Illinois State Water Survey Contract Report 2022-03
December 2022

Watershed-Specific Release Rate Analysis Phase III: Cook County, Illinois

Gregory Byard[1], Bruce Rhoads[2], Arthur Schmidt[3], Robert Hudson[4], Nikhil Sangwan[1], Cecilia Cullen[1], Devin Mannix[1], Walton Kelly[1], Tasneem Meem[2], Maggie Gardner[3], Leo Fouts[3], Hunter Gross[4], Armando Zavalza[4]

[1] Illinois State Water Survey
[2] University of Illinois - Department of Geography and Geographic Information Sciences
[3] University of Illinois - Department of Civil and Environmental Engineering
[4] University of Illinois - Department of Natural Resources and Environmental Sciences

Illinois State Water Survey
Prairie Research Institute
University of Illinois Urbana-Champaign

Prairie Research Institute
UNIVERSITY OF ILLINOIS URBANA-CHAMPAIGN

Executive Summary

The Cook County Watershed Management Ordinance (WMO) was approved by the Metropolitan Water Reclamation District of Greater Chicago (MWRD, or the District) Board of Commissioners on October 3, 2013, became effective on May 1, 2014, and was most recently amended on April 7, 2022. Article 208 of the amended ordinance directs the District to consider the "impacts of watershed specific release rates on disproportionately impacted communities, the impacts of release rates under existing and future development scenarios in collar counties on watersheds in the District, and the impact of volume control and watershed specific release rates on stream erosion and related water quality effects such as turbidity and sedimentation." The Illinois State Water Survey (ISWS), in consultation with MWRD, convened an interdisciplinary team that included the ISWS Coordinated Hazard Assessment and Mapping Program and Groundwater Science Section as well as the University of Illinois' Department of Geography and Geographic Information Sciences, Department of Civil and Environmental Engineering, and Department of Natural Resources and Environmental Sciences to consider these impact assessments.

Urban development is typically accompanied by an increase in impervious area that can lead to increased runoff and more severe flooding. Stormwater retention and detention policies are commonly employed by regulatory agencies to mitigate these potential development impacts within their jurisdiction. The hydrologic and hydraulic impacts of such policies have been explored widely in the scientific literature as well as specifically as they pertain to the greater Chicago Region and the MWRD WMO in *ISWS Contract Report 2019-06: Watershed-Specific Release Rate Analysis: Cook County, Illinois* by Flegel and coauthors in 2019 (http://hdl.handle.net/2142/103416). The analysis and report that follows consider how these management policies are related to issues of social equitability, issues of multi-jurisdictional watershed management, and issues of stormwater quality.

Impacts of Watershed-Specific Release Rates on Disproportionately Impacted Communities

The WMO's watershed-specific release rate regulations aim to mitigate any potential increase in peak water surface elevations during large flood events due to new development by requiring detention storage and controlled release of runoff from these sites. This study seeks to examine the equitability of these regulations by comparing associated costs (detention storage requirements) and benefits (potential flood mitigation) for "Disproportionately Impacted Areas" or DIA. To identify communities highly susceptible to flooding based on existing conditions, MWRD defines these DIAs as areas that have a Chicago Metropolitan Agency for Planning Urban or Riverine Flood Susceptibility Index mean value of 5-10 and a Low to Moderate Income level as defined by the U.S. Department of Housing and Urban Development. Detailed Watershed Plan modeling developed by MWRD and to which future conditions scenario-based modeling was added by Flegel and coauthors (2019) was re-analyzed. Detention storage was quantified by subbasin using hydrologic modeling of future conditions under both the existing WMO regulations and more restrictive management options, coupled with hydraulic modeling to evaluate the flood mitigation potential along a stream reach, and subsequently analyzed spatially to identify any systematic inequities between management policies on DIAs. The results of this analysis found that although District DIAs generally require marginally higher (~6% more) detention storage at the time of development or redevelopment, they enjoy moderately higher flood mitigation levels (~0.24 ft more) than Non-DIAs. Differences in storage requirements between DIA and Non-DIAs at watershed and community levels are also generally mild, though flood mitigation levels exhibit much larger intra-watershed and intra-community variations between DIA and Non-DIA. The results include spatial and summary data parsed at both the watershed and community scale to aid policymakers in understanding these differences throughout their areas of interest. Gains in flood mitigation levels by adopting more restrictive release rates and insight into how changes in design rainfall may affect the interpretation of these DIA impact assessments are also provided. An improved understanding of the impact of release rates on DIAs enable policymakers and watershed managers to better evaluate whether policies address prevalent inequities in flood risk.

Impacts of Watershed-Specific Release Rates in Collar Counties

Prior to the adoption of watershed-specific release rates in the WMO by the MWRD Board of Commissioners, the Illinois State Water Survey conducted an analysis of the hydrologic and hydraulic impacts of various potential policies (Flegel et al., 2019). Although the 2019 analysis included future conditions modeling throughout the greater Chicago region, including those areas tributary to District waterways, only the potential management decisions by the District were considered. In areas outside of District jurisdiction, future conditions were modeled assuming any such development would occur under existing stormwater management policies. This study seeks to examine the impacts of hypothetical future changes to extra-jurisdictional stormwater management policies on the ability of the District's watershed-specific release rates to mitigate future increases in peak water surface elevation due to future development. Four watersheds with significant drainage area outside of the District were studied including the North Branch Chicago River watershed and the Lower Salt Creek, Addison Creek, and the Buffalo Creek subwatersheds. Future conditions hydrologic and hydraulic modeling based on the MWRD Detailed Watershed Plan modeling were created. The *ISWS Contract Report 2019-06: Watershed-Specific Release Rate Analysis: Cook County, Illinois* by Flegel and coauthors in 2019 (http://hdl.handle.net/2142/103416) found that release rates for development along the main stem of the Des Plaines River in Cook County alone will not mitigate water surface elevation increases due to future development, even without accounting for the projected impacts of future development in Wisconsin. Therefore, although the Lower Des Plaines River includes a significant amount of drainage area outside of the District, future development of the Lower Des Plaines River watershed was neither included in the evaluation of watershed-specific release rates in the previous report nor in this analysis. Areas within MWRD jurisdiction were subject to the prescribed watershed-specific release rate and areas within Lake or DuPage Counties were subject to a range of release rates from 0.10 to 0.30 cubic feet per second per acre (cfs/ac), the same release rates considered by ISWS in 2019 within the District. The Addison Creek and Buffalo Creek subwatersheds demonstrated a level of resilience to increases in peak water surface elevation due to future development under more permissive release rates in tributary areas outside of Cook County. The effective watershed-specific release rates are expected to be effective at mitigating future increases in peak water surface elevation due to future development for release rates in adjoining counties as high as 0.30 cfs/ac for Addison Creek and 0.25 (to potentially 0.30) cfs/ac for Buffalo Creek. The North Branch Chicago River watershed and Lower Salt Creek subwatershed, however, demonstrated a high degree of sensitivity to the selection of release rates within the adjoining jurisdiction. Although minor increases in the DuPage or Lake County release rate to 0.15–0.20 cfs/ac would likely be mitigated by the WMO watershed-specific release rate, additional increases to 0.25–0.30 cfs/ac would not, and increases in peak water surface elevation would be expected. The selection of a release rate in collar counties was thus found to influence whether the watershed-specific release rates prescribed in the WMO Appendix B will continue to mitigate future increases in peak water surface elevation due to future development, but not all watersheds were found to be sensitive to such changes. Watersheds with a substantial proportion of drainage area falling outside the WMO jurisdiction and with low average base condition runoff rates were the most sensitive, and those with only small portions of the drainage area or high average base condition runoff rates were less sensitive. It is recommended that the relevant watershed management agencies coordinate any changes in their watershed management requirements for multi-jurisdictional streams in the future. Early communication will provide managers with the most flexibility in responding to changing watershed dynamics. Watershed managers could also consider whether uncertainty in management practices outside of their jurisdiction should influence management practices within their jurisdiction.

Stream Channel Dynamics in Urban Settings: A Literature Review

Assessing the impact of volume control and watershed-specific release rates on stream erosion and related water quality effects such as turbidity and sedimentation required a review of the scientific literature regarding the stream channel dynamics that govern these processes in an urban setting, the relations between watershed management strategies and stream erosion, turbidity, and sedimentation, the underlying chemistry of urban stormwater pollution, and the potential impacts to not only surface water systems, but also the groundwater of Cook County.

The literature review of the impacts of urbanization on urban stream channel dynamics found several recurrent themes that provide a basis for generalization as well as considerable details that highlight the complexity of these impacts. 1) Urbanization fundamentally alters the hydrology of urban landscapes by increasing rates of runoff and, to some extent, volumes of runoff. As a result, the magnitudes of peak discharges for a specific recurrence interval increase, particularly for the most frequent flows. 2) Whereas construction activities may deliver large amounts of fine sediment to urban streams during the construction phase of urbanization, the long-term effect of urbanization on sediment delivery is complex but often involves reductions in sediment delivery from the watershed because of widespread coverage of the landscape by impervious surfaces. Delivery of sediment from within streams may increase during the urbanized phase because of increases in channel erosion. 3) The increase in peak discharges, along with channelization of many urban streams, often increases the bed shear stress and stream power per unit area of flows, resulting in an increased potential for mobilization of channel bed material and erosion of streambanks. 4) Although net deposition of sediment may occur on floodplains or even within streams during the construction phase, the most prominent geomorphic response of streams to urbanization is erosional enlargement through either expansion (simultaneous erosion of the channel bed and banks) or incision (downcutting of the bed followed by widening). This erosional response reflects the potential for increased mobilization of bed and bank material related to increases in the bed shear stress and stream power per unit area caused by the effect of urbanization on stream hydrology and hydraulics. Locally, the response also reflects spatial variability in rates of bed-material transport, with erosional sites likely to occur where the rate of bed-material transport increases in the downstream direction. 5) Efforts to mitigate increased flooding by increasing retention and storage of stormwater, while effective at reducing peak discharges and achieving peak-matching goals for non-urbanized watersheds, may increase the durations of transport-effective discharges (as storage water is gradually released) that could promote erosion of streams. This issue is understudied and is only beginning to receive attention within the research community.

These general understandings are broadly relevant to urbanization that has occurred and is continuing to occur within the greater Chicago region. However, it must be emphasized that the geomorphic dynamics of rivers are a function of two major factors: 1) general erosional and depositional processes related to the flow of water and movement of sediment that determine the form of stream channels and 2) environmental context, which determines exactly how those processes operate in any particular geographic setting to produce adjustments between process and form. Most of the research that has been conducted on responses of streams to urbanization consists of case studies in particular geographic settings. Because environmental context is important, generalizing beyond case studies is often difficult. Just because a stream adjusted a specific way at a specific place does not mean it will do so in another. To understand the role of context in adjustment, it is vital to have good information on that context. The literature reviewed in this report indicates that very little work has been done on the geomorphology of streams in greater Chicago, nor has basic data on these streams been collected that could inform geomorphological analysis. The review did not identify any scientific studies of major importance that examined the geomorphological response of streams in greater Chicago to urbanization. A critical need exists for basic geomorphological information on these streams before judgments can be made about possible morphological responses to stormwater runoff policies. A generalization that can be made is that if the sediment transport capacity exceeds the availability of sediment (either coming into a reach from an upstream reach or from delivery of material to a reach by stormwater runoff into it), the channel will erode, as long as it does not have an inerodible bed and banks, which is another unknown for many streams in greater Chicago. This basic idea serves as the foundation for the stream-power approach to assessing channel stability that is considered in the pilot analysis. This analysis represents an important first step toward achieving an improved understanding of how various stormwater policies might affect channel stability.

Biogeochemical Processes in Stormwater Best Management Practices

The summary included in the chapter *Biogeochemical Processes in Stormwater Best Management Practices* provides an overview of the processes through which stormwater policy such as volume control or detention requirements influence downstream water quality. Stormwater is precipitation that acquires additional solutes and particles as it contacts natural and constructed surfaces on the way to surface waters. These additional pollutants may have accumulated on urban watershed surfaces between storm events or be derived from incremental dissolution into stormwater of components of the constructed surfaces themselves. The wide variety of materials and activities conducted in urban watersheds means that differences in stormwater composition are often observed between residential, commercial, and industrial catchments. Differences can also arise from the flow path taken to the best management practices (BMPs) of interest because, for example, concrete and asphalt release different solutes. Urban soils can also be heavily compacted, causing some to resist water infiltration nearly as much as impervious surfaces. Stormwater is only in contact with these impervious surfaces for a short time compared to water infiltrating into soils, however. Thus, urban stormwater can acquire distinctive water chemistry that is very different from runoff in natural landscapes.

Although stormwater control measures are primarily aimed at reducing flooding, they can also have a secondary purpose of improving water quality in the waterways into which they discharge. Evaluating the effectiveness of a particular practice, as discussed in subsequent sections, requires an understanding of how the processes acting within BMPs affect a wide variety of pollutants of interest. The summary provided discusses the key terms associated with BMP water quality and an overview of the water chemistry principles needed to understand the key processes within these BMPs. One key factor in understanding BMP water quality impacts is to understand that the water chemistry reactions that determine the fate of individual constituents are unique to individual or classes of constituents, are influenced by the other constituents in the water at the time, and may only be held in a temporary equilibrium rather than permanently converted to another form. These equilibrium speciation reactions include acid-base reactions, metal complexation, sorption including both absorption and adsorption, and mineral precipitation.

The chemical constituents borne by urban stormwater are transported into BMPs in both dissolved and suspended particle forms. Once stormwater enters a BMP, several physical and biogeochemical processes will determine the fate of a constituent. These processes include sedimentation, filtration, infiltration, volatilization, sorption, precipitation and dissolution, abiotic transformations, vegetative uptake, and microbial transformations. The chapter includes a summary of the types of constituents likely to be affected by each process as well as the processes that act on them in many common examples of BMPs within the District.

Since stormwater BMPs are highly effective at retaining suspended particles, constituents that occur primarily in the particulate fraction usually are also removed quite efficiently. On the other hand, primarily dissolved constituents may or may not be removed or retained. In order for a constituent dissolved in stormwater to be removed before discharge to waterways or groundwater, that constituent must either 1) sorb onto the filter media or underlying soils of BMPs, 2) be incorporated into sediments via sorption, precipitation reactions, or uptake by vegetation, or 3) be transformed into a form that escapes to the atmosphere or is otherwise immobilized. Otherwise, the dissolved constituent will be exported from the BMP, as in the case of chloride.

Of course, infiltration practices are designed to divert stormwater and any exported constituents it bears from waterways to shallow groundwater. Since the groundwater may eventually flow back into a stream, the net water quality impact of infiltration depends broadly on what the fate of the constituent is in the subsurface environment. Non-sorbing, conserved constituents such as chloride should reach streams as the infiltrated stormwater does, though they may be diluted somewhat and spread out over time. Conserved constituents that sorb will experience slower transport (retardation), but should eventually reach streams as well. If the subsurface transport pathway is long enough that a constituent can be transformed or immobilized *en route*, infiltration will provide excellent protection to surface waterways.

Any constituent removed from stormwater must accumulate within the BMP, aside from a few such as VOCs, nitrate, and mercury that occur in, or can be transformed into, a volatile form. Accumulated constituents may eventually find their way back into stormwater via resuspension of sediments or remobilization, as observed for internal loading of phosphorus to stormwater from pond sediments.

Finally, note that since transformation processes cause changes in the state of a system and the levels of constituents within it over time, the longer stormwater is held within a practice, the greater is the extent to which processes that remove pollutants from or release pollutants to that water can progress.

Relations between Watershed Management Strategies and Stream Erosion, Turbidity, and Sedimentation: A Literature Review

The literature review of the relations between watershed management strategies and stream erosion, turbidity and sedimentation, and related water quality effects provides a current state of the science to help define the mechanisms of potential impacts of these management strategies. The review summarizes the impact of stormwater management practices on the magnitude and frequency of flows, water levels, and other hydraulic parameters such as stream power or shear stress downstream from these practices. Additionally, the review summarizes the impacts of stormwater management practices on nutrients (nitrogen and phosphorous), total suspended solids (TSS), iron, silver, and chloride. The review included both 1) the impact of watershed management strategies on downstream hydraulic and hydrologic effects, particularly as related to factors that may affect stream erosion, such as peak discharge, flow duration, shear stress, or stream power, and 2) the impact of watershed management practices on downstream water quality. The review considers the impacts of detention/retention practices, wetlands, and distributed small-scale practices.

From the literature, the primary metric used to evaluate the impact of stormwater control measures on downstream hydraulics and hydrology was change in flooding, usually on either peak discharge or volume, but occasionally on baseflow. Many studies examined the impact of the size and location of such practices within the watershed, typically by using watershed modeling of hypothetical scenarios. The studies tend to agree that BMPs generally have a positive impact on hydrology downstream, but the placement of practices does matter, and some practices can exacerbate flooding. The review found detention and retention ponds often reduce peak discharge but extend the receding limb of the hydrograph. This relationship can have impacts on the sediment transport in receiving systems as considered by the pilot analysis. Alternative management strategies have been considered that include alternate detention pond sizes, a design for matching pre-development sediment transport, and the effectiveness of these practices over time and as development patterns intensify.

Considering how different stormwater management options affect hydrology, a major theme of the literature is timing. Green infrastructure options like grass swales, rain gardens, biofiltration, open space, and forested floodplains lead to increased lag times in peak flows and to longer duration flows. In general, neighborhoods that are designed with low-impact development and green infrastructure tend to have reduced peak flows, greater lag times, and more baseflow than their conventional counterparts. The literature also reports differences in the discharge per unit area in watersheds using distributed stormwater control measures as compared to those using centralized stormwater control measures with the distributed measures having lower discharge per unit watershed area than centralized stormwater control measures for small (< 3 cm) events, but greater discharge per unit watershed area than centralized stormwater control measures for large (> 3 cm) storm events.

The review also included an extensive number of studies examining the removal efficiencies of stormwater management practices. These studies often varied considerably in both the category of removal (which form of a particular constituent was monitored) and how removal rates were quantified, with some studies reporting reductions based on concentrations and other studies reporting reductions based on loads. The studies vary widely in their reported removal of nutrients, with some studies reporting an increase in downstream nutrients and others reporting nearly 100% removal. The general trend was that longer retention times typically provide greater removal efficiencies of most solids and nutrients than those with shorter retention times. The removal efficiencies were also generally higher for more frequent, lower intensity storm events than they were for large storm events that occasionally demonstrated a net export of constituents, as constituents captured during previous low flow events could be re-mobilized at high flow. Constituent removal varies with the design, location, and age of the facility, with the season, storm characteristics, and operation plan, and sequencing of practice. The review summarizes and compares results by stormwater practices for individual constituents.

Evaluating Stormwater Management Policies' Effects on Water Quality: Monitoring Options

The literature review of the relations between watershed management strategies and stream erosion, turbidity, and sedimentation, as well as a review of the various monitoring or sampling methodologies available for confirming the effectiveness of these practices provide the District with several options to consider when evaluating the performance of their stormwater management policies. A consistent theme of the literature is that volume control and stormwater management practices are effective at mitigating many of the negative hydrologic and hydraulic impacts associated with urbanization, particularly those related to flooding, and are, in general, effective at improving water quality for many of the constituents of interest to the District. Extended stormwater detention, such as that afforded by more restrictive release rates, were generally more effective at removing certain constituents. Urban stormwater BMP monitoring can occur at a range of spatial and temporal scales and use a variety of methodologies. Each of these approaches is tailored to the evaluation of the effectiveness of specific types of management objectives. These methodologies include:

- BMP performance assessment
    - This methodology would be recommended if the District has concerns about volume control and release rates effectiveness at removal of novel contaminants for which little scientific literature is currently available (e.g. PFAS, microplastics, pharmaceuticals), but would not be expected to significantly improve confidence in the effectiveness of removal for constituents that have already been extensively investigated (e.g. nitrogen, phosphorous, TSS).
- Synoptic surveys
    - This methodology would be recommended if the District seeks to generally characterize effluent from volume control and detention practices, to understand pollutant removal/buildup in sediment or the incidence of internal loading (pollutant release from sediments to stormwater). It could also help in developing relationships between geographical characteristics and pollutant loads.
- Small watershed studies
    - This methodology would be recommended if the District seeks to deepen understanding of pollutant budgets and processes for specific types of volume control or detention practices or constituents and would allow unintended impacts or the relationships to design parameters to be explored.
- Sewershed/Watershed Monitoring
    - This methodology would be recommended if the District seeks to characterize stormwater effluent reaching receiving streams to identify critical contaminants by watershed, to identify trends in contaminants by season or over time, or to establish a baseline by which to evaluate future changes to stormwater management policies.
- BMP Census
    - This methodology would be recommended if the District seeks to understand the spatial distribution of specific types of volume control or detention practices or inventory the designed capacity or current condition of such practices. Such a census would improve confidence in the conclusions drawn from other modeling methodologies, enable more detailed watershed assessments, and assist in watershed or site sampling plans.
- Modeling
    - Modeling and data analysis plays an important role in interpreting the data generated by environmental monitoring and in supporting decisions that are based on them. For urban stormwater, models can be applied at scales ranging from BMPs to watersheds. Modeling is an effective way to analyze the impacts of proposed policies, identify spatial and temporal trends in water quality, and

evaluate areas of contamination risk. Modeling is recommended as a means for the District to efficiently and judiciously leverage investments in field sampling to provide the greatest understanding and confidence in stormwater policy and processes.

Impact of Volume Control and Detention Practices on the Groundwater of Cook County

The Illinois State Water Survey Groundwater Science Section conducted a review of the scientific literature to understand the impact of stormwater infrastructure on groundwater resources in Cook County apart from the City of Chicago. Although most communities in Cook County using groundwater had switched water supply sources to Lake Michigan by the early 2000s, citizens continue to interact with groundwater indirectly through recreational activities and with interaction with ecosystems dependent on groundwater. Sufficiently uncontaminated and abundant groundwater recharge is important for wetland and river ecosystems in Cook County. Additionally, regional shallow groundwater flow in Cook County moves toward the southwest suburbs, an area where communities are still dependent on groundwater for water supply. To understand whether infiltrating stormwater to groundwater in order to reduce contaminant loading in surface water leads to undesirable groundwater quality and ecological impacts in the MWRD region, the ISWS strongly recommends monitoring and sampling groundwater throughout the area.

A 2022 U.S. Geological Survey (USGS) summary of impacts of green infrastructure to the Great Lakes catchments calls the potential infiltration of stormwater contaminants to groundwater "one of the greatest potential negative consequences of green infrastructure." As groundwater interactions with stormwater management structures are a relatively recent focus of research, most referenced studies in this review are outside of Illinois or with a different geology than Cook County. The groundwater literature review summarizes the geologic setting in greater Chicago and describes how factors unique to this area interact with stormwater infrastructure and how this interaction may lead to unintended consequences of a stormwater management policy. A number of contaminants of concern likely to be found in the District and directly related to infiltration from stormwater best management practices were reviewed, including chloride, per- and polyfluoroalkyl substances (PFAS), metals, nutrients such as phosphate and nitrate, and other relevant contaminants.

From the literature, five main factors that increase risk of groundwater contamination on a site basis were identified: 1) if known contamination sources are in the drainage area, 2) if transmissive sediments or bedrock are near the surface, 3) if stormwater structures are designed for infiltration, 4) if stormwater structures do not receive regular maintenance, and 5) if nearby water table elevations are high relative to stormwater detention. Additionally, contaminants are more likely to infiltrate groundwater if they are highly soluble, high in concentration, or the soil sorption capacity is limited. Although we expect chloride, phosphate, nitrate, metals, and PFAS to be the most relevant contaminants to Cook County stormwater based on the land-use history, we recommend a thorough groundwater water sampling campaign to assess the suite of contaminants relevant to stormwater.

Because shallow groundwater use for community drinking water supplies has been limited in Cook County in the past several decades, current groundwater data are limited. To quantify the impact of stormwater management practices on groundwater in Cook County, establishing a groundwater monitoring network is strongly recommended. Monitoring wells can be established near prominent volume control and retention structures to monitor for contaminant loads to shallow groundwater as well as adjacent to nearby habitats that may be receiving groundwater flow sourced in part from these structures. Nested wells, i.e., wells set at different depths at the same site, are recommended to evaluate the potential for groundwater movement between stormwater features, the water table, sand and gravel aquifers, and the underlying bedrock aquifer. This will help elucidate where contaminants are present and the extent of infiltration into the groundwater system. It would be of particular relevance to use this monitoring system to evaluate the performance of underdrains in limiting infiltration to deeper groundwater.

Though many of the criteria for evaluating contamination potential require site-specific information, we can approximate regional contamination potential from existing geologic records. Generally, the greatest contamination potential for the shallow aquifer system is likely to occur where fine sediments are thin or absent at the surface or where sand is significant at the surface. These higher vulnerability areas are characterized by the presence of coarse sediments such as sand and gravel near the surface as well as areas with limited overlying sediments that may indicate vulnerability to contamination in the bedrock aquifer. Sediment thickness over the aquifer bedrock is useful in considering contamination potential as greater amounts of overlying sediment provide a larger buffer between the land surface and groundwater resources. The geology suggests that the location of greatest contamination risk to groundwater in the MWRD region is where transmissive sediments are within 10 feet of the surface. These transmissive deposits and lack of overlying sediments are a historic remnant of the landscape, as this is where the post-glacial Lake Chicago abruptly burst 19,000 years ago. The escaping waters from the lake incised into the landscape, removing glacial deposits and other material overlying the aquifer. As this is the most geologically sensitive area, prioritizing this area for monitoring well installation or sampling efforts would be insightful.

The relative importance of sampling different constituents is considered throughout this report in the MWRD area. Many of the high sampling priorities overlap with constituents of high interest to MWRD. The relative importance of land use, seasonality, and basin management on contamination of groundwater will vary for each volume control and detention feature. This means that a robust sampling campaign including many constituents is best to understand the impact of retention basins and volume control measures on groundwater.

As long-term declines in habitat diversity are well documented in Cook County wetlands, especially associated with increasing salinization and invasion by salt-tolerant species, chloride is perhaps the highest priority for monitoring to understand the potential for groundwater contamination from stormwater control and detention structures. Although data are scarce, elevated chloride concentrations in Cook County groundwater is precedented. Outfitting monitoring wells with continuous electrical conductivity probes allows for continuous hourly collection of chloride and total dissolved solids (TDS) data, as conductivity is a proxy for chloride and TDS once a regression is established. These probes are relatively inexpensive and would be instrumental in determining the existence of links between chloride and TDS in groundwater, stormwater infrastructure, and salinization in sensitive wetland ecosystems. With wetland habitats being increasingly fragmented in the Chicago region, and wetlands being on average surrounded by over 50% developed land in this region, these habitats are undoubtedly increasingly vulnerable to local impacts and stormwater routing influences. Similarly, given the long history of industrial and commercial land use historically in this region, many of these wetlands may be adjacent to existing contamination that may be remobilized during rain events. We recommend establishing monitoring adjacent to nearby wetlands that might be impacted by stormwater structures.

When studying groundwater quality in urban areas, evaluating many potential sources of contamination is important to determine the relative influence of stormwater in the system. The literature points to leaky infrastructure as a significant contributor to urban groundwater, with sewage leakage as a pervasive and troubling contamination source. Any sampling campaign should consider including pathogens, boron (sometimes an indicator for detergents in sewage), or pharmaceuticals to detect the presence of either stormwater networks capturing sewage leakage or sewage infiltration outside the basin influencing water quality beneath the basin.

For preliminary sampling, establishing approximate groundwater ages will be valuable to validate the methodology for determining contamination potential (for example, correlating groundwater ages with transmissivity in soils and upper sediments). In complex flow systems, such as the region's shallow aquifer, recharge takes complex paths to the subsurface, and waters of different ages could reside in different geologic units along a vertical profile. To this end, we recommend sampling for water isotopes that will help indicate the age of the groundwater. Water isotopes will show how close water from groundwater is to recent precipitation and help determine if older groundwater exists in isolated lenses.

Recent literature reviews emphasize the need for studies on the impact of stormwater on the scale of watershed catchments. Some authors discuss the need for future work to model solute transport from infrastructure to groundwater, the need for groundwater modeling to improve at large spatial scales, and the need for studies to focus on larger scales (watershed scale instead of pond site specific). After establishing a basic understanding of Cook County's groundwater quality, geology in the shallow aquifer, and impact of stormwater infiltration on the groundwater resources, the ISWS can develop a groundwater flow and contaminant transport model of the region. The ISWS has a functional model of the shallow aquifer system in Will County, directly south of Cook County, that can be adapted to include Cook County. We recommend stepwise modeling, i.e., building model complexity during the data collection process and refining understanding of stormwater processes during the monitoring campaign. For an informed model, water levels, water quality, and detailed information about stormwater detention structures will be necessary. A first step for the model would be to simulate water movement from volume control and retention measures to groundwater, calibrated to both water level measurements and chloride time series from the proposed monitoring wells. Modeling would also help show how conservation efforts to protect groundwater quality at local scales can help aquifers and wetland ecosystems at regional scales.

Stormwater infrastructure is essential for preventing flooding on Illinois roads, homes, and businesses. However, potential impacts to groundwater quality are critical to assess. The

ISWS can propose many ways to study groundwater in this area, but first, monitoring wells would need to be installed, as Cook County does not have a well network large enough for sufficient spatial coverage. Geology and land use can guide where it would be most beneficial to install these monitoring wells to assess the impact of stormwater on groundwater quality. Installing monitoring wells and communication with property owners to maintain stormwater infrastructure to full functionality would be a step toward protecting groundwater supplies in the region and maintaining ecosystem health for groundwater-dependent habitats in the region.

Watershed Pilot Analysis

Results from the literature review of *Stream Dynamics in Urban Settings* and *Relations between Watershed Management Strategies* and *Stream Erosion, Turbidity, and Sedimentation* summarized previously indicate that 1) channel erosion is a common problem in urban streams and is often related to changes in the magnitudes of relatively frequent flood events and 2) implementation of stormwater detention measures to control peak discharge results in longer duration of elevated discharges as the flow recedes from the peak discharge to the baseflow conditions. These findings from the literature review suggest that efforts to control release rates of stormwater in urban environments should consider the trade-off between reducing peak discharges of extreme events and increasing the duration of flows of moderate size through stormwater release practices.

The literature review of *Stream Dynamics in Urban Settings* highlighted approaches that could be used to evaluate the potential for stream erosion based on results from the hydrologic and hydraulic models developed originally for the Detailed Watershed Plans and more recently used in the initial evaluation of watershed-specific release rates. In particular, this review showed that stream power, the time rate of energy expenditure of flowing water in a river, provides a fundamental metric for predicting rates of bed-material transport in natural rivers. Thus, the potential for erosion within a reach of an urban stream can be assessed by determining 1) whether the actual stream power of a flow exceeds the critical stream power required to mobilize particles of different sizes on the channel bed and 2) whether the excess stream power of the flow, that being the stream power in excess of the critical stream power, is increasing over distance along the stream.

The objective of the pilot watershed analysis was to develop an approach to evaluating stream erosion potential for different watershed-specific release rates based on the concept of excess stream power and excess total work, where excess power and work refer to the capacity of the flow to transport bed material. Future condition models developed for the previous watershed-specific release rate study were updated with a broad range of design storms for the Upper Salt Creek and Addison Creek watersheds. Automated routines were developed to extract hydraulic properties of flow at every modeled hydraulic cross section. Field samples were obtained from these two watersheds and analyzed for streambed particle size distributions. By analyzing the hydraulic properties of flow over time in relationship to the material properties of these streams, it was possible to determine the cumulative excess stream power and thus consider the impacts of watershed-specific release rates on stream erosion in District streams.

The results of the pilot analysis identified several important themes for consideration. 1) At a local scale, changes in release rate may increase durations of stream power in excess of the critical threshold for bed-material transport, thereby increasing the total transport capacity of flows of a given recurrence frequency. 2) Because the spatial pattern of excess stream power remains unchanged for different release rates, the potential for channel erosion was higher for more restrictive release rates where excess stream power is increasing over distance. 3) Excess stream power does not always increase with more restrictive release rates, and more work would be needed to determine the factors that cause increases or reductions in excess stream power at specific locations and instances. Given that bed material properties are not well defined in the scientific literature, field sampling and characterization of the material properties of District streams would be an important first step necessary to evaluate these erosion and depositional processes on a larger scale or to determine how well model predictions of high erosion potential conform to evidence of actual channel erosion.

Recommendations

- The results of this analysis find that although District DIAs generally require marginally higher (~6% more) detention storage at the time of development or redevelopment, they enjoy moderately higher flood mitigation levels (~0.24 ft more) than non-DIAs. With this improved understanding of the impact of release rates on DIAs, policymakers and watershed managers are encouraged to continue evaluating whether existing or proposed stormwater management policies address prevalent inequities in flood risk.
- The selection of a release rate in collar counties was found to influence whether the watershed-specific release rates prescribed in the WMO Appendix B will continue to mitigate increases in peak water surface elevation due to future development in some watersheds. It is strongly recommended that the relevant watershed management agencies coordinate any changes in their watershed management requirements for multi-jurisdictional streams in the future.
- A consistent theme of the literature is that volume control and stormwater release rates are effective at mitigating many of the negative hydrologic and hydraulic impacts associated with urbanization, particularly those related to flooding, and are, in general, effective at improving water quality for many of the constituents of interest to the District. Volume control and extended stormwater detention, such as that afforded by more restrictive release rates, were generally more effective at removing certain constituents, although unintended consequences of these practices were documented related to both stream erosion and to groundwater quality. Constituent removal varies by species and with the design, location, and age of the facility, with the season, storm characteristics, and operation plan, and sequencing of practice. If the District seeks to build upon the extensive scientific literature related to these topics, it is recommended that any field sampling and associated modeling prioritize those processes with the highest uncertainty or potential for unintended consequences. These include conducting field sampling to characterize the material properties of district streams, evaluating the unintended consequences to groundwater from stormwater management

practices, and evaluating constituent re-mobilization from stormwater management systems including those associated with stratification.

- The results of the pilot analysis, the review of the scientific literature regarding stream channel dynamics in urban settings, and the review of the relations between watershed management strategies and stream erosion, turbidity, and sedimentation show that although practices such as volume control and stormwater release rates are effective at mitigating many of the negative hydrologic and hydraulic impacts associated with urbanization, particularly those related to flooding, and are in general effective at improving water quality for many of the constituents of interest to the District, the potential remains that these practices do not mitigate increases in stream bed mobilization, nor subsequent stream erosion, because the extended duration of discharge downstream of stormwater detention practices may increase the duration of flow above the critical stream power. If the District would like to expand upon this pilot analysis to a watershed-scale assessment of erosion potential on District streams, additional sampling of bed material would be required. Accurate characterization of the sediment of stream reaches is critical to understanding excess stream power and the data available from the scientific literature are not adequate to provide this characterization. With this data and field investigation at critical locations within the study area, a detailed review of the hydraulic modeling could be performed to better understand the effects of local structures or channel features on the hydraulics and confirm the model predictions of high erosion potential conform to evidence of channel erosion in the field. The pilot analysis could then be expanded from the Upper Salt Creek and Addison Creek watersheds to other watersheds of interest to the District for which modeling was performed during the development of the watershed-specific release rates.
- The scientific literature clearly identifies the potential for unintended consequences to groundwater quality due to infiltration from stormwater systems within the District. A thorough groundwater sampling campaign to assess the suite of contaminants relevant to stormwater is, therefore, recommended. Shallow groundwater use for community drinking water supplies has been limited in Cook County in the past several decades, and current groundwater data in the region are very limited. To quantify the impact of stormwater management practices on groundwater in Cook County, establishing a groundwater monitoring network is strongly recommended. Specifically, we first recommend the development of a groundwater model to confirm areas with a high potential risk of groundwater contamination. This model would be used to site the location of a groundwater observation network. These wells would initially be monitored for a broad range of constituents, as described in the report, necessary for the calibration of continuously monitoring probes suitable for the constituent of highest concern, chloride. These initial observation wells would provide the data necessary to refine the groundwater model with the potential to identify areas where these unintended consequences of stormwater management practices could be observed and quantified in the field. Monitoring wells can be established near prominent volume control and retention structures to monitor for contaminant loads to shallow groundwater, as well as adjacent to nearby habitats that may be receiving groundwater flow sourced in part from

these structures. Nested wells, i.e., wells set at different depths at the same site, are recommended to evaluate the potential for groundwater movement between stormwater features, the water table, sand and gravel aquifers, and the underlying bedrock aquifer. This will help elucidate where contaminants are present and the extent of infiltration into the groundwater system. It would be of particular relevance to use this monitoring system to evaluate the performance of underdrains in limiting infiltration to deeper groundwater. Both surface water and groundwater samples would be required to quantify the constituent budgets at these stormwater management practices. Chloride, phosphate, nitrate, PFAS, and water isotopes are constituents with high sampling priority, though consideration should be given to copper, zinc, iron, manganese, lead, pesticides, microplastics, pharmaceuticals, and VOC if practicable. Additional details about the recommendations for groundwater monitoring can be found in Chapter 6.4.

- Volume control and stormwater detention are, in general, effective at improving water quality for many of the constituents of interest to the District, though the literature shows high variability in the removal efficiencies of constituents from stormwater management practices at the site level. Although constituent removal varies by species and with the design and location, with the season, storm characteristics, and operation plan, and sequencing of practices, the effect of aging stormwater management practices, and in particular the ability of constituents tied to sediment in these practices to be re-mobilized, is one common source of uncertainty for which little scientific data is currently available. If the District wishes to better understand re-mobilization of contaminants from sediment, we recommend prioritizing a two-stage sampling program. The first stage would be to collect periodic water quality samples at a large, representative number of stormwater management practices in the District. These samples would be used to identify sites known to contain constituents susceptible to re-mobilization and thus reduced removal efficiency (synoptic survey method). The sites most likely to exhibit this re-mobilization would then be sampled in detail to understand the impacts of stratification (BMP performance assessment and small watershed studies method).

Acknowledgments

This study was funded by the Metropolitan Water Reclamation District of Greater Chicago (MWRD) through research grants executed with the University of Illinois Board of Trustees and performed by the Illinois State Water Survey (ISWS), part of the Prairie Research Institute, in collaboration with the Department of Geography & Geographic Information Sciences, the Department of Civil and Environmental Engineering, and the Department of Natural Resources and Environmental Sciences.

The authors wish to gratefully acknowledge the contributions of the researchers in Phase I and Phase II of the Watershed-Specific Release Rate Analysis, including Amanda Flegel, Sally McConkey, and Nicole Gaynor as well as several other ISWS staff. Vlad Iordache completed model setup and Zoe Zaloudek completed GIS spatial analysis for the project and created report exhibits; Glenn Heistand, Chris Hanstad, and Laura Keefer provided guidance and advice, and Lisa Sheppard edited the report. We also thank MWRDGC staff, Ann Gray, Maureen Durkin, Nate Wolf, and Adam Witek as well as the Technical Advisory Committee, who provided feedback to the project.

Table of Contents

Introduction

The Cook County Watershed Management Ordinance (WMO) was approved by the Metropolitan Water Reclamation District of Greater Chicago (MWRD, or the District) Board of Commissioners on October 3, 2013, became effective on May 1, 2014, and was most recently amended on April 7, 2022. Article 208 of the amended ordinance directs the District to consider the "impacts of watershed specific release rates on disproportionately impacted communities, the impacts of release rates under existing and future development scenarios in collar counties on watersheds in the District, and the impact of volume control and watershed specific release rates on stream erosion and related water quality effects such as turbidity and sedimentation." The Illinois State Water Survey (ISWS), in consultation with MWRD, convened an interdisciplinary team that included the ISWS Coordinated Hazard Assessment and Mapping Program and Groundwater Science Section as well as the University of Illinois' Department of Geography and Geographic Information Sciences, Department of Civil and Environmental Engineering, and Department of Natural Resources and Environmental Sciences to consider these impact assessments.

Urban development is typically accompanied by an increase in impervious area that can lead to increased runoff and more severe flooding. Stormwater retention and detention policies are commonly employed by regulatory agencies to mitigate these potential development impacts within their jurisdiction. The hydrologic and hydraulic impacts of such policies have been explored widely in the scientific literature as well as specifically as it pertains to the greater Chicago Region and the MWRD WMO in ISWS Contract Report 2019-06: Watershed-Specific Release Rate Analysis: Cook County, Illinois by Flegel and coauthors in 2019 (http://hdl.handle.net/2142/103416). The analysis and report that follows consider how these management policies are related to issues of social equitability, multi-jurisdictional watershed management, and stormwater quality.

Each chapter has been grouped by research theme. Article 208.2 of the WMO focuses on the impacts of watershed-specific release rates on disproportionately impacted communities and is discussed in Chapter 1. The appendices include additional exhibits that provide additional detail regarding the spatial variability of detention requirements and potential mitigation benefits at the community scale. Article 208.3 of the WMO focuses on the impacts of watershed-specific release rates in collar counties on watersheds in the District and is discussed in Chapter 2. Article 208.4 of the WMO focuses on the impact of volume control and watershed-specific release rates on stream erosion and related water quality effects. Given the breadth of this topic, this theme is addressed in Chapters 3 through 8. Chapter 3 includes a literature review of stream channel dynamics in an urban setting. Chapter 4 includes a discussion of the water quality processes associated with constituents of concern to the District. Chapter 5 includes a literature review of the relationship between watershed management strategies and water quality. Chapter 6 discusses the impact of volume control and detention practices on the groundwater of Cook County. Finally, Chapter 7 discusses the monitoring options for evaluating stormwater management policies' effects on water quality, and Chapter 8 includes a watershed pilot assessment of stream erosional potential.

Chapter 1. Impacts of Watershed-Specific Release Rates on Disproportionately Impacted Communities [WMO Article 208.2]

1.1 Introduction

1.1.1 Background

Urban development is typically accompanied by an increase in impervious areas. This in turn leads to increased runoff and higher flood levels unless properly managed (Shuster et al., 2005; Oudin et al., 2016). Urban planners and engineers employ various methods to mitigate this increased flood risk (Ashley et al., 2007; Fenner et al., 2019). One such method involves requiring new developments to provide stormwater retention or detention facilities to attenuate peak runoff rates. Mandated with the task of promulgating the Watershed Management Ordinance (WMO) within Cook County, Illinois and protecting the public from flood damages in its service area, the Metropolitan Water Reclamation District of Greater Chicago (MWRD, or the District) has adopted this approach to mitigate the potential risk associated with new development projects. Section 504.3 (A) of the WMO specifies that the gross allowable release rate for a development-related detention facility shall be based on a watershed-specific release rate for the 100-year, 24-hour storm event. These regulatory release rates for various watersheds (Table 1) were determined by a methodology that the Illinois State Water Survey (ISWS) proposed in previous phases of this project (Flegel et al., 2019) and are listed in Appendix B of the WMO.

Table 1. Watershed-Specific Allowable Release Rates for the Storm Event Having a 1% Probability of Being Equaled or Exceeded in a Given Year. Adapted from WMO Appendix B.

Watershed Planning Area	*Gross Allowable Release Rate Cubic feet per second per acre (cfs/ac)*
Poplar Creek	0.25 cfs/ac
Upper Salt Creek	0.20 cfs/ac
Lower Des Plaines	0.20 cfs/ac
North Branch	0.30 cfs/ac
Calumet Sag Channel	0.30 cfs/ac
Little Calumet River	0.25 cfs/ac

1.1.2 Motivation

Flooding, the costliest natural disaster facing the nation, adversely impacts some communities more than others. Low-income and marginalized communities suffer disproportionately more from this natural hazard (Wing et al., 2022; Hallegatte et al., 2016; Mitchell et al., 2014). Heterogeneity in flood risk can be attributed to variability in its constituent elements, namely flood hazard probability, exposure, and vulnerability (FLOODsite, 2009; Schanze, 2016). Flooding probability of an area depends on various natural (climate, topography, soil, etc.) and anthropogenic (impervious cover, artificial drainage system, etc.) factors. Even at a similar flood hazard probability, unequal exposure and vulnerability levels could result in vastly different flood risks among various communities in an area. In fact, low-income households are more exposed to floods than the average urban population as they tend to occupy more flood-prone areas (Hallegatte et al., 2016; Frank, 2020; Fielding, 2018). Although exposure refers to the quantity of receptors, such as the people, assets, and activities that may be impacted by a hazard, the vulnerability of a system is a function of the susceptibility, value, and coping capacity of these exposed receptors (FLOODsite, 2009; Schanze, 2006). Factors such as fewer financial and infrastructural resources used to prepare, protect, respond, and recover from flood impacts make low-income households highly vulnerable to the hazard (Fothergill and Peek, 2004; Fialka, 2019). All these disparities have a compounding effect on inequities in flood risk where marginalized communities are disproportionally impacted. Furthermore, this inequity gap is expected to grow even wider in the future as the adverse impacts of climate change are projected to fall disproportionately on the low-income population (Wing et al., 2022; IPCC, 2022; USGCRP, 2018).

Cook County has its own share of high flood risk areas along with significant inequities (Katz, 2021; Flavelle et al., 2020; Keenan et al., 2019; Festing et al., 2014). According to a Center for Neighborhood Technology study (Keenan et al., 2019), a total of $433 million in flood claim payouts were made to Chicago residents during 2007–2017. Eighty-seven percent of these insurance claim payments pertained to households located in communities of color and low-income communities. The study also reported a strong negative relationship between the number of claims and median household income of a community, indicating disproportionately higher flood risk levels in the disadvantaged communities. Water quality issues are also critical here as combined sewer overflows during floods can discharge untreated water into streams posing a public health and environmental hazard (Keenan et al., 2019; Katz, 2021). Further, climate change projections suggest more frequent, heavier precipitation and consequently more intense flooding in the urban watersheds of the county in the future (Wuebbles et al., 2021; Angel et al., 2020; Markus et al., 2017; Frankson et al., 2017; Markus et al., 2016). Without any remedial measures, this higher hazard probability would translate into even larger flood risk and inequity levels for low-income communities because of their higher exposure and vulnerability to such events.

Government and non-government organizations at various levels are cognizant of these inequities and have proposed, advocated, or implemented certain remedial measures to work toward environmental justice. In this context, the United States Environmental Protection Agency (USEPA) defines environmental justice as "the fair treatment and meaningful involvement of all people regardless of race, color, national origin, or income with respect to the development, implementation and enforcement of environmental laws, regulations and policies." Fair treatment here means that no group should bear a disproportionate share of the negative environmental consequences resulting from various regulations and policies (USEPA, 2022). Federal Executive Order 12898, issued in 1994, directs federal agencies to address the disproportionately adverse effects of their actions on low-income populations with the objective of implementing environmental justice. Executive Order 14008 signed in January 2021 further instructs federal agencies to develop programs, policies, and activities toward this goal and creates a government-wide Justice40 Initiative to "deliver 40 percent of the overall benefits of relevant federal investments to disadvantaged communities." Several local environment groups and community leaders have also highlighted inequities and pushed for environmental justice in

Cook County (Harris and Simba, 2022; Campillo and Simba, 2021). The Chicago City Council allocated $200 million in the FY2022 budget toward climate mitigation and environmental justice projects. In March 2022, U.S. senators from Illinois, Dick Durbin and Tammy Duckworth, and the MWRD announced $1.5 million in funding to support flood risk mitigation efforts in four underserved communities in the region (MWRD, 2022). MWRD's own five-year 2021–2025 Strategic Plan and 2022 Budget were devised with equity as one of the key guiding principles (MWRD, 2021). Motivated by such guiding principles and community voices, the MWRD Board of Commissioners included Section 208.2 in the WMO amendment adopted on May 16, 2019 to examine the watershed-specific release rates from an equity lens.

1.1.3 Objectives

Section 208.2 of the WMO directs the District to study the impact of watershed-specific release rates on disproportionately impacted communities. Per a MWRD memorandum dated July 10, 2021, disproportionately impacted areas (DIA) are defined as "areas with Chicago Metropolitan Agency for Planning (CMAP) Urban or Riverine Flood Susceptibility Index (FSI) mean value of 5-10, as of July 24, 2018, and Low to Moderate Income level as defined by the U.S. Department of Housing and Urban Development (HUD)." The FSI was developed by using a frequency ratio approach based on the empirical relationship between the spatial distribution of reported flood locations and several flood-related factors such as the Topographic Wetness Index and impervious cover. Slightly different flood-related factors were considered for riverine and urban flooding indices (CMAP, 2018). Low to Moderate Income level areas are census tracts where the majority of households are low- to moderate-income families that HUD defines as those with incomes of less than 80% of the median family income for the area (HUD, 2020; HUD, 2022). Note that the chosen FSI range corresponds to a relatively high hazard probability and exposure (CMAP, 2018), whereas the low-moderate income level represents high vulnerability. DIAs thus, by definition, are at high flood risk levels. The WMO's watershed-specific release rates regulation aims to mitigate any potential increase in peak water surface elevations during large flood events due to new developments by requiring detention storage and controlled release of runoff from these sites. This study seeks to examine the equitability of these regulations by comparing associated costs (detention storage requirement) and benefits (flood mitigation) for DIA communities relative to Non-DIAs. It also evaluates shifts in flood mitigation levels at more restrictive release rates in study watersheds. This study does not involve any financial analysis; equity of the regulation is examined only from the hydrological viewpoint. More specifically, the objectives of this study are to compare the impacts of watershed-specific release rates on DIAs and Non-DIAs in terms of (a) detention storage requirements, and (b) reduction in peak water surface elevations during a 1% annual chance flood event.

1.2 Methods

The Phase I and Phase II hydrologic and hydraulic models (Figure 1) were used to compute storage requirements and peak water surface elevations for study area subbasins and cross sections, respectively, for both the prescribed release rate as well as more restrictive release rates that were previously considered. These data were used to evaluate how detention storage requirements and reductions in peak water surface elevations related to watershed-specific release rates prescribed in WMO Appendix B vary by community and DIA. Although a brief description of these models is provided below, readers are encouraged to refer to *Illinois State Water Survey Contract Report 2019-06: Watershed-Specific Release Rate Analysis: Cook County, Illinois* by Flegel et al. (2019) (https://hdl.handle.net/2142/103416) for a detailed review of these models and related concepts.

Hydrologic (HEC-HMS) and hydraulic (HEC-RAS) models for this project were obtained from MWRD's Detailed Watershed Plans (DWP). These models were calibrated to observed events and originally represented land-use conditions circa 2004. These models were updated as part of the 2019 ISWS study of watershed-specific release rates to reflect major stormwater infrastructure projects impacting watershed hydrology or hydraulics (henceforth referred to as base conditions). Future scenarios of 40% land development were simulated by dividing every base condition HEC-HMS subbasin further into two components–one with 40% of the original subbasin area representing future development and one with the remaining 60% area retaining base conditions. Runoff from all developed subbasin components, which assumes an average curve number of 88 and the application of volume control requirements, namely the first inch of runoff retained on site using the HEC-HMS canopy method, was routed through a detention basin using a linear outflow hydrograph formulation (Guo, 1999) such that it meets WMO volume control and release rate requirements. Four release rate scenarios, 0.15 cubic feet per second per acre (cfs/ac), 0.20 cfs/ac, 0.25 cfs/ac, and 0.30 cfs/ac, were considered for analysis for which synthetic storage-discharge relationships were developed using the 100-year, 24-hour design rainfall. HEC-HMS flows corresponding to a critical duration, 100-year return period storm were then routed through HEC-HMS and HEC-RAS unsteady state hydraulic models to obtain peak water surface elevations at various cross sections.

These models produce detention storage and peak water surface elevations at a spatial resolution (subbasins and cross section, respectively) that is different from the focus of this study, i.e., communities. The following sections describe the geospatial and statistical methods used to process model outputs for the detention storage and peak water surface elevation analyses at the community level.

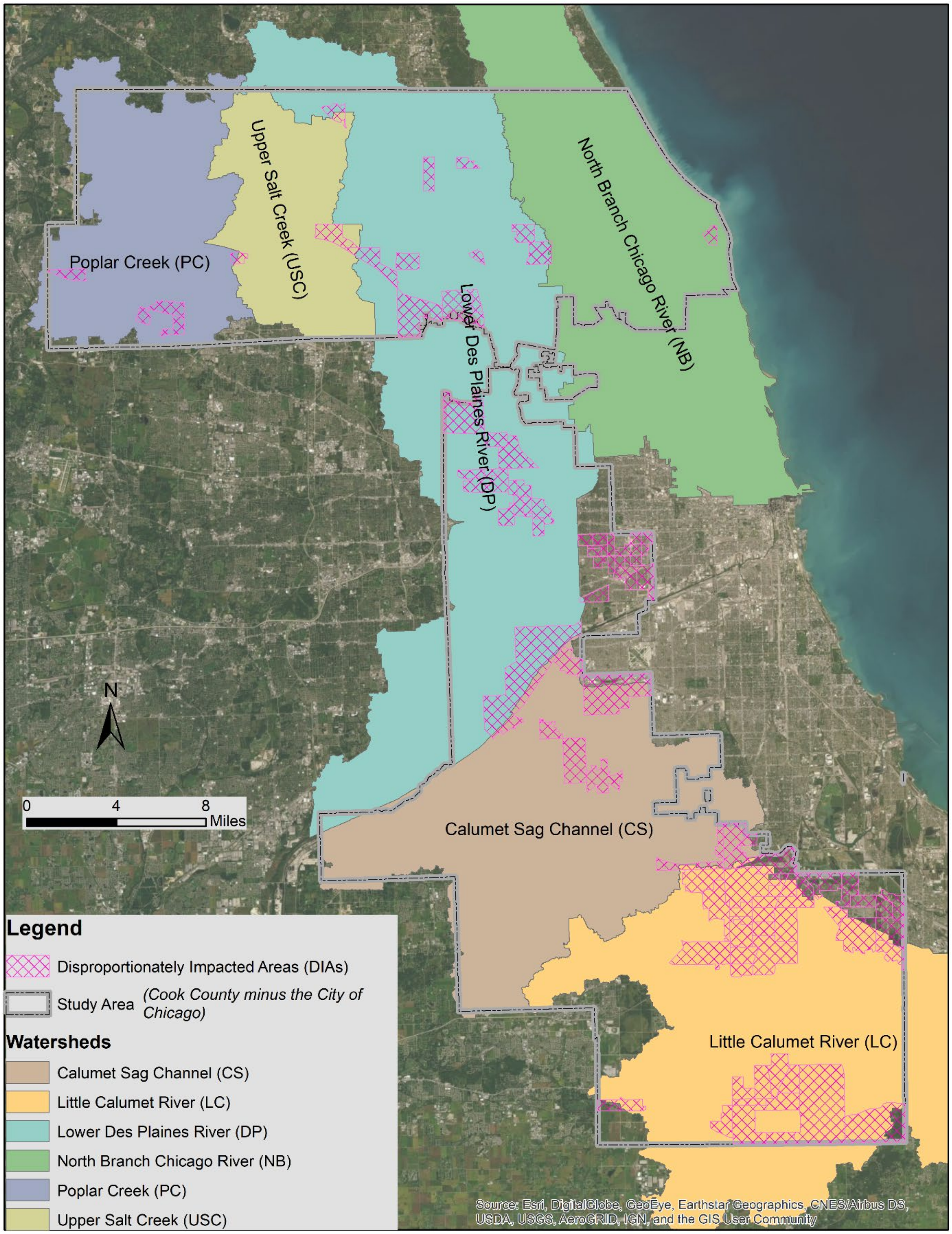


Figure 1. MWRD watersheds and DIAs and study area limits

1.2.1 Detention storage

HEC-HMS models were run for 100-year return period, 24-hour duration storms in select watersheds and subwatersheds during Phase II of this analysis. For this study, maximum storage values in reservoirs linked to developed subbasin components were tabulated from the simulation results. Since storage values (ac-ft) are highly dependent on the drainage area, they were normalized for the subbasin area and expressed in inches. During previous phases of the analysis, it was determined that select watersheds or subwatersheds, especially those with high base condition runoff rates, were likely to control whether a particular watershed-specific release rate was effective at mitigating future increases in peak water surface elevation from new development or redevelopment. These study areas were the focus of Phase II hydrologic and hydraulic modeling and are included in Figure 2. These select watersheds and subwatersheds provide a rich dataset that can be used to leverage additional information about subwatersheds not included in the Phase II analysis. This was particularly critical in Poplar Creek, Des Plaines, and Little Calumet River watersheds where a significant number of DIAs fell outside of areas studied in detail during Phase II. Base condition models, which were available for most of the study areas (except portions of the Cal Sag watershed), were used to estimate the detention storage requirements in these subbasins.

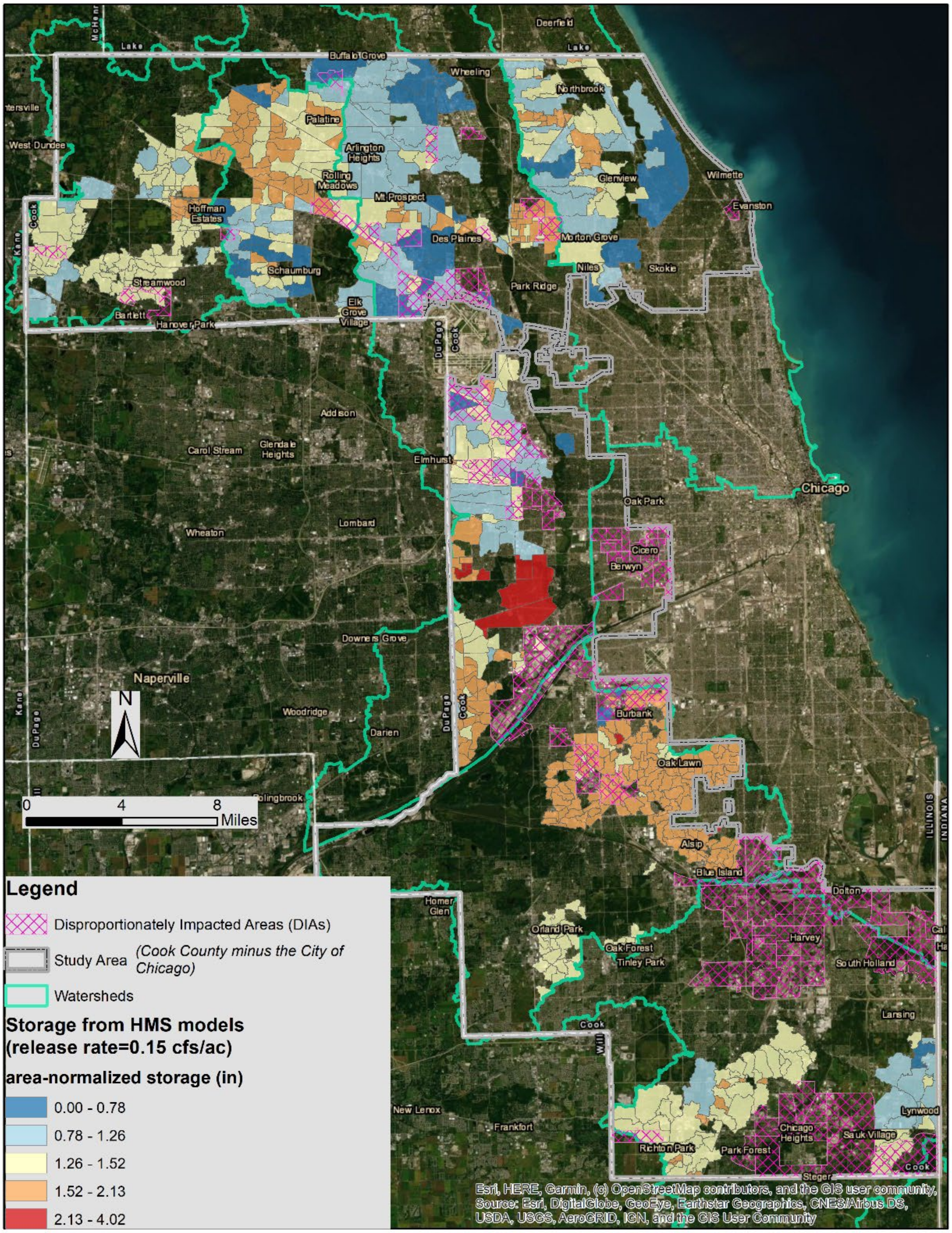


Figure 2. Subbasin detention storage information extracted from HEC-HMS models for release rate = 0.15 cfs/ac scenario

Considering the several commonalities in their meteorology and watershed characteristics (except for curve number and volume control storage), peak runoffs from base conditions and future condition models are expected to be strongly correlated. It is thus hypothesized that the storage required to meet various release rate requirements for future development and the peak subbasin runoff at base conditions are also strongly correlated. Plots of two quantities (Figure 3) –area-normalized storage and area-normalized peak runoff at base conditions (henceforth referred to as base conditions runoff rate)–for four release rate scenarios and regression results clearly validate this hypothesis. These regression relationships were then applied to calculate storage requirements for subbasins that were not explicitly studied in previous phases (Figure 4). Note that during Phase II, subbasins with more than 20% forest cover required special attention. These areas were excluded from regression and all subsequent analyses as lower future development rates (8–32%) in these green areas yield lower runoff and storage values that are not representative of the majority study area.

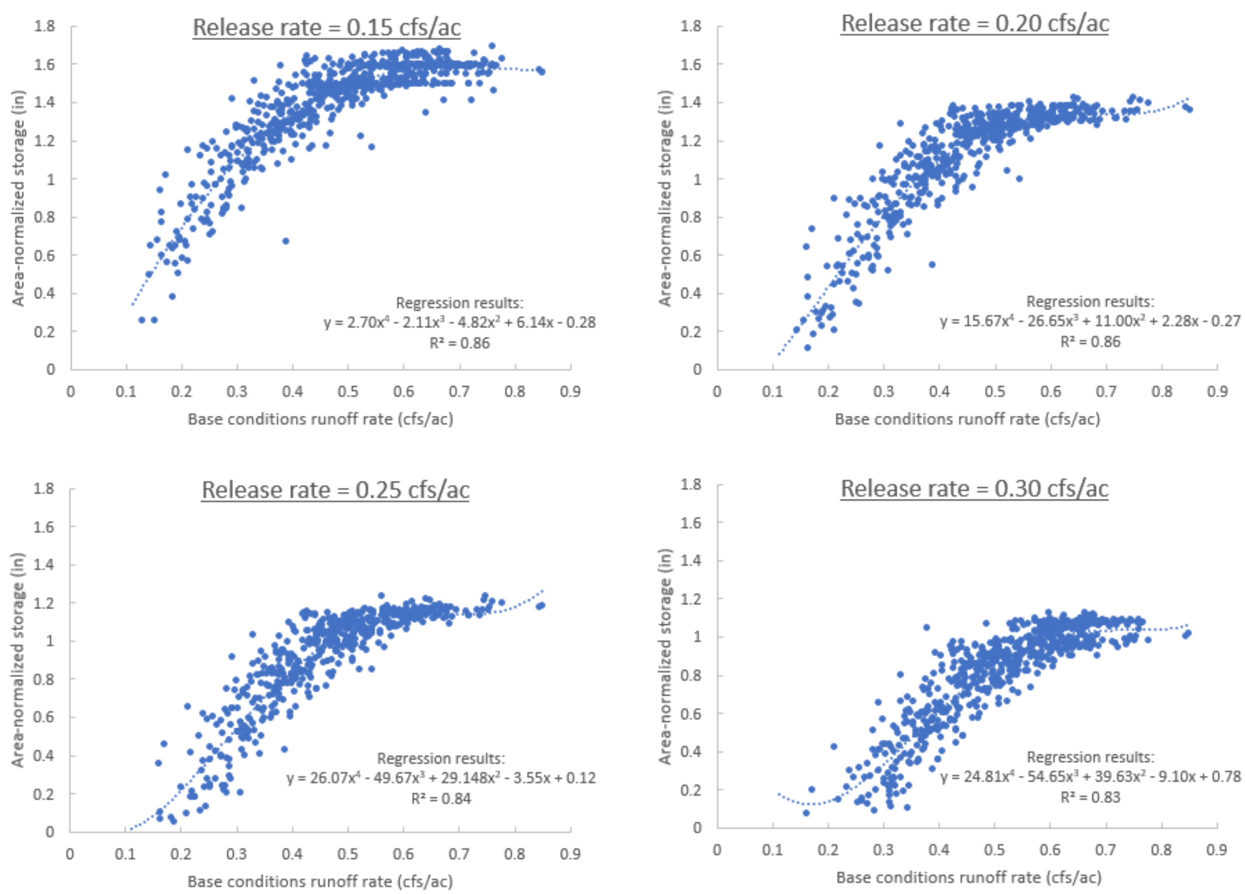


Figure 3. Regression between area-normalized subbasin detention storage at various release rates and base conditions runoff rate (i.e., area-normalized subbasin runoff at base conditions)

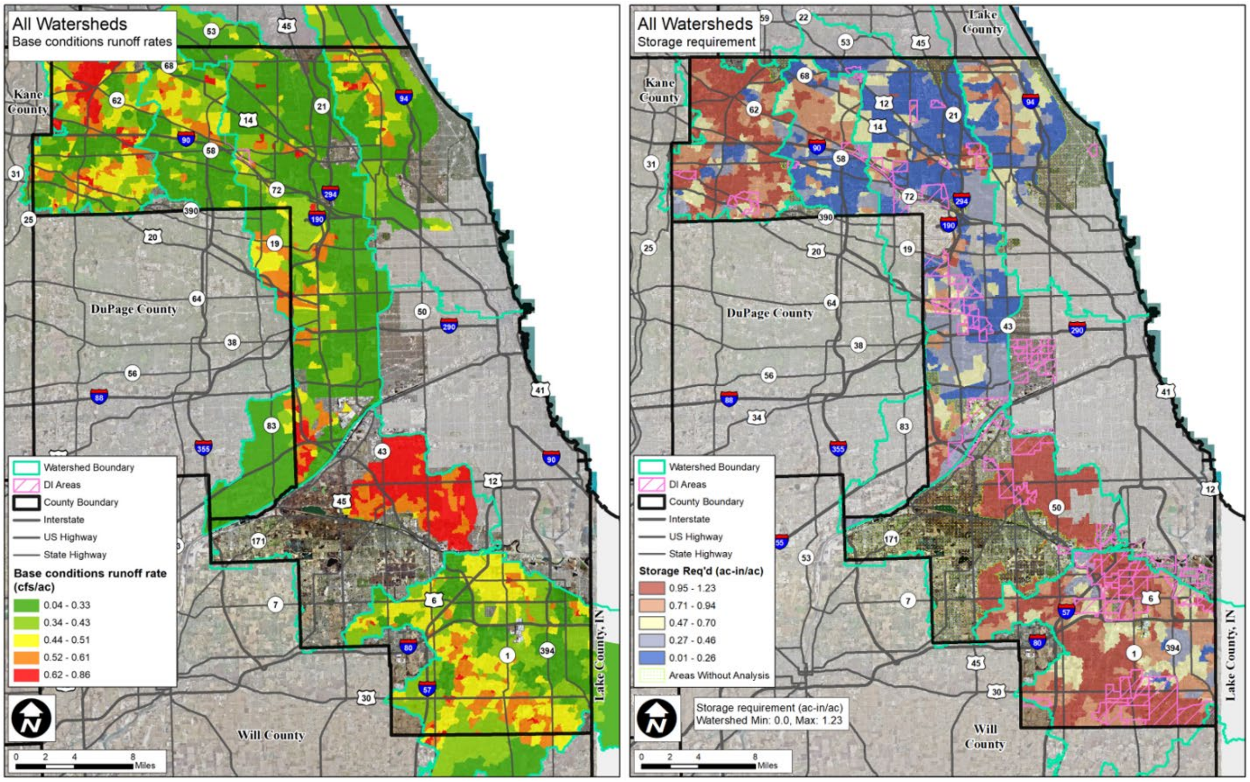


Figure 4. Subbasin storage database (right) updated using regression on base condition runoff rates (left). Storage values shown here represent detention storage volumes (required to meet the WMO's watershed-specific release rates) normalized by corresponding subbasin areas and thus are expressed in ac-in/ac units.

These subbasin storage data were then converted to a raster for comparative analysis. A gridded data format such as a raster was preferred for several reasons. It provides immense flexibility in calculating aggregated statistics for disparate geographical systems (e.g., subbasins, DIA census tracts, communities, watersheds) while properly accounting for spatial heterogeneity and area sensitivities of storage data. A raster is also suitable for statistical treatments such as group mean-comparison hypothesis testing and effect size estimation. Practical factors such as the spatial resolution of decision making and computational constraints governed the raster cell size selection of 200 feet. This scale is consistent with WMO article 201.1(D), which notes a watershed management permit is typically required for developments greater than 0.5 acres.

Aggregated average storage values were calculated for DIAs and Non-DIAs in the six watershed planning areas, and the two groups were compared statistically. During the group mean-comparison hypothesis testing, p-values–a measure of statistical significance–were found to be invariably small. This can be attributed to large sample sizes in tests; each cell represents a data point, and the total cell count is of the order of 10,000. Scientific literature indicates that p-values can be potentially misleading when very large (N > 1000) samples are involved (Demidenko, 2016; Travers et al., 2017). Also, although a p-value informs if there is an effect (presence of a significant difference between two groups), it does not reveal the effect size (Sullivan and Fienn, 2012). Thus, an effect size statistic, Cohen's *d,* was used to measure the standardized difference between two groups using the following formula:

$$d = \frac{M_1 - M_2}{s}$$

where M_1 and M_2 are means of the two groups, and s is the standard deviation of either group (Cohen, 1969; Coe, 2002). To aid the interpretation of Cohen's d value, the scientific literature defines three classes of effect size as shown in Figure 5 (Sullivan and Fienn, 2012; Coe, 2002). Cohen's d can be related to the overlap of the two groups' data distribution. The larger the effect size, the bigger the non-overlap region and the farther apart the two means would be.

Relative Size	Effect Size, *d*	Percentile	% of Non-overlap
	0.0	50	0
Small	0.2	58	15
Medium	0.5	69	33
Large	0.8	79	47
	1.0	84	55
	1.5	93	71
	2.0	97	81

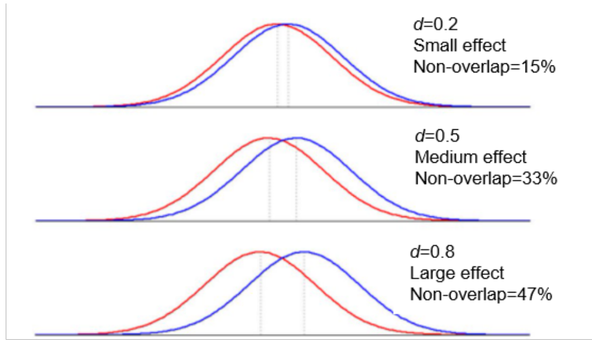


Figure 5. Interpretation of Cohen's d. Effect size can be characterized as small (d≤0.2), medium (d=0.3-0.7), or large (d≥0.8) depending on d value. It is proportional to the non-overlapping region between two groups' data distribution. Percentile field here represents the probability of correctly determining the group (DIA or Non-DIA) a subbasin belongs to from its storage value.

Next, storage requirements were examined for individual DIA communities. The term "DIA communities" here implies communities that have at least 1% of their area as DIA. Note that community boundaries are different from DIA boundaries that follow the census tract system. Average storage requirements were computed for all DIA communities and their DIA pockets. These were then compared to the average storage requirements in Non-DIAs of their respective watersheds. Some of the DIA communities span across multiple watersheds. The watershed containing the majority of DIA pockets (for which storage information was available) was assigned as the primary watershed for any such community.

As described in previous phase reports, hydrological analysis used in this study was based on design rainfall data obtained from the Illinois State Water Survey *Rainfall Frequency Atlas of the Midwest-Bulletin 71* (Huff and Angel, 1992). Since then, ISWS Bulletin 75 *Precipitation Frequency Study for Illinois* (Angel et al., 2020) has been published. Per this latest Bulletin, Cook County watersheds receive higher precipitation amounts than those used in this study. Anticipating the release of Bulletin 75, and to examine the effects of higher design precipitation on the selection of watershed-specific release rates, Phase II modeling included 500-year, 24-hour rainfall derived from Bulletin 71 within the Upper Salt Creek watershed. Expanding upon this dataset, differences in DIA and Non-DIA storage values corresponding to the 500-year return period and 100-year return period storms in the Upper Salt Creek watershed are compared. The Upper Salt Creek watershed was chosen for this analysis as it is the only watershed for which a rainfall event with a return period larger than the 100 year, 24-hour Bulletin 71 event was hydrologically modeled in previous phases. This larger return period, the 500-year storm, is treated as a proxy for Bulletin 75 rainfall, and the analysis is only intended to explore any expected general trends in variation from this study's results with a higher rainfall input.

1.2.2 Flood mitigation levels

HEC-HMS flows corresponding to base conditions and future 40% development scenarios meeting the WMO release rate requirements were routed through HEC-RAS unsteady state hydraulic models to obtain 100-year return period peak water surface elevations at various cross sections. The differences between base conditions' and future development scenarios' water surface elevations, often referred to as flood mitigation levels in this report, were tabulated for four release rates (rr = 0.15 cfs/ac, 0.20 cfs/ac, 0.25 cfs/ac, and 0.30 cfs/ac) at all cross sections as:

$$(dW_{rr} = WSE_{rr} - WSE_{base})$$

where
WSE_{base} = peak water surface elevation at base conditions,
WSE_{rr} = peak water surface elevation for 40% development scenario meeting the release rate requirement of rr cfs/ac, and
dW_{rr} = change in peak flood level offered by release rate rr at a cross section. Negative dW_{rr} values imply a reduction in peak flood levels.

Average flood mitigation levels, aggregated at the watershed scale, were then computed for DIAs and Non-DIAs in the six watershed planning areas. This analysis involved identification and assignment of appropriate watershed and DIA/Non-DIA status for every cross section. Flood mitigation levels were assigned weights equal to their respective reach lengths (defined as flow path length to next downstream cross section) in the calculation of averages to account for the non-uniform spacing between cross sections. Sparse cross section data in conjunction with limited DIAs meant that the DIA average in certain watersheds is based on few data points. Note that, along with water surface elevations, elements such as floodplain width and exposed property values also determine flood mitigation levels. Their evaluation, however, is beyond the scope of this study and thus were not considered.

Next, reductions in peak flood levels were examined for all communities where hydraulic model results were available. Reach-length weighted average flood mitigation levels were computed for various communities. In the case of DIA communities, besides an average value for an entire community, the average mitigation level was also calculated exclusively for its DIA component to enable intra-community assessment of flood mitigation levels. In a multi-watershed community, the watershed containing the majority of the hydraulic study reaches was assigned to the community for the analysis. This criterion is different from that used in the storage analysis, leading to five multi-watershed communities (Blue Island, Chicago Heights, Elk Grove, Oak Forest, and Niles) being assigned different watersheds in two analyses. Further, communities (e.g., Hickory Hills, Bridgeview) with a negligible amount of hydraulic cross sections (< 3) or total reach length < 0.5 miles were omitted from the analysis.

The flood mitigation level analysis methods described above consider WMO-specified release rates for six watersheds. The next treatment explores the potential benefits of adopting a more restrictive release rate for these watersheds. Although ISWS is not aware of plans to adopt more restrictive release rates for these watersheds, the information provided can help both the District and local communities understand the relationships between management strategies and community-level benefits at the subwatershed scale. For this, five classes of flood mitigation levels (Table 2) were first defined, and then the percentages of a watershed reach length belonging to each class were computed for the WMO-specified and more restrictive release rates. Shifts in the percentage composition across these risk mitigation classes across release rates were then analyzed for any trends and patterns for different watersheds. This analysis was repeated separately for DIAs as well.

Table 2. Five Classes of Potential Flood Risk Mitigation from Release Rates Implementation Compared to the Study Area Average

Potential risk mitigation	*Lower Limit*	*Upper Limit*
Much below average	-0.1 ft	∞
Moderately below avg.	-0.5 ft	-0.1 ft
Near average	-1.0 ft	-0.5 ft
Moderately above avg.	-1.5 ft	-1.0 ft
Much above average	-∞	-1.5 ft

1.3 Results

The analyses of detention storage requirements and the corresponding flood mitigation levels (dW) are presented and discussed in this section to evaluate the impact of watershed-specific release rates on the District's disproportionately impacted areas. DIA and Non-DIA results are compared at two different scales, the watershed and community. Tabular summaries of the two quantities aggregated at the watershed level are accompanied with watershed-wide maps to exhibit their spatial variation. For individual communities, relevant statistics are listed in a tabular format and further analyzed using box plots. Storage requirement maps of individual communities are included as Appendix A in this report.

1.3.1 Detention storage requirements

1.3.1.1 Storage requirements at the watershed scale

Detention storage needed to meet the WMO's watershed-specific release rate requirements were determined for study area subbasins (Figure 6 through Figure 12), and the aggregate average values computed for DIAs and Non-DIAs in six watersheds are presented in Table 3. It was observed that DIAs require higher average detention storage than Non-DIAs in all watersheds. The magnitude of this difference varies across watersheds. Cohen's *d,* a statistical measure of the effect size, suggests that the differences between DIA and Non-DIA means are small in the Cal Sag and Des Plaines River watersheds, and medium in Little Calumet, Poplar Creek, and Upper Salt Creek watersheds. Comparison is inapplicable in the case of the North Branch watershed as the watershed study area contains negligible slivers of DIA (Figure 10). Overall, DIAs in the District study area require marginally higher detention storage.

Table 3. Average Detention Storage (Area-Normalized) Needed in DIAs and Non-DIAs of Study Area Watersheds

Watershed	*DIA mean storage (in)*	*Non-DIA mean storage (in)*	*Effect size (Cohen's d)*
Cal Sag	1.06	1.05	Small (0.2)
Des Plaines	0.57	0.54	Small (0.1)
Little Calumet	0.91	0.82	Med (0.4)
North Branch (NB)	-	0.44	-
Poplar Creek	0.95	0.86	Med (0.4)
Upper Salt Creek	1.05	0.92	Med (0.4)
Overall (excluding NB)	0.82	0.77	Small (0.2)

Table 3 results are based on detention storage requirements normalized by total subbasin areas. Since the future development scenario assumes 40% homogeneous development in study areas, these results need to be divided by 0.40 to obtain storage needed per acre of new development. Further applying a unit conversion factor of 134.4 cubic yards per acre-inch, Table 3 transforms to Table 4, presenting the detention storage requirements (expressed in cubic yards) per acre of new development in study area watersheds. These results provide a more practical insight into storage requirements. Overall, DIAs require 6% more detention storage, which translates to extra storage of 0.13 ac-in (17.5 cubic yards) for every acre of new development. The percentage increase value varies from 1% (Cal Sag watershed) to 14% (Upper Salt Creek watershed). In absolute figures, the detention storage requirement is in general clearly lower for the Des Plaines River (Figure 8 and Figure 9) and the North Branch Chicago River (Figure 10) watersheds. This requirement is highest in the case of the Cal Sag watershed. Note that only two subwatersheds of the Cal Sag watershed were analyzed, Tinley Creek and Stony Creek (Figure 6), as the hydrologic and hydraulic modeling was not carried out in other Cal Sag subwatersheds during previous phases of the project.

Table 4. Approximate Detention Storage Needed Per Acre Area of New Development in DIAs and Non-DIAs of Study Area Watersheds

Watershed	*DIA storage (cu. yards)*	*Non-DIA storage (cu. yards)*	*Δ%* $\left(\frac{DIA - NonDIA}{NonDIA}\right)$
Cal Sag	356	354	1%
Des Plaines	192	183	6%
Little Calumet	307	276	11%
North Branch	-	148	-
Poplar Creek	320	289	10%
Upper Salt Creek	354	309	14%
Overall (excluding NB)	276	258	6%

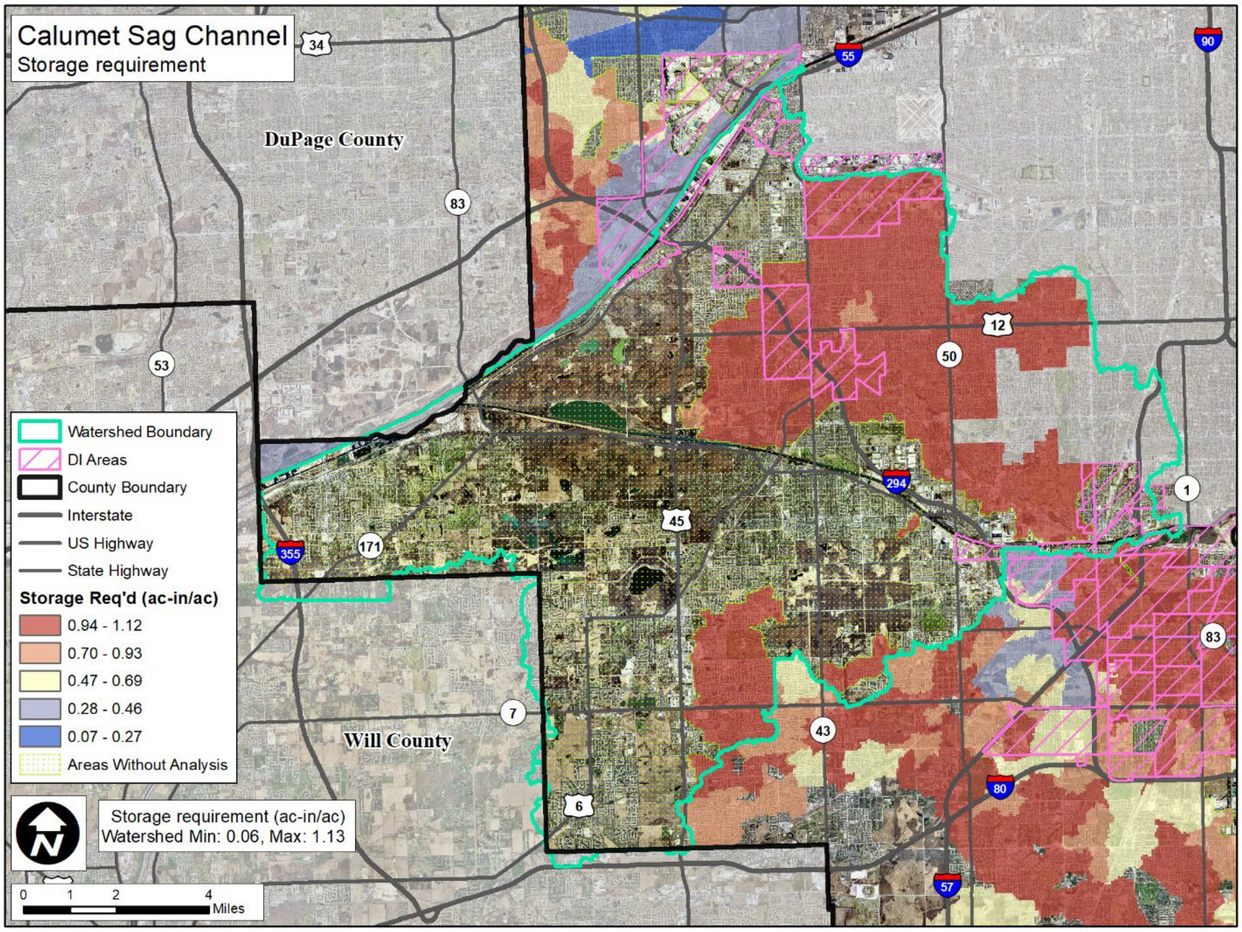


Figure 6. Detention storage requirements (area-normalized) in the Cal Sag watershed

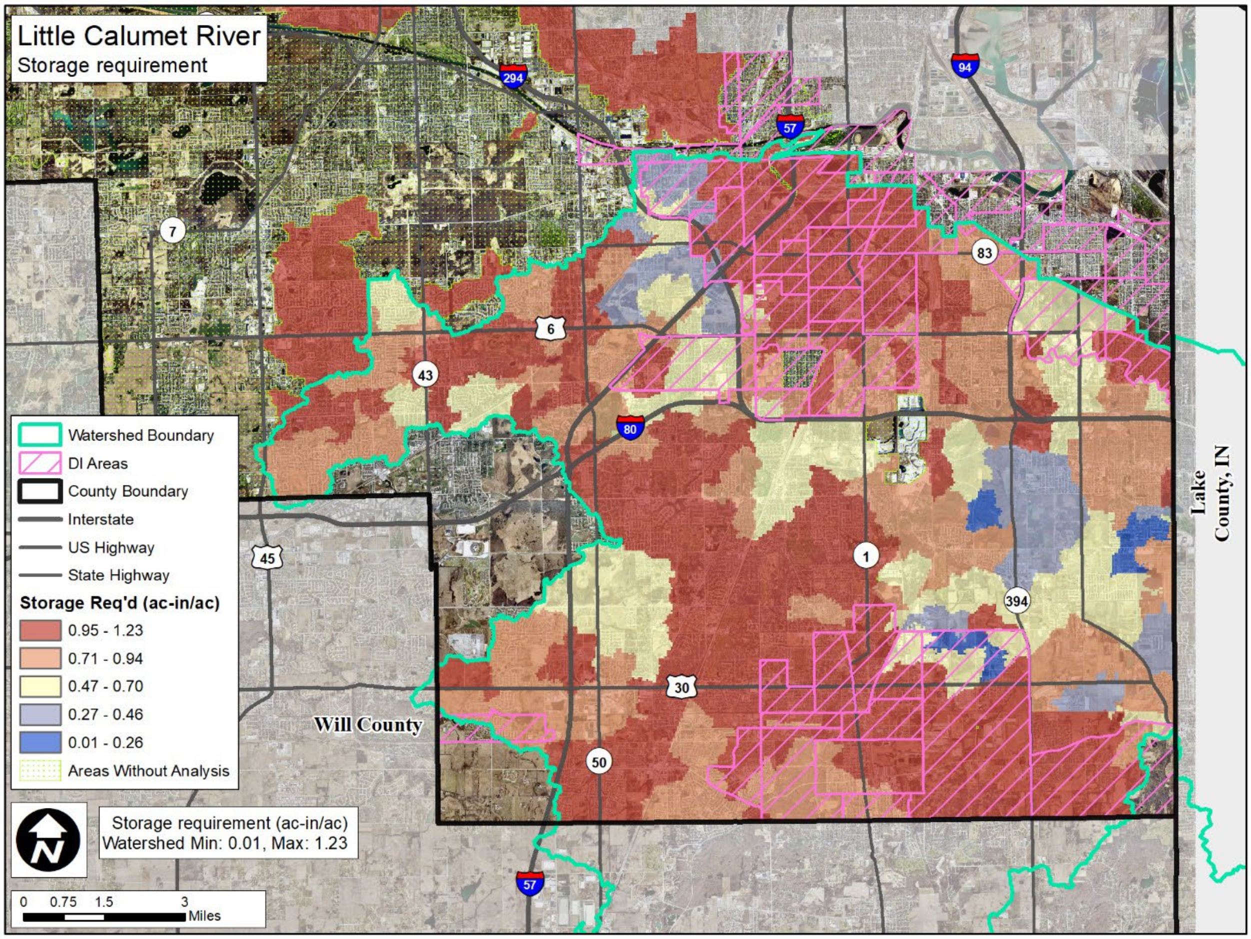


Figure 7. Detention storage requirements (area-normalized) in Little Calumet watershed

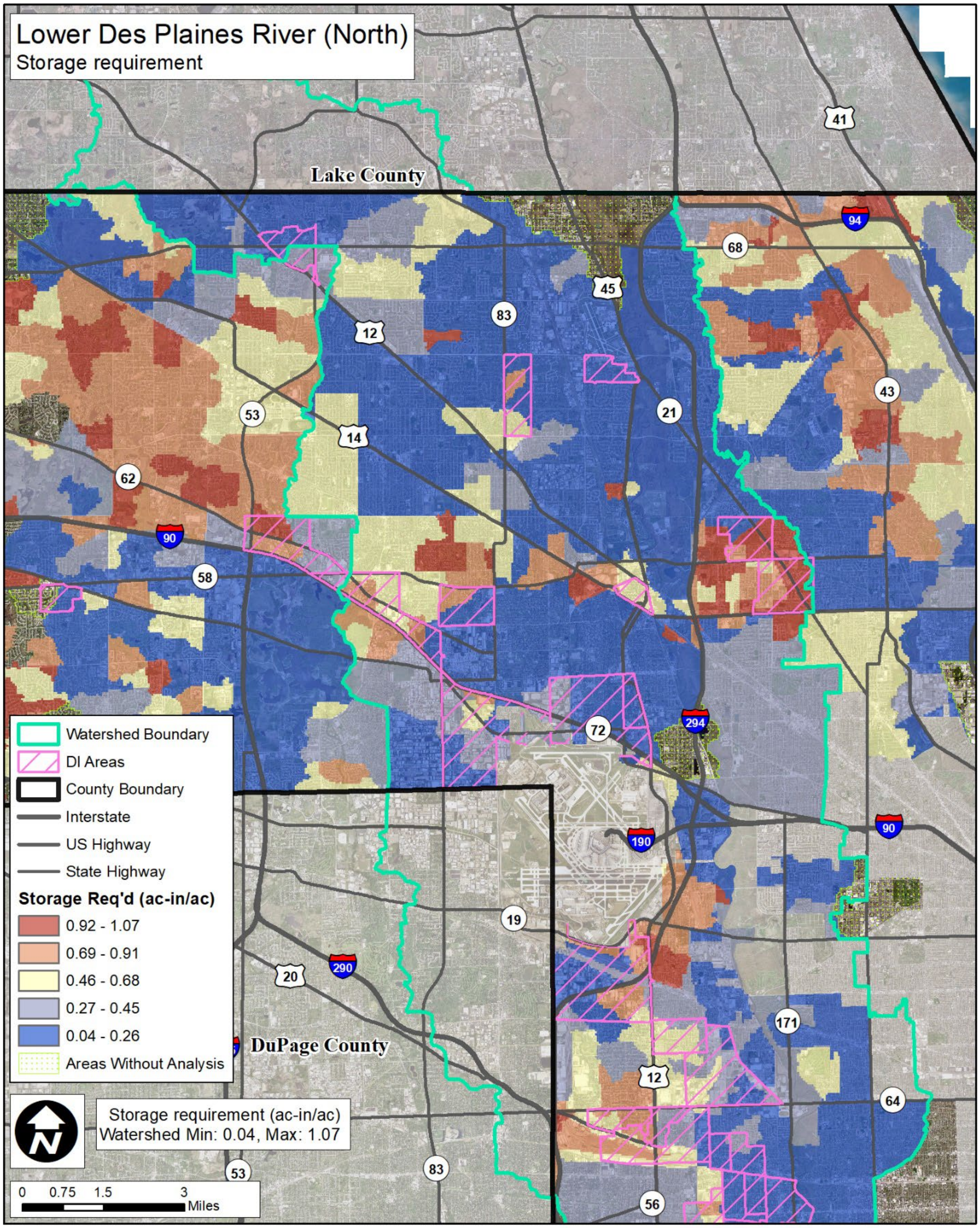


Figure 8. Area-normalized detention storage requirements in Des Plaines watershed (North)

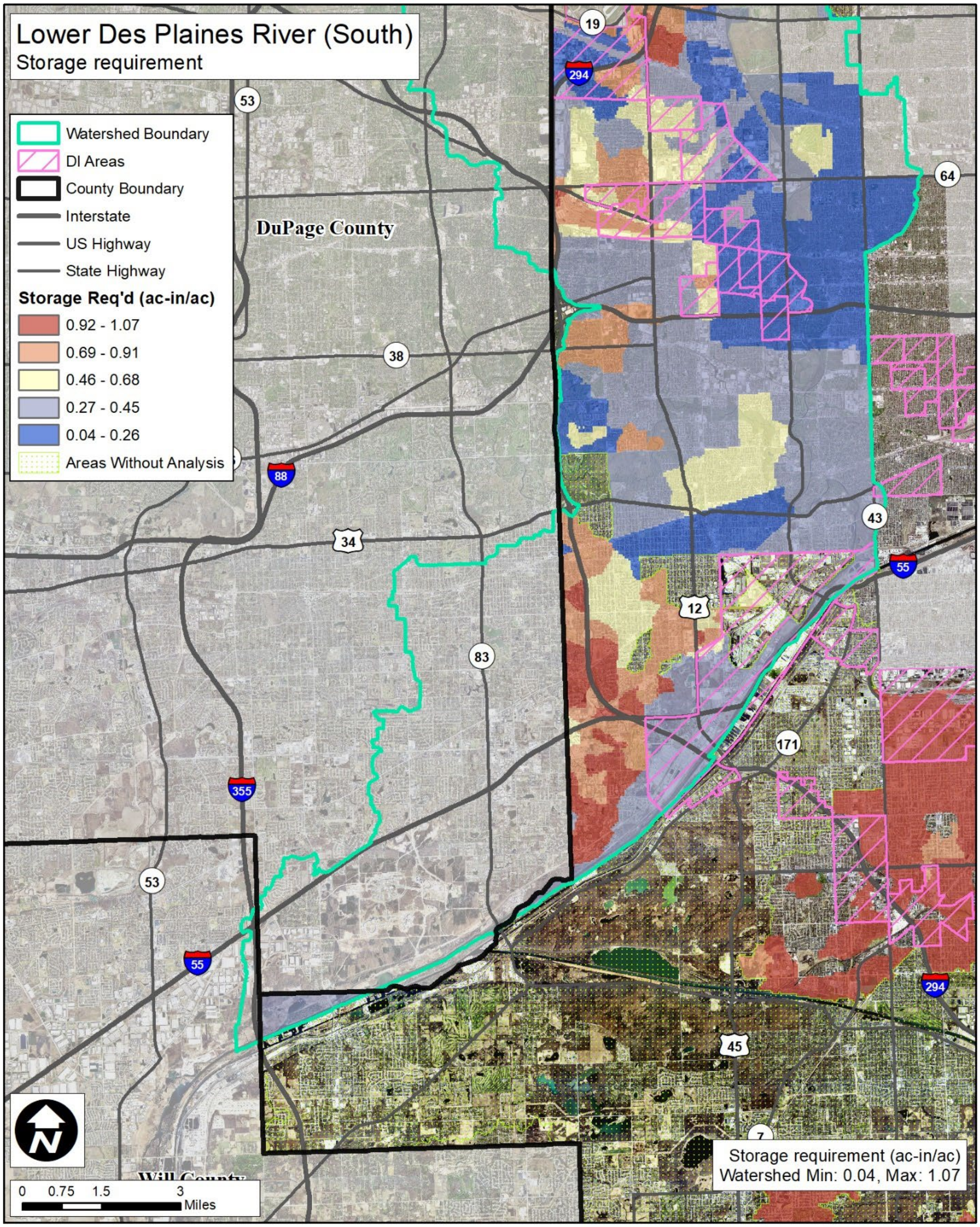


Figure 9. Area-normalized detention storage requirements in Des Plaines watershed (South)

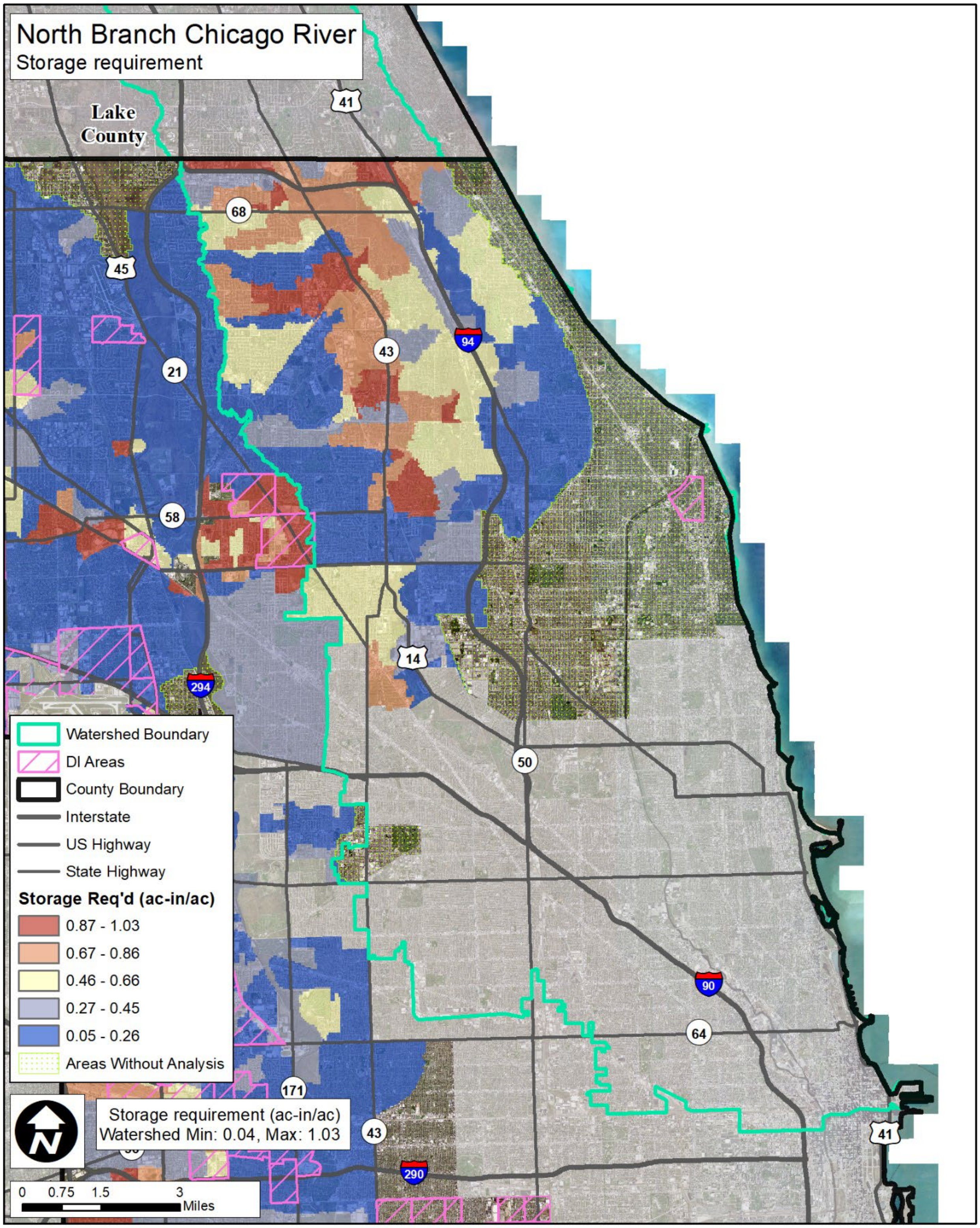


Figure 10. Detention storage requirements (area-normalized) in North Branch watershed

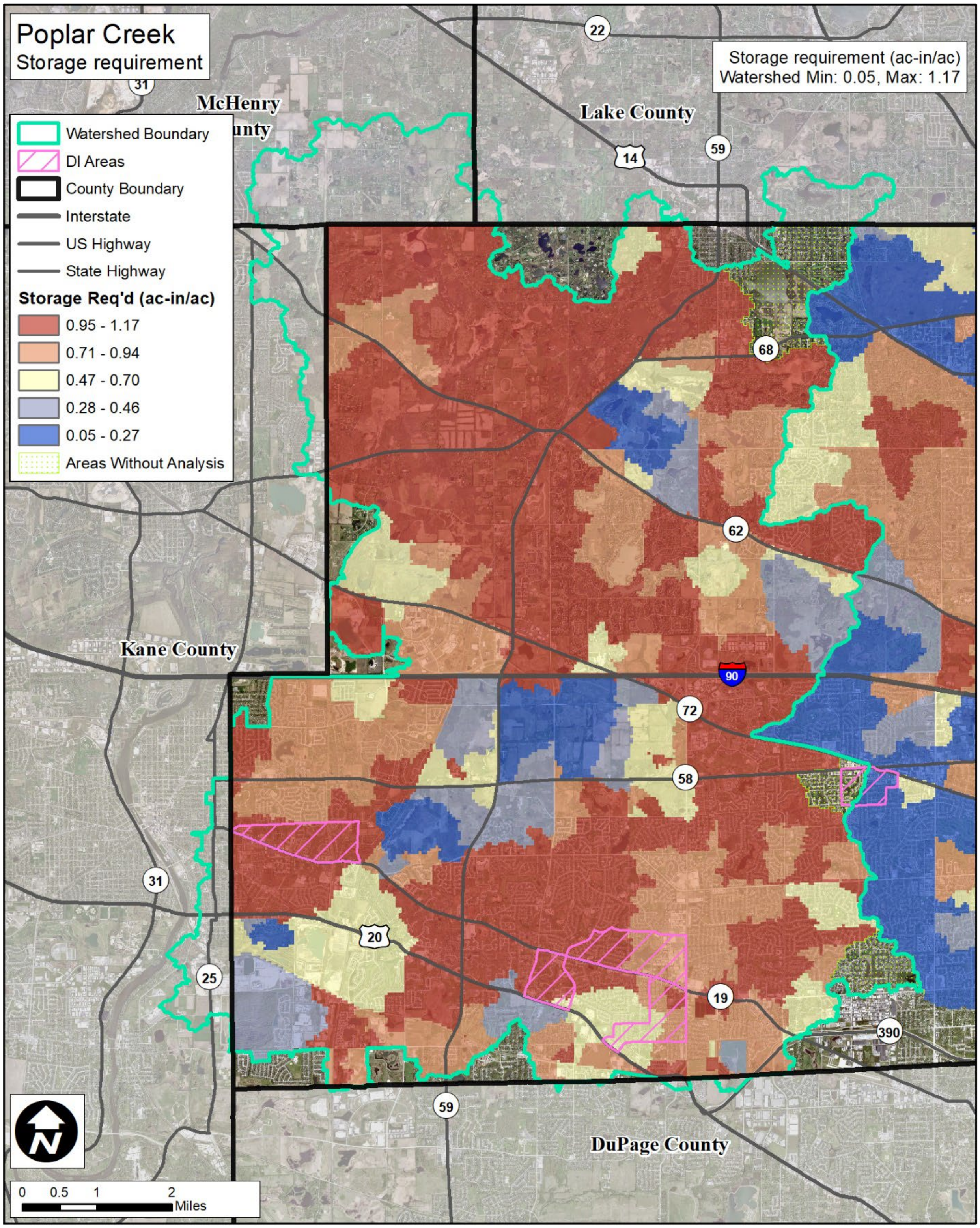


Figure 11. Detention storage requirements (area-normalized) in Poplar Creek watershed

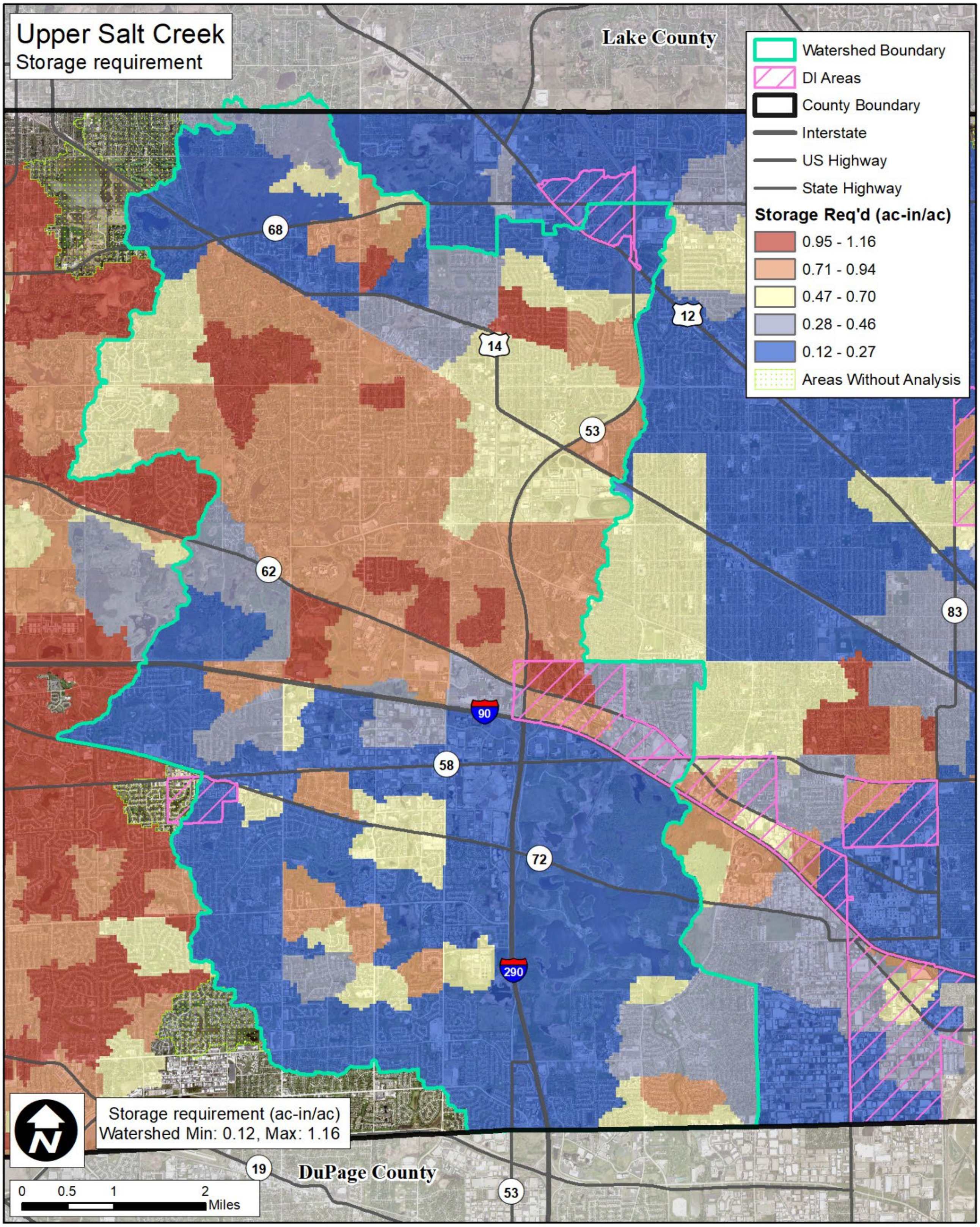


Figure 12. Detention storage requirements (area-normalized) in Upper Salt Creek watershed

Table 5. Area-Normalized Storage Requirements in Various DIA Communities and Their DIAs

Community	%DIA	Watershed Non-DIA Avg (WNA)		Whole Community		DIAs only	
		Watershed	WNA Storage (in)	Storage (in)	% of WNA	Storage (in)	% of WNA
Bedford Park	71	Cal Sag	1.05	1.00	95%	1.01	96%
Chicago Ridge	62			1.05	100%	1.05	100%
Bridgeview	33			1.07	101%	1.07	102%
Hickory Hills	31			1.07	102%	1.08	103%
Justice	19			1.07	102%	1.07	102%
Burbank	14			1.04	100%	1.04	99%
Palos Hills	3			1.06	101%	1.07	102%
Hodgkins	100	Des Plaines	0.54	0.05	10%	0.05	10%
McCook	100			0.78	144%	0.76	141%
Stone Park	100			0.76	141%	0.74	137%
Summit	74			0.02	3%	0.02	3%
Melrose Park	66			0.73	136%	0.72	134%
Maywood	64			0.03	6%	0.00	0%
Franklin Park	59			0.64	119%	0.81	150%
Willow Springs	26			0.39	72%	0.02	4%
Prospect Heights	26			0.51	95%	0.55	102%
Bellwood	25			0.91	168%	0.95	177%
Des Plaines	17			0.37	68%	0.34	63%
Northlake	16			1.03	190%	1.14	211%
Niles	16			0.64	119%	1.29	240%
Countryside	16			0.68	127%	0.01	3%
Rosemont	14			0.21	39%	0.22	41%
Elk Grove Village	14			0.76	140%	0.69	128%
Lyons	14			0.08	15%	0.01	3%
Mount Prospect	9			0.76	140%	0.68	126%
Arlington Heights	5			0.81	150%	1.10	203%
Bensenville	2			0.97	180%	0.50	93%
Dixmoor	100	Little Calumet	0.82	0.98	119%	0.98	119%
Phoenix	100			1.06	129%	1.06	129%
Posen	100			1.00	122%	1.00	122%
Riverdale	100			0.99	120%	0.99	121%
Harvey	99			0.96	117%	0.96	117%
Robbins	91			0.41	50%	0.41	50%
Calumet City	88			0.76	93%	0.78	95%
Blue Island	76			0.97	119%	0.92	112%
Chicago Heights	72			0.96	117%	0.97	118%
Dolton	71			0.91	111%	0.91	111%
Steger	68			0.95	116%	1.02	124%
Sauk Village	63			0.96	117%	1.04	127%
Ford Heights	60			0.72	88%	0.71	86%
Markham	60			0.64	79%	0.72	88%
Park Forest	27			0.96	118%	1.02	125%
South Holland	23			0.88	108%	0.90	110%
Hazel Crest	15			0.91	111%	0.75	91%
Crestwood	8			0.68	83%	0.37	45%
S. Chicago Heights	7			1.00	122%	1.10	134%
Matteson	6			0.91	111%	0.82	100%
Oak Forest	4			0.96	117%	0.93	113%
Streamwood	13	Poplar Creek	0.86	0.96	112%	0.96	111%
Hanover Park	10			0.84	97%	0.77	90%
Elgin	1			0.86	100%	1.09	127%
Rolling Meadows	22	Upper Salt	0.92	1.15	125%	1.15	125%
Palatine	3			0.98	107%	0.43	47%

1.3.1.2 Storage requirements at the community level

The next analysis examines the storage requirements for various DIA communities in the study area. Communities with more than 1% disproportionately impacted areas were considered for this analysis. Table 5 lists these communities and their respective average storage requirements. This information is also presented in map exhibits (Appendix A) prepared for each of these DIA communities. The community average includes both DIAs and Non-DIAs. Average storage values were computed exclusively for DIA pockets in these communities and are also reported in the table. Both sets of these averages–whole community and DIAs only–are also expressed as a percentage of average storage requirement in the Non-DIAs (termed Watershed Non-DIA Average or WNA) of their respective watersheds. This allows for comparison of storage requirement in DIA communities with Non-DIAs in a watershed. Communities with storage requirements much greater than the WNA (%WNA≫100%) are color coded in shades of red, those much lower than WNA (%WNA≪100%) are coded in shades of dark green, and the ones in the vicinity of WNA (%WNA≈100%) are in shades of yellow/lime.

Box and whisker plots were used to visualize the distribution of %WNA values for DIAs only portions (Figure 13). DIAs in almost all Cal Sag watershed communities have storage requirements very close to that of the watershed's Non-DIA average. Diverse deviations were observed in other watersheds' DIA communities. The Des Plaines River watershed has the widest range in the deviation from WNA–very low (0%, Maywood) to very high (240%, Niles) with 114% as a median value. The middle 50% of Des Plaines DIA communities have DIA storage requirements between 6% and 145% of the WNA. The distribution is left-skewed, indicating the presence of a larger number of DIA communities in the lower storage value region. Many of the lower value outliers–Hodgkins, Summit, Maywood, Willow Springs, and Lyons (all in Des Plaines watershed)–can be attributed to the application of the regression approach to meager base flow rates recorded in the hydrologic models of these areas. The Little Calumet watershed, on the other hand, exhibits a much smaller deviation from the WNA, ranging between 45% (Crestwood) and 134% (South Chicago Heights) with 113% as the median value. The middle 50% of Little Calumet DIA communities have moderate DIA storage requirements between 90% and 125% of WNA. Poplar Creek and Upper Salt Creek watershed results are not shown in Figure 13 as there are insufficient data points to justify a box and whisker plot for these areas. Barring Palatine, DIAs in these two watersheds' communities have storage requirements deviating only moderately (90% to 127%) from their respective WNA. Overall, across all watersheds, most of the study area DIA communities have DIA storage requirements between 90% and 125% of their respective WNA with 110% as the median value.

The above analysis compares DIA storage requirements with respect to the watershed Non-DIA average and does not examine the intra-community variations. Figure 14 facilitates the examination of intra-community heterogeneity in DIA communities' results. Here the DIA storage requirement in each DIA community of a watershed is expressed as the percentage of the corresponding community average storage, and the resulting distribution is plotted. As evident in Figure 14, the Cal Sag watershed DIA communities are remarkably homogeneous. DIA and Non-DIA pockets in DIA communities have nearly identical storage requirements. In the Des Plaines River watershed, however, the range in variations is quite large, going from very low (2 percent, Countryside) to very high (201%, Niles) with 98% as the median value. Little Calumet communities exhibit a much smaller variation. Except for Crestwood and Hazel Crest, DIAs in most communities require storage moderately close to their community averages. At a global level, DIAs in the District's DIA communities exhibit a trend similar to that observed in the Little Calumet watershed.

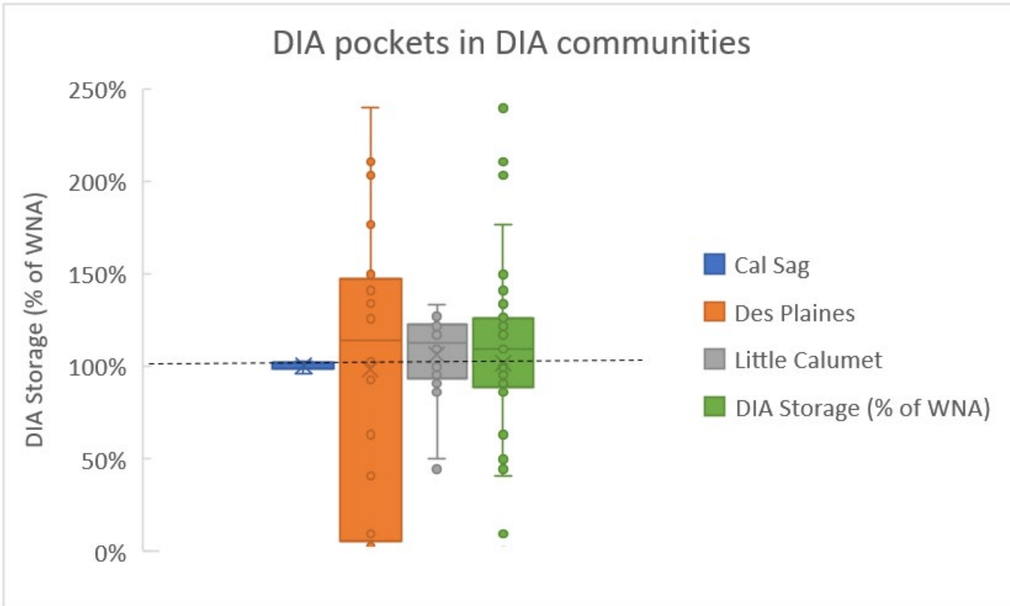


Figure 13. Distribution of DIA storage requirements in relation to respective watershed Non-DIA averages. Each data point represents aggregated result for the DIA portion of a DIA community.

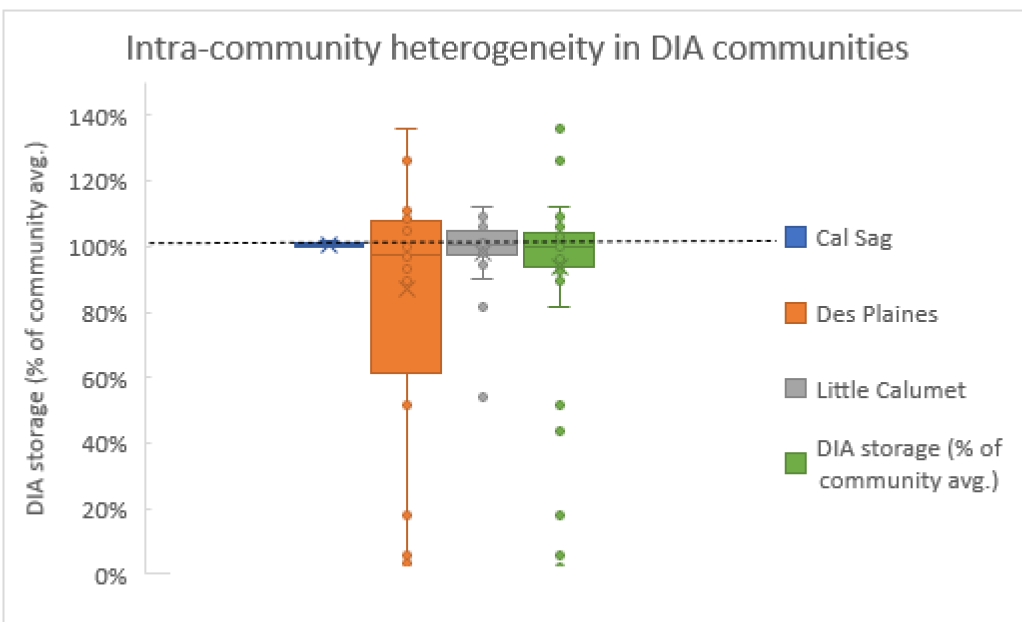


Figure 14. Distribution of DIA storage requirements in relation to their respective community average storage

1.3.1.3 Impact of higher rainfall events on storage requirements

As the previous phases of this study used the design rainfall effective at the time of the study and this analysis builds upon those results, it was explored how the storage analyses presented in this report would likely change with a regional increase in precipitation frequency and intensity as reported in Bulletin 75. The Bulletin 71 derived 500-year return period storm was previously analyzed for Upper Salt Creek and could serve as a proxy higher rainfall event for this analysis. Detention storage requirements in the Upper Salt Creek watershed areas corresponding to a 100-year return period (Table 6) storm were compared with the 500-year return period storm results (Table 7) used in this study. In both cases, DIAs require higher detention storage than Non-DIAs. However, the percent difference(s) between DIAs and Non-DIAs is lower for the 500-return period storm in all four release rate scenarios. Assuming similar relationships (a) between 100-year base condition flow rates and 500-year base flow rates (Figure 15) and (b) between storage and base condition flow rates (Figure 16), it is estimated that the aforementioned trend would likely hold true in the majority of the District watersheds. In other words, the percent difference(s) between DIAs and Non-DIAs is expected to reduce at higher return period storms in the majority of the study area. These assumptions may not be appropriate for the Cal Sag watershed where most of the studied subbasins lie in the far-right region of the storage-base flow conditions rate curve (Figure 16). These Cal Sag data points correspond to highly urbanized subbasins where storage requirements are less sensitive to change in base flow condition rates for a given storm. It must also be pointed out that only two Cal Sag subwatersheds were considered in the study; inclusion of the remaining subwatersheds may also affect any estimation of the trend for a higher rainfall event in the watershed.

Table 6. Area-Normalized Detention Storage Requirements in the Upper Salt Creek Watershed Areas for the 100-year Return Period Storm

Release Rate Scenario	*DIA storage (in)*	*Non-DIA storage (in)*	*Δ%* $\left(\frac{DIA - NonDIA}{NonDIA}\right)$
0.15 cfs/ac	1.33	1.20	11%
0.20 cfs/ac	1.05	0.92	14%
0.25 cfs/ac	0.79	0.68	16%
0.30 cfs/ac	0.56	0.48	17%

Table 7. Area-Normalized Detention Storage Requirements in the Upper Salt Creek Watershed Areas for the Increased Rainfall Proxy (500-Year Return Period Storm)

Release Rate Scenario	*DIA storage (in)*	*Non-DIA storage (in)*	*Δ%* $\left(\frac{DIA - NonDIA}{NonDIA}\right)$
0.15 cfs/ac	2.72	2.63	3%
0.20 cfs/ac	2.45	2.35	4%
0.25 cfs/ac	2.19	2.07	6%
0.30 cfs/ac	1.91	1.79	7%

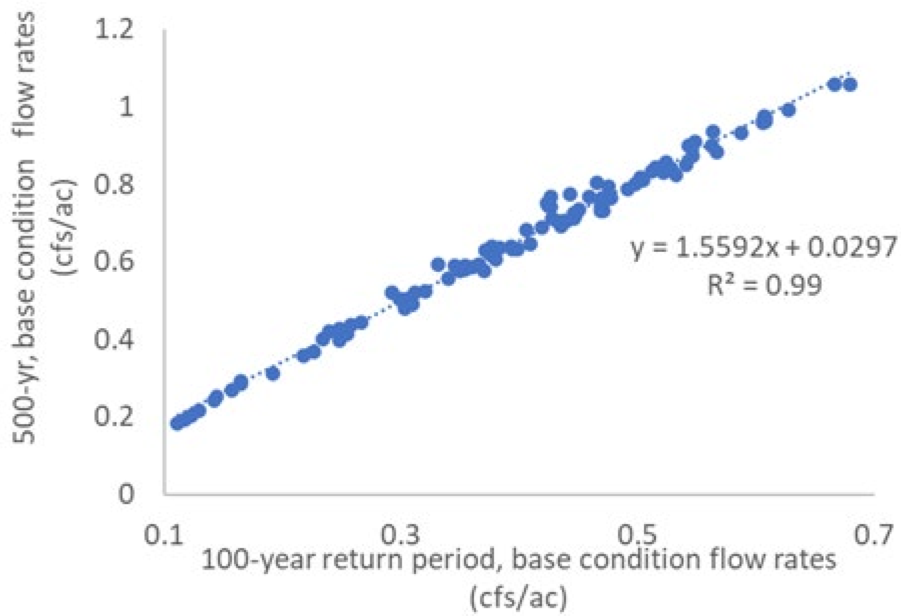


Figure 15. Relationship between base condition flow rates for 500-year and 100-year return period storms in the Upper Salt Creek watershed

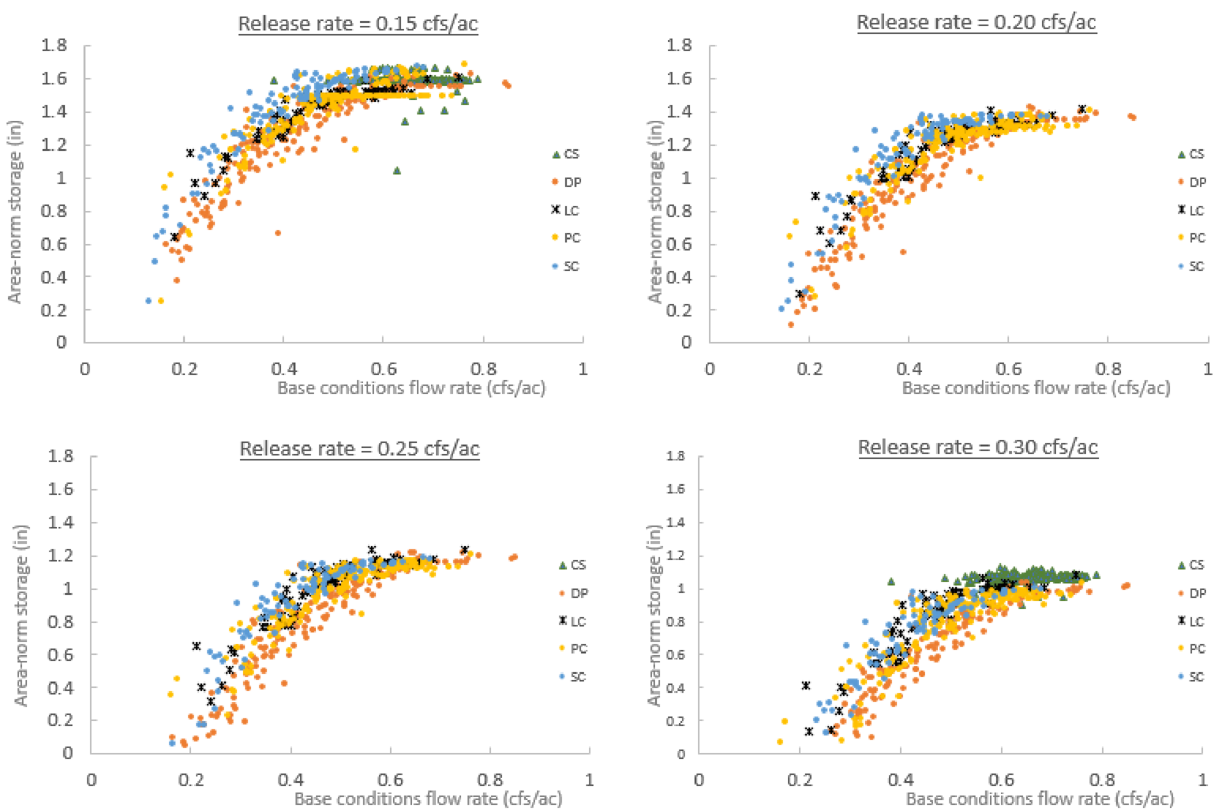


Figure 16. Relationships between area-normalized storage and 100-year return period base condition flow rates for five watersheds and four release rate scenarios

1.3.2 Flood mitigation levels

1.3.2.1 Flood mitigation levels at the watershed scale

Flood mitigation levels (dW), as defined in the methods section, were determined for 100-year return period storms at all cross sections on study reaches. The aggregate average values computed from this database for DIAs and Non-DIAs in six watersheds are presented in Table 8. Except for the Cal Sag and Poplar Creek watersheds, DIAs were found to have on average higher flood mitigation levels than Non-DIAs. As measured by Cohen's *d,* the difference in the two averages is statistically small for the Des Plaines watershed, whereas it is large for Little Calumet and Upper Salt Creek watersheds. Cal Sag watershed DIAs, on the other extreme, were observed to have on average much lower flood mitigation levels than Non-DIAs. No average difference was observed in the Poplar Creek watershed. Comparative analysis of dW is inapplicable in the case of the North Branch watershed as there are no DIAs in the watershed study area. Overall, DIAs in the District study area were found to have moderately higher flood mitigation levels than Non-DIAs.

Table 8. Average Reduction in Peak Flood Levels (dW) in DIAs and Non-DIAs of Study Area Watersheds When WMO's Watershed-Specific Release Rate Requirements are Met

Watershed	*DIA dW (ft)*	*Non-DIA dW (ft)*	*Effect size (Cohen's d)*
Cal Sag	-0.56	-0.80	Large (-0.7)
Des Plaines	-0.89	-0.78	Small (0.2)
Little Calumet	-0.63	-0.32	Large (1.0)
North Branch	-	-0.32	-
Poplar Creek	-0.30	-0.30	Zero (0.0)
Upper Salt Creek	-0.64	-0.41	Large (0.8)
Overall (excluding NB)	-0.75	-0.51	Med (0.4)

1.3.2.2 Flood mitigation levels at the community level

Flood mitigation levels were next examined at the community level. Average dW values computed for various communities are listed in Table 9 (DIA communities) and Table 10 (Non-DIA communities) and are also displayed in map exhibits (Appendix B). Since DIAs are the focus of this study, only DIA community results are analyzed in detail here. Average dW values for DIA communities and their DIAs, like in storage analysis, were expressed as a percentage of average flood mitigation levels in the Non-DIAs of their respective watersheds. Sparse cross section data in conjunction with limited DIAs mean that there are only a handful of data points for all but the Des Plaines watershed. Box and whisker plots (Figure 17 and Figure 18) are thus presented only for the Des Plaines watershed and global analysis. As evident in Table 9 results, DIAs in all three Cal-Sag DIA communities have flood mitigation levels that are 51%–75% of the watershed Non-DIA average (WNA). Little Calumet and Upper Salt Creek communities, on the other hand, have DIA flood mitigation levels almost 150–200% of the WNA, except for Palatine DIA where the flood mitigation level is 59% of the WNA. There is only one community in Poplar Creek, Elgin, for which the DIA dW information is available, and it is nearly equal to the WNA. A much wider variation was observed in the Des Plaines watershed (Figure 17). Flood mitigation levels in its DIAs range between meager (11%, Rosemont) and high (225%, Stone Park) of the WNA. With a median value of 112% WNA, most communities in the Des Plaines watershed (and globally) have DIAs exceeding WNAs in flood mitigation levels.

Table 9. Flood Mitigation Levels in Various DIA Communities and their DIAs

DIA Community	*%DIA*	*Watershed Non-DIA Avg. (WNA)*		*Whole Community*		*DIAs only*	
		Watershed	*WNA dW (ft)*	*dW (ft)*	*% of WNA*	*dW (ft)*	*% of WNA*
Blue Island	76	Cal Sag	-0.80	-1.03	129%	–[1]	–
Chicago Ridge	62			-0.62	77%	-0.60	75%
Bridgeview	33			-0.41	51%	-0.41	51%
Crestwood	8			-0.66	82%	–	–
Oak Forest	4			-0.37	46%	–	–
Palos Hills	3			-0.59	74%	-0.43	54%
Stone Park	100	Des Plaines	-0.78	-1.76	225%	-1.76	225%
Melrose Park	66			-1.09	140%	-1.48	190%
Franklin Park	59			-0.85	109%	-0.85	109%
Willow Springs	26			-0.91	117%	–	–
Prospect Heights	26			-0.28	36%	-0.16	20%
Bellwood	25			-1.22	156%	-0.95	122%
Des Plaines	17			-0.93	119%	-0.25	32%
Northlake	16			-1.37	176%	-1.63	209%
Countryside	16			-0.54	69%	–	–
Rosemont	14			-0.09	11%	–	–
Lyons	14			-0.21	27%	–	–
Mount Prospect	9			-1.11	142%	-1.00	128%
Arlington Heights	5			-0.47	60%	-0.51	66%
Sauk Village	63	Little Calumet	-0.32	-0.51	160%	-0.65	203%
Matteson	6			-0.42	131%	-0.49	153%
Niles	16	N. Branch/DP[2]	-0.32	-0.18	57%	-1.30	165%
Streamwood	13	Poplar Creek	-0.30	-0.45	150%	–	–
Elgin	1			-0.17	56%	-0.31	102%
Rolling Meadows	22	Upper Salt Ck.	-0.41	-0.62	150%	-0.85	206%
Elk Grove Village	14			-0.32	77%	-0.76	184%
Palatine	3			-0.50	122%	-0.24	59%

[1] En dash (–) implies that cross section data are not available in these areas.
[2] Most of Niles community is in the North Branch watershed, but its DIA lies almost entirely in the Des Plaines watershed. Thus, North Branch WNA was used in the "Whole Community" analysis, whereas Des Plaines WNA was considered for "DIAs only" analysis.

Table 10. Flood Mitigation Levels in Various Non-DIA Communities

Non-DIA Community	*Watershed Non-DIA Average (WNA)*		*Whole Community*	
	Watershed	*WNA dW (ft)*	*dW (ft)*	*% of WNA*
Oak Lawn	Cal Sag	-0.80	-1.02	127%
Alsip			-1.13	141%
Orland Hills			-0.37	46%
Orland Park			-0.77	96%
Park Ridge	Des Plaines	-0.78	-0.80	103%
Wheeling			-0.35	45%
Broadview			-0.35	44%
Brookfield			-0.70	89%
Buffalo Grove			-0.72	92%
Burr Ridge			-0.70	90%
Elmwood Park			-0.06	7%
Hinsdale			-1.26	161%
Indian Head Park			-1.18	152%
La Grange Park			-0.29	38%
North Riverside			-0.21	27%
Schiller Park			-1.21	156%
Westchester			-0.34	43%
Western Springs			-1.82	233%
Homewood	Little Calumet	-0.32	-0.38	121%
Glenwood			-0.34	108%
Lynwood			-0.14	44%
Lansing			-0.03	9%
Flossmoor			-0.38	120%
Olympia Fields			-0.49	154%
University Park			-0.33	104%
Glenview	North Branch	-0.32	-0.49	154%
Morton Grove			-0.27	85%
Deerfield			-1.16	366%
Glencoe			-0.24	75%
Golf			-0.22	71%
Northbrook			-0.34	106%
Northfield			-0.21	65%
Wilmette			-0.32	102%
Winnetka			-0.26	83%
Hoffman Estates	Poplar Creek	-0.30	-0.31	105%
South Barrington			-0.29	98%
Schaumburg	Upper Salt Creek	-0.41	-0.34	83%
Inverness			-0.33	80%

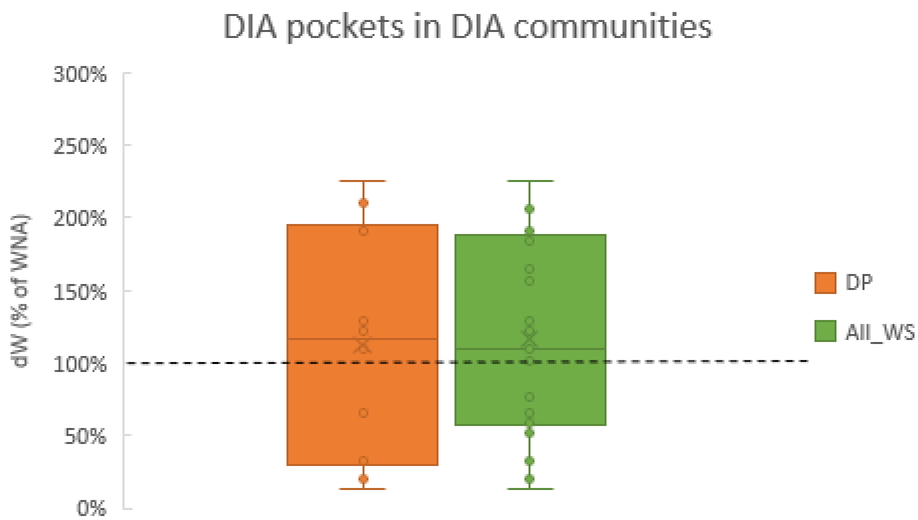


Figure 17. Distribution of DIA flood mitigation levels (dW) in relation to respective watershed Non-DIA averages (WNA). Each data point represents aggregated result for the DIA portion of a DIA community.

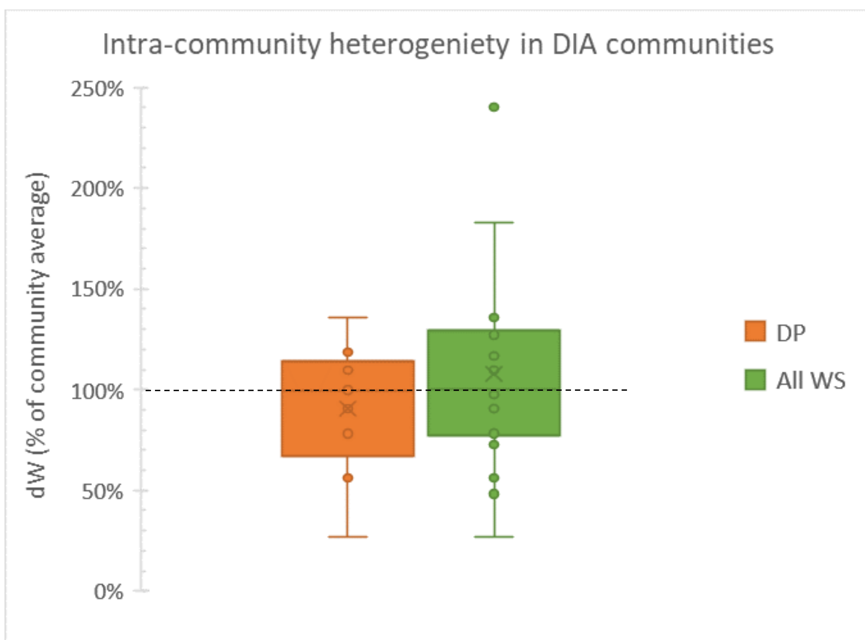


Figure 18. Distribution of DIA flood mitigation levels (dW) in relation to their respective community average dW. In other words, y-axis is $\frac{dW_{DIA\,only}}{dW_{whole\,community}}$ *expressed in percentage.*

Figure 18 facilitates the examination of intra-community heterogeneity in DIA communities' results. Here DIA flood mitigation levels in each DIA community are expressed as a percentage of the corresponding community average storage, and the resulting distribution is plotted for the Des Plaines watershed and all watersheds combined (Figure 18). Other watershed results were not plotted separately for the lack of enough data points. Table 9 and Figure 18 results revealed that there are a handful of communities where DIAs are either much smaller (e.g., Des Plaines and Prospect Heights in Des Plaines watershed, Palatine in Upper Salt Creek watershed) or much larger (e.g., Elk Grove Village in Upper Salt Creek watershed, Elgin in Poplar Creek watershed) than their respective community averages. DIAs in the remaining majority communities have flood mitigation levels fairly close to their community averages.

