water samples were analysed for temperature, chlorides and conductivity, and all springs were sampled for macroinvertebrates. The results of both chemical and ecological analysis were integrated to infer groundwater quality, with data interpretation enhanced through the implementation of laboratory-based acute and chronic toxicity testing of chloride on a range of species. Data indicated that chloride concentrations were highest in springs located close to urban salting sites with a maximum of 1,345 mg/L (mean 1,092 mg/L) reported for a spring adjacent to a bridge and highway in comparison to a mean chloride concentration of 100 mg/L for springs located in rural areas. The study uses the spring water data to conclude that chloride levels were 10–60 times higher in urban areas as direct result of salting activities. In terms of biotic impacts, results suggested it was not possible to categorise species as either high or low chloride tolerant across the range of concentrations determined in the field. However, ecotoxicity data did identify the amphipod *Gammarus pseudolimnaeus* as intolerant to both acute and chronic chloride exposure, with – for example – its reproductive behaviour disrupted by the latter. An aligned study by Betts et al., (2015) integrated data from open access sources – including groundwater wells used for the supply of drinking water, salt application rates and land use type – to identify salt vulnerable areas at several locations within the Grand River watershed (Canada). The relative vulnerability assessment method takes a mass balance approach using data on chloride inputs via surface runoff, shallow interflow and baseflow to predict groundwater chloride concentrations. The methodology was applied to 20 water supply wells, with results showing predictions strongly correlated with measured concentrations (R^2 = 0.84). The study concluded that the chloride application density (a function of the percentage area receiving salt multiplied by a land use-type salt application weighting factor) had the greatest influence on groundwater recharge chloride concentrations.

Two studies considered groundwater impacts from the perspective of nutrients. With a focus on phosphorous (P), Indris et al. (2020) analysed highway runoff and roadside soils for total and soluble P at two sites in Ontario, Canada, using this data to infer implications for receiving surface water and groundwater bodies. Whilst P concentrations in highway runoff were typically higher than those reported for other urban land use types, results also indicated there was no correlation between soil P concentrations and distance from the road or significant differences between roadside soil P profiles and those from non-roadside environments. Hence, the study concluded that P from traffic sources did not pose a risk to roadside soils and – on this basis – did not present a risk to groundwater. The study by Wu et al., (2016) coupled substance flow analysis with a Generalised Watershed Loading Functions model to consider the risks to groundwater posed by nitrogen (N) and P emitted from a range of urban sources under current and future scenarios. The study involved the prediction of N and P loads emitted from various sources (including traffic, parking, horse droppings, aerial deposition and vegetation) to an urban lake via permeable and impermeable surface runoff and groundwater flows. However, whilst the data for permeable and impermeable N and P runoff loads from various sources is assessed as ‘high quality’, groundwater data is reported as poor quality due to limited data availability. Results suggested that under current conditions groundwater flows contribute only a small proportion of total N and P loads to the investigated lake. However, under future scenarios considered (variations in the strength of identified sources and alternative climate predictions), groundwater transport of N was predicted to show a marked increase as a result of climate change.

With regard to pesticides, research by McKnight et al., (2015) integrated data on a range of pesticides in surface water and groundwater with historic data on pesticide usage. Whilst the study catchments are predominantly agricultural, selected sampling locations are in areas classified as urban and hence the study is included within this synthesis. The study includes groundwater pesticide data from previous studies which was integrated with stream data collected under differing hydrological conditions to enable the role of groundwater recharge as a route for pesticides to surface waters to be evaluated. Of the thirty pesticides and their degradation products determined, 21 were detected in groundwater samples with maximum concentrations recorded for groundwater (in comparison to stormwater and baseflow) for nine substances including atrazine (whose use was discontinued in Denmark in 1994), dichlobenil (a pesticide used in urban rather than agricultural applications) and its metabolite 2,6- dichlorobenzamide. Through an analysis of the relationship between groundwater, storm flow and baseflow data, the study concluded that groundwater is a key source of pesticides entering surface water bodies.

The impacts of a new section of the E18 (Stockholm, Sweden) highway on soil (direct measurements) and groundwater (impacts inferred) was the focus of a combined field monitoring and modelling study by Earon et al., (2012). Runoff, snow and soil samples were collected on up to three occasions with samples analysed for a range of basic physico-chemical parameters and metals, with resistivity sampling carried out on eight occasions from 2010–2011). Integration of sampling, resistivity and modelling data sets indicated the year round infiltration of surface runoff and snowmelt through road and road shoulder materials. This was identified as contributing to the environmental risks faced by groundwater, with the role of adsorption to soil materials identified as a risk mitigating factor. With a focus on exploring metal soil dynamics and fate, Bhavsar et al., (2008) couples together multi-species metal transport and speciation models to estimate transport rates and water sediment exchanges. The model is applied to nickel-contaminated mining site in Ontario, Canada, concluding that leaching from soil to groundwater is a major pathway by which Ni is removed from the soil. However, no further information on groundwater concentrations or impacts is provided.

Research by Conant et al., (2019) proposes a process-based framework to enable users to conceptualise groundwater-surface water interactions, identify key process and predict their impacts. The framework consists of a series of diagrams which users can adapt to local circumstances to support development of a conceptual model which can then be used to test hypotheses related to how groundwater-surface water systems function. More specifically, it supports users to consider and characterise the fundamental flow, biogeochemical and biological processes connecting above and below groundwater bodies at a site level, enabling spatial and temporal patterns of groundwater – surface water interactions to be identified and provide a basis for designing site investigations. The framework is applied to three case study sites of differing scales and baseline data availability, illustrating its use in identifying knowledge gaps, test hypotheses, and select locations, sampling times, and instrumentation to support ongoing field monitoring of the potential impacts of groundwater – surface water interactions.

A5.2 Studies considering groundwater recharge

Groundwater is a key water resource and maintaining is quantity is crucial for communities that rely on groundwater as their source of drinking water and for ecosystems that are wholly or partially fed by groundwater discharges (Bradford and Gharabaghi et al., 2004). Groundwater is replenished by precipitation e.g. rainfall, surface runoff and/or snowmelt or via discharge from surface water bodies e.g. lakes and rivers. Groundwater recharge refers to the processes by which water above the surface migrates through soils and rocks into groundwater bodies. Urbanisation spatially and temporally changes patterns of groundwater recharge, and the need to counteract these impacts has been recognised in stormwater management planning e.g. the Ontario stormwater management guidelines recommend use of a groundwater modelling approach to determine appropriate recharge criteria with a view to ensuring pre-development groundwater recharge rates are maintained post-development (Bradford and Gharabaghi et al., 2004).

Within the short-listed papers, eight articles consider groundwater recharge processes as either the sole focus of the study or as a component of an integrated water resource management perspective. For example, modelling the impacts of predicted climate change-driven change in rainfall patterns on groundwater recharge has been considered from seasonal (Livingston et al., 2020), spatial (Jyrkama and Sykes, 2007) and water resource availability (Saha, 2015) perspectives.

All three studies are set in a Canadian context and consider the impact of changes in precipitation and temperature on surface runoff volumes, steam flow and evaporation at a variety of scales, with results varying from decreases to increases in groundwater recharge depending on the climate scenarios included.

A further development of these hydrology-focussed studies is research by Liu et al., (2020) which linked surface water and groundwater flow models with a flow-biota model to evaluate the impact of alternative climate change scenarios on a Danish river basin dominated by groundwater flows. Results indicated that – whilst total water yield within the catchment was not predicted to change significantly under either scenario – spatial patterns of groundwater discharge to streams were affected and (together with predicted increases in irrigation demand) contributed to reduced stream flow with associated reductions in fish and macrophyte indices of up to 14 and 11 %, respectively (Liu et al., 2020). In a complimentary study, Boumaiza et al., (2022) used historic temperature and water budget data generated over a period of 100 years to evaluate how groundwater recharge potential has varied spatially and temporally at a regional scale. Results indicate that despite increasing trends in evapotranspiration and surface runoff volumes, potential groundwater recharge rates showed an average increase of 0.7±0.4mm/year as a function of increased precipitation patterns.

Research by Wijesekara et al., (2014) initially took an historic approach, modelling the impacts of previous land use changes on surface runoff volumes, evapotranspiration and infiltration. This knowledge was then applied to predict the impact of four alternative land use change scenarios on the identified hydrological processes at a catchment level. However, whilst results indicated that patterns of development impact hydrological processes at a sub-catchment level, at the time of implementation (2014) insufficient data was available to enable the impact of land-use change on the groundwater table to be assessed. With a focus on interactions between groundwater and surface water, Beckers and Frind (2000) and Mojarrad et al., (2023) developed and implemented different steady state modelling approaches to identify the role of baseflow in a swamp water balance and the impact of hyporheic flow on groundwater discharges, respectively. The water balance study integrated hydrostratigraphy data (soil types and conductivities) with a site specific water mass balance (including surface runoff and groundwater recharge components), enabling the impact of local changes in recharge behaviour on base flow to be quantified. The study by Mojarrad et al., (2023) also integrated hydrologic, geologic and geographic data sets within a physically-based model to investigate the impacts of hyporheic flow on groundwater discharge behaviour. The study included a comparative assessment of flow fields (i.e. with and without consideration of hyporheic flows), highlighting the need to include hyporheic flow within surface water groundwater interaction assessments.

Most of the studies discussed above refer to limitations in the availability of groundwater data sets. This is also raised in a review of urban hydrological sciences (Oswald et al., 2023) which refers to both the increasingly important role of urban groundwater bodies in providing resilience in water supply systems and that there is limited groundwater data available, as function of minimal monitoring of groundwater in urban areas and limited access to data that does exist due to security concerns over sharing information on actual/potential drinking water supply areas (Oswald et al., 2023).

A6. Conclusions

This review presents a synthesis of findings from a range of studies which – directly or indirectly – consider the impact of urban stormwater discharges on receiving water bodies within a cold country context. Over 200 papers were reviewed for this report, with findings from 77 papers synthesised to address the core research question: how does stormwater runoff impact the chemical quality, geomorphology and/or ecological status of receiving waters? A limitation of these studies is that only 28 of the 77 papers involved the physical collection of receiving water samples from study sites (26 papers related to surface waters and 2 papers related to groundwaters), with the remaining studies either drawing on historic data sets, inferring impacts from e.g. modelling studies or suggesting potential impacts based on stormwater runoff concentrations. Further, in the absence of a standard method for sampling and evaluating stormwater, these studies utilise a wide range of sampling methods, analytical protocols, experimental designs and test species. Hence it is not possible to undertake a meta-analysis of the available data sets. However, although a robust cross-comparison between studies is not currently feasible, it is possible to identify trends within the emerging evidence base as to the types and magnitudes of impacts that could be anticipated at sites receiving urban stormwater discharges.

The majority of research on urban stormwater-associated impacts within receiving waters to-date has been undertaken by Canadian researchers where the focus has been chlorides as a function of its seasonal application as a component of road salts and its highly soluble behaviour. The research base includes studies which have found a significant correlation between urban factors and the salinization of lake waters, identifying urban areas (i.e. road networks and stormwater drainage systems) as primary sources of chloride contamination. Surface water chloride concentrations in excess of acute and chronic receiving water threshold concentrations have been widely reported, with associated impacts including chloride-driven stratification of water bodies. Under such conditions, data indicates that lower water layers can become anoxic which can also lead to the release of previously bound P from sediments into the water column. Similar relationship between urbanisation and groundwater have been reported for chlorides, with groundwater chloride concentrations increasing in proximity to urban areas. The dynamic relationship between stormwater, groundwater and surface water has also received some attention, whereby stormwater transfers chlorides directly (via piped stormwater discharges) and indirectly (infiltration to groundwater which recharges surface water bodies) to surface waters. However, whilst stormwater runoff to surface and groundwaters is a seasonal input (linked to winter maintenance activities), groundwater recharge of surface waters can be a year-round process. Hence, a seasonal episodic input can effectively – through contamination of groundwater base flows – become a year round impact phenomenon. Whilst the evidence base is strongest for chlorides, a diversity of other substances has been reported to demonstrate similar behaviour trends i.e. elevated concentrations of e.g. metals, PAHs, PCBs, PFAS in surface waters and surface water sediments in areas adjacent to/downstream of urbanised areas. For example, studies have reported concentrations of several metals in receiving waters and sediments in excess of receiving water EQS/threshold values at sites receiving stormwater discharges. Similar trends have been reported for PAHs, with single studies identifying stormwater runoff as a key source of microplastics discharging to a lake and that an accidental release of PFAS and sprinkler water that discharged via a stormwater drainage system continued to have impacts in receiving water almost 10 years after the original spill event (when sampling stopped).

Although a reasonable degree of confidence can be associated with the statement stormwater runoff negatively impacts the chemical quality of receiving waters, the likelihood that elevated pollution concentrations will translate into negative ecological impacts is less certain. Several detailed field and laboratory-based studies employing a variety of test species have reported changes in a range of endpoints, from community composition and abundance to changes in growth, reproduction, gene expression and mortality. The reviewed studies typically focused on different pollutants or pollutant groups, with impacts linked to the presence of metals (especially Zn and Cu), chlorides, PAHs and PFAS. Results show that different species – and even life stages with a single species – can vary in their sensitivity to particular pollutants. However, whilst stormwater runoff-related toxicity studies show there is often an impact, there are also studies where no ecological impacts have been reported, even at sites where runoff concentrations exceed EQS. A range of factors driving this differential response have been proposed, including water hardness (studies have shown increasing toxicity with decreasing water hardness levels), 'upstream ecologies' acclimated to pollution sources further upstream), potential for antagonistic/synergistic effects associated with the unknown profile of stormwater discharges and natural geology. However, whilst such factors are suggested at high-level, much work remains to be done in relation to understanding the impact and interactions of pollution mixtures on receiving water biota (at both acute and chronic levels) before the locations/scenarios where urban stormwater discharges will have a deleterious effect on receiving water ecology can be robustly predicted. Only three studies on the hydrogeomorphological impacts of stormwater runoff within receiving waters could be sourced. Whilst each study adopted a different approach and scale, all three concluded that urbanisation impacts the hydrological cycle, with increased stormwater runoff volumes and flow rates leading to negative receiving water impacts including erosion, and contamination of receiving waters and their sediments.

In terms of understanding the impact of urban stormwater discharges on receiving surface waters and groundwaters in a cold climate context, the evidence base is growing. To-date the majority of studies have focused on surface water impacts from a chemical quality perspective, with groundwater impacts a relatively neglected topic. Most of the groundwater studies identified refer to limitations in the availability and/ or quality of groundwater data sets. Given the importance of both surface water and groundwater as a source of drinking water in many areas, this is a topic that requires urgent attention. However, even in relation to surface water impacts, whilst the occurrence – and to a lesser degree ecological impacts – of a range of stormwater pollutants have been more widely evaluated, there is a lack of depth within the emerging research base i.e. no parameter has been the subject of ≥5 independent studies in this specific context. Whilst understanding of impacts of stormwater discharges on surface water ecology have received some attention, this has typically focused on acute impacts as soluble pollutant loads are typically short in duration (i.e. a per event basis) with relatively little research addressing repeated exposures (e.g. multiple urban stormwater discharges) at either acute or chronic concentrations. Further, whilst 'good ecological status' under the EU Water Framework Directive is defined in relation to chemical, ecological and hydrogeological components, reviewed studies typically focus on chemicals or biota or hydrogeomorphology; no single study addressed the receiving water impacts of stormwater discharges from all three perspectives.

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Appendix B

Design, treatment performance and maintenance requirements

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B1. Summary

This report reviews the international peer-review literature relating to design, treatment performance and maintenance requirements of stormwater treatment technologies from a cold climate perspective. The research studies included in this report were identified using a systematic approach and consider treatment performance, design criteria and maintenance of stormwater technologies in cold climate. Several different treatment technologies were identified where the treatment efficiency for different pollutants was evaluated in full-scale facilities located in cold climate areas. Overall, there were only a few field studies available and too few to compare and assess the performance of different technologies to each other. The highest number of studies (six) was identified for bioretention systems followed by ponds and wetlands, reactive filters (three), and permeable pavements and swales (one). There were also examples of the performance of treatment trains (a combination of at least two technologies installed in series) and how older facilities can be retrofitted to improve treatment. Most studies investigated the removal of heavy metals such as copper, zinc and lead, suspended solids and nutrients (phosphorus and nitrogen) from stormwater. A few studies, but not for all types of facilities, have also investigated organic pollutants such as PAHs, oils and microplastics. Most facility types showed pollutant removals of more than 50 % for the heavy metals, suspended solids and phosphorus, except for swales which performed worse. Lowest removal was seen for nitrogen, rarely performing better than 50 % in any type of facility. In general, the removal was poorer for dissolved phase compared to total concentrations.

Considering design and construction stormwater treatment systems in cold climate, only studies for ponds and wetlands, biofilters and (reactive) filter materials and amendments were reported. For ponds and wetlands as well as biofilters the plants may play an important role and need to be selected to be tolerant to winter conditions and activities, and for optimal treatment performance. E.g., better performance was shown for planted biofilters compared to (non-planted) sand filters for phenolics, PAHs and hydrocarbons and microplastics and plants have also been shown to increase the removal of nitrogen in ponds. Nitrogen removal in biofilters can be improved by submerged saturated zones. Salt in general have negative impact on the retention of heavy metals. A wide range of reactive filter materials and amendments have been investigated in laboratory studies such as biochar, charcoal, iron-treated sand, compost, peat, clay, granulated activated carbon, pine bark, olivine, crushed limestone, zeolite, shell sand and wood chips. In addition to examining the materials themselves, the effect of salt and temperature was also evaluated in several of the studies.

Proper inspection and maintenance of stormwater treatment facilities is necessary to ensure that they continue to fulfil their function over time though this is often neglected. Mentioned maintenance activities in the reviewed studies were regular inspection and removal of trash and large regular removal of accumulated sediments particularly in pre-sedimentation units. Wetlands, ponds, biofilters and grassed swales need care for the vegetation including mowing, pruning, removal, weeding and replanting. Removal or replacement of soil or filter media should be done for grassed swales, biofilters and infiltration trenches. Permeable pavements need regular removal of accumulated particles to maintain their infiltration capacity and prevent them from clogging.

B2. Introduction

This report reviewed the peer-review literature to address the following core research questions (RQs):

- RQ 2.1: What is the treatment performance for different types of facilities and target pollutants?
- RQ 2.2: Are there design criteria and recommendations affecting the treatment performance and are these sensitive to differences in climate?
- RQ 2.3: Which type of maintenance is needed for the different types of facilities?

Use of a systematic approach (see Section B3) led to the identification of a long-list of 662 papers. The titles and abstracts were reviewed for relevance to the core questions, leading to the shortlisting of 279 papers for full review. Common reasons for excluding papers include studies were not about stormwater, stormwater quality or stormwater treatment, or they were not conducted in an urban context. Full review of the 279 papers led to the final selection of 33 papers in relation to addressing treatment performance, 51 papers to address the design criteria and sensitivity to climate and 26 papers to address the maintenance.

Findings are structured in relation to those that directly address each question (e.g. report treatment efficiency by different technologies, how climate and/or design may affect treatment performance and describe/assess maintenance needs for different technologies). The report concludes with an evaluation of the strength of the evidence base for each core question addressed.

B3. Methodology

The review activities undertaken in this report followed the PRISMA approach (Page et al., 2020), an established methodology to provide a transparent, complete and accurate account of how studies are identified and their characteristics reported within systematic reviews. The methodology used in this study involved two stages as follows:

- to facilitate alignment between the review activities (see also Appendices A and C), overarching keywords were identified and applied within a research database to identify a common 'longlist of papers'
- the longlist of papers was further interrogated/refined using keywords relevant to the focus of each review topic (see also Appendices A and C).

B3.1 Common approach

Keywords for the common approach were: stormwater or "storm water" or runoff. These keywords were selected from an inclusivity perspective i.e. to capture as many articles as possible within the targeted field. These pre-defined keywords were entered into SCOPUS, a research publications database which provides access to >90 million research documents including outputs from >29,750 peer-reviewed journals (SCOPUS, 2023). The initial search was undertaken in June 2023 and returned 133,504 hits. This initial longlist was then filtered using the keyword 'urban' leading to the identification of 42,124 papers, with this set of articles providing a common data pool for each review to work with (Table B1).

Table B1. Key words used in the common approach to align activities withinAppendices A, B and C

Search limited to	Key words	Number of hits
Article title, abstract, keywords	Stormwater or "storm water" or runoff	133,504
Search within results	Urban	42,124

B3.2 Methodology to identify articles to address 'Design, treatment performance and maintenance requirements' research

To identify relevant papers for this review, the 42,124 articles identified using the common approach were filtered using the keywords "blue-green", "nature-based", sustainable, "low impact development", "sustainable urban drainage system", "water sensitive urban design", "green infrastructure" and "sponge city" as these terms reflect the focus of this topic. Furthermore, "treatment" was added assuring not to exclude any type of technology. For each RQ, subsets of additional keywords were added to extract papers within their core focus. Further filters were applied to extract documents which were in English, in the form of peer-reviewed research or review article and where at least one co-author had a cold climate country affiliation. Initially the geographic scope was limited to the Nordic countries and Canada, but this was extended following feedback from the project's expert advisory panel to additionally include the Baltic countries. Results of applying filters are reported for each step and RQ in Table B2a–c.

Table B2a. Full and specific search record for RQ2.1

Search limited to	Key words	Number of hits
Article title, abstract, keywords	”blue-green” OR ”nature-based” OR sustainable OR “low impact development” OR “sustainable urban drainage system” OR “water sensitive urban desigh” OR “green infrastructure” OR “sponge city” OR treatment	12,914
Article title, abstract, keywords	Performance OR removal	4,372
Limit to ‘Language’	English	4,235
Limit to ‘Document type’	Articles (3,008), review articles (231)	3,432
Limit to ‘country/territory*	Canada (197), Denmark (65), Finland (20), Iceland (0), Norway (48), Sweden (100), Estonia (0), Latvia (0), Lithuania (0)	**439**

Table B2b. Specific search record for RQ2.2

Search limited to	Key words	Number of hits
Article title, abstract, keywords	Climate OR temperature OR season OR winter	3,397
Limit to ‘Language’	English	3,287
Limit to ‘Document type’	Articles (2,392), review articles	2,606
Limit to ‘country/territory’	Canada (160), Denmark (45), Finland (20), Iceland (0), Norway (54), Sweden (92), Estonia (0), Latvia (0), Lithuania (0)	**372**

Table B2c. Specific search record for RQ2.3

Search limited to	Key words	Number of hits
Article title, abstract, keywords	Longterm OR maintenance OR “asset management”	797
Limit to ‘Language’	English	771
Limit to ‘Document type’	Articles (471), review articles (61)	532
Limit to ‘country/territory’	Canada (34), Denmark (4), Finland (3), Iceland (0), Norway (5), Sweden (17), Estonia (0), Latvia (0), Lithuania (0)	**66**

After the three separate searches, the lists were merged and duplicates and triplicates were removed, resulting in a total of 662 papers.

Figure B1 provides an overview of how the 662 articles reported in Table B2a–c were managed in relation to the three core research questions (pertaining to treatment performance, design and climate, and maintenance) including reasons for exclusion. The initial screening of articles involved their export from SCOPUS and uploading into the open-access systematic review software Rayyan (www.rayyan.ai) which enables research teams to review the same set of articles and – through a blind reviewer mode – provides a mechanism for quality assurance by allowing more than one researcher to independently review a shortlist of articles and compare the consistency of decision-making. In terms of this review, two reviewers independently screened 20 abstracts and reached the same inclusion/exclusion decision on all papers providing confidence that decisions on inclusion/exclusion were being made consistently within the team undertaking this activity.

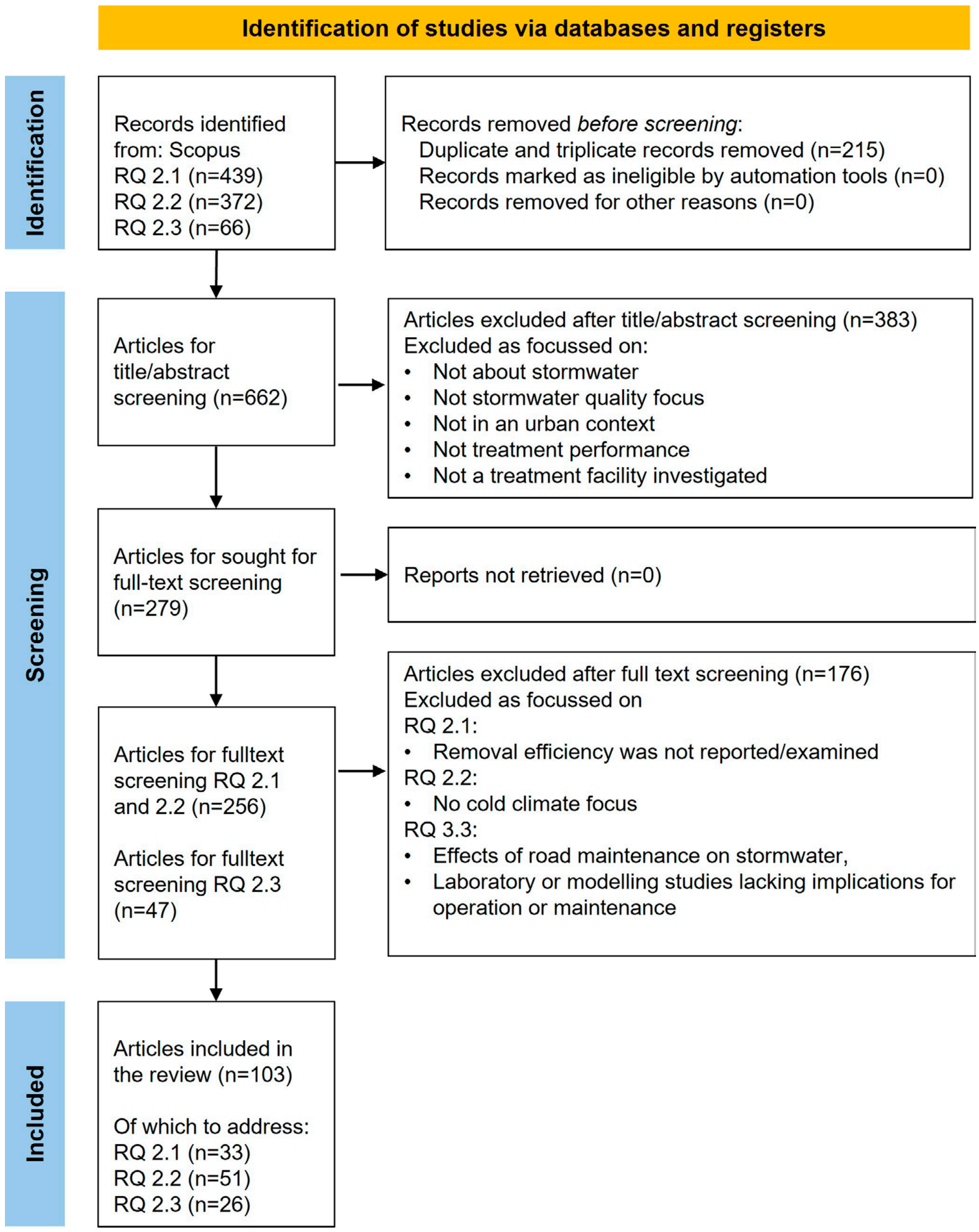

Figure B1. Prisma 2020 flow diagram outlining the screening and filtering steps applied to the 662 papers extracted from Scopus.

B4. Results

From the articles identified as relevant for this review study, information was found on a variety of stormwater treatment technologies and types of facilities such as i) ponds and wetlands, ii) biofilters/bioretention systems, iii) reactive and membrane filters, iv) grass swales, v) infiltration trenches, vi) coagulation/flocculation, vii) permeable pavements, viii) treatment trains, and ix) retrofitting of existing facilities. However, information on all technologies and types of facilities were not available for all three RQs. Table B3 indicates which technologies and facility types were reported on for each RQ.

Table B3. Stormwater treatment technologies and facility types reported on in the literature for each RQ 1–3.

Facility type	RQ 2.1 Treatment performance	RQ 2.2 Design and climate	RQ 2.3 Maintenance
Ponds and wetlands	X	X	X
Biofilters/bioretention systems	X	X	X
Grassed swales	X		X
Permeable pavements	X		X
Reactive filter materials and membrane filters	X	X	
Infiltration trenches			X
Coagulation/flocculation	X		
Treatment trains	X		
Retrofitting of existing facilities	X		

B4.1 RQ 2.1: What is the treatment performance for different types of facilities and target pollutants?

This section summarises the studies that reported on the treatment efficiency of different facility types and technologies in cold climates. Focus was on field studies that investigated the removal efficiency of stormwater treatment facilities. Studies that measured inflow and outflow pollutant concentrations during actual rainfall events and calculated the removal efficiency were considered the primary source of information. These studies provided insight into the performance under complex field conditions including for example, factors related to stormwater runoff quality, freeze-thaw cycles, and dry periods. Modelling and review studies are included if the objective of the study covered cold climate conditions.

B4.1.1 Wet stormwater ponds and constructed stormwater wetlands

Wet stormwater ponds are characterized with permanent pool of water enabling sedimentation whereas constructed stormwater wetlands (CSWs) can be designed for surface or subsurface flow and in comparison to ponds have smaller water depths (Taylor, 1992; Rochfort et al., 1997; Crowe et al., 2007). As summarized by Crowe et al. (2007), previous research agrees that the performance of CSWs is lower in cold weather due to reduced storage volume from thick ice, reduced oxygen transfer between water and atmosphere and reduced biological activity. Often the pond and CSW are combined to improve the performance, with the pond having better functionality during the winter/spring and the wetland during the summer and the plant growing season (Crowe et al., 2007). In particular, subsurface vertical flow wetlands are considered a good polishing step after detention facilities for nutrients and solids for northern climate, however, compared to the surface systems they are more prone to clogging and thus, maintenance is a crucial factor to be considered (Rochfort et al., 1997).

In general, the effectiveness of a pond and CSW system operating in cold climate, largely depends on the targeted pollutant. For example, the removal of Total Suspended Solids (TSS) in pond and wetlands is generally good (61–95 %) (Table B4). When it comes to dissolved pollutants the combination of a pond and CSW is more efficient compared to only pond. For example, 19-year-old pond and CSW in Växjö, in Southern Sweden was investigated and the mean removal efficiency of dissolved metals (Cd, Cu, Pb, Ni and Zn) was 50–66 % (for pond only) and for the pond/wetland system 64–81 % (Al-Rubaei et al., 2017a). Dissolved Ni had much lower mean removals for both a pond (–5 %) and the pond/wetland combination (8 %) compared to the mean removals of total concentrations 67 and 82 % for pond and the pond/wetland system, respectively (Al-Rubaei et al., 2017a). Moreover, during one year monitoring period (May 2013-April 2014, 13 rain events), total metal removal efficiency was stable whereas total nitrogen (TN) removal decreased during the winter season and for the events with short antecedent dry days (ADD) (Al-Rubaei et al., 2016). Comparing the load removal efficiency of pond with previous sampling conducted in the years of 1997 and 2003 when the system was nine and three years in operation respectively (Al-Rubaei et al., 2016), indicated that CSW can sustain good performance with time (Al-Rubaei et al., 2016). In another study, three wetlands in Ontario, Canada were monitored from May to October during years 1998–2001. Wetlands 1 and 2 were designed to operate in series, draining the catchment area of 5.84 ha. Their performance was compared to the performance of Wetland 3 servicing the catchment of 23 ha. During one event in September, the authors reported extremely high negative removals of studied contaminants (from –2221 % to –71 %) which was attributed to low inflow pollutant concentrations entering facility due to the frequent heavy rains (Farrel and Scheckenberger, 2003). Excluding this event, the median removal efficiency for the sampling period were calculated (Farrel and Scheckenberger, 2003). Removal of TSS, total P and total metals were comparable between the two facilities (Table B4) and the authors concluded that the primary removal occurs by adsorption to sediment which settle in facility forebays thus, increased detention volume and residence time that two wetlands in series provide, benefited more removals of Total Kjeldal Nitrogen, faecal streptococci and E. coli (Farrel and Scheckenberger, 2003).

