Stormwater Pond Literature Review

Draft Final Report

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Acknowledgments
We thank Megan Kocher and Julie Kelly from the University of Minnesota Libraries for their excellent guidance in this project and especially for the help in designing the search terms and setting up the literature review in Covidence.
I. Introduction

Impervious surfaces in urban environments prevent rainfall from infiltrating into the soil and, in turn, dramatically increase the volume of surface runoff generated during storms (Burns et al. 2005, Hancock et al. 2010). Urban environments are also major sources of sediment and chemical pollutants that can be transported to receiving waters via stormwater runoff, degrading water quality. Stormwater ponds are one of the most common structural stormwater control measures (SCMs, also known as stormwater best management practices or BMPs) used to mitigate the adverse effects of development on urban hydrology and water quality (Anderson et al. 2002, Song et al. 2015, Blecken et al. 2017). Ponds are a prominent feature of newly developed land (< 30 years) in the United States, with states like Georgia, the Carolinas, and Minnesota containing thousands of stormwater ponds (Blecken et al. 2017, Beckingham et al. 2019, Taguchi et al. 2020a). The prevalence of stormwater ponds will likely increase as states work to comply with Total Maximum Daily Load (TMDL) requirements for pollutant load reductions and to manage increased stormwater volumes associated with urbanization and climate change (Moore et al. 2016).

Stormwater ponds designs typically consist of an impervious depression in the landscape containing inlet and outlet structures, which together maintain a permanent pool of water and a reservoir for the temporary storage of runoff volumes (Al-Rubaei et al. 2017). The initial management goal of stormwater ponds was to maintain on-site peak flow rates at pre-development levels by capturing stormwater and slowly releasing it into streams and storm drain networks (Emerson et al. 2005, Hancock et al. 2010). The temporary detention of runoff in stormwater ponds can prevent urban stormwater from entering drainage networks all at once, which would otherwise cause surface flooding, stream erosion, and combined sewer overflows.

Stormwater pond designs were later modified to also manage water quality by promoting sedimentation of suspended solids in stormwater (Starzec et al. 2005). Sediments can increase water turbidity and thus inhibit plant growth and increase water temperatures. They can also blanket spawning grounds and reduce aquatic biodiversity (Shammaa et al. 2002). While sediments are aquatic pollutants themselves, they are also major vectors of nutrients, heavy metals, pathogens, pesticides, and other contaminants, which can bind to sediments and be transported in runoff. The ability of ponds to settle out sediments and sediment-bound pollutants allows them to address a variety of water quality concerns.

Stormwater ponds are not specifically designed to manage dissolved pollutants, but they can support organisms that can process dissolved chemicals. For example, ponds often contain diverse plant, algal, and microbial communities that can uptake and immobilize nutrients and metals. They can also support bacterial species, such as denitrifiers, that can permanently remove nitrate from stormwater by converting it to dinitrogen gas. The capacity of stormwater ponds to provide physical removal of particulate pollutants and biological treatment of dissolved pollutants has led to them to be considered an effective SCM for water quality improvement. Some municipalities require the use of stormwater ponds to treat runoff from all development projects that exceed certain areal coverage (Goff and Gentry 2006), and ponds are increasingly seen as a key tool in mitigating future stormwater challenges.

Despite their theoretical potential, ponds have exhibited variable hydrologic and water quality performances in the field. For example, studies investigating catchment-scale impacts of pond
implementation have often failed to observe significant reductions in peak flow rates (Emerson et al. 2005, Aulenbach et al. 2017). In some cases, ponds have actually increased peak flow rates in receiving waters (McCuen 1979, Aulenbach et al. 2017), raising fundamental questions about the efficacy of ponds for hydrologic control. Widespread implementation of ponds have also failed to improve water quality in some catchments (Gold et al. 2017). Moreover, ponds can display a wide diversity of biogeochemical processes which, in some cases, can promote the transformation and release of aquatic pollutants (Song et al. 2013, Taguchi et al. 2020a). Often, the processes that underlie effective pond functions are poorly understood leading to uncertainty in guidance for management and retrofit (Janke et al. 2022). Finally, although ponds can provide natural habitat for some organisms, they can function as ecological traps for others, and may promote spread of invasive species (Clevenot et al. 2018). As these examples suggest, it is not yet clear where and how stormwater ponds are providing services they were intended to perform, or how they influence other unintended services and disservices.

The large variability in pond performance observed in field studies suggests that environmental factors interact with pond designs to influence specific hydrologic and water quality outcomes. These relationships are as yet poorly known. While the indiscriminate or unguided use of ponds to manage all stormwater problems may lead to poor outcomes (James et al. 1987), improved design for new ponds and modification of the many existing aging ponds can lead to better performance outcomes. Useful information can be extracted from a rapidly growing research to more effectively inform pond design and management. The purpose of this review is to provide a systematic assessment of the scientific literature investigating the effectiveness of stormwater ponds with respect to multiple performance measures. The review summarizes the research that has been conducted on stormwater ponds to date to provide a synthetic understanding of pond functioning and to identify critical knowledge gaps where further research is needed. The four specific questions this review address include:

1. Are stormwater ponds functioning as designed and intended? Are they providing rate control, flood control, energy dissipation, and the removal of sediments and sediment-bound pollutants?
2. Are stormwater ponds performing other important functions or responsible for unintentional disservices that detract from their designed and intended purpose?
3. How do we accurately assess stormwater ponds to address Questions 1 and 2? Are current assessment protocols accurate or is research needed to develop accurate protocols?
4. Can stormwater ponds be redesigned and modified to better serve their original, current, and future objectives?

II. Methodology

Scope

This literature review is global in scope and therefore considered all peer-reviewed literature that has been published on stormwater ponds in the last 50 years. The review is primarily interested in designed and constructed stormwater ponds, which differ from the many natural ponds and wetlands that have been retrofitted to provide stormwater management. However, given the potential paucity of research on designed and constructed ponds, their similarity in function to
retrofitted natural ponds, and the fact that papers rarely indicate the exact nature of the study ponds, we took a relatively inclusive approach that is detailed in sections below.

**Approach**

The literature considered in this review was obtained from four citation databases: the Web of Science, CABI, Engineering Village, and Agricultural & Environmental Science Database. A search query (Appendix A) was developed to identify papers that contained specified keywords in their titles and abstracts. This search query yielded approximately 5,400 papers from the four databases. Exclusionary terms were not used in the search query to avoid the possibility of inappropriately omitting highly relevant papers.

We used Covidence (www.covidence.org), a systematic review management tool, to screen papers outside the scope of the literature review and to reduce the total number of papers from 5,400 down to a reasonable number. We conducted two rounds of sequential screening in Covidence with each paper receiving two reviewers. In Round 1, reviewers voted Yes or No to papers they wished to include or exclude, respectively, based solely on reading paper titles and abstracts. Papers that received two Yes votes moved on to Round 2, while papers that received two No votes were excluded from the review. Papers that received one Yes and one No vote were flagged as conflict papers, requiring reviewers to manually decide whether they should be included or excluded from Round 2. In Round 2, reviewers had access to the full text of papers and were again required to vote Yes or No on papers. However, decisions to exclude papers in Round 2 required a rationale. Papers that received two No votes in Round 2 were excluded from the review, while papers that received two Yes votes were identified as papers to be considered in the review.

Two screening tools were used to determine whether to include or exclude papers in Covidence. The first screening tool was a decision tree (Appendix B) in which reviewers were asked to answer a series of questions to determine whether to include or exclude a paper. However, this decision tree was fairly general, so a list of additional exclusion criteria (Appendix C) was developed to further screen papers deemed outside the scope of the review. The decision was made to include papers on certain topics that are relevant to stormwater ponds, but were deemed of lower priority to this particular review (e.g., the impact of ponds on biodiversity, floating treatment wetland technologies). These topics contributed a substantial number of papers to the final paper count but did not receive significant consideration in this review.

The final list of papers was organized into a spreadsheet, and the contents of the papers were summarized. The purpose of this review was to assess the number of studies that have focused on a particular stormwater pond topic and to summarize broad performance trends and mechanistic drivers. We did not set out to conduct a meta-analysis, so quantitative metrics on pond performance were not recorded or analyzed in this review. However, we did leverage existing databases on pond performance to provide quantitative metrics on pond performance when available.

This review was limited to assessing peer-reviewed literature and did not consider other sources of information. There are many gray literature studies that contain very important data on pond performance, but this literature was too vast and too varied to be systematically evaluated in this review. Limiting the scope of the review to peer-reviewed articles could introduce bias.
Therefore, additional gray literature reviews may be needed to determine whether the performance trends identified in gray literature align with those of the peer-reviewed literature.
III. Results

**Literature Review Summary**

In total, 593 peer-reviewed studies were used for our literature review. Of the 593 peer-reviewed studies, 184 studies were classified as low priority and beyond the scope of this exercise and were therefore not considered in the following analysis. The omitted studies included studies on biodiversity (115), floating treatment wetlands (48), public perceptions of stormwater ponds (8), stormwater treatment trains (8), economic factors (3), and optimal BMP placements (2). The remaining 409 peer-reviewed studies were published between 1975 and 2022. We considered studies on rate control, flood control, energy and erosion, and total or volatile suspended solids to be Question 1 papers. Studies on phosphorus, nitrogen, metals, chlorides, polycyclic aromatic hydrocarbons (PAHs), heat accumulation, and bacteria and pathogens were considered Question 2 papers.

Figure 1 shows that publications on stormwater ponds are increasing, with the majority of papers published in the last 10 years. Of the 409 included peer-reviewed studies, 68 studies pertained to Question 1 and 255 papers pertained to Question 2, indicating a large focus has been placed on water quality functions and performance of stormwater ponds. Eighty-six studies contained data on both categories. The number of studies by topics of interest along with the type of study (laboratory, field, modeling, other) are illustrated in Figure 2. A greater proportion of Question 1 papers were modeling papers, with more than 50% of rate control, volume control, and energy and erosion papers containing pond or watershed-scale models. By contrast, most of the water quality (Question 2) papers consist of field studies, especially for the slightly less common topics (chloride, PAHs, and bacteria/pathogens).

**Literature Review Outputs: Spreadsheet Database and Reference List**

The complete list of papers making it to Round 2 of the screening process can be found in an included reference list and a spreadsheet database, with the latter summarizing various information about the papers (e.g., study type, hydrologic or water quality variables investigated, geographic metadata, etc.). The lists include papers selected for the final review as well as those excluded for reasons described in the Methodology above (typically due to out-of-scope topics). The spreadsheet was not filled out exhaustively but should provide a valuable tool for locating references on specific topics or methodologies.
Figure 1. Total number of journal papers published every 5 years by subject. Question 1 contains papers about rate, flood, volume, and energy and erosion control as well as total and volatile suspended solids ($n = 68$). Question 2 contains papers about phosphorus, nitrogen, metals, polycyclic aromatic hydrocarbons (PAHs), chlorides, and bacteria and pathogens ($n = 255$). Papers that contained data from topics in both Question 1 and Question 2 were categorized as “Both” ($n = 86$).
Figure 2. Total number of journal papers reviewed by study type for each section. Papers that included data or information for more than one section (e.g. Phosphorus and Nitrogen) are included in all relevant sections. Field studies contain papers that are field studies or field studies with associated models. Lab studies contain papers that are solely lab studies or lab studies with associated models. Model studies are studies that were categorized as pond or watershed-scale model studies. The category for “other” contains meta analyses, reviews, remote sensing, and uncategorized papers.

Existing Pond Performance Summaries

Although we noted in our review the inclusion of particular types of performance outcomes assessed (e.g., removal efficiencies, volume reductions), it would have been well beyond the scope of this review to provide additional quantitative assessments of those performance outcomes. Methods of assessment in the references varied considerably (see section on Current Pond Assessment Methods, page 48), including both direct observations and completely- or partially-modeled results, and performance metrics included a range of parameters such as concentration reduction, mass reduction, sediment burial, peak flow attenuation and delay. Therefore, a comprehensive quantitative summary or meta-analysis was not feasible here, though we have made efforts to summarize general performance outcomes for many of the pollutants and hydrologic factors. In lieu of a meta-analysis, we highlight here several studies which included meta-analyses of pond performance or indicator data.
Performance summary from the International BMP Database

As a baseline, we first present the performance summary of the International BMP Database prepared by Clary et al. (2020). From that report, we have extracted pollutant reduction metrics for nutrients, metals, sediment, and fecal indicator bacteria, for the two SCM types relevant to our review (retention ponds, and wetland basins; results shown in Table 1). Clary et al. (2020) defined retention ponds as those with permanent pools, and wetland basins as retention ponds (permanent pool) with more than 50% surface coverage by emergent vegetation. A detailed description of the methods used in the summary can be found in Section 1 of the Clary et al. (2020) report, but a few details are worth noting. In particular, they note that SCMs required both influent and effluent data, and they excluded grab samples (except for bacteria samples) as well as baseflow samples. We calculate performance in terms of reductions in median concentrations at the far right column of Table 1; note that this is NOT the same as the median reduction performance of the ponds, nor is it pollutant mass reduction.

Selected Pond Meta-Analyses

We also include a subset of studies from the literature review that focused on meta-analyses of a large number of ponds (> ~50). We have made no effort to summarize the results across these studies, but highlight them as potentially useful for future research or for readers interested in studies with greater spatial scope.

- **Blaszczak et al. (2018):** a study of denitrification in 64 ponds across 8 U.S. cities; data collection primarily focused on nitrate, sediments, chloride, and metals.
- **Chiandet et al. (2016):** investigated 50 natural and constructed ponds in southern Ontario (Canada) and included a large data collection effort for nitrogen, phosphorus, dissolved oxygen, pond morphology, and watershed characteristics.
- **Gallagher et al. (2011):** sampled 68 ponds in the Baltimore metropolitan area, including water column and sediment, investigated metals, PAHs, and chloride.
- **Grauert et al. (2012):** sampled 70 Danish ponds for metals and PAHs in sediments.
- **Koch et al. (2014):** investigated N removal in several pond types using a literature meta-analysis, with published data from 72 wet detention ponds.
- **McEnroe et al. (2013):** profiles of temperature, conductivity, and dissolved oxygen, and water column N, P, and TSS for 45 ponds in southern Ontario (Canada).
- **Perron et al. (2020):** a study of dragonfly and damselfly communities in 41 stormwater ponds across Ottawa (Canada), including measurement of water quality variables (nutrients, metals, water chemistry).
- **Scheffers et al. (2013):** investigated 58 ponds in Edmonton (Canada) for amphibians, and collected water samples for nitrogen, phosphorus, and metals.
- **Soenderup et al. (2016):** study of 66 ponds across Denmark for environmental factors (age, pond type, watershed land use) affecting nitrogen, phosphorus, and organic matter.
- **Wakelin et al. (2003):** study of 58 ponds in Winnipeg (Canada), sampling for nutrients, chlorophyll-a, metals, bacteria, and oxygen.
- **Williams et al. (2016):** study of 184 ponds across a natural-urban gradient in the Great Lakes region (USA), focused on dissolved organic matter and anthropogenic effects.
Table 1. Summary of pollutant removal (as concentration reduction) of two pond SCM types in the International BMP Database, as compiled and analyzed by Clary et al. (2020). Included are several forms of N and P, as well as TSS, TDS, E. coli, fecal coliform, and metals (Cd, Cr, Cu, Fe, Pb, Ni, Zn). Interquartile range is 25th - 75th percentile. We report reduction in median concentration at the far right of the table. Trend significance based on three different tests performed by Clary et al. (left is for comparison of 95% confidence intervals; middle position is a Mann-Whitney rank-sum test, \( p < 0.05 \); and right is a Wilcoxon signed rank test; \( p < 0.05 \)). A ‘\(^\wedge\)’ indicates significant increase in pollutant, ‘\(^o\)’ indicates no significant in-out difference, and ‘\(^v\)’ indicates significant reduction in pollutant. For further details, see Clary et al. (2020).