1.3.2.3 Flood mitigation benefits with more restrictive release rates

Flood mitigation analyses presented above were based on the watershed-specific release rates mandated by WMO, 0.30 cfs/ac for Cal Sag and North Branch watersheds, 0.25 cfs/ac for Little Calumet and Polar Creek watersheds, and 0.20 cfs/ac for Des Plaines and Upper Salt Creek watersheds. In this segment, release rates smaller than that required by WMO for a watershed are also considered to evaluate the potential flood mitigation benefits offered by a set of more restrictive release rates. Mitigation levels were categorized based on a watershed's total reach lengths across five classes of flood mitigation levels (as defined in Table 2) for each watershed at the prescribed WMO release rate and more restrictive release rates. Figure 19 provides a visual account of this categorization of flood mitigation levels for six watersheds at relevant release rates. As seen here, a more restrictive release rate generally enhances potential risk mitigation by increasing the reach length with significant benefits and reducing the reach length with trivial benefits. The magnitude of this shift varies across release rates and watersheds. As expected, not all classes shrink or expand by the same amount. To simplify the evaluation, this segment primarily examines the shifts in two relatively inferior classes, *moderately below* (yellow) and *much below average* (red). Shrinkage of these two classes would imply that there are fewer study reaches with a reduction in peak flood levels smaller than 0.5 ft, thereby providing a reasonable measure of benefits associated with a more restrictive release rate. Per Figure 18, going from 0.30 cfs/ac to 0.15 cfs/ac offers substantial flood mitigation benefits in the Cal Sag watershed. Note that results corresponding to intermediate release rates, 0.25 cfs/ac and 0.20 cfs/ac, were not considered as these scenarios were modeled only for Tinley Creek subbasins in the watershed during the previous phases. The North Branch watershed is highly sensitive to changes in release rates; large shifts to superior dW classes were noted at all release rate alternatives. In the case of Little Calumet and Poplar Creek watersheds, moving to 0.20 cfs/ac offers considerable benefits; further reduction in the release rate to 0.15 cfs/ac leads to only marginal additional improvements. The Des Plaines watershed exhibits relatively moderate sensitivity to change in release rate; only marginal improvements are observed on moving from 0.20 cfs/ac to 0.15 cfs/ac. Unlike the Des Plaines watershed, gains in moving from 0.20 cfs/ac to 0.15 cfs/ac release rate are significantly large in the case of Upper Salt Creek.

Similar analysis was conducted for study reaches in DIAs. Results of this analysis are presented in Figure 20. In the case of the Cal Sag watershed, relatively large benefits in DIA flood mitigation levels are expected from reducing the release rate from 0.30 cfs/ac to 0.15 cfs/ac. This gain in flood mitigation levels is apparently greater than that expected in watershed Non-DIAs, which could help alleviate the large negative gap in flood mitigation levels between the two areas reported in Table 8. Results for 0.20 cfs/ac and 0.25 cfs/ac release rates are not available as these scenarios were not modeled for the watershed in previous phases of this project. Moving to 0.20 cfs/ac offers considerable benefits to Little Calumet DIAs; further reduction in the release rate to 0.15 cfs/ac leads to only marginal improvements. Poplar Creek

DIAs are highly sensitive to change in release rates; relatively large benefits were noted at both release rate alternatives. Des Plaines DIAs, like the rest of the watershed, does not stand to gain much with a reduction in release rate. Upper Salt Creek DIAs, on the other hand, are expected to see significantly large shifts in flood risk mitigation levels with a reduction in the release rate. However, it is important to note that even at the prescribed WMO release rate, i.e., 0.20 cfs/ac, almost all DIA reaches studied in the Upper Salt Creek watershed have already attained flood mitigation levels that fall into either the *near average* class or the *moderately above average* class.

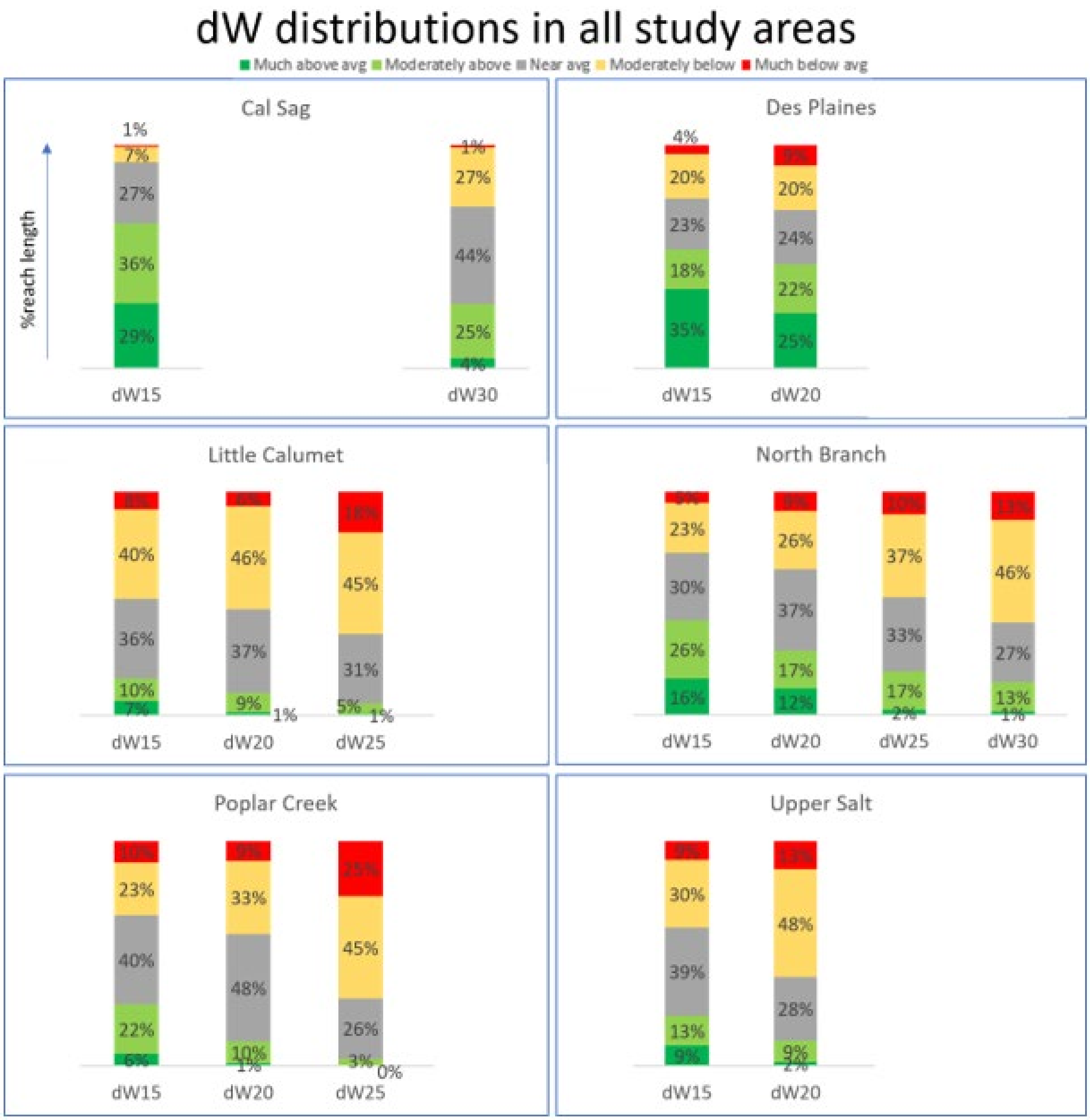


Figure 19. Distribution of study area flood mitigation levels across five potential risk mitigation classes at release rates equal to and less than that specified in WMO for various watersheds. Percentage values represent percentage of a watershed's study reach length belonging to a certain class of dW. dW15, dW20, dW25, and dW30 correspond to four release rate scenarios, 0.15 cfs/ac, 0.20 cfs/ac, 0.25 cfs/ac, and 0.30 cfs/ac, respectively.

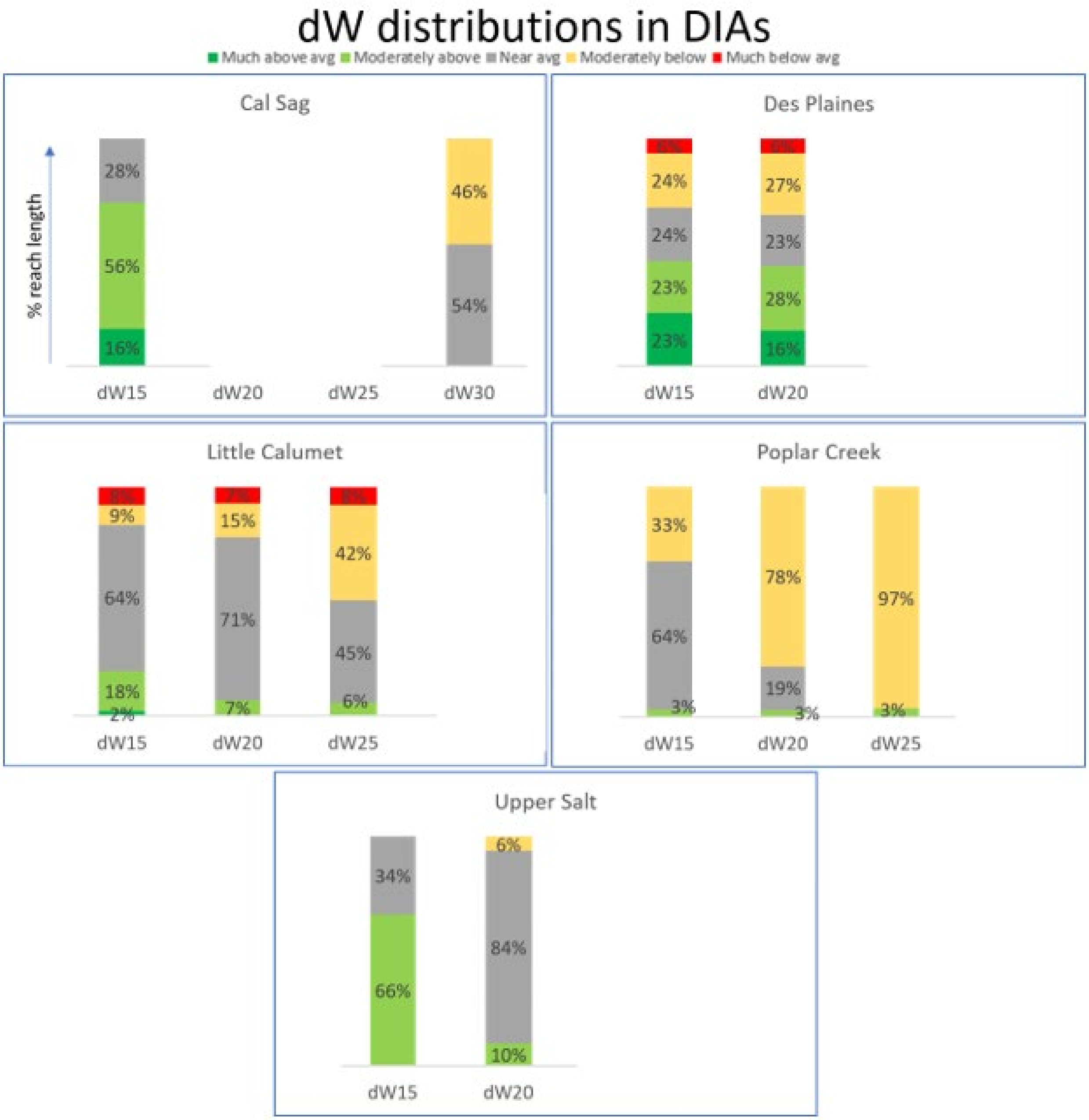


Figure 20. Distribution of DIA flood mitigation levels across five potential risk mitigation classes at release rates equal to and less than that specified in WMO for various watersheds. Percentage values represent percentage of a watershed's DIA reach length belonging to a certain class of dW. dW15, dW20, dW25, and dW30 correspond to four release rate scenarios, 0.15 cfs/ac, 0.20 cfs/ac, 0.25 cfs/ac, and 0.30 cfs/ac, respectively.

1.4 Summary and Conclusions

The relative impacts of watershed-specific release rates on DIAs and Non-DIAs in terms of detention storage requirements and flood mitigation levels were evaluated for the District. It was found that **District DIAs generally require marginally higher (~6% more) detention storage but enjoy moderately higher flood mitigation levels (~0.24 ft more) than Non-DIAs**. Differences in storage requirements between DIA and Non-DIA at watershed and community levels are also generally mild, unlike flood mitigation levels that exhibit much larger intra-watershed and intra-community variations between DIAs and Non-DIAs. This larger variation in flood mitigation levels is partly attributable to scarce hydraulic data, especially in DIAs.

DIAs in the Cal Sag watershed require marginally higher detention storage (~only 1% more) and attain significantly smaller average flood mitigation levels (~0.24 ft less) than its Non-DIAs. In this relative sense, DIAs in Cal Sag are the worst impacted by the watershed-specific release rates. Remarkable inter- and intra-community homogeneity was observed in the distribution of storage requirements. Note that only Tinley Creek and Stony Creek subwatersheds were included in the analyses. Any extrapolation of these results to other Cal Sag subwatersheds should thus be treated with due care.

In the Des Plaines River watershed, DIAs require marginally higher storage (~6% more) and attain marginally higher flood mitigation levels (~0.11 ft more) compared to Non-DIAs. Large spatial variations in relative performance indicators for both storage requirements and flood mitigation levels were found to be a characteristic feature of the watershed. In other words, even though on average DIAs and Non-DIAs have similar storage requirements and flood mitigation levels, there are a handful of communities where DIAs have much different (higher or lower) storage requirements and/or flood mitigation levels than Non-DIAs.

The relative performance of DIAs is perhaps best in the case of the Little Calumet watershed. In comparison to watershed Non-DIAs, DIAs require moderately higher storage (~11% more) and experience a significantly larger average reduction in peak flood levels (~0.31 ft more). This trend was observed uniformly throughout the watershed and within communities.

Poplar Creek watershed DIAs require moderately higher (~10% more) storage and observe identical flood mitigation levels relative to Non-DIAs. There are very few DIA communities in the watershed to comment on the spatial variation across communities.

In the case of the Upper Salt Creek watershed, DIAs on average require moderately higher (~14% more) storage and experience a much larger reduction in peak flood levels (~0.23 ft more) compared to Non-DIAs. Like in the Poplar Creek watershed, there are few data points to comment on the spatial variation of quantities across watershed communities.

The North Branch watershed study area does not have any DIAs in the modeled areas and is thus not discussed here.

A pilot analysis involving a comparison of detention storage values corresponding to 100-year and 500-year return period storms in the Upper Salt Creek watershed indicates that the trend of DIAs requiring larger storage than Non-DIAs is likely to hold true at a higher return period storm in most District areas. This is particularly relevant following the recent ISWS publication, Bulletin 75- *Precipitation Frequency Study for Illinois* (Angel et al., 2020), that predicts increased intensity and frequency of extreme precipitation events in Cook County watersheds.

Gains in flood mitigation levels on adopting more restrictive release rates were analyzed. Findings showed that significantly more reaches in four watersheds (Little Calumet, North Branch, Poplar Creek, and Upper Salt Creek) and their DIAs would attain a peak flood level reduction above 0.5 ft on moving to the next more restrictive release rate (i.e., 0.30 cfs/ac, 0.25 cfs/ac, 0.20 cfs/ac, and 0.15 cfs/ac, respectively); further reduction in release rates does not yield as much additional benefits in flood mitigation levels. The Des Plaines River watershed and its DIAs, on the other hand, are relatively less sensitive to changes in release rates. Since intermediate release (i.e., 0.20 cfs/ac and 0.25 cfs/ac) scenarios were not modeled for Cal Sag study areas, this analysis is incomplete for the watershed. Flood mitigation levels do, however, reduce drastically in the watershed and its DIAs with the adoption of the most restrictive release rate, i.e., 0.15 cfs/ac.

These comparative results provide a better understanding of physical infrastructure requirements (detention storage) and flood mitigation benefits (reduction in peak flood levels) of WMO's watershed-specific release rates within DIAs relative to Non-DIAs. Barring some local variations, storage requirements are only marginally more stringent for DIAs. Flood mitigation benefits, on the other hand, are on average moderately larger for them. An improved understanding of the impact of release rates on DIAs enable policymakers and watershed managers to better evaluate whether policies address prevalent inequities in flood risk. Tabular and map exhibits provided in this report serve as a communication tool to inform various communities, policymakers, and stakeholders about the specific impacts of watershed-specific release rates proposed in previous phases of the project.

It is important to note the limitations of this study. Due to limited availability of hydraulic data, flood mitigation level analysis was constrained to limited specific areas in the District, especially in the case of Poplar Creek, Cal Sag, and Little Calumet watersheds. Further note that DIAs are, by definition, already at higher risk, and therefore any policy or effort aimed at reducing flood risk inequities should be evaluated in this light. Because District DIAs generally require marginally higher detention storage but enjoy moderately higher flood mitigation levels than Non-DIAs, the findings of this study may give the impression that watershed-specific release rates are more favorable to DIAs and are thereby equitable. This, however, would perhaps be an oversimplification of several complexities in play. Firstly, a more thorough cost-benefit analysis would involve calculating the economic costs of providing detention storage facilities and assigning dollar values to flood mitigation levels depending on the exposed property value estimates. Secondly, spatial and temporal variations in results become important in such a cost-benefit analysis. The scope of this study is limited only to hydrology and hydraulic elements, however, and an examination of economic aspects is recommended for future studies.

1.5 References

Angel, J.R., Markus, M., Wang, K.A., Kerschner, B.M., & Singh, S. (2020). Precipitation Frequency Study for Illinois. Illinois State Water Survey Bulletin 75, Champaign, IL.

Ashley, R., Garvin, S., Pasche, E., Vassilopoulos, A., & Zevenbergen, C. (Eds.). (2007). Advances in urban flood management (1st ed.). CRC Press. https://doi.org/10.1201/9780203945988

Campillo, P., & Simba, I. (2021). From Toxic Fluff in Lincoln Park, to the Smoke that Blanketed Little Village: A Snapshot of Environmental Justice Issues in Chicago. Illinois Environmental Council, Published on Feb 8, 2021. https://ilenviro.org/snapshot-of-environmental-justice-issues-in-chicago/. Accessed on May 10, 2022.

CMAP or Chicago Metropolitan Agency for Planning. (2018). ON TO 2050 Strategy Paper- Stormwater and Flooding. https://www.cmap.illinois.gov/documents/10180/653821/FY18-0051+STORMWATER+AND+FLOODING_FINAL.pdf. Accessed on May 12, 2022.

Coe, R. (2002, September). It's the effect size, stupid. In British Educational Research Association Annual Conference (Vol. 12, p. 14).

Cohen, J. (1969). Statistical Power Analysis for the Behavioral Sciences. Hillsdale, NJ: Lawrence Erlbaum Associates.

Demidenko, E. (2016). The p-value you can't buy. The American Statistician, 70(1), 33-38.

Fenner, Richard, Emily O'Donnell, Sangaralingam Ahilan, David Dawson, Leon Kapetas, Vladimir Krivtsov, Sikhululekile Ncube, and Kim Vercruysse. 2019. "Achieving Urban Flood Resilience in an Uncertain Future." Water 11 (5): 1082. Doi: http://dx.doi.org/10.3390/w11051082.

Festing, H., Copp, C., Sprague, H., Wolf, D., Shorofsky, B., & Nichols, K. (2014). The Prevalence and Cost of Urban Flooding - A case study of Cook County, IL. CNT or The Center for Neighborhood Technology https://cnt.org/sites/default/files/publications/CNT_PrevalenceAndCostOfUrbanFlooding2014.pdf Accessed on May 10, 2022.

Fialka, J. (2019). "When Storms Hit Cities, Poor Areas Suffer Most." Scientific American, E&E News. https://www.scientificamerican.com/article/when-storms-hit-cities-poor-areas-suffer-most/. Accessed May 9, 2022

Fielding JL (2018) Flood risk and inequalities between ethnic groups in the floodplains of England and Wales. Disasters 42(1):101–123

Flavelle, C., Lu, D., Penney, V., Popovich, N., & Schwartz, J. (2020). New data reveals hidden flood risk across America. The New York Times, Published on June 29, 2020. https://www.nytimes.com/interactive/2020/06/29/climate/hidden-flood-risk-maps.html Accessed on May 10, 2022.

Flegel, A., Byard, G., McConkey, S., Hanstad, C., Gaynor, N., & Zaloudek, Z. (2019). Watershed-Specific Release Rate Analysis: Cook County, Illinois. Illinois State Water Survey.

FLOODsite (2009) Language of risk. Project definitions. 2nd ed. Report T32-04-01, www.floodsite.net

Fothergill, A., & Peek, L. A. (2004). Poverty and disasters in the United States: A review of recent sociological findings. Natural hazards, 32(1), 89-110. doi:10.1023/B:NHAZ.0000026792.76181.d9

Frank, T. (2020). Flooding disproportionately harms Black neighborhoods. Scientific American: Houston, TX, USA. https://www.scientificamerican.com/article/flooding-disproportionately-harms-black-neighborhoods/. Accessed on May 9, 2022.

Frankson, R., K. Kunkel, S. Champion, B. Stewart, D. Easterling, B. Hall, and J.R. Angel. (2017). Illinois State Climate Summary. NOAA Technical Report NESDIS 149-IL, 4 pp., https://statesummaries.ncics.org/il.

Guo, J. C. (1999). Detention storage volume for small urban catchments. Journal of water resources planning and management, 125(6), 380-382.

Hallegatte, S., Vogt-Schilb, A., Bangalore, M., & Rozenberg, J. (2016). Unbreakable: building the resilience of the poor in the face of natural disasters. World Bank Publications.

Harris, T., & Simba, I. (2022). A Look at Chicago's Environmental Justice Battles, Impacts and Solutions. Illinois Environmental Council, Published on Jan 5, 2021. https://ilenviro.org/a-look-at-chicagos-environmental-justice-battles-impacts-and-solutions/. Accessed on May 10, 2022.

HUD or U.S. Department of Housing and Urban Development, 2020. Memorandum, entitled "Uncapped" Income Limits for 2020, from the Office of Block Grant Assistance and the Office of Affordable Housing Programs. https://www.huduser.gov/portal/datasets/CDBG/2020-Uncapped-IncomeLmts-Memo.pdf. Accessed on July 7, 2022.

HUD or U.S. Department of Housing and Urban Development, 2022. ACS 5-Year 2011-2015 Low and Moderate Income Summary FAQs. https://www.hudexchange.info/programs/acs-low-mod-summary-data/acs-low-mod-summary-data-faqs-2011-2015/. Accessed on July 7, 2022.

Huff, F. A., & Angel, J. R. (1992). Rainfall frequency atlas of the Midwest. ISWS Bulletin 71.

IPCC, 2022: Climate Change 2022: Impacts, Adaptation, and Vulnerability. Contribution of Working Group II to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [H.-O. Pörtner, D.C. Roberts, M. Tignor, E.S. Poloczanska, K. Mintenbeck, A. Alegría, M. Craig, S. Langsdorf, S. Löschke, V. Möller, A. Okem, B. Rama (eds.)]. Cambridge University Press. In Press.

Katz, L. (2021). A racist past, a flooded future: formerly redlined areas have $107 billion worth of homes facing high flood risk—25% more than non-redlined areas. Redfin. https://www.redfin.com/news/redlining-flood-risk/ Accessed on May 10, 2022.

Keenan, M. B., Shankar, P., & Haas, P. (2019). Assessing disparities of urban flood risk for households of color in Chicago. Illinois Municipal Policy Journal, 2019, 4(1), 1-18.

Markus, M., Angel, J. R., Byard, G. J., Zhang, C., Zaloudek, Z., & McConkey, S. A. (2016). Communicating the Impacts of Potential Future Climate Change on the Expected Frequency of Extreme Rainfall Events in Cook County, Illinois. Illinois State Water Survey.

Markus, M., Angel, J., Wang, K., Byard, G., McConkey, S., & Zaloudek, Z. (2017). Impacts of potential future climate change on the expected frequency of extreme rainfall events in Cook, DuPage, Lake, and Will counties in northeastern Illinois. Illinois State Water Survey.

Mitchell, T., Guha-Sapir, D., Hall, J., Lovell, E., Muir-Wood, R., Norris, A., ... & Wallemacq, P. (2014). Setting, measuring and monitoring targets for reducing disaster risk. Recommendations for post-2015 international policy frameworks. ODI.

MWRD or Metropolitan Water Reclamation District of Greater Chicago (2021). Chicago's Water Reclamation District approves 2022 budget. MWRD News webpage. https://mwrd.org/chicagos-water-reclamation-district-approves-2022-budget-waterworld. Published on Dec 12, 2021. Accessed on May 12, 2022.

MWRD or Metropolitan Water Reclamation District of Greater Chicago (2022). Water infrastructure resiliency: U.S. Sens. Duckworth, Durbin support MWRD work to mitigate flooding, improve water quality. MWRD News webpage. https://mwrd.org/water-infrastructure-resiliency-us-sens-duckworth-durbin-support-mwrd-work-mitigate-flooding . Published on March 28, 2022. Accessed on May 12, 2022.

Oudin, L.; Salavati, B.; Furusho, C.; Ribstein, P.; Saadi, M. (2016). Hydrological impacts of urbanization at the catchment scale. J. Hydrol. 2018, 559, 774–786.

Schanze, J. (2006). Flood Risk Management & A Basic Framework, In: Schanze, J., Zeman, E. and Marsalek J. (eds.), Flood risk management Hazard, vulnerability and mitigation measures. Dortrecht, Springer, 1&20.

Schanze, J. (2016). Resilience in flood risk management–Exploring its added value for science and practice. In E3S Web of Conferences (Vol. 7, p. 08003). EDP Sciences.

Shuster, W.D.; Bonta, J.; Thurston, H.; Warnemuende, E.; Smith, D.R. Impacts of impervious surface on watershed hydrology: A review. Urban Water J. 2005, 2, 263–275.

Sullivan, G. M., & Feinn, R. (2012). Using effect size—or why the P value is not enough. Journal of Graduate Medical Education, 4(3), 279-282.

Travers, J. C., Cook, B. G., & Cook, L. (2017). Null hypothesis significance testing and p values. Learning Disabilities Research & Practice, 32(4), 208-215.

United States Environmental Protection Agency (USEPA), 2022. Accessed from https://www.epa.gov/environmentaljustice on July 5, 2022.

USGCRP, 2018: Impacts, Risks, and Adaptation in the United States: Fourth National Climate Assessment, Volume II [Reidmiller, D.R., C.W. Avery, D.R. Easterling, K.E. Kunkel, K.L.M. Lewis, T.K. Maycock, and B.C. Stewart (eds.)]. U.S. Global Change Research Program, Washington, DC, USA, 1515 pp. doi: 10.7930/NCA4.2018.

Wing, O.E.J., Lehman, W., Bates, P.D. et al. Inequitable patterns of US flood risk in the Anthropocene. Nat. Clim. Chang. 12, 156–162 (2022). https://doi.org/10.1038/s41558-021-01265-6

Wuebbles, D., J. Angel, K. Petersen, and A.M. Lemke (Eds.), 2021: An Assessment of the Impacts of Climate Change in Illinois. The Nature Conservancy, Illinois, https://doi.org/10.13012/ B2IDB-1260194_V1

Chapter 2. Impacts of Watershed-Specific Release Rates in Collar Counties [WMO Article 208.3]

2.1 Background

The Metropolitan Water Reclamation District of Greater Chicago (MWRD, or the District) first adopted the Watershed Management Ordinance (WMO) on May 1, 2014 and most recently amended the WMO on April 7, 2022. Article 208 of the amended ordinance directs that "The District shall initiate a study of certain current provisions of and potential amendments to this Ordinance. This study will be initiated by the end of 2019 with a targeted completion date of May 2025." Article 208.3 calls for a study on the "impacts of release rates under existing and future development scenarios in collar counties on watersheds in the District."

The Illinois State Water Survey (ISWS) developed a Watershed-Specific Release Rate methodology in its Phase I study, which was later applied more broadly in Phase II to determine regulatory release rates for all District watersheds. Regarding Article 208.3, the goal of this Phase III study was to expand upon this methodology to include additional assessment of watershed management decisions outside the WMO regulatory area that could impact potential future flood risks within Cook County, excluding the City of Chicago. The results of the Phase I and Phase II studies were published in the report titled "*Illinois State Water Survey Contract Report 2019-06: Watershed-Specific Release Rate Analysis: Cook County, Illinois*," (https://hdl.handle.net/2142/103416) authored by Flegel, Byard, McConkey, Hanstad, Gaynor, and Zaloudek in March 2019. Analysis for Article 208.3 was to include the application of the Phase I and Phase II methodology to areas that are outside the WMO regulatory area but contribute inflow to select tributaries within the WMO regulatory area.

In the Phase I and Phase II studies, ISWS evaluated release rates by comparing results from the MWRD Detailed Watershed Plan base models with results from future scenario models. Models of future development in Cook County simulated conditions at release rates ranging from 0.15 cubic feet per second per acre (cfs/ac) to 0.30 cfs/ac. Models of future development in portions of collar counties upstream of Cook County simulated conditions at the respective county's established stormwater release rate at the time of the study. With regard to Article 208.3, the goal of Phase III was to expand upon the methodologies developed in Phases I and II of the Watershed-Specific Release Rate Study and include an additional assessment of watershed management decisions outside of the District that could potentially impact future flood risks within Cook County, excluding the City of Chicago. As such, ISWS modeled future development conditions in portions of the neighboring (collar) counties upstream of Cook County at various release rates, ranging from 0.15 cfs/ac to 0.30 cfs/ac while using Watershed-Specific Release Rates specified in the May 7, 2020 WMO as amended for subbasins falling within the WMO jurisdiction. This analysis would help the District identify the sensitivity of potential flooding impacts due to future development outside of the District on streams within the District and estimate the magnitude of any increases.

2.2 Methods and Procedures

The methodology for selecting watershed release rates in Cook County was developed in Phase I of this project and broadly applied in Phase II (Flegel et al., 2019). For a complete review of the methodology employed, including the background, the technical and public outreach (including with the MWRD Technical Advisory Committee), the determination of future development scenarios, and the various sensitivity analyses that were performed to evaluate the methodology, please see "*Illinois State Water Survey Contract Report 2019-06: Watershed-Specific Release Rate Analysis: Cook County, Illinois*" by Flegel et al. (2019).

The impact of the selection of an allowable release rate for future development is evaluated by comparing the results from base models with the results from future scenario models. Base models were established from the available hydrologic and hydraulic models completed from previous Detailed Watershed Plans (DWP) prepared by and for MWRD. The future scenario models include adjustments of the base U.S. Army Corps of Engineers HEC-HMS hydrologic model parameters for a specified percentage of area assumed to be developed and meeting the WMO volume control and detention storage requirements. The HEC-HMS results of the future scenarios, with variations on the release rate used for storage determination, are then routed through the U.S. Army Corps of Engineers HEC-RAS unsteady state hydraulic models. The future scenario results are compared to the base model results to determine whether a particular release rate meets the objective of mitigating increases in peak flood levels due to future development.

Future scenarios for this Phase III study were modeled by revising the base model hydrologic parameters to incorporate future land development in the same manner as in Phase I and Phase II. Future scenario hydrologic parameters were based on the same average development parameters identified in Phase II of this project. The future scenario was modeled to represent average future development of 40% of the land area. An average developed curve number of 88 was used to model rainfall runoff. Transformation parameters, such as the time of concentration and routing coefficients, were kept the same for the future development and the original base models.

Modeling of the WMO stormwater management requirements for future development included both volume control and stormwater detention. The HEC-HMS canopy method was used to reduce the rainfall by 1 inch for the average impervious area of development (52%) to address volume control requirements. Detention storage was based on a linear outflow hydrograph to calculate storage discharge relationships.

In Phase II, the effective release rates used in the adjacent counties at the time of the analysis were unchanged for the future conditions analysis. For this Phase III analysis, the release rate in portions of the watersheds that fell outside of Cook County were varied to analyze the impacts to water surface elevations on streams within Cook County. The four release rates (0.15, 0.20, 0.25, and 0.30 cfs/ac) considered in Phase II within Cook County were analyzed in Phase III for each collar county area tributary to the selected subwatersheds. Modeling in areas subject to the WMO used the release rate determined in Phase II and now defined in Appendix B of the WMO for each watershed.

The criteria identified in Phase I and applied in Phase II were used to evaluate impacts to water surface elevation within the District due to alternative release rates used in the adjacent counties. Analysis criteria included the percentage of stream length with increases in peak water surface elevation greater than 0.1 feet, the maximum water surface increase at any cross section location, and the maximum water surface increase at any reservoir with model results showing an increased flood elevation.

Only those subwatersheds analyzed in detail as part of Phase II with significant amounts of drainage area outside of Cook County were considered for additional analysis in Phase III. This included (a) North Branch Chicago River watershed with drainage area including Lake County, Illinois, and (b) select subwatersheds within the Lower Des Plaines River watershed including Buffalo Creek subwatershed with drainage areas including Lake County, Illinois, as well as Addison Creek and Salt Creek with drainage areas including DuPage County, Illinois.

A quality-assurance review was performed at the completion of each modeling scenario to review the storage-outflow curves generated by the analysis, the model connectivity, and to identify numerical instabilities occasionally identified at isolated locations during Phase II. These numerical instabilities were addressed by further refining the HEC-RAS HTAB hydraulic parameters of the DWP base models at individual cross sections.

2.2.1 North Branch Chicago River watershed

Hydrologic and hydraulic analyses were completed for modeled streams in the North Branch watershed during Phase II. The hydrologic HEC-HMS models and single HEC-RAS hydraulic model from the North Branch Chicago DWP represented the drainage area upstream of the North Shore Channel and were used for the base condition model with updates for major stormwater projects. Although base runoff rates and the elongated watershed shape were indicators that a more restrictive (lower) release rate may have helped mitigate future flood hazards, Phase II found a more restrictive release rate was not necessary to mitigate increases in water surface elevation due to future development, but this finding relied on the assumption that existing volume control and stormwater detention practices remained unchanged within Lake County.

Approximately 50% of the DWP study drainage area lies outside Cook County for the North Branch Chicago River watershed. The area outside Cook County is also included in the DWP hydrologic model. To assess the impacts of release rates under existing and future development scenarios in collar counties on watersheds in the District, future condition scenarios were included in the models for the area outside of the Cook County WMO jurisdiction. The same future development assumptions that were applied for Cook County in the Phase II analysis (namely 40% development/redevelopment) were also applied to Lake County, and alternative release rate selections (0.15, 0.20, 0.25, and 0.30 cfs/ac) in Lake County were evaluated for impacts on water surface elevations within the District. The 0.15 cfs/ac scenario in Lake County, the current release rate for that jurisdiction, was analyzed in Phase II and was included for comparative purposes in Phase III as well.

Future scenarios included one future development assumption and four target release rates applied uniformly throughout the Lake County portion of the watershed, with the Cook County release rate modeled using the WMO Watershed-Specific Release Rate as defined in Appendix B of the WMO. These results were compared with the Phase II base model results to evaluate the impacts.

Prior to any new analysis for this phase of the project, the ISWS coordinated with MWRD to identify any new stormwater projects completed since Phase II that were expected to significantly impact the assessment of the release rates under evaluation. No such stormwater projects were identified during the review. A full discussion of the stormwater projects identified during Phase II that were expected to influence the selection of watershed-specific release rates and thus were incorporated into the Phase II and III modeling can be found in Flegel et al. (2019).

2.2.2 Lower Des Plaines River watershed

Hydrologic and hydraulic modeling analyses were completed for the modeled streams of the Lower Des Plaines River watershed during Phase II. The Des Plaines River watershed DWP includes separate models for each subwatershed draining to the Des Plaines River within Cook County, which was the foundation for the base condition modeling.

Of these subwatersheds, Buffalo Creek and Salt Creek (including Addison Creek) contain a significant amount of drainage area within Lake and DuPage Counties, respectively. The same 40% future development assumption applied in Cook County in the Phase II analysis was applied to Lake and DuPage Counties, and the impacts of alternative release rate selections in Lake and DuPage Counties within the WMO jurisdiction were evaluated under four release rate scenarios (0.15, 0.20, 0.25, and 0.30 cfs/ac) for impacts to water surface elevations within the District. The 0.15 cfs/ac scenario in Lake County was analyzed in Phase II for Buffalo Creek and was included for comparative purposes in Phase III as well.

The Phase II study found that release rates for development along the main stem of the Des Plaines River in Cook County alone will not mitigate water surface elevation increases due to future development, even without accounting for the projected impacts of future development in Wisconsin. Therefore, although the Lower Des Plaines River includes a significant amount of drainage area outside of the District, future development of the Lower Des Plaines River watershed was neither included in the evaluation of watershed-specific release rates in Phase II nor in this analysis.

Prior to any new analysis for this phase of the project, the ISWS coordinated with MWRD to identify any new stormwater projects completed since Phase II that were expected to significantly impact the assessment of the release rates under evaluation. No such stormwater projects were identified during the review. A full discussion of the stormwater projects identified during Phase II that were expected to influence the selection of watershed-specific release rates and thus were incorporated into the modeling can be found in Flegel et al. (2019) and remain in the Phase III modeling.

2.3 Results

The Watershed-Specific Release Rate Phase II study used a comparative technique to evaluate the impacts of a particular watershed management strategy. Peak water surface elevations during a critical duration storm event obtained from a base model using the current levels of development were compared with the peak water surface elevations from the same magnitude and duration storm using future development and watershed management assumptions. The same approach has been employed as part of this study. For each studied watershed or subwatershed, water surface elevation comparison maps and tabular summaries of the change in peak water surface elevations were prepared. Comparison maps include the peak water surface elevation at each hydraulic cross section and select reservoirs included in the DWP and future conditions modeling. These maps allow the user to identify spatial patterns in sensitivity to a particular watershed management practice. Tabular data were prepared using individual hydraulic cross sections and their associated downstream river stationing as well as peak stages in select reservoirs. The tabular data provide a holistic summary of the impacts within a watershed and help identify those scenarios that meet a particular watershed management objective. Flegel et al. (2019) include additional discussion of the development of these particular evaluations in collaboration with MWRD and the MWRD Technical Advisory Committee.

The results of this modeling are purely used to understand the sensitivity of peak flood elevations to changing watershed management practices outside of the District's control and do not indicate any awareness of potential future changes to release rates in adjoining counties, including Lake or DuPage Counties, Illinois.

2.3.1 North Branch Chicago River watershed

The North Branch Chicago River watershed modeling included two watershed management practices for release rates, one fixed for all scenarios and one variable to evaluate sensitivity. Within Cook County, areas subject to the WMO Watershed-Specific Release Rate were modeled using the prescribed release rate listed in the WMO Appendix B, namely 0.30 cfs/ac. Within areas tributary to Cook County located in Lake County, Illinois, a separate watershed management practice was used. In Lake County, the release rate was varied from 0.15 cfs/ac to 0.30 cfs/ac to understand the impacts of extra jurisdictional changes in watershed management practices on peak water surface elevations within the District.

The results of the 0.15 cfs/ac release rate scenario are shown in Figure 21 and Table 11. The 0.15 cfs/ac scenario within Lake County with a 0.30 cfs/ac release rate was first modeled as part of Phase II during the evaluation of Watershed-Specific Release Rates within Cook County, Illinois. As found during Phase II, the results indicate that increases in peak water surface elevation due to development throughout the watershed are mitigated under these management scenarios. During Phase II, the mitigation of rises due to future development was defined as changes in peak water surface elevation of less than 0.1 feet at hydraulic cross sections or less than 0.5 feet within a reservoir between the base model conditions and future model conditions for a particular release rate. The majority of the watershed would be expected to see minor comparative decreases in peak water surface elevations due to future development under these practices, except for portions of Skokie River near the Lake County line where little change is observed. No increased stages were identified within the North Branch Chicago River reservoirs under this management scenario due to future development.

The results of the 0.20 cfs/ac release rate scenario are shown in Figure 22 and Table 11. Similar to the 0.15 cfs/ac release rate, under the 0.20 cfs/ac release rate, most cross sections show comparable water surface elevations between the base and future condition models due to future development. Those cross sections that exhibited an increase in peak water surface elevation greater than 0.1 feet were 0.13 feet and isolated near a single restrictive structure on the Middle Fork North Branch of the Chicago River. The increases occurred at only 0.3% of the stream length studied. During Phase II, such isolated and localized increases were not considered significant and as such did not indicate any issues in the release rate's ability to mitigate increases in peak water surface elevation due to future development within the watershed. These changes fell well within the range of hydraulic stability sensitivity across the DWP models as a whole. No reservoirs were identified with more than 0.5 feet higher water surface elevation due to future development.

Table 11. Evaluation Criteria/Metrics for Collar County Release-Rate Impact Analysis in North Branch Chicago River Watershed

Criteria	*Collar County Release Rate*			
	0.15 (cfs/ac)	*0.20 (cfs/ac)*	*0.25 (cfs/ac)*	*0.30 (cfs/ac)*
Stream length with increase in peak water surface elevation (WSEL) > 0.1' (ft)	78	798	88,509	144,713
Stream length with increase in peak WSEL> 0.1' (%)	0.0%	0.3%	30.8%	50.4%
Maximum XS WSEL increase	0.06'	0.13'	0.30'	0.42'
Maximum reservoir WSEL increase	--	0.01'	0.02'	0.11'
Reservoirs with increases > 0.5'	--	--	--	--

At the 0.25 cfs/ac release rate within Lake County, the ability to mitigate future increases in peak water surface elevation due to future development drops significantly from the more conservative release rates discussed above. More than 30% of cross sections would be expected to experience increased peak water surface elevations. Although the peak increase in water surface elevation remains relatively small at 0.3 feet, it nonetheless would violate the selection criteria established during Phase II. As seen in Figure 23, these increases in peak water surface elevation under this management scenario are found primarily along the Skokie River and along the Middle Fork North Branch of the Chicago River near the confluence with Skokie River.

The inability of the modeled management practice to mitigate future increases in peak water surface elevation first identified at the 0.25 cfs/ac release rate becomes even more pronounced at 0.30 cfs/ac. More than half of the stream within the study area would be expected to experience increased peak flood elevations due to future development with the largest increase of over 0.4 feet as can be seen in Table 11. The expected location of increased water surface elevation follows a similar pattern as in 0.25 cfs/ac with the increases focused along the Skokie River and the Middle Fork North Branch Chicago River, but with additional increases extending downstream to the North Branch Chicago River main stem as shown in Figure 24.

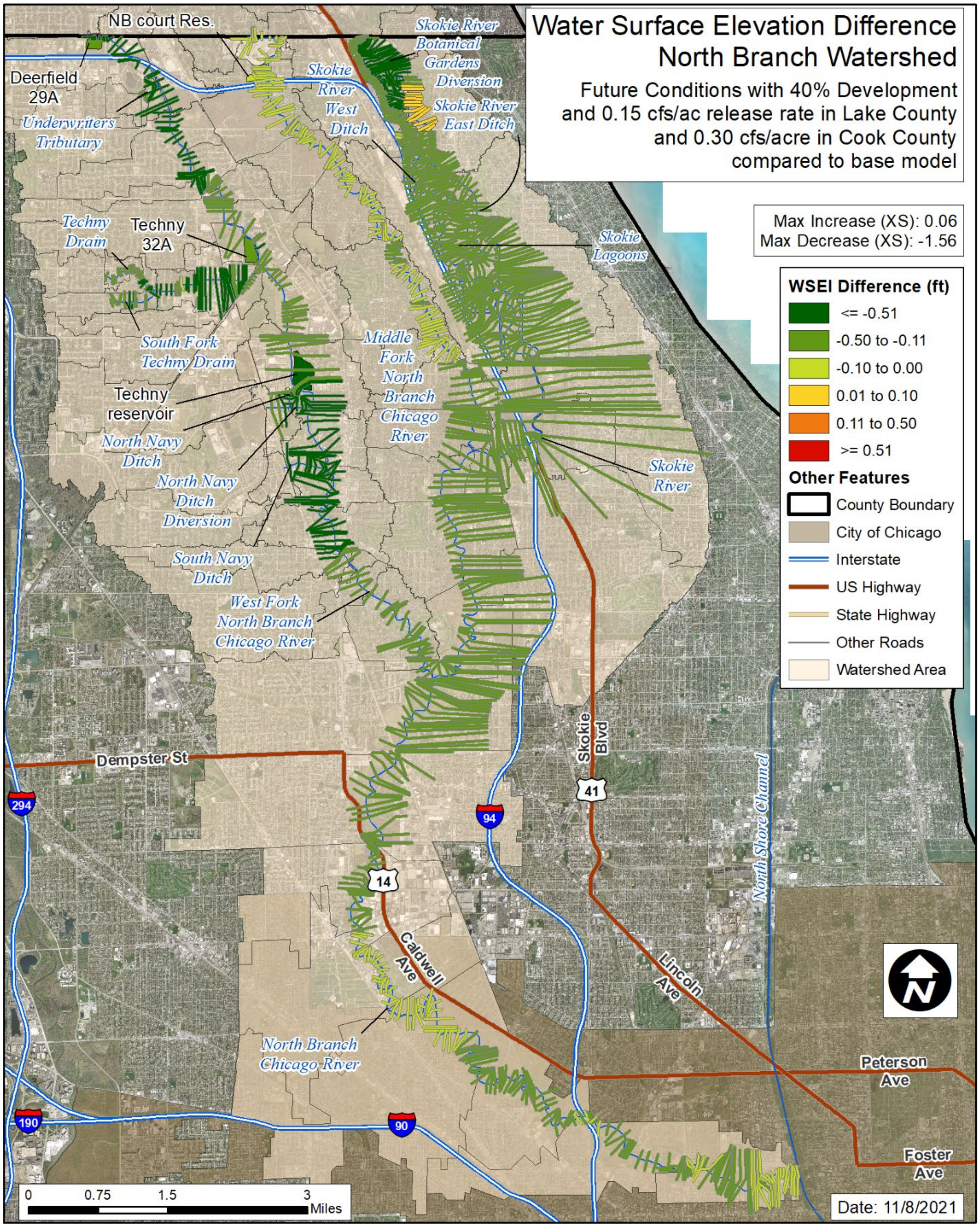


Figure 21. Water surface elevation differences by cross section between the base model and future conditions model using a 0.30 cfs/ac release rate within WMO jurisdiction and 0.15 cfs/ac within Lake County

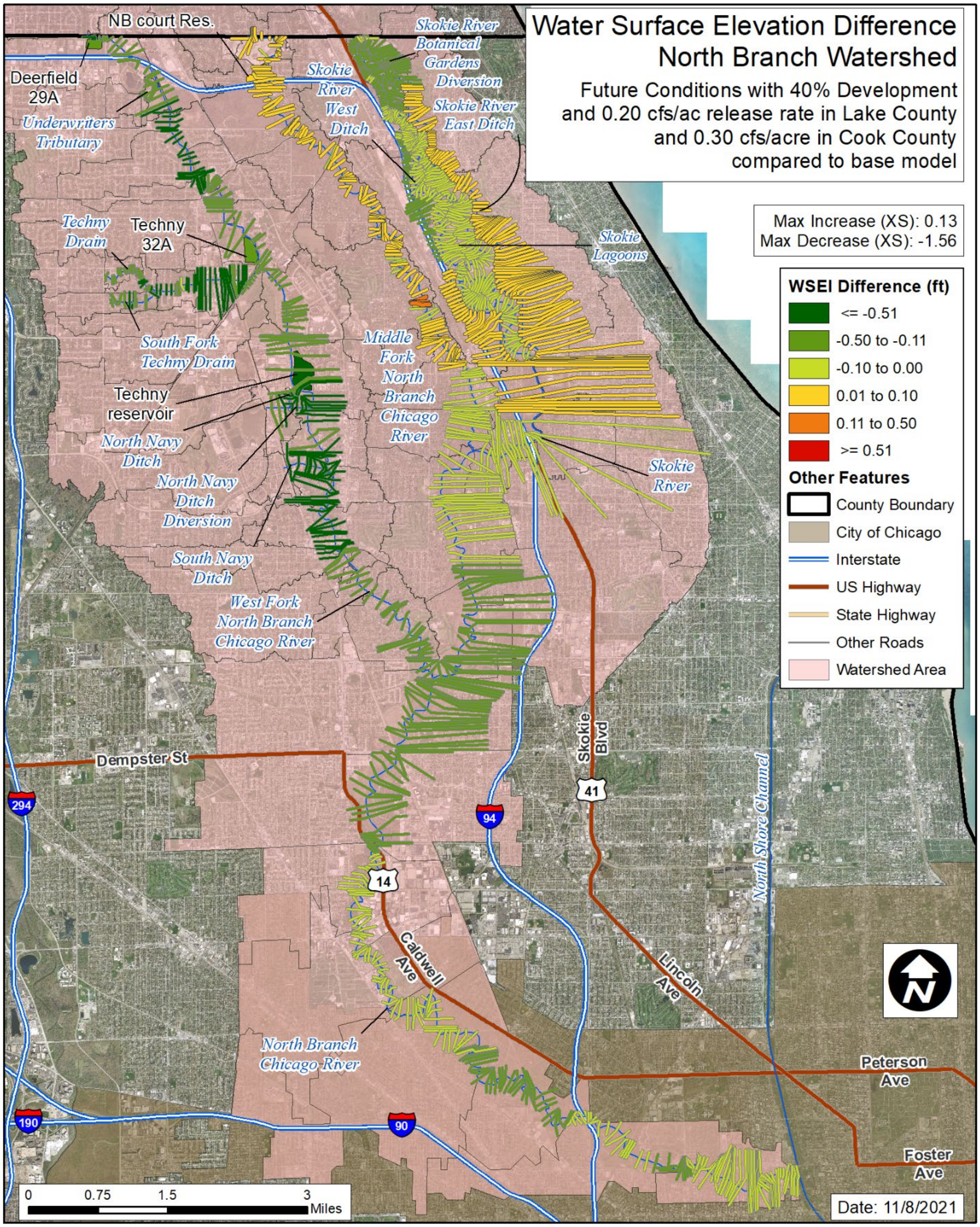


Figure 22. Water surface elevation differences by cross section between the base model and future conditions model using a 0.30 cfs/ac release rate within WMO jurisdiction and 0.20 cfs/ac within Lake County

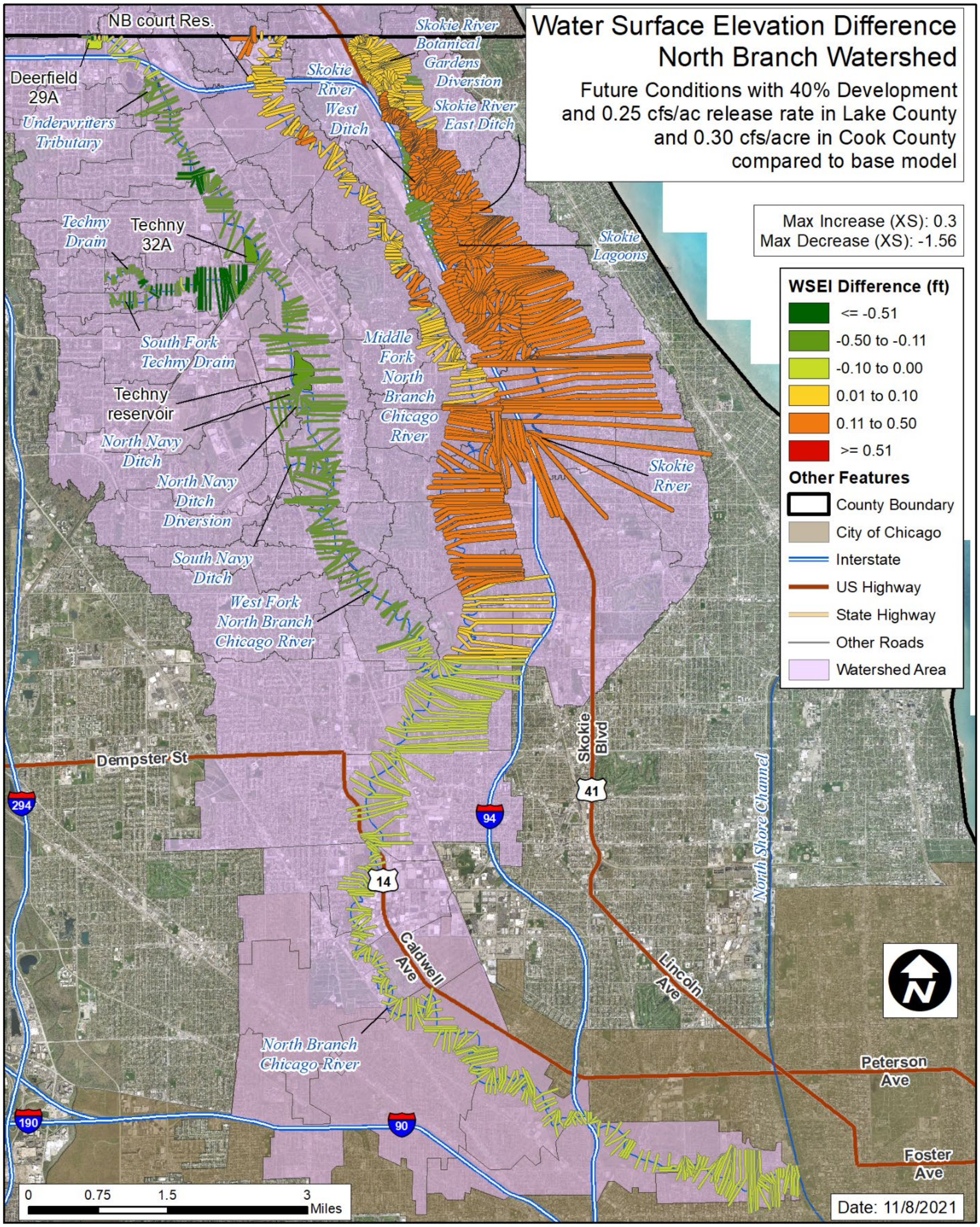


Figure 23. Water surface elevation differences by cross section between the base model and future conditions model using a 0.30 cfs/ac release rate within WMO jurisdiction and 0.25cfs/ac within Lake County

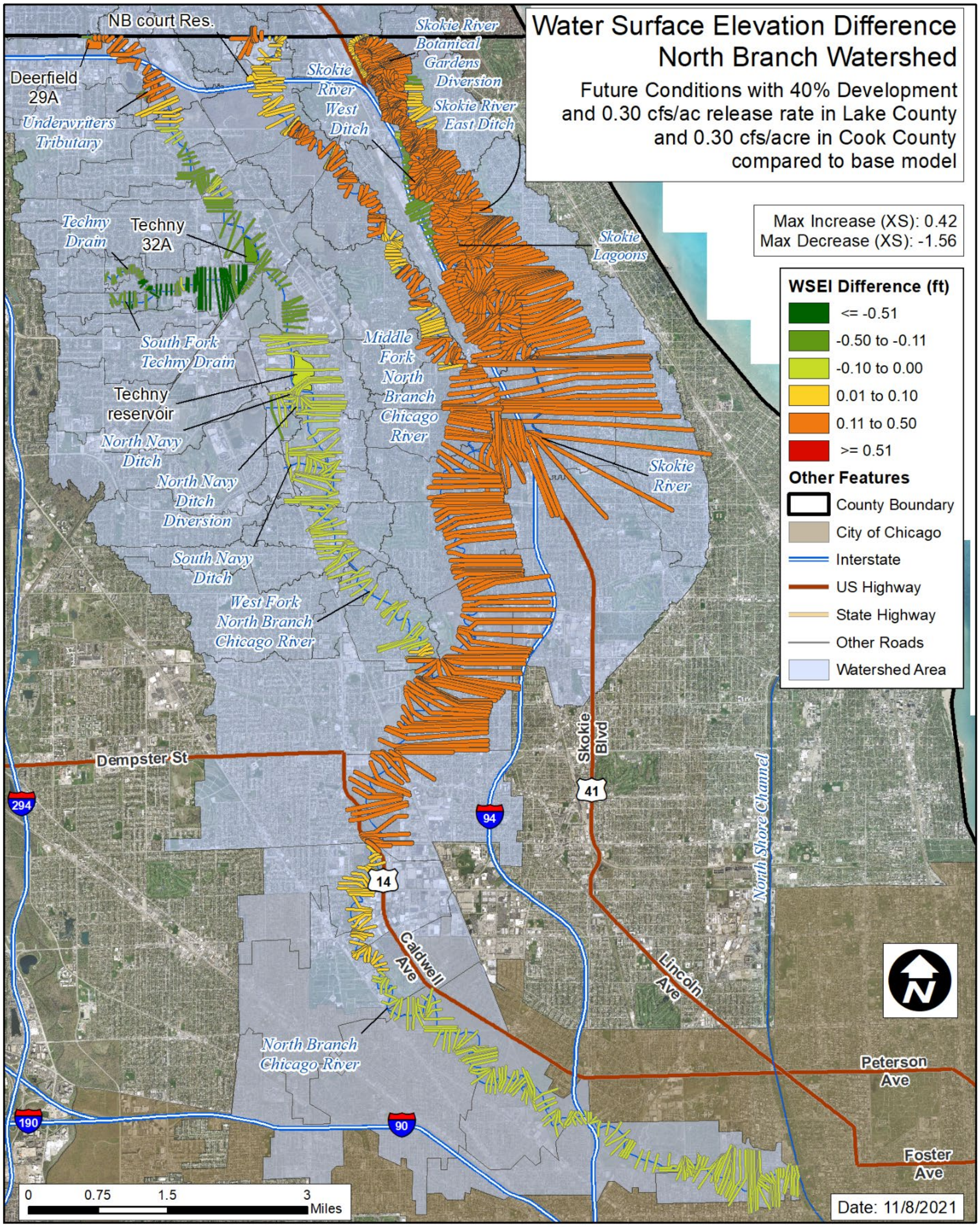


Figure 24. Water surface elevation differences by cross section between the base model and future conditions model using a 0.30 cfs/ac release rate within WMO jurisdiction and 0.30 cfs/ac within Lake County

2.3.2 Lower Des Plaines River watershed

Three subwatersheds within the Lower Des Plaines River watershed were considered for this analysis based on the percentage of the watershed falling under the stormwater release rate of a neighboring county. This includes Lower Salt Creek subwatershed, which includes the Addison Creek subwatershed as a tributary, and Buffalo Creek subwatershed.

The Lower Des Plaines River watershed modeling for this analysis included two watershed management practices for the release rate, one fixed for all scenarios and one variable to evaluate sensitivity. Within Cook County, areas subject to the WMO Watershed-Specific Release Rate were modeled using the prescribed release rate listed in the WMO Appendix B, namely 0.20 cfs/ac. Within areas tributary to Cook County located in Lake County or in DuPage County, Illinois, a separate watershed management practice was used. In these counties, the release rate was varied from 0.15 cfs/ac to 0.30 cfs/ac to understand the impacts of extra jurisdictional changes in watershed management practices on peak water surface elevations within the District.

The Addison Creek subwatershed drains portions of DuPage and Cook Counties. Discharge from Addison Creek joins Lower Salt Creek approximately 3 miles upstream of its confluence with the Des Plaines River.

Addison Creek was not sensitive to the selection of the release rate within DuPage County. As shown in Table 12 and Figure 25 through Figure 28, no cross sections or storage areas demonstrated significant increases in peak water surface elevation due to development under any of the tested release rates in DuPage County.

Although no significant increases were identified within the Addison Creek subwatershed, differences were observed in the maximum decrease in peak water surface elevations. As would be expected, the most restrictive release rate of 0.15 cfs/ac resulted in the greatest decrease in peak water surface elevations, and the 0.30 cfs/ac resulted in the smallest decrease in peak water surface elevation due to future development.

It is also important to note that while no changes in peak water surface elevation were observed within the Addison Creek subwatershed, Addison Creek is also tributary to the Lower Salt Creek subwatershed and, as such, the effects of the release rate within the DuPage County portion of Addison Creek could impact peak water surface elevations downstream of the confluence with Lower Salt Creek.

Table 12. Evaluation Criteria/Metrics for Collar County Release-Rate Impact Analysis in the Addison Creek Subwatershed

	Collar County Release Rate			
Criteria	*0.15 (cfs/ac)*	*0.20 (cfs/ac)*	*0.25 (cfs/ac)*	*0.30 (cfs/ac)*
Stream length with increase in peak water surface elevation (WSEL) > 0.1' (ft)	--	--	--	--
Stream length with increase in peak WSEL> 0.1' (%)	0.0%	0.0%	0.0%	0.0%
Maximum XS WSEL increase	--	--	--	--
Maximum reservoir WSEL increase	--	--	--	--
Reservoirs with increases > 0.5'	--	--	--	--

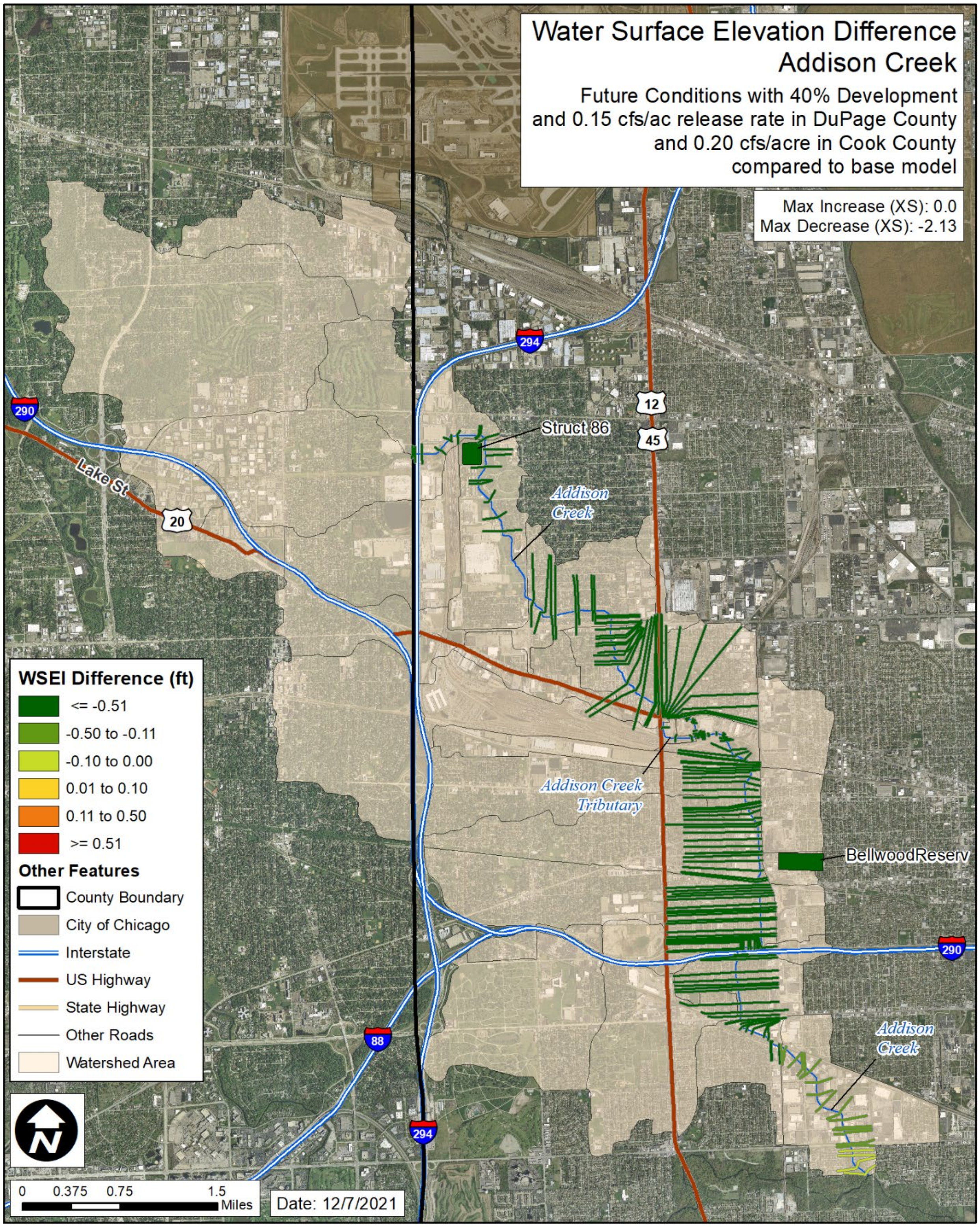


Figure 25. Water surface elevation differences by cross section between the base model and future conditions model using a 0.20 cfs/ac release rate within WMO jurisdiction and 0.15 cfs/ac within DuPage County

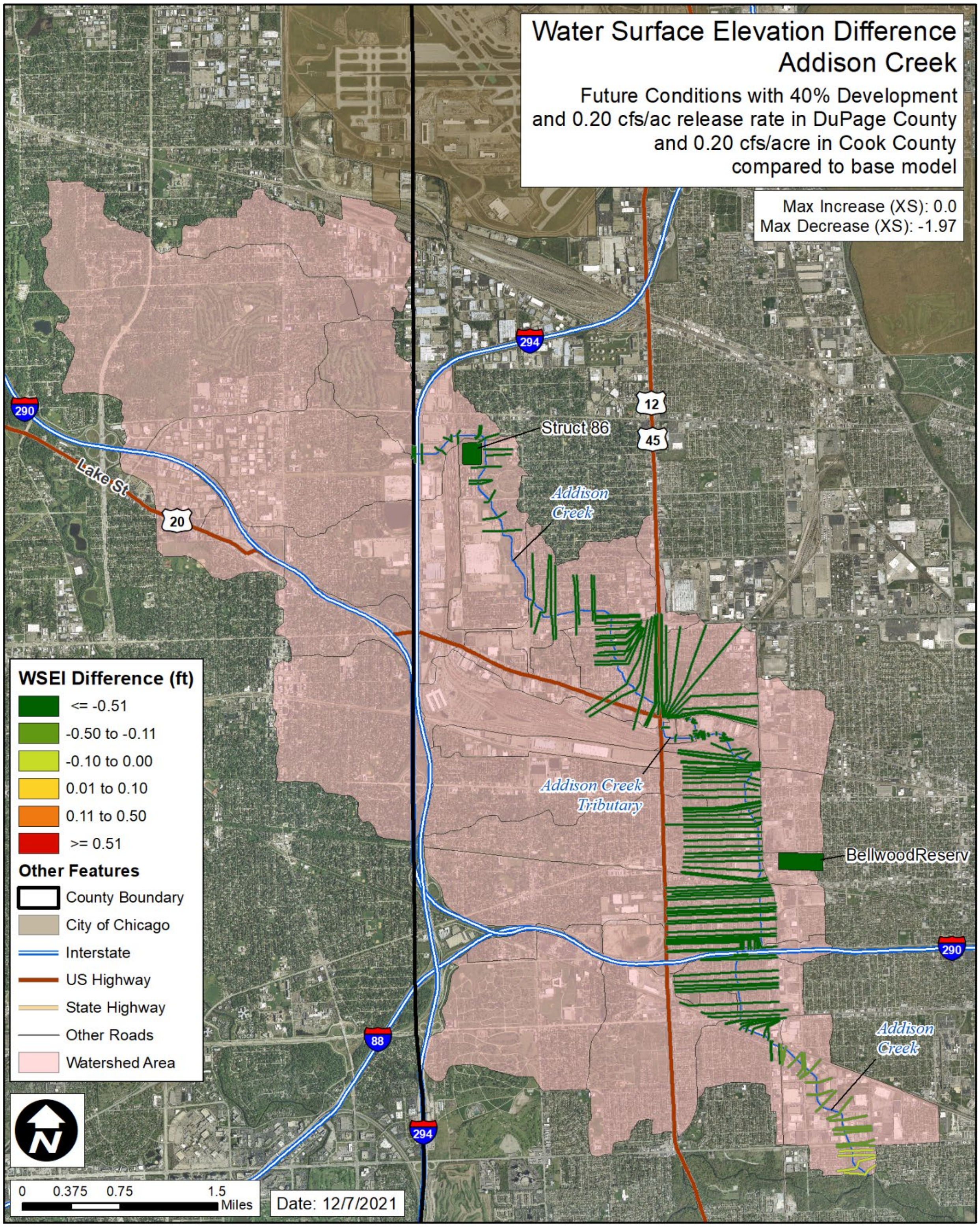


Figure 26. Water surface elevation differences by cross section between the base model and future conditions model using a 0.20 cfs/ac release rate within WMO jurisdiction and 0.20 cfs/ac within DuPage County

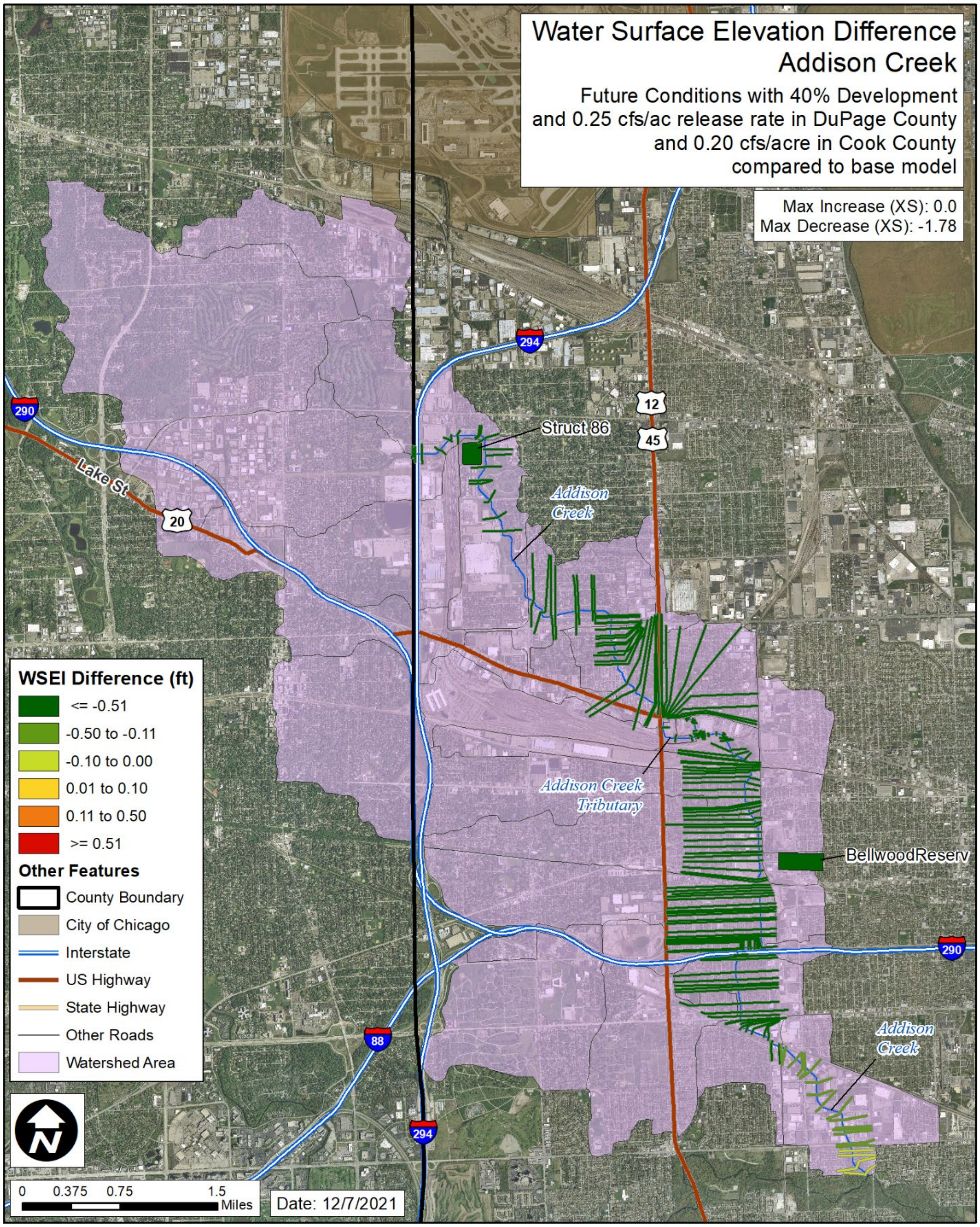


Figure 27. Water surface elevation differences by cross section between the base model and future conditions model using a 0.20 cfs/ac release rate within WMO jurisdiction and 0.25 cfs/ac within DuPage County

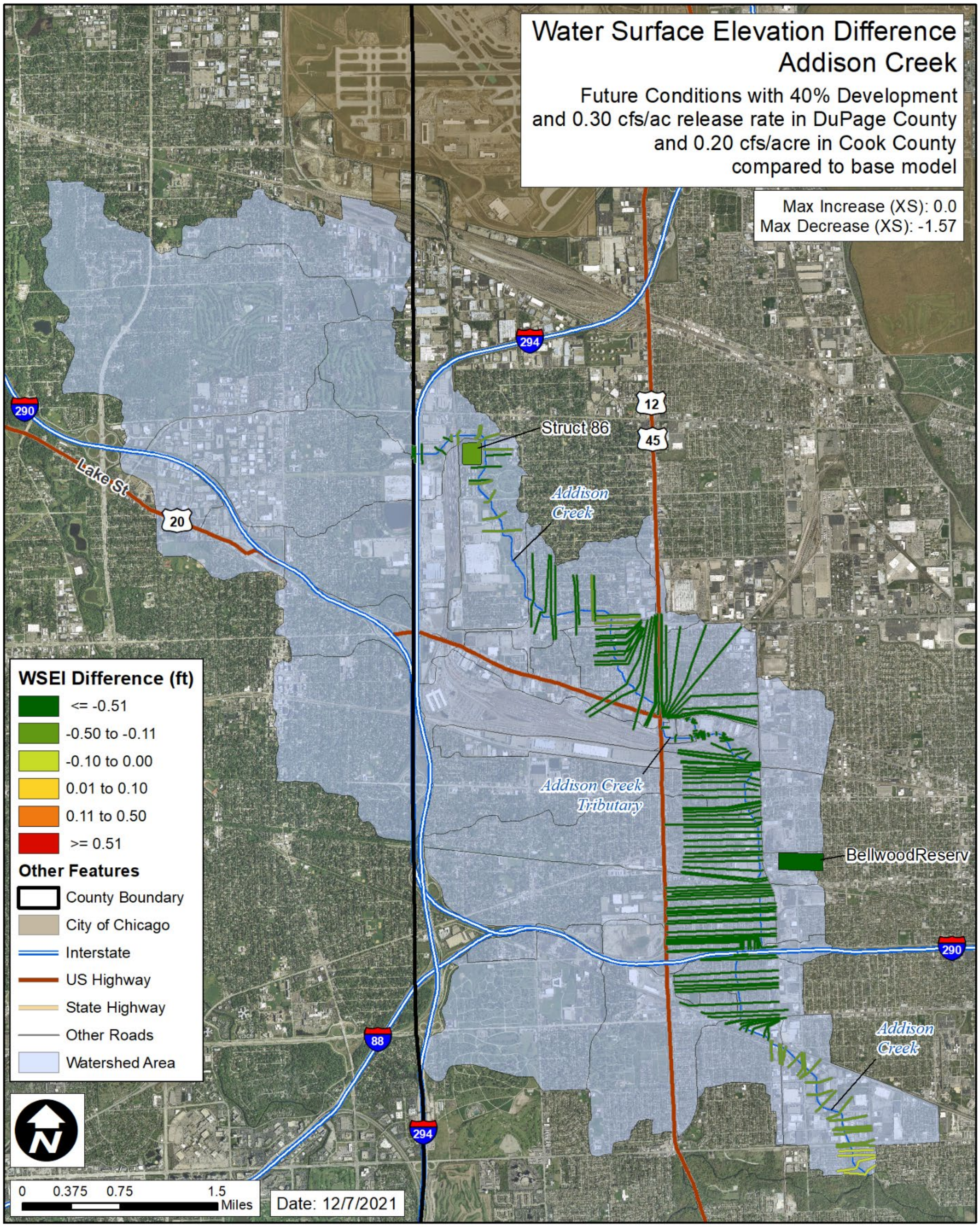


Figure 28. Water surface elevation differences by cross section between the base model and future conditions model using a 0.20 cfs/ac release rate within WMO jurisdiction and 0.30 cfs/ac within DuPage County

The Lower Salt Creek did demonstrate sensitivity to the release rate modeled within DuPage County. The Lower Salt Creek includes several tributary subwatersheds; therefore, a number of release rates were used during the analysis, but only the release rate within DuPage County was varied to measure sensitivity. The release rate within the Upper Salt Creek watershed was fixed at 0.20 cfs/ac as prescribed in the WMO Appendix B. Portions of the Addison Creek subwatershed in Cook County also used the prescribed 0.20 cfs/ac, but the release rate within the DuPage County portion of the watershed was set consistent to the DuPage County release rate being modeled within the Lower Salt Creek model. The results of the 0.15 cfs/ac release rate scenario are shown in Table 13 and Figure 29. The results indicate that increases in peak water surface elevation due to development throughout the watershed are mitigated under this management scenario. During Phase II, mitigation of rises due to future development was defined as changes in peak water surface elevation of less than 0.1 feet at hydraulic cross sections or less than 0.5 feet within a reservoir between the base model conditions and future model conditions for a particular release rate. The majority of the watershed would be expected to see a minor comparative decrease in peak water surface elevations under these practices.

Table 13. Evaluation Criteria/Metrics for Collar County Release-Rate Impact Analysis in the Salt Creek Subwatershed

	Collar County Release Rate			
Criteria	*0.15 (cfs/ac)*	*0.20 (cfs/ac)*	*0.25 (cfs/ac)*	*0.30 (cfs/ac)*
Stream length with increase in peak water surface elevation (WSEL) > 0.1' (ft)	--	970	23,384	29,170
Stream length with increase in peak WSEL> 0.1' (%)	0.0%	1.6%	38.4%	47.9%
Maximum XS WSEL increase	0.03'	0.12'	0.39'	0.45'
Maximum reservoir WSEL increase	--	--	--	0.10'
Reservoirs with increases > 0.5'	--	--	--	--

The results of the 0.20 cfs/ac release rate scenario are shown in Table 13 and Figure 30. Similar to the 0.15 cfs/ac release rate, under the 0.20 cfs/ac release rate, most cross sections show comparable water surface elevations between the base and future condition models due to future development. A single cross section returned an increase in peak water surface elevation greater than 0.1 feet at 0.12 feet. The increase at this single cross section accounted for and increased water surface elevation at 1.6% of the stream length studied. During Phase II, such isolated and localized increases were not considered significant and as such did not indicate any issues in the release rates' ability to mitigate increases in peak water surface elevation due to future development within the watershed. These changes fell well within the range of hydraulic stability sensitivity across the DWP models as a whole. Instances like this typically indicated a localized sensitivity that could be addressed through additional modeling or minor mitigation. The goal of the comparative analysis was to identify significant shifts in the effectiveness of a mitigation strategy. No reservoirs were identified with more than 0.5 feet higher water surface elevation due to future development.

At the 0.25 cfs/acre release rate within DuPage County, the ability to mitigate future increases in peak water surface elevation due to future development diverges rapidly from more conservative release rates previously discussed as can be seen in Table 13. More than 38% of cross sections would be expected to experience increased peak water surface elevations. The largest increase would be nearly 0.4 feet and would violate the selection criteria established during Phase II. As seen in Figure 31, these increases in peak water surface elevation under this management scenario are found primarily along the main stem of Lower Salt Creek from the DuPage-Cook County line to upstream of Manheim Road.

The inability of the modeled management practice to mitigate future increases in peak water surface elevation first identified at the 0.25 cfs/ac release rate becomes even more pronounced at 0.30 cfs/ac. Nearly half of the stream length (~48%) within the study area would be expected to experience increased peak flood elevations due to future development with the largest increase of 0.45 feet as shown in Table 13. The expected location of increased water surface elevation follows a similar pattern as in 0.25 cfs/ac with the increases from the DuPage-Cook County line to near Manheim Road, but with additional increases extending downstream to the Des Plaines River as can be seen in Figure 32. The increases downstream of Manheim Road include both significant increases greater than 0.1 feet as well as locations with elevations closer to those in the base model.

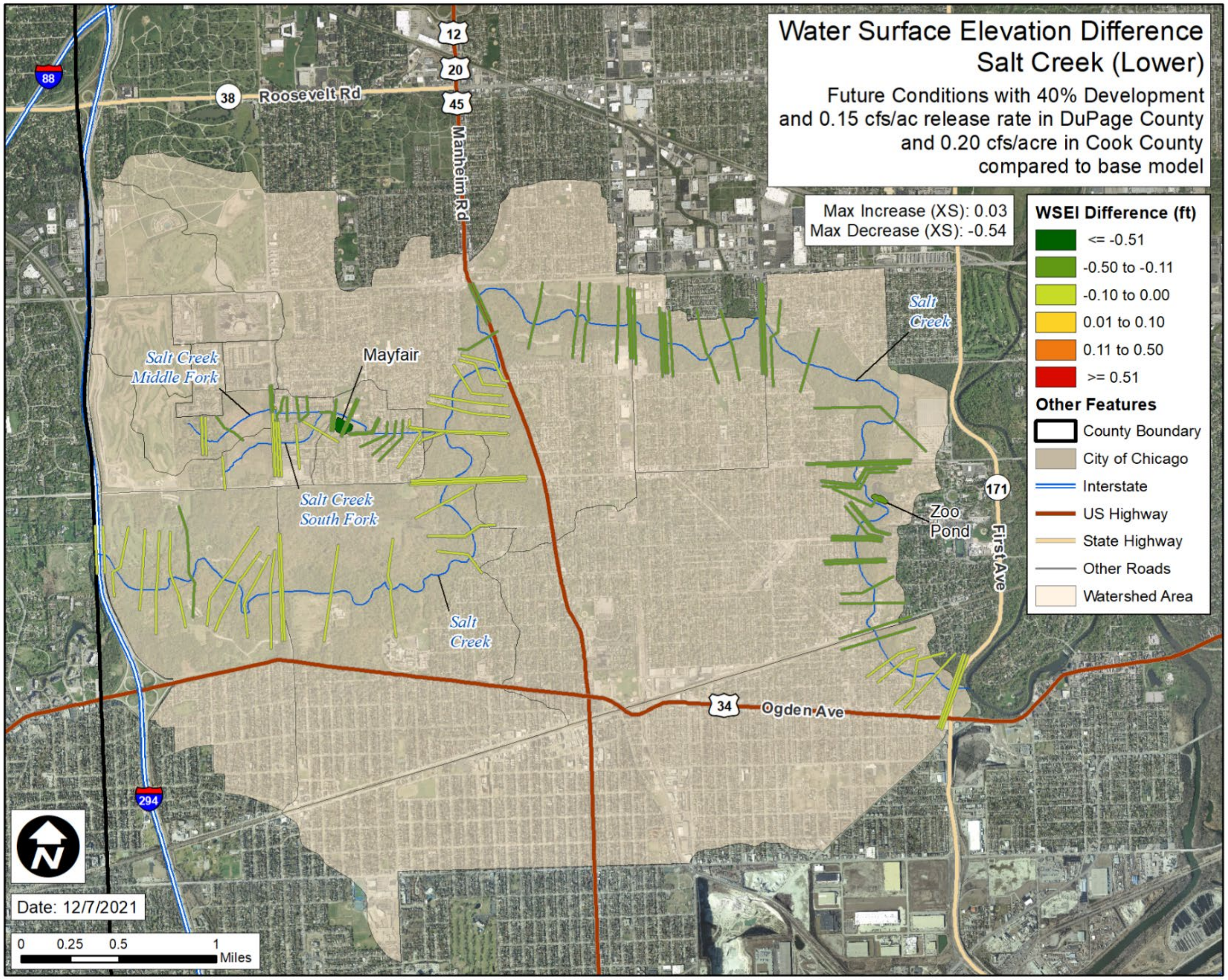


Figure 29. Water surface elevation differences by cross section between the base model and future conditions model using a 0.20 cfs/ac release rate within WMO jurisdiction and 0.15 cfs/ac within DuPage County

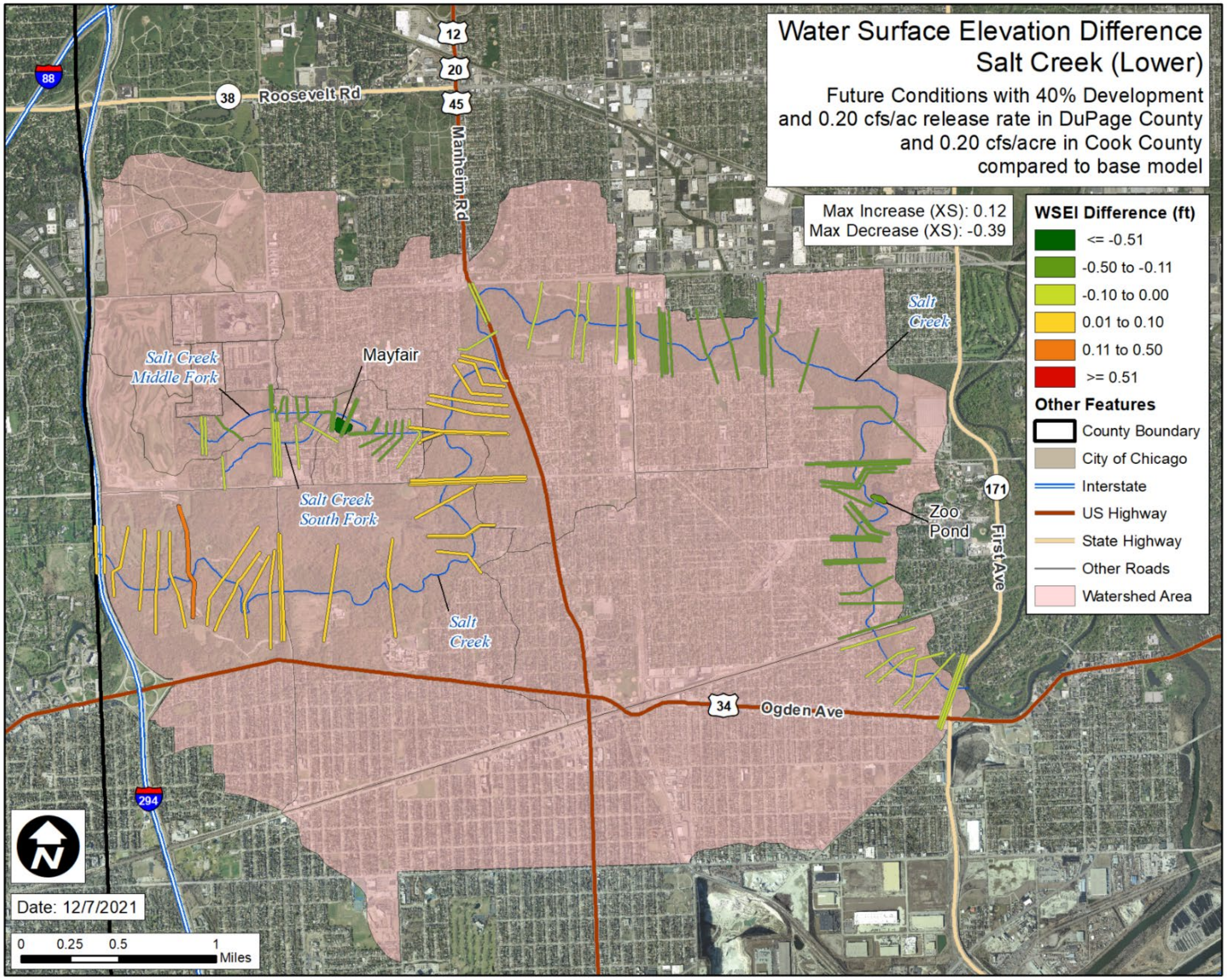


Figure 30. Water surface elevation differences by cross section between the base model and future conditions model using a 0.20 cfs/ac release rate within WMO jurisdiction and 0.20 cfs/ac within DuPage County

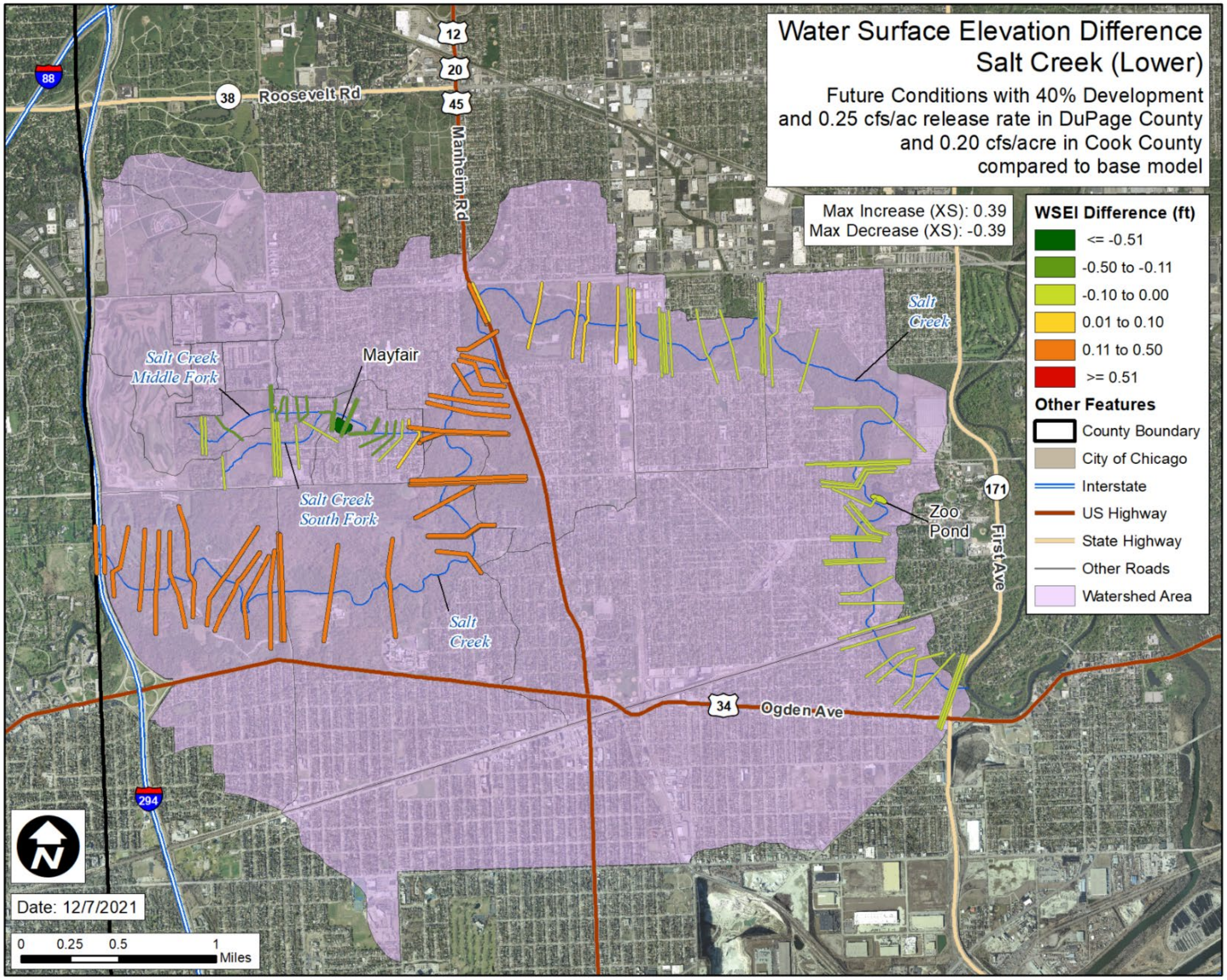


Figure 31. Water surface elevation differences by cross section between the base model and future conditions model using a 0.20 cfs/ac release rate within WMO jurisdiction and 0.25 cfs/ac within DuPage County

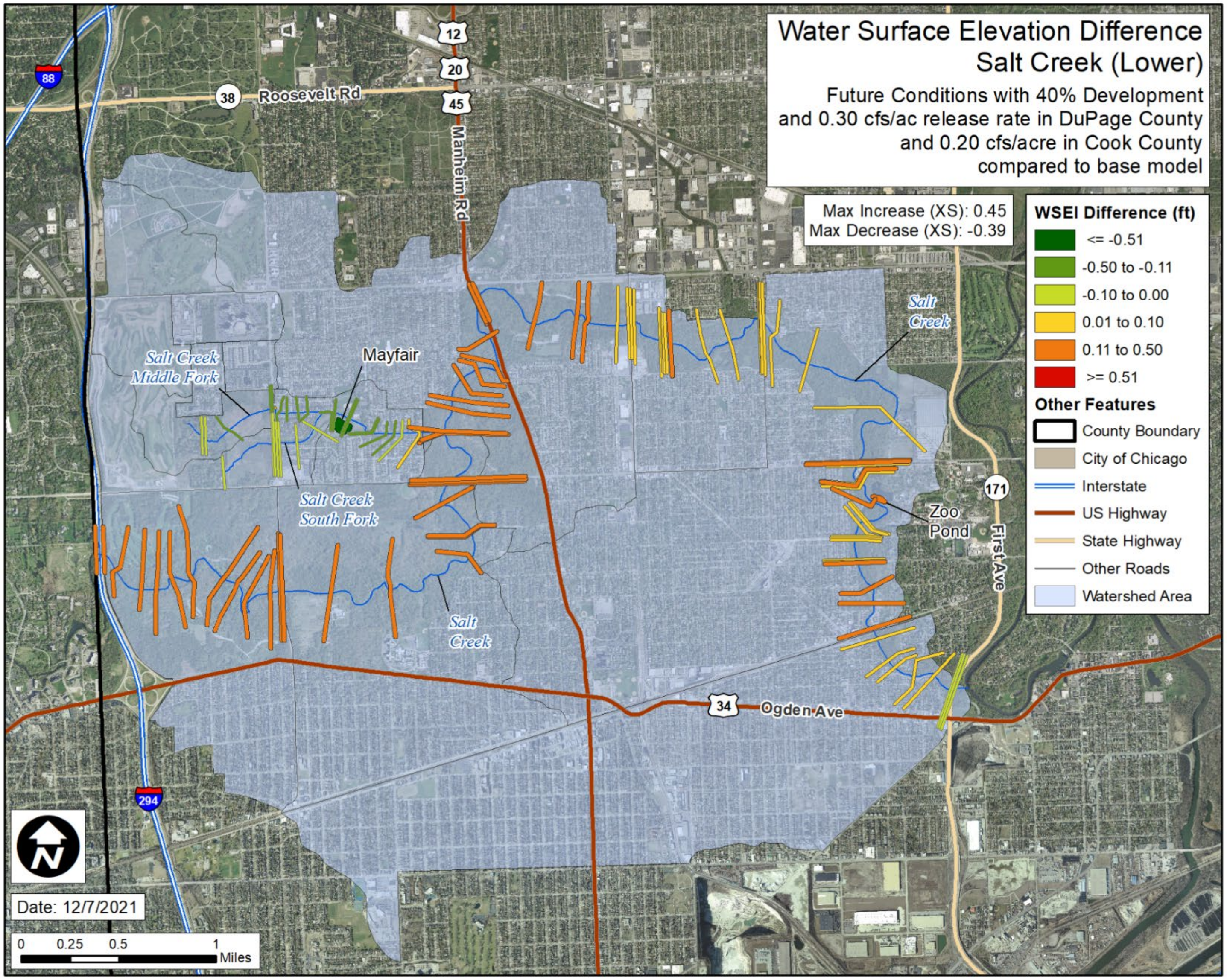


Figure 32. Water surface elevation differences by cross section between the base model and future conditions model using a 0.20 cfs/ac release rate within WMO jurisdiction and 0.30 cfs/ac within DuPage County

The Buffalo Creek subwatershed straddles the Lake-Cook County line with the creek generally flowing parallel to the border. This alignment results in Buffalo Creek flowing within Cook County before flowing into Lake County as it passes through the Buffalo Creek Reservoir before returning to Cook County before continuing southeast to the Lower Des Plaines River. The Buffalo Creek subwatershed modeling included two watershed management practices for the release rate, one fixed for all scenarios and one variable to evaluate sensitivity. Within Cook County, areas subject to the WMO Watershed Specific Release Rate were modeled using the prescribed release rate listed in the WMO Appendix B, namely 0.20 cfs/ac. Within areas tributary to Cook County but located in Lake County, Illinois, a separate watershed management practice was used. In Lake County, the release rate was varied from 0.15 cfs/ac to 0.30 cfs/ac to understand the impacts of extra jurisdictional changes in watershed management practices on peak water surface elevations within the District.

The results of the 0.15 cfs/acre release rate scenario are shown in Table 14 and Figure 33. The 0.15 cfs/ac scenario within Lake County with a 0.20 cfs/ac release rate was first modeled as part of Phase II during the evaluation of the Watershed-Specific Release Rates within Cook County, Illinois. As found during Phase II, the results indicate that increases in peak water surface elevations due to development throughout the watershed are mitigated under these management scenarios. During Phase II, the mitigation of rises due to future development was defined as changes in peak water surface elevation of less than 0.1 feet at hydraulic cross sections or less than 0.5 feet within a reservoir between the base model conditions and future model conditions for a particular release rate. The majority of the watershed would be expected to see minor comparative decreases in peak water surface elevations due to future development under these practices. No increased stages were identified within the Buffalo Creek reservoirs under this management scenario due to future development.

Table 14. Evaluation Criteria/Metrics for Collar County Release-Rate Impact Analysis in the Buffalo Creek Subwatershed

Criteria	*Collar county release rate*			
	0.15 (cfs/ac)	*0.20 (cfs/ac)*	*0.25 (cfs/ac)*	*0.30 (cfs/ac)*
Stream length with increase in peak water surface elevation (WSEL) > 0.1' (ft)	--	--	122	898
Stream length with increase in peak WSEL> 0.1' (%)	0.0%	0.0%	0.2%	1.3%
Maximum XS WSEL increase	0.01'	0.01'	0.16'	0.35'
Maximum reservoir WSEL increase	--	--	--	0.29'
Reservoirs with increases > 0.5'	--	--	--	--

The results of the 0.20 cfs/ac release rate scenario are shown in Table 14 and Figure 34. Similar to the 0.15 cfs/ac release rate, under the 0.20 cfs/ac release rate, most cross sections show comparable water surface elevations between the base and future condition models due to future development.

A similar result is observed for the 0.25 cfs/ac release rate as shown in Table 14 and Figure 35 with most cross sections showing comparable water surface elevations between the base and future condition models due to future development. Those cross sections that returned an increase in peak water surface elevation greater than 0.1 feet were 0.16 feet and isolated near a single restrictive structure on Buffalo Creek between the Buffalo Creek Reservoir and the confluence with the Lower Des Plaines River. The increases occurred at only 0.2% of the stream length studied. During Phase II, such isolated and localized increases were not considered significant and, as such, did not indicate any issues in the release rate's ability to mitigate increases in peak water surface elevation due to future development within the watershed as these changes fell well within the range of hydraulic stability sensitivity across the DWP models as a whole. No reservoirs were identified with an increase in peak water surface elevation due to future development.

At the 0.30 cfs/ac release rate (Figure 36), the ability to mitigate increases in future water surface elevation due to development is very near the selection criteria used during Phase II. Approximately 1.3% of stream length would be expected to experience increases, but unlike Lower Salt Creek, this difference occurs at a number of hydraulic cross sections and is not isolated to an individual location. The peak difference in water surface elevation is 0.35 feet and includes changes in the peak water surface elevation near the Buffalo Creek Reservoir. Although the reservoir itself is not expected to experience an increase of greater than the 0.5-foot threshold, the reservoir elevation controls the cross section peak water surface elevation for some distance upstream of the reservoir. The ability of a 0.30 cfs/ac release rate in Lake County and a 0.20 cfs/ac release rate within Cook County to mitigate future increases in peak water surface elevations due to development is thus questionable.

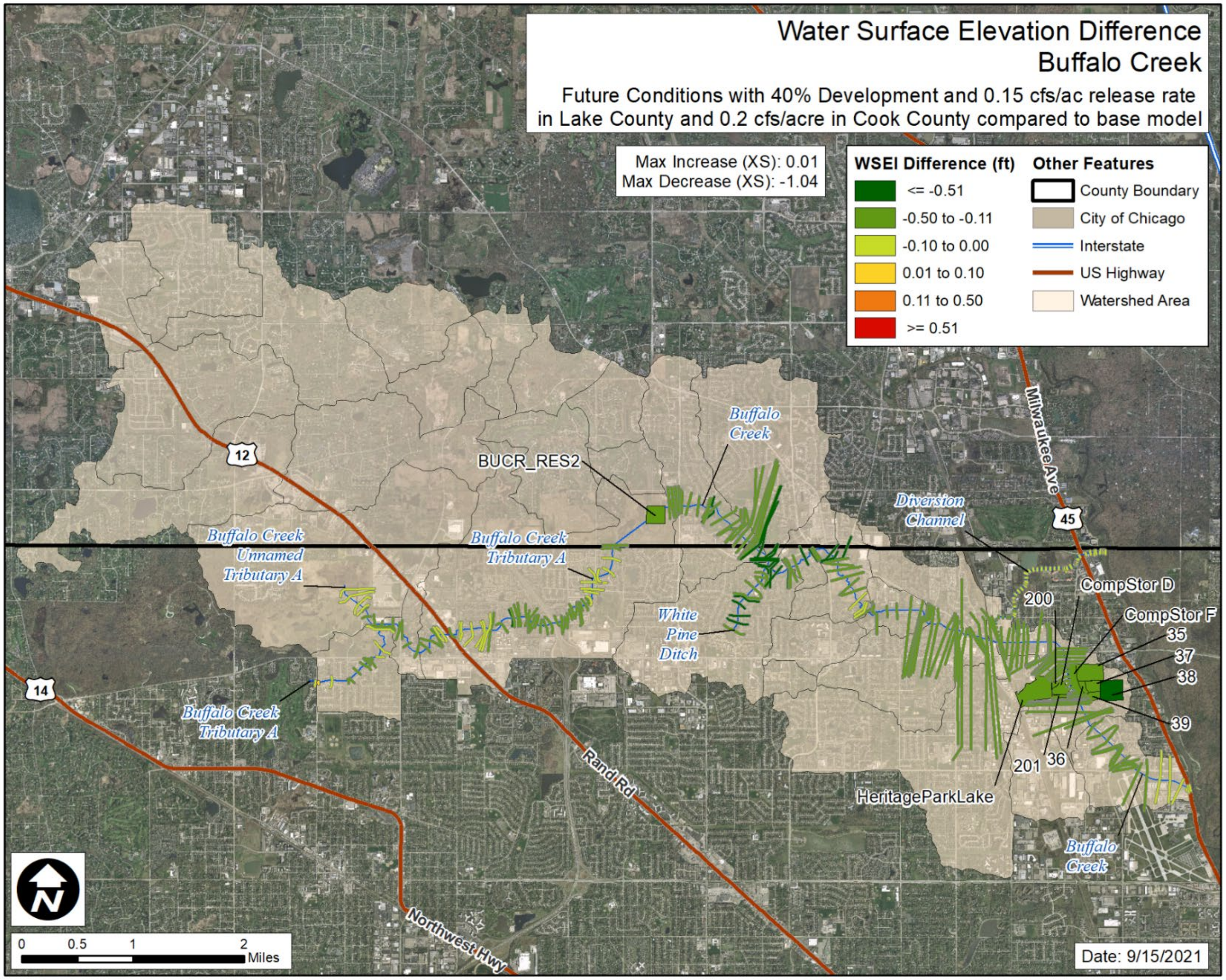


Figure 33. Water surface elevation differences by cross section between the base model and future conditions model using a 0.20 cfs/ac release rate within WMO jurisdiction and 0.15 cfs/ac within Lake County

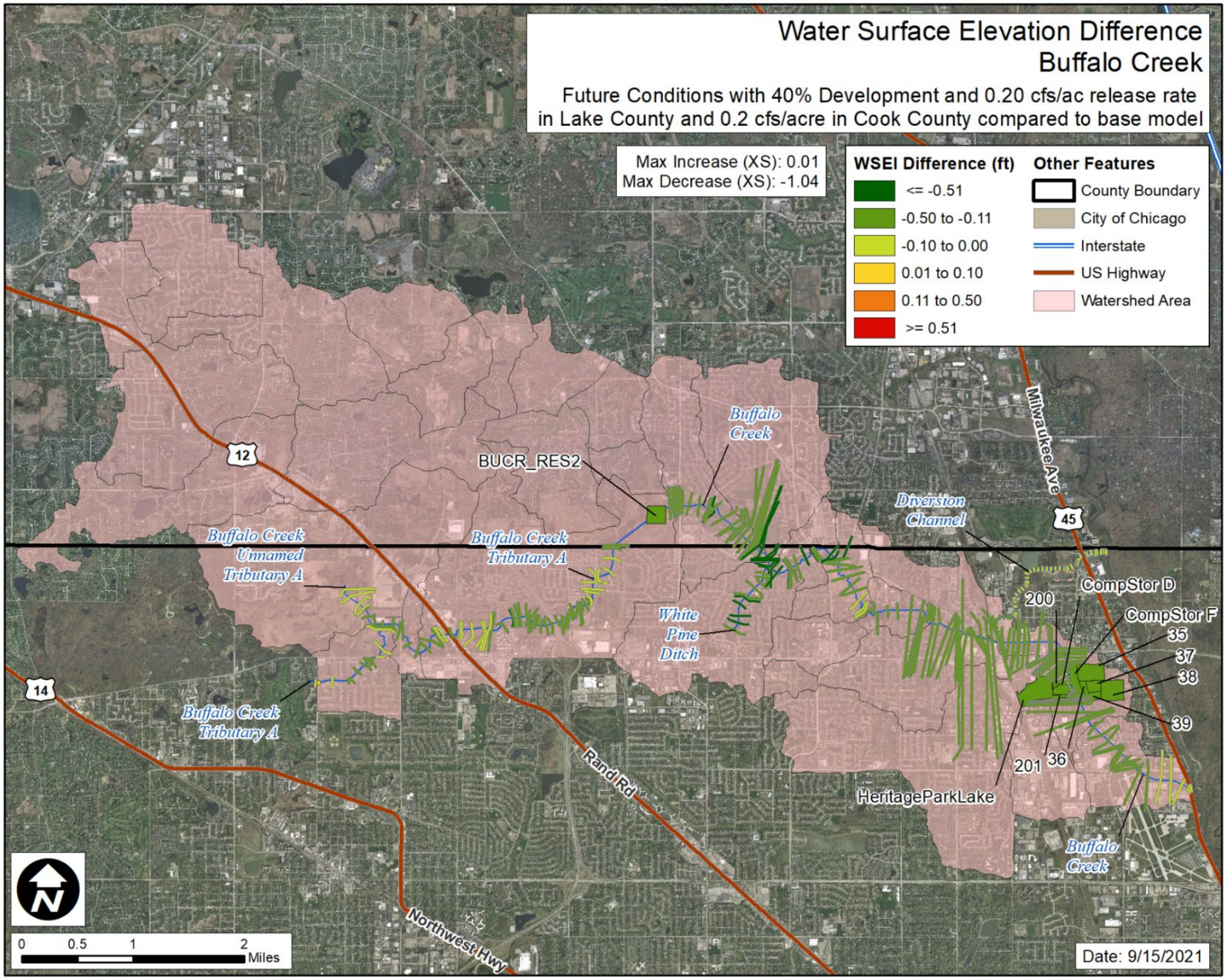


Figure 34. Water surface elevation differences by cross section between the base model and future conditions model using a 0.20 cfs/ac release rate within WMO jurisdiction and 0.20 cfs/ac within Lake County

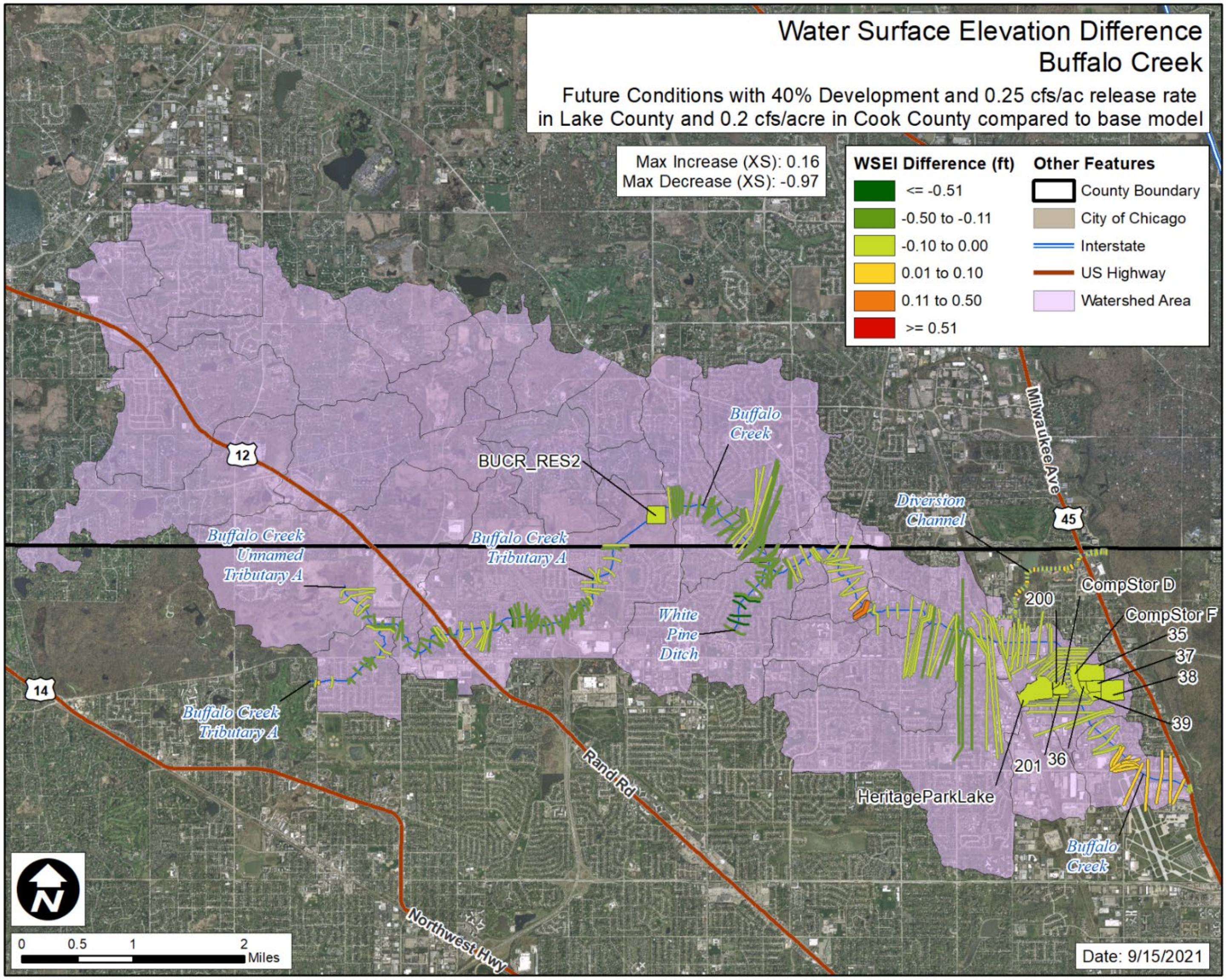


Figure 35. Water surface elevation differences by cross section between the base model and future conditions model using a 0.20 cfs/ac release rate within WMO jurisdiction and 0.25 cfs/ac within Lake County

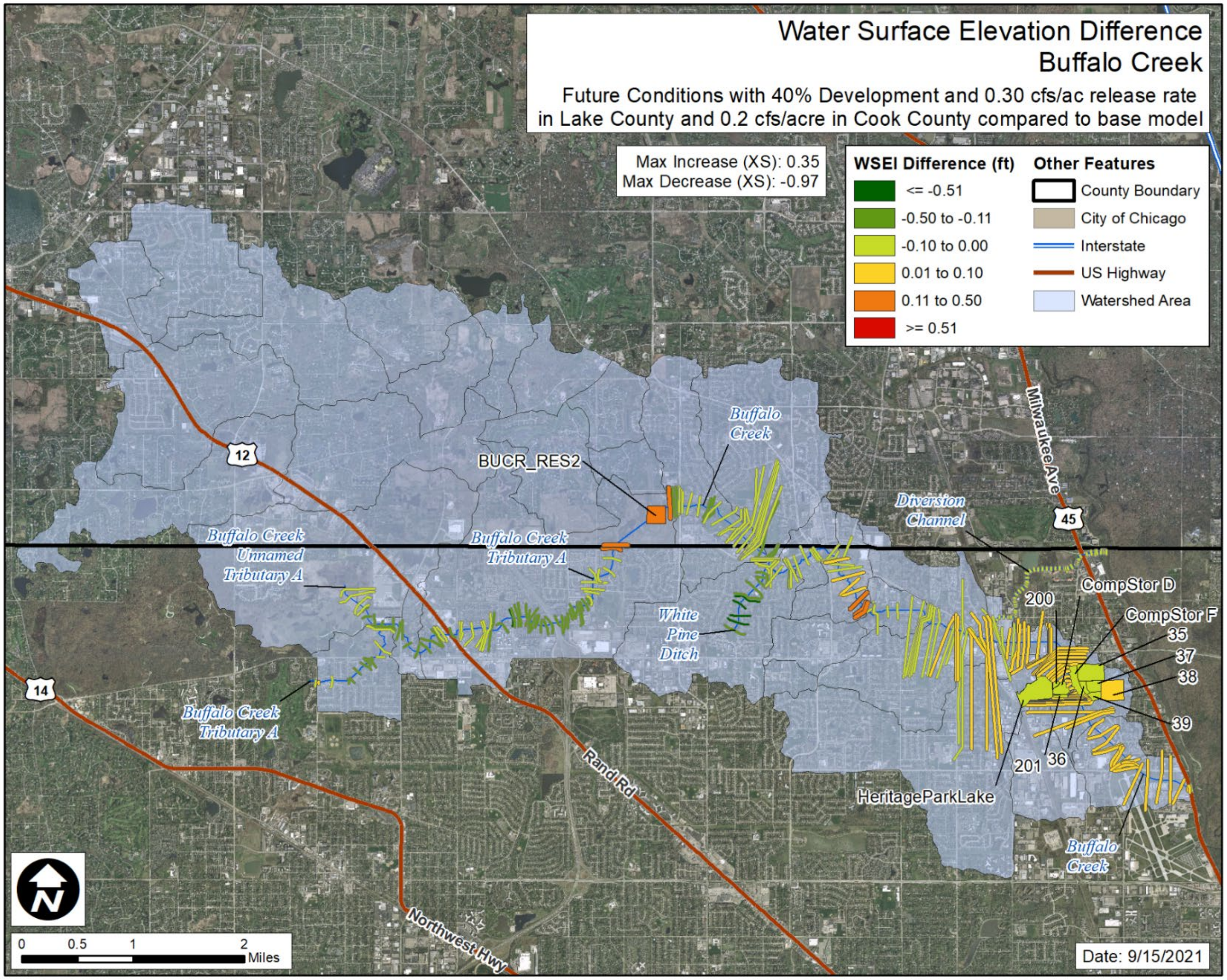


Figure 36. Water surface elevation differences by cross section between the base model and future conditions model using a 0.20 cfs/ac release rate within WMO jurisdiction and 0.30 cfs/ac within Lake County

2.4 Conclusions

The Illinois State Water Survey is not aware of any pending updates to the watershed management policies in counties that drain stormwater to streams within the District, yet the results of this analysis can provide policymakers and stakeholders with an understanding of the potential response to future development under alternative stormwater release rates and how the sensitivity of those changes can vary by both watershed and subwatershed.

Certain watersheds or subwatersheds demonstrate a high level of resilience to increases in peak water surface elevation due to future development under changes in the release rate in tributary areas outside the jurisdiction of the WMO. In the Addison Creek subwatershed, no increases in peak water surface elevation due to future development were identified regardless of the release rates selected for DuPage County. Although no increases were identified under the various scenarios modeled, this does not mean there would be no impact to peak water surface elevations within the Addison Creek subwatershed due to modifications to the DuPage County release rate. Rather, during the Phase II portion of the analysis, the appropriateness of a watershed-specific release rate, such as those now included in the WMO Appendix B, was based on whether increases to the peak water surface elevation were mitigated at cross sections and storage areas across the management area. However, certain areas were expected to experience *improvements or benefits*, namely relative decreases in peak water surface elevation due to future development under a particular release rate selection. As the release rate is increased within DuPage County, the magnitude of these anticipated relative decreases in peak water surface elevation due to future development would become smaller, though not so dramatic as to cause water surface elevation rises in Cook County. Examples of this occurrence are seen in the maximum decrease in peak water surface elevation due to future development of -2.13 feet at 0.15 cfs/ac, -1.97 feet at 0.20 cfs/ac, -1.78 feet at 0.25 cfs/ac, and a maximum decrease of -1.57 feet at 0.30 cfs/ac in DuPage County. This is highlighted not to suggest that the release rate would not mitigate future increases in peak water surface elevation, as these reduced benefits were not considered during Phase II, but rather to note the difference between no increases in peak water surface elevation due to future development and no change in peak water surface elevations due to a change in management strategy.

Addison Creek showed similar insensitivity to the selection of release rate chosen within Cook County during the Phase II analysis. In those scenarios, DuPage County release rates were kept fixed, and Cook County release rates varied from 0.15 to 0.30 cfs/ac. None of the modeled scenarios during Phase II produced increases in peak water surface elevations due to future development. Much of this insensitivity can be explained by the average base condition runoff rate, which was 0.45 cfs/ac. This means that any of the studied release rates would be expected to lower the watershed runoff rate following development under the WMO. However, as it is a tributary to Lower Salt Creek, the release rate within the Addison Creek subwatershed also has a direct impact on the water surface elevations within Lower Salt Creek as discussed below.

The Buffalo Creek subwatershed also shows a level of resilience to changes in release rates within Lake County. At the highest release rate studied, 0.30 cfs/ac within Lake County, there would be some question as to whether this release rate coupled with the release rate currently prescribed by the WMO Appendix B within Cook County would be sufficient to mitigate future increases in peak water surface elevation due to future development at every location. Yet the vast majority of cross sections in the model and all reservoirs would not be expected to be negatively impacted by future development. Release rates less than 0.30 cfs/ac within Lake County, in conjunction with the watershed-specific release rate prescribed by the WMO, are expected to be effective at mitigating future increases in peak water surface elevation due to future development.

As with the Addison Creek subwatershed, much of the sensitivity to the selected release rate in the Buffalo Creek subwatershed appears related to the average base condition runoff rate for the watershed. The average base condition runoff rate was 0.27 cfs/ac, meaning that release rates lower than this threshold would be likely to lower peak water surface elevations based on the general relationship observed during Phase II, where release rates lower than the average base condition runoff rate often resulted in lower modeled water surface elevations. As the results indicated, release rates in Lake County lower than this value (0.15, 0.20, and 0.25 cfs/ac) were effective mitigation strategies, while 0.30 cfs/ac would be expected to perform very near the limit of acceptability. A similar pattern was seen during Phase II when the same lower release rates were determined to be effective mitigation strategies for the particular subwatershed, while the performance declined for the 0.30 cfs/ac release rate.

The Lower Salt Creek subwatershed was shown to be highly sensitive to the selection of release rate within DuPage County. This sensitivity was related to the difference between the currently prescribed release rate and the release rates studied as part of this analysis. As was described in Phase II of this study, subwatersheds with drainage areas in surrounding counties that use a restrictive release rate often required a less restrictive release rate within Cook County than would have been necessary had the entire watershed been subject to a common release rate. During Phase II, Lower Salt Creek was not sensitive to the selection of release rate within Cook County. The reasons for this were likely twofold. First, because much of the drainage area was located upstream of Cook County, the amount of drainage area held at a constant release rate lessened the sensitivity to changes within the Cook County portion of the watershed, particularly since the prescribed DuPage County release rate was lower than those considered within Cook County. Second, the average base condition runoff rates in some of the tributary subwatersheds were quite high (0.45 cfs/ac in Addison Creek, 0.35 cfs/ac in Upper Salt Creek Arlington Heights Branch subwatershed, and 0.36 cfs/ac release rate in Upper Salt Creek main stem subwatershed). None of the analyzed release rates considered within Lower Salt Creek resulted in increased peak water surface elevations due to future development. This was despite the general trends between the average base conditions peak runoff rate and the selected release rate, suggesting that Lower Salt Creek would have been sensitive to release rates at or above 0.25 cfs/ac, which corresponds to its average base condition peak runoff rate within Cook County.

Minor increases in the DuPage County release rate to 0.15 or 0.20 cfs/ac would not be expected to cause increases in peak water surface elevation due to future development. Increasing the release rate to 0.25 or 0.30 cfs/ac while retaining the watershed-specific release rate in Cook County as prescribed in the WMO Appendix B, namely 0.20 cfs/ac, is not expected to be an effective strategy at mitigating increases in peak water surface elevation due to future development along the Lower Salt Creek main stem with 38% and 48% of the total reach length expected to experience increasing elevations, respectively.

The North Branch Chicago River watershed was also found to be highly sensitive to the selection of release rate within Lake County. This result was expected given that during the Phase II analysis, a reduction of peak discharge at the county boundary based on the Lake County release rate regulations was found to be the key factor impacting the acceptable release rate within Cook County. Without such regulation within Lake County, a 0.30 cfs/ac release rate within Cook County would not have been expected to mitigate increases in peak water surface elevation due to future development. This expectation during Phase II was supported by this additional analysis. For scenarios in which Lake County release rates were held at 0.15 or 0.20 cfs/ac, the 0.30 cfs/ac release rate prescribed by WMO Appendix B is expected to mitigate increases in peak water surface elevation. However, at Lake County release rates of 0.25 and 0.30 cfs/ac, the ability of the Cook County release rate to mitigate increases is substantially reduced with 31% and 50% of the studied stream length expected to experience increases greater than the allowable limit of a 0.1-foot rise.

The ability of the release rate to mitigate future increases is again correlated to the average base condition runoff rate within the North Branch Chicago River watershed. West Fork, Middle Fork, and North Branch upstream subwatersheds all had base condition runoff rates greater than 0.30 cfs/ac (0.41, 0.32, and 0.32 cfs/ac, respectively). In each of these subwatersheds, the modeled release rates would generally provide discharges lower than the natural condition. For the Skokie River, however, the base condition runoff rate was 0.27, with some subbasins as low as 0.12 cfs/ac. Areas with base condition runoff rates less than the release rate being modeled often required more restrictive release rates to mitigate future increases in peak water surface elevation due to future development. This was not found to be the case during Phase II because of the water surface elevation benefits afforded by the restrictive Lake County release rate, but accounts for much of the sensitivity observed during this study. Skokie River, with the lowest average base condition runoff, but without the benefit of more restrictive Lake County release rates, was the most sensitive to a change in management strategy.

The impacts of changes in watershed management strategies outside of the collar counties and into the State of Wisconsin were not considered as part of this analysis but were considered as part of Phase II. Based on the drainage areas upstream of Cook County, watershed management strategies in the State of Wisconsin would be expected to have the largest impact on water surface elevations in Cook County along the Des Plaines River. Although future development and its associated impervious areas within the Des Plaines River watershed upstream of Cook County are expected to increase future water surface elevations along the Des Plaines River, watershed-specific release rates applicable only in Cook County under the WMO were not identified such that these future increases would be mitigated. The total amount of drainage area upstream of Cook County, the low percentage of drainage area to the Des Plaines River originating in Cook County, and the long critical duration (10-day critical duration) of the Des Plaines River in Cook County all contribute to the ineffectiveness of a 24-hour storm watershed-specific release rate in mitigating future water surface elevation increases due to future development. Additional information about this topic can be found in the Phase II report.

As with the results of the Phase II analysis, it is important to note that this study was designed to analyze the impacts of various watershed management practices on peak water surface elevation due to future development alone. This study used a comparative approach to isolate the impacts of future development and the response of a watershed to future development such that the impacts are expected to follow similar spatial and temporal patterns even under updated design rainfalls as found during the Phase II sensitivity analysis. This study does not, however, consider the impacts of changes in extreme precipitation over time in an absolute sense. As such, future decreases in peak water surface elevation should be understood to be relative to a common extreme rainfall regime. The sensitivity of peak water surface elevations to changes in design rainfalls, such as between Illinois State Water Survey Bulletin 70 and Illinois State Water Survey Bulletin 75, is not considered in this analysis. It is likely that while a particular watershed management strategy may effectively mitigate future increases in peak water surface elevation due to future development under a particular rainfall regime, these practices are insufficient to mitigate both increases due to future development and precipitation due to changing rainfall patterns.

2.5 Summary

The selection of a release rate in collar counties can influence whether the watershed-specific release rates prescribed in the WMO Appendix B will continue to mitigate future increases in peak water surface elevation due to future development, but not all watersheds were found to be sensitive to such changes. Watersheds with a substantial proportion of drainage area falling outside the WMO jurisdiction and with low average base condition runoff rates were the most sensitive, and those with only small portions of the drainage area or high average base condition runoff rates were less sensitive.

The results of this study are predicated on each individual scenario's modeling assumptions, namely that the prescribed release rate is required *prior* to the future development in a watershed. As during Phase II, spatiotemporal trends in future development over short time periods were not considered, as the uncertainty in the timing of the redevelopment of a particular parcel of land is extremely high. Instead, the analysis focused on long-term development trends over broader spatial scales for which confidence is much higher. Therefore, this analysis considers only a change in collar county release rates assumed to be effective as of current conditions modeling. The impacts of a collar county changing a release rate or other watershed management practice at some point in the future after some amount of development has occurred in both Cook County and in the collar county will inherently influence the sensitivity to a particular change in management strategy.

It is recommended that the relevant watershed management agencies coordinate any changes in their watershed management requirements for multijurisdictional streams in the future. Early communication will provide managers with the most flexibility in responding to changing watershed dynamics. Watershed managers could also consider whether uncertainty in management practices outside of their jurisdiction should influence management practices within their jurisdiction.

Chapter 3. Stream Channel Dynamics in Urban Settings: A Literature Review [WMO Article 208.4]

3.1 Introduction

Urban areas cover only about 3% of land around the world (van Vliet et al., 2017), yet more than half of the world's population lives in these areas (United Nations, 2018). In Illinois, the Chicago metropolitan area accounts for only 16% of the total land of the state but contains 75% of the state's population. The process of urbanization, by necessity, dramatically transforms land cover through the establishment of a built environment required to sustain a high population density. This built environment includes buildings, roads, sidewalks, and parking lots. The replacement of native prairie, forest, or agricultural land with built infrastructure transforms hydrological processes associated with drainage of the landscape during wet-weather events. To accommodate transformed rates of stormwater runoff, artificial drainage systems, in particular storm sewer networks, are installed to help facilitate efficient drainage and prevent localized flooding. The result of changes in land-cover conditions to include impervious surfaces and of the implementation of stormwater drainage systems is increased rates of runoff to receiving waters, such as streams and rivers.

Over the past several decades, recognition of enhanced stream flooding caused by increased rates of runoff has led to efforts to increase stormwater retention and detention. In the District, the Cook County Watershed Management Ordinance (WMO) addresses concerns about the enhanced risk of flooding and the need to mitigate adverse impacts of urban flooding. The WMO seeks to reduce the potential for loss of property due to flood damage; manage and mitigate the effects of urbanization on stormwater drainage throughout Cook County; protect existing and new development by minimizing the increase of stormwater runoff volume beyond that experienced under existing conditions and reduce peak stormwater flows; and reduce or mitigate the environmentally detrimental effects of existing and future runoff to improve or maintain water quality. A key component of the WMO is the establishment of watershed-specific release rates to achieve these goals. Under the direction of the Chicago Metropolitan Water Reclamation District (MWRD), the Illinois State Water Survey has conducted analysis to help identify appropriate watershed-specific release rates. As this work has progressed, MWRD has become interested in the potential effects of stormwater-runoff mitigation on stream erosion. A WMO amendment adopted by the MWRD Board of Commissioners on May 16, 2019, directs, under Section 208.4, a study of the "impact of volume control and watershed specific release rates on stream erosion and related water quality effects such as turbidity and sedimentation." A first step in this study involves a comprehensive review of relevant literature on erosion of urban streams. More generally, the problem of stream erosion is a problem in fluvial geomorphology–the science of rivers as agents of erosional and depositional change of the Earth's surface. The purpose of this chapter is to provide an overview of state-of-the art science on urban stream channel dynamics, the role of erosion and deposition in these dynamics, and the connection of these dynamics to basic theory within the field of fluvial geomorphology.

The organization of the chapter centers around the core issue of concern within the WMO, i.e., the influence of urbanization on rates of stormwater runoff. The connection of this issue, a topic in urban hydrology (Section 3.2), to the problem of stream erosion occurs through the intermediary topics of sediment delivery (Section 3.3) and stream hydraulics (Section 3.4), which determine how land-use change influences the supply of sediment to urban streams and how hydrological changes govern the forces and power of flowing water in urban rivers. Ultimately, changes in hydraulics are related to how effectively urban streams can transport sediment. Erosion and deposition of stream channels is the result of sediment transport. These two processes are responsible for changes in the form of urban stream channels (Section 3.5). Urban channels will undergo morphological adjustment in response to urbanization, and, presumably, adjust to the urbanized state of the landscape over time.

3.2 Urban Hydrology

3.2.1 Underlying runoff processes

Morphological change of urban streams by erosion or deposition is typically related to changes in the hydrology of these streams caused by change in land cover associated with urbanization. Thus, to understand this morphological change, it is necessary to consider how urbanization changes stream hydrology. Most investigations of the effects of urbanization on stream hydrology have focused on the urbanized phase rather than the construction phase of urbanization.

During a storm event, the process of runoff in an undeveloped watershed depends on the intensity of precipitation and the infiltration characteristics of the soil that are determined by the hydraulic conductivity and antecedent moisture condition of the soil (Rhoads, 2020). Portions of the precipitation can thus reach streams through surface or subsurface runoff. Surface runoff can occur as infiltration excess overland flow (when rainfall intensity is greater than the infiltration capacity of unsaturated soil) and saturation excess overland flow (where precipitation falls onto saturated ground and is converted to runoff). Subsurface flow occurs when precipitation infiltrates into the soil. This type of flow can occur as throughflow (water moving laterally and downslope within soil particles) and groundwater flow (through underlying parent materials).

The hydrologic response of an undeveloped watershed can be profoundly altered by the effects of urban development. Urban-induced changes in land use and land cover, particularly the covering of natural soils with impervious surfaces like roofs, sidewalks, roads, and parking lots, as well as the installation of storm sewer systems will generally increase greatly the rates of surface runoff, particularly infiltration excess overland flow (Booth, 1991; Booth and Bledsoe, 2009). The abundant surface runoff in urban areas produced by the lack of infiltration by impervious surfaces is generally referred to as stormwater runoff. Amounts of subsurface flow through soil is typically drastically reduced by urban development because impervious surfaces prevent infiltration of precipitation into the soil. As a result, soil and groundwater storage are reduced over time (Booth, 1991; Leopold, 1968).

The combination of reduced infiltration and rapid and efficient routing of runoff through storm sewers increases the volume, rate, and magnitude of flow in streams. Volumes increase because less evapotranspiration occurs when vegetated soil is replaced by impervious surfaces. Increased runoff volumes combined with increased rates of runoff lead to increases in peak discharges in stream channels, resulting in more frequent flood events. Changes in the magnitude and frequency of discharges have important implications for the morphological stability of erodible stream channels. Typical responses include channel enlargement, incision, and overall degradation of the aquatic ecosystem (Gurnell et al., 2007; Paul and Meyer, 2001; Poff et al., 2006). The water quality of the receiving streams also degrades as urban runoff encounters various dissolved and suspended contaminant sources (Ferreira et al., 2018; Tsihrintzis and Hamid, 1997).

Fundamental changes in the underlying runoff processes in a particular region are thus a root cause that accounts for adverse hydrologic, geomorphic, ecologic, and water quality impacts that collectively have become known as the urban stream syndrome (Booth et al., 2015). Watersheds dominated by subsurface runoff prior to development, whether natural or artificially enhanced, such as by drainage tiles, will exhibit a more dramatic change in hydrology than those dominated by surface flow because urbanization changes more radically the underlying runoff processes in the former compared to the latter.

3.2.2 Runoff volume, lag time, peak discharge, and impervious surface cover metrics

Once the runoff-generating processes are altered, the effects of urbanization are then reflected through significant changes in three inter-linked parameters of stream hydrology—total runoff volume, lag time, and peak discharge (Rhoads, 1995). These changes are recognized as some of the "direct" hydrological modifications caused by urbanization (Chin et al., 2022). Before presenting the impacts of urbanization on runoff volume, lag time, and peak discharge separately, it is vital to understand the representation of these hydrological properties in terms of the unit hydrograph and index parameter (i.e., imperviousness) through which changes in those properties are intimately associated.

A hydrograph (Figure 37) is the time distribution graph of discharge associated with a particular mass of precipitation produced by a storm event (Linsley et al., 1958). The area under the curve of a hydrograph corresponds to the total runoff volume (discharge, or volume per unit time, integrated over time). The time between the center of mass of precipitation and the peak of the hydrograph indicates the lag time, whereas the peak of the hydrograph denotes the peak rate of runoff or peak discharge.

To recognize and quantify the urban-induced changes in hydrological parameters, the metric "total impervious area" (TIA) has been frequently used in the literature, and is the fraction of the watershed occupied by impervious surfaces (Arnold and Gibbons, 1996). The imperviousness of a drainage basin is calculated through the TIA and is an index of urban development. This metric, although commonly related to hydrological response, does not distinguish compacted pervious surfaces that can act like impervious surfaces, nor does it account for impervious surfaces that do not contribute directly to stormwater runoff to streams (Booth and Jackson, 1997). An "effective impervious area" (EIA) (Miller, 1978), often interchangeably referred to as a "directly connected impervious area" (DCIA) (Sytsma et al., 2020), refers to impervious areas that have a direct hydraulic connection to stream channels (May et al., 1997; Sohn et al., 2020). The direct measurement of the EIA is difficult and is commonly estimated through correlations with the TIA (Booth and Bledsoe, 2009). Hydrologic response is often impacted above a certain threshold of imperviousness surface cover, i.e., a threshold value over which the hydrological responses in a catchment become substantial. Previous studies indicated a wide range of threshold values, typically varying between 3 and

20% (Booth and Jackson, 1997; Brun and Band, 2000; May et al., 1997; Oudin et al., 2018; Schueler, 1995; Yang et al., 2010; Yeo and Guldmann, 2006).

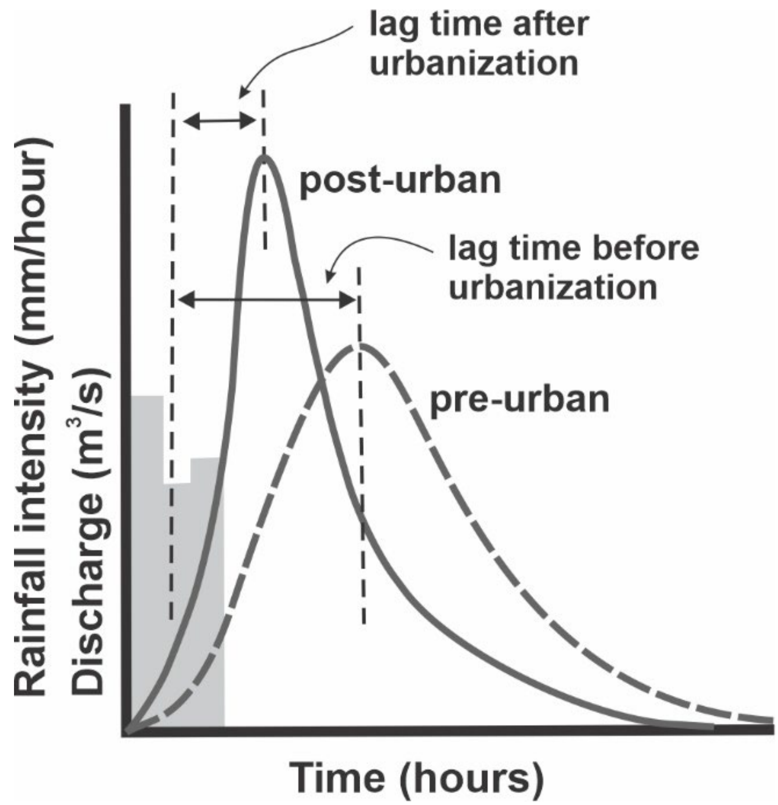


Figure 37. Hypothetical unit hydrograph conceptualizing the impact of urbanization through change in significant parameters. The dotted gray line and the solid gray line illustrate the shape of hydrograph for pre- and post-development conditions, respectively. The gray bars represent the temporal pattern of rainfall intensity.

3.2.2.1 Runoff volume

Stormwater runoff volume is mainly determined by the amount of precipitation, the infiltration capacity of soil, the percentage of the impervious cover, and the amount of evapotranspiration (Leopold, 1968; Schueler, 1995). Increases in the impervious surface cover, such as rooftops, streets, sidewalks, and parking lots, substantially increase the total volume of stormwater runoff by reducing the infiltration of precipitation into the soil (Harris and Rantz, 1964). Studies from urbanizing watersheds across the world have confirmed profound increases in runoff volume related to urbanization (Table 15). Short-duration intense storm events increase the runoff volume the greatest (Gregory, 1974); during storms greater than a 1- to 2-year recurrence interval, saturated catchments increasingly mimic the behavior of impervious surfaces regardless of urban development (Hollis, 1977). Depending on the hydrologic soil class, pervious surfaces of urban watersheds start contributing to runoff for infrequent events (e.g., 10-year storm with 6 hours duration) with at least 1.5 to 2 inches of rainfall over 6 hours (Miller and Viessman Jr., 1972).