Table B4. Mean concentration removal efficiency [%] of different pond and constructed stormwater wetland (CSW) systems. Removals are given for main pollutant groups TSS, metals (Zn, Cu, Pb) and nutrients (TP and TN). Removal of dissolved metals is indicated with bold font.

Ref	Monitoring period (# rain events)	Facility type	TSS	Zn	Cu	Pb	TP	TN
[1]	2013–2014 (13)	Pond	95	82	86	93	86	50
		Pond+CSW	96	90	91	96	89	59
	Summer 2003 (5)	Pond	61	53	56	55	nd	nd
	1997 (3)	Pond	86	73	61	84	58	25
		Pond+CSW	92	95	76	94	84	52
[2]	2013–2014 (13)	Pond	90 ± 7	84 ± 18 **64 ± 24**	88 ± 11 **58 ± 32**	89 ± 14 **50 ± 43**	76 ± 13	39 ± 25
		Pond+CSW	91 ± 7	92 ± 8 **81 ± 12**	91 ± 9 **69 ± 20**	83 ± 32 **64 ± 36**	80 ± 13	45 ± 27
[3]	May-October 1998–2001 (12)	Wetland 2[1]	−1520–97 (71)	−335–99 (57)	−458–96 (60)	−272–95 (66)	−249–88 (55)	nd
		Wetland 3	−32–99 (76)	−178–98 (65)	−41–97 (61)	−28–98 (66)	−17–95 (49)	nd

[1] Range for sampled events and median in brackets, calculated excluding the event with high negative removals. Water quality grab samples were taken at the outlet of Wetland 2.

[1] Al-Rubaei et al., (2016), [2] Al-Rubaei et al. (2017a), [3] Farrel and Scheckenberger (2003)

B4.1.2 Infiltration based facilities – Biofilters and bioretention

As for the ponds and wetlands, the effectiveness of the bioretention system operating in cold climate conditions largely depends on the targeted pollutant type (Table B5). In general removal of TSS in bioretention cells is good. For instance, positive concentration reductions were calculated for TSS 63 % (Spraakman et al., 2020b), 74.5 % (Géhéniau et al., 2015), 84 % (Brodeur-Doucet et al., 2021), > 78 % (Lange et al., 2022), 95 % for chalk-amended vegetated biofilter (BFC) and vegetated biofilter (BF), and 35–90 % (median = 70 ± 8 %) for non-vegetated sand filter (SF) (Beryani et al., 2023).

Table B5. Mean concentration removal efficiency [%] of bioretention cells. Removals are given for main pollutant groups TSS, total metals (Zn, Cu, Pb) and nutrients (TP and TN). Removal of dissolved (dis) and truly dissolved (TD) metals are specified, as well as mass reductions (MR) where applicable.

Reference and location	Monitoring period	TSS	Zn	Cu	Pb	TP	TN
[1], Canada	Spring/autumn (ARE=2–4)	63–98	71–73	−27–8	−68–59	56–75	−33–36
	Winter (ARE=3–4)	91–98	80–89	33–78	63–90	43–100	−57–71
[2], Canada	ARE=24	84	nd	nd	nd	6	−5
[3], Canada	May–October 2017–2018 (ARE=13)	63	86–96$_{MR}$	92$_{MR}$	24	nd	nd
[4], Canada	Jan–May 2013 and Sept 2013–Feb 2014	74.5	48.3	−14.1	54.3	−65.3	nd
[5], Sundsvall, Sweden	Sept–Dec 2020 (ARE=6)	> 78	94 77 (dis) 76 (TD)	81 22 (dis) 5.7 (TD)	> 76 6.6 (dis) nd (TD)	nd	nd
[6], Sundsvall, Sweden	ARE=11	95±3$_{BF}$ 95±3$_{BFC}$ 35–90$_{SF}$	nd				

[1] Pineau et al. (2021), [2] Brodeur-Doucet et al. (2021), [3] Spraakman et al. (2020), [4] Géhéniau et al. (2015), [5] Lange et al. (2022), [6] Beryani et al. (2023)

When it comes to metal reduction by bioretention, one study mentioned statistically non-significant removal due to low metal inflow concentrations (Spraakman et al., 2020b) and another study noticed difference dependent on metal species, i.e., metals Pb and Zn were effectively reduced but not Cu and Ni (Géhéniau et al., 2015). Only one field study from Sundsvall, Sweden investigated removal of total, dissolved and truly dissolved metals (Cd, Pb, Cu and Zn) in the field setup (Lange et al., 2022). The mean removal of pollutants after passage through the gross pollutant trap (GPT) followed by the bioretention system was > 76 % (Pb), 81 % (Cu), 94 % (Zn), 35 % (DOC), > 78 % (TSS), and −5.3 % (Cl) (Lange et al., 2022). The removal of dissolved metals Cd, Pb, Cu and Zn were 30 %, 6.6 %, 22 %, and 77 % respectively and truly dissolved fractions were 19 %, n.d., 5.7 %, and 76 % for Cd, Pb, Cu and Zn respectively. These results show that dissolved and truly dissolved Zn concentrations were reduced to a higher extent compared to other metals (Cu, Pb and Cd), possibly due to lower affinity to organic matter (Lange et al., 2022). One of the sampled events had increased Cl concentration due to the road salt application which increased particularly the (truly) dissolved metal concentration from the bioretention cell (Lange et al., 2022).

Regarding removal of other pollutants by bioretention, positive concentration reductions were calculated for nitrite (80 %), and ammonia/ammonium (56 %) and total nitrogen (24 %) (Spraakman et al., 2020b). Concentrations of chloride, phosphate, total phosphorus and nitrate plus nitrite were not reduced by the bioretention cell (Spraakman et al., 2020b). Total dissolved solids (TDS) had a negative removal and pH, conductivity, alkalinity and hardness also significantly increased at the outlet (Spraakman et al., 2020b). E.coli and faecal coliforms were reduced, leaching of TKN and phosphorus was observed and chloride concentrations in the effluent increased during the cold season (Géhéniau et al., 2015). Particularly the bioretention filter material consisted of a mix of 93 % sand, 6 % silt, and 1 % clay, with a gravel layer with a perforated pipe below and pervious geotextile membrane at the bottom (Géhéniau et al., 2015).

Among the limited number of field studies investigating the removal efficiency of organic contaminants in infiltration-based urban drainage facilities, the majority of substances examined were found to exhibit satisfactory removal efficiency. When comparing the removal efficiencies among substances, it is important to consider the potential for underestimation of performance for removing substances that were presented at concentrations close to the quantification limits at the inlets of the facilities. This applies to the investigation of e.g. PAHs and hydrocarbons in Beryani et al. (2023). However, instances of poor treatment performance have also been reported. For example, non-significant changes in concentrations between inlet and outlet for selected PAHs (ACY, ANT, BbF, Benzo[*e*]pyrene and CHR) were observed; significantly higher concentrations of 1-methylnaphthalene and BaP were identified in the outlet compared to the inlet; and the insufficient removal of dissolved contaminants, in general, was noted (Spraakman et al., 2020).

For microplastics specifically, a net-release of rubber, bitumen and other microplastic particles (microplastic fibres and fragments) was observed in two out of the nine monitoring events, where higher concentrations were identified in the outlet than in the inflow (Lange et al., 2021). These two events were characterised by medium-to-low rainfall depths (7.5 and 9 mm) and peak intensities (0.9 and 1.6 mm/h), in comparison to events with rainfall depths and peak intensities up to 23.6 mm and 8.8 mm/h respectively. Overall, the bioretention system described in Lange et al. (2021) demonstrated greater efficiency in removing rubber particles compared to bitumen or other microplastic particles (Figure B2).

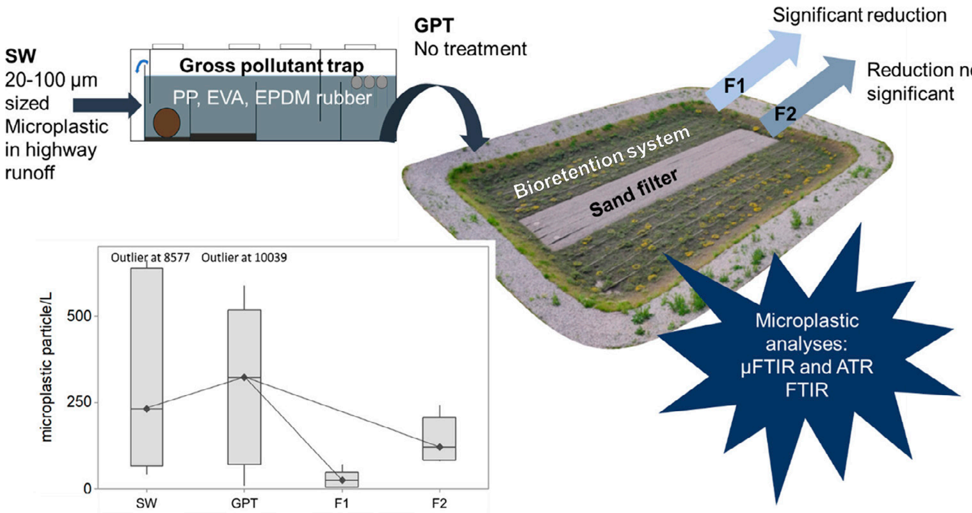

Figure B2. Treatment of microplastics (20–100 µm size range) in a gross pollutant trap (GPT) – bioretention system compared to a GPT-sand filter system. Graphical abstract from Lange et al. (2022). Published under Creative Commons Attribution (CC BY 4.0) license.

B4.1.3 Infiltration based facilities – Grassed swales

Swales operating in cold climate serve an additional important function, i.e., they are used for storing of snow during winter months and subsequent control of the snowmelt (Gavrić et al., 2021; Ekka et al., 2021). In a review paper on stormwater management in cold climate, different stormwater treatment facilities were classified based on their "winter performance" and grass swales/ditches were given the positive score (Bäckström and Viklander, 2000). In a review and design recommendation paper by Ekka et al. (2021), designs of swales for cold climate were explored. Based on the previous research, a number of operational challenges were noted: higher flow velocities due to the lower flow resistance from the dormant vegetation, reduced biological and chemical activity during melting period and lower soil infiltration (due to frozen soils) were suggested as factors reducing swale performance during snowmelt (Ekka et al., 2021). The literature review on grass swales and grass filter strips (Figure B3) (Gavrić et al., 2019) revealed that the majority of studies that measured stormwater quality in swales were conducted in the temperate climate, where facilities were not exposed to seasonal snow, with exception of the pioneering work by Bäckström et al. (2006).

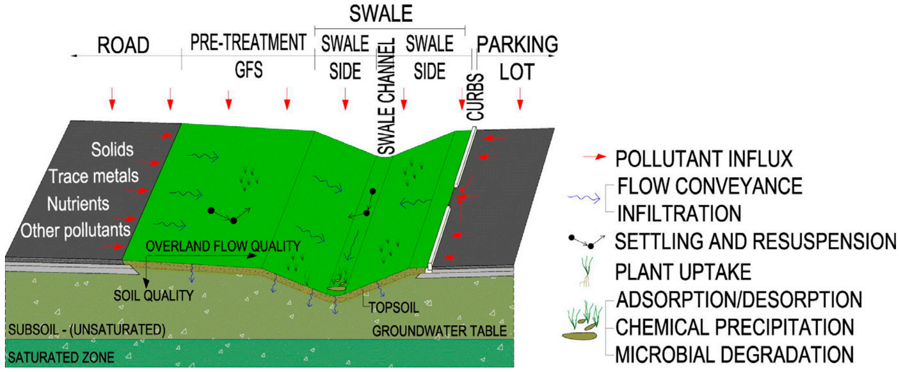

Figure B3. Processes taking place in different parts of a swale. Graphical abstract from Gavric et al. (2019). Published under Creative Commons Attribution (CC BY 4.0) license.

A grass swale in the city of Luleå, Sweden was investigated during May-June, after the surface sediment were brushed from swales during regular maintenance taking place in the beginning of May (Bäckström et al., 2006). The swale was investigated during 13 rainfall events from May to June 2000, of which four events were used for calculating removal efficiency of TSS and total and dissolved metals (Cu, Pb and Zn) (Bäckström et al., 2006). The authors observed that the inflow pollutant concentration is an important factor affecting removal rates (Bäckström et al., 2006). The results indicated that swales are not effective in TSS removal when inflow concentration is below 40 mg/L (Bäckström et al., 2006). In general, the investigated swale was not able to remove TSS and metals to a high extent, especially dissolved Cu was problematic since the leaching was observed (Bäckström et al., 2006). In addition, the swale had stable pH during the events with the minimum value of 6.0 (Bäckström et al., 2006).

Table B6. Removal efficiency of TSS and different metals. Removal of dissolved metals are marked with bold font.

Reference	TSS	Zn	Cu	Pb
(Bäckström et al., 2006)	−129–47	−35–40 **8–32**	−288–(−12) **−375–(−104)**	−186–12 **−51–13**

B4.1.4 Infiltration based facilities – Permeable pavements (PP)

Studies indicate good performance in TSS removal for both winter and non-winter conditions (Drake et al., 2014; Huang et al., 2016a). Different PP cells in Vaughan, Ontario, Canada were investigated over 24 months (from June 2010-June 2012) and samples representing spring, summer and autumn were investigated (Figure B4) (Drake et al., 2014). PP were able to reduce TSS and provide buffering of pH (Drake et al., 2014). Controlled field experiments in the city of Calgary (Alberta, Canada) were conducted on three permeable pavements (PP) types (Huang et al., 2016a). For each test, stormwater was taken from a stormwater retention pond and sediments collected from the roads were added to get the TSS inflow concentration of ~500 mg/L (Huang et al., 2016a). Tests that represented winter conditions were tests where the pavement temperature was less than 5 °C (Huang et al., 2016a). The range of TSS removal rates was 86.9–94.6 % indicating only small differences between winter and non-winter conditions (Huang et al., 2016a). It should be noted that TSS accumulation in the filter may cause clogging which calls for regular maintenance, further discuss in Section B4.3.5.

Table B7. Concentration removal efficiency of different pollutants. Mass removals are indicated.

Reference and location	TSS	Zn	Cu	Pb	TP	TN
Drake et al. (2014), Canada	> 80 $> 80_{MR}$	62–82 $89–93_{MR}$	50–62 $65–74_{MR}$	nd	9–82 $> 75_{MR}$	34–45 $47–59_{MR}$

No effect of pavement temperature was observed for removal of TP (74.6–84.4 %) and Cu, Pb and Zn (66.2–86.1 %) whereas TN removal varied more (2.9–40 %) and was affected more by temperature of the pavement (Huang et al., 2016a). The pollutant removal rates of the same three pavements, PA, PC, and PICP was investigated in the laboratory under room temperature between 19 and 22 °C (Huang et al., 2016b). The removals of TSS were 87–95 %, TP 75–89 %, and TN 3–10 % (Huang et al., 2016b).

In one field test of the performance of permeable pavements in Drake et al. (2014), the efficient removal of oil, grease and PAHs was identified, with these substances frequently detected in the influent but seldom in the effluent.

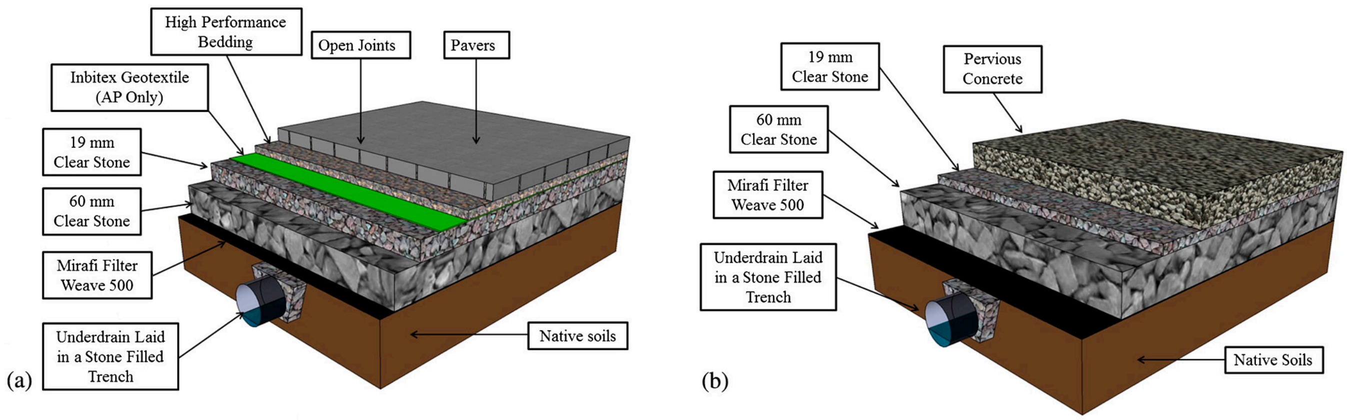

Figure B4. Cross sections of permeable interlocking concrete pavers (a) and pervious concrete (b). Reprinted from Journal of Environmental Management 139, Drake, J., Bradford, A., Van Seters, T., Stormwater quality of spring-summer-fall effluent from three partial-infiltration permeable pavement systems and conventional asphalt pavement. 69–79, Copyright 2014 with permission from Elsevier.

B4.1.5 Reactive and membrane filters

Different filter materials such as biochar, lightweight aggregates, sand, woodchip, zeolite etc. can be used as amendments in different stormwater treatment facilities (e.g., green roofs, biofilters, swales, infiltration trenches) (Westholm, 2023). Also materials can be part of treatment system as two examples from Sweden where **zeolite** was used for treating copper roof runoff (Milovanović et al., 2022), and **sand and a sand-biochar** filters were used in the treatment system for treating road runoff (Koivusalo et al., 2023). For the operation and installation of filters in the field there are several insights from previous research. Treated volume of water (rather than the amount of time the filter has been in operation) is influential factor for the filter aging (Milovanović et al., 2022). Moreover, the hydraulic loading rate is relevant factor affecting the filter performance since high hydraulic loading rate (e.g., pumping the water to the filtering system) reduces contact time between the stormwater and the filter, possibly limiting the treatment efficiency (Milovanović et al., 2022). Several things should be considered when installing filters in the field: i) installation at the optimal location so that the filter area is e.g., 2–5 % of the catchment contributing area, ii) preventing water bypass flows which means expertise in installation of filters, iii) selection of suitable materials as well as iv) design that enables measurements and surveys of influent and effluent waters in order that the treatment performance (for nutrients in this case) and its decline over time can be studied (Koivusalo et al., 2023). Installation can include placement of filters underground as a stack of horizontal layers of high and low porosity, through which flow is conveyed by gravity. Such design was implemented in a pilot plant in Copenhagen "dual porosity filtration (DPF)" (Jensen et al., 2011).

As shown in Table B8, filters have high removal of TSS (> 90 %) (Jensen et al., 2011) and in the study by Milovanovic et al. (2022) TSS concentrations were mostly below detection limit in the effluent, when the influent TSS concentrations were 3.4–19.3 mg/L. Generally, the removal efficiency of total metals ranged from 50–98 % (Table B8) however, the research points out that the despite high removals, effluent metal concentrations of the filter could still be concerningly high e.g., 350–600 µg Cu/L (Milovanovic et al., 2022).

Table B8. Removal efficiency [%] of Zn and Cu and where applicable other pollutant. Dissolved metals (diss) are indicated.

Ref.	Monitoring period (ARE)	Filter type	Zn [%]	Cu [%]	Other pollutants [%]
Jensen et al. (2011)	January–July 2007 (ARE=25)	Dual porosity filtration (DPF)	70.0–87.3	50.6–61.6	88.1–97.9 (Pb) 73.3–78.0 (TP) 91.5–98.9 (TSS)
Milovanovic et al. (2022)	Dec 2018–March 2020 (ARE=7)	Zeolite	51–94 48–94 (diss)	52–82 49–85 (diss)	20–68 (Turbidity) 16–68 (TOC)
Hallberg et al. (2022)	May 2021–Jan 2022 (ARE=24)	Sand	93 87 (diss)	67 19 (diss)	nd

A review paper by Ebrahimi Gardeshi et al. (2023) looked at the impact of de-icing salts on the environment and possible treatment techniques including stormwater treatment facilities, thermal distillation, membrane methods (e.g., reverse osmosis, nanofiltration and electrodialysis). The study concluded that a hybrid system where electrodialysis is combined with conventional stormwater treatment facilities (e.g., constructed wetlands, swales, sedimentation basins etc.) is a promising approach for removing and recovering salts from runoff (Ebrahimi Gardeshi et al., 2023).

B4.1.6 Coagulation/flocculation

In a laboratory study, coagulation/flocculation was tested for its potential to treat stormwater and snowmelt and remove total and dissolved metals (Cd, Cr, Cu, Ni, Pb, and Zn), organic content (TOC and DOC), aliphatics and PAHs (Nyström et al., 2020). In the study five coagulants were tested (alum, PAX-215, PAX-XL360, PIX-11 and chitosan), of which all resulted in high (>90 %) removal rates of total metals, PAH-H, and TOC concentrations (Nyström et al., 2020). However, the reduction of the dissolved fraction of metals, organic carbon and PAH-L were lower, e.g. Cu was on average reduced by 40 % and for DOC no additional reduction compared to the control was observed. Dissolved Zn was reduced only when using chitosan (~51 %), whereas iron chloride increased the dissolved concentration of zinc due to decrease in pH (Nyström et al., 2020). All the tested coagulants exhibited a significant removal of aliphatics (C16-C35 and C35-C40), the longer carbon chains the better the reduction. In a field study, two intermediate bulk containers for flocculation and a disc filter (with a 10 µm mesh) were investigated when an organic flocculant was added and without flocculant addition (Nielsen et al., 2015). Before one of the sampled events de-icing salts were used in the catchment which increased electrical conductivity and probably TSS (Nielsen et al., 2015). While the addition of flocculant increased removal efficiency of for example, turbidity and TSS, there was no statistical significance in particle size distribution between the inlet and the outlet (Nielsen et al., 2015).

B4.1.7 Treatment trains

Treatment trains consist of several systems installed in series to improve the quality of the final effluent (Anderson et al., 1997). The combination of pre-sedimentation pond, pond and CSW (Al-Rubaei et al., 2017a), as well as that of a gross pollutant trap and biofilter (Lange et al., 2022; Beryani et al., 2023) can also be considered to be treatment trains. However, these two combinations have been described in the Sections B4.1.1. and B4.1.2. and the following section focuses on pond/filter combinations.

In order to further improve water quality from ponds, a submerged aerobic biological filter (similar to biofilter) was investigated in controlled field tests (Anderson et al., 1997; Mothersill et al., 2000). Biofilter was able to remove TSS, NH4, particulate metals, organic carbon, and phosphorus however, high TSS removal efficiency (97 %) resulted in clogging and increased accumulation in top 15–20 cm of the biofilter (Mothersill et al., 2000). A facility in Odense, Denmark with a grit chamber, a wet retention pond, sand filter and fixed media sorption filter was investigated and shown to be effective in removal of nitrogen and phosphorus (Vollertsen et al., 2009). The sorption filters were important for removal of Total-P and orthophosphate (Vollertsen et al., 2009). Three full-scale wet detention ponds amended with sand filters (planted with Phragmites australis) at the outlet, in the cities of Aarhus, Silkeborg and Odense, in Denmark were investigated (Isteniç et al., 2012). In each system additional technology was tested: fixed-media sorption filter, iron-enriched bottom sediment, and dosing aluminium salts to form flocs that settle. All three wet detention ponds effective sedimentation occurred removing TSS, Tot-N, Tot-P and orthophosphate (PO4-P). Sand filters improved the removal of Zn in all three ponds and Cu in one system. The “fixed-media sorption filter” installed at one of the ponds, further improved PO4-P, Zn and Cu removal. Outflow from a stormwater pond in Padborg, Denmark is distributed to a crushed concrete filter and sand filter which performance was compared during 1 year (2013–2014) (Sønderup et al., 2015). Retention of the pond followed by filters was good for SS, organic matter (OM), particulate phosphorus (PP) and metals (Sønderup et al., 2015). A study from Canada investigated the effect of a treatment train where the runoff from a drainage ditch conveys the water to the detention basin after which the water is directed to the underground filtering bed (80 % limestone and 20 % dolomite) and CSW (Guesdon et al., 2016). The system was investigated from March 2012 to November 2013 to study removal of de-icing salts (NaCl) and Cd and Pb (Guesdon et al., 2016). The removal efficiency of Pb was 35.8 % in the summer compared to the spring (6.8 %), and for Cd it was 10.3 % (spring) compared to –6.7 % (summer) (Guesdon et al., 2016). Moreover, five bioretention cells forming a treatment train with a wet pond in Canada, were found efficient in removing TSS in winter conditions, where winter events were characterized by snow on the ground and mean air temperature below 10 °C (Pineau et al., 2021). The range of TSS concentration removal for four winter sampling days was 91–98 %, compared to spring/autumn sampling days 63–98 % (Pineau et al., 2021). TSS removal by the whole treatment train was 85–98 (three winter days) and 81 % for one spring event (Figure B5) (Pineau et al., 2021). Finally, stormwater and de-icer from the airport in Luleå, Sweden was treated by oil trap and infiltration pond system prior its discharged to the receiving water (Jia et al., 2018). The amount of nitrogen released from the pond relative to the initial nitrogen amount used for de-icing (converted from urea used in the de-icers) was reported to be 0.62–40.8 % (Jia et al., 2018).

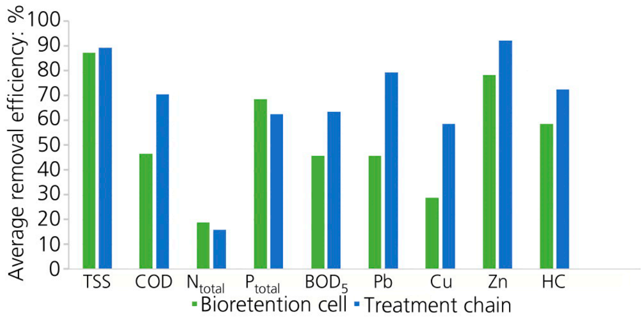

Figure B5. Removal efficiencies of a bioretention cell (green bars) and the full treatment train including bioretention cells and a pond (blue bars). Reproduced from Environ. Eng. Sci. 16, Pineau, B., Brodeur-Doucet, C., Corrivault-Gascon, J., Arjoon, D., Lessard, P., Pelletier, G., Duchesne, S., Performance of green infrastructure for storm water treatment in cold climate (Canada), Copyright 2021, with permission from the licensor through PLSclear 2021.

Table B9. Mean concentration removal efficiency [%] of different stormwater treatment train including end-of-pipe systems. Removals are given for main pollutant groups TSS, metals (Zn, Cu, Pb) and nutrients (TP and TN).

Reference	Treatment train	Removal efficiency of different pollutants [%]
Guesdon et al. (2016)	Detention basin + filter + CSW	Pb (6.8$_{spring}$), Pb (35.8$_{summer}$)
Sønderup et al. (2015)	Pond	TSS (60), Zn (42–92), TP (39, 65$_{summer}$ and 18$_{winter}$) TN (39)
	Pond + sand filter	TSS (77), TP (78)
	Pond + crushed concrete filter	TSS (78), TP (80)
Pineau et al. (2021)	5 Bioretention cells + Pond	TSS (81$_{spring/autumn,}$ 85–98$_{winter}$), Zn (90–95$_{winter}$) Cu (27–78$_{winter}$) Pb (63–90$_{winter}$), TP (29–100$_{winter}$) TN (–71–71$_{winter}$)

Three stormwater ponds, each featuring different retrofitted designs – such as the installation of additional sand filters with plants or adsorption filters at the outflow, and the dosing of aluminium salt at the inflow – were examined in Istenič et al. (2011) for their efficacy in removing PAHs. These facilities were concluded to be efficient in removing total PAHs, with concentrations generally above the detection limit in the inflow but below the detection limit in the effluent, therefore no removal efficiency was specified.

B4.1.8 Retrofitting existing facilities

Improving stormwater quality of the pond incoming water (control at the catchment), as well as pre-treatment of pond influent using e.g., grass filter strips and sedimentation basins is recommended (Marsalek et al., 1992). Retrofits at the pond could include: i) increasing the pond volume, ii) modifying pond layout, depth and flow paths, iii) modifying the outlet (e.g., having a multi-level outlet structure (Shammaa et al., 2002)), iv) infiltration inlets after the sediment trap, and v) enhancing biosorption by rooted plants in shallow waters (Marsalek et al., 1992). Replacing the "dual inlet/outlet structures" with separate inlet and outlet was also mentioned (Shammaa et al., 2002). Another retrofitting option is lengthening the flow by baffles (Marsalek et al., 1992), which can benefit the flow regime (avoid problems with short-circuiting and dead zones) and retention time with potential to increase pollutant removal through sedimentation (Matthews et al., 1997). For instance, in a modelling study, removal of SS in ponds was shown to be improved when installing baffles (Figure B6) (German et al., 2005). Moreover, another retrofitting option is to further treat pond effluent by e.g., wetlands, sand filters and disinfections (Figure B7) (Marsalek et al., 1992).

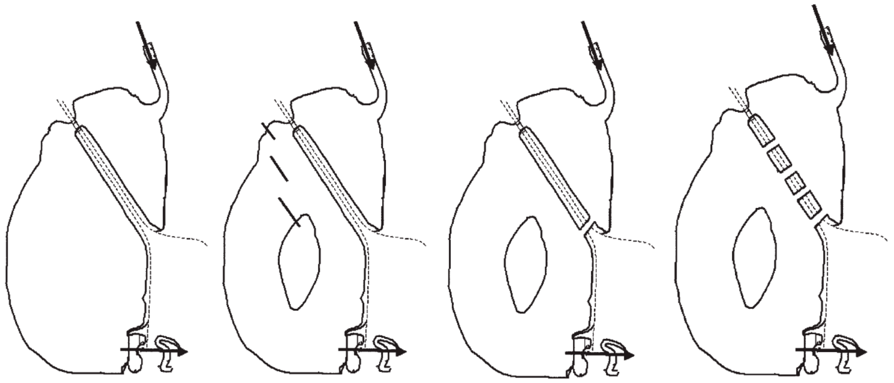

Figure B6. Four suggestions how to retrofit and improve removal efficiency of a pond modelled by German et al. (2005). From left: island removed, baffles installed, one culvert under the causeway and four culverts under the causeway. Reprinted from Water Science & Technology, volume 52, Issue number 5, pages 105-112, with permission from the copyright holders, IWA Publishing.