<table>
<thead>
<tr>
<th>Pond Type</th>
<th>No. Studies (Samples)</th>
<th>Concentration</th>
<th>Trend Signif.</th>
<th>% Reduction in Median Conc.</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Median (Interquartile Range)</td>
<td></td>
<td>In</td>
</tr>
<tr>
<td>Total Suspended Solids (TSS), mg/L</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Retention Pond</td>
<td>72 (1199)</td>
<td>74 (1191)</td>
<td>49.0 (15.0 - 150)</td>
<td>12.0 (5.0 - 32.9)</td>
</tr>
<tr>
<td>Wetland Basin</td>
<td>31 (601)</td>
<td>30 (563)</td>
<td>35.5 (14.0 - 89.0)</td>
<td>14.0 (4.69 - 32.0)</td>
</tr>
<tr>
<td>Total Dissolved Solids (TDS), mg/L</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Retention Pond</td>
<td>16 (169)</td>
<td>16 (156)</td>
<td>122 (69.0 - 180)</td>
<td>178 (78.3 - 364)</td>
</tr>
<tr>
<td>Wetland Basin</td>
<td>5 (65)</td>
<td>5 (38)</td>
<td>127 (77.0 - 197)</td>
<td>149 (92.0 - 238)</td>
</tr>
<tr>
<td>Total Phosphorus (TP), mg/L</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Retention Pond</td>
<td>71 (1161)</td>
<td>75 (1138)</td>
<td>0.246 (0.0996 - 0.542)</td>
<td>0.120 (0.0500 - 0.263)</td>
</tr>
<tr>
<td>Wetland Basin</td>
<td>27 (690)</td>
<td>27 (647)</td>
<td>0.170 (0.106 - 0.319)</td>
<td>0.122 (0.0660 - 0.222)</td>
</tr>
<tr>
<td>Soluble Reactive Phosphorus (SRP), mg/L</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Retention Pond</td>
<td>42 (734)</td>
<td>43 (687)</td>
<td>0.0856 (0.0288 - 0.243)</td>
<td>0.0340 (0.0099 - 0.127)</td>
</tr>
<tr>
<td>Wetland Basin</td>
<td>13 (482)</td>
<td>14 (454)</td>
<td>0.0371 (0.199 - 0.0832)</td>
<td>0.0370 (0.0130 - 0.0798)</td>
</tr>
<tr>
<td>Total Dissolved Phosphorus (TDP), mg/L</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Retention Pond</td>
<td>20 (396)</td>
<td>23 (435)</td>
<td>0.129 (0.0700 - 0.212)</td>
<td>0.0642 (0.030 - 0.144)</td>
</tr>
<tr>
<td>Wetland Basin</td>
<td>9 (338)</td>
<td>8 (311)</td>
<td>0.0550 (0.0320 - 0.101)</td>
<td>0.0460 (0.0250 - 0.0815)</td>
</tr>
</tbody>
</table>
Table 1, continued. Nitrogen and Bacteria.

<table>
<thead>
<tr>
<th>Pond Type</th>
<th>No. Studies (Samples)</th>
<th>Concentration</th>
<th>% Reduction in Median Conc</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Median (Interquartile Range)</td>
<td>Trend Signif.</td>
</tr>
<tr>
<td></td>
<td>In</td>
<td>Out</td>
<td>In</td>
</tr>
<tr>
<td><strong>Total Nitrogen (TN), mg/L</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Retention Pond</td>
<td>35 (618)</td>
<td>37 (602)</td>
<td>1.63 (1.05 - 2.66)</td>
</tr>
<tr>
<td>Wetland Basin</td>
<td>14 (471)</td>
<td>14 (477)</td>
<td>1.43 (0.970 - 1.96)</td>
</tr>
<tr>
<td><strong>Total Kjeldahl Nitrogen (TKN), mg/L</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Retention Pond</td>
<td>47 (654)</td>
<td>52 (704)</td>
<td>1.35 (0.820 - 2.30)</td>
</tr>
<tr>
<td>Wetland Basin</td>
<td>15 (188)</td>
<td>17 (274)</td>
<td>1.01 (0.593 - 1.39)</td>
</tr>
<tr>
<td><strong>Nitrate-Nitrite (NO₃), mg/L</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Retention Pond</td>
<td>60 (958)</td>
<td>62 (932)</td>
<td>0.400 (0.160 - 0.771)</td>
</tr>
<tr>
<td>Wetland Basin</td>
<td>22 (561)</td>
<td>22 (523)</td>
<td>0.370 (0.156 - 0.655)</td>
</tr>
<tr>
<td><strong>Ammonium (NH₄), mg/L</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Retention Pond</td>
<td>42 (654)</td>
<td>45 (644)</td>
<td>0.110 (0.0425 - 0.229)</td>
</tr>
<tr>
<td>Wetland Basin</td>
<td>16 (475)</td>
<td>16 (476)</td>
<td>0.0770 (0.0338 - 0.160)</td>
</tr>
<tr>
<td><strong>E. coli, MPN/100mL</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Retention Pond</td>
<td>7 (103)</td>
<td>7 (100)</td>
<td>4110 (854 - 26500)</td>
</tr>
<tr>
<td>Wetland Basin</td>
<td>9 (106)</td>
<td>11 (97)</td>
<td>6210 (774 - 21,400)</td>
</tr>
<tr>
<td><strong>Fecal coliform, cfu/100mL</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Retention Pond</td>
<td>15 (163)</td>
<td>18 (211)</td>
<td>5500 (1030 - 30,800)</td>
</tr>
<tr>
<td>Wetland Basin</td>
<td>7 (65)</td>
<td>8 (53)</td>
<td>15,000 (3080 - 37,500)</td>
</tr>
</tbody>
</table>
### Table 1, continued. Metals.

<table>
<thead>
<tr>
<th>Pond Type</th>
<th>No. Studies (Samples)</th>
<th>Concentration</th>
<th>Trend Signif.</th>
<th>% Reduction in Median Conc.</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>In</td>
<td>Out</td>
<td>Median</td>
<td>(Interquartile Range)</td>
</tr>
<tr>
<td><strong>Total Cadmium (Cd), µg/L</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Retention Pond</td>
<td>33 (518)</td>
<td>35 (545)</td>
<td>0.400 (0.159 - 1.00)</td>
<td>v v v</td>
</tr>
<tr>
<td>Wetland Basin</td>
<td>11 (180)</td>
<td>11 (169)</td>
<td>0.271 (0.101 - 0.700)</td>
<td>v v o</td>
</tr>
<tr>
<td><strong>Total Chromium (Cr), µg/L</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Retention Pond</td>
<td>19 (252)</td>
<td>18 (231)</td>
<td>4.00 (2.00 - 8.00)</td>
<td>v v v</td>
</tr>
<tr>
<td>Wetland Basin</td>
<td>na</td>
<td>na</td>
<td>na</td>
<td>na</td>
</tr>
<tr>
<td><strong>Total Copper (Cu), µg/L</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Retention Pond</td>
<td>52 (934)</td>
<td>54 (922)</td>
<td>9.59 (4.76 - 18.3)</td>
<td>v v v</td>
</tr>
<tr>
<td>Wetland Basin</td>
<td>14 (298)</td>
<td>14 (258)</td>
<td>7.40 (4.27 - 11.8)</td>
<td>v v v</td>
</tr>
<tr>
<td><strong>Total Lead (Pb), µg/L</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Retention Pond</td>
<td>51 (832)</td>
<td>52 (850)</td>
<td>9.00 (2.79 - 26.0)</td>
<td>v v v</td>
</tr>
<tr>
<td>Wetland Basin</td>
<td>12 (200)</td>
<td>12 (174)</td>
<td>3.48 (1.51 - 10.0)</td>
<td>v v v</td>
</tr>
<tr>
<td><strong>Total Nickel (Ni), µg/L</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Retention Pond</td>
<td>13 (187)</td>
<td>14 (169)</td>
<td>3.37 (2.00 - 7.90)</td>
<td>o o v</td>
</tr>
<tr>
<td>Wetland Basin</td>
<td>na</td>
<td>na</td>
<td>na</td>
<td>na</td>
</tr>
<tr>
<td><strong>Total Zinc (Zn), µg/L</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Retention Pond</td>
<td>60 (1032)</td>
<td>63 (995)</td>
<td>50.0 (27.3 - 100)</td>
<td>v v v</td>
</tr>
<tr>
<td>Wetland Basin</td>
<td>19 (342)</td>
<td>19 (308)</td>
<td>52.5 (34.1 - 94.6)</td>
<td>v v v</td>
</tr>
</tbody>
</table>
Q1.1 Rate Control and Q1.2 Flood Control

Background and Performance Trends

Urbanization can dramatically increase the volume of runoff generated during storms and shorten the time required for runoff to travel into natural and engineered drainage systems (Burns et al. 2005, Galster et al. 2006). Large volumes of runoff entering urban drainage networks at approximately the same time can cause high peak flow rates, leading to surface flooding, streambank erosion, and sewer overflows (Moore et al. 2016). Consequently, stormwater ponds have been designed with two primary purposes in mind: “rate control”, typically defined as maintaining on-site peak flow rates to levels at or below pre-development levels for large storm events (return intervals of 2-100 years) (Emerson et al. 2005), and “flood control”, which is less well-defined and typically refers to the delayed transport of peak flows, with the presumed benefit of reduced inundation downstream. However, the role of ponds in reducing inundation extent and duration was very rarely assessed in any of the papers reviewed, thus this section focuses primarily on pond functions of peak flow reduction and detention.

We also note that while volume reduction could provide a measure of peak reduction and flood control, this is not typically an intended design or function of wet detention ponds. Accordingly, volume reduction is rarely assessed, but is typically very low (Hancock et al. 2010, 7% - 29% in Wissler et al. 2020, Birkinshaw et al. 2021, 5.7% in Nayeb Yazdi et al. 2021). Winter volume reduction is especially low (Vollertsen et al. 2009a). Dry detention basins and stormwater wetlands may provide some volume reduction via infiltration and evapotranspiration (23% - 71%; Wissler et al. 2020) but these pond types were not a primary focus of this review.

Despite widespread use of ponds for rate control, this review found relatively few field-based studies that have evaluated the efficacy of ponds in achieving peak flow reductions (though see Birkinshaw et al. 2021). Most studies assessing rate control in stormwater ponds have done so using models (Figure 2). Furthermore, many water quality focused field studies have monitored pond inflow and outflow rates, but have failed to report peak flow reduction values. Field studies that have reported peak flow values generally show that ponds effectively reduce the peak flow rates of outflows relative to inflows (Hancock et al. 2010, Merriman and Hunt 2014, Krivtsov et al. 2020). These results align with modeling study results (Giacomoni et al. 2014, Guan et al. 2018) as well as past pond monitoring efforts associated with the National Urban Runoff Program (NURP) (US EPA 1983).

Although ponds effectively reduce pond outflow rates relative to inflow rates, they can fail to achieve their primary goal of maintaining pre-development peak outflow rates (Hancock et al. 2010). This is particularly true for smaller, more frequent storms (Fennessey et al. 2001, Emerson et al. 2005). Furthermore, studies show that the widespread use of ponds to manage urban development may not provide effective rate control at the watershed-level (Emerson et al. 2005, Aulenbach et al. 2017) and can even increase stream peak flow rates in some instances (McCuen 1979, Emerson et al. 2005).

Peak flow reductions also have tradeoffs with flow duration, such that reducing peak flow for a given storm size necessarily extends its period of flow beyond the expected duration, which can erode channels and transport stream sediments. Booth et al. (1997) and others acknowledge that while pre-development peak flow reduction can sometimes be achieved, post-development volumes are greater and therefore achieving simultaneous reductions in flow duration are
difficult. Drawdown times in ponds are often required to be 24-48 hours (Nehrke and Roesner 2004, Hancock et al. 2010, others), but this may not be sufficiently long to reduce erosive flow duration in receiving streams. Further, some have found highly variable retention times in ponds (e.g., 15-36 hours; Nayeb Yazdi et al. 2021).

**Mechanistic Drivers of Performance**

The demonstrated ability of ponds to reduce site-scale peak flow rates is governed largely by pond size, storage volume, and the design of outflow structures (Booth et al. 1997, Nehrke and Roesner 2004). However, their varied abilities to maintain pre-development peak flow rates may be influenced by a variety of factors. First, ponds are often designed to treat static “design” storms, which do not capture the range of rainfall conditions that ponds can experience. Such conditions include high intensity rainfall events that deliver volumes of runoff into ponds almost instantaneously (Hancock et al. 2010, Emerson et al. 2005), as well as smaller storms that make up the majority of storms that occur within a year – which are more frequent than the large and infrequent storms for which ponds are designed (Fennessey et al. 2001, Emerson et al. 2005). Second, pond storage volumes and hydraulic functions change as ponds accumulate sediments over time, and modeling efforts show that the rate control effectiveness of ponds decreases as they fill with sediment (Guo 1997, Ahilan et al. 2019). Third, ice formation on ponds may also decrease rate control effectiveness during winter months in cold climates (Vollertsen et al. 2009a). Finally, instances where ponds do maintain on-site pre-development peak flow rates may be inaccurate due to overestimation of pre-development peak flow rates (Fennessey et al. 2001). Although ponds often achieve peak flow reductions immediately downstream from a development, their ability to restore pre-development flow regimes can be limited for the majority of storms.

Stormwater ponds are designed to manage runoff from developed sites and consequently, their performance is often evaluated at the site-level. However, the overarching goals of SCMs are often applicable only at the watershed level, within main channels. There is a perception that if all developments release runoff at pre-development levels, then watershed-level peak flow rates will be maintained at pre-development levels as well. Unfortunately, stormwater released from multiple ponds can combine downstream to affect peak flow rates of the watershed. Many studies have shown that widespread implementation of stormwater ponds and other SCMs can have negligible impacts on peak flow rates at watershed outlets (Emerson et al. 2005, Goff and Gentry 2006, Aulenbach 2017). In fact, stormwater ponds have been shown to increase peak flow rates when implemented near watershed outlets because delayed runoff from lower portions of the watershed can converge with undelayed runoff from the headwaters (McCuen 1979, Emerson et al. 2005). Such undesirable outcomes can arise from the indiscriminate use of ponds to treat all development, without considering how pond outflows interact with flow from other parts of the watershed. Accordingly, many studies (Booth 1997, Goff and Gentry 2006, Ayalew et al. 2015, Mullapudi et al. 2020) have demonstrated greater effectiveness for watershed-level flood control and peak reduction when ponds are distributed in watershed headwaters, rather than using relatively larger ponds in downstream portions of the watershed.

Achieving catchment-scale hydrologic goals using detention-based technologies like stormwater ponds is difficult because such management approaches do not reduce overall runoff volumes (Emerson et al. 2005). Additionally, flow reductions in one part of the stream hydrograph will increase flow rates in a different part of the stream hydrograph. This tradeoff can significantly
increase the duration of high flow rates within a watershed, even if peak flow rates are effectively reduced (Booth et al. 2002, Emerson et al. 2005). Prolonged high flow rates can cause significant channel erosion and sediment transport if they exceed critical flow thresholds (Hawley et al. 2017). The inability of stormwater ponds to reduce flow volumes may prevent them from performing key hydrologic functions like channel protection or sediment control, especially in highly developed watersheds with impervious surfaces greater than 40% (Fan and Li 2004).

**Design Innovations**

Most retrofit options that target rate control involve modifying pond outflow structures to alter the timing of stormwater release from ponds. Many studies have investigated the effects of different weir and orifice shapes on outflow hydrographs (Headley and Tanner 2012) and clogging likelihood (Cox et al. 2015) to improve rate control performance. Emerson et al. (2005) showed that restricting the diameter of the lowest pond orifice could improve peak flow rates at the watershed level. Increasing pond size and storage volume would also prolong detention times and affect outflow rates. These and other pond modifications could be applied relatively easily to ponds, but it is unclear how such modifications would interact with flow from other parts of the watershed to influence downstream peak flow rates. Watershed-specific models would likely need to be developed to assess the potential effectiveness of these retrofit options.

Real-time control is an emerging technology that uses hydraulic sensors and controllable outlet valves to actively manage stormwater infrastructure. Ponds equipped with real-time control can use environmental data (i.e., rainfall forecasts, stream flow rates, pond depths) to inform adaptive pond management. For example, real-time control can be used to drain portions of ponds before large storm events to enable greater stormwater capture (Gaborit et al. 2016). It may also be used to hold water within ponds if downstream flow rates are too high. Various studies have used models to assess the impact of real-time control on peak flow rates and found that this technology can substantially reduce volumes (Parolari et al. 2018, Bowes et al. 2021) and peak flow rates (Muschalla et al. 2014, Bilodeau et al. 2018, Mullapudi et al. 2020, Schmitt 2020, Naughton et al. 2021). A couple modeling studies even demonstrated reduction in inundation extent and duration from storage and release of runoff from ponds throughout a watershed using machine learning optimization approach (Giacomoni et al. 2014, Ngo et al. 2016). Real time control also shows potential for adapting to climate change by allowing existing ponds to be managed in a way to handle changing storm intensity, frequency, and duration (Parolari et al. 2018, Sharior et al. 2019). However, real-time control is still a new technology that faces significant barriers to adoption, including cost (monitoring and transmitting data), choice of modeling or optimization approach, substantial amounts of training data (for machine learning approaches especially), and lack of regulatory credits (Mullapudi et al. 2020, Brasil et al. 2021, Naughton et al. 2021).