Table 15. Examples of Changes in Runoff in the Urbanizing Areas of the World

Study Location	*Direction/Quantification of Change*	*Runoff Metrics*	*Reference*
Nassau County, Long Island, New York	123.1% increase	direct runoff	(Sawyer, 1961)
Santa Clara County, California	44% increase (from 1.18 in 1945 to 1.70 times in 1958)	ratio of outflow to inflow	(Harris and Rantz, 1964)
Sacramento, California	2.29 times greater	runoff volume	(James, 1965)
Austin, Texas	190%, 210%, and 240% increase (38th, 23rd street station, and the area between them, respectively)	runoff	(Espey Jr et al., 1965)
East Meadow Brook, Nassau County, Long Island, New York	270% increase (920 acre-feet/year in 1943 to 3,400 acre-feet/year in 1962)	average annual direct runoff	(Seaburn, 1969)
NE Exeter, Devon	0.9% increase	runoff	(Gregory, 1974)
Harlow, Essex	increase	water yield	(Hollis, 1977)
NE Exeter, Devon	2-4 times greater	storm runoff volume	(Walling, 1979)
Little Sugar Creek basin, Charlotte, North Carolina	2 times increase (400 mm in 1962 to 800 mm in 1995)	annual runoff	(Smith et al., 2002)
NASA's John F. Kennedy Space Center (KSC) and Indian River Lagoon watershed (IRL), Florida	49%, and 113% increase (for KSC and IRL, respectively)	average annual runoff	(Kim et al., 2002)
Atascadero Creek watershed, southern coast of California	350% increase	average runoff depth	(Beighley et al., 2003)
Yangtze River Delta region, China	11.3% increase	surface runoff	(Zhou et al., 2013)
South Carolina Sandhills, USA	More than an order of magnitude increase	runoff volume	(Hung et al., 2018)
central Missouri, USA	400% increase	runoff volume	(Wei et al., 2018)
Belo Horizonte, MG, Brazil	37.3% (2-year return period) & 20.1% increase (75-year return period)	runoff volume	(Rosa et al., 2020)

3.2.2.2 Lag time

Urbanization typically results in a decrease in lag time with the amount of decrease increasing as the degree of imperviousness increases (Figure 37, Table 16) (Leopold, 1991). The addition of stormwater drainage networks to the existing natural drainage network increases the drainage density of the watershed, thereby efficiently collecting runoff and rapidly delivering it to streams (Figure 38) (Graf, 1977a). The decrease in lag time contributes to an increase in flashiness, or the rate of rise or fall of the discharge hydrograph (Rosburg et al., 2017). Thus, the tendency for flash floods is enhanced by urbanization.

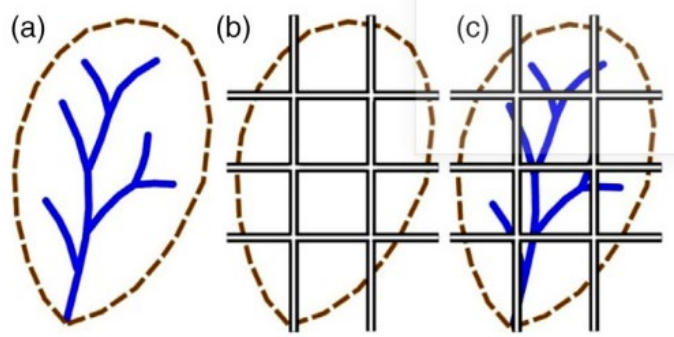


Figure 38. Urban drainage system (c) = Combination of natural (a) plus artificial (b) network configurations. [Source: Adapted from Ress et al. (2020) and Graf (1977a)]

Table 16. Examples of Changes in Lag Time in the Urbanizing Areas of U.S.

Study Location	*Direction/Quantification of Change*	*Hydrograph Metric*	*Reference*
Washington, D.C.	decrease	Lag time	(Carter, 1961)
Pennsylvania	decrease	Lag time	(Leopold, 1968)
East Meadow Brook, Nassau County, Long Island, New York	decrease	Flood peak widths	(Seaburn, 1969)
Detroit, Michigan	20% decrease	Lag time	(Brater and Sangal, 1969)
NE Exeter, Devon	50% decrease	Lag time	(Gregory, 1974)
NE Exeter, Devon	50% decrease	Lag time	(Gregory, 1976)
White Rock Creek watershed, Collin and Dallas counties, Texas	25% decrease (3.27 hour in the 1960s to 2.45 hour in the 2000s)	Average lag time	(Vicars-Groening and Williams, 2007)
Rocky Branch watershed, Columbia, South Carolina	decrease	Lag time	(Ress et al., 2020)

3.2.2.3 Peak discharge

The integrated outcome of increases in stormwater runoff volume and decreases in lag time is an increase in peak discharge (Rhoads, 1995) (Figure 37). The rapid runoff transit times produced by storm-sewer systems in a watershed decreases the travel time for stormwater runoff to reach streams, enhancing the convergence of flow from different parts of the watershed over a shortened interval. This effect, when combined with increased runoff volume from impervious cover, produces an enlarged flood peak (Anderson, 1970). Recent data analysis for 280 stream gages unaffected by dams or other upstream storage throughout the United States indicates that for every percentage point increase in impervious surface cover in a watershed, the magnitude of the annual flood, the peak discharge for any given year, increases by 3.3% (Blum et al., 2020).

Runoff from saturated permeable surfaces can also contribute substantially (30 to 60%) to flood peaks in urban areas (Skaugen et al., 2020), with one study indicating a 10-fold increase in peak flow rate per unit area (Burges et al., 1998). As a result, antecedent moisture conditions, which often are discounted in urban areas given greater concern about the effects of impervious surfaces, can have a substantial influence on urban flood response and should be accounted for in assessing impacts of urbanization on flooding, especially in the light of potential shifts in soil moisture regimes associated with climate change (Hettiarachchi et al., 2019). This influence may, however, be spatially heterogeneous given that some studies have found little or no impact of antecedent moisture conditions on runoff volume and timing in urban areas (Miller and Hess, 2017; Zhou et al., 2017). Also, unraveling the combined effects of climate change, particularly changes in the frequency and intensity of precipitation, and the growth of impervious surface cover related to increasing density of development are often difficult to isolate in urban environments (Hodgkins et al., 2019).

The effect of highly developed urban areas on runoff has been demonstrated to not only amplify the flood peaks, but also create completely new flood events (Booth, 1991). Hydrologic modeling of the Hylebos Creek watershed in the state of Washington for complete forest cover versus urban land use showed that flood events equivalent in magnitude to the 5-year flood for the forested condition occurred 39 out of 40 years for the urbanized condition (Booth, 1991). In other words, the magnitude of peak discharges of flows with specific recurrence intervals greatly increases compared to pre-urbanization conditions. As a result, the frequency of flooding increases in urban environments. Studies from around the world (Table 17) indicate that peak discharge typically increases from 1.5 to as high as 8 times as a result of urbanization. The relative increase in peak discharge caused by urbanization is greater for high-frequency events than for infrequent events (Hollis, 1975). Because the relative increase in floods induced by urban development declines with increasing recurrence intervals (Hollis, 1975), urbanization generally tends to reduce the variability of peak discharges (Anderson, 1970). Data analysis for the Chicago region confirms this type of response to urbanization. Data for Salt Creek, Illinois (Figure 39), indicate that increasing urbanization has increased the magnitude of small floods by about 200% (from about 400 cubic feet per second, or ft^3/s, to 1200 ft^3/s), whereas large floods have increased by only about 100% (from about 1000 ft^3/s to about 2000 ft^3/s) (Konrad, 2003). Similarly, analysis of data for 103 stream gages in the northeastern part of the state, including the Chicago region, using data through 1977 show that the effect of impervious surface cover on flood peaks of particular recurrence intervals decreases as the recurrence interval increases (Allen and Bejek, 1979) and that the magnitude of this effect increases as impervious surface cover increases. More recent analysis of peak-discharge data for 143 stream gages in the same region from 1945 to 2009 confirms that, under the assumption of an average impervious surface cover of 30% for urban land defined on the basis of housing density, increases in impervious surface cover (achieved through increases in housing density) have a greater effect on the magnitude of frequent floods compared to infrequent floods (Over et al., 2016) (Figure 40). Moreover, in contrast to the relations derived by Allen and Bajek (1979), which predict decreasing rates of increase in peak discharge as the percentage of impervious surface cover increases, the relations developed by Over et al. (2016) predict increasing rates of increase in peak discharge as the percentage of impervious surface cover increases (Figure 40).

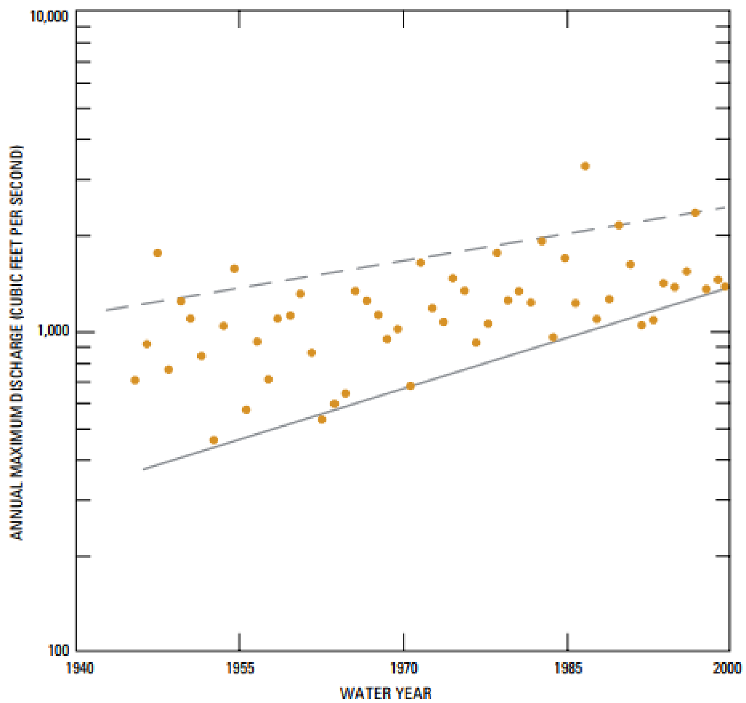


Figure 39. The relative increase in annual maximum discharge in Salt Creek, Illinois, (USGS gaging station 05531500) (Konrad, 2003). Solid and dashed lines denote small floods (less than 95% of the annual peaks) and large floods (more than 95% of the annual peaks), respectively.

The urban impact on flood peaks varies seasonally, causing larger peaks during times when land surfaces are typically dry (lack antecedent moisture) during summer and fall than during wet seasons of winter and spring (Yang et al., 2011). Similarly, increases in peak discharge are relatively more pronounced in dry years than in wet years because an increase in the impervious surface cover amplifies runoff to a greater extent when soils are unsaturated versus saturated (Schütte and Schulze, 2017). Results of hydrologic modeling suggest that increased storm drain networks and stormwater management ponds influence the peak discharge more significantly than the proportion of impervious surfaces in urban areas (Meierdiercks et al., 2010). Recent work has also pointed out the importance of the walls of high-rise buildings in urban areas; these walls can influence the peak discharge more than the rooftop component. An experimental study showed that for a building with a ratio of wall area (windward-facing) to rooftop area of 3:1, the walls can increase peak flows by 14.2% to 17% for a wind speed of 1.1 meters per second (Yoo et al., 2021). If the ratio of the wall area to the rooftop area increases to 10, the amount of rainwater from the building wall can be larger than that from the rooftop, even if the wind speed is only 1 m/s (Cho et al., 2020). Changes in flood magnitude and frequency are an important component of the urban stream syndrome that can increase stream erosion and negatively impact stream ecology (Konrad and Booth, 2005; Walsh et al., 2005).

Table 17. Examples of Changes in Peak Discharge in the Urbanized Areas of the World

Study Location	*Direction/Quantification of Change*	*Flow metrics*	*Reference Study*
Washington, D.C.	1.8 times larger	Flood peak	(Carter, 1961)
Nassau County, Long Island, New York	2.3 times increase	Peak discharge (occurrence interval 1.15 years)	(Sawyer, 1961)
Northern New Jersey, Michigan, Pennsylvania, and Virginia	3-4 greater (average)	Peak runoff	(Waananen, 1961)
Harris County, Houston, Texas	2-5 times greater	Flood peak	(Van Sickle, 1962)
Waller Creek watershed, Austin, Texas	51% and 6% increase (for 23rd & 38th Street stations, respectively)	Peak discharge	(Espey Jr et al., 1965)
Jackson, Mississippi	4.5 times larger	Mean maximum annual flood peaks	(Wilson, 1967)
Tokyo (northern part)	Increased by a factor of 3	Flood peak	(Kinosita and Sonda, 1967)
East Meadow Brook, Nassau County, Long Island, New York	2.5 times increase (from 313 to 776 cubic meters per second)	Average peak discharge	(Seaburn, 1969)
Metropolitan area, Washington, D.C., USA	2-8 times increase	Flood peak	(Anderson, 1970)
Upper Santa Anna Valley, California, USA	3-6 times increase	Peak discharge (recurrence interval 2 years)	(Durbin, 1974)
Canon's Brook Catchment, Harlow, England	Approximately 150% increase (from 0.057 m^3/s to 0.142 m^3/s)	Median flow	(Hollis, 1977)
Peachtree Creek, Atlanta region, Georgia, USA	30% to more than 100% greater	Peak discharge	(Rose and Peters, 2001)
McMullen Creek basin, Charlotte, North Carolina	3.4% increase	Flood peak	(Smith et al., 2002)
White River basin, Indiana, USA	19% increase	Frequency of high flow	(Yang et al., 2010)
Haydon Wick catchment, Swindon, UK	Over 400% increase (from 0.31 $m^{3/}s$ to 1.65 $m^{3/}s$)	Peak flow	(Miller et al., 2014)
Belo Horizonte, MG, Brazil	Increased by 30% on average	Peak flow at catchment outlet	(Rosa et al., 2020)

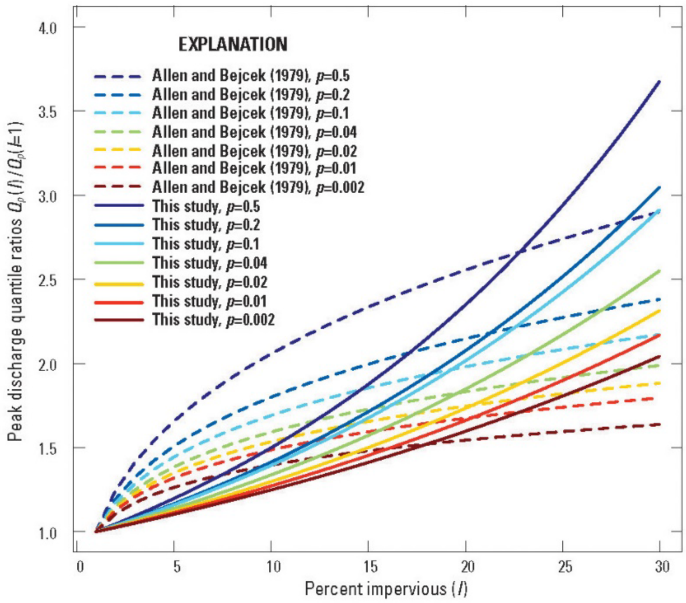


Figure 40. Comparison of increases in peak discharge of different probabilities (inverse of recurrence interval) for increasing impervious surface cover in the Chicago region (above a base value of 1% for impervious conditions) as determined by Allen and Bejcek (1979) versus Over et al. (2016) (referred to as this study) (from Over et al., 2016)

The spatial distribution of urban development within the basin also influences the peak discharge. In large watersheds (1000–10000 km^2), development locations close to the basin outlet, i.e., where the increased stormwater runoff has short travel times and therefore precedes the main flood peak of the basin as a whole, have small effects on the peak magnitude compared to development locations within the center of a watershed that correspond to the modal travel time of the entire watershed, which has the maximum effect on flood peaks (Yang et al., 2011). In small watersheds (< 200 km^2) the increase in peak discharge and the decrease in lag time both become greater as the urbanized portion of the watershed shifts progressively closer to the watershed outlet, an effect that can be captured by the correlation between either peak flow and lag time and a geometric index called the "Relative Nearness of Imperviousness to the Catchment Outlet" (RNICO) (Roodsari and Chandler, 2017). Thus, the increase in peak discharge observed in small watersheds tends to be moderated in large watersheds, i.e., the impacts decrease with the watershed scale, and large watersheds are less sensitive to an increased degree of development (Roodsari and Chandler, 2017; Rougé and Cai, 2014). Dual peaks are sometimes observed in hydrographs, representing largely separate urban and rural areas within a watershed, the relative size of which depends on the degree of urbanization (Sheeder et al., 2002).

3.3 Sediment in Urban Streams

3.3.1 Conceptualization of the impact of urbanization on sediment delivery

Urban development, by changing land cover conditions, often has a marked impact on the sediment dynamics of watersheds. Changes in land cover affect the extent to which soil is exposed to erosion by surface runoff, thereby resulting in changes in rates of delivery of eroded soil to streams. Moreover, changes in the hydrology and hydraulics of streams caused by urbanization will influence the potential for stream channel erosion or deposition, which also will affect the amount of sediment transported by streams.

Early geomorphological research in the 1960s produced what has become a well-known conceptual model characterizing temporal change in sediment dynamics of streams undergoing urbanization (Figure 41). This model was developed based on an analysis of data for urban areas in the eastern United States, specifically the Baltimore area. The model shows how land-use changes over time affect the sediment yield of streams, i.e., the average mass of sediment exported from a contributing watershed per year. In the eastern United States, most land was initially forested. Once the forest was cleared for agriculture, sediment yields increased as both the increased exposure and working of the soil increased its susceptibility to erosion. As urban areas expanded, they encroached on agricultural land, transforming it through the process of urbanization. This process and its effects on sediment dynamics can be characterized by two phases: the construction phase and the urbanized phase. During the construction phase, land is cleared, soil is exposed to erosion, and sediment yields increase dramatically, exceeding those associated with agricultural land use. The length of this phase is variable, but it can last several years in large-scale projects, such as major housing developments. Percentages of exposed soil at project sites can reach 100% during early parts of the construction phase and may remain at 50% for more than two years (Russell et al., 2020) (Figure 42). As urbanization progresses to the urbanized phase, much of the exposed soil is covered by various types of impervious surfaces, including parking lots, roofs, sidewalks, and streets. Other areas, particularly in suburban settings, are landscaped with shrubs, grasses, or scattered trees. According to the conceptual model, soil erosion during this phase decreases markedly to levels less than those associated with agricultural land use (Figure 41).

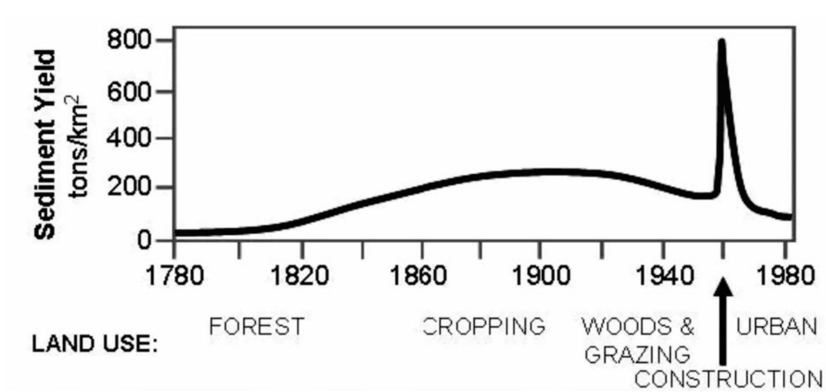


Figure 41. Change in sediment yield over time in the eastern United States with changes in land use (from Lord et al., 2009, adapted from Wolman, 1967)

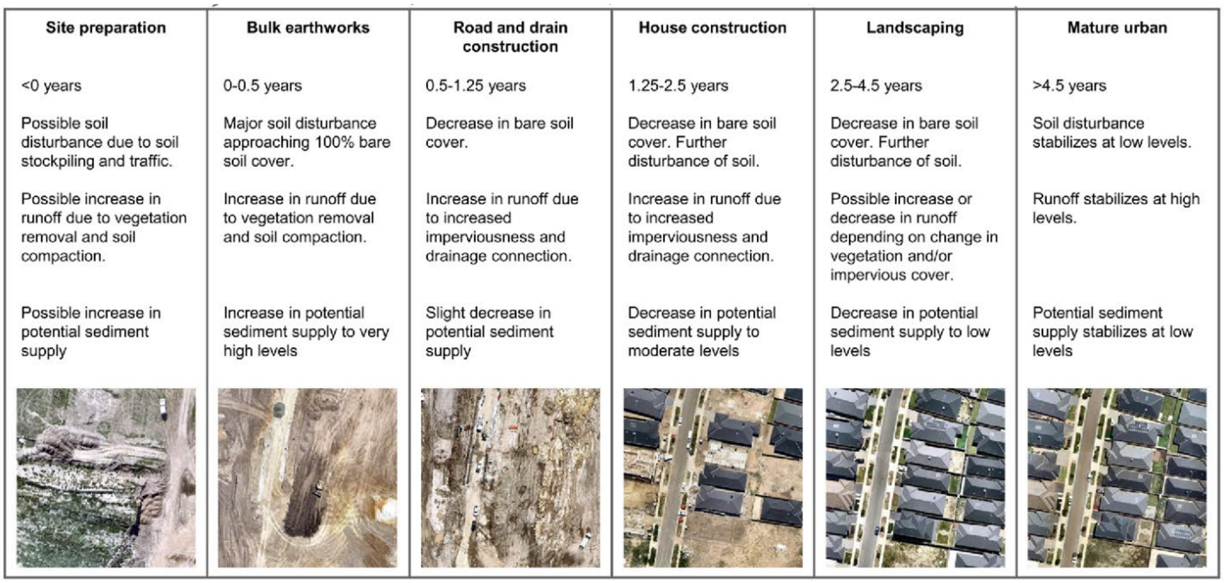


Figure 42. Detailed sequence of land-cover changes during urbanization associated with residential housing development and the relation of each element in the sequence to sediment dynamics. The first three elements of the sequence are components of the construction phase, the next two elements are transitional components, and the sixth element represents the urbanized phase (from Russell, 2021).

3.3.2 Studies of the impact of urbanization on sediment delivery

Although this two-phase model of urban sediment dynamics is prominent, evidence to support its validity is rather limited. Early studies showed that sediment yields from urban areas under construction can be as much as two orders of magnitude greater than those for nearby agricultural watersheds of the same size (Wolman and Schick, 1967; Walling and Gregory, 1970). Overall, the increase in sediment yield related to construction tends to decrease with watershed size given that the construction effect is often localized. Considerable dilution of this effect occurs as runoff from areas not undergoing active construction is included in the total area under consideration. These early studies were completed before the widespread implementation of sediment erosion-control practices. These practices can be highly effective at reducing erosion of exposed soil. The use of mulches, polyacrylamides, rolled erosion control products, compost, or compaction reduces soil erosion rates by 20 to 95% compared to rates for untreated bare soil (Tyner et al., 2011).

The discharge of sediment to streams from construction sites in Illinois is regulated under Sections 401 (Water Quality Certification), 402 (National Pollutant Discharge Elimination System), and 404 (Dredge or Fill Permitting) of the Clean Water Act of 1977. Of particular relevance is the NPDES Permit for Construction Activities administered by the Illinois Environmental Protection Agency (https://www2.illinois.gov/epa/topics/forms/water-permits/storm-water/Pages/construction.aspx). Given this regulatory environment, the use of erosion-control best management practices (BMPs) is now common at construction sites. Nevertheless, recent research demonstrates that BMP implementation is not entirely effective at mitigating the problem. For the period 1989–2009, soil erosion rates at construction sites on the fringe of the expanding Phoenix metropolitan area generally were 1.3 to 3.1 times greater than those for undeveloped grazed land (Jeong and Dorn, 2019). In small urbanizing watersheds in South Carolina with active construction during 2004–2007, sediment yields of streams were 60 to 90 times greater than those for an undeveloped reference watershed (Santikari and Murdoch, 2019). These elevated yields occurred despite the implementation of erosion-control BMPs such as silt fences, check dams, inlet protection devices, sediment control basins, hydroseeding, and 6-meters-wide riparian buffers on either side of the streams. Moreover, sediment yields of the urban streams were six times greater than the reference watershed in completely developed areas where construction was complete and impervious surfaces constitute the majority of land cover (Santikari and Murdoch, 2019). In fact, when examined at a global scale, the existing data indicate that construction does lead to large increases in sediment yield, but these yields tend to exceed rates associated with agricultural land use, even during the second urbanized phase of urbanization (Russell et al., 2017). Thus, the classic model (Figure 41) is being revised to recognize that urban watersheds, whether under construction or in a developed condition, are characterized by elevated average sediment yields (Figure 43).

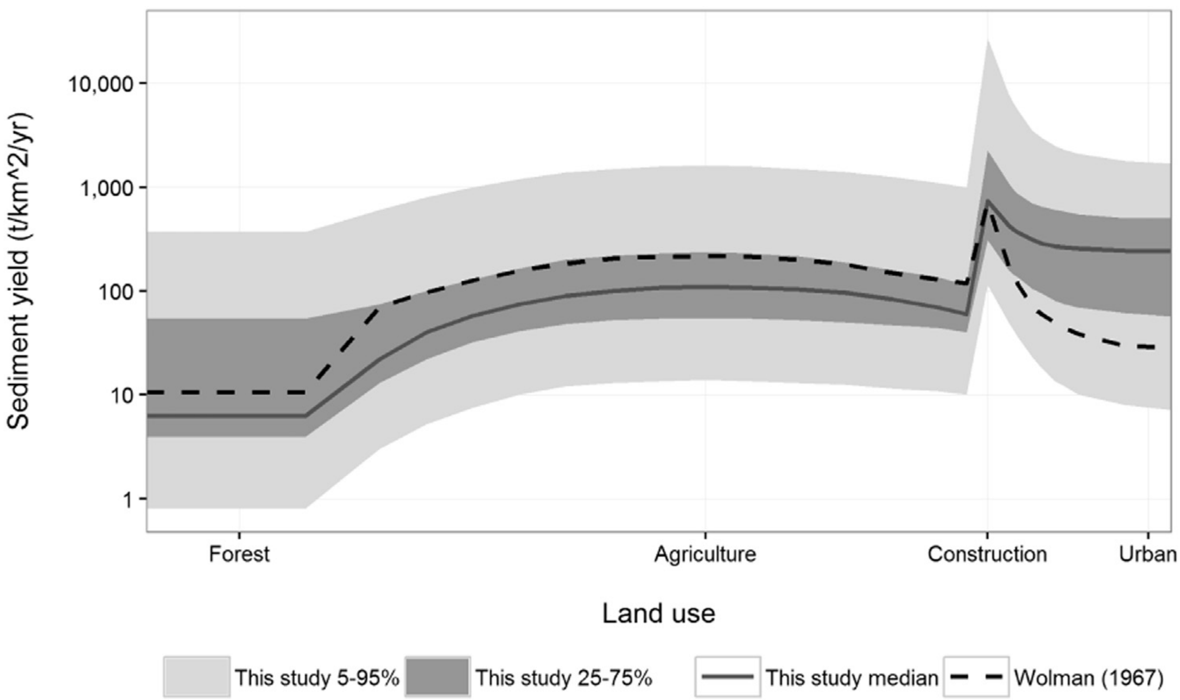


Figure 43. Revised conceptual model of influence of urbanization on sediment yield based on survey of available data (Russell et al., 2017). Original model (Figure 41) shown as dashed line.

A particular focus of recent urban stream research on sediment dynamics has been to determine the origin of sediment in these streams. If yields remain high once construction is completed, where is the sediment coming from? One possibility is that surface wash off of the sediment from impervious surfaces is a more substantial source than was previously recognized. Further research is needed on this issue, but field studies show that wash off from impervious surfaces is complex and varies greatly among runoff events (Bai and Li, 2013). The use of radionuclides of beryllium (^{7}Be) and lead (^{210}Pb) to trace sources of sediment in a small, urbanized watershed draining to the Chesapeake Bay showed that only 15% of the total sediment load was produced by runoff from paved surfaces (Gellis et al., 2020). Another potential source of sediment that has received considerable attention is in-channel material, particularly sediment eroded from the bed or banks of urban streams. Increases in stream power associated with increases in flow magnitudes and durations as well as with changes in channel form that promote increases in flow velocities often lead to enhanced erosion of urban streams. This eroded material serves as an important source of suspended sediment and can also be transported along the bottom of the river as bedload. Several studies have shown that stream erosion can contribute a substantial fraction, if not the majority, of suspended sediment to urban streams (Table 18).

Table 18. Contributions of Within-Channel Sources of Sediment to Suspended Sediment Flux in Urban Streams

Channel Erosion Contribution to Sediment Flux	*Source of Sediment*	*Stream*	*Reference*
66%	channel erosion	San Diego Creek, California	Trimble, 1997
20%	bank erosion	Issaquah Creek, Washington	Nelson and Booth, 2002
43%	bank erosion	Valley Creek, Pennsylvania	Fraley et al., 2009
91%	bank erosion	Upper Difficult Run, Virginia	Cashman et al., 2018
57% ± 15%	bank erosion	Dead Run, Maryland	Gellis et al., 2020

Although most research on urban sediment dynamics has focused on fine (sand and smaller) sediment transported in suspension, recent studies have begun to examine delivery and transport of coarse (gravel and larger) sediment to urban streams. Although much sediment coarser than sand in urban streams comes from streambank erosion, some of this material can be supplied from urban sources in the form of debris from disintegrating concrete or other surfacing material (Russell et al., 2018). Changes in hydrology and hydraulics associated with urbanization can increase bedload transport capacities (Russell et al., 2020) and the frequency at which coarse bed material is mobilized (Plumb et al., 2017), resulting in greater bedload sediment yields of urban streams compared to yields of nearby rural streams (Russell et al., 2018). In urbanized environments, frequent events become more effective in mobilizing bed material and redistributing it throughout the stream system. Such changes can be important ecologically because mobilization of coarse material may represent a form of disturbance for benthic organisms. Geomorphologically, the movement of bed material plays an important role in shaping the form of stream channels; more frequent movement of bed material will potentially produce more frequent changes in channel form. Detailed analysis of bedload transport in a gravel-bed urban stream in southern Ontario indicates that urbanization has increased the frequency and distance of movement of coarse bed material, which probably accounts for active enlargement of the channel (Papangelakis et al., 2019). By contrast, a nearby stream with stormwater management in the form of several offline and online retention ponds as well as a high flow diversion weir has less frequent bed-material movement because the attenuated peaks now do not have the competence to move a wide range of sizes of bed material.

3.3.3 Sediment budgets as a framework for analyzing changes in urban sediment dynamics

Recent attempts to fully characterize the sediment dynamics of urban watersheds draw upon the methodology of sediment budgeting, a powerful analytical framework for assessing in detail how sediment delivery and storage interact to yield a net output of sediment from a watershed. Sediment budgets are accounting schemes used to evaluate patterns of sources of sediment, storage areas for sediment, and connectivity between sources and sinks (Reid and Dunne, 2016; Rhoads, 2020).

A sediment budget developed for the urbanizing watershed of the Good Hope Tributary in Maryland for 1951 to 1996 shows that upland erosion delivered to small headwater tributaries was 70% of the total amount of sediment leaving the watershed over that same period (Allmendinger et al., 2007) (Figure 44). Even though sediment was being supplied to the tributaries, the channels of these tributaries enlarged, producing additional eroded sediment equal to 40% of the total exported load. All this eroded sediment (8900 m^3) was delivered to the Good Hope Tributary, the main stream in the watershed, where erosion of the stream channel generated additional sediment equal to 40% of the total amount of exported sediment. A large volume of sediment delivered from the headwater tributaries or from erosion of the main stream channel was deposited on the floodplain of the Good Hope Tributary, an amount equal to 50% of the total exported sediment. In other words, floodplain storage helped to mitigate the large amounts of sediment delivered to this stream and to reduce the amount of exported sediment (8100 m^3) to a value slightly below the amount delivered to the main stream from erosion of uplands and within tributaries (8900 m^3).

A sediment-budget approach was used to assess the supply and storage (sinks) of coarse sand (> 0.5 mm) and gravel (> 2mm) within a suburban stream system near Melbourne, Australia (Russell, 2019a, 2019b) (Figure 45). Sources and sinks of sediment in this system include hillslopes, the stormwater network, and stream channels. Most sediment eroded from hillslopes was either redeposited on hillslopes or extracted from the system through street sweeping. About 67% of the total amount of sand and gravel supplied to the storm sewer network was transported out of the watershed–a high percentage compared to surrounding forested watersheds (Russell, 2018). This efficiency of sediment export was attributed to the high transportability of sediment through the network of pipes comprising the stormwater system and to the lack of floodplain or in-channel storage along highly modified streams that were disconnected from adjacent floodplains. Despite the low percentage of stored sediment within the storm sewer system, frequent cleanouts are required given the large volumes of material delivered to this system from hillslope erosion. Also, the substrate of the main stream system is thin and of uniform texture–a condition that impedes the development of instream geomorphological features, such as pools and riffles, which provide habitat for aquatic organisms.

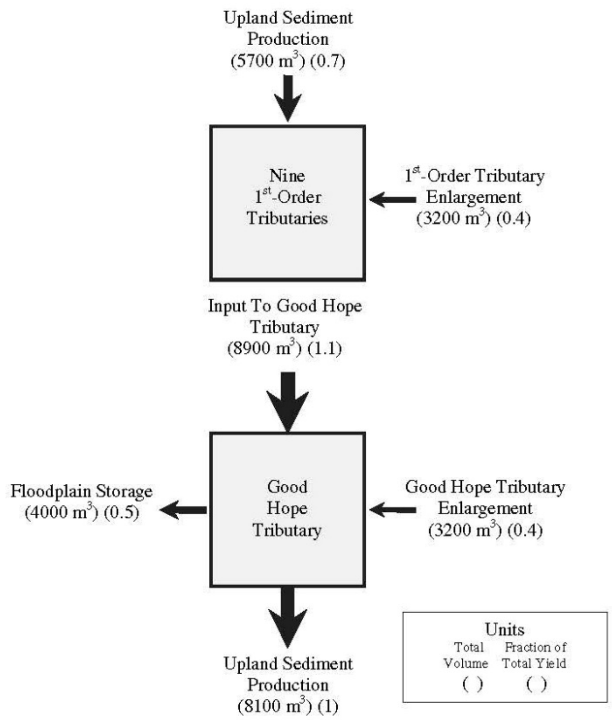


Figure 44. Sediment budget for the Good Hope Tributary watershed in Maryland (from Allmendinger et al., 2007)

Sediment budgets, while useful for determining the sources and sinks of sediment in urban watersheds, typically require a considerable investment of time and effort to develop and are typically prone to substantial uncertainties (Reid and Dunne, 2016). Direct measurements of sediment fluxes related to sources and sinks are costly and usually can only be sustained for short periods of time within relatively small watersheds. Sediment fluxes in urban settings are typically characterized by high levels of variability over short timescales and small spatial scales (Kemper et al., 2019), adding to uncertainty in accurately estimating the average values of these fluxes. In most cases, the construction of sediment budgets is not based on direct measurements of sediment fluxes, but instead relies on the estimation of budget components using indirect methods. Often, sediment fluxes on hillslopes are estimated based on models that can simulate soil erosion and deposition. Net deposition at potential sites of sediment storage, such as detention ponds and floodplains, can be estimated through coring. Changes of sediment storage within stream channels usually rely on repeat surveying of channel cross sections over time. This method, if used to estimate a sediment budget over a period of many years or decades, depends on the availability of historical surveys of channel cross sections.

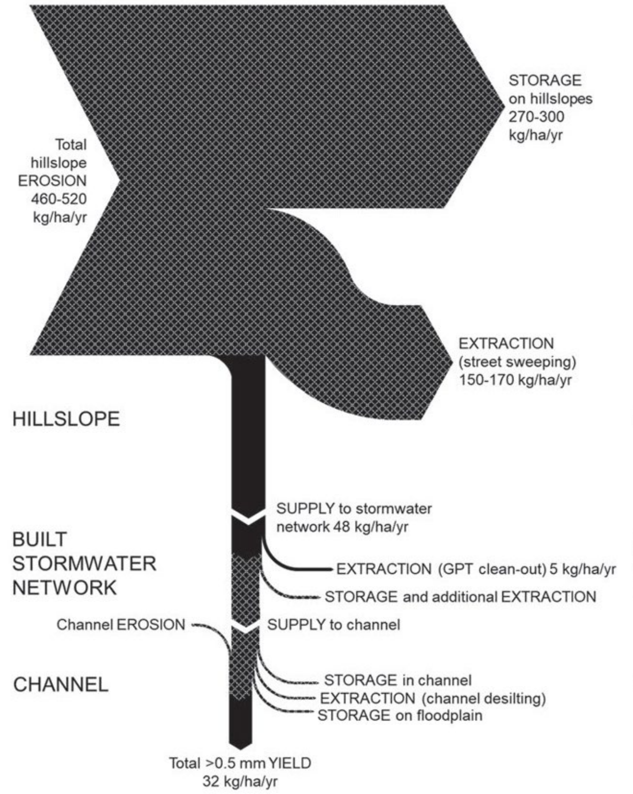


Figure 45. Sediment budget developed for coarse sand and gravel (> 0.5 mm) in the Scotchman's Creek watershed near Melbourne, Australia (from Russell et al., 2019b)

3.4 Urban Stream Hydraulics

3.4.1 Impacts of urbanization on stream hydraulics

Urban development can profoundly influence the hydraulic conditions of streams. Although hydraulic impacts of urbanization on streams have focused mainly on ecological degradation, understanding of this phenomenon is crucial geomorphologically as it provides key mechanistic insight into linkages between bed-material transport and changes in channel morphology by erosion or deposition. Once urbanization takes place, changes in the quantity and rate of runoff (altered catchment hydrology) and in channel form (changes in the geometry or alignment of stream channels) are the dominant factors influencing hydraulic properties, such as flow velocity, depth, and width. Changes in these hydraulic properties in turn are linked to changes in the energy, power, and force of the flowing water. Such changes play a key role in mobilizing sediment comprising the channel boundary and in determining the capacity of the stream to transport mobilized sediment (Russell et al., 2020).

3.4.2 Bed shear stress and stream power per unit area as useful hydraulic metrics

Two major factors produce changes in the hydraulics of urban streams: change in watershed hydrology that tends to promote increases in the volumes and rates of runoff (see section of this report on urban hydrology) and changes in the physical characteristics of streams, known as channelization, including channel straightening, widening, deepening, and lining of stream beds or banks with artificial materials, such as sheet piling, gabions, or concrete

(Brookes, 1988). Changes in hydraulics associated with these two factors can be best understood through the variable of discharge, which links changes in hydrology and channel form to changes in hydraulics. Discharge (Q) equals the product of the flow width (W), depth (D), and mean velocity (U):

$$Q = WDU$$

If the magnitude of a discharge of a particular recurrence frequency (for example, the 5-year flood) increases as a result of urbanization, this increase in discharge will be accommodated, at least in part, by increases in the velocity and depth of flow. If this same discharge is confined to an enlarged, straightened channel with a relatively smooth bed and banks, rather than being able to spread laterally across the floodplain of what was formerly a natural alluvial stream with a variable bed and vegetated banks, the depth and velocity will further increase.

Increases in depth and velocity, along with the increase in channel slope (S) that can accompany channel straightening, increase the shear stress and power of the flow acting on the channel boundary, which, in turn, increase the likelihood of mobilization of this boundary and the accompanying potential for channel erosion. The bed shear stress (τ) is defined as:

$$\tau = \gamma DS$$

where γ is the specific weight of water. Flowing water as it moves downslope expends energy in overcoming frictional resistance (internal and boundary) and in eroding and transporting sediment. The time rate of energy expenditure is known as stream power (Bagnold, 1966; Rhoads, 1987). Stream power per unit area of the bed (ω) equals the bed shear stress multiplied by the flow velocity, or:

$$\omega = \gamma DUS = \tau U$$

Both τ and ω have served as fundamental metrics for predicting rates of bed-material transport (q_b) in natural rivers. Numerous formulations of q_b have been developed, but generally these formulations relate rates of bed-material transport to values of τ or ω in excess of critical values of these two metrics required to mobilize specific size fractions (i) of the bed material (τ_{ci} or ω_{ci}) (Rhoads, 2020). Moreover, these formulations typically are nonlinear power functions with exponents greater than 1, indicating that the bed-material transport rate increases rapidly as values of τ or ω exceed τ_{ci} or ω_{ci}. Although a similar approach can be used to evaluate the potential for particles in channel banks to be mobilized, in many cases bank material is cohesive, limiting the value of relations based on the mobilization of individual non-cohesive particles. Nevertheless, the potential for cohesive banks to erode often correlates strongly with stream power and corresponding bed-material transport rates because the movement of non-cohesive bed material at the base of cohesive banks can destabilize these banks (Alber and Piégay, 2017; Larsen et al., 2006; Nanson and Hickin, 1986). Given its strong connection to bed-material transport, stream power has been viewed as a primary metric for assessing the potential for erosion and deposition in stream channels (Bizzi and Lerner, 2015).

Urbanization-induced increases in runoff can result in extensive increases in magnitudes of discharges of specific recurrence intervals, resulting in corresponding increases in bed shear stress and stream power throughout drainage networks. As a result, the bed-material transport capacity of streams will increase compared to pre-urbanized conditions, enhancing the potential for stream erosion. In channelized urban streams (i.e., large rectangular or trapezoidal flood control channels), confinement of floods within enlarged channels increases flow depths and velocities, whereas channel straightening increases channel slopes. As a result, channelization generally increases the bed shear stress and stream power, enhancing the bed-material transport capacity (Bagnold, 1966) and the potential for erosion (Bledsoe and Watson, 2001; Rhoads, 1995), even in channels lined with concrete (Vaughn, 1990).

Channel morphological conditions can often vary spatially in urban environments because of differences in management of different segments of a river within different political jurisdictions. Channelized reaches may differ from one another or may be juxtaposed with more natural reaches maintained within forest preserves, parks, or other environmentally protected areas. Local effects, such as bridge crossings, may also produce spatial variation in channel cross sections and slopes. This spatial variation in channel form results in spatial variation in hydraulic conditions. As flowing water is conveyed through morphologically fragmented urban fluvial systems, it develops spatial gradients in bed shear stress and stream power. Where bed shear stress or stream power are increasing over distance, the bed-material transport capacity will also increase and, assuming that sediment supply remains constant, the channel boundary will erode if the material comprising this boundary can be mobilized by the flow (Figure 46). Conversely, where bed shear stress or stream power are decreasing over distance, the bed-material transport capacity will decrease and, assuming that the sediment load is at capacity, deposition will occur within the stream channel (Figure 46). In other words, spatial gradients in the bed-material transport capacity, which are intrinsically linked to the hydraulic conditions of the stream, will determine the mechanism of channel morphological changes over time (Rhoads, 2020). This basic principle is captured nicely by the Exner equation for conservation of bed material, which relates changes in the elevation of the stream bed (η) over time (t) to variations in the bed-material transport rate q_b over distance (x):

$$(1-p)\frac{\partial \eta}{\partial t} = -\frac{\partial q_b}{\partial x}$$

where p is porosity. If q_b decreases over distance ($-\partial q_b / \partial x$), deposition will occur, and the bed elevation will increase, whereas if q_b increases over distance ($\partial q_b / \partial x$), erosion will occur, and the bed elevation will decrease.

Recent studies have corroborated the use of stream power and bed shear stress as important metrics for assessing the potential for morphological change in fluvial systems affected by the urban stream syndrome (Walsh et al., 2005), including erosion potential. These studies also confirm the importance of considering spatial variation in bed-material transport capacity, the maximum amount of bed material a stream can transport given its hydraulic conditions, in assessing the potential for morphological change. For nine headwater streams with varying degrees of urbanization in Melbourne, Australia, cumulative bed-material transport capacity estimated over a one-year period using a bed-material transport function based on excess bed shear stress (Wilcock and Kenworthy, 2002) was found to be one to three orders of magnitude higher for urban and peri-urban streams than for non-urban reference streams (Russell et al., 2020). Most of this difference was associated with higher discharges during the same storm events in urban streams compared to rural streams. In addition, the estimated bed-material transport capacity exceeded measured bed-material transport rates by factors of 50 to 350 in urban streams, indicating that supply-limited conditions exist in these streams. These conditions occur when the supply, or availability, of sediment does not equal the transport capacity of the flow. Most alluvial streams with abundant amounts of sand and fine gravel are transport limited. In these streams the transport of material is limited only by the capacity of the stream to transport sediment, not by the supply of sediment. If the supply of sediment locally is less than the capacity for transport, mobilization of bed material and net removal of this material, i.e., channel erosion, will occur to meet the demand of the sediment-starved flow.

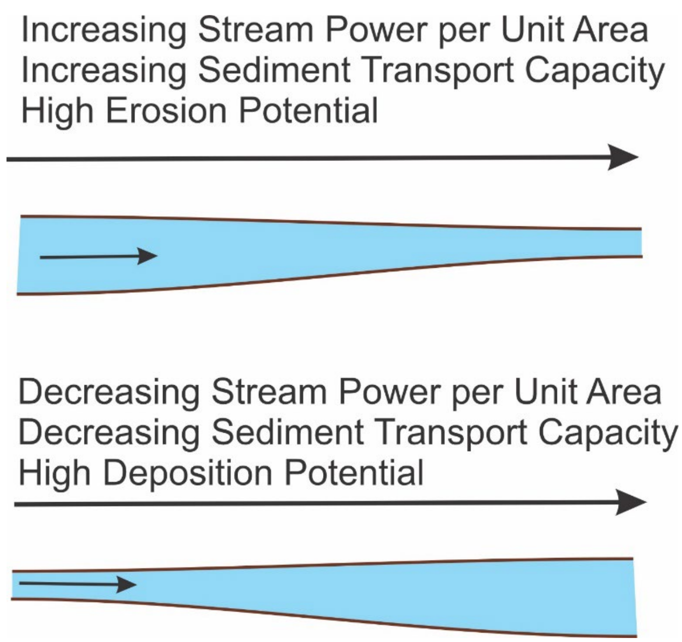


Figure 46. Diagram looking down on a stream from above showing (top) narrowing of the flow, which should increase depth and velocity, leading to an increase in stream power and (bottom) widening of the flow, which should decrease depth and velocity, leading to a decrease in stream power

Using the metric of stream power per unit area, a spatial decision-support model, called "Stream Power Index for Networks" (SPIN), has been developed to map the spatial distribution of stream power per unit area throughout drainage networks (Ghunowa et al., 2021; Papangelakis et al., 2022). This work builds on earlier work on mapping the spatial distribution of stream power per unit length throughout drainage networks (Vocal Ferencevic and Ashmore, 2012). The SPIN tool can be used to assess how existing conditions or potential future scenarios, including plans for stormwater management and stream restoration, affect the spatial distribution of stream power in urban networks. This information, in turn, can provide the basis for evaluating how channel stability and mobility of bed material may vary spatially throughout the network by comparing stream power values to power thresholds governing channel change and bed-material mobilization (Ghunowa et al., 2021; Papangelakis et al., 2022). The tool is readily integrated with data generated by hydraulic models, including HEC-RAS.

Another approach to the application of stream power per unit area to urban stream assessment is to try to relate stream power to the *regime* or equilibrium condition of streams, where regime refers to a dynamically stable state in which sediment flux into and out of the reach of the river under consideration is approximately balanced over time. Under these conditions, channel morphology, although it might change slightly, should not evolve systematically through net erosion or deposition. Drawing upon data for 733 stream channels considered to be "in regime," a machine learning approach was used on two-thirds of these data to generate a model predicting stream power of the two-year flood based on median particle size, channel slope, and bankfull channel depth and width (MacKenzie et al., 2022). The model was then applied to the remaining one-third of the data to see how well it performed in predicting the observed stream power for the two-year flood for these streams that were not included in model development. Overall, the model predicted observed values accurately with an R^2 of 0.85 and limited scatter (Figure 47). Values of stream power per unit area for the two-year flood for several urban streams in southern Ontario plotted outside of the envelope of regime stream power, indicating that these channels likely are not in regime and have excess stream power, which should result in erosional instability (Figure 47).

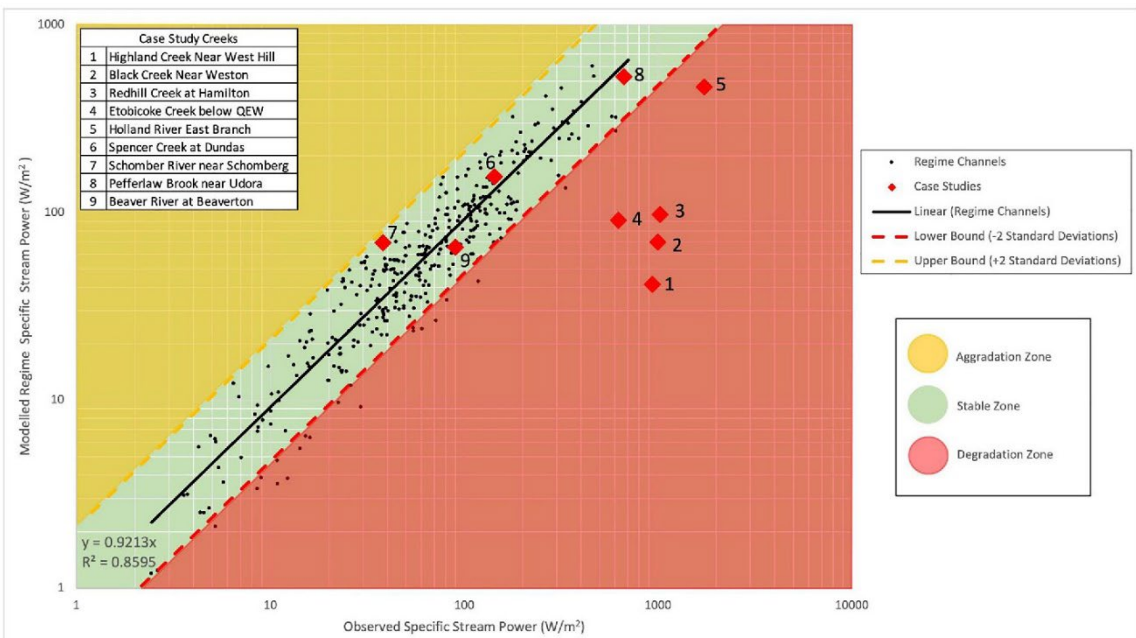


Figure 47. Plot of observed stream power per unit area versus predicted stream power per unit area for regime streams (modified regime-specific stream power) showing domain of stable streams (green), streams likely to aggrade (yellow), and streams likely to erode (red). Of nine case-study urban streams for southern Ontario, five (1-5) plot outside of the stability envelope, indicating that these streams have excessive stream power relative to regime conditions and are likely to erode (from McKenzie et al., 2022).

The idea of integrating stream power over time to determine the total amount of work that a sequence of flows performs on streams has been the focus of some recent studies (Ibrahim and Rouhi, 2021; Soar et al., 2017). A River Energy Audit Scheme involves integrating stream power per unit area with flow duration to investigate network-scale distributions of annual geomorphic energy (Soar et al., 2017). For each reach throughout the network, excess stream power per unit area for different size fractions of the bed material is determined as an integrated measure, which is then scaled up to the channel width and integrated over time to yield a total annual measure of energy (Ea) or work (Wa):

$$\mathbf{W_a} = \mathbf{E_a} = \left\{ \sum_{j=1}^{m} f_j W_j \left[\sum_{i=1}^{n} p_i \left(\omega_j - \omega_{ci} \right) \right] \right\} * t$$

where p_i is the fraction of the total amount of bed material found in size class i of the particle-size frequency distribution, n is the total number of particle-size classes, ω_j is the stream power associated with discharge class j of the discharge frequency distribution, W_j is the flow width for each discharge class (j), f_j is the fraction of the total flow duration associated with discharge-class j, m is the total number of discharge classes, and t is time (for annual work, the number of seconds in a year). Mapping of the spatial distribution of $\mathbf{W_a}$ by reach throughout a network can show where work is exceptionally high or low and where spatial transitions occur in work (high to low or low to high), both of which may indicate channel instability in the form of erosion or deposition. This approach provides insight into the system-wide potential for channel instability and sensitivity of streams to changes in land-use or climate, thereby informing strategic planning for river channel management.

A somewhat similar approach proposes an integration of stream power but actually involves integration of momentum flux, rather than power, to get an index of the total momentum of the flow over a series of events (Ibrahim and Rouhi, 2021). A weakness of this approach is that it is not limited to flows that exceed the threshold for particle mobility, i.e., those capable of actually reshaping the channel boundary, but includes all flows regardless of size. Nevertheless, in a specific application of the approach, the cumulative momentum of all flows accounted for about 70% of the variance in observed channel changes at stormwater outfalls in Fairfax County, Virginia.

A network-scale event-based simulation tool based on the stream power concept has been developed to assess channel evolution over time scales of years to decades (Lammers and Bledsoe, 2018). Simulations can be based either on actual records of flows to evaluate how channels have responded to these flows or on a hypothetical sequence of flows to explore how channels may respond to such flows. The model incorporates algorithms to account for bed and bank erosion; it can also account for bed aggradation where deposition is a concern. Although not tested in an urban setting, the model has been proposed as a useful tool for evaluating urban stream evolution.

Related work has attempted to determine the cumulative bed-material transport capacity over a series of flows for urban streams and then compare that capacity to the capacity of the same flows for a nonurban reference stream (Russell et al., 2020). For small urban streams in Australia, erosion potential, the ratio of the cumulative transport capacity of urban streams to the transport capacity of the reference stream, ranged from 27 to 1117, indicating the dramatic increase in cumulative transport capacity caused by urbanization (Russell et al., 2020). Because the vast majority of the increase for these particular urban streams was related to changes in hydrology, rather than channel morphology, only the mitigation of excess runoff could reduce transport capacity toward values characteristic of the reference stream.

Other work related to aquatic habitat assessments has mainly examined the value of using bed shear stress, either in a dimensional or dimensionless form, to assess the potential of bed material to be mobilized. This approach emphasizes that aquatic organisms are likely to be disturbed when bed mobility is common as is often the case in urban streams subjected to increased runoff and channelization. The assessment of bed mobility potential based on actual shear stress versus critical shear stress for Cardinia Creek in southeast Melbourne, Australia indicated that the actual shear stress was above the threshold for entrainment four times longer for an urbanized reach compared to a non-urbanized reach (Anim et al., 2018). Based on results of 2D hydrodynamic model simulations, the duration of maximum bed shear stress equaling or exceeding the critical bed shear stress for the urban reach was 120 days/year, whereas the period of exceedance was only 35 days/year for the non-urban reach. Similar findings for McMahons Creek in Melbourne revealed increased frequency of an unstable channel bed exhibiting partial or full mobility (Anim et al., 2019a).

Conventional stormwater control measures that are focused only on a "peak matching" strategy, i.e., a strategy aimed at matching peak discharges in urban areas to those that occurred prior to urbanization or that satisfy desired flood-inundation requirements, typically lead to an increased duration of erosive flows, i.e., flows that exceed the critical discharge (Q_c) for bed particle entrainment, and thus increase the frequency and duration of excess shear stress (Bledsoe, 2002; Hawley et al., 2017). These erosive hydraulic environments contribute to geomorphic and ecological degradation, commonly referred to as the urban disturbance regime (Hawley and Vietz, 2016). This work emphasizes the importance of considering not just the magnitude of the peak discharges in stormwater management, but also how the management of hydrological conditions is connected to hydraulic conditions, which in turn strongly influence ecological quality and geomorphic stability.

3.4.3 Hydraulics and physical habitat

The extents of floodplain inundation and shallow slow-water habitat (SSWH) are two hydraulically relevant metrics that are critical to the sustenance of biota but typically are in short supply in urban streams. Floodplain inundation for urban streams typically is less in frequency, duration, and area than for rural streams because of increased channel capacity (channelization) and the flashy nature of urban high flows (Anim et al., 2019a; Anim et al., 2018). Lack of variable bed structure (e.g., pool, riffle, runs) in urban streams (Hawley et al., 2013) results in less variability in flow velocity and depth (Anim and Banahene, 2021), thereby limiting hydraulic habitat. The combined effect of uniform channels and frequent high discharges cause urban streams to have uniformly high flow depths and fast-flowing water, reducing the availability of SSWH areas (Anim et al., 2019a; Anim et al., 2018; Anim et al., 2019b).

Incorporating in-stream variability of substrate and bed morphology into the stormwater mitigation practices has the potential to reduce the frequency of bed mobility, a key concern related to channel stability, and support habitat and ecosystem functioning (Anim et al., 2019a). Because erosive conditions in urban settings reflect changes both in channel morphology as well as in flow regime, restoration of pre-development hydrology alone, a common focus of stormwater management practices, will not necessarily improve the bed disturbance regime (Anim et al., 2019b). From an ecological standpoint, both key stressors, altered hydrology and morphology, must be addressed and incorporated into the mitigation goals of urban stream management to reduce the ecological and geomorphological effects of the urban stream syndrome (Anim and Banahene, 2021).

3.5 Effects of Urbanization on Stream Channel Form

3.5.1 Basic conceptual model

Alteration in drainage basin hydrology (Section 3.2), hydraulic properties (Section 3.4), and sediment regime (Section 3.3) associated with urbanization ultimately leads to changes in the form of urban stream channels. Changes in the morphology of streams due to urban development was initially related to the classic conceptual model of changes in the urban sediment regime (Figure 48) (Wolman, 1967). The model, based on the environmental setting of the eastern United States, depicts how human-induced watershed-scale changes in the sediment regime have led to adjustment of the channel form. Prior to European settlement, the landscape was forested, sediment yields were low, and streams were morphologically stable. As settlement commenced, land was cleared for farming, sediment yields increased, and aggradation (i.e., net deposition) occurred, particularly on floodplains. Implementation of conservation practices in the mid-20th century reduced sediment yields, leading to net erosion of streams. As urban areas expanded into agricultural land, sediment yields increased dramatically during the construction phase of urbanization, producing some aggradation of stream systems, again primarily on floodplains. As urban development progressed, the widespread coverage of land by impervious surfaces reduced sediment yield and increased runoff, resulting in channel enlargement through scour. Thus, according to this conceptual model, morphological change in urban streams typically involves aggradation during the construction phase and erosion during the urbanized phase of urbanization. Since the 1960s, this model has had a large, strong influence on research on morphological change of streams in urban environments, serving as a framework for interpreting results of this research and for refining knowledge of urban impacts on stream geomorphology (Chin, 2006; Chin et al., 2022).

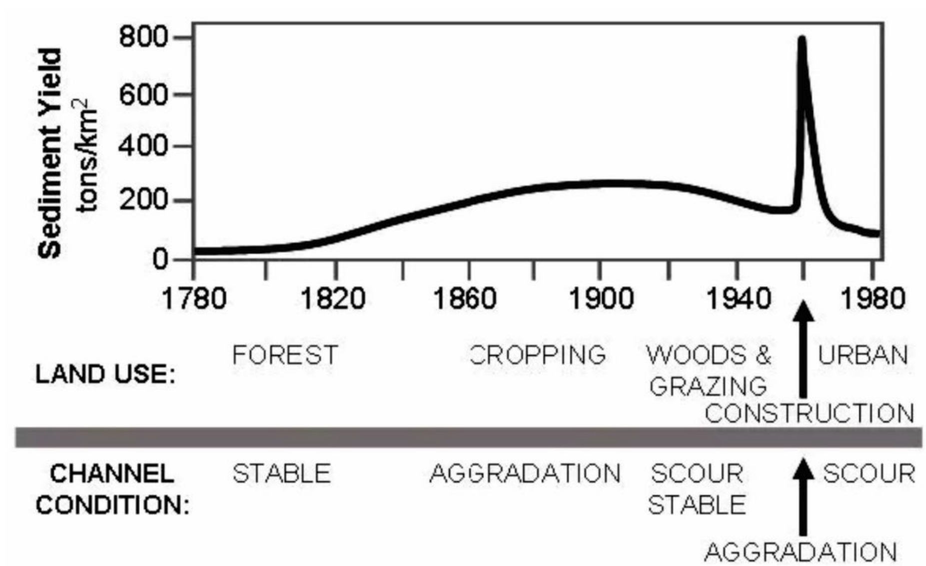


Figure 48. Relation of changes in channel form (channel condition) to changes in sediment regime (from Lord et al., 2009, adapted from Wolman, 1967)

3.5.2 Channel aggradation

Several studies have documented the downstream reduction of channel dimensions induced during the construction phase of urbanization when streams cannot transport the increased sediment load produced from erosion of exposed soil (Ebisemiju, 1989b; Keen-Zebert, 2007; Leopold, 1973; Leopold et al., 2005; Nanson and Young, 1981; Odemerho, 1992; Yousefi et al., 2019). Aggradation can reduce channel capacity, the area of the channel cross section at the top of the channel banks, by 13 to 47% compared to the original or undisturbed state of the stream channel [Table 19, (Ebisemiju, 1989b; Leopold, 1973)].

Forty-one years of monitoring of the 9.5 km^2 Watts Branch in Maryland indicated that during the first 20 years as land surfaces were cleared for construction, increased sediment production from the erosion of soil resulted in deposition along the channel boundaries and development of higher banks, narrower channel widths, and massive point bars (Leopold, 1973; Leopold et al., 2005). Similar adjustments were observed in the headwater streams in the Ado-Ekiti region of southwestern Nigeria based on the analysis of surveyed channel cross sections (Ebisemiju, 1989b). Results revealed that over a one-year period, cross-sectional area decreased by 4% mainly as a result of aggradation of the channel bed (16% decrease in channel depth). Although urban development increased the runoff and peak discharge, excessive sediment production and the stream's incapacity to transport the large amount of sediment delivered to it caused rapid aggradation of the channel bed (Ebisemiju, 1989b). A net decrease in mean depth due to channel bed aggradation, ranging from 17 to 85%, was documented in two urban streams in Fayetteville, Arkansas from cross-sectional measurement over a period of 18 months (Keen-Zebert, 2007). These changes occurred despite the implementation of erosion-control practices on construction sites. Sedimentation and consequential reduction in mean depth was enhanced by the presence of flow obstructions, such as woody debris, bridges, and culverts–a finding supported by results of research on urban streams in Nigeria (Odemerho, 1992). The conceptual model (Figure 48) emphasizes the time-dependency of channel adjustment and the need to consider lags in adjustment in relation to evolving land-use conditions (covered in Section 3.5.6). Downstream decreases in channel depth within streams in northeastern Puerto Rico have been attributed to intensive agricultural land use that produced net deposition within streams. Nevertheless, the aggraded condition has persisted during a transition to urbanization, despite a reduction in sediment supply (Clark and Wilcock, 2000). This state may be transient as accumulated sediment is gradually removed over time; indeed, some local areas of downstream increases in channel size occur where streams traverse heavily urbanized areas without stormwater controls.

In addition to bed aggradation, channel response to excessive sediment production often leads to floodplain deposition. Eroded sediments from upstream are often deposited as floodplain alluvium in response to high runoff events, where vertical accumulation of sediment leads to expansion of the floodplain or creation of new floodplain areas (Graf, 1975). Moreover, aggradation of the channel bed decreases the depth and thus increases the frequency of overbank flow, which causes deposition outside the channel on floodplains (Odemerho, 1992). The presence, connectivity, and hydraulic and geomorphic functionality of adjacent floodplains have been identified as important factors determining the extent of the downstream reduction of channel width or depth in response to increased sediment yield from upstream disturbance

(Nanson and Young, 1981; Odemerho, 1992). Along the Ikpoba River in Nigeria, channel size along an urbanized segment of the river was smaller than channel size upstream and downstream of the urban area (Odemerho, 1992). The availability of extensive floodplains along the channel helped to dissipate the excess energy of the frequent urban high flows by temporarily storing floodwaters and preventing downstream propagation of sedimentation by promoting deposition of sediment on the floodplain.

The extent to which aggradation of river channels during the construction phase of urbanization is a common response remains unclear. Evidence to support this type of response is rather limited. Early studies in the United States (Wolman, 1967; Leopold, 1973; Graf, 1975) or those from international settings (Ebisemiju, 1989b) have examined situations in which modern sediment erosion-control practices were not implemented at construction sites. This generalized response may not occur if such practices are used. Moreover, how streams respond to an increased supply of sediment, even when caused by construction, will depend on whether sediment supply-limited or transport-limited conditions exist in the stream system (Phillips and Scatena, 2013). Under transport-limited conditions, the stream is transporting sediment at maximum capacity, and the amount of transported material is, on average, equal to the amount supplied. Any increase in supply will exceed the maximum transport capacity and cause channel aggradation. In a supply-limited situation, the stream is not transporting at maximum capacity, and the amount of transport is limited by the supply. Thus, the stream has "excess" transport capacity, and the addition of sediment from construction may not necessarily cause aggradation. Most alluvial streams, i.e., those formed in the material they transport, are transport limited, whereas streams bounded by bedrock, concrete, or other high-resistant material (e.g., glacial till) are often supply limited. The size (diameter) of sediment supplied from upstream is another contributing factor that can determine downstream channel aggradation (Clark and Wilcock, 2000). In many cases eroded soil from construction sites consists mainly of fines or silt and clay. In such cases, this fine-grained eroded material supplied to streams will be transported as wash load (i.e., material suspended in the flow that will not deposit on the channel bed unless velocities are equal to or close to zero). Streams have virtually an unlimited capacity to transport wash load, but fine-grained suspended sediment can be deposited on floodplains during floods without affecting the channel bed morphology. Overbank deposition of fine sediment can increase the bank height and subsequently the channel depth. Bed aggradation will likely occur only in response to an extreme increase in the supply of fine sediments. If the increased supply mostly contains coarse bed material, deposition will likely occur on the bed and as channel bars because the material will move as bed-material load (e.g., sediment transported along the bed of the stream or by being temporarily suspended off the bed), rather than as wash load.

Table 19. Summary of Studies on Channel Change Produced by Urbanization

Study location	*Morphologically relevant metrics*	*Direction of change*		*Cause of change*	*Citing literature*
		Qualitative	*Quantitative*		
Watts Branch, Rockville, MD	Channel area (average)		13% decrease	Increased sediment load from urban development	(Leopold, 1973)
Meadow Hills, Southeast Denver	Floodplain area		270% increase	Increased sediment production during construction	(Graf, 1975)
Piedmont province of Baltimore, Maryland-Washington, D.C.	Channel cross-sectional area		2 times increase	Increased magnitude and frequency of flood flows	(Robinson, 1976)
	Width/depth ratio		1.7 times increase		
Illawarra region, New South Wales, Australia	Cross-sectional area		2 to 3 times increase	Channel modification and urban peak runoff	(Nanson and Young, 1981)
Harry's Brook, New Jersey	Widening and degradation	increase		Increased bedload carrying capacity	(Whipple Jr. and DiLouie, 1981)
Sawmill Brook, central Connecticut, USA	Channel pattern	Meandering to braided		Increased frequency of moderate floods + decreased bedload flux	(Arnold et al., 1982)
	Bank erosion	increase			
	Bedload sediment discharge	decrease			
Armidale, New South Wales, Australia	Bank erosion rate		3.6 times greater	Increased total runoff	(Neller, 1988)
	Knickpoint retreat rate		2.4 times greater		
	Channel enlargement		4 times larger (average)	Increased storm runoff	(Neller, 1989)

Table 19. Summary of Studies on Channel Change Produced by Urbanization, Continued

Study location	*Morphologically relevant metrics*	*Direction of change*		*Cause of change*	*Citing literature*
		Qualitative	*Quantitative*		
Avondale stream basin, Harare, Zimbabwe	Drainage density (channel network + stormwater drains)		(0.35-0.80) km/km^{-2} to 3.15 km/km^{-2}	Increased runoff + peak discharge	(Whitlow and Gregory, 1989)
	Channel widening		1.7 times increase		
	Bank erosion		0.33 m/year (average rate)		
Ado-Ekiti, Southwestern Nigeria	Channel capacity		47% smaller	Increased rates of sediment production and delivery to streams	(Ebisemiju, 1989b)
Monks Brook, central southern England	Channel capacity		2 to 2.5 times increase	Urbanization (increase in peak discharge)	(Gregory et al., 1992)
	Widening		Up to 2.2 times increase		
	Bed lowering		0.21 meters (Average)		
Fairmount Park, Philadelphia, Pennsylvania	Pools		31% shallower	Increased runoff	(Pizzuto et al., 2000)
	Median width		26% larger		
	Median sinuosity		8% lower		
Fountain Hills, Arizona	Width-depth ratio	Low (immediately d/s of road crossing) and high (farther d/s, before next road-crossing)		Increased urban runoff	(Chin and Gregory, 2001)

Table 19. Summary of Studies on Channel Change Produced by Urbanization, Continued

Study location	*Morphologically relevant metrics*	*Direction of change*		*Cause of change*	*Citing literature*
		Qualitative	*Quantitative*		
Humid regions, USA	Channel instability	increase		Increased discharge and stream power	(Bledsoe and Watson, 2001)
Scull and Mud Creeks, Fayetteville, AR	Mean depth	decrease		Construction during urban development	(Keen-Zebert, 2007)
Little Lehigh Creek, PA	Width		Average increase of 3.6 ± 0.6 m	Increased peak discharge	(Galster et al., 2008)
Southeastern Coastal Plain streams, NC	Bankfull cross-sectional area		Approximately 1.78 times larger	Increased stormwater runoff associated with increased impervious areas	(O'Driscoll et al., 2009)
Jakarta, Nigeria	Capacity ratio		2.36 times larger	Increased peak discharge	(Nabegu, 2010)
	Sinuosity		Decreased by 68.58%		
	Channel density		Increased by 28.6%		
Southern Ontario, Canada	Bankfull depth and width	Smaller or no significant trend		Increased frequency of bankfull discharge + decreased bed material supply	(Annable et al., 2012)
Southern California	Channel pattern	Single thread to braided		Increased urban flow	(Hawley et al., 2012)
	Channel width		2 to 3-fold increase		

Table 19. Summary of Studies on Channel Change Produced by Urbanization, Continued

Study location	*Morphologically relevant metrics*	*Direction of change*		*Cause of change*	*Citing literature*
		Qualitative	*Quantitative*		
Southern California	Cross-sectional area		14 times increase	Increased cumulative sediment transport capacity	(Hawley and Bledsoe, 2013)
NE Puerto Rico	Average bankfull cross-sectional area		1.5 times larger	Increased peak flow	(Phillips and Scatena, 2013)
Yzeron River, France	Bankfull cross-sectional area		1.8 times larger	Local anthropogenic factors	(Navratil et al., 2013)
	Bankfull width and depth		1.3 times increase		
Northern Kentucky	Bankfull Cross-sectional area		Increased at average rate of 0.075 m^2/year	High energy of urban flow regime causing excess bed material transport	(Hawley et al., 2013)
	Pool length		Increased by 1%/year and deepened by 0.45 cm/year		
	Riffle length		Decreased by 0.15 m/year		
	Median particle size		Increased at average rate 1.7%/year		
Melbourne, Australia	Width-depth ratio	Smaller (incised channels)		Effective imperviousness (EI) leading to increased discharge	(Vietz et al., 2014)
	Bar, benches, and large woody debris	Less common			
Fountain Hills, Arizona	Channel capacity		10 times larger	Increased urban runoff	(Chin et al., 2017)
	Width		3 times increase		
	Depth		7 times deeper		

Table 19. Summary of Studies on Channel Change Produced by Urbanization, Continued

Study location	*Morphologically relevant metrics*	*Direction of change*		*Cause of change*	*Citing literature*
		Qualitative	*Quantitative*		
Wilket Creek, Toronto, Canada	Channel enlargement ratio		Ranging between 2.6 and 8.2	Urban flow regime	(Bevan et al., 2018)
	Enlargement rate		2.3 m^2/year		
	Incision		40-50 cm (average)	Increased flow + increased slope (channelization)	
Los Laureles Canyon (LLCW), Tijuana, Mexico	Cross-sectional area (downstream of hardpoints)		64 times greater	Urbanization and in-channel structures	(Taniguchi et al., 2018)
Talar River, Iran	Width		Decreased by 84%		(Yousefi et al., 2019)
	Length of alluvial bar		Decreased by 70%		
Sand River, Aiken South Carolina	Incision and widening		0.2 m/year	Increased flow energy	(Sullivan et al., 2020)
Northern Kentucky	Incision		Increased from 0.5 cm/year to 1.5 cm/year	Increased runoff	(Hawley et al., 2020)
	Widening		An order of magnitude increase (from 1 cm/year to 9.4 cm/year)		
Karoon River, Iran	Width		17% increase	High discharge from an extreme flood	(Yousefi et al., 2021)
	Depth		25% decrease		
	Thalweg length		Decreased about 126 m		

3.5.3 Channel erosion

Although long-term investigations of the effects of urban development on stream channel adjustment have focused on both aggradation and erosion as embodied in the conceptual model of urban channel response, erosion is widely reported as the predominant morphological response of streams to urbanization. Drawing on the result of 46 studies of the effects of urban development on channel morphology, most (72%) studies documented the enlargement of stream channels (Gregory, 2006). Net erosion, which results in channel enlargement, can occur by two mechanisms: gradual channel expansion (or quasi-equilibrium channel expansion), where both the bed and banks are eroded simultaneously, increasing the size of the bankfull stream channel, and channel incision, where the bed is lowered first, banks become steep and high, and the channel subsequently widens (Booth, 1991; Booth, 1990; Henshaw and Booth, 2000). In cases of incision, the increase in channel size may be much greater than in the cases of expansion.

3.5.3.1 Channel expansion

Channel expansion or quasi-equilibrium expansion occurs when an increase in discharge produces an approximately proportional increase in the bankfull channel cross-sectional area, i.e., part of the increased flow is gradually accommodated by the increase in channel depth and width (Booth, 1990). The terms "expansion" and "enlargement" are frequently used interchangeably, typically indicating an increase in channel cross-sectional area or channel capacity or channel dimensions (depth and width).

Expansion or enlargement is typically viewed as a form of adjustment in which the river system strives to maintain a balanced, or equilibrium, relationship among flow, sediment transport, and channel form. The notion that stream systems adjust to prevailing environmental conditions to attain an equilibrium or a morphologically stable state is a well-established conceptual framework with fluvial geomorphology, but this notion is not without controversy (Rhoads, 2020). The notion often is associated with the related concept of a dominant discharge. This concept proposes that the bankfull dimensions (width and depth) of river systems are related to a discharge of a particular magnitude and frequency that governs these dimensions. The dominant discharge has been equated to the bankfull discharge, which for many perennial river systems in humid-temperate environments has a recurrence interval of about one to two years (Rhoads, 2020). In other words, in many river systems, runoff events with a peak discharge on the order of a 1- to 2-year recurrence interval will fill the stream channel to the top of its banks; these events are viewed as dominant ones governing the size of the channel. Correspondingly, changes in the magnitude of dominant discharges should result in changes in channel size. When urban development occurs, amounts of runoff from storm events, particularly the runoff for more frequent events, will increase, sometimes dramatically. As a result, the magnitudes of peak discharges for events with 1- to 2-year recurrence intervals will also increase. Given that these events represent dominant discharges, the channel dimensions will enlarge through erosion to accommodate this change in hydrological regime.

Numerous studies from around the world (Table 19) have reported the enlargement of urban streams, based on documented changes in different channel metrics, including the bankfull width and depth, width-to-depth ratio, channel capacity or cross-sectional area, and enlargement ratio (ratio of the cross-sectional area of the current channel to that of the reference/pre-urban channel). The cross-sectional area of urban streams is 1.5 to 64 times greater than that of reference or adjacent rural streams (Chin et al., 2017; Gregory et al., 1992; Hawley and Bledsoe, 2013; Nanson and Young, 1981; Navratil et al., 2013; O'Driscoll et al., 2009; Phillips and Scatena, 2013; Robinson, 1976; Taniguchi et al., 2018). The smallest increase in channel size has been documented in humid tropical streams of Puerto Rico (1.5 times increase), where channels prior to urbanization were already strongly affected by frequent high-magnitude storm events (Phillips and Scatena, 2013). Bankfull areas of small streams in a suburban watershed in France were 1.8 times larger than those of adjacent rural streams, whereas bankfull width-depth ratios of the suburban streams were 1.3 times greater than that of the rural streams (Navratil et al., 2013). Both ratios decreased with increasing drainage area, indicating that differences diminished as the effects of urbanization within the urbanized watershed were filtered by inclusion of less intensively urbanized areas. Enlargement ratios for small urban streams (0.11 to 8.63 km^2) in New South Wales, Australia were about 3.77, which also showed a decreasing trend with an increasing drainage area (Neller, 1989). The enlargement ratio of semi-alluvial Wilket Creek (contributing watershed 14.8 km^2) in Toronto, Canada, which has experienced more than 50 years of watershed urbanization, varied between 2.6 to 8.2 when compared with its rural counterparts (Bevan et al., 2018). Channel widening via bank erosion is also a widely reported adjustment process in urban streams undergoing expansion in response to urban-induced increases in peak flows (Chin et al., 2017; Galster et al., 2008; Grable and Harden, 2006; Gregory et al., 1992; Hawley et al., 2012; Hawley et al., 2020; Pizzuto et al., 2000; Whitlow and Gregory, 1989; Yousefi et al., 2021). The width of urban streams is reported to be 1.17 to 3 times larger than the width of nearby non-urban channels (Table 19).

Whether all these reported values of channel enlargement represent pure expansion–the enlargement of the bankfull channel to attain a new equilibrium with changed hydrological conditions–cannot be ascertained conclusively based on the available information. Some of the more extreme values of enlargement probably represent incision, which can produce channels that are not adjusted to the prevailing hydrological regime. Distinctions between expansion and incision, while useful conceptually for ascertaining different mechanisms of enlargement, cannot always be rendered after the fact when channels have already enlarged.

Enlargement is most commonly attributed to increases in the magnitudes of frequent flows (Table 19), but other factors, such as changes in resistance to erosion, can also play a role. Urban development often encroaches on vegetated floodplains along riparian corridors. Through clearing of floodplain vegetation, the stream system is deprived of stabilizing forests that help to dissipate the high energy of increased urban flows by adding roughness to the channel boundary and floodplain (Booth and Jackson, 1997; Finkenbine et al., 2000). High-magnitude urban flows and riparian clearing, either alone or in combination, can enhance susceptibility to erosion, causing increases in channel depth and width (Booth, 1991). Such changes can occur either gradually in response to a gradual increase in flow or abruptly during a single large event (Booth, 1991).

Ultimately, the erosion of streams must occur through the mobilization of sediment, particularly sediment on the bed, which is often linked to bank erosion. The increase in the magnitude of flows with recurrence intervals of one to two years also increases the bedload transport capacity of urban streams, promoting erosion of the channel bed and banks. Modeling of bedload transport capacity for two urbanizing watersheds in New Jersey, USA revealed that transport capacity during a 15-year flood event had been at least doubled from the pre-developed condition, which was evident through widely observed channel widening and deepening (Whipple Jr. and DiLouie, 1981). In semi-arid suburbanizing (10.4% imperviousness) watersheds of southern California, an approximate 360% increase in cumulative sediment transport capacity relative to the pre-developed stage has caused a 14-fold increase in channel cross-sectional area (Hawley and Bledsoe, 2013).

Gullying induced by road crossings and realignment of channels can sometimes initiate enlargement with the potential for enlargement of this type increasing with the increasing channel slope (Neller, 1989). Channel change in urban environments can also be caused by large floods that occur independently of urbanization. An exceptional regional flood along the Karoon River, Iran that exceeded the magnitude of the 100-year flood by a factor of two changed the morphology of a meander loop in the urbanized city of Ahvaz through bank erosion and widening, causing about a 43% increase in the active channel area (Yousefi et al., 2021).

3.5.3.2 Channel incision

Incision is characterized by rapid downcutting of the channel bed that is disproportional to the increased magnitude of discharge (Booth, 1991; Booth, 1990). A channel susceptibility to incision is influenced by flow and sediment parameters as well as channel characteristics including slope and roughness (Booth, 1990). The extent to which the bed is erodible compared to the stream banks as well as the depth of alluvial bed material are also important. Streams with thick layers of highly mobile, non-cohesive bed material and cohesive bank material are particularly prone to incision. Such streams are common throughout the midwestern United States (Simon and Rinaldi, 2000).

Incision is generally initiated when an imbalance is created between the sediment transport capacity of the flow and the amount of sediment the flow can transport (Rhoads, 2020). In urban environments, such conditions commonly develop through the impacts of urban development on flow hydrology, flow hydraulics, and sediment supply. Urbanization increases peak discharges, decreases sediment supply through expansion of impervious surfaces, and often enhances flow velocities, bed shear stresses, and stream power through channelization. These effects increase the sediment transport capacity of the flow, initiating excessive mobilization of bed material and downcutting of the channel bed. The degree of incision in low-order coastal plain streams in North Carolina, USA was found to be positively correlated with the extent of total impervious area (TIA) and stormwater runoff (Hardison et al., 2009). Rapid incision increases bank heights and slopes, leading to bank failure via mass wasting (Simon, 1989), which in turn constantly introduces new sediment to the channel. Although bank failures can help to alleviate the imbalance between transport capacity and sediment influx, they contribute greatly to enlargement of the channel as ongoing bank failures lead to channel widening. Because widening only begins after banks reach a critical height and slope for failure, incision of the channel bed can be considered a process that is intrinsically associated with excess bed-material transport capacity.

Channel slope is an important factor in determining whether incision, once initiated, will continue or cease. Incision, by lowering the elevation of the stream bed, reduces the channel slope, thereby reducing the sediment transport capacity of the flow. This mode of adjustment inherently inhibits the occurrence of further incision. In most cases, incision occurs rapidly at first and then slows as this inhibitory effect becomes important (Simon, 1989). Where the gradient of a stream is already low, an additional reduction in slope through incision will result in a significant reduction in transport capacity, thereby limiting incision. By contrast, rapid incision is most pronounced and sustained in channels with steep gradients. Large woody debris (LWD), by adding substantial hydraulic roughness to the channel, also diminishes a susceptibility to incision. The presence of vertical grade control structures are also documented to influence the extent of channel incision, such as driving incipient lateral channel responses or increasing the incision depth when moving upstream from the hardpoint/control structure (Hawley et al., 2012).

Previous studies have distinguished between a stable and unstable reach by quantitatively assessing different stability indicators (Doyle et al., 2000). Streams that are experiencing little or no erosion of the bed or banks are commonly considered stable, whereas those experiencing severe incision and widening are considered unstable (Booth and Jackson, 1997). Excess shear stress, calculated as the difference between measured bed shear stress and critical shear stress, is a stability indicator metric that can potentially evaluate the impact of urbanization on stream channels (Baker et al., 2008). Excess shear stress was used to indicate the instability of incised urban streams (Doyle et al., 2000; O'Driscoll et al., 2009). When plotted against depth, excess shear stress was found to be greater for the full channel stage (full incised channel) than an estimate of the bankfull stage, indicating less stability of the incised channels (O'Driscoll et al., 2009). When the timing of urbanization varies within the watershed, sediments eroded from upstream incising reaches can be a source of deposition at downstream reaches, leading to unstable upstream and stable or aggrading downstream reaches (Doyle et al., 2000). Thus, channel response (stable or unstable), in addition to degree of urbanization, depends on the timescale of urbanization and the location of the reach of interest in relation to the location of development within the watershed.

Incision rates in urban stream channels of 0.5 cm/year to 0.2 m/year have been reported (Hawley et al., 2020; Sullivan et al., 2020). Sand River, a cohesive ephemeral river in Aiken, South Carolina, experienced rapid channel degradation in response to over 100 years of land use change (Sullivan et al., 2020). Up to 35 m of river incision occurred between 1930 and 1992 in response to a 131% increase in urbanized land within the watershed. The river incised an additional 2.5 meters and widened by 3 meters between 2002 and 2012, indicating that it is still actively adjusting to urbanization despite widespread implementation of stormwater control practices in 1992. This dramatic level of incision has produced a radically altered channel morphology that is still evolving to human disturbance and that does not represent an equilibrium adjustment between flow and form. The incision was caused by increased discharge from the artificial stormwater drainage network that increased flow energy of the river. As a result, flow exceeded the natural transport capacity and initiated incision of the channel bed (Sullivan et al., 2020). In a related study, the channel incision ratio, which is measured as the ratio of the height of the pre-disturbance channel banks (i.e., from the thalweg or deepest point of the channel to the floodplain) to the height of the current bankfull indicator (i.e., from thalweg to the floodplain), varied between 1 and 5.7 for streams in small coastal watersheds (less than 5 km^2) in eastern North Carolina, indicating significant incision of urban streams compared to rural streams (O'Driscoll et al., 2009).

Another way rivers near rapidly growing cities experience incision is when sand and gravel are extracted as valuable resources for urban construction (Arrospide et al., 2018; Yousefi et al., 2019). Downstream from a mining location, the river is often "sediment-starved" so that the enhanced sediment transport capacity induces incision of the riverbed (Rinaldi et al., 2005; Yousefi et al., 2019). Elevation analysis of the Maipo River in the metropolitan region of Santiago, Chile revealed that gravel mining has caused incision of up to 20 m in 31 years (Arrospide et al., 2018). This response has substantially reduced the braided pattern of the river, even transitioning some braided reaches to single threads. The braiding index, a metric of the total length of bars in a reach divided by the total length of the reach, decreased by 70%, from 1.51 in 1955 to 0.46 in 2013 because of sand mining of the alluvial bars in the Talar River, Iran (Yousefi et al., 2019).

Incision, by deepening and enlarging the channel, substantially increases the capacity of the channel to contain floods. As a result, the extent of floodplain inundation is reduced, allowing less deposition of alluvium along the floodplain or valley bottom (O'Driscoll et al., 2009). Thus, incision often reduces the amount of sediment deposited on floodplains. This disconnection of incised streams from floodplains can also cause drops in the riparian water table (Hardison et al., 2009; O'Driscoll et al., 2009). Incised reaches often lack interaction between the stream surface and riparian vegetation near floodplains. Moreover, the degree of incision and its effect on riparian vegetation often interact to influence bank erosion rates. Changes in water-table conditions associated with incision can influence the type of vegetation growing on banks, which can affect resistance to bank erosion. Along an urban stream in Philadelphia, the highest bank erosion rates occurred in incised reaches with high banks where the channel was disconnected from the floodplain and knotweed grew on the banks instead of trees (Arnold and Toran, 2018). Although some work indicates that water quality and physical habitat are degraded in incised streams (Shields et al., 2010), other research suggests that incision alone is not necessarily a sign of ecological impairment (Duncan et al., 2011).

3.5.3.3 Evolution of incised channels

Morphological adjustments in incising channels, whether caused by changes in land use or direct human impacts, such as channelization, often follow a predictable trajectory. The regular sequence of change in such channels has given rise to the concept of channel evolution models or CEMs (Hawley et al., 2012). Although channel response to land-use change associated with urbanization is highly complex, many studies have developed CEMs by simplifying the complex processes into predictable trajectories of morphological changes (Bevan et al., 2018; Booth and Fischenich, 2015; Colosimo and Wilcock, 2007; Hawley et al., 2012; Hawley et al., 2020; James and Lecce, 2013; O'Driscoll et al., 2009). CEMs thus provide a conceptual framework for understanding the observed changes (CEM as a diagnostic tool) and predicting the future changes (CEM as a predictive tool) that may occur in urban streams, which is critical to sustainable management of these evolving systems (Booth and Fischenich, 2015). Most CEMs developed for urban settings draw upon the foundational work on CEMs by Schumm et al. (1984) and Simon (1989) that focused on the sequence of adjustment processes observed in single-thread incising channels. The classic CEM (Schumm et al., 1984) is composed of five stages (Figure 49). The initial stage represents the pre-urban condition with an undisturbed channel that has stable banks and a well-connected floodplain (Stage 1: stable or pre-urban or pre-developed). Following urbanization, incision, or erosion of the channel bed, increasing bank heights and steepness occur (Stage 2: incision or degradation). When the banks become over-steepened and exceed the critical height for bank failure, the channel widens by mass wasting in the form of bank collapse (Stage 3: widening). Channel widening continues until the reduced sediment transport capacity promotes aggradation within the channel (Stage 4: aggradation), leading to the establishment of stable banks along with a stable channel inset within a newly formed floodplain, both of which lie at the bottom of a trench. At the top of the trench is the abandoned former floodplain, now a terrace (Stage 5: re-stabilization or quasi-equilibrium). An additional phase, termed "constructed" is often included after Stage 1, making a total of six stages within the evolution model (Booth and Fischenich, 2015; O'Driscoll et al., 2009; Simon, 1989) (Figure 50). The constructed stage is included when the undisturbed channel is first channelized before it begins to incise. Channelization may itself trigger incision or, if accompanied by land-use change in the form of urbanization, enhance the effects of land-use change.

The basic channel evolution model not only includes a temporal component but also a spatial component (Figure 50, bottom). When considering adjustment spatially, the locus of maximum disturbance is important. In the case of urbanization, this locus may be the core of urban development along the stream, perhaps accompanied by channel modification in the form of channelization. Usually, incision is initiated close to the locus of maximum disturbance and, if erosion of the bed is not constrained by some sort of inerodible material, progresses upstream from this location in the form of a migrating headcut (an abrupt vertical drop in the bed elevation) or knickpoint (a locally steep section of the channel bed). The depth of incision usually diminishes as the headcut/knickpoint migrates headward. The net effect of this adjustment process is to lower both the elevation of the channel bed and the channel slope. Moreover, sediment excavated from the zone of erosion at and upstream of the location of incision is transported downstream, resulting in net deposition when the delivery of this material to undisturbed reaches downstream of the incision zone exceeds the transport capacity of these reaches. The combination of erosion upstream and deposition downstream contributes to the overall decrease in channel slope. At any point in time, the different temporal stages can be identified spatially along the length of the adjustment reach (Figure 50, bottom). Those farthest downstream may have already completed adjustment through Stage VI and reach a quasi-equilibrium condition, whereas those farthest upstream may have yet to be affected by adjustment. In between, the channel system is undergoing major incisional changes.

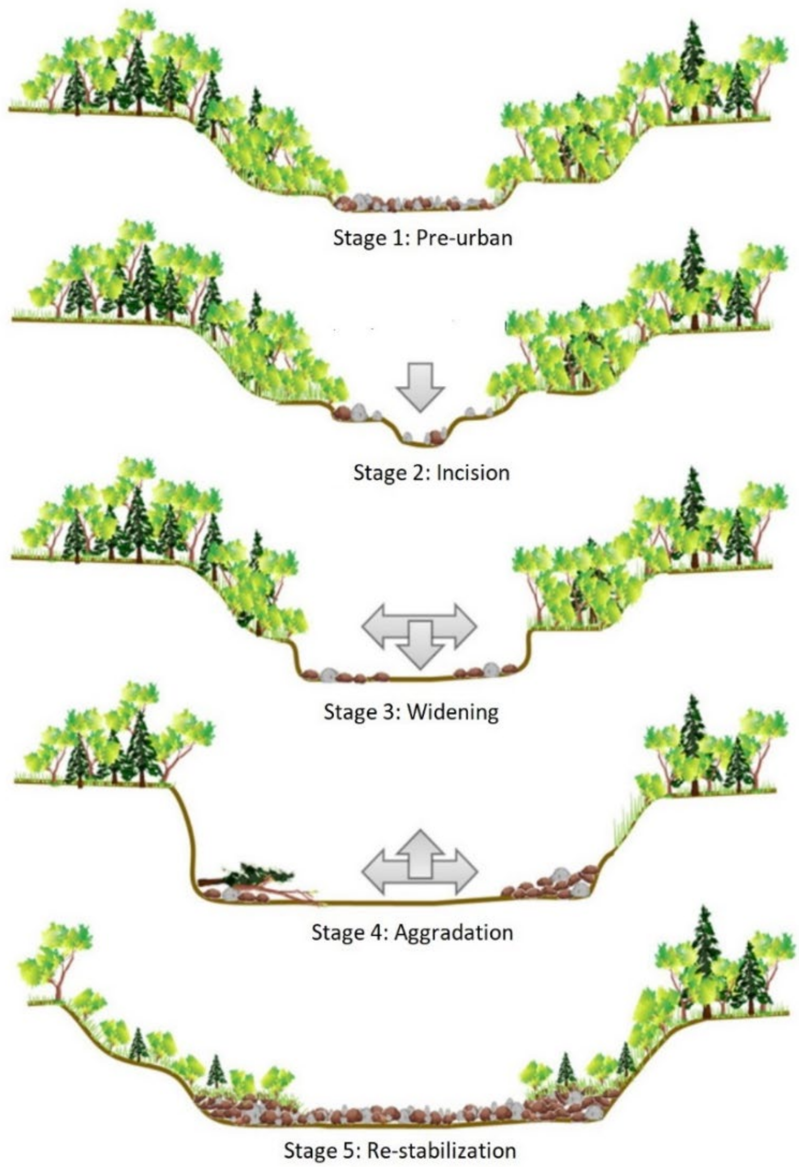


Figure 49. The classic CEM showing the five stages of channel adjustment (from Hawley et al., 2020, modified from Schumm, 1984)

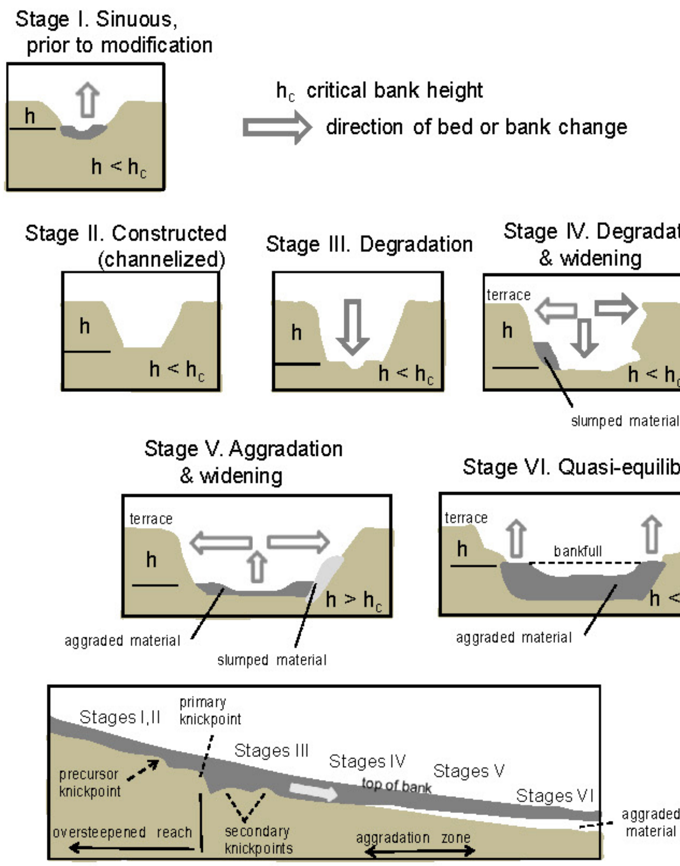


Figure 50. Channel evolution model showing constructed (channelized) stage (II), erosion upstream, aggradation downstream, and the spatial distribution of the different stages along the length of the affected stream at a particular point in time of evolutionary recovery. h in the model refers to bank height and h_c is critical height for failure (from Wohl et al. 2016, modified from Simon, 1989).

3.5.3.4 Urban channel evolution

The classic CEM (Figure 49 and Figure 50) assumes that incision occurs for a single-thread alluvial channel and does not take into account possible differences in adjustment processes in different environmental settings, including urban settings, nor does it consider potential disturbances to adjustments in urban streams. As a result, not all incising urban streams necessarily follow the sequence of adjustment processes described in the classic CEM. Work in urban fluvial geomorphology has used the classic CEM as a benchmark against which actual adjustments can be compared to more accurately understand urban channel evolution.

Analysis of time-series survey data revealed that the evolution trajectory of gravel/cobble bed streams in northern Kentucky subjected to suburbanization is mostly consistent with the classic CEM (Schumm et al., 1984), except that incision (Stage 2, Figure 49) is accompanied by streambed coarsening (Hawley et al., 2020). During the early phases of channel adjustment, the excess sediment transport capacity of the increased urban flow leads to gradual coarsening of the channel bed through winnowing of fine material (Hawley et al., 2013; Hawley et al., 2020). The bankfull depth and median bed particle size of suburban streams increased by approximately 121% and 29%, respectively. The period of bed coarsening and incision was followed by widening and sedimentation (Stages 3 and Stage 4). In addition to mass failure of banks, widening of bedrock substrate reaches occurred because the excess erosive power of the flow eroded the banks rather than the bed. Although most reaches were still evolving, one site approached the re-stabilization state (Stage 5) and was possibly adjusting to the altered catchment hydrology because of an upstream stormwater retrofit. This response illustrates the effect of a resistance substrate on erosional adjustment.

A CEM for semi-arid channels in southern California showed both similarities and substantial departures from the classic CEM (Hawley et al., 2012). A major departure involved a change in planform from a single-thread channel to a multithread braided channel, instead of adjusting to the altered flow regime solely as a single-thread channel. The shift in channel pattern was induced by both lateral erosion and incision. A braided planform can develop following lateral erosion with only minor incision or after major incision. In both cases, channel erosion at upstream locations induced by increases in urban flow increase the supply of sediment downstream and initiate braiding via central bar formation, local aggradation, or channel widening through bank erosion. These braided systems experienced about a two- to three-fold increase in active channel width compared to reference streams. The presence of relatively erodible banks and downstream grade control structures or hard points, which restrict vertical incision, are important factors for driving the laterally based trajectories. Braided states, once developed, continue to evolve through adjustments involving incision, widening, and aggradation. These states do not represent static endpoints of a channel evolutionary sequence.

A regional CEM based on the evolution of Wilket Creek, an urban stream in Toronto, Canada has been developed to address the role of glacial materials and till exposure in channel adjustment to urbanization (Bevan et al., 2018) (Figure 51). The exposure of till is common in streams in Illinois, including the Chicago region. Channel evolution in Wilket Creek following urbanization is influenced strongly by a geologic control point that is marked by a local convexity in the longitudinal profile of the stream. This convexity represents a local increase in channel slope. It occurs at the transition from lacustrine deposits formed by an ancestral glacial lake and a valley complex downstream of these deposits that includes coarse sediment (boulders > 1 m diameter) embedded in glacial till as well as interbedded silts and sands. The bed of the creek at the control point is characterized by abundant coarse particles. The response of the stream to urbanization differs upstream and downstream of the control point. Upstream, channel adjustments followed the classic CEM (Schumm et al., 1984) (Figure 49), but incision is limited by the presence of the control point, which acts as a control on erosion of the channel bed. Downstream of the control point channel, evolution involves five phases conditioned by the glacial control (Figure 51). Following urbanization, the substrate of the pre-urban channel (Stage 1) coarsened, the channel widened through bank erosion, and meanders along the creek extended in response to urban-induced increases in runoff and peak discharge (Stage 2). Extreme meander extensions led to channel avulsions in the form of cutoffs of highly sinuous meander loops. Avulsions through cutoff involve sudden shifts in the course of the stream to a new path while abandoning the old path (Stage 3). The high stream power of flows within the steepened channels produced by avulsion initiates incision, which progresses upstream as a migrating knickpoint (Stage 4). Following incision, the channel continues to enlarge through widening as flow within the relatively straight channel is directed locally into the banks, forcing dramatic erosion and removal of riparian forest (Stage 5). During this last stage, the abandoned channels may become almost completely infilled with sediment. Overall, the adjustment-produced channel-enlargement ratios (post-urban cross-sectional area versus pre-urban cross-sectional area) are as high as 8.2.

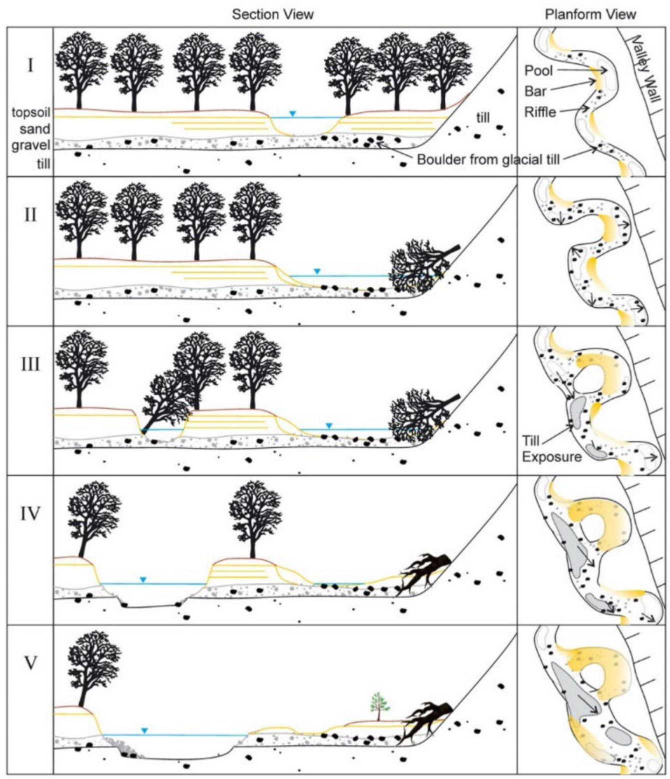


Figure 51. Channel evolution model showing stages of adjustment in Wilket Creek, Ontario downstream of a geologic control point caused by a change in glacial materials within the valley of the creek (from Bevan et al., 2018)

Response of low-order urban streams in the piedmont of North Carolina, USA was explained based on the trend found in three morphological variables – maximum width (W_{max}), maximum depth (D_{max}), and the ratio of maximum width and maximum depth (W_{max}/D_{max}) that best correlated with the age of urbanization (Johnson and Royall, 2019). Data analysis showed an increasing trend in the W_{max}/D_{max} ratio over time, which was largely driven by the decrease in D_{max}. Reworking the classic CEM and following the trend in these morphological variables, the adjustment sequence for southern piedmont streams is explained with a simplified three-phase model. Most of the small streams in the study area are characterized by aggradation from upland agricultural erosion (prior to urbanization) in addition to urban construction erosion. These streams are also underlain by bedrock. The streams first incise into the aggraded bed sediments, resulting in an increase in D_{max} (Phase 1). After that, widening (Phase 2) through bank erosion occurs as incision becomes restricted because of the presence of underlying bedrock, which prevents banks from reaching a critical height for mass failure. As widening continues, decreased sediment transport capacity leads to accumulation of the eroded coarse bank materials within the bed, which results in a gradual decrease in D_{max} and a major increase in the W_{max}/D_{max} ratio. Slow widening (gradual increase in W_{max}) and aggradation (decreasing D_{max}) of coarse sediment in the bed continues, leading to a gradual increase in the W_{max}/D_{max} ratio and the establishment of a new quasi-equilibrium state (Phase 3).

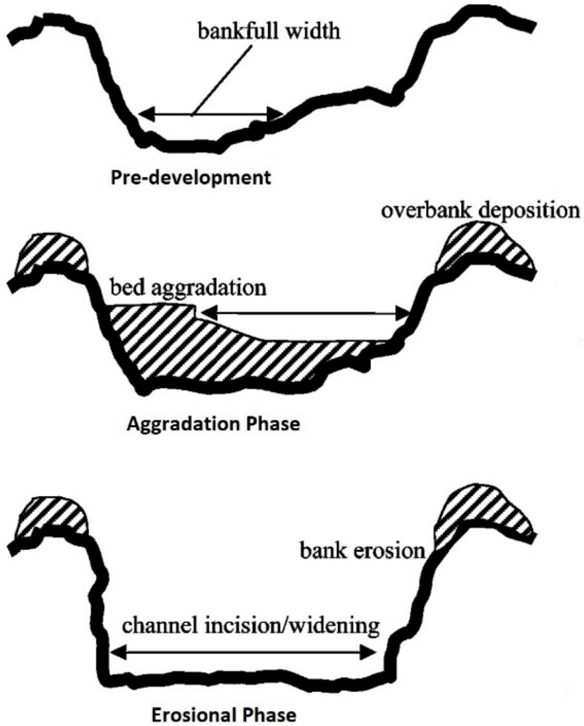


Figure 52. Urban channel evolution model based on Wolman's (1967) conceptual model (Figure 1). Channel evolution model from Paul and Meyer (2001).

Although not strictly related to incision, urban channel evolution is sometimes recognized through the conceptual model of Wolman (1967) that described channel adjustments based on the urban-induced changes in water and sediment input into the watershed (Figure 52). Similar to the classic CEM, Stage 1 can be recognized as a stable stream. Aggradation from increased sediment yield from urban construction defines Stage 2, and erosion resulting from increasing urban runoff and declining sediment yield defines Stage 3. Refined from the Wolman (1967) model, a CEM has been developed that categorizes the adjustment processes into three phases – an aggraded phase (similar to (Wolman, 1967) Stage 2) and two erosion phases (modified from Wolman (1967) Stage 3): early erosion and late erosion (Colosimo and Wilcock, 2007). The evolution model was used to describe the channel adjustments occurring in the urbanizing Gwynns Falls, Maryland watershed based on the sediment stored in the channel. In this model, the aggraded stage is characterized by lateral and point bars formed mainly by deposition of fine sediments. Aggradation during this stage can also be caused by deposition of sediment eroded from upstream channel enlargement. The early erosion stage is characterized by an increase in cross-sectional area, lack of fine sediment, and bars that are smaller than those observed in the aggraded stage. In the late erosion stage, fine sediments are completely removed, and channels are significantly larger than in the previous two stages. Degree, timing, and location of urban development and the presence of grade control structures produced variability in these channel adjustments.

Drawing upon the importance of including local conditions in addition to the implicit assumptions of the classic CEM, a comprehensive CEM has been developed that emphasizes a range of potential disturbances characteristic of an urban stream and its contributing watershed (Booth and Fischenich, 2015) (Figure 53, Table 20). The sequence of channel responses has been described based on direct channel modifications (e.g., channel straightening, channel confinement via levee building, bank and bed armoring, removal of large woody debris and riparian vegetations), and watershed-scale modifications (e.g., changes to water and sediment input). The evolution of an urban channel can be influenced by differences in regional factors, such as climate, watershed relief, rate of sediment delivery, as well as by watershed factors, such as local channel slope, presence or absence of floodplain, in-channel woody debris, and bank vegetation (Booth et al., 2015). Overall, the contribution of these regional and local differences to channel response is usually minor relative to the urban-induced impacts to the channel and watershed (Booth and Fischenich, 2015). Moreover, urban stream burial (by limiting the sediment supply) and infrastructures or road crossings (by limiting the upstream migration of knickpoint caused by incision) may act as important factors to consider in urban channel evolution (O'Driscoll et al., 2009). As urbanization is often an ongoing process, continuous change in land use can reinitiate channel evolution, causing variability in the evolution phases (Simon and Rinaldi, 2000).

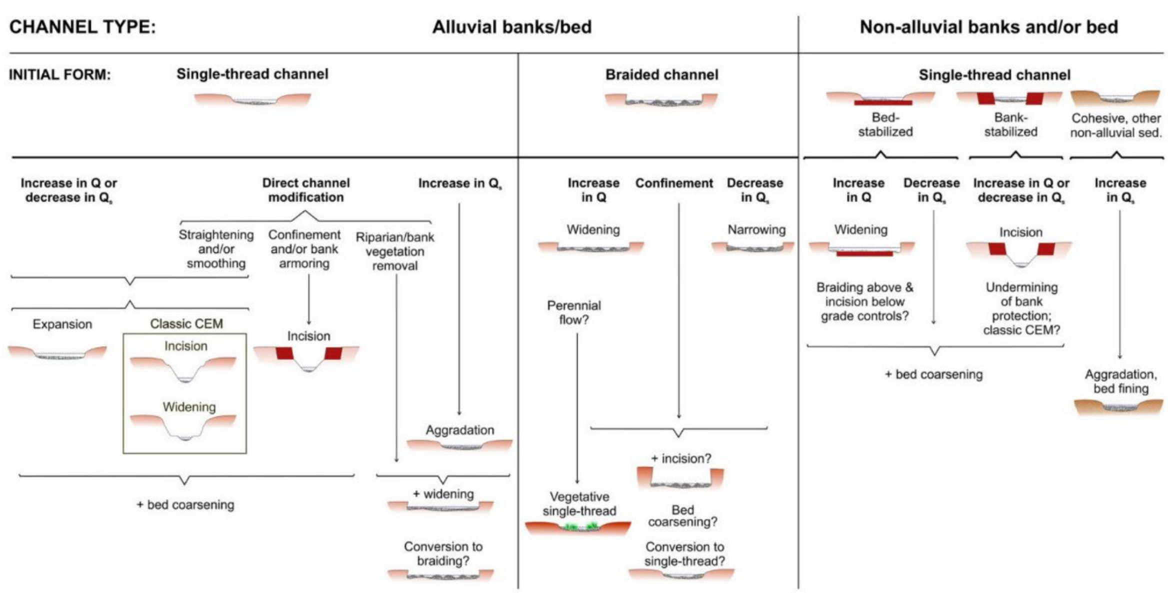


Figure 53. Comprehensive CEM depicting urban channel adjustment according to channel type and type of disturbance (from Booth and Fischenisch, 2015)

Table 20. Urban Channel Adjustment According to Channel Type and Type of Disturbance (From Booth and Fischenisch, 2015)

Channel type	*Anticipated urban disturbance(s)*	*Primary response; other responses/considerations*	
Alluvial, single-thread	Increased Q or decreased Q_s; straightening, reduced roughness	If low-magnitude disturbance: expansion (in both width and depth)	
		If high-magnitude disturbance: incised-channel (classic) CEM, bed coarsening	
	Riparian vegetation removal	Widening	Potential conversion to multi-thread
	Confinement	Incision, bed coarsening	
	Increased Q_s	Aggradation, widening, bed fining	Potential conversion to multi-thread
Alluvial, braided	Increased Q	Incision and/or widening	Potential conversion to single thread; bed coarsening
	Decreased Q_s	Narrowing and/or incision	
	Confinement	Incision	
Non-alluvial: bed-stabilised	Increased Q	Widening, bed coarsening	Braiding/incision may occur above/below intermittent grade control(s)
	decreased Q_s	Bed coarsening	Incision may occur below grade control(s)
Non-alluvial: bank-stabilised	Increased Q or decreased Q_s	Incision, bed coarsening	Progression of classic CEM depends on magnitude of incision relative to bank protection
Non-alluvial (any)	Increased Q_s	Aggradation, bed fining	

Notes: 'Alluvial' channels are those able to adjust their form and size both laterally and vertically; 'non-alluvial' channels are constrained by natural or artificial structures in one or more dimensions. See Figure 5 for a graphical representation of this application. CEM = channel evolution model; 'Q' = discharge; 'Q_s' = sediment discharge

According to the comprehensive model, expansion and incision occur in alluvial, single-thread channels in response to increases in discharge (Q) and/or decreases in sediment load (Q_s). Expansion is more typical of modest changes in Q and Q_s less severe disturbance, whereas incision that conforms with the classic CEM response is more common for large changes in Q and Q_s. Expansion and incision may be accompanied by coarsening of the bed material in gravel-bed streams. If the channel is confined artificially, e.g., by riprap or sheet piling, incision and bed coarsening will occur, but widening will be constrained. Removal of the riparian vegetation with substantial changes in Q and Q_s can lead to channel widening through bank erosion. Aggradation, widening, and fining of bed material often accompany large increases in sediment load typical of the construction phase of urbanization. If widening by vegetation removal or aggradation is pronounced, the channel may become braided (i.e., multithread). The model also includes responses of braided streams to urbanization, which can potentially be converted to single-thread channels through incision or narrowing. In single-thread channels with inerodible beds, widening and bed coarsening are the primary responses to increases in *Q*, whereas incision and bed coarsening without widening is the primary response if the channel banks are inerodible.

3.5.4 Relationship of channel change to driving factors of change

Although urban-induced changes in land use are the driving forces for modifying the morphology of stream channels, not all channels are susceptible to change. Moreover, local factors can influence channel response, and despite attempts to develop general models of response (e.g., Figure 53, Table 20), responses can vary widely, not only from one geographic region to another, but also locally. Studies across the world have evaluated the magnitude, rates, and causes of channel change (Table 19) resulting from urban development, but relationships between measured channel change and channel or watershed characteristics are difficult to generalize. Several factors, such as slope, position of the stream relative to the watershed, timing and degree of urban development, riparian vegetation, lithology of bed and bank, and sediment transport characteristics determine channel response (aggradation or degradation) to watershed urbanization (Bledsoe and Watson, 2001; Fitzpatrick et al., 2005).

3.5.4.1 Altered hydrology and sediment transport capacity

Urban-induced change in the hydrologic regime manifested as an increase in stormwater runoff and peak discharge is the most fundamental driving factor for triggering change in channel dimensions, specifically channel enlargement (Abali et al., 2021; Arnold et al., 1982; Bevan et al., 2018; Chin et al., 2017; Galster et al., 2008; Gellis et al., 2017; Gregory, 1976; Gregory et al., 1992; Hawley et al., 2012; Hawley et al., 2020; Hollis and Luckett, 1976; Jeje and Ikeazota, 2002; Nanson and Young, 1981; Neller, 1988; Neller, 1989; O'Driscoll et al., 2009; Phillips and Scatena, 2013; Pizzuto et al., 2000; Robinson, 1976; Russell et al., 2020; Sullivan et al., 2020; Vietz et al., 2014; Whitlow and Gregory, 1989). An examination of the effect of hydrology on channel form in 16 gravel-bed streams in Puget Lowland, Washington revealed that urban development disproportionately increased the frequency but decreased the duration of high flows (Konrad et al., 2005). In other words, the hydrological response became more flashy. This change in hydrological regime was captured by the metric $T_{0.5}$, the cumulative fraction of time that stream flow equaled or exceeded the peak of a 0.5-year flood event ($Q_{0.5}$), i.e., the peak streamflow exceeded on average twice per year. $T_{0.5}$ varied among streams from 0.002 to 0.004 at the highest levels of urban development to around 0.03 at the lowest levels, indicating the brief duration of frequent high flow events in urban streams. On the other hand, the magnitude of the dimensionless bed shear stress for $Q_{0.5}$, a metric of bed mobility, was inversely related to $T_{0.5}$, indicating that bed disturbance by $Q_{0.5}$ is greatest for streams with low values of $T_{0.5}$. Thus, flashy urban streams are prone to high levels of bed disturbance. On the other hand, stormwater detention structures, built on a “peak matching” strategy, typically lead to stream erosion because the increased duration of erosive flows exceeds the frequency and duration of critical shear stress for bed particle entrainment (Bledsoe, 2002). Thus, the evidence is somewhat contradictory; some studies indicate that increased peak flows from increased flashiness are more likely to cause stream erosion than prolonged flow durations, whereas other studies indicate that prolonged flow durations associated with reductions of peak flows enhance erosion. Ultimately, enlargement through erosion typically occurs whenever the sediment transport capacity of the flow is increased relative to its pre-developed or reference conditions (Hawley and Bledsoe, 2013; Hawley et al., 2013).

3.5.4.2 Channel response and impervious area or degree of development/urbanization

Numerous studies have linked observed channel responses to watershed imperviousness. In particular, erosion is largely driven by impervious surface cover and its influence on stormwater runoff. Early work on this problem showed that channel enlargement in particular is strongly affected by impervious surface cover directly connected to the storm sewer network, which delivers stormwater runoff directly to streams (Hammer, 1972). Regression analysis based on 40 stream sites in northern Kentucky showed a strong correlation between channel instability, defined as the percentage increase in bankfull cross-sectional area, and the percentage of impervious cover (Hawley et al., 2013). A study of enlargement ratio for 59 stream reaches in a West Sussex study in England revealed that a 10% increase in impervious area had increased the downstream channel size by about 1.7 times (Hollis and Luckett, 1976). The TIA explained 65–72% of channel enlargement of southeastern coastal plain streams in North Carolina (O'Driscoll et al., 2009). Watershed (geology, soils, precipitation) and channel conditions (slope, particle size) aside, impervious cover was found to be as the only statistically significant predictor of change in cross-sectional area of semi-arid channels of southern California (Taniguchi and Biggs, 2015). In the Piedmont streams of North Carolina, those in watersheds with greater than a 10% impervious surface cover have substantially larger cross-sectional areas, bankfull widths, and bankfull depths than those in rural watersheds (Doll et al., 2002).

Overall, assessments of the influence of impervious surface cover on channel response indicate that net erosion (i.e., enlargement) can occur when the total impervious surface cover exceeds 10 to 20% of the total watershed area (Bledsoe and Watson, 2001; Chin, 2006). Streams in humid U.S. regions displayed substantial instability due to increased stream power inducing from low levels of imperviousness (10 to 20%) (Bledsoe and Watson, 2001). Streams in semi-arid environments, which are inherently more dynamic morphologically than those in humid-temperate environments, may be more sensitive to urban-induced changes in surface runoff. Approximately 2 to 10% watershed imperviousness was sufficient to initiate incision and braiding in semi-arid streams of southern California (Hawley et al., 2012).

Building on the early findings of Hammer (1972), the importance of effective impervious surface cover, the proportion of impervious cover directly connected to streams through stormwater drainage systems, rather than the total impervious area has been emphasized in some studies. Substantial channel degradation of lowland streams in western Washington occurred at low levels of imperviousness, approximately 10% effective impervious area (EIA) (Booth and Jackson, 1997). Effective imperviousness (EI) was found to be a better predictor than TIA of substantial geomorphic changes at 17 stream sites in Australia (Vietz et al., 2014). At these sites, detectable changes occurred for watersheds with EIAs as small as 2 to 3% (Vietz et al., 2014).

Other metrics related to urban development, such as age, location, time, and extent of development, have also been related to channel instability. In southwestern Nigeria, channel response varied directly with the intensity of urban development and with the location of urban development within the watershed (Ebisemiju, 1989a; Ebisemiju, 1989b). The physical condition of a stream is approximately equally affected if the extent of urbanization extends across the entire watershed of the stream or is concentrated only on part of the watershed close to the stream (McBride and Booth, 2005).

Despite the common association of channel erosion with increasing urbanization, some studies have not found a strong link between channel conditions and urban development (Kang and Marston, 2006; Kang et al., 2010; Navratil et al., 2013). In particular, several investigations have shown that imperviousness is not significantly correlated to channel change (Finkenbine et al., 2000; Cianfrani et al., 2006; Nabegu, 2014; Ramírez et al., 2009; Taniguchi et al., 2018). An analysis of 21 urban and suburban streams in western Washington watersheds (0.1 to 20 km^2) yielded no significant relationship between the magnitude of channel change and channel topography or watershed condition, such as channel gradient, and degree of urban development measured as an effective impervious area (EIA) (Booth and Henshaw, 2001). Small basins (draining few tens of hectares area) displayed the most dramatic downstream effects at relatively low levels of development, particularly where discharge is locally increased because of flow concentration from road crossings or ditches. Dimensions of urban streams in southern Ontario, Canada actually decreased with increasing urbanization (Annable et al., 2012). Monitoring of 12 gravel-bed streams in urban and urbanizing watersheds over a 15-year period revealed that increasing urban development has increased the frequency of bankfull discharge events, has not affected the total volume of runoff, and has decreased the bed-material sediment supply. No significant differences exist between the bankfull width and depth of urban versus rural streams. The urban streams are well connected to adjacent floodplains, which allows high flows to spill onto floodplains, thereby reducing velocities and buffering the stream systems against erosion.

3.5.4.3 Variations in bank/bed material and riparian vegetation

Variations in the lithology of watersheds can alter the channel slope, properties of beds and materials, and vegetation growth, which in turn can influence the channel response. Analysis of chalk and shale watersheds in Dallas, Texas showed that chalk channels, in response to urbanization, are more susceptible (12–67% greater) to erosion and enlargement than shale channels (Allen and Narramore, 1985). The susceptibility of urban streams in small watersheds (< 10 sq mi) in northern Texas to erosion was also associated with the nature of bed and bank materials, which varied from alluvial to bedrock to mixed alluvial and bedrock (Allen et al., 2002). Loose erodible materials were among one of the various factors responsible for the increased dimensions of the urban channel in southeastern Nigeria (Jeje and Ikeazota, 2002). In southern California, sand-bedded streams experienced incision and enlargement (enlargement ratio up to 115), whereas gravel-bedded streams experienced widening and were less enlarged (enlargement ratio less than 7) (Taniguchi and Biggs, 2015). In western Washington, rates of morphological adjustments for a given degree of urbanization were greater for channels having granular sandy bed materials than for those formed in cohesive silt and clay (Booth and Henshaw, 2001). Bedrock channels with coarse bed and bank materials can act as a geologic control point, stabilizing channels locally despite decades of urbanization (Nelson et al., 2006).

Riparian buffer zones can substantially influence channel responses to urbanization (Cianfrani et al., 2006). Urban riparian corridors with diverse forest vegetation along the channel banks help to stabilize these banks against erosion (Keen-Zebert, 2007). Where vegetation is absent or sparse, banks are often highly erodible, leading to channel instability (Taniguchi et al., 2018). A study of 45 streams in the Chicago area revealed that the presence of a riparian buffer zone had no attenuating impact on urban stream stability because in many cases storm drains and outlets from detention structures directly bypass the riparian corridors (Fitzpatrick et al., 2005).

3.5.4.4 Structures on channel beds or banks

Local structures such as hardpoints, grade controls, culverts, stormwater pipes/drains, road crossings, and bridges can in some cases have strong local effects on channel stability. Erosion of streams in the rapidly urbanizing watershed of Los Laureles Canyon, Mexico, some of which had channel cross-sectional areas up to 64 times larger than those for reference streams (Taniguchi et al., 2018), eroded more severely downstream of local channel hardpoints (concrete flumes and culverts). The proximity to channel hardpoints also affects the evolutionary trajectory of urban streams. Channel enlargement through increased incision depth can occur upstream from channel hardpoints, such as artificial (e.g., concrete or riprap) or natural (e.g., bedrock) grade controls (Hawley and Bledsoe, 2013; Hawley et al., 2012). Bedrock outcrops act as natural grade control structures that enhance resistance to incision and facilitate in-channel aggradation (Colosimo and Wilcock, 2007). Artificial straightening and deepening of urban rivers increases channel slope and flow depths, increasing stream power and triggering erosion within oversized, straight channels (Brookes et al., 2005). Increased channel dimensions caused by scouring downstream of stormwater outfall sites is common in urban streams (Gregory, 2006; O'Driscoll et al., 2009). Plunge pools or scour holes are also common at transitions between elevated pipe outlets and streams and at road-crossing culverts (Allan and Estes, 2005). Bridges can also cause localized constrictions that promote downstream scour, including enlarged channels (capacity increased up to four times to that of the bridge opening) (Douglas, 1985). Road crossings can further degrade urban streams by becoming point sources of stormwater discharge into streams, thereby contributing locally to channel erosion (McBride and Booth, 2005). Thus, proximity to road crossings, road sewers, or storm sewers can be a major factor influencing morphological adjustments in small urban streams (< 5 km^2) (Navratil et al., 2013). Road crossings fragment stream channel adjustments, often causing adjustments that produce deeper and narrower channels downstream of crossings compared to upstream (Chin et al., 2017; Chin and Gregory, 2001). Other infrastructures, such as drop structures or sedimentation ponds, can trap sediment, thereby starving downstream locations of bed material and increasing downstream channel degradation (Jordan et al., 2010).

3.5.4.5 Climate change

Changes in climate can alter the flow and sediment regimes of streams, which in turn can either amplify or ameliorate the effects typically associated with the urban stream syndrome (Hale et al., 2016). As climate change can lead to non-stationarity in hydrologic conditions, it adds a confounding factor to the understanding of urban channel adjustments (Gregory, 2006). Given that changes in climate, even those induced by humans, generally occur over years, if not decades, and that the geomorphic response of streams to changes in climate occurs over a similar timescale, attempts to understand the impacts of future climate change on urban stream adjustments has only begun. As a result, evidence of potential effects of such adjustments, independent of other effects, has yet to emerge.

3.5.5 Other geomorphic responses

Besides incision and enlargement, changes in water and sediment input into streams associated with urban-induced changes in land cover can yield a broad spectrum of geomorphic responses (Nabegu, 2014; Paul and Meyer, 2001). Alteration in drainage basin hydrology associated with urban development can cause a change in the drainage network, channel pattern and channel geometry (Gregory, 1976).

3.5.5.1 Change in planform

In cases where the vertical adjustment of streams through incision or enlargement is limited, a stream may change its planform to accommodate changes in flow and sediment regime. Rapid geomorphic change in response to urbanization along Sawmill Brook, Connecticut caused this stream to change from a meandering to a braided planform (Arnold et al., 1982). High rates of bank erosion and lateral migration caused by increased high flows from urbanization introduced a large amount of sediment in the stream, increasing the bedload sediment flux. As a result, the channel bed aggraded by developing mid-channel bars. The bars further accelerated bank erosion by deflecting flow against the banks, eventually establishing a braided pattern. Similarly, in southern California, 7 out of 33 study reaches shifted from a predominantly single-thread channel to a fully braided state following urbanization (Hawley et al., 2012). The shift in planform was attributed to increased peak flows that caused incision and widening and increased the sediment supply to downstream reaches, resulting in central bar formation and bed aggradation.

3.5.5.2 Channel shape

Many natural streams exhibit trapezoidal or U-shaped channel cross sections with gently sloping banks. Scouring of urban streams often produces channels of rectangular shape, characterized by steep, nearly vertical banks and a relatively flat bed (Yorke and Herb, 1978). Channel shape, measured as the ratio of total width to maximum depth of the channel, can reflect the effects of urbanization (Keen-Zebert, 2007). Measured cross sections of rural channels in Arkansas have a parabolic or U-shaped channel form, whereas urban channels have a rectangular form. The rectangular shape is most pronounced in streams where urbanization has historically been the dominant land use, reflecting the scoured nature of these channels.

3.5.5.3 Physical attributes of channels: pool, riffles, bars, large wood, sediment grain size

Urbanization not only alters the channel dimensions, but also modifies the physical attributes of streams that are critical to the sustenance of the aquatic ecosystem. The geomorphology and ecology of urban streams are linked to effectively address the relationship between physical habitat and geomorphic characteristics (Gregory, 2011). Urban rivers are more homogeneous than reference or less-disturbed streams in terms of morphological attributes and functionality (Booth et al., 2015). Channelized urban streams that have been deepened for flood control and armored for erosion control typically are disconnected from floodplains. As a result, these channels have less large woody debris, poorly developed or widely spaced pools, and less sediment storage in the form of bars, exhibiting more simplified morphologies (Segura and Booth, 2010).

Geomorphic analysis of urban streams in northern Kentucky indicated that stream riffle lengths have shortened and pool lengths and depths have increased (Hawley et al., 2013). This change has been caused by the upstream migration of a series of headcuts initiated by the urban flow regime and even occurs in reaches with prevalent grade control structures. For every 1% of impervious cover in the watershed, the average rate of decrease in riffle length was 0.15 m/yr, the average rate of increase in pool length was approximately 1%/yr, and the average rate of increase in pool depths was about 0.45 cm/yr (Hawley et al., 2013). By contrast, pool depths decreased by 31% in gravel-bed urban streams in southeastern Pennsylvania (Pizzuto et al., 2000).

An investigation of 17 urban streams in Australia with varying degrees of watershed urbanization has revealed that bars and benches formed by the deposition of bedload and suspended load, respectively, are now less common (Vietz et al., 2014). Bars also generally are not observed in urban streams undergoing erosion (Colosimo and Wilcock, 2007). Moreover, incision due to sand and gravel mining for urban construction has substantially reduced the size of channel bars in the Talar River, Iran (Yousefi et al., 2019). However, the introduction of extensive sand and silt during the construction phase of urbanization can lead to an increase in sand bars and sand dunes (Chin, 2006).

Most urban streams do not contain abundant large wood compared to rural streams (Booth and Jackson, 1997; Chin, 2006; Finkenbine et al., 2000; O'Driscoll et al., 2009; Vietz et al., 2014). Wood, if it does enter urban streams, is often viewed as an obstacle to flow and may be intentionally removed to clear the channel of woody debris. The absence of wood reduces hydraulic roughness, which can enhance flow velocities and stream power, thereby promoting erosion and channel enlargement.

Bed material coarsening, particularly in gravel-bed streams, is often an initial response to increased flows caused by urbanization (Hawley et al., 2013). The median particle size of bed material in urban streams in northern Kentucky increased 1.7%/yr for every 1% increase in impervious cover (Hawley et al., 2013). If available, urban channels typically contain more coarse particles (gravel) than rural streams (Finkenbine et al., 2000; O'Driscoll et al., 2009). High-magnitude floods have been observed to change the size of bed material by transporting large angular rocks exposed at upstream construction sites (Leopold et al., 2005). The size of riffle particles were significantly larger in urban piedmont streams than in their rural counterparts, but this size difference of riffle substrate was not observed in coastal plain streams, which were sandier than the piedmont streams (Utz and Hilderbrand, 2011). An increase in bed material size has in some cases been attributed to the introduction of anthropogenic debris in urban streams (Grable and Harden, 2006). Moreover, proportions of fine sediment can increase with ongoing construction in urbanizing watersheds (Phillips and Scatena, 2013). Over time, the high transport capacity of urban streams could remove coarse particles, leading to fining of bed material texture, but often bank erosion provides a supply of coarse material so that urban gravel-bed streams often maintain a texture similar to that of rural gravel-bed streams (Pizzuto et al., 2000).

3.5.5.4 Floodplain riparian corridor

Changes in hydrology associated with urbanization can often change floodplain riparian conditions. Streams in relatively dry climates may change from intermittent to perennial, leading to the expansion of riparian vegetation along the channel and floodplain (White and Greer, 2006). Such a change reflects increased storm runoff from impervious surfaces as well as increased dry season flow supplied by excess irrigation water conveyed into streams via municipal stormwater systems. Runoff from urban water use in precipitation-limited regions also can augment dry season flow, thereby increasing the amount of water availability in urban riparian zones (Solins and Cadenasso, 2022).

However, urban development can also transform perennial streams into an ephemeral condition by lowering the water table through channel degradation (Jordan et al., 2010; Solins and Cadenasso, 2020). This phenomenon, termed as "riparian hydrologic drought," affects the growth of riparian vegetation and the riparian ecosystem (Groffman et al., 2003). Incision of the coastal plain streams in North Carolina, driven by increased urban runoff, was responsible for declining riparian groundwater table and causing drier conditions in riparian zones, especially during the summer season (Hardison et al., 2009).

Urban development encroaches the riparian corridor by depriving the stream of stabilizing forests and replacing deep-rooted trees with shallow-rooted grass or ornamental plants that reduce the resistance against channel bed and bank erosion (Booth and Jackson, 1997). Riparian vegetation, if unaffected by urbanization, can help to stabilize channel morphology by stabilizing banks against the erosive effect of enhanced peak flows (Hession et al., 2003).

3.5.5.5 Drainage density, sinuosity, and slope

The introduction of storm drains and other artificial channels can modify greatly the drainage networks contributing runoff and sediment to urban streams. The addition of storm drains and discontinuous channels that were later channelized to the drainage network of the Avondale basin in Zimbabwe increased the drainage density of this watershed by 808% (from 0.35 km/km^2 to 3.15 km/km^2) (Whitlow and Gregory, 1989). The drainage density of South Branch of Ralston Creek, Iowa increased by more than 50% compared to its natural condition during the process of suburbanization (Graf, 1977a).

The channelization of urban streams is common, and this practice strongly affects channel sinuosity (channel length/valley length) and slopes. Because channelization often results in channel straightening, the sinuosity of urban streams typically is less than that of rural streams. The sinuosity of urban streams in southeastern Pennsylvania is 8% less than that of rural streams (Pizzuto et al., 2000). When urban streams are free to adjust erosionally, bank erosion may increase the sinuosity of channelized streams. In the Avondale basin, Nigeria, widening via bank slumping of the lower portions of a straight channel (channelized reach), particularly downstream of bridges, gradually increased channel sinuosity (Whitlow and Gregory, 1989).

Channel straightening also is a common practice that increases the slope of urban streams (Brookes et al., 2005; Grable and Harden, 2006; Phillips and Scatena, 2013). Reductions in slope can occur through erosion upstream of the channelized reach and through the deposition of eroded material downstream of the channelized reach (Figure 50).

3.5.5.6 Stream burial and urban stream deserts

Stream burial, a pervasive consequence of growing urban development, occurs when streams are routed through culverts, ditches, underground pipes, or concrete-lined channels or when streams are completely eliminated from the natural stream network by infilling or paving over (Napieralski and Welsh, 2016; Weitzell et al., 2016). This practice represents a major human impact on the geomorphology of urban streams by completely eliminating them as a form of open channel. It negatively affects the biodiversity of the aquatic habitat by fragmenting the ecology of the headwater streams. The intensity of stream burial is correlated to impervious surface cover associated with urbanization (Itsukushima and Ohtsuki, 2021; Weitzell et al., 2016). Analysis of stream burial patterns across the Potomac River Basin (38,000 km^2) in the United States has revealed high ratios of stream burial in urban areas with impervious cover greater than 30% (Weitzell et al., 2016). The burial of small headwater streams or first order streams is more common than larger streams during urban development (Elmore and Kaushal, 2008; Han et al., 2020). Mapping of stream burial for the rapidly urbanizing city of Detroit in Michigan revealed that it has lost at least 80% of its stream channels over the past century (Napieralski and Welsh, 2016). Moreover, lowland river infilling during infrastructure and building construction has decreased the drainage density of the Yinfeng plain in the Yangtze River Delta region by 20%, leading to reduced storage and flood control capacity (Yang et al., 2016).

With excessive stream burial in response to rapid development, the concept of an urban stream desert has emerged, referring to riverless urban areas within a watershed (Napieralski et al., 2015; Napieralski and Carvalhaes, 2016). Urban stream deserts constitute 6.2% of the urban areas (11,490 km^2) within 11 different regions of the U.S. with Detroit and Chicago being some of the largest stream deserts within the Great Lakes region (Napieralski and Carvalhaes, 2016).

3.5.6 Timescale of channel adjustment

Altered hydrological regimes and channel configurations related to urban development can clearly trigger associated changes in hydraulic conditions that often lead to channel instability, particularly channel enlargement through erosion. An important management concern related to this issue is the time scale of channel adjustment, i.e., the time required for streams to evolve in response to urbanization to reach a new stable configuration. Scattered redevelopment after initial development often makes urbanization an ongoing process, resulting in both spatially and temporally varied channel morphological adjustments (Chin et al., 2022; Walsh et al., 2016). This concern is important given that climate change can lead to trending hydrological conditions to which urban streams must also adjust (Hung et al., 2018). Understanding the time component of channel adjustment after disturbance in natural land surface processes is geomorphologically important to assess the impact of human activity on fluvial systems (Graf, 1977b) and to determine appropriate restorative measures (Finkenbine et al., 2000).

The time required for urban streams to adjust to the post built-out condition is described using reaction time and recovery time (Graf, 1977b; Simon, 1989). The reaction time is defined by the time period between disruption of a geomorphic system and the initiation of system change due to that disruption (Graf, 1977b) (Figure 54). Contextually, disturbance indicates the urban-induced changes in land, marked initially by the clearing of land for construction and subsequently by the transformation of the land into a built environment, whereas recovery time refers to the time required to achieve a new adjusted or stable state following the disturbance (Graf, 1977b). A variety of studies have examined the timing involved in channel adjustments initiated by urbanization (Figure 54 and Table 21); however, the general lack of historical data documenting long-term channel adjustments following the onset of urbanization precludes detailed understanding of the factors that govern recovery times and prediction of recovery times in specific circumstances. Reaction times generally are relatively short, spanning from months to a couple of years (Chin, 2006) (Table 21). Recovery times, on the other hand, can be quite long, often spanning many years or even decades (Table 21).

In urban streams, channel morphological response depends on the stage of urban development. Two stages have been recognized as important: the construction phase, in which land is cleared for construction of buildings or other urban infrastructure, and the urbanization phase, which follows the construction phase and is characterized by widespread impervious surface cover. Early work during the 1960s indicated that the construction phase of urbanization can sometimes lead to sedimentation within stream channels because of increased sediment yield from erosion of exposed soil (aggradation phase). Once land is cleared, soil erosion from construction sites, if not controlled properly, can increase suspended sediment concentrations in adjacent streams, potentially resulting in sedimentation. High concentrations of suspended sediment and associated sedimentation in Esrom Creek, Australia during the construction phase was recorded within five to six months of urbanization (Hannam, 1979). In Iowa, the creation of new floodplains and expansion of old floodplains by vertical accretion were observed within two years of urban construction because of the streams' inability to carry large quantities of newly available sediment (Graf, 1975). Removal of sediment deposited on floodplains once construction ceases is difficult to achieve and may take years to decades. Although the extent of the aggradation phase is usually short-lived, it can extend longer for downstream channels when sediment is sourced from eroding upstream reaches (Colosimo and Wilcock, 2007; Trimble, 1997). Widespread implementation of erosion-control practices associated with urban construction has substantially reduced problems associated with excessive sediment delivery to streams from urban construction, particularly in the United States. Thus, problems related to aggradation noted in early studies conducted during the 1960s and 1970s have become less pronounced since that time.