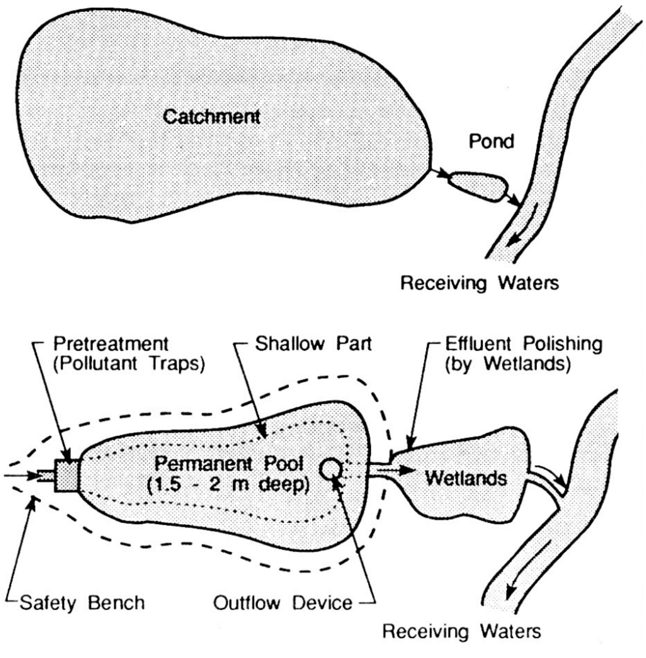

Figure B7. Schematic representation of a pond in relation to its catchment (top) and how it can be retrofitted to pond-wetland system for improved treatment (bottom). From Marsalek et al. (1992). Reprinted from Water Quality Research Journal, volume 27, Issue number 2, pages 403–422, with permission from the copyright holders, IWA Publishing.

In an experimental setup, stormwater standard sumps were retrofitted to improve sediment removal efficiency (Ma and Zhu, 2014). Several simple structures were tested, and the best performance was obtained with a structure containing “oblique baffles, two horizontal plates and a porous screen” (Ma and Zhu, 2014). The effect was avoiding “short circuit,” and increasing the path of sediments, however the authors mentioned the need to test the applicability in the field.

B4.2 RQ 2.2: Are there design criteria and recommendations affecting the treatment performance and are these sensitive to differences in climate?

In addition to the field studies carried out in cold climates, which provided data on treatment efficiency and were summarised in Section B4.1, a number of controlled field, laboratory, and modelling studies investigating various climate-related factors and activities were identified. In general, such conditions, mentioned in previous research included:

- Usage of grit and de-icing salts
- Asphalt wear due to studded tires, cold starts of car engines, exhaust fumes
- Frost in the biofilter media and soil, freeze-thaw cycles
- Ice cover on the ponds
- Snow on the ground in the catchment
- Significant snowmelt volumes in spring
- Low air temperature and low evapotranspiration
- Non-growing or short growing season as well as dormant plants

B4.2.1 Ponds & wetlands

ROLE OF PLANTS

Several studies investigated pollutant uptake of wetland plants. For instance, in the controlled experiment by Scholz and Hedmark (2010), the role of macrophytes (*Phragmites australis* (Cav.) Trin. ex Steud.), in nitrogen removal (ammonia-nitrogen, nitrate-nitrogen, nitrite-nitrogen and ortho-phosphate-phosphorus) was assessed by introducing sieved gully pot (GP) water artificially polluted with nitrogen in the form of ammonium chloride into the wetland filters. Removal of total inorganic nitrogen (TIN) was higher for planted filters (93.8–97.8 %) compared to unplanted filters (15.1–50.5 %). The study also considered the seasonal affect by having inflow water temperature range from 5.8 to 20.7 °C (i.e., sampling GP liquor from May to December), for which the removal rate of TIN was more stable in planted filters compared to unplanted filters. *P. australis* was harvested at the end of the growing season and the study concluded that the plant uptake was the main removal mechanism for nitrogen followed by nitrification and denitrification (Scholz and Hedmark, 2010). The effects of salt (0, 100, and 1000 mg NaCL/L) and temperature (5, 15 and 25 °C) on the metal (Cd, Cu, Pb, and Zn) and Cl uptake by wetland plants was investigated in the greenhouse chambers (Schück and Greger, 2023). Low temperatures decreased the removal of all metals and increased salinity decreased the removal of Cd and Pb but had no effect on the removal of Zn or Cu. From three plants investigated, *C. pseudocyperus* was effective in removing Cu and Pb and *Phalaris arundinacea* was effective in removing Cd, Zn, and Cl^-. The findings indicate that plant species selection is important to maximize the removal of targeted pollutant (Schück and Greger, 2023). Thirty-four wetland plants native to Sweden were screened from which *C. riparia* and *P. arundinacea* were considered good candidates due to high Cl^- tolerance, large biomass, and high accumulation of Cl^- (Schück and Greger, 2022). The plants *Atriplex patula* and *Typha angustifolia* were selected for designing CSW and the laboratory study investigated the effect of nutrient supply and hydraulic residence time (during plant biomass development) on salt adsorption of plants (Morteau et al., 2015). The study found that plants exposed to one week residence time and treated with high nutrient supply using Hoagland solution diluted four times (compared to the low supply using synthetic road stormwater) had the highest aboveground biomass and salt removal (Morteau et al., 2015). Another technology that uses floating rafts with plants, called floating treatment wetlands (FTWs) was investigated in the study from Stockholm, Sweden (Boynukisa et al., 2023). Planted floating rafts were placed in two stormwater ponds operating in cold climate for 12 weeks in order to investigate the metal uptake (Boynukisa et al., 2023). Three native wetland species were investigated *Carex riparia*, *C. pseudocyperus*, and *Phalaris arundinacea*. Higher metal uptake was observed for the plants with higher biomass indicating that the good growing conditions are important and *Phalaris arundinacea* was consider a good candidate for floating treatment wetlands (Boynukisa et al., 2023).

Choosing adequate plant species that will be able to grow in cold climate conditions implies that the proper maintenance is important in in order to prevent problems with overgrown vegetation (Al-Rubaei et al., 2017b). In the city of Winnipeg (Manitoba, Canada) conventional stormwater retention basins (CSRBs) were experiencing regular overgrowth of aquatic macrophytes and algae blooms (Badiou et al., 2019). Aquatic vegetation was removed mechanically by paddlewheel harvester and/or chemically by addition of the aquatic herbicide. Concentrations of water quality parameters such as TP, SRP, ammonia and chlorophyll-a (indicator of algal blooms) increased in the basins subject to vegetation removal compared to other basins and constructed wetlands. For example, Chl-a of 405 µg/L, 128 µg/L and 33 µg/L was measured for the CSRBs that had vegetation removed, the ones without vegetation removed and CSWs respectively. The study concluded that CSW were a good alternative for stormwater management (Badiou et al., 2019).

EFFECT ON WATER TEMPERATURE

Though beyond the scope of this study, it is worth mentioning that end-of-pipe systems such as ponds and wetlands has been shown to affect the water temperature of the permanent water pool during warmer months. This type of thermal pollution may be of concern for example for systems that discharge to receiving waters with cold water fisheries (Papa et al., 1999). Underground stormwater detention chambers were investigated as an alternative to sun-exposed detention facilities (Drake et al., 2016). This was shown to reduce the temperature of the effluent water with on average 4 °C.

B4.2.2 Infiltration based facilities – biofilters

Biofilters have, in comparison to other stormwater treatment technologies, been extensively investigated in controlled field and laboratory studies. Typically studied factors are the effect of temperature and salt on the removal of metals, submerged zones and temperature for reduction of nitrogen, the role of plants and filter material composition.

CONTROLLED FIELD AND LAB STUDIES OF THE IMPACT OF COLD CLIMATE CONDITIONS

In controlled experiments, a bioretention box design with a layer of gravel (10 cm), sand (55 cm) and mulch (10 cm) with the additional 15 cm space for standing water was investigated in late winter/early spring and summer (Muthanna et al., 2007). Late winter/early spring season was characterized by frozen soil and dormant vegetation, and summer season was characterized with fully established vegetation cover (Muthanna et al., 2007). Comparing the performance in April (4 °C) and August (13.2 °C), Zn mass removal was the same (90 %), Pb mass removal increased from 83 to 89 % in the summer, whereas Cu mass removal was 60 and 75 % for April and August respectively (Muthanna et al., 2007). 15 biofilter columns were investigated in controlled experiments to study the removal of total and dissolved metals across different air temperatures (2–20 °C) (Blecken et al., 2011). The study observed high metal removals indicating that the low temperature is not limiting the mechanical filtration of particulate bound metals, plant uptake and adsorption capacity of the top layer (Blecken et al., 2011). Only in the case of dissolved Cu the temperature was a significant factor i.e., higher temperature had more dissolved Cu in the biofilter effluent (Blecken et al., 2011). Higher temperature probably led to higher biological activity and decomposition of organic matter (e.g., plant roots, bacteria) leading to leaching of Cu complexed with dissolved organic matter (DOM) (Blecken et al., 2011). In the controlled bioretention column experiments removals of total and dissolved Cd, Cu and Zn were investigated under warm conditions (19.4 °C) and cold conditions (3.6 °C) as well as under the salt-pulse experiment (Paus et al., 2014b). Low temperature did not impede dissolved metal sorption except for Cu, for which removal was significantly reduced under low temperature (Paus et al., 2014b). Metal leaching was observed when NaCl was introduced and salt had higher effect in the warm conditions (Paus et al., 2014b). The effect of salt on biofilters subject to low and high temperature was examined in controlled experiments (Søberg et al., 2014). Out of 12 biofilter mesocosms, with a sand-based filter material planted with native vegetation adapted to Nordic climate, six columns were placed in a cold room (4.6 ± 0.6 °C) and the rest in an indoor laboratory (17.1 ± 1.5 °C) (Søberg et al., 2014). The effect of salt was apparent in removal of dissolved metals particularly Cu and Pb; however, the fraction of dissolved Pb was very small, thus, overall Pb removal was not much affected (Søberg et al., 2014). When it comes to removal of TSS neither salt nor temperature had an effect on the removals (Søberg et al., 2014). 24 pilot-scale bioretention columns were tested during 18 weeks to investigate the effect of salt, low temperature and inclusion of water-saturated submerged zone (SZC) in the metal removal (Søberg et al., 2017). Leaching of Cu and Pb occurred for salt experiments, top 10 cm of biofilter accumulated the most metals and inclusion of SZC had beneficial effect on TSS and metal removal (Søberg et al., 2017). In a bioretention column laboratory study, 12 bioretention columns with different plant species adapted to cooler climates (i.e., Northen

Europe climates) were investigated to study the effect of the plant species and salt on removal of total, dissolved and truly dissolved metals (Lange et al., 2020). Total metal (Cd, Pb, Cu, Zn) removal was >95 %. Salt had negative impact on the removal of Cd and Cu in all fractions and Zn only in unfiltered sampled (Lange et al., 2020). Biofilter outflow contained more colloidal and truly dissolved metals (Lange et al., 2020). Finally, vegetation had no significant effect on the metal removal (except Cd in salt experiment) and fractionation (Lange et al., 2020).

Similarly, effects of temperature (typical range 1.5–25 °C) and salt were investigated for the removal of nutrients. The effect of temperature, water saturated submerged zone (SZC), and different length of ADD on the removal of nutrients were investigated (Søberg et al., 2021). The experiment included 16 bioretention columns out of which eight were placed in the greenhouse with 25 °C and eight in a refrigerated container at 1.5 °C and the measured contaminants included ammonium-nitrogen (NH4-N), nitrite/nitrate-nitrogen (summarized as NOx–N) and total nitrogen (TN) (Søberg et al., 2021). Bioretention cells with SZC at low temperature 1.5 °C had the highest nitrogen removal (Søberg et al., 2021). Longer ADD period decreased NOx-N and thus TN removal for standard bioretention cells however, the design with SZC eliminated this effect (Søberg et al., 2021). Moreover, sand-based bioretention columns with or without SZC were subject to low (4.6 °C) or high (17.7 °C) temperatures and exposed to stormwater with NaCl to study phosphorus removal (Søberg et al., 2020). High temperature and salt decreased TP removal and salt had no effect on DP removal (Søberg et al., 2020). Inclusion of SZC improved TP removal however, it was regarded to be of little practical significance since the standard columns also performed well (Søberg et al., 2020).

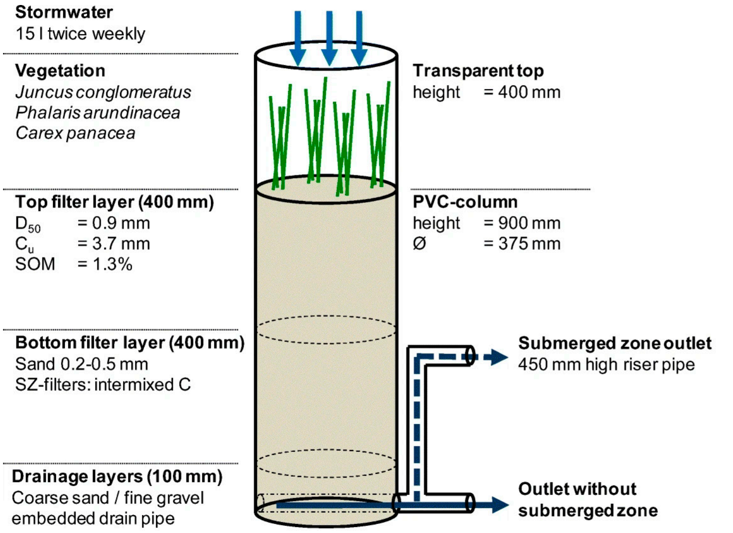

Figure B8. Bioretention column design used in laboratory testing by Søberg et al. (2020). Published under Creative Commons Attribution (CC BY 4.0) license.

Although the cold temperature and high salt loads are assumed to have a negative effect on the performance of organic removal, this assumption has not been definitively substantiated (Kratky et al., 2017). For example, in a laboratory-based adsorption/desorption study, the cold temperature (min of 4 °C) and high salinity (3 g/L) did not negatively affect the adsorption of benzotriazole by bioretention soils (Rhodes-Dicker and Passeport, 2019).

DESIGN IMPLICATIONS

There are several design implications that could be summarized. As reviewed by Kratky et al., (2017), pollutants that are removed by mechanical filtration such as TSS and particular metals are not affected by the cold climate conditions (if media is not frozen). In general, for bioretention systems that operate in cold climate regions, recommendation is to use coarse filter material that is well drained after the rain event preventing ice blockage in the filter and, thus, allowing infiltration even after/during temperature below zero (Blecken et al., 2011; Beryani et al., 2021). Another implication of this is prevention of filter clogging due to higher TSS concentrations in the snowmelt (Kratky et al., 2017). Still, the material should not be too coarse so that filtration and adsorption processes still can take place. Particle size distribution is an important design factor for the biofilter with higher efficiency with greater particle size (Taleban et al., 2009). A total of 50 influent experiment with standard and high SS loads were carried on a 10 types of bioretention columns to investigate permeability (Gong et al., 2022). The study showed the effect of SS loading on the decreased permeability and clogging of the system (Gong et al., 2022). However, more research is needed to investigate clogging of pores and creation of pores by plant growth and decay over time (Skorobogatov et al., 2020).

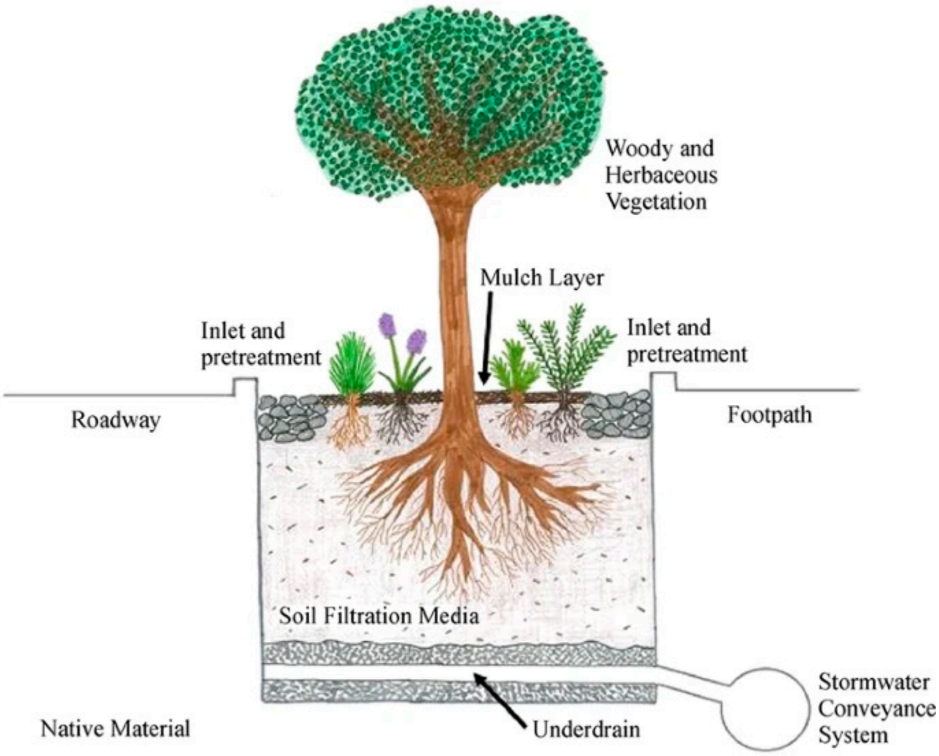

Figure B9. Schematic representation of a bioretention design. Reproduced from Kratky et al. (2017) with permission from Springer Nature.

When it comes to dissolved pollutants, the removal is more dependent on the bioretention design e.g., inclusion of the submerged zone , soil amendments, contact time etc. (Kratky et al., 2017). Lefevre et al. (2015) reviewed previous research on removal of dissolved pollutants in bioretention cells and found that denitrification in bioretention cells increases with residence time, organic carbon content and temperature. Even though denitrification is limited during short retention times in biofilters (Robertson et al., 2018), temporarily creating internal water storage zone can promote denitrification and reduce leaching of nutrients which can also be used in combination with the usage of filter amendments (Zhang et al., 2023). During ponding in bioretention systems, particulate phosphorus can settle and soluble phosphorus can be sorbed onto mulch and bioretention soil particles as well as colloids suspended in stormwater or soil pore water (Roy-Poirier et al., 2010b). In addition to good hydraulic performance, bioretention design should minimize leaching and support plant growth (Skorobogatov et al., 2020).

Sagrelius et al. (2023) compared various bioretention types to gain a better understanding of the relative sustainability (social, economic, and technical-environmental criteria). Twelve bioretention design configurations were selected from bioretention systems constructed between 2015 and 2019 in Sweden. The system with a water-saturated zone and a variety of plant species outperformed the other systems in the technical-environmental criteria (Sagrelius et al., 2023). Dependent on the targeted pollutant the usage of salt can have a negative effect for the bioretention removal efficiency (Kratky et al., 2017). Low temperature and salt may affect plant and bacteria growth which is important for the dissolved metals and nutrients thus, in areas where salt is applied for winter road maintenance, salt tolerant vegetation is needed (Kratky et al., 2017).

In fact, a review paper recommends more research in the role of plants since they are a “key feature” in bioretention design (Spraakman et al., 2020a). Another review paper also stresses the role of vegetation in bioretention design especially interaction with media (Skorobogatov et al., 2020). The review paper focused on examining the importance of plants in bioretention systems concludes that plant effectiveness depends on which pollutant is being considered e.g., nitrogen (runoff retention, plants are important) or TSS (physical filtration, plants less important), dissolved phosphorus (chemical adsorption, plants less important) (Dagenais et al., 2018). Thus bioretention performance is affected by plant choice mainly root characteristics and growth rate (Dagenais et al., 2018). The authors concluded that research on plants in bioretention needs to be broaden to include range of regions and climates (Dagenais et al., 2018) especially when plants in bioretentions operating in cold climate next to roads are exposed to “periodic inundation, de-icing salts, road dust, splashes of water from the road, freezing and thawing of soil, and periods with ice cover during the winter” (Laukli et al., 2022). For example, investigation of 31 plant species showed that *Iris pseudacorus* had high tolerance to de-icing salts, and *Amsonia tabernaemontana* and *Baptisia australis* were also mentioned (Laukli et al., 2022).

Figure B10. Plants identified by Lauki et al. (2022) to be tolerant to salt. From left to right: *Iris pseudacorus*[5], *Amsonia tabernaemontana*[6] and *Baptisia australis*[7].

A general recommendation is that the design of bioretention in cold climate considers local characteristics and concerns. For example, if the major concern is the flooding of the catchment the design objective should be peak flow and volume reduction, and combination of the biofilter with detentions systems (Kratky et al., 2017). However, optimizing water quantity control lowers effectiveness in reducing dissolved concentrations of nutrients, organic compounds, metals (Kratky et al., 2017; Okaikue-Woodi et al., 2020). In addition, care should be taken to prevent contamination of groundwater (e.g., lining the bioretentions and having an underdrain) since removal of chloride is limited (Kratky et al., 2017).

The biofilters investigated exhibited various designs, and their performance in the removal of organic contaminants was compared across different types, such as sand-, vegetated- and chalk-amended vegetated biofilter in Beryani et al. (2023). Similarly, the removal of microplastics was compared between sand- and vegetated biofilters by Lange et al. (2021) and Lange et al. (2022).

Among the organic contaminants, variations in the removal efficiency between biofilters of different designs were partially attributed to the physicochemical properties of the contaminants. For example, less efficient removal of less hydrophobic substances, such as BPA and OP, was identified for sand biofilter, compared with other substances of higher hydrophobicity, such as PAH-M, PAH-H and long-chain (C_{16}-C_{40}) alkanes (Beryani et al., 2023). Filtering was considered a dominant mechanism in the removal of these hydrophobic substances in the sand biofilter. In comparing the performance between different biofilter designs, vegetated biofilters were shown to substantially enhance the removal of substances including phenolics, PAHs and hydrocarbons. However, no statistically significant difference in the performance of organic contaminant removal between chalk-enhanced vegetated and vegetated biofilters was observed (Beryani et al., 2023).

Mechanical filtration was suggested as the key mechanism for the removal of microplastics in biofilters (Lange et al., 2021, 2022; Smyth et al., 2021). The vegetated biofilter was identified as presenting better performance in the removal of 20–100 µm microplastics, where the median effluent particle concentration from the vegetated

[5] Photo obtained from: https://www.flickr.com/photos/coanri/98621275,

[6] Photo obtained from: https://commons.wikimedia.org/wiki/File:Amsonia_nadrenska_Amsonia_tabernaemontana.jpg

[7] Photo obtained from: https://commons.wikimedia.org/wiki/File:Baptisia_australis_kz04.jpg

biofilter (26.5 particles/L) was approximately five times lower than that from the sand biofilter (121 particles/L) (Lange et al., 2022). However, both sand- and vegetated biofilters demonstrated equally good performance in removing rubber and bitumen particles in the size range of 100–300 µm (Lange et al., 2021). Although the vegetated biofilter was identified as statistically significantly more effective at removing other microplastics compared with the sand biofilter, the difference in effluent particle concentrations between both types of biofilters (0.33 and 0.86 particles/L) was considered marginal and of limited practical significance.

Three bioretention design parameters were suggested to substantially impact the removal of organic contaminants: 1) submerged zones, 2) bioretention media composition, and 3) vegetation (LeFevre et al., 2015). As an example, the presence of a topsoil layer composed of sand to silty sand on the surface of a vegetated biofilter in Beryani et al. (2023) noticeably reduced the infiltration rate, thereby enhancing the filtration effect at the beginning of the runoff event. Besides, the amendment of organic mulch in the topsoil layer to support the growth of plants, along with the accumulation of organic matter from decaying vegetation, may also enhance the adsorption of contaminants to the soil media, thus improving the biofilter's treatment efficiency. In a laboratory biofilter column study by LeFevre et al. (2012), sorption was identified as the dominant mechanism for the removal of NAP (56–73 %), followed by biodegradation (12–18 %) and plant uptake (2–23 %). However, the amendment of organic mulch in the substrate of infiltration-based facilities may not necessarily lead to an enhanced adsorption performance for contaminants. In a laboratory-based adsorption/ desorption study, benzotriazole was found to be more efficiently adsorbed by normal bioretention soil (organic matter content of 13 ± 1 %) than by the hardwood mulch (organic matter content of 92 ± 4 %) (Rhodes-Dicker and Passeport, 2019). The presence of polar functional groups, such as carboxyl-, phenoxy- and hydroxyl-groups associated with the protein faction in soil, favours better adsorption of polar compounds like benzotriazole, compared to the lignin component of hardwood mulch.

Essentially, the large molecule size, low solubility, and high hydrophobicity of the majority of organic substances pose challenges to the plant uptake processes, due to the limited capacity of these substances to pass through cell membranes (LeFevre et al., 2015). Instead, the role of plants in providing favourable conditions for bacterial activities is considered the main contribution to enhancing the removal of organic contaminants in infiltration-based facilities. However, the root system of the plants may create macropores in the biofilter media, which can lead to the formation of preferential flow paths (Dagenais et al., 2018; Skorobogatov et al., 2020). This can limit the interaction between the contaminants and the bulk media, potentially leading to the breakthrough of contaminants.

The facility's construction material (e.g. stormwater pipeline, cell separator membranes and geotextiles) may also constitute an essential source of contaminants. For example, the negative removal rate of NP reported in Beryani et al. (2023), may be attributed to the underground ethylene propylene diene monomer rubber membranes and/or polypropylene-made geotextile.

MODELLING STUDIES

When modelling the hydrology of a sub catchment it is important to consider the cold climate conditions, i.e., snow cover build-up and meltdown since this will affects the bioretention performance (Gougeon et al., 2023). SWMM was used to simulate the quantity and treatment performance (TSS, Cr, Pb, Zn and Cl^-) of bioretention cells during snowmelt (Gougeon et al., 2023). The results showed that the bioretention performance in managing snowmelt was dependent on the land use type, impervious surface type and snow management practices (Gougeon et al., 2023). The case study catchment was in Quebec, Canada consisting of natural undeveloped parcel, residential, industrial and snow storage area. Assessing several BGIs scenarios, the model suggested the largest improvement of snowmelt quantity and quality when installing bioretention in industrial catchments and around snow storage sites (Gougeon et al., 2023).

Another study modelled hydrological performance and nutrient removal of bioretention considering cold climate conditions, i.e., snowmelt, spring runoff events and freeze-thaw cycles (Yu et al., 2023). The chosen model was HYDRUS 1D which was calibrated and validated for the ponding depth and outflow hydrograph data collected in the laboratory study on different bioretention column designs that considered cold climate conditions (Yu et al., 2023). The modelling study recommended using loamy sand and soil media thickness of 60–80 cm (Yu et al., 2023). Simulated results and results from controlled lab experiments showed ability for phosphate, nitrite and ammonium load removal of > 90 % but leaching for chloride and nitrate caused by de-icing salts and nitrification (Yu et al., 2023).

In a field study from Canada, six 20 m long separate cells, lined with one of three tested liner types were investigated (Trenouth et al., 2018). The facility was considered enhanced facility compared to the typical swale. The field monitoring took place in years 2013/2014 and 2014/2015 over the months October to March during which time 17 precipitation and snowmelt events were captured. HYDRUS-1D was used and the model was calibrated on the field data and was used to estimate the thickness of an amendment (IRON stockpile) layer required to treat Cu, selected as the case study metal. Simulation results showed that ~20 cm thick layer was needed to bring the effluent Cu concentrations below the effluent guideline value after 20 year operation, a typical life spam for roadside ditch. Moreover, comparison of HYDRUS-1D simulated underdrain chloride concentrations and measured ones from two compacted clay liners in the filtration system indicated that the model could be used to stimulate chloride attenuation by the filtration system (Trenouth et al., 2018).

B4.2.3 Filter materials

CONTROLLED FIELD AND LABORATORY INVESTIGATIONS

Previous research showed that in the case of metals, Total-P etc., relatively simple, vegetated sand-based filters perform well. However, a wide range of other filter materials and amendments have been investigated and these studies are discussed in this section. Majority of the studies were controlled field or (more often) laboratory investigations where sorption and desorption in different materials were tested by column and batch adsorption tests. Such studies although in controlled environment different from real world conditions are relevant since they give an idea about which materials are good candidates with the potential to be upscaled to filed and further tested. The following section summarizes the relevant findings.

Ability of different biochar to retain Cd, Zn, Cu, Pb, and de-icing salts (Na) was investigated as well as the optimal amount of **biochar** in soil mixture (comprising of biochar, soil, and compost) (Seguin et al., 2018). It was found that compared to the control mixture (soil, 7.5 % compost, 0 % biochar), soil mixture which included 7.5 % biochar (by dry weight), increases the soil sorption capacity for Cd, Na and has the similar sorption capacity for Cu, Zn and Pb and was recommended as an useful amendment for urban soils such as street tree pits (Seguin et al., 2018). **Compost**-amended sands used for the bioretention media were studied and they were found to be able to retain metals however, there was a concern for P leaching (Paus et al., 2014a). For minimizing release of dissolved P using an iron-amended sand layer below the compost as well as having a balance between rapid draining (large grain sizes) and effective pollutant removal was recommended (Paus et al., 2014a). The type of gravel (with and without iron oxide mineral coating) was investigated for its efficiency to adsorb metals (Norris et al., 2013). The experiments showed that microgabbro gravel that has clay minerals on it surface is suitable for gravel used in filter drains without need for iron oxide mineral coating (Norris et al., 2013).

EFFECT OF SALT

Effect of salt was investigated in several controlled studies. Solutions with NaCl addition reduced performance of **peat** (Kalmykova et al., 2009). Dissolved organic carbon (DOC) was leached in the initial phase of the filter operation and decrease in metal removal was recorded although, sorption was restored quickly for Zn, more gradually for Pb and Ni, and not for Cu (Kalmykova et al., 2009). Metal retention was not significantly affected by the rise of pH from 6.7 to 8.0, by the rise in metal concentrations from 100 µg/L to 1 mg/L and by freezing (Kalmykova et al., 2009). However, the metals which were not effectively removed by peat columns were As and Cr (Kalmykova et al., 2009). Peat was recommended as suitable material to be used for treatment of Cd, Cu, Zn, Ni and Pb from stormwater (Kalmykova et al., 2009). DOC was used as a surrogate parameter to measure performance of filters (**crushed clay and granular activated carbon**) to treat organic de-icing chemicals, and no adsorption of DOC was observed (Raspati et al., 2018). In the same study, the filtration experiment was conducted on two filters with crushed clays which showed that the DOC degradation occurs in the top 20 cm layer however, the authors note the importance of studying other factors not covered in the experiments, including freezing condition (Raspati et al., 2018). Metal adsorption of **sand filters and three adsorbents charcoal, pine bark, and olivine** were compared during high hydraulic loads and the application of de-icing salts (Monrabal-Martinez et al., 2017). The sand filters amended with pine bark and olivine had the best performance (Monrabal-Martinez et al., 2017). The effect of salt on already retained metals was studied and addition of NaCl (1 g/L) did not negatively impact desorption of already adsorbed metals except for charcoal in the case of Ni (Monrabal-Martinez et al., 2017). Combining two adsorbents, **pine bark and olivine**, was recommended to have the optimal performance and the cost as well as having a pre-sedimentation step to reduce clogging, however with a need to be tested in the field conditions (Monrabal-Martinez et al., 2017). Eight adsorbent materials i.e., bottom ash, fly ash, pine bark, seaweed, olivine, sawdust, charcoal and zeolite were investigated during rapid batch adsorption tests using synthetic stormwater (Ilyas and Muthanna, 2017). Experiments with synthetic stormwater with NaCl addition (from 10 and 1200 mg/l) showed that salt had very little effect on adsorption of metal ions on **olivine and charcoal** (Ilyas and Muthanna, 2017). Multi-criteria modeling (MCM) was used to decide which materials are the best for the upscaling which showed that **pine bark, olivine and charcoal** were the best available options for upscaling (Ilyas and Muthanna, 2017). In another study five low-cost sorption materials **(crushed limestone, shell-sand, zeolite, and two granulates of olivine)** were investigated for their ability to improve the quality of artificial stormwater containing phosphorus, arsenic, and metals (Cd, Cr, Cu, Ni, Pb, Zn) (Wium-Andersen et al., 2012). Sorption capacity was investigated by batch experiments and desorption was analyzed by sequential extraction with three extraction steps: i) dionized water, ii) sodium chloride and iii) sodium hydroxide (Wium-Andersen et al., 2012). The two final steps were ment to simulate Danish winter conditions, where NaCl is used as deicing agent on roads during winter road maintenance (Wium-Andersen et al., 2012). Considering different factors (e.g., unit cost of the material, soprtion capacity, change in pH etrc.), different materials had different strengths and there was no ideal material that met all criteria (Wium-Andersen et al., 2012). The authors mention that the important thing is to have a targeted pollutant in mind when designing a filter e.g., "filter designed to protect a sensitive lake towards phosphorus" (Wium-Andersen et al., 2012). Trenouth and Gharabaghi (2015) tested metal adsorption of locally available materials (slag from steel mill, a drinking water treatment residual, overburden from a coal mine and woodchips from landscaper) using shaker and soil column tests. The study concluded that the materials (**coal mine overburden, steel mill slag and wood chips**) could be amended into soils to improve dissolved metal removal and that metal remobilization was not recorded for high chloride concentrations applied in the column tests on materials basic oxygenated furnace, blast furnace and goethite-rich overburden (Trenouth and Gharabaghi, 2015).