Beyond pond-level retrofits, studies have investigated how watershed characteristics interact with pond size and location to influence downstream flow rates (James et al. 1987, Goff and Gentry 2006, Meierdiercks et al. 2010). These studies suggest that complex, site-specific dynamics govern watershed responses to SCMs. Further, studies have shown pond performance may not scale predictably with watershed-level changes. For example, studies by Booth et al. (1997) and Aulenbach et al. (2017) suggest that as watershed imperviousness increases, a proportionally greater amount of watershed area needs to be treated by ponds to maintain pre-
development peak discharges. In this vein, authors have criticized policies that require the indiscriminate use of ponds to treat runoff from all development (James et al. 1987, Emerson et al. 2005, Hancock et al. 2010) and have suggested watershed-level planning to inform pond placement and design. Others have developed methods for optimizing pond placement and design using multiobjective genetic algorithms for integrated watershed planning (Yeh and Labadie 1997, Muleta and Nicklow 2004). While theoretically compelling, watershed-level planning needs to occur before land is developed and complete hydrologic analyses are costly and require coordination among local governments (Goff et al. 2006).

Knowledge Gaps and Research Needs

The hydraulic processes that govern outflow rates from ponds are well understood. The challenges ponds face in achieving effective rate control are two-fold. First, how can passive and static stormwater ponds manage precipitation events for which they were not originally designed? Second, how can stormwater ponds designed for on-site performance also meet watershed-level flow reduction goals? Real-time control technologies have significant potential to move existing ponds from static control systems to adaptive control systems capable of managing the hydrologic uncertainties associated with climate change. Modifying pond outflow structures can also alter pond flow performances, but such changes should be informed by watershed models or stream data to ensure that site-scale changes have positive downstream flow impacts.

The results also suggest a need to understand the interaction among different SCM types at watershed scale, and optimizing quantity, location, and sizing to take advantage of the strengths of various SCM types for rate and flood control. In particular, we note that several studies called out the sometimes poor performance of ponds for reducing peak flows or durations of small or frequent events, instead demonstrating that distributed low impact development (LID)-type practices (e.g., green roofs, rain gardens, bioretention) are much more effective at volume and rate control of small events (Emerson et al. 2005, Tillinghast et al. 2012, Giacomoni et al. 2014). This watershed-scale optimization of ponds and other SCM types is a knowledge gap with applicability not just to flood and rate control, but likely to water quality control as well.

Q1.3 Energy Dissipation

Background and Performance Trends

Energy dissipation is primarily required at entrances to a pond, where velocities can be higher than desirable. These high velocities can scour sediment and remove plants. Energy dissipation has been a topic of interest since hydraulic structures were first invented, and there are many design specifications that can be applied to pond inlets and outlets. Specifically, the U.S. Agricultural Research Service has developed general design guidelines that can be applied to ponds (e.g., Deal et al. 1997).

Mechanistic Drivers of Performance

The shear stress associated with high liquid velocities is the primary mechanistic driver of erosion and suspension of particles from the sediment bed. Energy dissipation is required to reduce the momentum of the flow and the shear stress generated by that momentum. This can be
accomplished with the placement of concrete structures or graded riprap. Once velocities have been reduced sufficiently, plants can provide excellent energy dissipation (Gu et al. 2017). High energy stormwater inflows are the primary drivers of sediment resuspension, particularly in shallow ponds that lack energy dissipation structures and have accumulated substantial pond sediment (Guan et al. 2018). Wind has also been shown to agitate water enough to resuspend pond sediments (Bentzen et al. 2009).

**Design and Maintenance Innovations**

Baffles, flow obstacles, pond bottom structures, and emergent vegetation have been explored as relatively low cost options for improving energy dissipation of inflows in ponds (e.g., Eckert et al. 2018, Li et al. 2019). Farjood et al. (2015) investigated the hydraulic impacts of different types and configurations of baffles in ponds and found that porous baffles near the inlet can most effectively reduce energy of inflows. Green infrastructure has been used for energy dissipation in the last 20 or so years, typically composed of logs surrounded by plants to reduce momentum and help maintain soil around the logs. Strategic placement of rooted and floating vegetation near pond inlets has been explored as an energy dissipation strategy (Gu et al. 2017) and has also been shown to increase sedimentation rates and prevent resuspension (Braskerud 2001, Tanner and Headley 2011). Li et al. (2019) noted that installation of riprap on the pond bottom did not improve energy dissipation.

**Knowledge Gaps and Research Needs**

There is much information in the gray literature on energy dissipation that can be applied to inlets and outlets. If energy dissipation is a concern in ponds, a thorough search of gray literature can identify the applicable designs.

**Q1.4 Sediment Removal**

**Background and Performance Trends**

Sediments and total suspended solids (TSS) can have deleterious effects on freshwater ecosystems. By increasing water turbidity, suspended solids can inhibit plant growth, increase water temperatures, and reduce dissolved oxygen concentrations (Shammar et al. 2002). Developed areas are major sources of sediment because they contain large amounts of bare soil, often associated with urban construction. Developed areas also tend to generate large volumes of high energy surface runoff that can transport urban sediment to surface waters, degrading water quality.

While stormwater ponds are designed to detain surface runoff for prolonged periods to mitigate downstream flooding and peak flow rates (Emerson et al. 2005), the extended retention of stormwater in ponds can also improve water quality by allowing sediments to settle out of suspension. Sedimentation is the primary sediment removal mechanism operating in stormwater ponds (Beckingham et al. 2019), and it tends to be effective, with a significant reduction (inflow vs outflow) in median TSS of 61-76% (Table 1; Clary et al. 2020). Stormwater ponds are particularly effective at removing coarse mineral sediments, which settle out of suspension much faster than organic sediments and fine mineral sediments (Guan et al. 2018). Despite their high
overall TSS removal performance, ponds can exhibit low and even negative sediment removal rates, particularly as ponds accumulate sediment over time (Soenderup et al. 2016). Removal of finer sediments is especially poor, with median TDS significantly increasing by 46% in wet ponds in the International BMP Database (Table 1; Clary et al. 2020). Clary et al. 2020 note that TDS typically includes sediment finer than 0.45 - 2.0 um, and may be composed of colloidal sediments as well as dissolved ions (such as chloride).

**Mechanistic Drivers of Performance**

Sediments exhibit different settling rates depending on their size, shape, and density. Large, circular, high density mineral particles can settle out of suspension in a matter of seconds, while small, oblong, low density organic particles can take days or weeks to settle (Zhu et al. 2020). The capacity of stormwater ponds to remove sediments therefore depends on the interaction between pond detention times and particle settling times, which can vary with pond designs (storage volume, outlet structure) and drainage area properties (soil type, land use). Ponds that receive mostly coarse sediment loads may require very short retention times to be effective, while ponds that receive fine colloidal sediments may be ineffective even when designed for extended detention (Selbig et al. 2016). Water column agitation from wind and animal activity can further hinder sedimentation in stormwater ponds (Kasper and Jenkins 2007, Andradóttir and Mortamet 2016).

Factors that reduce the detention time of sediment in stormwater ponds have the potential to significantly reduce the sediment removal performance of ponds. A common phenomenon that reduces the effective detention time in ponds is hydraulic short-circuiting (when inflow nearly directly flows to the outlet), which can result from high temperature differences between pond water and inflow water (Hendi et al. 2018), hydraulic “dead zones” produced by pond shape and layout (Walker 1998, Guzman et al. 2018), and failures to dissipate stormwater inflow energies (Li et al. 2019). Effective detention times can also be reduced by processes that reduce the storage volume of ponds. For example, hydrodynamic models show that sediment accumulation in ponds can reduce sediment removal by decreasing detention times (Ahilan et al. 2019). Ice cover on ponds can also have a major impact on sediment removal rates. Stormwater traveling over bare ice can flow directly to receiving waters with minimal sedimentation, while water that flows under ice can become pressurized, reducing residence times and sometimes causing scouring (Semadeni-Davies 2006, Natarajan and Davis 2015).

Pond design has been shown to exert a relatively strong influence on sedimentation and resuspension rates. The resuspension of formerly settled solids is another phenomenon that can produce poor sediment removal performance in stormwater ponds. Sediment removal efficiency is generally higher in deeper ponds, with permanent depths > 0.9 m (Fennessey and Jarrett 1997). Long and narrow ponds also tend to remove sediments better than short and wide ponds due to reduced short-circuiting (Marsalek et al. 1992, Walker 1998). Ponds that meander and have flow obstacles have also been shown to have high rates of TSS removal (Guzman et al. 2018). Finally, young ponds have been shown to have better sediment removal rates than old ponds, likely due to having less internal sediments (Soenderup et al. 2016).
Design and Maintenance Innovations

One way to improve sediment removal rates in ponds, as well as pond hydrologic functioning, is to intercept sediments before they enter ponds using pretreatment devices. A variety of pond pretreatment devices exist, including vegetative filter strips, hydrodynamic separators, and sediment forebays (Marsalek et al. 1992, Taguchi et al. 2020b). These devices can be relatively inexpensive to install and, if maintained, can significantly delay the need for costly pond dredging. Regular sediment removal from pretreatment devices also represents a substantial pollutant removal mechanism that can reduce contaminant accumulation rates and internal loading. However, such devices require routine maintenance, and there may be costs associated with sediment disposal, particularly if deemed hazardous waste due to the presence of toxic metals and PAHs.

Another way to improve sediment removal performance is to dissipate stormwater inflow energy, which can resuspend sediments and lead to hydraulic short-circuiting and decreased retention times. Long, narrow, and deep ponds with a degree of meandering have been shown to dissipate inflow energy and remove TSS much better than short, wide and shallow ponds without any flow obstacles (Conn and Fiedler 2006, Lightbody et al. 2009). Ponds can therefore be lengthened or deepened in order to enhance retention times and sediment removal. However, pond lengthening and deepening is often space and cost prohibitive, so baffles, flow obstacles, pond bottom grid structures (Milovanović et al. 2020), and emergent vegetation have been explored as relatively low cost options for improving detention times and sediment removal. Eckert et al. (2018) used a baffle dike to increase the flow length of a pond where inflow and outflow structures were, by necessity, located near each other and observed 83% TSS removal. Khan et al. (2017) and Farjood et al. (2015) investigated the hydraulic impacts of different numbers, types, and placements of baffles in ponds and found that certain baffle configurations can significantly improve effective retention times, while others can exacerbate the tendency for short-circuiting. Furthermore, Guzman et al. (2018) explored the hydrologic impacts of multiple in-pond flow obstacles (pinch points, deflector islands, berms) arrangements and found that a cluster of deflector islands located near the pond inlet provided the greatest reduction in short-circuiting. Strategic placement of rooted and floating vegetation near pond inlets has also been explored as an energy dissipation strategy (Gu et al. 2017) and has been shown to increase sedimentation rates and prevent resuspension (Braskerud 2001, Tanner and Headley 2011). Although stormwater inflows are the primary driver of sediment resuspension, wind also has the capacity to resuspend fine particles. Wind shelter belts and strategic placement of vegetation have been explored as a way to decrease wind-induced resuspension (Bentzen et al. 2009). However, wind also helps aerate and cool ponds, which is important in phosphorus control.

Other methods that increase the retention time of ponds can also improve sediment removal rates. Pond lengthening and deepening are two design modifications that can significantly increase retention times, but are costly to implement. However, modifications to outflow structures can influence the pond depth at which outflow begins and thus the actual detention times of ponds. Such retrofits can involve sluice gates (Carpenter et al. 2014), weir modifications (Headley and Wyrick 2010), and real-time control (Gaborit et al. 2016).

Finally, many pond designs are based on a generic stormwater particle size distribution (PSD) obtained by the National Urban Runoff Program (NURP). However, field monitoring efforts
show that measured PSDs can vary greatly from the NURP PSDs. The use of a fixed PSD for all stormwater contexts can lead ponds to be smaller or larger than needed, causing them to be ineffective or unnecessarily expensive (Selbig et al. 2016). Consequently, some studies argue that local soil type and particle size distributions should be a criterion in pond design (McBurnie et al. 1990, Selbig et al. 2016).

Knowledge Gaps and Research Needs

TSS removal in ponds is relatively high, the mechanisms that underlie this function are relatively well known, and a wide variety of pond design and retrofit options exist to improve it. Routine cleaning of sediment forebays is a highly effective practice for maintaining the hydrologic functions of ponds and removing pollutant sources before they enter a pond. However, stormwater sediments are often considered hazardous waste, so research similar to McNett and Hunt (2010) and Geronimo et al. (2019) is necessary to assess the toxicity of sediments in forebays and landscape drivers of toxicity. Also needed is research on methods to remediate sediments classified as hazardous waste. Baffles are a promising retrofit option in existing ponds, especially ponds that are deemed to be too small. The effect of baffles on short-circuiting in frozen ponds and ponds with high temperature inflows should be explored (Hendi et al. 2018). Real-time control applications focused on hydrologic variables will also have impacts on sedimentation rates that should be considered before pursuing such technologies.

Q1.5 Sediment-Bound Pollutant Removal

Background and Performance Trends

Sediments and suspended solids are aquatic pollutants themselves, but they can also serve as vectors of other pollutants, such as nutrients, heavy metals, hydrocarbons, pesticides, and pathogens. These pollutants can bind to both mineral and organic sediments and be transported with them downstream via surface runoff, thereby posing a variety of water quality challenges. Stormwater ponds have the potential to mitigate pollutant loading by settling out suspended solids and the pollutants attached to them. The ability of ponds to remove sediment-bound pollutants via sedimentation gives them the capacity to significantly improve stormwater quality, as the majority of pollutants contained in urban runoff are particulate bound (Liu and Davis 2014, Song et al. 2015, Clary et al. 2020, Janke et al. 2022). However, relatively few studies quantify the removal of the particulate pollutant fraction separately because it requires a full chemical speciation of water samples (i.e., particulate and dissolved forms), which is time and resource intensive. Consequently, removal performances are reported for the total concentration (particulate + dissolved) and dissolved concentration (Clary et al. 2020), and sediment-bound or particulate pollutant removals are often omitted from meta-analyses (Koch et al. 2014, Valenca et al. 2021). The vast majority of studies assessing particulate pollutants in this review measured the pollutant contents of pond sediments to evaluate their toxicity risk to aquatic organisms or potential for land applications. Such assessment methods do not allow for performance calculations, however. A few studies assessing particulate pollutant removal by ponds showed that sediment-bound pollutant removal by ponds is modest and exceeds that of dissolved pollutant removal (Stanley 1996, Egemose et al. 2015).
Mechanistic Drivers of Performance

The challenges facing sediment-bound pollutant removal are similar to those of sediment removal in ponds in that fine particles require prolonged detention times to settle out and can be resuspended by runoff inflows or by wind-driven mixing. This suggests that shallow and/or unsheltered ponds may be more susceptible to resuspension. Since fine particles can contain higher concentrations of pollutants (e.g., metals) than coarse particles (Lee et al. 1997, Ferreira and Stenstrom 2013, El-Mufleh et al. 2014), complete sediment-bound pollutant removal can be difficult to achieve. Different particle types tend to attract different pollutant types and can exhibit differential behaviors in ponds. For example, metals and PAHs tend to associate with organic sediments (Wood and Shelley 1999, Zohar et al. 2017), which tend to have lower densities than mineral sediments and thus may take longer to settle and are more susceptible to resuspension. Negatively charged pollutants like phosphate and arsenate tend to associate with positively charged metal hydroxides contained in mineral sediments. Accordingly, sedimentation may be a more effective mechanism for removing pollutants that are associated with mineral sediments. Ultimately, the sediment-bound pollutant removal performance of ponds depends on interactions between particle types, particle sizes, pollutant chemistries, and pond designs, which may vary dramatically between ponds.

Variations in settling rates, pollutant concentrations, and particle sizes can also influence the spatial distribution of sediment toxicity in ponds. For example, coarse sediments rapidly settle out of suspension and tend to accumulate near pond inlets, while fine sediments take longer to settle out and tend to accumulate near pond outlets (Dominic et al. 2016). Because coarse particles often exhibit lower concentrations of heavy metals than fine particles, sediments dredged from locations near pond inlets (e.g., sediment forebays) may qualify for different disposal techniques (e.g., land application) than sediments dredged near pond outlets (confined disposal). Such spatial patterns in sediment toxicity have important implications for stormwater pond maintenance.

With respect to frequency of dredging, which is relevant to removal of sediments and sediment-bound pollutants, recommendations in the literature were inconsistent. Soenderup et al. (2016) recommended a dredging interval of 5 years, based on observed decline in TP removal of ponds older than 5 years, while Ahilan et al. (2019) suggested an 8-10 year interval would be sufficient to maintain high sediment removal performance. The longest interval, at 25 years, was suggested by Yousef et al. (1994) to maintain sedimentation and rate reduction performance, based on observed sediment accumulation rates and an estimated loss of 10-15% of pond volume over that interval.