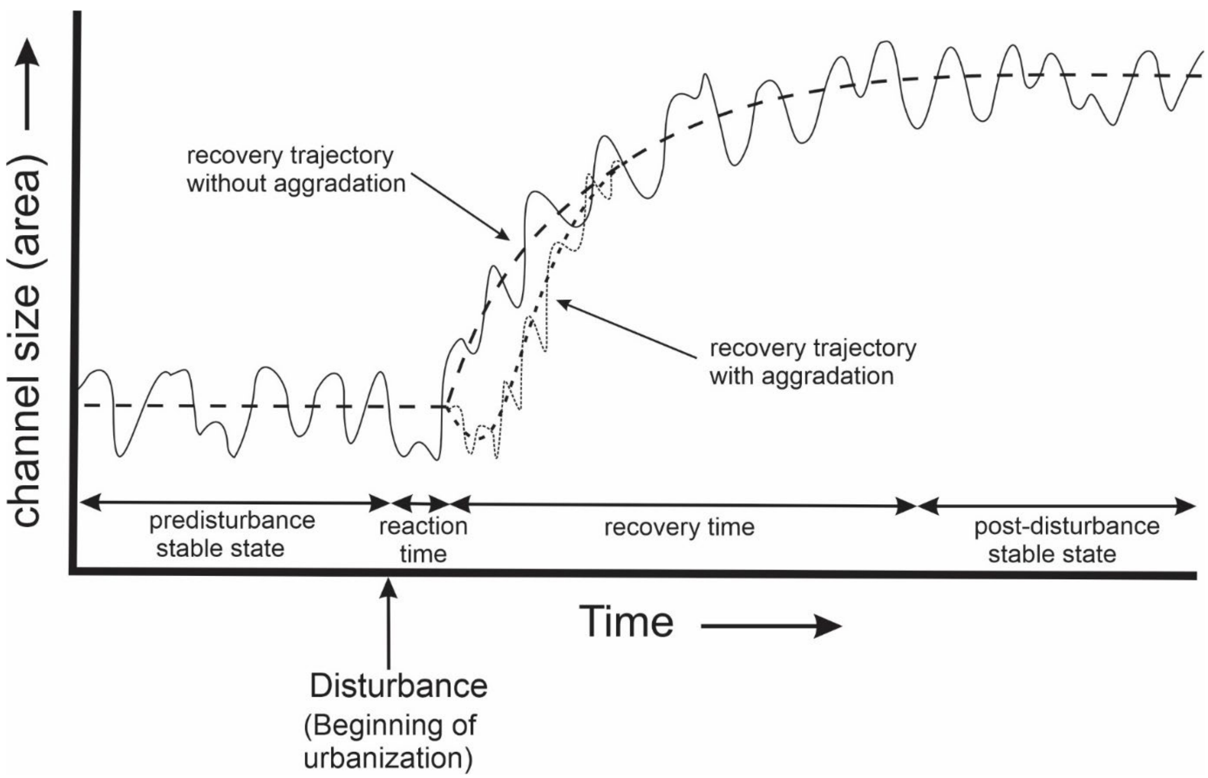


Figure 54. Conceptual diagram illustrating the typical morphological response of streams to urbanization over time, including trajectories with and without aggradation that may result if large amounts of sediment are delivered to streams during the initial construction phase of urbanization

By contrast, channel erosion related to increased flood magnitudes during the urbanization phase, characterized by widespread coverage by impervious surfaces, is the most prominent type of disturbance to urban streams. Erosion is related to both increases in flood magnitudes of moderate frequency events (2–5-year recurrence interval) as well as to decreases in sediment supply associated with the spread of impervious surfaces (Chin, 2006). As construction wanes and impervious surfaces become ubiquitous, sediment delivery decreases, discharge increases, and stored sediment may be gradually flushed from the system (Chin, 2006). Historical measurements of Baltimore streams showed that the high loads of sediment introduced during the construction phase were removed within seven years (Wolman, 1967; Wolman and Schick, 1967). Cross-section surveys (from 1953 to 1972) of Watts Branch near Rockville, Maryland revealed reduced channel capacity due to the large increase in sediment load for the first 12 years, which subsequently followed a trend of erosional regime as urban high flows frequently exceeded the channel capacity (Leopold, 1973). Despite the shift to erosion, the observed net adjustment based on 20 years of surveying was a net decrease in channel size relative to its original form. From 1973 to 1993 the channel did widen relative to its width in 1972 (Leopold et al., 2005). Similarly, the case of Kuala Lumpur confirmed that the time span of 17 years was sufficient to decrease the high sediment yield and alter the adjustment sequence from aggrading to eroding (Douglas, 1985). Conversely, the case study of Canon's Brook, UK revealed that after 14 years of urbanization, little or no substantial change occurred in channel morphology (Hollis and Luckett, 1976).

Incision and enlargement of urban channels typically occur over timespans of several years to decades for the streams (Chin, 2006). The time to complete the enlargement phase varies with urbanization age and location of urbanization. Three decades were sufficient for major morphological adjustments to occur in the dryland stream channels (Chin et al., 2017). Study results around the world (Table 21) indicate that the time required for the adjustment process to be completed could be as short as 5 years (Neller, 1988) or up to almost 40 years (Johnson and Royall, 2019). However, the case studies of Sawmill Brook or San Diego Creek in USA showed that even after longer periods of adjustment (almost 40–50 years), the streams continue to be erosionally unstable (Arnold et al., 1982; Trimble, 1997). A period of 41 years of observation of the Watts Branch in Maryland was still "too short" for the stream to complete the urbanization cycle; at the end of this period erosion was still occurring (Leopold et al., 2005). Assessment of long-term data from Sand River in Aiken, SC, USA indicated that the cohesive channel did not attain a stable state despite over 82 years of urban development (Sullivan et al., 2020).

Table 21. Adjustment Times of Streams to Urbanization

Study location	*Adjustment phase*	*Time period of adjustment (after urban development)*	*Stabilized?*	*Citing literatures*
Baltimore, USA	Recovery - shift from initial aggradation to subsequent erosion	5-7 years	no	(Wolman, 1967; Wolman and Schick, 1967)
Philadelphia, USA	Recovery - Shift from initial aggradation to subsequent erosion	More than 4 years	yes	(Hammer, 1972)
	Recovery - channel enlargement	Approx. 30 years		
Watts Branch, Maryland, USA	Recovery - channel aggradation	12 years	no	(Leopold, 1973)
	Recovery - shift from aggradation to subsequent erosion	More than 20 years		
Southeast Denver, Colorado, USA	Reaction time	Less than 2 years	N/A	(Graf, 1975)
Canon's Brook, Harlow, Essex, UK	Recovery - shift from aggrading to eroding regime	Greater than 14 years	no	(Hollis and Luckett, 1976)
West Bathurst, New South Wales, Australia	Reaction time	5 to 6 months	N/A	(Hannam, 1979)
Sawmill Brook, Connecticut, USA	Recovery - channel enlargement	More than 40 years	no	(Arnold et al., 1982)
Sungai Anak Ayer Batu in Kuala Lumpur, Malaysia	Recovery - channel aggradation	17 years	no	(Douglas, 1985)
	Recovery - shift from aggradation to erosion	More than 17 years		
Armidale, New South Wales, Australia	Recovery - channel enlargement	5 years after completion of urban development	yes	(Neller, 1988)
San Diego Creek, Southern California, USA	Recovery - channel enlargement	More than 50 years	no	(Trimble, 1997)
Vancouver, British Columbia, Canada	Recovery - channel enlargement	Approx. 20 years	yes	(Finkenbine et al., 2000)
Puget Sound lowlands, western Washington, USA	Recovery - channel enlargement	10 to 20 years	yes	(Henshaw and Booth, 2000)
Fountain Hills, Arizona, USA	Recovery - channel enlargement	More than 30 years	no	(Chin and Gregory, 2001)
Southern Piedmont of USA	Recovery - channel enlargement	Greater than 40 years	yes	(Johnson and Royall, 2019)
Sand River, Aiken South Carolina, USA	Recovery - channel enlargement	82 years	no	(Sullivan et al., 2020)

The temporal sequence of adjustments in streams that are in the process of accommodating the increased volume of urban runoff in some cases conforms to the classic channel evolution model (CEM) that has been developed to characterize erosional adjustments of river systems (Schumm, 1984; Simon, 1989). This conceptual model depicts the typical sequence of adjustments, including a phase of channel incision followed by channel widening (Figure 49 and Figure 50). Recovery occurs when excessive widening reduces the sediment-transport capacity, and the system is stabilized through deposition within the widened channel (Hawley et al., 2020). Streams in urban watersheds in the southern Piedmont of USA can be characterized according to the time since urbanization using a three-phase model (Johnson and Royall, 2019) adapted from the CEM. A period of more than 40 years was required to establish a relatively stable condition in these streams, which exhibited substantial erosional adjustments during the first 20 to 35 years following urbanization (Johnson and Royall, 2019). Wilket Creek in Canada was still in a state of recovery after 50-plus years of constant urban development (Bevan et al., 2018). In northern Kentucky, urban streams are beginning to reach fairly stable configurations several decades after urbanization–a condition attributed to upstream stormwater retrofits that reduced the rate of urban runoff from detention facilities (Hawley et al., 2020). Although characterization of the adjustment stage using a CEM can be useful in some circumstances, spatial and temporal variations in water and sediment delivery from the surrounding watershed often produce complex patterns of spatially varied morphological adjustments along urban streams, which may preclude meaningful application of a CEM to characterize the stage of evolutionary adjustment at particular locations and particular times (Colosimo and Wilcock, 2007).

The erosive regime will continue until changes in channel form (depth, width, slope) produce changes in hydraulic conditions (shear stress and stream power) that reduce the sediment transport capacity to match sediment supply (Morisawa and Laflure, 1979). In general, no universal predictive relation can be applied to define the period of restabilization of urban streams because it depends on the combination of hydrologic and geomorphic characteristics such as geologic substrate (Booth, 1990) and riparian vegetation (Keen-Zebert, 2007) of the channel and its contributing watershed, rather than the degree or rate of urbanization (Henshaw and Booth, 2000). Moreover, the stability of streams depends on stable land-cover conditions, which may be hard to achieve in urban environments (Booth and Henshaw, 2001). Although the time period required to establish a new stable state seems to be relatively long, yet highly variable, it is assumed that most stream channels eventually adjust to the process of urbanization (Chin, 2006). Nevertheless, this adjustment can be complex and is not guaranteed. Six factors have been identified as important in governing the adjustment of streams to urbanization. Both the rate of sediment delivery and the hydrological regime need to stabilize to provide an opportunity for the stream system to adjust to new inputs of sediment and water. If these conditions continue to evolve over time, it will be difficult for the system to achieve stability. Characteristics of bed and bank materials, as well as riparian vegetation, can vary locally, leading to spatial variability in adjustment. Also, the timescale of recovery may reflect the proximity of a portion of the stream system to locations with the greatest amount of disturbance to sediment delivery or hydrological regime. Those closest to the foci of intense disturbance generally will be impacted for longer periods of time than those farther away from these areas.

Thus, the magnitude of disturbance is also important because highly impacted sites will tend to take longer to recover than those that are less impacted. Of course, trends in climate or climate variability can also influence the adjustments given that these trends will lead to non-stationarity in hydrological conditions or to a change of variability in these conditions. Finally, management of urban streams can both exacerbate disturbance (e.g., channelization that increases erosive potential) or facilitate recovery (e.g., restoration aimed at alleviating the effects of disturbed sediment delivery and hydrological regime).

3.6 Summary and Recommendations

Current understanding of the impacts of urbanization on stream channels can be summarized by noting that the body of research on this topic includes recurrent themes that provide a basis for generalization as well as considerable details that highlight the complexity of these impacts. General themes include:

1) Urbanization fundamentally alters the hydrology of urban landscapes by increasing rates of runoff and, to some extent, volumes of runoff. As a result, the magnitudes of peak discharges for a specific recurrence interval increase, particularly for the most frequent flows.
2) Whereas construction activities may deliver large amounts of fine sediment to urban streams during the construction phase of urbanization, the long-term effect of urbanization on sediment delivery is complex but often involves reductions in sediment delivery from the watershed because of widespread coverage of the landscape by impervious surfaces. Delivery of sediment from within streams may increase during the urbanized phase because of increases in channel erosion.
3) The increase in peak discharges, along with channelization of many urban streams, often increases the bed shear stress and stream power per unit area of flows, resulting in an increased potential for mobilization of channel bed material and erosion of streambanks.
4) Although net deposition of sediment may occur on floodplains or even within streams during the construction phase, the most prominent geomorphic response of streams to urbanization is erosional enlargement through either expansion (simultaneous erosion of the channel bed and banks) or incision (downcutting of the bed followed by widening). This erosional response reflects the potential for increased mobilization of bed and bank material related to increases in the bed shear stress and stream power per unit area caused by the effect of urbanization on stream hydrology and hydraulics. Locally, it also reflects spatial variability in rates of bed-material transport, with erosional sites likely to occur where the rate of bed-material transport increases in the downstream direction.
5) Efforts to mitigate increased flooding by increasing retention and storage of stormwater, while effective at reducing peak discharges and achieving peak-matching goals for non-urbanized watersheds, may increase the durations of transport-effective discharges (as storage water is gradually released) that could promote erosion of streams. This issue is understudied and is only beginning to receive attention within the research community.

These general understandings are broadly relevant to urbanization that has occurred and is continuing to occur within the greater Chicago region. However, it must be emphasized that the geomorphic dynamics of rivers are a function of two major factors: 1) general erosional and depositional processes related to the flow of water and movement of sediment that determine the form of stream channels and 2) environmental context, which determines exactly how those processes operate in any particular geographic setting to produce adjustments between process and form. Most of the research that has been conducted on responses of streams to urbanization consists of case studies in particular geographic settings. Because environmental context is important, generalizing beyond case studies is often difficult. Just because a stream adjusted a specific way at a specific place does not mean it will do so in another. To understand the role of context in adjustment, it is vital to have good information on that context. The literature reviewed in this report indicates that very little work has been done on the geomorphology of streams in Chicago, nor has basic data on these streams been collected that could inform geomorphological analysis. The review did not identify any scientific studies of major importance that examined the geomorphological response of streams in Chicago to urbanization. A critical need exists for basic geomorphological information on these streams before judgments can be made about possible morphological responses to stormwater runoff policies. A generalization that can be made is that if the sediment transport capacity exceeds the availability of sediment (either coming into a reach from an upstream reach or from delivery of material to a reach by stormwater runoff into it), the channel will erode, as long as it does not have an inerodible bed and banks, which is another unknown for many streams in Chicago. This basic idea serves as the foundation for the stream-power approach to assessing channel stability that is being pursued in the optional pilot analysis. This analysis represents an important first step toward achieving an improved understanding of how various stormwater policies might affect channel stability.

3.7 References

Abali, T.P., Inko-Tariah, I.M., Nsiegbe, D.K., Nkii, L.B., Ideki, O., 2021. A comparative analysis of morphometry and morphology of Otamiri watershed, south-eastern Nigeria. Zien Journal of Social Sciences and Humanities, 1(1), 211-225.

Alber, A., Piégay, H., 2017. Characterizing and modelling river channel migration rates at a regional scale: Case study of south-east France. Journal of Environmental Management, 202, 479-493. https://doi.org/10.1016/j.jenvman.2016.10.055.

Allan, C.J., Estes, C.J., 2005. A morphological and economic examination of plunge pools as energy dissipaters in urban stream channels. Journal of the American Water Resources Association, 41(1), 123-133. https://doi.org/10.1111/j.1752-1688.2005.tb03722.x.

Allen, H.E., Bejcek, R.M., 1979. Effects of urbanization on the magnitude and frequency of floods in northeastern Illinois. U.S. Geological Survey Water-Resources Investigations Report 79-36. doi:10.3133/wri7936.

Allen, P.M., Arnold, J.G., Skipwith, W., 2002. Erodibility of urban bedrock and alluvial channels, north Texas. Journal of the American Water Resources Association, 38(5), 1477-1492. doi:10.1111/j.1752-1688.2002.tb04360.x.

Allen, P.M., Narramore, R., 1985. Bedrock controls on stream channel enlargement with urbanization, north central Texas. Journal of the American Water Resources Association, 21(6), 1037-1048. https://doi.org/10.1111/j.1752-1688.1985.tb00199.x.

Allmendinger, N.E., Pizzuto, J.E., Moglen, G.E., Lewicki, M., 2007. A sediment budget for an urbanizing watershed, 1951-1996, Montgomery County, Maryland, U.S.A. Journal of the American Water Resources Association, 43(6), 1483-1498. https://doi.org/10.1111/j.1752-1688.2007.00122.x.

Anderson, D.G., 1970. Effects of urban development on floods in northern Virginia. US Geological Survey Water Supply Paper, 2001C. http://pubs.er.usgs.gov/publication/wsp01C.

Anim, D.O., Banahene, P., 2021. Urbanization and stream ecosystems: the role of flow hydraulics towards an improved understanding in addressing urban stream degradation. Environmental Reviews, 29(3), 401-414. doi: 10.1139/er-2020-0063.

Anim, D.O., Fletcher, T.D., Pasternack, G.B., Vietz, G.J., Duncan, H.P., Burns, M.J., 2019a. Can catchment-scale urban stormwater management measures benefit the stream hydraulic environment? Journal of Environmental Management, 233, 1-11. https://doi.org/10.1016/j.jenvman.2018.12.023.

Anim, D.O., Fletcher, T.D., Vietz, G.J., Pasternack, G.B., Burns, M.J., 2018. Effect of urbanization on stream hydraulics. River Research and Applications, 34(7), 661-674. doi: 10.1002/rra.3293.

Anim, D.O., Fletcher, T.D., Vietz, G.J., Pasternack, G.B., Burns, M.J., 2019b. Restoring in-stream habitat in urban catchments: Modify flow or the channel? Ecohydrology, 12(1), e2050. https://doi.org/10.1002/eco.2050.

Annable, W.K., Watson, C.C., Thompson, P.J., 2012. Quasi-equilibrium conditions of urban gravel-bed stream channels in southern Ontario, Canada. River Research and Applications, 28(3), 302-325. doi:10.1002/rra.1457.

Arnold, C.L., Boison, P.J., Patton, P.C., 1982. Sawmill Brook: an example of rapid geomorphic change related to urbanization. Journal of Geology, 90(2), 155-166. doi: 10.1086/628660.

Arnold, C.L., Gibbons, C.J., 1996. Impervious surface coverage: the emergence of a key environmental indicator. Journal of the American Planning Association, 62(2), 243-258. doi: 10.1080/01944369608975688.

Arnold, E., Toran, L., 2018. Effects of bank vegetation and incision on erosion rates in an urban stream. Water, 10(4), 482, doi:10.3390/w10040482.

Arrospide, F., Mao, L., Escauriaza, C., 2018. Morphological evolution of the Maipo River in central Chile: Influence of instream gravel mining. Geomorphology, 306, 182-197. doi: 10.1016/j.geomorph.2018.01.019.

Bagnold, R.A., 1966. An approach to the sediment transport problem from general physics. US Geological Survey Professional Paper 422-I. http://pubs.er.usgs.gov/publication/pp422I.

Bai, S., Li, J., 2013. Sediment wash-off from an impervious urban land surface. Journal of Hydrologic Engineering, 18(5), 488-498. doi:10.1061/(ASCE)HE.1943-5584.0000654.

Baker, D.W., Pomeroy, C.A., Annable, W.K., MacBroom, J.G., Schwartz, J.S., Gracie, J., 2008. Evaluating the effects of urbanization on stream flow and channel stability - state of practice. World Environmental and Water Resources Congress 2008, American Society of Civil Engineers, pp. 1-10.

Beighley, R.E., Melack, J.M., Dunne, T., 2003. Impacts of California's climatic regimes and coastal land use change on streamflow characteristics. Journal of the American Water Resources Association, 39(6), 1419-1433. https://doi.org/10.1111/j.1752-1688.2003.tb04428.x.

Bevan, V., MacVicar, B., Chapuis, M., Ghunowa, K., Papangelakis, E., Parish, J., Snodgrass, W., 2018. Enlargement and evolution of a semi-alluvial creek in response to urbanization. Earth Surface Processes and Landforms, 43(11), 2295-2312. doi: 10.1002/esp.4391.

Bizzi, S., Lerner, D.N., 2015. The use of stream power as an indicator of channel sensitivity to erosion and deposition processes. River Research and Applications, 31(1), 16-27. https://doi.org/10.1002/rra.2717.

Bledsoe, B.P., 2002. Stream erosion potential and stormwater management strategies. Journal of Water Resources Planning and Management, 128(6), 451-455. doi: 10.1061/(asce)0733-9496(2002)128:6(451).

Bledsoe, B.P., Watson, C.C., 2001. Effects of urbanization on channel instability. Journal of the American Water Resources Association, 37(2), 255-270. https://doi.org/10.1111/j.1752-1688.2001.tb00966.x.

Blum, A.G., Ferraro, P.J., Archfield, S.A., Ryberg, K.R., 2020. Causal effect of impervious cover on annual flood magnitude for the United States. Geophysical Research Letters, 47(5). doi: 10.1029/2019gl086480.

Booth, D.B., 1990. Stream-channel incision following drainage-basin urbanization. Journal of the American Water Resources Association, 26(3), 407-417. https://doi.org/10.1111/j.1752-1688.1990.tb01380.x.

Booth, D.B., 1991. Urbanization and the natural drainage system – impacts, solutions, and prognoses. Northwest Environmental Journal, 7, 93-118.

Booth, D.B., Bledsoe, B.P., 2009. Streams and urbanization. In: L.A. Baker (Ed.), The Water Environment of Cities. Springer US, Boston, MA, pp. 93-123.

Booth, D.B., Fischenich, C.J., 2015. A channel evolution model to guide sustainable urban stream restoration. Area, 47(4), 408-421. doi:10.1111/area.12180.

Booth, D.B., Henshaw, P.C., 2001. Rates of channel erosion in small urban streams. Water Science and Application, 2, 17-38.

Booth, D.B., Jackson, C.R., 1997. Urbanization of aquatic systems: degradation thresholds, stormwater detection, and the limits of mitigation. Journal of the American Water Resources Association, 33(5), 1077-1090. https://doi.org/10.1111/j.1752-1688.1997.tb04126.x.

Booth, D.B., Roy, A.H., Smith, B., Capps, K.A., 2015. Global perspectives on the urban stream syndrome. Freshwater Science, 35(1), 412-420. doi:10.1086/684940.

Brater, E.F., Sangal, S., 1969. Effects of urbanization on peak flows. In: W.L. Moore, C.W. Morgan (Eds.), Effects of Watershed Changes on Stream Flow. Water Resources Symposium, vol. 2., University of Texas, Austin, pp. 201–213.

Brookes, A., 1988. Channelized Rivers: Perspectives for Environmental Management. Wiley, Chichester, New York.

Brookes, A., Chalmers, A., Vivash, R., 2005. Solving an urban river erosion problem on the Tilmore Brook, Hampshire (UK). Water and Environment Journal, 19(3), 199-206. https://doi.org/10.1111/j.1747-6593.2005.tb01587.x.

Brun, S.E., Band, L.E., 2000. Simulating runoff behavior in an urbanizing watershed. Computers, Environment and Urban Systems, 24(1), 5-22. https://doi.org/10.1016/S0198-9715(99)00040-X.

Burges, S.J., Wigmosta, M.S., Meena, J.M., 1998. Hydrological effects of land-use change in a zero-order catchment. Journal of Hydrologic Engineering, 3(2), 86-97. doi:10.1061/(ASCE)1084-0699(1998)3:2(86).

Carter, R.W., 1961. Magnitude and frequency of floods in suburban areas. US Geological Survey Professional Paper 424-B, B9–B11. https://pubs.er.usgs.gov/publication/pp424B.

Cashman, M.J., Gellis, A., Sanisaca, L.G., Noe, G.B., Cogliandro, V., Baker, A., 2018. Bank-derived material dominates fluvial sediment in a suburban Chesapeake Bay watershed. River Research and Applications, 34(8), 1032-1044. https://doi.org/10.1002/rra.3325.

Chin, A., 2006. Urban transformation of river landscapes in a global context. Geomorphology, 79(3-4), 460-487. doi: 10.1016/j.geomorph.2006.06.033.

Chin, A., Gidley, R., Tyner, L., Gregory, K., 2017. Adjustment of dryland stream channels over four decades of urbanization. Anthropocene, 20, 24-36. doi: 10.1016/j.ancene.2017.11.001.

Chin, A., Gregory, K.J., 2001. Urbanization and adjustment of ephemeral stream channels. Annals of the Association of American Geographers, 91(4), 595-608.

Chin, A., Gregory, K.J., O'Dowd, A.P., 2022. Urbanizing river channels. In: J.F. Shroder (Ed.), Treatise on Geomorphology (Second Edition). Academic Press, Oxford, pp. 1255-1276. https://doi.org/10.1016/B978-0-12-409548-9.12500-X

Cho, E., Yoo, C., Kang, M., Song, S.-u., Kim, S., 2020. Experiment of wind-driven-rain measurement on building walls and its in-situ validation. Building and Environment, 185, 107269. https://doi.org/10.1016/j.buildenv.2020.107269.

Cianfrani, C.M., Hession, W.C., Rizzo, D.M., 2006. Watershed imperviousness impacts on stream channel condition in southeastern Pennsylvania. Journal of the American Water Resources Association, 42(4), 941-956. https://doi.org/10.1111/j.1752-1688.2006.tb04506.x.

Clark, J.J., Wilcock, P.R., 2000. Effects of land-use change on channel morphology in northeastern Puerto Rico. Geological Society of America Bulletin, 112(12), 1763-1777. doi: 10.1130/0016-7606(2000)112<1763:eoluco>2.0.co;2.

Colosimo, M.F., Wilcock, P.R., 2007. Alluvial sedimentation and erosion in an urbanizing watershed, Gwynns Falls, Maryland. Journal of the American Water Resources Association, 43(2), 499-521. https://doi.org/10.1111/j.1752-1688.2007.00039.x.

Doll, B.A., Wise-Frederick, D.E., Buckner, C.M., Wilkerson, S.D., Harman, W.A., Smith, R.E., Spooner, J., 2002. Hydraulic geometry relationships for urban streams throughout the piedmont of North Carolina. Journal of the American Water Resources Association, 38(3), 641-651. https://doi.org/10.1111/j.1752-1688.2002.tb00986.x.

Douglas, I., 1985. Hydrogeomorphology downstream of bridges: one mechanism of channel widening. Applied Geography, 5(2), 167-170. https://doi.org/10.1016/0143-6228(85)90040-2.

Doyle, M.W., Harbor, J.M., Rich, C.F., Spacie, A., 2000. Examining the effects of urbanization on streams using indicators of geomorphic stability. Physical Geography, 21(2), 155-181. doi: 10.1080/02723646.2000.10642704.

Duncan, W.W., Goodloe, R.B., Meyer, J.L., Prowell, E.S., 2011. Does channel incision affect in-stream habitat? Examining the effects of multiple geomorphic variables on fish habitat. Restoration Ecology, 19(1), 64-73. doi: 10.1111/j.1526-100X.2009.00534.x.

Durbin, T.J., 1974. Digital simulation of the effects of urbanization on runoff in the upper Santa Ana Valley, California. US Geological Survey Water-Resources Investigations Report, 41-73. http://pubs.er.usgs.gov/publication/wri7341.

Ebisemiju, F.S., 1989a. Patterns of stream channel response to urbanization in the humid tropics and their implications for urban land use planning: a case study from southwestern Nigeria. Applied Geography, 9(4), 273-286. https://doi.org/10.1016/0143-6228(89)90028-3.

Ebisemiju, F.S., 1989b. The response of headwater stream channels to urbanization in the humid tropics. Hydrological Processes, 3(3), 237-253. https://doi.org/10.1002/hyp.3360030304.

Elmore, A.J., Kaushal, S.S., 2008. Disappearing headwaters: patterns of stream burial due to urbanization. Frontiers in Ecology and the Environment, 6(6), 308-312. https://doi.org/10.1890/070101.

Espey Jr, W.H., Morgan, C.W., Masch, F.D., 1965. A study of some effects of urbanization on storm runoff from a small watershed. Texas Water Development Board Report, 23. http://pubs.er.usgs.gov/publication/70047174.

Ferreira, C.S.S., Walsh, R.P.D., Ferreira, A.J.D., 2018. Degradation in urban areas. Current Opinion in Environmental Science & Health, 5, 19-25. https://doi.org/10.1016/j.coesh.2018.04.001.

Finkenbine, J.K., Atwater, J.W., Mavinic, D.S., 2000. Stream health after urbanization. Journal of the American Water Resources Association, 36(5), 1149-1160. https://doi.org/10.1111/j.1752-1688.2000.tb05717.x.

Fitzpatrick, F.A., Diebel, M.W., Lutz, M.A., Richards, K.D., Harris, M.A., Arnold, T.L., 2005. Effects of urbanization on the geomorphology, habitat, hydrology, and fish index of biotic integrity of streams in the Chicago area, Illinois and Wisconsin. American Fisheries Society Symposium, 2005(47), 87-115.
Fraley, L.M., Miller, A.J., Welty, C., 2009. Contribution of in-channel processes to sediment yield of an urbanizing watershed. Journal of the American Water Resources Association, 45(3), 748-766. https://doi.org/10.1111/j.1752-1688.2009.00320.x.
Galster, J.C., Pazzaglia, F.J., Germanoski, D., 2008. Measuring the impact of urbanization on channel widths using historic aerial photographs and modern surveys. Journal of the American Water Resources Association, 44(4), 948-960. doi:10.1111/j.1752-1688.2008.00193.x.
Gellis, A.C., Fuller, C.C., Van Metre, P.C., Mahler, B.J., Welty, C., Miller, A.J., Nibert, L.A., Clifton, Z.J., Malen, J.J., Kemper, J.T., 2020. Pavement alters delivery of sediment and fallout radionuclides to urban streams. Journal of Hydrology, 588, 124855. https://doi.org/10.1016/j.jhydrol.2020.124855.
Gellis, A.C., Myers, M.K., Noe, G.B., Hupp, C.R., Schenk, E.R., Myers, L., 2017. Storms, channel changes, and a sediment budget for an urban-suburban stream, Difficult Run, Virginia, USA. Geomorphology, 278, 128-148. doi: 10.1016/j.geomorph.2016.10.031.
Ghunowa, K., MacVicar, B.J., Ashmore, P., 2021. Stream power index for networks (SPIN) toolbox for decision support in urbanizing watersheds. Environmental Modelling & Software, 144, 105185. https://doi.org/10.1016/j.envsoft.2021.105185.
Grable, J., Harden, C., 2006. Geomorphic response of an Appalachian valley and ridge stream to urbanization. Earth Surface Processes and Landforms, 31, 1707-1720. doi: 10.1002/esp.1433.
Graf, W.L., 1975. The impact of suburbanization on fluvial geomorphology. Water Resources Research, 11(5), 690-692. https://doi.org/10.1029/WR011i005p00690.
Graf, W.L., 1977a. Network characteristics in suburbanizing streams. Water Resources Research, 13(2), 459-463. https://doi.org/10.1029/WR013i002p00459.
Graf, W.L., 1977b. Rate law in fluvial geomorphology. American Journal of Science, 277(2), 178-191. doi:10.2475/ajs.277.2.178.
Gregory, K.J., 1974. Streamflow and building activity. In: K. Gregory, W. J., D.E. (Eds.), Fluvial Processes in Instrumented Watersheds. Insitute of British Geographers, London, pp. 123–139.
Gregory, K.J., 1976. Drainage basin adjustments and man. Geographica Polonica, 34, 155–173.
Gregory, K.J., 2006. The human role in changing river channels. Geomorphology, 79(3), 172-191. https://doi.org/10.1016/j.geomorph.2006.06.018.
Gregory, K.J., 2011. Wolman MG (1967) A cycle of sedimentation and erosion in urban river channels. Geografiska Annaler 49A: 385–395. Progress in Physical Geography: Earth and Environment, 35(6), 831-841. 10.1177/0309133311414527.
Gregory, K.J., Davis, R.J., Downs, P.W., 1992. Identification of river channel change to due to urbanization. Applied Geography, 12(4), 299-318. https://doi.org/10.1016/0143-6228(92)90011-B.
Groffman, P.M., Bain, D.J., Band, L.E., Belt, K.T., Brush, G.S., Grove, J.M., Pouyat, R.V., Yesilonis, I.C., Zipperer, W.C., 2003. Down by the riverside: urban riparian ecology. Frontiers in Ecology and the Environment, 1(6), 315-321. https://doi.org/10.1890/1540-9295(2003)001[0315:DBTRUR]2.0.CO;2.
Gurnell, A., Lee, M., Souch, C., 2007. Urban rivers: hydrology, geomorphology, ecology and opportunities for change. Geography Compass, 1(5), 1118-1137.
Hale, R.L., Scoggins, M., Smucker, N.J., Suchy, A., 2016. Effects of climate on the expression of the urban stream syndrome. Freshwater Science, 35(1), 421-428. doi: 10.1086/684594.
Hammer, T.R., 1972. Stream channel enlargement due to urbanization. Water Resources Research, 8(6), 1530-1540. https://doi.org/10.1029/WR008i006p01530.
Han, L., Xu, Y., Deng, X., Li, Z., 2020. Stream loss in an urbanized and agricultural watershed in China. Journal of Environmental Management, 253, 109687. https://doi.org/10.1016/j.jenvman.2019.109687.
Hannam, I., 1979. Urban soil erosion: an extreme phase in the Stewart subdivision, West Bathurst. Journal of the Soil Conservation Service of New South Wales, 35(1), 19-25.

Hardison, E.C., O'Driscoll, M.A., DeLoatch, J.P., Howard, R.J., Brinson, M.M., 2009. Urban land use, channel incision, and water table decline along coastal plain streams, North Carolina. Journal of the American Water Resources Association, 45(4), 1032-1046. https://doi.org/10.1111/j.1752-1688.2009.00345.x.
Harris, E.E., Rantz, S.E., 1964. Effect of urban growth on streamflow regimen of Permanente Creek, Santa Clara County, California. US Geological Survey Water Supply Paper, 1591B. http://pubs.er.usgs.gov/publication/wsp1591B.
Hawley, R.J., Bledsoe, B.P., 2013. Channel enlargement in semiarid suburbanizing watersheds: A southern California case study. Journal of Hydrology, 496, 17-30. doi: 10.1016/j.jhydrol.2013.05.010.
Hawley, R.J., Bledsoe, B.P., Stein, E.D., Haines, B.E., 2012. Channel evolution model of semiarid stream response to urban-induced hydromodification. Journal of the American Water Resources Association, 48(4), 722-744. doi: 10.1111/j.1752-1688.2012.00645.x.
Hawley, R.J., Goodrich, J.A., Korth, N.L., Rust, C.J., Fet, E.V., Frye, C., MacMannis, K.R., Wooten, M.S., Jacobs, M., Sinha, R., 2017. Detention outlet retrofit improves the functionality of existing detention basins by reducing erosive flows in receiving channels. Journal of the American Water Resources Association, 53(5), 1032-1047. https://doi.org/10.1111/1752-1688.12548.
Hawley, R.J., MacMannis, K.R., Wooten, M.S., 2013. Bed coarsening, riffle shortening, and channel enlargement in urbanizing watersheds, northern Kentucky, USA. Geomorphology, 201, 111-126. doi:10.1016/j.geomorph.2013.06.013.
Hawley, R.J., MacMannis, K.R., Wooten, M.S., Fet, E.V., Korth, N.L., 2020. Suburban stream erosion rates in northern Kentucky exceed reference channels by an order of magnitude and follow predictable trajectories of channel evolution. Geomorphology, 352, 106998. https://doi.org/10.1016/j.geomorph.2019.106998.
Hawley, R.J., Vietz, G.J., 2016. Addressing the urban stream disturbance regime. Freshwater Science, 35(1), 278-292. 10.1086/684647.
Henshaw, P.C., Booth, D.B., 2000. Natural restabilization of stream channels in urban watersheds. Journal of the American Water Resources Association, 36(6), 1219-1236. https://doi.org/10.1111/j.1752-1688.2000.tb05722.x.
Hession, W.C., Pizzuto, J.E., Johnson, T.E., Horwitz, R.J., 2003. Influence of bank vegetation on channel morphology in rural and urban watersheds. Geology, 31(2), 147-150. doi:10.1130/0091-7613(2003)031<0147:iobvoc>2.0.co;2.
Hettiarachchi, S., Wasko, C., Sharma, A., 2019. Can antecedent moisture conditions modulate the increase in flood risk due to climate change in urban catchments? Journal of Hydrology, 571, 11-20. doi:10.1016/j.jhydrol.2019.01.039.
Hodgkins, G.A., Dudley, R.W., Archfield, S.A., Renard, B., 2019. Effects of climate, regulation, and urbanization on historical flood trends in the United States. Journal of Hydrology, 573, 697-709. doi:10.1016/j.jhydrol.2019.03.102.
Hollis, G.E., 1975. The effect of urbanization on floods of different recurrence interval. Water Resources Research, 11(3), 431-435. https://doi.org/10.1029/WR011i003p00431.
Hollis, G.E., 1977. Water yield changes after the urbanization of the Canon's Brook Catchment, Harlow, England. Hydrological Sciences Bulletin, 22(1), 61-75. doi: 10.1080/02626667709491694.
Hollis, G.E., Luckett, J.K., 1976. The response of natural river channels to urbanization: Two case studies from southeast England. Journal of Hydrology, 30(4), 351-363. https://doi.org/10.1016/0022-1694(76)90118-9.
Hung, C.-L.J., James, L.A., Carbone, G.J., 2018. Impacts of urbanization on stormflow magnitudes in small catchments in the Sandhills of South Carolina, USA. Anthropocene, 23, 17-28. https://doi.org/10.1016/j.ancene.2018.08.001.
Ibrahim, Y.A., Rouhi, A., 2021. Role of hydrodynamic forces in shaping the dynamics of channel evolution in urban watershed. Journal of Hydrologic Engineering, 26(8), 05021019. doi:10.1061/(ASCE)HE.1943-5584.0002111.
Itsukushima, R., Ohtsuki, K., 2021. A century of stream burial due to urbanization in the Tokyo Metropolitan Area. Environmental Earth Sciences, 80(7), 237. doi:10.1007/s12665-021-09524-7.
James, L.A., Lecce, S.A., 2013. Impacts of land-use and land-cover change on river systems. In: J.F. Shroder (Ed.), Treatise on Geomorphology. Academic Press, San Diego, pp. 768-793.

James, L.D., 1965. Using a digital computer to estimate the effects of urban development on flood peaks. Water Resources Research, 1, 223-234.

Jeje, L.K., Ikeazota, S.I., 2002. Effects of urbanisation on channel morphology: The case of Ekulu River in Enugu, south eastern Nigeria. Singapore Journal of Tropical Geography, 23(1), 37. 10.1111/1467-9493.00117.

Jeong, A., Dorn, R.I., 2019. Soil erosion from urbanization processes in the Sonoran Desert, Arizona, USA. Land Degradation & Development, 30(2), 226-238. https://doi.org/10.1002/ldr.3207.

Johnson, P., Royall, D., 2019. Evaluating the effects of urbanization age on the morphology of low-order urban streams in the U.S. southern Piedmont. Physical Geography, 40(1), 1-27. doi: 10.1080/02723646.2018.1487635.

Jordan, B.A., Annable, W.K., Watson, C.C., Sen, D., 2010. Contrasting stream stability characteristics in adjacent urban watersheds: Santa Clara Valley, California. River Research and Applications, 26(10), 1281-1297. doi: 10.1002/rra.1333.

Keen-Zebert, A., 2007. Channel Responses to Urbanization: Scull and Mud Creeks in Fayetteville, Arkansas. Physical Geography, 28(3), 249-260. doi:10.2747/0272-3646.28.3.249.

Kemper, J.T., Miller, A.J., Welty, C., 2019. Spatial and temporal patterns of suspended sediment transport in nested urban watersheds. Geomorphology, 336, 95-106. https://doi.org/10.1016/j.geomorph.2019.03.018.

Kim, Y., Engel, B.A., Lim, K.J., Larson, V., Duncan, B., 2002. Runoff impacts of land-use change in Indian River Lagoon watershed. Journal of Hydrologic Engineering, 7(3), 245-251. doi:10.1061/(ASCE)1084-0699(2002)7:3(245).

Kinosita, T., T. Sonda., 1967. Change of runoff due to urbanisation, Int. Symp. on floods and their computation, UNESCO, Leningrad, U.S.S.R. August 1967. 13 pp., illus.

Konrad, C., 2003. Effects of urban development on floods. U.S. Geological Survey Fact Sheet 076-03. https://pubs.usgs.gov/fs/fs07603/pdf/fs07603.pdf.

Konrad, C.P., Booth, D.B., 2005. Hydrologic changes in urban streams and their ecological significance. American Fisheries Society Symposium, 2005(47), 157-177.

Konrad, C.P., Booth, D.B., Burges, S.J., 2005. Effects of urban development in the Puget Lowland, Washington, on interannual streamflow patterns: Consequences for channel form and streambed disturbance. Water Resources Research, 41(7). doi: 10.1029/2005wr004097.

Lammers, R.W., Bledsoe, B.P., 2018. A network scale, intermediate complexity model for simulating channel evolution over years to decades. Journal of Hydrology, 566, 886-900. https://doi.org/10.1016/j.jhydrol.2018.09.036.

Larsen, E.W., Premier, A.K., Greco, S.E., 2006. Cumulative effective stream power and bank erosion on the sacramento river, California, USA. Journal of the American Water Resources Association, 42(4), 1077-1097. https://doi.org/10.1111/j.1752-1688.2006.tb04515.x.

Leopold, L.B., 1968. Hydrology for urban land planning - A guidebook on the hydrologic effects of urban land use. U.S. Geological Survery Circular 554, http://pubs.er.usgs.gov/publication/cir554.

Leopold, L.B., 1973. River channel change with time : An example. Geological Society of America Bulletin, 84(6), 1845-1860. doi: 10.1130/0016-7606(1973)84<1845:RCCWTA>2.0.CO;2.

Leopold, L.B., 1991. Lag times for small drainage basins. Catena, 18(2), 157-171. https://doi.org/10.1016/0341-8162(91)90014-O.

Leopold, L.B., Reed, H., Miller, A., 2005. Geomorphic effects of urbanization in forty-one years of observation. Proceedings of the American Philosophical Society, 149(3), 349-371.

Linsley, R.K., Kohler, M.A., Paulhus, J.L.H., 1958. Hydrology for Engineers. McGraw-Hill, New York.

Lord, M.L., Germanoski, D., Allmendinger, N.E., 2009. Fluvial geomorphology: Monitoring stream systems in response to a changing environment. In: R. Young, L. Norby (Eds.), Geological Monitoring. Geological Society of America, Boulder, Colorado, pp. 69-103.

MacKenzie, K.M., Gharabaghi, B., Binns, A.D., Whiteley, H.R., 2022. Early detection model for the urban stream syndrome using specific stream power and regime theory. Journal of Hydrology, 604, 127167. https://doi.org/10.1016/j.jhydrol.2021.127167.

May, C., Horner, R., Karr, J., Mar, B., Welch, E., 1997. Effects of urbanization on small streams in the Puget Sound Lowland ecoregion. Watershed Protection Techniques, 2.

McBride, M., Booth, D.B., 2005. Urban impacts on physical stream condition: Effects of spatial scale, connectivity, and longitudinal trends. Journal of the American Water Resources Association, 41(3), 565-580. https://doi.org/10.1111/j.1752-1688.2005.tb03755.x.

Meierdiercks, K.L., Smith, J.A., Baeck, M.L., Miller, A.J., 2010. Analyses of urban drainage network structure and its impact on hydrologic response. Journal of the American Water Resources Association, 46(5), 932-943. doi:10.1111/j.1752-1688.2010.00465.x.

Miller, C.R., Viessman Jr., W., 1972. Runoff volumes from small urban watersheds. Water Resources Research, 8(2), 429-434. https://doi.org/10.1029/WR008i002p00429.

Miller, J.D., Hess, T., 2017. Urbanisation impacts on storm runoff along a rural-urban gradient. Journal of Hydrology, 552, 474-489. doi:10.1016/j.jhydrol.2017.06.025.

Miller, J.D., Kim, H., Kjeldsen, T.R., Packman, J., Grebby, S., Dearden, R., 2014. Assessing the impact of urbanization on storm runoff in a peri-urban catchment using historical change in impervious cover. Journal of Hydrology, 515, 59-70. https://doi.org/10.1016/j.jhydrol.2014.04.011.

Miller, R.A., 1978. The hydraulically effective impervious area of an urban basin, Broward County, Florida. Proceedings of the International Symposium on Urban Stormwater Management. In: Haan, C.T. (ed). University of Kentucky, Lexington, 259-261.

Morisawa, M., Laflure, E., 1979. Hydraulic geometry, stream equilibrium and urbanization. In: Rhodes, D.D. and Williams, G., Adjustments of the Fluvial System, Kendall Hunt, Dubuque, IA., pp. 333-350.

Nabegu, A., 2010. Response of the Jakara stream channel to urbanisation. African Scientist, Vol. 11.

Nabegu, A., 2014. Impact of urbanization on channel morphology: some comments. IOSR Journal of Environmental Science, Toxicology and Food Technology, 8, 40-45. doi: 10.9790/2402-08424045.

Nanson, G.C., Hickin, E.J., 1986. A statistical analysis of bank erosion and channel migration in western Canada. GSA Bulletin, 97(4), 497-504. doi: 10.1130/0016-7606(1986)97<497:asaobe>2.0.co;2.

Nanson, G.C., Young, R.W., 1981. Downstream reduction of rural channel size with contrasting urban effects in small coastal streams of southeastern Australia. Journal of Hydrology, 52(3), 239-255. https://doi.org/10.1016/0022-1694(81)90173-6.

Napieralski, J., Keeling, R., Dziekan, M., Rhodes, C., Kelly, A., Kobberstad, K., 2015. Urban Stream deserts as a consequence of excess stream burial in urban watersheds. Annals of the Association of American Geographers, 105(4), 649-664.

Napieralski, J.A., Carvalhaes, T., 2016. Urban stream deserts: Mapping a legacy of urbanization in the United States. Applied Geography, 67, 129-139. https://doi.org/10.1016/j.apgeog.2015.12.008.

Napieralski, J.A., Welsh, E.S., 2016. A century of stream burial in Michigan (USA) cities. Journal of Maps, 12(sup1), 300-303. 10.1080/17445647.2016.1206040.

Navratil, O., Breil, P., Schmitt, L., Grosprêtre, L., Albert, M.B., 2013. Hydrogeomorphic adjustments of stream channels disturbed by urban runoff (Yzeron River basin, France). Journal of Hydrology, 485, 24-36. doi: 10.1016/j.jhydrol.2012.01.036.

Neller, R.J., 1988. A comparison of channel erosion in small urban and rural catchments, Armidale, New South Wales. Earth Surface Processes and Landforms, 13(1), 1-7. https://doi.org/10.1002/esp.3290130102.

Neller, R.J., 1989. Induced channel enlargement in small urban catchments, Armidale, New South Wales. Environmental Geology and Water Sciences, 14(3), 167-171. doi:10.1007/BF01705127.

Nelson, E.J., Booth, D.B., 2002. Sediment sources in an urbanizing, mixed land-use watershed. Journal of Hydrology, 264(1-4), 51-68. doi:10.1016/s0022-1694(02)00059-8.

Nelson, P.A., Smith, J.A., Miller, A.J., 2006. Evolution of channel morphology and hydrologic response in an urbanizing drainage basin. Earth Surface Processes and Landforms, 31(9), 1063-1079. doi:10.1002/esp.1308.

O'Driscoll, M.A., Soban, J.R., Lecce, S.A., 2009. Stream channel enlargement response to urban land cover in small coastal plain watersheds, North Carolina. Physical Geography, 30(6), 528-555. doi:10.2747/0272-3646.30.6.528.

Odemerho, F.O., 1992. Limited downstream response of stream channel size to urbanization in a humid tropical basin. The Professional Geographer, 44(3), 332-339. doi:10.1111/j.0033-0124.1992.00332.x.

Oudin, L., Salavati, B., Furusho-Percot, C., Ribstein, P., Saadi, M., 2018. Hydrological impacts of urbanization at the catchment scale. Journal of Hydrology, 559, 774-786. https://doi.org/10.1016/j.jhydrol.2018.02.064.

Over, T.M., Saito, R.J., Soong, D.T., 2016. Adjusting annual maximum peak discharges at selected stations in northeastern Illinois for changes in land-use conditions. U.S. Geological Survey Scientific Investigations Report 2016-5049, 33p. http://dx.doi.org/10.3133/sir20165049.

Papangelakis, E., MacVicar, B., Ashmore, P., 2019. Bedload sediment transport regimes of semi-alluvial rivers conditioned by urbanization and stormwater management. Water Resources Research, 55(12), 10565-10587. https://doi.org/10.1029/2019WR025126.

Papangelakis, E., MacVicar, B., Ashmore, P., Gingerich, D., Bright, C., 2022. Testing a watershed-scale stream power index tool for erosion risk assessment in an urban river. Journal of Sustainable Water in the Built Environment, **8**(3). doi: 10.1061/jswbay.0000989.

Paul, M.J., Meyer, J.L., 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics, 32, 333-365.

Phillips, C.B., Scatena, F.N., 2013. Reduced channel morphological response to urbanization in a flood-dominated humid tropical environment. Earth Surface Processes and Landforms, 38(9), 970-982. doi:10.1002/esp.3345.

Pizzuto, J.E., Hession, W.C., McBride, M., 2000. Comparing gravel-bed rivers in paired urban and rural catchments of southeastern Pennsylvania. Geology, 28(1), 79-82. doi: 10.1130/0091-7613(2000)028<0079:cgripu>2.0.co;2.

Plumb, B.D., Annable, W.K., Thompson, P.J., Hassan, M.A., 2017. The impact of urbanization on temporal changes in sediment transport in a gravel bed channel in southern Ontario, Canada. Water Resources Research, 53(10), 8443-8458. https://doi.org/10.1002/2016WR020288.

Poff, N.L., Bledsoe, B.P., Cuhaciyan, C.O., 2006. Hydrologic variation with land use across the contiguous United States: Geomorphic and ecological consequences for stream ecosystems. Geomorphology, 79, 264-285. doi:10.1016/j.geomorph.2006.06.032.

Ramírez, A., Jesús-Crespo, R.D., Martinó-Cardona, D.M., Martínez-Rivera, N., Burgos-Caraballo, S., 2009. Urban streams in Puerto Rico: what can we learn from the tropics? Journal of the North American Benthological Society, 28(4), 1070-1079. doi:10.1899/08-165.1.

Reid, L.M., Dunne, T., 2016. Sediment budgets as an organizing framework in fluvial geomorphology. In: H.P. G. Mathias Kondolf (Ed.), Tools in Fluvial Geomorphology. John Wiley & Sons, Chichester, UK, pp. 357-380.

Ress, L.D., Hung, C.-L.J., James, L.A., 2020. Impacts of urban drainage systems on stormwater hydrology: Rocky Branch Watershed, Columbia, South Carolina. Journal of Flood Risk Management, 13(3), e12643. https://doi.org/10.1111/jfr3.12643.

Rhoads, B.L., 1987. Stream power terminology. The Professional Geographer, 39(2), 189-195. doi:10.1111/j.0033-0124.1987.00189.x.

Rhoads, B.L., 1995. Stream power: a unifying theme for urban fluvial geomorphology. In: E.E. Herricks (Ed.), Stormwater Runoff and Receiving Systems: Impact, Monitoring, and Assessment. Lewis Publishers, Boca Raton, FL, pp. 65-75.

Rhoads, B.L., 2020. River Dynamics: Geomorphology to Support Management. Cambridge University Press, Cambridge, UK.

Rinaldi, M., Wyżga, B., Surian, N., 2005. Sediment mining in alluvial channels: physical effects and management perspectives. River Research and Applications, 21(7), 805-828. https://doi.org/10.1002/rra.884.

Robinson, A.M., 1976. Effects of urbanization on stream channel morphology. National Symposium on Urban Hydrology, Hydraulics, and Sediment Control, Proc(111), 115-127.

Roodsari, B.K., Chandler, D.G., 2017. Distribution of surface imperviousness in small urban catchments predicts runoff peak flows and stream flashiness. Hydrological Processes, 31(17), 2990-3002. https://doi.org/10.1002/hyp.11230.

Rosa, D.W.B., Nascimento, N.O., Moura, P.M., Macedo, G.D., 2020. Assessment of the hydrological response of an urban watershed to rainfall-runoff events in different land use scenarios – Belo Horizonte, MG, Brazil. Water Science and Technology, 81(4), 679-693. doi:10.2166/wst.2020.148.

Rosburg, T., Nelson, P., Bledsoe, B., 2017. Effects of urbanization on flow duration and stream flashiness: A case study of Puget Sound streams, western Washington, USA. Journal of the American Water Resources Association, 53. doi:10.1111/1752-1688.12511.
Rose, S., Peters, N., 2001. Effects of urbanization on streamflow in the Atlanta Area (Georgia, USA): A comparative hydrological approach. Hydrological Processes, 15, 1441-1457. doi:10.1002/hyp.218.
Rougé, C., Cai, X., 2014. Crossing-scale hydrological impacts of urbanization and climate variability in the Greater Chicago Area. Journal of Hydrology, 517, 13-27. https://doi.org/10.1016/j.jhydrol.2014.05.005.
Russell, K., 2021. Potential sediment supply fluxes associated with greenfield residential construction. Anthropocene, 35. https://doi.org/10.1016/j.ancene.2021.100300.
Russell, K.L., Vietz, G.J., Fletcher, T.D., 2017. Global sediment yields from urban and urbanizing watersheds. Earth-Science Reviews, 168, 73-80. https://doi.org/10.1016/j.earscirev.2017.04.001.
Russell, K.L., Vietz, G.J., Fletcher, T.D., 2018. Urban catchment runoff increases bedload sediment yield and particle size in stream channels. Anthropocene, 23, 53-66. https://doi.org/10.1016/j.ancene.2018.09.001.
Russell, K.L., Vietz, G.J., Fletcher, T.D., 2019a. A suburban sediment budget: Coarse-grained sediment flux through hillslopes, stormwater systems and streams. Earth Surface Processes and Landforms, 44(13), 2600-2614. https://doi.org/10.1002/esp.4685.
Russell, K.L., Vietz, G.J., Fletcher, T.D., 2019b. Urban sediment supply to streams from hillslope sources. Science of The Total Environment, 653, 684-697. https://doi.org/10.1016/j.scitotenv.2018.10.374.
Russell, K.L., Vietz, G.J., Fletcher, T.D., 2020. How urban stormwater regimes drive geomorphic degradation of receiving streams. Progress in Physical Geography: Earth and Environment, 44(5), 746-778. doi:10.1177/0309133319893927.
Santikari, V.P., Murdoch, L.C., 2019. Effects of construction-related land use change on streamflow and sediment yield. Journal of Environmental Management, 252, 109605. https://doi.org/10.1016/j.jenvman.2019.109605.
Sawyer, R.M., 1961. Effect of urbanization on storm discharge in Nassau County, Long Island, New York. USGS Professional Paper 475-C, 62-117, 185–187. http://pubs.er.usgs.gov/publication/ofr62117.
Schueler, T., 1995. The importance of imperviousness. Watershed Protection Techniques, 1(3), 100-111.
Schumm, S.A., Harvey, M.D., Watson, C.C., 1984. Incised Channels: Morphology, Dynamics, and Control. Water Resources Publications, Littleton, CO.
Schütte, S., Schulze, R.E., 2017. Projected impacts of urbanisation on hydrological resource flows: A case study within the uMngeni Catchment, South Africa. Journal of Environmental Management, 196, 527-543. https://doi.org/10.1016/j.jenvman.2017.03.028.
Seaburn, G.E., 1969. Effects of urban development on direct runoff to East Meadow Brook, Nassau County, Long Island, New York. Professional Paper 627B. http://pubs.er.usgs.gov/publication/pp627B.
Segura, C., Booth, D.B., 2010. Effects of Geomorphic setting and urbanization on wood, pools, sediment Storage, and bank Erosion in Puget Sound streams. Journal of the American Water Resources Association, 46(5), 972-986. https://doi.org/10.1111/j.1752-1688.2010.00470.x
Sheeder, S.A., Ross, J.D., Carlson, T.N., 2002. Dual urban and rural hydrograph signals in three small watersheds1. Journal of the American Water Resources Association, 38(4), 1027-1040. https://doi.org/10.1111/j.1752-1688.2002.tb05543.x.
Shields, F.D., Lizotte, R.E., Knight, S.S., Cooper, C.M., Wilcox, D., 2010. The stream channel incision syndrome and water quality. Ecological Engineering, 36(1), 78-90. doi: 10.1016/j.ecoleng.2009.09.014.
Simon, A., 1989. A model of channel response in disturbed alluvial channels. Earth Surface Processes and Landforms, 14(1), 11-26. https://doi.org/10.1002/esp.3290140103.
Simon, A., Rinaldi, M., 2000. Channel instability in the loess area of the midwestern United States. Journal of the American Water Resources Association, 36(1), 133-150. https://doi.org/10.1111/j.1752-1688.2000.tb04255.x.
Skaugen, T., Lawrence, D., Ortega, R.Z., 2020. A parameter parsimonious approach for catchment scale urban hydrology – Which processes are important? Journal of Hydrology X, 8, 100060. https://doi.org/10.1016/j.hydroa.2020.100060.

Smith, J.A., Baeck, M.L., Morrison, J.E., Sturdevant-Rees, P., Turner-Gillespie, D.F., Bates, P.D., 2002. The regional hydrology of extreme floods in an urbanizing drainage basin. Journal of Hydrometeorology, 3(3), 267-282. 10.1175/1525-7541. doi:(2002)003<0267:TRHOEF>2.0.CO;2.
Soar, P.J., Wallerstein, N.P., Thorne, C.R., 2017. Quantifying river channel stability at the basin scale. Water, 9(2), 133. https://doi.org/10.3390/w9020133.
Sohn, W., Kim, J.-H., Li, M.-H., Brown, R.D., Jaber, F.H., 2020. How does increasing impervious surfaces affect urban flooding in response to climate variability? Ecological Indicators, 118, 106774. https://doi.org/10.1016/j.ecolind.2020.106774.
Solins, J.P., Cadenasso, M.L., 2022. Urban runoff and stream channel incision interact to influence riparian soils and understory vegetation. Ecological Applications, n/a(n/a), e2556. https://doi.org/10.1002/eap.2556.
Sullivan, J., Grubb, S., Willis, R., Boozer, D., Flickinger, B., Dixon, C.E., 2020. Cohesive channel response to watershed urbanization: Insights from the Sand River, Aiken SC. Water (Switzerland), 12(12), 1-16. doi:10.3390/w12123441.
Sytsma, A., Bell, C., Eisenstein, W., Hogue, T., Kondolf, G.M., 2020. A geospatial approach for estimating hydrological connectivity of impervious surfaces. Journal of Hydrology, 591, 125545. https://doi.org/10.1016/j.jhydrol.2020.125545.
Taniguchi, K.T., Biggs, T.W., 2015. Regional impacts of urbanization on stream channel geometry: A case study in semiarid southern California. Geomorphology, 248, 228-236. doi:10.1016/j.geomorph.2015.07.038.
Taniguchi, K.T., Biggs, T.W., Langendoen, E.J., Castillo, C., Gudino-Elizondo, N., Yuan, Y., Liden, D., 2018. Stream channel erosion in a rapidly urbanizing region of the US–Mexico border: documenting the importance of channel hardpoints with Structure-from-Motion photogrammetry. Earth Surface Processes and Landforms, 43(7), 1465-1477. https://doi.org/10.1002/esp.4331.
Trimble, S.W., 1997. Contribution of stream channel erosion to sediment yield from an urbanizing watershed. Science, 278(5342), 1442-1444. doi:10.1126/science.278.5342.1442.
Tsihrintzis, V.A., Hamid, R., 1997. Modeling and management of urban stormwater runoff quality: A review. Water Resources Management, 11(2), 136-164. doi:10.1023/A:1007903817943.
Tyner, J.S., Yoder, D.C., Chomicki, B.J., Tyagi, A., 2011. A review of construction site best management practices for erosion control. Transactions of the ASABE, 54(2), 441-450. https://doi.org/10.13031/2013.36447.
United Nations, 2018. World Urbanization Prospects: The 2018 Revision [key facts]. https://population.un.org/wup/Publications/Files/WUP2018-KeyFacts.pdf.
Utz, R.M., Hilderbrand, R.H., 2011. Interregional variation in urbanization-induced geomorphic change and macroinvertebrate habitat colonization in headwater streams. Journal of the North American Benthological Society, 30(1), 25-37. doi:10.1899/10-007.1.
Van Sickle, D., 1962. The effects of urban development on storm runoff: The Texas Engineer. v. 32, no. 12.
van Vliet, J., Eitelberg, D.A., Verburg, P.H., 2017. A global analysis of land take in cropland areas and production displacement from urbanization. Global Environmental Change-Human and Policy Dimensions, 43, 107-115. doi:10.1016/j.gloenvcha.2017.02.001.
Vaughn, D.M., 1990. Flood dynamics of a concrete-lined, urban stream in Kansas City, Missouri. Earth Surface Processes and Landforms, 15(6), 525-537. https://doi.org/10.1002/esp.3290150605.
Vicars-Groening, J., Williams, H., 2007. Impact of urbanization on storm response of White Rock Creek, Dallas, TX. Environmental Geology, 51, 1263-1269. doi:10.1007/s00254-006-0419-6.
Vietz, G.J., Sammonds, M.J., Walsh, C.J., Fletcher, T.D., Rutherfurd, I.D., Stewardson, M.J., 2014. Ecologically relevant geomorphic attributes of streams are impaired by even low levels of watershed effective imperviousness. Geomorphology, 206, 67-78. https://doi.org/10.1016/j.geomorph.2013.09.019.
Vocal Ferencevic, M., Ashmore, P., 2012. Creating and evaluating digital elevation model-based stream-power map as a stream assessment tool. River Research and Applications, 28(9), 1394-1416. https://doi.org/10.1002/rra.1523.
Waananen, A.O., 1961. Hydrologic effects of urban growth - some characteristics of urban runoff. USGS Professional Paper 424-C, 353-356.
Walling, D.E., 1979. The hydrological impact of building activity: a study near Exeter. In: G.E. Hollis (Ed.), Man's Impact on the Hydrological Cycle in the United Kingdom. Geobooks, Norwich, pp. 181–198.

Walling, D.E., Gregory, K.J., 1970. The measurement of the effects of building construction on drainage basin dynamics. Journal of Hydrology, 11(2), 129-144. https://doi.org/10.1016/0022-1694(70)90099-5.
Walsh, C.J., Fletcher, T.D., Vietz, G.J., 2016. Variability in stream ecosystem response to urbanization: Unraveling the influences of physiography and urban land and water management. Progress in Physical Geography: Earth and Environment, 40(5), 714-731. doi:10.1177/0309133316671626.
Walsh, C.J., Roy, A., Feminella, J., Cottingham, P., Groffman, P., Morgan Ii, R., 2005. The Urban stream syndrome: Current knowledge and the search for a cure. Journal of the North American Benthological Society, 24, 706-723. doi:10.1899/0887-3593(2005)024\[0706:TUSSCK\]2.0.CO;2.
Wei, L., Hubbart, J.A., Zhou, H., 2018. Variable streamflow contributions in nested subwatersheds of a US midwestern urban watershed. Water Resources Management, 32(1), 213-228. doi:10.1007/s11269-017-1804-5.
Weitzell, R.E., Kaushal, S.S., Lynch, L.M., Guinn, S.M., Elmore, A.J., 2016. Extent of stream burial and relationships to watershed area, topography, and impervious surface area. Water, 8(11), 538, doi:10.3390/w8110538.
Whipple Jr., W., DiLouie, J., 1981. Coping with increased stream erosion in urbanizing areas. Water Resources Research, 17(5), 1561-1564. https://doi.org/10.1029/WR017i005p01561.
White, M.D., Greer, K.A., 2006. The effects of watershed urbanization on the stream hydrology and riparian vegetation of Los Peñasquitos Creek, California. Landscape and Urban Planning, 74(2), 125-138. https://doi.org/10.1016/j.landurbplan.2004.11.015.
Whitlow, J.R., Gregory, K.J., 1989. Changes in urban stream channels in Zimbabwe. Regulated Rivers: Research & Management, 4(1), 27-42. https://doi.org/10.1002/rrr.3450040104.
Wilcock, P.R., Kenworthy, S.T., 2002. A two-fraction model for the transport of sand/gravel mixtures. Water Resources Research, 38(10), 12-11-12-12. https://doi.org/10.1029/2001WR000684.
Wilson, K.V., 1967. A preliminary study of the effects of urbanization on floods in Jackson, Mississippi. USGS Professional Paper 575-D, 259–261.
Wohl, E., Mersel, M.K., Allen, A.O., Fritz, K.M., Kichefski, S.M., Lichvar, R.W., Nadeau, T.-L., Topping, B.J., Trier, P.H., Vanderbilt, F.B., 2016. Synthesizing the scientific foundation for ordinary high water mark delineation in fluvial systems. U.S. Army Corp of Engineers, Engineering Research and Development Center, Report ERDC/CREEL SR-16-5.
Wolman, M.G., 1967. A cycle of sedimentation and erosion in urban river channels. Geografiska Annaler: Series A, Physical Geography, 49(2-4), 385-395. doi:10.1080/04353676.1967.11879766.
Wolman, M.G., Schick, A.P., 1967. Effects of construction on fluvial sediment, urban and suburban areas of Maryland. Water Resources Research, 3(2), 451-464. https://doi.org/10.1029/WR003i002p00451.
Yang, G., Bowling, L.C., Cherkauer, K.A., Pijanowski, B.C., 2011. The impact of urban development on hydrologic regime from catchment to basin scales. Landscape and Urban Planning, 103(2), 237-247. https://doi.org/10.1016/j.landurbplan.2011.08.003.
Yang, G., Bowling, L.C., Cherkauer, K.A., Pijanowski, B.C., Niyogi, D., 2010. Hydroclimatic response of watersheds to urban intensity: An observational and modeling-based analysis for the White River Basin, Indiana. Journal of Hydrometeorology, 11(1), 122-138. doi:10.1175/2009jhm1143.1.
Yang, L., Xu, Y., Han, L., Song, S., Deng, X., Wang, Y., 2016. River networks system changes and its impact on storage and flood control capacity under rapid urbanization. Hydrological Processes, 30(13), 2401-2412. https://doi.org/10.1002/hyp.10819.
Yeo, I.-Y., Guldmann, J.-M., 2006. Land-Use Optimization for Controlling Peak Flow Discharge and Nonpoint Source Water Pollution. Environment and Planning B: Planning and Design, 33(6), 903-921. doi:10.1068/b31185.
Yoo, C., Cho, E., Na, W., Kang, M., Lee, M., 2021. Change of rainfall–runoff processes in urban areas due to high-rise buildings. Journal of Hydrology, 597, 126155. https://doi.org/10.1016/j.jhydrol.2021.126155.
Yorke, T.H., Herb, W.J., 1978. Effects of urbanization on streamflow and sediment transport in the Rock Creek and Anacostia River basins, Montgomery County, Maryland, 1962-74. Professional Paper 1003. http://pubs.er.usgs.gov/publication/pp1003.

Yousefi, S., Moradi, H.R., Keesstra, S., Pourghasemi, H.R., Navratil, O., Hooke, J., 2019. Effects of urbanization on river morphology of the Talar River, Mazandarn Province, Iran. Geocarto International, 34(3), 276-292. 10.1080/10106049.2017.1386722.

Yousefi, S., Pourghasemi, H.R., Rahmati, O., Keesstra, S., Emami, S.N., Hooke, J., 2021. Geomorphological change detection of an urban meander loop caused by an extreme flood using remote sensing and bathymetry measurements (a case study of Karoon River, Iran). Journal of Hydrology, 597, 125712. https://doi.org/10.1016/j.jhydrol.2020.125712.

Zhou, F., Xu, Y., Chen, Y., Xu, C.Y., Gao, Y., Du, J., 2013. Hydrological response to urbanization at different spatio-temporal scales simulated by coupling of CLUE-S and the SWAT model in the Yangtze River Delta region. Journal of Hydrology, 485, 113-125. https://doi.org/10.1016/j.jhydrol.2012.12.040.

Zhou, Z., Smith, J.A., Yang, L., Baeck, M.L., Chaney, M., Ten Veldhuis, M.-C., Deng, H., Liu, S., 2017. The complexities of urban flood response: Flood frequency analyses for the Charlotte metropolitan region. Water Resources Research, 53(8), 7401-7425. doi:10.1002/2016wr019997.

Chapter 4. Biogeochemical Processes in Stormwater Best Management Practices [WMO Article 208.4]

4.1 Introduction

Water pollution is defined in the Clean Water Act as the discharge of any substance that alters "the chemical, physical, biological, or radiological integrity of a water." Although the Clean Water Act mainly regulates pollution discharged from point sources such as wastewater reclamation plants or industrial facilities, urban stormwater discharges can come under regulation as well.

In its report on urban water quality issues, the National Research Council (2009) defines *regulated* urban stormwater as follows:

> "Stormwater" runoff is the water associated with a rain or snowstorm that can be measured in a downstream river, stream, ditch, gutter, or pipe shortly after the precipitation has reached the ground. For small and highly urban watersheds, the interval between rainfall and measured stormwater discharges may be only a few minutes. From a regulatory perspective, stormwater must pass through some sort of engineered conveyance, be it a gutter, a pipe, or a concrete canal. If it simply runs over the ground surface, or soaks into the soil and soon reemerges as seeps into a nearby stream, it may be water generated by the storm but it is not regulated stormwater. [Our] attention is focused mainly on that component of stormwater that emanates from those parts of a landscape that have been affected in some fashion by human activities ("urban stormwater").

Pollution of urban stormwater arises from this diversion of a large proportion of rainfall from its natural hydrologic flowpath, infiltrating into soils and slow groundwater discharge to streams, to routes that run over constructed surfaces (with their associated pollutants), through storm sewer systems, and discharge directly into surface waters. The stormwater acquires pollutants that accumulate on these constructed surfaces and together with the increased volume and rate of flow contribute to "urban stream syndrome" which refers to the seriously degraded condition of many urban streams and rivers. A survey of Maryland Piedmont watersheds found that they are measurably impacted when total impervious surface area exceeds 12% and severely degraded when imperviousness exceeds 30% (Klein, 1979).

From a water quality perspective, stormwater is precipitation (or snowmelt) that acquires additional solutes and particles as it contacts natural and constructed surfaces on the way to surface waters (Figure 55). These additional constituents may have accumulated on urban watershed surfaces between storm events or be derived from incremental dissolution of components of the constructed surfaces themselves into stormwater. The wide variety of materials and activities conducted in urban watersheds means that differences in stormwater composition are often observed between residential, commercial, and industrial catchments. Differences can also arise from the flowpath taken to the best management practices (BMPs) of interest because, for example, concrete and asphalt release different solutes. Urban soils can also be heavily compacted, causing some to resist water infiltration nearly as much as impervious surfaces.

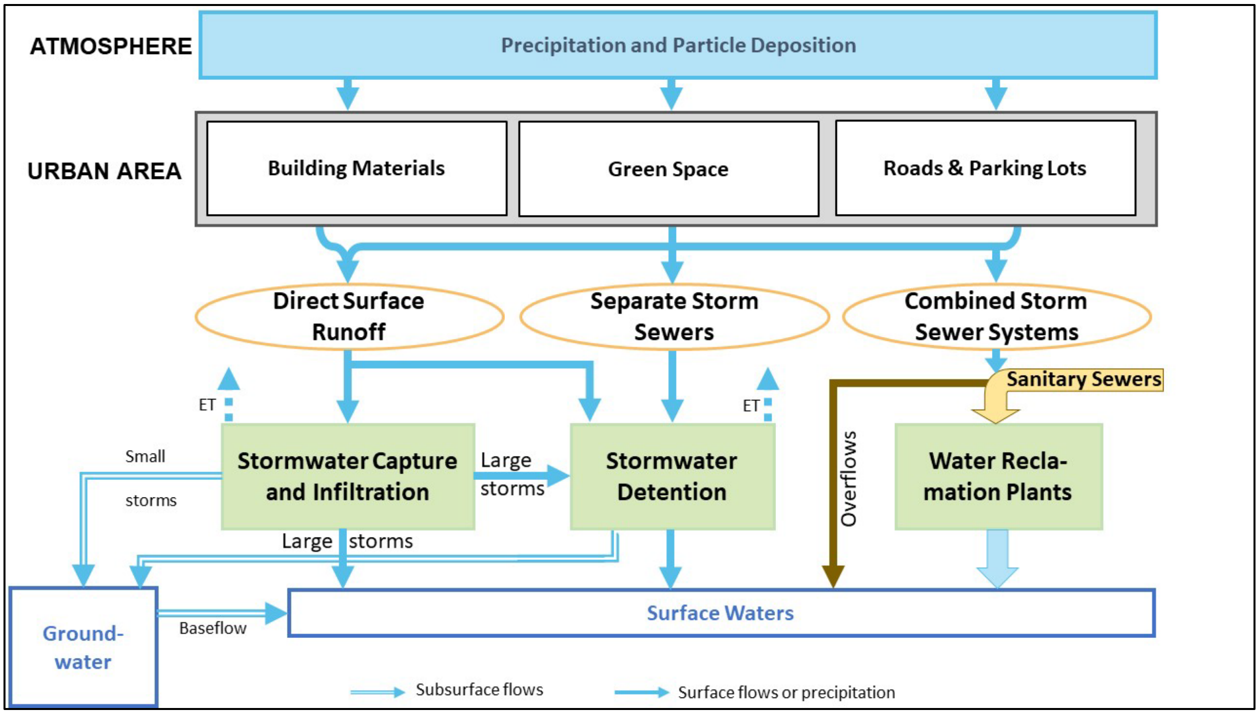


Figure 55. Pollutant sources and transport by stormwater in the urban hydrologic cycle. Atmospheric deposition includes substances emitted far away as well air pollution generated locally by vehicular traffic and industrial emissions to the atmosphere. The stormwater BMPs under consideration here affect stream water quality by a) enhancing infiltration or b) retaining stormwater in systems where further processing of pollutants can occur.

Stormwater is only in contact with these impervious surfaces for a short time compared to the time infiltrated water is in contact with soils, however. Thus, urban stormwater can acquire distinctive "pollutant cocktails" that are very different from runoff in natural landscapes (Kaushal et al., 2020), but it is usually diluted in the major solutes relative to stream water and shallow groundwater (Table 22), with the notable exception of snowmelt.

Table 22. Comparison of Some Water Quality Parameters at Various Stages of Stormwater Cycle

		Precipitation	*Stormwater Event Mean Concentration (EMC)*			
Constituent	*Units*	*Constituents (2000-2010)*	*without Snowmelt[a]*	*with Snowmelt[b]*	*Road Runoff[c]*	*Roof Runoff[c]*
Alkalinity ($CaCO_3$)	mg/L	-0.9	23.3			
*p*H		4.9	7.1		7.4	5.7
Conductivity (κ)	µS/cm	20	88		440	141
Cl^-	mg/L	0.19	6.1	397	130	7.7
DOC	mg/L	~1	10			
BOD as O_2	mg/L	Low	8.0	11.5	11-32	12
COD as O_2	mg/L	Low	52.4	44.7	106	66
Oil and Grease	mg/L	Low	4.0	3	4.5	0.7
TOC	mg/L	~1	12	15.2		
Turbidity	NTU	Low	19.4			
Total Susp Solids	mg/L			54.5	160	

[a] (Pamuru et al., 2022) [b] (Watershed Protection, 2003) [c] (Göbel et al., 2007)).

Mitigating urban stormwater pollution is an important goal because the concentrations of some pollutants in stormwater can exceed those in treated wastewater (LeFevre et al., 2015). As a result, stormwater is a major cause of water pollution in the United States and in the Chicago area. According to the U.S. Environmental Protection Agency's 2000 National Water Quality Inventory, "urban runoff/storm sewers" was identified as the cause of impairments in 13% of rivers and 18% of lakes officially designated as having water quality poorer than necessary to attain their designated uses (EPA, 2005). Total Maximum Daily Loads (TMDLs) in the area also identify stormwater as a significant contributor to surface water impairments for chloride and other pollutants in the Chicago area (See Chapter 7).

Measures taken to control stormwater, herein referred to as BMPs, are primarily aimed at reducing flooding, but have a secondary purpose of improving water quality in the waterways into which they discharge. The effectiveness of BMPs at reducing pollution loads has been documented in many cases (See Chapters 5 and 7). This chapter is aimed at building an understanding of how the processes acting within BMPs affect a wide variety of pollutants of interest. The discussion is necessarily general as the efficacy of BMPs regarding stormwater pollution varies with multiple factors, including BMP design, local conditions such as climate and soil properties, and the nature of the pollutant.

4.2 Chemical Constituents in the Urban Water Cycle

4.2.1 Introduction

Quantifying BMP performance with respect to water quality requires deriving a mass budget for pollutants of interest. For such budgets, it is generally adequate to compare measured inputs and outputs of the total mass of an element for inorganic pollutants, or the compound of interest for organic pollutants. The differences reflect their removal or retention. Of course, the distinction between pollutants in dissolved and suspended particulate forms is important to understand their performance as well. However, there are additional distinctions in the physicochemical states of pollutants and terms used to describe them that are needed to explain the inner workings of stormwater BMPs. This section is intended as an aid to those for whom these distinctions are not already well known.

To meet the monitoring requirements of the Clean Water Act, urban surface waters are commonly sampled and chemically analyzed to assess compliance with water quality standards. Less commonly, the composition of stormwater and groundwater in urban environments is characterized as well. Because these measurements are obtained to address different water quality issues at different levels of detail, there is no single set of analyses that is universally performed. Although this complicates interpretation of the data, it is possible to obtain a coherent understanding provided that the nature of the measurements and their relationships to the fundamental chemical entities present in the water are understood.

To this end, it is necessary to define what is meant by a water quality "constituent" and some additional terms that are used to characterize the various physicochemical states of a constituent as it may occur at different places and times in the urban water cycle. This is important because a *constituent* may be defined to include a variety of distinct states or *species* and it is the concentrations of these *species,* not necessarily the constituent, that determines how it reacts and is transported.

The term "constituent" is related to but not identical to a result of water analysis. Each analytical result typically expresses the concentration of an analyte, while a "constituent" may comprise just one analyte or may reflect the "totality of an element present without regard to the species." Constituents such as calcium and cadmium typically comprise only one analyte since each element only exists in the form of the elements' divalent cation (and related complexes). Total nitrogen levels in water, on the other hand, are often derived from data for at least two different analytes: nitrate + nitrite (NO_x-N), and total Kjeldahl nitrogen (TKN). Different phosphorus constituents may be reported as orthophosphate, reactive phosphorus, or total phosphorus. These distinctions reflect the occurrence of N and P in multiple forms that are not in equilibrium with each other. Other constituents, such as total dissolved solids, may be a composite of several different analytes that don't share a common element.

Another distinction that arises in monitoring water quality is between sample fractions. The reported concentration of any constituent must reflect the fraction of the sample that is analyzed, be it a whole, unfiltered water sample, the water that passes through a filter, or the suspended particles retained by a filter after passing the sample through it. The distribution of a constituent between dissolved and suspended particle fractions is crucial to assess both how it is transported and how reactive it is.

Finally, some constituents comprise a group of species related to each other through a system of equilibrium reactions, such as acid-base reactions, involving a single parent species (also called the component in water quality modeling). At least in cases where the concentrations and properties of the other constituents that engage in the side reactions are accurately known, the relative proportions of the parent and daughter species can be calculated from well-known mass conservation and equilibrium mass law equations (Brezonik and Arnold, 2022). Such calculations enable more accurate predictions of constituent reactivity, but only in cases where poorly-characterized organic matter is not of great importance.

4.2.2 Key definitions

Here, we introduce and define terms that appear throughout this chapter:

Fraction: a part of a sample defined by the process(es) it has been subjected to when preparing the sample for analysis, e.g., dissolved (filterable), suspended (retained by a filter), and total (or whole water sample).

Analyte: a substance whose concentration is measured by chemical analysis.

Constituent: Any distinct chemical substance, whether discharged as a pollutant or derived from natural sources. A constituent can comprise a number of parent and daughter species that are related by reactions at or near equilibrium. Constituents can also be defined to include multiple forms of an element not at equilibrium with one another.

Species: A distinct physicochemical state of a constituent distinguished by stoichiometry, charge, molecular structure, or phase. Common chemical formulas generally convey enough information to define one species. For example, the common species of water (H_2O) in the different phases of vapor, liquid, and solid ice are written as $H_2O(g)$, $H_2O(l)$, and $H_2O(s)$, respectively. For our purposes, oxygen (O_2) commonly occurs as a gas, $O_2(g)$, and an aqueous species dissolved in water, O_2(aq). Species can also be ionic or neutral, such as the aqueous species Hg^{2+}(aq) or Hg^0(aq).

Pollutant: a substance with deleterious effects on the integrity of a water that is present at elevated levels due to human discharges or perturbation of the environment.

Phase: a substance of homogenous chemical composition that exists in a contiguous volume.

Speciation: the quantitative description of a constituent's distribution among *species* at equilibrium with one another or an element between a variety of constituents that it forms.

4.2.3 Constituents and analytes

It might seem that the simplest way to describe the composition of water is by measuring the concentrations of the different elements it contains. However, total concentrations are not sufficient to describe the chemical behavior of some elements. Even aside from carbon, which occurs in a vast number of compounds, many elements occur in a variety of oxidation states. Each oxidation state may comprise one or more species distinguished by the numbers of H and O atoms contained or by net charge. Since the element oxygen has a higher affinity for electrons than nearly all other common elements, most elements effectively lose two electrons to each oxygen bonded to the element of interest. Similarly, hydrogen has the lowest affinity for electrons, and so H atoms increase the electron density of the main element in an inorganic compound. This net deficit or excess of electrons relative to its elemental state defines the main element's formal oxidation state in the constituent/parent species (Table 23) and exerts a large influence on the chemical behavior of the element. Thus, for elements that can be found in the environment in multiple oxidation states, chemists define and analyze a constituent specific to their corresponding parent species.

The most abundant dissolved constituents in urban waters are the inorganic ions formed from a small number of elements: C, N, S, Ca, Mg, Na, K, and Cl. The elements Ca, Mg, K, and Na all occur exclusively as monoatomic cations in a single oxidation state and are therefore referred to as major cations. Many trace metals, such as Cd, Pb, and Zn, also occur as monoatomic cations with single oxidation states, but others such as Fe, Mn, and Cu are commonly found in two or more oxidation states. The only molecular cation that can be found at levels comparable to the major ions is ammonium (NH_4^+), which can be quantitatively significant in precipitation.

In contrast, of the elements making up the major anions in natural waters, only one, Cl, exists almost solely in a single oxidation state (Cl^-), except when added as chlorine to disinfect wastewater. Nitrogen, sulfur, and carbon all occur in multiple oxidation states, though all three are dominated by their oxy-anions, NO_3^-, SO_4^{2-}, and HCO_3^-, in surface waters. Thus, each must be quantified separately from the total concentration of the element in surface waters.

Recall that reactions that convert an element from one stable oxidation state to another are termed "reduction/oxidation" or redox reactions. Experimental studies of these reactions under conditions where they are reversible, have yielded precise knowledge of their energetics. However, most of the redox reactions of interest in water quality occur slowly, if at all, and are effectively irreversible within a single environmental compartment. This stands in stark contrast to acid-base reactions, which equilibrate almost instantaneously. Because they are not at equilibrium, it is common to consider the redox reactions of an element as "transformations" of one constituent to another. It is also important to know that while there are some relevant abiotic redox reactions, such as the reactions of reduced iron (Fe^{II}) and copper (Cu^{I}) with molecular oxygen and related radicals, the most important redox reactions in natural and urban waters are mediated by microbes.

The key environmental factor determining which redox reactions take place within a system is the abundance of molecular oxygen. Where oxygen is in abundant supply, a condition termed oxic or aerobic, redox processes tend to oxidize elements. When oxygen is not present, a condition termed anoxic or anaerobic, the chemical state of certain elements can change dramatically. The shift between aerobic and anaerobic conditions is governed by the extent to which water is in contact with the atmosphere. Water in contact with the atmosphere is usually close to the temperature-dependent saturation level. When a soil layer or strata in a water body that contains organic matter is out of contact for enough time, oxygen will become depleted (see below).

*Table 23. Key Elements in Surface Water Chemistry and the Constituents Formed from Them. Cells in the table contain the parent species and oxidation states. All species without an indication of predominant phase are aqueous. "Major" cations and anions are in **bold print**.*

Element	*Parent Species $\mid$ Oxidation State*				
Ca	**Ca^{2+} $\mid$ II**				
Mg	**Mg^{2+} $\mid$ II**				
Na	**Na^{+} $\mid$ I**				
K	**K^{+} $\mid$ I**				
Cl	HClO $\mid$ +I	**Cl^{-} $\mid$ -I**			
O	O_2 $\mid$ 0	H_2O $\mid$ -II			
S	**SO_4^{2-} $\mid$ VI**	SO_3^{2-} $\mid$ IV	S_8(s) $\mid$ 0		H_2S $\mid$ -II
C	**HCO_3^{-} $\mid$ IV**	CO $\mid$ II	"CH_2O" $\mid$ 0	CH_3OH $\mid$ -II	CH_4(g) $\mid$ -IV
N	**NO_3^{-} $\mid$ V**	NO_2^{-} $\mid$ III	N_2(g) $\mid$ 0	**NH_4^{+} $\mid$ -III**	DON $\mid$ -III
P	PO_4^{3-} $\mid$ V	$(PO_4)_n^{n-}$ $\mid$ V	R-OPO_3^{2-} $\mid$ V		
Fe	Fe^{3+} $\mid$ III	Fe^{2+} $\mid$ II	Fe(metal) $\mid$ 0		
Mn	MnO_2(s) $\mid$ IV	Mn^{2+} $\mid$ II			
Cu	Cu^{2+} $\mid$ II	Cu^{+} $\mid$ I	Cu(metal) $\mid$ 0		
Cr	CrO_4^{2-} $\mid$ VI	Cr^{3+} $\mid$ III	Cr(metal) $\mid$ 0		

Organic compounds, both natural and anthropogenic, exhibit a diversity that is orders of magnitude greater than inorganic. To assess the impact of a BMP on any pesticide or organic pollutant, one simply treats it as a distinct constituent. In other cases, constituents may reflect an aggregate measure. Examples include dissolved organic carbon (DOC), biochemical oxygen demand (BOD), or metal-binding ligand concentration (L_1), which are all regarded as appropriate constituents in context.

4.2.4 Particulate and dissolved sample fractions

In this context, the most relevant distinction in the physicochemical state of a constituent in water is between its dissolved and suspended particle fractions. The importance of this distinction lies in the profoundly different behavior of the dissolved and particulate fractions of the constituent within BMPs. Note that because the particulate fraction is operationally defined and because suspended particles are a mixture of different materials, it is more accurate to refer to a particulate "fraction" than "phase."

"Dissolved" substances are defined as the fraction of a water sample capable of passing through whatever filter was employed in processing the water sample. Typical filters have nominal pore sizes in the range 0.4 to 1.0 micrometer (μm). Of course, the dissolved fraction includes constituents in their most chemically active state: true solutes. True solutes are ionic and molecular species that can exert direct effects on aquatic life, engage in reactions, volatilize, sorb onto or into particulate matter, and precipitate to form solids. Pollutants and solutes of natural origin vary greatly in their tendencies to enter those processes, and the differences between them will be discussed later.

Colloidal matter comprises very small particles that remain in suspension in stagnant water and can pass through standard-sized filters. Colloids are typically less reactive than true solutes but more reactive than large particles on account of their surface-to-volume ratio. This also makes them effective sorbents. Some colloids can even move through porous media.

Suspended particulate matter carried by urban stormwater exhibits a wide range of physical and chemical characteristics, as depicted in Figure 56.

- Suspended solids are typically taken to include particles with sizes from 0.5 to 60 μm. These include soil minerals, organic particles derived from the decomposition of plant debris or aquatic microorganisms, and particles derived from vehicles or the built environment. Note that minerals, which are denser than water, can be transported in suspension.
- Except for viruses, microorganisms generally fall in the same size range as detrital and mineral particles. Accordingly, the constituents of their biomass and pollutants they have absorbed are included in measurements of particulate matter. In stormwater, the main class of microorganisms are bacteria derived from fecal material deposited on surface soils by wildlife or human pets.
- Microplastics are plastic particles smaller than 5 mm in size. Microplastics are increasingly recognized as important pollutants in themselves, but they can also absorb hydrophobic compounds and aid in their transport (Werbowski et al., 2021)

- Gross particulate matter (> 5 mm) that floats on water comprises large pieces of litter discarded by humans as well as vegetation debris (Laurenson et al., 2013). In Australian cities, the former makes up 25–30% of the total gross pollutant load, while the remaining 70–75% is human-derived litter (Allison et al., 1998).

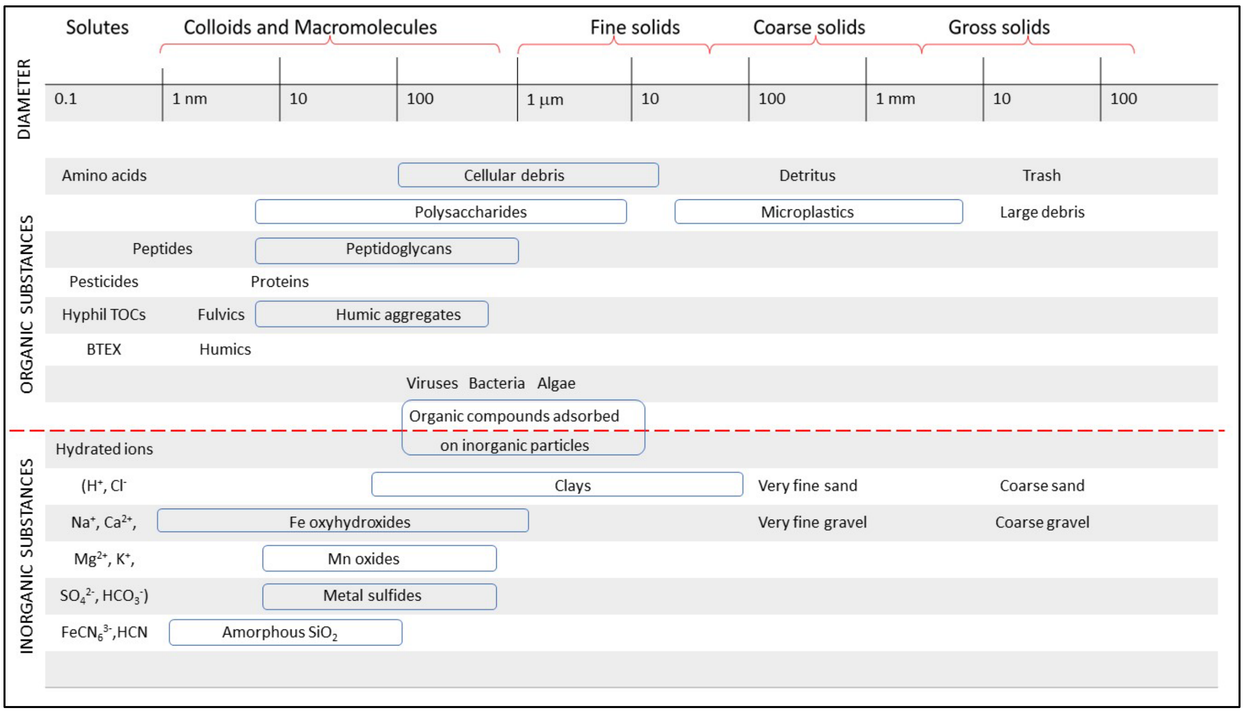


Figure 56. Physicochemical state of organic and inorganic substances commonly found in surface waters organized by size of molecule or the particle it is associated with (Lead and Wilkinson, 2006)

Finally, we note that fats, oil, and grease (FOG) also occur in surface waters in a variety of forms. Since these compounds are sparingly soluble and less dense than water, they tend to occur in films or blobs on the surface of water. FOG occurs in sanitary sewers as a by-product of food preparation and may be deposited on the walls of sewage systems. In stormwater, it can be derived from leakage from automobiles onto roadways and bridges or discharge from mishandling in automobile workshops (Husain et al., 2014; Bakr et al., 2020).

4.3 Equilibrium Speciation Reactions

4.3.1 Overview

A complete description of the chemistry of an element requires not only distinguishing between its different constituents, but also between species considered part of the same constituent. Chemists typically consider these to comprise a group with a parent (also called the component in water quality modeling) and daughter species formed by reversible reactions. At least in cases in which the concentrations and properties of the other constituents that engage in the side reactions are accurately known, the relative proportions of the parent and daughter species can be calculated from well-known mass conservation and equilibrium mass law equations (Brezonik and Arnold, 2020). Such calculations enable more accurate predictions of constituent reactivity. The full speciation of each constituent then is the quantitative description of its distribution among *species* at equilibrium with one another.

Note that organic constituents engage in some of these equilibrium reactions as well. Many undergo acid-base reactions and most sorb to some extent. Certain natural and synthetic compounds also make very good ligands that bind metal ions. Just as for inorganic species, these distinctions are important for understanding how different processes affect the transport and fate of an organic compound.

Although a full description of a constituent's speciation can get quite involved, certain distinctions made in water quality work may be quite familiar to the reader. These may include:

i) Acid-base speciation, which describes the relative concentrations of protonated and deprotonated forms of a substance. For example, ammonia occurs as ammonium and ammonia, NH_4^+ and NH_3, and orthophosphate species have several different protonation states: $H_3PO_4^0$, $H_2PO_4^-$, HPO_4^{2-}, and PO_4^{3-}.

ii) Complexation speciation is the extent to which certain solutes in water called ligands form "complexes" with a metal ion affect the speciation of both metal and ligand. The reversible "coordination bonds" that bind metals to ligands usually reduce both the toxicity (bioavailability) of metal ions and their tendency to sorb onto particle surfaces. Humic and fulvic acids are particularly effective natural ligands.

iii) Sorption speciation describes the relative concentrations of a substance in its dissolved and particle-associated forms (exclusive of any pure solid precipitate phases). It is well known that many metals tend to bind strongly to both mineral and organic particles, and hydrophobic organic compounds are mainly absorbed in particulate organic matter. Since such sorbed pollutants are transported as particles, it is important to distinguish the relative amounts of dissolved and sorbed species in water.

iv) Minerals and certain organic compounds can form distinct solid phases and thus be included in particulate fractions. A well-known example is iron oxide, which can occur as the mineral ferrihydrite ($Fe(OH)_3$) or in a variety of dissolved iron species in the +III oxidation state.

It is also important to remember that some constituents may not remain fixed in the form of one species or even oxidation state as it passes from precipitation to stream water. A constituent may be transported in stormwater mainly as one species but exert its effects after re-equilibrating or even transforming into a different constituent within a BMP. For example, a metal adsorbed to suspended soil particles might be carried in stormwater but then exert an impact after desorbing as an aqueous cation.

4.3.2 Acid-base reactions

Since a hydrogen ion is simply a proton with no electron cloud around it, it can approach negative ions and electron-dense parts of molecules closely due to the large attractive forces. As a result, hydrogen ions don't exist free in aqueous solution, but only in association with ions and molecules known as "bases" or "proton acceptors." Water itself has two lone pairs of electrons, making it an excellent "base." Many other inorganic and organic compounds, especially ionic compounds, are bases as well. The bases in an aqueous solution all compete for the supply of available protons, as in this reaction for a generic base (A^-):

$$H_3O^+ + A^- \longleftrightarrow HA^0 + H_2O$$

Protonated molecules, such as HA^0 are generally capable of donating protons to other proton-accepting molecules, which makes them "acids."

In water quality work, the reactions of the carbonate species comprise a well-known acid-base system as are the three protonated daughter species formed from the parent orthophosphate ion (PO_4^{3-}). The ammonium/ammonia conjugate acid-base pair engage in another important acid-base reaction that governs the water quality standard for aquatic life.

$$NH_4^+ \longleftrightarrow NH_3(aq) + H^+$$

The *p*H of a water influences, to varying degrees, most of the water quality constituents of interest here or is influenced by them in some way. Some of the main interactions are listed in Table 24.

Table 24. pH Interactions with Water Quality Parameters of Concern

Category	*Monitored parameters*	*Effects and interactions with pH*
Solids	Turbidity/Appearance	*p*H affects coagulation of particles
	Suspended solids	*p*H affects tendency of metals to adsorb to particles
Salts	Chloride	Chloride may enhance weathering and thus alter *p*H
Inorganics	Iron	Low *p*H strongly increases Fe(III) solubility High *p*H speeds oxidation of Fe(II)
	Silver	
	Cyanide	Hydrogen cyanide (HCN) is more volatile and more toxic than CN^-, which is prevalent at higher *p*H
Nutrients	Nitrogen (various forms)	Nitrification of ammonia lowers *p*H Denitrification raises *p*H Ammonia speciation affects toxicity
	Phosphorus	Adsorption of orthophosphate is *p*H-dependent
Organic compounds	Fats, Oils, Grease	
	Hydrocarbons	Little effect
	Pesticides	Adsorption of ionizable compounds affected by *p*H Hydrolysis rates affected by *p*H
	BTEX	Little effect
Other	*p*H	Governed by alkalinity, dissolved inorganic carbon, and other weak acids
	DO	Photosynthesis and respiration affect dissolved CO_2, which affects *p*H

4.3.3 Metal complexation

In water, metal cations undergo easily reversible "coordination" reactions with other ions and molecules that are capable of sharing electrons with them. Some metal-ligand interactions are largely ionic in nature and lead to the formation of ion-pairs:

$$Na^{+} + Cl^{-} \longleftrightarrow NaCl^{0}$$

As noted above, water molecules have two lone pairs of electrons that can be shared with metal ions. Due to the abundance of water molecules, metals become hydrated (form coordination bonds with water) when dissolving into aqueous solution. In some cases, the metal-water bond is strong enough to cause the hydrated water molecule to lose a proton to other water molecules in solution. Such is the case for Fe^{3+}:

$$Fe^{3+} + H_2O \longleftrightarrow FeOH^{2+} + H^{+}$$
$$Fe^{3+} + 2H_2O \longleftrightarrow Fe(OH)_2^{+} + 2H^{+}$$
$$Fe^{3+} + 3H_2O \longleftrightarrow Fe(OH)_3^{0} + 3H^{+}$$
$$Fe^{3+} + 4H_2O \longleftrightarrow Fe(OH)_4^{-} + 4H^{+}$$

Other inorganic ions, especially carbonate, phosphate, and ammonia, can also form coordination bonds. One of the strongest inorganic complexes, aside from some formed with bisulfide, is the reaction of chloride with ionic mercury (Hg^{2+}):

$$Hg^{2+} + Cl^{-} \longleftrightarrow HgCl^{+}$$
$$Hg^{2+} + 2Cl^{-} \longleftrightarrow HgCl_2^{0}$$

In addition to the above two complexed mercury species, the $HgCl_3^{-}$ and $HgCl_4^{2-}$ complexes also form at high chloride levels. However, mercury and silver are among the very few trace metals for which chloride complexes constitute a significant fraction of the total dissolved metal concentrations at the chloride levels in stream water.

When the concentration of inorganic constituents that act as significant ligands, mostly carbonate, chloride, phosphate, and hydroxide (*p*H), is known, it is possible to make accurate calculations of the equilibrium speciation of metal ions. However, such calculations are frequently complicated by the presence of organic ligands. In treated wastewater, synthetic ligands such as Ethylenediaminetetraacetic acid (EDTA) likely are present, though rarely analyzed. In runoff from soils or wetlands, natural dissolved organic matter (DOM) is generally present. It is well established that the anions of humic and fulvic acids in DOM also act as metal-binding ligands, such as shown in this reaction of the copper(II) ion.

$$Cu^{2+} + H - Humate^{-} \longleftrightarrow Cu - Humate + H^{+}$$

4.3.4 Sorption

Sorption is a chemical process that causes constituents to accumulate on reactive surfaces (adsorption) or within particulate matter (absorption). The extent of *adsorption* is often constrained by a finite capacity of the sorbent (particles or solid media) to bind the pollutant under consideration; *absorption* is less often constrained in this way. Although the precipitation of an insoluble mineral or organic compound to form a separate phase is not the same as sorption, a precipitated constituent is difficult to distinguish from one that is sorbed in routine monitoring data.

Sorption onto mixed particles is commonly modeled as a reversible process at equilibrium with an aggregate "partition coefficient" that represents all the phases present:

$$X(aq) + Particle \xleftrightarrow{K_P} X(part)$$

The equilibrium equation is written:

$$K_P \equiv \{X_{part}\} / [X_{aq}]$$

where $\{X_{part}\}$ is the concentration of X in particulate matter (mg-X kg-particles^{-1}), and $[X_{aq}]$ is its concentration in the dissolved phase (mg-X L^{-1}).

4.3.4.1 Absorption

Hydrophobic organic compounds are those that are uncharged with minimal polar parts. They absorb most strongly into particles with a high organic matter content, in the same way that organic solvents extract hydrophobic compounds from water. As a result, the mathematical relationship between the concentrations of dissolved and particulate fractions of a substance can be described accurately by a linear isotherm with a slope that is often described as an equilibrium "partition coefficient" or K_P. The partition coefficients (K_p) used to model the distribution of hydrophobic organic compounds between water and soil or sediment particles have been shown to depend on the organic matter content of the particles (Schwarzenbach et al., 2002):

$$K_p = K_{oc} \cdot f_{oc}$$

Furthermore, a compound's tendency to absorb (K_{oc}) into natural particulate organic matter (POM) is often related to its octanol-water partition coefficient (K_{ow}) via a simple linear free energy relationship, such as:

$$\log K_{oc} = A \cdot \log K_{ow} + B$$

Note that changes in the protonation state of a neutral organic constituent can cause it to acquire a charge and make it less prone to hydrophobic absorption.

4.3.4.2 Adsorption

Not every organic compound of interest here is hydrophobic, of course (Spahr et al., 2020). Some neutral organics are hydrophilic, i.e., polar enough that they remain as aqueous species rather than partition into organic solvents or POM. Other organic molecules are weak acids or bases with significant fractions present in the form of charged ions at ambient *p*H. Such molecules prefer to remain in solution, but can bind to reactive sites on mineral, especially metal oxide, surfaces. Often, the same sites also bind anions such as phosphate and arsenate. Metal ions can also bind to such mineral surfaces and to surface groups in particulate organic matter (Brezonik and Arnold, 2022).

For the constituents that bind to particular sites on particle surfaces rather than become absorbed into a hydrophobic phase, the adsorption process is qualitatively different. Adsorption of metal ions is often described as "surface complexation," which reflects the distinct chemical species that form on the surfaces of sorbent minerals. Here, the reactions of a surface group on ferric oxyhydroxide particles (≡Fe-OH) required to form a surface complex of copper are shown:

$$\equiv Fe-O-H \longleftrightarrow \equiv Fe-O^{-} + H^{+}$$

$$Cu^{2+} + \equiv Fe-O^{-} \longleftrightarrow \equiv Fe-O-Cu^{+}$$

As implied by these equations, the affinity of ferric oxyhydroxide surfaces for sorbates depends on their protonation state, and hence on the solution *p*H and the extent of their rection with other adsorbable species. Surface complexation of metal cations is also aided by the development of a negative surface charge. Of course, cations that are associated with surfaces of opposite charges but not in an actual surface complex are said to be satisfying the cation exchange capacity of the particulate media (soil).

The fact that adsorption occurs via binding to particulate sites implies that such mineral particles have a finite capacity to bind metal ions and other solutes. Since the mineral surface is saturable, the relationship between dissolved and adsorbed species is non-linear and often described by a Langmuir isotherm (Brezonik and Arnold, 2022; Erickson et al., 2013).

Finally, it is important to note that the partition coefficient for constituents that are adsorbed by surface complexation are quite complicated to calculate from the first principles. They must reflect 1) the aqueous speciation of metal ions and acid-base speciation of other ionizable sorbates, 2) the effects of competing sorbates, and 3) the surface speciation and charge of the mineral surface. Perhaps the single most important factor to recall is that extensive complexation in the aqueous phase will cause metal ions to sorb less.

4.3.4.3 Summary

At equilibrium, constituents attain a distribution between aqueous solution and the different forms of particulate matter suspended in or in contact with the water. Although enough is known to predict which constituents are generally most likely to sorb, the properties of the sorbent phase are often poorly known, even when the concentrations of particulate organic matter and inorganic elements are established. For example, particulate iron occurs in a variety of phases with different surface properties and affinities for sorbates. Thus, it is difficult to predict how strongly metal and phosphate ions will bind to inorganic surfaces.

Nevertheless, the qualitative understanding that comes from this knowledge can help explain trends in field data, such as differences in the distribution of a constituent between particulate and dissolved phases under different *p*H or dissolved oxygen conditions.

Finally, it should be recalled that rapid equilibrium sorption steps are often followed by 1) slow diffusion within particle aggregates, or 2) formation of increasingly strong solute-particle bonds that causes the slow formation of passivated forms of pollutants.

4.3.5 Mineral precipitation

Most inorganic constituents are capable of precipitating as a salt or mineral phase. For major ions, with the exception of calcium carbonate, the solubility limits are too high to be relevant in freshwater environments. The most relevant inorganic precipitate is ferric iron, which is the dominant oxidation state of iron in oxic waters and is highly insoluble. Iron(III) can also form colloidal particles in association with humic and fulvic acids that exert effects on the reactivity of other metals.

4.4 Acid-Base Effects in River and Stream Water Quality

4.4.1 Introduction

Long-term water quality monitoring data are widely used to assess changes in pollutant loadings from both point and non-point sources. Since multiple sources typically contribute to the load of most constituents in stream water, accurately distinguishing them requires careful construction of constituent budgets. For the case of urban stream water, it is clear that substantial pollutant loads must come from stormwater, but the high variability in observed pollutant concentrations and the large number of different discharges make quantification difficult.

A second approach to identifying a stormwater fingerprint in water quality is to examine how constituent concentrations depend on stream discharge. Such a dependence is expected because of the different paths taken to the stream by rainwater that enters the storm sewer networks and that which infiltrates into soils and is discharged as baseflow (groundwater).

Hydrologists commonly divide streamflow time-series data, known as hydrographs, into stormflow and baseflow components, with the former comprising the peaks following rainstorms and the latter the relatively steady flows that occur between storms. National datasets even assign average fractions of streamflow that is baseflow to every point on the national stream network (Wolock and McCabe, 1999). Regardless of the exact fraction at any location in the District, clearly stormwater has some effect, and it should be most apparent under relatively high flow conditions.

4.4.2 Acid-base equilibria and *p*H

Acid-base reactions comprise the quintessential equilibrium reactions in water. Through their control of the *p*H, they affect 1) the exchange of reactive gases with the atmosphere, 2) the dissolution and precipitation of minerals and concrete, 3) the rates of many transformation reactions, 4) the toxicity of metals and other ionizable toxicants, and 5) nearly every other class of equilibrium reactions noted above.

4.4.2.1 Definition

*p*H is a widely reported measure of the chemical potential of hydrogen ion (H^+) in natural and engineered waters. It is defined as the negative logarithm of the activity of hydrogen ions in that water:

$$pH \equiv -\log\{H^+\}$$

In precipitation and most streams and bodies of freshwater, the numerical values of H^+ activity and molar concentration are essentially equivalent, so it is common to define *p*H in fresh waters in terms of the molar concentration:

$$pH \cong -\log[H^+]$$

Since a hydrogen ion is simply a proton with no electron cloud around it, it is orders of magnitude smaller in size than all other ions. For this reason, the proton can approach other ions and molecules more closely, making the force attracting it to any lone pair of electrons large. As a result, hydrogen ions don't exist free in aqueous solution, but only in association with ions and molecules known as “bases” or “proton acceptors.”

Water itself has two lone pairs of electrons, making it an excellent base. As the most abundant base in any aqueous solution is water, chemists define a proton in association with water, or the hydronium ion (H_3O^+), as its reference state. Therefore, while the *p*H is commonly taken to reflect the concentration of protons, it worthwhile to recall that:

$$pH \cong -\log[H^+] = -\log[H_3O^+]$$

Other molecules in surface waters that protons commonly bind to include carbonate and bicarbonate ions, ammonia, phosphate, and many organic bases. The bases in an aqueous solution all compete for the supply of available protons, as in this generic base (A^-) reaction:

$$H_3O^+ + A^- \xleftrightarrow{1/K_a} HA^0 + H_2O$$

Protonated molecules, such as HA^0, are generally capable of donating protons to other proton-accepting molecules, which makes them acids. Chemists call the two species A^- and HA^0 a "conjugate acid-base pair" with A^- being the base and HA^0 the acid.

4.4.2.2 Acid-base reactions

The proton transfer reactions between acids and bases in water rapidly reach equilibrium, so it is simple to calculate the distribution of protons among their various states of protonation using simple chemical equilibrium models. For such models, it is convenient to express all equilibrium reactions as dissociation of acids:

$$HA^0 + H_2O \xleftrightarrow{K_a} H_3O^+ + A^-$$

Strong acids have pK_a values near or less than zero. Strong bases have pK_a values approaching 14. To evaluate the protonation state of an acid at a particular *p*H value, chemists generally speak of the negative logarithm of this acid dissociation constant (K_a) or pK_a:

$$pK_a = -\log K_a$$

When the *p*H is less than an acid's pK_a, the acid is the predominant form of the acid-base couple, e.g., HA^0. When the *p*H is greater than an acid's pK_a, the predominant form is the conjugate base, e.g., A^-.

Most of the major ions in surface waters are derived from strong bases or strong acids that dissociate completely (See Major Ions section below). The acid anions are formed from the dissociation of strong acids, such as sulfuric acid formed in the atmosphere:

$$H_2SO_4^0 \longrightarrow 2H^+ + SO_4^{2-}$$

None of the major acid anions – SO_4^{2-}, NO_3^-, and Cl^- – is protonated to a significant extent in the *p*H range of most natural waters (*p*H 4-9). The base cations – Ca^{2+}, Mg^{2+}, Na^+, K^+ – are derived from the weathering of primary and secondary minerals that typically yield bicarbonate ions when in contact with water and atmospheric CO_2:

$$2Albite + 2CO_2 + 11H_2O \longrightarrow 2Na^+ + 2HCO_3^- + 4H_4SiO_4 + Kaolinite$$

$$CaCO_3(s) + CO_2 + H_2O \longrightarrow Ca^{2+} + 2HCO_3^-$$

Thus, the terms "base cation" means it was added to water as a base, not that the cation is a base. Similarly, "acid anion" means the anion was added to solution as an acid, not that the anion is an acid.

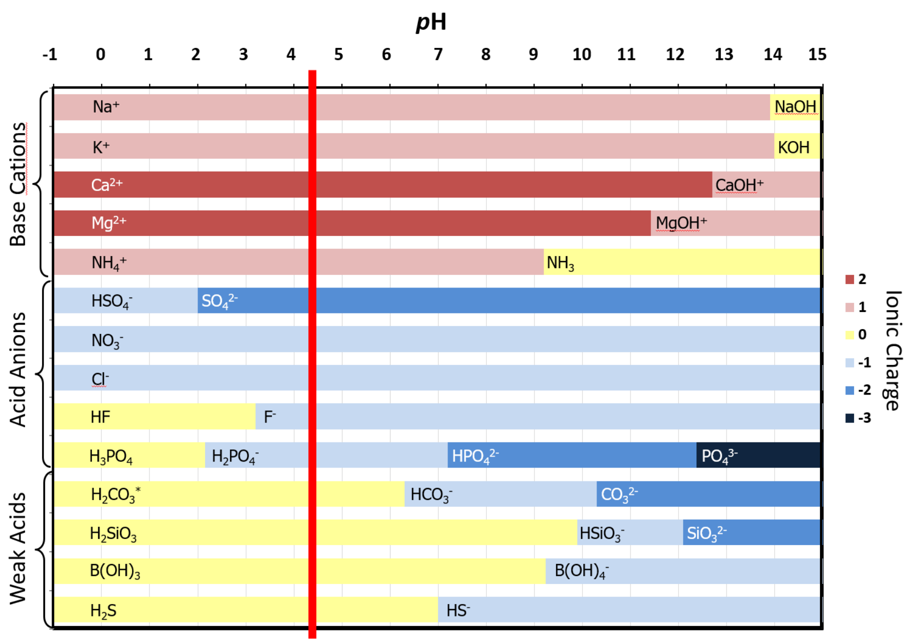


Figure 57. pH ranges over which different acid-base species of major ions and key weak acids in surface waters predominate. Note that the more highly protonated species (acidic) of a group are on the left side of the figure and the most deprotonated (basic) are on the right. The red line corresponds roughly to the CO_2 equivalence point, which defines the completion of an alkalinity titration.

4.4.2.3 pH scale

*p*H values mostly fall in the range from 0 to 14. A *p*H value of 7 is considered neutral, as it is the *p*H of pure water without added acids or bases. *p*Hs lower or higher than 7 are considered acidic or alkaline, respectively. However, a pure water at *p*H 7 is not the relevant reference for natural surface water systems since water in contact with the atmosphere absorbs CO_2. At equilibrium with the atmosphere, dissolved CO_2 into pure water lowers its *p*H to the mildly acidic range near ~5.6.

4.4.2.4 Measurement

*p*H can easily be measured using potentiometric or colorimetric methods. The potentiometric method employs a voltmeter to measure the potential of an ion-selective electrode that can be immersed in water. Using a ruggedized probe, *p*H can also be monitored in situ. In some cases, it is convenient to measure *p*H using indicator dyes that change color depending on the *p*H. Such dyes can either be added to a solution or embedded in paper test strips.

4.4.2.5 Effects of pH alteration

The *p*H of stormwater can exert a multitude of effects on aquatic life in receiving waters, both via its direct effects on the solutes and surfaces in contact with the water or via the dissolution and precipitation of materials on the way to streams. Different effects on the many constituents in natural waters are typically observed depending on whether the *p*H is raised or lowered (USEPA, 2017). Both raised and lowered *p*H can harm aquatic life by causing a worsened condition, slower growth, changed behavior, and an increased susceptibility to other stressors. These effects can cause mortality, reduced reproductive success, and ultimately population sizes and community structure.

Although there are cases in which stormwater can acquire acidic constituents, more typically it has a higher *p*H and alkalinity than rainwater due to the basic solutes it gains as it contacts concrete and other surfaces in urban watersheds.

The effects on biota of *p*H changes arise from the fact that the *p*H determines the relative abundance within groups of species, comprising many water quality constituents, that reversibly interconvert among themselves. Examples of such constituents include carbonate, ammonia, aluminum ion, and many organic compounds (Figure 57).

4.4.2.6 Toxicity of neutral species

Many constituents that undergo reversible protonation reactions have one electroneutral species. Such species can passively diffuse through cell membranes and thus can be absorbed into organisms whether they are beneficial or not. Ions, on the other hand, are generally much less permeable and are absorbed by specific ion carriers that are under physiological control.

Consider the case of ammonia toxicity. The water quality standard for ammonia defines the maximum concentration considered safe for aquatic life. Note that the standard is *p*H dependent (Figure 58) with more stringent (lower acute exposure) standards at a higher *p*H. This results from the following acid-base equilibrium and unregulated, passive uptake of the neutral NH_3 species through the gills, while the charged NH_4^+ cannot:

$$NH_4^+ \longleftrightarrow NH_3(aq) + H^+$$
$$NH_3(aq) \xleftrightarrow{\text{Passive Transport}} NH_3(fish)$$

The plot of the fraction of $[NH_4]_T$ present in neutral species shows that the fractional abundance of $NH_3(aq)$ rises sharply over the same *p*H range that the WQ standard declines.

Other water quality constituents of interest also exhibit *p*H-dependent toxicity. For example, the neutral, unionized forms of several phenols were more toxic to a guppy than their anionic forms. As a result, they were more toxic at low *p*H than at neutral to high *p*H (Saarikoski and Viluksela, 1981). Similarly, the neutral hydrogen cyanide acid species (HCN) predominates *p*H values less than its pK_a of 9 and is about twice as toxic as its anionic conjugate base (CN^-). The latter only comprises an appreciable fraction of the total cyanide above about *p*H 8 (Rand, 1995).

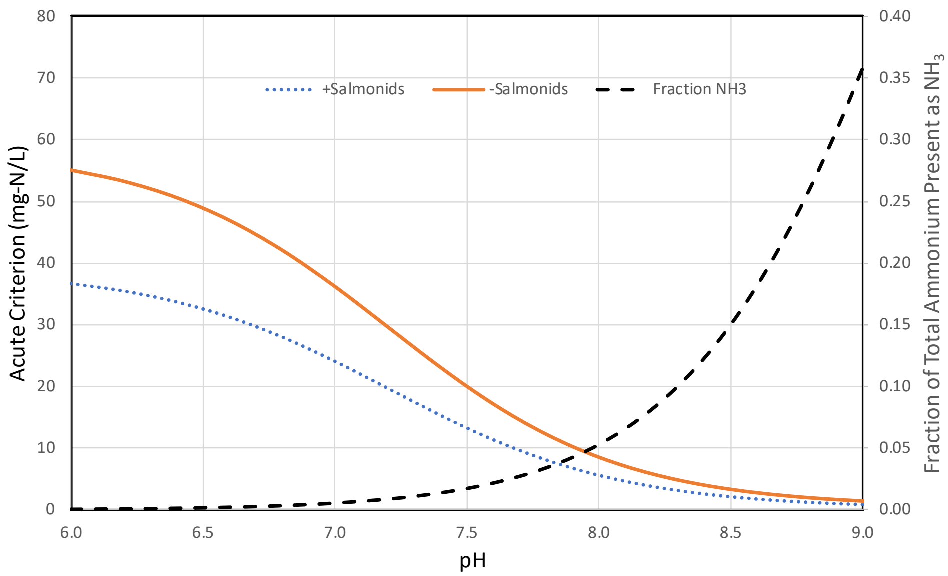


Figure 58. Influence of ammonia speciation on its acute toxicity. As the fraction of total ammonia present in the neutral NH3 form increases with increasing pH, the overall toxicity of the total ammonia increases. Thus, the criterion value must be set lower to protect fish and other aquatic life at higher pH.

4.4.2.7 Metal solubility

Common interactions to consider with *p*H involve metals (e.g., aluminum, copper, zinc). Acidic runoff mobilizes and leaches metals from soils and surfaces into streams, resulting in orders of magnitude higher dissolved metal concentrations at low *p*H. Metals whose solubilities are limited by the formation of hydroxide minerals, such as aluminum and iron(III) become increasingly bioavailable with decreasing *p*H (< 6.0) according to this solubility equilibrium for gibbsite:

$$Al(OH)_3(\mathrm{s}) + 3H^+ \longleftrightarrow Al^{3+} + 3H_2O$$

The Al^{3+} ion undergoes further hydrolysis reactions at *p*H values near neutrality:

$$Al^{3+} + H_2O \longleftrightarrow AlOH^{2+} + H^+$$
$$Al^{3+} + 2H_2O \longleftrightarrow Al(OH)_2^+ + 2H^+$$
$$Al^{3+} + 3H_2O \longleftrightarrow Al(OH)_3^0 + 3H^+$$
$$Al^{3+} + 4H_2O \longleftrightarrow Al(OH)_4^- + 4H^+$$

Although the formation of Al-hydroxy complexes at higher *p*H significantly offsets the extremely strong *p*H dependence, circumneutral waters typically do not contain significant levels of dissolved aluminum due to the limit $Al(OH)_3$(s) formation places on its solubility.

As with the deprotonation of hydrogen cyanide, the hydrolysis of aluminum ions results in lower toxicity of dissolved Al to algae at neutral than at low *p*H (Gensemer and Playle, 1999). This effect is incorporated in the biotic ligand model.

4.4.2.8 Adsorption onto particles

The *p*H exerts a strong influence on the protonation state of ionizable surface moieties of many types of particles and solids in contact with water, which in turn affects their surface charge. Both the charge and protonation state of surface groups strongly affect the tendency of metals and other ionic substances to bind to solid surfaces. (See Metals section below).

4.4.2.9 Gas exchange

Important volatile compounds that also happen to be ionizable acids or bases include carbon dioxide, ammonia, and many organic compounds. Since only neutral compounds can be transferred from water to air, the volatilization of ammonia is strongly favored at *p*H near or above its pK_a of 9.25:

$$NH_4^+ \longleftrightarrow H^+ + NH_3(aq) \xleftrightarrow{volatilization} NH_3(g)$$

Of course, the most important *p*H-dependent gas exchange process is that of carbon dioxide (see below), which is favored at low *p*H. Other compounds that undergo *p*H-dependent volatilization include cyanide and hydrogen sulfide.

4.4.2.10 Bioavailability of metal ions

The concentration of hydrogen ions in water exerts multiple effects on metal uptake by aquatic biota. As postulated by the Biotic Ligand Model, hydrogen ions both directly compete for uptake with metals at the gill surface and shift the fraction of metal ions in water present in their most bioavailable, free ionic forms. Thus, just as ammonia becomes more toxic at high *p*H, some metals become more toxic at low *p*H. (See Metals section below.)

4.4.2.11 Factors governing the pH

Many materials in the environment dissolve upon contact with stormwater, forming ions and molecules that potentially change its *p*H. Solutes that completely dissociate with a net release or consumption of protons are considered “strong” acids or bases, respectively, and those that partially dissociate are considered “weak.” The excess of strong bases over acids in a water can be measured by titration and is reported as its alkalinity (or equivalently its acid-neutralizing capacity):

$$ALK \equiv [OH^-] - [H^+] + [HCO_3^-] + 2[CO_3^{2-}]$$

When the alkalinity of a water is negative, its excess of strong acids over bases is referred to as its “mineral acidity.” Usually, the largest source of strong acids in stormwater is atmospheric deposition (both wet and dry), which contains nitric and sulfuric acids derived from the combustion processes employed in energy production and transportation. However, there are also processes occurring within urban stormwater systems and soils that yield acids, including the oxidation of ammonia to form nitrate:

$$NH_4^+ + 2O_2 \xrightarrow{microbes} NO_3^- + 2H^+ + H_2O$$

and the oxidation of reduced sulfur minerals to produce sulfate. Though it would not be common for urban systems, pyrite in soil can become exposed and oxidized, leading to the production of sulfuric acid in some watersheds.

Strong bases are mostly derived from the dissolution of minerals in dust, soil, and concrete, which can make stormwater more alkaline than precipitation. Concrete contains a mixture of mineral components, including $CaO \cdot SiO_2 \cdot 2H_2O$ and $Ca(OH)_2$ or $CaO{\cdot}H_2O$ (Jiang et al., 2014; Xie et al., 2004). The calcium oxide (CaO) they contain can be weathered by acidic precipitation to yield Ca^{2+} ion and sulfate in equivalent amounts:

$$CaO(\text{concrete}) + H_2SO_4 \longrightarrow Ca^{2+} + SO_4^{2-} + H_2O$$

Weak acids such as CO_2 can also acidify the extremely basic porewater in concrete and contribute to CaO(concrete) dissolution as well.

Strong acids and bases determine a water's alkalinity, but the observed *p*H is equally dependent on the concentrations of weak acids and bases. The main weak acid that affects surface water resides in the atmosphere. Atmospheric carbon dioxide (CO_2) exchanges rapidly enough between the air and any water exposed to air that concentrations in air and water tend to remain near equilibrium with each other (Figure 59):

$$CO_2(g) \longleftrightarrow CO_2(aq)$$

Since the alkalinity of most surface waters falls in the range from 5 to 250 mg-$CaCO_3$/L, their *p*H values tend to remain between about 6 and 9. Note that without carbon dioxide, the *p*H of waters with typical positive alkalinities of over 50 mg-$CaCO_3$/L would be near 11. Below alkalinities of about 5 mg-$CaCO_3$/L, the *p*H drops steeply to the 5-6 range, as observed in current precipitation. In waters with negative alkalinity, such as most pre-2000 precipitation, the *p*H is so low that carbon dioxide does not dissociate and so mostly doesn't influence the *p*H.

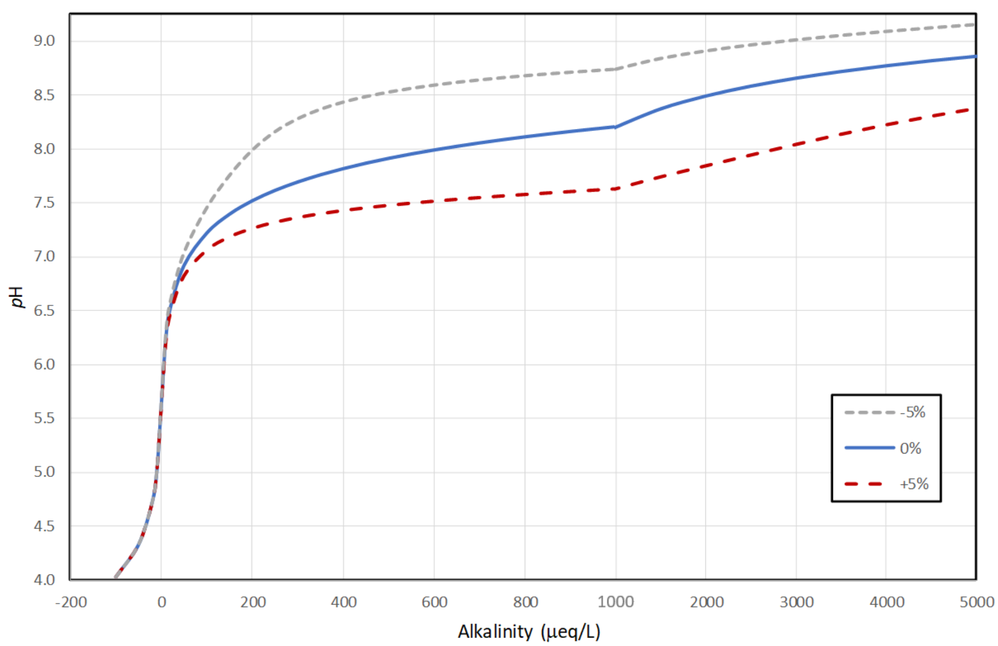


Figure 59. Dependence of pH on alkalinity for water at equilibrium with atmospheric CO_2 at a concentration of 400 parts per million by volume (solid blue line). Dashed lines are plotted for 5% more dissolved inorganic carbon (DIC) (red long-dashed line) or 5% less DIC (short-dashed grey line). Note that alkalinity (ALK) of 5000 meq/L equals 250 mg-CaCO3/L and that for ALK>0, [HCO3-] approximately equals ALK.

Carbon dioxide buffering occurs because CO_2 reacts with water and protons to yield a group of inter-related ions (species) called the carbonate system:

$$CO_2(aq) + H_2O \longleftrightarrow H_2CO_3^0$$
$$H_2CO_3^0 \longleftrightarrow H^+ + HCO_3^-$$
$$HCO_3^- \longleftrightarrow H^+ + CO_3^{2-}$$

The aggregate concentration of carbonate species defines the water quality parameter known as dissolved inorganic carbon (DIC):

$$DIC = [CO_2(aq)] + [H_2CO_3] + [HCO_3^-] + [CO_3^{2-}]$$

The influence of inorganic carbon on *p*H can be inferred in streams from the diurnal fluctuations in *p*H and dissolved oxygen (Figure 60). These fluctuations are driven by photosynthesis, which consumes CO_2 during daylight hours, and respiration, which produces it at night. These biogeochemical processes incorporate inorganic carbon into biomass and subsequently release it back into the water, as represented by this approximate equation:

$$HCO_3^- + H^+ + H_2O \underset{\text{Respiration}}{\overset{\text{Photosynthesis}}{\rightleftarrows}} CH_2O_{BIOMASS} + O_2$$

The magnitude of these fluctuations depends on the alkalinity of the water and the productivity, which is controlled by the supply of nutrients.

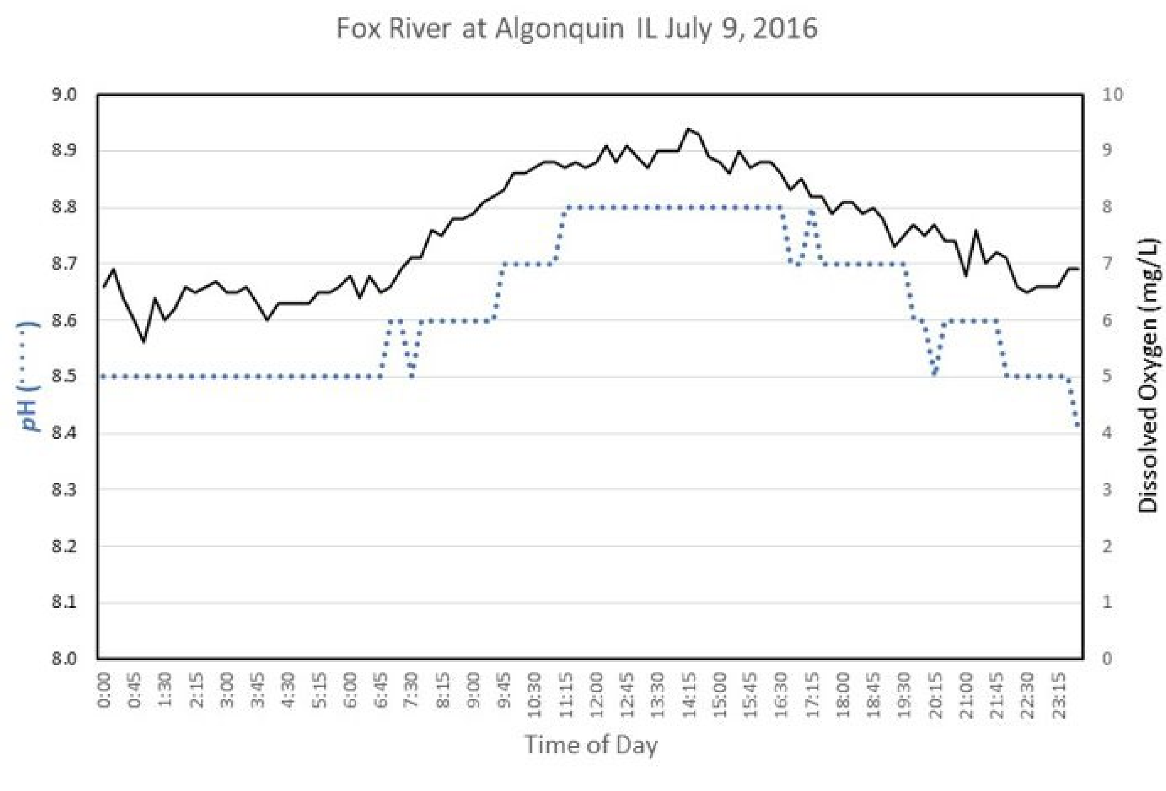


Figure 60. Continuous monitoring of pH and dissolved oxygen (DO) in the Fox River at Algonquin, IL. Note that the water temperature rose about 2 ℃ from 7 am to 3 pm, but this would make oxygen less soluble so photosynthesis must have caused the increase in DO, and the commensurate reduction in DIC raised the pH. River discharge was roughly constant at 430 ± 20 cfs. An alkalinity of 226 mg-CaCO3/L was measured on 7/5/2016.

In most cases, the alkalinity (ALK) and DIC together determine the *p*H of a natural water as depicted in Figure 59 and Figure 60, but there are exceptions. For example, in wetlands and surface soils where the decomposition products of plant litter include significant concentrations of organic acids, the *p*H may drop well below that expected for air-equilibrated water.

4.4.3 Spatial and temporal trends in river *p*H: Urbanization

4.4.3.1 Urbanization effects on pH

Over the past decade or two, studies of riverine water quality data have identified urbanization as a contributor to positive temporal trends in *p*H, alkalinity, and related parameters (Kaushal et al., 2014, 2018; Stets et al., 2014, 2020). Although the decline in the acidity of atmospheric deposition is an important contributor to the trend, increasing urbanization has contributed as well. "Urbanization" in this context refers to a combination of factors that do not all affect *p*H in the same way. For example, the reduced infiltration into soils in urban areas leads to a lower alkalinity production by mineral weathering in soils, but the high weatherability of concrete can more than offset that loss.

Comparative studies across watersheds have shown that urbanized watersheds export more alkalinity and their streams have higher *p*Hs (Barnes and Raymond, 2009). A strong ($p < 0.01$) positive correlation between *p*H and both percent urban land cover and impervious surface cover was observed in coastal New Jersey watersheds (Conway, 2007). The rural New Jersey systems exported poorly-buffered, organic-rich water and so may have been more sensitive to urbanization.

Precipitation naturally acquires ions, mostly base cations, as it flows over and through the soils of undisturbed watersheds (Driscoll et al., 2007). The same is true of urban watersheds, with the difference that these ions are acquired upon contact with impervious surfaces, e.g., buildings and roads. In fact, the contributions of some cations apparently are greater than in natural soils due to the high weatherability of concrete, as shown by 1) the positive correlation of *p*H, alkalinity, and specific conductance with ISC and urban land use (Conway, 2007) and 2) the negative correlation with forest land cover consistent with rapid weathering of concrete and other materials in the urban environment. This phenomenon is increasingly being referred to as "salinization syndrome" (Kaushal et al., 2018).

Examining *p*H data obtained from monitoring of precipitation and stream water allows us to understand the effects of constructed surfaces on the *p*H of stormwater in the region.

4.4.3.2 Precipitation

During the mid- to late 20th century, precipitation was a significant source of acids to urban and rural watersheds. However, environmental regulations limiting atmospheric sulfur emissions led to a striking decline in the sulfate concentrations in rainfall and in the acidity of rainfall (Figure 61). Thus, in recent years the average acid-base balance of precipitation has tended towards near neutral, yielding *p*H values of 6.0 ± 0.5.

The decline in acidity of precipitation ultimately makes streams and rivers less acidic as well. However, in some cases, the decline in acidity also appears to have caused a decline in base production via mineral weathering, so the net impact on river alkalinity is not as great as the decline in acid anion inputs.

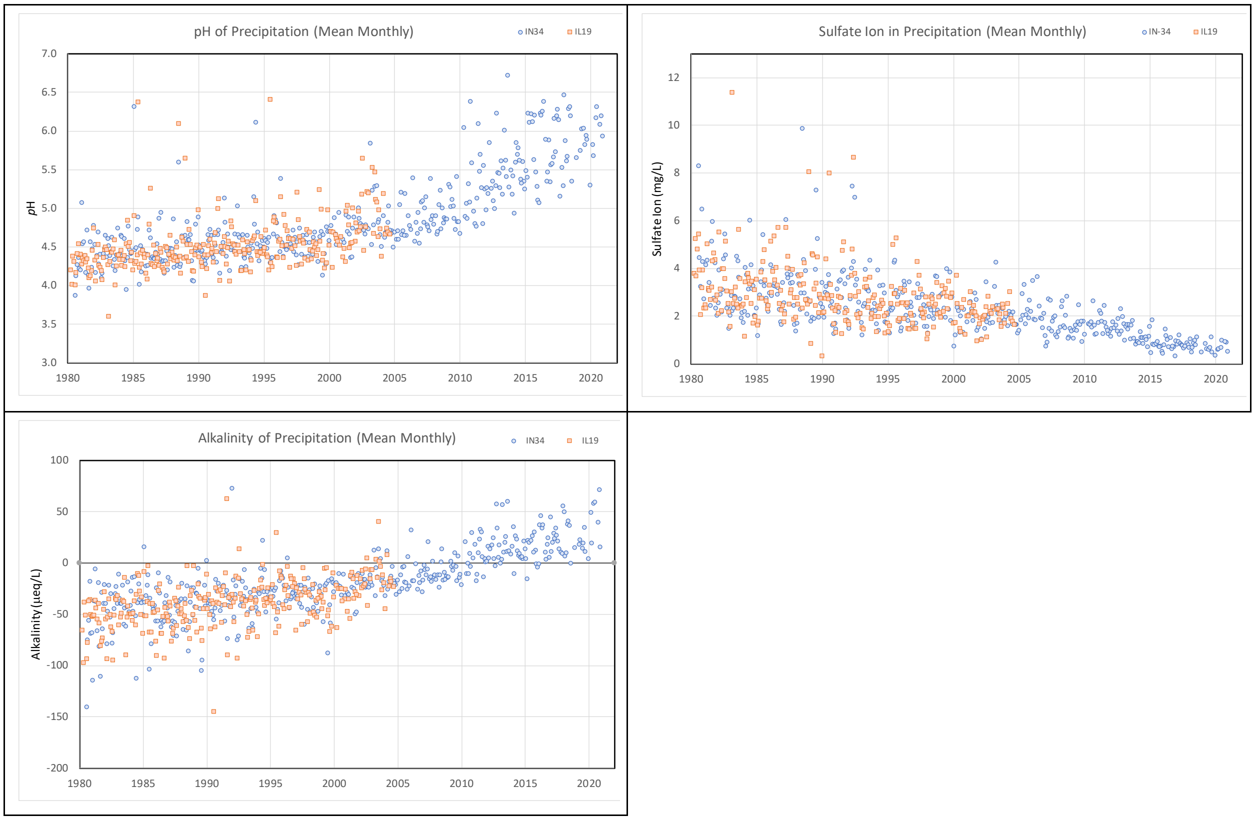


Figure 61. pH and levels of sulfate ion and charge balance alkalinity of wet deposition (precipitation) at National Atmospheric Deposition Program sites in the study area. The increasing trend in alkalinity was mainly caused by the decrease in sulfate ion content of the precipitation, which was the result of national policies requiring that atmospheric sulfur emissions be reduced beginning in the 1990s, and by a lesser extent to a decline in nitrate.