EFFECT OF TEMPERATURE

When it comes to temperature, it is recognized that the temperature is expected to affect the adsorption to various amendments, and consequently the efficacy of amendments in mitigating nutrient leaching in cold winter needs to be investigated (Zhang et al., 2023). Effect of temperature varies dependent on the material in question. Metal solutions (single- and multi-solute) were used in adsorption tests as well as two samples of real stormwater at different temperatures (5, 15 and 40 °C) to study removal of metals (Cd, Cu, Ni, Pb, and Zn) in cold climate conditions. The study investigated used compost as an adsorbent and concluded that lower temperatures do not affect the adsorption process (Pennanen et al., 2020). However, as previously described using compost leads to phosphorus leaching (Paus et al., 2014a) and brings the concern of compost degrading over time, thus it is not recommended. Ten, vegetated bioretention cells (five different designs) were investigated in the field experiment with synthetic stormwater from May through November 2010 at University campus in Guelph, Canada to study the removal of nutrients (Randall and Bradford, 2013)**.** The designs included: i) sandy soil mix with organic matter and sandy soil mix with some amendments ii) shredded newsprint, iii) lanthanum-modified bentonite product, iv) alum-based drinking water treatment residuals and v) oxide-coated media. Bioretention cells were planted with plants which can tolerate well drained sandy soils, periodic inundation, low temperatures and salt. Sandy soil mix had satisfactory performance 75.5 % of TP and 53.4 % of TN and type of OM in a bioretention garden was an important factor in achieving nutrient removal (Randall and Bradford, 2013).

B4.3 RQ 2.3 Which type of maintenance is needed for the different types of facilities?

As with all facilities, proper inspection and maintenance of blue-green infrastructures for stormwater management is necessary in order to ensure that they continue to fulfil their function over the long-term (Blecken et al., 2017; Langeveld et al., 2022). However, several studies show that it is often neglected, both in Sweden (Al-Rubaei et al., 2017b; Beryani et al., 2021; Blecken et al., 2017; Starzec et al., 2005) and in other cold-climate countries (Langeveld et al., 2022; Mooselu et al., 2022). As decentralized and multifunctional facilities, sometimes located on privately-owned land, BGI require organizations to find new strategies to ensure maintenance than those used for the maintenance of the centralized, single-purpose, publicly-owned facilities typically associated with the water sector (e.g. sewers or treatment plants). In this context, organizational issues (e.g. insufficient communication and unclear responsibilities between the numerous actors involved) can result in insufficient maintenance (Blecken et al., 2017; Langeveld et al., 2022; Wang et al., 2023). In addition, there is a lack of knowledge on the timing and type of maintenance measures necessary to maintain the various technical functions of these relatively new types of facilities; faced with this this uncertainty, the needs for maintenance are frequently ignored at the time of construction, resulting in a lack of planning or attributed funding (Blecken et al., 2017; Langeveld et al., 2022). The following section summarizes what is currently known about the operation and maintenance needs of different types of BGI, followed by a brief discussion on approaches for better predicting the required frequency of maintenance activities.

Table B10. Recommended maintenance activities according to facility type

Stormwater control measure	Inspection of inlet and/or outlet structures	Removal of trash and large debris	Removal of sediments	Care for vegetation (mowing, pruning, removal, weeding, replanting)	Removal and/or replacement of surface layer of soil or filter media	Removal of particles from surface layer of structure
Wet pond	X[1,2]	X[1,2,3]	Forebay and/or whole facility[1,4,5,6,7,8]	X[1,3,5]		
Constructed wetland	X[1]	X[1,3,7]	Forebay and/or whole facility[1,7,9,10]	X[1,3,5]		
Grassed swale/ vegetative filter strip				X[11,12,13]	X[5,13]	
Biofilter	X[1,14]	X[1,14]	Forebay[1,12,14]	X[1,14]	X[1,5,14,15,16]	
Infiltration trench					X[1,5,17]	
Permeable pavement						X[1,5,18]

[1]Blecken et al., 2017, [2]Starzec et al., 2005, [3]Badiou et al., 2019, [4] Gavric et al., 2022, [5]Langeveld et al., 2022, [6]Olding et al., 2004, [7]Suits et al., 2023, [8]Wik et al., 2008, [9]Han et al., 2014, [10]Mungasavalli and Viraraghavan, 2006, [11]Al-Rubaei et al., 2017b, [12]Ekka et al., 2021, [13]Mooselu et al., 2022, [14]Beryani et al., 2021, [15]Furén et al., 2023, [16]Kratky et al., 2017, [17]Bergman et al., 2011, [18]Drake et al., 2013.

B4.3.1 Wet pond / constructed wetland

Wet ponds and constructed wetlands have similar maintenance requirements. Inlet and outlet structures must be inspected on a regular basis (e.g. annually) (Blecken et al., 2017; Starzec et al., 2005) to verify that they are not obstructed; at this time, any observed trash or debris should be removed (Blecken et al., 2017; Starzec et al., 2005) .

As plants play an important role in the water quality performance of constructed wetlands, maintaining a functioning ecosystem, which may require adjusting plantings over time (Badiou et al., 2019), is of high importance, and the water quality performance of these facilities can improve over time as vegetation establishes (Al-Rubaei et al., 2016; Guesdon et al., 2016). Cutting and removal of vegetation may be necessary in both ponds and wetlands to avoid overgrowth (Blecken et al., 2017; Langeveld et al., 2022) and the proliferation of mosquitoes (Blecken et al., 2017). A study in Canada identified significant environmental costs (water quality and greenhouse gas emissions) associated with control of vegetation (removal of aquatic macrophytes and filamentous algal mats by harvesting or herbicide application) in wet ponds, motivated by aesthetic expectations of residents; because constructed wetlands incorporate diverse vegetation into their design, less maintenance is needed to achieve the desired aesthetic, resulting in less required maintenance and better overall performance (Badiou et al., 2019).

To maintain function long-term, sediments must be removed from both wet ponds and constructed wetlands periodically. Failure to remove sediment can result in reduced hydraulic capacity and high discharges of suspended solids (Olding et al., 2004), particularly under turbulent conditions (Starzec et al., 2005). Both constructed wetlands and stormwater ponds are often constructed with a forebay, which can achieve a high removal of particles upstream of the main part of the facility (Han et al., 2014). Forebays should be constructed to facilitate sediment removal, which must be undertaken regularly (Al-Rubaei et al., 2017b; Blecken et al., 2017). Well-designed and well-maintained forebays reduce the necessary frequency of sediment removal from the rest of the facility and help to sustain long-term performance (Al-Rubaei et al., 2016). For example, removal of sediment from the forebay of a constructed wetland on average every five years allowed sustained performance of a constructed wetland over a 19-year period (Al-Rubaei et al., 2016). The necessary frequency of sediment removal at a given facility is difficult to predict (Langeveld et al., 2022; Suits et al., 2023), as it depends on both facility design and catchment properties, including activities which may change over time (e.g. construction activities higher suspended solid inputs) (Olding et al., 2004). Proposed approaches for identifying when removal is needed include annual professional inspection of facilities to verify functioning (Blecken et al., 2017), computational fluid dynamics (CFD) modelling of particle settling and sediment accumulation (Han et al., 2014) and the use of water-quality sensors to monitor suspended particle concentrations at the outlet (Suits et al., 2023).

The sediment removed from ponds and wetlands may be toxic (Wik et al., 2008) and contains both organic substances and metals, which can be mobilized during removal (Gavrić et al., 2022; Karlsson et al., 2016). These sediments need to be dried before removal (Mungasavalli and Viraraghavan, 2006), which is often done near the facility, discharging leachate water back to the facility, which is counterproductive to the function of a pond (i.e. loads of leached pollutants will only have been postponed)

(Blecken et al., 2017). The quality of sediment can vary significantly between ponds; therefore sediment quality must be evaluated and an environmental risk assessment conducted to make decisions regarding disposal method (Gavrić et al., 2022).

B4.3.2 Grassed swale / vegetative filter strips

Grassed swales and vegetative filter strips require relatively little maintenance. The most frequent maintenance activity is mowing of the grass (Blecken et al., 2017; Mooselu et al., 2022), which is typically carried out twice a year (Mooselu et al., 2022). Because grassed swales and vegetative filter strips receive both more water and more nutrients than typical turf areas, they typically do not require fertilization. Care should be taken not to mow these facilities after major rains as they are typically soggier than other turf surfaces and mowing at such times can cause rutting and scarring of the surface and vegetation (Blecken et al., 2017). In cold climate locations, swales and vegetative filter strips used for snow storage in winter may accumulate traction grit, which may be cleaned mechanically in the spring using loader-mounted rotating street sweepers (Blecken et al., 2017).

Grassed swales and vegetative filter strips typically receive runoff as overland flow from an impervious area such as a road. Over time, road subsidence, sediment accumulation and turf growth will cause the surface of the swale or vegetative filter strips to rise relative to the road surface (Langeveld et al., 2022), leading to a blockage of this overland flow (Ekka et al., 2021; Langeveld et al., 2022; Mooselu et al., 2022). Further, infiltration capacity may be lost, and pollutants may accumulate in the top layer of soil (Langeveld et al., 2022). Therefore, periodically, the turf and top layer of soil/sediment must be removed to restore the level relative to the surface of the road (Langeveld et al., 2022; Mooselu et al., 2022).

B4.3.3 Biofilter

The most frequent maintenance required for biofilter systems is the maintenance of vegetation, which may involve pruning (Beryani et al., 2021; Blecken et al., 2017), replanting and removal of weeds, particularly in during the first two years of operation (Blecken et al., 2017), replacement of mulch (Kratky et al., 2017), and (if existent) emptying of the sediment forebay. Inlet and outlet structures should also be verified regularly and cleaned to remove trash or large debris that may block flow (Beryani et al., 2021; Blecken et al., 2017).

Over time, sediments and pollutants accumulate in the biofilter (predominantly on the surface) (Furén et al., 2023). Modelling studies have shown that theoretically, over time, a biofilter may lose its ability to remove dissolved pollutants due to the saturation of all sorption sites in its filter media (Rodríguez-Gómez et al., 2021). However, field studies have demonstrated that clogging is likely to be more limiting to the effective lifespan of biofilter facilities than saturation of sorption sites (Furén et al., 2023), Clogging can often be remediated by replacing the top layer of the filter (Beryani et al., 2021; Blecken et al., 2017; Gong et al., 2022; Langeveld et al., 2022; Marvin et al., 2020), which also typically removes the most contaminated filter media (Furén et al., 2023). In some cases, replacement of all filter media may be necessary (Blecken et al., 2017).

Proposed frequencies for the replacement of the top layer of filter media to prevent clogging vary from 2–3 years to > 25 years (Furén et al., 2023). The sediment loads reaching the biofilter are more important to the duration before clogging than hydraulic loads (Gong et al., 2022). The loads reaching the main part of the biofilter can be limited by constructing a forebay at the biofilter inlet, which requires regular sediment removal (Blecken et al., 2017; Ekka et al., 2021). Besides this, the duration to clogging depends on various design factors. The presence of plants, particularly those with thicker rhizomes and fewer fibrous roots, reduces the clogging potential of biofilters as compared with non-vegetated filters (Gong et al., 2022). Filter media composition can also influence the duration to clogging: filter media with very high permeability tends to clog more rapidly (Gong et al., 2022) and some amendments (e.g. Ca amendments, alum, fly ash and Phoslock, an additive for phosphorus removal) may increase clogging potential (Marvin et al., 2020). Biofilters with a thinner filter media layer tend to clog more quickly than those with a thicker layer, as to those with a submerged zone (Gong et al., 2022).

B4.3.4 Infiltration trench

Infiltration trenches can lose permeability over time, both on the bottom and on side walls, limiting performance as the facility ages (Bergman et al., 2011). This clogging can occur within only a few years of construction; one study found that clogging occurred due to leaf litter after only 1.5 years of operation (Blecken et al., 2017). Reducing the sediments and leaf detritus entering these systems, for example, by constructing them downstream of a pretreatment device, such as a vegetative filter strip, may increase the duration to clogging (Blecken et al., 2017). When it occurs, clogging can be remediated by removing the top layer of soil (Blecken et al., 2017; Langeveld et al., 2022) or by tilling the infiltration surface. Sometimes infiltration trenches are topped by permeable paving systems; this is problematic as the devices recommended for the maintenance of permeable pavements (e.g. vacuum cleaning, see next section) may not be able to access these facilities (Blecken et al., 2017).

B4.3.5 Permeable pavement

The main maintenance issue related to permeable pavement (PP) is clogging by traffic-related particles (Blecken et al., 2017; Drake et al., 2013; Langeveld et al., 2022), The duration to clogging can vary from less than a year (PP can clog in only one winter in cold climates where sand is applied for traction) to multiple years. It depends on the surrounding landscape, the interface between permeable and impermeable pavements, the presence of “dirty vehicles”, the proximity of snow disposal, and the prevalence of sources of particles such as construction sites or soil disturbance (Blecken et al., 2017), as well as on PP design (e.g. installing a geotextile at the bottom of the structure may cause it to clog at the bottom, while including plants in the structure can help to sustain permeability) (Drake et al., 2013). Current knowledge does not allow the duration to clogging to be predicted. The development of models predicting clogging may improve this situation (Drake et al., 2013); alternatively, piezometers, pressure transducers or temperature sensors could be used to identify the occurrence of clogging (Suits et al., 2023) or annual inspections of function could be made (Blecken et al., 2017).

Various techniques have been proposed to unclog permeable pavements, including vacuum sweeping, power washing, sweeping with a push broom and washing surfaces with a large-diameter hose. The effectiveness of maintenance techniques depends on the type of permeable pavement: while some studies have found vacuum washing to be most effective, others have found pressure washing to be most effective. In general, cleaning partially rather than entirely restores initial infiltration rates and clogging may be irreversible (Drake et al., 2013). Preventative vacuum-cleaning in combination with pressure washing (i.e. cleaning before clogging occurs) is recommended to maintain long-term function (Blecken et al., 2017).

Another issue that may cause failure of permeable pavement is structural failure (i.e. excessive heaving, rutting, cracking or revealing), which is a common problem with pavements in cold climates, particularly where there are many freeze-thaw cycles. Permeable pavements may be less subject to structural failure than traditional pavements because (1) they are designed to drain rapidly and are unlikely to contain water when they freeze and (2) the infiltration of water may keep them warmer and reduce the frequency of freezing. For the same reasons, clogged permeable pavements are more vulnerable to frost than unclogged permeable pavements, so maintenance to control clogging is also a protection against this type of failure (Drake et al., 2013).

B4.3.6 Overcoming uncertainties in the required frequency of maintenance

For all BGI facility types, there is significant uncertainty in the duration before which heavy maintenance activities, including sediment removal or replacement of the top layer of soil or filter media, are needed. This makes it difficult to plan maintenance activities (Blecken et al., 2017; Langeveld et al., 2022) and creates a significant source of uncertainty when evaluating the environmental impact, sustainability and life-cycle cost of these systems (Brudler et al., 2019). Models have been proposed as a method for evaluating when and where maintenance may be required (Drake et al., 2013; Han et al., 2014; Rodríguez-Gómez et al., 2021); however, existing models have not been validated for this use, as the necessary data sets (long-term monitoring of aging facilities) do not exist. In addition, generalized models are also difficult to develop because BGI systems are unique (i.e. not standardized) systems located in an uncontrolled urban environment (Langeveld et al., 2022). In the absence of reliable models, systems must be monitored to identify when maintenance is necessary. Some authors have proposed regular (i.e. annual) inspections to identify when and which maintenance is necessary (Blecken et al., 2017), while others look to the use of sensors to aid in monitoring, while also noting that sensors themselves require maintenance (Langeveld et al., 2022; Suits et al., 2023).

B5. Conclusions

Conclusions drawn from this review related to each research question are separately summarised below.

B5.1 RQ 2.1 What is the treatment performance for different types of facilities and target pollutants?

For RQ 2.1, field studies carried out in cold climate on stormwater facilities such as ponds and wetlands (3 studies), biofilters (6 studies), grassed swales (1 study), permeable pavements (1 study), reactive filters (3 studies) and treatment trains (3 studies) were identified. In addition, a laboratory study evaluating the potential to apply coagulation/flocculation was also reviewed. Overall, these studies are too few to compare and assess the performance of different technologies to each other.

Most studies have investigated the removal of heavy metals (mainly copper, zinc and lead), sediments (measured as TSS) and nutrients (total phosphorus and total nitrogen and sometimes their speciation) from stormwater. A few studies, but not for all types of facilities, have also investigated organic pollutants (e.g. PAHs, oils, microplastics, etc.).

In general, most of the studies, independent of technology, reported removal efficiencies for total copper, zinc, lead, TSS and phosphorous of 50 % or better on event basis. Poorer removals were observed for the dissolved phase in comparison to total concentrations. Lowest removal efficiencies were obtained for the one study on swales where the studied swale was not able to remove TSS or metals to a high extent, often negative removals were reported. Extremely negative removals (–2221 % to –71 %) were occasionally also observed for a wetland. In both studies, these observations were mainly associated with low inflow concentrations. In most types of facilities, nitrogen reduction is in comparison to other pollutants quite low, rarely above 50 % and more frequently negative removal rates were observed (i.e. increased concentration) such as in several of the tested biofilters.

B5.2 RQ 2.2 Are there any design criteria and recommendations affecting the treatment performance and are these sensitive to differences in climate?

Several factors and activities were mentioned in the reviewed articles to be taken into account when designing and construction stormwater treatment systems in cold climate. These included the directly climate related factors such as low air temperature, ice cover on ponds and snow on the ground, frost in biofilter media and soils and freeze-thaw cycles, significant snowmelt volumes in spring, non-growing or short growing season and dormant plants. In addition, activities related to the cold seasons included usage of grit and de-icing salt for winter road maintenance, increased asphalt wear due to studded tyres and cold starts of car engines.

For ponds and wetlands, the role of plants has frequently been investigated. Plants have been shown to increase the removal of nitrogen from the water. Large biomass of the plants may also increase the uptake of heavy metals and chloride. On the contrary, low temperatures and high salt concentrations may affect the removal of (especially dissolved) metals negatively. Furthermore, it should be acknowledged that the plant selection can influence and improve the removal of targeted pollutants. It is also important that the plants are tolerant to salt.

The function of biofilters in cold climate and factors affecting their performance have been studied to a large extent in laboratory studies. TSS and particle bound pollutants are mainly retained by physical filtration processes and is little affected by temperature, salt or any other factors related to cold climates. However, due to high concentration of TSS in snowmelt, coarse and well-draining filter materials are promoted to prevent the filters from clogging. For dissolved pollutants the removal is more dependent on the design of the biofilter, e.g. inclusion of a saturated zone, soil amendments and contact time. Salt in general have negative impact on the retention of heavy metals. From the metal species usually studied (Zn, Cu, Pb, Cd) it seems like Cu, Pb and to some extent Cd were the most sensitive to salt and may be desorbed from the particulate phase and pass the filter bed. Dissolved Cu was also sensitive to temperature with higher temperatures leading to increased release of Cu mainly attributed to increased degradation of organic matter and its association with Cu. Nitrogen removal can be improved by submerged saturated zones and lower temperatures has been shown to have a positive effect on the removal of nitrogen as well as phosphorus. Longer dry periods affect nitrogen and metal removal negatively. For organic pollutants, in general, climatic effects are less investigated though it is assumed that cold temperature and high salt loads have a negative effect on the removal but there are too few studies to make any conclusions. Plants in biofilters may affect the treatment performance. Better performance has been shown for planted biofilters compared to (non-planted) sand filters for phenolics, PAHs and hydrocarbons and to some extent also for microplastics. However, the root systems may create macro pores in the filter media leading to the formation of preferential flow paths and breakthrough of contaminants. Finally, the materials used for the construction of biofilters should also be considered with respect to sustainability and since they may contain pollutants which can leach and be released with the effluents.

A wide range of filter materials have been investigated in laboratory studies (batch and column tests) with the purpose of using the materials as amendments in biofilters, tree pits etc. to increase treatment efficiency. Materials investigated include biochar, charcoal, iron-treated sand, compost, peat, clay, granulated activated carbon, pine bark, olivine, crushed limestone, zeolite, shell sand and wood chips. In addition to examining the materials themselves, the effect of salt and temperature was also evaluated in several of the studies. The most common substances to be studied in this type of experiments were limited to heavy metals (Cu, Zn, Pb and occasionally Cd, Cr, Ni) and phosphorus. Most studies showed no effects on the adsorption capacity when salt was added to the test water, with the exception of peat.

B5.3 RQ 2.3 Which type of maintenance is needed for the different types of facilities?

As was pointed out in the reviewed studies, proper inspection and maintenance of stormwater treatment facilities is necessary to ensure that they continue to fulfil their function over time. Despite this, several studies reported that this is often neglected. Recommended maintenance activities mentioned for different technologies were regular inspection and removal of trash and large debris for wet ponds and constructed wetlands as well as biofilters. These types of facilities also need regular removal of accumulated sediments, particularly the forebays which are designed to promote pre-sedimentation. Furthermore, for wetlands, ponds, biofilters and grassed swales, care for the vegetation including mowing, pruning, removal, weeding and replanting is needed. Removal or replacement of soil or filter media should be done for grassed swales, biofilters and infiltration trenches. Permeable pavements need regular removal of accumulated particles to maintain their infiltration capacity and prevent them from clogging.

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Appendix C

Understanding co-benefits and overcoming barriers for increased societal acceptance of sustainable stormwater management

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C1. Summary

This report reviews the international peer-review literature relating to understanding co-benefits and overcoming barriers for increased societal acceptance of sustainable stormwater management. The research studies included in this report were identified using a systematic approach and consider who is involved in sustainable stormwater management, what co-benefits different stakeholder groups ascribe to blue-green solutions, as well as the aims, incentives, barriers and conflicts of sustainable stormwater management. While conventional stormwater management was originally only concerned with flood mitigation, sustainable stormwater management is performed to reach multiple aims and manage stormwater while providing a wide range of co-benefits to stakeholders, striving to create multifunctional blue-green solutions and spaces. This approach has increased in recent decades, and involves a wide range of solutions, and thus also actors. While few of the reviewed articles describe processes of stakeholder participation specifically in the development of blue-green solutions, stakeholder participation in the development of public urban green spaces in general is more widely studied. This calls for a shift in both practice and research towards viewing the implementation and management of blue-green solutions as part of the governance and management of urban green spaces, rather than as a separate issue. In this way, lessons learnt from studies on green space management in general (e.g. how to conduct participation processes and overcome barriers and challenges) can be applied to sustainable stormwater management.

Different stakeholders attach different values and benefits to blue-green solutions. These are important to understand in order to create socially inclusive blue-green solutions. While municipal managers emphasise general and holistic values such as which solution is best for the whole city, the environment or in the future, local residents tend to emphasise place-specific values related to the use and experience of the blue-green solution. What co-benefits that are created and valued also depend on the blue-green solution in question, with different blue-green solutions differing in their ability to provide specific co-benefits. A number of barriers and conflicts that hinder the successful implementation and management of blue-green solutions were identified. The most common barriers described were regulatory, organisational, knowledge and economic barriers. Other barriers mentioned relate to the physical environment, land ownership and social barriers. To overcome the identified barriers, there is a need for changed stormwater regulations that approach stormwater as a ‘resource’ rather than a ‘hazard’, and that are more adaptive and less reliant on quantitative outcomes to allow for more integrative, collaborative planning and decision-making. Above all, there is a need to improve communication and collaboration between urban stormwater stakeholders and to overcome departmental silos within local authorities for a more holistic approach to stormwater implementation and management. The shift towards more sustainable stormwater management requires visionary government leadership at local, regional and national levels, with authorities supporting the process as drivers, coordinators and capacity builders.

C2. Introduction

This report reviewed the peer-review literature to address the following research questions (RQs):

- RQ1. What **actors and stakeholders** are involved in sustainable stormwater management?
- RQ2. Which **co-benefits** are created through sustainable stormwater management **according to different actors and stakeholders**?
- RQ3. What are the underlying **aims and incentives** for sustainable stormwater management?
- RQ4. What **barriers and challenges** to sustainable stormwater management and what **conflicts** between its stakeholders exist? How can these be overcome and resolved for **increased societal acceptance** of sustainable stormwater management?

C3. Methodology

The review followed the PRISMA approach (Page et al., 2020), an established methodology to provide a transparent, complete and accurate account of how studies are identified and their characteristics reported within systematic reviews. The methodology involved two stages as follows:

- to facilitate alignment between the review activities (see also Appendix A and B), overarching keywords were identified and applied within a research database to identify a common 'longlist of papers'
- the longlist of papers was further interrogated/refined using keywords relevant to the focus of each review topic (see also Appendix B and C).

C3.1 Common approach

Keywords for the common approach were: stormwater or "storm water" or runoff. These keywords were selected from an inclusivity perspective i.e. to capture as many articles as possible within the targeted field. These pre-defined keywords were entered into SCOPUS, a research publications database which provides access to > 90 million research documents including outputs from > 29,750 peer-reviewed journals (SCOPUS, 2023). The initial search was undertaken in June 2023 and returned 133,504 hits. This initial longlist was then filtered using the keyword 'urban' leading to the identification of 42,124 papers, with this set of articles providing a common data pool for each review to work with (Table C1).

Table C1. Key words used in the common approach to align activities within Appendices A, B and C

	Key words	Number of hits
Article title, abstract, keywords	Stormwater or "storm water" or runoff	133,504
Search within results	Urban	42,124

C3.2 Methodology to identify articles to address the research questions

To identify relevant papers for this review, the 42,124 articles identified using the common approach were filtered using different terms describing stormwater systems with vegetation as keywords, for example 'blue-green'; 'water sensitive urban design' and 'green infrastructure'. This filtering yielded 12, 914 records, which were then filtered through different terms for stakeholders as the actor perspective is the core focus of this report. Further filters were applied to extract documents that were in English and in the form of peer-reviewed research or review articles. Results of applying search terms and filters are reported for each step in Table C2.

Table C2. Overview of applied search terms and filters

	Refining criteria and filters	Number of hits
Article title, abstract, key words	”blue-green” OR ”nature-based” OR sustainable OR ”low impact development” OR ”sustainable urban drainage system” OR ”water sensitive urban design” OR ”green infrastructure” OR ”Sponge City” OR treatment	12,914
Article title, abstract, key words	stakeholder* OR user* OR actor* OR governance	807
Language	Limit to English	788
Document type	Limit to peer-review research articles (548) and review articles (48)	597

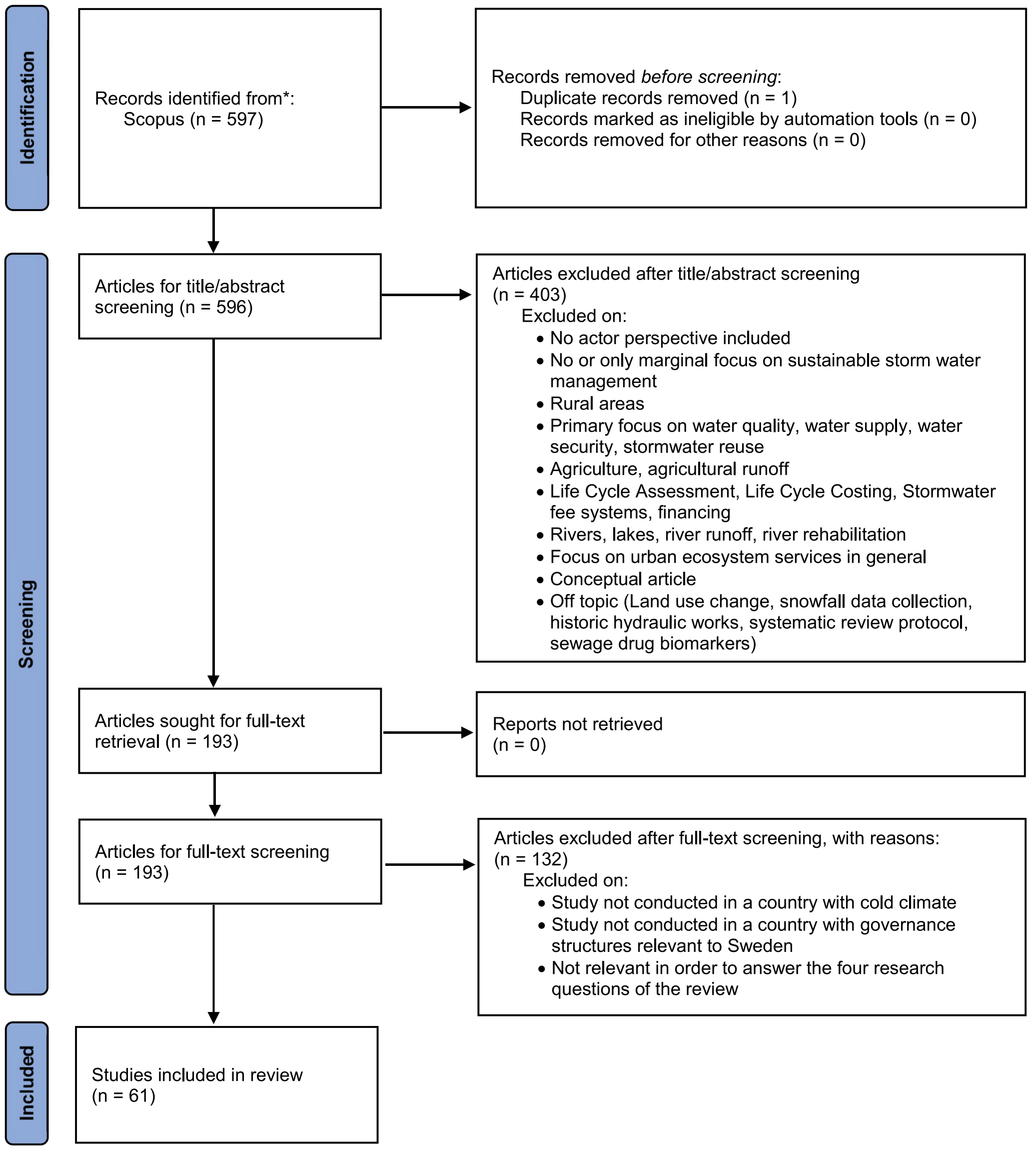

Figure C1. PRISMA 2020 flow diagram outlining the screening and filtering steps applied to the 597 papers extracted from SCOPUS.