The ultimate fate of sediment-bound pollutants in stormwater ponds depends on the chemical stability of the pollutant in question and the environmental conditions it is subject to within ponds. Stormwater ponds can fluctuate between multiple biogeochemical states throughout the year, exhibiting thermal stratification, chemical stratification, and hypoxia, and undergo freeze-thaw cycles. They also receive intense solar radiation and support diverse biological activity. These variable conditions can transform and mobilize many pollutants commonly contained in pond sediments. Pond sediment incubation and flux studies are needed to determine the long-term stability of pollutants in pond sediments.
Design Innovations

Pollutants that enter stormwater ponds in sediments either remain bound to sediments or are converted to dissolved forms. If bound to sediments, then all of the design innovations that apply to sediment removal also apply to sediment-bound pollutant removal (Q1.4). If they are converted to dissolved forms, then all of the design innovations that apply to the subsections of Q2 also apply to the dissolved forms of pollutants that were formerly sediment-bound. Consequently, design innovations for sediment-bound pollutant removal are not discussed here.

Knowledge Gaps and Research Needs

Relatively few studies have measured sediment-bound pollutant removal. TSS removal is an inaccurate proxy for particulate pollutant removal because fine particles can exhibit substantially higher pollutant concentrations than coarse particles and finer particles may not necessarily be retained by settling. These fine particles are also the most susceptible to resuspension. If certain pollutants like metals are primarily associated with particulate and colloidal particles, then pond retrofit methods that target coarse particle removal (e.g., sediment forebays and vegetative filter strips) may be ineffective tools for managing these pollutants. Stormwater practitioners could therefore benefit from an improved understanding of the extent to which fine particle dynamics influence pollutant loads for different contaminant groups.

Many studies consider the total nutrient forms (particulate plus dissolved) rather than the various dissolved and particulate forms separately, with removal assumed or observed to arise primarily from particulate removal and capture. We acknowledge that particulate-bound pollutant removal represents an intended function of wet ponds, but to reduce repetition and for sake of organization, we discuss knowledge gaps and design innovations for both particulate and dissolved pollutant forms under their respective Q2 sections below.
Q2.1 Nitrogen

**Background and Performance Trends**

Nitrogen (N) is a co-limiting nutrient in freshwater and marine ecosystems, and excessive N loading can contribute to eutrophication and harmful algal blooms in surface waters (Carpenter et al. 1998). Sources of N in urban environments include lawn fertilizer, pet waste, plant litter, atmospheric deposition, and fossil fuel combustion (Collins et al. 2010, Hobbie et al. 2017). Stormwater ponds have the capacity to reduce stormwater N loads through a variety of mechanisms and are consequently considered effective SCMs for N management. The primary N removal mechanisms operating in ponds are sedimentation, biological uptake, and denitrification, but N can also be removed via chemical adsorption and volatilization. N removal efficiencies in ponds are generally low to moderate, with median concentration reduction of 26%, 59%, and 29% for total N, nitrate, and ammonium, respectively (Table 1), which also suggest a low retention of organic N. However, removal performances are highly variable, with many studies reporting negative removal efficiencies (Koch et al. 2014). The mechanistic drivers responsible for this performance variability differ in space and time and for the various chemical species of N.

**Mechanistic Drivers**

Particulate organic N (PON) can account for 50% or more of total N loads in urban stormwater (Li and Davis 2014, Janke et al. 2022). Because ponds are specifically designed to promote sedimentation, particulate N removal tends to be effective in ponds (Griffiths and Mitsch 2020, Janke et al. 2022). Instances of poor particulate N removal may be due to the difficulty associated with removing fine particles from suspension, which take longer to settle out than coarse particles and can contain high concentrations of N. Fine particles are also susceptible to resuspension from stormwater inflows, which can lead to negative N removal rates in older ponds that contain large amounts of sediment (Griffiths and Mitsch 2020).

Settling PON in pond sediments does not guarantee its permanent removal from the watershed. For example, PON can be converted to dissolved organic nitrogen (DON) via biodegradation or to ammonium via N mineralization, and ammonium can be nitrified or assimilated and released as DON in microbial or plant exudates (Lusk and Toor 2016). Multiple studies have reported substantially higher DON concentrations in pond outflows than inflows (Jani et al. 2020, Gold et al. 2021), with DON accounting for as much as 78% of total N in outflow loads (Lusk and Toor 2016). Outflow DON can also exhibit much lower aromaticity and molecular weight than inflow DON, suggesting that ponds increase the bioavailability of DON (Lusk and Toor 2016).

Plants and microbes also have the capacity to uptake and immobilize dissolved N. However, biological uptake may be a temporary removal mechanism as N immobilized in biomass can be remineralized when organisms die and decompose or be excreted as exudates (Nichols 1983, Hoagland et al. 2001). Plant harvesting has potential to permanently remove N from stormwater, but this practice simply relocates the N problem and has been shown to increase pond N concentrations in some studies (Badiou et al. 2019). Furthermore, N inputs to a pond may overwhelm plant N demands (Griffiths et al. 2021), particularly in large ponds with relatively small plant coverage.
Finally, denitrification can permanently remove N from stormwater by converting nitrate to nitrogen gas. For denitrification to occur, portions of a pond must be anaerobic, and the ponds must contain sufficient amounts of nitrate and a carbon source (Bettez and Groffman 2012). These conditions are often satisfied in stormwater ponds, yet nitrate removal efficiencies can be low (Koch et al. 2014, Valenca et al. 2021). There are many reasons why denitrification may not be a substantial N removal mechanism in all ponds. First, it is a biologically mediated process that has specific temperature requirements to be effective. Consequently, denitrification can be completely absent during winter months in cold climates (Collings et al. 2020) or suppressed in hot summer months, even when ponds are anaerobic (Gold et al. 2021, Zhong et al. 2010).

Second, biological reactions often take longer to occur than chemical reactions, so denitrification may not be able to treat water quality volumes due to a short detention time (e.g., 24-48 hours) in ponds (Valenca et al. 2021). Third, denitrification has been shown to increase with pond age, which suggests that the community of denitrifying bacteria takes time to establish in pond sediments (Hohman et al. 2021). Finally, denitrification can be limited by nitrate availability (Rahman et al. 2019). Although nitrate makes up a portion of N inputs into ponds, other N species must be converted to nitrate via nitrifying bacteria, which have their own environmental requirements. While ponds have potential to facilitate N removal via denitrification, it may require specific conditions to effectively reduce stormwater N loads.

The effectiveness of N removal in stormwater ponds is influenced by many factors, including pond design, climate, and watershed properties. Pond design (depth, vegetation, retention time) can exert major control over the physical and biogeochemical characteristics of a pond and thus the efficacy of the different N removal mechanisms. Climate can also have major effects on N removal in ponds given that climate governs biological activity and N cycling is a biologically mediated process. Finally, watershed properties (area, imperviousness, N sources) determine the amount of stormwater and N that ponds receive. A recent meta-analysis (Valenca et al. 2021) investigated trends in pond N removal across multiple climates to determine whether pond design or climate exerted greater control over N removal in stormwater ponds. Although data constraints limited this analysis to hot-summer humid continental climates and humid subtropical climates, it found that N removal was more influenced by pond design than climate. In particular, it found that N removal was highest in shallow ponds with high vegetation coverage and relatively small drainage areas. Whether this trend applies to cold climate regions is unclear.

**Design Innovations**

A major theme in designing ponds for N control is to increase pond retention times. Increasing retention times can promote sedimentation of particulate N and allow more time for biological N uptake and denitrification. Many of the pond design innovations mentioned above would achieve increased retention time, such as increased pond depth and length (Marsalek et al. 1992, Mallin et al. 2002), sluice gates (Carpenter et al. 2014), real-time control (Naughton et al. 2021), outflow modifications (Headley and Wyrick 2010, Schwartz et al. 2017), and designs to reduce short circuiting (Eckert et al. 2018, Guzman et al. 2018, Khan et al. 2017). Studies investigating the impacts of increased retention time on pond N dynamics have observed variable results, but generally suggest that increased retention time moderately increases N removal (Toet et al. 1990, Chaindet and Xenopoupolos 2011, Carpenter et al. 2014). However, our review did not identify any controlled pond experiments that could isolate the effect of retention time on N removal over long periods of time.
Vegetation is another pond design factor that could influence N removal. If ponds are relatively deep, rooted vegetation may only exist on aquatic shelves or pond perimeters. Such vegetation may play an important role in discouraging waterfowl activity and N inputs from their excretion (Collins et al. 2010), but plant N demands would likely be small relative to N inflows. Different plant species can influence denitrification activity in pond sediments (Gordon et al. 2020), but vegetation may only exist in shallow portions of the pond where sediments are largely aerobic. Floating macrophytes, such as duckweed or plants contained on floating treatment wetland technologies, have potential to extract dissolved N from the water column (Perniel et al. 1998). Many studies have investigated floating treatment wetland technologies, but few have used experimental controls or investigated the impacts of floating treatment wetlands on pond-scale outcomes. Overall, pond vegetation can influence N dynamics by affecting microbial communities, sedimentation rates, and via direct uptake of dissolved N. The magnitude of these effects relative to N inputs tends to be low, however, particularly for ponds in cold climates.

Ammonium and nitrate are charged ions that can adsorb to oppositely charged mineral and organic surfaces. Consequently, studies have investigated the use of sand, biochar, and other media amendments to filter dissolved N from ponds. Chang et al. (2016) developed a solar-powered floating media filter to remove N from ponds during dry periods, and others have explored gravity-fed filters to treat storm flows (Rushton and Teague 2005). Filters have shown some capacity to remove ammonium in lab trials, but very little capacity to remove nitrate. Furthermore, filters tend to clog and need to be coarse to filter large pond volumes in reasonable time frames.

Finally, pretreatment practices like sediment forebays may be particularly effective for N removal because the majority of stormwater N loads are particulate bound (Li and Davis 2014). Unlike other pollutants, coarse organic sediments can contain very high concentrations of N and be effectively settled in forebays (Geronimo et al. 2019). Forebays have been shown to contain higher concentrations of dissolved N in sediments than the main pond (Chiandet and Xenopoulos 2016). Installing and regularly maintaining sediment forebays may therefore be an effective practice for N control.

**Knowledge Gaps and Research Needs**

Stormwater ponds were not originally designed to control dissolved N loads, but a reduction in median concentration of 30% - 60% dissolved N (Table 1) by ponds indicates a substantial water quality benefit. It is also important to note that unlike media-based stormwater SCMs (e.g., bioretention), stormwater ponds start with variable and often low N and organic stores, so function for N removal changes over time, as seen in recent studies (Hohman et al. 2021). In light of their often high N removal performance, research to improve N removal in ponds may be a relatively low priority. However, while N removal may be acceptable on average, high variability in removal (Collins et al. 2010, Koch et al. 2014, Valenca et al. 2021) as well as in the large range of inflow and outflow concentrations in ponds in Clary et al. (2020) indicates a need to better understand controls on N cycling in ponds. The capacity of ponds for NH₄ production and loss, and for conversion of PON to bioavailable DON, pose eutrophication risks to downstream waterbodies and are a likely source of high variability in observed N removal by ponds. Consequently, the most immediate research needed on pond N dynamics is an investigation into NH₄ and DON-PON. In particular, research that investigates the rate,
magnitude, and environmental drivers of these forms would help to contextualize the eutrophication risk it poses.

To further improve N management in ponds, additional research into the magnitude of denitrification and plant uptake within ponds (Blaszczak et al. 2018, Hohman et al. 2021) would help to determine whether management strategies that augment these processes (e.g., increasing pond volumes, detention times, and vegetation coverage) are worth pursuing. Many studies have quantified plant N uptake and measured denitrification, but few have attempted pond mass balances for N to determine the magnitude of these processes relative to N inputs (Ahlgren et al. 1994). Such research would provide a critical first step in determining whether costly pond retrofits to promote N removal will achieve their intended goal.

The impact of on N cycling and retention in ponds of stressors known to affect retention of other pollutants (e.g., phosphorus, metals) may represent another research need. For example, the impact of cold weather periods (when denitrification and other biologically-driven processes will occur at lower rates) and snowmelt (when runoff and transport rates can be relatively intense) are under-studied. Similarly, the impact of warming climate and changes in rainfall frequency or extended dry periods will impact both nitrogen and phosphorus cycling in ponds. Other important stressors could include interactions with chloride (road salt) and metals, and the impact of pond age, which may not have the same detrimental effects on retention of N as for P, but still represents a potential research need.
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Q2.2 Phosphorus

Background and Performance Trends

Phosphorus (P) is an essential nutrient used by most organisms for growth and metabolism and is typically present as in organic matter and phosphates bound to various organic and inorganic substrates (EPA 1999, Clary et al. 2020). In stormwater, P is characterized as total P and as particulate and dissolved forms based on passage through a filter (usually 0.45 to 0.7-um nominal pore size) (Clary et al. 2020). Soluble reactive P (SRP) can also be distinguished from dissolved P by analysis before and after chemical digestion. The particulate fraction of P includes mostly organic material (living and non-living) as well as soil particles, and is generally the majority of total P in stormwater (~50% - 75% of influent TP to ponds; Clary et al. 2020 and Table 1). Dissolved P includes SRP, a largely inorganic and bioavailable portion that can be rapidly assimilated by plants and microorganisms (comprising ~20% - 35% of influent TP to ponds; Clary et al. 2020 and Table 1), with the remaining dissolved fraction is composed of larger, more recalcitrant organic compounds. In general, P tends to be associated with finer particle sizes (e.g., clay and silt) and more organic forms of sediment (Greb and Bannerman 1997); thus, P removal tends to be lower than particulate removal due to poor retention for finer sediments and the presence of dissolved P (Greb and Bannerman 1997).

Many processes govern transformation and cycling of P in ponds. With respect to removal processes from the water, plant and microbial uptake can transform SRP into particulate P, and phosphates adsorbed to sediment or metal substrates can settle. Particulate P will also settle out in the water column over time (sedimentation). Transformation processes also occur in ponds, as bioavailable SRP can be produced from mineralization of (particulate) organic P as well as from redox reactions of inorganic phosphate complexes and metal oxides, which are mediated by pH, temperature, and oxygen conditions. Organic particulate P may also be converted to other dissolved forms via decomposition and photodegradation. Therefore “burial” of P as sediment is not always a permanent removal from the water column in ponds. Further, P also does not have a gaseous form that’s relevant in the context of stormwater (unlike nitrogen), so all P that enters a pond must be retained indefinitely to achieve effective long-term P removal.

Per the International BMP Database (Clary et al. 2020; Table 1), ponds provide substantial TP removal, with roughly 50% reduction in median TP concentration in wet retention ponds and 28% median reduction in stormwater wetlands. For dissolved forms (TDP and SRP) removal is significant in wet ponds (50% - 60% removal) but much lower, and not significant, in the stormwater wetlands (0.3% - 16%; Table 1). In all cases, results are highly variable, suggesting that while median performance is more or less in line with expectations (for wet ponds), a range of factors are likely affecting performance in ponds of all types. Dissolved species of P are especially variable across the studies and pond types in the International BMP Database and in the literature, and poorly retained in some instances (e.g., for stormwater wetlands). Thus, many recent innovations in pond design and treatment have focused on managing the dissolved form of P, in acknowledgement of its generally poor retention and potential for release from sediments under conditions (anoxia, high productivity) common in many ponds.
Mechanistic Drivers

While stormwater detention ponds were originally constructed primarily to alleviate flooding issues in urbanizing watersheds, understanding of the water quality control function of ponds, especially with respect to sediment and phosphorus and reducing stormwater contribution to eutrophication and algae blooms, has also been subject to decades of research. In particular, increased hydraulic residence time in ponds has been demonstrated to provide sediment removal through settling and burial in the bottom of ponds (Walker 1998), with associated removal of phosphorus attached to sediment particles (Weiss et al. 2010). Therefore, wet detention pond construction has been widely used for stormwater management over the past several decades, with assumed and observed benefits to sediment and phosphorus removal.

Among studies included in the review, total phosphorus (TP) removal was influenced by several pond and watershed factors, in particular those with direct or indirect effects on water retention and residence time. Higher TP removal was associated with greater flowpath length (Comings et al. 2000, Mallin et al. 2002), treatment volume relative to drainage area (Yousef et al. 1994, Wu et al. 1996, Comings et al. 2000), and water retention (Janke et al. 2022), with benefits largely attributed to increased residence time providing time for solids to settle (Walker 1996). For example, Soenderup et al. (2016) found that removal of both particulate P and dissolved P (as TDP and DIP) were improved by increasing flow path length and residence time in ponds. Accordingly, TP removal was found to be driven by high sedimentation rates, especially of finer materials (Ferrara and Witkowski 1983, Greb and Bannermen 1997, Griffiths and Mitsch 2020, others). Ponds with “cleaner” standing water were also found to provide benefits from dilution of P-rich inflows (Ferrara and Witkowski 1983). By contrast, poor TP removal performance was observed during winter and snowmelt periods (Vollertsen et al. 2009a, Natarajan and Davis 2016, Nayeb Yazdi et al. 2021), caused by lower storage capacity in frozen ponds, high loading rates, and low biological processing rates in colder temperatures. Seasonality was also observed, with higher inflow TP during spring leaf-out and late summer or fall, during leaf drop (Natarajan and Davis 2016, Janke et al. 2022).