4.4.3.3 Stormwater

Little data on the *p*H of stormwater in the greater Chicago area exist. Alkalinity and *p*H event mean concentration (EMC) values reported in the literature (Table 22) are greater than that of precipitation, but certainly not as well buffered against *p*H drops as groundwater. This conclusion is consistent with *p*H observations in stream water in the region (Figure 62). Currently, in Poplar Creek, *p*H exhibits a decreasing trend at flows above 3 cubic meters per second ($m^3\ s^{-1}$), which is when the river contains the highest proportion of stormwater. At flows above 10 $m^3\ s^{-1}$, the *p*H drops to roughly 7.1 to 7.8. During the 1980s, a similar decreasing trend is observed, but the *p*H dropped to 6.9 to 7.4 in the high flow range. This observation is more consistent with the lower alkalinity and *p*H of precipitation during the 1980s than at the present. Thus, we can infer that the stormwater alkalinity reflects precipitation chemistry, though most likely it gains enough alkalinity upon contacting urban watershed surfaces to raise the alkalinity from negative to slightly positive values.

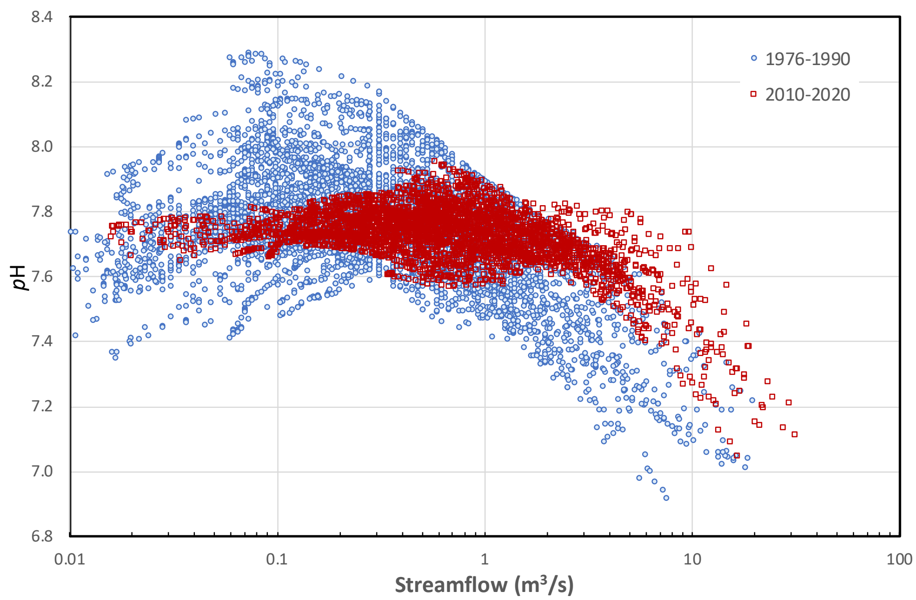


Figure 62. Dependence of stream water pH on stream discharge in Poplar Creek at Elgin, IL. Points are calculated daily values for the periods 1976–1990 (circles) and 2010–2020 (red squares) estimated using USGS EGRET Weighted-Regression in Time Discharge and Season (WRTDS) analysis of monitoring data. Note that the lower pH values predicted at high flows are consistent with stormwater with intermediate alkalinity values between that of rainwater and groundwater.

4.4.3.4 Stream water

The two stream monitoring sites that we have examined so far show strikingly different trends with respect to flow-normalized *p*H (Figure 63). Recall that flow-normalization is intended to remove the effect of fluctuations, both daily to interannual, in streamflow on the variable being analyzed. The analysis suggests that Poplar Creek exhibits little to no trend over time, while a remarkable increase in *p*H has occurred in Addison Creek. At this preliminary stage, we suggest that alkalization of Addison Creek likely reflects the change in atmospheric deposition chemistry, but increased weathering of concrete in this highly developed watershed may have contributed significantly as well. Careful consideration of other factors such as changes in flow and chemistry of wastewater discharges must be made before a firm conclusion can be reached.

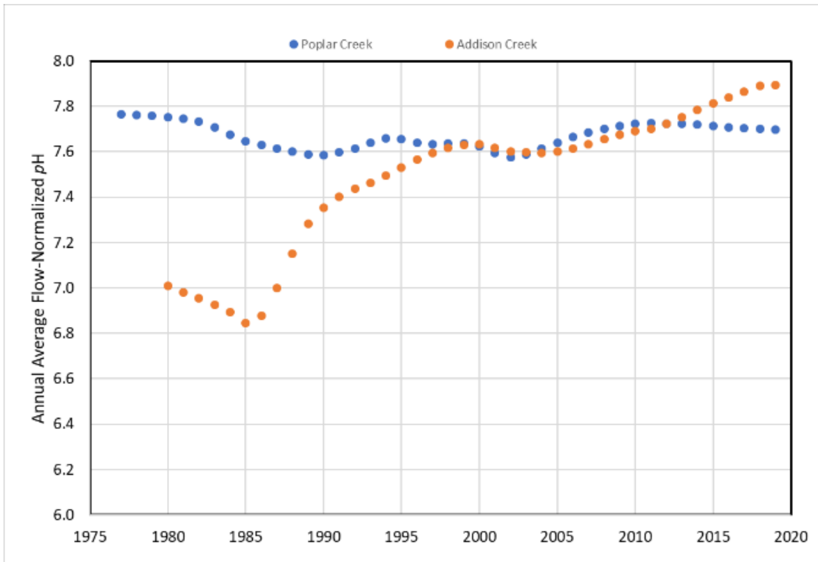


Figure 63. Annual average flow-normalized pH in area streams. Although the flow-normalized pH in Poplar Creek varied over the period of record, there was no significant temporal trend. In Addison Creek, however, the flow-normalized pH increased nearly a full pH unit from 1985 to the present. We are currently exploring the reasons for this trend.

4.4.4 Water quality standards and impairments

In surface waters, the U.S. Environmental Protection Agency (USEPA) defines *p*H values below 6 to 6.5 to be "low" while values over 9 are considered "high." This *p*H standard was set to protect aquatic life, as many aquatic organisms are *p*H-sensitive. *p*H values in this range don't affect the suitability of water for contact recreation. If such high or low values occur frequently or for long enough in a water, *p*H could be considered a "candidate cause" for a water quality impairment (USEPA, 2017). Illinois' standards for "general use" waters follow USEPA guidelines and require that "*p*H shall be within the range of 6.5 to 9.0 except for natural causes."

The Illinois EPA previously listed *p*H as a cause of impairment in portions of Upper Salt Creek and both East and West Branches of the DuPage River. However, according to the 2019 TMDL for these watersheds, no violations were observed at least after 2001.

4.4.5 Summary

*p*H is influenced by stormwater contacting natural and constructed materials capable of contributing ionizable substances. Both acidification (decreases in mean *p*H) and alkalization (increases in mean *p*H) can contribute to degradation of aquatic ecosystems. In this region, there is evidence for stream water alkalization in Addison Creek. This rise does not yet put the *p*H into the impaired range, but the causes of this trend at this site and any other where it may be occurring should be explored further. It isn't clear how stormwater control measures (BMPs) designed to detain and infiltrate stormwater will affect long-term trend in some streams towards higher *p*H.

In addition, stream water *p*H dips up to about 0.5 units under high flow conditions, suggesting that stormwater has lower alkalinity than the average river water or base flow. However, the *p*H remains well within the 6.5–9.0 range set forth in Illinois' water quality criteria. BMPs that enhance stormwater infiltration should reduce the magnitude of *p*H dips in the streams.

4.5 Key Processes in Best Management Practices for Stormwater

Once stormwater enters a BMP, the constituents/pollutants it transports can be 1) transmitted to streams, 2) diverted to groundwater, 3) retained via the trapping of the particles they are associated with or via reversibly sorbing to media, 4) volatilized, or 5) transformed into a different chemical constituent. Again, the extent to which each process affects a constituent depends on the type of BMP and of course the chemistry of the constituent (See Chapter 7). Here, a general description of how intra-BMP processes affect the constituents of interest is presented.

4.5.1 Sedimentation

Since particles tend to become suspended or remain in suspension in flowing water, stormwater can acquire considerable particulate loads. Conversely, by slowing or stopping the flow of stormwater entering them, BMPs facilitate the sedimentation of suspended particles. Particles and any associated constituents that settle out of pooled water collected in a BMP are retained at least temporarily within the structure.

The rate of sedimentation in a BMP depends non-linearly on particle density and size as well as water temperature and the speed with which entering stormwater mixes with water already collected in the BMP (Erickson et al., 2013). Particles of a larger size and greater density settle more rapidly and are retained more efficiently. Mineral particles of a size greater than 1-mm settle out essentially completely.

For constituents that are either insoluble (minerals) or adsorbed onto mineral particles and dense aggregates, sedimentation efficiently reduces the loads transmitted to surface waters during storm events. Particle-associated constituents known to be retained in BMPs via sedimentation include phosphorus, heavy metals, and hydrophobic organic compounds. In theory, a BMP's efficacy in reducing the transmission of a constituent should be roughly proportional to its fraction in the particulate matter sedimented within the BMP.

Constituent retention via sedimentation is not necessarily permanent, however. Particles can disaggregate into more easily resuspended sizes and be transported in subsequent storms (McFarland et al., 2019). Furthermore, persistent organic compounds and elements are not degraded over time, so they may redissolve and become subject to transport via subsequent storm events. Many organic compounds, however, can be degraded or volatilized over time. Similarly, nitrogen cycling can transform particle-associated forms of nitrogen into gases that are released to the atmosphere.

Some of the largest particles are plastic litter or plant debris, which have densities low enough to float. Although such gross particles may float rather than sediment out, they can nevertheless be retained if water exits the BMP via a filtration process.

4.5.2 Filtration

Filtration is the straining of suspended particles from stormwater as it flows through granular media and results in their becoming trapped within BMPs. Relevant granular media used in BMPs include gravel, sand, or even organic materials in underdrains or filtration practices. The effectiveness of filtration depends largely on the size of the suspended particles relative to the size of the pore spaces within a BMP's granular media. In addition to filtration by media, vegetation can reduce stormwater transport of coarse particulate matter as water flows through dense vegetation such as grass (Stagge et al., 2012)

Constituents associated with particles trapped by filtration are at least temporarily retained within the media rather than transported with the water. Depending on how strongly they are bound to the particles, constituents will be retained more or less effectively. Reversibly-sorbed substances may be released back into water flowing through the media if the concentration in the water drops sufficiently. Constituents can also be released when particles dissolve, such as when a mineral becomes undersaturated or when organic particles decompose. Persistent constituents and inorganic species are thus prone to being released over time, while some organic pollutants may degrade or even be mineralized into inorganic constituents comprised of the elements from which they are made.

4.5.3 Infiltration

Enhanced infiltration is the principal means by which BMPs achieve volume control. Infiltrated stormwater is absorbed by the ground, where it percolates downward toward the water table or flows laterally toward streams. This results in the diversion of a portion of stormwater from an event away from direct discharges to surface waters. Shallow groundwater can flow into streams but the combination of infiltration and subsurface flow lengthens the time needed to reach the streams considerably and pollutant transport can be retarded by sorption to soils.

Infiltration causes stormwater to be filtered via movement through the medium of the soil, which typically comprises relatively fine particles with small pore spaces. Thus, it effectively removes suspended particles as well. However, colloids have been detected in groundwater and so presumably colloidally bound constituents are able to move through soils as well.

Since the suspended particles in stormwater typically comprise different materials than soils in infiltration basins, dissolved constituents at equilibrium with particles in the water may still be retained by adsorption to the soils. The mass of suspended particles relative to water in flowing stormwater is much lower than the ratio of soil solids to water during infiltration, which also encourages sorption. However, constituents retained during infiltration are prone to being released back into solution just as with filtration.

4.5.4 Volatilization

Volatilization can result in the transfer of substances that have truly dissolved and are electroneutral from the water to the atmosphere. Unsurprisingly, volatile organic compounds (VOCs) are especially subject to volatilization. Dissolved gases, such as carbon dioxide, oxygen, dinitrogen, and elemental mercury are as well. Rates of volatilization are fastest from flowing water because the turbulence in the water speeds transport to the water surface and subsequent transfer to the gas phase, but it can occur in ponded water as well. Volatilization is not likely to significantly mitigate most stormwater constituents when ponded only for a few days but may well have effects in wet-bottom retention systems.

The rate of volatilization also depends on molecular properties of the pollutant, including its equilibrium air-water partition coefficient (Henry's Law constant) and its rate of diffusion within water and air. These factors divide organic compounds into those with volatilization rates that are limited by diffusion in the liquid and gaseous phases. The former behaves like oxygen, but the latter are more akin to the evaporation of water itself.

4.5.5 Sorption

The reactions involved in constituent sorption have been described in Section 4.3.4. Within BMPs, sorption is relevant in two different contexts. One context is within the water column, as a constituent's species re-equilibrate between suspended particle and dissolved fractions. The second context occurs as the stormwater exits a BMP via infiltration and potentially via filtration as well. This second type of sorption is also called immobilization or fixation (Laurenson et al., 2013). Like adsorption on suspended particles, sorption on particulate media can cause pollutants to accumulate on reactive surfaces (adsorption) or within organic matter (absorption).

During infiltration, soil serves as the sorbent medium and retards constituent transport relative to the water. Filtration media within or underlying BMPs can also be selected for their ability to sorb constituents/pollutants and retard transport. As described above, the extent of sorption and type of pollutant adsorbed depends on the composition of the soils or other media. For constituents that are strongly sorbed, this process efficiently reduces the loads transmitted to surface waters during storm events.

In filtration practices, the type of media exerts a significant influence on which constituents are retained. BMPs that drain stormwater through organic-rich media tend to retard the transport of hydrophobic organic compounds efficiently (Laurenson et al., 2013). Note also that the retention of organic particles adds to a BMPs capacity for absorption of hydrophobic organic compounds. Metals can bind to both the organic and mineral sorbents of soils and pond sediments, filtration media, and soils underlying BMPs. The chemistry of the water within the BMP also affects the efficiency of sorption. For example, most organic compounds will sorb most strongly in the *p*H range where they are uncharged and most hydrophobic, while metal ions tend to adsorb more strongly at higher *p*H and are often released into solution at low *p*H. Of course, metals adsorbed onto iron oxides can be released when iron minerals dissolve under anaerobic conditions.

Pollutant/constituent retention by adsorption does not necessarily act as a permanent sink, however. Persistent compounds and elements are not degraded and could redissolve. However, even temporary sorption gives other processes time to act on some significant constituents. Organic compounds can be degraded or volatilized over time, and nitrogen cycling can lead to the release of this element from BMPs.

4.5.6 Precipitation and Dissolution

Constituents that can form a pure solid phase have maximum solubility limits that can exert an important influence on water quality. When this limit is exceeded, the substance is said to be "supersaturated," and precipitation causes the dissolved concentration to decrease over time. When the dissolved concentration is below the limit, the substance is said to be "undersaturated" and dissolution causes the dissolved concentration to increase over time in water that is in contact with the pure solid phase.

The equilibrium reactions and associated mathematical expressions for solubility limits differ for solids that exist mainly as ions in solution and those that exist mainly as neutral compounds. Ionic substances have a solubility limit defined by their solubility product (K_{sp}):

$$AB_n(s) \longleftrightarrow A^{z+} + nB^{z/n-}$$
$$K_{sp} \equiv [A^{z+}][B^{z/n-}]^n$$

where A is a generic cation of positive charge z and B is a generic anion with charge $-z/n$. As noted above, the precipitation of ferric hydroxide limits the concentration of dissolved iron in both surface and groundwaters where oxygen is present.

Neutral organic compounds simply have a defined saturated concentration (K_{sat}):

$$N(s) \xleftrightarrow{K_{sat}} N(aq)$$
$$K_{sat} \equiv [N(aq)]_{sat}$$

Although the solubility limit of a substance can be defined with mathematical precision, in reality few saturable solids exist exactly at equilibrium with the solutions they are in contact with. In the context of stormwater, the difference in the flow of stormwater relative to the solid phase, which may be fixed in place, is a common reason that equilibrium is not attained.

Finally, we note infiltration of stormwater may cause some soil minerals to dissolve, while others may precipitate from changes in solution composition within the media, e.g., *p*H within soils.

4.5.7 Abiotic transformations

4.5.7.1 Reduction/Oxidation

For elements that can exist in multiple oxidation states (Table 23), their fate in BMPs may vary greatly depending on redox reactions within the practice. The single most important determinant of elemental oxidation states in any compartment of a BMP is the supply of oxygen relative to the microbial demand for oxygen in aerobic metabolism. Since ponded water is in contact with the atmosphere, generally abiotic oxidation reactions predominate in BMPs. Abiotic oxidation of Fe(II) is a well-studied *p*H-dependent reaction in surface waters. Since Fe(II) is much more soluble than Fe(III), the net effect of its oxidation is to increase Fe retention by BMPs. Reductive processes tend to be microbially driven, as described below.

4.5.7.2 Hydrolysis

Hydrolysis occurs when a water or hydroxide molecule forms a bond with an organic compound (Brezonik and Arnold, 2022). Hydrolysis can lead to the transformation of organic halides by replacing the halide with a hydroxyl group:

$$R-Cl+OH^- \longrightarrow R-OH+Cl^-$$

Another example of a hydrolysis reaction is the addition of water to an ester, thus forming a hydroxylic acid:

$$R_1-COO-R_2 \longrightarrow R_1-COOH+R_2-OH$$

The rates of different hydrolysis reactions may be enhanced by the presence of H^+ (acid-catalyzed) or OH^- (base-catalyzed).

4.5.7.3 Photolysis

Sunlight absorbed by various substances in water can cause chemical reactions that degrade organic compounds or change the oxidation state of inorganic compounds. Direct photolysis of a compound of interest occurs when it absorbs a photon that is energetic enough to alter its electronic state and ultimately alter its chemical structure. Indirect photolysis occurs when one molecule, such as natural organic matter, absorbs a photon and then transfers the excess energy to another. Some transfers form oxygen and hydroxyl radicals, which can enter into a wide variety of reactions with organic and inorganic compounds. Other transfers cause the secondary molecules to alter its structure. Atrazine is an example of a compound that enters into both types of photochemical reactions (Brezonik and Arnold, 2022). Iron, especially when bound to certain organic ligands, undergoes photochemical reduction reactions that make it more soluble (Nagai et al., 2007).

For photolysis to have an impact on water quality, the water must be exposed to sunlight for extended periods. Thus, it tends to be most significant in constructed wetlands and to a lesser degree in wet-bottom detention basins.

4.5.8 Vegetation processes

BMPs can be designed to take advantage of plants' effects on constituent fate and transport. Nutrients, organic compounds, and even metals can be affected. Vegetation can divert flows in ways that enhance particle filtration (Stagge et al., 2012). Plants can directly take up many constituents and facilitate the degradation of others by fostering the development of soils with active microbial populations (Laurenson et al., 2013). Certainly, they are directly responsible for transpiration and their roots increase soil porosity, which enhances infiltration. Both processes reduce waterborne pollutant transport out of the BMP. The net effect is to turn the BMP into a small ecosystem that provides water quality benefits.

Naturally, the growth of vegetation directly removes constituents that contain the major nutrients N and P from stormwater. Small amounts of some metals such as iron, manganese, copper, and zinc are also essential for plant growth, but plants can absorb non-essential metals as well. Constituents absorbed by vegetation are retained within the BMP, but their longer-term impact depends on the fate of the biomass produced. Those in plant litter that is removed from the BMPs will not be released at the site of uptake, but those in plant litter remineralized within the BMP soils or sediments will be re-released. Nevertheless, the long-term re-release of metals is likely preferable to pulses of metals in stormwater entering streams directly as the metals will be associated with organic matter, thus mitigating their toxicity (Laurenson et al., 2013).

Plants can directly degrade organic compounds that they take up. Organic compounds may be passively absorbed via roots and then be subject to degradation by enzymes within the plant or simply become tied up in biomass. Plant roots also exude a variety of compounds that influence metal speciation and transport. Plants also take up volatile substances, even Hg, and emit them to the atmosphere.

The organic residues of vegetation are the main source of energy for the microbial communities in BMP soils. As more biologically active soil populations generally are more likely to have active metabolic and cometabolic processes, they typically have water quality benefits. The microbial degradation of plant litter usually requires immobilization of some additional inorganic nitrogen (Schlesinger and Bernhardt, 2020). The higher organic loadings in vegetated BMPs increase the chances of anaerobic microsites or larger oxygen-depleted zones forming, allowing anerobic bacteria to grow (See next section).

4.5.9 Microbial transformations

Microbes carry out many of the most important transformation processes of both organic and inorganic water quality constituents in surface waters and sediments. Some of these reactions supply microbes with energy and are thus involved in microbial metabolism. Other reactions take place without microbes gaining energy from the process. Such "cometabolism" occurs when compounds that are somewhat similar to others involved in metabolism or assimilation are inadvertently acted upon by microbial enzymes. The breakdown of organic compounds via either pathway is termed "biodegradation," while inorganic microbial transformations tend to have process-specific names.

Readers are most likely to be familiar with microbial degradation as expressed in the water quality parameter "biochemical oxygen demand." Oxygen demand is exerted by microbial communities as they metabolize degradable organic matter and reduced nitrogen species, especially ammonium. Similar reactions occur in water, sediments, and soils. Their dissimilatory metabolic pathways ultimately break down the organic matter to the inorganic constituents of which it is comprised, i.e., mainly oxidized forms of C, N, and P. The dissimilatory degradation of organic pollutants, also called biomineralization, can take place via enzymatic reactions occurring both intra- and extra-cellularly (Laurenson et al., 2013). Assimilatory metabolism, in contrast, incorporates fragments of organic and inorganic molecules into biomass.

In dissimilatory metabolism, microbes catalyze the transfer of electrons from the reduced carbon atoms in organic compounds to available oxidants or "terminal electron acceptors," in a manner that allows them to obtain energy for maintenance and growth. The oxidative mineralization reaction for generic organic matter (for the purposes of describing microbial metabolism, natural organic matter is often written as "CH_2O") can be written as:

$$"CH_2O" \xrightarrow{Microbes} CO_2 + 4e^- + H_2O$$

Note that in this reaction, the oxidation state of the carbon atom increases from near zero, the mean oxidation state in natural organic matter derived from plant litter, to a value of +IV in carbon dioxide. Microbes must transfer the electrons derived from the mineralization of the reduced organic compound to an electron acceptor or oxidant, typically via a series of reactions involving multiple enzymes and molecules that serve as electron carriers. The final oxidant that accepts electrons from the respiratory processes of a microbial population or consortium (TEA_{ox}) is the "terminal electron acceptor" for that process (Brezonik and Arnold, 2022).

Within streams, water columns of ponds and lakes, and surficial sediments, the dioxygen molecule (O_2) is far and away the predominant terminal electron acceptor. The microbially mediated reduction of dioxygen also consumes four protons and leads to the formation of water:

$$O_2(aq) + 4H^+ + 4e^- \longrightarrow 2H_2O$$

Together, these two reactions make up aerobic respiration:

$$"CH_2O" + O_2(aq) \xrightarrow{Microbes} CO_2(aq) + H_2O + Energy$$

The concentration of degradable organic matter that can be broken down aerobically in a finite period of time, typically five days, is reported as "biochemical oxygen demand" for water samples and sediment oxygen demand for sediments. Biochemical oxygen demand depends on the concentration and type of organic matter in a water.

In some environmental compartments, such as sediments and stratified water columns, microbes can deplete oxygen to very low concentrations, even effectively reaching zero. In such locales, microbes must employ other terminal electron acceptors. The most quantitatively significant alternative terminal electron acceptors are the oxidized forms of the major elements, such as NO_3^-, $Fe(OH)_3(s)$, $MnO_2(s)$, SO_4^{2-}, and CO_2. The redox cycling of the oxidized and reduced forms of these elements creates the prominent zonation of redox processes at oxic/anoxic interfaces in sediments and density-stratified waters (Brezonik and Arnold, 2022).

This redox processing has important consequences for the sedimentary fate of metals and nutrients as such conditions can occur in sediments and stratified water columns of wet retention systems (Taguchi et al., 2020). Nitrate can be consumed, leading to a true loss from the system via denitrification. Orthophosphate, on the other hand, can be remobilized to the water column if the iron oxyhydroxide minerals it is adsorbed to in pond sediments becomes reduced.

The conceptual basis for understanding these biogeochemical cycles and the resulting sedimentary zonation lies in the differences in free energy available for microbial metabolism with different terminal electron acceptors. The terminal electron accepting processes (TEAPs) that are quantitatively important for the oxidation of organic matter in sediments are, in order of decreasing energy yield:

Denitrification. $$NO_3^- + 6H^+ \xrightarrow{5e^-} \tfrac{1}{2}N_2 + 3H_2O$$

Mn reduction. $$MnO_2(s) + 4H^+ \xrightarrow{2e^-} Mn^{2+} + 2H_2O$$

Fe reduction. $$Fe(OH)_3(s) + 3H^+ \xrightarrow{e^-} Fe^{2+} + 3H_2O$$

Sulfate reduction. $$SO_4^{2-} + 10H^+ \xrightarrow{8e^-} H_2S + 4H_2O$$

Methanogenesis. $$CO_2 + 4H^+ \xrightarrow{4e^-} CH_4 + H_2O$$

Homoacetogenesis. $$2CO_2 + 7H^+ \xrightarrow{8e^-} CH_3COO^- + 2H_2O$$

The net reaction for organic matter decomposition occurring within a particular zone of the sediments can be written by combining the oxidation of "CH_2O" with one of the above terminal electron-accepting (reduction) processes.

The subsequent reactions of the reduced inorganic compounds formed via reaction are also important in sedimentary environments. Some (Fe^{2+} and S^{2-}) may be sequestered as mineral phases (FeS or FeS_2). Others diffuse into overlying waters (acetate, NH_4^+, Mn^{2+}, H_2S, H_2, CH_4) and/or are transported in bubbles (CH_4 and H_2) upward where they come into contact with oxidized species, most notably oxygen. The resulting oxidation reactions can be abiotic or microbially mediated.

Other terminal electron acceptors that can be environmentally significant, if not typically dominant, in sediments are humic acids and organochlorine compounds (RCl_n), which are transformed via reductive dechlorination:

$$RHCl + 2e^- + H^+ \longrightarrow RH_2 + \tfrac{1}{2}Cl^-$$

Reductive dechlorination is an important pathway in the breakdown of otherwise persistent, highly chlorinated compounds such as PCBs.

Finally, there are multiple trace elements of interest from a water quality perspective that have biogeochemical cycles that are coupled to the major element redox cycles, including As, Cr, Hg, and Se. Not only are these processes potentially significant in wet pond sediments, but also wetting and drying cycles within soils and BMP sediments facilitate such reactions (Laurenson et al., 2013).

4.6 Summary: Biogeochemical Processes in BMPs

4.6.1 Overview

The chemical constituents borne by urban stormwater are transported into best management practices (BMPs) in both dissolved and suspended particle forms. Naturally, the distribution between these two fractions varies greatly with the source of the constituent as well as its chemistry. Since stormwater BMPs are highly effective at retaining suspended particles, constituents that occur primarily in the particulate fraction usually are also removed quite efficiently. On the other hand, primarily dissolved constituents may or may not be retained. For a constituent dissolved in stormwater to be removed before discharge to waterways or groundwater, that constituent must either 1) sorb onto the filter media or underlying soils of BMPs, 2) be incorporated into sediments via sorption, precipitation reactions, or uptake by vegetation, or 3) be transformed into a form that escapes to the atmosphere or is otherwise immobilized. Otherwise, the dissolved constituent will be exported from the BMP, just as chloride is.

Infiltration practices are designed to divert stormwater and any exported constituents it bears from waterways to shallow groundwater. Since the groundwater may eventually flow back into a stream, the net water quality impact of infiltration depends broadly on what the fate of the constituent is in the subsurface environment. Non-sorbing, conserved constituents such as chloride should reach streams as the infiltrated stormwater does, though they may be diluted somewhat and spread out over time. Conserved constituents that sorb will experience slower transport (retardation) but eventually should reach streams as well. If the subsurface transport pathway is long enough that a constituent can be transformed or immobilized *en route*, infiltration will provide excellent protection to surface waterways.

Any constituent removed from stormwater via one of the non-transformative processes described above must accumulate within the BMP over time. Such accumulations may eventually find their way back into stormwater via resuspension of sediments or remobilization, as observed for internal loading of pond sediments.

Finally, note that since processes cause changes in the state of a system and its constituents over time, the length of time stormwater remains within a practice can affect the extent to which a process can remove constituents in stormwater.

4.6.2 Summary by BMP Type

The key processes occurring in several types of BMPs are summarized in Table 25. Note that it may be helpful to also consult the summary of BMP hydrologic fluxes in Table 53 of Chapter 7.

Table 25. Key Processes in BMPs that affect constituent fate. "w/UD" means BMP type from the immediately preceding row with underdrain. Adapted from McFarland et al. (2019). The number of "+" symbols reflects the relative magnitude of the process.

BMP	*Sedimentation*	*Filtration*	*Infiltration*	*Sorption*	*Trans-formations*	*Biouptake*
Release Rate						
Dry detention basin	+++		++	+		
Constructed wetland	+++				+++	+++
Wet retention pond	+++		+	+	++	++
Cistern	+++		++			
Volume Control						
Pervious pavement			+++	++		
w/UD		++	+	++		
Bioretention			+	++	++	++
w/UD	++	++	++	++	+	+
Bioswale	++		+++	++	++	++
w/UD	++	++	+	++	++	++
Rain garden			+++	++	++	++
w/UD	++	++	+	++	+	+
Infiltration trench			+++	++	+	++

Process Definitions

Sedimentation	Settling of particles in stagnant or slow-moving water
Filtration	Straining of particles from water flowing through media
Infiltration	Transport of dissolved constituents into soils
Sorption	Retention of dissolved constituents on media during filtration or infiltration
Transformations	Degradation of organics or alteration of inorganics (redox); mostly microbial metabolism or cometabolism.
Biouptake	Uptake by higher plants and incorporation into biomass

4.6.3 Summary by Constituents

Table 26 to Table 34 summarize several aspects of the chemistry of key water quality constituents, their interactions, and their fate in BMPs.

Please note that a separate table for Cyanide (CN^-) is not included. The constituent $Fe(CN)^{3-}$ is covered under iron. This compound is a component of road salt and its dissociation would be a source of CN^- in the environment (Exall et al., 2011). However, it is not commonly detected in Chicago region waterways.

Table 26. Chemistry of Suspended Solids

	Silt and Clay	*Fine Sand*	*Particulate Org. Matter*
Pollutant Description			
Fraction	Particulate	Particulate	Particulate
General description	Particles up to 63 μm diameter	Particles 0.063-0.25 μm diameter	Detrital material from plants and algae
Constituent or Analyte	TSS		POC
Interactions	Sorbs metals and TrOCs		Source of BOD and TrOCs
Overall effect of SCMs	Strongly retained	Strongly retained	Strongly retained
Sources			
	Erosion & scouring	Erosion & scouring	Plant litter
Phase behavior			
Sorption	Sorbent for Me & P	Sorbent for Me & P	Sorbent for Me & TrOC
Solubility	Clay precip	SiO2 precip	
Volatilization			
Physical Processes			
Sedimentation	Retained	Retained	Retained
Filtration	Retained	Retained	Retained
Infiltration (soils)			
Resuspension	Resuspendable		Resuspendable
Biological Processes			
Microbes			Decomposed to DOM
Higher plants	Enhanced sedimentation		Product of plant death
Algae			Product of algal death
Chemical Processes			
Aqueous equilibria	*p*H-dependent sorbent	*p*H-dependent sorbent	*p*H-dependent sorbent
Redox			
Photochemistry			
Abiotic degradation			
Overall effect of SCMs	Retained/resuspended	Retained	Retained/resuspended

Table 27. Chemistry of Biochemical Oxygen Demand

	Carbonaceous BOD		*Nitrogenous BOD*		*Other*	
	Urban runoff OM	*Natural OM*	*Ammonia*	*Organic N*	*Sulfides*	*Iron(II)*
Pollutant Description						
Fraction	Dissolved & Particulate	Dissolved & Particulate	Dissolved	Dissolved & Particulate	Dissolved & Particulate	Dissolved
General description	High BOD/ TOC ratio	Lower BOD/ TOC ratio				
Constituent or Analyte			NH3-N	DON	Sulfide	Ferrous Iron
Interactions	Dissolved Oxygen	Dissolved Oxygen				
Overall effect of SCMs			Oxidation	Oxidation		
Phase behavior						
Sorption			Sorbed to clays			
Precipitation					Metal-Sulfides	FeS(s)
Volatilization			At *p*H > 9		At *p*H < 7	
Physical Processes						
Sedimentation	Retention	Retention			Retained	
Filtration	Retention	Retention				
Infiltration	Retardation	Retardation	Retardation	Retardation	Retardation	Retardation
Resuspension	Resuspendable	Resuspendable				
Biological Processes						
Microbes	Mineralization; Dissolution	Mineralization; Dissolution	Ammonia oxidation	Mineralizatio n	Sulfide oxidation	Dissimilatory Fe reduction
Higher plants		Plant litter	Uptake	Uptake		Uptake
Algae		Algal growth	Uptake	Uptake		Uptake
Chemical Processes						
Aqueous equilibria			Acid-base		Acid-base	
Redox					Oxidation	Oxidation & reduction
Photochemistry						Photoreduction of Fe(III)
Abiotic degradation						

Table 28. Chemistry of Fats, Oil, and Grease

	Fats	*Oils*	*Grease*
Pollutant Description			
Fraction	Dissolved, NAP	Dissolved, NAP	Dissolved, NAP
General description	Slightly soluble; Distinct phase floats on water; sticks to surfaces	Slightly soluble; Distinct phase floats on water; sticks to surfaces	Slightly soluble; Distinct phase floats on water; sticks to surfaces
Constituent or Analyte	Various	Oils	
Interactions	Contributes to BOD; Solubilizes hydrophobic organics	Contributes to BOD; Solubilizes hydrophobic organics	Contributes to BOD; Solubilizes hydrophobic organics
Overall effect of SCMs	Retained	Retained	Retained
Phase behavior			
Adsorption	Mulch helps remove	Mulch helps remove	Mulch helps remove
Precipitation	Solubility limit can be reached	Solubility limit can be reached	Solubility limit can be reached
Volatilization		Some oils are somewhat volatile	
Physical Processes			
Sedimentation	High retention	High retention	High retention
Filtration (retention by granular media)	Retained	Retained	Retained
Infiltration (soils)	Likely sorbed	Likely sorbed	Likely sorbed
Thermal processes			
Biological Processes			
Microbes	Mineralization	Mineralization	Mineralization
Higher plants			
Algae			
Chemical Processes			
Aqueous equilibria			
Redox			
Photochemistry			
Abiotic degradation			

Table 29. Chemistry of Hydrocarbons

	PAHs	*BTEX*
Chemical States		
Fraction	Dissolved/Adsorbed	Dissolved/Adsorbed/NAP
General description	Hydrophobic	Slightly hydrophobic/volatile
Constituent or Analyte		
Interactions		
Overall effect of SCMs	Retain	Retain & degrade
Phase behavior		
Adsorption	Adsorbs strongly	Adsorbs weakly
Precipitation	Limited solubility	Limited solubility
Volatilization		Yes
Physical Processes		
Sedimentation	Sorbed X settles	Sorbed X settles
Filtration (retention by granular media)	Sorbed X retained	Sorbed X retained
Infiltration (soils)	Dissolved fraction	Dissolved fraction
Thermal processes		
Biological Processes		
Microbes	Somewhat degraded	Readily degraded
Higher plants	Uptake	Uptake
Algae	Uptake	
Chemical Processes		
Aqueous equilibria		
Redox		
Photochemistry	Direct and indirect	Direct and indirect
Volatilization		Highly volatile
Abiotic decay		

Table 30. Chemistry of Pesticides

	Hydrophilic	*Hydrophobic*
Chemical States		
Fraction	Dissolved/Sorbed	Dissolved/Sorbed/NAPL
General description		
Constituent or Analyte	Individual compounds	Individual compounds
Interactions		
Overall effect of SCMs	Slight retention	Strong retention
Phase behavior		
Adsorption	Some	Strongly sorbing
Precipitation		Limited solubility
Volatilization		
Physical Processes		
Sedimentation	Sorbed settles	Sorbed settles
Filtration (retention by granular media)	Retardation	Retardation
Infiltration (soils)	Dissolved fraction	Dissolved fraction
Thermal processes		
Biological Processes		
Microbes	Some degraded	Some degraded
Higher plants	Some taken up	Absorbed
Algae	Some taken up	Absorbed
Chemical Processes		
Reversible Reactions	Acid/Base	Acid/Base
Redox		
Photochemistry	Direct and indirect	
Volatilization		Some are volatile
Abiotic decay		

Table 31. Chemistry of Chloride

	Chloride Ion	*Chlorine (HOCl)*
Pollutant Description		
Fraction	Dissolved	Dissolved
General description	Anion (-1)	Strong oxidant
Constituent or Analyte	Chloride; TDS	Chlorine
Interactions	Binds some metals	
Overall effect of SCMs	Negligible	Consumed
Phase behavior		
Sorption		
Precipitation	Highly soluble	
Volatilization		Volatile
Physical Processes		
Sedimentation		
Filtration		
Infiltration	Not retarded	
Resuspension		
Biological Processes		
Microbes		Disinfectant
Higher plants	High levels are toxic	
Algae		
Chemical Processes		
Aqueous equilibria		Acid-base
Redox		Reduced to chloride
Photochemistry		
Abiotic degradation		

Table 32. Chemistry of Nitrogen

	Ammonia (NH3)	*Nitrate/Nitrite*	*Dissolved Organic N*	*Particulate Organic N*
Chemical States				
Fraction	Dissolved	Dissolved	Dissolved	Organic particles
General description	Cation (+1) and neutral	Anion (-1)	Neutral/Cation (+1)	
Constituent or Analyte	Ammonia-N	Nox-N	DON	PON
Interactions	Adds to BOD		Adds to BOD	Adds to BOD
Overall effect of SCMs	Retention	Retention	Retention	Strong retention
Phase behavior				
Sorption	Sorbs on clays	None		
Precipitation				
Volatilization	At *p*H > 9			
Physical Processes				
Sedimentation	+	---		+++
Filtration (retention by granular media)	++			+++
Infiltration (soils)		+++		
Thermal processes				
Biological Processes				
Microbes	Mineralization; Oxidation	Denitrification	Mineralization	Decomposition
Algae	Uptake	Uptake	Uptake	Primary Production
Higher plants	Uptake	Uptake	Uptake	From detritus
Chemical Processes				
Reversible Reactions	Acid/Base		Acid/Base	
Redox				
Photochemistry		Photochemically active		
Volatilization	Volatile at high pH			
Abiotic decay				

Table 33. Chemistry of Phosphorus

	Orthophosphate	*Polyphosphates*	*Organic P (DOP)*
Chemical States			
Fraction	Dissolved/Adsorbed	Dissolved/Adsorbed	Dissolved/adsorbed
General description	Anion (-1 to -3)	Anion	
Constituent or Analyte	Orthophosphate		Dissolved Organic P
Interactions			
Overall effect of SCMs	Strong retention	Strong retention	Strong retention
Phase behavior			
Adsorption	Adsorbs on metal oxides		
Precipitation	Forms minerals with Ca, Fe, Al		
Volatilization			
Physical Processes			
Sedimentation	Retains adsorbed OP		
Filtration (retention by granular media)	Retains adsorbed OP		
Infiltration (soils)	Strong retardation		
Thermal processes			
Biological Processes			
Microbes	SRC: Mineralization of organic P		Mineralizes to OP
Higher plants	Nutrient Uptake		Nutrient Uptake
Algae	Nutrient Uptake		Nutrient Uptake
Chemical Processes			
Reversible Reactions	Acid/Base		
Redox	Released by reduction/solubilization of Fe-oxides		
Photochemistry			
Volatilization			
Abiotic decay			

Table 34. Chemistry of Iron

	Ferric	*Ferric Oxide*	*$FeCN_6^{3-}$*	*Ferrous*	*Elemental*
Chemical States					
Fraction	Dissolved/Particulate	Particulate	Dissolved	Dissolved	Metal
General description	Variable (+3 to -1)	Mineral particles	Anion (-3)	Cation (+2)	Solid metal
Constituent or Analyte		Particulate Fe	Ferricyanide		
Interactions	Binds to DOM; Sorbent for P	Dissolves if no O2			Oxidized by DO
Overall effect of SCMs	Strong removal	Strong removal			
Phase behavior					
Adsorption	Adsorbs on most particles	Sorbent	Anion exchange		
Precipitation	Insoluble (forms FeOHx)	Insoluble			
Volatilizaton					
Physical Processes					
Sedimentation	Particles sediment	Particles sediment			Particulate
Filtration	Particles filtered	Particles filtered			
Infiltration (soils)	Strong retardation	Doesn't infiltrate	Transportable	Infiltrates	
Thermal processes					
Biological Processes					
Microbes	Reduced to Fe(II) in anoxic zones	Reduced to Fe(II) in anoxic zones		Oxidized in oxic waters	Dissolves
Higher plants	Uptake			Uptake	
Algae	Uptake			Uptake	
Chemical Processes					
Aqueous equilibria	Acid/Base; complexation	Dissolves at low *p*H			
Redox				Released by reduction/solubilization of Fe-oxides	
Photochemistry	Direct & indirect; forms Fe(II)	Direct photoreduction	Releases HCN		
Abiotic degradation					

4.7 References

Bakr AR, Fu GY, Hedeen D. 2020. Water quality impacts of bridge stormwater runoff from scupper drains on receiving waters: A review. Sci. Total Environ. 726:138068

Barnes RT, Raymond PA. 2009. The contribution of agricultural and urban activities to inorganic carbon fluxes within temperate watersheds. Chem Geol. 266(3-4):318–27

Brezonik PL, Arnold WA. 2022. Water chemistry: the chemical processes and composition of natural and engineered aquatic systems. Oxford University PressNew York

Conway TM. 2007. Impervious surface as an indicator of *p*H and specific conductance in the urbanizing coastal zone of New Jersey, USA. J. Environ. Manage. 85(2):308–16

Driscoll CT, Driscoll KM, Roy KM, Dukett J. 2007. Changes in the chemistry of lakes in the Adirondack region of New York following declines in acidic deposition. Applied Geochemistry. 22(6):1181–88

EPA US. 2005. National Management Measures to Control Nonpoint Source Pollution from Urban Areas. EPA-841-B-05-004, US Environmental Protection Agency

Erickson AJ, Weiss PT, Gulliver JS. 2013. Stormwater Treatment Processes. In Optimizing stormwater treatment practices, pp. 23–34. New York, NY: Springer New York

Gensemer RW, Playle RC. 1999. The bioavailability and toxicity of aluminum in aquatic environments. Crit. Rev. Environ. Sci. Technol. 29(4):315–450

Göbel P, Dierkes C, Coldewey WG. 2007. Storm water runoff concentration matrix for urban areas. J Contam Hydrol. 91(1-2):26–42

Husain IAF, Alkhatib MF, Jammi MS, Mirghani MES, Bin Zainudin Z, Hoda A. 2014. Problems, control, and treatment of fat, oil, and grease (FOG): a review. J. Oleo Sci. 63(8):747–52

Jiang G, Wightman E, Donose BC, Yuan Z, Bond PL, Keller J. 2014. The role of iron in sulfide induced corrosion of sewer concrete. Water Res. 49:166–74

Kaushal SS, Likens GE, Pace ML, Utz RM, Haq S, et al. 2018. Freshwater salinization syndrome on a continental scale. Proc. Natl. Acad. Sci. USA. 115(4):E574–E583

Kaushal SS, McDowell WH, Wollheim WM. 2014. Tracking evolution of urban biogeochemical cycles: past, present, and future. Biogeochemistry. 121(1):1–21

Kaushal SS, Wood KL, Galella JG, Gion AM, Haq S, et al. 2020. Making "Chemical Cocktails" - Evolution of Urban Geochemical Processes across the Periodic Table of Elements. Appl. Geochem. 119:1–104632

Klein RD. 1979. Urbanization and stream quality impairment. J Am Water Resources Assoc. 15(4):948–63

Laurenson G, Laurenson S, Bolan N, Beecham S, Clark I. 2013. The role of bioretention systems in the treatment of stormwater. In Vol. 120, pp. 223–74. Elsevier

Lead JR, Wilkinson KJ. 2006. Aquatic colloids and nanoparticles: current knowledge and future trends. Environ. Chem. 3(3):159

McFarland AR, Larsen L, Yeshitela K, Engida AN, Love NG. 2019. Guide for using green infrastructure in urban environments for stormwater management. Environ. Sci.: Water Res. Technol.

Nagai T, Imai A, Matsushige K, Yokoi K, Fukushima T. 2007. Dissolved iron and its speciation in a shallow eutrophic lake and its inflowing rivers. Water Res. 41(4):775–84

National Research Council. 2009. Urban stormwater management in the united states. Washington, D.C.: National Academies Press

Pamuru ST, Forgione E, Croft K, Kjellerup BV, Davis AP. 2022. Chemical characterization of urban stormwater: Traditional and emerging contaminants. Sci. Total Environ. 813:151887

Saarikoski J, Viluksela M. 1981. Influence of *p*H on the toxicity of substituted phenols to fish. Arch Environ Contam Toxicol. 10(6):747–53

Schlesinger W, Bernhardt E. 2020. Biogeochemistry: An Analysis of Global Change. Elsevier

Schwarzenbach RP, Gschwend PM, Imboden DM. 2002. Environmental Organic Chemistry. Hoboken, NJ, USA: John Wiley & Sons, Inc.

Spahr S, Teixidó M, Sedlak DL, Luthy RG. 2020. Hydrophilic trace organic contaminants in urban stormwater: occurrence, toxicological relevance, and the need to enhance green stormwater infrastructure. Environ. Sci.: Water Res. Technol. 6(1):15–44

Stagge JH, Davis AP, Jamil E, Kim H. 2012. Performance of grass swales for improving water quality from highway runoff. Water Res. 46(20):6731–42

Stets EG, Kelly VJ, Crawford CG. 2014. Long-term trends in alkalinity in large rivers of the conterminous US in relation to acidification, agriculture, and hydrologic modification. Sci. Total Environ. 488-489:280–89

Stets EG, Sprague LA, Oelsner GP, Johnson HM, Murphy JC, et al. 2020. Landscape drivers of dynamic change in water quality of U.S. rivers. Environ. Sci. Technol. 54(7):4336–43

Taguchi VJ, Olsen TA, Natarajan P, Janke BD, Gulliver JS, et al. 2020. Internal loading in stormwater ponds as a phosphorus source to downstream waters. Limnol. Oceanogr. 5(4):322–30

USEPA. 2017. Causal Analysis/Diagnosis Decision Information System (CADDIS): Volume 2. US Environmental Protection Agency, Washington, DC

Watershed Protection Center for. 2003. Impacts of Impervious Cover on Aquatic Systems. 1, Center for Watershed Protection

Werbowski LM, Gilbreath AN, Munno K, Zhu X, Grbic J, et al. 2021. Urban stormwater runoff: A major pathway for anthropogenic particles, black rubbery fragments, and other types of microplastics to urban receiving waters. ACS EST Water. 1(6):1420–28

Wolock DM, McCabe GJ. 1999. Explaining spatial variability in mean annual runoff in the conterminous United States. Clim. Res. 11:149–59

Xie S, Qi L, Zhou D. 2004. Investigation of the effects of acid rain on the deterioration of cement concrete using accelerated tests established in laboratory. Atmos. Environ. 38(27):4457–66

Chapter 5. Relations between Watershed Management Strategies and Stream Erosion, Turbidity, and Sedimentation: A Literature Review [WMO Article 208.4]

5.1 Introduction

The objectives of this chapter are to assess the current state of the science regarding the relations between watershed management strategies (volume control and watershed-specific release rates) and stream erosion, turbidity, and sedimentation and to help define the mechanisms of potential impacts of these management strategies. This chapter summarizes results from a review of the scientific literature on the impact of stormwater management practices on the magnitude and frequency of flows, water levels, and other hydraulic parameters such as stream power or shear stress downstream from these practices. Additionally, this chapter summarizes results from the literature describing the impact of stormwater management practices on nutrients (nitrogen and phosphorus), total suspended solids (TSS), iron, silver, and chloride.

The first part of this review focuses on the impact of watershed management strategies on downstream hydraulic and hydrologic effects, particularly as related to factors that may affect stream erosion, such as peak discharge, flow duration, shear stress, or stream power. The second part of this review focuses on the impact of watershed management strategies on downstream water quality. Both sections will examine the impacts of detention and retention practices, wetlands, and distributed small-scale practices.

5.2 Impact of Stormwater Control Measures on Downstream Hydraulics and Hydrology

This review identified 96 studies that examined the impact of stormwater control measures on downstream hydraulics and hydrology. The primary metric used in most studies on the impact of watershed management strategies was the change in flooding as represented by peak discharge or volume of flooding. Many studies examined the impact of the size and the location in the watershed of the practices and how these affected downstream peak discharge and flooding. Several studies considered the impact on flow hydrographs in addition to the peak discharge. Some studies examined the impact of watershed management strategies on baseflow, and some examined the impact on the overall flow duration curve. All studies that examined characteristics in addition to peak discharge were based on model simulations of hypothetical scenarios. A few studies examined the impact of watershed management strategies on stream erosion. The following sections describe the impacts of wetlands, detention and retention, and distributed stormwater controls on downstream hydraulics and hydrology. Studies that considered the potential impact on erosion have been highlighted in the following sections.

5.3 Impact of Wetlands on Downstream Hydraulics and Hydrology

Table 35 summarizes 13 studies describing the effects of wetlands on hydrology. The studies often use the National Wetland Inventory or the National Hydrography Dataset or both when studying wetlands over a large area to identify wetlands in the United States. The definition of a wetland used by the National Wetland Inventory, based on Cowardin et al. (1979), is:

> Lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water. For the purposes of this classification wetlands must have one or more of the following attributes: (1) at least periodically, the land supports predominantly hydrophytes; (2) the substrate is predominantly undrained hydric soil; and (3) the substrate is nonsoil and is saturated with water or covered by shallow water at some time during the growing season each year (Federal Geographic Data Committee, 2013).

The studies tend to agree that generally wetlands have a positive impact on hydrology downstream (Blanchette et al., 2019; Javaheri and Babbar-Sebens, 2014; Mitchell et al., 2018; Tang et al., 2020), but the placement of wetlands does matter and some wetlands can exacerbate flooding (Wu et al., 2020b). As wetland acreage increases, peak flow and floods tend to decrease (Blanchette et al., 2019; Demissie and Khan, 1993; Mitchell et al., 2018; Tang et al., 2020; Wu et al., 2020b) and low flow increases (Blanchette et al., 2019; Demissie and Khan, 1993). One study that compared wetlands to no-wetlands scenarios found that wetlands had fewer instances of discharges above 5% exceedance and below 75% exceedance, but the duration of these events lasted longer (Smakhtin and Batchelor, 2005). Another study found that the loss of wetlands increased flooding more than restoring wetlands decreased it (Al-Weshah et al., 1993). Notably, in two studies wetlands upstream seemed to have a bigger impact on hydrology (Wu et al., 2020b) than those downstream, and as storage capacity of the wetlands increased, the downstream flood area, flood depth, and flood duration decreased (Tang et al., 2020). In contrast, one study found that encroachment on downstream main stem wetlands associated with fifth-order streams had a larger impact than upstream wetlands associated with first-, second- and third-order streams (Ogawa and Male, 1986).

Table 35. Summary of Literature Describing the Effect of Wetlands on Hydrology

Authors	*Year*	*Description*
Al-Weshah et al.	1993	Removing wetlands has a bigger negative impact on flooding than the positive impact of restoring them.
Ayub et al.	2010	A wetland was constructed on a college campus as part of a larger BMP project to serve as a national pilot project. The study determined that the wetland was capable of storing water as intended and controlled the quantity of water that flowed downstream.
Blanchette et al.	2019	The study modeled two scenarios–1978 and 2014 with and without their wetlands–to determine the impact of land-cover changes. Without wetlands, the 1978 scenario would have 2–14% higher low flow support and a 15–26% reduction in high flow attenuation. Without wetlands, the 2014 scenario would have 3–20% higher low flow support and 16–20% reduction in high flow attenuation.
Demissie and Khan	1993	As wetlands increase in the landscape, peak flow and flood volume tend to decrease. For each 1% increase in wetland area, on average low flow increased 7.9%, peak flow to average precipitation ratio decreased 3.7%, and the flood flow volume to total precipitation ratio decreased 1.4%.
Evenson et al.	2018	Loss of larger wetlands (> 3 hectares) results in decreased mean daily cumulative inundated areas within wetlands. Loss of depressional wetlands has a greater impact on cumulative simulated residence time. Loss of wetlands far from a stream increases runoff-contributing areas. Any wetland loss affects peak flow and increases flooding risk downstream, but loss of larger wetlands had a bigger impact than smaller wetlands.
Javaheri and Babbar-Sebens	2014	The study found that wetlands can reduce the peak flow up to 42%, flood areas up to 55%, and maximum velocity up to 15%. The study noted that these reductions are not necessarily simultaneous, as a subbasin that had a maximum peak flow reduction for a specific design storm did not also have its maximum reduction in the flood inundation area. It also found that deeper wetlands are more effective at reducing the impacts of storms with higher return periods. For example, a 1.8 m deep wetland reduced peak flows for a 500-year storm by 20%, while a 0.5 m wetland only reduced it by 11%.
Mitchell et al.	2018	This study modeled the impact of changes in river discharge from different water-retention site implementation scenarios. Overall, the study found that water storage in the form of wetlands and detention can control high flows: as the water-retention site extent increases, high flows tend to decrease. The way this model was set up, water retention sites could only lose water through evaporation, overflowing, and seepage. Thus, the most important factor in site performance was hydraulic conductivity, which could enhance seepage. The study did not find a significant relationship between water retention site placement and performance.

Table 35. Summary of Literature Describing the Effect of Wetlands on Hydrology, Continued

Authors	*Year*	*Description*
Ogawa and Male	1986	The study simulated what happens if existing wetlands in a watershed were encroached upon. It found that encroachment of 25% results in minimal impact on peak flows, but complete 100% encroachment led to a 200% increase in peak flows in 38% of cases. Downstream main stem wetlands have a disproportionate larger effect on peak flows downstream than on upstream wetlands. This means that peak flow changes due to encroachment of downstream main stem wetlands can be seen at further distances from the encroached wetland than the changes in peak flow from upstream wetlands.
Smakhtin and Batchelor	2005	This study examined wetland hydrological functions using continuous streamflow records and flow duration curves. The study selected a discharge level that only 5% of flows exceed. This level was exceeded 144 times by a wetland and 187 times by the no-wetland scenario. However, the mean duration of the events was 3.5 days and 2 days, respectively. Essentially, the wetland had fewer higher discharges, but they lasted longer. The same pattern occurred below the 75% exceeded level. The wetland scenario had 65 occurrences below that level and the no-wetland scenario had 165 occurrences, but the mean durations were 38 days and 21 days, respectively.
Tang et al.	2020	Wetlands more upstream result in a smaller flood area, shallower flood depth, and shorter flood duration downstream. As wetland storage capacity increases, downstream flood area, flood depth, and flood duration decrease.
Wang et al.	2010	Loss of 10–20% of the wetlands study area would drastically increase peak discharge and loadings of sediment, total phosphorus, and total nitrogen. A loss of 10% of wetlands results in a 40% increase in peak discharge.
Wu et al.	2020a	Wetlands have a bigger impact on baseflow and quickflow in summer, and cumulative effects of wetlands increase with increasing wetland area.
Wu et al.	2020b	Wetlands can reduce peak flows by 24%, mean flows by 12%, event duration by 4%, and flow volume by 17%. Placement matters and wetlands can also exacerbate flooding. Upstream wetlands considerably decreased downstream peak flow, mean flow, and flow volume with a higher reduction efficiency but increased the downstream duration and flow volume on the falling limb of the hydrograph.

5.4 Impact of Retention and Detention on Downstream Hydraulics and Hydrology

Table 36 summarizes 25 studies on how retention and detention affect hydrology. In this instance, a detention pond refers to a method of stormwater control where water is temporarily stored with water draining from the detention pond in between storm events. A detention pond is synonymous with a dry retention pond and dry detention pond. A retention pond refers to a method of stormwater control where water is permanently stored within the pond and does not fully drain in between storm events. A retention pond is synonymous with a wet retention pond and wet detention pond. Detention and retention ponds have the largest impact when placed upstream in a watershed rather than downstream (Ayalew et al., 2015; Goff and Gentry, 2006; Su et al., 2010). Although detention and retention ponds often reduce peak discharge (Hess and Inman, 1994; Soong et al., 2009; Su et al., 2010; Thomas et al., 2016), they also extend the receding limb of the hydrograph (Bledsoe, 2002; Damodaram et al., 2010; Ferguson, 1995; Ravazzani et al., 2014; Soong et al., 2009; Su et al., 2010). One study did find that while developed areas with detention basins exceeded critical shear stress more than undeveloped conditions, the developed areas with detention basins minimized the exceedance compared to developed areas without control (Rohrer et al., 2005). Two authors found that shorter detention ponds (designed to release stored water within 24 hours) controlled flow duration characteristics and low durations better than longer detention ponds (48 hours) (Fan and Li, 2004; Li and Fan, 2010). When peak discharges are delayed by detention basins, the peaks can add up downstream with the delayed peaks of other streams, leading to more severe flooding (McCuen, 1974). In a study examining the impact of parallel detention basins (basins that only affect their own subcatchment) and detention basins in a series (detention basins placed on waterways that are subtended by another basin downstream), the total efficiency of flood peak reduction is related to the sum of each individual basin for parallel detention basins, while the total efficiency for basins in a series is much more complex and can even lead to lower efficiencies overall (Del Giudice et al., 2014).

Several studies have indicated that detention ponds should be designed larger to control for downstream channel erosion caused by increased shear stress (Bledsoe, 2002) and to account for matching 2- and 10-year peak discharges when continuous modeling is used (Booth and Jackson, 1997). In one case study, however, a tributary with a detention pond had severe channel incision, while another tributary draining an area of similar urban development was relatively unchanged (Booth and Henshaw, 2001). Focusing on matching pre-development sediment transport may lead to better erosion management than focusing on flow exceedances alone (Hawley et al., 2012). Detention ponds lose effectiveness as development increases (Goff and Gentry, 2006) and as they age because of sedimentation (Guo, 1997). One study even notes that after 40% urbanization, flow duration curves cannot recover to pre-development levels (Li and Fan, 2010). Damodaram et al. (2010) compared low-impact development (LID) practices (rainwater harvesting and permeable pavements on parking lots) with detention. They determined that only the detention pond resulted in a reduction of peak flows from design (2-year) events to pre-development peak flows, but the infiltration-based LID was more effective at reducing the volume from small storms.

Table 36. Summary of Literature Describing the Effect of Retention and Detention on Hydrology

Authors	*Year*	*Description*
Ahmadisharaf et al.	2021	In a comparison between larger and smaller retention ponds, large ponds have a bigger impact on flood reduction.
Ayalew et al.	2015	Retention ponds placed upstream in a catchment provide better flood protection; as they are moved downstream, the impact decreases. With two retention ponds, the maximum reduction of peak discharges of low exceedance probability occurs when the upstream pond is emptied first or has a larger storage capacity.
Bledsoe	2002	Detention ponds result in lower discharge of longer duration, even if peak flow of pre-development is met. In detention focused on peak flow reduction, there is an increase of 50% of flows exceeding critical shear stress from pre-development. A detention pond sized 61% larger would be necessary to control for erosion.
Booth and Henshaw	2001	A study examining stream erosion found that there was an overall absence of general relationships between the channel changes and physical parameters of the stream and watershed. Geologic parameters did have some influence; those with cohesive silt clay substrates had low rates of channel adjustment. The study rejected its hypothesis that urban development consistently increased the rate of channel change. An example of note is two tributaries that drain areas of intense urban development. The discharge into one goes through a detention pond, the other does not. The one with the detention pond has severe channel incision, while the one without was relatively unchanged.
Booth and Jackson	1997	In a continuous model, authors found that detention ponds would need to be up to 50% greater in volume than the Soil Conservation Service design for the 2–10 standard (matching two- and ten-year peak discharges).
Damodaram et al.	2010	The study used a modified runoff curve number to simulate impacts of low-impact development (LID) options (permeable pavement, rainwater harvesting, and green roofs) and compared with detention ponds for five events ranging from about 24% of the 1-year event to the 100-year event. Between LID and detention pond, only the detention pond resulted in reduction of peak flows to pre-development peak flows. Detention pond had longer duration, while LID had shorter duration. Detention is better at larger storms and LID is better at smaller storms.
Del Giudice et al.	2014	The study examined how a detention basin performs and how that performance is affected by multiple basins both in parallel and in series. They found that for parallel detention basins (detention basins placed in positions where they only affect their own subcatchment), the overall efficiency for flood peak reduction is related to the sum of the individual efficiencies. The series of detention basins (detention basins placed along different river branches subtended by another basin further downstream) is much more complex. Detention basins in a series can reduce efficiency in some cases.
Emerson et al.	2005	In a catchment model, over 100 detention ponds had little impact on streamflow. A possible reason: 75% of basin did not drain through detention ponds and so detention ponds and storms rarely exceeded the outflow rate of detention.

Table 36. Summary of Literature Describing the Effect of Retention and Detention on Hydrology, Continued

Authors	*Year*	*Description*
Fan and Li	2004	A study examining flow duration and flow regime through computer modeling found that flow duration changed significantly after urbanization. Extended detention basins with short detention times (24 hours) controlled flow duration characteristics at low flow rates better than those with longer detention times (48 hours). The study recommended both source control measures and extended detention basin to help manage the hydrograph.
Ferguson	1995	Urban stormwater hydrographs with varying levels of infiltration and detention routed through hypothetical drainage networks had increased flow duration for detention and reduced flow volume with shortened flow duration for infiltration. Neither reduced peak rate of flow to the degree they were designed.
Goff and Gentry	2006	This study examined how watershed and development characteristics were related to detention ponds and their impacts. Detention ponds were able to maintain pre-development flows for low-, medium-, and high-intensity development on second-order streams, but no scenario met pre-development flows on first-order streams. Detention is most effective at maintaining pre-development flows throughout the watershed when development is located upstream in the catchment and less effective when development is located downstream. Detention also decreases in effectiveness as development percentage increases and detention is less effective in watersheds that are elongated in shape.
Guo	1997	A study examined an 18-year-old dry detention basin and found that due to sedimentation, it had decreased its volume control from a 13-year storm to a 4-year storm.
Hawley et al.	2012	The study examined effectiveness of stormwater controls to reduce erosion impacts in two watersheds. The stormwater controls included above and below ground retention and detention, downspout disconnections, curb and walk filter media, bioswales, infiltration trenches, pervious pavement, and underground storage in streets. Focusing control on sediment transport (matching to pre-development) may be more effective than matching only the duration of flow exceedances because it may better represent channel stability. Of the options examined, only multi-stage detention fits the budget listed by the utility in the area of study and could store the required volume.
Hess and Inman	1994	Hydrographs have 140% increased peak flow with removal of detention.
Karuppasamy et al.	2009	The study examined how leaving detention ponds out of modeling for a hazard plan affected the final result. The study modified the Federal Emergency Management Agency models used by adding in detention, updating land use and storm sewer information, refining rating curves and changing the time step computation from 5 minutes to 1 minute. The impact was an increase of peak flows by up to 57% in some places and a decrease by up to 32% in others. The research attributes the decrease in peak flows to upstream detention.

Table 36. Summary of Literature Describing the Effect of Retention and Detention on Hydrology, Continued

Authors	*Year*	*Description*
Kohn et al.	2014	The study modeled different scenarios to attenuate peak flows and reduce sediment in a watershed using HEC-RAS. The 14 scenarios included nine scenarios with increasing numbers of detention ponds (from 7 to 44 detention ponds), one scenario with a watershed diversion weir, one scenario with a reservoir on the main stem, one scenario with stream armoring, and one scenario with stream widening. The scenarios that performed the best (attenuated peak flow and reduced sediment transport better than alternatives) were the scenarios with the maximum number of retention ponds (44) and the reservoir located on the main stem.
Li and Fan	2010	Extended detention ponds with short detention times (24 hours) controlled flow duration characteristics at low durations better than longer detention times (48 hours). After 40% urbanization, flow duration curves could not recover to pre-development.
McCuen	1974	An individual approach to detention can cause flooding when delayed peaks add up downstream. A regional approach would be preferable.
McCuen	1979	Stormwater management increases the duration of bankfull flow and can cause downstream flooding and channel degradation. Urbanization alters the timing of runoff by decreasing the time to peak. Detention basins attempt to reverse this, but it may not return the timing to its predevelopment timing and may cause flooding downstream. The study argues that detention basins cannot replace natural storage prior to development specifically related to timing. The study notes that measuring peak flow at the outlet of detention can overlook this phenomenon and that detention basin effectiveness should be examined using a regional lens to avoid increased duration of bankfull flows and associated erosion downstream.
Mullapudi et al.	2018	A stormwater detention pond retrofitted with a smart sensor can control the water release to control streamflow downstream. The study indicates that at the time of publishing the control site could be constructed for less than $3500 if the detention basins do not require structural modification.
Ravazzani et al.	2014	This study looked at a network of detention facilities in a heavily urbanized river basin. A dam had been installed to prevent downstream flooding, and the study looked at how 7 on-stream or 7 off-stream basins would affect the dam downstream. The model ran with just on-stream and just off-stream detention basins. They led to similar peak flow reductions (36% on stream and 31% off stream), but on-stream detention led to a shift ahead in time to peak while off stream was in phase with undisturbed timing. Off-stream detention basins had lower hydrograph volume because the model assumed that reservoir was released after the end of the flood. In on-stream detention basins, critical duration increases linearly with the area of the basin, while off-stream critical duration was approximately constant with increasing area. For a 100-year storm, not implementing detention far exceeds the discharge allowable at the dam and on-stream detention reduces peak flow, but not enough. Off-stream detention leads to the best scenario for the dam but is still above the discharge allowable.

Table 36. Summary of Literature Describing the Effect of Retention and Detention on Hydrology, Continued

Authors	*Year*	*Description*
Rohrer et al.	2005	This study used 53 years of historical precipitation data to model hydrologic and hydraulic conditions to determine best design conditions for detention basins. It determined that pre-development and post-development stress curves are different because there is inherently a greater volume of water after development. If the duration of flow above critical shear stress is maintained, then erosion potential does not increase. In its simulation of a watershed as developed with uncontrolled conditions (no detention), developed with controlled conditions (peak shaving of 1- and 10-year storms and extended detention with 40-hour drawdown) and undeveloped conditions, the duration of time that the critical shear stress is exceeded is 0.23% for developed, uncontrolled conditions, 0.03% for developed, controlled conditions, and 0.02% for undeveloped. For the lower critical shear stress, 0.08% is exceeded by undeveloped, 0.54% by developed with controlled, and 2.15% developed with uncontrolled conditions.
Soong et al.	2009	When specific release rates for detention ponds were simulated within a watershed, peak flows were reduced, but duration was longer.
Su et al.	2010	The study examined detention pond design and evaluated effectiveness at reducing peak discharges at various downstream locations. Detention ponds upstream were generally most effective at reducing peak discharges at downstream locations. Peak flow decreases, but duration increases. Detention can reduce downstream peak flows to pre-development levels but meeting the standard of no downstream impact would require an extra-large detention which is typically unrealistic and too expensive. Options to address this in other ways is to include a collection ditch at the bottom of the detention pond to cut off the “long tail flow” or offline detention ponds to prevent adding to peaks downstream.
Thomas et al.	2016	Detention ponds resulted in peak flow reduction of 3–17%, but the benefits decreased after approximately 100 km^2 drainage (~2 km downstream).

5.5 Impact of Distributed Stormwater Management Options on Downstream Hydraulics and Hydrology

Table 37 summarizes 33 studies on how different stormwater management options affect hydrology. A major theme of the literature is timing. Green infrastructure options like grass swales, rain gardens, biofiltration, open space, and forested floodplains lead to increased lag times in peak flows (Bell et al., 2020; Dixon et al., 2016; Hood et al., 2007; Jamrussri and Toda, 2017). Green infrastructure options also tend to lead to longer duration flows (James and Dymond, 2012; Pennino et al., 2016). In general, neighborhoods that are designed with low-impact development and green infrastructure tend to have reduced peak flows (Bedan and Clausen, 2009; Hood et al., 2007; Hopkins et al., 2020; Selbig and Bannerman, 2008; Williams and Wise, 2006; Zahmatkesh et al., 2015), greater lag times (Hood et al., 2007; Williams and Wise, 2006), and more baseflow (Zimmer et al., 2007) than their conventional counterparts. One study suggested basing design standards for stormwater control on sediment size could lead to less streambank erosion (Tillinghast et al., 2011), though another study indicated that it is difficult to maintain pre-development erosion levels in all stream reaches (Elliott et al., 2010). A related study found that when implemented on a large enough scale, stormwater control measures that emphasize infiltration, retention, and harvesting of surface runoff can reduce bed mobility potential towards pre-development levels (Anim et al., 2019). Based on four years of survey data, another study found that as impervious surface increases, so too does the bankfull cross sectional area (Hawley et al., 2013). One study sums up green infrastructure as working best with short-duration/low-intensity events, less well with long-duration/low-intensity events, and worst at high-intensity events (Tao et al., 2017). One study (Hopkins et al., 2017) indicated that distributed stormwater control measures had lower discharge per unit watershed area than centralized stormwater control measures for small (< 3 cm) events but greater discharge per unit watershed area than centralized stormwater control measures for large (> 3 cm) storm events. This study also indicated that centralized stormwater control measures showed a longer duration of runoff than distributed measures or forested watersheds.

Table 37. Summary of Literature Describing Effect of Distributed Stormwater Management Options on Hydrology

[In the table below, LID means low impact development, BMP means best management practices, and SCM means stormwater control measures. Studies marked with a * symbol indicate that the study focuses on infiltration trenches or infiltration basins.]

Authors	*Year*	*Description*
Acreman et al.	2003	Embanking a river increases peak flows by 50–150% downstream and increases peak water levels 0.5–1.6 m.
Ahiablame and Shakya	2016	The study examined how development affects flood risk and how rain barrels, porous pavement, and rain gardens could mitigate the effect. The practices were able to reduce annual runoff by 3–40% depending on what level they were deployed (25%, 50%, 75%, and 100%). Pervious pavement was most effective followed by rain gardens for parking areas, then rain barrels, and finally rain gardens for roofs.
Anim et al.	2019	A two-dimensional hydraulic model simulated the impact of stormwater control measures (bioretention systems, rainwater tanks, harvesting of stormwater pipes for off-stream storage and non-potable uses) in an urban catchment. When stormwater control measures are implemented on a large enough scale, they could restore in-stream hydraulics to close to natural levels. The study specifically modeled scenarios aiming to reduce surface runoff volume by 30%, 45%, and 65%. They reduced bed mobility potential, encouraged close to natural hydraulic diversity, and resulted in improvement of retentive habitat availability.
Aulenbach et al.	2017	For every 1% increase in impervious surface, there needs to be an increase of 2.6% peak streamflow, 1.1% stormwater yield, and 1.5% streamflow runoff treated by BMPs.
Bedan and Clausen	2009	Researchers compared two neighborhoods, one with LID and one without. Flow increased in the conventional neighborhood after construction, and there was a 42% decrease in flow in the LID neighborhood. LID included grassed bioretention swales, rain gardens, pervious pavement, and open space.
Bell et al.	2020	This study sought to establish a relationship between stormwater control measures and the change in hydrology by conducting a literature review. It found for each 1% of stormwater control measure mitigated impervious area in a watershed, an additional decrease in runoff of 0.43% and 0.60% reduction in peak flows occurred. The more impervious a watershed, the larger the reduction per percent mitigated.
*Bergman et al.	2011	The study investigated two infiltration trenches 15 years after implementation. The study found that the infiltration rate had decreased significantly (statistically significant). If the clogging rate continues, the study estimates that in 100 years there will be 60% overflow of total incoming runoff.
Bizzi and Lerner	2015	A study examined how total and specific stream power (TSP and SSP) can be used to identify dominant processes such as erosion, transport, and deposition within a stream channel. A decrease in both TSP and SSP from upstream to downstream indicates a deposition-dominated stream while local stream power drives local erosion. The minimum energy to trigger erosion is TSP=1648 W/m and SSP=34 W/m^2.

Table 37. Summary of Literature Describing Effect of Distributed Stormwater Management Options on Hydrology, Continued

Authors	*Year*	*Description*
Black et al.	2021	Lag times (centroid of rainfall and hydrograph peak) increases of 2.6–7.3 hours in headwater catchments up to 26 km^2 where leaky wood structures, online ponds, and riparian plantings were implemented. Larger catchments downstream and those with just riparian plantings did not have significant increases in lag time.
Dixon et al.	2016	Logjams have highly variable results, forested floodplains result in reduced peak magnitude at the outflow if in the middle or upper catchment, and riparian forest restoration at 20–40% results in 19% peak magnitude reduction due to desynchronized timing.
Elliott et al.	2010	The Storm Water Management Model (SWMM) was modified to include on-site flow control devices and then used to model hydraulic habitat suitability and erosion potential. Urbanization increased erosion potential by a factor of 1.58–9.32, but erosion control ponds and detention tanks could significantly reduce or return it to pre-development levels. Due to poor soil drainage, infiltration was not effective at reducing erosion. The erosion potential curves predicted no sediment movement at below 0.7 m^3/s, 0.03 m^3/s, and 0.05 m^3/s for the three reaches analyzed. The authors suggest it is difficult to maintain erosion below pre-development in all stream reaches.
Fitzpatrick et al.	2005	Examined urban indicators, landscape characteristics, geomorphic characteristics, habitat characteristics, hydrologic characteristics, and fish. Below 30% watershed urban land, unit area discharge for a 2-year flood increased with increasing urban land; above 30% urban land, unit area discharges were variable. Channel enlargement happened in urban streams with a high percentage of watershed clayey surficial deposits.
Flegel et al.	2019	Illinois State Water Survey methodology to determine watershed-specific release rates.
Garcia-Cuerva et al.	2018	This study looked at the benefits of implementing green infrastructure (specifically bioretention cells and/or rain harvesting) within marginalized communities. It found that full catchment deployment would be required to significantly mitigate runoff. As storm size increased, the study found the size of the implemented green infrastructure and the coverage of the watershed have a greater impact on volume and peak flow reduction. The study found a decentralized scenario was more efficient than a centralized one, but in these scenarios, decentralized had greater coverage, while centralized had more routing of pervious flows.
Hawley and Bledsoe	2013	Case study examined stream that increased its channel significantly (14-fold) and found that cross sectional channel enlargement was very dependent on the ratio of post- to pre-urban sediment transport capacity over cumulative duration simulations of 25 years (accounted for almost 60% of variance). This highlights the importance of focusing on a range of flows for managing stream stability. The study recommends that management focus on magnitude and duration of all flows above the critical flow for entrainment of the channel bed material.

Table 37. Summary of Literature Describing Effect of Distributed Stormwater Management Options on Hydrology, Continued

Authors	*Year*	*Description*
Hawley and Vietz	2016	Study established a model that can be used to predict an order of magnitude approximation of the critical discharge for bed particle entrainment using bed material class and 2-year peak discharge.
Hawley et al.	2013	The study monitored streams over four years. They found that higher impervious surface catchments led to stream cross sections becoming larger, stream riffle lengths shortening, their pools becoming longer and deeper, and bed composition coarsening. One study site increased the bankfull area by 47% in just over a year after the mostly forested watershed experienced an increase of 4% imperviousness. For every 1% of impervious cover, the bankfull cross-sectional area increased significantly ($p< 0.5$) at an average rate of 0.075 m^2. The watersheds examined had a variety of stormwater control policies including no control and peak matching, but neither seemed to produce channel stability.
Hawley et al.	2020	When streams reach above 5% total impervious area, they begin to coarsen (stream bed) and widen. The only streams with greater than 5% total impervious area that did not coarsen or widen (4 of 45 streams) had stabilization efforts through stream restoration, stormwater retrofits, or appeared to be entering the stage of coarsening and widening. It seemed that developed watersheds led to more widening than undeveloped watersheds.
Hood et al.	2007	LID had significantly greater lag times for storms of less than 25.4 mm, storms with less than 4-hour duration, and antecedent moisture conditions less than 25.4 mm than traditional development. LID also had lower peak discharge than conventional. LID included grass swales, rain gardens, and biofiltration along with open space.
Hopkins et al.	2017	Compared a forested watershed (3% impervious) with developed watersheds (30–39% impervious) treated with centralized and distributed stormwater control measures. Showed lower peak discharge and discharge per unit area for distributed measures for storms less than 3 cm but greater runoff for larger storms. Showed longer duration of runoff for all events for centralized measures than for distributed measures.

Table 37. Summary of Literature Describing Effect of Distributed Stormwater Management Options on Hydrology, Continued

Authors	*Year*	*Description*
*Hopkins et al.	2020	The study compared four watersheds, a forested control, an urban control with 40% impervious surface and detention-based stormwater control measures (SCM), and urban treatment (1) with 33% impervious surface and infiltration-based SCMs (recharge chambers and infiltration trenches) and an urban treatment (2) with 44% impervious surface and infiltration-based SCMs (tree boxes, infiltration trenches, and underground detention) using long-term monitoring data. For the precipitation events from 1 to10 mm, the urban control had significantly higher peak streamflow, runoff yield, and runoff ratios than the urban treatments or forested control. For events from 11 to 20 mm, the urban treatments were not significantly different from each other. For events from 21 to 50 mm, the urban treatment 1 was similar to the forested control, while urban treatment 2 was not significantly different from the urban control. Duration and time to peak was typically greater in urban treatment 2 than in urban treatment 1. The study concludes that its results indicate better hydrologic performance from high-density infiltration-based SCMs than low-density detention-focused SCMs (urban control).
James and Dymond	2012	A bioretention cell can reduce peak flow, but volumes do not return to predevelopment levels. Flows also remain higher than average for a longer period of time than predevelopment.
Jamrussri and Toda	2017	The study examined how installing non-structural methods, reforestation, retention areas, and reforestation + retention areas, would affect flash floods. It found that these non-structural methods could be effective at reducing peak discharges and flood volumes.
Jarden et al.	2016	The study looked at two streets where there was voluntary participation in green infrastructure (GI), one street with 32.2% and one with 13.5% of lots with GI installed over 2 years. On the street with smaller lots and lower participation, GI reduced peak discharge by up to 33% and total storm runoff by up to 40%. On the street with larger lots and larger participation, there was no significant reduction, which may be due to contemporaneous street repairs. GI included bioretention cells, rain gardens, and rain barrels.
Loperfido et al.	2014	Although distributing BMPs across the landscape rather than centralizing them results in better response, forest land and impervious surface area still appear more important than BMP placement.
*Natarajan and Davis	2015	Researchers examined an infiltration basin that has shifted over time to resemble a wetland. They found that peak flows were reduced by 67%, flow volume by 67%, and overall flow rates were observed.

Table 37. Summary of Literature Describing Effect of Distributed Stormwater Management Options on Hydrology, Continued

Authors	*Year*	*Description*
Ogden et al.	2011	The study examines how different factors affect flooding in a watershed in Maryland. It determines that imperviousness is relatively unimportant in storms at 100-year recurrence or more in terms of runoff efficiency and volume but can affect peak flow. Width function (a measure of how many stream links located the same distance from an outlet) has a large effect on flood peaks. The watersheds with more stream density further away from the outlet had significantly higher first and second peak discharges than watersheds with more stream density closer to the outlet for both simulations with impervious surfaces and without. Subsurface drainage networks also have an impact, as they increase the magnitude of the first hydrograph peak.
Pennino et al.	2016	More stormwater green infrastructure (detention ponds, shallow marshes, wet ponds, sand filters, infiltration trend/basins, bioretention, and swales) results in less flashy hydrology, lower peak runoff, less frequent runoff events, and less variable runoff. It also means a longer hydrograph duration.
*Selbig and Bannerman	2008	A comparison of two catchment basins, traditional development versus LID, results in less volume of runoff and lower peak flows in LID. LID included grass swales, a detention pond, and an infiltration basin. The infiltration basin performed particularly well.
Sohn et al.	2020	This study examined the effects of impervious surfaces on urban flooding and streamflow. It found that the control of total impervious area (TIA) and directly connected impervious area (DCIA) for volume control were most effective for monthly rainfall of a 5–10% probability of exceedance, and peak flow reduction was most effective if the 24-hour peak storm in a month had a 2–5% exceedance. These results supported the idea that DCIA mitigation should be prioritized over TIA mitigation for both volume and peak flow control in high-risk flooding.
*Tao et al.	2017	This study modeled green stormwater infrastructures (infiltration trenches, stormwater bumpouts, and stormwater tree trenches) in a watershed to determine impacts on CSOs and flooding. It found that for short duration and low intensity storms, green stormwater infrastructure has the best performance, and thus, helps reduce stormwater in the combined sewer area. GSI does less well at low intensity long duration events with lower efficiency for volume and peak flow reduction. GSI has the worst performances for high intensity events.
Tillinghast et al.	2011	The study looked at potential impacts of sub-bankfull flows resulting from stormwater control measures on stream stability. It found that most detention-based stormwater control was designed for the 2- to 10-year discharge events, but in this study, 94% of the calculated critical discharges for the d_{65} substrate size were below the 2-year discharge. When designing stormwater control, basing design on sediment size could reduce stream bank erosion.

Table 37. Summary of Literature Describing Effect of Distributed Stormwater Management Options on Hydrology, Continued

Authors	*Year*	*Description*
Webber et al.	2020	Researchers created a framework to model the effectiveness of green infrastructure (green roofs, rainwater capture, tree pits, permeable pavement, rain gardens, enhanced catchment storage). They found that intensive application of green infrastructure could reduce flood depth and velocity, but there are still residual risks. No GI completely prevented surface water flooding in the areas studied during any rainfall event.
Williams and Wise	2006	When traditional development, cluster development, partial LID, and full LID are compared, a combination of land preservation and infiltration-based stormwater management results in response closest to natural conditions, maintaining peak flow near pre-development levels. LID shifted the peak timing. For 2-year storms, the change is small, but for 25-year storms it could cause unanticipated flooding downstream. LID includes infiltration-based measures with some detention and retention ponds.
Yang and Li	2013	The study looks at two watershed-scale community projects in Houston, one with green infrastructure and one conventional. It found that the impervious cover percentage in the GI site (32.3%) is more than twice that of the conventional site (13.7%). The GI site's precipitation streamflow ratio maintains a steady, and there is minimal correlation of nutrient loading with impervious surface cover. The conventional site has a fluctuation in streamflow ratio (30–66%), implying a flashy stream condition, and the nutrient loading is significantly correlated with impervious cover percentage.
Yang et al.	2011	Flood peaks tend to be smaller and arrive earlier with urbanization if development occurs in areas with shorter travel time, so the largest impacts are not necessarily seen immediately downstream.
Zahmatkesh et al.	2015	A study modeled climate change and LID in an urban New York watershed with SWMM. It included rainwater harvesting, porous pavement, and bioretention. It found that climate change increased historical annual runoff volume by 48%, but that LID provided an average 41% reduction for their modeled scenario. Between 10% and 90% of each of the subwatersheds were considered for LID implementation with 0–30 units of each LID per hectare. Application of LID controls also showed promise at reducing peak flows by 8–13% on average and decreasing watershed runoff by 28 and 14% for 2-year and 50-year return periods, respectively.
Zimmer et al.	2007	In comparing an 1871 scenario, existing urban development in 2004 (no low impact development (LID)) scenario, and a retrofit of a subdivision with LID (disconnecting impervious surface, street swales, bioretention, forest), LID produces less runoff but more baseflow than the 1871 scenario. It also reduced the duration of erosive flowrates in the test watershed. Increasing forest cover and bioretention infiltration had the largest roles in decreasing peak flow rate and total flow volume.

5.6 Impact of Stormwater Control Practices on Downstream Water Quality

This review identified 57 studies that examined the impact of wetlands on the removal of nutrients and 19 studies that examined the impact of detention and retention on the removal of nutrients. This review identified 52 studies that examined the removal of TSS, 16 studies that examined the removal of iron, 8 studies that examined the removal of chloride, and 3 studies that examined the removal of silver by stormwater management practices. There are five categories of nitrogen removal included: nitrates (NO_3), nitrates and nitrites ($NO_3 + NO_2$), total ammonia nitrogen ($TAN=NH_3 + NH_4^+$), total Kjeldahl nitrogen (TKN), and total nitrogen (TN). Of the five categories, more studies (34) listed removal rates for TN than for any other category. Although TAN has the fewest at two studies, many studies did report on either NH_3 or NH_4^+ removal but did not report the combined NH_3 or NH_4^+ removal that makes up TAN. The studies are not consistent in how removal rates were quantified, with some studies reporting reductions based on concentrations and other studies reporting reductions based on loads. The studies vary widely in their reported removal of nutrients, with some studies reporting an increase in downstream nutrients and others reporting nearly 100% removal. Figures showing the reported removal of phosphorus and nitrogen categories are provided to illustrate the range of reported removals.

5.7 Impact of Wetlands on Nitrogen Removal

Table 38 summarizes 49 studies on the effect of wetlands on nitrogen removal. The studies vary widely on reported TN removal; some reach above 90% removal (Luederitz et al., 2001; Rodríguez and Brisson, 2015) and some export almost 50% more than was input (Carleton et al., 2001; Lenhart and Hunt, 2011). Figure 64 through Figure 68 depict the removal rates by nitrogen removal. Several studies indicated that nitrogen removal was seasonal (Beutel et al., 2009; Horne, 2001; Lu et al., 2009). As retention time increased, pollutant removal increased (Huang, 2000; Tanner et al., 1995) and during storm events, wetlands struggled to maintain removal rates, instead often exporting nutrients (Raisin et al., 1997; Spieles and Mitsch, 1999). Another study found that placing wetlands in a series was not effective as only the first wetland consistently removed nutrients (Hathaway and Hunt, 2010), echoing another study that found placing a wetland after a primary facultative pond performed better than a wetland placed after a series of facultative ponds (Senzia et al., 2003). There were studies with contrasting results. One study found that using concentration instead of load removals could underestimate TN removal by up to 100% (Moustafa et al., 1996). Of those studies examined here that listed removal percentages for both load and concentration calculated from the same data in an experiment, three studies had load removal higher than concentration (Al-Rubaei et al., 2016; Lenhart and Hunt, 2011; Moustafa et al., 1996), three studies had overlapping ranges of load and concentration removals (Hoffmann et al., 2012; Kovacic et al., 2006; Luederitz et al., 2001), and one study had load removals that were lower than concentration removals (Carleton et al., 2000).

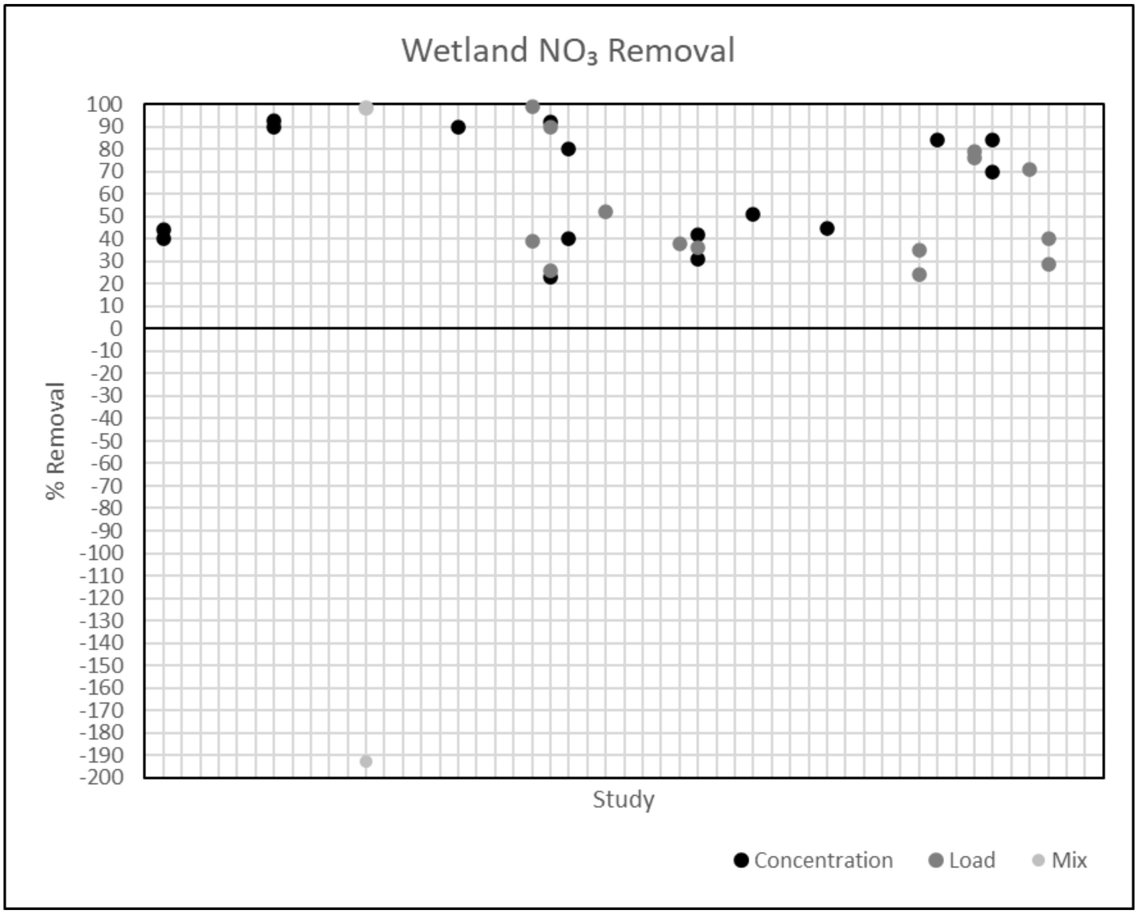


Figure 64. Scatterplot of wetland NO_3 removals

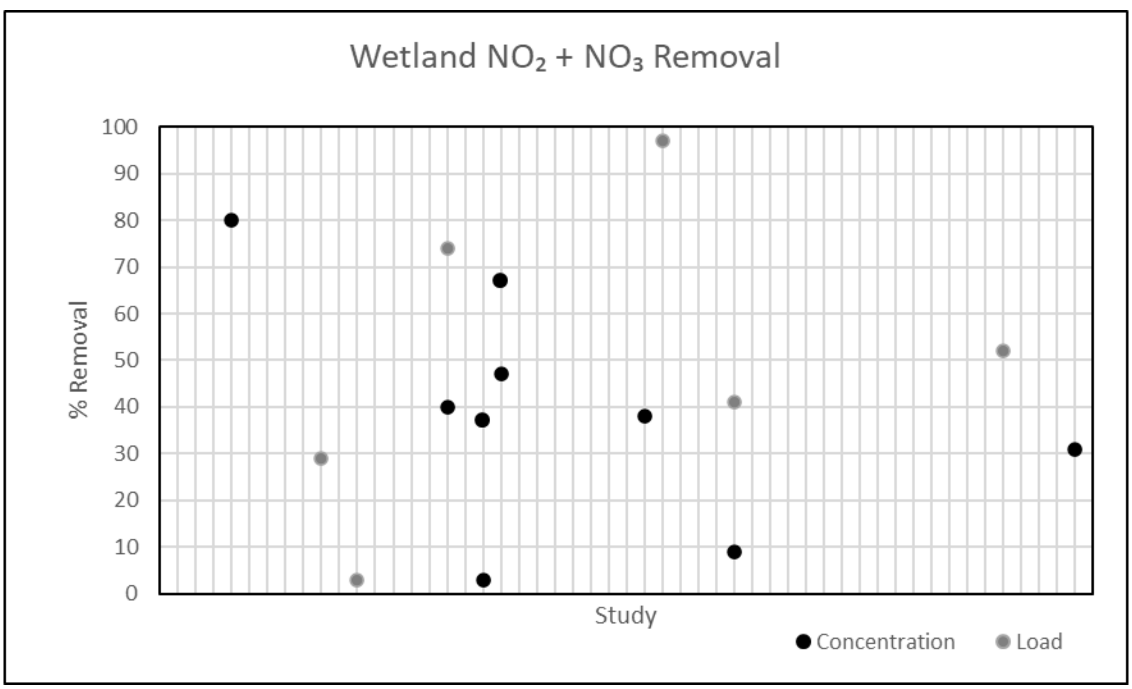


Figure 65. Scatterplot of wetland NO_2 + NO_3 removals

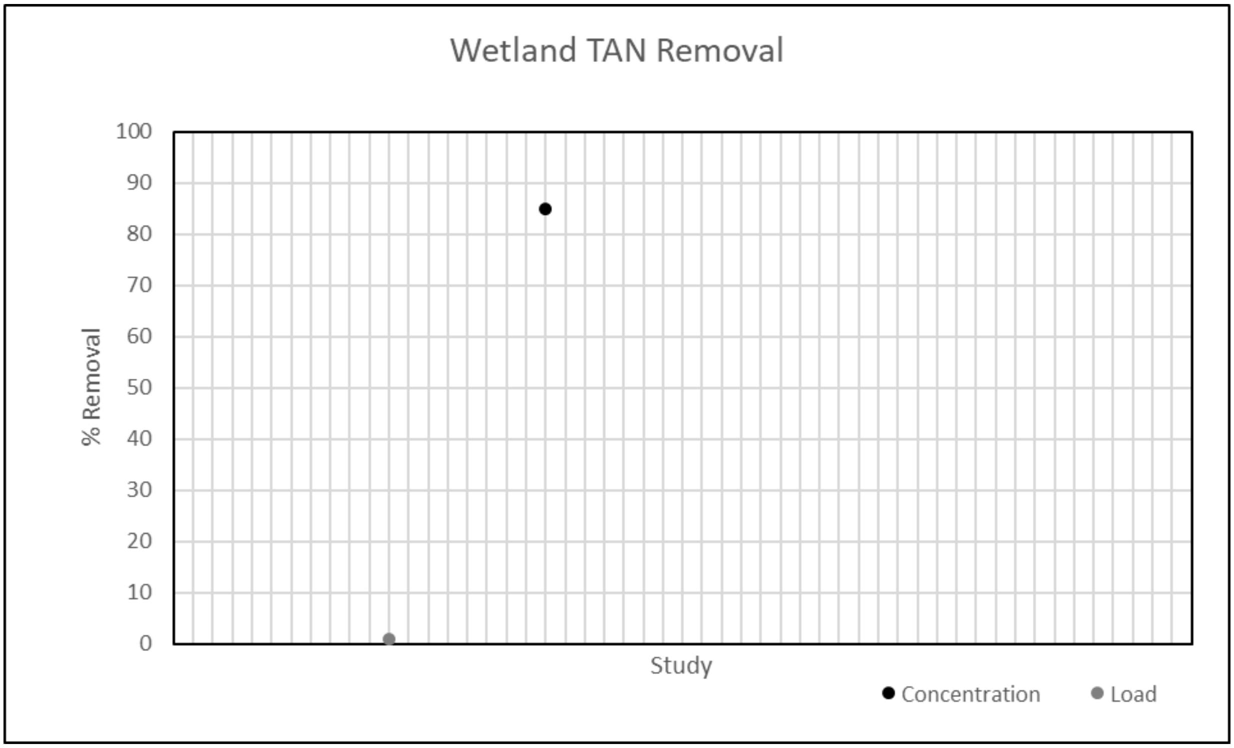


Figure 66. Scatterplot of wetland TAN removals

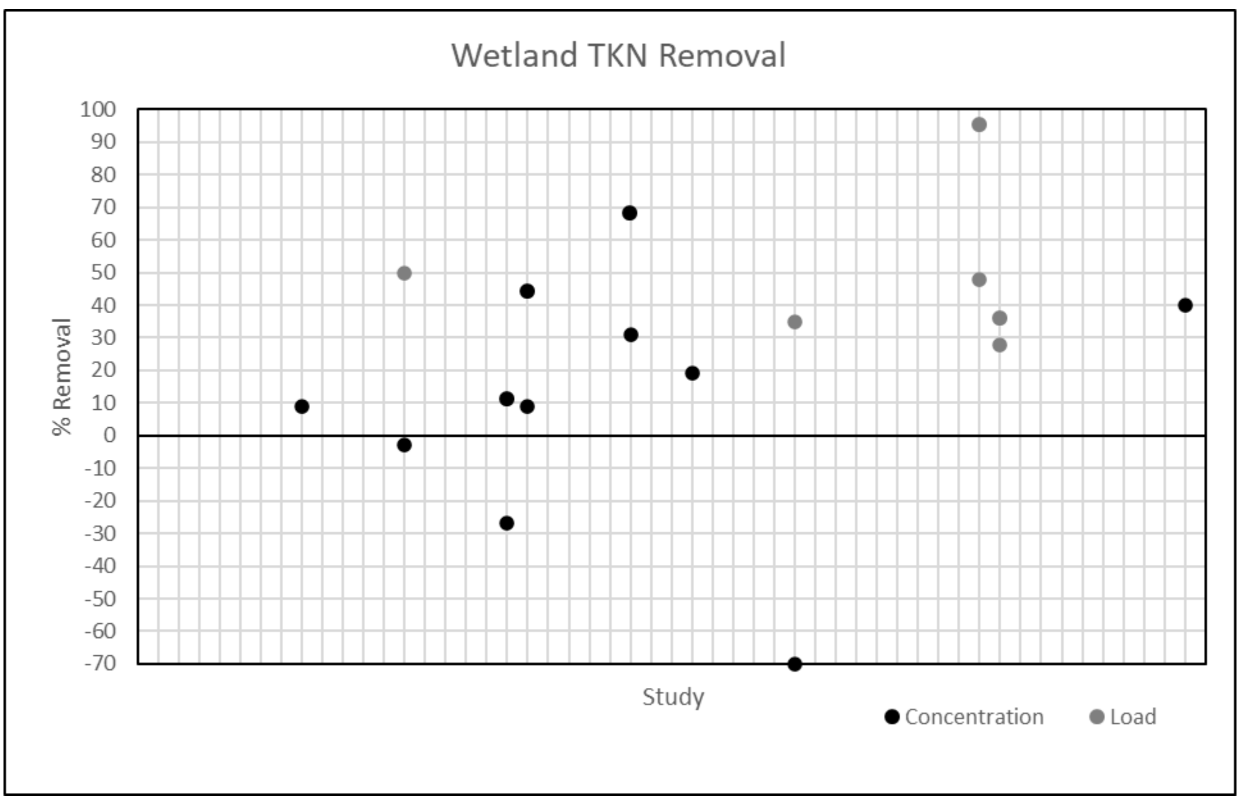


Figure 67. Scatterplot of wetland TKN removals

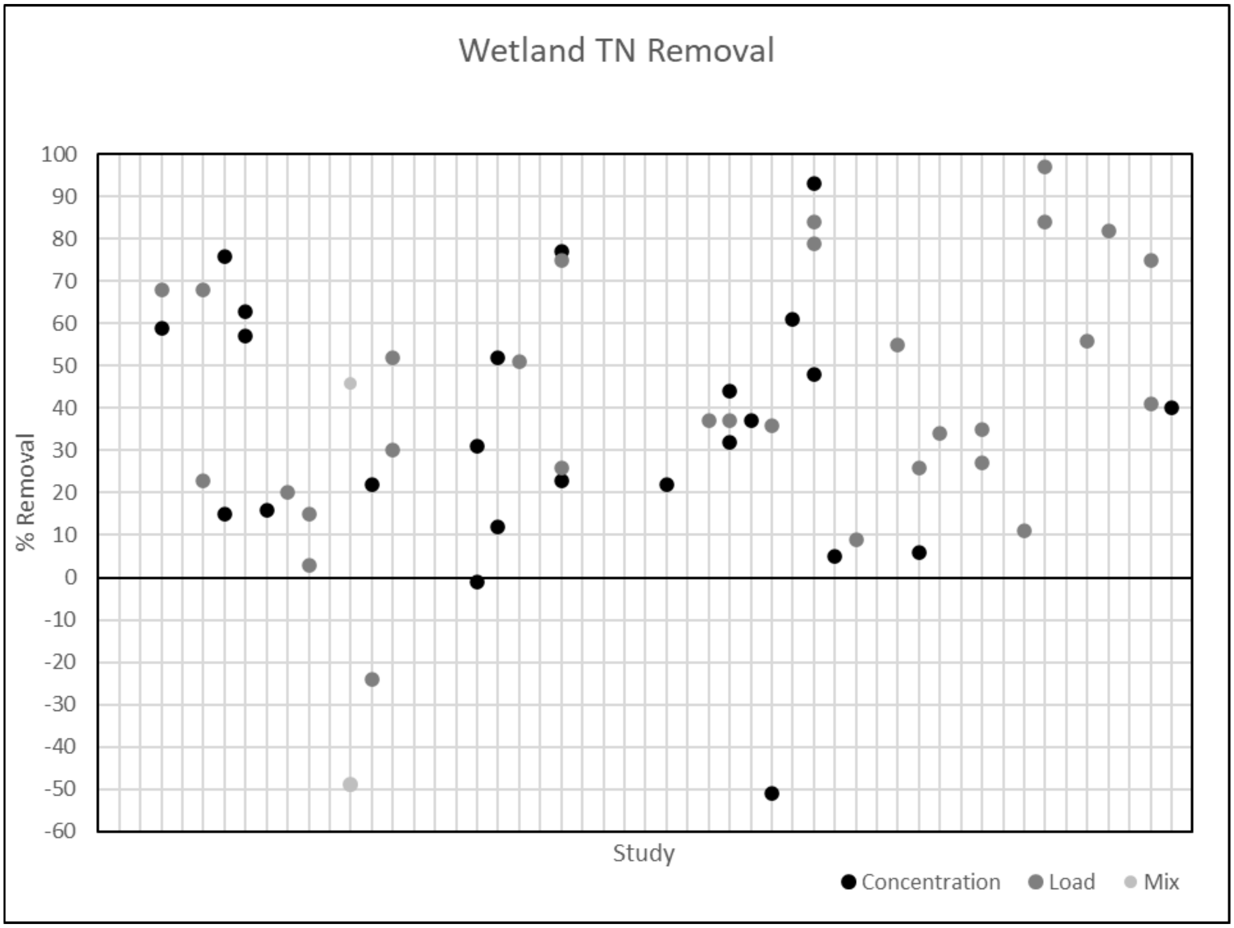


Figure 68. Scatterplot of wetland TN removals

In Figure 64 through Figure 68, each vertical line represents one of the studies listed in Table 38. The points represent the percent removal using concentration, load, or a mix of concentration and load methods. Mean and median values are not differentiated in these figures. Multiple points on a single vertical line represent a different value reported by the study. Multiple points of the same color on the same line indicate that the study provided a range, and the points represent the upper and lower bound of the range.

Table 38. Summary of Literature Describing the Effect of Wetlands on Nitrogen Removal

[There are five categories of nitrogen removal listed: nitrates (NO_3), nitrates + nitrites (NO_2 + NO_3), total ammonia nitrogen (TAN=NH_3 + NH_4^+), total Kjeldahl nitrogen (TKN), and total nitrogen (TN). Nitrogen removal is calculated by load (mass) or by concentration, denoted with an "l" or "c", respectively. For studies that indicated their calculations were mean or median percentages, the removals are marked with an "m" or "d", respectively. Negative percentages are enclosed in parenthesis (-) and indicate that the wetland served as a source of nitrogen. Ranges are indicated with a dash in between the values. The words "not sig" are an abbreviation for not significant.]

Authors	*Year*	*Description*	*NO_3*	*NO_2 + NO_3*	*TAN*	*TKN*	*TN*
Adler et al.	1996	To optimize phosphorus removal, researchers made phosphorus the limiting nutrient and harvested plant biomass regularly.	40-44% c				
Al-Rubaei et al.	2014	A 19-year-old constructed wetland and pond with no maintenance performed since construction still recorded nutrient and metal concentration reductions.					61% c
Al-Rubaei et al.	2016	A constructed wetland and pond with no maintenance performed still reduced nutrient concentrations after 19 years without maintenance and compared to its removal rates at three and nine years old, it performed more efficiently and stably.					59% c 68% l
Álvarez et al.	2013	A laboratory experiment of a compost-based constructed wetland handling gold mine effluent was able to remove nitrate and nitrite.		80% c			
Andersson et al.	2005	The performance of four large surface wetlands (20–28 hectares) removed both total phosphorus and total nitrogen.					23-68% l

Table 38. Summary of Literature Describing the Effect of Wetlands on Nitrogen Removal, Continued

Authors	*Year*	*Description*	*NO_3*	*$NO_2 + NO_3$*	*TAN*	*TKN*	*TN*
Babatunde et al.	2011	A wetland constructed using waste product from drinking water production (dewatered alum sludge) removed nutrients and became more efficient over time.					15-76% c
Beutel et al.	2009	A constructed treatment wetland with a desedimentation basin and two surface flow wetlands removed nitrogen. Rates appear to fluctuate seasonally. Only wetland removal listed.	90-93% c				57-63% c
Birch et al.	2004	A wetland that drains a residential urban catchment reduced metal and nutrient concentrations, but still did not meet water quality standards for boating.				9% c, m	16% c, m
Borden	2001	Two wet detention ponds and one pond wetland system had varying levels of success for nutrient removal. Only wetland system listed.		29% c			20% c
Braskerud	2002	Four cold temperate climate surface flow constructed wetlands retained about 3–15% of total N input.					3-15% l
Brown	1984	An urban wetland was less effective at removing dissolved pollutants as sedimentation was the key process of removal, but it still removed some nutrients.		3% l, m	1% l, m		
Carleton et al.	2000	A constructed wetland treating stormwater runoff was able to remove some nutrients and metals to varying degrees depending on the measurement method used. It was not effective at removing TKN by median concentration or load or TN by median load.				(-3%) c, d (-50%) l, d	22% c, d (-24%) l, d

Table 38. Summary of Literature Describing the Effect of Wetlands on Nitrogen Removal, Continued

Authors	*Year*	*Description*	*NO_3*	*$NO_2 + NO_3$*	*TAN*	*TKN*	*TN*
Carleton et al.	2001	An analysis of 35 wetland studies found removal rates varied widely.	(-193)-98% mix c, l				(-49)-46% mix c, l
Coveney et al.	2002	In a wetland constructed to reduce nutrients in a eutrophic lake, particulate matter was reduced by 90%, but soluble inorganic compounds increased (though levels were low).					30-52% l
Fink and Mitsch	2004	A 1.2 hectare created/restored wetland receiving groundwater and stormwater flows exported more phosphorus in precipitation events than dry weather flows, but there was no significant increase in nitrate-nitrite exports.		40% c, m 74% l, m			
Gessner et al.	2005	A wetland examined for its ability to reduce cyanide and hydrocarbons was also found to reduce nitrogen and phosphorus.	90% c				
Guerrero et al.	2020	A comparison between two regional detention facilities with wetlands.		(-3)-37% c, m 13-16% c, d		(-27)-11% c, m 2-3% c, d	(-1)-7% c, m 2-31% c, d
Hathaway and Hunt	2010	In a series of three wetlands, the first wetland removed at least 80% of the total concentration for all pollutants, and no pollutant was significantly reduced from the outlet of wetland 2 to the outlet of wetland 3. The removal efficiencies are listed in order of wetland (1 at the top, 3 at the bottom).		67% c 47% c not sig c	85% c not sig c not sig c	44% c 9% c not sig c	52% c 12% c not sig c
Healy and Cawley	2002	A recently constructed surface flow wetland was investigated as a potential tertiary treatment option, and while it performed well at N reduction, it was less effective at P reduction.					51% l

Table 38. Summary of Literature Describing the Effect of Wetlands on Nitrogen Removal, Continued

Authors	*Year*	*Description*	*NO_3*	*$NO_2 + NO_3$*	*TAN*	*TKN*	*TN*
Hey et al.	1994	Four experimental wetlands were examined for percent removals, and researchers found that the wetlands all had similar outlet concentrations despite different loading rates.	39-99% l				
Hoffmann et al.	2012	Two restored wetlands that received drainage water from agricultural fields rich in nitrate were monitored five years later and found to perform well at nitrogen removal.	23-92% c 26-90% l				23-77% c 26-75% l
Horne	2001	Two case studies of constructed wetlands in California indicate that nitrate removal was seasonal with less removed in winter than in summer.	80% c summer 40% c winter				
Huang	2000	In constructed wetland laboratory experiment, increased residence time resulted in decreased NH4 and TKN concentrations.				31-67% c	
Jordan et al.	2003	In a wetland receiving inflows from a 14-acre agricultural watershed, the wetland performed better in the first year of the two-year study due to a drying period.	52% l				
Kadlec et al.	2010	Four free-surface wetlands were examined and, due to high loading, did not reduce nutrient concentrations by a large degree, but did remove 98 t/yr of nitrogen and 3.6 t/yr of phosphorus.		38% c		19% c	22% c
Kohler et al.	2004	A four-year study on golf course wetlands indicated the wetlands were able to efficiently remove N-NO3/NO2, N-NH3, and P. Pesticide was only detected once, and it was a type not used on golf courses.		97% l			

Table 38. Summary of Literature Describing the Effect of Wetlands on Nitrogen Removal, Continued

Authors	*Year*	*Description*	*NO_3*	*$NO_2 + NO_3$*	*TAN*	*TKN*	*TN*
Kovacic et al.	2000	Three treatment wetlands receiving tile drainage in addition to runoff were effective at reducing NO3-N, but less so at TP. Adding a buffer between wetlands and river increased removal efficiency.	38% l				37% l
Kovacic et al.	2006	Two runoff wetlands handling agricultural runoff were able to reduce both P and nitrate nitrogen.	36% l 31-42% c				37% l 32-44% c
Land et al.	2016	93 articles on created or restored freshwater wetlands were reviewed and median removal rates were found for TN and TP.					37% c, d
Lenhart and Hunt	2011	Four different metrics of performance were examined for a constructed wetland. Concentrations of major nutrients increased, but loads of nutrients decreased.		9% c, m 41% l, m		(-70%) c, m 35% l, m	(-51%) c, m 36% l, m
Lu et al.	2009	A free surface constructed wetland handling agricultural runoff for nitrogen removal capabilities found removal rates varied by season.	standard deviation larger than values				spr: 62% sum: 65% fall: 55% win: 56%
Luederitz et al.	2001	Several different types of wetlands, horizontal flow wetland, a sloped horizontal flow wetland, a larger horizontal flow wetland, a stratified vertical flow wetland, and an unstratified vertical flow wetland, were compared and all removed N and P.					48-93% c 79-84% l
Martin	1988	A system of a detention pond with wetlands was able to reduce nutrients and suspended solids. Only wetland removal listed.					5% c

Table 38. Summary of Literature Describing the Effect of Wetlands on Nitrogen Removal, Continued

Authors	*Year*	*Description*	*NO_3*	*$NO_2 + NO_3$*	*TAN*	*TKN*	*TN*
Meuleman et al.	2002	Removal efficiencies of biomass harvesting in a wastewater treatment wetland and a natural wetland indicated that timing matters. Harvesting in September–October would likely have led to 20–25% removal rates for both nitrogen and phosphorus rather than lower winter rates.					9% l
Mitsch et al.	2005	In a model to determine how much wetland would need to be created to remove 40% nitrogen load to the Gulf of Mexico, given an inflow rate of 60 g N/ (m^2 yr), the average wetland reduces nitrate nitrogen concentrations by 45%.	45% c				
Moustafa	1999	The Everglades nutrient removal project was assessed over three years and found to have average retention rates of 82% for TP and 55% for TN.					55% l
Moustafa et al.	1996	A subtropical constructed freshwater wetland was examined, and nutrient loading rates and nutrient retention rates were strongly correlated for TP but not for TN. Also, the analysis indicated that concentration reductions (rather than mass balances) could underestimate mass retention by 50% for TP and 100% for TN.					26% l 6% c
Moustafa et al.	1998	A small constructed freshwater wetland was examined and mean annual removal rates were found for TP and TN.					34% l

Table 38. Summary of Literature Describing the Effect of Wetlands on Nitrogen Removal, Continued

Authors	*Year*	*Description*	*NO_3*	*$NO_2 + NO_3$*	*TAN*	*TKN*	*TN*
Naylor et al.	2003	Researchers attempted to combine plants to remove N and steel slag/limestone to remove P in one wetland and found that it was best to do a two-part wetland (basin 1 planted and basin 2 unplanted with p adsorbing substrate), as the high *p*H of the steel slag/limestone inhibits plant growth.				48-95% l	
Oberts and Osgood	1991	A detention/wetland system including a detention pond followed by six in-line wetlands was found to have high removal rates of nutrients for combined snowmelt and rain events.	24-35% c			28-36% c	27-35% c
Rai et al.	2013	A subsurface flow constructed wetland was investigated for its ability to remove contaminants from on-site sewage at varying retention rates. The listed rates are for 36-hour retention.	84% c				
Raisin et al.	1997	A small, constructed wetland was found to have varying retention rates throughout the year depending on seasonal conditions and hydrological events. In some cases, the wetland served as a source, especially for larger storm events where there were large volumes of water. Over a year, the wetland retained more nitrogen than it exported.					11% l
Rodríguez and Brisson	2015	A comparison of wetlands planted with native and European phragmites indicated that native phragmites showed potential for treatment removal, and they have the potential to outperform the European variety.	79% l				84-97% l

Table 38. Summary of Literature Describing the Effect of Wetlands on Nitrogen Removal, Continued

Authors	*Year*	*Description*	*NO_3*	*$NO_2 + NO_3$*	*TAN*	*TKN*	*TN*
Schulz and Peall	2001	A constructed wetland was installed along a tributary and investigated for its ability to remove agricultural runoff.	70-84% c				
Senzia et al.	2003	The performance of six subsurface flow constructed wetlands that received effluent from primary facultative ponds was investigated. The wetlands were placed immediately after the primary facultative pond and after a string of facultative ponds and a maturation point. The wetlands immediately after the primary facultative pond performed the best. Only wetlands after the primary facultative pond listed.		51% l			56% l
Sim et al.	2008	A series of wetland cells (24), part of a 200-hectare constructed wetland system, were investigated for nutrient removal and compared to a pilot tank system with a common reed (phragmites karka) and Tube Sedge. The wetland cells performed better than the tank experiments. Only wetland field cells listed.	71% l				82% l
Spieles and Mitsch	1999	Two constructed wetlands receiving ambient river water were compared to constructed municipal wastewater treatment wetland. All three struggled with flood events during which they could export as high as 400% the nitrate inflow despite having positive mean reduction rates over the two-year study period.	29-40% l average 50-60% l summer <10% l spring/winter				

Table 38. Summary of Literature Describing the Effect of Wetlands on Nitrogen Removal, Continued

Authors	*Year*	*Description*	*NO_3*	*$NO_2 + NO_3$*	*TAN*	*TKN*	*TN*
Tanner et al.	1995	The effect of influent loading rates on mass removal of N and P from dairy wastewaters was compared in four pairs of planted and unplanted gravel bed wetlands. As retention times increased from 2 to 7 days, the removal of TN and TP increased as well. Planted wetlands performed better than unplanted wetlands.					41-75% l
Vymazal and Kröpfelová	2009	A review of 900 annual means of 300 systems broke systems into categories to determine how industrial, municipal, agricultural, and landfill leachate wetlands performed at nitrogen removal.		31% c, d		40% c, d	40% c, d

5.8 Impact of Wetlands on Phosphorus Removal

Table 39 summarizes 47 studies on the effect of wetlands on total phosphorus (TP) removal. These studies indicated phosphorus removal efficiencies varied widely (Figure 69), with some studies showing that the wetlands served as a source of phosphorus, increasing the concentration or load by 50% or more from the input (Carleton et al., 2001; Hoffmann et al., 2012) and some studies showing removal efficiencies greater than 90% (Adler et al., 1996; Al-Rubaei et al., 2016; Andersson et al., 2005; Babatunde et al., 2011; Hey et al., 1994; Luederitz et al., 2001; Mitsch et al., 2005; Rodríguez and Brisson, 2015). Although several studies examined potential factors that may predict or influence phosphorus removal efficiencies, results were largely inconclusive and sometimes contradictory. For instance, one study indicated that TP removal was linear with the loading rate (Dunne et al., 2012), and another study found that regardless of loading rate, four experimental wetlands usually ended up having the same outlet concentration (Hey et al., 1994). Similarly, one study found that using concentration rather than load could underestimate TP removal by 50% (Moustafa et al., 1996), but of the seven studies that listed both concentration and load metrics, four did have load removals higher than concentration removals (Al-Rubaei et al., 2014; Lenhart and Hunt, 2011; Luederitz et al., 2001; Moustafa et al., 1996), two had higher concentration than load removals (Carleton et al., 2000; Fink and Mitsch, 2004), and one study had a range of concentration that includes values above and below the load (Kovacic et al., 2006). One study indicated that placing wetlands in a series is not an effective means of increasing TP removal as the removal efficiency significantly dropped between wetlands (Hathaway and Hunt, 2010). Another study indicated that TP removal may increase over time (Al-Rubaei et al., 2016).

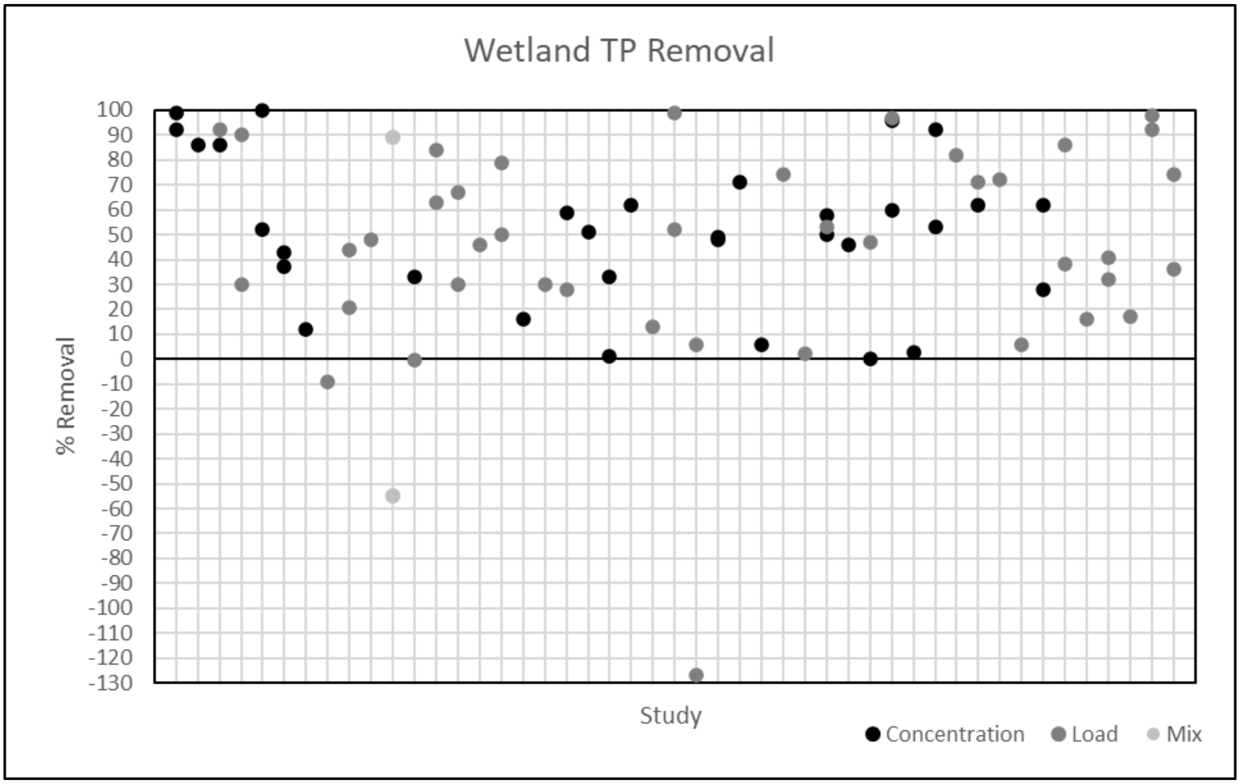


Figure 69. Scatterplot of wetland TP removals. Each vertical line represents one of the studies listed in Table 39. The points represent the percent removal using concentration, load, or mix of concentration and load methods. Mean and median values are not differentiated in this figure. Multiple points on a single vertical line represent multiple values reported by the study. Multiple points of the same color on the same line indicate that the study provided a range, and the points represent the upper and lower bound of the range.

Table 39. Summary of Literature Describing the Effect of Wetlands on Phosphorus Removal

[Total phosphorus (TP) removal is calculated by load (mass) or by concentration, denoted with an "l" or "c", respectively. For studies that indicated their calculations were mean or median percentages, the removals are marked with an "m" or "d", respectively. Negative percentages indicate that the wetland served as a source of phosphorus. Ranges are indicated with a dash in between the values EXCEPT when both numbers in the range are negative, between which the word "to" is used. The words "not sig" is an abbreviation for not significant.]

Authors	*Year*	*Description*	*TP Removal*
Adler et al.	1996	To optimize phosphorus removal, phosphorus was made the limiting nutrient, and plant biomass was harvested regularly in constructed wetlands.	92-99% c
Al-Rubaei et al.	2014	A 19-year-old constructed wetland and pond with no maintenance performed since construction still recorded nutrient and metal concentration reductions.	86% c
Al-Rubaei et al.	2016	A constructed wetland and pond with no maintenance performed still reduced nutrient concentrations after 19 years without maintenance. Compared to its removal rates at 3 and 9 years old, it performed more efficiently and stably.	89% c 92% l
Andersson et al.	2005	The performance of four large surface wetlands (20–28 hectares) removed both total phosphorus and total nitrogen.	30-90% l
Babatunde et al.	2011	A wetland constructed using waste product from drinking water production (dewatered alum sludge) removed nutrients and became more efficient over time.	52-100% c
Beutel et al.	2014	A constructed treatment wetland with a desedimentation basin and two surface flow wetlands removed phosphorus. Muskrats reduce nutrient removal efficiency. Only wetland removal listed.	37-43% c, m
Birch et al.	2004	A wetland that drains a residential urban catchment reduced metal and nutrient concentrations but still did not meet water quality standards for boating.	12% c, m
Borden	2001	Two wet detention ponds and one pond wetland system had varying levels of success for nutrient removal. Only wetland system listed.	(-9%) c
Braskerud	2002	Four cold-temperate-climate surface flow constructed wetlands retained TP.	21-44% l, m
Brown	1984	An urban wetland was less effective at removing dissolved pollutants as sedimentation was the key process of removal, but it still removed some nutrients.	48% l, m

Table 39. Summary of Literature Describing the Effect of Wetlands on Phosphorus Removal, Continued

Authors	*Year*	*Description*	*TP Removal*
Carleton et al.	2001	An analysis of 35 wetland studies found removal rates varied widely.	(-55%)-89% mix c, l
Carleton et al.	2000	A constructed wetland treating stormwater runoff was able to remove nutrients and metals.	33% c, d (-0.3%) l, d
Chen et al.	2015	Some of the largest stormwater treatment areas in the world (constructed wetlands) retained TP. As TP load increases, so does TP export, and to maintain consistent results, hydraulic loading rate, inflow TP rate, and phosphorus loading rate should be kept within optimal ranges.	63-84% l
Coveney et al.	2002	In a wetland constructed to reduce nutrients in a eutrophic lake, particulate matter was reduced by 90% but soluble inorganic compounds increased (though levels were low).	30-67% l
DeBusk et al.	2004	A mesocosm study on wetlands with a limestone bed found TP removal.	46% l
Dierberg et al.	2005	Submerged aquatic vegetation wetlands reduced phosphorus levels at increasing levels as hydraulic residence time increased.	50-79% l
Dunne et al.	2012	A constructed wetland adjacent to a eutrophic lake reduced TP linearly with loading rates.	30% l, d
Dunne et al.	2015	A constructed treatment wetland receiving eutrophic lake water reduced phosphorus levels in large part due to sedimentation.	16% c, m
Fink and Mitsch	2004	A 1.2 hectare created/restored wetland receiving groundwater flows and stormwater flows exported more phosphorus in precipitation events than dry weather flows, but there was no significant increase in nitrate-nitrite exports.	59% c, m 28% l, m
Gessner et al.	2005	A wetland examined for its ability to reduce cyanide and hydrocarbons was also found to reduce nutrients.	51% c
Guerrero et al.	2020	In a comparison between two regional detention facilities with wetlands, the larger wetland had lower outlet pollutant loads for NOx, TN, and TP.	1-9% c, m 17-33% c, d
Hathaway and Hunt	2010	In a series of three wetlands, the first wetland removed at least 80% of the total concentration for all pollutants, and no pollutant was significantly reduced from the outlet of wetland 2 to the outlet of wetland 3. Removals listed in order of wetlands.	62% not sig not sig c

Table 39. Summary of Literature Describing the Effect of Wetlands on Phosphorus Removal, Continued

Authors	*Year*	*Description*	*TP Removal*
Healy and Cawley	2002	A recently constructed surface flow wetland was investigated as a potential tertiary treatment option, and while it performed well at N reduction, it was less effective at P reduction.	13% l
Hey et al.	1994	Four experimental wetlands were examined for percent removals, and researchers found that the wetlands all had similar outlet concentrations despite different loading rates.	52-99% l
Hoffmann et al.	2012	Two restored wetlands that received drainage water from agricultural fields rich in nitrate were monitored five years later and found to perform well at nitrogen removal, but not phosphorus removal.	(-127%) to (-6%) l
Kadlec	2003	In an examination of 21 wastewater treatment systems that use wetlands, the median removals were found for NH4-N, TSS, and TP. Phosphorus showed a seasonal, but not temperature effect.	48% c, d 49% c, m
Kadlec	2016	In an examination of studies of > 40-hectare wetlands focused on phosphorus removal the median removal concentration was determined.	71% c, d
Kadlec et al.	2010	Due to high loading, four free surface wetlands did not reduce nutrient concentrations by a large degree but did remove 98 t/yr of nitrogen and 3.6 t/yr of phosphorus.	6% c
Kohler et al.	2004	A four-year study on golf course wetlands indicated the wetlands were able to efficiently remove N-NO3/NO2, N-NH3, and P. Pesticide was only detected once, and it was a type not used on golf courses.	74% l
Kovacic et al.	2000	Three treatment wetlands receiving tile drainage in addition to runoff were effective at reducing NO3-N, but less so for TP. Adding a buffer between wetlands and river increased removal efficiency.	2% l
Kovacic et al.	2006	Two runoff wetlands handling agricultural runoff were able to reduce both P and nitrate nitrogen.	53% l 50-58% c
Land et al.	2016	93 articles on created or restored freshwater wetlands were reviewed and median removal rates for TN and TP were found.	46% c, d
Lenhart and Hunt	2011	Four different metrics of performance were examined for a constructed wetland. Concentrations of major nutrients increased but loads of nutrients decreased.	0% c, m 47% l, m
Luederitz et al.	2001	Several different types of wetlands, horizontal flow wetland, a sloped horizontal flow wetland, a larger horizontal flow wetland, a stratified vertical flow wetland, and a unstratified vertical flow wetland, were compared and all removed N and P.	60-96% c 97% l

Table 39. Summary of Literature Describing the Effect of Wetlands on Phosphorus Removal, Continued

Authors	*Year*	*Description*	*TP Removal*
Martin	1988	A system of a detention pond with wetlands was able to reduce nutrients and suspended solids. Only wetland removal listed.	3% c
Meuleman et al.	2002	Removal efficiencies of biomass harvesting in a wastewater treatment wetland and a natural wetland indicated that timing matters. Harvesting in September–October would likely have led to 20-25% removal rates for nitrogen and phosphorus rather than lower winter rates.	6% l
Mitsch et al.	1995	In a comparison of four constructed freshwater riparian marshes, phosphorus was removed in low and high flow conditions.	53-90% c low flow 64-92% c high flow
Moustafa	1999	The Everglades nutrient removal project was assessed over three years and average retention rates were found for TN and TP.	82% l
Moustafa et al.	1996	A subtropical constructed freshwater wetland had nutrient loading rates and nutrient retention rates that were strongly correlated for TP but not for TN. The analysis indicated that concentration reductions (rather than mass balances) could underestimate mass retention by 50% for TP and 100% for TN.	71% l 62% c
Moustafa et al.	1998	A small constructed freshwater wetland was examined and mean annual removal rates were found for TP and TN.	72% l
Nairn and Mitsch	1999	In two wetland ponds, one planted and one not planted with vegetation, phosphorus removal was related to decreases in turbidity and level of biological activity.	58-62% c
Naylor et al.	2003	Results from combining plants to remove N and steel slag/limestone to remove P in one wetland indicate more effectiveness in doing a two-part wetland (basin 1 planted and basin 2 unplanted with p adsorbing substrate) as the high *p*H of the steel slag/limestone inhibits plant growth.	38- > 86% l
Niswander and Mitsch	1995	A two-year-old constructed riparian wetland was observed and used to create a model to simulate the hydrology, phosphorus retention, and tree growth.	16% l
Oberts and Osgood	1991	A detention/wetland system including a detention pond followed by six in line wetlands was found to have high removal rates of nutrients for combined snowmelt and rain events.	32-41% c
Raisin et al.	1997	A small, constructed wetland was found to have varying retention rates. In some cases, the wetland served as a source, especially for larger storm events where there were large volumes of water.	17% l

Table 39. Summary of Literature Describing the Effect of Wetlands on Phosphorus Removal, Continued

Authors	*Year*	*Description*	*TP Removal*
Rodríguez and Brisson	2015	A comparison of wetlands planted with native and European phragmites indicated that native phragmites showed potential for treatment removal and that it has the potential to outperform the European variety.	92-98% l
Tanner et al.	1995	The effect of influent loading rates on mass removal of N and P from dairy wastewaters was compared in four pairs of planted and unplanted gravel bed wetlands. As retention times increased from two to seven days, the removal of TN and TP increased as well. Planted wetlands performed better than unplanted wetlands.	36-74% l

5.9 Impact of Detention and Retention on Nitrogen Removal

Table 40 summarizes 12 studies on the effect of detention on nitrogen removal. In this instance, a detention pond refers to a method of stormwater control where water is temporarily stored with water draining from the detention pond in between storm events. A detention pond is synonymous with a dry retention pond and dry detention pond. There are five categories of nitrogen removal included, nitrates (NO_3), nitrates and nitrites (NO_3 + NO_2), total ammonia nitrogen (TAN=NH_3 + NH_4^+), total Kjeldahl nitrogen (TKN), and total nitrogen (TN). There appears to be a wide variety of removal rates across nitrogen species as well as within nitrogen species. TAN (TAN=NH_3 + NH_4) is the least reported of the five species within the studies examined with no studies reporting TAN removal for detention, though a number of studies do report NH_3 or NH_4^+ values. Figure 70 to Figure 73 visually depict the spread of the nitrogen removal rates reported in the studies. Of the studies that report both load and concentration-based metrics for a nitrogen species, the load tends to report higher removal rates than concentration (Harper et al., 1999; House et al., 1993; Wissler et al., 2020a), but sometimes that does not hold up for all species of nitrogen (House et al., 1993). One reason for the discrepancy between load and concentration is the loss of water through infiltration which reduces both the mass of the nitrogen species and water exiting the pond through the outlet (Harper et al., 1999).

Of the nitrogen species, only TKN did not serve as a source of nitrogen in any of the studies listed below that reported it. TKN ranged from 12 to 78% removal. In contrast, NO_3, NO_2 + NO_3, and TN all had at least one instance of negative removal rates. They ranged from (-116%) -98%, (-46%) -91%, and (-10%) -86%, respectively. Of note, the nitrogen removal rate appears to be seasonal (Rosenzweig et al., 2011). On the effects of aging detention basins, one study looked at two unmaintained detention basins and found that they were still effective at removing NO_2 + NO_3 and TKN (Wissler et al., 2020b); however, another study found that over time, the detention basin lost storage as sedimentation occurred (Stanley, 1996).

In Figure 70 to Figure 73, each vertical line represents one of the studies listed in Table 40. The points represent the percent removal using concentration or load methods. Mean and median values are not differentiated in these figures. Multiple points on a single vertical line represent multiple values reported by a study. Multiple points of the same color on the same line indicate that the study provided a range, and the points represent the upper and lower bound of the range.