Figure C1 provides an overview of how the 597 articles reported in Table C2 were managed in relation to the four core research questions including reasons for exclusion. During the full-text screening, a geographical scope was applied, only including studies conducted in countries (and American states) with cold climate and with governance structures relevant to Sweden. Globally, there are five major climatic domains: boreal, polar, temperate, subtropical and tropical (FAO and UNEP, 2020). For this report, cold climate was defined as boreal and temperate areas in the northern hemisphere. This slightly broader definition of cold climate than the one used in Appendices A and B was employed in this review due to its focus on governance aspects, which may be comparable within a larger geographical area. Thus, limiting the current review to cold climate studies as defined in Appendices A and B i.e. Canada, Scandinavia and the Baltic countries, would imply excluding several studies relevant to the topic of this report. The initial screening of articles involved their export from SCOPUS and uploading into the open-access systematic review software Rayyan (www.rayyan.ai) which enables research teams to review the same set of articles and – through a blind reviewer mode – provides an mechanism for quality assurance by allowing more than one researcher to independently review a shortlist of articles and compare the consistency of decision-making. In terms of this review, two reviewers independently screened 20 abstracts and reached the same inclusion/exclusion decision on all papers providing confidence that decisions on inclusion/exclusion were being made consistently within the team undertaking this activity.

C4. Results

C4.1 Context description: Geographical scope of included papers

The reviewed studies encompass a variety of 12 countries (Figure C2), with the majority of studies conducted in the USA (n = 28) (Figure C3). Other studies included in the review were conducted in Norway (n = 5), the UK (n = 5), Sweden (n = 4), Denmark (n = 4), or are global studies (n = 6). Overall results also indicate how the research area of sustainable stormwater management from a stakeholder perspective, as included in this review, has increased between 2004 and 2023, with a presumed peak in 2018 (Figure C4).

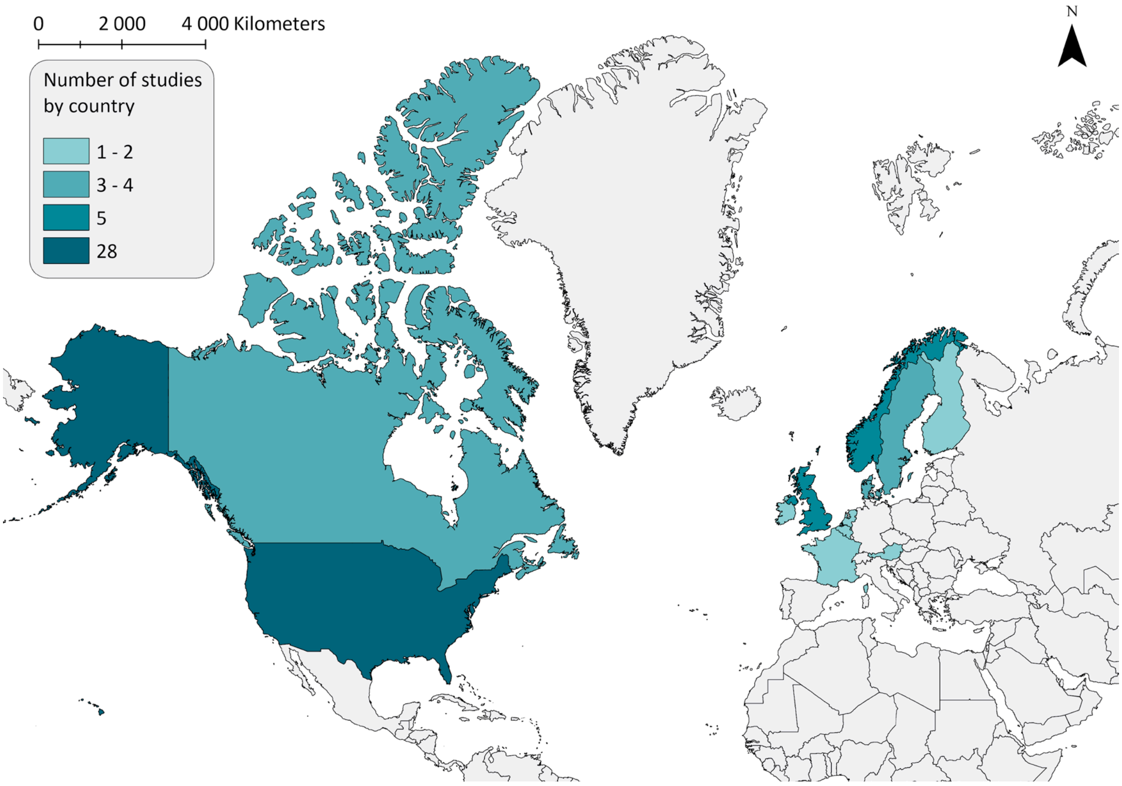

Figure C2. Location by country of the studies in the 61 reviewed articles.

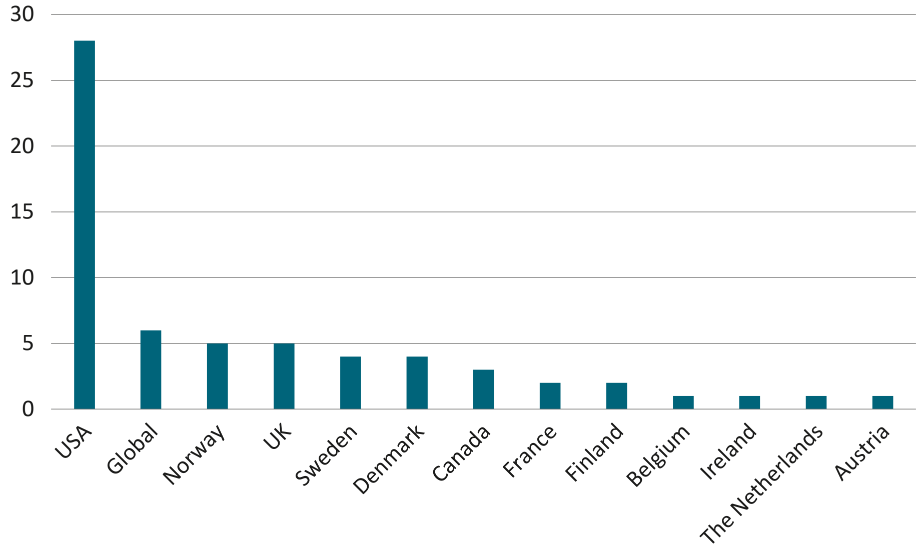

Figure C3. Number of reviewed studies by country. Since some of the included articles reported on studies conducted in several countries, the total number of studies (n=63) exceeds the number of reviewed articles (n=61).

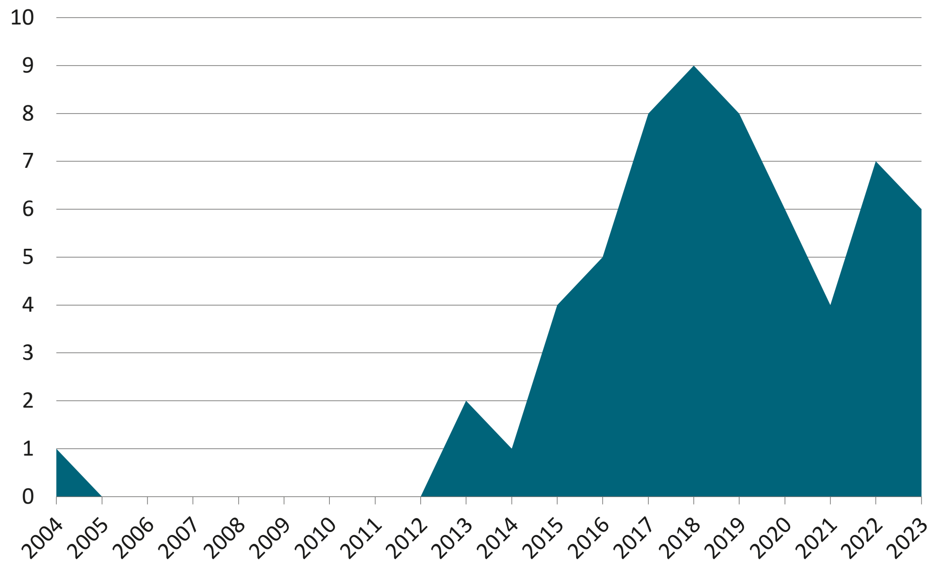

Figure C4. Timeline of reviewed studies based on year of publication.

Today, the conventional principles and grey solutions of stormwater management are increasingly being challenged. There is an emerging shift in stormwater management from viewing stormwater as a nuisance to viewing it as a resource, and from focusing solely on flood control to seeking multifunctional solutions that recognise the many potential co-benefits of sustainable stormwater management (Gimenez-Maranges et al., 2023). The various co-benefits of blue-green solutions beyond stormwater management are often used as a justification for adoption, but in reality, meeting volume or water quality targets drives actual investment in blue-green solutions and decisions about where to locate them (Turner et al., 2022). Holistic approaches remain challenging for professionals, with the result that a limited environmental engineering perspective seems to prevail in practice, preventing the implementation of alternative blue-green solutions (Gimenez-Maranges et al., 2023). There are a number of reasons for this, which will be discussed in this report. Synthesis findings are structured according to the four research questions, beginning by looking at who is involved in sustainable stormwater management (RQ1), followed by what co-benefits different stakeholder groups ascribe to blue-green solutions (RQ2), continuing with the aims and incentives (RQ3) and barriers and conflicts of sustainable stormwater management (RQ4).

C4.2 Sustainable stormwater management systems

Urban stormwater management has become increasingly complex in recent decades and, as a result, the associated terminology has become more diverse (Fletcher et al., 2015). Existing terms commonly used to describe sustainable stormwater management systems include Low Impact Development (LID), Green Infrastructure (GI), Best Management Practices (BMPs), Source Control, Water Sensitive Urban Design (WSUD), Integrated Urban Water Management (IUWM), Water Sensitive Cities, Sustainable Urban Drainage Systems (SUDS) or Sustainable Drainage Systems (SuDS), Alternative Techniques, Stormwater Control Measures (SCM) and Nature-based Solutions (NBS), with the preferred term varying between regions and countries (Fletcher et al., 2015). While it may seem desirable to have a universal terminology in relation to urban drainage, many of the terms used evolve locally and thus reflect locally shared and nuanced understandings of what principles are most appropriate to address place-specific problems within the local institutional context, and thus play an important role in establishing awareness and credibility of new approaches (Fletcher et al., 2015). This means that there are individual differences between these sustainable stormwater management systems in terms of focus and precise definitions. In a broad sense, the focus of the different terms ranges from those describing specific techniques to those describing overarching principles or a conceptual approach, and from those applied specifically to urban stormwater to those applied to the whole urban water cycle management (Fletcher et al., 2015).

The general trend in urban drainage shows a broadened focus from the 1960s, when urban stormwater was initially managed with the aim of flood mitigation, to today, when the aim is also to treat stormwater to protect the water quality of receiving waters and to integrate treatment into the urban landscape, while also providing a range of co-benefits (Fletcher et al., 2015). This alternative approach involves disconnecting some of the sealed surfaces from the conventional urban drainage system and instead managing stormwater runoff locally so that water is retained (permanently held back) through infiltration and evaporation, or detained (temporarily held back) for later discharge at a reduced flow rate to other drainage elements (Liu and Jensen, 2017). Some of these newer systems may include blue-green solutions such as green roofs, rain gardens and swales. While these have shown to reduce local flooding, economical loss and discomfort at storm events with medium or frequent return periods, it is important to note that these small-scale installations have little impact on the large-scale catastrophic rain events such as river flooding, seaside flooding or very intense cloud bursts that pose the greatest danger to urban infrastructure and communities (Emilsson & Ode Sang, 2017). In this report, we will use the term sustainable stormwater management when referring to entire process of planning, designing, constructing and managing urban stormwater runoff by using blue-green solutions.

C4.3 Blue-green solutions studied

Stormwater control measures describe the concrete physical solutions for sustainable stormwater management. They are divided into structural stormwater control measures, meaning engineered or built infrastructure such as bioretention systems and green roofs, and non-structural measures, meaning operational or procedural practices such as residential downspout disconnection programmes, maintenance procedures, minimising the use of chemical fertilisers and pesticides, or public education (Fletcher et al., 2015). Another term commonly used in Sweden to describe concrete physical solutions for sustainable stormwater management is blue-green solutions. Blue-green solutions implies combining green infrastructure with urban water management to create multifunctional solutions that provide co-benefits such as thermal climate regulation, increased biodiversity, reduced flood risk and places for social interaction and recreation (IVL, 2024). In this report we will use the term blue-green solutions when referring to physical solutions for sustainable stormwater management. In new urban developments, it may be possible to use them as stand-alone solutions, while when implemented through retrofitting in existing urban areas, they often need to be combined with conventional stormwater solutions (Liu and Jensen, 2017). Swales are an important part of stormwater run-off management in areas where space is limited, particularly roads and car parking areas (Pierre et al., 2019). Table C3 provides an overview of different types of blue-green solutions.

Table C3. Overview of blue-green solutions for sustainable stormwater management, based on Fletcher et al. (2015), Revitt et al. (2019), Scholes et al. (2008), Eisenberg et al. (2022), and authors' own notes.

Basins and wetlands	Detention basins	Dry most of the time and able to store rainwater during wet conditions; often possess a grassed surface
	Extended detention basin	Dry most of the time and able to store rainwater during wet conditions for up to 24 h; grassed surface and may have a low basal marsh
	Retention ponds	Contain some water at all times and retains incoming stormwater; frequently with vegetated margins
	Constructed wetlands • Sub-surface flow	Vegetated system with extended retention time: Sub-surface flow wetlands typically contain a gravel substrate, planted with reeds, through which the water flows.
	• Surface flow	Surface flow wetlands typically contain a soil substrate, planted with reeds, over which the water flows.
	Rain gardens	Dry most of the time and able to store rainwater during wet conditions. Often include wetland edge vegetation, such as wildflowers, sedges, rushes, ferns, shrubs and small trees.
Filter strips and swales	Filter strips	Grassed or vegetated strip of ground that stormwater flows across
	Swales	Vegetated broad shallow channels for transporting stormwater
Bio-engineered solutions	Green roofs	Divided into extensive and intensive green roofs
	Green walls	Includes both constructed living walls, and green facades with plants growing in the ground
Infiltration systems	Filter drains	Gravelled trench systems where stormwater can drain through the gravel to be collected in a pipe; unplanted but host to algal growth
	Soakaways	Underground chamber or rock-filled volume: stormwater soaks into the ground via the base and sides; unplanted but host to algal growth
	Infiltration trenches	A long thin soakaway; unplanted but host to algal growth
	Infiltration basins	Detains stormwater above ground which then soaks away into the ground through a vegetated or rock base
Porous surfaces/ permeable paving	Porous surface without sub-storage	A porous material e.g. porous asphalt but no sub-storage units (e.g. no stormwater crates)
	Porous surface with sub-storage	A porous material e.g. grasscrete with sub-storage units
Green spaces and trees	Public urban green spaces	Publically accessible areas within cities, with natural or cultivated vegetation, that are deliberately managed to provide specific environmental, social, and economic benefits.
	Green corridors	Connections between multiple urban green spaces (public and private)
	Urban private gardens	Usually private residential gardens, but may also include outdoor spaces around private multi-family housing.
	Street trees	Individual trees along streets
	Groups of trees	Groups of trees growing in public urban green spaces and urban woodlands.

C4.4 (RQ1) Responsibilities and participation in sustainable stormwater management – Who is involved and how?

Development of public urban green spaces, including blue-green solutions, is commonly divided into the four phases of planning, design, construction and management (CE, 2000; Rodiek, 2006; van den Brink et al., 2016; Jansson et al., 2020). This process of landscape development typically begins with a plan set by authorities at national, regional and local levels, followed by the production of more detailed designs at various scales, which are then implemented through construction and planting (Jansson et al., 2020). Long-term management is the final phase and involves both operational maintenance and further enhancement or development of user qualities in a long-term perspective (Jansson & Lindgren, 2012; Dempsey & Smith, 2014). The long-term management is time wise the longest of the phases and involves replanning, redesign, reconstruction and continued maintenance in response to emerging societal challenges and new demands from citizens and users (Randrup & Persson, 2009).

Typical actors involved in sustainable stormwater management include planners, managers and civil engineers from municipal departments (e.g. Water and Sewerage; Urban Environment, Parks and Recreation; and Planning and Building), as well as private consultants with expertise in construction; planning; engineering; green space maintenance; and water and sewerage (Kvamsås, 2023). Many sustainable stormwater management initiatives are still carried out through a top-down and expert-driven process for site selection and design, despite acknowledgement of the importance of public engagement (Campbell-Arvai & Lindquist, 2021). This tends to lead to suboptimal outcomes in terms of sustaining these projects over the long-term, achieving multifunctional and inclusive spaces and seizing opportunities to increase public participation in urban planning (Campbell-Arvai & Lindquist, 2021). At the same time, there are signs of an emerging paradigm shift in stormwater management, moving away from traditional top-down approaches towards the implementation of additional decentralised blue-green solutions and the practice of involving local residents to also manage stormwater from their private properties (Wilfong et al., 2023).

In the following section, we identify the actors and stakeholders involved in sustainable stormwater management and look specifically at cases where a wider range of stakeholders are involved in the development of blue-green solutions, e.g. local residents or researchers. Here, we describe participatory processes relating to development of blue-green solutions found in the reviewed articles as well as involvement approaches employed (e.g. public workshops or surveys). Table C4 provides an overview of all cases of participatory sustainable stormwater management described in the literature reviewed. The section is organised according to the phase in which participation took place (i.e. planning, design, construction and/or long-term management including operational maintenance), but in several of the cases reviewed, participation took place in more than one phase.

Table C4. Overview of all cases of participatory sustainable stormwater management described in the literature reviewed. Yellow dots indicate the stakeholders involved in the participatory processes related to the development of blue-green solutions found in the reviewed articles, and blue dots indicate the phase in which the participation took place (i.e. planning, design, construction and/or long-term management including operational maintenance).

Reference	Country	Municipal departments	Regional agencies	The public water utility company	Water consultants/ stormwater engineers	Architecture offices	Professional mediator	The public/ local residents	Children	Neighbourhood associations; Community advisory councils	NGOs and non-profit companies	Local businesses	Researchers	Planning	Design	Construction	Long-term management
Ellis et al., 2004	France	•			•			•				•		•			
Fryd et al., 2013	Denmark	•											•	•			
Shandas, 2015	USA (Oregon)	•						•									•
Geaves and Penning-Rowsell, 2016; 2015	UK	•									•						•
Dhakal & Chevalier, 2016	USA (Washington)	•						•	•	•				•			•
Dhakal & Chevalier, 2016	USA (Oregon)	•			•			•	•	•		•		•			•
Dhakal & Chevalier, 2016	USA (Pennsylvania)	•						•						•			•
Dhakal & Chevalier, 2016	USA (Illinois)	•									•			•			•
Vierikko & Niemelä, 2016	Finland	•			•		•	•						•	•		
Fitzgerald & Laufer, 2017	USA (Pennsylvania)	•			•			•			•			•		•	•
Madsen et al., 2017	Denmark	•		•						•						•	
Dobre et al., 2018	Belgium	•	•			•		•		•	•		•	•	•	•	•
Meenar, 2019	USA (Pennsylvania)		•					•			•		•		•		
Pierre et al. 2019	Canada	•			•			•							•		
Laatikainen et al., 2020	Finland							•					•				•
Campbell-Arvai & Lindquist, 2021	USA (Michigan)							•			•		•	•			

C4.4.1 Participation in planning

The municipalities are usually the project owners, and as such tend to steer the planning processes, sometimes involving researchers in the early planning stages. In Denmark, researchers participated in the development of a flood mitigation strategy for Copenhagen (Fryd et al., 2013). Participants can also be involved in an integrated process of both planning and design of concrete blue-green solutions. This was the case in Detroit, USA, where researchers worked with two local non-governmental organisations (NGOs) to run workshops with local residents who were also members of the two NGOs. A series of five workshops were held with two groups of 11 and nine participants respectively. The purpose of the workshops was to (i) identify and clarify goals related to green infrastructure installations, (ii) provide verbal and written feedback on a specific 3D design software as it was being developed, (iii) create GI designs for vacant and abandoned properties in the local neighbourhoods, and (iv) evaluate the software (Campbell-Arvai & Lindquist, 2021). Similarly, local residents were also involved in Helsinki, Finland, where consultants from an environmental consultant company and managers from the municipal public works department collaborated on a stormwater management plan (Vierikko & Niemelä, 2016). The local residents were involved through a participatory planning process that included two public workshops, led by a professional mediator, where participants worked on maps, planning together with the consultants and managers where to locate which blue-green solutions (Vierikko & Niemelä, 2016). This participatory planning process for a stormwater management plan can be described as the public participating in the planning and design phase. In France, local residents were involved in the selection of a construction site for a county retention basin (Ellis et al., 2004). First, county services engineers conducted a multi-criteria analysis, on the basis of which they presented the strengths and weaknesses of each site to the elected county and municipal officials. The officials then presented and explained the results of the analysis to local residents and local business representatives. This was followed by a public debate, which revealed that local visual and operational impacts were the most important issues for local residents, and therefore one of the sites was eliminated, despite its high hydraulic performance (Ellis et al., 2004).

C4.4.2 Participation in design

In Hamilton, Canada, municipal landscape architects arranged a pop-up park event to gather stakeholder input on a streetscape design that proposed a system of swales to replace existing grey infrastructure (Pierre et al., 2019). Passers-by were invited to chat about the street and write suggestions for creating a more welcoming environment through the use of green infrastructure. The landscape architects then used 3D design software to incorporate stakeholder suggestions into a design, and then gathered additional stakeholder feedback on the final design, which was publicly displayed via ArcGIS and shared on social media, resulting in over 400 views. The design generated through the software still required further input, alteration and refinement by city planners, stormwater engineers and other design professionals (Pierre et al., 2019).

In the city of Philadelphia, USA, local residents of a neighbourhood with residents of Latin American cultural or ethnic identity participated in the co-design of two vacant lots in the area (Meenar, 2019). The design charrette, led by the local community development corporation, Asociación Puertorriqueños en Marcha for Everyone, also involved professional experts representing government agencies, Philadelphia-wide environmental non-profits, and researchers. The resulting site designs aimed to both manage stormwater and provide social co-benefits for local residents. Meenar (2019) argues that integrating placemaking and co-benefit provision into the design of blue-green solutions, rather than focusing solely on stormwater management, can make the design more attractive to marginalised communities living in disadvantaged neighbourhoods.

C4.4.3 Participation in construction

As part of a climate adaptation effort in the city of Copenhagen, Denmark, private residential associations, the municipality and the utility company all have implemented LAR (Lokal Anvendelse af Regnvand), seeing it as part of the solution to dealing with cloudbursts (Madsen et al., 2017). LAR is the Danish term for Local Use of Rainwater or Local Rainwater Drainage, which is defined as any initiative that controls rainwater and stormwater locally, thereby reducing the amount of water that goes into the piped sewerage system (Madsen et al., 2017).

The implementation of the ambitious stormwater management plan of Philadelphia, USA, called Green City-Clean Waters, was a collaboration between three city departments (Water; Parks and Recreation; Streets), engineering consultants (who participated in the early stages of plan development), residents (who, for example, installed rain barrels and downspout planters to manage water on their property), commercial property owners, and local organisations (Fitzgerald & Laufer, 2017). One of these organisations was a horticultural society that was tasked with managing the Rain Check programme for local residents, including conducting stormwater audits on private properties and organising regular workshops for residents to learn about the role their properties can play in implementing the city's stormwater management plan (Fitzgerald & Laufer, 2017). One of the incentive programmes implemented by the water department required commercial property owners to agree to a 45-year operation and maintenance agreement to ensure that facilities are maintained to meet regulatory requirements (Fitzgerald & Laufer, 2017).

C4.4.4 Participation in long-term management, including operational maintenance

Long-term environmental monitoring data on the performance of blue-green solutions is important to identify changes over time as a basis for strategic management decisions. Local residents are an untapped resource for this. In a pilot study conducted by researchers in Kajaani, Finland, local residents were involved in water quality monitoring, collecting data on sources of pollution and levels of contamination in runoff (Laatikainen et al., 2020). The five participating households first attended a practical training session and then used visual observation, reagent strips and a smartphone application for data transfer to observe organoleptic (appearance, odour, colour) and chemical parameters (pH, nutrient content) at selected sites near a lake twice a week for eight weeks (Laatikainen et al., 2019). From the study it was concluded that stakeholder participation in environmental monitoring provides valuable additional data to complement monitoring by environmental authorities and industry representatives. Authors also concluded that stakeholders can potentially carry out monitoring on a more regular basis than can be afforded through monitoring by contracted experts, thus providing more frequent data and allowing monitoring of larger and sometimes environmentally sensitive areas at relatively low cost, while increasing the information available to the general public (Laatikainen et al., 2020; 2019). After the pilot study, the local authorities decided to use citizen-based environmental monitoring to prove the efficiency of a newly constructed biological filtration system for nutrient removal (Laatikainen et al., 2020).

Local residents could also be a valuable resource in the operational maintenance of blue-green solutions. In the city of Portland, Oregon, USA, residents participate in maintaining blue-green solutions by adopting a green street planter (City of Portland, 2024). These volunteers, called 'Green Street Stewards', supplement regular municipal maintenance. Their main role is to check planters before, during and after storms and remove sediment, leaves and debris around curb openings and drains to allow water to flow, but they also regularly pick up litter and water plants during dry summer weather (City of Portland, 2024). Online guides with maintenance instructions are provided to participants. In studying the Portland case, Shandas (2015) sent out surveys to local residents to find out why people chose to participate in stormwater management in a public space. Factors that were found to explain residents' willingness to maintain public stormwater facilities included living in the neighbourhood for less time, previous experience of participating in environmental projects, higher levels of education, rating the quality of their neighbourhood as high, having high quality neighbourhood associations, and perceiving that it is possible for local people to make a positive impact on their neighbourhood (Shandas, 2015). Survey respondents who felt that there were too few quality green spaces in their neighbourhood were more likely to be involved in public stormwater management, possibly seeing new stormwater facilities as a positive addition to greening the neighbourhood (Shandas, 2015).

Another example of when local residents participate in the maintenance comes from the UK, where numerous 'flood action groups' have formed over the last few decades in response to major flood events and associated changes in legislation that have devolved responsibility from central government to local authorities (Geaves and Penning-Rowsell, 2016). The groups are often quite small, consisting of a handful of self-selected, community-supported activists who contribute to flood risk reduction through fundraising, hands-on river maintenance including producing leaflets with key contacts and advice in the event of a flood, and lobbying to persuade authorities to invest in major flood risk reduction measures (Geaves and Penning-Rowsell, 2016; 2015).

The cities Portland, Oregon; Seattle, Washington; Philadelphia, Pennsylvania; Chicago, Illinois are among the (early) leader cities in the US for implementing blue-green solutions (Dhakal & Chevalier, 2016). They have worked with public participation in stormwater management in different ways. In Seattle, Washington, Seattle Public Utilities involves the public through community advisory councils and green infrastructure partnerships, runs educational programmes for school children aged 5–18, and organises annual watershed forums and public tours for residents. The public can also participate in cleaning up stormwater facilities through programmes such as 'Adopt-a-Street' and 'Adopt-a-Drain'. In Portland, Oregon, the Bureau of Environmental Services runs educational programmes for school children aged 5–18, community stewardship programmes, and collaborates with neighbourhood associations and riparian property owners. The city has also established a committee of stakeholder representatives, including community, professional and business groups, who provide input into the development review process. In Philadelphia, Pennsylvania, the Philadelphia Water Department engages stakeholders through outreach, education and clean-up programmes. In addition, local residents provide input into goal setting and help the city identify vacant land suitable for implementing blue-green solutions. In Chicago, Illinois, the Department of Water Management involves community NGOs including the Metropolitan Planning Council, Chicago Wilderness, Space to Grow, and the Center for Neighborhood Technology (Dhakal & Chevalier, 2016).

The transition to more sustainable stormwater management in a municipality in Brussels, Belgium, involved a wide range of actors (Dobre et al., 2018). The regional government and the regional water agencies, which are responsible for the management and maintenance of the combined sewer systems, worked together with the municipal water department. In addition, the local authorities formed a local working group to exchange knowledge between experts from the local authorities and representatives of the public water agencies. Neighbourhood committees worked to raise awareness of water-related problems. Two local universities conducted research on the causes of flooding in Brussels, the potential of technical solutions and the interaction between actors. In addition, an NGO involved citizens in water-related issues and, in cooperation with the local authorities and the neighbourhood committees, organised discussions of water-related issues between state and non-state actors and developed the concept of 'catchment solidarity' to improve cooperation between actors in the same urban catchment area. Further, they recreated urban streams by using small devices to collect, infiltrate and drain stormwater on the surface, and developed a project to integrate vegetated swales and rain gardens at street level. The NGO, neighbourhood committees, architecture offices and researchers together organised Map-it sessions and guided tours to gather information and design scenarios on water-related issues. The Brussels Environmental Agency initiated the dredging of urban streams to increase water flow (Dobre et al., 2018).

C4.4.5 Local residents: an untapped resource in management of blue-green solutions

In most of the cases of participatory sustainable stormwater management identified in this study, the processes of developing and implementing blue-green solutions are rather hierarchical and top-down, driven by local authorities. The fact that local authorities lead the development of blue-green solutions is in line with what has long been described in the literature on public space planning and management (see e.g. de Magalhães & Carmona, 2009; Jansson & Randrup, 2020).

Only 14 out of the 61 reviewed articles described participatory processes related to development of blue-green solutions, despite that increased user involvement is one of the proposed solutions to barriers and conflicts within sustainable stormwater management (see *4.7 (RQ 4) Barriers and challenges to sustainable stormwater management*). Further, in the cases where participation processes occurred, non-public actors such as local residents and researchers mainly participated in the planning phase, where the municipality informed them about a future stormwater management plan or consulted them in order to obtain feedback on the plan and ideas on alternatives. Information and consultation are types of participation that imply that the municipality retains most of the power, while participants have limited influence on the end result (Ambrose-Oji et al., 2011). The few identified cases of stakeholder participation in the construction and management phases come from the USA, UK, Belgium, Finland and Denmark. The lack of Swedish studies in this area suggests that local residents are an untapped resource for the long-term management of existing blue-green solutions in Sweden. However, looking beyond stormwater management alone, stakeholder participation in green space management in general has been widely described in the literature (see e.g. Fors et al., 2021 and Mattijssen et al., 2017). Randrup et al. (2021) found that municipal planners and managers were very supportive of involving local residents, including in the long-term management of green spaces, but that they generally found it time and resource consuming. The existence of scientific studies on user participation in green space development in the scientific literature in general means that it cannot be concluded that stakeholder participation in the development of blue-green solutions is uncommon, simply because we did not identify such studies in this review, which focused specifically on stormwater management.

C4.5 (RQ2) Co-benefits from blue-green solutions as perceived by different stakeholders

In this section, we focus on which co-benefits that are created through sustainable stormwater management according to different actors and stakeholders. The main function of structural stormwater control measures is to handle urban stormwater through purification, infiltration, retention, conveyance and detention. However, when designed as blue-green solutions, they can have several co-benefits, including water savings, energy savings, improvement of air quality, carbon sequestration, biodiversity protection, nature conservation, recreational opportunities and public health benefits (Kvamsås, 2023). These are often referred to as ecosystem services, which are typically categorised as supporting, provisioning, regulating and cultural ecosystem services (MEA, 2005). In particular, blue-green solutions support biodiversity and provide cultural ecosystem services, as explained by Vierikko & Niemelä, (2016) and Fryd et al. (2013), who mention that flood systems consisting of ponds, streams and lakes provide habitats for species and are valued by the public as places for recreation, restoration, relaxation, enjoyment and education. Vegetation in blue-green solutions also provides regulating ecosystem services. For example, bioretention cells and wetlands contribute to local thermal regulation, trees in bioretention cells reduce air pollutants and process carbon dioxide, stormwater wetlands sequester carbon (Cizek & Fox, 2015), and soil media and plants in swales can remove pollutants from road runoff (Pierre et al., 2019).