In-pond concentrations of P were similarly influenced by a range of factors, with some variability in the results. For example, in studies comparing multiple ponds, higher TP and dissolved P (DP) were associated with greater age (Chianet and Xenopoulos 2016), suspended sediment (Greb and Bannermen 1997, Chianet and Xenopoulos 2016), benthic organic matter (Song et al. 2017), stratification strength (Song et al. 2013), and temperature (Wanek et al. 2021). Lower TP and DP were observed in newer ponds (Yousef et al. 1986) and in those with greater watershed area and complex morphology (surface area:perimeter) (Chianet and Xenopoulos 2016) and with higher sand content in sediments (Greb and Bannermen 1997). Sediments in ponds and wetlands were generally rich in organic P (Song et al. 2017, Taguchi et al. 2020a) compared to lakes.

A major management concern in ponds is the release of sediment-bound SRP under low oxygen conditions, a process occurring in eutrophic lakes that has also been observed in stormwater ponds (Yousef et al. 1986, Song et al. 2013, Taguchi et al. 2020a). SRP tended to be lower in wet ponds relative to treatment wetlands (Woodcock et al. 2010) and was also observed to be taken up more rapidly by duckweed than algae (Sims and Hu 2013). Conversely, in ponds with well-
oxygenated water columns, SRP removal rates were high and sediment release was low (Yousef et al. 1986).

Sedimentation of particulates, while a primary function of wet ponds for sediment and nutrient removal, over long term could lead to lower P removal rates (Oberts and Osgood 1991, Merriman and Hunt 2014). Sedimentation may also primarily impact allochthonous inputs (e.g., coarse sediment from the watershed) and provide less removal for in-pond production (e.g., detritus, algal biomass) (Qualls and Heyvaert 2017, Schroer et al. 2018). Accordingly, Song et al. (2015) showed that ponds slightly increase particulate P concentrations but demonstrated that this was due to increased algal biomass rather than ineffective sedimentation.

Aquatic vegetation may also be important for P retention, through the mechanisms of uptake by phytoplankton and macrophytes (Griffiths et al. 2021), as well as through promoting sedimentation or preventing re-suspension (Griffiths and Mitsch 2020). Effects of vegetation on performance was variable; Vincent and Kirkwood (2014) found no correlation of surface TP and emergent vegetation cover in a broad sampling study, while Martin (1988) and Mallin et al. (2002) observed high TP and SRP removal by ponds with open water and emergent littoral vegetation, and Griffiths and Mitsch (2020) showed that vegetated stormwater wetlands significantly reduced particulate P loads via sedimentation (34.5% of TP removal). P removal was dependent on plant species (Weiss et al. 2006, Lu et al. 2010), with Weiss et al. (2006) further finding that P was generally higher in above-ground biomass than roots in several species. Studies of floating plants (duckweed and Wolffia) were rare, but demonstrated high P uptake rates (Perniel et al. 1998), including an ability to outcompete algae for SRP (Sims and Hu 2013). Further, Griffiths et al. (2021) found that algal and plant uptake increased with higher temperatures and rainfall, suggesting a sensitivity to climate change. With respect to other pond types, Wissler et al. (2020) found that overgrown dry detention basins promoted TP and SRP removal, while vegetated treatment wetlands showed poor removal of DP and SRP (Martin 1988, Oberts and Osgood 1991, Merriman and Hunt 2014, Perron and Pick 2020), with some wetlands even being a net source of phosphorus. Perron and Pick (2020) suggested high productivity and diurnal DO fluctuations as primary causes of poor dissolved P retention.

Assessment Methods

A wide range of methods were used in the included studies for assessment of pond phosphorus dynamics and removal performance, including most common or standard methods for water, vegetation, and sediments (see section on Current Pond Assessment Methods, page 48). Several unique approaches included methods using diffusive gradients in thin films (Trowsdale and Arnold 2007), paired watershed studies (Gold et al. 2017), concentrations in invertebrates (Stephansen et al. 2016), and comprehensive phosphorus mass balances that included assessment of sediment, vegetation, and water (Griffiths and Mitsch 2020, Griffiths et al. 2021).

Design Recommendations and Innovations

Pond Design for P Removal

Several of the studies suggested designing ponds with sinuous or complex geometries and greater length-to-width ratios to increase residence time. However, Chiandet and Xenopoulos
(2016) found TDP and TP in ponds increased with more sinuous shorelines (i.e., surface area-to-perimeter), a counter-intuitive result given that they suggest complex shorelines promote greater macrophyte growth for trapping and assimilating P. Song et al. (2013) found that ponds with high surface area-perimeter ratio tended to stratify more often, which could promote release of P from anoxic sediments. Song et al. (2015, 2017) raised a similar question of shallow ponds with longer flow paths, suggesting that in highly productive ponds with organic, P-rich sediments, even with the likely benefit of better oxygenation there is high potential for internal P loading.

The tradeoffs of increased residence time, internal loading, oxygenation, and re-suspension in shallower or morphologically complex ponds represents a knowledge gap in pond design.

Pond forebays emerged as a frequent recommendation for improving water quality in the main body of ponds (Taguchi et al. 2020b). The primary advantage of a forebay is to rapidly slow incoming flows to promote rapid sedimentation of particulates, increasing residence time in the main pond for settling of finer particulates and for assimilation of dissolved forms (Schueler 2000). Chiandet and Xenopoulos (2016) found ponds with forebays had significantly lower TP than ponds without forebays, and also observed significantly lower TP and TDP (as well as TDN and TSS) in the main pool of ponds vs. their forebays. Forebays also provided high sedimentation in stormwater treatment wetlands (Griffiths and Mitsch 2020). Song et al. (2017) observed high algal biomass and sediment organic P in forebays, suggesting they may be more productive than the main pond pools, and thus also pose a potential flushing risk in big storms (Chiandet and Xenopoulos 2016). In light of this, Merriman and Hunt (2014) and others highlighted the need to dredge forebays frequently to maintain their effectiveness. Forebays appear to improve sediment and P removal, provided they are maintained appropriately. Further research may be needed to better understand maintenance needs and expected water quality benefits.

Maintenance, Retrofits, and Treatments

Dredging is a maintenance option to directly counter the negative effects of sedimentation and accumulation of organic, P-rich sediment, though it can be expensive, especially if sediments are contaminated with heavy metals or hydrocarbons and must be disposed of as hazardous waste. Dredging was shown to be effective for improving P removal by ponds (Oberts and Osgood 1991, Lürling et al. 2017), with Waajen et al. (2016) highlighting the importance of combining dredging with other measures (biomanipulation, filters) to improve effectiveness across all forms of P. Soenderup et al. (2016) found much poorer removal of dissolved and particulate P beyond 5 years since dredging, suggesting a dredging interval of 5 years. Yousef et al. (1994) recommended dredging every 25 years, based on an estimated 10-15% loss in storage volume over that time span.

Floating treatment wetlands (FTW), a practice adapted from wastewater treatment, were a common topic in the initial set of papers for the review but were excluded as beyond scope. However, we included a few papers that described case studies of pond retrofits with floating vegetation islands. Results of those studies showed that floating islands provided modest to significant TP and TSS removal versus control ponds (Winston et al. 2013, Schwammberger et al. 2017, Walker et al. 2017). A unique study using artificial roots as a control in mesocosms (Tanner and Headley 2010) demonstrated greater removal of TDP and fine sediment by the live
plants, suggesting that plants affect conditions at very small scale (perhaps through promoting biofilm growth to improve sorption or attachment of fine sediment to plant material). One major concern with the use of FTW in northern climates is over-winter survival of plants; Tharp et al. (2019) demonstrated winter survival of at least a couple plant species in a Vermont (USA) mesocosm study.

In recognition of the P retention benefits stemming from increasing residence time and reducing energy of incoming flows to ponds, a few studies investigated the use of baffles and islands in stormwater ponds, often implemented as retrofits. Eckert et al. (2018) found that ponds with baffles provided 59% removal of TP (which would exceed the expected removal of ~50% for wet ponds; Clary et al. 2020), while German and Svensson (2005) observed an additional 10% removal of TP in ponds with baffles vs. control ponds without baffles. These structural retrofits have applicability to a number of pollutants found primarily in particulate form (sediment, metals, PAHs, and nitrogen), and could be a useful topic for future research. While an obvious enhancement, however, they are not generally applied, possibly due to their cost and potential maintenance requirements.

Filtration has been used to provide additional removal of both particulate and dissolved forms of phosphorus through sorption, and primarily implemented as a media bed that pond water flows or infiltrates through. Many types of media have been tested in field and laboratory settings, with sand being perhaps the most common type. Sand filters have provided removal of P when used as an inline filter (e.g., as a bench between a forebay and main pond pool) or as an outlet filter bench with underdrain (Vollertsen et al. 2009b, Ryan et al. 2010, Soenderup et al. 2016). Sand is often amended with iron filings to provide additional SRP removal (e.g., Belden and Fossum 2018), with one study adding plants to provide further P removal (Istenic et al. 2012). Other effective filter media included limestone (Postila et al. 2017), crushed concrete (Soenderup et al. 2015), Oyster shells and olivine (Vollertsen et al. 2009b, Istenic et al. 2012), and titanium oxide (Chen et al. 2019). Spent lime has also been used, with mixed results (RPBCWD 2021). Water treatment residuals, which contain aluminum oxide, a research topic in stormwater filtration (O’Neill and Davis 2012a, 2012b), have not been attempted in the exit of a pond. In terms of maintenance, Egemose (2018) noted that most P was found in the upper 5 cm of the filter bed and recommended swapping out the upper layer regularly. Other concerns with filters include the need to occasionally dry out (in the case of iron amendments, needed to form iron oxides to bind SRP). Filtration appears to be a promising pond modification, yet knowledge gaps exist on long-term performance, maintenance needs, and optimization of amendments like iron filings, spent lime or water treatment residuals.

A wide range of sediment treatments have also been tested on ponds, with the goal of preventing mobilization of dissolved forms of P from sediments into the water column. Sediment amendments include alum and lanthanum-enhanced bentonite (Osgood 2012, Waajen et al. 2016, 2017), which have proven to be effective on short time scales. Further, Na and Park (2004) found that an addition of ocher pellets and CaCO₃ reduced sediment P flux. In terms of indirect benefits, alum treatments also successfully reduced *Wolffia* abundance (Osgood 2012) and iron sulfate provided a reduction of algae (Istenic et al. 2012). Iron filings have also been used recently in the Twin Cities, with results pending (Natarajan and Gulliver 2022). Sediment treatment and amendments have been successfully demonstrated in eutrophic lakes and ponds,
yet more research appears needed on dosing, cost-effectiveness, and especially on the duration of benefits in ponds with high sedimentation rates.

Additional Knowledge Gaps and Research Needs

Ponds are “hot spots” of biogeochemical processing (Woodcock et al. 2010, Williams et al. 2016, Frost et al. 2019). Given the importance of oxygen and temperature to microbial and geochemical processes affecting the mobility and transformation of P, many recent studies have focused on understanding how these processes are affected by oxygen dynamics, mixing, and stratification in ponds (e.g., McEnroe et al. 2013, Song et al. 2013, Song et al. 2017, Taguchi et al. 2020a). In particular, understanding the extent, magnitude, and controls on internal loading of P, especially from anoxic sediments, is a major knowledge gap. Even shallow ponds are often stratified and anoxic, and duration of stratification may be more important than stratification strength to P release (Song et al. 2013, Taguchi et al. 2020a). Other studies have shown that fresh inputs of organic matter and warm temperatures lead to high microbial decomposition in ponds (Woodcock et al. 2010, Williams et al. 2013, Song et al. 2015, Song et al. 2017, Frost et al. 2015), providing another likely source of internal loading. Song et al. (2015) also showed that while landscape P tended to be well deposited and converted near pond inlets, P produced in the ponds tended to be exported.

Similarly, hydrologic and physical mixing processes (from wind or runoff inputs) impact stratification and oxygen dynamics in ponds. While ponds tend to be small, and are often well sheltered by vegetation or building, Wakelin et al. (2003) found wind to be an important factor in TP and SRP levels in Canadian ponds. Chiaudet and Xenopoulos (2016) also highlight wind mixing as a research need. Song et al. (2013), by contrast, suggested that runoff inputs and thermal processes (surface heat flux) were more important for mixing than wind events. A strong interaction of hydrology and biogeochemical processes has also been observed (Song et al. 2015, Schroer et al. 2018); Janke et al. (2022) observed P retention of all forms being driven by pond hydrologic retention, which was enhanced by high water loss during dry periods. Song et al. (2013) also suggested that episodic or seasonal mixing could also potentially enhance movement of sediment-released P to top of the water column. Together, these results indicate a better understanding is needed of the internal complexities of ponds: the interactions of mixing forces (wind, runoff, heat), pond characteristics (depth, volume, sediment properties), hydrology, and oxygen dynamics on P release and burial.

The importance of pond vegetation (macrophytes, phytoplankton, and floating vegetation like duckweed) for P cycling and removal is evident from existing research (Martin 1988, Perniel et al. 1998, Mallin et al. 2002, Merriman and Hunt 2014, Vincent and Kirkwood 2014, Griffiths et al. 2021, Nakhaei et al. 2021). The high variability in those results suggest that more work is needed to understand the role of vegetation in ponds and how plants might be managed or manipulated to better remove P. For example, several studies have suggested harvesting of plant material (both floating and rooted) as a means to permanently remove P, though some studies showed mixed results (Badiou et al. 2019, Wakelin et al. 2003). Further, floating vegetation (e.g., duckweed and Wolffia) may be important for management due to its rapid growth (Perniel et al. 1998, Sims and Hu 2013) and potential to be flushed from ponds during high flow events. The impact of all types of pond vegetation on annual nutrient budgets, especially P that may be released during senescence, the impact of long wet or dry periods, and long-term storage in
above-ground vs. below-ground biomass (Griffiths et al. 2021), remain under-studied in the context of ponds.

An emerging concern with respect to pond vegetation is the occurrence of harmful algal blooms. Waajen et al. (2014) pointed out that blooms may be more widespread and more toxic than realized, in part because ponds are not as widely used for recreation. Excess phosphorus can contribute to algae blooms in ponds, but in the few studies investigating HAB in ponds found relationships between P and microcystin concentration or cyanobacteria abundance to be highly variable and insignificant (Vincent and Kirkwood 2014, de La Cruz et al. 2017). Greenfield et al. (2017) found that a few bacteria species were correlated with P, and that these species often correlated with higher temperatures and presence of HAB in the study ponds. Further research is needed to better understand conditions that contribute to susceptibility of ponds to HABs.

The effect of pond age on P retention is a management and assessment concern, and was the subject of many studies. In particular, several studies demonstrated high removal in new ponds of DP or SRP (Yousef et al. 1986, Vollertsen et al. 2009a, Gold et al. 2017) and TP (Yousef et al. 1986, Stanley 1996, Vollertsen et al. 2009a) and in recently-dredged treatment wetlands (Oberts and Osgood 1991). These studies all acknowledged that while performance was acceptable in these new or renovated ponds, there were concerns for degraded performance as the ponds fill with sediment, trash, and vegetation. Aging stormwater ponds are numerous, and the intensive monitoring necessary for computing complete water and P budgets are time-consuming and potentially expensive. Thus, a need exists to rapidly and cost-effectively assess the function and performance of stormwater ponds. Such methods could include assessments like those developed by Tahsin and Chang (2016), which relied on a combination of TN, TP, and secchi measurements to compute a Trophic Index for ponds.

Annual scale studies were rare, with P removal most often assessed for warmer periods only. Monitoring programs typically began after ice-out and snowmelt due to the difficulty during freezing conditions and usually do not extend far into the fall to capture turnover (Natarajan and Davis 2016). Thus some periods of high P transport or transformation are not well represented in the literature (Janke et al. 2022). Several studies highlighted the need to better understand the impacts of winter on P retention in ponds: Wakelin et al. (2003) observed high in-pond P in spring due to high rates of over-winter decomposition, while others (Comings et al. 2000, Vollertsen et al. 2009a, Natarajan and Davis 2016, Nayeb Yazdi et al. 2021) observed poor winter performance or worse removal during cold weather than during warm weather. Oberts and Osgood (1991) and Mayer et al. (1996) noted that ice cover may compress flows under ice and cause resuspension during snowmelt, and Vollertsen et al. (2009a) recommend that ponds be sized to account for ice thickness to improve snowmelt storage. In northern climates where chloride from road de-icer is a major input to ponds, salt may impact stratification (Taguchi et al. 2020a, Janke et al. 2022) and P and metals cycling (Mayer et al. 2008, Tromp et al. 2012). In general, substantial knowledge gaps exist for pond performance on annual time scales, and on the role of winter-related processes (snowmelt, freeze-thaw, and road salt accumulation). Such information and understanding is especially important in the face of climate change, and the likelihood of warmer, wetter winters with more frequent runoff events in Minnesota.