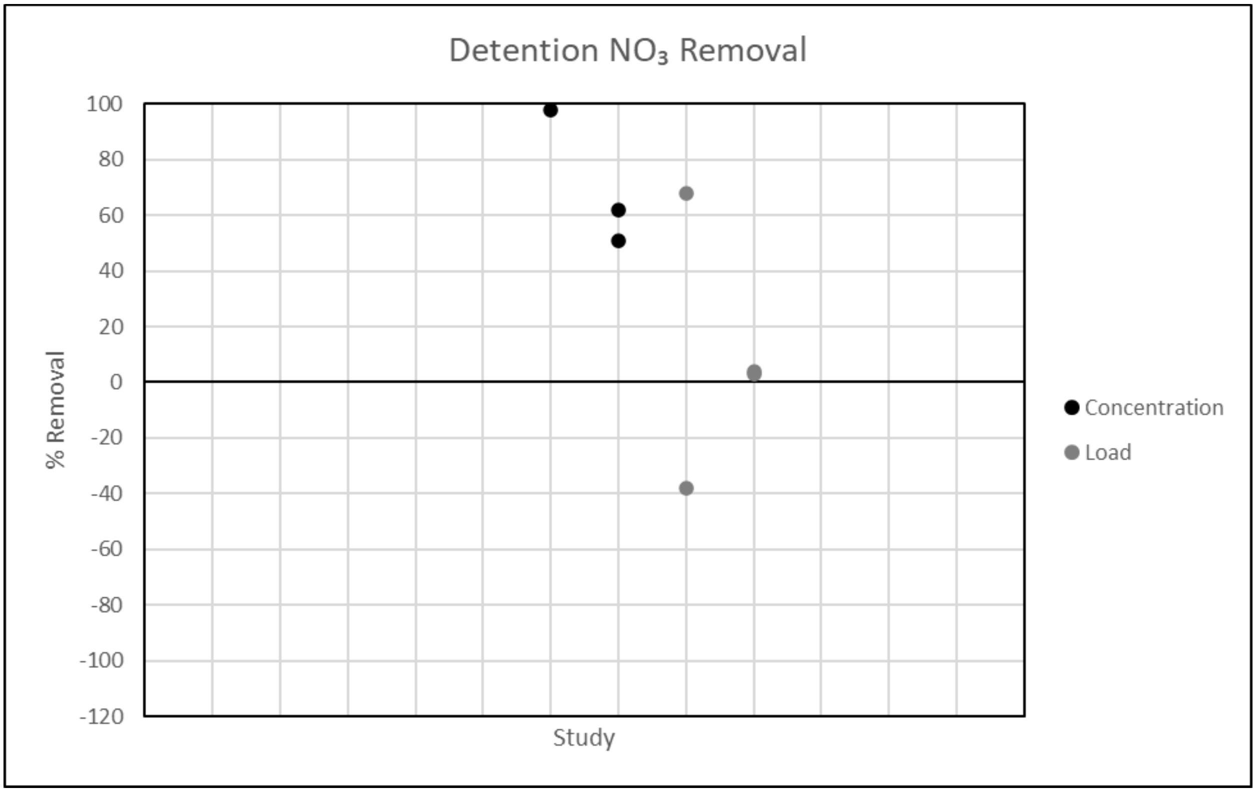


Figure 70. Scatterplot of detention pond nitrate removals

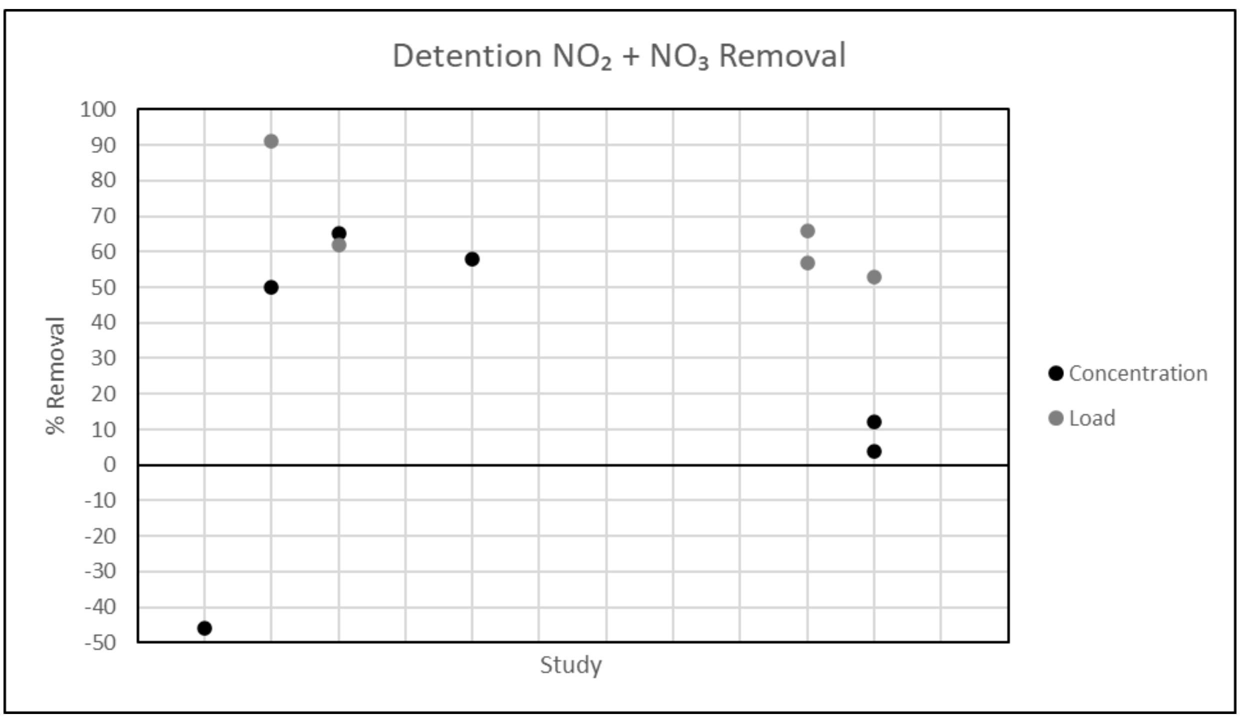


Figure 71. Scatterplot of detention pond nitrite and nitrate removals

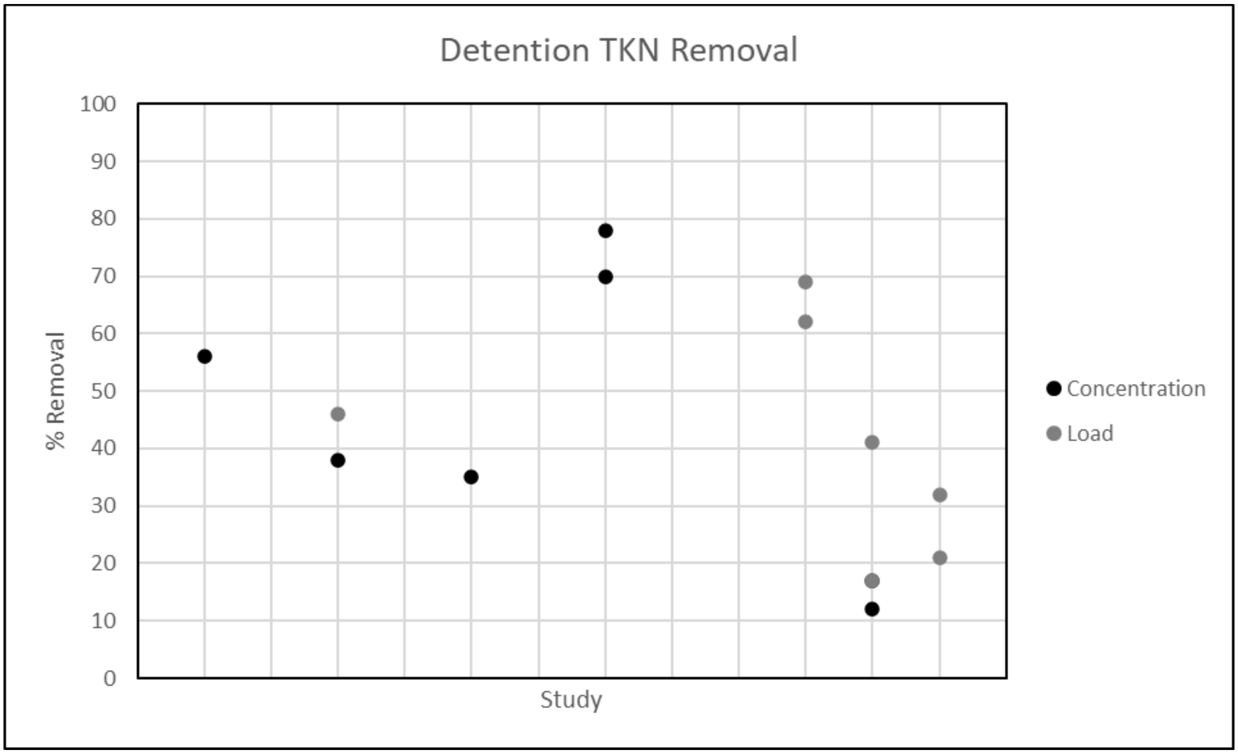


Figure 72. Scatterplot of detention pond TKN removals

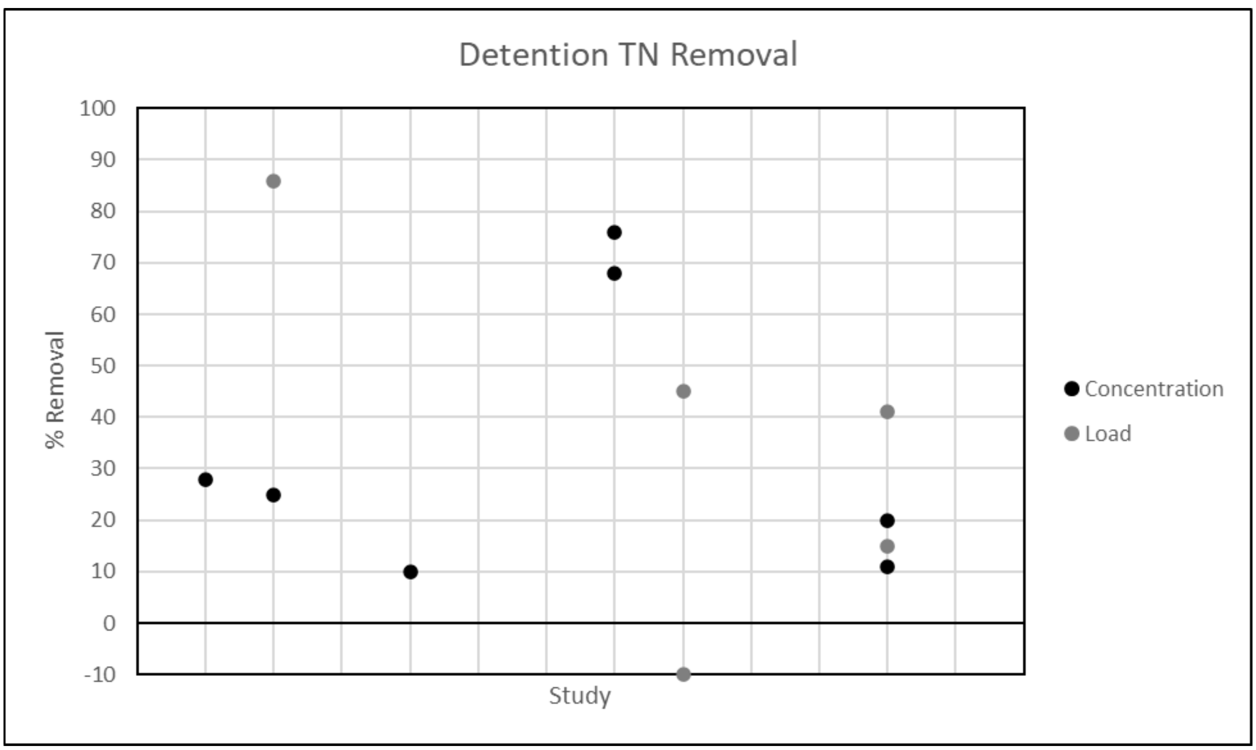


Figure 73. Scatterplot of detention pond total nitrogen removals

Table 40. Summary of Literature Describing the Effect of Detention on Nitrogen Removal

[There are five categories of nitrogen removal listed- nitrates (NO_3), nitrates + nitrites ($NO_2 + NO_3$), total ammonia nitrogen ($TAN=NH_3 + NH_4^+$), total Kjeldahl nitrogen (TKN), and total nitrogen (TN). Nitrogen removal is calculated by load (mass) or by concentration, denoted with an "l" or "c", respectively. For studies that indicated their calculations were mean or median percentages, the removals are marked with an "m" or "d", respectively. Negative percentages indicate that the wetland served as a source of nitrogen and are enclosed in parenthesis. Ranges are indicated with a dash in between the values EXCEPT when both numbers are negative, which substitutes the word "to". The words "not sig" are an abbreviation for not significant and indicate the study reported results that were not significant.]

Author	*Year*	*Description*	*TAN*	*NO_3*	*$NO_2 + NO_3$*	*TKN*	*TN*
Birch et al.	2006	The nutrient removal performance of a detention pond alongside a motorway was variable. The means disguise a wide range of values.			(-46%) c, m ((-85)-1%)	56% c, m (26-82%)	28% c, m ((-1)-59%)
Harper et al.	1999	A detention pond had higher mass removal efficiencies due to groundwater seepage than concentration removal efficiencies.			50% c 91% l		25% c 86% l
House et al.	1993	An urban stormwater detention pond near a lake was able to decrease both loads and concentrations.			65% c, d 62% l	38% c, d 46% l	
Martin	1988	A system of a detention pond with wetlands was able to reduce nutrients and suspended solids. Only detention removal listed.					10% c
Middleton and Barrett	2008	By attaching an automated valve/controller to an extended detention basin, the overall pollutant removal efficiency was improved. This created a batch type detention basin that increased the residence time.			58% c	35% c	
Morse et al.	2017	One detention and one retention basin were monitored for stormwater quality, and the dry basin performed better at nitrogen removal. Only detention removal listed.		98% c			

Table 40. Summary of Literature Describing the Effect of Detention on Nitrogen Removal, Continued

Author	*Year*	*Description*	*TAN*	*NO_3*	*$NO_2 + NO_3$*	*TKN*	*TN*
Oberts and Osgood	1991	A detention/wetland system including a detention pond followed by six in line wetlands was found to have high removal rates of nutrients for combined snowmelt and rain events.		51-62% c		70-78% c	68-76% c
Rosenzweig et al.	2011	A stormwater detention pond had seasonal variation of nitrogen influent and retention.		(-38)-68% l Dec: (-38%) July: 68%			(-10)-45% l Dec: (-10%) May: 45%
Stanley	1996	A detention pond was monitored and found to remove pollutants, but in almost all storms its capacity was exceeded, and it also lost 0.16% storage each year from sedimentation.		4% l, m 3% l, d			
Wissler et al.	2020a	Two maintained detention basins were examined for performance and found to reduce pollutants, but not enough for water to be safe given the water quality.			4-12% c, d 53% l, d	12-17% c, d 17-41% l, d	11-20% c, d 15-41% l, d
Wissler et al.	2020b	Two unmaintained detention basins were examined for performance and found to reduce pollutants despite no maintenance.			57-66% l, d	62-69% l, d	
Wu Wu et al.	1989 1996	Three urban detention ponds not originally designed for water quality were examined for water quality, and it was found that 1-2% of watershed area should be used for siting to maintain 70% removal of sediment. Presence of geese increases pollutant levels of ponds.				21-32% l	

Table 41 summarizes eight studies on the effect of retention on nitrogen removal. In this instance, a retention pond refers to a method of stormwater control where water is permanently stored within the pond and does not fully drain in between storm events. A retention pond is synonymous with a wet retention pond and wet detention pond. There are five categories of nitrogen removal included, nitrates (NO_3), nitrates and nitrites ($NO_3 + NO_2$), total ammonia nitrogen ($TAN=NH_3 + NH_4^+$), total Kjeldahl nitrogen (TKN), and total nitrogen (TN). Figure 74 to Figure 78 depict the nitrogen removal rates visually. For the retention pond studies examined that reported nitrogen removals, the most reported nitrogen species was TN. In studies where there was a comparison between retention ponds, it was found that younger ponds performed better at removing nitrogen (Sønderup et al., 2016), sand filters improved nitrogen retention (Sønderup et al., 2016), and retention ponds with high length-to-width ratios also performed better at nitrogen removal (Mallin et al., 2002). The removal rates often do not provide a full picture of the variation across the season (Chrétien et al., 2016).

In Figure 74 to Figure 78, each vertical line represents one of the studies listed in Table 41. The points represent the percent removal using concentration or load methods. Mean and median values are not differentiated in these figures. Multiple points on a single vertical line represent multiple values reported by a study. Multiple points of the same color on the same line indicate that the study provided a range, and the points represent the upper and lower bound of the range.

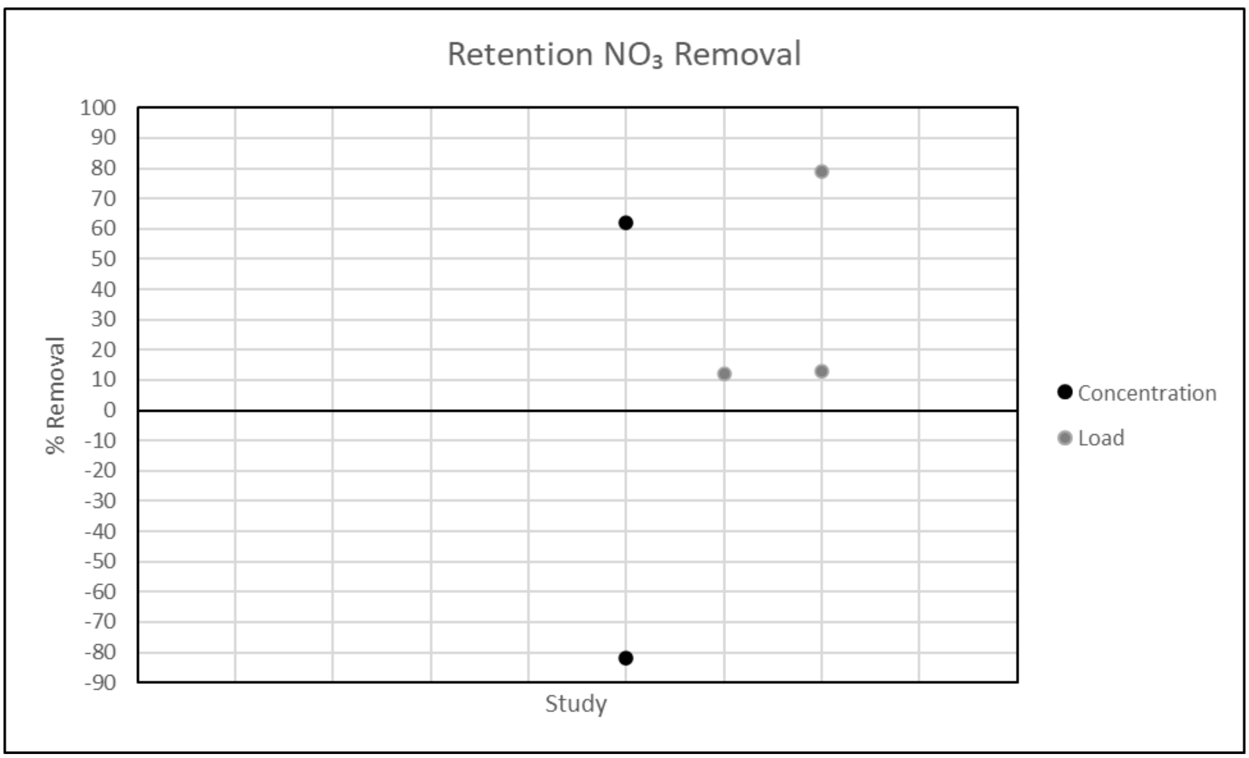


Figure 74. Scatterplot of retention pond nitrate removals

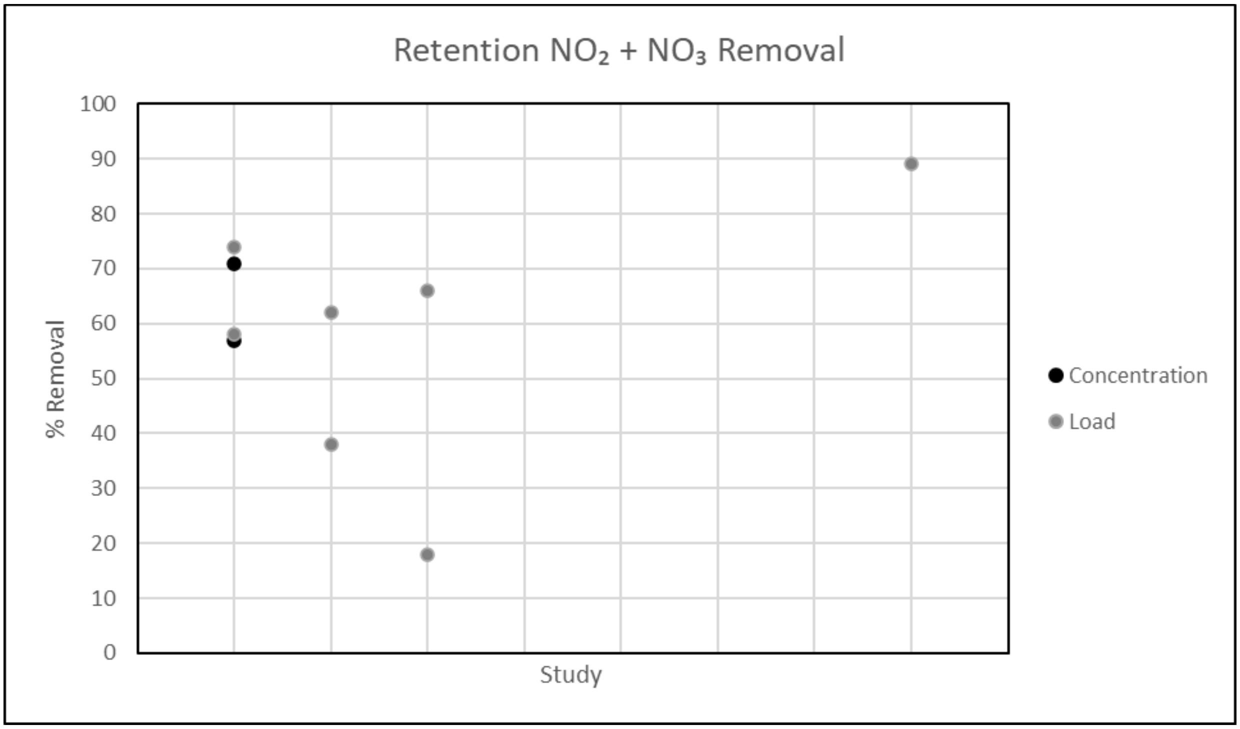


Figure 75. Scatterplot of retention pond nitrite and nitrate removals

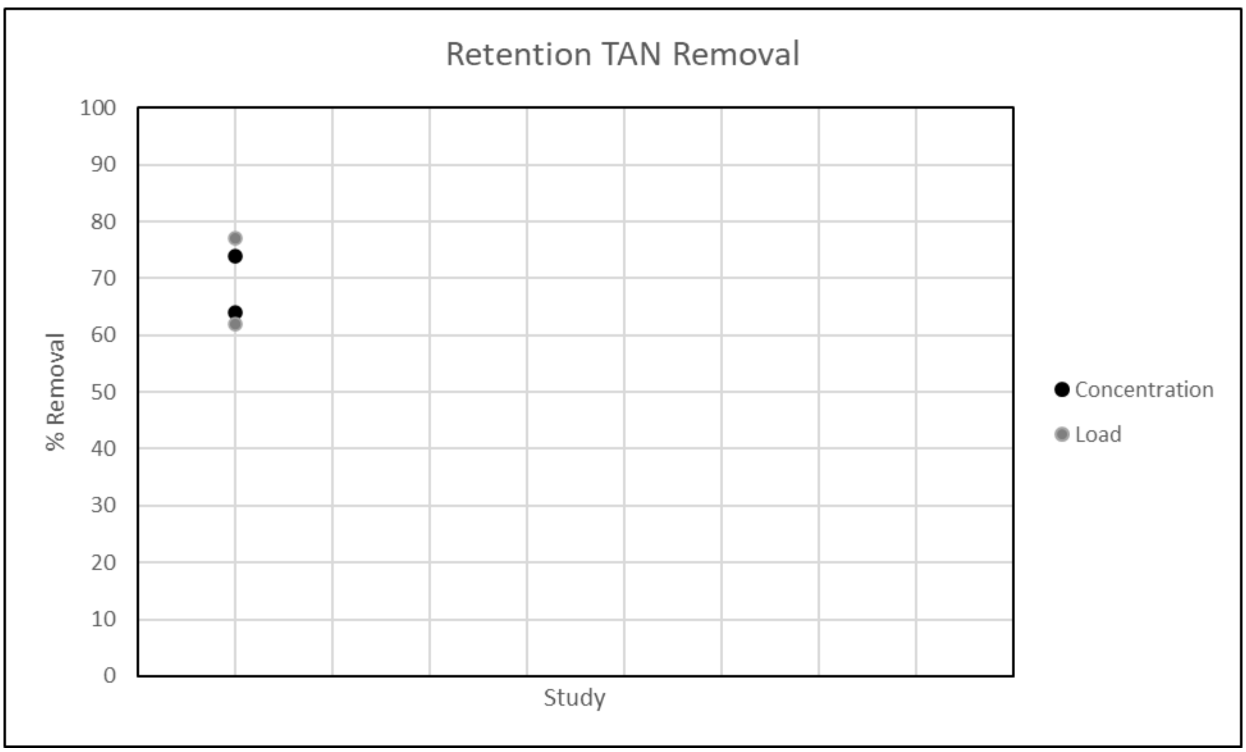


Figure 76. Scatterplot of retention pond TAN removals

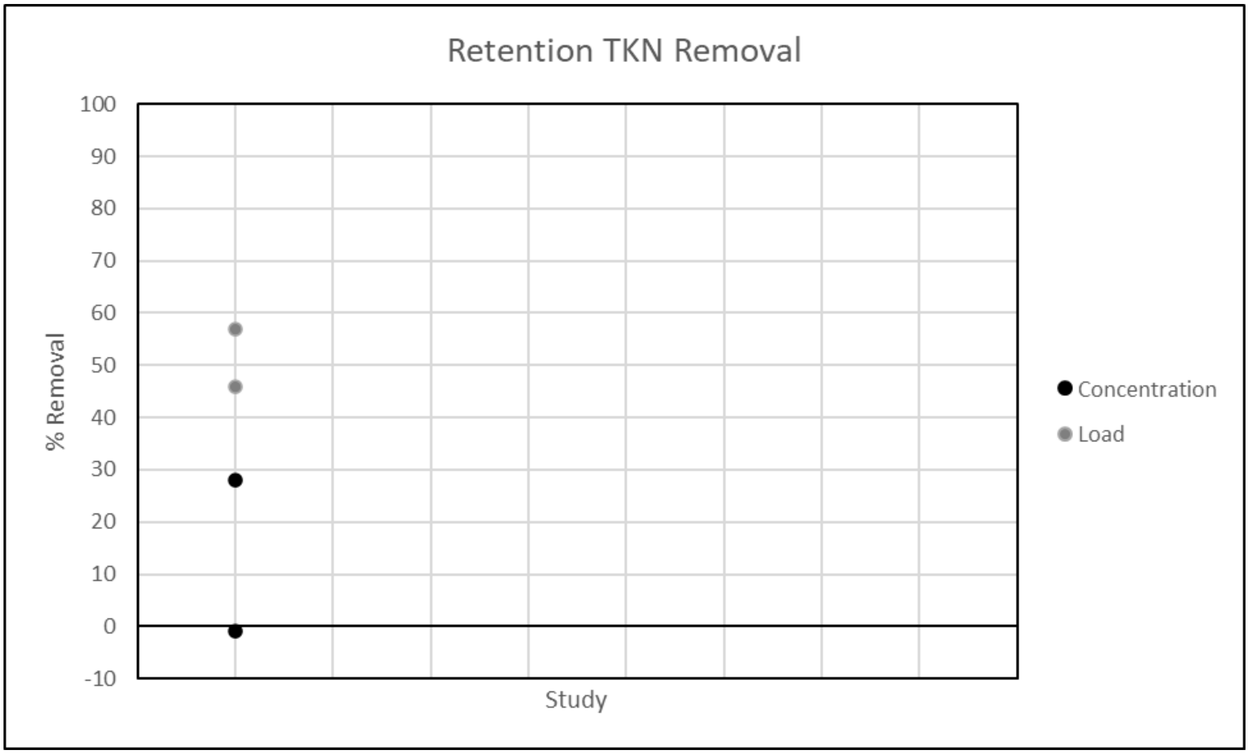


Figure 77. Scatterplot of retention pond TKN removals

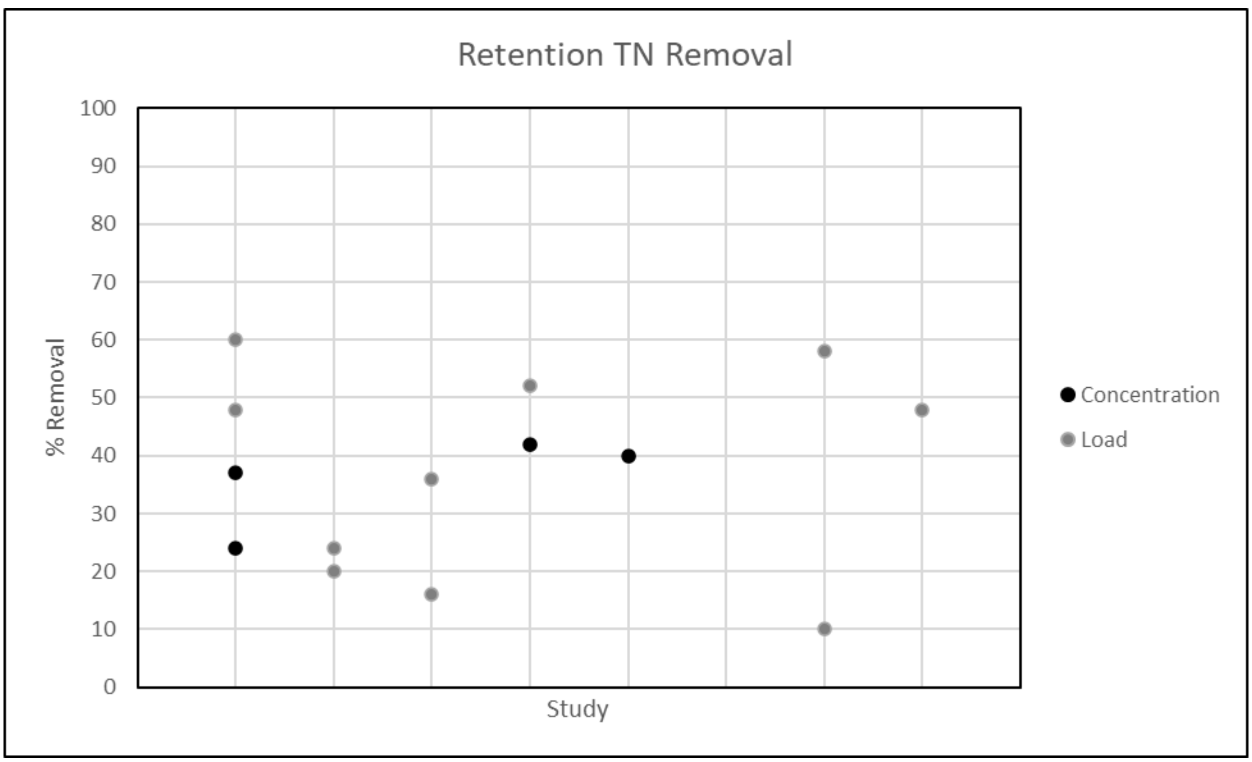


Figure 78. Scatterplot of retention pond TN removals

Table 41. Summary of Literature Describing the Effect of Retention on Nitrogen Removal

[There are five categories of nitrogen removal listed nitrates (NO_3), nitrates + nitrites ($NO_2 + NO_3$), total ammonia nitrogen ($TAN=NH_3 + NH_4^+$), total Kjeldahl nitrogen (TKN), and total nitrogen (TN). Nitrogen removal is calculated by load (mass) or by concentration, denoted with an "l" or "c", respectively. For studies that indicated their calculations were mean or median percentages, the removals are marked with an "m" or "d", respectively. Negative percentages indicate that the retention/detention facility served as a source of phosphorus. Ranges are indicated with a dash in between the values EXCEPT when both numbers are negative which substitutes the word "to". The words "not sig" are an abbreviation for not significant and indicate the study reported results that were not significant.]

Author	*Year*	*Description*	*TAN*	*NO_3*	*$NO_2 + NO_3$*	*TKN*	*TN*
Baird et al.	2020	Two infiltrating wet retention ponds designed to allow for both infiltration and retention were tested and found to reduce annual pollutant loads.	64-73% c, m 65-74% c, d 62-77% l		57-66% c, m 58-71% c, d 58-74% l	not sig c, m -1-28% c, d 46-57% l	27% c, m 24-37% c, d 48-60% l
Borden	2001	Two wet detention ponds and one pond wetland system had varying levels of success for nutrient removal. Only detention pond removals listed.			38-62% l		20-24% l
Borden et al.	1997	Two retention ponds with different influent pollutant concentrations were examined for removal efficiency.			18-66% l		16-36% l
Chrétien et al.	2016	A retention pond receiving runoff from agriculture was examined for pollutant removal efficiency. The mean removal efficiencies do not show the wide range of variation over the study period.					42% c 52% l

Table 41. Summary of Literature Describing the Effect of Retention on Nitrogen Removal, Continued

Author	*Year*	*Description*	*TAN*	*NO_3*	*$NO_2 + NO_3$*	*TKN*	*TN*
Mallin et al.	2002	Three "wet detention ponds" (retention ponds) were examined for performance, and generally, the most successful pond had a high length-to-width ratio and lots of native macrophyte species.		Not sig c 62% c -82% c			not sig c 40% c not sig c
Morse et al.	2017	One detention and one retention basin were monitored for stormwater quality, and the dry basin performed better at nitrogen removal. Only retention pond removal listed.		12% l			
Sønderup et al.	2016	Retention ponds combined with sand filters had higher retention rates than wet ponds alone, and young retention ponds had higher retention than older ones.		12.7% l, d no filter 78.5% l, d sand filter			9.5% l, d no filter 57.6% l, d sand filter
Stormwater Academy	2010	A wet detention pond (retention) was designed, constructed, and monitored for water quality improvement.			89% l		48% l

5.10 Impact of Detention and Retention on Phosphorus Removal

Table 42 summarizes 11 studies on the effect of detention on total phosphorus (TP) removal. In this instance, a detention pond refers to a method of stormwater control where water is temporarily stored with water draining from the detention pond in between storm events. A detention pond is synonymous with a dry retention pond and dry detention pond. For those studies that reported both concentration and load-based removal rates, load removal tended to be higher (Harper et al., 1999; House et al., 1993; Hussain et al., 2005; Wissler et al., 2020a). Most studies listed positive removal rates. Only 1 of the 11 studies found that a detention pond served as a source of TP (Birch et al., 2006). The removal rates ranged from -5 to 84% (Figure 79).

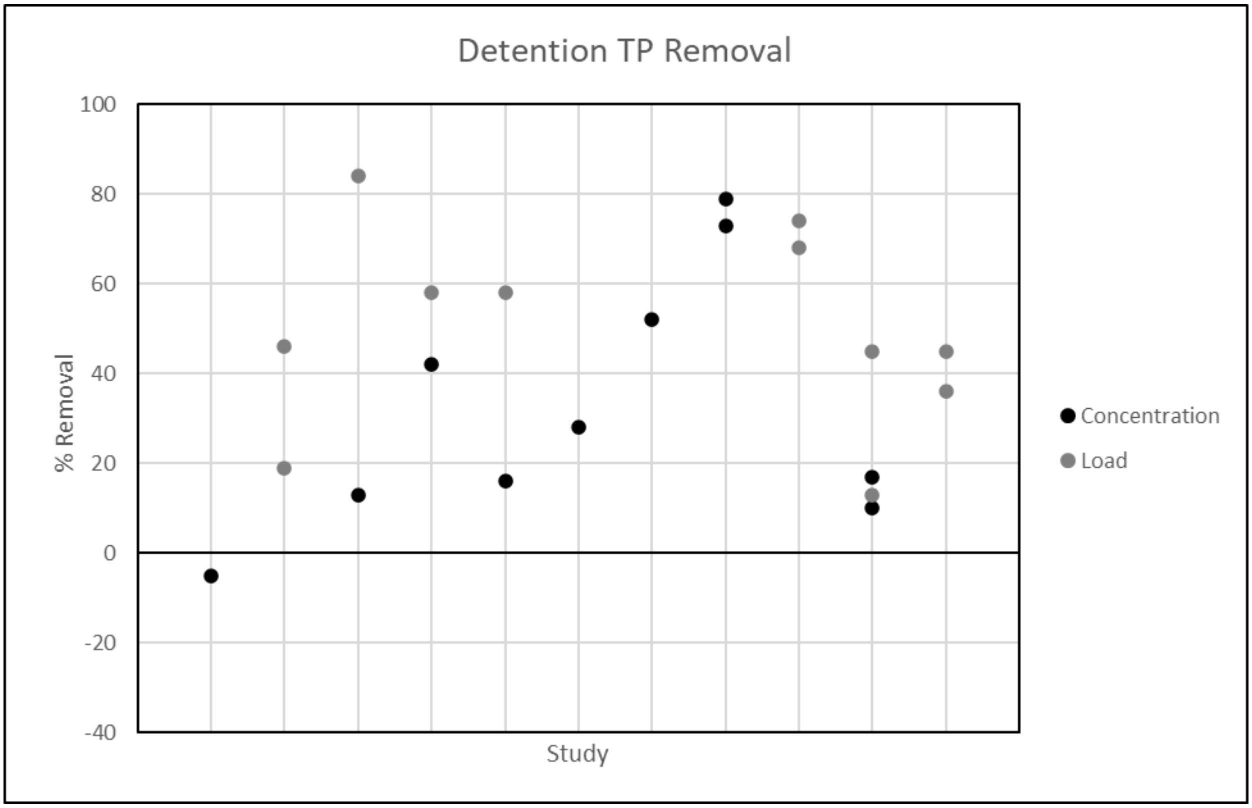


Figure 79. Scatterplot of detention pond TP removals. Each vertical line represents one of the studies listed in Table 42. The points represent the percent removal using concentration or load methods. Mean and median values are not differentiated in this figure. Multiple points on a single vertical line represent multiple values reported by the study. Multiple points of the same color on the same line indicate that the study provided a range, and the points represent the upper and lower bound of the range.

Table 42. Summary of Literature Describing the Effect of Detention on Phosphorus Removal

[Total phosphorus (TP) removal is calculated by load (mass) or by concentration, denoted with an “l” or “c”, respectively. For studies that indicated their calculations were mean or median percentages, the removals are marked with an “m” or “d”, respectively. Negative percentages indicate that the retention/detention facility served as a source of phosphorus. Ranges are indicated with a dash in between the values EXCEPT when both numbers in the range are negative, between which the word “to” is used. The words “not sig” are an abbreviation for not significant and indicate the study reported results that were not significant.]

Author	*Year*	*Description*	*TP*
Birch et al.	2006	The nutrient removal performance of a detention pond alongside a motorway was variable. The mean removal efficiency does not show the wide range of values (-61) -76%.	-5% c, m
Comings et al.	2000	Two "wet detention ponds" (retention ponds) were investigated for performance and the one that was newer, focused on water quality, and had a longer detention time performed better.	20% l 46% l
Harper et al.	1999	A detention pond had higher mass removal efficiencies due to groundwater seepage than concentration removal efficiencies.	13% c 84% l
House et al.	1993	An urban stormwater detention pond near a lake was able to decrease both loads and concentrations.	42% c, d 58% l
Hussain et al.	2005	Detention ponds with under-drains demonstrated pollutant removal capabilities for both load and concentration.	16% c 58% l
Martin	1988	A system of a detention pond with wetlands was able to reduce nutrients and suspended solids. Only wetland removal listed.	28% c
Middleton and Barrett	2008	By attaching an automated valve/controller to an extended detention basin, the overall pollutant removal efficiency was improved. This created a batch type detention basin that increased the residence time.	52% c
Oberts and Osgood	1991	A detention/wetland system including a detention pond followed by six in line wetlands was found to have high removal rates of nutrients for combined snowmelt and rain events.	73-79% c
Wissler et al.	2020a	Two unmaintained detention basins were examined for performance and found to reduce pollutants despite no maintenance.	68-74% l, d
Wissler et al.	2020b	Two maintained detention basins were examined for performance and found to reduce pollutants, but not enough for water to be safe.	10-17% c, d 13-45% l, d
Wu Wu et al.	1989 1996	Three urban detention ponds not originally designed for water quality were examined for water quality. It was found that 1-2% of watershed area should be used for siting to maintain 70% removal of sediment. Presence of geese increases pollutant levels of ponds.	36-45% l

Table 43 summarizes eight studies on the effect of retention on total phosphorus (TP) removal. In this instance, a retention pond refers to a method of stormwater control where water is permanently stored within the pond and does not fully drain in between storm events. A retention pond is synonymous with a wet retention pond and wet detention pond. Of the studies that listed both concentration and load-based removal, load removal tended to be higher (Baird et al., 2020; Chen et al., 2009; Chrétien et al., 2016). The removal rates ranged from -35 to 88% (Figure 80). The mean values reported for each study do not necessarily show the possible wide variation of values over the year (Chrétien et al., 2016).

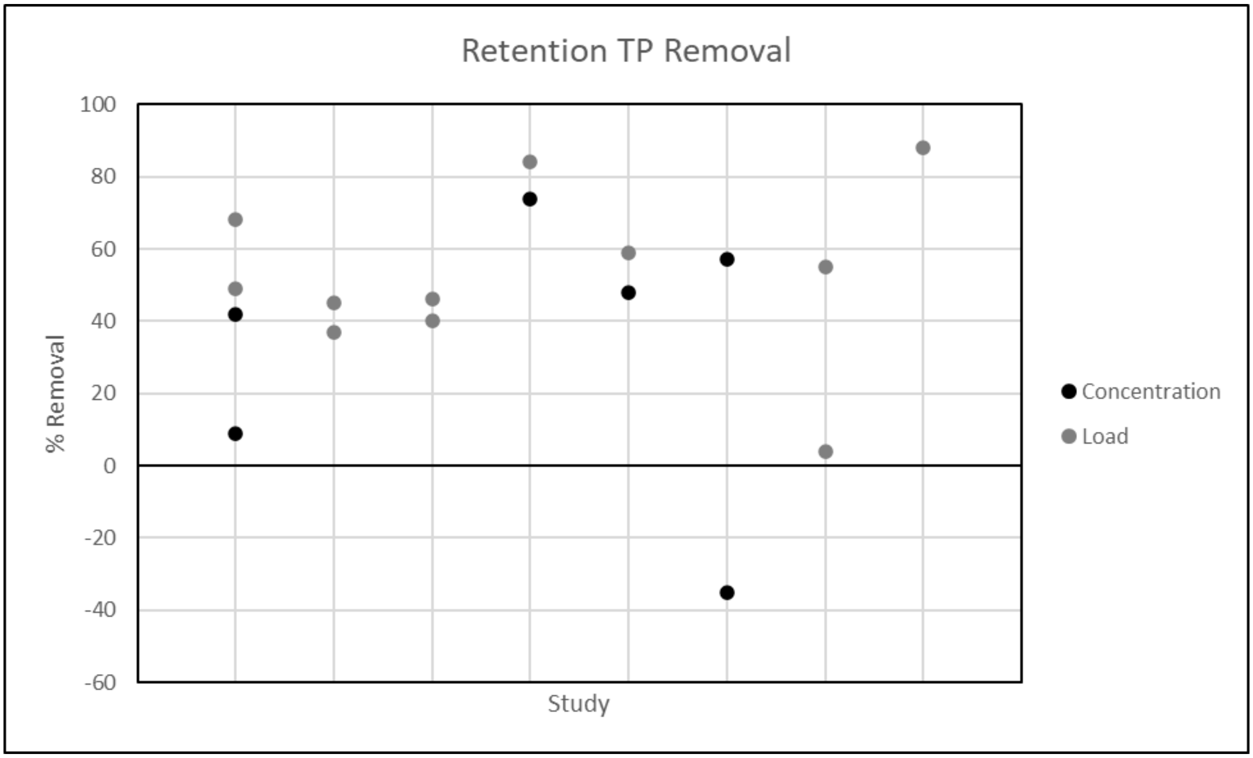


Figure 80. Scatterplot of retention pond TP removals: Each vertical line represents one of the studies listed in Table 43. The points represent the percent removal using concentration or load methods. Mean and median values are not differentiated in this figure. Multiple points on a single vertical line represent multiple values reported by the study. Multiple points of the same color on the same line indicate the study provided a range, and the points represent the upper and lower bound of the range.

Table 43. Summary of Literature Describing the Effect of Retention on Phosphorus Removal

[Total phosphorus (TP) removal is calculated by load (mass) or by concentration, denoted with an "l" or "c", respectively. For studies that indicated their calculations were mean or median percentages, the removals are marked with an "m" or "d", respectively. Negative percentages indicate that the retention/detention facility served as a source of phosphorus. Ranges are indicated with a dash in between the values EXCEPT when both numbers in the range are negative, between which the word "to" is used. The words "not sig" are an abbreviation for not significant and indicate the study reported results that were not significant.]

Author	*Year*	*Description*	*TP*
Baird et al.	2020	Two infiltrating wet retention ponds designed to allow for both infiltration and retention were tested and found to reduce annual pollutant loads.	28-38% c, m 9-42% c, d 49-68% l
Borden	2001	Two wet detention ponds and one pond wetland system had varying levels of success for nutrient removal. Only detention ponds listed.	37-45% l
Borden et al.	1997	Two retention ponds with different influent pollutant concentrations were examined for removal efficiency and neither met design objective of 85% TSS removal.	40-46% l
Chen et al.	2009	A train of treatment ponds (retention pond 1- retention pond 2- eco pond- gravel filter bed- limestone filter bed- vegetative buffer) was examined for pollutant removal efficiency. By calculating removal efficiency using different methods, a range of values was found.	74% c 84% l
Chrétien et al.	2016	A retention pond receiving runoff from agriculture was examined for pollutant removal efficiency. The mean removal efficiencies do not show the wide range of variation over the study period.	48% c 59% l
Mallin et al.	2002	Three "wet detention ponds" (retention ponds) were examined for performance, and generally, the most successful pond had a high length-to-width ratio and lots of native macrophyte species.	not sig c 57% c -35% c
Sønderup et al.	2016	Retention ponds combined with sand filters had higher retention rates than wet ponds alone, and young retention ponds had higher retention than older ones. The retention pond only removal rate is listed first, followed by the retention pond with sand filter removal rate.	3.8% l, d no filter 54.7% l, d sand filter
Stormwater Academy	2010	A wet detention pond (retention) was designed, constructed, and monitored for water quality improvement.	88% l

5.11 Impact of Wetlands on TSS Removal

Table 44 summarizes 29 studies on the effects of wetlands on the removal of total suspended solids. Three studies (marked with an asterisk) used "SS" as their measure, but the abbreviation stood for suspended solids rather than the alternative of settleable solids. For that reason, all measures of suspended solids have been lumped together into the category TSS. Figure 81 shows a visual representation of the removal rates of the studies examined. Most of the studies reported positive removal rates of TSS, but there were five studies that reported negative removal rates. One removal rate was exceptionally negative at (-311)%, but the other negative removal rates were between (-4)% and (-30)% (Birch et al., 2004; Carleton et al., 2001; Hoffmann et al., 2012; Koskiaho, 2003; Lenhart and Hunt, 2011). On the reverse side, 11 studies reported ranges that included 90% removal or higher, including the study that reported a negative removal rate of (-311)% (Al-Rubaei et al., 2016, 2014; Babatunde et al., 2011; Brown, 1984; Carleton et al., 2001; Coveney et al., 2002; Healy and Cawley, 2002; Hey et al., 1994; Naylor et al., 2003; Rodríguez and Brisson, 2015; Senzia et al., 2003). Several studies indicated that TSS removal improved over time (Al-Rubaei et al., 2016, 2014; Babatunde et al., 2011), but there was also a study that found that in later periods, the TSS removal efficiency showed a slight decreasing trend (Dunne et al., 2012). Similarly, for seasonality, one study found that there was a distinct seasonal trend with TSS removal (Kadlec et al., 2010), whereas another found no seasonal trend in its TSS removal efficiency (Kadlec, 2003). Two studies found that despite different loading rates, there were similar outlet concentrations of TSS (Hey et al., 1994; Schulz and Peall, 2001). Other studies found that removal efficiencies varied depending on the type of storm event; extremely high flow events or events that exceeded the capacity of the wetland led to lower removal efficiencies (Birch et al., 2004; Carleton et al., 2000). One study found that placing wetlands in a series did not substantially improve the TSS removal efficiency instead of just having a single wetland. The first wetland had a removal efficiency of 80%, and the following two wetlands had removal efficiencies that were not significant (Hathaway and Hunt, 2010).

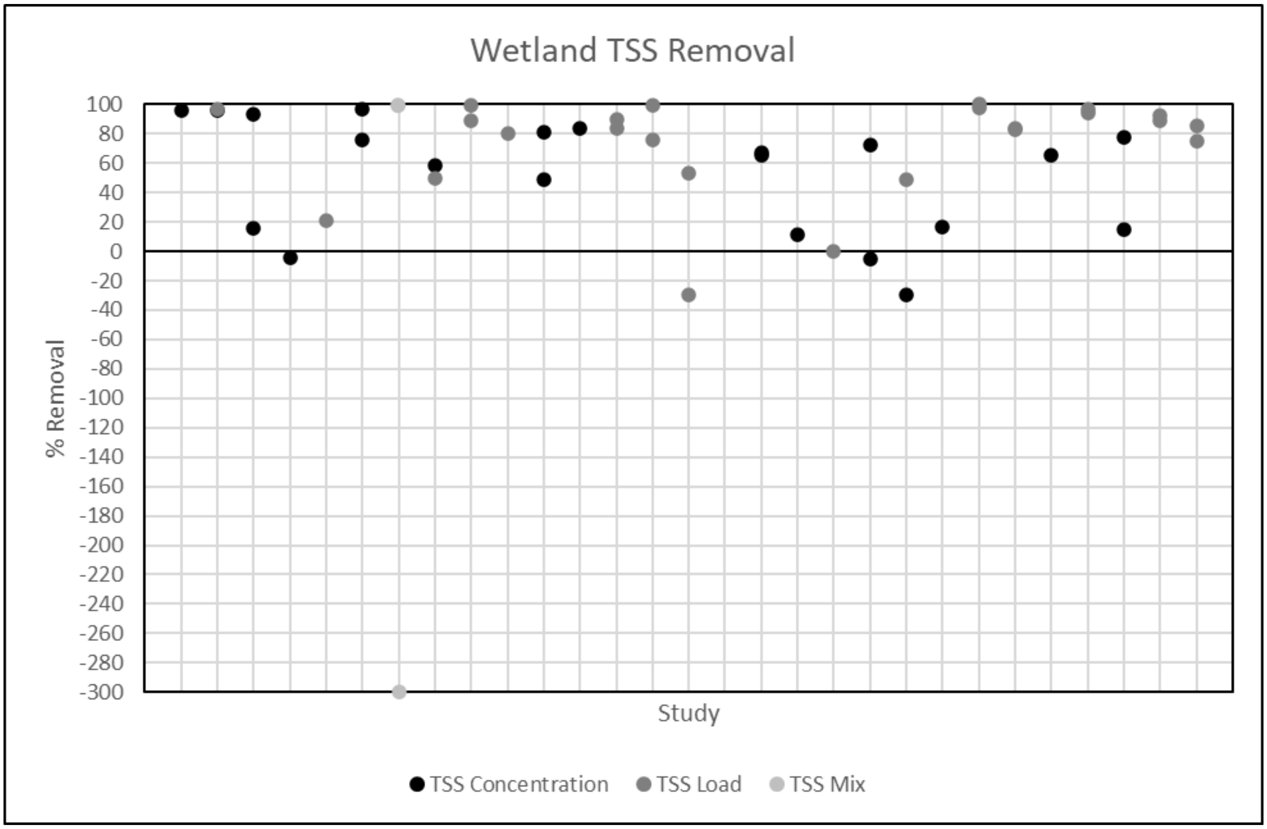


Figure 81. Scatterplot of wetland TSS removals: *Each vertical line represents one of the studies listed in Table 44. The points represent the percent removal using concentration or load methods. Mean and median values are not differentiated in this figure. Multiple points on a single vertical line represent multiple values reported by the study. Multiple points of the same color on the same line indicate that the study provided a range, and the points represent the upper and lower bound of the range.*

Table 44. Summary of Literature Describing the Effect of Wetlands on Total Suspended Solids

Author	*Year*	*Description*	*TSS*
Al-Rubaei et al.	2014	A 19-year-old constructed wetland and pond with no maintenance performed since construction still recorded nutrient and metal concentration reductions.	96% c
Al-Rubaei et al.	2016	A constructed wetland and pond with no maintenance performed still reduced nutrient concentrations after 19 years without maintenance. When compared to its removal rates at 3 and 9 years old, the wetland performed more efficiently and stably when compared to newly constructed.	96% c 97% l
Babatunde et al.*	2011	A wetland constructed using dewatered alum sludge from drinking water production removed nutrients and became more efficient over time.	16-93% c
Birch et al.	2004	A wetland that drains a residential urban catchment reduced metal and nutrient concentrations, but still did not meet water quality standards for boating. TSS concentrations varied based on event. During four high-flow events, the removal efficiency ranged from 9 to 46%, but during two very high-flow events, the efficiency ranged from (-98) to (-76%).	(-4%) c, m
Borden	2001	Two wet detention ponds and one pond wetland system had varying levels of success for nutrient removal. TSS removal was not well correlated with other pollutant removals. Only the pond-wetland system removal listed.	21% l
Brown	1984	Researchers found an urban wetland was less effective at removing dissolved pollutants than total pollutants, as sedimentation was the key process of removal, but it still removed some nutrients.	non-volatile 97% l volatile 76% l
Carleton et al.	2001	An analysis of 35 wetland studies found removal rates varied widely.	(-300)-99.6% mix c, l
Carleton et al.	2000	A constructed wetland treating stormwater runoff was investigated. The median load removals were higher for the subset of storms that did not overflow the maximum volume of the wetland.	57.9% c, d 49.6% l, d

Table 44. Summary of Literature Describing the Effect of Wetlands on Total Suspended Solids, Continued

Author	*Year*	*Description*	*TSS*
Coveney et al.	2002	In a wetland constructed to reduce nutrients in a eutrophic lake, particulate matter was reduced by 90% but soluble inorganic compounds increased (though levels were low).	89-99% l
Dunne et al.	2012	A constructed wetland adjacent to a eutrophic lake reduced TSS, but the removal rate approached an asymptote as loading increased over 3 kg/(m^2 year). The removal efficiency of TSS was always above 80% but showed a slight decreasing trend at later periods.	>80% l, d
Guerrero et al.	2020	A comparison between two regional detention facilities with wetlands.	49-56% c, d 60-81% c, m
Hathaway and Hunt	2010	In a series of three wetlands, the first wetland removed at least 80% of the total concentration for all pollutants, and no pollutant was significantly reduced from the outlet of wetland 2 to the outlet of wetland 3. The removal efficiencies are listed in order of wetland.	84% c not sig c not sig c
Healy and Cawley	2002	A recently constructed surface flow wetland was investigated as a potential tertiary treatment option, and while it performed well at N reduction, it was less effective at P reduction. The major mechanism of nutrient removal was the settlement of particulates. TSS removal efficiency was high.	84-90% l
Hey et al.	1994	Four experimental wetlands were examined for percent removals, and researchers found that the wetlands all had similar outlet concentrations despite different loading rates.	76-99% l
Hoffmann et al.	2012	Two restored wetlands that received drainage water from agricultural fields rich in nitrate were monitored five years later and found to perform well at nitrogen removal.	(-30)-53% l
Jordan et al.	2003	In a wetland receiving inflows from a 14-acre agricultural watershed, the wetland performed better in the first year of the two-year study due to a drying period.	not sig l
Kadlec	2003	In an examination of 21 wastewater treatment systems that use wetlands, the median removals were found. TSS did not appear to be affected by season.	67% c, d 65% c, m

Table 44. Summary of Literature Describing the Effect of Wetlands on Total Suspended Solids, Continued

Author	*Year*	*Description*	*TSS*
Kadlec et al.	2010	Four free surface wetlands, part of a treatment system that included a deactivated sludge wastewater treatment plant, were examined. The wetlands were intended to control TSS. The TSS in this system did have strong seasonality; TSS was higher in winter in all parts of the system. Some of the TSS exported from the wetland during winter can be attributed to waterfowl (water is warm and does not freeze). The TSS export is mostly inorganic, but the input is mostly organic.	11% c, m winter 3% c spring/sum/fall 20% c
Kohler et al.*	2004	A four-year study of golf course wetlands indicated the wetlands were able to efficiently remove nutrients and metals, but the relative reduction of suspended solids was 0%. Removals were based on the first 15-minute interval when samples taken.	0% l
Koskiaho	2003	Two constructed wetlands were investigated. An elongated shape appeared to help maintain high hydraulic efficiency. The wetland covered approximately 5% of its watershed area performed better than its counterpart covering 0.5%.	Large 43-72% c small (-5)-7% c
Lenhart and Hunt	2011	Researchers examined a constructed wetland by comparing four different metrics of performance. For TSS, the removal efficiency was positive for load and negative for concentration.	(-30%) c, m 49% l, m
Martin	1988	A system of a detention pond with wetlands was able to reduce nutrients and suspended solids. Only wetland removal listed.	17% c
Naylor et al.	2003	Researchers attempted to combine plants to remove N and steel slag/limestone to remove P in one wetland and found that it was best to do a two-part wetland (basin 1 planted and basin 2 unplanted with p adsorbing substrate) as the high *p*H of the steel slag/limestone inhibits plant growth. TSS removal for all treatments was above 98%.	98-100% l
Oberts and Osgood	1991	A detention/wetland system including a detention pond followed by six in line wetlands was found to have high removal rates of nutrients for combined snowmelt and rain events.	83-84% l
Rai et al.	2013	A subsurface flow constructed wetland was investigated for its ability to remove contaminants from onsite sewage at varying retention rates. The listed rate is for 36-hour retention.	65% c

Table 44. Summary of Literature Describing the Effect of Wetlands on Total Suspended Solids, Continued

Author	*Year*	*Description*	*TSS*
Rodríguez and Brisson	2015	A comparison of wetlands planted with native and European phragmites indicated that native phragmites showed the potential for treatment removal and that it has the potential to outperform the European variety.	94-97% l
Schulz and Peall	2001	A constructed wetland was installed along a tributary and investigated for its ability to remove agricultural runoff. During wet periods, the TSS concentrations were much higher at the inlet, but the concentration at the outlet was constant during dry and wet periods.	Dry: 15% c Wet: 78% c
Senzia et al.	2003	The performance of six subsurface flow constructed wetlands that received effluent from primary facultative ponds was investigated. The wetlands were placed immediately after the primary facultative pond and after a string of facultative ponds and a maturation pond. The wetlands immediately after the primary facultative pond had mostly higher removal levels of nitrogen species and TSS than the one after the maturation pond.	89-92% l
Tanner et al.*	1995	This study examined how loading rate affected the mass removal of suspended solids. The loading rate of SS varied from 60 to 250 g/m^3. It found that the mean annual SS removal rates (%) remained within the same range regardless of loading rates.	75-85% l

5.12 Impact of Detention and Retention on TSS Removal

Table 45 summarizes 16 studies describing the effect of detention on TSS removal. In this instance, a detention pond refers to a method of stormwater control where water is temporarily stored with water draining from the detention pond in between storm events. A detention pond is synonymous with a dry retention pond and dry detention pond. There was one study (marked with an asterisk) that used "SS" as its measure, but the abbreviation stood for suspended solids rather than the alternative of settleable solids. For that reason, all measures of suspended solids have been lumped together into the category TSS. All the studies examined reported positive average removal rates for TSS ranging from 4 to 93% (Figure 82). Many studies focused on how real-time control can improve TSS removal efficiency. This can be accomplished by opening or closing the outlet valve based on forecast (Gaborit et al., 2013), closing the outlet valve when the water level increases beyond a threshold (Sharior et al., 2019), or using the outlet valve to increase residence time (Middleton and Barrett, 2008). One study indicated that the TSS removal varied widely in part because of a wide range of TSS loadings that aren't apparent in the average TSS removal rate (Birch et al., 2006). High TSS removal can cause a loss in capacity of the detention pond as sedimentation occurs (Stanley, 1996). In contrast, another study found that unmaintained detention basins still performed well (Wissler et al., 2020b).

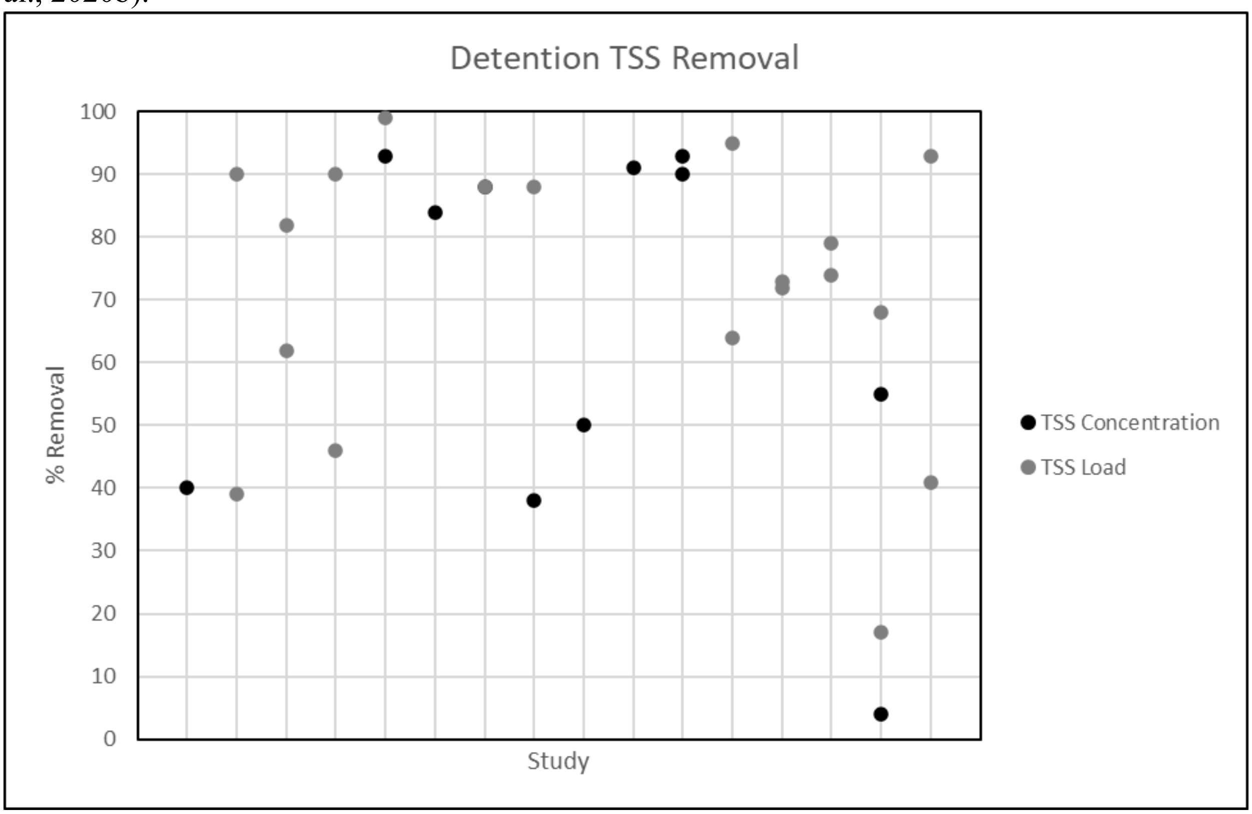


Figure 82. Scatterplot of detention pond TSS removals: Each vertical line represents one of the studies listed in Table 45. The points represent the percent removal using concentration or load methods. Mean and median values are not differentiated in this figure. Multiple points on a single vertical line represent multiple values reported by the study. Multiple points of the same color on the same line indicate that the study provided a range, and the points represent the upper and lower bound of the range.

Table 45. Summary of Literature Describing the Effect of Detention on Total Suspended Solids Removal

Author	*Year*	*Description*	*TSS*
Birch et al.	2006	The nutrient removal performance of a detention pond alongside a motorway was variable. The range of TSS removal is large (-12 to 93%) due to variable stormwater input.	40% c, m
Carpenter et al.	2014	A detention basin retrofitted with a new outlet to increase retention had increased efficiency.	39- 90% l
Comings et al.	2000	Two "wet detention ponds" (retention ponds) were investigated for performance and the one that was newer, focused on water quality, and had a longer detention time performed better.	62-82% l
Gaborit et al.	2013	Real Time Control (RTC) of a stormwater detention basin using an outfall valve and rainfall forecasts (opening or closing the outfall valve based on the forecast) increased the TSS removal efficiency when modeled using SWMM5.	46% l original 90% l RTC
Harper et al.	1999	A detention pond had higher mass removal efficiencies due to groundwater seepage than concentration removal efficiencies.	93% c 99% l
Hossain et al.	2005	The efficiency and flow regime of a stormwater detention pond was investigated. The flow regime varied with its changing surface topography.	84% c
House et al.*	1993	An urban stormwater detention pond near a lake was able to decrease both median loads and concentrations.	88% c, d 88% l
Hussain et al.	2005	Detention ponds with under-drains demonstrated pollutant removal capabilities for both load and concentration.	38% c 88% l
Martin	1988	A system of a detention pond with wetlands was able to reduce nutrients and suspended solids. Only detention removal listed.	50% c
Middleton and Barrett	2008	By attaching an automated valve/controller to an extended detention basin, the overall pollutant removal efficiency was improved. This created a batch type detention basin that increased the residence time.	91% c
Oberts and Osgood	1991	A detention/wetland system including a detention pond followed by six in line wetlands was found to have high removal rates of nutrients for combined snowmelt and rain events.	90-93% c

Table 45. Summary of Literature Describing the Effect of Detention on Total Suspended Solids Removal, Continued

Author	*Year*	*Description*	*TSS*
Sharior et al.	2019	A model was used to assess how real-time active control of detention basin outflow affected water quality. The real-time control increased the retention compared to the passive outlet. Of the three controls used, the one where the valve is closed when TSS is above a threshold performed best.	64% l passive 77-95% l RTC
Stanley	1996	A detention pond was monitored and found to remove pollutants, but in almost all storms its capacity was exceeded, and it also lost 0.16% storage each year (due to TSS).	73% l, d 72% l, m
Wissler et al.	2020a	Two maintained detention basins were examined for performance and found to reduce pollutants, but not enough for water to be safe.	4-55% c, d 17-68% l, d
Wissler et al.	2020b	Two unmaintained detention basins were examined for performance and found to reduce pollutants despite no maintenance.	74-79% l, d
Wu Wu et al.	1989 1996	Three urban detention ponds not originally designed for water quality were examined for water quality, and it was found that 1-2% of watershed area should be used for siting to maintain 70% removal of sediment. Presence of geese increases pollutant levels of ponds.	41-93% l

Table 46 summarizes nine studies on the effect of retention ponds on the removal efficiency of total suspended solids (TSS). In this instance, a retention pond refers to a method of stormwater control where water is permanently stored within the pond and does not fully drain in between storm events. A retention pond is synonymous with a wet retention pond and wet detention pond. Two studies (marked with an asterisk) used “SS” as their measure, but the abbreviation stood for suspended solids rather than the alternative of settleable solids. For that reason, all measures of suspended solids have been lumped together into the category TSS. Of the nine studies examined, eight reported positive removal rates of TSS ranging from 11 to 87% (Figure 83), and one study reported no significant results for TSS removal. Some of the studies found that one large retention pond performed better at TSS removal and flood mitigation than a series of smaller retention ponds, but retention ponds on the whole performed better at TSS removal and peak flow reduction for storms of a smaller depth (Ahmadisharaf et al., 2021). In addition, mean removal rates of TSS for retention ponds could obfuscate a wide range of variability (Chrétien et al., 2016), the retention rate of TSS showed a seasonal influence (Semadeni-Davies, 2006), and young retention ponds had higher removal rates for TSS than older ones (Sønderup et al., 2016).

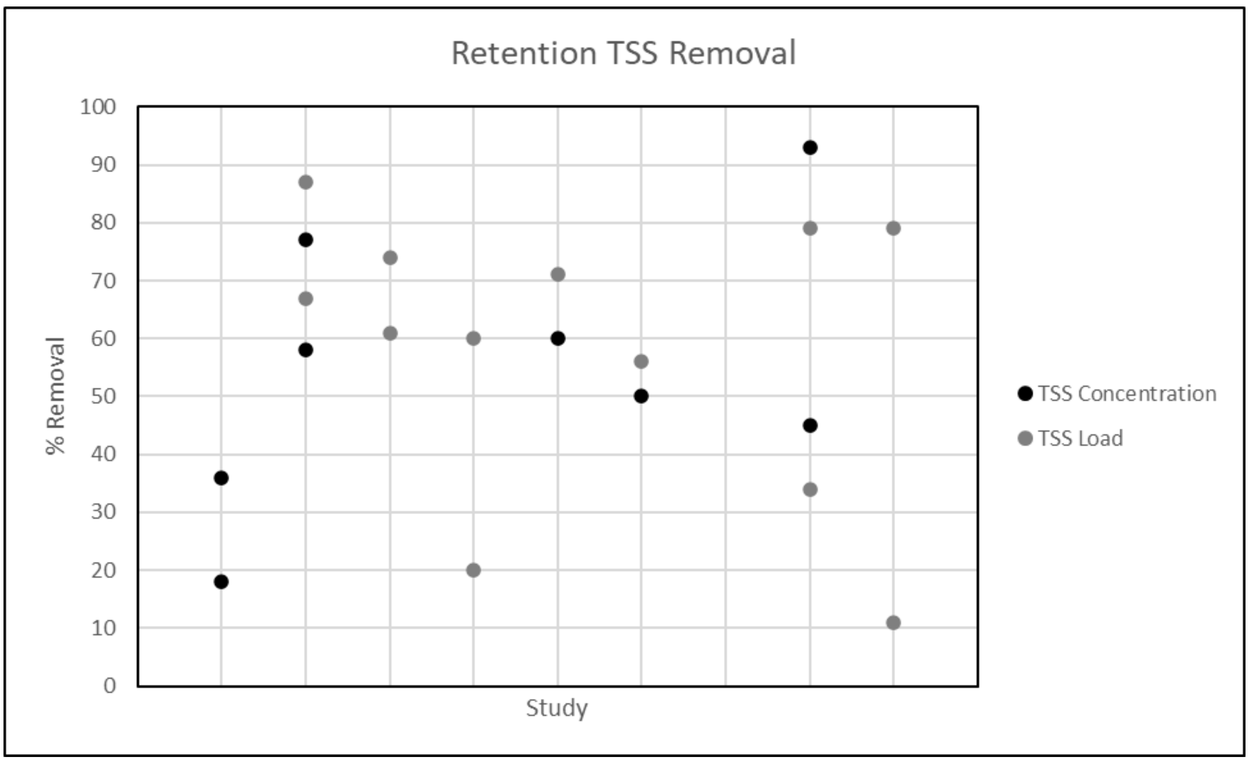


Figure 83. Scatterplot of retention pond TSS removals: Each vertical line represents one of the studies listed in Table 46. The points represent the percent removal using concentration or load methods. Mean and median values are not differentiated in this figure. Multiple points on a single vertical line represent multiple values reported by the study. Multiple points of the same color on the same line indicate that the study provided a range, and the points represent the upper and lower bound of the range.

Table 46. Summary of Literature Describing the Effect of Retention on Total Suspended Solids Removal

Author	Year	Description	TSS
Ahmadisharaf et al.	2021	In a model of hypothetical retention ponds, a large retention pond was more efficient than a series of small ponds for flood mitigation and TSS removal, and retention ponds more effectively reduced peak flow and TSS in smaller storm depths.	18-36% c
Baird et al.	2020	Two infiltrating wet retention ponds designed to allow for both infiltration and retention were tested and found to reduce annual pollutant loads.	73-74% c, d 58-77% c, m 67-87% l
Borden	2001	Two wet detention ponds and one pond wetland system had varying levels of success for nutrient removal. Only detention ponds listed.	61-74% l
Borden et al.	1997	Two retention ponds with different influent pollutant concentrations were examined for removal efficiency and neither met the design objective of 85% TSS removal.	20-60% l
Chen et al.*	2009	A train of treatment ponds (retention pond 1- retention pond 2- eco pond- gravel filter bed- limestone filter bed- vegetative buffer) was examined for pollutant removal efficiency. By calculating removal efficiency using different methods, a range of values was found.	60% c 71% l
Chrétien et al.	2016	A retention pond receiving runoff from agriculture was examined for pollutant removal efficiency. The mean removal efficiencies do not show the wide range of variation over the study period.	50% c 56% l
Mallin et al.	2002	Three "wet detention ponds" (retention ponds) were examined for performance, and generally, the most successful pond had a high length-to-width ratio and lots of native macrophyte species. There were no significant results reported for TSS removal.	not sig not sig not sig
Semadeni-Davies	2006	Stormwater ponds were investigated to determine the winter-spring removal rates. The inflow rate of TSS varied widely but tended to be lower during the winter than the summer, potentially due to melt events having less intensity than rain events. The removal rates varied month to month during the winter/spring. The overall rate removal efficiency of winter was 49% compared to 79% in the summer.	summer 73% c 79% l winter/spring 45-93% c 34-74% l
Sønderup et al.*	2016	Retention ponds combined with sand filters had higher retention rates than wet ponds alone, and young retention ponds had higher retention than older ones.	retention 11.1% l, d + sand filter 79.2% l, d

5.13 Impact of Stormwater Control Measures on Iron Removal

Table 47 summarizes four studies on the effect of wetlands on iron removal. Iron (Fe) is a metal that was not as commonly reported in the literature examined for this literature review as other metals such as zinc or copper. Of the four studies that did report iron removal for wetlands, two studies reported negative removal efficiencies (Arroyo et al., 2010; Birch et al., 2004), one study reported a range that included both negative and positive removal efficiencies (Hoffmann et al., 2012), and one study reported a positive, but relatively low, removal efficiency (Kohler et al., 2004). In general, wetland iron removal is not highly successful.

Table 47 also summarizes three studies (two papers by the same author detailed the same study) on the effect of detention ponds on iron removal. In this instance, a detention pond refers to a method of stormwater control where water is temporarily stored with water draining from the detention pond in between storm events. A detention pond is synonymous with a dry retention pond and dry detention pond. Two of the studies reported iron removal of greater than 50% (Harper et al., 1999; Wu, 1989; Wu et al., 1996). The third study had a relatively low removal rate of 3%, but even more notably, the range of iron removal varied greatly from (-60%) removal to 89% removal (Birch et al., 2006). One study found that a retention pond sized to have a surface-to-area ratio of 1–2% would have about a 60% iron removal rate (Wu, 1989; Wu et al., 1996). In one of the other studies, groundwater seepage played a major role in the removal efficiencies as the retention pond had an underdrain that regularly clogged (Harper et al., 1999).

Table 47 also summarizes four studies on the effect of retention ponds on iron removal. In this instance, a retention pond refers to a method of stormwater control where water is permanently stored within the pond and does not fully drain in between storm events. A retention pond is synonymous with a wet retention pond and wet detention pond. The studies reported a wide range of iron removal rates. The lowest reported was (-329%) and the highest was 77% removal; both were reported in a study that looked at three different retention ponds. The third pond had a removal rate that was not significant (Mallin et al., 2002). The other values reported across the studies had a much smaller range from about (-4%) removal to 38% removal.

Table 47 also summarizes five studies describing the effect of biofiltration on iron removal. The studies examined different methods of biofiltration including grass swales, vegetative filter strips, and bioretention cells. One of the studies reported an iron removal rate of (-13,000%), which the study noted was likely attributable to soil leaching (Hunt et al., 2006). The only other study that found a negative iron removal rate was by the same primary author at a rate of (-330%) (Hunt et al., 2008). Both studies that reported negative removal rates were bioretention cells. The other studies reported removal rates above 50%. These studies included results from grass swales, vegetated filter/buffer strips, and a bioretention cell.

Table 47 also includes the only study examined that described the effect on an infiltration basin on iron removal (Birch et al., 2005). Their findings indicate that an infiltration basin was moderately to highly efficient at removing suspended particulate matter from stormwater. The study attributes the increase of Fe in the outflow of the infiltration basin potentially to leaching of clay minerals in the topsoil. The study reported a (-81%) iron removal rate, which was attributed potentially to leaching clay minerals in the topsoil.

Table 47. Summary of Literature Describing the Effect of Stormwater Control Measures on Iron Removal

Author	*Year*	*Description*	*Fe*
		Effect of wetlands on iron removal	
Arroyo et al.	2010	The study examined a pilot wetland treating wastewater to determine metal removal efficiency (11 different metals). Generally, the study found that the constructed wetland did have as high of removal efficiencies as other reported studies in the literature.	-12% c
Birch et al.	2004	A wetland that drains a residential urban catchment reduced some metal and nutrient concentrations, but not iron concentrations. However, the effluent concentrations of contaminants were still higher than the Public Health standards for secondary contact.	-84% c, m
Hoffmann et al.	2012	Two restored wetlands that received drainage water from agricultural fields rich in nitrate were monitored five years later and found to perform well at nitrogen removal.	(-13)-42% c (-11)-46% l
Kohler et al.	2004	A four-year study on golf course wetlands indicated the wetlands were able to efficiently remove 11 of 17 nonzero parameters of nutrients and metals. Removals based on first 15-minute interval when samples taken.	11% l
		Effect of detention on iron removal	
Birch et al.	2006	The nutrient removal performance of a detention pond alongside a motorway was variable for the nutrients and metals examined.	3% c, m ((-60)-89%)
Harper et al.	1999	A detention pond had higher mass removal efficiencies due to groundwater seepage than concentration removal efficiencies. This particular detention pond had a filter underdrain that regularly clogged without backwashing (every couple of weeks), so without groundwater seepage, the pond would not have remained dry in between storms.	64% c 94% l
Wu Wu et al.	1989 1996	Three retention ponds were examined for water quality and a relationship between performance and surface-to-area ratio was found. Generally, a 1-2% ratio of surface to area resulted in 60% removal of iron.	52-87% l

Table 47. Summary of Literature Describing the Effect of Stormwater Control Measures on Iron Removal, Continued

Author	*Year*	*Description*	*Fe*
		Effect of retention on iron removal	
Borden	2001	Two wet detention ponds and one pond wetland system had varied removal efficiencies for the parameters tested. One pond had Fe removal efficiencies that varied 33-37% over the seasons, while the other varied 24-48%. Only detention ponds listed.	35-38% l
Borden et al.	1997	Two retention ponds with different influent pollutant concentrations were examined for removal efficiency. The major design goal of the retention ponds was 85% TSS removal. Regardless of iron removal, the ponds did not achieve that goal.	(-4)-29% l
Mallin et al.	2002	Three "wet detention ponds" (retention ponds) were examined for performance, and generally, the most successful pond had a high length-to-width ratio and lots of native macrophyte species.	(-329%) c 77% c not sig c
Sønderup et al.	2016	Retention ponds combined with sand filters tend to have higher retention rates for nutrients and metals than wet ponds alone and young retention ponds tend to have higher retention than older ones. Total iron is an exception.	-0.2% l, d no filter -4.0% l, d sand filter
		Effect of biofiltration on iron removal	
Barrett et al.	1998	The study examined grass swales and vegetative filter strips along highways to determine removal efficiency. They found that medians with side slopes less than 12% and with a length of at least 8 m from the pavement edge to the center were able to reduce storm water loads from highways.	75-79% c
Glass and Bissouma	2005	The study examined a bioretention cell to determine its pollutant removal efficiency. The study found removals that were less than what was previously reported for this particular cell, but it had generally high removal rates for the pollutants examined.	51% c
Hunt et al.	2006	The study examined three bioretention cells for pollutant removal, two with conventional drainage and one with an underdrain. Iron has an extremely high increase, which in this study can likely be attributed to soil leaching.	(-13,000%) l
Hunt et al.	2008	The study examined a bioretention cell in an urban setting to determine peak flow mitigation and pollutant removal efficiency. The study found that in an urban setting, bioretention can reduce peak flow runoff for small to medium storms and can reduce pollutant concentrations.	(-330%) c

Table 47. Summary of Literature Describing the Effect of Stormwater Control Measures on Iron Removal, Continued

Author	*Year*	*Description*	*Fe*
Walsh et al.	1998	The study constructed a grass swale (GS) to identify how swale length, water depth, and season impacted removal efficiency as well as conducted field experiments to measure efficiency of vegetated buffer strips (VB). Most of the removal by the constructed grass swale occurred in the first 20 m of the 40 m channel, and increasing water depth and velocity reduced removal efficiencies. The study recommended including vegetated buffer strips or grassed swales in a highway design and ignoring seasonal effects on efficiency for design considerations.	GS: 74% l VB: 79-83% l
		<u>Effect on an infiltration basin on iron removal</u>	
Birch et al.	2005	An infiltration basin was moderately to highly efficient at removing suspended particulate matter from stormwater. The study attributes the increase of Fe in the outflow of the infiltration basin potentially to leaching of clay minerals in the topsoil.	-81% c, m

5.14 Impact of Stormwater Control Measures on Removal of Chloride and Silver

Eleven studies examined the impact of wetlands, detention and retention, and bio-infiltration on the removal of chloride and silver from stormwater runoff. Table 48 and Table 49 summarize the results of these studies.

Table 48 describes the effects of different stormwater best management practices on chloride (Cl-) removal. The stormwater practices include wetlands, detention, retention, and biofiltration. The uniting theme across the different techniques is a wide range of reported chloride removals. Every technique had both positive and negative removals, despite each having three or fewer studies examined per technique. In addition, the removal rates tended to have high absolute values.

Table 49 describes the effect of wetlands and retention on silver (Ag) removal. Silver removal was not frequently reported in the literature examined. Much more frequent were metals such as copper or zinc. The wetlands reported relatively high silver removals. The two studies examined reported removal rates above 80% (Auvinen et al., 2017; Crites et al., 1997). The one study that reported silver for retention ponds reported a rate of 22%; however, in most months, the data indicated that silver was below detection limits (Borden et al., 1997).

5.15 Impact of Distributed Stormwater Control Measures on Nutrient Removal

Sixteen studies that examined the impact of distributed stormwater control measures on removal of sediments and nutrients from urban runoff are summarized in Table 50. These studies all focused on specific, localized implementation of distributed stormwater control measures. The studies consistently showed a reduction in loads of TSS, with values ranging from 38 to 99% reduction. Studies showed a wide variation in the effect of distributed practices on nitrogen and phosphorus loads, with some studies showing an increase in loads (particularly when baseflow is considered) and others showing reductions of up to 99%. Hopkins et al. (2017) indicated that distributed stormwater control measures had 70% lower sediment discharge per unit watershed area than centralized stormwater control and event mean sediment concentrations that were 70% lower than centralized stormwater control measures for 40 storm events over a two-year period. This study also indicated that runoff from the watershed with distributed stormwater control measures showed lower event-mean particulate phosphorus concentrations than from centralized measures, except for large storm events, where the runoff from the watershed with distributed stormwater control measures showed larger event-mean phosphorus. Hunt et al. (2008) monitored the inflow to and outflow from a bioretention basin that received runoff from a municipal parking lot for 23 storms over 25 months. Their monitoring showed that bioretention can reduce concentrations of many urban pollutants, including nutrients, sediment, metals, and fecal indicator bacteria. Dutta et al. (2021) developed SWMM model parameters based on case studies in the literature and then used the model to determine removal efficiencies for TSS, TN, and TP for bioretention, vegetative swales, and permeable pavements. Their results indicated removal efficiencies between 19 and 23% for all three pollutants for all practices, except for no removal of TN by permeable pavements.

Table 48. Summary of Literature Describing the Effect of Wetlands on Chloride Removal

Author	*Year*	*Description*	*Cl^-*
		Effect of wetlands on chloride removal	
Kohler et al.	2004	A four-year study on golf course wetlands indicated the wetlands were able to efficiently remove nutrients and metals. Removals based on first 15-minute interval when samples taken.	77% l
Martin	1988	A system of a detention pond with wetlands was able to reduce nutrients and suspended solids. Only the wetland removal rate is listed.	(-10%) c
Moustafa	1999	The Everglades Nutrient Removal Project was assessed over three years. The average Cl- mass entering the wetland and exiting was nearly identical.	Not sig
		Effect of detention on chloride removal	
Harper et al.	1999	A detention pond had higher mass removal efficiencies due to groundwater seepage than concentration removal efficiencies.	42% c 89% l
House et al.	1993	An urban stormwater detention pond near a lake was generally able to decrease both median loads and concentrations; however, chloride event mean concentrations were usually higher in the outflow than inflow.	-245 c, d -58% l
Martin	1988	A system of a detention pond with wetlands was able to reduce most nutrients and metals examined, though chloride did not have a high removal efficiency. Only detention removal listed.	1% c
		Effect of retention on chloride removal	
Semadeni-Davies	2006	Stormwater ponds were investigated to determine the winter-spring removal rates. Some removals remained consistent, but many removal rates dropped in winter compared to summer. For chloride specifically, the retention pond could retain up to 80% for a snowmelt-generated flow event, but the study found chloride was flushed in between events. Chloride loads were also much higher in the winter inflow (58,000 kg for a period of nine days in January versus 3,500 kg for five days in April).	(-90)-67% c (-130)-77% l

Table 48. Summary of Literature Describing the Effect of Wetlands on Chloride Removal, Continued

Author	*Year*	*Description*	*Cl^-*
		Effect of biofiltration on chloride removal	
Burgis et al.	2020	The study monitored a bioretention cell and bioswale to determine its impact on salt. Both significantly reduced Cl-, but the study suggests that the majority of salt removed from surface water infiltrates into groundwater.	bioretention: 40% c 80% l bioswale: 4% c 76% l
Li and Davis	2009	The study monitored two bioretention cells for 15 months to determine the impact on different contaminant parameters. The study found that runoff volume reduction and pollutant mass removal went hand in hand. One cell significantly reduced chloride and the other significantly increased chloride concentration/load.	(-400)-0% c, d (-154)-97% l, d

Table 49. Summary of literature describing the effect of wetlands on silver removal

Author	*Year*	*Description*	*Silver (Ag)*
		Effect of wetlands on silver removal	
Auvinen et al.	2017	In an investigation of silver removal in a constructed wetland treating wastewater, TSS and silver removal were correlated, indicating silver was likely bound to solids.	80-90% l
Crites et al.	1997	A constructed free surface wetland was monitored for a year and a half. Silver was not the focus, but the influent and effluent concentrations of it are provided.	97% c, m
Borden et al.	1997	**Effect of retention on silver removal** Two retention ponds with different influent pollutant concentrations were examined for removal efficiency and neither met design objective of 85% TSS removal. Only one of the ponds was monitored for silver and most months, total silver was below the detection limit.	22% l

Table 50. Summary of Literature Describing the Effect of Distributed Stormwater Control Measures on Nutrient and Sediment Removal

Author	*Year*	*Description*
Ahiablame et al.	2012	Reviewed literature related to low-impact development (LID) stormwater control measures. Reviewed both field and laboratory studies and numerical model studies. Field and laboratory studies showed removal efficiencies from 47 to 99% for TSS, from 32 to 99% for TN, from -3 to 99% for TP, from 43 to 99% for copper, from 31 to 98% for lead, and from 62 to 97% for zinc. Discussed three commonly used models (Long-Term Hydrologic Impact Assessment–Low Impact Development (L-THIA-LID); Storm Water Management Model (SWMM), and System for Urban Stormwater Treatment and Analysis INtegration (SUSTAIN)). L-THIA-LID is strictly a practice-based approach using the runoff curve number and event-mean concentration to calculate runoff and water quality. SWMM and SUSTAIN allow limited process-based simulation, allowing power function, exponential function, saturation function, first-order decay, and continuously stirred reactor functions to estimate water quality processes.
Brodeur-Doucet et al.	2021	Examined runoff from treatment trains serving a 2.4 ha public market in Longueuil, Quebec. One train consisted of a series of five 70 m^2 bioretention cells followed by a wet retention pond receiving runoff from a 0.91 ha parking lot. The bioretention alone reduced runoff volumes to retention basin by 8–10, reduced peak flows by an average of 95%, reduced TSS by 84%, increased TN by 5%, and reduced TP by 6%. The retention basin alone increased TSS by 20% and reduced TN by 28% and TP by 40%. The treatment train of bioretention followed by retention pond showed a TSS removal of 89%.
Brown et al.	2012	Parking lot (0.89 ha total area) was retrofit with 0.53 ha of pervious pavement followed by a 500 m^2 bioretention cell. 33 of 80 rainfall events in 17-month period produced runoff; samples collected for 17 of these events. Total runoff volume reduced 69%; sediment and nutrient loads reduced 89% (TSS), 49% (TN), and 51% TP. However, when baseflow was considered, TN load was actually 64% larger than the runoff loads.

Table 50. Summary of Literature Describing the Effect of Distributed Stormwater Control Measures on Nutrient and Sediment Removal, Continued

Author	*Year*	*Description*
Chapman and Horner	2010	Examined runoff from a cascading series of 12 bioretention basins with a total surface area of 400 m^2 that captures runoff from an urban area with 0.81 ha of impervious area. Examined runoff volumes and concentrations of TSS, TN, total and dissolved phosphorus, total and dissolved copper, lead, and zinc, and motor oil. Showed at least 48% of inflows captured. Showed pollutant removal efficiencies of 87% of TSS, 63% of TN, 67% of TP, 80% of copper, 80% of Zinc, 86% of lead, and 92% of motor oil.
Davis et al.	2006	Examined data from controlled runoff events in two field sites in Maryland. Showed removal efficiencies of TP 65-87%, TKN 52-67%, and TN 49-59%
Davis	2007	Examined data for 12 runoff events at two bioretention sites on the University of Maryland College Park campus. Showed removal efficiencies of 47% (TSS), 76% (TP), 83% (TN), 57% (copper), 83% (lead), and 62% zinc.
DeBusk and Wynn	2011	Examined runoff from a 35 m^2 bioretention cell capturing runoff from a 1600 m^2 impervious parking lot from 28 runoff-producing events over a 6.5-month period. Showed 97% reduction in runoff volume. Showed cumulative mass reductions greater than 99% for TSS, TN, and TP. However, only three out of 28 storms sampled for sediment and nutrient in outflow, so load values may be biased by large number of storms with no outflow sample.
Dutta et al.	2021	Provided an analysis of the literature on distributed stormwater controls ("Nature-based solutions") and of the tools to simulate these practices and developed a "decision matrix" to select a model, practices, and pollutants to test. Developed model parameters based on case studies in the literature and then used a model to determine removal efficiencies for TSS, TN, and TP for bio-retention, vegetative swales, and permeable pavements. Their results indicated removal efficiencies between 19 and 23% for all three pollutants for all practices, except no removal of TN by permeable pavements.

Table 50. Summary of Literature Describing the Effect of Distributed Stormwater Control Measures on Nutrient and Sediment Removal, Continued

Author	*Year*	*Description*
Eckart et al.	2017	Provided a review of the literature related to the performance of LID practices, particularly as an adaptation strategy for the impacts of urbanization and climate change.
Glass and Bissouma	2005	Collected inflow and outflow samples from 15 runoff-producing events at a 27.1 m^2 bioretention cell receiving runoff from a 12.1 ha impervious parking lot. Showed removal efficiencies of 98% (TSS), ammonia 65% (NH_3-N), 75% (copper), 71% (lead), and 80% (Zinc).
Hopkins et al.	2017	Compared a forested watershed (3% impervious) with developed watersheds (30–39% impervious) treated with centralized and distributed stormwater control measures for 40 paired storm events. All watersheds located in Montgomery County, MD. Showed sediment yield from watershed with distributed stormwater control measures was 70% lower than centralized stormwater control watershed. Showed event mean sediment concentrations from watershed with distributed stormwater controls was 20 mg/L and from watershed with centralized measures was 67 mg/L. Showed that event mean particulate phosphorus from watershed with distributed stormwater control measures 0.05 mg/L and event mean concentration from watershed with centralized measures was 0.13 mg/L. Indicated that mean particulate phosphorus concentrations from watershed with distributed control measures exceeded that from the watershed with centralized stormwater control measures for 40% of storm events, mainly the largest storm events.
Hunt et al.	2008	Samples from 23 runoff-producing events from a 229 m^2 bioretention basin that received runoff from a 0.37 ha municipal parking lot in Charlotte, NC. Concentrations measured in effluent were lower than influent concentrations by 59.5% (TSS), 32.2% (TN), 44.3% (TKN), 72.3% (NH_4-N), 31.4% (TP), 54% (copper), 31.4% (lead), and 77.0% (zinc). Also showed reduction in fecal coliform of 69%.

Table 50. Summary of Literature Describing the Effect of Distributed Stormwater Control Measures on Nutrient and Sediment Removal, Continued

Author	*Year*	*Description*
Jayakaran et al.	2019	Parking lot at Washington State University retrofit with 0.32 ha of pervious pavement. Parking lot was spiked with "street dirt" from local street sweeping and a spiked solution that included sediments, metals, oils, and nutrients applied before 9 of the 12 monitored storm events. TSS removal was 97% for surface outlets and 93% for underdrains. TKN removal was 61% for at surface outlet and 67% for underdrains. TP removal was 83% for at surface outlet and 88% for underdrains. Total copper removal was 81% for at surface outlet and 83% for underdrains. Total lead removal was 96% for at surface outlet and 98% for underdrains. Total zinc removal was 95% for at surface outlet and 98% for underdrains. Motor oil removal was 91% for both surface outlet and for underdrains.
Pagotto et al.	2000	Examined runoff from a 3200 m^2 section of motorway in the Nantes (France) metropolitan area. Compared runoff characteristics before and after resurfacing the road with 30 mm of porous asphalt. Average TSS concentrations reduced from 68 mg/L for conventional pavement to 13 mg/L for porous pavement. Loads of total metals lowered from 35% (copper) to 78% (lead).
Roseen et al.	2006	Examined runoff from 11 runoff-producing events from a bioretention basin with design volume equivalent to 23 mm rainfall over the 4000 m^2 watershed area. Showed removal efficiencies of 96.5% (TSS), 99 % (TP), and 99% (zinc).
Wilson et al.	2015	Compared runoff for 47 storms between adjoining commercial areas. Conventional area was .76 ha, 61% directly connected impervious area (DCIA) treated by 0.14 ha dry detention basin. Low impact development (LID) basin is 2.53 ha, 84% DCIA, treated by three cisterns (163 m^3), 140 m of grassed bioswales, a 60 m^2 bioretention cell, and a 1325 m^3 underground detention cell with 760 m of gravel-filled infiltration trench. For 47 monitored storms, runoff reduction from the conventional system was 51% and from the LID system was 98%. For four monitored storms, the conventional system showed a reduction of 38% in TSS and increases in TN and TP. For six monitored storms, the LID site showed a reduction of 67% in TSS and 50% in TP, but no reduction in TN.

5.16 References

Acreman, M.C., Riddington, R., Booker, D.J., 2003. Hydrological impacts of floodplain restoration: a case study of the River Cherwell, UK. Hydrol. Earth Syst. Sci. 7, 75–85. https://doi.org/10.5194/hess-7-75-2003

Adler, P.R., Summerfelt, S.T., Glenn, D.M., Takeda, F., 1996. Evaluation of a wetland system designed to meet stringent phosphorus discharge requirements. Water Environment Research 68.

Ahiablame, L.M., Engel, B.A. & Chaubey, I., 2012. Effectiveness of Low Impact Development Practices: Literature Review and Suggestions for Future Research. Water, Air, and Soil Pollution 223, 4253–4273. https://doi.org/10.1007/s11270-012-1189-2

Ahiablame, L., Shakya, R., 2016. Modeling flood reduction effects of low impact development at a watershed scale. Journal of Environmental Management 171, 81–91. https://doi.org/10.1016/j.jenvman.2016.01.036

Ahmadisharaf, E., Alamdari, N., Tajrishy, M., Ghanbari, S., 2021. Effectiveness of Retention Ponds for Sustainable Urban Flood Mitigation across Range of Storm Depths in Northern Tehran, Iran. J. Sustainable Water Built Environ. 7, 05021003. https://doi.org/10.1061/JSWBAY.0000946

Al-Rubaei, A.M., Engström, M., Viklander, M., Blecken, T., 2014. Long-Term Treatment Efficiency of a Constructed Stormwater Wetland: Preliminary Results 8.

Al-Rubaei, A.M., Engström, M., Viklander, M., Blecken, G.-T., 2016. Long-term hydraulic and treatment performance of a 19-year old constructed stormwater wetland—Finally maturated or in need of maintenance? Ecological Engineering 95, 73–82. https://doi.org/10.1016/j.ecoleng.2016.06.031

Álvarez, R., Ordóñez, A., Loredo, J., Younger, P.L., 2013. Wetland-based passive treatment systems for gold ore processing effluents containing residual cyanide, metals and nitrogen species. Environ. Sci.: Processes Impacts 15, 2115. https://doi.org/10.1039/c3em00410d

Al-Weshah, R., Keefer, L., Demissie, M., 1993. The Role of Wetlands in Stormwater Runoff for the Flint and Mutton Creek Watersheds, Lake County, Illinois. Illinois State Water Survey 71.

Andersson, J.L., Bastviken, S.K., Tonderski, K.S., 2005. Free water surface wetlands for wastewater treatment in Sweden: nitrogen and phosphorus removal. Water Science and Technology 51, 39–46. https://doi.org/10.2166/wst.2005.0283

Anim, D.O., Fletcher, T.D., Pasternack, G.B., Vietz, G.J., Duncan, H.P., Burns, M.J., 2019. Can catchment-scale urban stormwater management measures benefit the stream hydraulic environment? Journal of Environmental Management 233, 1–11. https://doi.org/10.1016/j.jenvman.2018.12.023

Arroyo, P., Ansola, G., de Luis, E., 2010. Effectiveness of a Full-Scale Constructed Wetland for the Removal of Metals from Domestic Wastewater. Water Air Soil Pollut 210, 473–481. https://doi.org/10.1007/s11270-009-0272-9

Aulenbach, B.T., Landers, M.N., Musser, J.W., Painter, J.A., 2017. Effects of Impervious Area and BMP Implementation and Design on Storm Runoff and Water Quality in Eight Small Watersheds. J Am Water Resour Assoc 53, 382–399. https://doi.org/10.1111/1752-1688.12501

Auvinen, H., Kaegi, R., Rousseau, D.P.L., Du Laing, G., 2017. Fate of Silver Nanoparticles in Constructed Wetlands—a Microcosm Study. Water Air Soil Pollut 228, 97. https://doi.org/10.1007/s11270-017-3285-9

Ayalew, T.B., Krajewski, W.F., Mantilla, R., 2015. Insights into Expected Changes in Regulated Flood Frequencies due to the Spatial Configuration of Flood Retention Ponds. J. Hydrol. Eng. 20, 04015010. https://doi.org/10.1061/(ASCE)HE.1943-5584.0001173

Ayub, K.R., Zakaria, N.A., Abdullah, R., Ramli, R., 2010. Water balance: case study of a constructed wetland as part of the bio-ecological drainage system (BIOECODS). Water Science and Technology 62, 1931–1936. https://doi.org/10.2166/wst.2010.473

Babatunde, A.O., Zhao, Y.Q., Doyle, R.J., Rackard, S.M., Kumar, J.L.G., Hu, Y.S., 2011. Performance evaluation and prediction for a pilot two-stage on-site constructed wetland system employing dewatered alum sludge as main substrate. Bioresource Technology 102, 5645–5652. https://doi.org/10.1016/j.biortech.2011.02.065

Baird, J.B., Winston, R.J., Hunt, W.F., 2020. Evaluating the hydrologic and water quality performance of novel infiltrating wet retention ponds. Blue-Green Systems 2, 282–299. https://doi.org/10.2166/bgs.2020.010

Barrett, M.E., Walsh, P.M., Malina, J.F., Charbeneau, R.J., 1998. Performance of Vegetative Controls for Treating Highway Runoff. Journal of Environmental Engineering 124, 1121–1128. https://doi.org/10.1061/(ASCE)0733-9372(1998)124:11(1121)

Bedan, E.S., Clausen, J.C., 2009. Stormwater Runoff Quality and Quantity From Traditional and Low Impact Development Watersheds. JAWRA Journal of the American Water Resources Association 45, 998–1008. https://doi.org/10.1111/j.1752-1688.2009.00342.x

Bell, C.D., Wolfand, J.M., Panos, C.L., Bhaskar, A.S., Gilliom, R.L., Hogue, T.S., Hopkins, K.G., Jefferson, A.J., 2020. Stormwater control impacts on runoff volume and peak flow: A meta-analysis of watershed modelling studies. Hydrological Processes 34, 3134–3152. https://doi.org/10.1002/hyp.13784

Bergman, M., Hedegaard, M.R., Petersen, M.F., Binning, P., Mark, O., Mikkelsen, P.S., 2011. Evaluation of two stormwater infiltration trenches in central Copenhagen after 15 years of operation. Water Science and Technology 63, 2279–2286. https://doi.org/10.2166/wst.2011.158

Beutel, M.W., Morgan, M.R., Erlenmeyer, J.J., Brouillard, E.S., 2014. Phosphorus Removal in a Surface-Flow Constructed Wetland Treating Agricultural Runoff. Journal of Environmental Quality 43, 1071–1080. https://doi.org/10.2134/jeq2013.11.0463

Beutel, M.W., Newton, C.D., Brouillard, E.S., Watts, R.J., 2009. Nitrate removal in surface-flow constructed wetlands treating dilute agricultural runoff in the lower Yakima Basin, Washington. Ecological Engineering 35, 1538–1546. https://doi.org/10.1016/j.ecoleng.2009.07.005

Birch, G.F., Matthai, C., Fazeli, M.S., Suh, J.Y., 2004. Efficiency of a constructed wetland in removing contaminants from stormwater. Wetlands 24, 459–466. https://doi.org/10.1672/0277-5212(2004)024[0459:EOACWI]2.0.CO;2

Birch, G.F., Fazeli, M.S., Matthai, C., 2005. EFFICIENCY OF AN INFILTRATION BASIN IN REMOVING CONTAMINANTS FROM URBAN STORMWATER. Environmental Monitoring and Assessment 16.

Birch, G.F., Matthai, C., Fazeli, M.S., 2006. Efficiency of a retention/detention basin to remove contaminants from urban stormwater. Urban Water Journal 3, 69–77. https://doi.org/10.1080/15730620600855894

Bizzi, S., Lerner, D.N., 2015. The Use of Stream Power as an Indicator of Channel Sensitivity to Erosion and Deposition Processes: SP AS AN INDICATOR OF EROSION AND DEPOSITION. River Res. Applic. 31, 16–27. https://doi.org/10.1002/rra.2717

Black, A., Peskett, L., MacDonald, A., Young, A., Spray, C., Ball, T., Thomas, H., Werritty, A., 2021. Natural flood management, lag time and catchment scale: Results from an empirical nested catchment study. Journal of Flood Risk Management n/a, e12717. https://doi.org/10.1111/jfr3.12717

Blanchette, M., Rousseau, A.N., Foulon, É., Savary, S., Poulin, M., 2019. What would have been the impacts of wetlands on low flow support and high flow attenuation under steady state land cover conditions? Journal of Environmental Management 234, 448–457. https://doi.org/10.1016/j.jenvman.2018.12.095

Bledsoe, B.P., 2002. Stream Erosion Potential and Stormwater Management Strategies. J. Water Resour. Plann. Manage. 128, 451–455. https://doi.org/10.1061/(ASCE)0733-9496(2002)128:6(451)

Booth, B., Henshaw, P.C., 2001. Rates of Channel Erosion in Small Urban Streams, in: Land Use and Watersheds: Human Influence on Hydrology and Geomorphology in Urban and Forest Areas, AGU Monograph Series, Water Science and Application. pp. 17–38.

Booth, D.B., Jackson, C.R., 1997. URBANIZATION OF AQUATIC SYSTEMS: DEGRADATION THRESHOLDS, STORMWATER DETECTION, AND THE LIMITS OF MITIGATION. J Am Water Resources Assoc 33, 1077–1090. https://doi.org/10.1111/j.1752-1688.1997.tb04126.x

Borden, R.C., 2001. PERFORMANCE EVALUATION OF REGIONAL WET DETENTION PONDS AND A WETLAND FOR URBAN NONPOINT SOURCE CONTROL (No. UNC-WRRI-2001-335). Water Research Institute of the University of North Carolina Chapel Hill.

Borden, R.C., Dorn, J.L., Stillman, J.B., 1997. Evaluation of wet ponds for protection of public water supplies (No. UNC-WRRI-97-311). Water Research Institute of the University of North Carolina Chapel Hill.

Braskerud, B.C., 2002. Factors affecting phosphorus retention in small constructed wetlands treating agricultural non-point source pollution. Ecological Engineering 19, 41–61. https://doi.org/10.1016/S0925-8574(02)00014-9

Brodeur-Doucet, C., Pineau, B., Corrivault-Gascon, J., Arjoon, D., Lessard, P., Pelletier, G., Duchesne, S., 2021. Seasonal hydrological and water quality performance of individual and in-series stormwater infrastructures as treatment trains in cold climate. Water Quality Research Journal, 56 (4): 205–217. https://doi.org/10.2166/wqrj.2021.026

Brown, R.G., 1984. Effects of an urban wetland on sediment and nutrient loads in runoff. Wetlands 4, 147–158. https://doi.org/10.1007/BF03160493

Brown, R.A., Line, D.E., Hunt,W.F., 2012. LID treatment train: pervious concrete with subsurface storage in series with bioretention and care with seasonal high water tables. Journal of Environmental Engineering, 138 (6):689–697. http://dx.doi.org/10.1061/(ASCE)EE.1943-7870.0000506.

Burgis, C.R., Hayes, G.M., Henderson, D.A., Zhang, W., Smith, J.A., 2020. Green stormwater infrastructure redirects deicing salt from surface water to groundwater. Science of The Total Environment 729, 138736. https://doi.org/10.1016/j.scitotenv.2020.138736

Carleton, J.N., Grizzard, T.J., Godrej, A.N., Post, H.E., Lampe, L., Kenel, P.P., 2000. Performance of a Constructed Wetlands in Treating Urban Stormwater Runoff. Water Environment Research 72, 295–304. https://doi.org/10.2175/106143000X137518

Carleton, J.N., Grizzard, T.J., Godrej, A.N., Post, H.E., 2001. Factors affecting the performance of stormwater treatment wetlands. Water Research 35, 1552–1562. https://doi.org/10.1016/S0043-1354(00)00416-4

Carpenter, J.F., Vallet, B., Pelletier, G., Lessard, P., Vanrolleghem, P.A., 2014. Pollutant removal efficiency of a retrofitted stormwater detention pond. Water Quality Research Journal 49, 124–134. https://doi.org/10.2166/wqrjc.2013.020

Chapman, C., & Horner, R. R. (2010). Performance assessment of a street-drainage bioretention system. Water Environment Research, 82(109), 109–119. https://doi.org/10.2175/106143009X426112

Chen, C.-F., Lin, J.-Y., Huang, C.-H., Chen, W.-L., Chueh, N.-L., 2009. Performance evaluation of a full-scale natural treatment system for nonpoint source and point source pollution removal. Environ Monit Assess 157, 391–406. https://doi.org/10.1007/s10661-008-0544-7

Chen, H., Ivanoff, D., Pietro, K., 2015. Long-term phosphorus removal in the Everglades stormwater treatment areas of South Florida in the United States. Ecological Engineering 79, 158–168. https://doi.org/10.1016/j.ecoleng.2014.12.012

Chrétien, F., Gagnon, P., Thériault, G., Guillou, M., 2016. Performance Analysis of a Wet-Retention Pond in a Small Agricultural Catchment. J. Environ. Eng. 142, 04016005. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001081

Comings, K.J., Booth, D.B., Horner, R.R., 2000. Storm Water Pollutant Removal by Two Wet Ponds in Bellevue, Washington. J. Environ. Eng. 126, 321–330. https://doi.org/10.1061/(ASCE)0733-9372(2000)126:4(321)

Coveney, M.F., Stites, D.L., Lowe, E.F., Battoe, L.E., Conrow, R., 2002. Nutrient removal from eutrophic lake water by wetland filtration. Ecological Engineering 19, 141–159. https://doi.org/10.1016/S0925-8574(02)00037-X

Cowardin, L.M., Carter, V., Golet, F.C., Laroe, E.T., 1979. Classification of Wetlands and Deepwater Habitats of the United States. Fish and Wildlife Service, U.S. Department of the Interior.

Crites, R.W., Dombeck, G.D., Watson, R.C., Williams, C.R., 1997. Removal of metals and ammonia in constructed wetlands. Water Environment Research 69, 132–135. https://doi.org/10.2175/106143097X125272

Damodaram, C., Giacomoni, M.H., Prakash Khedun, C., Holmes, H., Ryan, A., Saour, W., Zechman, E.M., 2010. Simulation of Combined Best Management Practices and Low Impact Development for Sustainable Stormwater Management1: Simulation of Combined Best Management Practices and Low Impact Development for Sustainable Stormwater Management. JAWRA Journal of the American Water Resources Association 46, 907–918. https://doi.org/10.1111/j.1752-1688.2010.00462.x

Davis, A. P., 2007. Field performance of bioretention: Water quality. Environmental Engineering Science, 24(8), 1048– 1063. http://doi.org/10.1089/ees.2006.0190

Davis, A. P., Shokouhian, M., Sharma, H., & Minami, C,. 2006. Water quality improvement through bioretention media: nitrogen and phosphorus removal. Water Environment Research, 78, 284–293. https://doi.org/10.2175/106143005X94376

DeBusk, K.M. and Wynn, T.., 2011. Storm-water bioretention for runoff quality and quantity mitigation. Journal of Environmental Engineering 137 (9), 800–808. doi:10.1061/(ASCE)EE.1943-7870.0000388.

DeBusk, T.A., Grace, K.A., Dierberg, F.E., Jackson, S.D., Chimney, M.J., Gu, B., 2004. An investigation of the limits of phosphorus removal in wetlands: a mesocosm study of a shallow periphyton-dominated treatment system. Ecological Engineering 23, 1–14. https://doi.org/10.1016/j.ecoleng.2004.06.009

Del Giudice, G., Rasulo, G., Siciliano, D., Padulano, R., 2014. Combined Effects of Parallel and Series Detention Basins for Flood Peak Reduction. Water Resour Manage 28, 3193–3205. https://doi.org/10.1007/s11269-014-0668-1

Demissie, M., Khan, A., 1993. Influence of Wetlands on Streamflow in Illinois. Illinois State Water Survey 57.

Dierberg, F.E., Juston, J.J., DeBusk, T.A., Pietro, K., Gu, B., 2005. Relationship between hydraulic efficiency and phosphorus removal in a submerged aquatic vegetation-dominated treatment wetland. Ecological Engineering 25, 9–23. https://doi.org/10.1016/j.ecoleng.2004.12.018

Dixon, S.J., Sear, D.A., Odoni, N.A., Sykes, T., Lane, S.N., 2016. The effects of river restoration on catchment scale flood risk and flood hydrology: The Effects of River Restoration on Catchment Scale Flood Risk. Earth Surf. Process. Landforms 41, 997–1008. https://doi.org/10.1002/esp.3919

Dunne, E.J., Coveney, M.F., Marzolf, E.R., Hoge, V.R., Conrow, R., Naleway, R., Lowe, E.F., Battoe, L.E., 2012. Efficacy of a large-scale constructed wetland to remove phosphorus and suspended solids from Lake Apopka, Florida. Ecological Engineering 42, 90–100. https://doi.org/10.1016/j.ecoleng.2012.01.019

Dunne, E.J., Coveney, M.F., Hoge, V.R., Conrow, R., Naleway, R., Lowe, E.F., Battoe, L.E., Wang, Y., 2015. Phosphorus removal performance of a large-scale constructed treatment wetland receiving eutrophic lake water. Ecological Engineering 79, 132–142. https://doi.org/10.1016/j.ecoleng.2015.02.003

Dutta A, Torres AS, Vojinovic Z., 2021. Evaluation of pollutant removal efficiency by small-scale nature-based solutions focusing on bio-retention cells, vegetative swale and porous pavement. Water 13(17):2361. https://doi.org/10.3390/w13172361

Eckart, K., McPhee, Z., Bolisetti, T., 2017. Performance and Implementation of Low Impact Development–A Review. Science of the Total Environment, vol. 607.,pp. 413–432. https://doi.org/10.1016/j.scitotenv.2017.06.254

Elliott, A.H., Spigel, R.H., Jowett, I.G., Shankar, S.U., Ibbitt, R.P., 2010. Model application to assess effects of urbanisation and distributed flow controls on erosion potential and baseflow hydraulic habitat. Urban Water Journal 7, 91–107. https://doi.org/10.1080/15730620903447605

Emerson, C.H., Welty, C., Traver, R.G., 2005. Watershed-Scale Evaluation of a System of Storm Water Detention Basins. J. Hydrol. Eng. 10, 237–242. https://doi.org/10.1061/(ASCE)1084-0699(2005)10:3(237)

Evenson, G., Golden, H., Lane, C., McLaughlin, D., D'Amico, E., 2018. Depressional wetlands affect watershed hydrological, biogeochemical, and ecological functions. Ecol Appl 28, 953–966. https://doi.org/10.1002/eap.1701

Fan, C., Li, J., 2004. A Modelling Analysis of Urban Stormwater Flow Regimes and their Implication for Stream Erosion. Water Quality Research Journal 39, 356–361. https://doi.org/10.2166/wqrj.2004.048

Federal Geographic Data Committee, 2013. Classification of wetlands and deepwater habitats of the United States FGDC-STD-004-2013.

Ferguson, B.K., 1995. DOWNSTREAM HYDROGRAPHIC EFFECTS OF URBAN STORMWATER DETENTION AND INFILTRATION. Proceedings of the 1995 Georgia Water Resources Conference 4.

Fink, D.F., Mitsch, W.J., 2004. Seasonal and storm event nutrient removal by a created wetland in an agricultural watershed. Ecological Engineering 23, 313–325. https://doi.org/10.1016/j.ecoleng.2004.11.004

Fitzpatrick, F.A., Diebel, M.W., Harris, M.A., Arnold, T.L., Lutz, M.A., Richards, K.D., 2005. Effects of Urbanization on the Geomorphology, Habitat, Hydrology, and Fish Index of Biotic Integrity of Streams in the Chicago Area, Illinois and Wisconsin 30.

Flegel, A., Byard, G., McConkey, S., Hanstad, C., Gaynor, N., Zaloudek, Z., 2019. Watershed-specific release rates analysis: Cook County, Illinois (No. 2019– 06). Illinois State Water Survey Prairie Research Institute.

Gaborit, E., Muschalla, D., Vallet, B., Vanrolleghem, P.A., Anctil, F., 2013. Improving the performance of stormwater detention basins by real-time control using rainfall forecasts. Urban Water Journal 10, 230–246. https://doi.org/10.1080/1573062X.2012.726229

Garcia-Cuerva, L., Berglund, E.Z., Rivers, L., 2018. An integrated approach to place Green Infrastructure strategies in marginalized communities and evaluate stormwater mitigation. Journal of Hydrology 559, 648–660. https://doi.org/10.1016/j.jhydrol.2018.02.066

Gessner, T.P., Kadlec, R.H., Reaves, R.P., 2005. Wetland remediation of cyanide and hydrocarbons. Ecological Engineering 25, 457–469. https://doi.org/10.1016/j.ecoleng.2005.07.015

Glass, C., Bissouma, S., 2005. Evaluation of a parking lot bioretention cell for removal of stormwater pollutants. WIT Transactions on Ecology and the Environment 81, 10.

Goff, K.M., Gentry, R.W., 2006. The Influence of Watershed and Development Characteristics on the Cumulative Impacts of Stormwater Detention Ponds. Water Resour Manage 20, 829–860. https://doi.org/10.1007/s11269-005-9010-2

Guerrero, J., Mahmoud, A., Alam, T., Chowdhury, M.A., Adetayo, A., Ernest, A., Jones, K.D., 2020. Water Quality Improvement and Pollutant Removal by Two Regional Detention Facilities with Constructed Wetlands in South Texas. Sustainability 12, 2844. https://doi.org/10.3390/su12072844

Guo, Q., 1997. Sediment and Heavy Metal Accumulation in Dry Storm Water Detention Basin. Journal of Water Resources Planning and Management 123, 295–301. https://doi.org/10.1061/(ASCE)0733-9496(1997)123:5(295)

Harper, H.H., Herr, J.L., Baker, D., Livingston, E.H., 1999. Performance evaluation of dry detention stormwater management systems. Sixth Biennial Stormwater Research & Watershed Management Conference.

Hathaway, J.M., Hunt, W.F., 2010. Evaluation of Storm-Water Wetlands in Series in Piedmont North Carolina. J. Environ. Eng. 136, 140–146. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000130

Hawley, R.J., and Bledsoe, B.P., 2013. Channel enlargement in semiarid suburbanizing watersheds: A southern California case study. Journal of Hydrology 496, 17–30. https://doi.org/10.1016/j.jhydrol.2013.05.010

Hawley, R.J., and Vietz, G.J., 2016. Addressing the urban stream disturbance regime. Freshwater Science 35, 278–292. https://doi.org/10.1086/684647

Hawley, R.J., Wooten, M.S., Vatter, B.C., Onderak, E., Lachniet, M.J., Schade, T., Grant, G., Groh, B., DelVerne, J., 2012. Integrating stormwater controls designed for channel protection, water quality, and inflow/infiltration mitigation in two pilot watersheds to restore a more natural flow regime in urban streams. Watershed Science Bulletin: Journal of the Association of Watershed & Stormwater Professionals 3, 16.

Hawley, R.J., MacMannis, K.R., Wooten, M.S., 2013. Bed coarsening, riffle shortening, and channel enlargement in urbanizing watersheds, northern Kentucky, USA. Geomorphology 201, 111–126. https://doi.org/10.1016/j.geomorph.2013.06.013

Hawley, R.J., MacMannis, K.R., Wooten, M.S., Fet, E.V., Korth, N.L., 2020. Suburban stream erosion rates in northern Kentucky exceed reference channels by an order of magnitude and follow predictable trajectories of channel evolution. Geomorphology 352, 106998. https://doi.org/10.1016/j.geomorph.2019.106998

Healy, M., Cawley, A.M., 2002. Nutrient Processing Capacity of a Constructed Wetland in Western Ireland. J. Environ. Qual. 31, 1739–1747. https://doi.org/10.2134/jeq2002.1739

Hess, G.W., Inman, E.J., 1994. Effects of urban flood-detention reservoirs on peak discharges and flood frequencies, and simulation of flood-detention reservoir outflow hydrographs in two watersheds in Albany, Georgia. Us Geological Survey Water Resources Investigations Report 94-4158 37.

Hey, D.L., Kenimer, A.L., Barrett, K.R., 1994. Water quality improvement by four experimental wetlands. Ecological Engineering 3, 381–397. https://doi.org/10.1016/0925-8574(94)00008-5

Hoffmann, C.C., Heiberg, L., Audet, J., Schønfeldt, B., Fuglsang, A., Kronvang, B., Ovesen, N.B., Kjaergaard, C., Hansen, H.C.B., Jensen, H.S., 2012. Low phosphorus release but high nitrogen removal in two restored riparian wetlands inundated with agricultural drainage water. Ecological Engineering 46, 75–87. https://doi.org/10.1016/j.ecoleng.2012.04.039

Hood, M.J., Clausen, J.C., Warner, G.S., 2007. Comparison of Stormwater Lag Times for Low Impact and Traditional Residential Development. J Am Water Resources Assoc 43, 1036–1046. https://doi.org/10.1111/j.1752-1688.2007.00085.x

Hopkins, K.G., Loperfido, J.V., Craig, L.S., Noe, G.B., and Hogan, D. M., 2017, Comparison of sediment and nutrient export and runoff characteristics from watersheds with centralized versus distributed stormwater management, Journal of Environmental Management, 203, 286-298, http://dx.doi.org/10.1016/j.jenvman.2017.07.067

Hopkins, K.G., Bhaskar, A.S., Woznicki, S.A., Fanelli, R.M., 2020. Changes in event-based streamflow magnitude and timing after suburban development with infiltration-based stormwater management. Hydrological Processes 34, 387–403. https://doi.org/10.1002/hyp.13593

Horne, A.J., 2001. Potential value of constructed wetlands for nitrate removal along some large and small rivers. SIL Proceedings, 1922-2010 27, 4057–4062. https://doi.org/10.1080/03680770.1998.11901757
Hossain, M.A., Alam, M., Yonge, D.R., Dutta, P., 2005. Efficiency and Flow Regime of a Highway Stormwater Detention Pond in Washington, USA. Water Air Soil Pollut 164, 79–89. https://doi.org/10.1007/s11270-005-2250-1
House, L.B., Waschbusch, R.J., Hughes, P.E., 1993. Water quality of an urban wet detention pond in Madison, Wisconsin, 1987-1988 (Open-File Report No. 93–172), Open-File Report. U.S. Geological Survey.
Huang, J., 2000. Nitrogen removal in constructed wetlands employed to treat domestic wastewater. Water Research 34, 2582–2588. https://doi.org/10.1016/S0043-1354(00)00018-X
Hunt, W.F., Jarrett, A.R., Smith, J.T., Sharkey, L.J., 2006. Evaluating Bioretention Hydrology and Nutrient Removal at Three Field Sites in North Carolina. J. Irrig. Drain Eng. 132, 600–608. https://doi.org/10.1061/(ASCE)0733-9437(2006)132:6(600)
Hunt, W.F., Smith, J.T., Jadlocki, S.J., Hathaway, J.M., Eubanks, P.R., 2008. Pollutant Removal and Peak Flow Mitigation by a Bioretention Cell in Urban Charlotte, N.C. J. Environ. Eng. 134, 403–408. https://doi.org/10.1061/(ASCE)0733-9372(2008)134:5(403)
Hussain, C.F., Brand, J., Gulliver, J.S., Weiss, P.T., 2005. Water Quality Performance of Dry Detention Ponds with Under-Drains (No. MN/RC-2005-23). Minnesota Department of Transportation.
James, M.B., Dymond, R.L., 2012. Bioretention Hydrologic Performance in an Urban Stormwater Network. J. Hydrol. Eng. 17, 431–436. https://doi.org/10.1061/(ASCE)HE.1943-5584.0000448
Jamrussri, S., Toda, Y., 2017. Simulating past severe flood events to evaluate the effectiveness of nonstructural flood countermeasures in the upper Chao Phraya River Basin, Thailand. Journal of Hydrology: Regional Studies 10, 82–94. https://doi.org/10.1016/j.ejrh.2017.02.001
Jarden, K.M., Jefferson, A.J., Grieser, J.M., 2016. Assessing the effects of catchment-scale urban green infrastructure retrofits on hydrograph characteristics: Hydrologic Effects of Catchment-Scale Green Infrastructure Retrofits. Hydrol. Process. 30, 1536–1550. https://doi.org/10.1002/hyp.10736
Javaheri, A., Babbar-Sebens, M., 2014. On comparison of peak flow reductions, flood inundation maps, and velocity maps in evaluating effects of restored wetlands on channel flooding. Ecological Engineering 73, 132–145. https://doi.org/10.1016/j.ecoleng.2014.09.021
Jayakaran, A.D., Knappenberger, T., Stark, J.D., Hinman, C., 2019. Remediation of stormwater pollutants by porous asphalt pavement. Water 11, 520. https://doi.org/10.3390/w11030520
Jordan, T.E., Whigham, D.F., Hofmockel, K.H., Pittek, M.A., 2003. Nutrient and Sediment Removal by a Restored Wetland Receiving Agricultural Runoff. Journal of Environment Quality 32, 1534. https://doi.org/10.2134/jeq2003.1534
Kadlec, R.H., 2003. Pond and wetland treatment. Water Science and Technology 48, 1–8. https://doi.org/10.2166/wst.2003.0266
Kadlec, R., 2016. Large Constructed Wetlands for Phosphorus Control: A Review. Water 8, 243. https://doi.org/10.3390/w8060243
Kadlec, R.H., Cuvellier, C., Stober, T., 2010. Performance of the Columbia, Missouri, treatment wetland. Ecological Engineering 36, 672–684. https://doi.org/10.1016/j.ecoleng.2009.12.009
Karuppasamy, E., Postel, N., Pomeroy, C.A., Jacobs, T.A., 2009. The Impact of Smaller Detention Basins on Flood Hazard Areas in Lenexa, Kansas, in: World Environmental and Water Resources Congress 2009. Presented at the World Environmental and Water Resources Congress 2009, American Society of Civil Engineers, Kansas City, Missouri, United States, pp. 1–10. https://doi.org/10.1061/41036(342)459
Kohler, E.A., Poole, V.L., Reicher, Z.J., Turco, R.F., 2004. Nutrient, metal, and pesticide removal during storm and nonstorm events by a constructed wetland on an urban golf course. Ecological Engineering 23, 285–298. https://doi.org/10.1016/j.ecoleng.2004.11.002
Kohn, M.S., Fulton, J.W., Williams, C.A., Stogner, Sr., R.W., 2014. Remediation scenarios for attenuating peak flows and reducing sediment transport in Fountain Creek, Colorado, 2013 (Scientific Investigations Report), Scientific Investigations Report. U.S. Geological Survey.
Koskiaho, J., 2003. Flow velocity retardation and sediment retention in two constructed wetland-ponds. Ecological Engineering 13.

Kovacic, D.A., David, M.B., Gentry, L.E., Starks, K.M., Cooke, R.A., 2000. Effectiveness of Constructed Wetlands in Reducing Nitrogen and Phosphorus Export from Agricultural Tile Drainage. J. environ. qual. 29, 1262–1274. https://doi.org/10.2134/jeq2000.00472425002900040033x
Kovacic, D.A., Twait, R.M., Wallace, M.P., Bowling, J.M., 2006. Use of created wetlands to improve water quality in the Midwest—Lake Bloomington case study. Ecological Engineering 28, 258–270. https://doi.org/10.1016/j.ecoleng.2006.08.002
Land, M., Granéli, W., Grimvall, A., Hoffmann, C.C., Mitsch, W.J., Tonderski, K.S., Verhoeven, J.T.A., 2016. How effective are created or restored freshwater wetlands for nitrogen and phosphorus removal? A systematic review. Environ Evid 5, 9. https://doi.org/10.1186/s13750-016-0060-0
Lenhart, H.A., Hunt, W.F., 2011. Evaluating Four Storm-Water Performance Metrics with a North Carolina Coastal Plain Storm-Water Wetland. J. Environ. Eng. 137, 155–162. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000307
Li, H., Davis, A.P., 2009. Water Quality Improvement through Reductions of Pollutant Loads Using Bioretention. J. Environ. Eng. 135, 567–576. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000026
Li, J., Fan, C., 2010. Evaluation of Stormwater Management Practices for Stream Erosion, in: World Environmental and Water Resources Congress 2010. Presented at the World Environmental and Water Resources Congress 2010, American Society of Civil Engineers, Providence, Rhode Island, United States, pp. 3237–3247. https://doi.org/10.1061/41114(371)333
Loperfido, J.V., Noe, G.B., Jarnagin, S.T., Hogan, D.M., 2014. Effects of distributed and centralized stormwater best management practices and land cover on urban stream hydrology at the catchment scale. Journal of Hydrology 519, 2584–2595. https://doi.org/10.1016/j.jhydrol.2014.07.007
Lu, S., Zhang, P., Jin, X., Xiang, C., Gui, M., Zhang, J., Li, F., 2009. Nitrogen removal from agricultural runoff by full-scale constructed wetland in China. Hydrobiologia 621, 115–126. https://doi.org/10.1007/s10750-008-9636-1
Luederitz, V., Eckert, E., Lange-Weber, M., Lange, A., Gersberg, R.M., 2001. Nutrient removal efficiency and resource economics of vertical flow and horizontal flow constructed wetlands. Ecological Engineering 18, 157–171. https://doi.org/10.1016/S0925-8574(01)00075-1
Mallin, M.A., Ensign, S.H., Wheeler, T.L., Mayes, D.B., 2002. Pollutant Removal Efficacy of Three Wet Detention Ponds. J. ENVIRON. QUAL. 31, 7.
Martin, E.H., 1988. Effectiveness of an Urban Runoff Detention Pond-Wetlands System. Journal of Environmental Engineering 114, 810–827. https://doi.org/10.1061/(ASCE)0733-9372(1988)114:4(810)
McCuen, R.H., 1974. A regional approach to urban storm water detention. Geophysical Research Letters 1, 321–322. https://doi.org/10.1029/GL001i007p00321
McCuen, R.H., 1979. Downstream Effects of Stormwater Management Basins. J. Hydr. Div. 105, 1343–1356. https://doi.org/10.1061/JYCEAJ.0005300
Meuleman, A.F.M., Beekman, J.Ph., Verhoeven, J.T.A., 2002. Nutrient retention and nutrient-use efficiency in Phragmites australis stands after wastewater application. Wetlands 22, 712–721. https://doi.org/10.1672/0277-5212(2002)022[0712:NRANUE]2.0.CO;2
Middleton, J.R., Barrett, M.E., 2008. Water Quality Performance of a Batch-Type Stormwater Detention Basin. Water Environment Research 80, 172–178. https://doi.org/10.2175/106143007X220842
Mitchell, N., Kumarasamy, K., Cho, S., Belmont, P., Dalzell, B., Gran, K., 2018. Reducing High Flows and Sediment Loading through Increased Water Storage in an Agricultural Watershed of the Upper Midwest, USA. Water 10, 1053. https://doi.org/10.3390/w10081053
Mitsch, W.J., Cronk, J.K., Wu, X., Nairn, R.W., Hey, D.L., 1995. Phosphorus Retention in Constructed Freshwater Riparian Marshes. Ecological Applications 5, 830–845. https://doi.org/10.2307/1941991
Mitsch, W.J., Day, J.W., Zhang, L., Lane, R.R., 2005. Nitrate-nitrogen retention in wetlands in the Mississippi River Basin. Ecological Engineering 24, 267–278. https://doi.org/10.1016/j.ecoleng.2005.02.005
Morse, N.R., McPhillips, L.E., Shapleigh, J.P., Walter, M.T., 2017. The Role of Denitrification in Stormwater Detention Basin Treatment of Nitrogen. Environ. Sci. Technol. 51, 7928–7935. https://doi.org/10.1021/acs.est.7b01813

Moustafa, M.Z., 1999. Nutrient retention dynamics of the Everglades Nutrient Removal Project. Wetlands 19, 689–704. https://doi.org/10.1007/BF03161705

Moustafa, M.Z., Chimney, M.J., Fontaine, T.D., Shih, G., Davis, S., 1996. The response of a freshwater wetland to long-term "low level" nutrient loads - marsh efficiency. Ecological Engineering 7, 15–33. https://doi.org/10.1016/0925-8574(95)00063-1

Moustafa, M.Z., Fontaine, T.D., Guardo, M., James, R.T., 1998. The response of a freshwater wetland to long-term 'low level' nutrient loads: nutrients and water budget. Hydrobiologia 13.

Mullapudi, A., Bartos, M., Wong, B., Kerkez, B., 2018. Shaping Streamflow Using a Real-Time Stormwater Control Network. Sensors 18, 2259. https://doi.org/10.3390/s18072259

Nairn, R.W., Mitsch, W.J., 1999. Phosphorus removal in created wetland ponds receiving river overflow. Ecological Engineering 14, 107–126. https://doi.org/10.1016/S0925-8574(99)00023-3

Natarajan, P., Davis, A.P., 2015. Hydrologic Performance of a Transitioned Infiltration Basin Managing Highway Runoff. J. Sustainable Water Built Environ. 1, 04015002. https://doi.org/10.1061/JSWBAY.0000797

Naylor, S., Brisson, J., Labelle, M.A., Drizo, A., Comeau, Y., 2003. Treatment of freshwater fish farm effluent using constructed wetlands: the role of plants and substrate. Water Science and Technology 48, 215–222. https://doi.org/10.2166/wst.2003.0324

Niswander, S.F., Mitsch, W.J., 1995. Functional analysis of a two-year-old created in-stream wetland: Hydrology, phosphorus retention, and vegetation survival and growth. Wetlands 15, 212–225. https://doi.org/10.1007/BF03160701

Oberts, G.L., Osgood, R.A., 1991. Water-quality effectiveness of a detention/wetland treatment system and its effect on an urban lake. Environmental Management 15, 131–138. https://doi.org/10.1007/BF02393844

Ogawa, H., Male, J.W., 1986. Simulating the Flood Mitigation Role of Wetlands. Journal of Water Resources Planning and Management 112, 114–128. https://doi.org/10.1061/(ASCE)0733-9496(1986)112:1(114)

Ogden, F.L., Raj Pradhan, N., Downer, C.W., Zahner, J.A., 2011. Relative importance of impervious area, drainage density, width function, and subsurface storm drainage on flood runoff from an urbanized catchment: FLOODING IN URBANIZED CATCHMENTS. Water Resour. Res. 47. https://doi.org/10.1029/2011WR010550

Pagotto, C., Legret, M., & Le Cloirec, P., 2000. Comparison of the hydraulic behavior and the quality of highway runoff water according to the type of pavement. Water Research, 34(18), 4446–4454. https://doi.org/10.1016/S0043-1354(00)00221-9

Pennino, M.J., McDonald, R.I., Jaffe, P.R., 2016. Watershed-scale impacts of stormwater green infrastructure on hydrology, nutrient fluxes, and combined sewer overflows in the mid-Atlantic region. Science of The Total Environment 565, 1044–1053. https://doi.org/10.1016/j.scitotenv.2016.05.101

Rai, U.N., Tripathi, R.D., Singh, N.K., Upadhyay, A.K., Dwivedi, S., Shukla, M.K., Mallick, S., Singh, S.N., Nautiyal, C.S., 2013. Constructed wetland as an ecotechnological tool for pollution treatment for conservation of Ganga river. Bioresource Technology 148, 535–541. https://doi.org/10.1016/j.biortech.2013.09.005

Raisin, G.W., Mitchell, D.S., Croome, R.L., 1997. The effectiveness of a small constructed wetland in ameliorating diffuse nutrient loadings from an Australian rural catchment. Ecological Engineering 9, 19–35. https://doi.org/10.1016/S0925-8574(97)00016-5

Ravazzani, G., Gianoli, P., Meucci, S., Mancini, M., 2014. Assessing Downstream Impacts of Detention Basins in Urbanized River Basins Using a Distributed Hydrological Model. Water Resour Manage 28, 1033–1044. https://doi.org/10.1007/s11269-014-0532-3

Rodríguez, M., Brisson, J., 2015. Pollutant removal efficiency of native versus exotic common reed (Phragmites australis) in North American treatment wetlands. Ecological Engineering 74, 364–370. https://doi.org/10.1016/j.ecoleng.2014.11.005

Rohrer, C.A., Postel, N.A., O'Neill, P.A., Roesner, L.A., 2005. Development of Design Criteria for Regional Stormwater Management Facilities to Maintain Geomorphic Stability in Cedar Creek, in: Impacts of Global Climate Change. Presented at the World Water and Environmental Resources Congress 2005, American Society of Civil Engineers, Anchorage, Alaska, United States, pp. 1–10. https://doi.org/10.1061/40792(173)112

Roseen, R. M., Ballestero, T. P., Houle, J. J., Avelleneda, P., Wildey, R., and Briggs, J., 2006. Stormwater low-impact development, conventional structural, and manufactured treatment strategies for parking lot runoff. Transportation Research Record: Journal of the Transportation Research Board, Washington, D.C.: Transportation Research Board of the National Academies, vol. 1984, 135–147.

Rosenzweig, B.R., Smith, J.A., Baeck, M.L., Jaffé, P.R., 2011. Monitoring Nitrogen Loading and Retention in an Urban Stormwater Detention Pond. J. Environ. Qual. 40, 598–609. https://doi.org/10.2134/jeq2010.0300

Schulz, R., Peall, S.K.C., 2001. Effectiveness of a Constructed Wetland for Retention of Nonpoint-Source Pesticide Pollution in the Lourens River Catchment, South Africa. Environ. Sci. Technol. 35, 422–426. https://doi.org/10.1021/es0001198

Selbig, W.R., Bannerman, R.T., 2008. A comparison of runoff quantity and quality from two small basins undergoing implementation of conventional and low impact development (LID) strategies: Cross Plains, Wisconsin, water years 1999-2005 (Scientific Investigations Report No. 2008–5008), Scientific Investigations Report. U.S. Geological Survey.

Semadeni-Davies, A., 2006. Winter performance of an urban stormwater pond in southern Sweden. Hydrological Processes 20, 165–182. https://doi.org/10.1002/hyp.5909

Senzia, M.A., Mashauri, D.A., Mayo, A.W., 2003. Suitability of constructed wetlands and waste stabilisation ponds in wastewater treatment: nitrogen transformation and removal. Physics and Chemistry of the Earth, Parts A/B/C 28, 1117–1124. https://doi.org/10.1016/j.pce.2003.08.033

Sharior, S., McDonald, W., Parolari, A.J., 2019. Improved reliability of stormwater detention basin performance through water quality data-informed real-time control. Journal of Hydrology 573, 422–431. https://doi.org/10.1016/j.jhydrol.2019.03.012

Sim, C.H., Yusoff, M.K., Shutes, B., Ho, S.C., Mansor, M., 2008. Nutrient removal in a pilot and full scale constructed wetland, Putrajaya city, Malaysia. Journal of Environmental Management 88, 307–317.

Smakhtin, V.U., Batchelor, A.L., 2005. Evaluating wetland flow regulating functions using discharge time-series. Hydrol. Process. 19, 1293–1305. https://doi.org/10.1002/hyp.5555

Sohn, W., Kim, J.-H., Li, M.-H., Brown, R.D., Jaber, F.H., 2020. How does increasing impervious surfaces affect urban flooding in response to climate variability? Ecological Indicators 118, 106774. https://doi.org/10.1016/j.ecolind.2020.106774

Sønderup, M.J., Egemose, S., Hansen, A.S., Grudinina, A., Madsen, M.H., Flindt, M.R., 2016. Factors affecting retention of nutrients and organic matter in stormwater ponds: Nutrients and Organic Matter in Stormwater Ponds. Ecohydrol. 9, 796–806. https://doi.org/10.1002/eco.1683

Soong, D.T., Murphy, E.A., Straub, T.D., 2009. Effect of detention basin release rates on flood flows- Application of a model to the Blackberry Creek watershed in Kane County, Illinois (Scientific Investigations Report No. 2009–5106), Scientific Investigations Report. U.S. Geological Survey.