However, blue-green solutions can also cause ecosystem disservices, i.e. negative effects or nuisances, such as pests and diseases, damage to infrastructure, allergic reactions, health problems caused by poisonous plant species, and obstructed views due to vegetation, which can cause traffic hazards (Lyytimäki and Sipilä, 2009). Examples of such disservices include the perceived disamenity caused by an occasionally flooded urban green space (Fryd et al., 2013).

Table C5. Overview of reviewed articles reporting on co-benefits ascribed to blue-green solutions by different stakeholders.

Reference	Local residents/ citizens/ the public	Children	NGOs	Municipal planners and managers	Regional authorities	Politicians	Water engineers and industry professionals	Researchers	Champions	Traditional technocrats
Ellis et al., 2004	•			•	•		•			
Cizek & Fox, 2015		•								
Vierikko & Niemelä, 2016	•			•		•				
Madsen et al., 2017	•			•			•	•	•	
Heckert & Rosan, 2018	•			•						
Pierre et al., 2019	•									
Miller and Montalto, 2019	•			•						
Elliott et al., 2020								•		
Campbell-Arvai & Lindquist, 2021	•									
Buffam et al., 2022			•	•	•		•	•		
Turner et al., 2022									•	•
Gimenez-Maranges et al., 2023	•			•			•			
Thodesen et al., 2023	•			•						

While the general concepts of ecosystem services and disservices are now well established in research and practice, less is known about how the values and key co-benefits attributed to blue-green solutions differ between different stakeholders and actors involved in sustainable stormwater management. This plurality of values is important to consider, as mapping the different values that different stakeholders ascribe to blue-green solutions can facilitate the identification of mutual values and the understanding of disagreements between stakeholders (Vierikko & Niemelä, 2016). Furthermore, using the concept of ecosystem services in the planning and management of urban green spaces may fail to recognise non-monetary or intangible values (e.g. Vierikko & Niemelä, 2016). Therefore, in this section we will map the plurality of values attributed to blue-green solutions by different stakeholders by including all types and definitions of co-benefits and values described in the reviewed articles. In Table C5, we list the stakeholder groups studied in the reviewed articles in terms of which co-benefits these groups ascribe to blue-green solutions. In the following text we describe, discuss and extract the many, diverse and overlapping values mentioned in the literature.

C4.5.1 Place-specific user values vs. general and holistic values

There are large differences between different stakeholder groups in terms of which types of values they may ascribe to blue-green solutions. In a study in Finland, it was explored which values and meanings local residents, managers (from the municipal Public Works Department and stormwater management consultants) and politicians ascribed to a public urban green space and an urban brook (Vierikko & Niemelä, 2016). Vierikko & Niemelä (2016) employed an integrative value mapping and found that the involved stakeholders expressed 47 perceived values in total, which were divided into four types: (1) *use and experience values* (non-consumptive direct use; psychological, aesthetic or other direct response to the environment), (2) *existence values* (knowledge of the existence of a specific ecosystem aspect or other element), (3) *symbolic values* (abstract meanings assigned to the area), and (4) *bequest and moral values* (values relating to the idea of maintaining the environment for the benefit of other people or for environmental reasons). As green space users with strong place identity and attachment to the place, the local residents assigned more *use and experience values* (e.g. enjoying nature and beautiful trees; learning from nature; outdoor activities; picnicking; social relation) and *symbolic values* (e.g. spiritual value of the brook; cultural heritage; the last remnant of natural brook) than did other stakeholder groups. In addition, local residents ascribed more *existence values* to specific parts of the places studied, such as wild animals or habitats for wildlife, than did managers. Instead, politicians and managers ascribed the same *bequest and moral values*, such as protecting the environment and having an entire city perspective, but unlike the values ascribed by local residents, the moral values of politicians and managers were universal rather than locally ascribed to the urban brook or public urban green space. Negative values shared by local residents, managers and politicians concerned the water condition such as brown colour and sudden foaming, and flooding. While managers expressed more abstract negative values such as ‘waste land’ or ‘human-destroyed’, local residents expressed negative values connected to environmental concerns and serious concerns about the effects of foaming on human health. Fewer negative values were ascribed to the area by the politicians than by the local residents or by the managers (Vierikko & Niemelä, 2016).

This difference between local residents emphasising place-specific values related to the use and experience of the blue-green solution, and municipal managers emphasising general values such as which solution is best for the whole city, the environment or in the future, has also been noted in other studies. In a study of the restoration of an urban stream in Trondheim, Norway, diverging priorities and perspectives between different stakeholders were identified (Thodesen et al., 2023). The most prominent difference was that the municipal managers responsible for implementation had a strong long-term focus and made efforts to plan for the future, while local residents stated that things were fine as they were and saw no need to change things in their local area. The municipality's concerns for the studied stream restoration were to ensure long-term macro-planning for climate change adaptation and environmental sustainability, while local residents were more interested in children's safety around the water (Thodesen et al., 2023). Research from Miller and Montalto (2019) confirms this difference between local residents and municipal managers. They found that while stormwater management is the primary municipal driver for investment in sustainable stormwater management systems in New York City, local residents value co-benefits more highly than actual stormwater management. Similar place-specific, user-oriented values of blue-green solutions were mentioned by local residents in Michigan, USA. To guide the development of a 3D design software for user participation in green infrastructure design, researchers asked these local residents, representing local NGOs, about their desired objectives for the blue-green solutions in their neighbourhood. The local residents highlighted (in order of importance) a desire for reduced effort and costs for residents, increased quality of life, educational opportunities, sustaining the green infrastructure over the long-term, improvements in health, improvements in the 'look' of the neighbourhood, increase in community vitality, enticing youth to stay in the community, and improvements in safety (Campbell-Arvai & Lindquist, 2021).

Another example is the study of members of the public who provided feedback on a proposed streetscape swale design using 3D design software, who mainly commented on the aesthetics, appreciating the increase in green space and additional seating along the street (Pierre et al., 2019). They also raised concerns about the health of the plant material in the swales during dry periods and the reduced capacity of the car parks. In the planning process of a retention basin in France, county service engineers were most interested in new construction works, real time control, the quality of receiving waters, and the overall technical performance and coherence of the project, while county and municipal elected officials mainly focused on achieving effective drainage and community environmental benefits, and local residents were concerned with local surface water flooding and neighbourhood environmental protection (Ellis et al., 2004).

The Philadelphia Water Department also identified bequest values, stating that blue-green solutions provide economic benefits through job creation, co-benefits through facilitating recreation and improving public health, and environmental benefits through improved air quality and carbon sequestration (Heckert & Rosan, 2018). From the same study, it was also concluded that blue-green solutions may be perceived differently by local residents from different socio-economic and cultural groups (Heckert & Rosan, 2018). Members of a community advisory board from different parts of the city of Philadelphia were found to generally agree on the overall potential socio-economic and community benefits of blue-green solutions, but disagreed on the level of importance of these benefits, the extent to which specific solutions offered them benefits, and which solutions would be best combined in their neighbourhood to e.g. create community amenities and improve aesthetics (Heckert & Rosan, 2018). This variation between different members of the public highlights the importance of engaging different stakeholders groups in stormwater management, also traditionally marginalised ones, to tailor solutions to their varied and specific needs.

In Linz, Austria and Toulouse, France, a survey was sent to local residents and professionals involved in stormwater management (including responses from professionals at a municipal planning department; the construction sector, and a landscape architecture office). It showed that both residents and professionals generally considered stormwater to be a very valuable part of the urban ecosystem and that stormwater management should: 1) preferably be handled on site where stormwater is generated; 2) be adapted to the biophysical characteristics of each site; and 3) be based on a close link with natural processes. There were also some differences between the two groups of respondents with regard to the function of stormwater, where local residents were positive towards the use of stormwater harvesting for watering, while professionals instead considered ecological functions related to stormwater quantity and quality to be most important, putting aside the social functions of stormwater, as they perceived a number of barriers to stormwater harvesting. In addition, professionals preferred a less "wild-looking" nature for stormwater management than local residents (Gimenez-Maranges et al., 2023). This is an interesting finding, as contemporary literature in general describes the transition towards a wilder and more biodiverse urban environment, primarily driven by austerity and a general concern for biodiversity loss (Randrup et al., 2021; Oke et al., 2021).

C4.5.2 Co-benefits for children

Children are a user group with specific needs for blue-green solutions to be safe for them, but they, like other local residents, gain place-specific values from these solutions. Bioretention cells, rain gardens, stormwater wetlands, and rainwater harvesting are often small, cost-effective landscape elements that can be integrated throughout a site (Cizek & Fox, 2015). As children have less access to nature, and the places they frequent often have impervious surfaces that may be suitable for blue-green solutions, incorporating these solutions into children's outdoor play spaces can be a way to manage stormwater locally while providing opportunities for children to interact with nature (Cizek & Fox, 2015). According to Cizek & Fox (2015), bioretention cells and rain gardens benefit children by offering education and learning through seasonal changes; opportunities for unstructured exploration; diverse sensory stimulation; and familiarisation with outdoor environments. Further, wetlands benefit children by fostering positive and cooperative relationships with others, promoting inclusion, promoting creative play; creating multi-generational settings; and fostering environmental awareness. Rainwater harvesting benefit children by offering education, demonstrating important concepts such as the water cycle; hands on water play; providing a sense of local, environmental context;

fostering positive and cooperative relationships with others; promoting inclusion; promoting creative play; and fostering environmental awareness. Negative aspects include the danger of drowning in permanently ponded water in wetlands, and the risk of children eating the mulch in the bioretention cell and being accidentally exposed to metal toxins if the runoff comes from roads with heavy traffic (Cizek & Fox, 2015).

C4.5.3 Co-benefits according to sustainable stormwater management champions

What values that are ascribed to blue-green solutions also depend on whether the person is a sustainable stormwater management champion (i.e. driving and promoting implementation) or not. Madsen et al. (2017) analysed the development of LAR (the Danish term for Local Use of Rainwater or Local Rainwater Drainage), over time in Copenhagen, Denmark, through the lens of a framework for technological stabilisation. As a part of their study, they conducted eight interviews with public managers, industry professionals, researchers and citizens, out of which three interviewees were considered LAR frontrunners or champions. Rather than specifying which meanings were ascribed to LAR by which stakeholder group, Madsen et al. (2017) distinguish between meanings ascribed by LAR frontrunners and by all interviewees. Further, they describe how the main meanings ascribed to LAR has varied over time during the technological stabilisation process. Meanings attributed to LAR by LAR frontrunners included *Environmental Protection* (arguing that LAR systems provide habitats for flora and fauna); *Aesthetics* and *Community* (arguing that LAR provides better and more beautiful urban, natural and green expressions and better opportunities for community activities than conventional underground solutions); and *Groundwater Recharge* (arguing for further infiltration of rainwater to recharge groundwater aquifers to benefit the groundwater resource). Other meanings mentioned by interviewees in general were *Groundwater Recharge* and improved *Water Quality* (arguing that LAR reduces the amount of water entering the combined sewage system, thereby lowering the numbers of overflows to recipients); *Water Harvesting* (arguing for supplementing use of groundwater with stormwater); *Health* (arguing that the vegetation improves air quality and urban climate); and *Economic Efficiency* (arguing that the construction costs are lower for LAR than for traditional sewer systems). The interviewees also mentioned two negative meanings: *Economic Inefficiency* (arguing against economic efficiency, stating that maintenance costs are higher for LAR than for traditional sewer systems, seeing sewer pipes as an effective and cheap solution); and *Groundwater Threat* (expressing worries about the quality of the stormwater infiltrating into the soil and further pollution from percolation through polluted soil, stating that LAR does not provide any treatment, resulting in that the water bypasses directly to the secondary groundwater table). Around 2007–08 the water utilities were corporatised in Denmark, which sparked a new interest in the meanings of *Economic Efficiency* and *Integrated* (arguing that LAR is an economically efficient holistic water management method) and the meaning *Cloudburst Flooding* (arguing that LAR can help mitigate large and rare cloudburst events) and the directly opposite meaning *Nuisance Flooding* (arguing that LAR can only manage small and frequent events that lead to “nuisance floods”) (Madsen et al., 2017).

A survey of stormwater professionals in Cleveland, Ohio and Denver, Colorado found that the environmental value orientation of stormwater managers influenced their co-benefit priorities (Turner et al., 2022). In line with the findings of Madsen et al. (2017), they found differences between so-called *champions* (government and non-profit staff who are strongly committed to green infrastructure and help drive green infrastructure adoption at the city level) and *traditional technocrats* (in contrast to champions, who do not drive innovation in an organisation) (Turner et al., 2022). Champions were more likely to support GI and prioritise the environmental co-benefits of stormwater control measures, such as habitat and reduced air temperature, having concerns about stormwater runoff degrading habitats. Technocrats, on the other hand, supported grey infrastructure and prioritised human co-benefits such as property values, having concerns about runoff damage to personal property, that stormwater management challenges will result in higher taxes and altruistic concerns about damage to others' property (Turner et al., 2022).

C4.5.4 Provided values depend on the specific blue-green solution

Different blue-green solutions differ in their ability to provide specific co-benefits. In a study by Elliott et al. (2020), academic experts from a wide range of disciplines were asked to identify which ecosystem services are provided by which type of blue-green solutions within the context of New York City. According to the experts, public urban green spaces, wetlands and community gardens were the three most beneficial types in terms of the number of ecosystem services provided. Conversely, the lowest ranked types were rain barrels, permeable paving and vacant land. This variation means that it is important to differentiate between different types of blue-green solutions in planning frameworks that aim to deliver co-benefits from blue-green solutions in addition to stormwater management. While experts identified cultural ecosystem services as the most important co-benefits of blue-green solutions, these benefits are under-represented in planning, evaluation and research (Elliott et al., 2020).

Also in terms of economic feasibility, the performance of different blue-green solutions varies. Disconnection of impermeable areas and implementation of basins were found to be the most cost-effective solutions in a case study of a catchment in London, UK (Ossa-Moreno et al., 2017). In addition, roof disconnection and infiltration strips were found to be the most cost-effective ways to promote blue-green solutions and reduce flood risk to vulnerable infrastructure assets such as police stations and schools (Ossa-Moreno et al., 2017).

C4.5.5 Delivering co-benefits where they are most needed

It is also important to consider how to provide desired benefits and co-benefits *in the right place*. Heckert & Rosan (2016) developed an equity index to be used to identify communities that lack amenities that blue-green solutions could help promote, and to target investments in blue-green solutions to neighbourhoods with the greatest need. The index was based on variables related to disadvantage and vulnerability, such as the percentage of people who are minority, low-income, and over the age of 64, as well as environmental factors related to exposure to environmental risks, such as proximity to traffic, ozone levels, and particulate matter, and access to environmental amenities, such as access to green spaces and tree canopy cover. Heckert & Rosan (2016) emphasise that the equity index should be used by someone with local knowledge to interpret the results correctly. The objective of the implementation, i.e. the needs of the area in question, determines the choice of the blue-green solution. For example, in a neighbourhood with underperforming schools, the focus should be on greening schoolyards, and where unemployment rates are high, the creation of green jobs related to stormwater management could be prioritised (Heckert & Rosan, 2016).

C4.5.6 Regional differences in co-benefits

There may be regional differences in stakeholder values and preferred blue-green solutions. Buffam et al. (2022) studied green infrastructure in Addis Ababa (Ethiopia), Cincinnati (USA) and Malmö (Sweden) and found large similarities in ecosystem service priorities between cities, but also significant differences (e.g. to prioritise production of local produce in Ethiopia). They concluded that certain key ecosystem service priorities may apply broadly to cities regardless of socio-cultural context (Buffam et al., 2022). However, regional differences in preferred blue-green solutions between two American cities have also been identified, which may be due to different policy contexts (Turner et al., 2022), rather than overall climate and socio-economic differences.

C4.5.7 Co-benefits play a role in economic viability

Based on their assessment of the economic benefits of blue-green solutions, Ossa-Moreno et al. (2017) found that the economic feasibility of implementing blue-green solutions improved significantly when considering not only flood reduction but also co-benefits, as all net present values increased when the latter were included. The co-benefits included in the study were air quality, biodiversity and ecology, groundwater recharge, rainwater harvesting, wastewater treatment and reduction of surface water charges (Ossa-Moreno et al., 2017).

C4.5.8 Several factors affect which co-benefits that are obtained

Table C6 summarises the general picture of differences between municipal managers and local residents in terms of the values they ascribe to blue-green solutions and the benefits they consider most important. While these findings provide some guidance on the aspects that need to be considered when implementing blue-green solutions, the general public is itself a diverse stakeholder group. Therefore, in order to take full account of local needs and perspectives, it may be beneficial for local authorities to involve different stakeholder groups in the planning, design, construction and management of blue-green solutions. It is particularly important to involve traditionally marginalised societal groups, such as children, the elderly, immigrants, LGBTQ people and people with disabilities, in order to contribute to more socially inclusive cities and blue-green solutions adapted to different needs (Fors et al., 2021).

Table C6. General differences between municipal managers and local residents in the values they ascribe to blue-green solutions.

Municipal managers appreciate...	Local residents appreciate...
Moral values and values for future generations	Values related to current use and experience of the blue-green solution
General and holistic values	Place-specific values
Values that benefit the city as a whole or the environment, both now and in the future	Existential and symbolic values
...and see stormwater management as the primary driver for investment in blue-green solutions.	...and value co-benefits higher than actual stormwater management.

However, as can be concluded from this section, who you ask is not the only factor that will affect the co-benefits that can be obtained from a blue-green solution. In Figure C5, we summarise the main factors identified in the reviewed articles.

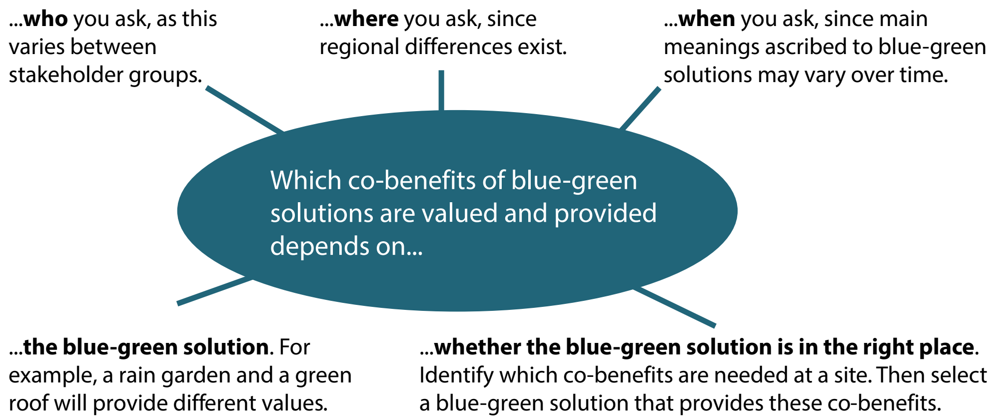

Figure C5. Factors affecting which co-benefits that are valued and provided.

C4.6 (RQ3) Underlying aims and incentives for sustainable stormwater management

In this section, we analyse the underlying aims and incentives for sustainable stormwater management according to different actors and stakeholders. We mainly focus on the aims and incentives of municipalities, as they are the main responsible actor in sustainable stormwater management in public urban areas.

C4.6.1 Handling stormwater runoff

Unsurprisingly, handling of stormwater runoff is generally the main objective of sustainable stormwater management in the studies reviewed. An example of this from Norway, where the planning processes for two stormwater sector plans were led by municipal water officials who actively involved actors from other relevant municipal sectors, such as planning and urban environmental departments, representing other interests and values. The ambition was to develop new knowledge; locally appropriate solutions; aiming for holistic planning by focusing not only on stormwater management but also on co-benefits and multifunctionality, such as preserving biodiversity and supporting ecosystem services. However, it was found that in practice, other functions such as parking lots were often given higher priority than blue-green solutions. The study concluded that when municipal water actors initiate holistic implementation of sustainable stormwater management systems, the traditional stormwater management aspects dominate (Kvamsås, 2023). Stormwater management was also in focus in France, where a county retention basin was constructed with the main aim of providing effective drainage and treatment of stormwater runoff from a local motorway (Ellis et al., 2004). On a smaller, site-specific scale, the City of Portland adopted 'green streets' as part of a broader watershed management plan that aimed to address deteriorating pipe infrastructure, and in doing so, involved citizens in stewardship efforts to develop decentralised stormwater facilities that capture stormwater and infiltrate into soils (Shandas, 2015). The national technical guideline for handling of stormwater through implementation of blue-green solutions, introduced by Norwegian Water in 2015, had general sustainability goals, aiming to reduce the adverse impacts of climate change and urbanisation on the community and turn water problems into resources by applying sustainable measures (Nie, 2016).

C4.6.2 Creating multifunctional spaces

In addition to stormwater management, stakeholders in the reviewed studies also describe important additional reasons for implementing blue-green solutions. The ambition to create multifunctional spaces could be seen in several cases. In a study of green roofs in the USA, telephone interviews were conducted with hospital administrators and sustainability professionals from hospitals with green roofs (Starry et al., 2022). Respondents always had multiple reasons for initially investing resources in a green roof, with the main objective being the patient experience, with patients visiting the roof, seeing it as a relaxing area, or having a view of the green roof. However, only 60 % of the 105 American hospitals identified with green roofs had accessible green roofs, so making more of them accessible would increase their usage. Many respondents also stated that green roofs were part of the hospital's marketing, attracting prospective students and patients. Other objectives included the green roof contributing to energy savings, patient therapy, ecological values, stormwater management and educational programmes such as medicinal plant cultivation (Starry et al., 2022).

The US Environmental Protection Agency sees stormwater management as the central benefit of interest for green stormwater infrastructure, but also recognises the many co-benefits of blue-green solutions, such as water supply, reduction of ground-level ozone and particulate pollution, heat island reduction, habitat provision and connectivity, and cultural services such as aesthetics and recreation (Turner et al., 2022). Similarly, the *Plan for a stronger, more resilient New York* mentions GI, stating that it has the ability to simultaneously "absorb storm water, mitigate local flooding, decrease urban heat island effect, increase pedestrian and traffic safety, and beautify neighborhoods" (Meerow, 2020).

Creating multifunctional spaces was also in focus in the first Green Roof Strategy of Oslo, Norway, which identified the following ecosystem service objectives for green roofs in the city: better stormwater management; preservation of the city's biodiversity; spaces for recreation, socialising, learning and experiences; local food production; aesthetics; temperature regulation in and between buildings; improved air quality and noise reduction; and CO_2 sequestration (Venter et al., 2021).

In a Copenhagen case, the municipality aimed to improve possibilities for recreation by using WSUD retrofits to reduce number of combined sewage overflows in order to achieve bathing water quality in an estuary (Fryd et al., 2013). Employing an economic perspective, the city of Detroit, USA, promoted the installation of GI both as a remedy for property flooding and as a way to reduce or eliminate the city's drainage fee, which adds a charge to water bills based on the proportion of impervious surface on a property – a fee that was very unpopular with residents (Campbell-Arvai & Lindquist, 2021).

In a study of two watersheds in Maryland, USA, the primary goal of stormwater management for local authorities was to meet increasingly strict legal requirements to control and treat stormwater prior to discharge to improve downstream water quality. For local residents, however, the goals of stormwater management were more related to the broader objective of creating liveable, healthy, resilient communities, and they were frustrated with the way stormwater was being managed. The study concluded that the objectives of local authorities are not always aligned with the needs of local residents (Wilfong et al., 2023).

C4.6.3 The policy context affects stormwater management aims

A survey of stormwater professionals in Cleveland, Ohio, and Denver, Colorado, found regional differences where Cleveland respondents mainly focused on water quantity, selecting peak flow/flood and volume reduction as primary goals, while Denver respondents mainly focused on water quality and erosion control/stream stability as primary goals (Turner et al., 2022). This was consistent with the regional context in terms of regulatory policy and existing infrastructure, showing that policy context shape regional priorities for stormwater control measures (Turner et al., 2022).

C4.6.4 Need to consider different perspectives

The objectives of sustainable stormwater management according to different stakeholders were not a prominent part of the reviewed articles. However, what is evident is the ongoing transition from a focus on flood mitigation only to new approaches and the promotion of multiple values and benefits. This finding is consistent with the development of stormwater management as described by Fletcher et al. (2015), which shows a broadened focus from the 1960s, when urban stormwater was initially managed with the aim of flood mitigation, to the multiple aims of sustainable stormwater management today (see Figure C6).

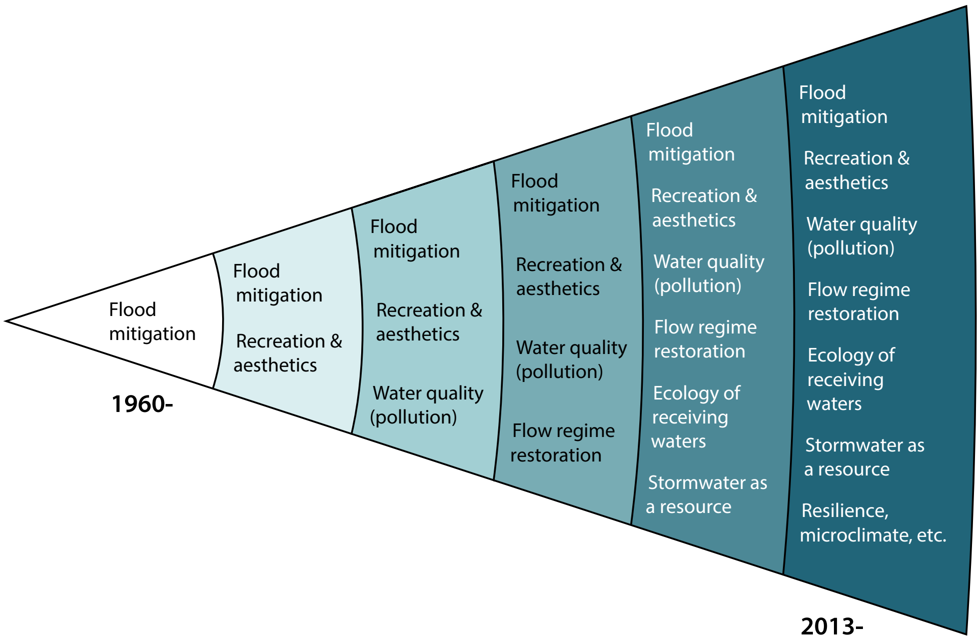

Figure C6. Overview of the increasing integration and complexity of urban stormwater management over time (after Fletcher et al., 2015).

If the many desired objectives of sustainable stormwater management outlined above are to be achieved, several different perspectives need to be considered simultaneously when deciding how to implement and manage blue-green solutions. For example, removal of litter and debris, weeding and cleaning and removing build-up of geotextile or filters may be appropriate maintenance activities for a blue-green solution if only the main function of flood mitigation is considered (Knapik et al., 2024). However, to facilitate the delivery of co-benefits and the creation of a multi-functional blue-green solution, other maintenance measures are required. For example, a detention basin that is used as a sunken football pitch when not flooded requires different maintenance and has different key stakeholders whose needs need to be addressed than a detention basin that is not intended to provide co-benefits. It is therefore important to consider the blue-green solution as part of a whole, taking into account all the functions, values and stakeholders relevant to the site.

Like the governance and management of public urban green spaces in general, the governance and management of blue-green solutions needs to be place and context specific for successful performance (Knapik et al., 2024; Qiao et al., 2018). This involves taking into account aspects related to the physical characteristics of the site and local conditions, such as soil type and moisture content, topography, local climate and ecology, and selecting plant species that prefer and thrive in these conditions (Knapik et al., 2024). Equally important, however, are the needs of local residents and other stakeholders, and the functions that a specific blue-green solution provides for them in addition to flood mitigation. Aspects such as number of visitors, desired co-benefits and key stakeholders need to be considered in both implementation and long-term management. The fact that blue-green solutions, unlike underground pipes, are visible to local residents means that the appearance of the system needs to be a priority, as public acceptance and involvement can be affected by negative perceptions of blue-green solutions implemented (Knapik et al., 2024).

C4.7 (RQ4) Barriers and challenges to sustainable stormwater management

Despite the fact that blue-green solutions are increasingly valued in urban planning, their implementation is still slow. Sufficient and efficient implementation of blue-green solutions, scaling-back traditional grey stormwater infrastructure requires that collaboration between various stakeholders, with varied aims and interests, is sustained over long periods of time and that the solutions are adapted to local conditions and needs, while ensuring that they perform in a just as reliable and cost-effective was as grey solutions (Montalto et al., 2013). Sometimes other functions and values such as parking, universal design (i.e. urban environments that provide universal access for all) and cultural heritage considerations are given higher priority (Kvamsås, 2023). There are also a range of other existing barriers and challenges to sustainable stormwater management. In this section, we map barriers and challenges to sustainable stormwater management, as they have been described in the literature, and how these can be overcome.

The most common barriers to sustainable stormwater management identified were regulatory barriers, organisational barriers, knowledge barriers and economic barriers (Figure C7). Other barriers mentioned multiple times include the influence of the physical environment, and land ownership and social barriers.

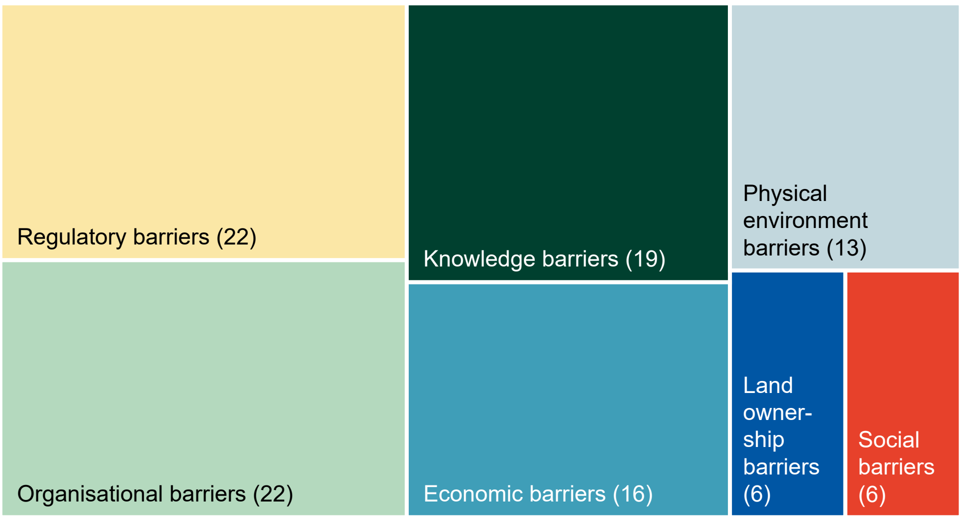

Figure C7. Overview of barriers to sustainable stormwater management mentioned in the reviewed articles. Number of articles mentioning barriers within each category in brackets.