Finally, given the rarity of annual scale studies and the importance of climate and hydrology to pond performance more generally, we also see a need for more research on the resilience of
pond function under changing climate. This is a research need that is relevant across all water quality (Q2) and hydraulic topics (Q1).
Q2.3: Heavy Metals

Background and Performance Trends

Metals are ubiquitous in the environment and can be present at concentrations in urban stormwater runoff that are potentially harmful to aquatic life and human health. Metals in urban runoff primarily originate from vehicular sources (automobile exhausts, vehicle wear, brake pads), buildings and rooftops, and from atmospheric deposition. The most common metals found in urban runoff and that are frequently reported in literature are cadmium (Cd), chromium (Cr), copper (Cu), lead (Pb), nickel (Ni), and zinc (Zn). Metals in urban runoff occur in particulate, dissolved or colloidal forms, depending on their origin and other water quality parameters that influence speciation. Most metals are predominantly associated with particles, especially the finer organic matter fraction, although a significant fraction of some metals (copper, cadmium, zinc) may exist in dissolved phases (Pitt et al. 1995 and several others). The partitioning characteristics of different metals on different particle size ranges (Sansalone and Buchberger 1997, Johnson et al. 2003) influence the extent to which particles and associated metals will be retained by particle settling in stormwater ponds.

Per the International BMP Database (Clary et al. 2020; also see Table 1), the removal of total metals (i.e., particulate + dissolved forms) in runoff is largely achieved and is significant through detention basins, retention ponds and wetland basins. The concentration reductions, however, depend on the metal as some metals are more commonly detected and present at higher concentrations (zinc, copper, chromium, lead) than others (cadmium). Dissolved metal removal performance in ponds is mixed with either marginal or no reduction observed for most metals (Sebastian et al. 2014, Egemose et al. 2015). However, dissolved metal concentrations in the influent runoff often tend to be low, even below the laboratory detection limits, which makes the interpretation of removal performance difficult (Clary et al. 2020). On the other hand, such low metal concentrations also mean lower potential toxicity risks.

In the review of literature, a majority of studies evaluated pond sediments for metal pollution (concentrations, spatial and vertical patterns, relationships to watershed factors) when compared to studies determining concentration reductions in runoff flows. Although sediment assessment does not directly quantify the runoff metal removal performance, the overall results of metal accumulation levels in pond sediments strongly indicate that most stormwater ponds are functioning well to trap metals found in runoff.

Mechanistic Drivers

The primary removal mechanism for metals in ponds is via sedimentation; the fraction of metals that can be removed by sedimentation is dependent on the partitioning and particle size associations of the metals (Ferreira and Stenstrom 2013, Clary et al. 2020, many others). Our review of studies found that metal concentrations in pond sediments vary substantially among ponds, driven by the runoff particle characteristics, metal species, pond and sediment characteristics, and land use type and watershed setting (Bishop et al. 2000, Liebens 2001, Clozel et al. 2006, Crawford et al. 2010, Frost et al. 2015). Within the pond area, outlet sediments were found to contain higher metal concentrations than inlet and sediment forebay, since the finer particle fractions contain the highest concentrations and accumulate near the outlet (Clozel et al.
Substantial metal accretion in the upper sediments and decrease in concentration with depth was a common observation in several ponds, although the concentration-depth relationships varied by metal species and were influenced by the sediment type and size the metals associated with (Nightingale 1975, Lee 1997, Karlsson et al. 2010). Organic material increased metal sequestration in the sediments, which possibly prevents downward migration of metals when compared to sandier soils (Pernak and Leschber 1992, Zohar et al. 2017). Land use and pond location in the watershed had a substantial effect on sediment metal concentrations, since higher contamination was found at industrial ponds than at residential ponds combined with proximity to urban sources of metals (Wigington et al. 1983, Mayer et al. 1996, Polta et al. 2005, Crawford et al. 2010, Grauert et al. 2012, many others).

Plant uptake of metals (phytoremediation) has been investigated for dissolved metal removal in ponds and floating treatment wetlands and was found to be a function of plant type and density among other factors such as contact time with plants (de Beauregard and Mahy 2002, Weiss et al. 2006, Ladislas et al. 2013, Perron and Pick 2020). A higher accumulation of metals in the belowground mass (roots) than the aboveground mass (shoots) has been observed (Ladislas et al. 2012, Laffont-Schwob et al. 2015); this has important implications for plant management in ponds because plant harvesting will provide temporary sequestration of metals that may re-release after plant senescence while plant uprooting will enable the complete removal of metals from the system. Plant uptake may play a smaller role than sediment trapping for metal removal (Laffont-Schwob et al. 2015), but plants can be used as bioindicators for metal pollution in stormwater ponds (Ladislas et al. 2012).

Release of accumulated metals may occur due to changes in pH and redox conditions, and due to organic matter degradation, thereby affecting the overall and seasonal metal removal performance (Yousef et al. 1990, Clary et al. 2020). The metal removal performance in ponds was observed to drop by half during winter (Semadeni-Davies 2006). Trends associated with seasonality of metal removal have also been attributed to pond concentrations of chloride from road salts. Road salt can impact metals solubility and oxygen demand in pond sediments through association with volatile constituents (Tsavdaris et al. 2013). Higher salt concentrations coinciding with higher metal levels in pond sediments were observed by Gallagher (2011), suggesting the possibility of combined toxicity risks due to metal-salt interaction. Toxicity from metals in water and in sediments on invertebrates and other biota has been studied, although the negative impacts and toxicity risks can be site-specific as bioavailability is influenced by the sediment conditions (Stephansen et al. 2016). If present at highly toxic levels in sediments, the reuse and disposal of sediments removed from ponds and sediment forebays must be carefully assessed (McNett and Hunt 2011).

**Assessment Methods**

Metals in stormwater ponds are commonly assessed by sampling sediments and water. Metal accumulations in vegetation for phytoremediation studies (de Beauregard and Mahy 2002, Weiss et al. 2006, Perron and Pick 2020) and in invertebrates and other biota for ecotoxicity studies (Stephansen et al. 2018) have also been performed.
Design Recommendations

Generally, higher metal retention is achieved by higher sediment accumulation. Pond designs that promote longer hydraulic retention times and particle settling are therefore recommended (see discussion in the section on Sediment Removal). The presence of upstream swales and sediment forebays can reduce the total metal load input to ponds (Roinas et al. 2014, Schifman et al. 2018). Plants can provide additional reduction in dissolved metal forms (Tanner and Headley 2011).

Knowledge Gaps and Research Needs

Metal concentrations and distributions in urban runoff, and metal removal and accumulation in ponds has been extensively researched. The impact of high chloride concentrations on metal complexation and mobility suggests that more research is needed to establish joint trends in seasonality of chloride and metal removal in ponds. Only a few studies have examined relationships between metal and PAH accumulation in ponds (Kamalakkan et al. 2004). Phytoremediation is a helpful mechanism for metal removal, but a more comprehensive dataset on assimilation of both metals and nutrients in different plant species will be helpful for pollutant and plant management in ponds.

Q2.4: Polycyclic Aromatic Hydrocarbons

Background and Performance Trends

Polycyclic aromatic hydrocarbons (PAHs) originate from natural and anthropogenic sources in the environment. In urban settings, the primary cause of PAH pollution is often due to anthropogenic sources including combustion products (incomplete combustion of petroleum, oil, coal and wood), motor vehicles (leaking motor oil, gasoline, tire and brake wear), road debris, road surface abrasion, seal coat materials for asphalt paved parking lots, and industrial processes (Polta et al. 2006). An increased degradation of sediment quality due to PAH contamination has been observed in urban watersheds due to increasing urban sprawl over the past two decades (Van Metre et al. 2000). PAHs are pollutants of great concern due to their toxic and carcinogenic effects, persistent nature and ability to bioaccumulate.

In stormwater ponds, PAHs will mostly settle out to the bottom sediments. Due to their extremely low solubility and hydrophobic nature, most PAHs, especially the heavy-weight PAHs, are associated primarily with particulate matter and are thus more predominant in urban stormwater runoff than their dissolved forms (Bathi et al. 1999, Hwang and Foster 2006). In the review of literature, several studies have documented the presence of PAHs in stormwater pond sediments in Minnesota (Polta et al. 2006, Crane 2014, Huang et al. 2019) and beyond (Kamalakkan et al. 2005, Flanagan et al. 2021), indicating ponds act as sinks for PAHs. The studies have largely focused on evaluating the sediments for PAH contamination and thus reported the sediment concentrations of PAHs, including the type of PAHs, the spatial and vertical distribution in the pond and nature of particle associations; however, a quantitative measure of performance (i.e., percent reduction in runoff) has not been reported.
Overall, the studies report an extreme variability in PAH concentrations in pond sediments. Typically, higher concentrations of PAHs are measured in finer size particles than the coarser fraction since PAHs are mostly associated with organic aggregates rather than mineral forms (El-Mufleh et al. 2013). The differences in particle associations not only results in spatial differences in the amount and type of PAHs from inlet to the outlet of the pond, but also increases the possibility of dispersion and accumulation of PAHs beyond the pond (Kamalakkan et al. 2005). While some pond sediments are dominated by heavy PAHs (4-6 molecular rings), other sediments contain light-weight PAHs (2-3 rings), suggesting the presence of different sources of PAHs to the ponds (i.e., combustion processes vs. fossil fuel leaks vs. coal tar sealcoat). The influence of major land use type on the PAH source apportionment and contamination levels are possible, i.e., lower PAHs in ponds in residential catchments than in industrial/commercial and road catchments (Kamalakkan et al. 2005, Benardin-Souibgui et al. 2018), although significant correlations with land use and pond characteristic were not always found (Crane 2014, Huang et al. 2017, Flanagan 2021). The deposition of PAHs relative to heavy metals (Cr, Mo, Mn, Co, Zn, Fe, etc.) was also not conclusive in one study (Kamalakkan et al. 2005) likely due to different physical and chemical characteristics and transport processes. The variabilities and lack of trends could be due to the complex chemical structure of PAHs that dictates their behavior, distribution, and fate in the environment.

Mechanistic Drivers

The persistence of PAHs in the environment is dependent on a variety of factors, such as the chemical structure, concentration, dispersion, and bioavailability of the PAH. Additionally, environmental factors such as pH, temperature, soil type and structure, and presence of adequate levels of oxygen, nutrients and water for the activity of the PAH-degrading microbial community (bacteria and fungi) control the fate and persistence of PAHs (Bamforth and Singleton 2005). PAHs are readily degraded by photooxidation and reaction with atmospheric oxidants. However, PAHs can be associated with other pollutants such as hydrocarbons and heavy metals (Kamalakkan et al. 2005) that are more readily degraded aerobically than PAHs, which increases the overall residence time of PAH in the environment (Bamforth and Singleton 2005). Under anaerobic conditions, PAH degradation is severely limited which means, PAHs will be very stable and have prolonged existence in an anoxic environment such as pond sediments.

Assessment Methods

The most common method for PAH assessment is by sediment sampling, as PAHs are mostly bound to sediments. Sediments collected by dredging or coring are subject to analysis for a suite of PAHs and evaluated in terms of benzo[a]pyrene equivalent concentration (which is determined using the relative potency factors provided by the MPCA; Polta et al. 2006) that is compared to industrial soil reference value (SRV) to evaluate the extent of contamination. The dissolution (bioavailability) of the PAHs are typically assessed separately via bench-scale or laboratory studies.

Design Recommendations and Treatments

Accumulation of PAHs in pond sediment has important implications for sediment management (i.e., removal and disposal). Design measures, such as sediment forebays, to reduce sediment
load input to ponds could limit the PAH accumulation in the pond (Schifman et al. 2018, Flanagan et al. 2021).

Bioremediation of PAH-contaminated sediments by indigenous microbial population can be achieved by adjusting the water, air, and nutrient supply (biostimulation) and by inoculating with microorganisms with known pollutant-transforming abilities (bioaugmentation). Examples of these processes are land-farming and biopiling (several studies cited in Bamforth and Singleton 2005). Kyser et al. (2010) conducted bench-scale experiments using compost to biodegrade PAHs in pond sediments but the results were less successful. Huang et al. (2019) conducted a preliminary treatability study in the laboratory that showed reduction in PAH bioavailability by the addition of powdered activated carbon to sediments. Studies outside the pond literature have studied the ability of a consortium of bacteria and fungi for the bioremediation of PAHs (Bamforth and Singleton 2005, Sheng et al. 2009). While aerobic degradation of PAH is well documented, there seems to be minimal possibility of biodegradation under denitrifying, sulfate-reducing and methanogenic conditions that are often found in stormwater ponds. One potential disadvantage of in situ anaerobic bioremediation of PAH, however, would be the concomitant release of redox-sensitive nutrient species such as phosphate from the sediments.

Phytoremediation may be difficult and inadequate due to sensitivity of plant species to PAHs (Díaz-Ramirez et al. 2003), thus combining the use of plants and plant growth promoting bacteria for clean-up of PAH-contaminated soil has been suggested (Glick 2003, Sheng et al. 2009).

Knowledge Gaps and Research Needs

One of the consistent findings in the literature is the effect of material composition (organic vs. inorganic) of runoff particulates and sediments on the PAH associations. While some studies have focused on the nature of PAHs in different fractions of the sediment, only a few studies have investigated the nature of the particles themselves (El-Mufleh et al. 2013). Methods to identify the specific signatures of the sources of anthropogenic particles in the pond sediments could be used as a surrogate for chemical analysis for PAH (Flanagan et al. 2021).

Remediation measures for PAHs in stormwater ponds was not found in the current literature search, and only selected research reports (Kyser et al. 2010, Huang et al. 2019) were included in this review. There is a substantial research need for PAH remediation measures for pond sediments. Currently, the main means of disposing of dredged sediments contaminated with PAHs is disposal in a confined disposal facility.

Q2.5: Road Salt and Chloride Accumulation and Transport

Background and Performance Trends

Road salts are frequently applied to control ice in northern temperate regions, particularly in Minnesota. Chloride is a large component of road salt, and road salts are the most common source of chloride in stormwater ponds. Stormwater ponds, especially in northern temperate regions like Minnesota, tend to have extremely high levels of chloride. However, stormwater ponds are not designed to reduce chloride loads, so most of the chloride entering stormwater ponds is eventually exported downstream or enters the groundwater.
**Mechanistic Drivers**

Designed and constructed ponds were not designed to remove chloride, and like most dissolved constituents, chloride is simply exported downstream. Stormwater ponds do serve as a way to delay the timing of chloride release, which allows for some dilution of chloride but also can extend the time period in which toxic levels of chloride enter downstream waterbodies.

Stormwater ponds have some of the highest salt concentrations among freshwater waterbodies (Casey et al. 2013), and these elevated concentrations of chloride are toxic for the biota within the stormwater ponds (Gallagher et al. 2011, Tixier et al. 2012). Taxa found in stormwater ponds are correlated with chloride concentrations and are generally tolerant of higher levels of pollution (Meland et al. 2020).

Toxicity due to chloride concentrations can be increased by interactions with other pollutants including metals. Elevated levels of chloride in stormwater ponds can lead to increased metal mobilization from the sediments (Mayer et al. 2008) and ultimately allow export of metals during heavy salting periods (Tromp et al. 2012). The combination of metals and salts can lead to interactive effects between the two, further exacerbating the toxicity of stormwater ponds. Road salts can also impact oxygen demand in the sediment (Tsavdaris et al. 2013).

High concentrations of chloride can lead to freezing point depression in stormwater ponds. Ice that experienced a freezing point depression of 0.15 ºC melts 26% faster than ice without a freezing point depression at an ambient temperature of 0 ºC (She 2016). The effects of freezing point depression decrease as the ambient temperature increases, so the impacts of freezing point depression are reduced at higher and lower ambient temperatures.

**Assessment Methods**

Chloride concentrations in stormwater pond can be monitored directly through continuous sensors but are often approximated using continuous sensors for specific conductivity.

**Design Innovations**

Our review did not identify any design innovations associated with chloride reduction or removal within stormwater ponds. The most effective way to reduce chloride concentrations would be to reduce the amount of chloride applied as road salts. In order to reduce existing high chloride concentrations, dilution appears to be the best option.