Spieles, D.J., Mitsch, W.J., 1999. The effects of season and hydrologic and chemical loading on nitrate retention in constructed wetlands: a comparison of low- and high-nutrient riverine systems. Ecological Engineering 14, 77–91. https://doi.org/10.1016/S0925-8574(99)00021-X

Stanley, D.W., 1996. Pollutant removal by a stormwater dry detention pond. Water Environment Research 68, 1076–1083. https://doi.org/10.2175/106143096X128072

Stormwater Academy, 2010. Poppleton Creek Wet Detention Pond. BMP Trains Research and Publications 25.

Su, D., Fang, X., Fang, Z., 2010. Effectiveness and Downstream Impacts of Stormwater Detention Ponds Required for Land Development, in: World Environmental and Water Resources Congress 2010. Presented at the World Environmental and Water Resources Congress 2010, American Society of Civil Engineers, Providence, Rhode Island, United States, pp. 3071–3081. https://doi.org/10.1061/41114(371)314

Tang, Y., Leon, A.S., Kavvas, M.L., 2020. Impact of Size and Location of Wetlands on Watershed-Scale Flood Control. Water Resour Manage 34, 1693–1707. https://doi.org/10.1007/s11269-020-02518-3

Tanner, C.C., Clayton, J.S., Upsdell, M.P., 1995. Effect of loading rate and planting on treatment of dairy farm wastewaters in constructed wetlands—I. Removal of oxygen demand, suspended solids and fecal coliforms. Water Research 29, 17–26. https://doi.org/10.1016/0043-1354(94)00139-X

Tao, J., Li, Z., Peng, X., Ying, G., 2017. Quantitative analysis of impact of green stormwater infrastructures on combined sewer overflow control and urban flooding control. Front. Environ. Sci. Eng. 11, 11. https://doi.org/10.1007/s11783-017-0952-4

Thomas, N.W., Arenas Amado, A., Schilling, K.E., Weber, L.J., 2016. Evaluating the efficacy of distributed detention structures to reduce downstream flooding under variable rainfall, antecedent soil, and structural storage conditions. Advances in Water Resources 96, 74–87. https://doi.org/10.1016/j.advwatres.2016.07.002

Tillinghast, E.D., Hunt, W.F., Jennings, G.D., 2011. Stormwater control measure (SCM) design standards to limit stream erosion for Piedmont North Carolina. Journal of Hydrology 411, 185–196. https://doi.org/10.1016/j.jhydrol.2011.09.027

Vymazal, J., Kröpfelová, L., 2009. Removal of nitrogen in constructed wetlands with horizontal sub-surface flow: a review. Wetlands 29, 1114–1124. https://doi.org/10.1672/08-216.1

Walsh, F.M., Barrett, M.E., Malina, J.F., Charbeneau, R.J., 1998. Use of Vegetative Controls for Treatment of Highway Runoff (No. FHWA/TX-7-2954-2). Center for Transportation Research, Austin, TX.

Wang, X., Shang, S., Qu, Z., Liu, T., Melesse, A.M., Yang, W., 2010. Simulated wetland conservation-restoration effects on water quantity and quality at watershed scale. Journal of Environmental Management 91, 1511–1525. https://doi.org/10.1016/j.jenvman.2010.02.023

Webber, J.L., Fletcher, T.D., Cunningham, L., Fu, G., Butler, D., Burns, M.J., 2020. Is green infrastructure a viable strategy for managing urban surface water flooding? Urban Water Journal 17, 598–608. https://doi.org/10.1080/1573062X.2019.1700286

Williams, E.S., Wise, W.R., 2006. HYDROLOGIC IMPACTS OF ALTERNATIVE APPROACHES TO STORM WATER MANAGEMENT AND LAND DEVELOPMENT. J Am Water Resources Assoc 42, 443–455. https://doi.org/10.1111/j.1752-1688.2006.tb03849.x

Wilson, C.E., Hunt, W.F., Winston, R.J., Smith, P., 2015. Comparison of runoff quality and quantity from a commercial low-impact and conventional development in Raleigh, North Carolina. J. Environ. Eng. 141 (2):5014005. http://dx.doi.org/10.1061/(ASCE)EE.1943-7870.0000842.

Wissler, A.D., Hunt, W.F., McLaughlin, R.A., 2020a. Water Quality and Hydrologic Performance of Two Dry Detention Basins Receiving Highway Stormwater Runoff in the Piedmont Region of North Carolina. J. Sustainable Water Built Environ. 6, 05020002. https://doi.org/10.1061/JSWBAY.0000915

Wissler, A.D., Hunt, W.F., McLaughlin, R.A., 2020b. Hydrologic and water quality performance of two aging and unmaintained dry detention basins receiving highway stormwater runoff. Journal of Environmental Management 255, 109853. https://doi.org/10.1016/j.jenvman.2019.109853

Wu, J.S., 1989. Evaluation of detention basin performance in the Piedmont region of North Carolina (No. 248). Water Research Institute of the University of North Carolina Chapel Hill.

Wu, J.S., Holman, R.E., Dorney, J.R., 1996. Systematic Evaluation of Pollutant Removal by Urban Wet Detention Ponds. Journal of Environmental Engineering 122, 983–988. https://doi.org/10.1061/(ASCE)0733-9372(1996)122:11(983)

Wu, Y., Zhang, G., Rousseau, A.N., Xu, Y.J., 2020a. Quantifying streamflow regulation services of wetlands with an emphasis on quickflow and baseflow responses in the Upper Nenjiang River Basin, Northeast China. Journal of Hydrology 583, 124565. https://doi.org/10.1016/j.jhydrol.2020.124565

Wu, Y., Zhang, G., Rousseau, A.N., Xu, Y.J., Foulon, É., 2020b. On how wetlands can provide flood resilience in a large river basin: A case study in Nenjiang river Basin, China. Journal of Hydrology 587, 125012. https://doi.org/10.1016/j.jhydrol.2020.125012

Yang, B., Li, S., 2013. Green Infrastructure Design for Stormwater Runoff and Water Quality: Empirical Evidence from Large Watershed-Scale Community Developments. Water 5, 2038–2057. https://doi.org/10.3390/w5042038

Yang, G., Bowling, L.C., Cherkauer, K.A., Pijanowski, B.C., 2011. The impact of urban development on hydrologic regime from catchment to basin scales. Landscape and Urban Planning 103, 237–247. https://doi.org/10.1016/j.landurbplan.2011.08.003

Zahmatkesh, Z., Burian, S.J., Karamouz, M., Tavakol-Davani, H., Goharian, E., 2015. Low-Impact Development Practices to Mitigate Climate Change Effects on Urban Stormwater Runoff: Case Study of New York City. J. Irrig. Drain Eng. 141, 04014043. https://doi.org/10.1061/(ASCE)IR.1943-4774.0000770

Zimmer, C.A., Heathcote, I.W., Whiteley, H.R., Schroter, H., 2007. Low-Impact-Development Practices for Stormwater: Implications for Urban Hydrology. Canadian Water Resources Journal 32, 193–212. https://doi.org/10.4296/cwrj3203193

Chapter 6. Impact of Volume Control and Detention Practices on the Groundwater of Cook County [WMO Article 208.4]

6.1 Background

Urbanization transforms areas of vegetated soils to impervious surfaces, a transition that leads to increased volumes of stormwater runoff. When runoff cannot infiltrate into natural soil horizons, communities design a variety of measures to direct the stormwater and route it to groundwater (Freeborn et al., 2012). Designed stormwater routing is critical to prevent flooding, especially in a flat landscape such as northeastern Illinois without gravity-driven stream networks to route overland flow (Lai & Anders, 2018). Despite the necessity of infrastructure to route stormwater, this runoff water can negatively impact freshwater ecosystems and potentially degrade the groundwater drinking water supply of downgradient communities, as is the case south and west of Cook County. Indeed, a U.S. Geological Survey (USGS) summary of impacts of green infrastructure to the Great Lakes catchments calls the potential infiltration of stormwater contaminants to groundwater "one of the greatest potential negative consequences of green infrastructure" (USGS, 2022). To understand the impact of stormwater infrastructure on groundwater resources in Cook County apart from the Chicago proper, the Metropolitan Water Reclamation District of Greater Chicago (MWRD) has sponsored this Illinois State Water Survey (ISWS) review of published studies relevant to the topic.

By the early 2000s, most communities in Cook County using groundwater had switched water supply sources to Lake Michigan; shallow groundwater use is mostly restricted to areas of southernmost and northwestern Cook County. Citizens of Cook County interact with groundwater indirectly through recreational activities: fishing, swimming, and visiting parks in the region with ecosystems dependent on groundwater. Sufficiently uncontaminated and abundant groundwater recharge is important for wetland and river ecosystems in Cook County. Howard & Gerber (2018) discuss at length how contaminants in northeast Illinois groundwater can easily be mobilized and reemerge in discharge to surface water, contributing to eutrophication in aquatic ecosystems and potentially becoming a problem for water supplies. Northeast Illinois is home to sensitive species dependent on groundwater seepage to wetland habitats, such as the endangered Hines Emerald Dragonfly (Kay et al., 2018) and Blanding's turtle. In their fundamental review, Pasterski and coauthors (2020) found that declining water quality could help explain the population decline of many bivalve, snail, and fish species in wetlands throughout Cook County. High chloride concentrations in groundwater seepage can damage wetlands through salinization of soil zones, rendering the habitat preferentially livable for salt-tolerant species (Panno, et al., 1999; Chicago Tribune, 2021), such as cattails and invasive species of phragmites. Indeed, invasive species are an ongoing problem to the Chicago metropolitan region. A 2018 study found that invasive species are present in 99% of the region's wetlands and that 35% of total species were invasive (Skutely and Matthews, 2018). Additionally, regional shallow groundwater flow in Cook County moves toward the southwest suburbs, an area with communities still dependent on groundwater for water supply (Abrams and Cullen, 2020). To understand whether infiltrating stormwater to groundwater in order to reduce contaminant loading in surface water leads to undesirable groundwater quality and ecological impacts in the MWRD region, the ISWS strongly recommends monitoring and sampling groundwater throughout the area.

In Cook County, the aquifer closest to the surface and most likely to be affected by stormwater contamination is the interconnected Silurian dolomite and overlying glacial sands aquifer system, referred to collectively as the shallow aquifer. Most of the county is underlain by the Silurian dolomite (Willman, 1973), a geologic unit where water is most transmissive in fractures throughout the rock (Leetaru et al., 2004; Roadcap et al., 1993). Fractures in the dolomite are typically largest near the bedrock surface and thin with depth (Roadcap et al., 1993; Muldoon et al., 2001). Preferential flow through the fractures are the fastest transit pathways within the rock. Primary permeability in the bedrock is generally small but plays an important role in serving as a reservoir for potential contaminants (Roadcap et al., 1993).

The overlying glacial sediments are typically 0 to 200 feet thick through the county (United States Geological Survey, 2018). Sands can be locally abundant within stream and river valleys in the region (Morrow & Sharpe, 2009). Glacial sediments are thinnest in the east-central and southern parts of the county in the area formerly occupied by a post-glacial lake, are frequently less than 75 feet thick, and are primarily fine-grained materials (Leetaru et al., 2004), with the thinnest areas where the glacial lake burst and eroded other sediments (Buschbach & Heim, 1972). The distribution of sands and fine-grained materials is important as the Silurian aquifer is generally not transmissive where overlain by fine-grained materials (Roadcap et al., 1993; Csallany & Walton, 1963), whereas the presence of basal sands indicates higher shallow aquifer transmissivity and well productivity (Csallany and Walton, 1963). Figure 84 shows the spatial distribution of basal sands overlying bedrock and the specific capacity of wells, a proxy for their productivity, determined from well-installation reports in the ISWS wells database. The more transmissive areas on this map are largely consistent with the fastest recharge rates observed in Tunnel and Reservoir Plan (TARP) monitoring wells in the area (Kay, 2016).

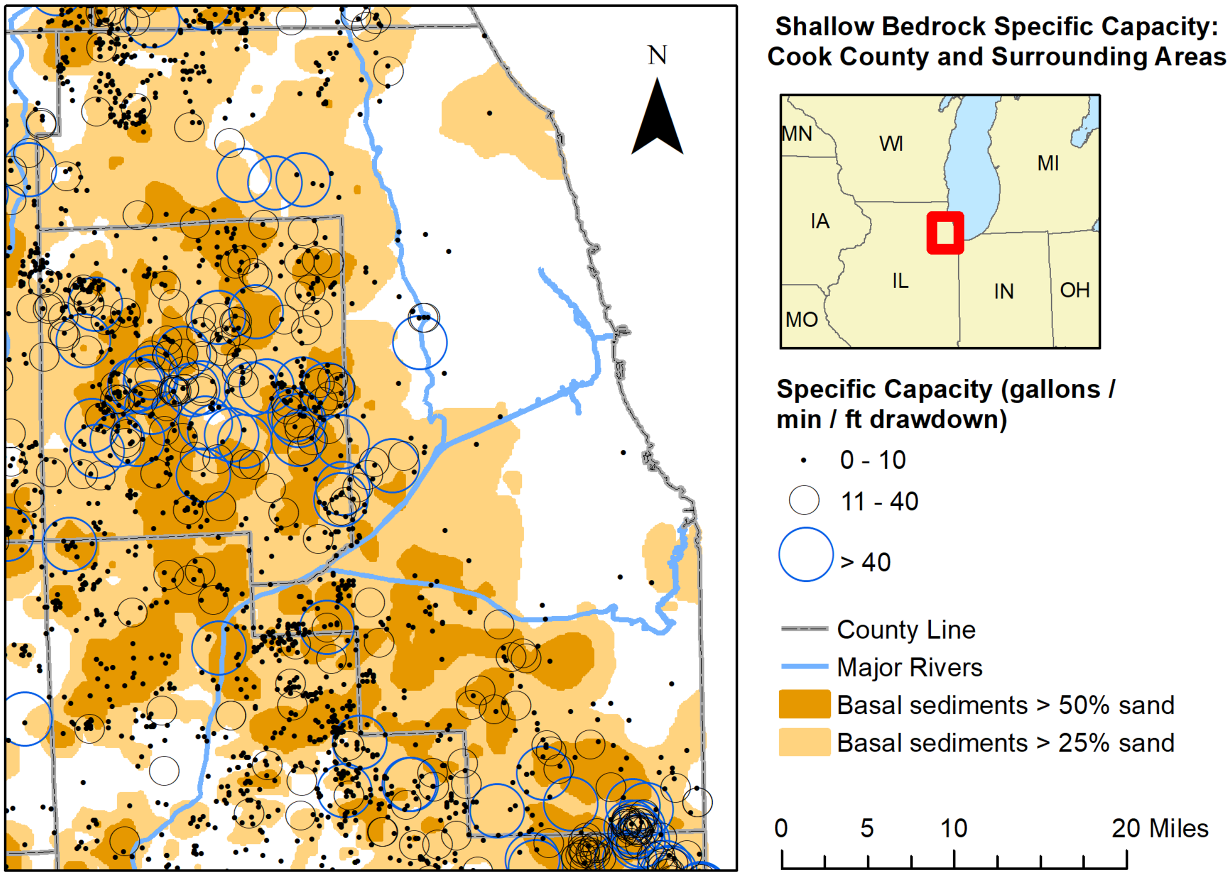


Figure 84. Distribution of basal sands overlying bedrock and specific capacity of water wells in the Chicago, IL region, from Illinois State Water Survey (ISWS) and Illinois State Geological Survey well records

Wastewater and stormwater infrastructure, for sewage transport, water volume control, and detention, radically alter the hydrogeology of a landscape and are interconnected in complex ways. For those reasons, some researchers refer to heavily urbanized groundwater systems as "urban karst" (Kaushel and Belt, 2012; Bonneau et al., 2017). This urban karst is composed of a nebulous network of surface and subsurface channels for stormwater and wastewater infrastructure that can inflate recharge to groundwater in areas otherwise limited by impervious surfaces, whether intentionally through infiltration design or unintentionally through leaky conduits. Precipitation enters these leaky networks and the extra water stresses the piping network and mobilizes contaminants (Ascott et al., 2018). Studies have found wide ranges of urban leakage compared to an area's natural recharge, with urban leakage values from 10% (Minnig et al., 2018) to 1000% (Wakode et al., 2018) of natural recharge rates. This amount of urban leakage has ramifications for solute concentrations and redox potential in both shallow groundwater and surface waters (Kaushal & Belt, 2012). At this time, no published study has compared urban rates of recharge to natural infiltration in the Chicago region.

Stormwater runoff can constitute a large fraction of total recharge to groundwater. For instance, a water budget study in Barcelona that focused on sources of groundwater recharge found that stormwater runoff composed 20% of the total, 22% from municipal pipe leakage, 30% from sewer leakage, and the remaining 28% from rainfall in non-urbanized areas and water exchanged from a nearby river (Vázquez-Suñé et al., 2010). Sewer and municipal leakage are problematic, as they can be subsequently captured by and routed through stormwater networks (Ascott et al., 2016; Roy et al., 2018; Wakode et al., 2018). Although recharge rates still vary spatially and temporally, on the whole, urbanization often increases recharge and creates highly permeable groundwater flow paths that render groundwater contaminant remediation more difficult (Sharp, 2010). The MWRD Watershed Management Ordinance (WMO) prescribes certain water volume control and detention requirements, but we acknowledge many municipal features can impact subsurface water flow, particularly buried wastewater pipes.

Stormwater infrastructure design and maintenance impacts how groundwater interacts with these structures (Parlov et al., 2019). In Cook County, the MWRD WMO focuses on two stormwater practices: volume control and detention. Volume control refers to practices that capture and hold on site the first inch of runoff from surfaces to infiltrate to groundwater, and the remainder can be released to receiving stormwater systems at the prescribed release rate. This is accomplished from a variety of practices, such as infiltration basins, bioswales that promote infiltration, or gravel-bottomed underground storage vaults that allow for infiltration. Not all stormwater is required to be captured by volume control. The remaining water falls under stormwater detention, where water is detained and not intended to infiltrate to groundwater. Examples include wet- or dry-bottomed detention basins and underground storage vaults that do not infiltrate or have water in excess of their infiltration capacity. These features are subject to the MWRD WMO Watershed Specific Release Rate that regulates how stormwater will reach streams and rivers.

6.2 How Water Volume Control and Detention Impacts Groundwater

As groundwater interactions with stormwater management structures are a relatively recent focus of research, most referenced studies in this review are outside of Illinois or with a different geology than Cook County. Recharge to groundwater in urban areas is challenging to estimate because of the complex intersection of land use, infrastructure, and geology (Lerner, 2002). In an effort to avoid too many confounding factors, many studies choose a small study site or focus on a single contaminant or class of contaminants. Every stormwater volume control or detention basin is a unique feature interacting with groundwater; the degree of interaction of stormwater and groundwater varies seasonally with water table levels, precipitation frequency, precipitation amounts, and the presence of underground sewer lines (Thompson et al., 2021). Despite this complexity, many papers examining stormwater infrastructure effects on groundwater have overlapping or synergistic results and are included here despite having different hydrogeological settings. In the following sections, we discuss common mechanisms, stormwater detention and volume control, that facilitate contaminants' emergence in groundwater.

Groundwater mounding, here referring to the water table rising at a spatially limited area associated with localized recharge, is perhaps the most obvious evidence of stormwater infiltration to groundwater beneath stormwater management features. In the context of stormwater, the extent of mounding is related to infiltration rate (Machusick et al., 2009), and infiltration diminishes with distance away from the retention area (Bonneau et al., 2018; Zhang and Chui, 2019). Limited infiltration outside of the retention area may be attributed to seasonal evapotranspiration losses or interception by subsurface drains (Bonneau et al., 2018). In addition to these infiltration limiting factors, in the MWRD region areas surrounding the detention and infiltration basins are highly impervious. Contaminants present in infiltrated water can be concentrated in soil and the vadose zone because of evapotranspirative losses, impacting vegetation or later remobilizing during recharge events (Grimaldi et al., 2009; Sullivan et al., 2014). Groundwater modeling has demonstrated that mounding near ponds can decrease the time it takes for contaminants to reach the groundwater (Nimmer et al., 2010; Novotny et al., 2009). Indeed, mounding enables greater movement of contaminants such as chloride (Novotny et al., 2009) or per- and polyfluoroalkyl substances (PFAS) (Cáñez et al., 2021); the latter tend to be immobile in unsaturated conditions (Brusseau et al., 2019).

Mounding and infiltration are often not uniform across the retention infrastructure. Fischer and co-authors (2003) found that infiltration rates vary across a single retention pond and are related to the rate and extent of sedimentation in the pond. They found that infiltration was highest at the infrastructure inlet. However, infiltration rates were generally high, ranging from 0.2 to 2.4 inch/minute, and served to dilute background groundwater constituent concentrations while loading the aquifer with stormwater-laden constituents. A review of green infrastructure's impact on shallow groundwater found that detention basins should be placed at least 1 meter above the groundwater table to limit mounding and contaminant transport (Zhang and Chui, 2019); regular pond maintenance and seasonal variation must be considered when evaluating this threshold. The MWRD WMO widely employs underdrains in stormwater volume control and retention because of the abundant clay presence in soils, intended to limit long-term basin retention and reserve runoff capacity by routing shallow groundwater into pipes routed away from the pond (Zhang and Chui, 2019). Though this should, in theory, limit infiltration to deeper groundwater, in practice, poor basin maintenance may lead to the underdrains no longer performing as intended.

Fine sediments, carried by stormwater into volume control or retention basins, can accumulate and impact the functionality of the infrastructure. Datry et al. (2004) found that fine sediments had clogged much of the bed sediments in their retention basin, causing most infiltration to occur along the margins of the basin where sediment build-up was thinner. This has implications for performance of underdrains, as poorly maintained basins could focus most infiltration away from the center of the basin where underdrains are likely to be placed, which may increase infiltration to deeper groundwater. Fine sediments can also play a role in contaminant transport through colloidal transport, an important mechanism for movement of metals through soils into shallow groundwater (Massoudieh and Ginn, 2008). Very fine colloids with large surface areas accumulate through atmospheric or water deposition in basins. Because of their large surface area, metals such as lead, copper, and zinc with a strong affinity for the solid phase are found to sorb to these fine particles and follow preferential flow to the shallow water table (Massoudieh and Ginn, 2008). Though the authors note the extent of movement observed in the study was limited, the movement of metals into different redox zones within soils can facilitate further transport.

Although not directly related to stormwater volume control and retention, wastewater infrastructure associated with subsurface pipes is also found to be a direct and pervasive source of many contaminants to groundwater. Leaking wastewater pipes and stormwater conveyance networks often contribute a sizeable portion of urban “recharge” (Sharp, 2010). In extreme cases, such as one in India, urban leakage from water mains and sewage was estimated to be more than 10 times the natural recharge rate (Wakode et al., 2018). London area studies have found that pipe leakage is most likely to occur in areas where pipes are particularly stressed, areas under heavily trafficked roads, and areas with clay bedrock (Ascott et al., 2016). The same research team found that the greatest contamination from pipe leakage occurred during the wettest time of the year (Ascott et al., 2018).

Similarly, high permeability trenches, where gravel fill is placed when installing pipes and other infrastructure, can allow leakage to travel along unexpected pathways (Bonneau et al., 2017, 2018). As many Chicago suburbs had preexisting development prior to the current land use, unmapped subsurface structures may be of concern as potential transport pathways where they intersect groundwater flow paths. Considering land use and geology is key to building an understanding of what contaminants are more likely and where contamination potential is greatest across the region.

6.3 Contaminants of Concern Likely to be Found in the MWRD Area

The most recent groundwater studies for the region include Roadcap et al. (1993), a study of groundwater quality of southernmost Cook and Will Counties, and Kay (2016), a report about Silurian Dolomite Aquifer water quality adjacent to the Tunnel and Reservoir Plan (TARP) tunnels through Cook County. In 2020, the Illinois-Indiana Sea Grant announced funding for a new project on chloride transport through stormwater infrastructure (Sea Grant, 2020), but the results of that study are not yet published. Most existing water quality monitoring for shallow groundwater in the county is from the TARP area that detects combined sewer flow seepage into groundwater. After combined sewer flow events, water quality in the vicinity of TARP tunnels generally returns to background levels (Kay, 2016). Recent groundwater quality data are only sporadically available across Cook County because of the limited number of active supply and monitoring wells. However, insights on contaminants of greatest concern will need to be drawn from these limited observations as well as from published studies of areas with geology and land use histories similar to Cook County. The prioritization of contaminants varies between studies. Clark & Pitt (2010) highlight the need to monitor for contaminants with the greatest mobilization (nitrate, some pesticides, pathogens, and chloride), and other studies such as Taguchi et al., 2020 focus on contaminants they identify as the greatest concern to public health, namely metals, pathogens, and chloride. Warner et al. (2016) note many classes of contaminants of concern for urban groundwater (Figure 85), and from this we can broadly classify contaminants based on their behavior and relevance for Cook County.

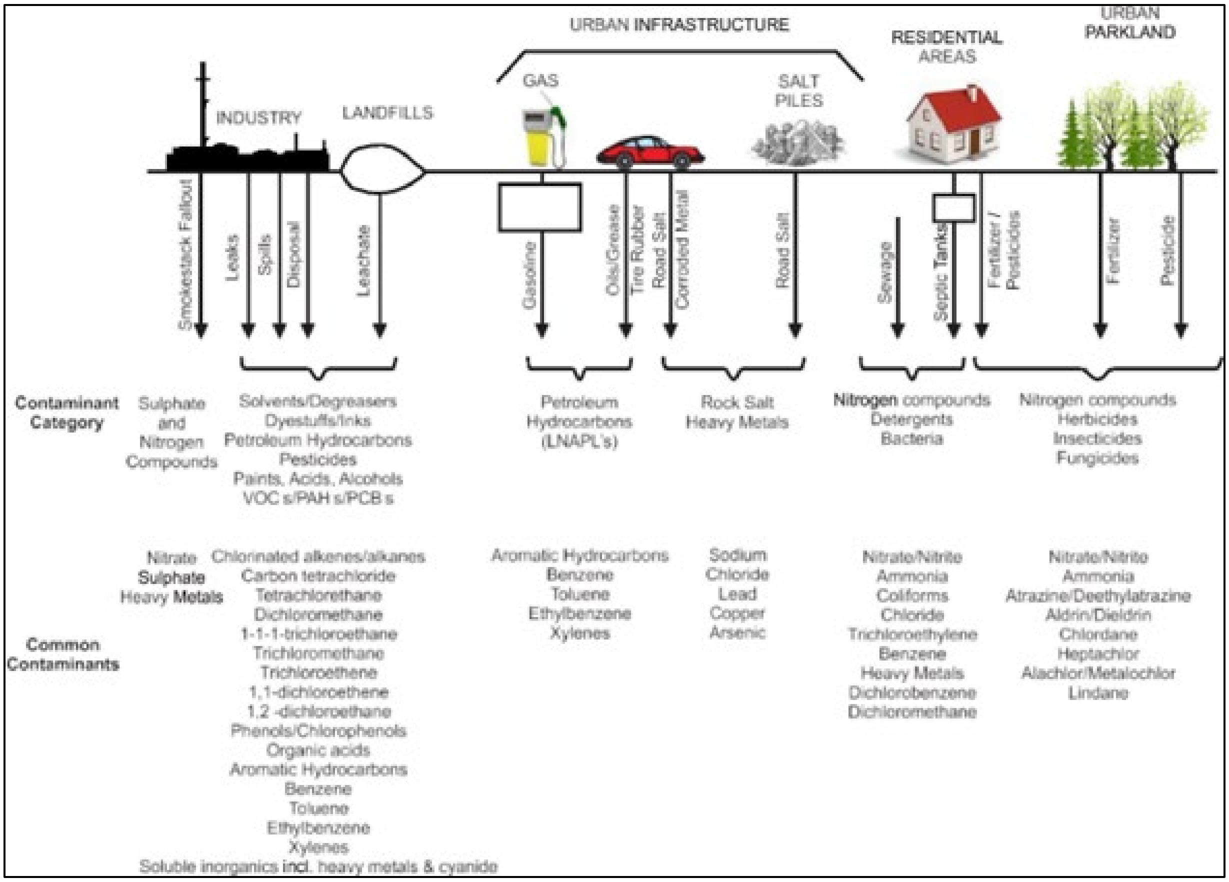


Figure 85. Common sources of urban groundwater contamination (from Warner et al., 2016; modified from Howard, 1997)

6.3.1 Chloride

Groundwater salinization is a global concern (Foster & Chilton, 2003), and in Illinois specifically, chloride levels near Chicago have been rising for decades (Kelly, 2008; Kelly, 2020) primarily attributed to winter deicer applications on paved surfaces. Chloride is a conservative contaminant that generally remains in solution as it travels along groundwater flow pathways. This makes it an ideal tracer to indicate urban leakage in developed areas and potential contamination pathways. Furthermore, high chloride concentrations can increase metal mobility and exacerbate the potential for heavy metal contamination (Taguchi et al., 2020). A Twin Cities, Minnesota study examining the impacts of municipal ponds on chloride found that 72% of chloride applied remains in the watershed whether as salt or dissolved in groundwater (Novotny et al., 2009). Watersheds draining areas with more stormwater basins have been found to have greater chloride in baseflow and higher peak conductivity, suggesting that stormwater management basins introduce year-round chloride to streams and rivers (Snodgrass et al., 2017). Indeed, numerous studies have found that stormwater detention or retention basins can be a source of chloride to an aquifer year round (Casey et al., 2013; Lembcke et al., 2017; Lam et al., 2020; Burgis et al., 2020).

High chloride concentrations in stormwater ponds originate from high salt application rates on paved urbanized areas (Lembcke et al., 2017). Areas that underwent major land use change in areas with thin (< 50 feet) cover over the bedrock aquifer saw a rapid increase in chloride levels from the mid-1990s to modern times in Will County, particularly prevalent in areas with heavily paved surfaces and stormwater basins (Cullen et al., [in revisions]). Although Cullen and co-authors cannot definitively state that chloride from stormwater infrastructure is impacting groundwater resources in Will County, their model indicates a need for "point source recharge" of chloride in localized areas that would function similarly to recharge from stormwater retention basins. This hypothesized source of chloride to the drinking water supplies of nearby communities is currently being researched by the ISWS.

Both Roadcap et al. (1993) and Kay (2016) documented elevated chloride in some areas of Cook County, suggesting winter deicer infiltration to the bedrock aquifer. Figure 86 shows chloride concentrations from Kay (2016), covering a larger area of the county than Roadcap et al. (1993) and featuring more recent data collected through 2013. Natural concentrations of chloride in northeast Illinois groundwater are below 15 mg/L (Panno et al., 2006). In 2013, several areas of the county experienced elevated concentrations clustered around the southeasternmost and north-central parts of the county. Samples west of the McCook reservoir had the highest concentrations of chloride, with four samples exceeding 100 mg/L. While not a comprehensive dataset covering the entire county, the variability in chloride concentrations suggests that infiltration pathways exist within portions of the county.

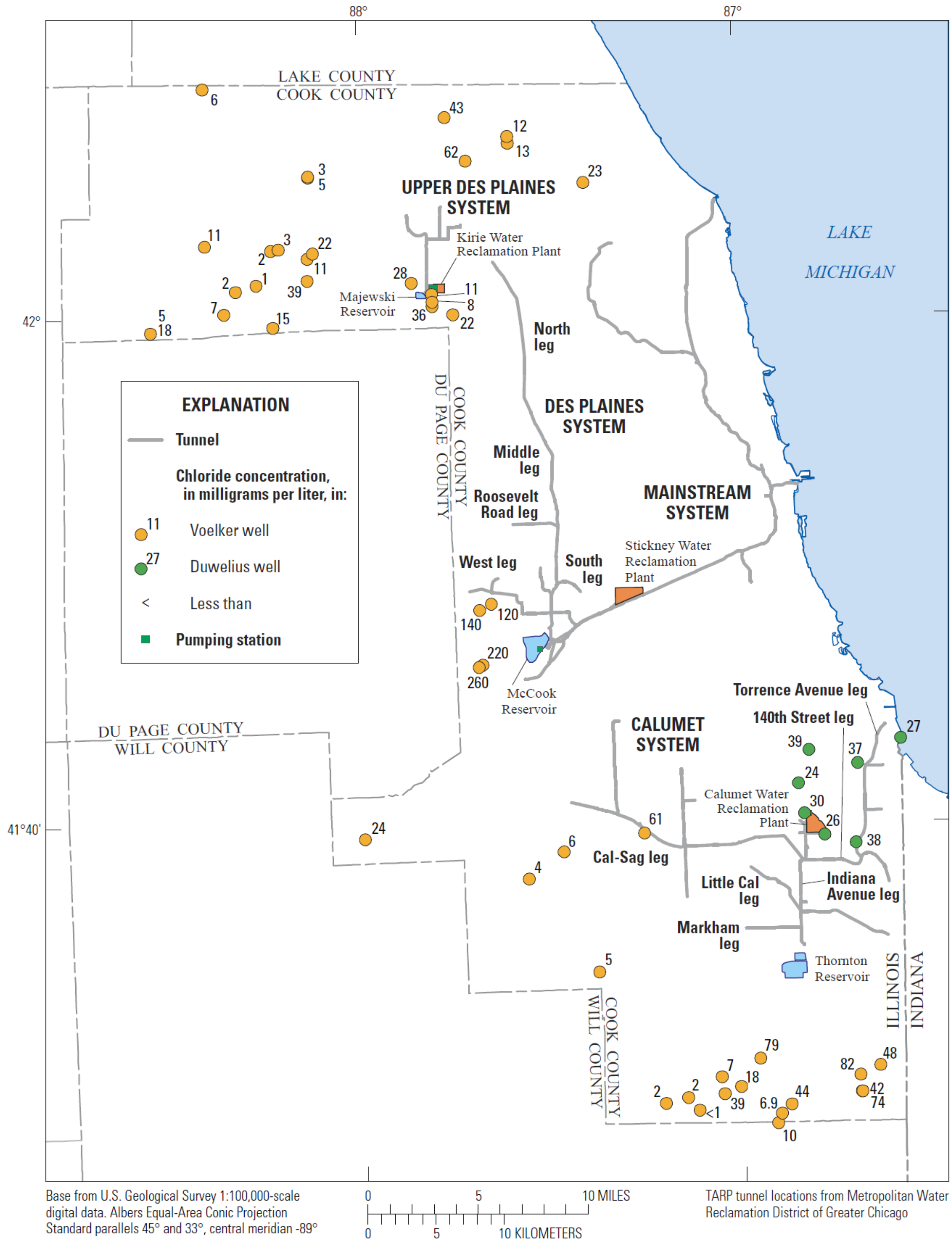


Figure 86. From Kay (2016), concentrations of chloride in water from the Silurian aquifer in Cook County, Illinois

6.3.2 PFAS

Per- and polyfluoroalkyl substances (PFAS) are a group of manufactured chemicals characterized by a carbon-fluorine backbone; the carbon-fluorine bond is one of the strongest and least degradable in organic chemistry (USEPA, accessed 2022). This bond makes them useful for many industrial and commercial applications, such as non-stick coating, flame retardants, food packaging, and cosmetics, but also means that PFAS is persistent in the environment, a trait that has earned these compounds the nickname "forever chemicals." PFASs such as perfluorooctanoic acid (PFOA) and perfluorooctane sulfonic acid (PFOS) have been associated with negative health effects in humans, including cancer, liver damage, weakened immune systems, and decreased vaccine effectiveness (Pelch et al., 2022). Despite this, there are no official federal regulatory standards for PFAS concentrations in water, although the USEPA recently released a lifetime health advisory limit (HAL) for four different PFASs (USEPA, accessed 2022). PFAS compounds have entered the hydrosphere through direct atmospheric deposition from emissions, usage of PFAS-containing foams in firefighting activities, and wastewater discharge from accumulated household and industrial sources. As this is one of many emerging contaminants, PFAS behavior in stormwater detention has limited research to date. Olmsted et al. (2021) detected PFAS compounds in stormwater sediments in both rural and urban catchments in Florida, ranging in concentrations from 7.2 to 4,800 nanograms per kilogram; Cáñez et al. (2021) found that an infiltration basin in Arizona introduced PFAS compounds to the aquifer sourced from wastewater treatment plant reclaimed water. PFAS concentrations were correlated with groundwater elevation, as PFAS is bound in unsaturated soil horizons and primarily mobile in saturated conditions (Brusseau et al., 2019; Cáñez et al., 2021). The water table elevation beneath retention basins is likely to be the critical factor in whether PFAS moves from stormwater detention to groundwater.

The Illinois Environmental Protection Agency (IEPA) recently completed a PFAS sampling campaign for Illinois drinking water supplies and published their results as an interactive map (IEPA, accessed 2022). However, coverage for PFAS sampling in the MWRD area is poor, mostly clustered in southeast Cook County. Based on detections elsewhere in Chicago's suburbs, Cook County's past and current industrial land use, and high volumes of wastewater discharge (not tested as part of the IEPA study), PFAS is likely to be present in some stormwater detention features. Further study is needed to understand its fate and transport to shallow groundwater in Cook County, but it is possible that PFAS is mobilized into groundwater through stormwater detention.

6.3.3 Metals

Metals in stormwater can originate from vehicle components, vehicle fuels and oils, tire residue, and industrial waste (Barbosa et al., 2012). Most metals are widely known to preferentially sorb and be sequestered within basin sediments and typically are only mobilized into groundwater when the soil sorption capacity is exceeded (Weiss et al., 2008). Sorption capacity exceedance is typically caused by chronic loads or lack of maintenance in bed sediments. As mentioned in the water table mounding discussion, colloidal transport can be important, whereby metals sorb to particles in the water column, including micro- and nano-plastics, and then in some circumstances mobilize into groundwater (Massoudieh & Ginn, 2008).

Understanding how stormwater behaves in the vadose zone is critical for assessing the risk of contamination from contaminants retained by soils, such as metals (Clark & Pitt, 2010). As Cook County is the site of many current and former industries, metal concentrations and speciation would most likely be tied to land use; for example, old tanneries can be locations of high chromium concentrations or lead contamination in the sediments of the Lake Calumet industrial area.

6.3.4 Phosphate and nitrate

Phosphate and nitrate together are one of the more prevalent concerns for watersheds in the Chicago region, as urban regions are the highest source per area for both nitrate and phosphorous (Hobbie et al., 2017; McIsaac, 2019). The Des Plaines River watershed is the largest contributor of nitrate and phosphate in Illinois, most originating from point-source discharge (treated wastewater and combined sewer overflows), with significant non-point source contributions (McIsaac, 2019). Though agricultural contributions are not expected across most of Cook County, major sources of nitrate could include fertilizer application, leaching from vegetated areas, sewage and septic leakage, lawn fertilizer, and animal waste (Aitkenhead-Peterson & Volder, 2010; Hobbie et al., 2017). Phosphate loading in urban areas can also be associated with leaky water infrastructure associated with phosphate-based lead inhibitor additives common in water treatment (Ascott et al., 2016, 2018). Geochemical analysis is needed to determine whether phosphate and nitrate are sourced from leaky pipes routed to stormwater runoff networks in the MWRD area.

Stormwater detention is typically thought to remove nitrate and phosphate from stormwater discharge and sequester it in basin sediments and plant material, yet in many cases it can be a net source to the groundwater system. Datry et al. (2004) observed loading of phosphate in groundwater beneath a retention basin; they hypothesized that this was caused by increased reaction times from the accumulation of fine sediments in the basin, increasing travel times overall. Similar behavior has also been observed in wetlands (Montgomery & Eames, 2008), suggesting care must be taken to limit phosphate loading from long-term degradation of performance in retention areas and slow infiltration times. Generally, limiting the saturation time of bed sediments is a best practice to minimize phosphate being mobilized into solution (Aitkenhead-Peterson & Volder, 2010; Hunt & Lord, 2006), although slow transit times promote denitrification and removal of nitrate (Aitkenhead-Peterson & Volder, 2010). Because of these properties, it may be challenging to target both nitrate and phosphate reductions to groundwater. Similar to its behavior in basin sediments and the behavior of metals, if sufficient loads of phosphate are present, the system may become saturated and phosphate can be mobilized in groundwater to streams and wetlands (Aitkenhead-Peterson & Volder, 2010; Domagalski & Johnson, 2011).

6.3.5 Other relevant contaminants

Pharmaceuticals, pesticides, and volatile organic compounds (VOC), while originating from different sources, are all common in urbanized areas and often studied together. Masoner et al. (2019) examined stormwater quality across several states and found that stormwater runoff has high concentrations of pharmaceuticals (prescription and nonprescription), pesticides, fossil fuel and combustion products, industrial chemicals, and household chemicals. Although concentrations in stormwater become diluted when mixed with groundwater, the authors urged communities to monitor groundwater for contaminant build-up. Similar calls have been made for Great Lakes' basins specifically, with Howard & Gerber (2018) noting the wide variety of organic contaminants present in urban areas. A study in Lyon, France set up monitoring wells up- and down-gradient of four different stormwater detention ponds and sampled the stormwater to analyze for polar chemicals, such as pesticides and pharmaceuticals. After continuing this study for three winters, they found four types of pesticides (two fungicides, one herbicide, and one insecticide) and lamotrigine, an antidepressant, had concentrations that increased with stormwater runoff (Pinasseau et al., 2020). Similarly, Hensen et al. (2018) documented the first case of biocides, typically applied to building facades to suppress microbial growth, entering groundwater via stormwater infiltration structures. Fischer et al. (2003) found benzene and toluene were detected above background levels in groundwater beneath retention basins, especially in the winter months, which they attribute to lower volatility in colder temperatures. They also found some pesticide concentrations, such as atrazine and desthyl-atrazine, varied seasonally in both retention basins and groundwater related to application rates. As organic contaminants are diverse in their behavior and sources, it is difficult to predict where they might occur in groundwater from stormwater detention structures, but they are likely to be found adjacent to land use associated with their application. When present, it is often at trace levels, but they have been known to impact aquatic biota and other sensitive species at low concentrations (Roy et al., 2018).

Microplastics, an understudied contaminant in groundwater, have been found in Illinois groundwater (Panno et al., 2019). These are defined as plastics smaller than 5 millimeters (approximately the size of a sesame seed) and occur through the physical breakdown of larger plastics into smaller sizes (NOAA, 2021). A study of karst basins in northwest and southwest Illinois found that the presence of microplastics, along with high chloride, nitrogen, and ortho-phosphate, could indicate contamination from septic systems, but the authors did not rule out stormwater runoff as the source of microplastics (Panno et al., 2019). Indeed, microplastics have been detected in stormwater (Koutnik et al., 2022; Liu et al., 2019; Ziajahromi et al., 2020). A Denmark study found a strong link between land use and the abundance of microplastics in retention basins, with retention ponds in commercial and industrial areas having higher microplastic concentrations than ponds in residential and highway areas (Liu et al., 2019). Overall, microplastics are primarily retained in the first 2 inches of sediment in stormwater infrastructure, but shape and size of particles have an impact on their mobility through the sediment (Koutnik et al., 2022). Another study examining microplastics originating from tires at the inlet and outlet of a stormwater wetland found that the outlet had a greater number and smaller sized microplastics (Ziajahromi et al., 2020). The MWRD region is likely to also have microplastics in its volume control and retention basins, particularly in basins near commercial and industrial sectors.

Pathogens, from bacteria and viruses, are an emerging concern for water quality, with groundwater hypothesized as an environmental reservoir for pathogens, where dominant strains vary with depth (Smith et al., 2013). Animals common in urbanized areas such as cats, dogs, and birds can contribute to biodegradable organic matter and pathogens to stormwater runoff (Barbosa et al., 2012). Shallow groundwater resources are sensitive to contamination from microbes in stormwater infiltration ponds, especially during heavy rainfall events, a period when precipitation amounts and microbial loads in stormwater have been found to be positively correlated (de Lambert et al., 2021). This is consistent with fecal coliform detections in bedrock aquifers adjacent to TARP tunnels following combined sewer flow events (Kay, 2016). Viruses have even been found to emerge in groundwater at depths of 220–300 meters merely weeks after high flow events because of leaky infrastructure (Bradbury et al., 2013). The distribution of pathogens can indicate which species are responsible for contaminating water in an area; for example, bird excrement was found to be the most pernicious in stormwater and groundwater in a North Dakota study (Olson et al., 2021). Although wetland habitats in Cook County are increasingly fragmented (Pasterski et al., 2020), groundwater could be acting as a storage and transport mechanism of species-specific pathogens (Borchardt et al., 2017). Because no study has been published on this in northeast Illinois, groundwater's potential for moving pathogens between habitats needs to be investigated. In short, pathogens should be an important consideration for monitoring and maintaining good drinking water quality in the southern and northwestern portions of the county, studying broader environmental consequences on habitats, and for their role as tracers for determining sources of water to the groundwater system.

6.4 Recommendations for Protecting Groundwater Quality

From the literature, we can see five main factors that increase risk of groundwater contamination on a site basis: 1) if known contamination sources are in the drainage area, 2) if transmissive sediments or bedrock are near the surface, 3) if stormwater structures are designed for infiltration, 4) if stormwater structures do not receive regular maintenance, and 5) if nearby water table elevations are high relative to stormwater detention. Figure 87 depicts a flow chart for evaluating contamination potential at an individual stormwater control or detention structure based on these criteria. Additionally, contaminants are more likely to infiltrate groundwater if they are highly soluble (Clark & Pitt, 2010), high in concentration, or the soil sorption capacity is limited. Although we expect that chloride, phosphate, nitrate, metals, and PFAS to be the most relevant contaminants to Cook County stormwater based on the land-use history, we recommend a thorough groundwater water sampling campaign to assess the suite of contaminants relevant to stormwater.

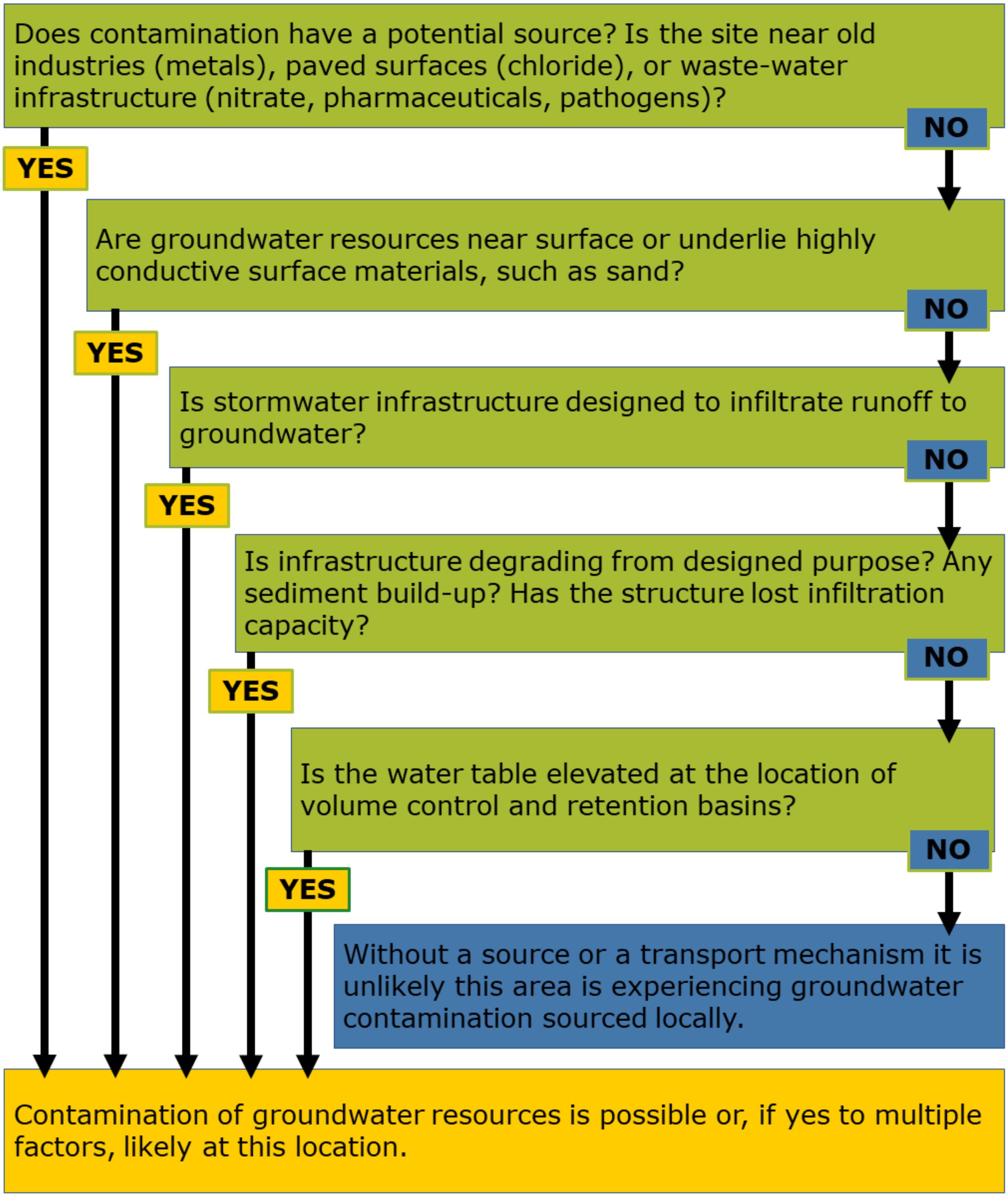


Figure 87. Assessing the likeliness of a site's groundwater to experience contamination

Because shallow groundwater use for community drinking water supplies has been limited in Cook County in the past several decades, current groundwater data are limited. To quantify the impact of stormwater management practices on groundwater in Cook County, establishing a groundwater monitoring network is strongly recommended. Monitoring wells can be established near prominent volume control and retention structures to monitor for contaminant loads to shallow groundwater, as well as adjacent to nearby habitats that may be receiving groundwater flow sourced in part from these structures. Nested wells, i.e., wells set at different depths at the same site, are recommended to evaluate the potential for groundwater movement between stormwater features, the water table, sand and gravel aquifers, and the underlying bedrock aquifer. This will help elucidate where contaminants are present and the extent of infiltration into the groundwater system. It would be of particular relevance to use this monitoring system to evaluate the performance of underdrains in limiting infiltration to deeper groundwater.

Though many of the criteria for evaluating contamination potential in Figure 87 require site-specific information, we can approximate regional contamination potential from existing geologic records. Generally, the greatest contamination potential for the shallow aquifer system is likely to occur where fine sediments are thin or absent at the surface (Dimitriou et al., 2008) or where sand is significant at the surface. In Figure 88, these higher vulnerability areas are characterized by the presence of coarse sediments such as sand and gravel near the surface as well as areas with limited overlying sediments that may indicate vulnerability to contamination in the bedrock aquifer. These areas are largely consistent with those highlighted as having highest infiltration potential by the USGS for the MWRD area (Morrow and Sharpe, 2009). Although broadly similar, the USGS study is more suited for assessing contamination potential in the narrow bands of sand found in stream and river valleys, and the ISWS map was initially created as a geologic interpolation for a regional groundwater model (Abrams et al., 2018). This map is more suited to identifying potential risk to the dolomite bedrock. Unlike the USGS study, which focused on continuous sand units at least 20 feet thick, we are considering bulk sand volume exceeding 25% as a potential contamination pathway. Sediment thickness over the aquifer bedrock is useful in considering contamination potential as greater amounts of overlying sediment provide a larger buffer between the land surface and groundwater resources. The geology suggests that the location of greatest contamination risk to groundwater in the MWRD region is the forest green colored area on Figure 88 (left) where transmissive sediments are within 10 feet of the surface. These transmissive deposits (Figure 88 left) and lack of overlying sediments (Figure 88 right) is a historic remnant of the landscape, as this is where the post-glacial Lake Chicago abruptly burst 19,000 years ago (Curry et al., 2018). The escaping waters from the lake incised into the landscape, removing glacial deposits and other material overlying the aquifer. As this is the most geologically sensitive area, prioritizing this area for monitoring well installation or sampling efforts would be insightful.

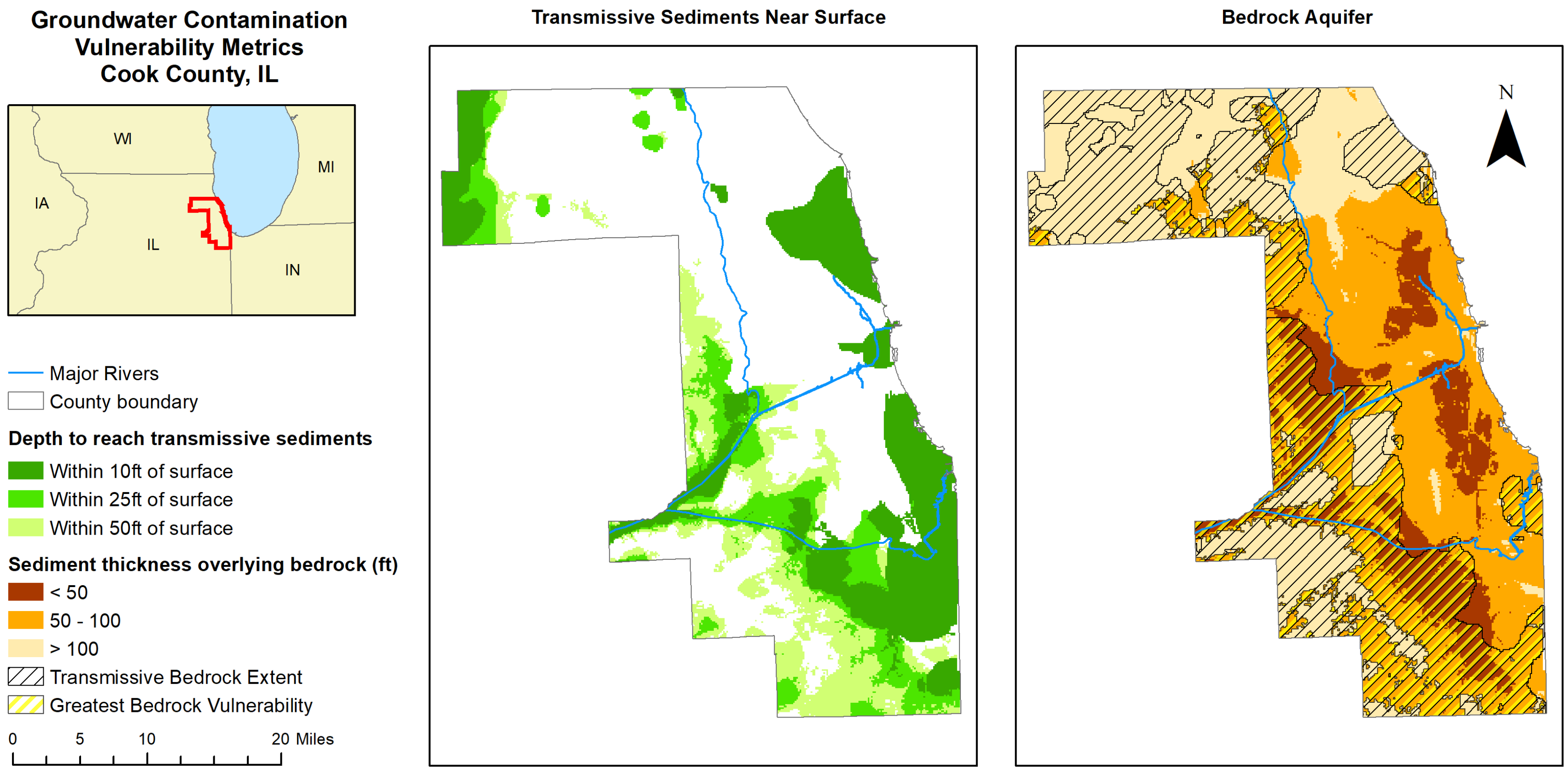


Figure 88. Areas of greater groundwater contamination vulnerability in Cook County, as indicated by presence of transmissive sediments near surface (left), and vulnerable bedrock recharge areas, as indicated by areas with transmissive bedrock overlain by less than 100 ft of sediments (right). Note "transmissive sediments" is a conservative estimate indicating greater than 25% coarse material (sand and gravel) by volume based on interpolated well lithological logs.

Table 51 shows the relative importance of sampling different constituents discussed throughout this report in the MWRD area. Many of the high sampling priorities overlap with constituents of high interest to MWRD (Table 51). The relative importance of land use, seasonality, and basin management on contamination of groundwater will vary for each volume control and detention feature. This means that a robust sampling campaign including many constituents is best to understand the impact of retention basins and volume control measures on groundwater.

Table 51. Sampling Recommendation for the MWRD Area

Constituent	*Identified as priority by MWRD?* [1]	*Sampling Priority*
Chloride	Yes	High[2]
Phosphate	Yes	High
Nitrate	Yes	High
PFAS[3]	No[4]	High
Water Isotopes[5]	No	High
Copper[6]	No	Moderate
Zinc[6]	No	Moderate
Iron[6]	Yes	Moderate
Manganese[6]	No	Moderate
Lead[6]	No	Moderate
Pesticides[7]	Yes	Moderate
Microplastics	No	Moderate
Pharmaceuticals and VOC	Yes[8]	Moderate
Silver	Yes	Low
Pathogens	No	Low

[1]See in Appendix A of MWRD Contract Phase III agreement.
[2]Chloride risk is a high sampling priority and needs to be continuously sampled year-round to be properly understood.
[3]PFAS requires a more costly and difficult sampling process than anything else on the table, so select and targeted sampling would be valuable.
[4]MWRD identified hydrocarbons as a priority concern, while PFAS is classified as fluorocarbons.
[5]Sampling for water isotopes is not sampling for a contaminant but used to assess the age and source of the groundwater at a given location.
[6]Often, many of these metals are included together in the same analysis.
[7]Specifically, we propose sampling glyphosate as that is commonly applied to lawns.
[8]VOCs were identified as contaminants of concern by MWRD, but not pharmaceuticals.

As long-term declines in habitat diversity are well documented in Cook County wetlands, especially associated with increasing salinization and invasion by salt-tolerant species (Panno et al., 1999; Pasterski et al., 2020; Price et al., 2014; Skultety & Matthews, 2017, 2018), chloride is perhaps the highest priority for monitoring to understand the potential for groundwater contamination from stormwater control and detention structures. Although data are scarce, elevated chloride concentrations in Cook County groundwater is precedented (Roadcap et al., 1993; Kay, 2016). Outfitting monitoring wells with continuous electrical conductivity probes allows for continuous hourly collection of chloride and total dissolved solids (TDS) data, as conductivity is a proxy for chloride and TDS once a regression is established. These probes are relatively inexpensive and would be instrumental in determining the existence of links between chloride and TDS in groundwater, stormwater infrastructure, and salinization in sensitive wetland ecosystems. With wetland habitats being increasingly fragmented in the Chicago region (Pasterski et al., 2020), and wetlands being on average surrounded by over 50% developed land in this region (Skultely and Matthews, 2018), these habitats are undoubtedly increasingly vulnerable to local impacts and stormwater routing influences. Similarly, given the long history of industrial and commercial land use historically in this region, many of these wetlands may be adjacent to existing contamination that may be remobilized during rain events. The Forest Preserves of Cook County identifies 66 wetland areas on their web map application (Forest Preserves of Cook County, accessed 2022); we recommend establishing monitoring adjacent to nearby wetlands that might be impacted by stormwater structures.

When studying groundwater quality in urban areas, evaluating many potential sources of contamination is important to determine the relative influence of stormwater in the system. The literature points to leaky infrastructure as a significant contributor to urban groundwater, with sewage leakage as a pervasive and troubling contamination source. Any sampling campaign should consider including pathogens, boron (sometimes an indicator for detergents in sewage), or pharmaceuticals to detect the presence of either stormwater networks capturing sewage leakage or sewage infiltration outside the basin influencing water quality beneath the basin.

For preliminary sampling, establishing approximate groundwater ages will be valuable to validate the methodology for determining contamination potential (for example, correlating groundwater ages with transmissivity in soils and upper sediments). In complex flow systems, such as the region's shallow aquifer, recharge takes complex paths to the subsurface, and waters of different ages could reside in different geologic units along a vertical profile (Shishaye et al., 2021). To this end, we recommend sampling for water isotopes that will help indicate the age of groundwater. Water isotopes will show how close water from groundwater is to recent precipitation and help determine if older groundwater exists in isolated lenses.

Recent literature reviews emphasize the need for studies on the impact of stormwater on the scale of watershed catchments (Zhang and Chui, 2019; USGS, 2022). Zhang and Chui (2019) discuss the need for future work to model solute transport from infrastructure to groundwater, the need for groundwater modeling to improve at large spatial scales, and the need for studies to focus on larger scales (watershed scale instead of pond site specific). After establishing a basic understanding of Cook County's groundwater quality, geology in the shallow aquifer, and impact of stormwater infiltration on the groundwater resources, the ISWS can develop a groundwater flow and contaminant transport model of the region. The ISWS has a functional model of the shallow aquifer system in Will County, directly south of Cook County (Abrams et al., 2018), that can be adapted to include Cook County. We recommend stepwise modeling, i.e., building model complexity during the data collection process and refining understanding of stormwater processes during the monitoring campaign. For an informed model, water levels, water quality, and detailed information about stormwater detention structures will be necessary. A first step for the model would be to simulate water movement from volume control and retention measures to groundwater, calibrated to both water level measurements and chloride time series from the proposed monitoring wells. Modeling would also help show how conservation efforts to protect groundwater quality at local scales can help aquifers and wetland ecosystems at regional scales.

Stormwater infrastructure is essential for preventing flooding on Illinois roads, homes, and businesses. However, potential impacts to groundwater quality are critical to assess. The ISWS can propose many ways to study groundwater in this area, but first monitoring wells would need to be installed, as Cook County does not have a well network large enough for sufficient spatial coverage. Geology and land use can guide where it would be most beneficial to install these monitoring wells to assess the impact of stormwater on groundwater quality. Installing monitoring wells and communication with property owners to maintain stormwater infrastructure to full functionality would be a step toward protecting groundwater supplies in the region and maintaining ecosystem health for groundwater-dependent habitats in the region.

6.5 References

Abrams, D. B., & Cullen, C. (2020). Analysis of Risk to Sandstone Water Supply in the Southwest Suburbs of Chicago (No. Contract Report 2020-04) (Vol. Contract Report 2020-04, p. 51). Champaign, IL: Illinois State Water Survey, University of Illinois. Retrieved from file:///C:/Users/wkelly/Downloads/ISWS_CR_2020-04_Final_Web.pdf

Abrams, D. B., Mannix, D. H., Hadley, D. R., & Roadcap, G. S. (2018). Groundwater Flow Models of Illinois: Data, Processes, Model Performance, and Key Results. https://www.ideals.illinois.edu/handle/2142/102968

Aitkenhead-Peterson, J., & Volder, A. (2010). Urban Ecosystem Ecology. American Society of Agronomy, Inc. https://acsess.onlinelibrary.wiley.com/doi/book/10.2134/agronmonogr55

Ascott, M. J., Gooddy, D. C., Lapworth, D. J., Davidson, P., Bowes, M. J., Jarvie, H. P., & Surridge, B. W. J. (2018). Phosphorus fluxes to the environment from mains water leakage: Seasonality and future scenarios. Science of The Total Environment, 636, 1321–1332. https://doi.org/10.1016/j.scitotenv.2018.04.226

Ascott, M. J., Gooddy, D. C., Lapworth, D. J., & Stuart, M. E. (2016). Estimating the leakage contribution of phosphate dosed drinking water to environmental phosphorus pollution at the national-scale. Science of The Total Environment, 572, 1534–1542. https://doi.org/10.1016/j.scitotenv.2015.12.121

Barbosa, A. E., Fernandes, J.N., David, L.M. (2012). Key issues for sustainable urban stormwater management. Water Research, 46 (12) p 6787-6798. https://doi.org/10.1016/j.watres.2012.05.029.

Bonneau, J., Fletcher, T. D., Costelloe, J. F., & Burns, M. J. (2017). Stormwater infiltration and the 'urban karst' – A review. Journal of Hydrology, 552, 141–150. https://doi.org/10.1016/j.jhydrol.2017.06.043

Bonneau, J., Fletcher, T. D., Costelloe, J. F., Poelsma, P. J., James, R. B., & Burns, M. J. (2018). Where does infiltrated stormwater go? Interactions with vegetation and subsurface anthropogenic features. Journal of Hydrology, 567, 121–132. https://doi.org/10.1016/j.jhydrol.2018.10.006

Borchardt, M. A., Spencer, S. K., Hubbard, L. E., Firnstahl, A. D., Stokdyk, J. P., & Kolpin, D. W. (2017). Avian Influenza Virus RNA in Groundwater Wells Supplying Poultry Farms Affected by the 2015 Influenza Outbreak. Environmental Science & Technology Letters, 4(7), 268–272. https://doi.org/10.1021/acs.estlett.7b00128

Bradbury, K. R., Borchardt, M. A., Gotkowitz, M., Spencer, S. K., Zhu, J., & Hunt, R. J. (2013). Source and transport of human enteric viruses in deep municipal water supply wells. Environ Sci Technol, 47(9), 4096–4103. https://doi.org/10.1021/es400509b

Brusseau, M. L., Yan, N., Van Glubt, S., Wang, Y., Chen, W., Lyu, Y., Dungan, B., Carroll, K. C., & Holguin, F. O. (2019). Comprehensive retention model for PFAS transport in subsurface systems. Water Research, 148, 41–50. PubMed. https://doi.org/10.1016/j.watres.2018.10.035

Burgis, C. R., Hayes, G. M., Henderson, D. A., Zhang, W., & Smith, J. A. (2020). Green stormwater infrastructure redirects deicing salt from surface water to groundwater. Science of The Total Environment, 729, 138736. https://doi.org/10.1016/j.scitotenv.2020.138736

Buschbach, T. C., & Heim, G. E. (1972). Preliminary geologic investigations of rock tunnel sites for flood and pollution control in the greater Chicago area. https://www.ideals.illinois.edu/handle/2142/78965

Cáñez, T. T., Guo, B., McIntosh, J. C., & Brusseau, M. L. (2021). Perfluoroalkyl and polyfluoroalkyl substances (PFAS) in groundwater at a reclaimed water recharge facility. Science of The Total Environment, 791, 147906. https://doi.org/10.1016/j.scitotenv.2021.147906

Casey, R. E., Lev, S. M., & Snodgrass, J. W. (2013). Stormwater ponds as a source of long-term surface and ground water salinisation. Urban Water Journal, 10(3), 145–153. https://doi.org/10.1080/1573062X.2012.716070

Ceisel, E., J., K.J. Van Meter, "Road Salt Legacies: Quantifying Fluxes of Chloride to Groundwater and Surface Water across the Chicago MSA," in prep, will be submitted to WRR

Clark, S. E., & Pitt, R. (2010). Groundwater Contamination Potential from Infiltration of Urban Stormwater Runoff. EFFECTS OF URBANIZATION ON GROUNDWATER, 46.

Csallany, S., & Walton, W. C. (1963). Yields of shallow dolomite wells in Northern Illinois. https://www.ideals.illinois.edu/handle/2142/101987

Curry, B. B., Lowell, T. V., Wang, H., and Anderson, A.C., 2018, Revised time-distance diagram for the Lake Michigan Lobe, Michigan Subepisode, Wisconsin Episode, Illinois, USA, in Kehew, A.E., and Curry, B. B., eds., Quaternary Glaciation of the Great Lakes Region: Process, Landforms, Sediments, and Chronology: Geological Society of America Special Paper 530, p. 69–101, https://doi:10.1130/2018.2530(04)

Datry, T., Malard, F., & Gibert, J. (2004). Dynamics of solutes and dissolved oxygen in shallow urban groundwater below a stormwater infiltration basin. Science of the Total Environment, 15.

de Lambert, J. R., Walsh, J. F., Scher, D. P., Firnstahl, A. D., & Borchardt, M. A. (2021). Microbial pathogens and contaminants of emerging concern in groundwater at an urban subsurface stormwater infiltration site. Science of The Total Environment, 775, 145738. https://doi.org/10.1016/j.scitotenv.2021.145738

Dimitriou, E., Karaouzas, I., Sarantakos, K., Zacharias, I., Bogdanos, K., & Diapoulis, A. (2008). Groundwater risk assessment at a heavily industrialised catchment and the associated impacts on a peri-urban wetland. Journal of Environmental Management, 88(3), 526–538. https://doi.org/10.1016/j.jenvman.2007.03.019

Domagalski, J. L., & Johnson, H. M. (2011). Subsurface transport of orthophosphate in five agricultural watersheds, USA. Journal of Hydrology, 409(1), 157–171. https://doi.org/10.1016/j.jhydrol.2011.08.014

Fischer, D., Charles, E. G., Baehr, A. L., & Scientist, S. (2003). Effects of Stormwater Infiltration on Quality of Groundwater Beneath Retention and Detention Basins. 8.

Foster, S. S. D., & Chilton, P. J. (2003). Groundwater: The processes and global significance of aquifer degradation. Philosophical Transactions of the Royal Society B: Biological Sciences, 358(1440), 1957–1972. https://doi.org/10.1098/rstb.2003.1380

Freeborn, J. R., Sample, D. J., & Fox, L. J. (2012). RESIDENTIAL STORMWATER: METHODS FOR DECREASING RUNOFF AND INCREASING STORMWATER INFILTRATION. Journal of Green Building, 7(2), 15–30. https://doi.org/10.3992/jgb.7.2.15

Grimaldi, C., Thomas, Z., Fossey, M., Fauvel, Y., & Merot, P. (2009). High chloride concentrations in the soil and groundwater under an oak hedge in the West of France: An indicator of evapotranspiration and water movement. Hydrological Processes, 23(13), 1865–1873. https://doi.org/10.1002/hyp.7316

Hensen, B., Lange, J., Jackisch, N., Zieger, F., Olsson, O., & Kümmerer, K. (2018). Entry of biocides and their transformation products into groundwater via urban stormwater infiltration systems. Water Research, 144, 413–423. https://doi.org/10.1016/j.watres.2018.07.046

Hobbie, S. E., Finlay, J. C., Janke, B. D., Nidzgorski, D. A., Millet, D. B., & Baker, L. A. (2017). Contrasting nitrogen and phosphorus budgets in urban watersheds and implications for managing urban water pollution. Proceedings of the National Academy of Sciences, 114(16), 4177–4182. https://doi.org/10.1073/pnas.1618536114

Howard, K., & Gerber, R. (2018). Impacts of urban areas and urban growth on groundwater in the Great Lakes Basin of North America. Journal of Great Lakes Research, 44(1), 1–13. https://doi.org/10.1016/j.jglr.2017.11.012

Hunt, W. F., & Lord, W. G. (2006). Bioretention Performance, Design, Construction, and Maintenance. NC State Extension Publications. https://content.ces.ncsu.edu/bioretention-performance-design-construction-and-maintenance

Kaushal, S. S., & Belt, K. T. (2012). The urban watershed continuum: Evolving spatial and temporal dimensions. Urban Ecosystems, 15(2), 409–435. https://doi.org/10.1007/s11252-012-0226-7

Kay, R. T. (2016). Hydrogeology and groundwater quality at monitoring wells installed for the Tunnel and Reservoir Plan System and nearby water-supply wells, Cook County, Illinois, 1995–2013. In Hydrogeology and groundwater quality at monitoring wells installed for the Tunnel and Reservoir Plan System and nearby water-supply wells, Cook County, Illinois, 1995–2013 (USGS Numbered Series No. 2015–5186; Scientific Investigations Report, Vols. 2015–5186, p. 357). U.S. Geological Survey. https://doi.org/10.3133/sir20155186

Kay, R. T., Gahala, A. M., & Bailey, C. (2018). Assessment of water resources in areas that affect the habitat of the endangered Hine's emerald dragonfly in the Lower Des Plaines River Valley, Illinois (Report No. 2018–5074; Scientific Investigations Report, p. 118). USGS Publications Warehouse. https://doi.org/10.3133/sir20185074

Kelly, W.R. (2008.) "Long-term trends in chloride concentrations in shallow aquifers near Chicago." Ground Water 46(5):772-781.

Kelly, W.R. (2020). "Recent Trends in Chloride and Total Dissolved Solids in Silurian Wells in the Southwest Water Planning Group Region: Indicators of Groundwater Contamination within the Silurian Dolomite Aquifer. Illinois State Water Survey." Illinois State Water Survey Assessed Accessed at http://hdl.handle.net/2142/107224

Knobeloch, L. M., Zierold, K. M., & Anderson, H. A. (2006). Association of arsenic-contaminated drinking-water with prevalence of skin cancer in Wisconsin's Fox River Valley. Journal of Health, Population, and Nutrition, 24(2), 206–213.

Koutnik, V. S., Leonard, J., Glasman, J. B., Brar, J., Koydemir, H. C., Novoselov, A., Bertel, R., Tseng, D., Ozcan, A., Ravi, S., & Mohanty, S. K. (2022). Microplastics retained in stormwater control measures: Where do they come from and where do they go? Water Research, 210, 118008. https://doi.org/10.1016/j.watres.2021.118008

Lai, J., and Anders, A. M. (2018). Modeled Postglacial Landscape Evolution at the Southern Margin of the Laurentide Ice Sheet: Hydrological Connection of Uplands Controls the Pace and Style of Fluvial Network Expansion. Journal of Geophysical Research: Earth Surface, 123(5), 967-984. 10.1029/2017JF004509

Lam, W. Y., Lembcke, D., & Oswald, C. (2020). Quantifying chloride retention and release in urban stormwater management ponds using a mass balance approach. Hydrological Processes, 34(23), 4459–4472. https://doi.org/10.1002/hyp.13893

Leetaru, H. E., Sargent, M., L., & Kolata, D. R. (n.d.). Illinois State Geological Survey Geologic Atlas of Cook County for Planning Purposes, Cook County, Illinois (No. 2004–12). Illinois State Geological Survey. Retrieved April 5, 2022, from https://isgs.illinois.edu/maps/county-maps/cook-geologic-atlas

Lembcke, D., Thompson, B., Read, K., Betts, A., & Singaraja, D. (2017). REDUCING ROAD SALT APPLICATION BY CONSIDERING WINTER MAINTENANCE NEEDS IN PARKING LOT DESIGN. Journal of Green Building, 12(2), 1–12. https://doi.org/10.3992/1943-4618.12.2.1

Lerner, D. N. (2002). Identifying and quantifying urban recharge: A review. Hydrogeology Journal, 10(1), 143–152. https://doi.org/10.1007/s10040-001-0177-1

Liu, F., Olesen, K. B., Borregaard, A. R., & Vollertsen, J. (2019). Microplastics in urban and highway stormwater retention ponds. Science of The Total Environment, 671, 992–1000. https://doi.org/10.1016/j.scitotenv.2019.03.416

Ma, Y., Egodawatta, P., McGree, J., Liu, A., & Goonetilleke, A. (2016). Human health risk assessment of heavy metals in urban stormwater. Science of The Total Environment, 557–558, 764–772. https://doi.org/10.1016/j.scitotenv.2016.03.067

Machusick, M. D., Traver, R. G., & Engineer, E. (2009). The Observed Effects of Stormwater Infiltration on Groundwater. World Environmental and Water Resources Congress, 10.

Masoner, J. R., Kolpin, D. W., Cozzarelli, I. M., Barber, L. B., Burden, D. S., Foreman, W. T., Forshay, K. J., Furlong, E. T., Groves, J. F., Hladik, M. L., Hopton, M. E., Jaeschke, J. B., Keefe, S. H., Krabbenhoft, D. P., Lowrance, R., Romanok, K. M., Rus, D. L., Selbig, W. R., Williams, B. H., & Bradley, P. M. (2019). Urban Stormwater: An Overlooked Pathway of Extensive Mixed Contaminants to Surface and Groundwaters in the United States. Environ. Sci. Technol., 12.

Massoudieh, A., & Ginn, T. R. (2008). Modeling Colloid-Enhanced Contaminant Transport in Stormwater Infiltration Basin Best Management PracticesAll rights reserved. No part of this periodical may be reproduced or transmitted in any form or by any means, electronic or mechanical, including photocopying, recording, or any information storage and retrieval system, without permission in writing from the publisher. Vadose Zone Journal, 7(4), 1261–1268. https://doi.org/10.2136/vzj2007.0179

McIsaac, G. (2019). Nitrate and Total Phosphorus Loads in Illinois Rivers: Update Through the 2017 Water Year (p. 68). Illinois Environmental Protection Agency.

Minnig, M., Moeck, C., Radny, D., & Schirmer, M. (2018). Impact of urbanization on groundwater recharge rates in Dübendorf, Switzerland. Journal of Hydrology, 563, 1135–1146. https://doi.org/10.1016/j.jhydrol.2017.09.058

Montgomery, J. A., & Eames, J. M. (2008). Prairie Wolf Slough Wetlands Demonstration Project: A Case Study Illustrating the Need for Incorporating Soil and Water Quality Assessment in Wetland Restoration Planning, Design and Monitoring. Restoration Ecology, 16(4), 618–628. https://doi.org/10.1111/j.1526-100X.2008.00492.x

Morrow, W. S., & Sharpe, J. B. (2009). Preliminary Assessment of the Potential for Inducing Stormwater Infiltration in Cook County, Illinois. In Preliminary Assessment of the Potential for Inducing Stormwater Infiltration in Cook County, Illinois (USGS Numbered Series No. 2009–1212; Open-File Report, Vols. 2009–1212). U.S. Geological Survey. https://doi.org/10.3133/ofr20091212

Nimmer, M., Thompson, A., & Misra, D. (2010). Modeling Water Table Mounding and Contaminant Transport beneath Storm-Water Infiltration Basins. Journal of Hydrologic Engineering, 15(12), 963–973. https://doi.org/10.1061/(ASCE)HE.1943-5584.0000256

NOAA. What are microplastics? National Ocean Service website, https://oceanservice.noaa.gov/facts/microplastics.html, 02/26/2021.

Novotny, E. V., Sander, A. R., Mohseni, O., & Stefan, H. G. (2009). Chloride ion transport and mass balance in a metropolitan area using road salt: CHLORIDE ION TRANSPORT. Water Resources Research, 45(12). https://doi.org/10.1029/2009WR008141

Olmsted, J. L., Ahmadireskety, A., Da Silva, B. F., Robey, N., Aristizabal-Henao, J. J., Bonzongo, J.-C. J., & Bowden, J. A. (2021). Using Regulatory Classifications to Assess the Impact of Different Land Use Types on Per- and Polyfluoroalkyl Substance Concentrations in Stormwater Pond Sediments. Journal of Environmental Engineering, 147(10), 06021005. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001906

Olson, E., Hargiss, C. L. M., & Norland, J. (2021). Escherichia coli levels and microbial source tracking in stormwater retention ponds and detention basins. WATER ENVIRONMENT RESEARCH, 16.

Panno, S. V., Nuzzo, V. A., Cartwright, K., Hensel, B. R., & Krapac, I. G. (1999). Impact of urban development on the chemical composition of ground water in a fen-wetland complex. Wetlands, 19(1), 236–245. https://doi.org/10.1007/BF03161753

Panno, S. V., Hackley, K. C., Hwang, H. H., Greenberg, S. E., Krapac, I. G., Landsberger, S., & O'Kelly, D. J. (2006). Source identification of sodium and chloride in natural waters: Preliminary results. Ground Water, 44(2), 176–187.

Panno, S. V., Kelly, W. R., Scott, J., Zheng, W., McNeish, R. E., Holm, N., Hoellein, T. J., & Baranski, E. L. (2019). Microplastic Contamination in Karst Groundwater Systems. Groundwater, 57(2), 189–196. https://doi.org/10.1111/gwat.12862

Parlov, J., Kovač, Z., Nakić, Z., & Barešić, J. (2019). Using Water Stable Isotopes for Identifying Groundwater Recharge Sources of the Unconfined Alluvial Zagreb Aquifer (Croatia). Water, 11(10), 2177. https://doi.org/10.3390/w11102177

Pasterski, M. J., Bellagamba, A., Chancellor, S., Cunje, A., Dodd, E., Gefeke, K., Hsieh, S., Schassburger, A., Smith, A., Tucker, W., & Plotnick, R. E. (2020). Aquatic landscape change, extirpations, and introductions in the Chicago Region. Urban Ecosystems, 23(6), 1277–1288. https://doi.org/10.1007/s11252-020-01001-6

Pelch, K.E., Reade, A., Kwiatkowski, C.F., Merced-Nieves, F. M., Cavalier, H., Schultz, K., Wolffe, T., Varshavsky, J. (2022). The PFAS-Tox Database: A systematic evidence map of health studies on 29 per- and polyfluoroalkyl substances, Environment International, Volume 167, 107408, ISSN 0160-4120, https://doi.org/10.1016/j.envint.2022.107408.

Pinasseau, L., Wiest, L., Volatier, L., Mermillod-Blondin, F., & Vulliet, E. (2020). Emerging polar pollutants in groundwater: Potential impact of urban stormwater infiltration practices. Environmental Pollution, 266, 115387. https://doi.org/10.1016/j.envpol.2020.115387

Price, A. L., Fant, J. B., & Larkin, D. J. (2014). Ecology of Native vs. Introduced Phragmites australis (Common Reed) in Chicago-Area Wetlands. Wetlands, 34(2), 369–377. https://doi.org/10.1007/s13157-013-0504-z

Roadcap, G. S., Cravens, S. J., & Smith, E. C. (1993). Meeting the Growing Demand for Water: An Evaluation of the Shallow Ground-Water Resources in Will and Southern Cook Counties, Illinois. https://www.ideals.illinois.edu/handle/2142/75859

Roy, J. W., Grapentine, L., & Bickerton, G. (2018). Ecological effects from groundwater contaminated by volatile organic compounds on an urban stream's benthic ecosystem. Limnologica, 68, 115–129. https://doi.org/10.1016/j.limno.2017.01.004

Sea Grant. (2020). 'Illinois-Indiana Sea Grant anoounces funding for five new research projects'. Available at https://iiseagrant.org/illinois-indiana-sea-grant-announces-funding-for-five-new-research-projects/

Sharp, J. M. Jr. (2010). The impacts of urbanization on groundwater systems and recharge. Aqua Mundi, 1, 51–56. https://doi.org/10.44409/Am-004-10-0008

Shishaye, H. A., Tait, D. R., Maher, D. T., Befus, K. M., Erler, D., Jeffrey, L., Reading, M. J., Morgenstern, U., Kaserzon, S., Mueller, J., & De Verelle-Hill, W. (2021). The legacy and drivers of groundwater nutrients and pesticides in an agriculturally impacted Quaternary aquifer system. Science of The Total Environment, 753, 142010. https://doi.org/10.1016/j.scitotenv.2020.142010

Skultety, D., & Matthews, J. W. (2017). Urbanization and roads drive non-native plant invasion in the Chicago Metropolitan region. Biological Invasions, 19(9), 2553–2566. https://doi.org/10.1007/s10530-017-1464-7

Skultety, D., & Matthews, J. W. (2018). Human land use as a driver of plant community composition in wetlands of the Chicago metropolitan region. Urban Ecosystems, 21(3), 447–458. https://doi.org/10.1007/s11252-018-0730-5

Smith, R. J., Jeffries, T. C., Roudnew, B., Seymour, J. R., Fitch, A. J., Simons, K. L., Speck, P. G., Newton, K., Brown, M. H., & Mitchell, J. G. (2013). Confined aquifers as viral reservoirs. Environmental Microbiology Reports, 5(5), 725–730. https://doi.org/10.1111/1758-2229.12072

Snodgrass, J. W., Moore, J., Lev, S. M., Casey, R. E., Ownby, D. R., Flora, R. F., & Izzo, G. (2017). Influence of Modern Stormwater Management Practices on Transport of Road Salt to Surface Waters. Environmental Science & Technology, 51(8), 4165–4172. https://doi.org/10.1021/acs.est.6b03107

Sullivan, P. L., Price, R. M., Miralles-Wilhelm, F., Ross, M. S., Scinto, L. J., Dreschel, T. W., Sklar, F. H., & Cline, E. (2014). The role of recharge and evapotranspiration as hydraulic drivers of ion concentrations in shallow groundwater on Everglades tree islands, Florida (USA). Hydrological Processes, 28(2), 293–304. https://doi.org/10.1002/hyp.9575

Taguchi, V. J., Weiss, P. T., Gulliver, J. S., Klein, M. R., Hozalski, R. M., Baker, L. A., Finlay, J. C., Keeler, B. L., & Nieber, J. L. (2020). It Is Not Easy Being Green: Recognizing Unintended Consequences of Green Stormwater Infrastructure. 33.

Thompson, J. M. (2020). The effect of stormwater infiltration and surrounding built infrastructure on local groundwater dyna. 12.

USEPA. Per- and Polyfluoroalkyl Substances (PFAS). Accessed July 2022 at https://www.epa.gov/pfas

United States Geological Survey. Soller, D.R., Garrity, C.P. (2018). 'Map of Quaternary Sediment Thickness'. US Department of the Interior Reston, VA. https://pubs.usgs.gov/sim/3392/sim3392_sheet1.pdf

United States Geological Survey. Baker, N.T., Sullivan, D.J, Selbig, W.R., Haefner, R.J., Lampe, D.C., Bayless, R., McHale, M.R. (2022). 'Green Infrastructure in the Great Lakes—Assessment of Performance, Barriers, and

Unintended Consequences'. US Department of the Interior Reston, VA. https://pubs.usgs.gov/circ/1496/cir1496.pdf
Vázquez-Suñé, E., Carrera, J., Tubau, I., Sánchez-Vila, X., & Soler, A. (2010). An approach to identify urban groundwater recharge. Hydrology and Earth System Sciences, 14(10), 2085–2097. Scopus. https://doi.org/10.5194/hess-14-2085-2010
Wakode, H. B., Baier, K., Jha, R., & Azzam, R. (2018). Impact of urbanization on groundwater recharge and urban water balance for the city of Hyderabad, India. International Soil and Water Conservation Research, 6(1), 51–62. https://doi.org/10.1016/j.iswcr.2017.10.003
Warner, K., Howard, K., Gerber, R., Soo Chan, G., & Ford, D. (2016). Impacts of urban development on groundwater. Groundwater Science Relevant to the Great Lakes Water Quality Agreement: A Status Report. Final Version, May, 2016, 46–57. Scopus.
Weiss, P. T., LeFevre, G., & Gulliver, J. S. (2008). Contamination of Soil and Groundwater Due to Stormwater Infiltration Practices, A Literature Review [Report]. St. Anthony Falls Laboratory. http://conservancy.umn.edu/handle/11299/115341
Willman, H. B. (1973). Rock stratigraphy of the Silurian system in northeastern and northwestern Illinois. https://www.ideals.illinois.edu/handle/2142/44692
Zhang, K., & Chui, T. F. M. (2019). A review on implementing infiltration-based green infrastructure in shallow groundwater environments_ Challenges, approaches, and progress. Journal of Hydrology, 579, 15. https://doi.org/10.1016/j.jhydrol.2019.124089
Ziajahromi, S., Drapper, D., Hornbuckle, A., Rintoul, L., & Leusch, F. D. L. (2020). Microplastic pollution in a stormwater floating treatment wetland: Detection of tyre particles in sediment. Science of The Total Environment, 713, 136356. https://doi.org/10.1016/j.scitotenv.2019.136356

Chapter 7. Evaluating Stormwater Management Policies' Effects on Water Quality: Monitoring Options

7.1 Introduction

Evaluating the effectiveness of urban stormwater management policies at achieving water quality goals requires both an understanding of the fundamental science of the processes and mechanisms by which hydrology and hydraulics interact with surface water chemistry and an understanding of the applied methods that can be used to confirm this foundational understanding within a management area. Building upon previous chapters that outline the fundamental science, this chapter describes potential urban stormwater Best Management Practice (BMP) monitoring strategies for evaluating the effectiveness of various management policies and provides examples of these monitoring approaches previously employed in other urban areas. While the ultimate design of any monitoring program depends on the specific management program goals, it is assumed herein that the overarching purposes of potential District monitoring programs would be to a) confirm that the BMPs are effective at improving water quality in greater Chicago, b) help identify the BMPs that reduce pollutant loads to the "maximum extent practicable," and c) explore how monitoring can support restoring the integrity of streams in the region via the Total Maximum Daily Load (TMDL) process.

Urban stormwater BMP monitoring can occur at a range of spatial and temporal scales and use a variety of methodologies. These include BMP-scale removal efficiency testing, stormwater characterization studies, small watershed studies, synoptic surveys, inspection and performance monitoring, as well as modeling studies. Each of these approaches are tailored to the evaluation of the effectiveness of specific types of management objectives.

7.1.1 Monitoring BMP Effectiveness (Efficiency Rates) – General Trends

Existing urban stormwater-quality management policies in other regions of the country can provide a framework for evaluating the effectiveness of District management policies and can suggest some general relationships likely to hold for standard BMPs in similar hydrologic settings. In particular, the BMP efficiency rates adopted by the Chesapeake Bay Program (CBP) strongly support the District proceeding under the assumption that BMPs have a positive effect on water quality. Over the past two decades, the CBP has had panels of experts define a set of "accepted" BMP pollutant removal efficiencies for use in their TMDL program based on the type of BMP, its capacity relative to the drainage area, and the underlying soil hydrologic group (Table 52). For several of the most important conventional pollutants – suspended solids and major nutrients – the most common stormwater BMPs are clearly effective at reducing pollutant loads to waterways. In addition, expert opinion elicited by the CBP on the effectiveness of BMPs for toxics management concluded that while buffer strips were most effective, stormwater detention and retention practices still provided benefits (Comstock et al., 2015).

The CBP guidance also includes adjustments to the long-term average BMP-specific efficiency rates that account for the ratio of stormwater volume detained/retained to impervious area drained (Figure 89). Aside from the greater efficiency of volume control practices (RR on Figure 89) relative to release rate practices (ST on Figure 89), it is worth noting that the District's volume control requirement of 1-inch corresponds to a relatively high level of BMP effectiveness, i.e., within 20% of the maximum for both RR and ST practices. Since stormwater treated in retention/detention systems will have already been passed through volume control structures, it can be assumed that the efficacy of RR practices should lie to the right of the 1-inch marker on these curves, i.e., increase a modest amount.

Table 52. Chesapeake Bay Program-Approved BMP Pollutant Removal Efficiencies (Comstock et al., 2015)

URBAN BMP		Total Nitrogen	Total Phosphorus	TSS
		MASS LOAD REDUCTION (%)		
Wet Ponds and Constructed Wetlands		20	45	60
Dry Detention Ponds		5	10	10
Dry Extended Detention Ponds		20	20	60
Infiltration		80-85	85	95
Filtering Practices (Sand Filters)		40	60	80
Bioretention	C & D w/UD	25	45	55
	A & B w/ UD	70	75	80
	A & B w/o UD	80	85	90
Permeable Pavement	C & D w/UD	10–20	20	55
	A & B w/ UD	45–50	50	70
	A & B w/o UD	75–80	80	85
Grass Channels	C & D w/o UD	10	10	50
	A & B w/o UD	45	45	70
Bioswale	aka dry swale	70	75	80

A broader review of stormwater BMP performance is embodied in the International BMP Database project, which is led by the Water Research Foundation (WRF). It is also sponsored by the U.S. Environmental Protection Agency (USEPA), the Environmental and Water Resources Institute (EWRI) of ASCE, the Federal Highway Administration (FHWA), and others. The 2020 Statistical Summary of the International BMP Database (WRF, 2020) (Table 53), like the CBP Efficiency Rate table, shows significant reduction in concentrations of TSS, TN, and TP between BMP inlets and outlets for the most prevalent types of BMPs in use in greater Chicago. This summary, which includes the results of multiple BMP tests in the Chesapeake Bay region, also shows that total metal levels are effectively reduced; dissolved metals are generally reduced but not as efficiently as the particulate fraction that drives total metal removal. In contrast, concentrations of total dissolved solids (mostly salt) were not found to be reduced by BMPs. The increases in TDS in Table 53 may result from net dissolution of organic substrates or merely be an artifact of mixing salts over time in ponds and wetlands.

Note that Table 53 reports the changes in pollutant concentration from inlet to outlet, rather than the change in loads to surface waterways as in Table 52. Since the load is computed as the product of concentration times flow, diversion of stormwater to infiltration will reduce loads even if concentrations do not change. This likely explains why the results for total phosphorus and orthophosphate in bioretention, grass strips, and grass swales differ from the CBP accepted values.

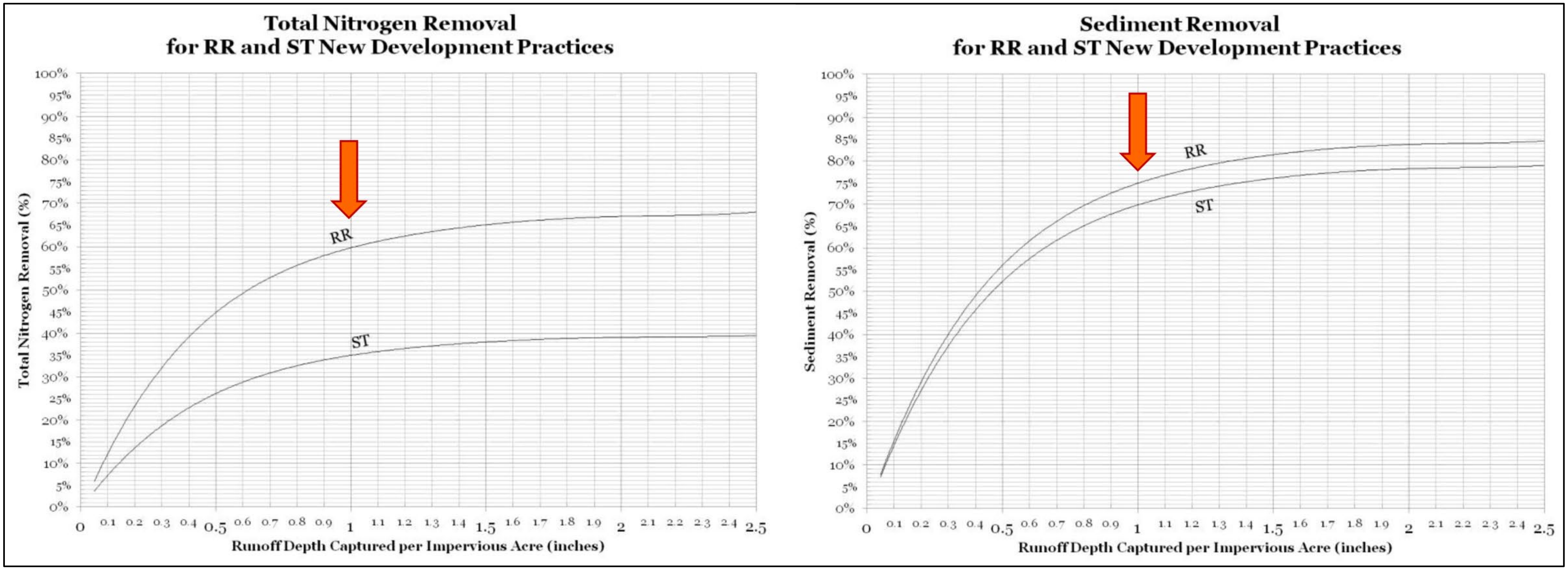


Figure 89. Chesapeake Bay Program "Adjustor Curves" to account for sizing of BMPs relative to impervious area drained. BMP Efficiency Rate (Removal Percent) is a function of impervious surface runoff depth captured for volume control (RR) and retention/detention (ST) BMPs. The arrows correspond to the District's volume control requirement. The phosphorus removal curves fall between the nitrogen and suspended sediment curves. Comstock et al. (2015)

Table 53. BMP effectiveness for various pollutants from Statistical Summary of International BMP Database (WRF, 2020). Symbols indicate statistical significance of concentration reductions from inlet to outlet for three different statistical tests.

Pollutant	*Detention Basin*	*Retention Pond*	*Constructed Wetland*	*Bioretention*	*Grass Strip*	*Grass Swale*	*Porous Pavement*	*Hydro-dynamic Separator*	*Media Filter*
TSS	▼▼▼	▼▼▼	▼▼▼	▼▼▼	▼▼▼	▼▼▼	▼▼▼	▼▼▼	▼▼▼
TDS	◇◇◇	△△△	◇◇△	△△◇	△△△	◇◇◇	NA	◇◇△	△△△
Fecal Coliforms	▼▼▼	◇▼▼	▼▼▼	▼▼▼	◇△◇	◇◇◇	NA	◇◇△	◇▼▼
Total Phosphorus	▼▼▼	▼▼▼	▼▼▼	◇△△	△△△	△△△	▼▼▼	◇▼▼	▼▼▼
Orthophosphate	◇◇◇	▼▼▼	◇◇▼	△△△	△△△	△△△	◇◇△	◇◇◇	▼▼▼
Total Nitrogen	◇◇▼	▼▼▼	◇◇◇	▼▼▼	◇◇▼	◇▼◇	NA	◇◇◇	◇▼▼
Total Cu	▼▼▼	▼▼▼	▼▼▼	▼▼▼	▼▼▼	▼▼▼	▼▼▼	◇◇▼	▼▼▼
Total Fe	▼▼▼	NA	NA	◇◇△	▼▼▼	◇▼▼	NA	NA	▼▼▼
Total Pb	▼▼▼	▼▼▼	▼▼▼	▼▼▼	▼▼▼	▼▼▼	▼▼▼	◇▼▼	▼▼▼

▼ influent/effluent comparison test indicates significant reduction in concentrations

◇ influent/effluent comparison test indicates no significant difference in concentrations

△ influent/effluent comparison test indicates significant increase in concentrations

NA not available or less than three studies for BMP/constituent

Chapter 5 of this report lists many additional studies that provide support for the findings of the CBP and WRF for a range of pollutants from solids, nutrients, bacteria, metals and some organic compounds. Although there is relatively little BMP efficiency data for organic compounds, both the WRF synthesis and the review provided in Chapter 5 suggest that BMPs that allow for settling, filtration, or infiltration can reduce the transport of at least particle-associated organic pollutants to area waterways. Spahr et al. (2020) report that BMPs sediment and absorb hydrophobic trace organic compounds (alkylphenols and alkylphenol ethoxylates) far better than hydrophilic compounds (diuron, isoproturon, simazine, atrazine, and others) because of their greater tendency to absorb in particles and media. Thus, atrazine and other hydrophilic trace organics were not well removed in a detention basin but were somewhat retained (20-50%) in biofiltration BMPs. Glyphosate was removed much more effectively due to transformation processes or possibly by virtue of its ability to form surface complexes.

7.1.2 BMP Effectiveness – Impact of Hydrologic Processes

Hydrologic processes play a central role in defining BMP efficiency rates. Upon detaining stormwater, the BMPs slowly release the water via one of three paths: i) discharge directly to waterways (which has the greatest potential for pollutant transport), ii) infiltration to groundwater, or iii) evaporation or transpiration by vegetation (Table 54). Infiltration reduces the transport load without necessarily affecting the pollutant concentration in water remaining within the BMP. Evapotranspiration (ET) reduces discharge but, acting alone, it would raise constituent concentrations without reducing loads transmitted to waterways. However, the longer water is retained in a structural BMP, the greater is the potential for processes that retain or degrade pollutants to act. Finally, some organic pollutants may degrade, and others may be taken up, settled out, or adsorbed more completely at long detention times.

Infiltration BMPs – permeable pavement, bioretention, and bioswales – have very high pollutant load reduction efficiencies (Table 52) since they divert most stormwater flow before it reaches surface waterways. Filtering BMPs also achieve good load reduction rates. Wet ponds and wetlands are less effective than infiltration and filtering BMPs, but far more effective than dry detention systems that rely only on sedimentation of particulate pollutants. Notice also that the addition of underdrains reduces the effectiveness of infiltration BMPs for all pollutants considered because they transmit outflowing water from infiltration back to surface waterways.

Table 54. Hydrologic function and processes of stormwater infrastructure. Hydrologic fate percentages are approximate over water over storm events (Bell et al., 2019).

	Function		*Hydrologic Fate of Stormwater (Percent of Inflow)*		
SCM Type	*Stormwater Detention*	*Volume Control*	*Infiltration*	*Evapotranspiration*	*Transmission*
Cisterns	+++		30-80	0-10	20-70
Wet retention pond	+++		0-25	0-70	30-95
Constructed wetland	+++		0-60	0-40	25-95
Dry detention basin	+++		15-70	10-35	10-70
Porous pavement		+++	5-90	5-20	0-80
Bioretention		+++	5-20	5-90	0-80
Green roof		+++	0	30-80	20-70
Infiltration trench		+++	60-80	0-10	20-40

7.1.3 BMP Effectiveness – Age and Maintenance

As they age, effective BMPs necessarily accumulate persistent pollutants and can experience a decline in their ability to retain some pollutants. In particular, the performance of ponded BMPs is affected by i) accumulation of organic matter in sediments, ii) growth of trees that block wind, and iii) infestation with aquatic vegetation that inhibits mixing (e.g. duck weed) (Janke et al., 2017; Taguchi et al., 2020). All of these increase the potential for water column stratification, even in shallow ponds. As is well known from the study of lakes, this can cut off the supply of oxygen to sediments that would otherwise come via mixing of oxygenated surface water and bottom waters. Continued microbial respiration in sediments and bottom waters under such conditions can result in seasonal oxygen depletion. Anoxia can have significant negative impacts on the pond bottom waters, which often bear high levels of undesirable constituents such as ammonia, phosphate, hydrogen sulfide, iron, and other metals. Release of pollutants such as phosphorus from sediments via this process is termed "internal loading," (Taguchi et al., 2020). Mercury methylation, which produces the most bioaccumulative species of Hg, also occurs much more rapidly in anoxic than oxic compartments of freshwater systems (Monson, 2007). On the other hand, denitrification is enhanced in low-oxygen waters, so there may be an additional net sink for nitrogen under these conditions. In addition, certain chlorinated compounds may be anaerobically respired via reductive dehalogenation.

An additional cause of shallow pond stratification in areas like the Chicago region is the accumulation of saline snowmelt at the bottom of stormwater retention basins and constructed wetlands (Marsalek, 2003). The net effect of roadsalt-polluted water retention in ponds is to spread out spikes in chloride export to waterways over time. However, multiple studies have found that the dense water layer near the sediments in BMPs can take on the chemical signature of anoxic lake hypolimnia (Janke, 2021; Marsalek, 2003; Taguchi et al., 2020). For the same reason, saline runoff turned a dimictic Michigan lake into a meromictic system that doesn't turnover either in fall or spring (Mayer et al., 2008). Note that this could also happen in storage vaults.

Fortunately, there are solutions to these problems. Proper SCM maintenance, which includes dredging, can be employed to reduce sediment oxygen demand and remove the source of some contaminants.

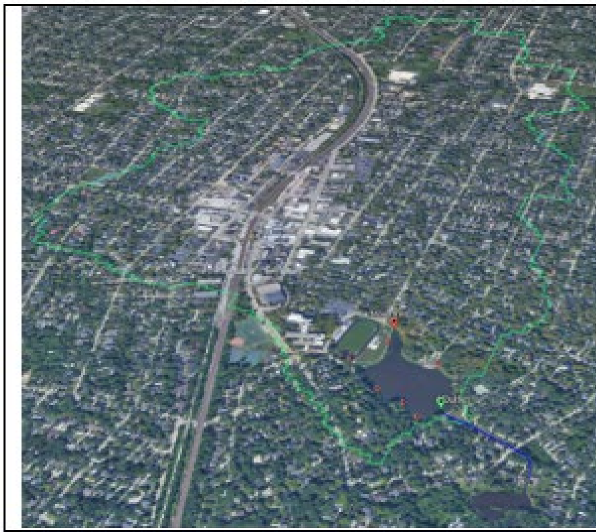
7.1.4 Applicability to the District

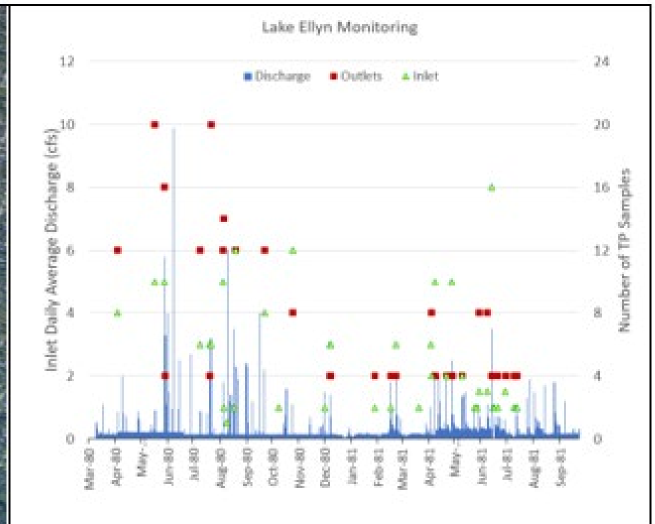
Inevitably, one must ask how well the previously cited BMP literature applies to greater Chicago since few of those BMP studies were performed in the area. The answer depends on how much the factors that affect BMP performance differ between greater Chicago and the locations where BMPs have previously been assessed. As Chapter 6 shows, the Chicago metro region's geological setting is unique in ways that need to be explored since they may affect infiltration and related impacts, but the dominant factors affecting BMP performance are similar enough to examples from the literature that they can serve as proxies for local studies. For example, the soil hydrologic groups in Chicago region (B, C, D) are also considered in the CBP efficiency rate summary, and the northern end of the Chesapeake watershed has a climate similar to Chicago's.

There is one study reported in the International BMP Database that was conducted a short distance outside the District in DuPage County. The study is worth considering in some detail because it illustrates the traditional approach of comparing inlet and outlet event mean concentrations for multiple storms in order to assess BMP efficacy. It was conducted at Lake Ellyn, which is a wet retention pond whose 300-ha sewershed comprises residential areas and much of downtown Glen Ellyn (Figure 90A). Flows were measured and water samples obtained at the main inlet, Municipal Separate Storm Sewer System (MS4) outfall, and from the surface and submerged depths at the outlet. Six storm sewer discharges that drained small basins were not monitored.

The Lake Ellyn study was conducted over several months in 1980 and 1981. Samples were collected on 42 different days. Since some storms spanned multiple days, a total of 19 events were analyzed. As many as 20 outlet and 16 inlet samples were collected during a single day (Figure 90B). The parameters measured included suspended solids, multiple forms of nitrogen, phosphorus, and metals in different sample fractions (dissolved, suspended, and unfiltered) as well as basic parameters such as *p*H, DO, specific conductance and chloride. In all, nearly 14,000 measurements were reported (Figure 90C).

Examining the final event mean concentration data for total phosphorous (TP) illustrates how a relatively homeostatic system can exhibit seemingly disparate performance metrics. The ratio of inlet to outlet event mean concentrations (EMC) ranges from 0.12 to 1.12 for total phosphorus (Figure 90D). However, it is clear that the TP in pondwater is maintained in a relatively narrow range of 0.1 to 0.28 mg-P/L over the study period. Thus, the variability in removal efficiency, which results from the variation in inlet concentrations, is real but is mainly caused by the performance metric chosen. Because so many events were sampled, the confidence intervals of the mean inlet-outlet ratio of 0.40 ± 0.12 (95% CI) are small enough to indicate statistically significant treatment of the stormwater. However, a smaller number of events reported in this way may not be. Similar behavior was reported for nitrogen.





Number of Measurements Made During Study		
Parameter	Main Inlet	Outlets
TSS	S 108	T 130
TDS	T 297	T 331
TP	D 104/T 202	D 126/T 249
TN	D 103/T 203	D 126/T 249
TKN	D 104/S 98/T 99	D 123/T 123
NOx	D 104/T 105	D 126/T 126
NH4	D 206/ T 206	D 252/ T 252
Iron	D 104/S 96/T 104	D 126/S 110/T 126
Many others		
Total	6485	>7000

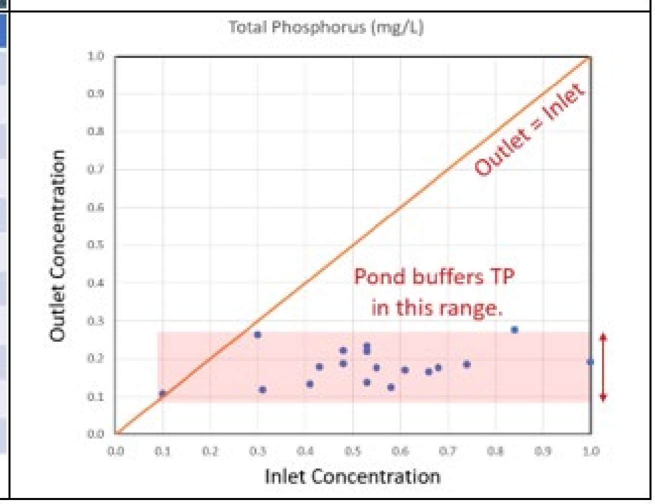


Figure 90. A (upper left): Aerial photo image of Lake Ellyn and watershed. Large red marker is main inlet. Large green marker indicates outlet location. Small red markers indicate unmonitored storm drains. B (upper right): Daily discharge at inlet with number of samples collected per day at inlet and outlet. C (lower right): Number of measurements for each parameter reported in NWIS. Letters indicate sample fraction analyzed (Dissolved, Suspended, Total). D (lower right): Comparison of inlet and outlet event mean concentrations for total phosphorus. Data from NWIS via waterqualitydata.us. Imagery from Google Earth with watershed boundary (green line) from StreamStats.usgs.gov.

Note that while the inlet-outlet concentration ratio suggests that the phosphorus removal efficiency was about 60% in Lake Ellyn, that is only the case if evaporation from the pond was negligible. If 50% of the inflow was evaporated (see Table 54), then the actual reduction in load may have been as high as 80%. Both rates are higher than the standard efficiency assumed for wet ponds by the CBP (Table 52).

As the list of analyses suggests (Figure 90C), removal efficiencies were also assessed for nitrogen, suspended and dissolved solids, and several metals in the same study. Except for TDS (chloride), both total and dissolved outlet-inlet ratios were favorable for pollutant removal in the wet retention system (Table 55), although removal of dissolved constituents was less effective. Excellent removal of suspended solids is indicated by the 0.09 outlet to inlet ratio. The lack of TDS (mainly salt) removal is expected for a non-adsorbing, not-reactive constituent. Events with outlet to inlet ratios greater than one had high antecedent levels of deicing salt in the pond.

The excellent removal efficiencies observed in the Lake Ellyn study suggest that there will be water quality benefits from implementing stormwater BMPs in Chicago comparable in magnitude to that observed elsewhere (Table 53). What is open to question is how much the stormwater loads of other pollutants of importance in the District are reduced by BMPs and whether there are any unintended negative consequences of BMPs or factors known to hurt BMP performance that apply specifically to greater Chicago.

Table 55. Summary of Event Mean Concentrations for Lake Ellyn BMP Study. Inlet and outlet concentrations are geometric means for 18 events. Outlet: Inlet ratio is average of all 18 outlet: inlet ratios per constituent (±1 SD).

Constituent	*Sample Fraction*	*Inlet Concentration*	*Outlet Concentration*	*Outlet: Inlet Ratio*
Total Suspended Solids	Suspended	239	17.4	0.09 ± 0.07
Total Dissolved Solids	Dissolved	194	491	3.34 ± 3.09
Chloride	Total	36.5	140	4.87 ± 3.39
Nitrogen, All forms	Total	3.85	1.65	0.50 ± 0.28
Phosphorus	Total	0.49	0.23	0.40 ± 0.23
Phosphorus	Dissolved	0.08	0.03	0.59 ± 0.42
Iron (Fe)	Total	7260	431	0.07 ± 0.04
Iron (Fe)	Dissolved	118	55.3	0.44 ± 0.56
Copper (Cu)	Total	47.6	5.92	0.15 ± 0.12
Copper (Cu)	Dissolved	9.41	4.36	0.52 ± 0.26
Zinc (Zn)	Total	228	26.6	0.13 ± 0.07

7.1.4.1 Unintended consequences of stormwater infiltration

The review in Chapter 6 raises concerns about one unintended consequence: infiltration of saline meltwater at infiltration based BMPs. This loading of salts likely contributes to the salinization of shallow groundwater, which raises chloride levels in greater Chicago streams during baseflow conditions and potentially impacts down-gradient groundwater aquifers. Both potential impacts require investigation and complicate the assessment of water quality benefits from infiltration based BMPs.

Since earlier work by ISWS showed that salinization of Chicagoland waterways was apparent by 2010 (Kelly et al., 2010, 2012), surface water records were reviewed to determine whether the same trends continued over the last decade using USGS' EGRET statistical software package (Zhang and Hirsch, 2019). EGRET includes the Weighted-Regression in Time Discharge and Season (WRTDS) tool, which is intended for analyzing trends in stream water quality using flow-normalized concentrations (Figure 91). WRTDS regressions remove the variability in water quality trends that arise solely from inter-annual variations in stream discharge, thus allowing the effects of constituent loading to watersheds, such as road salt applications, to be seen more clearly. The "2Q" variant of WRTDS allows one to explore differences in concentrations within different flow classes, such as baseflow (low-flow) or stormflow (high-flow) versus overall average conditions (Zhang et al., 2021).

Applying the WRTDS-2Q package to measurements of chloride in Poplar Creek near Elgin over the period 1978-2020 yielded intriguing results. In Poplar Creek, road salt applications caused increases in average stream water chloride until it reached 290 mg/L in 2005, confirming the earlier trends reported in Chapter 6 for rivers in Chicagoland. Average chloride levels have been declining slowly ever since that time, most likely because less road salt has been applied. The stream water is saltiest under winter low-flow conditions and reached chloride levels of 500 mg/L by 2004. They have declined only ~10% since. Winter high-flow chloride levels reached nearly 400 mg/L but have declined over 25% since that time. High chloride in September is indicative of salt in shallow groundwater. Note that September high-flow chloride levels peaked at about 150 mg/L in 2011, which was double the level around 1980. Even clearer evidence that shallow groundwater has been affected is the continuing trend in the September low-flow chloride levels which were still increasing through 2019 to 274 mg/L. Since September low-flow stream water contains the highest proportion of baseflow, i.e., discharged shallow groundwater, the results suggest that even with the current reduced salt applications, pollution of the shallow groundwater with chloride is continuing.

Similar trends are observed in other streams, such as Addison Creek, though it is more difficult to separate stormwater and point source effects under baseflow conditions (results not shown). Note also that the EPA chronic exposure criterion for chloride is 230 mg/L, which clearly is exceeded during winter and around 25% of the time during the fall in Poplar Creek. These results suggest the buildup of salt in groundwater is certainly worth investigating.

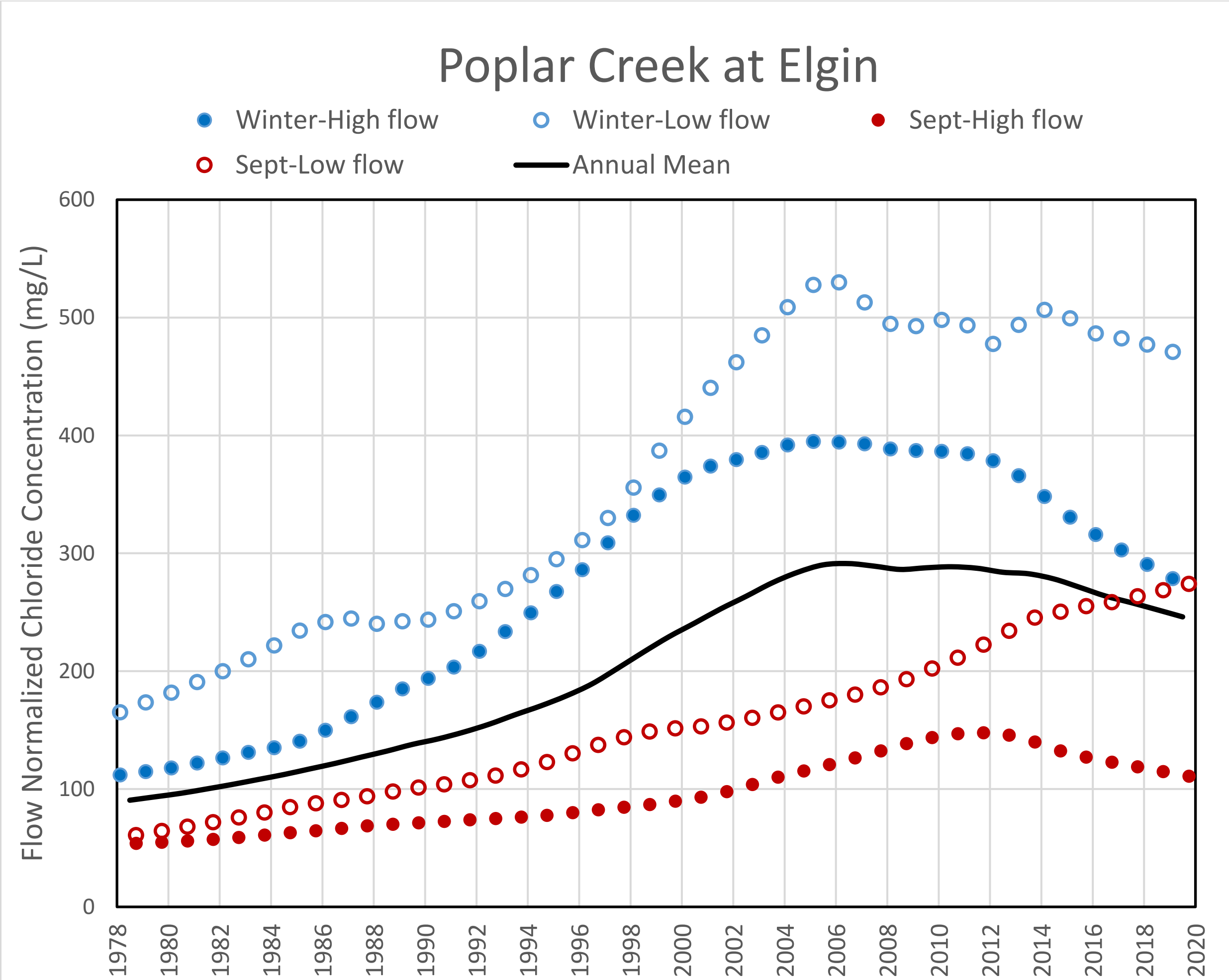


Figure 91. Average of daily flow-normalized chloride concentrations in Poplar Creek at Elgin over the months of a) January-March and b) September. Calculated from daily concentrations derived from USGS Weighted-Regression on Time Discharge and Season-2Q (Zhang and Hirsch, 2019) fitted to chloride monitoring data from USGS, ILEPA, and MWRD. Stream discharge data obtained from USGS-NWIS for gage station 05550500. NWIS data from 1977 were excluded due to anomalously high chloride levels. High-flow corresponds to flows >75th percentile. Low-flow corresponds to flows <25th percentile on each day of the specified time period.

7.2 BMP-Scale Removal Efficiency Testing

The traditional test of pollutant removal by BMPs involves selecting one or two BMPs that drain a small catchment or sewershed and quantifying the pollutant input-output budgets during the course of multiple natural storms (Figure 92). Typical watershed sizes in these studies range from a few to several hundred hectares. Tests typically quantify the pollutant input-output budgets of operating BMPs during the course of multiple natural storms, but some studies span seasons or even years (See Figure 90).

Normally the main pollutant input is borne by stormwater with land use and impervious surface cover within the drainage basin the key variables used to describe the expected pollutant loads. In a few cases, concentrations of some pollutants are high enough in precipitation or airborne particles that direct atmospheric deposition inputs must be considered as well. Of course, the main output is defined by the water flow and concentration of the pollutant in the effluent. Losses to infiltration can be estimated when the water budget is sufficiently well-defined or measured more accurately by sampling groundwater under the BMP. The balance of the pollutant budget can be assigned to degradation and/or retention within the BMP.

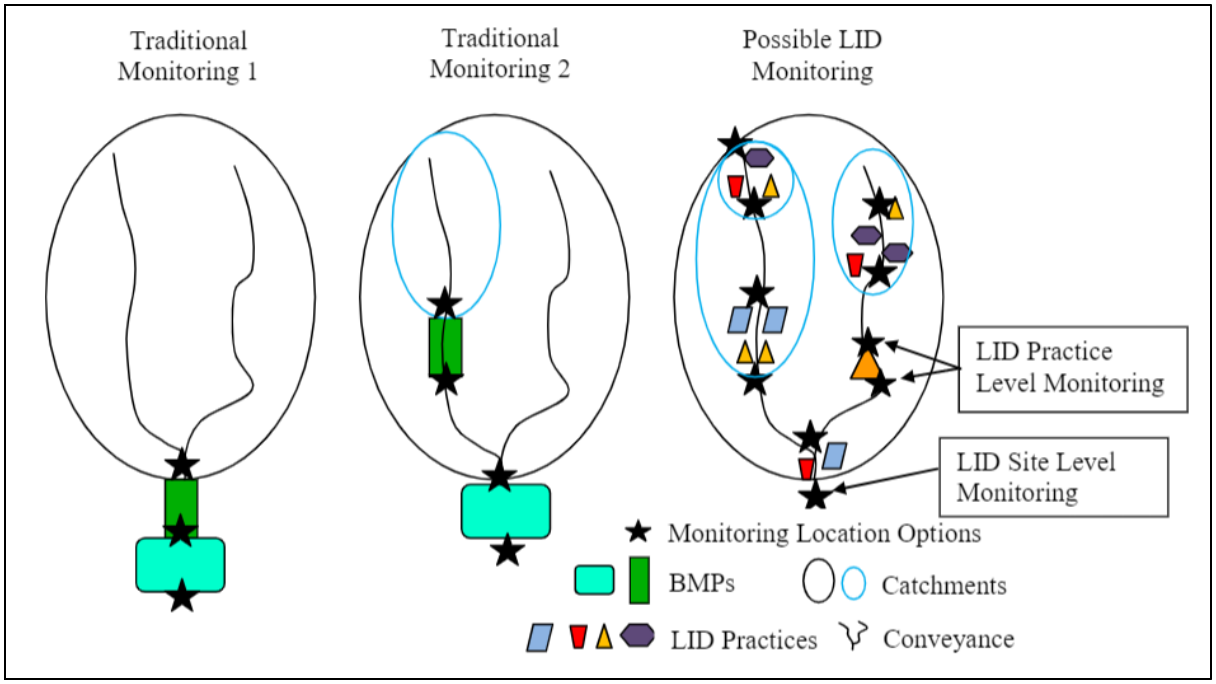


Figure 92. BMP and Low Intensity Development (LID) monitoring project designs. Monitoring locations are where flow is measured and water quality samples obtained (Geosyntec Consultants & Wright Water Engineers 2009)

The details of any such testing protocol depend on the type of BMP. To start with, BMPs vary in the difficulty of accessing inlets and outlets in order to obtain water samples. Some BMPs have well-defined inlets and outlets, while others have one but not the other. Diffuse inputs or outputs make studies of some systems difficult. Low-impact development BMPs can be more difficult to sample and therefore are often studied in groups. Some sedimentation practices – wet and dry ponds – can be monitored as well as constructed wetlands and filtration practices with underdrains. Some that are difficult to monitor are underground vaults, infiltration trenches, permeable pavement, filter strips, and bioretention practices (rain gardens).

Erickson and coauthors (2013) describe four levels of protocols for testing BMPs, each with increasing information gained and cost incurred. The two simplest tiers are i) visual inspection to look for obvious signs of malfunction and ii) hydraulic capacity testing (see Section 7.6). These protocols are primarily intended to identify factors that may impede BMP performance or indicate needs for maintenance; they will be discussed later under BMP Census (Section 7.6). Quantification of pollutant removal by BMPs is addressed in the third and fourth levels: synthetic stormwater trials and event-based monitoring. The third- and fourth-level protocols are two different ways of measuring performance related to water quality. They are aimed at quantifying the difference between the pollutant mass entering the BMP via the inlet and leaving via the outlet over multiple events.

7.2.1 Synthetic Runoff Testing

Although synthetic runoff testing is not a commonly used protocol, it is worth considering if the goal of a management program is to verify performance in multiple BMPs using a standard test. For this protocol, a known amount of pollutant of concern (or a suitable surrogate) is mixed into a known amount of water and it is discharged into a BMP. Since the mass of pollutant in the influent is accurately defined, one only needs to measure its concentration in the effluent to determine the removal efficiency. Of course, this approach is not suitable for toxic pollutants, but should prove to be useful for nutrients and for carefully selected substances that serve as surrogates for toxicants. It may also prove to be a good way to test novel designs for BMPs.

7.2.2 BMP Monitoring

Studies focused on quantifying pollutant retention during events have shown that in addition to differences between BMPs and pollutants, BMP efficacy can vary with storm intensity, pollutant load, and season. As a result, event-based measures of pollutant retention are highly variable. A key factor in designing such monitoring programs is the number of events sampled. There are standard procedures for determining the required number based on the expected uncertainty in the input-output comparisons and the desired precision of the result. For example, designing a study to measure phosphorus removal to within 10% when the measurements are accurate to 30% requires that ~40 events be monitored (Erickson et al., 2013).

While event-based studies are appropriate for systems with little long-term internal storage of stormwater, event-based performance data may be difficult to interpret in designs with more storage (WRF 2020). For example, the effluent samples from ponds during small storms are most likely to be displaced water captured previously and thus do not reflect treatment of stormwater from the current event. As a result, the WRF recommends studies that quantify a seasonal mass balance, which requires accumulating inputs and outputs over multiple storm events as well as between storms to obtain an average removal efficiency. This requires identifying sites where sample collection equipment and instrumentation can be left in place for an extended period.

7.2.2.1 Water Budget

To fully interpret a test result, both synthetic runoff testing and monitoring require that all significant terms in the water budget be quantified. Precipitation, inflow, and outflow are best measured using continuous in situ measurement methods such as gages and weirs. Water levels should be monitored in ponds in order to measure the change in water stored within the BMP. It may be possible to infer infiltration rates from the difference between inlet and outlet hydrographs or from hydraulic gradients in the groundwater table.

7.2.2.2 Water Quality Sampling

The types of water quality sampling employed depend on the constituent under consideration (Erickson et al., 2013). Some are amenable to measurement by continuous in situ sensors (oxygen, specific conductance, and others), which is the best option where feasible. Grab samples can be used for many parameters when personnel can remain on-site during events, but some analyses (dissolved oxygen and temperature) must be determined immediately while others must be analyzed later in a laboratory. Automated samplers are ideal for flow-proportional sampling schemes, which aside from continuous monitoring yield the most accurate measure of pollutant concentrations over a storm event.

There are many details to be considered in designing BMP efficiency rate investigations that depend on the specifics of the BMP type and location. When formulating specific designs, the reader should consult the many detailed discussions of monitoring program design in *Optimizing Stormwater Treatment Practices by* Erickson and coauthors (2013) and/or *Urban Stormwater Performance Monitoring* by Geosyntec Consultants & Wright Water Engineers (2009).

7.2.3 Augmented BMP Testing

It may be possible to augment relatively conventional studies of BMPs efficiency in ways that make it possible to address secondary goals efficiently. For example, the dependence of BMP efficiency on the residence time of water within them may prove relevant to decisions about release rate requirements. Also, augmented testing regimes could efficiently address aspects of the secondary questions raised in previous sections, i.e., salinization of groundwater and aging of wet ponds.

7.2.3.1 Process Studies

Investigations of how differences in design and/or management of a BMP affects performance could be used to address a wide range of questions. Of particular relevance to the decisions being faced in this context would be an exploration of what happens to water quality with longer detention times or a range of volumes of inflow. Such tests would require manipulating the flows of stormwater into a BMP or adjusting BMP features that regulate the rate of outflow.

As wet ponds are commonly employed in Chicagoland, the suggestion of Janke and coauthors (2022) to draw down water levels in retention ponds prior to storms could also be explored. Those workers showed that pollutant retention was strongly dependent on the antecedent volume of the retention pond. In other words, ponds that had water levels low enough to retain most of the stormwater removed a greater fraction of the nutrient load than ponds that quickly filled and overflowed. This would be consistent with greater benefits from lower release rates for the nutrients that were considered.

7.2.3.2 Coordination with groundwater monitoring recommendations

The study of saline water infiltration from BMPs into groundwater proposed in Chapter 6.4 would require a substantial new investment in monitoring equipment and field work. Coupling such an effort with BMP efficacy studies should make for an efficient use of resources. In particular, the existing Detailed Watershed Plan and previous watershed specific release rate study's hydrologic investigations would greatly aid in determining BMP water budgets over storms, seasons, and annual periods at certain scales and thereby make pollutant budgets more accurate.

Studies of systems designed to regulate release-rates -- dry detention systems, wet-bottom retention basins, and constructed wetlands -- would all be valuable. An intensive BMP characterization would be more detailed than either event-based or continuous monitoring of BMP described by Erickson if it were coordinated with studies of infiltration to groundwater proposed in Chapter 6.4. Herb and coauthors (2017) performed a detailed watershed study of the transport of road salt that could be integrated with a groundwater study.

Besides the opportunity to investigate saline snowmelt effects on stratification in these BMPs, other pollutants known to be influenced by hypoxia and anoxia in wet bottom ponds could be studied. Studies in the Twin Cities may serve as an example. The survey reported by Taguchi and coauthors (2020) included detailed studies of stormwater retention ponds in St Paul, MN that could serve as a model for within-pond investigations. Initially, the seasonal variation in water column stratification could be assessed by periodically measuring profiles of specific conductance, temperature, dissolved oxygen and *p*H. Then, the impact of anoxia on concentrations of impacted pollutants could be assessed by comparing concentrations in samples obtained in the surface and deep layers of the water column. If the systems behave as they did in Minnesota, elevated concentrations of phosphorus and metals would indicate internal loading of previously accumulated phosphate. Sediment incubations under oxic and anoxic conditions were also useful in illuminating the potential for internal loading in different wet-bottom ponds.

Coordinated studies of detention basins are necessarily different. Composited samples of pond sediments could be analyzed directly or after extraction to identify locations with a history of high loadings. Such monitoring in publicly accessible basins would help assure communities that potential threats to public health are managed. Resuspending sediments in synthetic stormwater could serve as a test of pollutant release to stormwater.

7.2.4 Summary: BMP-Scale Testing

Though hundreds of BMP studies have been listed in the Stormwater BMP Database and the reports discussed in Chapter 5, very few of them were performed in metro Chicago (WRF, 2020). Thus, it does seem imminently reasonable for MWRD to perform studies to confirm that the most important types of BMPs deployed in Chicagoland confer the expected water quality benefits. Expectations of such studies should, however, be modest. Given the variability in results at a single site and between sites, a handful of new studies are unlikely to greatly improve precision in presumed extent of pollutant removal by BMPs in the Chicago region. It would take a large number of events at a range of locations to distinguish Chicagoland-specific loads and BMP performance from those at other locations reported in the literature cited previously.

Additional value in BMP efficacy studies should be achievable by coupling them with a secondary purpose, such as i) testing advances in BMP technologies or operating practice especially suitable for the Chicago region, ii) studying BMP efficacy for a pollutant for which there is insufficient preexisting knowledge, or iii) investigating unintended negative consequences.

Urban Stormwater Performance Monitoring (Geosyntec Consultants & Wright Water Engineers, 2009) lists a number of possible goals for BMP testing of the type described above (Table 56). MWRD’s stated goal of discerning whether the BMPs benefit water quality approximates the broadest option. Further narrowing the objectives to BMP performance differences i) between select pollutants or ii) between key BMP types would help in making design decisions.

Table 56. Potential goals of BMP performance monitoring studies. Bold = highest priority for Chicago. After (Geosyntec Consultants & Wright Water Engineers, 2009)

- ***Does this BMP help achieve compliance with water quality standards?***
- ***How does this BMP's performance compare with the performance of other BMPs?***
- ***Does performance improve, decay, or remain stable over time?***
- ***How does this performance vary from pollutant to pollutant?***

- *How do BMP design variables affect performance?*
- *How does performance vary with different operation?*

- *What degree of pollution control or effluent quality does the BMP provide under normal conditions (i.e., representative storm types)?*
- *How does hydrology for developed conditions compare with pre-development hydrology in terms of peak flow rates, runoff volume, peak timing, site infiltration capacity, etc.?*
- *How does this normal performance vary with large or small storm events?*
- *How does this normal performance vary with rainfall intensity?*
- *Does performance vary seasonally? (For example, to what extent is infiltration reduced during cold temperatures?) Does performance vary seasonally? (For example, to what extent is infiltration reduced during cold temperatures?)*

7.3 Synoptic Surveys

Although the classic measure of individual BMP performance involves quantifying pollutant retention during individual storm events, there are other approaches that are more suitable for wider surveys of BMP condition. These involve synoptic sampling schemes designed to identify factors that govern the state of a medium to large number of systems, such as structural BMPs. Such studies do not directly evaluate BMP performance at individual sites, but they can yield data relevant to characterizing pollutant transport to streams, which is useful for modeling efforts to separate stormwater influence from groundwater and point source influences.

An example of such a survey was reported by (Taguchi et al., 2020). That study measured total phosphorus (TP) in 98 older stormwater retention ponds in the Twin Cities and compared the concentration distribution to the measured TP levels in stormwater. Each pond was sampled 6 times over a three-year period. They found that 40% had TP higher than the average TP in stormwater in the area, suggesting that those ponds would exhibit negative or near zero removal rates on average. Taguchi et al. (2020) invoked phosphorus remobilization from sediments to explain the results and investigated a subset of the ponds in sufficient detail to show that density stratification cause by saline snowmelt contributed to the formation of anoxic bottom waters that caused internal loading (see below). Note that despite the lack of specificity due to not measuring inlet or outlet TP during storms, the results argue against a high TP removal overall in those older ponds since stormwater leaving the ponds will have mixed with the pondwater.

7.3.1 Ponds and Wetlands

A study of the potential for internal loading of wet stormwater systems in the Chicago region should begin with a preliminary investigation to confirm that stratification does develop, as found in studies such as Janke and coauthors (2022) and Taguchi and coauthors (2020). Their work suggests that ponds with depths of about 2 meters, trees that shelter them from wind, and in close proximity to roads are most likely to be stratified. Assuming the preliminary investigation confirms the likelihood of stratification, then one could identify a representative set of wet-bottom retention ponds and wetlands to survey. Sites could be selected by the following criteria: pond type (natural, constructed wetland, retention), age of the ponds, their surface area, watershed impervious surface cover and density of roads, and density of storm sewer networks and land use in the watershed.

Field work would be performed seasonally (early spring, mid-summer, and fall). Field workers would use in situ probes to profile temperature, specific conductance (SC), dissolved oxygen (DO), and *p*H in order to characterize stratification. Grab samples of surface and bottom water would be obtained to measure pollutants of interest. The main goal would be to detect a difference between stratified and unstratified ponds and identify which factors contribute to the ponds being stratified.

Other studies have recovered sediments from ponds to explore the release of accumulated of pollutants (Taguchi et al., 2020) and rates of key biogeochemical processes (Blaszczak et al., 2018).

7.3.2 Dry detention basins

Dry detention basins cannot develop long-lasting stratified water columns, so all of the above factors, including saltwater should have a diminished effect. However, the possibilities for a survey with grab samples of ponded stormwater from many sites should be explored. It would give a reasonable representation of effluent from individual BMPs and identify common pollutants.

Spatial studies of detention basin sediments could also be conducted as an approach to surveying pollutant loadings to BMPs with different dominant land uses in their drainage basins. In addition, samples could be treated with synthetic stormwater to detect release of pollutants of concern. Other studies have employed sequential extraction to identify phases associated with particular pollutants. Methods of sediment sampling are explained in Erickson et al. (2013) and Geosyntec and Wright (2009).

7.3.3 Baseflow Water Quality

Fanelli and coauthors (2019) conducted a study in Baltimore that included baseflow sampling in an urban area to test for shallow groundwater salinization. They concluded that there was a strong negative effect of chloride on biological integrity. A similar study designed to sample small streams in the area under baseflow conditions would be directly relevant to assessing the impact of saline meltwater infiltration on shallow groundwater.

7.4 Small watershed studies

Stormwater pollution is well-suited to study at the scale of small watersheds, i.e., basins somewhat larger to much larger than BMP-scale basins. Like much of the rich environmental science literature on small watersheds, urban watershed studies typically employ continuous monitoring for periods of a year or longer. They can be designed to elucidate intra-watershed processes or to explore the relationships between spatial variations in water quality, land use, and other geographical variables, such as stormwater infrastructure. Note that since the District has extensively monitored stream water quality in the region for decades and since the streams and rivers monitored by the District undoubtedly contain stormwater, those data can serve as the backbone for watershed-scale studies in the region. However, the current sampling program design emphasizes monitoring the impact of point sources, making it difficult to isolate the role of stormwater. A long-term program of sampling stormwater from small watersheds across the region could be designed to span a range of different land uses, geological settings (as identified in Chapter 6.1), and perhaps types or ages of BMPs.