C4.7.1 Regulatory barriers

The majority of the regulatory barriers relates to lack of supportive policies. This is expressed in differences in law and design standards relating to urban drainage and land use (Nie, 2016; Rooney & Gill, 2018), which can be exemplified in the impracticalities of using rain barrels on private land due to local precipitation policies, or threshold requirements of the minimum development area that is required to install and apply blue-green solutions (Dhakal & Chevalier, 2017). Such legal restrictions discourage the allocation of funds for implementation of blue-green solutions on private properties, resulting in stormwater management funds primarily supporting grey infrastructure development on public land (Dhakal & Chevalier, 2017). Thus, lack of regulatory drivers (Johns, 2019) or following current legislation might force local authorities to act in ways that are contradictory to policies for blue-green solutions or best practices (Rooney & Gill, 2018). The challenge of recognising the multidimensional benefits of blue-green solutions is exacerbated by poor integration of regulatory bodies (Wihlborg et al., 2019). Additionally, sustainable initiatives relying on voluntary adoption may face implementation difficulties for developers and private sectors due to the absence of supportive policies (Zhang & He, 2021). Individuals may find it difficult to economically motivate implementation on their property; a local community will receive main and co-benefits from blue-green solutions as a collective group, but the individual household will not gain tangible economic benefits from implementing blue-green solutions on their property, unless they can participate in a stormwater incentive programme (Heidari et al., 2022). The implementation of blue-green solutions is further hampered by time constraints due to political pressure for immediate solutions, even though it takes time to develop new solutions (Liu & Jensen, 2017).

C4.7.2 Knowledge barriers

The primary knowledge barrier to sustainable stormwater management is the lack of documented knowledge regarding the benefits of blue-green solutions. This includes the lack of monitoring and performance evaluations (Li et al., 2020; Qiao et al., 2018; Rooney & Gill, 2018), which impedes practitioners from comprehending effectiveness of blue-green solutions and improving practices (Li et al., 2020). These challenges extend beyond implementing blue-green solutions in new areas to include the maintenance and replacement of aging infrastructure (Upadhyaya et al., 2014).

Lack of knowledge of the tools required can also be a barrier. For example, the modelling of blue-green solutions in a 3D design software to facilitate public participation was only fast after a significant learning process when used by landscape architects who are not experts in Geographic Information Systems (GIS) or coding (Pierre et al., 2019). Another knowledge barrier is the lack of engineering expertise, where the dominance of grey infrastructure engineering expertise over green infrastructure knowledge inhibits the adoption of sustainable stormwater management practices (Cousins, 2018; Johns, 2019). The lack of understanding, education, and awareness of sustainable stormwater management and its practices is still evident among various stakeholders, including private property owners, the general public, and professionals (Li et al., 2020). This deficiency can lead to a lack of trust, where local authorities may doubt the ability of private stakeholders to effectively manage stormwater, who in turn may be reluctant to participate in sustainable stormwater management practices (Qiao et al., 2018). The knowledge gaps hinder informed decision-making and effective management of stormwater infrastructure (Rooney & Gill, 2018), potentially leading to the implementation of a one-size-fits-all approach suggested by design standards (Zuniga-Teran et al., 2020).

C4.7.3 Economic barriers

The main economic barrier is insufficient funds at national, regional, and local government level which hinders the implementation of new blue-green solutions (Carlson et al., 2015; Li et al., 2020; Qiao et al., 2019; 2018) as well as the long-term management and restoration of existing stormwater infrastructure (Rooney & Gill, 2018; Upadhyaya et al., 2014). Currently, funding allocation for blue-green solutions is smaller compared to traditional grey infrastructure, and the use of funding tools such as stormwater fees is currently limited (Johns, 2019).

Detailed cost estimations that consider broader benefits beyond stormwater management such as maintenance costs (Li et al., 2020) and ecosystem services (Dhakal & Chevalier, 2017) are lacking (Qiao et al., 2018; Zuniga-Teran et al., 2020) due to the absence of a suitable valuation tool. This leads to reluctance from authorities, private stakeholders and developers to invest in blue-green solutions due to perceived cost risks.

Additionally, silo thinking within different municipal departments, each with its own budget and responsibilities, can hinder holistic planning and implementation due to budget constraints and lack of flexibility (Qiao et al., 2019). The funding is also an issue for implementation of blue-green solutions for private households, where low-income and marginalised households may not have financial resources to fund projects and instead have to depend on grant programs (Carlson et al., 2015; Matsler et al., 2023). Today, these are primarily directed towards institutions rather than individual residential or commercial property owners, limiting options for upfront funding for stormwater infrastructure on private property (Matsler et al., 2023) and leading to a lack of economic incentives for private investment in blue-green solutions (Ossa-Moreno et al., 2017). Cost was also mentioned as a key barrier to installation of additional green roofs at hospitals in the USA (Starry et al., 2022).

C4.7.4 Physical environment barriers

The physical environment barriers relate to challenges following urbanisation, where vegetated areas historically have been replaced by impervious infrastructure such as buildings and streets (Nie, 2016). The current densification of urban areas as a result of continued urbanisation has created diverging needs and interests for infrastructure development (Buffam et al., 2022), with housing needs often being prioritised over blue-green solutions, resulting in lack of space to implement new blue-green solutions (Buffam et al., 2022; Ellis et al., 2004; Liu & Jensen, 2017; Nie, 2016; Qiao et al., 2018; Upadhyaya et al., 2014). Conflicting land use interests hinder both new development of sustainable stormwater management systems and retrofitting in densely populated areas (Upadhyaya et al., 2014). The widespread implementation required to achieve the desired benefits is challenged by the diversity of urban surfaces in terms of construction, ownership, neighbourhood character, and underlying soils and geology, requiring local adaptation of solutions (Montalto et al., 2013). Additionally, physical barriers can hinder implementation of new blue-green solutions, such as the railway that hindered discharging floodwaters to the ocean in a study of Copenhagen, Denmark (Liu & Jensen, 2017). Other hindering aspects of the physical environment, found in a checklist used in France, include susceptibility of the soil nature to water, groundwater vulnerability to rainfall, high water table level, high risk of water pollution, low overburden capacity, low soil surface permeability, low permeability of the soil at depth, risk of water being polluted with fine suspended solids, and heavy traffic, all of which make infiltration solutions more difficult or even impossible (Ellis et al., 2004). Further, lack of an existing permanent outlet, low water table level and no existing permanent water flow obstruct use of retention techniques, and mean to high site slopes requires installation of check dams (Ellis et al., 2004). Also, historic urban landfills, such as solid waste, fly ash and rubble, make infiltration rates and lateral flow highly variable, reducing the reliability with which blue-green solutions can be expected to perform on these soils (Montalto et al., 2013).

The properties of the physical environment may hinder the use of blue-green solutions as a stand-alone solution, especially when implemented through retrofitting in existing urban areas (Liu and Jensen, 2017). A similar conclusion was drawn from a study of three cases of innovative flagship projects for nature-based climate adaptation and stormwater management in Copenhagen, Denmark, which stated that managing urban stormwater through natural processes alone is not a viable option (Jørgensen et al., 2022). The projects studied were essentially nature-based, but all were supported by technical grey solutions for economic or practical reasons, which means that from a technical point of view, the projects were hybrids. However, all projects generated co-benefits for several user groups in addition to flood protection, which is an important aspect included in the definition of NBS, meaning that they were nature-based in that respect (Jørgensen et al., 2022).

C4.7.5 Organisational barriers

Urban planning, zoning, development control and the construction and maintenance of green spaces have a major impact on stormwater management. When responsibility for these functions is spread across a number of different municipal departments rather than under one stormwater management department, it leads to fragmented governance and the risk of developing conflicting policies and programmes (Dhakal & Chevalier, 2016). The majority of the studies found the lack of collaboration and coordination among stakeholders and between different municipal departments (the silo effect) to be a major organisational barrier to implementation of blue-green solutions (Buffam et al., 2022; Carlson et al., 2015; Cousins, 2018; Dhakal & Chevalier, 2017; Fitzgerald & Laufer, 2017; Johns, 2019; Liu & Jensen, 2017; Nie, 2016; Upadhyaya et al., 2014). One example of this is the prevailing fragmented governance structure of the system, where drinking water, wastewater and stormwater are managed in three different municipal silos, making it difficult to bring all stakeholders together and integrate all parts of the urban water system into a single entity, which is a desirable goal that has been conceptualised as the 'One Water' approach (Pokhrel et al., 2022). Another challenge to the widespread adoption of the One Water approach is that the regulatory structure differs between the three different components of the urban water system. Furthermore, insufficient stakeholder engagement within and between water utilities and other institutions poses a serious challenge to the successful implementation of this approach (Pokhrel et al., 2022). Different stakeholders often have competing priorities and goals (Dhakal & Chevalier, 2017; Upadhyaya et al., 2014) resulting in difficulties to collaborate and in developing a comprehensive view of stormwater management issues (Carlson et al., 2015).

The lack of collaboration includes a lack of communication to solve problems and share information between local, regional and national levels of government, or between authorities and utility companies (Johns, 2019; Upadhyaya et al., 2014).

Within municipalities, the missing collaboration between water and planning departments (Johns, 2019) and sectors responsible for transportation, infrastructure, emergency planning and electric supply (Nie, 2016) is crucial for the planning and implementation of sustainable stormwater management systems that are resilient to heavy flooding events. In their study of the implementation of a stormwater management plan in Philadelphia, USA, Fitzgerald & Laufer (2017) found that cross-departmental collaboration was hampered by communication issues, competing departmental values (engineers focused on function versus planners and designers focused on aesthetics), and different regulatory environments at different departments (Fitzgerald & Laufer, 2017). The fact that blue-green solutions have multiple purposes and that the actors responsible for these different purposes are located in different siloed departments also makes collaboration difficult, as the division of responsibility for construction, operation and maintenance easily becomes unclear (Bohman et al., 2020).

Organisational culture may also hinder implementation of blue-green solutions. Even when there is a willingness to change, path-dependent, institutional patterns may inhibit the development of sustainable solutions due to that they are not fully compatible with the new goals of blue-green solutions compared to the aims of grey solutions that they were designed for (Bohman et al., 2020).

Vollaers et al. (2021) documented 70 cases in 11 Dutch municipalities to identify the causes of failures of sustainable stormwater management systems in practice. The results showed that failures often occurred at the interfaces between the blue-green solutions and other urban systems, such as when heavy traffic damaged a permeable pavement. Fifty per cent of the failures occurred in the design phase, and the rest in the construction and maintenance phases. This means that each phase needs attention to develop well performing blue-green solutions (Vollaers et al., 2021). The underlying causes of the failures were mainly socio-institutional in nature (Vollaers et al., 2021). Examples of causes include deep-rooted practices of the actors involved that lead to incorrect design of the blue-green solution, such as the habit of constructing raised edges around green spaces; poor communication between different actors and between different project phases; lack of knowledge about the performance and maintenance of blue-green solutions and how they interact with other urban systems; lack of experience in the construction of blue-green solutions; poor maintenance because parts of the blue-green solution are inaccessible or it is unclear who is responsible for maintenance; and people's actual use of the blue-green solution, such as putting a flower pot in a gutter, which blocks the stormwater flow path (Vollaers et al., 2021).

The lack of intergovernmental collaboration has been identified as particularly lacking between regional and local authorities (Johns, 2019), and is currently influenced by another organisational barrier: a lack of leadership (Johns, 2019; Qiao et al., 2018; 2019). The absence of clear leadership hampers the development and implementation of blue-green solutions due to unclear responsibilities between stakeholders such as municipalities, consultants, property owners, and developers (Li et al., 2020; Qiao et al. 2019). Unclear roles and responsibilities increase the confusion between these actors, leading to inefficient implementation (Qiao et al., 2019). In addition, weak governance strategies (Li et al., 2020) further exacerbate organisational barriers, as the current governance strategies that are often used were created to support grey infrastructure and are therefore not suitable for implementing blue-green solutions (Dhakal & Chevalier, 2017). These challenges are not specific to stormwater management, but apply to green space management in general.

C4.7.6 Land ownership barriers

Various types of land ownership barriers also hinder the expansion of sustainable stormwater management initiatives. One of these is the lack of access to privately owned land (Buffam et al., 2022). The large proportion of land being privately owned is a common barrier to sustainable stormwater initiatives in the USA (Johns, 2019), where the US Constitution prevents local authorities from implementing blue-green solutions on private land (Dhakal & Chevalier, 2017). This is a problem, since managing stormwater on public land alone will not be enough. A modelling study comparing different possible scenarios for implementing blue-green solutions in a specific neighbourhood of Philadelphia, USA, showed that the city would not be able to achieve its stormwater management goals unless it could engage more private property owners in managing stormwater locally on private land (Zidar et al., 2017).

Another barrier related to land ownership concerns municipal boundaries. River catchments often cross municipal boundaries, contributing to inconsistent and conflicting sustainable stormwater management strategies within a county when coordination is lacking (Rooney & Gill, 2018) and requiring blue-green solutions to be implemented within the catchments associated with the drainage network, rather than within the watersheds associated with local topography (Montalto et al., 2013).

C4.7.7 Social barriers

The level of interest in implementing blue-green solutions on their property varies between different owners, depending on the characteristics of the physical property, public or private ownership, and the socio-economic, cultural and demographic background of the owners (Montalto et al., 2013). Other challenges arise when designing blue-green solutions in the vicinity of spaces for children, particularly regarding safety concerns. Some blue-green solutions, such as wetlands, have permanent ponding and are not suitable in young children's spaces as they are considered a drowning hazard (Cizek & Fox, 2015). In addition, the blue-green solutions within children's spaces should not be designed to receive runoff from impervious surfaces that are heavily trafficked due to high pollutant levels (Cizek & Fox, 2015).

Another challenge is the equity aspect of access to blue-green solutions. There is currently a disproportionate access to blue-green solutions, where wealthier neighbourhoods often are prioritised over low-income neighbourhoods (Zuniga-Teran et al., 2020). This presents an environmental justice issue, which needs to be considered when planning and implementing blue-green solutions. In cities experiencing rapid green gentrification of low-income neighbourhoods, municipal plans to invest in blue-green solutions can be perceived as a precursor to lower-income residents being pushed out of the neighbourhood as property values rise, fostering mistrust between residents and local authorities that makes implementation difficult (Heckert & Rosan, 2018).

Distributional justice is when everyone has access to high quality urban green space. In this line, Starry et al. (2022) state that while we know that green roofs manage stormwater and may provide habitat for insects, the fact that less open space will be available due to urban densification raises the question of whether green roofs might additionally provide similar therapeutic benefits as urban green space, but at a lower cost, potentially making therapeutic benefits easily accessible to the general public. Procedural justice is about ensuring that all citizen groups, including traditionally marginalised ones, have the opportunity to participate in decision-making related to urban green space development. Current top-down technocratic systems of stormwater governance and management have low levels of procedural justice and constrain increased user participation in stormwater management, leaving participants with a sense of lack of autonomy in defining their roles, duties and responsibilities and with conflicts, tensions and inequities as a result (Wilfong et al., 2023).

C4.7.8 Overcoming barriers and challenges to sustainable stormwater management

To address the identified barriers and challenges to sustainable stormwater management, various strategies can be employed. First of all, **regulatory barriers** could be overcome by advocating for changes in legislation, regulation, and planning guidelines to support sustainable stormwater management (Li et al., 2020; Rooney & Gill, 2018). One approach is to remove specific technical requirements and reintroduce the possibility of imposing requirements such as water management capacity, through legislation. Another radical approach could be to mandate that each parcel of land manages a certain percentage of rainwater that lands on it, although this may face challenges in densely built areas (Wihlborg et al., 2019). It is also important that governmental policies and regulations become more adaptive and flexible and less reliant on quantitative outcomes to allow for more integrative, collaborative planning and decision-making involving the diverse range of urban stormwater stakeholders (Ossa-Moreno et al., 2017; Wilfong et al., 2023). A study of green leader cities in the US, i.e. cities that are early adopters of blue-green solutions, found that the critical factor that led them to plan significant investments in GI was their ability to take advantage of the 'policy window' that opened when the US Environmental Protection Agency issued a memorandum encouraging the use of blue-green rather than grey solutions to meet permitting requirements. They did not miss the opportunity and immediately incorporated blue-green solutions into their management plans (Hopkins et al., 2018).

It is also important to adopt a type of regulation that encourages citizen participation in sustainable stormwater management and the installation of blue-green solutions on private property. A study on how different policy instruments encourage citizens to participate in sustainable stormwater management, focusing on Onondaga County, New York, USA, showed that the type of policy instrument influences the level of citizen engagement (Lieberherr & Green, 2018). According to the case study, incentive-based policies ('carrots'), such as providing residents with rain barrels or financial reimbursement for installing green roofs or rain gardens on their property, and outreach ('sermon') policies, such as outreach and public education on sustainable stormwater management, can play an important role in encouraging citizen participation when regulatory instruments (sticks) are lacking (Lieberherr & Green, 2018).

Further, there is a need for new regulations that approach stormwater as a 'resource' rather than a 'hazard', and stormwater management as an 'opportunity' rather than a 'liability', to open up new possibilities for stormwater management and a more holistic range of desired regulatory outcomes and objectives for stormwater management (Wilfong et al., 2023). In practice, this shift towards seeing stormwater as a resource has been described as a move towards more integrated and hybrid forms of water governance in a number of ways (Cousins, 2018). First, GI could be viewed as a resource through economic valuation tools that monetise the value of stormwater and blue-green solutions, including their primary and co-benefits, helping public officials to see the financial opportunities and consequences of their decisions (Cousins, 2018). Another way of reimagining stormwater as a resource is to see GI as insurance, as well-functioning GI could help cities avoid the costs of floods and extreme events and build resilient cities (Cousins, 2018). However, the monetisation of resilience should be used with caution, as it is based on a highly simplified understanding of ecosystems and can create the misleading notion that investing in ecosystem resilience is equivalent to buying financial insurance (Cousins, 2018). Second, most legal and regulatory frameworks have historically defined stormwater as a nuisance, which prevents planners and managers from managing stormwater as a resource. For this to change, there is a need to discursively redefine stormwater as a resource (e.g. as a water supply issue) in laws and regulations, and to legally recognise the benefits of stormwater (Cousins, 2018). In addition, decentralised forms of governance that foster new forms of subject-making and urban environmental citizenship are needed to transform society's relationship with stormwater and realise its value as a resource. This may include new forms of public participation in sustainable stormwater management, such as rainwater harvesting programmes that recast stormwater as an alternative source of supply, or education campaigns that encourage behavioural change by educating local residents about their impact on the water cycle and how they can reduce their impact by installing rain barrels, rain gardens and cisterns on their property (Cousins, 2018).

Addressing **knowledge barriers** requires a comprehensive strategy. Current approaches to assessing sustainability of blue-green solutions often overlook considerations such as public health, well-being, and other co-benefits. Quantifying the risks in economic terms (Carlson et al., 2015) and establishing new evaluation criteria and standards for measuring the performance of blue-green solutions is essential for documenting, communicating and promoting the benefits of blue-green solutions and potentially integrating them into legislation (Qiao et al., 2018). Incorporating a hierarchical assessment approach that considers the functionality, resilience, and sustainability of blue-green solutions could provide a more thorough evaluation of the effectiveness of each initiative (Buffam et al., 2022; Upadhyaya et al., 2014).

To contribute to the lack of cost calculations for blue-green solutions compared to grey solutions, Heidari et al. (2022) conducted a quantitative and comparative analysis of rain gardens and permeable pavements and the benefits they provide, including nutrient uptake from stormwater and air pollutant deposition, to help justify spending on blue-green solutions. Using modelling for a case study watershed in Maryland, USA, they assessed stormwater quality improvement, life-cycle costs, and air pollutant deposition throughout the life-cycle of the two selected blue-green solutions, calculating benefit-cost ratios and stormwater nutrient removal costs at multiple spatial scales: household, subwatershed, and watershed. The results showed that rain gardens are much more efficient in treating stormwater at the household scale, while at the watershed scale, large dry or wet basins are more efficient if land and resources are available (Heidari et al., 2022). Furthermore, when comparing nutrient removal costs between permeable pavement and rain gardens, they found that implementing numerous rain gardens throughout a watershed is more cost-effective than installing permeable pavement in commercial car parks (Heidari et al., 2022). This is because permeable pavements are generally less efficient at treating stormwater and are significantly more expensive to construct and maintain (Heidari et al., 2022). They also identified households and sub-watersheds where nutrient removal costs were lower than at the watershed scale, calling for fine-scale optimisation of which blue-green solution to choose and where, using such optimal locations within a watershed (Heidari et al., 2022). In addition, the cost of nutrient removal correlates with the distance to the watershed outlet, with higher costs for upstream subwatersheds and those on the watershed boundary than for downstream subwatersheds (Heidari et al., 2022). This is because upstream runoff has already been treated by blue-green solutions, so downstream solutions only need to reduce nutrient concentrations in already treated water, which implies lower costs. This is a reason to implement blue-green solutions in upstream sub-watersheds (Heidari et al., 2022).

To guide multi-stakeholder decision-making, Liu et al. (2023) developed a framework to facilitate spatial prioritisation of blue-green solutions and maximise co-benefits at the catchment level. The framework was based on the CatchWat-SD model, developed to simulate an integrated multi-catchment water cycle, and applied to the Norfolk region of the UK. Aiming to bridge the urban-rural divide in catchment water management, three rural (runoff attenuation features, regenerative agriculture, floodplain) and two urban (urban green space, constructed wastewater wetlands) blue-green solutions were compared at different scales of implementation. To understand the impact of the studied blue-green solutions on the integrated water cycle, they assessed the system-level benefits of the studied solutions in terms of water availability, water quality and flood management at the catchment scale, and the associated economic costs. They found that rural blue-green solutions had more significant impacts across the catchment, which increased with the scale of implementation. They concluded that integrated urban-rural planning of blue-green solutions can improve water availability, water quality and flood management simultaneously, but that there may be trade-offs between different objectives. For example, phosphorus levels were found to be best reduced by expansion of urban green space to reduce loading on combined sewer systems, but this was at the trade-off of water availability, flooding, nitrogen and suspended solids (Liu et al., 2023).

Another knowledge related issue is the needed quality control over the construction, operation and maintenance of blue-green solutions through regular inspections or enforcement of standards of operation to ensure that all the decentralised stormwater facilities perform with high quality (Montalto et al., 2013). For this, training of local volunteers could be an option (Montalto et al., 2013).

Moreover, utilising spatial modelling allows planners to understand the impact of different land use decisions on ecosystem services, facilitating informed decision-making (Buffam et al., 2022). In addition, conducting more pilot studies to test solutions in different contexts (Li et al., 2020), extensive institutional lobbying (Montalto et al., 2013), and implementing widespread education programmes (Dhakal & Chevalier, 2017; Johns, 2019; Montalto et al., 2013) are critical to raising awareness of the benefits of blue-green solutions over grey infrastructure and ensuring implementation at a sufficient pace (Montalto et al., 2013).

The facilitation of knowledge exchange and collaboration between various stakeholders, including local authorities, urban planners, water professionals, and private stakeholders, can be achieved through the involvement of researchers from universities or research institutions (Qiao et al., 2019; 2018). Modelling blue-green solutions in 3D design software could be an effective communication tool for user participation processes. However, as this modelling has been found to require a significant learning process before it can be used by landscape architects, it may be helpful for this group to know which aspects of such software can be learned quickly so that they can use such software when involving the public in public decisions on stormwater management (Pierre et al., 2019).

Several strategies are suggested to overcome the **economic barriers** hindering sustainable stormwater management. Clearly, defining the level of ambition in each project requires agreement from all parties, as well as clarifying operational responsibilities and long-term funding. While sustainable stormwater management systems may require a higher initial investment, their long-term benefits often outweigh those of conventional stormwater systems (Zhang & He, 2021), and increased knowledge and quantification of the environmental and social benefits over time could convince owners and contractors to invest. Securing additional federal and regional funding is crucial to effectively support new initiatives (Upadhyaya et al., 2014; Rooney & Gill, 2018), while ensuring adequate maintenance funding is equally important for maintaining the long-term performance (Qiao et al., 2019).

Smart technologies that use sensing, controls, communications, and computing can improve the performance of blue-green solutions, but before investing in smart GI, it is important to understand what features and capabilities are desired by potential users. Meng & Hsu (2019) conducted a national survey in the US to understand what type of smart GI water utility and agency officials preferred for stormwater management. They found that these officials were willing to accept higher construction costs for smart GI if it meant less labour-intensive long-term management and lower long-term management costs. For example, water agencies were willing to pay 12.1 % more to construct a typical rain garden in order to reduce maintenance costs by 20 %, and 12.9 % more to add the function of self-irrigation when needed. This means that, if used in a way that meets stakeholder preferences, smart GI can help finance the long-term management of blue-green solutions. It was also found that agencies responsible for large service areas and those with prior experience with GI were more likely to adopt smart GI (Meng & Hsu, 2019).

Economic incentives, such as cost-sharing agreements or impact fees (Qiao et al., 2018), are considered essential to encourage private stakeholders to participate in the implementation of blue-green solutions in both new developments and retrofit projects (Rooney & Gill, 2018). For economic incentive schemes to be successful, clear guidance on the technical requirements for obtaining and maintaining incentives is important (Ossa-Moreno et al., 2017). Additionally, incentives such as fines for non-compliance with water quality standards and subsidies for installation of blue-green solutions can encourage household-level implementation, with support from grants, loans, or contributions from the private sector (Zuniga-Teran et al., 2020). Economically motivating property owners with fewer impervious surfaces to pay lower water fees can also boost the economic drive for implementing blue-green solutions (Wihlborg et al., 2019). Some public water utilities charge landowners a fee based on the amount of impervious surface on their property, with the fee increasing as owners pave more, and use the money to fund the implementation and management of more blue-green solutions (Carlson et al., 2015; Fitzgerald & Laufer, 2017; Ossa-Moreno et al., 2017). Reforming stormwater drainage charges in this way, moving away from a system based on water supply to one based on the impact of the property on the stormwater network, is an important institutional change towards a more equitable charging system that follows the polluter pays principle (Ossa-Moreno et al., 2017). Under the old system, a store with a large car park would have a small bill because of its low water consumption, while under the new system it would have a much larger bill because of its large area of impervious cover (Fitzgerald & Laufer, 2017). Education and outreach to property owners is important for this arrangement to work (Carlson et al., 2015).

To enable multiple funding and increase the implementation of blue-green solutions, one idea could be to identify all the stakeholders that would benefit from the intervention and share the implementation and management costs between them in proportion to the benefits they each receive (Ossa-Moreno et al., 2017). Stakeholders could include, for example, local residents, schools, public roads and the water utility. This could be operationalised through the establishment of public-private partnerships, where, for example, reduced fees from the water utility would be an incentive for private stakeholders to participate (Ossa-Moreno et al., 2017).

Philadelphia is a leading city in the US in climate action planning, having implemented a green stormwater management plan based almost entirely on green infrastructure through collaboration between several city departments (Fitzgerald & Laufer, 2017). To persuade private property owners to increase the implementation of blue-green solutions on their properties, the Philadelphia Water Department has introduced several programmes to reduce the economic burden. One of them allowed commercial property owners to apply for stormwater credits for pre-existing tree canopy, roof or downspout disconnection, porous pavement, green roofs, and sidewalk disconnection, for which they received a reduced stormwater fee (Fitzgerald & Laufer, 2017). However, despite the potential for significant savings, few property owners applied because the payback period for green infrastructure is up to 15–20 years, which is beyond the time horizon of most businesses (Fitzgerald & Laufer, 2017). Another programme provided grants and low-interest financing for the implementation of stormwater runoff reduction systems. A third programme focused on stormwater management at the district level, providing grants to project aggregators who were able to develop a stormwater management plan for multiple properties and obtain agreements with all property owners to agree to a 45-year operation and maintenance agreement with the water authority (Fitzgerald & Laufer, 2017). The programmes were funded by revenue from stormwater charges (Fitzgerald & Laufer, 2017). Local residents were offered free rain barrels and the opportunity to apply for the Rain Check programme (Fitzgerald & Laufer, 2017). In order to receive the financial support to manage water on their property, residents were required to attend a workshop to learn about the role their properties could play in implementing the city's stormwater management plan. Blue-green solutions installed by residents through the programme include downspout planters, rain garden plantings, pavement removal projects, and rain barrel installations (Fitzgerald & Laufer, 2017).

The general lack of space in urban areas necessitates the use of vacant land for the implementation of blue-green solutions (Qiao et al., 2018), which is not only a way to overcome space constraints, but also the creation of new business opportunities. Repurposing vacant land in the city is one way to address the **physical**

environment barrier of lack of green space to implement new blue-green solutions. The Great Lakes region of the USA has large amounts of vacant land due to population loss and urban sprawl. Such small, dispersed vacant lots could potentially be used to manage stormwater closer to where it is generated. Assessing the suitability of vacant lots for stormwater management requires highly detailed data, which is often lacking (Albro et al., 2017). A project in Buffalo, New York, generated the necessary data by developing a standardised protocol for assessing the suitability of vacant land, including parameters such as the location of high and low points on each lot, including areas of standing water and specific invasive hydrophilic plant species, soil compaction at randomised sampling locations, soil conductivity, and soil texture (Albro et al., 2017). The selection of suitable vacant sites for stormwater management for another project in the region, called *Vacant to Vibrant*, was based not only on the stormwater management potential of the site, but also on the potential for recreational use and the ease of project installation and management, including likelihood of societal acceptance (Albro et al., 2017).

Overcoming **organisational barriers** requires enhancing communication and collaboration among critical infrastructure systems. This includes urban planning, building operations, water and road systems, as well as community social and emergency services, to improve resilience to urban floods and disasters (Nie, 2016). Effective stormwater governance requires communication and collaboration across municipal departments, experimentation and a strategy for organisational learning, and that these three elements of governance are built into the planning process for blue-green solutions (Fitzgerald & Laufer, 2017). Also, Hopkins et al. (2018) found that building momentum and support for blue-green solutions occurred through a series of stages, from experimentation to demonstration to full transition from grey to blue-green approaches. Similarly, Rijke et al. (2013) describe the transition of urban stormwater management in Australian cities as consisting of three stages, with typical activities in the early stages including network formation, learning, experimentation, responding to a crisis and establishing policy decisions.

Building on the work of Rijke et al. (2013) and confirming their conclusions, Dobre et al. (2018) studied changes in stormwater management policies and practices in a municipality in Brussels, Belgium, in terms of how the technical and governance characteristics of alternative actions influence the shift from one transition stage to another in stormwater management (see Figure C8). The results show that in the early transition phase, soft actions such as changes in guidelines, legislation and economic incentives prevail (Dobre et al., 2018). The soft actions have the capacity to support and promote blue-green solutions in the planning, design and implementation stages. Dobre et al. (2018) highlight that policies should recognise that not only grey and blue-green solutions, but also soft actions are crucial to achieve a transition in stormwater management. In the first phase of transition, decentralised processes and informal networks create conditions for network formation, learning and experimentation, and promote the adoption of alternative actions (Dobre et al., 2018). In a next phase of transition, collaboration between formal institutions and informal networks is essential to increase the implementation of innovative stormwater management actions through replication (i.e. duplication of similar scale but different location and time), spatial or temporal upscaling (i.e. expanding the scope of an action in scale and time) and vertical upscaling (i.e. institutionalisation), and to ensure continuity over time (Dobre et al., 2018; Rijke et al., 2013). Local authorities are positioned as key actors to strengthen the collaboration between formal and informal networks (Dobre et al., 2018). The third stage of the transition process is reached when the integration of innovation reaches the status quo of the system (Rijke et al., 2013). In the Brussels case study, soft actions were associated with the typical activities of learning, network formation, experimentation and policy making, the implementation of blue-green solutions with experimentation and the implementation of grey solutions with responding to a crisis (Dobre et al., 2018). It was further concluded that safe implementation of blue-green solutions often needs to be preceded by soft actions, such as regulatory changes, which lead to the societal and institutional changes necessary for increased implementation of blue-green solutions (Dobre et al., 2018).