**Knowledge Gaps and Research Needs**

Stormwater ponds currently lack the design to manage or reduce chloride concentrations, hence modifications (pond depth, vegetation) to reduce impacts from chloride runoff are needed. Further evaluation on how timing and duration of chloride export from existing ponds may affect downstream water quality is also necessary. There is a need to evaluate chloride-induced chemostratification in ponds and its impact on pond processes, including seasonal stratification, mixing, oxygen dynamics, and nutrient availability. Chloride can interact with metals at the sediment-porewater interface and complex with metals from the sediments, ultimately allowing for the export of metals to downstream water bodies. More research on impacts to pollutant
retention due to increased chloride levels and possible interactive and toxicity effects on pond biota is therefore recommended.
Q2.6 Bacteria and Pathogens

Background and Performance Trends

Bacteria and pathogens are understudied in stormwater ponds. Stormwater ponds were not designed to manage bacteria and pathogens, but they are present in many stormwater ponds. Clary et al. (2020) showed that ponds and stormwater wetlands generally provided significant reduction of both E. coli and fecal coliforms (44% - 95% reduction in median concentrations across pond types; Table 1), though studies that included these indicators were relatively rare.

Mechanistic Drivers of Performance

Bacteria and pathogens in water are mainly measured through indicator bacteria, primarily E. coli and fecal coliform. There are few studies on bacteria and pathogens in stormwater ponds, but the studies encountered in this literature review found that indicator bacteria were retained by pond and wetland mesocosms, but could also be a source of E. coli and fecal coliform (Struck and Borst 2008, Hathaway et al. 2009). Ponds had higher microbial diversity during storm events and tended to provide high retention of microbes but did not reduce their concentrations (Vander Meer et al. 2021). Stormwater runoff can transport animal fecal matter, which is one of the main sources of microbes in stormwater to stormwater ponds (Pettersson and Astrom 2009). Sewage overflows can also be of concern. Saxton et al. (2016) found that geography and land use did not seem to influence bacterial or viral communities, but rather that site-specific factors had the greatest impact on bacterial and viral communities in stormwater ponds. Overall, bacteria concentration decrease with increased retention times and exposure to sunlight (Vergeynst et al. 2012).

Knowledge Gaps and Research Needs

We found very little research on bacterial and pathogen populations and dynamics within our literature review, but research indicates that exposure to UV light through extended residence time as well as sorption, sedimentation and filtration are all viable treatment options for indicator bacteria. The optimization of bacteria removal and reduction is a research need. The dynamics of bacteria and pathogens are not well understood, and bacteria and pathogens in stormwater ponds face a variety of outcomes: some wet ponds show up to a 70% reduction in indicator bacteria while others may be sources of indicator bacteria to receiving waterbodies (Hathaway et al. 2009, Hathaway and Hunt 2012). It may be possible to optimize bacteria removal through real time control (Vergeynst et al. 2012). Real time control could allow for increased residence time and exposure to sunlight, which could reduce indicator bacteria concentrations. Overall, we need a better understanding of the mechanisms contributing to indicator bacteria persistence in stormwater ponds in order to reduce their concentrations.
Q2.7: Heat Accumulation

Background and Performance Trends

Heat accumulation in stormwater ponds is a broad topic with implications for a number of within pond processes. Heat is both added to ponds from warm runoff from pavements and increased during storage of water within the pond. Thermal impacts affect pond biota, stratification and mixing regimes within the pond in addition to downstream impacts on water quality. In this review, we considered all aspects of heat accumulation within ponds, including inputs and outputs as well as changes in heat due to water storage and stratification, and the impact heat and water temperature has on other pond processes.

Mechanistic Drivers of Performance

Stormwater ponds primarily receive stormwater runoff from roads and other impervious surfaces. In some cases, the incoming water is warmer than the water within the pond. Hester and Bauman (2013) observed large temperature fluctuations in ponds and streams after summer rainfall, which they attributed to the warming effect of pavement runoff. Large fluctuations in temperature are detrimental to aquatic life and are of particular concern to coldwater streams and habitats which require cooler water to maintain biodiversity and habitat quality. Some ponds are constructed specifically to mitigate the impacts of warm water runoff, and all ponds, intentionally or not, provide habitat for numerous taxa.

Within pond processes can be impacted by temperature fluctuations as well. Changes in temperature can lead to short circuiting, impacting pond hydraulics and reducing the effectiveness of baffles. Counter currents can develop, which could cause resuspension of sediments, also reducing the effectiveness of designed and constructed ponds to capture and retain sediments (Hendi et al. 2018).

Urban ponds are much hotter than natural ponds and have higher temperature fluctuations (Brans et al. 2018). This is due in part to the urban heat island effect, which traps heat in urban areas and causes an increase in air temperature that is also translated to an increase in water temperature for urban ponds. Ponds are also small and shallow water bodies, so less energy is required to increase water temperature. There are several ways that can help reduce the impacts of the urban heat island (see Knowledge Gaps, below).

Wet ponds add total heat energy to runoff but reduce the rate of heat flow by reducing the runoff rate at pond outlets (Herb et al. 2009). Similarly to chlorides, this leads to a tradeoff between immediate impact and duration of the impact of thermal fluctuations. Shallower ponds have the most pronounced temperature fluctuations, with water temperature most varied in surface depths (0 to 1.2 m). Sensitivity analyses on a computational model (Sabouri et al. 2016) indicated that increasing the permanent pond volume from 2,000 m³ to 4,000 m³ resulted in an average increase of 0.5 °C in outlet event mean temperature due to an increase in surface area with the associated access to solar radiation, while a similar increase in outflow temperature (0.6 °C) arose through a simulated doubling of pond flow path length.
Design Recommendations and Innovations

Ponds that are less than 1.2 m deep experience the greatest temperature impacts (Sabouri et al. 2016) and wind sheltering can also increase heat retention (Herb et al. 2009). The literature recommends the use of vegetated pond borders, non-artificial substrates, and increased depth to reduce heat impacts. If heat impacts to downstream water bodies are of concern, deeper ponds with bottom draw outlets have a lower heat impact on receiving water bodies (Sabouri et al. 2016). However, this strategy presents new complications for maintenance and water quality as a number of pollutants (including chloride) accumulate at the bottom of stormwater ponds.

Knowledge Gaps and Research Needs

Increasing temperatures due to climate change will have a significant impact on aquatic ecosystems. When climate change is coupled with the Urban Heat Island Effect in shallow waterbodies like stormwater ponds, the effects are intensified. Water temperature affects mixing and stratification, thereby impacting nutrient cycling and settling times. Further research is needed to identify the impacts that increased temperature will have on within pond processes. Increased temperature is likely to impact nutrient cycling and dissolved oxygen availability as well as rates of decomposition and burial. The interactions between heat and pond processes are not well researched, and understanding the interactions between heat, depth, and mixing and stratification is necessary for predicting future pond performance.
Q3: Current Pond Assessment Methods

A variety of methods are used to assess pond function and performance. We summarize the most commonly encountered methods here, including data collection techniques for hydrologic and water quality variables (Table 2) as well as approaches for analyzing data and estimating pond performance (Table 3). We list some of the primary advantages and disadvantages of these common assessment techniques. Particularly novel or unique methods are discussed in the topic sections above, i.e., for phosphorus, metals, and PAHs.

The most common hydrologic assessment methods were direct monitoring of water level or inflows and outflows at ponds. A few studies investigated hydrologic impacts to receiving waters, typically streams where gaging stations were located downstream of where pond(s) discharged to the streams. Most hydrologic data were collected continuously or on an event basis, as is typical for pond and stream monitoring programs using automatic samplers.

For water quality assessment, methods were more varied. Autosamplers or grab sampling were used for water sampling, with roughly equal numbers of studies using either method; sediment sampling by dredge or by core extraction was also very common. A few studies sampled plant tissues or used continuous sensors (dissolved oxygen or conductivity most commonly, or phosphate or nitrate). Nearly two-thirds of the studies that included water quality assessment used multiple assessment methods, reflecting efforts to more completely characterize pond performance. Note that we did not take notes on the use of hand-held water quality instruments (Sonde-type), though these were used in the majority of studies with field or laboratory components to record temperature, oxygen, conductivity, oxidation-reduction potential, turbidity, etc. Water quality sampling intervals were also quite varied, though most samples were collected on the basis of storm events or regular intervals (e.g., weekly), with the more spatially extensive studies typically using a single (or at most, a few) sampling efforts in a large number of ponds.
Table 2. Assessment methods for data collection. Data collection refers to any data that is directly collected in the field and includes spatial or temporal measurements.

<table>
<thead>
<tr>
<th>Method</th>
<th>Constituent/Process Assessed</th>
<th>Advantages</th>
<th>Limitations</th>
</tr>
</thead>
<tbody>
<tr>
<td>Automatic sampler (e.g., ISCO sampler)</td>
<td>Concentrations and load of constituents, and hydrology</td>
<td>Characterize fluxes at a range of temporal scales</td>
<td>Labor intensive with higher equipment and analytical needs</td>
</tr>
<tr>
<td>Continuous sensors</td>
<td>PO₄, NH₄, conductivity, dissolved oxygen, temperature, chloride, turbidity, and others</td>
<td>Easily used in the field for immediate readings; continuous monitoring data collection. Temperature and conductivity are most common.</td>
<td>Discretized vertical profile are needed given strong gradients in chemical conditions. PO₄ and NH₄ are very expensive for a single sensor</td>
</tr>
<tr>
<td>Water level</td>
<td>Volume or pressure change over time</td>
<td>Can be used to estimate inflow and outflow volumes over time.</td>
<td>Calibration with other hydrology data needed to ensure accuracy</td>
</tr>
<tr>
<td>Aerial monitoring</td>
<td>Extreme storm volume, flooding, morphology, floating plant distribution, land cover</td>
<td>Typically high temporal resolution and spatial coverage; can record changes in land cover through time.</td>
<td>Field validation needed to ensure accuracy. Spatial resolution may be limited by the type of sensors.</td>
</tr>
<tr>
<td>Grab samples</td>
<td>Concentrations of dissolved constituents and suspended solids</td>
<td>Field sampling equipment is not required.</td>
<td>Does not capture storm event dynamics, and determining event mean concentration is not possible. Load measurement is not possible</td>
</tr>
<tr>
<td>Sediment sampling</td>
<td>Sediment characteristics and concentrations</td>
<td>Can be completed in a short period of time; integration and retrospective information</td>
<td>Complex relationship to water column conditions and concentrations</td>
</tr>
<tr>
<td>Plant tissue sampling</td>
<td>Plant mass and characteristics</td>
<td>Much of key nutrients can be contained in plants; improved mass balance.</td>
<td>Complex relationships between plant and water column concentrations</td>
</tr>
</tbody>
</table>
Table 3. Assessment methods for performance analysis. Performance analysis refers to the way collected data is interpreted and includes techniques such as paired watershed studies and inferences made from receiving waterbody data.

<table>
<thead>
<tr>
<th>Method</th>
<th>Constituent/Process Assessed</th>
<th>Advantages</th>
<th>Limitations</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mass-balance/budgets</td>
<td>Primarily for nutrients, also metals and other constituents</td>
<td>Load calculations</td>
<td>Need for intensive monitoring of hydrology and water sampling and analyses</td>
</tr>
<tr>
<td>Modeling</td>
<td>Any process or constituent</td>
<td>Integration, scaling and synthesis of information</td>
<td>“All models are wrong, but some are useful” - G. Box</td>
</tr>
</tbody>
</table>
Research Needs and Knowledge Gaps

We identified several areas of broad research needs from the literature review, generally with relevance across several intended (Q1) or unintended water quality (Q2) functions of constructed stormwater ponds. In Part I, we present these knowledge gaps related to understanding of environmental factors affecting pond function. In Part II, we present research needs related to assessment methods (Q3) and the modification and redesign of existing ponds for improved function (Q4).

Part I: Improving Predictive and Mechanistic Understanding of Ponds for Water Quality

While pond hydraulic functions (peak flow reduction and delay, energy dissipation) are largely well known, decades of research on water quality function of ponds have revealed that many drivers of pond nutrient and pollutant processing and retention remain poorly understood. We highlight several of these broad research needs (addressing Q1 and Q2) to improve predictive understanding of pond function and allow for more effective development and implementation of assessment, remediation, and modification approaches (described in Part II below).

(1) Magnitude and environmental drivers of biogeochemical processes (uptake, transformation, burial, and release of pollutants) facilitated by vegetation and microbes

- **Organic Nitrogen**: For nitrogen, research is needed on the fate and transformation of organic nitrogen inputs, including conversion of PON to DON and NH4, to better understand the eutrophication risk for receiving waters, and optimize N removal functions of ponds.

- **Denitrification**: Where N management is a concern, additional research into the magnitude and controls of denitrification within ponds would help determine how management strategies (e.g., increasing pond volumes, detention times, or vegetation coverage) might be optimized to promote N removal via denitrification. In general, the conditions necessary for high rates of denitrification are present in wet ponds and treatment wetlands (carbon, low oxygen, and nitrate).

- **Vegetation**: more work is needed to better understand the role of all types of pond vegetation (macrophytes, phytoplankton, floating plants) on annual nutrient budgets (N, P, metals), and especially how plants might be managed or manipulated to better retain nutrients and other pollutants. The impacts of long wet or dry periods, long-term storage in above-ground vs. below-ground biomass (Griffiths et al. 2021), and the fate of nutrients during and after fall senescence (of both floating and rooted plants (Schroer et al. 2018), remain under-studied in the context of ponds.

- **Internal Loading of P**: Understanding and predicting the extent, magnitude, and controls on internal loading of P, especially from anoxic sediments, is a major knowledge gap. Given often high rates of microbial activity in ponds, breakdown and conversion of organic P is another potential source of internal loading that warrants further study (e.g., Song et al. 2017, Frost et al. 2019).

- **Nutrient Burial**: For N, P and carbon, more information is needed to understand stabilization and sedimentary burial of these elements, the intended and desirable fate for these elements for watershed nutrient management.
• **Chloride and Metals**: Interactions of metals and chloride may lead to mobilization of metals, which are typically relatively stable once buried in pond sediments. A greater understanding is needed of these interactions and, where metal contamination is a concern, development of methods to reduce potential release and export of metals during periods of high salt accumulation.

• **Greenhouse Gasses**: Although not a focus of this review, ponds are hotspots of greenhouse gas production, especially for CO₂ and CH₄ (Holgerson et al. 2018, Rosentreter et al. 2021, Goeckner et al. 2022). Management that reduces these fluxes by promoting C burial rather than mineralization is likely to be a priority in the future.

(2) **Influence of physical processes on pollutant and oxygen dynamics, especially the role of hydrology (evapotranspiration, groundwater inputs), stratification, and physical mixing**

• **Shallow or morphologically complex ponds**: The tradeoffs on P (and N) removal of increased residence time and oxygenation, versus internal loading and re-suspension risk, represents a knowledge gap in pond designs pushing for more complex and shallow geometries that increase residence time.

• **Stratification and Mixing**: the interactions of mixing forces (wind, runoff, heat), pond characteristics (depth, volume, sediment properties), hydrology (surface and groundwater levels, discharge), and oxygen dynamics to influence the release and burial of nutrients (P and N especially) was a major knowledge gap identified in the literature (McEnroe et al. 2013, Song et al. 2017, Frost et al. 2019).

• **Temperature**: A better understanding of temperature dynamics is needed, especially in the context of climate change and observation of pronounced temperature gradients and persistent stratification in many shallow ponds.

• **Hydrology and Nutrient Retention**: Volume reduction by wet ponds may lead to enhanced removal of P and N (e.g., Janke et al. 2022), and accumulation of sediment, P, and N was affected by impervious runoff (Schroer et al. 2018), suggesting strong hydrologic controls on pollutant removal that would benefit from further assessment. The use of active or real time control (RTC) to improve pollutant removal via hydrologic retention seems an especially promising direction of study.

• **Chloride**: Chloride inflow from the use of road salt in the winter and early spring tends to develop an intense chemo-stratification. The resulting influence on stratification and mixing dynamics and its persistence into the summer or later, is a related research need.

(3) **Harmful Algal Blooms (HABs): risk indicators and factors that contribute to HABs in ponds and downstream, towards better predictability and management**

HABs in lakes are a major water quality and health issue due to impairments to recreation and harmful effects to pets and wildlife. Ponds are often used as a recreational or aesthetic amenity by nearby residents, yet HABs in ponds are less widely studied and represent an emerging issue due to their abundance in the eutrophic conditions of ponds. Ponds may also play a role as incubators of potentially harmful cyanobacteria (the primary bacteria in HABs) that can be flushed into receiving waters. Ponds have conditions that are distinct from lakes and thus controls of HAB formation are likely to be substantially different, and these controls warrant investigation since ponds remain relatively understudied compared to lakes. Existing studies
were largely inconclusive, with few patterns emerging between microcystin concentration or cyanobacteria abundance and pond or watershed indicators.

(4) Understanding the temporal dynamics of hydrologic and water quality responses in ponds: winter impacts, annual and interannual variations, and resilience to climate change

- **Winter Impacts**: The role of winter-related processes (snowmelt, freeze-thaw, and road salt accumulation) on pond performance for nutrient (N, P) and pollutant (metals, sediment) removal is a substantial knowledge gap. Very few studies investigated year-round or early spring (snowmelt) performance of ponds, yet these periods may be important due to reduced storage capacity, enhanced inputs, and low microbial activity. This understanding seems especially important in the face of climate change, and the likelihood of warmer, wetter winters with more frequent runoff events in Minnesota.

- **Annual studies**: given the rarity and difficulty of winter monitoring, annual performance estimates of ponds in northern climates were rare in the literature review. These assessments are necessary to study the legacy effects of seasonal transitions (e.g., the impact of a cold winter or a prolonged summer on late season performance). Methods of direct or indirect assessment of annual removal performance for all water quality variables are needed.

- **Climate Change**: given the importance of climate and hydrology to pond performance, we also see a need for more research on the resilience of pond function under climate change, especially for hitherto unforeseen precipitation intensities or frequencies. This is a research need that is relevant across all water quality (Q2) and hydraulic function topics (Q1). Understanding the potential long-term (year-to-year) legacy impacts of extreme climate events such as record storms, prolonged drought, or severe (or mild winters), represents a related knowledge gap.
Part II: Improving Assessment (Q3) and Modification or Redesign of Ponds (Q4)

In this section, we describe the research need for better assessment techniques, which is most relevant to Q3, as well as summarize the knowledge gaps with relevance to applied technologies or innovations in stormwater ponds, which are most relevant to Q4. These technologies include a range of structural modifications (e.g., baffles, filters, forebays), chemical treatments (e.g., alum), active control, and watershed-scale planning approaches.

(1) Pond assessment techniques: how to assess the hydrologic and water quality function of numerous stormwater ponds

A need exists to rapidly and cost-effectively assess the function and performance of stormwater ponds, especially those that are aging and potentially suffering from reduced storage and removal capacity due to sedimentation. This issue of reduced capacity is most applicable to pollutant retention (sediment and P especially) as well as to issues of rate control and flood control (e.g., Guo 1997, Ahilan et al. 2019). Some evidence suggests that older ponds retain or have improved N retention capacity compared to newer ponds. Overall a more complete development and evaluation is needed for assessment approaches for all water quantity and quality aspects of ponds. Such understanding should be included in future work addressing research needs identified here.

(2) Chemical Treatments: Dosing, cost effectiveness, and long-term performance

Sediment treatments by chemical addition (e.g., alum, iron filings, lime, and lanthanum) for controlling internal loading of phosphorus have been successfully demonstrated in eutrophic lakes and a few ponds, yet more research appears needed on the dosing, cost-effectiveness, and longevity of treatment benefits, especially in ponds with high sedimentation rates. We encountered few, if any, studies discussing sediment treatments for remediation or burial of nitrogen, metals, or PAHs.

(3) Oxygen Dynamics in Ponds

- **Mixing with aeration devices**: It is challenging to bring much oxygen into shallow ponds with aeration devices, but they can be effective mixing devices. We found no guidelines or examples of aerator mixing in the literature. Mixing with aerators seems to be an art, installed and adjusted through trial and error. The placement of the aerator above the poorly defined sediment surface to avoid resuspension of the sediments is one reason that specialists are required. Mixing with aeration devices could be an effective means of bringing oxygen to the pond sediment surface, and is a research need. It is our opinion that fountains are not cost-effective for mixing but may be used if an amenity is desired for ponds. It is much more cost-effective, on a mixing basis, to use sub-surface aerators. Optimizing aeration to improve oxygenation while minimizing the risk of resuspension, including the choice and locations of mixing devices, is also a research need for shallow ponds.

- **Enhanced wind mixing**: Many ponds are well sheltered from the wind by pond banks and/or vegetation growing upwind from the pond. Mitigation of wind sheltering in order to mix ponds was largely not present in the literature and is a research need. The potential for sediment resuspension with enhanced wind mixing should also be considered.
• **Improving oxygen conditions with submerged macrophytes:** Aquatic plants have a complex relationship with the oxygen dynamics of ponds, because photosynthesis creates dissolved oxygen in the water, but plant respiration uses oxygen (Odum, 1956). This can cause large diurnal swings in dissolved oxygen concentration. During the growing period in late spring and early summer, plants can overall add oxygen to the water, but during late summer and fall, the respiration of plants overtakes photosynthesis and aquatic plants will reduce overall oxygen concentration in the water. Floating plants are of special concern with regard to oxygen dynamics in ponds, because most of the photosynthesis for this group may occur above the water surface, but much of the respiration takes place below surface. Control of plants to maintain dissolved oxygen is not common practice, probably due to the complexity and lack of understanding of methods.

(4) **Structural Modifications and Filtration Technology: Cost effectiveness, maintenance needs, and long-term performance**

• **Outflow structures:** pond outflow characteristics can be manipulated by changing outlet structure or configuration, but such changes should be informed by watershed models or stream data to ensure that site-scale changes (both individually and aggregated across a watershed’s multiple ponds and/or SCMs) have positive downstream flow impacts.

• **Baffles:** in-pond baffles are a potentially useful retrofit to improve sedimentation and reduce energy of incoming flows, especially for under-sized ponds, though more studies are needed to confirm their effectiveness in field conditions. The effect of baffles on short-circuiting in frozen ponds and ponds with high temperature inflows should be explored (Hendi et al. 2018).

• **Water Quality:** Structural retrofits (baffles, islands) have applicability to a number of pollutants found primarily in particulate form (P, N, sediment, metals, and PAHs), and their effectiveness could be a useful topic for future research.

• **Fine particulates:** Stormwater practitioners would benefit from an improved understanding of the extent to which fine or colloidal particle dynamics influence pollutant removal for different contaminant groups, and especially how these finer particles may or may not be removed by filtration and baffle modifications.

• **Filtration and amendments:** Filtration appears to be a promising pond modification, yet knowledge gaps exist on long-term performance and maintenance needs. The optimization of filter materials (e.g., sand, crushed concrete) as well as amendments (e.g., iron filings, water treatment residuals, spent lime, plants) is vital.

• **Filter maintenance:** maintenance needs of filtration technologies were not commonly discussed, though one study (Egemose et al. 2018) noted that most phosphorus was found in the upper 5 cm of the sand filter bed and recommended swapping out the upper layer regularly. Further study of filter maintenance requirements for optimal performance are needed.

(5) **Sediment forebays: water quality impacts, maintenance needs, and cost effectiveness**

Sediment forebays were a common recommendation in the reviewed literature for improved removal of sediment and sediment-bound pollutants (coarser particles especially) and a more manageable dredging volume relative to an entire pond. Several knowledge gaps were identified.
• **Water quality benefits**: sediment and phosphorus removal appeared to be enhanced by sediment forebays (e.g., Chiandet and Xenopoulos 2016, Griffiths and Mitsch 2020), but further assessment under field conditions for phosphorus is warranted. The benefits to removal of other nutrients and other pollutants (metals, PAH) were also rarely assessed and would benefit from additional study.

• **Fine particulates**: Certain pollutants are primarily associated with fine or colloidal particles, so pond retrofit methods such as forebays, which may be more effective for coarser materials, may be less effective for pollutant reduction. This issue requires further research, and is relevant to other pond retrofits (filters, baffles, islands) that enhance pollutant removal via enhanced sedimentation.

• **Maintenance**: Most studies suggested regular dredging of forebays would be needed (e.g., Merriman and Hunt 2014) to maintain effectiveness, at potentially more frequent intervals than the main pond. Further research may be needed to better understand maintenance needs and resulting cost effectiveness. Further, assessment of the water quality benefits associated with routine maintenance are needed, especially with respect to removal of metals and PAHs, as well as organic N (mitigating the potential consequences of conversion to dissolved forms).

• **Toxicity**: if effective at promoting sedimentation, forebays are likely to have sediments containing toxic pollutants (metals and PAHs) at concentrations exceeding their soil reference values (SRVs), which poses additional maintenance and sediment disposal concerns. Predictability of pollutant toxicity based on pond or watershed features is a further research need, relevant to ponds more generally.

• **Cost effectiveness**: We found no studies on the cost effectiveness of installing a sediment forebay relative to sediment and phosphorus removal, maintenance or their impact on sediment toxicity.

(6) **Real-time and Active Control: site control and watershed management, benefits beyond peak flow reduction and flood control**

• **High priority data needs**: Substantial amount of training data generally is required for adaptive control optimization at the watershed scale (using machine-learning approaches or similar), and a wide range of algorithms and approaches exist even for site-scale optimization. Before this technology can be widely implemented, simplification or facilitation of control setup is needed.

• **Field testing**: real-time control practices, and especially the algorithms and machine-guided approaches used to optimize their use at watershed scale, were tested primarily using extensive modeling exercises. Only a few were implemented and tested in the field (Kerkez et al. 2016), highlighting a need for more real-world demonstration of a promising technology.

• **Climate change**: Real-time control technologies have significant potential to move existing ponds from static control systems to adaptive control systems capable of managing the hydrologic uncertainties associated with climate change, helping address a substantial management need.

• **Winter and Snowmelt**: Active control on pond outlet structures has high potential for managing snowmelt if ponds are drawn down prior to ice-in, but while such an approach was mentioned in the literature, we are not aware of any studies putting it into practice.
• **Water Quality**: Real-time control applications have initially focused on hydrologic variables but will also have impacts on sedimentation rates and nutrient or pollutant removal (e.g., Middleton and Barrett 2008). Potential water quality benefits or impacts should be considered while pursuing such technologies.

(7) **Expanding Spatial Scope: Watershed-level planning and coordination of pond and SCM management**

Haphazard implementation of ponds to treat urban runoff has not been effective (James et al. 1987, Emerson et al. 2005, Hancock et al. 2010), and recent work has emphasized the need for watershed-level planning to inform pond placement and design. The impact at watershed scale of not just ponds, but all distributed SCM types, is a broad research need for more effective management of urban watershed hydrology and runoff pollution. In particular, several research needs are evident:

• **Local vs. Watershed scale**: stormwater ponds designed for on-site performance can also advance watershed-level flow reduction goals. Greater benefits for peak flow reduction appear to be realized when ponds are used in headwaters versus downstream portions of watersheds, with the latter placement likely resulting in under-sized ponds. How well these observations can be generalized across a range of watersheds remains an open question.

• **Optimization**: Several studies have developed methods for optimizing watershed-level pond placement and design using multi-objective genetic algorithms for integrated watershed planning (Yeh and Labadie 1997, Muleta and Kicklow 2004 - citations in Goff and Gentry 2006). Other optimization techniques can be used to support additional studies.

• **Interaction with other SCM types**: Watershed infiltration (LID-type practices distributed throughout the watershed) had a greater effect on volume reduction than modification of pond outlet structures, and also provided a greater benefit in small storms (Emerson et al. 2005, Giacomoni et al. 2014). More research is needed to help with planning and implementation across wider spatial scales, including the optimization of different SCM types.

• **Flooding**: Very few studies among the included papers assessed the potential benefits of ponds for reduction of local or downstream inundation extent and depth (though we acknowledge that a few papers excluded from the review included broader flooding assessments of multiple SCM types, typically using models and with little emphasis on ponds). Where flooding may be a concern, this represents a knowledge gap.

• **Cooperation and Coordination**: With a need to plan and implement at watershed scale, there is a potential social research need here with respect to improving cross-jurisdictional cooperation and coordination.
References


(5) Andradottir, H.; Mortamet, M. Impact of Wind on Storm-Water Pond Hydraulics. JOURNAL OF HYDRAULIC ENGINEERING 2016, 142 (10).


(151) Nightingale, H. I. Lead, Zinc and Copper in Soils of Urban Storm-Runoff Retention Basins. JOURNAL AMERICAN WATER WORKS ASSOCIATION 1975, 67 (8), 443–446.


Appendices

Note: The complete list of papers for Round 2 of the screening process can be found in an included reference list and a spreadsheet database, with the latter summarizing various information about the papers (e.g., study type, hydrologic or water quality variables investigated, geographic metadata). These resources include papers selected for the final review as well as those excluded for reasons described in the Methodology above, typically out-of-scope topics. In its current format, the sheet provides a valuable tool for locating references on specific topics or methodologies; it can be enhanced with additional information from future activities and inputs.

Appendix A - Literature Search Query

(Stormwater OR storm water OR retention OR detention) NEAR/2 (pond* OR basin*)
treatment wetland*
Dry (pond* or basin*) AND (stormwater OR storm water OR runoff)
Urban (pond* OR basin*) AND (stormwater OR storm water OR runoff)
constructed (pond* OR basin* OR wetland*) AND (stormwater OR storm water OR runoff)
Floating (pond* OR wetland*) AND (stormwater OR storm water OR runoff)
Appendix B - Decision Tree

Title and Abstract Inclusion Terms

**Stormwater Pond Terms:** stormwater pond/basin, retention pond/basin, detention pond/basin, wet pond, urban pond, constructed pond, designed pond, treatment wetland, floating treatment wetland

**Hydrologic Parameters:** rate control, peak flow rate, flood control, flood risk, volume control, retention/detention time, energy dissipation, stratification

**Water Quality Parameters:** sediment, nitrogen, phosphorus, organic matter, metals, chloride, PAH’s, pathogens, bacteria

**Additional Parameters of Interest:** impacts on biodiversity, aesthetic value, vegetation impacts, resilience to climate change, design innovations, oil/grease removal, impacts of maintenance and operations
Appendix C - Additional Exclusion Criteria

<table>
<thead>
<tr>
<th>Research Contexts</th>
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<tbody>
<tr>
<td>Wastewater (mining, industrial, domestic)</td>
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<tr>
<td>Agricultural runoff</td>
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<tr>
<td>Sediment basins for construction sites</td>
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<td>Stabilization ponds</td>
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<td>Temporary ponds</td>
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<td>Urban ponds without stormwater inputs</td>
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<tr>
<td>Hydro-energy production</td>
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<td>On-stream ponds</td>
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<td>Treatment wetlands that do not treat urban stormwater</td>
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<td>Floating treatment wetlands when not applied to urban stormwater ponds</td>
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<td>Substrates for pollutant removal not tested in a stormwater pond</td>
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<tr>
<td>Stormwater ponds designed to treat combined sewer overflows</td>
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<tr>
<td>Greenhouse gases, microplastics, biocides, pharmaceuticals</td>
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<tr>
<th>Paper Types</th>
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<tbody>
<tr>
<td>Conference proceedings</td>
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<tr>
<td>Books and book chapters</td>
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<tr>
<td>Theses and Dissertations</td>
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<tr>
<td>News articles</td>
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<tr>
<td>Bulletins</td>
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<tr>
<td>Technical Fact Sheets</td>
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<tr>
<td>Local city projects and future development plans</td>
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<tr>
<td>Design Manuals, Aids, Rulebooks, and Guidelines</td>
</tr>
<tr>
<td>Reports and other grey literature</td>
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<tr>
<td>Concept papers unrelated to pond design innovations</td>
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<tr>
<td>Methodology papers unrelated to pond monitoring</td>
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<tr>
<td>Studies with an exclusive economic or legal focus</td>
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<tr>
<td>Papers that mention stormwater ponds in passing, but don’t offer any performance data</td>
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<tr>
<th>Modeling Contexts</th>
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<tbody>
<tr>
<td>Modeling processes that occur in ponds, but the paper does not focus on stormwater ponds</td>
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<tr>
<td>Overly focused on the modeling approach</td>
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<tr>
<td>When stormwater ponds are used as a generic BMP to illustrate model applications</td>
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<tr>
<td>Calibration and validation of model results</td>
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<tr>
<td>One of the many “new” and “simplified” modeling methods for the design and sizing stormwater ponds</td>
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<tr>
<th>Additional Contexts</th>
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<tbody>
<tr>
<td>Papers that mention ponds in passing as an example stormwater control measure</td>
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<tr>
<td>Flood mitigation strategy papers where stormwater ponds are one of the strategies</td>
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<tr>
<td>All of the different approaches to designing stormwater ponds</td>
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<tr>
<td>Applications of new models</td>
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<td>Decision support tools and frameworks</td>
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<tr>
<td>Risk-based analysis for pond design - costs, probability of failure, consequences of failure</td>
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<tr>
<td>Pond inflow sampling to measure stormwater runoff quality, but no focus on the pond itself</td>
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