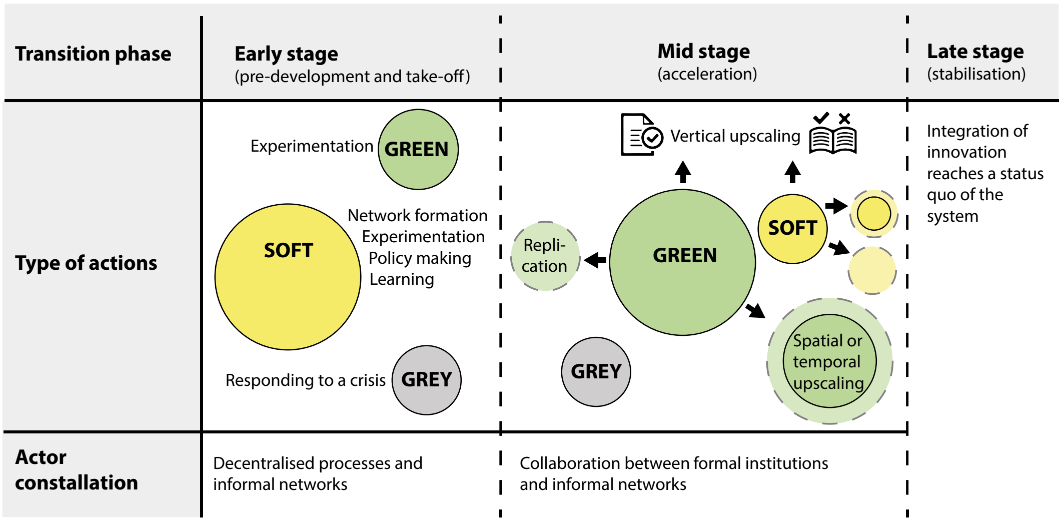

Figure C8. Soft actions in the early stage of transition are crucial for increased implementation of blue-green solutions in the mid stage. Authors' visualisation of findings by Dobre et al. (2018).

It is also important to get everyone on board from the start. The implementation of the *Green City, Clean Waters* stormwater management plan in Philadelphia, USA, required a major shift in policies and procedures. Acceptance of the cross-departmental cooperation required would have been greater if representatives from all affected departments had been involved in the planning process from the outset through an interdepartmental working group, to help everyone understand what was to be gained by adding more blue-green solutions, and to identify early on the barriers and changes needed in their own departments for smoother implementation of the new approach to stormwater management (Fitzgerald & Laufer, 2017). Similarly, for successful implementation it is important to involve maintenance staff already in the planning and design phase, as this increases the likelihood that they will support the blue-green solution (Starry et al., 2022). Creating mechanisms for communication, interaction, and coordination within government agencies and with external stakeholders is essential for facilitating implementation and long-term management of blue-green solutions (Dhakal & Chevalier, 2017; Qiao et al., 2018). This includes faster implementation and collaboration among key institutions (Liu & Jensen, 2017), along with education and public awareness programs to encourage retrofitting with blue-green solutions (Rooney & Gill, 2018). Fitzgerald & Laufer (2017) emphasise that interdepartmental collaboration is an ongoing process that needs to be prioritised for success, for example by assigning dedicated staff to maintain relationships with other departments, a role also described as a 'bridge builder' (Sunding & Randrup, 2024).

There are tools that can facilitate the multi-stakeholder collaboration needed to make a widespread transition from grey to blue-green solutions. One of these is the Geographic Information Systems (GIS)-based Gray to Green (G2G) GI planning tool, which can help both technical and non-technical users create and communicate a plan for GI implementation by identifying natural pathways for water on a site, treating stormwater runoff as close to the source as possible, and restoring a more natural water balance to the site, similar to what it was before development (Tsegaye et al., 2019). The G2G planning process consists of four steps: 1) mapping the existing hydrology and green infrastructure, 2) designing and adapting a development/redevelopment project to the site, 3) selecting appropriate blue-green solutions for the site, and 4) evaluating how well the blue-green solutions fit the site (Tsegaye et al., 2019).

There are a number of key actors who can lead the processes of increased collaboration between stakeholders that are needed for increased implementation of blue-green solutions. Landscape architects are trained to work collaboratively and are therefore uniquely positioned both to lead cross-departmental collaboration within a municipality and to combine cross-disciplinary collaboration in design with online user participation tools and shareable digital models, thereby significantly increasing user involvement in public decisions on stormwater management (Pierre et al., 2019). The presence of a driving spirit or champion can be critical to the implementation of blue-green solutions. An American study of hospital green roofs found that if there was a vocal champion advocating for a green roof, the hospital was more likely to invest resources, especially if that champion held a senior position, but also generally vocal advocates at all levels who felt they had the authority or freedom to experiment positively influenced green roof implementation (Starry et al., 2022). Further, based on interviews with American city, state, and regional staff on how they work with GI implementation, Harrington & Hsu (2018) concluded that government leadership at both the national and local levels is central to sustainable stormwater management. Broad adoption of blue-green solutions in the U.S. was found to be supported by *government as a driver* (through policy and local political support for GI), *government as a coordinator* (contributing knowledge by providing common definitions and best practices, and measuring the impact and effectiveness of blue-green solutions), and *government as a capacity builder* (where the U.S. Environmental Protection Agency (EPA) funded projects and programmes that build local capacity by supporting local collaboration and partnerships) (Harrington & Hsu, 2018). For example, the US EPA has funded the 'CLEAN Center' at Colorado State University, which aims to bring together different stakeholders and foster collaboration across sectors. Non-governmental actors are leading the way in information sharing. Harrington & Hsu (2018) argue that funding and management strategies should strengthen, rather than separate, the relationship between non-governmental and governmental actors, as both stakeholder groups are essential for adaptive governance of blue-green solutions.

Additionally, the promotion of governance structures that facilitate implementation of blue-green solutions are proposed to overcome organisational challenges (Li et al., 2020). This could be achieved by implementing a two-tier governance system based on hydrologic districts (Dhakal & Chevalier, 2017) or by establishing watershed partnerships or a single organisation responsible for water management at the catchment level that support municipalities and increase interaction between stakeholders (Carlson et al., 2015; Rooney & Gill, 2018). Fostering a decentralised governance strategy characterised by openness, accountability, and transparency is suggested (Dhakal & Chevalier, 2017; Li et al., 2020) and has been recognised as fundamental to achieve effective stormwater management in the USA and Australia (Rooney & Gill, 2018). Active NGO and community participation and interaction with robust local government institutions are essential for effective, long-term stormwater management (Carlson et al., 2015; Rooney & Gill, 2018). To achieve the desired co-benefits, it is important to involve both private and public stakeholders in formulating the goal of the intervention and in co-designing the solution (Ossa-Moreno et al., 2017).

Based on workshops, interviews and a survey with Swedish planners and water managers, Bohman et al. (2020) identified institutional changes needed to facilitate planning for more sustainable stormwater management. One challenge they identified was that from a traditional planning perspective, stormwater management is still largely seen as a site-specific technical issue to be addressed by engineers and water professionals within the boundaries of each local plan, an approach that works well for managing grey solutions. In order to instead adopt the landscape perspective needed for successful management of blue-green solutions, Bohman et al. (2020) suggest that watershed analysis should be institutionalised, making the assessment of stormwater risks, needs and benefits at a watershed level a mandatory step and the basis for comprehensive and local planning. They argue that this will make stormwater concerns more visible in negotiations over strategic land use decisions and therefore more easily integrated into urban planning. They also saw a need to develop cross-sectoral working practices in relation to blue-green solutions, as working in silos leads to unclear responsibilities and mandates, and budgets being limited to sectoral investments. As a solution, Bohman et al. (2020) suggest transferring leadership and funding for joint sustainable stormwater solutions from sectoral bodies to a central city office with a mandate to unite actors around common goals and distribute funds for earmarked projects. They also saw a strong need for vertical alignment within the municipal organisation to ensure that the visions related to stormwater management in the comprehensive plan are integrated into local plans, implemented in the construction phase and monitored in the long-term management phase. To achieve vertical alignment, Bohman et al. (2020) suggest integrating drinking water, wastewater and stormwater plans to avoid misaligned strategic objectives, establishing vertical interaction networks to facilitate learning and dialogue, and institutionalising evaluation of how planning intentions have been implemented in practice, including the allocation of economic resources for this activity.

The needed increased collaboration across stakeholder groups can also be organised as Integrated Urban Water Management, also referred to as the 'One Water' approach, which means that all parts of the urban water cycle, including urban water supply, sanitation, stormwater and wastewater, are considered as an integrated system and managed holistically, rather than separately, to achieve sustainable economic, social and environmental goals (Kirshen et al., 2018). If managers treat each part of the urban water system separately when adapting to climate and societal change, there is a risk that the strategies they implement will worsen the performance of other parts of the system, whereas integrated management can reduce overall adaptation costs and provide co-benefits at scale (Kirshen et al., 2018). Wider adoption of the One Water approach could be achieved through increased data sharing between water utilities and the use of new technologies, such as artificial intelligence, to promote better data analysis and improve decision-making to minimise energy costs, conserve water resources and better organise staff (Pokhrel et al., 2022). In addition, visionary leadership at different levels of government has been identified as a key driver for the implementation of the One Water approach (Pokhrel et al., 2022).

Social barriers can be handled in different ways. To overcome safety concerns related to blue-green solutions in play and learning environments, it is crucial to integrate diverse expertise in the design process. Collaboration between engineers, landscape architects, public health representatives and horticultural experts can lead to the development of appropriate design solutions (Cizek & Fox, 2015), such as designing a series of bioretention cells upstream of play and learning areas where pollutants have been removed upstream, with children only having access to the most downstream bioretention cell with treated stormwater. Another example is to limit the maximum flooding depths associated with some blue-green solutions with permanently ponded water, such as wetlands, to safe thresholds. If the deepest part of a wetland does not exceed 45 cm, wetlands should not be a concern in public urban green spaces or primary schools, according to a study by Cizek & Fox (2015).

To achieve equity goals in stormwater management, there is a need to adopt more holistic approaches to stormwater management that allow individuals and communities to better advocate for their goals and desires in decision-making related to sustainable stormwater management and blue-green solutions (Wilfong et al., 2023). Particularly in neighbourhoods where local residents resist any form of urban development, for example because they want to preserve the current character of the area, it is important to involve local residents in all stages of planning, design, construction and management, so that blue-green solutions can be adapted to the needs of different stakeholder groups (Montalto et al., 2013). To prevent the implementation of blue-green solutions from leading to the green gentrification of low-income neighbourhoods, local authorities can use planning tools such as the GI Equity Index (Heckert & Rosan, 2016) to identify neighbourhoods at risk of gentrification, combined with 'place staying efforts' such as the promotion of affordable housing or property tax relief for low-income homeowners (Heckert & Rosan, 2018).

C4.8 (RQ4) What conflicts between involved stakeholders exist?

In addition to the barriers mentioned above, conflicts between different stormwater stakeholders can also hinder the successful implementation and management of blue-green solutions. Stakeholders with fixed mindsets are one source of conflict, as stormwater management responsibilities are spread across a variety of agencies and individuals (Fryd et al., 2013; Cousins, 2018). Conflicts between different stakeholders are seen mainly because there are different ways of thinking and acting in relation to sustainable stormwater management. In this section, we map existing conflicts between stakeholders related to sustainable stormwater management and how these can be resolved.

In particular, conflicts arise when discussing economic models and definitions of economic efficiency and inefficiency of blue-green solutions. The main focus of such debates has been described by Madsen et al. (2017) as whether sustainable stormwater management systems can manage cloudburst events or, on the contrary, only small and frequent stormwater events, resulting in disagreements between stakeholders on whether or not to include blue-green solutions in cloudburst mitigation strategies. In densified cities, the desire for high urban land use may conflict with the space needed for stormwater management, depending on the frequency of flooding (Kvamsås, 2023). Decision-makers must therefore decide how to allocate resources, and this often leads to conflicting choices, as the level of flood mitigation is influenced by the selected and diverse sites within the city (Cousins, 2017).

Stormwater stakeholders in a study by Cousins (2017) were found to belong to one of two identified knowledge groups, with one group believing that sustainable stormwater management should be achieved through stricter laws and regulations, coupled with more scientific and data-driven approaches, while the other group saw new institutions and regulations to promote integrated management approaches, as well as economic instruments capable of assigning a value to stormwater, as the solution. Such conflicting and contrasting perspectives lead to different views on whether green or grey infrastructure is the preferred choice for stormwater management. This may hinder truly collaborative sustainable stormwater management, as collaboration remains situated within the possibilities of engineering science in technocratic environments (Cousins, 2017).

Despite an ambitious process, local residents involved in the development of a stormwater management plan in Finland ended up disagreeing on how to deal with the stormwater problem; the consultants and municipal managers suggested building retention ponds in the public urban green space to comply with the authority's planning limits, while the residents suggested locating bio-filtering elsewhere, upstream and outside the green space (Vierikko & Niemelä, 2016). Vierikko & Niemelä (2016) argue that the reason why the different stakeholders proposed different stormwater management solutions was that they ascribe different values to blue-green solutions. Such conflicts related to interests and concerns, about current and future land use, were also described by Fryd et al., (2013). In their case, use of private property was a part of an overall solution, which added to the complexity of the potential implementation. In line with Vierikko & Niemelä (2016), they pointed out that there is an increasing risk of environmental conflicts arising in relation to management, and therefore socio-cultural values need to be taken into account when implementing stormwater strategies based mainly on techno-ecological information. A study in New York City showed that it can also lead to conflict when the municipality invests and focuses exclusively on blue-green solutions for stormwater management, with designs developed to maximise stormwater capture, when residents, on the contrary, value the many co-benefits more than the actual stormwater management (Miller and Montalto, 2019).

Further, construction expenses and transaction charges provide a significant conflict issue, as smaller (private) facilities tend to be less cost-effective than larger facilities (Madsen et al., 2017), which makes them less appealing from a planning standpoint for private property owners to implement (Fryd et al., 2013). The present regulation system in e.g. Denmark provides an inefficient incentive scheme that makes households reluctant to implement blue-green solutions on private properties (Fryd et al., 2013).

Responsibility for maintenance is an issue that is often at the heart of many conflicts related to the long-term management of blue-green solutions. Public action resistance groups in Trondheim, Norway, cited a lack of funding for maintenance as a primary concern for the implementation of future projects (Thodesen et al., 2022). In particular, within distributed systems, which often also have distributed management, conflicts can arise due to a lack of clarity about who is responsible for different aspects of maintenance (Thodesen et al., 2022).

C4.8.1 Resolving conflicts between stakeholders in sustainable stormwater management

The general lack of consensus on the development, implementation and long-term management of blue-green solutions may pose a challenge to integrated governance approaches for many cities, and it is therefore important to make these differences empirically visible so that stakeholders can engage more constructively (Cousins, 2017). With many cities around the world facing similar sustainable stormwater management challenges, further research into how grey and blue-green approaches complement each other in addressing water quality, water quantity and providing co-benefits may help to resolve some of the disagreements between different stormwater stakeholders (Cousins, 2017). There are a number of research methods that could facilitate bringing the different perspectives of different actors to the fore, and mapping the different values they place on blue-green solutions. The *Q-methodology* provides a tool for identifying areas of conceptual agreement and disagreement that can inform policy (Cousins, 2017). Q-methodology (or Q-sort) is a research method often used in the social sciences to assess different points of view. This is achieved by bringing different participants to a workshop where they work in groups to sort and prioritise a number of claims, called Q-sets (see e.g. Buffam et al., 2022). Vierikko & Niemelä (2016) suggested that *integrated value mapping* can improve understanding of the different perspectives and needs of different actors. Here, different ecosystem service values are linked together, but not reduced to a single metric. Based on their case study, they suggested that careful value mapping should be undertaken to identify the socio-cultural values of key stakeholders (demands) and define the associated ecosystem services (supply) before the actual stormwater management plan is developed. One way to obtain stakeholder values is through interviews, as these are considered to be better at identifying negative values than surveys or value scaling (Vierikko & Niemelä, 2016).

Different governmental actors, e.g. in different municipal departments, and non-governmental actors, e.g. local residents, NGOs or private stormwater consultants, place different values on blue-green solutions and bring different important perspectives to the issue of sustainable stormwater management. This wide range of perspectives needs to be taken into account. A strong place identity among citizens supports the achievement and maintenance of the resilience of socio-ecological systems, because place identity increases environmental awareness, which in turn supports pro-environmental behaviour and improves self-esteem and feelings of belonging to society or a cultural group (Madsen et al., 2017).

Municipalities need to show that they take the values identified by local stakeholders into account. In a study in which New York City residents were found to appreciate co-benefits more than the actual stormwater management, the authors conclude that municipal programmes that emphasise the creation of multifunctional blue-green solutions, and in particular the locally valued co-benefits they provide, may gain broader societal acceptance than those that focus solely on stormwater management (Miller and Montalto, 2019).

Establishing sustainable stormwater management networks can also be a way to increase implementation of blue-green solutions. The 'climate adaptation networks' described by Thodesen et al. (2022) consisted of partnerships between government, academia and industry. These were set up at the national level to raise awareness at the municipal level of how cities could deal with the challenges of climate change. Municipalities involved in these networks undertook significantly more adaptive planning and implemented more measures than those not involved in such networks.

Negotiating diverse perspectives on institutional and infrastructure interventions, bridging political and jurisdictional gaps, developing financial mechanisms, and integrating fragmented governance structures is crucial for success (Kvamsås, 2023). However, Madsen et al. (2017) also note that departmental silos may not adequately explain differences in stakeholder perspectives. Instead, their findings demonstrate the need to forge (mis)connections between different stakeholder groups through shared ways of constructing solutions to a problem, thus enabling and constraining technical and institutional interventions. Here, long-term management responsibilities can be established and clarified in the planning stages to avoid conflicts regarding lack of funding for maintenance and to gain clarity about who is responsible for the different aspects of long-term maintenance (Thodesen et al., 2022).

C5. Conclusions – Avenues towards increased societal acceptance of sustainable stormwater management

C5.1 RQ1.What actors and stakeholders are involved in sustainable stormwater management?

Historically, urban drainage as a discipline was part of civil engineering. Conventional stormwater management originally involved stormwater engineers in the municipal water department and aimed to deal with flood mitigation only, using one primary stormwater management solution: underground pipes. This resulted in high degree of overview and control for the municipality (Figure C9). In recent decades, however, societal interest in urban stormwater management has increased. Sustainable stormwater management typically involves planners and managers from municipal departments such as Water and Sewerage; Urban Environment, Parks and Recreation; and Planning and Building as well as private consultants with expertise in, e.g., construction; planning; engineering; landscape gardening and water and sewerage. It is performed to reach multiple aims and manage stormwater while providing a wide range of co-benefits to stakeholders, striving to create multifunctional blue-green solutions and spaces. Achieving this is not about a single solution, but rather choosing from a range of different solutions, adapting them to the local site and context, and choosing the right solution depending on the desired co-benefits from the site. This still relatively new reality implies a lower degree of overview and control for the municipality. It is not straightforward to measure the performance of blue-green solutions in terms of flood mitigation and pollutant removal and to get an overview of implementation and management costs, maintenance requirements and co-benefits. This is also reflected in the many barriers and conflicts identified in this study that hinder the successful implementation and management of blue-green solutions.

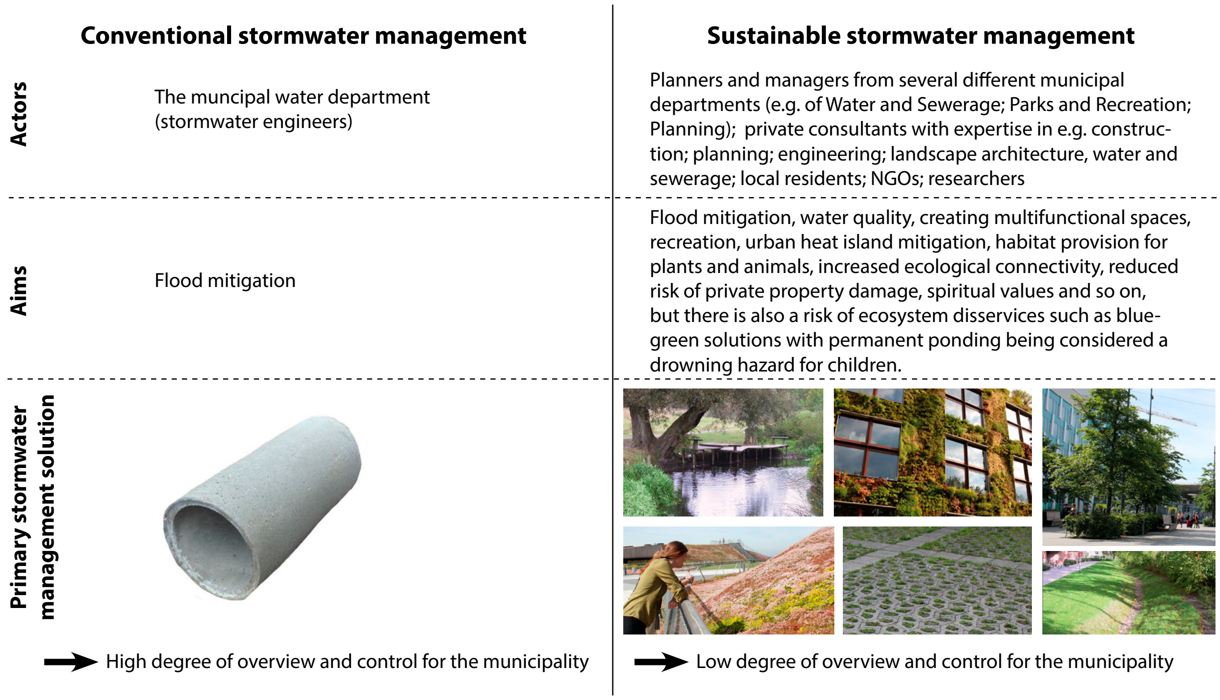

Figure C9. Sustainable stormwater management is more complex than conventional stormwater management and involves multiple actors, aims and types of physical stormwater management solutions.

While few of the reviewed articles describe processes of stakeholder participation specifically in the development of blue-green solutions, stakeholder participation in the development of public urban green spaces in general is more widely studied. This calls for a shift in both practice and research towards viewing the implementation and management of blue-green solutions as part of the governance and management of urban green spaces, rather than as a separate issue. In this way, lessons learnt from studies on green space management in general (e.g. how to conduct participation processes and overcome barriers and challenges) can be applied to sustainable stormwater management.

The lack of Swedish studies on participatory sustainable stormwater management suggests that local residents are an untapped resource for the long-term management of existing blue-green solutions in Sweden. Inspiration for increased stakeholder participation in Sweden in the future can be found in the reviewed studies from the USA, UK, Finland, Denmark and Belgium, where residents participate through Rain Check and other educational programmes; by installing rain barrels and downspout planters to manage water on their property; by managing blue-green solutions on public land, for example by volunteering as Green Street Stewards and maintaining rain gardens along streets or through programmes such as Adopt-a-Street and Adopt-a-Drain; and through citizen-based environmental monitoring of, for example, water quality.

C5.2 RQ2. Which co-benefits are created through sustainable stormwater management according to different actors and stakeholders?

Different stakeholders attach different values and benefits to blue-green solutions. These are important to understand in order to create socially inclusive blue-green solutions. Municipal managers tend to emphasise general and holistic values such as which solution is best for the whole city, the environment or in the future, while local residents tend to emphasise place-specific values related to the use and experience of the blue-green solution. While municipal managers see stormwater management as the primary driver for investment in blue-green solutions, local residents may value co-benefits more than actual stormwater management. While these findings provide some guidance on the aspects that need to be considered when implementing blue-green solutions, the general public is itself a diverse stakeholder group. Therefore, in order to take full account of local needs and perspectives, it may be beneficial for local authorities to involve different stakeholder groups, and especially marginalised groups, in the planning, design, construction and management of blue-green solutions. While we found several studies on what co-benefits local residents value, the needs and perspectives of traditionally marginalised groups in relation to blue-green solutions have not been studied, with the exception of the marginalised groups of children (Cizek & Fox, 2015), Latin American communities in disadvantaged neighbourhoods (Meenar, 2019), and local residents from different socio-economic and cultural groups (Heckert & Rosan, 2018). What values that are ascribed to blue-green solutions also depend on whether the person is a sustainable stormwater management champion (i.e. driving and promoting implementation) or not.

Several factors affect which co-benefits that are provided from blue-green solutions and valued by people, including who, where and when you ask, since this varies between stakeholder groups, regions and over time. It is also important to identify which co-benefits are needed at a site and then select a blue-green solution that provides these co-benefits in order to select the most appropriate type of blue-green solution for the place in question and to deliver co-benefits in neighbourhoods where they are most needed. In addition, what co-benefits that are created and appreciated depends on the blue-green solution in question. Different blue-green solutions differ in their ability to provide specific co-benefits. The study by Elliott et al. (2020) showed that public urban green spaces, wetlands and community gardens were the three most beneficial types of blue-green solutions in terms of the number of ecosystem services provided. Conversely, the lowest ranked blue-green solution types were rain barrels, permeable paving and vacant land. This variation means that it is important to differentiate between different types of blue-green solutions in planning frameworks that aim to deliver co-benefits in addition to stormwater management.

Focusing on the provision of co-benefits when designing blue-green solutions, rather than just stormwater management, can make the design more attractive to local residents and increase societal acceptance of sustainable stormwater management. In addition, considering co-benefits rather than just flood reduction when assessing the economic benefits of blue-green solutions significantly improves the economic viability of implementing blue-green solutions.

C5.3 RQ3. What are the underlying aims and incentives for sustainable stormwater management?

Sustainable stormwater management is driven by the desire of municipalities to plan blue-green solutions in a holistic way and their ambitions to increase environmental sustainability by contributing to urban biodiversity, supporting ecosystem services and climate adaptation; social sustainability by creating multifunctional spaces for recreation and improving bathing water quality of receiving waters, spaces important for public health; and economic sustainability. However, despite these aims, the implementation of blue-green solutions is still slow.

If the many desired objectives of sustainable stormwater management are to be achieved, several different perspectives need to be considered simultaneously when deciding how to implement and manage blue-green solutions. Like the governance and management of public urban green spaces in general, the governance and management of blue-green solutions needs to be place and context specific to create blue-green solutions that both manage stormwater and provide co-benefits. This involves taking into account a range of aspects related to both the physical conditions of the site and the needs of local residents and other stakeholders. The fact that blue-green solutions, unlike underground pipes, are visible to local residents means that the appearance of the system needs to be a priority, as public acceptance and involvement can be affected by negative perceptions of implemented blue-green solutions.

C5.4 RQ4. What barriers and challenges to sustainable stormwater management and what conflicts between its stakeholders exist? How can these be overcome and resolved for increased societal acceptance of sustainable stormwater management?

Sustainable stormwater management is hindered by a lack of supportive policies, lack of documented knowledge on the benefits of blue-green solutions, and lack of physical space for blue-green solutions in densified cities. Further, insufficient funds at national, regional, and local government level hinders implementation and not the least long-term management of blue-green solutions, which often lacks sufficient funding. Land ownership may also be a barrier, with the lack of access to privately owned land being the most prominent. Organisational responsibility for sustainable stormwater management is spread across a number of different institutions and actors, making increased collaboration and knowledge sharing crucial for increased implementation of blue-green solutions. Another challenge is the current equity issue, where marginalised groups often do not have access to blue-green solutions and their benefits, and this needs to be addressed in the planning and implementation of blue-green solutions.

To overcome the identified barriers, there is a need for changed stormwater regulations that approach stormwater as a 'resource' rather than a 'hazard', and that are more adaptive and less reliant on quantitative outcomes to allow for more integrative, collaborative planning and decision-making. In practice, this implies viewing blue-green solutions as a resource through economic valuation tools that monetise their primary and co-benefits; seeing blue-green solutions as insurance, as well-functioning GI could help cities avoid the costs of floods and extreme events and build resilient cities; and discursively redefining stormwater as a resource (e.g. as a water supply issue) in laws and regulations to legally recognise the benefits of stormwater (Cousins, 2018). To transform society's relationship with stormwater and realise its value as a resource, new forms of public participation in sustainable stormwater management are needed. Adopting holistic approaches to sustainable stormwater management that enable communities to participate in stormwater decision making, better advocate for their goals and aspirations, and interact with robust local government institutions is essential for effective and equitable long-term stormwater management.

Establishing new evaluation criteria and standards for measuring performance of blue-green solutions that also include current and long-term co-benefits is vital both for increased societal acceptance of blue-green solutions and to convince owners and contractors to invest. Economic incentives such as cost-sharing agreements, subsidies for installing blue-green solutions, or lower water fees for property owners with fewer impervious surfaces can motivate private stakeholders to implement more blue-green solutions. Cost calculations and performance assessments of different blue-green solutions can help to decide on the right blue-green solution for the right place, as different solutions will perform differently depending on local physical and social site conditions and needs, and where in the catchment they are located. These factors make spatial prioritisation of blue-green solutions crucial to achieving the desired primary and co-benefits, and such decisions require a landscape perspective.

Above all, there is a need to improve communication and collaboration between urban stormwater stakeholders. Different departments within local authorities are more likely to achieve successful implementation and long-term performance of blue-green solutions if they overcome departmental silos and get, for example, green space planners and managers, landscape architects, water engineers and public health experts working together, each contributing with their expertise to a holistic approach to stormwater implementation and management. In addition to communication and collaboration across municipal departments, effective stormwater governance requires experimentation and a strategy for organisational learning, and that these three elements of governance are built into the planning process for blue-green solutions. Successful implementation of blue-green solutions often needs to be preceded by soft actions, such as regulatory changes, that lead to the societal and institutional changes necessary for increased implementation of blue-green solutions.

It is also important to get everyone on board from the start. This implies involving all stakeholders from the beginning of the planning process. It is particularly important to involve maintenance staff and establish long-term management responsibilities at the planning stage to build support and avoid conflicts over lack of funding for maintenance of blue-green solutions, and to clarify who is responsible for different aspects of their maintenance.

Visionary government leadership at local, regional and national levels is central to successful sustainable stormwater management, with authorities supporting the process as drivers, coordinators and capacity builders. Interdepartmental collaboration is an ongoing process that needs to be prioritised for success. To overcome municipal departmental silos, leadership and funding could be transferred from sectoral departments to a central city office or person who takes on the role of a 'bridge builder'. Another solution could be the creation of catchment partnerships or a single organisation responsible for water management at the catchment level to support municipalities and increase interaction between stakeholders, or even bridge the urban-rural divide in stormwater management. The necessary increased cooperation between stakeholders can also be organised through the 'One Water' approach, which means that all parts of the urban water cycle, including urban water supply, sanitation, stormwater and wastewater, are considered as an integrated system and managed holistically, rather than separately, to achieve sustainable economic, social and environmental goals. There is also a need for vertical alignment within the municipal organisation, which can be achieved, for example, by establishing vertical interaction networks to facilitate learning and dialogue, and by institutionalising evaluation of how planning intentions have been implemented in practice, including the allocation of economic resources to this activity.

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Urban stormwater research – An evidence synthesis

Development of a holistic understanding of current technical, environmental and social/institutional knowledge with regard to urban stormwater research

This report presents the findings of a review of the international peer-review literature on the impacts of stormwater runoff on receiving water recipients from a cold climate perspective, the design, treatment performance and maintenance requirements of sustainable stormwater management systems, and the incentives for and societal acceptance of their use. Key findings are mapped against the requirements and policy objectives set out in the 2024 EU Urban Wastewater Treatment Directive, the UN Sustainable Development Goals (2015) and the Swedish environmental goals (2000). The report concludes with a series of research and policy recommendations which aim to support Sweden in its transition to a stormwater management approach that embraces opportunities to address stormwater quality, quantity and societal well-being objectives. This review is financed by the Swedish Environmental Protection Agency and the Swedish Agency for Marine and Water Management.

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