7.4.1 Intensive watershed studies

The aforementioned studies in the Twin Cities have been analyzed using comparative watershed methods (Hobbie et al., 2017; Janke et al., 2014, 2017). The study of chloride transport in the 433-ha Lake McCarrons watershed in St. Paul is a fine example of a watershed process study in an urban setting (Herb et al., 2017). Those workers monitored roadway runoff, baseflow, and inlet-outlet investigations of wet retention ponds within the watershed as well as the inlet and outlet of the lake over an 18-month period. The study showed that chloride salts retained in roadside ditches could be detected in stream water as late as the following fall.

7.4.2 Comparative watershed studies

Two recent examples of urban watershed studies that compare different watersheds selected to span a gradient in spatial properties are those by (Fanelli et al., 2017) and (Blaszczak et al., 2019). Those investigations quantified the influence on water quality of gradients in road density, storm sewer density, forest land cover, and other factors. Interestingly, both found that differences in specific conductance (and temporal variability in specific conductance) were important factors in assessing the impacts of urbanization on aquatic life.

7.5 Stormwater Characterization: Large Sewershed Scale

Currently, sampling and analysis of stormwater quality is not required for monitoring purposes in the same way that NPDES point source effluent and receiving water quality data are. Nevertheless, there is a large collection of data from Municipal Separate Storm Sewer System (MS4) outfalls collected mainly under programs sponsored by USEPA and USGS in the years 1980-1984 and 1992-2004 called the National Stormwater Quality Database (NSQD) (Pitt, 2018). These data make a good starting point for characterizing expected stormwater quality in a region. There are a limited number of results from DuPage County from the early 1980s and a substantial number from Minneapolis, Milwaukee, and Madison. The database does not include many results of newer studies, such as in the Twin Cities (Capitol Region Watershed District 2016; Janke et al., 2017). It also does not include results from numerous case studies of BMPs which generally include measurements of inlet stormwater quality.

Sampling stormwater can be more challenging than continuously flowing streams because constituent concentrations vary rapidly during stormwater runoff events. Some, but not all, pollutants exhibit a high pollutant concentration early in the event, termed a first-flush (Maestre et al., 2004). More recent analysis has shown that this phenomenon varies systematically between different pollutants (Reinholdt Jensen et al., 2022) Furthermore, pollutant concentrations in stormwater are lognormally distributed over wide ranges, meaning that many events need to be sampled in order to characterize a source watershed. Still, there are statistically significant differences in many pollutants, including nutrients, TSS, and metals according to the dominant land use in the watershed drained, season, and geographic location (especially EPA rainfall zone) (Maestre and Pitt, 2005). Other studies have found some evidence of land-use related differences in several trace organic pollutants (Burant et al., 2018). For example, polycyclic aromatic hydrocarbons (PAHs) were elevated in a commercial site likely due to the higher traffic there.

Monitoring of effluent from Chicago area MS4s would be helpful in TMDL and modeling efforts directed at distinguishing the expected loads from the land use classes identified as important in the NSQD – commercial, industrial, residential, institutional, freeways, and open space. Such efforts could take the studies performed in the Twin Cities, which included multi-season monitoring over an extended period, including at least 2004-2015 (Janke et al., 2017), as a model (Figure 93). Four agencies monitored stormwater quality by collecting water samples for nearly 2400 storms in 19 mainly residential watersheds (sewersheds). The Capitol Region Watershed Management District monitored much of St. Paul over that period, performing sampling during baseflow, stormflow, and snowmelt periods. In total, water quality was monitored at 11 MS4 outfalls and two wetland sites where automatic, flow-proportional samplers were deployed along with flow-measurement devices. Flow-only and pond levels were monitored at 5 stormwater ponds, 2 lakes, and an additional wetland site; six rain gages were operated. Water samples were analyzed to determine concentrations for nutrients, sediment, metals, and bacteria. Their purpose was "to characterize overall watershed health and water quality trends over time, which in turn [informed] management decisions for continued improvement of District water resources." (Capitol Region Watershed District, 2016).

7.5.1 Summary

MS4 monitoring would be invaluable for identifying the sources of pollutants of greatest concern in District stormwater. Of course, BMP-scale studies could also involve sampling small-scale MS4 sources, rather than the much larger sewershed sizes suggested here.

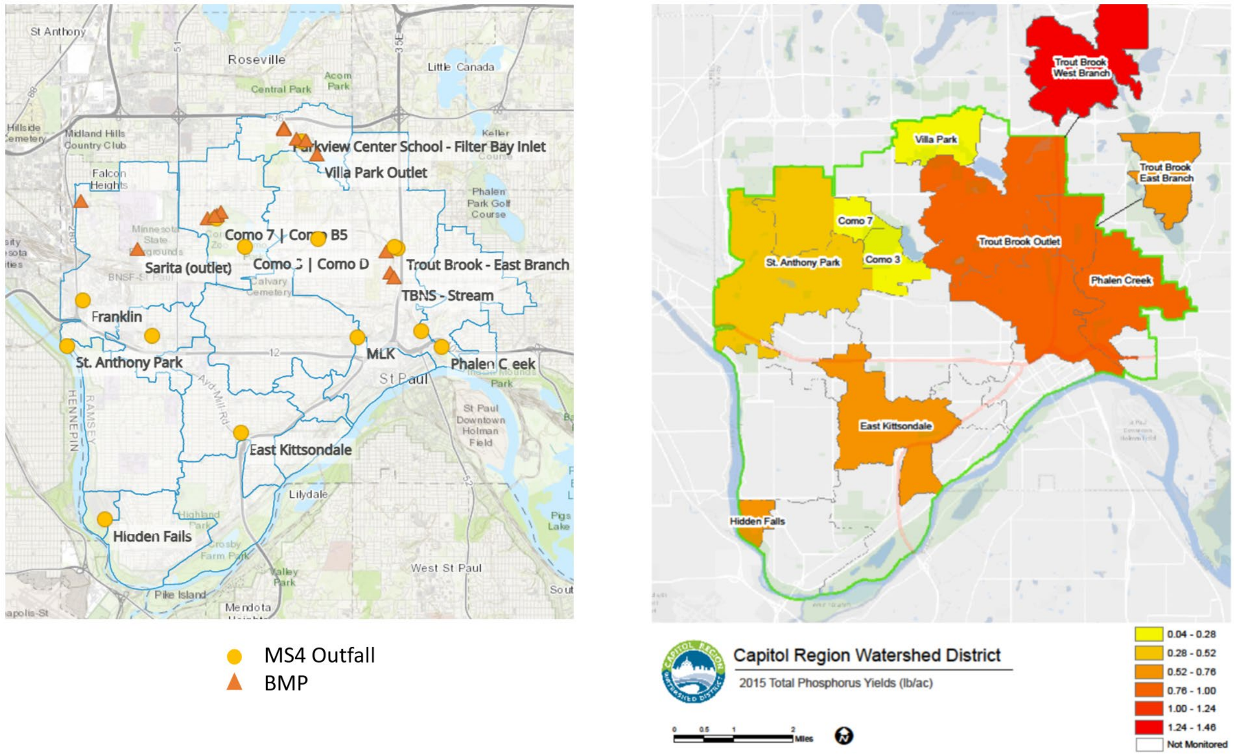


Figure 93. Stormwater quality and quantity monitoring in St. Paul's Capitol Region Watershed District. A) MS4 and BMP monitoring locations from Water Data Reporting Tool. B) Phosphorus yields (lb/ac/yr) from sewersheds monitored as indicated in A). https://waterdata.capitolregionwd.org/applications/public.html and (Capitol Region Watershed District 2016)

7.6 BMP Census

Stormwater monitoring studies focused on generating water quality data at BMPs are important, but they are also expensive and likely to be small in number compared to the number of installed BMPs. Two reasons to set up a monitoring/inspection program of BMPs that would cover most, or all of the BMP types operated in the District are: i) assure proper functioning and ii) build a database to support spatial modeling

Erickson et al. (2013) provides detailed descriptions of protocols for inspecting and monitoring SCMs. For example, finding standing water in a rain garden or dry pond long after a storm would indicate that it wasn't functioning properly. Visual inspections should also check for structural integrity, vegetation condition, sediment buildup, pipe clogging, and more. Ideally, such inspections should be performed annually. Checklists for visual inspection of a variety of SCM types are included in Erickson et al. (2013).

Hydrologic performance monitoring is the second tier in Erickson's system. Tests of the rate at which water infiltrates into the soil and extent of capture by underdrains can diagnose clogging of soils in older infiltration SCMs. The results of many such tests would also be useful for modeling BMP interactions with groundwater. Again, details of such tests are described in Erickson et al. (2013). The goal of such efforts is to make sure that BMP performance is sustained and would help identify when maintenance activities such as dredging are necessary.

7.7 Modeling and Data Analysis

Modeling and data analysis plays an important role in interpreting the data generated by environmental monitoring and in supporting decisions that are based on them. For urban stormwater, models can be applied at scales ranging from BMPs to watersheds. Some examples of each are described in the following subsections.

7.7.1 BMP Models

Process-based BMP models such as EPA's SWMM are designed to simulate the fate of pollutants after they enter BMPs during storm events. Thus, they must simulate the hydrology as well as pollutant chemistry. The representations of chemical processes in the model have frequently been upgraded from the very simple representation that was initially employed.

One example of a beneficial application of BMP modeling comes from San Jose, CA. There, workers sampled actual stormwater from sites of interest and conducted laboratory studies of how different sorbents affect pollutant sorption and transport. The goal was to assist in the design of improved BMPs in order to help meet TMDL requirements for reduced pesticide loading to surface waters.

Generally, a model that is calibrated to data from a real system that closely resembles that expected in the application of interest is a good way to predict how the BMP will behave under somewhat different conditions. For example, it would be useful to model a carefully-studied BMP and investigate the sensitivity of predicted pollutant retention under scenarios that reflect the different stormwater management policies of interest by the District, such as release rate requirements or stormwater volume control.

7.7.2 Watershed Models

Pollution of inland waterways and waterbodies can be conceptualized as occurring in two steps: i) loading to the watershed and ii) transport from the land surface to waterbodies. Of course, loadings vary with land use and the associated differences in human activities, causing the composition of stormwater to depend on land use. The land-to-water pollutant transport process, which also depends on geography, has been the subject of many studies and models analyzing the spatial variations in water quality. Stormwater BMPs, of course, are part of the land-to-water transport process. Thus, creating a database to record the characteristics of stormwater BMPs and their spatial distribution is important for modeling the watershed-scale impacts on water quality.

Watershed models that are commonly employed in support of TMDLs need to take into account the BMPs that are deployed across the watershed. For example, the Chesapeake Bay Program's CAST model serves as a central accounting system in support of the nutrient and sediment TMDLs for the bay. The model accounts for both nutrient loading processes and land-to-water transport (Chesapeake Bay Program, 2022; Shenk and Linker, 2013).

A well-regarded model used to interpret riverine water quality data is the USGS SPARROW model (Schwarz et al., 2006). It uses multivariate regression on watershed characteristics to predict constituent transport loads in rivers. The SPARROW model structure could be used to analyze data from the Chicago region in greater detail and derive estimates of pollutant loadings from different land use types.

A monitoring program could be designed to allow a SPARROW-like modeling effort to discriminate better between pollutant loads from particular land uses and the effects of various landscape metrics identified in other spatial studies (impervious surface cover, storm sewer density, etc). The effort would be greatly aided by detailed development of a database for land use (to model spatial distribution of sources) and stormwater infrastructure, similar to that available from Naperville's Open Data Portal (City of Naperville, 2020).

7.8 Summary

7.8.1 Monitoring Options

This chapter has briefly described a variety of potential monitoring programs with different specific goals that would support efforts to characterize and mitigate urban stormwater pollution in the District (Table 57).

Table 57. Summary of Options for BMP Monitoring

BMP-Scale Investigations (Section 7.2)
a. Validate performance (efficiency rate) assumptions for particular BMPs b. Establish performance (efficiency rate) metrics for particular pollutants (e.g., PFAS or Hg) c. Investigate intra-BMP processes in wet ponds: Development of anoxic conditions Internal loading from sediment accumulation of P and metals Mercury methylation due to anoxia Denitrification Reductive dehalogenation of PCBs d. Measure effects of operating conditions: Water level in ponds prior to storm Residence time (as affected by release rate criteria)
Survey properties of BMPs across District (Section 7.3)
a. Assess water quality in effluent from a particular type of BMPs (e.g., wet ponds) How does BMP age affect performance? Does pondwater quality vary with land use in watershed/sewershed? b. Test for internal loading c. Test for excessive accumulation of pollutants (e.g., in dry detention pond sediments); identify spatial location of pollutant sources d. Test BMP efficiency rate using synthetic stormwater methodology
Small watershed studies (Section 7.4)
a. Follow NRC (2009) recommendation to study BMPs in watershed context b. Develop thorough understanding of pollutant budgets and process rates c. Explore unintended impacts of BMPs via joint study with groundwater investigation d. Compare watersheds to investigate dependencies on key spatial properties of watersheds
Sewershed/watershed-scale monitoring (long-term, larger-scale) (Section 7.5)
a. Quantify MS4 loads to rivers to measure compliance with TMDL goals Investigate spatial distribution of sources b. Quantify riverine pollutant loads at locations upstream of WRPs in order to distinguish point and non-point source loads better
BMP Census (Section 7.6)
a. Develop database of BMP types and locations to document compliance with maintenance schedule and maintenance of proper functioning b. Develop database of BMP locations and properties to serve as a basis for modeling
Modeling (Section 7.7)
a. Process-based modeling of BMPs to assist with designing new technologies b. Regional pollutant loading/riverine transport model (a la SPARROW)

7.8.2 Prioritization of Options

The example of the Chesapeake Bay Program may provide a helpful illustration of how to address the uncertainty inherent in BMP implementation. The CBP consulted an expert panel, which recommended that routine, BMP-scale development decisions be based on the extant evidence that volume control and stormwater detention are, in general, effective at improving water quality for the pollutants subject to the Chesapeake Bay TMDL. While accepting these recommendations, the CBP also took steps to verify that their overall goals were being achieved over time. Thus, the CBP also implemented i) a requirement that BMPs report the results of regular inspections and maintenance, ii) a stream and river sampling program designed to monitor pollutant transport at a reasonably high level of resolution across the watershed, and iii) a modeling program to integrate observations and knowledge from the BMP to watershed scales.

The specific question of whether volume control and release rates benefit water quality in the District does not require as comprehensive a program as a TMDL does, but it can be approached in an analogous fashion. The BMP efficiency rates adopted by the CBP and net benefits of BMPs documented in other reviews could be taken as sufficient rationale to expect that the direct export of many pollutants to waterways via stormwater can be reduced by BMPs. However, two significant issues have been raised that suggest the question requires a more nuanced answer in the District. Namely, there may be i) salinization of groundwater by infiltration enhancing BMPs and ii) reduced pollutant removal efficiency in aging wet retention ponds due to internal loading. If these prove to be the salient concerns, additional policies specifying which BMP types or maintenance practices to employ may be needed to address them. To determine the magnitude of these additional concerns, the District could conduct pilot studies.

The recommendations in Chapter 6.4 and Chapter 7.4 above describe studies that would address the groundwater salinization issue. It was suggested that the groundwater work be coupled with a watershed-scale study and more intensive intra-BMP process investigations.

In order to better understand re-mobilization of pollutants from aging wet pond sediments, otherwise known as internal loading, we recommend prioritizing a two-stage sampling program. The first stage would be to collect periodic water quality samples at a large, representative number of stormwater management practices in the District. These samples would be used to identify sites that exhibit signs of pollutant re-mobilization and thus reduced removal efficiency (synoptic survey method). The sites most likely to exhibit this re-mobilization would then be sampled in detail to understand the impacts of stratification (BMP performance assessment and perhaps small watershed studies method).

In addition, it should be noted that since all BMPs age, monitoring regimes should be in place to make sure their performance is maintained and that the accumulation of pollutants within them doesn't pose a hazard. BMPs mostly accumulate pollutants over the long-term. This is a benefit until they start leaking via internal loading. Thus, a plan for monitoring pollutant buildup would be one way to manage the potential problem.

7.8.3 Causes of Impairments

The pollutants that Illinois EPA has listed with USEPA (303(d)-listed) as having pollutant levels that exceed or otherwise are in violation of water quality standards in some smaller rivers of the Chicago region are summarized in Table 58. These eight waterways are representative of the region and can reasonably be assumed to have significant impacts from stormwater relative to point source inputs. With the exception of chloride and *p*H, these pollutants should be mitigated to some extent by stormwater BMPs. Thus, there is good reason to expect a wide range of benefits to surface water quality from stormwater BMPs in greater Chicago, but any pollutant currently causing multiple impairments should be addressed in any monitoring the District implements.

Table 58. Causes of 303(d)-List water quality impairments in selected streams and smaller rivers in the Chicago region. Source: USEPA ATTAINS (2022).

Impairment	*Creek or River*							
	Addison	*Buffalo*	*Little Calumet*	*NB Chicago*	*Poplar*	*EB DuPage*	*Thorn*	*Upper Salt*
Sediment	X	X	X	X	X	X	X	X
Oxygen	X	X	X	X		X	X	X
Nitrogen	X		X	X		X	X	X
Phosphorus	X	X	X	X			X	X
Pathogens	X	X	X	X	X		X	X
Chloride	X	X		X	X	X	X	X
Metals & Metalloids	X			X	X		X	X
Oil	X		X				X	
Pesticides	X			X		X	X	X
Other Organics	X			X			X	X
*p*H						X		X

"NB" = North Branch; "EB" = East Branch.

Metals are a cause of impairments in most of these waterways (Table 59), as is common in urban stormwater (McFarland et al., 2019). The particular metals and metalloids reported as impairing water quality in these streams include arsenic, cadmium, chromium, copper, lead, mercury, silver, and zinc. Silver is not common in urban stormwater, so it may be a legacy pollutant that originated from a point source. Mercury is a well-known non-point pollutant (Mason and Sullivan, 1998) and is also a cause of impairments in the Des Plaines River from Brookfield to the Wisconsin border. Arsenic pollution may be a result of its use as an herbicide (Whitmore et al., 2008). Note that iron is not a listed cause of impairment.

Table 59. Metals causing water quality impairments in selected streams and smaller rivers in the Chicago region. Source: (USEPA 2022)

	Creek or River							
Pollutant	*Addison*	*Buffalo*	*Little Calumet*	*NB Chicago*	*Poplar*	*EB DuPage*	*Thorn*	*Upper Salt*
As						X	X	X
Ag				X				
Cd				X				
Cr	X							
Cu	X			X				
Hg				X				X
Ni	X			X				X
Pb				X				
Zn							X	

"NB" = North Branch; "EB" = East Branch

A handful of organic compounds are also listed as causes of impairments. Hexachlorobenzene and PCBs are persistent chlorinated compounds derived from legacy pollution and/or atmospheric transport (Table 60). The remainder are insecticides and herbicides. Note that none of the 16 pesticides most commonly detected in urban waterways and therefore designated as "urban signature pesticides" (Nowell et al., 2021), are listed as causes of impairments, although USGS surveys have detected all 16 in the Chicago Shipping and Sanitary Canal. The recent review of pesticides in urban stormwater by Spahr et al. (2020) suggests that diuron, terbutryn, bromacil, atrazine, and simazine are often responsible for the bulk of toxicity of pesticides to algae.

Table 60. Organic compounds and pesticides causing water quality impairments in selected streams and smaller rivers in the Chicago region. Source: USEPA ATTAINS (2022).

	Creek or River							
Pollutant	*Addison*	*Buffalo*	*Little Calumet*	*NB Chicago*	*Poplar*	*EB DuPage*	*Thorn*	*Upper Salt*
Aldrin	X			X				
α-BHC								
Chlordane				X			X	
DDT	X			X			X	
Dieldrin						X	X	
Endrin				X			X	
Hexachloro-benzene	X			X			X	X
Methoxychlor						X		X
PCBs	X			X			X	X

"NB" = North Branch; "EB" = East Branch

7.8.4 Prioritization of Constituents to Include in Monitoring

There can be several reasons to monitor any given constituent, including the level of risk it poses as a pollutant and its significance for understanding the processes and systems involved (Table 61). Assuming a general sampling plan is desired rather than one aimed at a particular pollutant, most of the MWRD priority pollutants are worth further investigation, though not always as a pollutant. Reasons to not include a 303(d)-listed pollutant would be if it wasn't expected to be in stormwater. Silver is an example of this. Also, since pharmaceuticals are not frequently detected in stormwater except in locations where leakage from sanitary sewers to storm sewers is suspected (Masoner et al., 2019), it would not be necessary to monitor them in locations known to not have such leaks.

As there are mercury impairments on a significant stretch of the Des Plaines River (USEPA ATTAINS, 2022) and two of the streams considered here (Table 59), monitoring methylmercury, the toxicologically most relevant form of mercury, is advisable. Masoner et al., (2019) also reported substantial – mean 0.25 ng/L – concentrations of methylmercury in a survey of stormwater sampled at 20 different conveyance structures in 17 states including Indiana, Wisconsin, and Minnesota.

Table 61. Recommendations and rationale for monitoring constituents.

Constituent	*Identified as priority by MWRD?*	*Groundwater Sampling Priority*	*IEPA 303(d) Priority Rank*	*Rationale for Including Constituent*
Chloride	Yes	High	Medium/Low	Key pollutant
Phosphate	Yes	High	Medium/Low	Key pollutant
Nitrate	Yes	High	Medium	Key pollutant
PFAS	No	High	N/A	Emerging pollutant
Water Isotopes	No	High	N/A	Scientific
Copper	No	Moderate	Medium	Current pollutant
Zinc	No	Moderate	Low	Current pollutant
Iron	Yes	Moderate	No	Scientific
Manganese	No	Moderate	No	Scientific
Lead	No	Moderate	Medium	Legacy pollutant
Nickel	No		Medium	Current pollutant
Chromium	No		Medium	Current pollutant
Arsenic	No		Medium	Current pollutant
Mercury & Methylmercury	No		Medium	Current and legacy pollutant
Pesticides	Yes	Moderate	Yes	Current pollutant [a]
PCBs	Yes	N/A	Medium	Legacy pollutant
Microplastics	No	Moderate	N/A	Current pollutant
Oil and Grease	Yes		Medium	Current pollutant
Pharmaceuticals and VOC	Yes	Moderate	Medium (HCB)	HCB a current pollutant [b]
Silver	Yes	Low	Low/Medium	Low priority
Pathogens	No	Low	Low/Medium	Current pollutant
*p*H (Alkalinity)	Yes		Low	Scientific
Dissolved oxygen (BOD/COD)	Yes		Low/Medium	Scientific

[a] *Include all 303(d)-listed pesticides plus urban signature pesticide*

[b] *Note that BTEX compounds were rarely detected in stream monitoring and may therefore may not need to be included here. Hexachlorobenzene should be studied.*

7.9 References

Bell CD, Spahr K, Grubert E, Stokes-Draut J, Gallo E, et al. 2019. Decision Making on the Gray-Green Stormwater Infrastructure Continuum. J. Sustainable Water Built Environ. 5(1):

Blaszczak JR, Delesantro JM, Zhong Y, Urban DL, Bernhardt ES. 2019. Watershed urban development controls on urban streamwater chemistry variability. Biogeochemistry. 144(1):61–84

Blaszczak JR, Steele MK, Badgley BD, Heffernan JB, Hobbie SE, et al. 2018. Sediment chemistry of urban stormwater ponds and controls on denitrification. Ecosphere. 9(6):e02318

Burant A, Selbig W, Furlong ET, Higgins CP. 2018. Trace organic contaminants in urban runoff: Associations with urban land-use. Environ. Pollut. 242(Pt B):2068–77

Capitol Region Watershed District. 2016. 2015 Stormwater Monitoring Report . Capitol Region Watershed District

Chesapeake Bay Program. 2022. Chesapeake Assessment Scenario Tool. https://cast.chesapeakebay.net

City of Naperville. 2020. City of Naperville Open Data Poral. Open Data Naperville. https://data.naperville.il.us

Comstock S, Crafton S, Greer R, Hill P, Hirschman D, et al. 2015. Recommendations of the Expert Panel to Define Removal Rates for New State Stormwater Performance Standards . Chesapeake Bay Program

Erickson AJ, Weiss PT, Gulliver JS. 2013. Stormwater Treatment Processes. In Optimizing Stormwater Treatment Practices, pp. 23–34. New York, NY: Springer New York

Fanelli RM, Prestegaard KL, Palmer MA. 2019. Urban legacies: Aquatic stressors and low aquatic biodiversity persist despite implementation of regenerative stormwater conveyance systems. Freshwater Science. 38(4):818–33

Fanelli R, Prestegaard K, Palmer M. 2017. Evaluation of infiltration-based stormwater management to restore hydrological processes in urban headwater streams. Hydrol. Process. 31(19):3306–19

Geosyntec Consultants, Wright Water Engineers. 2009. Urban Stormwater BMP Performance Monitoring. Urban Water Resources Research Council. Water Environment Research Federation

Herb W, Janke B, Stefan H. 2017. Study of De-icing Salt Accumulation and Transport Through a Watershed. Minnesota Department of Transportation, Research Services & Library

Hobbie SE, Finlay JC, Janke BD, Nidzgorski DA, Millet DB, Baker LA. 2017. Contrasting nitrogen and phosphorus budgets in urban watersheds and implications for managing urban water pollution. Proc Natl Acad Sci USA. 114(16):4177–82

Janke BD, Finlay JC, Hobbie SE, Baker LA, Sterner RW, et al. 2014. Contrasting influences of stormflow and baseflow pathways on nitrogen and phosphorus export from an urban watershed. Biogeochemistry. 121(1):209–28

Janke BD, Finlay JC, Hobbie SE. 2017. Trees and streets as drivers of urban stormwater nutrient pollution. Environ. Sci. Technol. 51(17):9569–79

Janke BD, Finlay JC, Taguchi VJ, Gulliver JS. 2022. Hydrologic processes regulate nutrient retention in stormwater detention ponds. Sci. Total Environ. 823:153722

Janke BD. 2021. Detecting phosphorus release from stormwater ponds to guide management and design. 597, St Anthony Falls Laboratory, Minneapolis

Kelly WR, Panno SV, Hackley KC, Hwang H-H, Martinsek AT, Markus M. 2010. Using chloride and other ions to trace sewage and road salt in the Illinois Waterway. Applied Geochemistry. 25(5):661–73

Kelly WR, Panno SV, Hackley KC. 2012. Impacts of road salt runoff on water quality of the chicago, illinois, region. Environmental & Engineering Geoscience. 18(1):65–81

Maestre A, Pitt R, Durrans R, Chakraborti S. 2004. Stormwater Quality Descriptions Using the Three-Parameter Lognormal Distribution. University of Alabama, Tuscaloosa

Maestre A, Pitt R. 2005. Nonparametric Statistical Tests Comparing First Flush and Composite Samples from the National Stormwater Quality Database. University of Alabama, Tuscaloosa

Marsalek J. 2003. Road salts in urban stormwater: an emerging issue in stormwater management in cold climates. Water Sci. Technol. 48(9):61–70

Masoner JR, Kolpin DW, Cozzarelli IM, Barber LB, Burden DS, et al. 2019. Urban stormwater: an overlooked pathway of extensive mixed contaminants to surface and groundwaters in the united states. Environ. Sci. Technol. 53(17):10070–10081

Mason RP, Sullivan KA. 1998. Mercury and methylmercury transport through an urban watershed. Water Res. 32(2):321–30
Mayer T, Rochfort Q, Borgmann U, Snodgrass W. 2008. Geochemistry and toxicity of sediment porewater in a salt-impacted urban stormwater detention pond. Environ. Pollut. 156(1):143–51
McFarland AR, Larsen L, Yeshitela K, Engida AN, Love NG. 2019. Guide for using green infrastructure in urban environments for stormwater management. Environ. Sci.: Water Res. Technol.
Monson B. 2007. Effectiveness of Stormwater Ponds/Constructed Wetlands in the Collection of Total Mercury and Production of Methylmercury. Minnesota Pollution Control Agency
National Research Council. 2009. Urban stormwater management in the united states. Washington, D.C.: National Academies Press
Nowell LH, Moran PW, Bexfield LM, Mahler BJ, Van Metre PC, et al. 2021. Is there an urban pesticide signature? Urban streams in five U.S. regions share common dissolved-phase pesticides but differ in predicted aquatic toxicity. Sci. Total Environ. 793:148453
Pitt R. 2018. The National Stormwater Quality Database (NSQD), Version 4.02. Department of Civil and Environmental Engineering, University of Alabama
Reinholdt Jensen DM, Sandoval S, Aubin J-B, Bertrand-Krajewski J-L, Xuyong L, et al. 2022. Classifying pollutant flush signals in stormwater using functional data analysis on TSS MV curves. Water Res. 217:118394
Schwarz GE, Hoos AB, Alexander RB, Smith RA. 2006. The SPARROW Surface Water-Quality Model: Theory, Applications and User Documentation. U.S. Geological Survey, Techniques and Methods. 6-B3:1–29
Shenk GW, Linker LC. 2013. Development and application of the 2010 Chesapeake Bay watershed total maximum daily load model. J. Am. Water Resour. Assoc. 49(5):1042–56
Spahr S, Teixidó M, Sedlak DL, Luthy RG. 2020. Hydrophilic trace organic contaminants in urban stormwater: occurrence, toxicological relevance, and the need to enhance green stormwater infrastructure. Environ. Sci.: Water Res. Technol. 6(1):15–44
Taguchi VJ, Olsen TA, Natarajan P, Janke BD, Gulliver JS, et al. 2020. Internal loading in stormwater ponds as a phosphorus source to downstream waters. Limnol. Oceanogr. 5(4):322–30
USEPA. 2022. Get Data: Access Public ATTAINS Data | US EPA. https://www.epa.gov/waterdata/get-data-access-public-attains-data
Water Research Foundation. 2020. International stormwater bmp database: 2020 summary statistics. Project No. 4968, Water Research Foundation
Whitmore TJ, Riedinger-Whitmore MA, Smoak JM, Kolasa KV, Goddard EA, Bindler R. 2008. Arsenic contamination of lake sediments in Florida: evidence of herbicide mobility from watershed soils. J. Paleolimnol. 40(3):869–84
Zhang Q, Hirsch RM. 2019. River water-quality concentration and flux estimation can be improved by accounting for serial correlation through an autoregressive model. Water Resour. Res. 55(11):9705–23
Zhang Q, Webber JS, Moyer DL, Chanat JG. 2021. An approach for decomposing river water-quality trends into different flow classes. Sci. Total Environ. 755(Pt 2):143562

Chapter 8. Watershed Pilot Analysis [WMO Article 208.4]

8.1 Introduction

8.1.1 Background

Results from the literature review of *Stream Channel Dynamics in Urban Settings* and *Relations between Watershed Management Strategies and Stream Erosion, Turbidity, and Sedimentation* indicate that 1) channel erosion is a common problem in urban streams and that this erosion is often related to changes in the magnitudes of relatively frequent flood events and 2) implementation of stormwater management measures to control peak discharge—particularly detention-based measures—results in longer duration of elevated discharges as the flow recedes from the peak discharge to the baseflow conditions. These findings from the literature review suggest that efforts to control release rates of stormwater in urban environments should consider the trade-off between reducing peak discharges of extreme events and increasing the duration of flows of moderate size through stormwater release practices. Increases in duration of flows of moderate size can potentially lead to increases in the length of time that flow exceeds a critical threshold for sediment movement compared to pre-development conditions. The literature also indicates that the relation between stormwater management measures implemented to attain watershed specific release rates and the corresponding impact on streambed erosion and sedimentation is complex and dependent on a range of channel and watershed specific factors. Factors that can affect this relation include the type and size of stormwater management measures, their arrangement in the watershed relative to the stream network, the geometric properties of the stream channels (slope, width, cross section shape), and the properties of sediment on the streambed.

The literature review of *Stream Channel Dynamics in Urban Settings* highlighted approaches that could be used to evaluate the potential for stream erosion based on results from the HEC-RAS models developed for the initial evaluation of watershed specific release rates. In particular, this review showed that stream power, the time rate of energy expenditure of flowing water in a river, provides a fundamental metric for predicting rates of bed-material transport in natural rivers. As discussed in the literature review, spatial and temporal variability of flow, as well as spatial heterogeneity in channel form, result in spatial and temporal variability in hydraulic conditions. For any given amount of flow, spatial variability in hydraulic conditions produces spatial gradients in bed shear stress and stream power. Where bed shear stress or stream power is increasing over distance, the capacity of the flow to transport bed material will also increase along that section of the channel. This spatial increase in bed-material transport capacity is likely to produce erosion if the stream exceeds the critical stream power for bed-material mobilization because the "sediment-hungry" flow will satisfy its increased capacity for transport by mobilizing material on the streambed. The critical steam power depends on the average size as well as the range of sizes of sediment on the streambed. Generally, the larger the average size of the bed material the larger the critical stream power required to mobilize the bed. However, the range of sizes is also important because fine particles on the streambed will move more readily (i.e. have a lower critical stream power) than coarse particles. Thus, the potential for erosion within a reach of an urban stream can be assessed by determining 1) whether the actual stream power of a flow exceeds the critical stream power required to mobilize particles of different sizes on the channel bed and 2) whether the excess stream power of the flow, or the stream power in excess of the critical stream power, is increasing over distance along the stream.

As was indicated in the literature review, an important consequence of stormwater management measures for control of peak discharge is the potential to increase the duration of erosive flows. Erosion is affected not only by the peak flow and associated stream power, but also by the time distribution of flows that exceed a critical value of stream power required to mobilize bed material. Thus, determining how changes in flow duration associated with volume control measures may affect the potential for sediment movement is an important aspect of assessing channel erosion potential. Recent studies (Ibrahim and Rouhi, 2021; Soar et al., 2017) have integrated excess stream power over time to determine the total amount of work that a sequence of flows performs in transporting bed material. This approach provides a basis for using the existing HEC-HMS and HEC-RAS models developed for the initial evaluation of watershed specific release rates (Flegel et al., 2019) to examine how different release rates may affect stream erosion potential following future development. The models provide time series of hydraulic characteristics of the flow throughout the stream network for a given design storm and release rate scenario. They also incorporate detention basins with linear outflow hydrograph formulation (Guo, 1999) at the spatial resolution of subbasins such that WMO volume control and release rate requirements are achieved for four release rate scenarios (0.15 cfs/ac, 0.20 cfs/ac, 0.25 cfs/ac, and 0.30 cfs/ac). The potential for changes in stream erosion potential can be evaluated by using the hydraulic data in these models to estimate excess stream power and then integrating this excess power over the time series of flow to determine the total excess work of the flows in transporting bed material. Locations where excess work increases along the streams can then be identified as sites of high erosion potential.

8.1.2 Objectives and Scope

The objective of the pilot watershed analysis is to develop an approach to evaluating stream erosion potential for different watershed specific release rates based on the concept of excess stream power and excess total work, where excess power and work refer to the capacity of the flow to transport bed material. The approach builds upon both existing concepts within the scientific literature on the relation of stream erosion to relevant hydraulic metrics and on the hydraulic modeling already performed within previous phases of this analysis to evaluate the influence of different watershed release rate scenarios on water surface elevations. Results are aimed at demonstrating the value of the approach for assessing stream erosion potential based on analysis of a few reaches in the upper Salt Creek watershed. Additional analysis for Addison Creek is included as supplemental materials to this report. All results are preliminary and may need further refinement to confirm their accuracy.

8.2 Methods

For the pilot analysis of the effect of watershed specific release rates on erosion potential, the HEC-HMS and HEC-RAS models developed for the initial evaluation of watershed specific release rates (Flegel et al., 2019) were applied to determine the stream power per unit area of the streambed (ω) associated with the flow for every cross section. Design storm hyetographs with return periods from 2 months to 100 years were developed based on Illinois State Water Survey *Rainfall Frequency Atlas of the Midwest-Bulletin 71* (Huff and Angel, 1992). These hyetographs were used with the HEC-HMS model for each watershed specific release rate to develop boundary conditions for the HEC-RAS model. The HEC-RAS models developed for the corresponding watershed specific release rate were used to determine the hydraulic properties of the flow at every cross section in the HEC-RAS model. The HEC-RAS Controller (Goodell, 2014) is a library of functions that allow external programs to access computational elements within HEC-RAS simulations. These computational elements include geometric components of the channel network and computational results from the simulation runs. For the pilot watershed analysis, Visual Basic for Applications (VBA) scripts were developed to read a Microsoft Excel spreadsheet that contains bed-sediment data for reaches where samples were collected. The script uses the HEC-RAS Controller functions to extract the time series of hydraulic parameters for each cross section in the reach and use these parameters to calculate the cumulative excess stream power.

Samples of the bed sediment were collected from seven location in Upper Salt Creek (Figure 94) and seven locations in Addison Creek (Figure 95) and analyzed to determine the distribution of particle sizes. Each sample was sieved to determine the percent of the total mass of sample that was finer than 20 sieve sizes ranging from 0.063 mm (the smallest size of sand) to 32 mm. These sediment size distributions provided input to the Microsoft Excel scripts that calculated the cumulative excess stream power.

Stream power per unit area (ω, W/m^2), which is the bed shear stress multiplied by the velocity, is defined as:

$$\omega = \rho g U D S = \tau U$$

where ρ is the density of the fluid (assumed to be 1000 kg/m^3), g is the acceleration of gravity (assumed to be 9.81 m/s^2), U is the cross-section mean velocity of the flow (m/s), D is the mean depth of the flow (m), S is the slope of the energy grade line (m/m), and τ is the bed shear stress of the flow (N/m^2). The stream power per unit area defines the time rate of energy expenditure of the flow as it moves from higher elevations to lower elevations. Therefore, integrating the stream power over the duration of a hydrograph gives the total energy expenditure, or work, associated with that storm event.

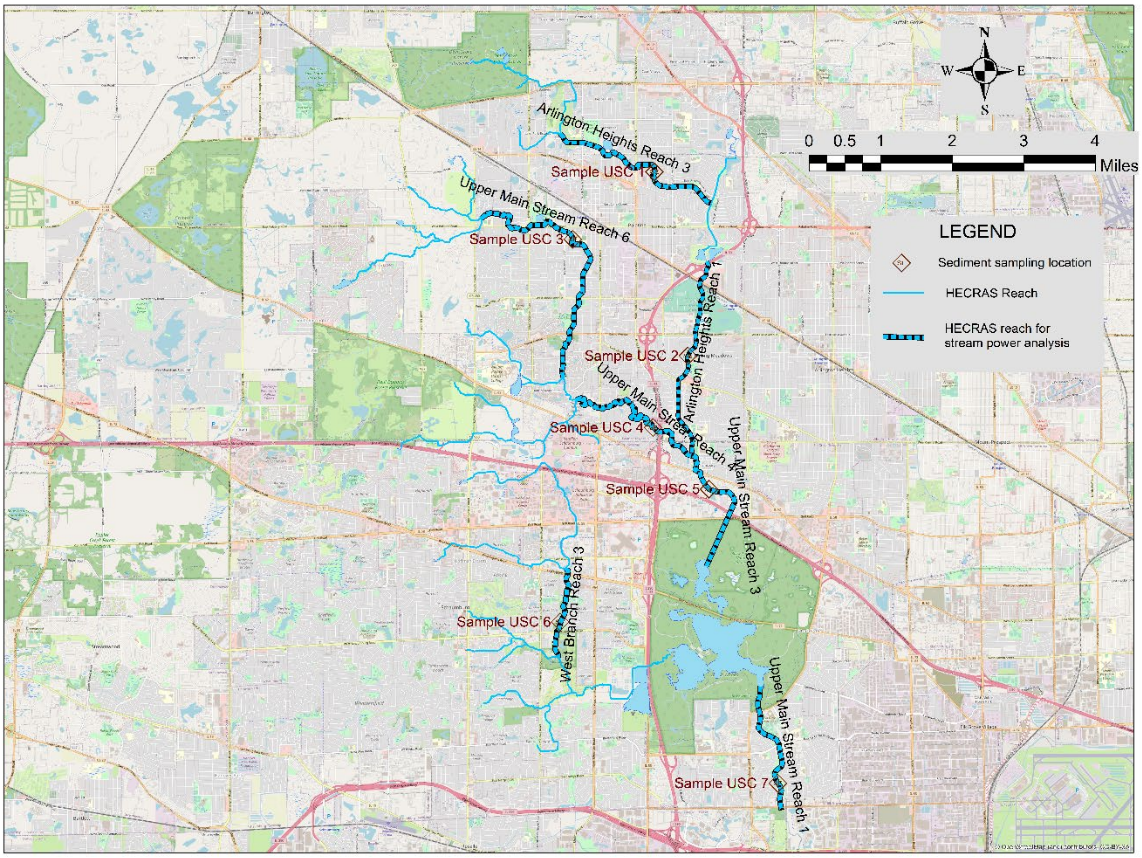


Figure 94. Map showing the location of sampling sites and associated HEC-RAS model reaches for the Upper Salt Creek watershed.

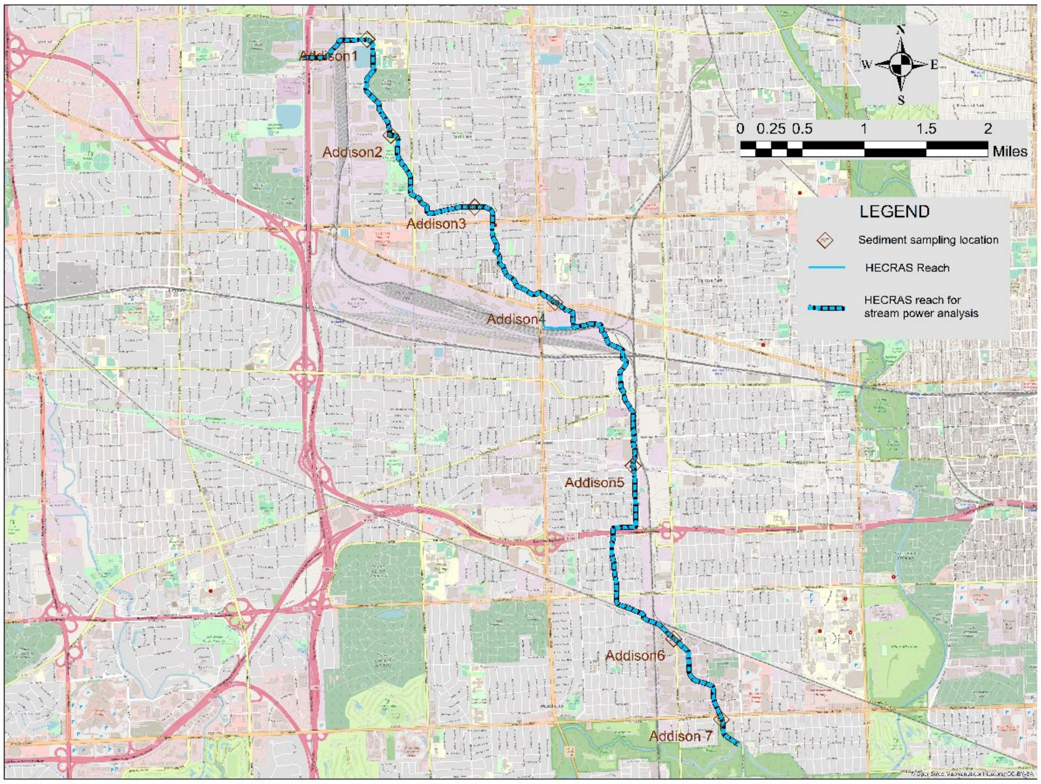


Figure 95. Map showing the location of sampling sites and associated HEC-RAS model reaches for the Addison Creek watershed.

The critical stream power for a given size fraction of the sediment was calculated as (Lammers and Bledsoe, 2018):

$$\omega_c = \omega_{(c,d_{50})}\left(\frac{d_i}{d_{50}}\right)^{-\beta}\rho\left(gsd_i\right)^{1.5}$$

where d_{50} is the median sediment size (m), $\omega_{(c,d_{50})}$ is the critical stream power for the median sediment size (W/m^2), s is a factor to adjust the density (ρ, kg/m^3) to equal the submerged specific density of the sediment (1.65) , and d_i is the size fraction of the sediment (m). Values of β and $\omega_{(c,d_{50})}$ were set at 0.8 and 0.24, respectively, based on based on values reported by Lammers and Bledsoe (2018) for bed material with size characteristics similar to those of streams in the Chicago region.

Soar et al. (2017) recommended integrating the excess stream power (defined as stream power in excess of the critical stream power) for different discharges corresponding to a flow duration curve and multiplying by time (number of seconds in a year) to estimate average annual total energy or work in excess of the critical value. This research adapts this method by integrating the excess stream power over the entire hydrograph of a design storm. The HEC-RAS simulations of the design storms provide data at a time interval of 1 hour that describe the hydraulic properties of the flow at every cross section. For each cross section the critical stream power is determined for each of the 20 sediments sizes based on the equation above. The model then calculates the excess stream power for that sediment size. This excess stream power is multiplied by the fraction of the total sediment material in that size fraction. The process is repeated for all size fractions to determine a total excess stream power for that time step that is weighted by the sediment size distribution. This process is repeated for all time steps in the hydrograph simulation to determine a cumulative excess stream power for that cross section. The equation used for these calculations is:

$$E_{cum} = \sum_{j=1}^{m} \left[\sum_{i=1}^{n} p_i \left(\omega_j - \omega_{ci} \right) \right] \Delta t$$

where E_{cum} is the cumulative excess stream power per unit area, or excess energy per unit area (J/m^2), for the storm at the given cross section; subscripts i and j are indices for the sediment size classes and the time steps in the hydrograph simulation, respectively; n and m are the total number of sediment size classes and time steps, respectively; p_i is the fraction of the total sediment at the given cross section in the i^{th} sediment size class; ω_j is the stream power per unit area of the flow for the j^{th} time step; and ω_{ci} is the critical stream power for the i^{th} sediment size fraction, and Δt is the time step duration.

8.3 Results

Streambed sediment samples were collected at seven sites in the Upper Salt Creek Watershed and seven sites in the Addison Creek watersheds. These sites showed wide variability in the streambed material, with some sites having no particles larger than 16mm and other sites with more than 25% of the sample larger than 32 mm (Table 62 through Table 68). Similar results for Addison Creek bed sediment samples are included as supplemental materials to this report.

Table 62. Sediment size classification for Upper Salt Creek sampling location 1.

HEC-RAS River: Arlington Height *d_{50}: 7.6 mm* *ϕ scale*	*HEC-RAS Reach: 3* *Size (mm)*	*Approx. Station: 26500* *Percent of total weight*
-5.00	31.5	9.72
-4.50	22.4	9.30
-4.00	16.0	9.91
-3.50	11.2	8.76
-3.00	8.0	7.53
-2.50	5.6	6.95
-2.00	4.0	5.49
-1.50	2.8	4.82
-1.00	2.0	4.49
-0.5	1.4	4.21
0	1.0	4.28
0.5	0.710	4.63
1	0.500	5.84
1.5	0.355	6.29
2	0.250	4.41
2.5	0.180	1.79
3	0.125	0.86
3.5	0.090	0.46
4	0.063	0.28

Table 63. Sediment size classification for Upper Salt Creek sampling location 2.

HEC-RAS River: Arlington Height *d_{50}: 1.6 mm* *ϕ scale*	*HEC-RAS Reach: 1* *Size (mm)*	*Approx. Station: 8000* *Percent of total weight*
-5.00	31.5	0.00
-4.50	22.4	0.00
-4.00	16.0	1.79
-3.50	11.2	5.79
-3.00	8.0	7.47
-2.50	5.6	7.98
-2.00	4.0	7.67
-1.50	2.8	7.21
-1.00	2.0	6.65
-0.5	1.4	5.14
0	1.0	3.78
0.5	0.710	3.34
1	0.500	4.59
1.5	0.355	9.10
2	0.250	15.01
2.5	0.180	9.60
3	0.125	3.38
3.5	0.090	1.07
4	0.063	0.43

Table 64. Sediment size classification for Upper Salt Creek sampling location 3.

HEC-RAS River: Upper Main Stream *d_{50}: 9.5 mm*	*HEC-RAS Reach: 6*	*Approx. Station: 61500*
ϕ scale	*Size (mm)*	*Percent of total weight*
-5.00	31.5	0.00
-4.50	22.4	10.59
-4.00	16.0	17.30
-3.50	11.2	13.54
-3.00	8.0	8.62
-2.50	5.6	6.55
-2.00	4.0	5.52
-1.50	2.8	4.90
-1.00	2.0	4.60
-0.5	1.4	3.89
0	1.0	3.61
0.5	0.710	3.63
1	0.500	4.51
1.5	0.355	4.94
2	0.250	3.95
2.5	0.180	1.81
3	0.125	0.94
3.5	0.090	0.78
4	0.063	0.33

Table 65. Sediment size classification for Upper Salt Creek sampling location 4.

HEC-RAS River: Upper Main Stream *d_{50}: 14.4 mm*	*HEC-RAS Reach: 4*	*Approx. Station: 37300*
ϕ scale	*Size (mm)*	*Percent of total weight*
-5.00	31.5	24.52
-4.50	22.4	11.17
-4.00	16.0	8.40
-3.50	11.2	7.10
-3.00	8.0	5.35
-2.50	5.6	6.14
-2.00	4.0	5.51
-1.50	2.8	4.45
-1.00	2.0	4.15
-0.5	1.4	3.51
0	1.0	2.99
0.5	0.710	2.64
1	0.500	2.99
1.5	0.355	3.88
2	0.250	3.75
2.5	0.180	1.96
3	0.125	0.86
3.5	0.090	1.07
4	0.063	0.43

Table 66. Sediment size classification for Upper Salt Creek sampling location 5.

HEC-RAS River: Upper Main Stream *d_{50}: 1.9 mm* *ϕ scale*	*HEC-RAS Reach: 3* *Size (mm)*	*Approx. Station: 29400* *Percent of total weight*
-5.00	31.5	0.00
-4.50	22.4	0.00
-4.00	16.0	0.80
-3.50	11.2	0.87
-3.00	8.0	3.21
-2.50	5.6	8.03
-2.00	4.0	11.03
-1.50	2.8	10.93
-1.00	2.0	9.46
-0.5	1.4	8.24
0	1.0	5.24
0.5	0.710	3.87
1	0.500	4.81
1.5	0.355	9.93
2	0.250	14.19
2.5	0.180	6.38
3	0.125	2.11
3.5	0.090	1.07
4	0.063	0.43

Table 67. Sediment size classification for Upper Salt Creek sampling location 6.

HEC-RAS River: West Branch *d_{50}: 10.5 mm* *ϕ scale*	*HEC-RAS Reach: 3* *Size (mm)*	*Approx. Station: 22200* *Percent of total weight*
-5.00	31.5	7.93
-4.50	22.4	7.33
-4.00	16.0	14.38
-3.50	11.2	13.43
-3.00	8.0	9.17
-2.50	5.6	7.14
-2.00	4.0	6.24
-1.50	2.8	5.44
-1.00	2.0	4.64
-0.5	1.4	4.37
0	1.0	3.94
0.5	0.710	4.26
1	0.500	4.97
1.5	0.355	3.94
2	0.250	1.77
2.5	0.180	0.53
3	0.125	0.25
3.5	0.090	1.07
4	0.063	0.43

Table 68. Sediment size classification for Upper Salt Creek sampling location 7.

HEC-RAS River: Upper Main Stream *d_{50}: 4.6 mm* *ϕ scale*	*HEC-RAS Reach: 1* *Size (mm)*	*Approx. Station: 1300* *Percent of total weight*
-5.00	31.5	0.00
-4.50	22.4	0.00
-4.00	16.0	3.31
-3.50	11.2	8.88
-3.00	8.0	12.26
-2.50	5.6	13.16
-2.00	4.0	11.57
-1.50	2.8	9.39
-1.00	2.0	7.52
-0.5	1.4	6.32
0	1.0	4.60
0.5	0.710	4.11
1	0.500	5.14
1.5	0.355	6.96
2	0.250	4.94
2.5	0.180	1.31
3	0.125	0.36
3.5	0.090	1.07
4	0.063	0.43

Cumulative excess stream power for the 2-year design storm, the pre-development (base scenario) condition, and all four candidate watershed specific release rates are shown for the seven Upper Salt Creek reaches where bed sediment samples were collected in Figure 96 and those similar that follow. In each of these figures, the graph on the left indicates the cumulative excess stream power per unit area. Because the excess stream power varies with the flow rate, the geometry of the channel, and the size distribution of the sediment, results shown on these figures exhibit wide variation in cumulative excess stream power. To better examine the impact of watershed specific release rate, the graph on the right of each of these figures shows the differences between each release rate scenario and the base scenario. The base scenario refers to the pre-development condition (no split subbasins, no change in curve number) without volume control or release rate applied. Figure 97, and those similar that follow, show profiles of the stream bed, water surface, and energy grade line for each of the seven reaches. These figures also show the locations of hydraulic structures such as bridges and culverts in these reaches. These figures clearly demonstrate how spatial differences in local channel and flow properties result in variability in cumulative excess stream power. For example, the relatively large cumulative excess stream power near station 15600 of Arlington Heights Reach 1 is related to high velocity flows being released from the Twin Lakes Reservoir outlet structure immediately upstream from this location. Similarly, the large cumulative excess stream power near station 63800 in Upper Main Stream reach 6 is related to a relatively steep slope coupled with large velocities just downstream from the Bridgeview St. culvert.

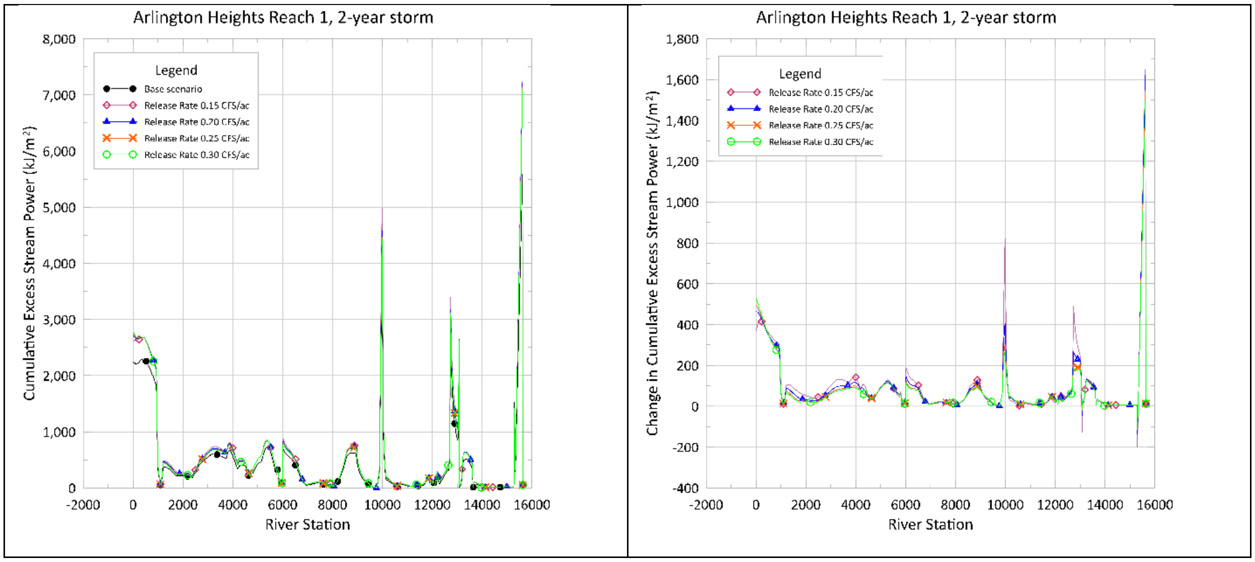


Figure 96. Graphs showing excess stream power 10-day cumulative stream power and difference between cumulative stream power for base scenario and four watershed specific release rates for 2-year design storm for Arlington Heights Reach 1 of Upper Salt Creek Watershed.

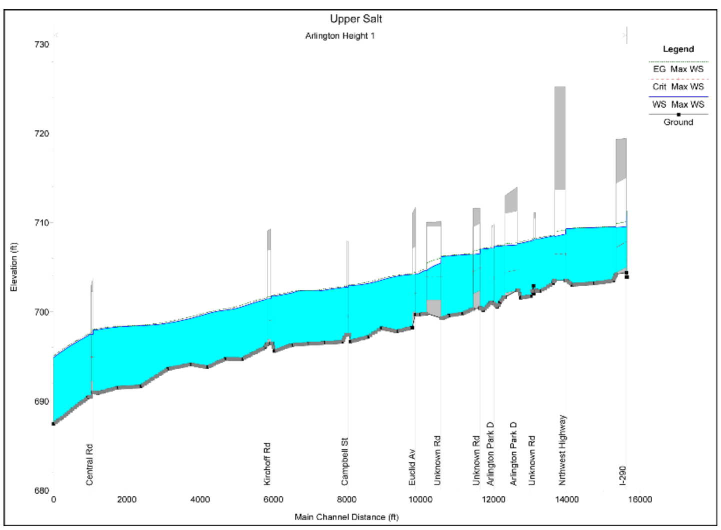


Figure 97. Graph showing profiles of water surface, energy grade line, and bed and locations of hydraulic structures in Arlington Heights Reach 1 of Upper Salt Creek Watershed.

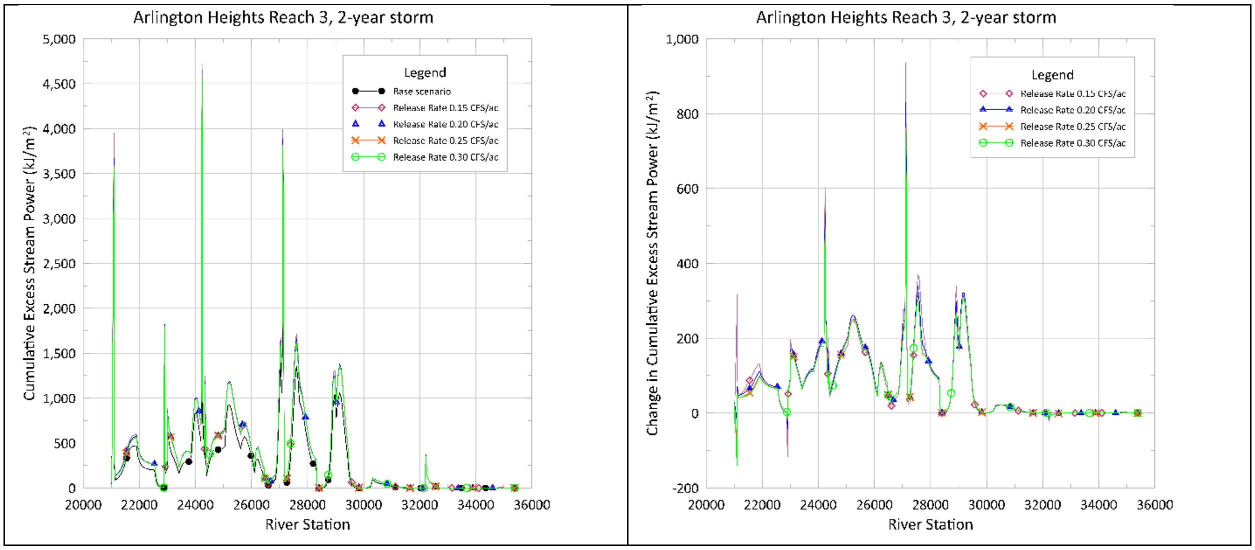


Figure 98. Graphs showing excess stream power 10-day cumulative stream power and difference between cumulative stream power for base scenario and four watershed specific release rates for 2-year design storm for Arlington Heights Reach 3 of Upper Salt Creek Watershed.

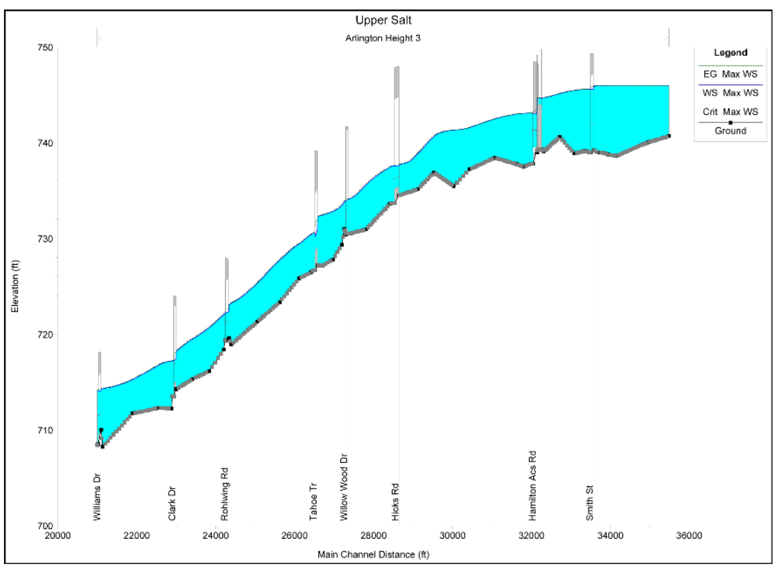


Figure 99. Graph showing profiles of water surface, energy grade line, and bed and locations of hydraulic structures in Arlington Heights Reach 3 of Upper Salt Creek Watershed.

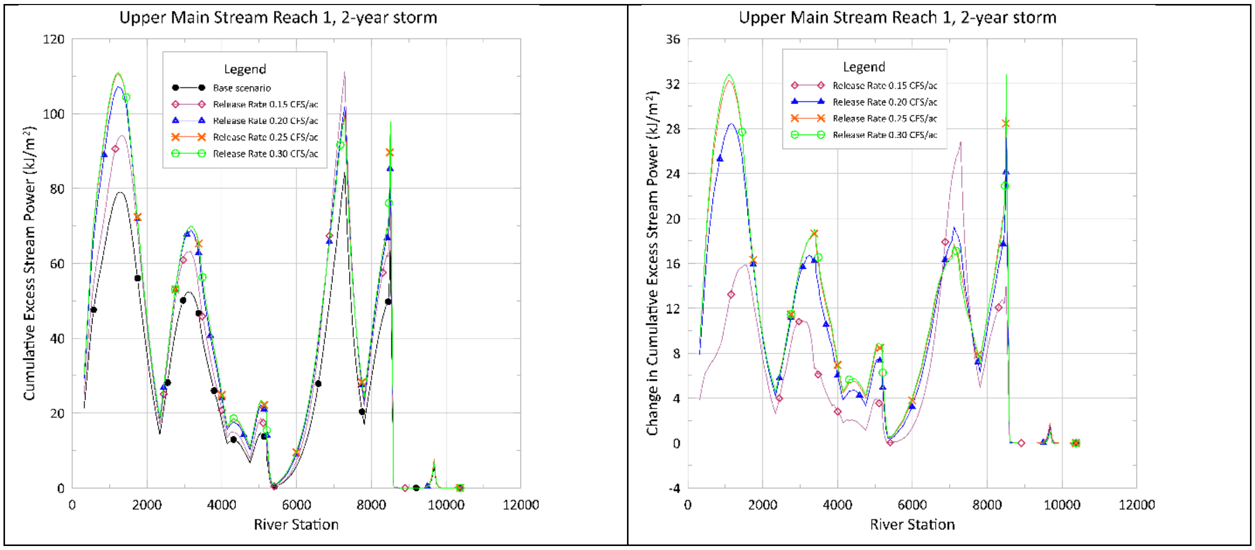


Figure 100. Graphs showing excess stream power 10-day cumulative stream power and difference between cumulative stream power for base scenario and four watershed specific release rates for 2-year design storm for Upper Main Stream Reach 1 of Upper Salt Creek Watershed.

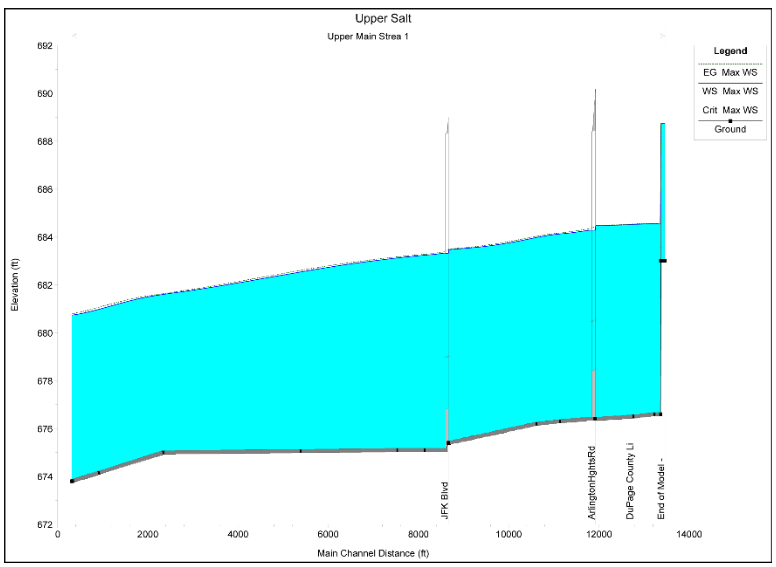


Figure 101. Graph showing profiles of water surface, energy grade line, and bed and locations of hydraulic structures in Upper Main Stream Reach 1 of Upper Salt Creek Watershed.

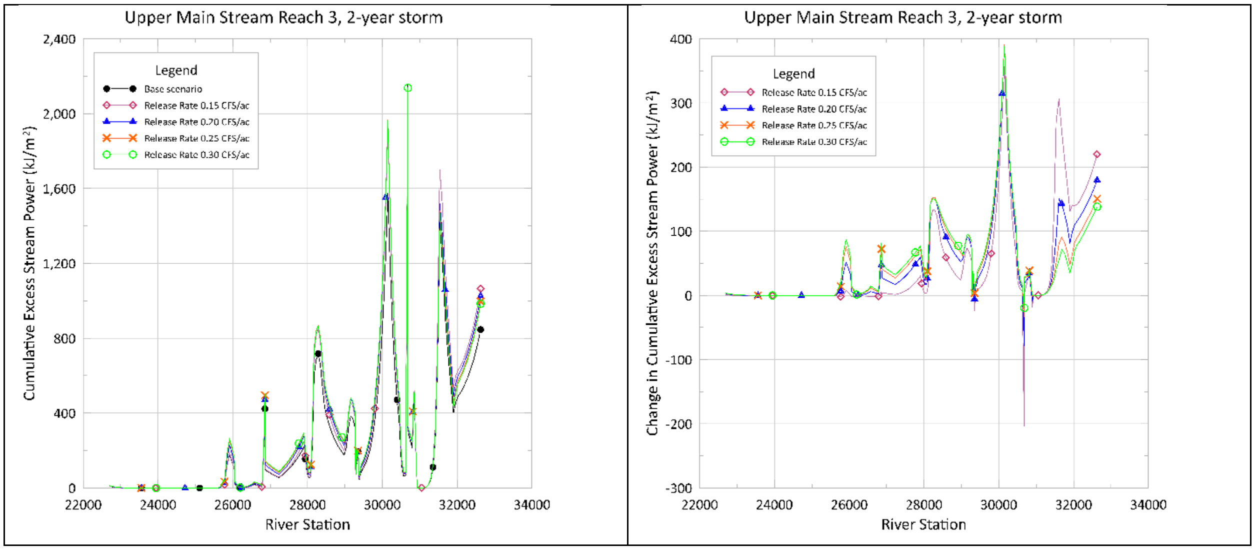


Figure 102. Graphs showing excess stream power 10-day cumulative stream power and difference between cumulative stream power for base scenario and four watershed specific release rates for 2-year design storm for Upper Main Stream Reach 3 of Upper Salt Creek Watershed.

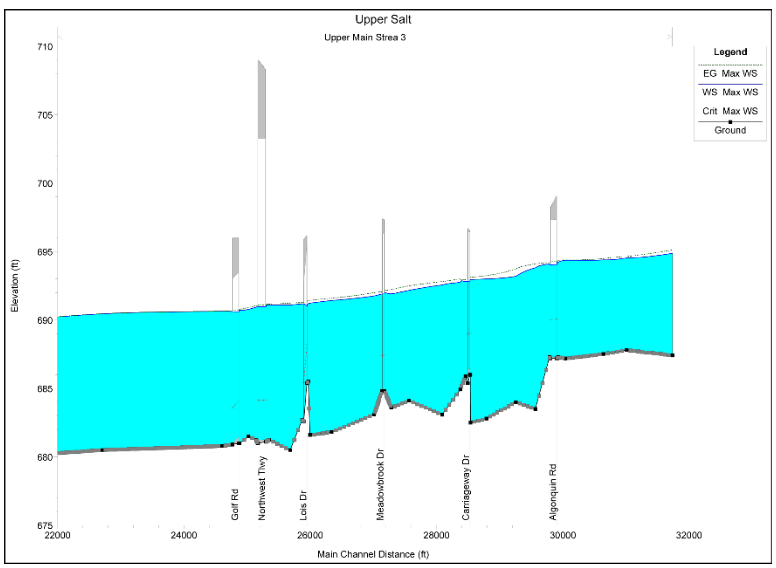


Figure 103. Graph showing profiles of water surface, energy grade line, and bed and locations of hydraulic structures in Upper Main Stream Reach 3 of Upper Salt Creek Watershed.

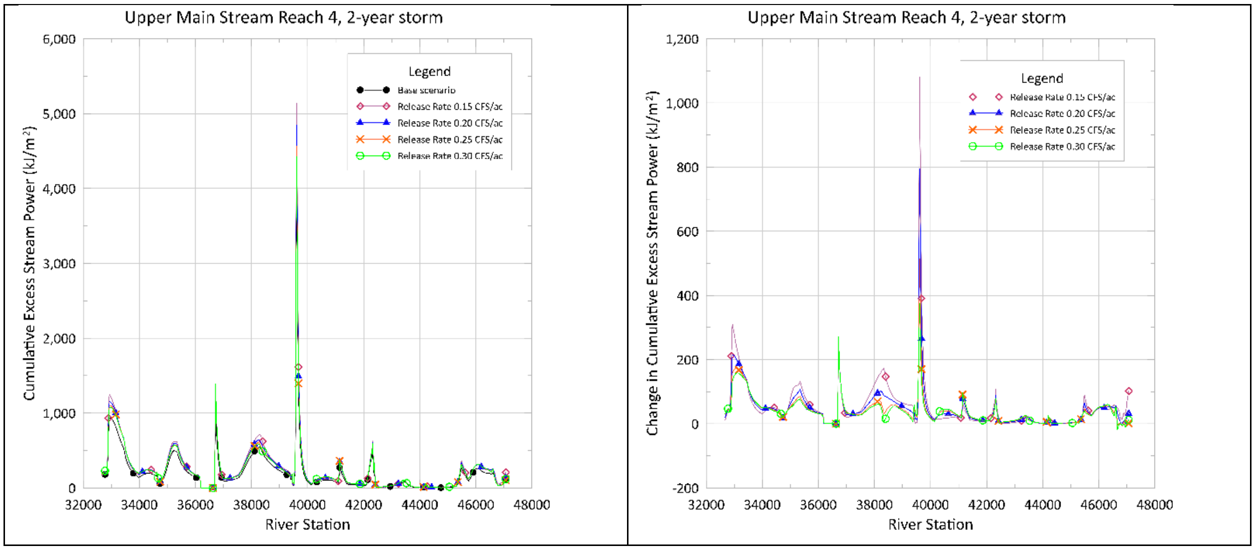


Figure 104. Graphs showing excess stream power 10-day cumulative stream power and difference between cumulative stream power for base scenario and four watershed specific release rates for 2-year design storm for Upper Main Stream Reach 4 of Upper Salt Creek Watershed.

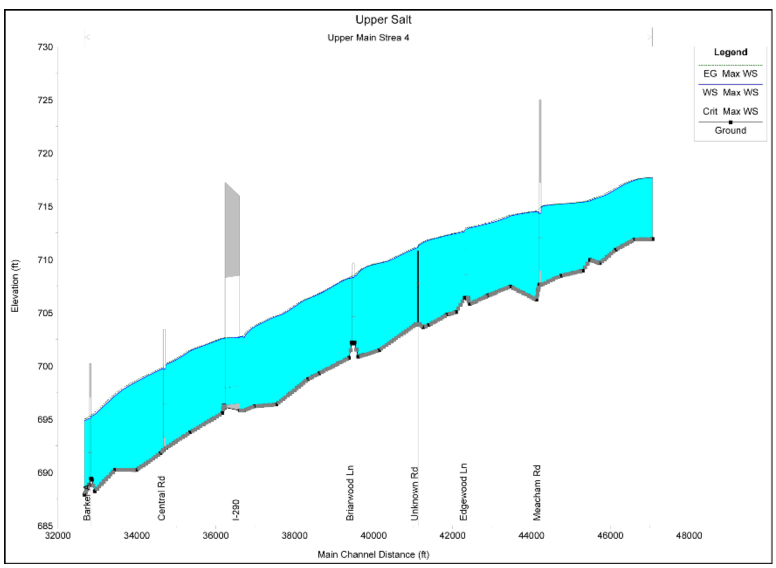


Figure 105. Graph showing profiles of water surface, energy grade line, and bed and locations of hydraulic structures in Upper Main Stream Reach 4 of Upper Salt Creek Watershed.

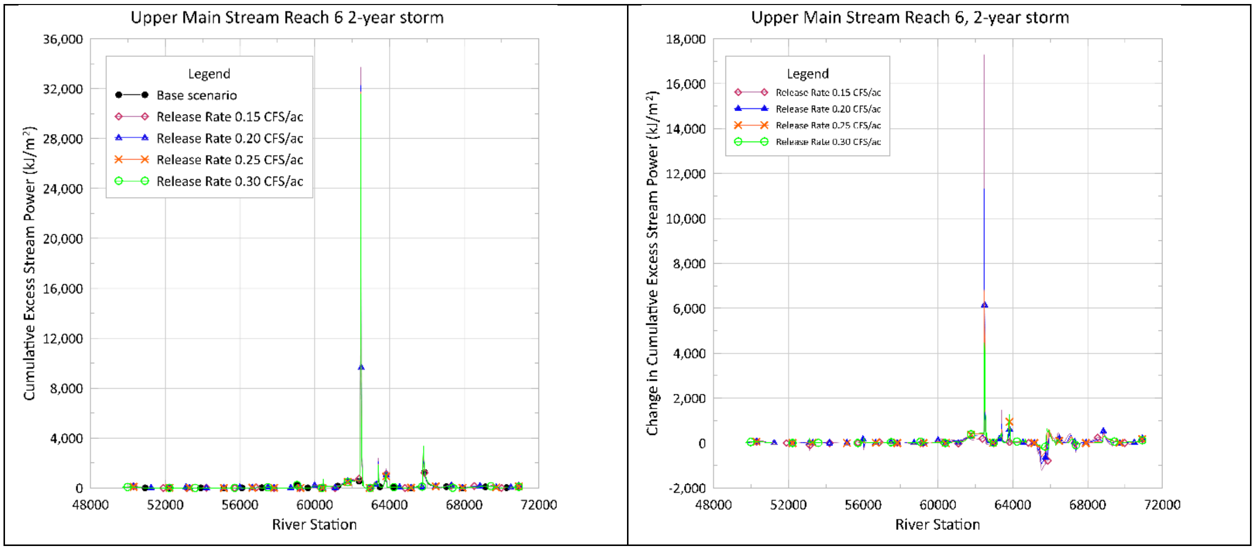


Figure 106. Graphs showing excess stream power 10-day cumulative stream power and difference between cumulative stream power for base scenario and four watershed specific release rates for 2-year design storm for Upper Main Stream Reach 6 of Upper Salt Creek Watershed.

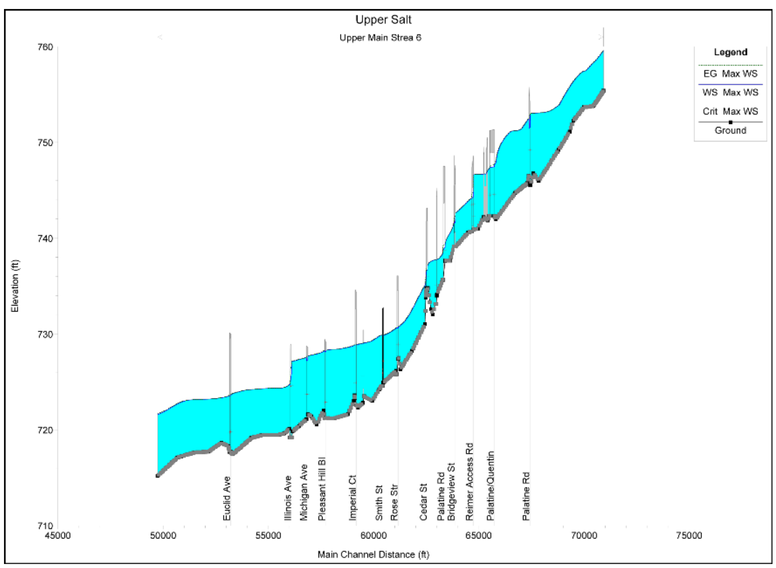


Figure 107. Graph showing profiles of water surface, energy grade line, and bed and locations of hydraulic structures in Upper Main Stream Reach 6 of Upper Salt Creek Watershed.

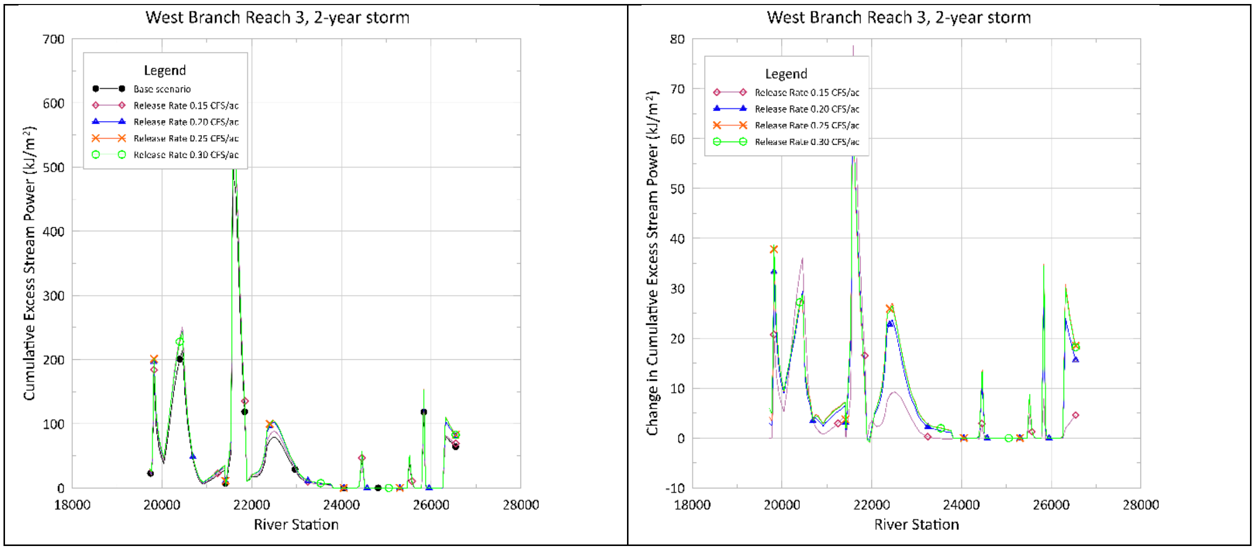


Figure 108. Graphs showing excess stream power 10-day cumulative stream power and difference between cumulative stream power for base scenario and four watershed specific release rates for 2-year design storm for West Branch Reach 3 of Upper Salt Creek Watershed.

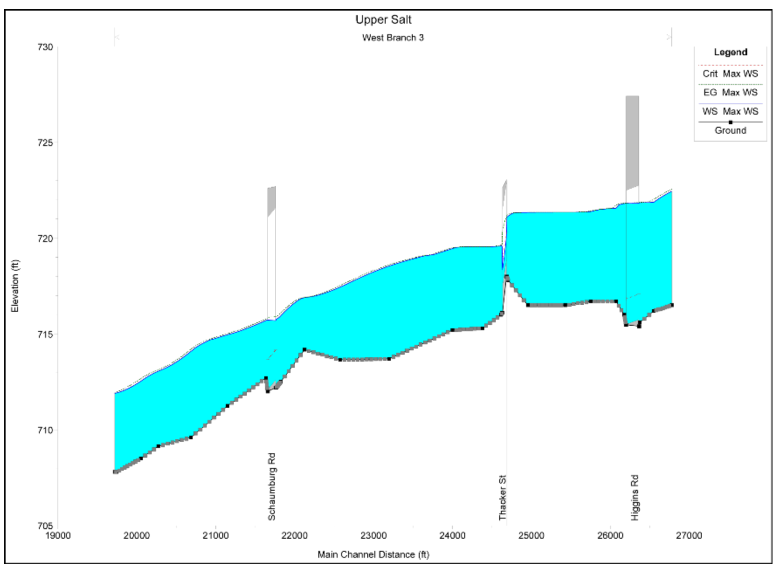


Figure 109. Graph showing profiles of water surface, energy grade line, and bed and locations of hydraulic structures in West Branch Reach 3 of Upper Salt Creek Watershed.

Graphs showing differences between the four watershed specific release rate scenarios and the base scenario indicate that the change in cumulative excess stream power among release rate scenarios is small compared to the cumulative excess stream power. In other words, for most stream cross sections, the effect of changing the release rate scenario is small compared to the total magnitude of cumulative excess stream power, and hence the erosion potential. In general, the difference between the cumulative excess stream power for the base scenario and the watershed specific release rate scenarios is largest for the 0.15 cfs/acre scenario. For most reaches, more restrictive watershed specific release rates have larger cumulative excess stream power compared to less restrictive release rates. This increase in cumulative excess power with more restrictive release rates indicates that stormwater management practices that greatly restrict release rates by storing large volumes of water and then releasing it gradually are less effective at mitigating stream erosion potential than less restrictive practices. However, for some locations, such as the downstream end of Upper Main Stream Reach 1, the more restrictive release rate scenarios show a decrease in cumulative excess stream power. These locations are often located downstream from in-line storage structures that affect the hydraulics of the flow. Locations where the cumulative excess stream power decreases for future development and watershed specific release rate scenarios may indicate locations where sedimentation could increase under the future scenarios.

Figure 110 to Figure 116 show the cumulative excess stream power and difference in cumulative excess stream power from the base scenario for the 50-year design storm. These figures show similar behavior to the 2-year design storm, except the magnitude of the cumulative excess stream power is larger than that for the 2-year design storm. Profile plots are not shown for the 50-year storm because the channel geometry and location of hydraulic structures is unchanged between different design storms.

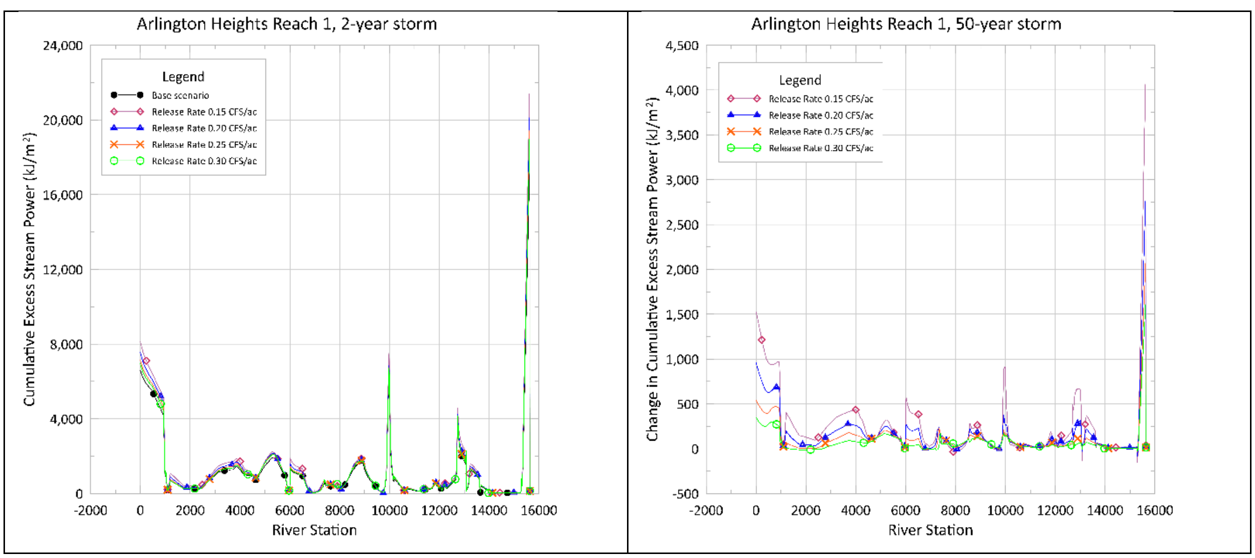


Figure 110. Graphs showing excess stream power 10-day cumulative stream power and difference between cumulative stream power for base scenario and four watershed specific release rates for 50-year design storm for Arlington Heights Reach 1 of Upper Salt Creek Watershed.

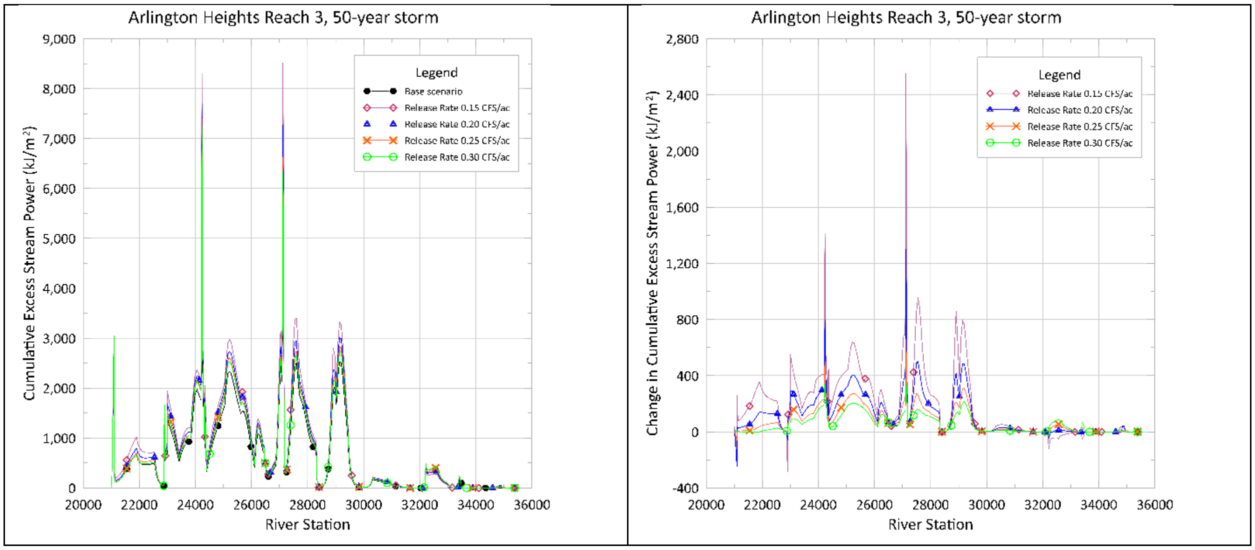


Figure 111. Graphs showing excess stream power 10-day cumulative stream power and difference between cumulative stream power for base scenario and four watershed specific release rates for 50-year design storm for Arlington Heights Reach 3 of Upper Salt Creek Watershed.

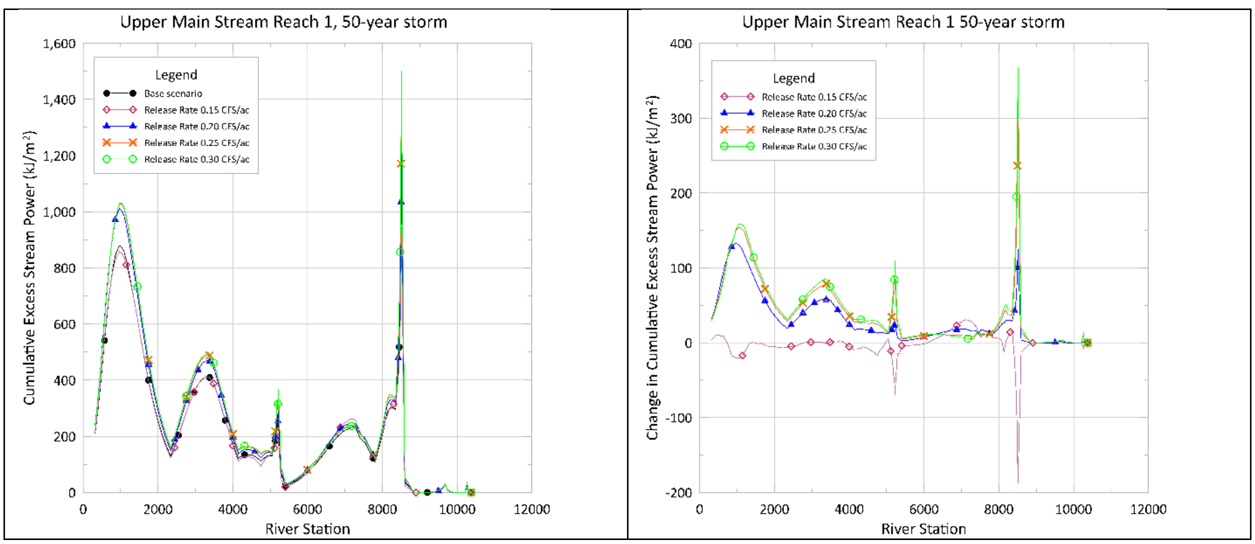


Figure 112. Graphs showing excess stream power 10-day cumulative stream power and difference between cumulative stream power for base scenario and four watershed specific release rates for 2-year design storm for Upper Main Stream Reach 1 of Upper Salt Creek Watershed.

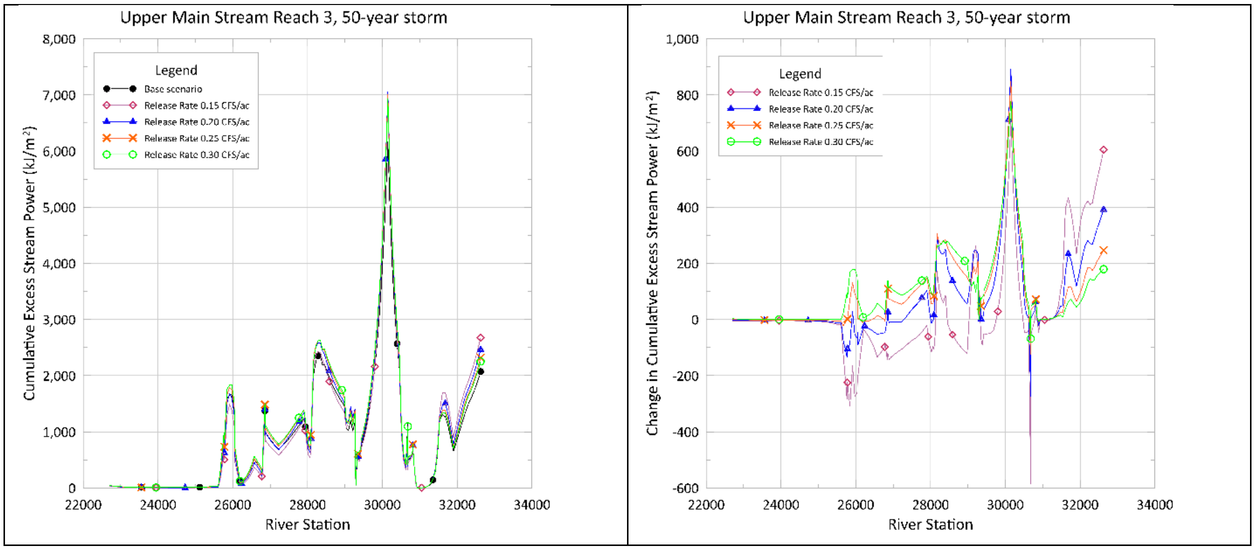


Figure 113. Graphs showing excess stream power 10-day cumulative stream power and difference between cumulative stream power for base scenario and four watershed specific release rates for 2-year design storm for Upper Main Stream Reach 3 of Upper Salt Creek Watershed.

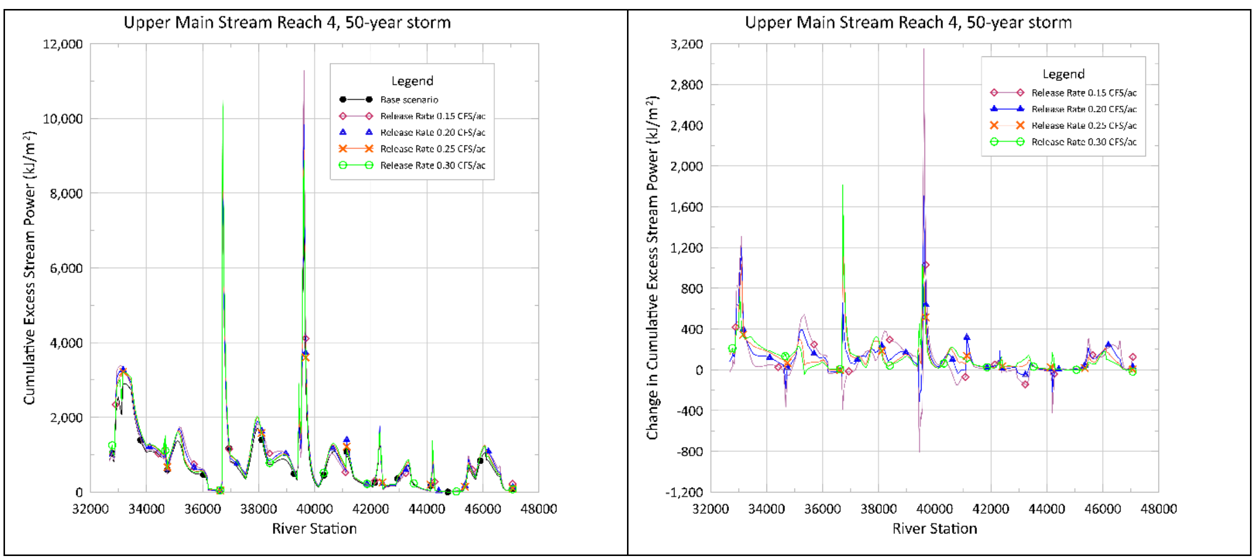


Figure 114. Graphs showing excess stream power 10-day cumulative stream power and difference between cumulative stream power for base scenario and four watershed specific release rates for 2-year design storm for Upper Main Stream Reach 4 of Upper Salt Creek Watershed.

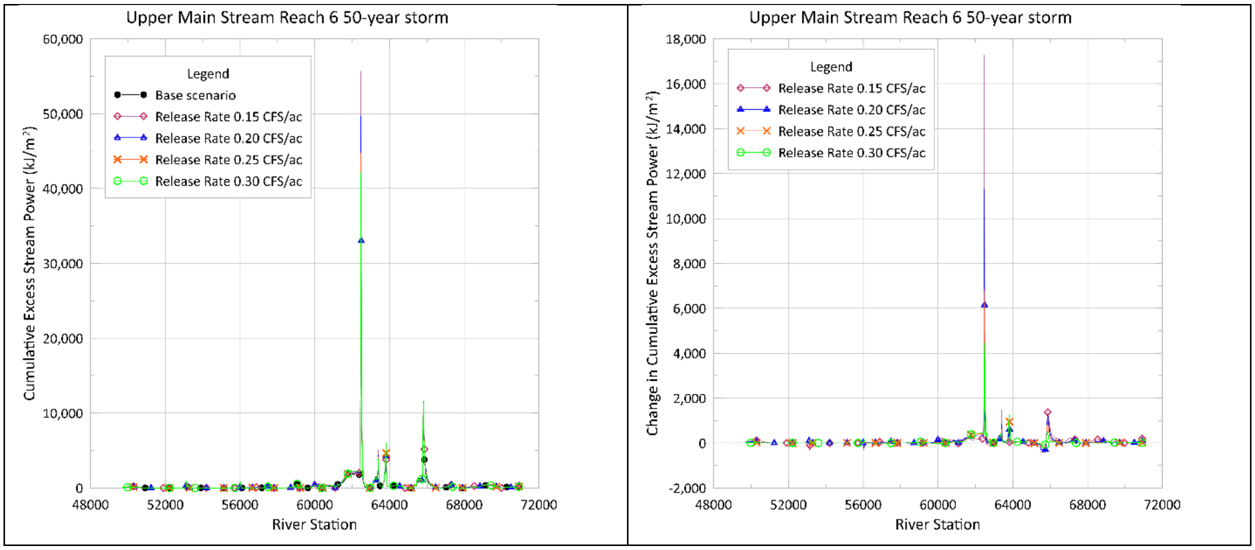


Figure 115. Graphs showing excess stream power 10-day cumulative stream power and difference between cumulative stream power for base scenario and four watershed specific release rates for 2-year design storm for Upper Main Stream Reach 6 of Upper Salt Creek Watershed.

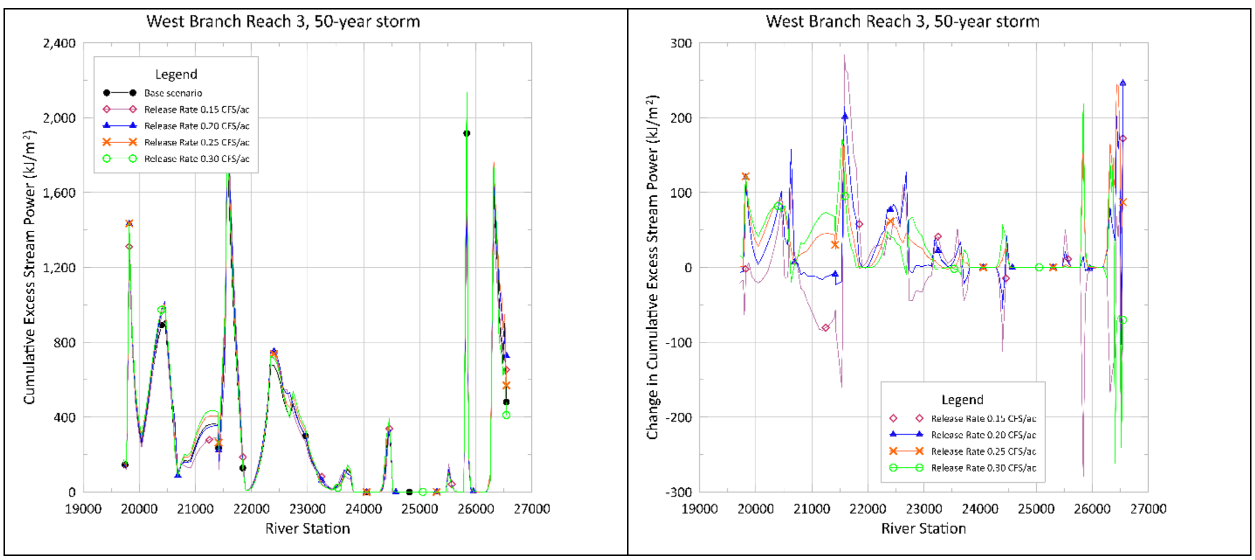


Figure 116. Graphs showing excess stream power 10-day cumulative stream power and difference between cumulative stream power for base scenario and four watershed specific release rates for 2-year design storm for West Branch Reach 3 of Upper Salt Creek Watershed.

To evaluate the effect of storm return period on cumulative excess stream power, values of cumulative excess stream power were generated for multiple design storms with return periods ranging from 3-months to 50 years. Values of cumulative excess stream power from these design storms for each watershed specific release rate scenario provide an estimate of how effective different watershed specific release rate scenarios are at mitigating the impact of future development on stream erosion potential for a range of storm event magnitudes. Figure 117 shows the cumulative excess stream power from these design storms for each watershed specific release rate scenario fit to a log-normal exceedance probability distribution for a 300-foot subreach of Arlington Heights Reach 1 where bed sediment sample was collected on Upper Salt Creek. As was observed in Figure 96 to Figure 109, the change in erosion potential with increasing storm magnitude is much larger than the differences among watershed specific release rate scenarios. Furthermore, while the difference in erosion potential among release rate scenarios is small for the more frequent (smaller return period) events, the more restrictive release rate scenarios show a larger increase in erosion potential for the largest magnitude events. This result indicates that stormwater management for future development based on watershed-specific release rates is least effective at mitigating impacts on stream erosion potential for the most extreme storm events.

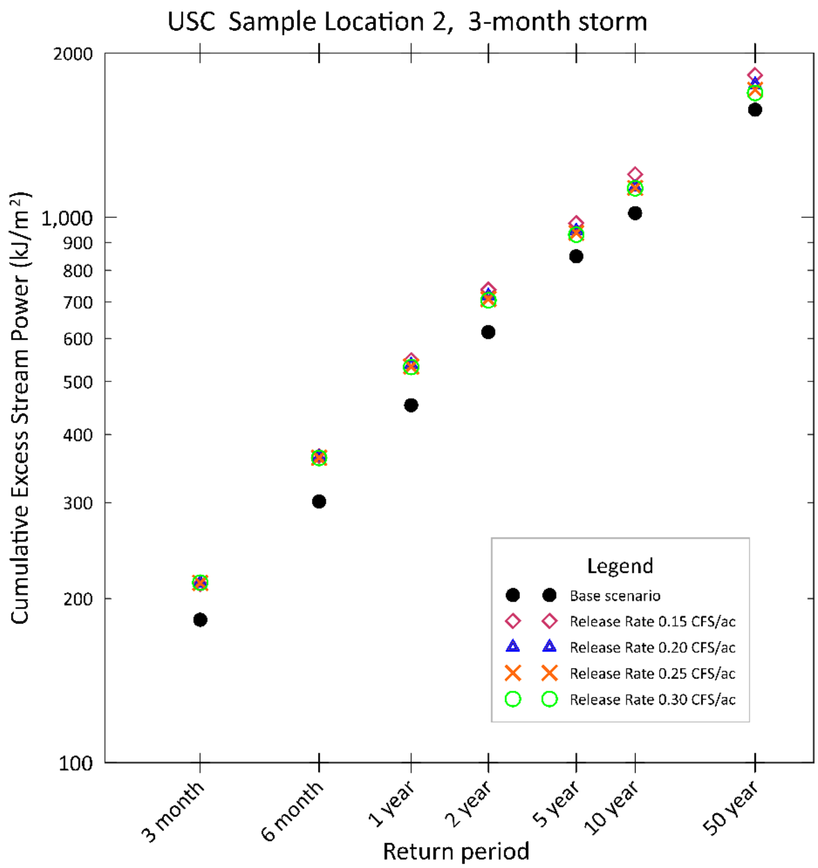


Figure 117. Graph showing exceedance probability of excess stream power 10-day cumulative stream power and difference between cumulative stream power for base scenario and four watershed specific release rates for 3-month to 50-year design storms for location of bed sediment sample Upper Salt Creek 2 in Arlington Heights Reach 1 of Upper Salt Creek Watershed.

An analysis of the cumulative excess stream power for reaches in Addison Creek near bed-sediment sampling locations is provided as supplemental materials to this report.

8.4 Conclusions

The results of this pilot analysis to establish a method for assessing erosion potential of streams in the greater Chicago metropolitan area under different watershed-release rate scenarios lead to the following conclusions:

1. Highly restrictive release rates may be less effective at mitigating the impacts of future development on erosion potential than less restrictive release rates. As release rates become more restrictive, durations of stream power in excess of critical threshold for bed-material transport may locally increase, thereby increasing the total transport capacity of flows of a given recurrence frequency.
2. Because the spatial pattern of excess stream power remains unchanged for different release rates, stormwater management based on watershed specific release rates does not fundamentally change the spatial pattern of stream erosion potential. For both the base scenario and the various watershed-specific release rates, erosion potential is greatest where excess stream power is increasing over distance.
3. Excess stream power does not increase for all locations in the stream for future development scenarios and for all watershed specific release rate scenarios. More work would be needed to determine the factors that cause increases or reductions in excess stream power at specific locations and instances.
4. Some locations show a decrease in cumulative excess stream power for future development and watershed specific release rate scenarios. These locations may be associated with infrastructure that promotes water storage during flows of all magnitudes, which in turn could increase sedimentation under the future scenarios.

Extending the pilot analysis to an assessment of stream erosion potential for all watersheds examined in this and previous phases of analyses would require several considerations. First, bed material samples would need to be collected and analyzed for numerous locations throughout each watershed given that information on the size distribution of bed material is necessary for calculating values of critical stream power. Second, the HEC-RAS simulations used in the pilot analysis were based on information describing flow and channel characteristics used to determine the influence of watershed release rate scenarios on water surface elevations in previous phases of this analysis. No refinement of the HEC-RAS information was undertaken in the pilot analysis. In this sense, the results are preliminary and detailed interrogation of cumulative excess stream power estimates would be necessary to ensure that seemingly “anomalous” results (exceptionally high or low values of cumulative excess stream power) are not artifacts of the modeling and represent real effects of local structures or channel features on hydraulic conditions. Moreover, the analysis is based on Water Survey Bulletin 71 (Huff and Angel, 1992). Recently, Illinois State Water Survey Bulletin 75 has been published (Angel et al., 2020). This report shows larger values of precipitation (and thus discharge) for design storms across all return periods. The analysis herein could be applied to the new design storm information. Finally, field investigations should be conducted to link the spatial distribution of stream power values to actual channel conditions. Such investigations would provide the basis for determining how well model predictions of high erosion potential conform to evidence of actual channel erosion.

8.5 References

Angel, J.R., Markus, M., Wang, K.A., Kerschner, B.M., & Singh, S. (2020). Precipitation Frequency Study for Illinois. Illinois State Water Survey Bulletin 75, Champaign, IL.

Ibrahim, Y.A., Rouhi, A., 2021. Role of hydrodynamic forces in shaping the dynamics of channel evolution in urban watershed. Journal of Hydrologic Engineering, 26(8), 05021019. doi:10.1061/(ASCE)HE.1943-5584.0002111.

Lammers, R.W., Bledsoe, B.P., 2018. A network scale, intermediate complexity model for simulating channel evolution over years to decades. Journal of Hydrology, 566, 886-900. https://doi.org/10.1016/j.jhydrol.2018.09.036.

Soar, P.J., Wallerstein, N.P., Thorne, C.R., 2017. Quantifying river channel stability at the basin scale. Water, 9(2), 133. https://doi.org/10.3390/w9020133.

Flegel, A., Byard, G., McConkey, S., Hanstad, C., Gaynor, N., & Zaloudek, Z. (2019). Watershed-Specific Release Rate Analysis: Cook County, Illinois. Illinois State Water Survey.

Huff, F. A., & Angel, J. R. (1992). Rainfall frequency atlas of the Midwest. Bulletin (Illinois State Water Survey) no. 71.

Goodell, Christopher. (2014). Breaking the HEC-RAS Code - A User's Guide to Automating HEC-RAS.

Guo, J. C. (1999). Detention storage volume for small urban catchments. Journal of water resources planning and management, 125(6), 380-382.

Appendix A. Map exhibits of detention storage requirements in DIA communities

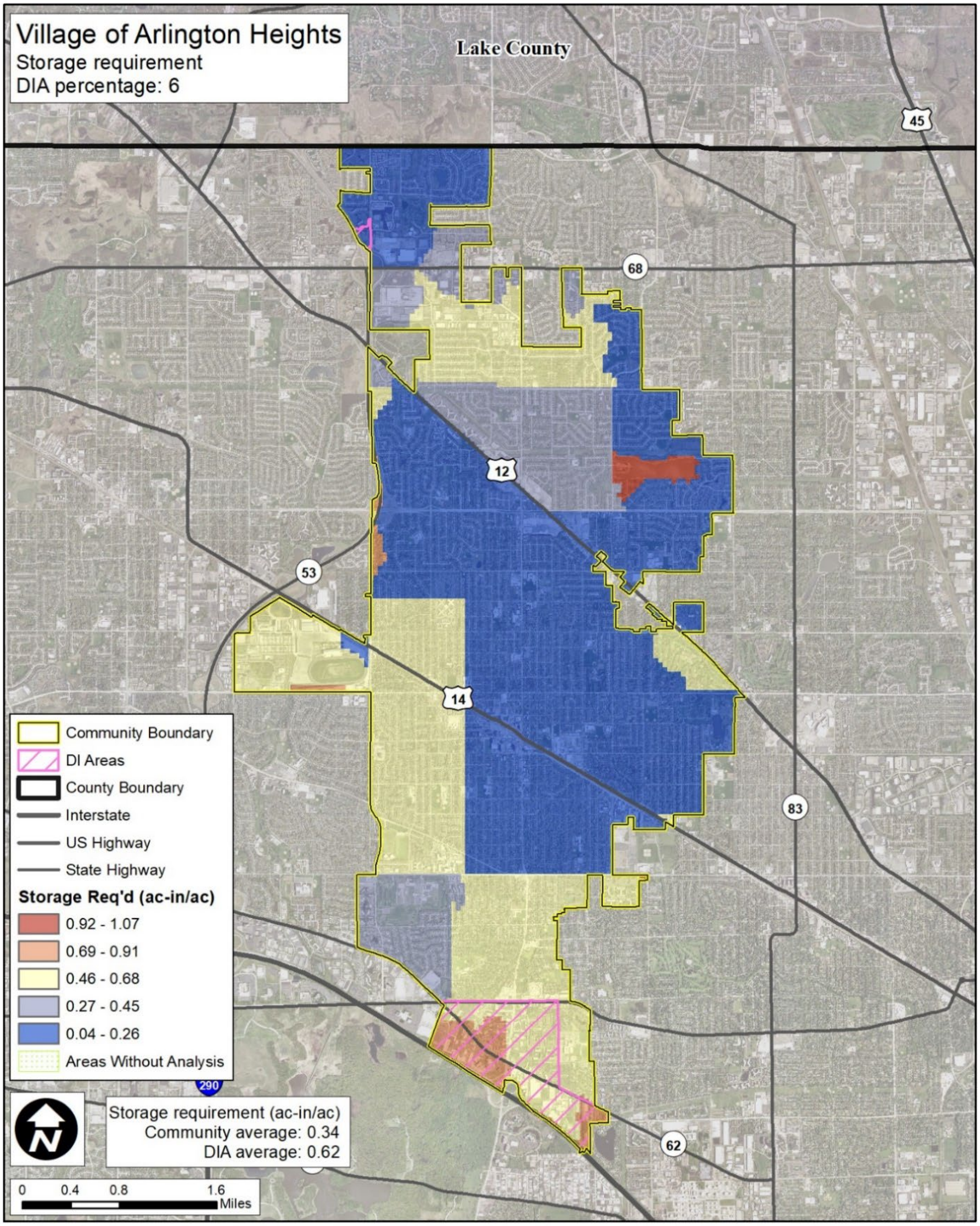
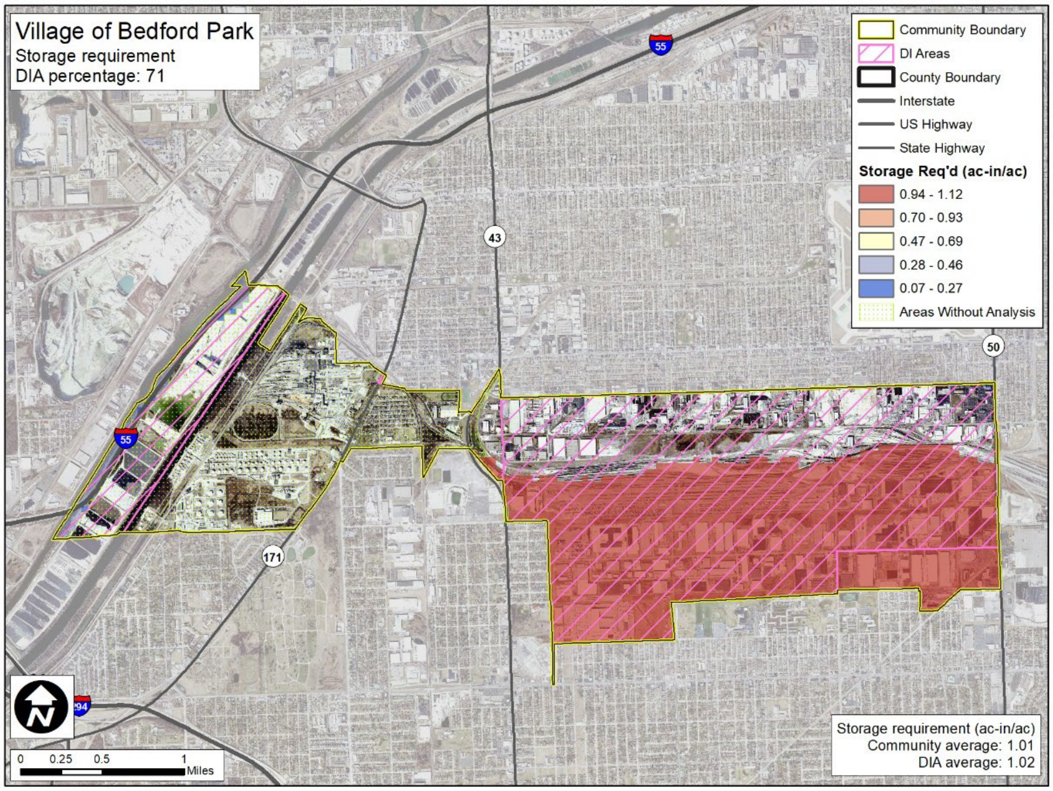


Figure A1



Village of Bedford Park
Storage requirement
DIA percentage: 71
Community Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.94 - 1.12
0.70 - 0.93
0.47 - 0.69
0.28 - 0.46
0.07 - 0.27
Areas Without Analysis
55
43
50
171
294
0 0.25 0.5 1 Miles
Storage requirement (ac-in/ac)
Community average: 1.01
DIA average: 1.02

Figure A2

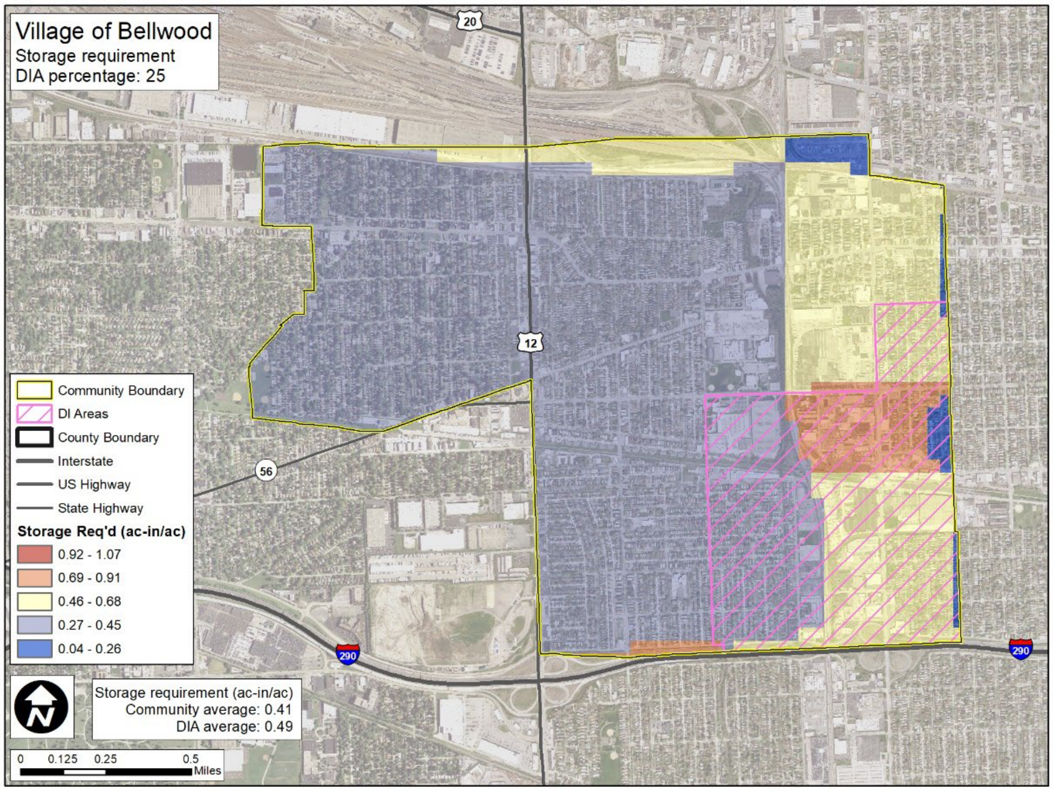

Village of Bellwood
Storage requirement
DIA percentage: 25
Community Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.92 - 1.07
0.69 - 0.91
0.46 - 0.68
0.27 - 0.45
0.04 - 0.26
Storage requirement (ac-in/ac)
Community average: 0.41
DIA average: 0.49
0 0.125 0.25 0.5 Miles
20
12
56
290
290

Figure A3

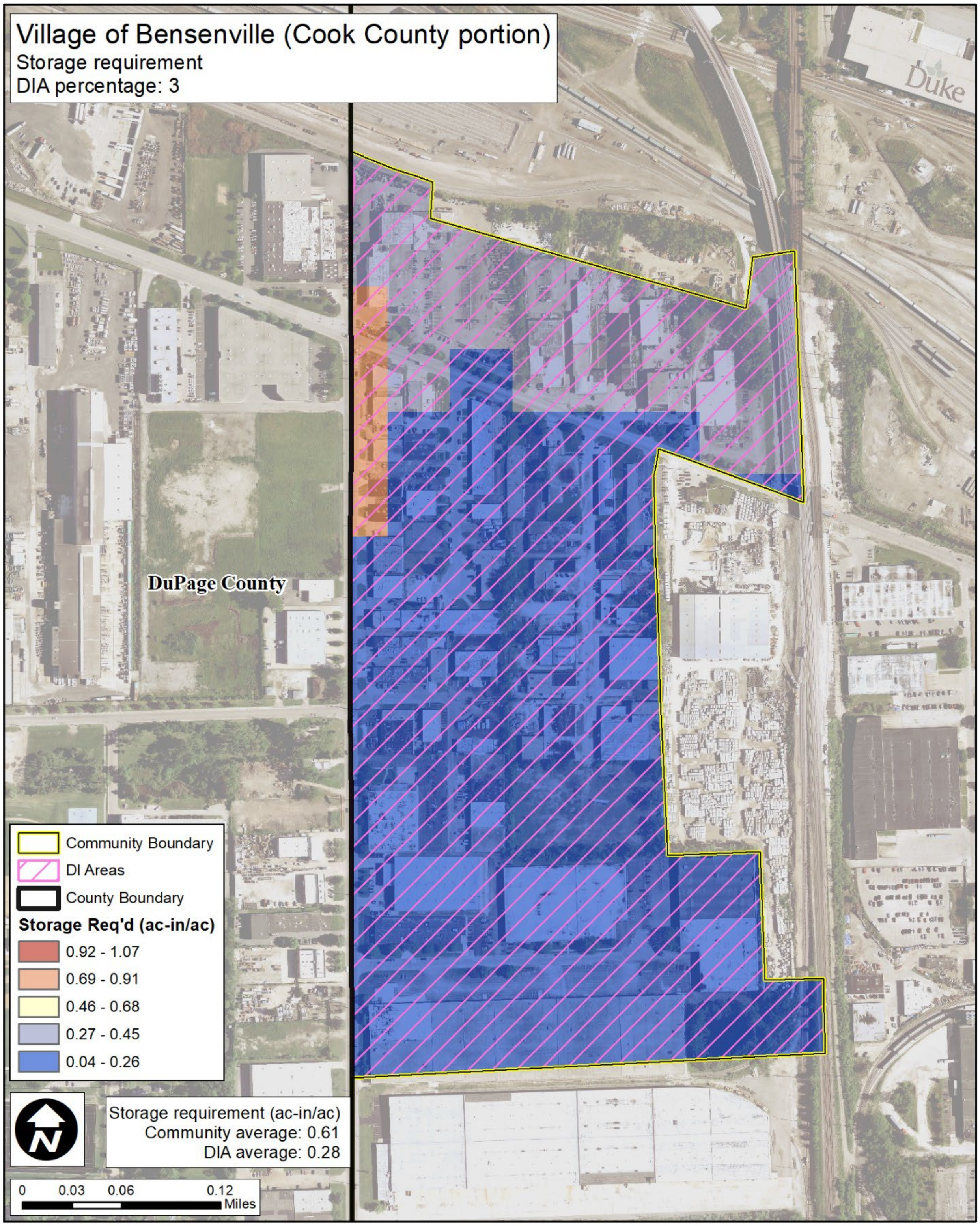

Village of Bensenville (Cook County portion)
Storage requirement
DIA percentage: 3
DuPage County
Duke
Community Boundary
DI Areas
County Boundary
Storage Req'd (ac-in/ac)
0.92 - 1.07
0.69 - 0.91
0.46 - 0.68
0.27 - 0.45
0.04 - 0.26
N
Storage requirement (ac-in/ac)
Community average: 0.61
DIA average: 0.28
0 0.03 0.06 0.12
Miles

Figure A4

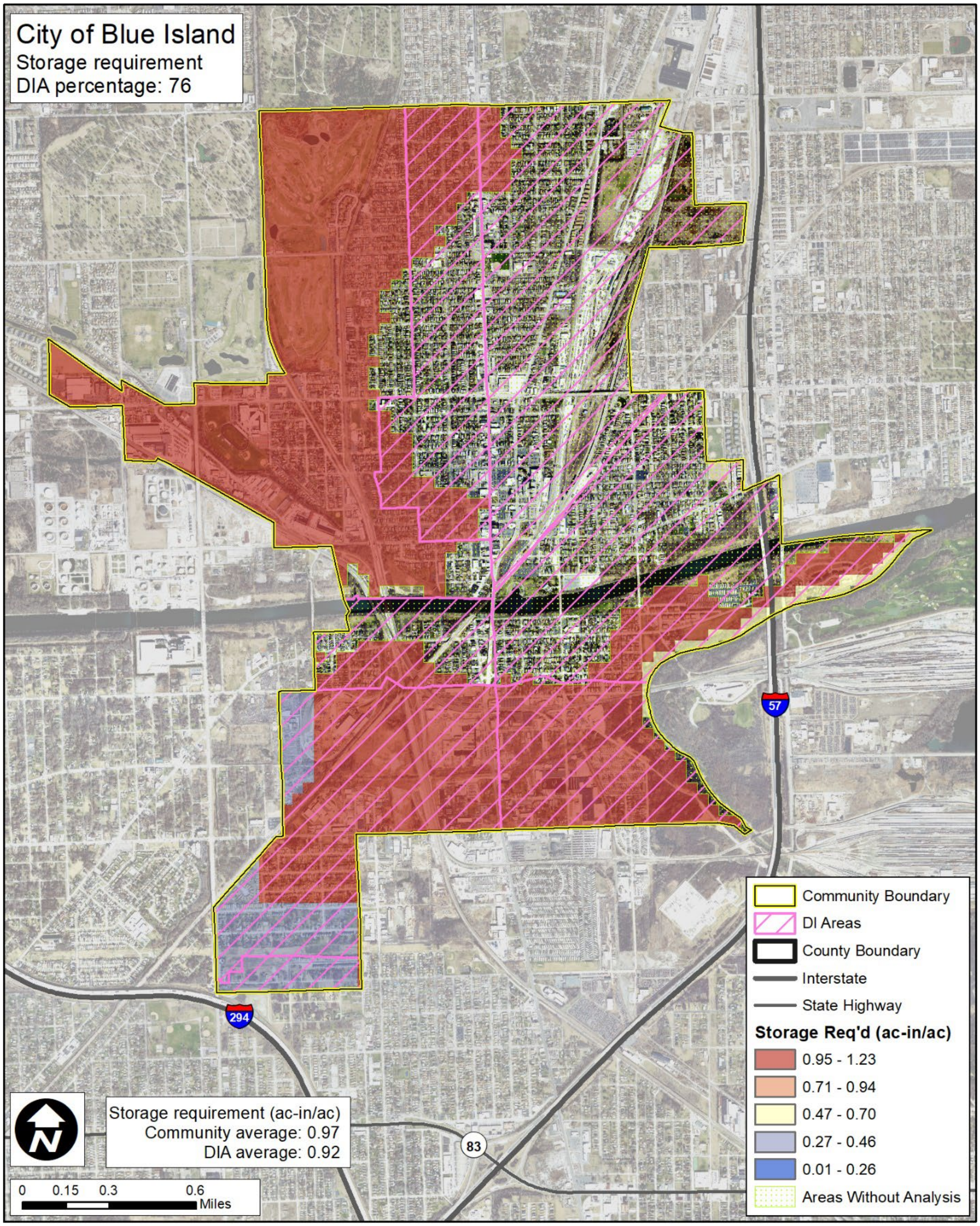
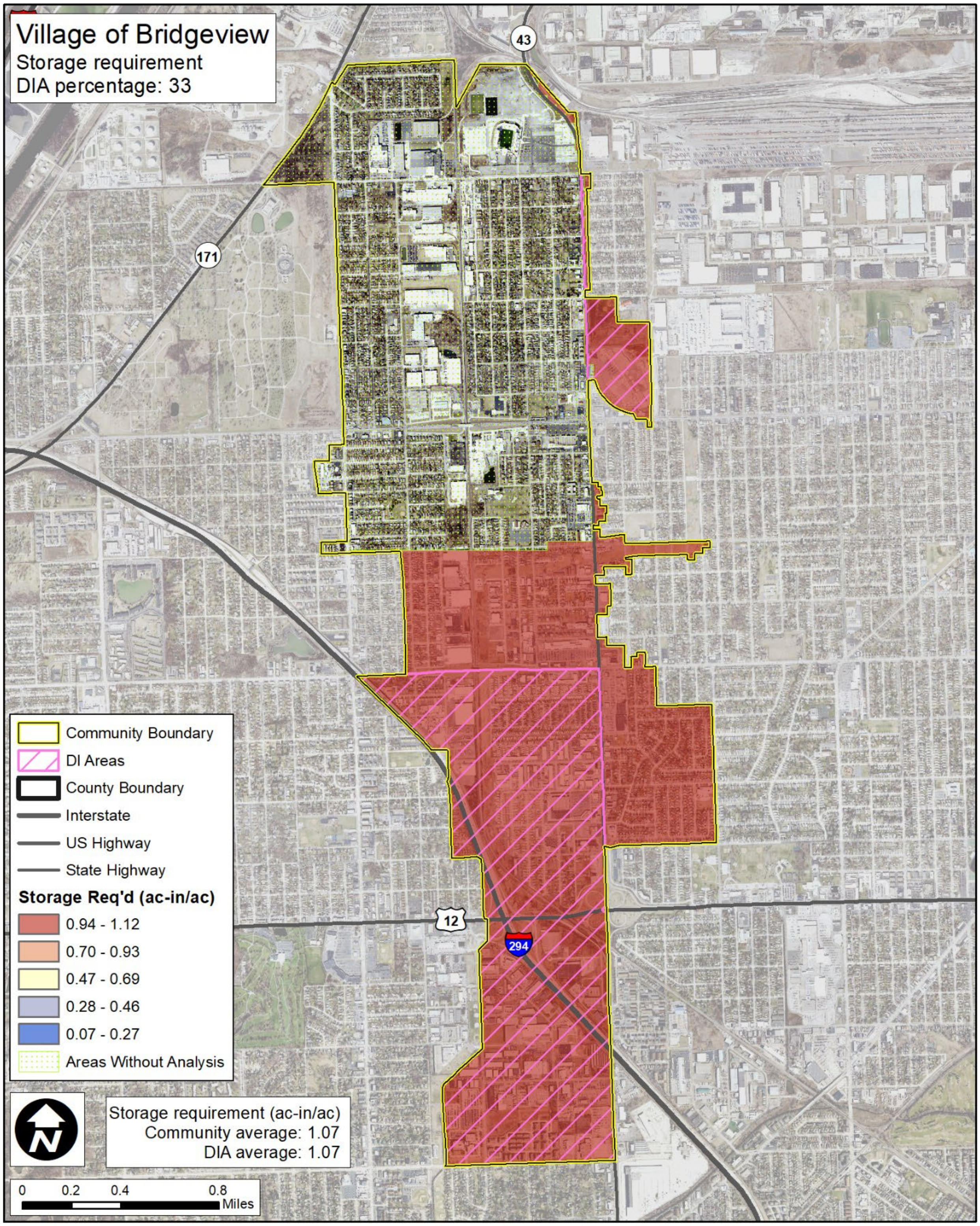

City of Blue Island
Storage requirement
DIA percentage: 76
57
294
83
Community Boundary
DI Areas
County Boundary
Interstate
State Highway
Storage Req'd (ac-in/ac)
0.95 - 1.23
0.71 - 0.94
0.47 - 0.70
0.27 - 0.46
0.01 - 0.26
Areas Without Analysis
Storage requirement (ac-in/ac)
Community average: 0.97
DIA average: 0.92
N
0 0.15 0.3 0.6
Miles

Figure A5



Village of Bridgeview
Storage requirement
DIA percentage: 33
43
171
12
294
Community Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.94 - 1.12
0.70 - 0.93
0.47 - 0.69
0.28 - 0.46
0.07 - 0.27
Areas Without Analysis
N
Storage requirement (ac-in/ac)
Community average: 1.07
DIA average: 1.07
0 0.2 0.4 0.8
Miles

Figure A6

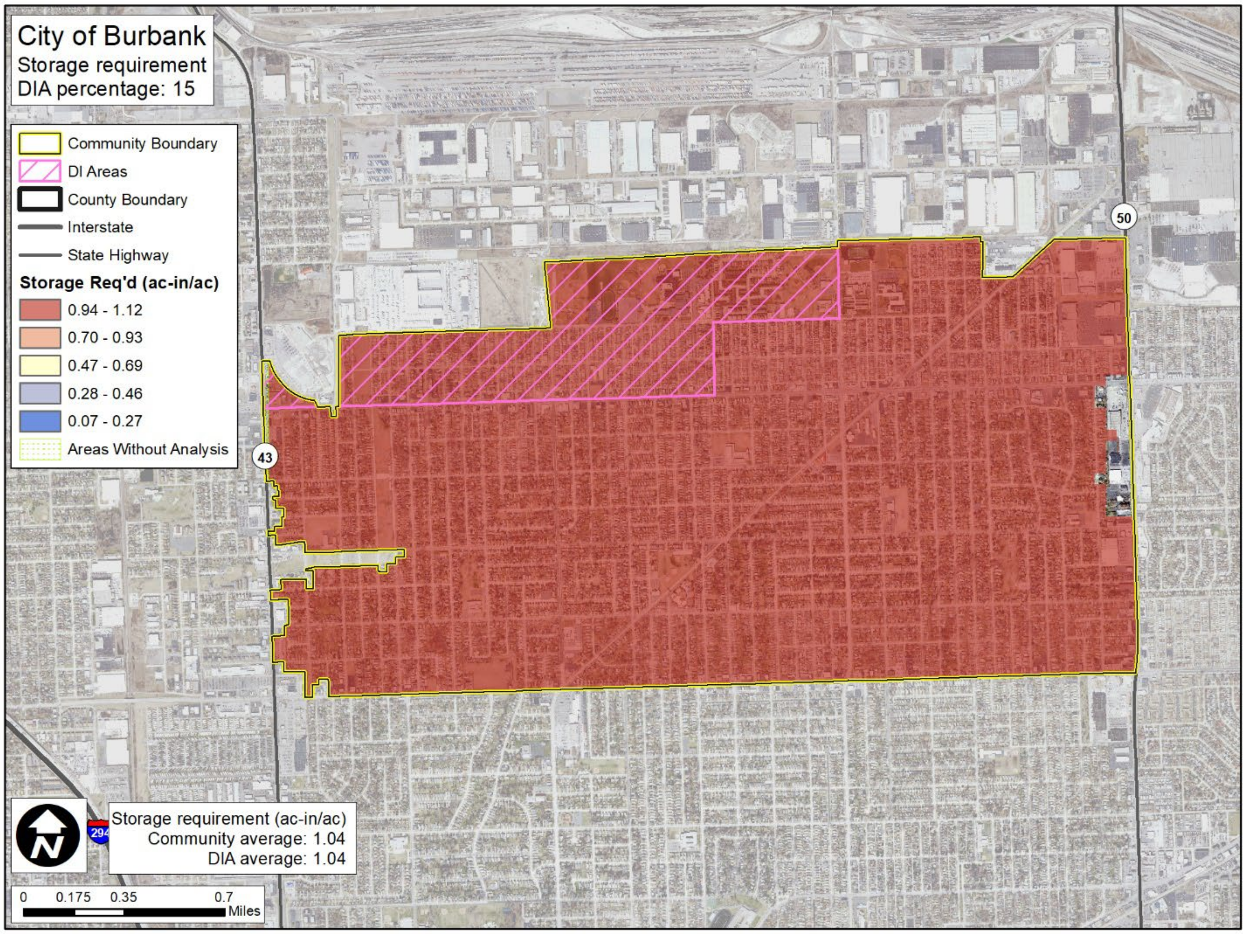

City of Burbank
Storage requirement
DIA percentage: 15
Community Boundary
DI Areas
County Boundary
Interstate
State Highway
Storage Req'd (ac-in/ac)
0.94 - 1.12
0.70 - 0.93
0.47 - 0.69
0.28 - 0.46
0.07 - 0.27
Areas Without Analysis
50
43
294
Storage requirement (ac-in/ac)
Community average: 1.04
DIA average: 1.04
0 0.175 0.35 0.7
Miles

Figure A7

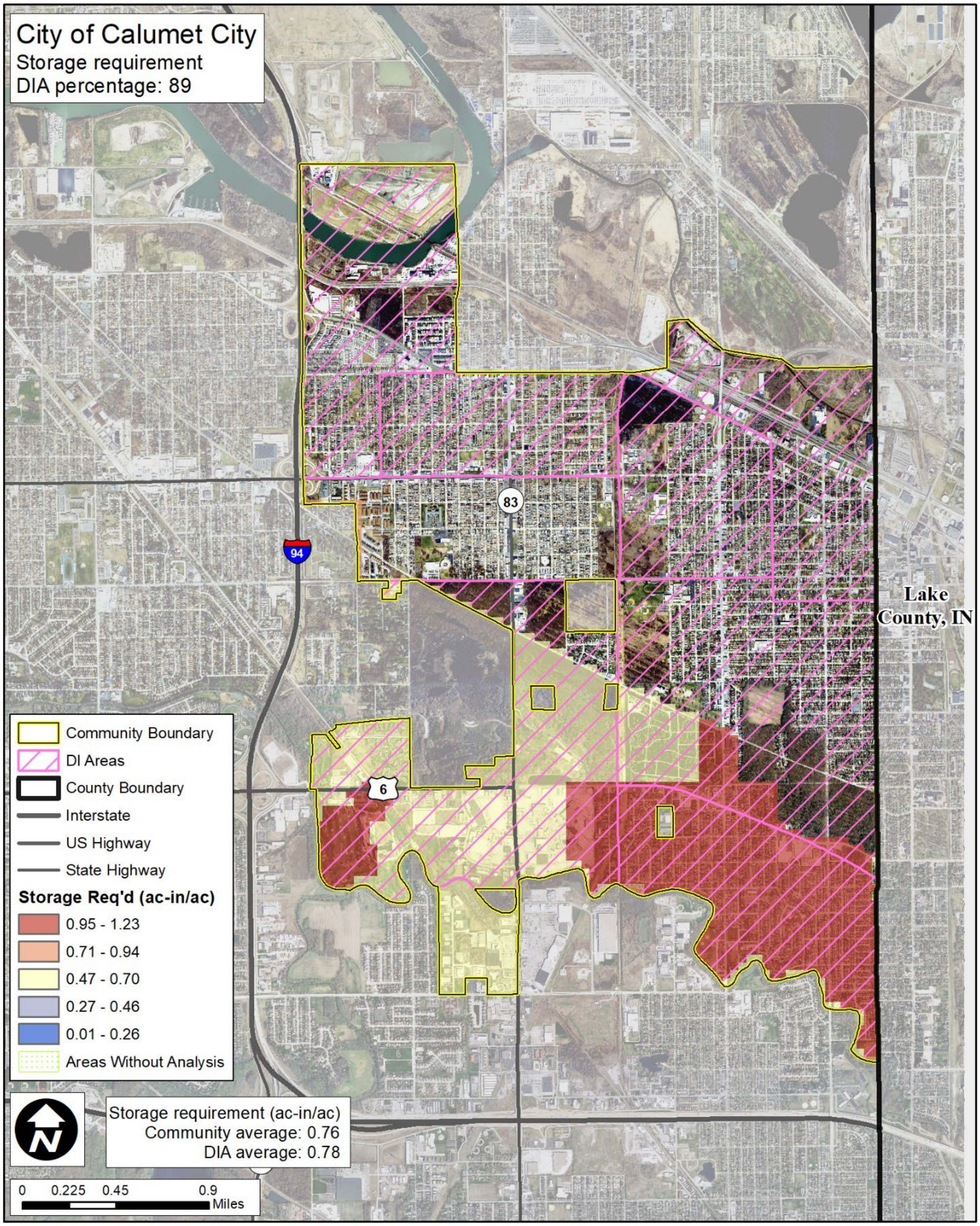
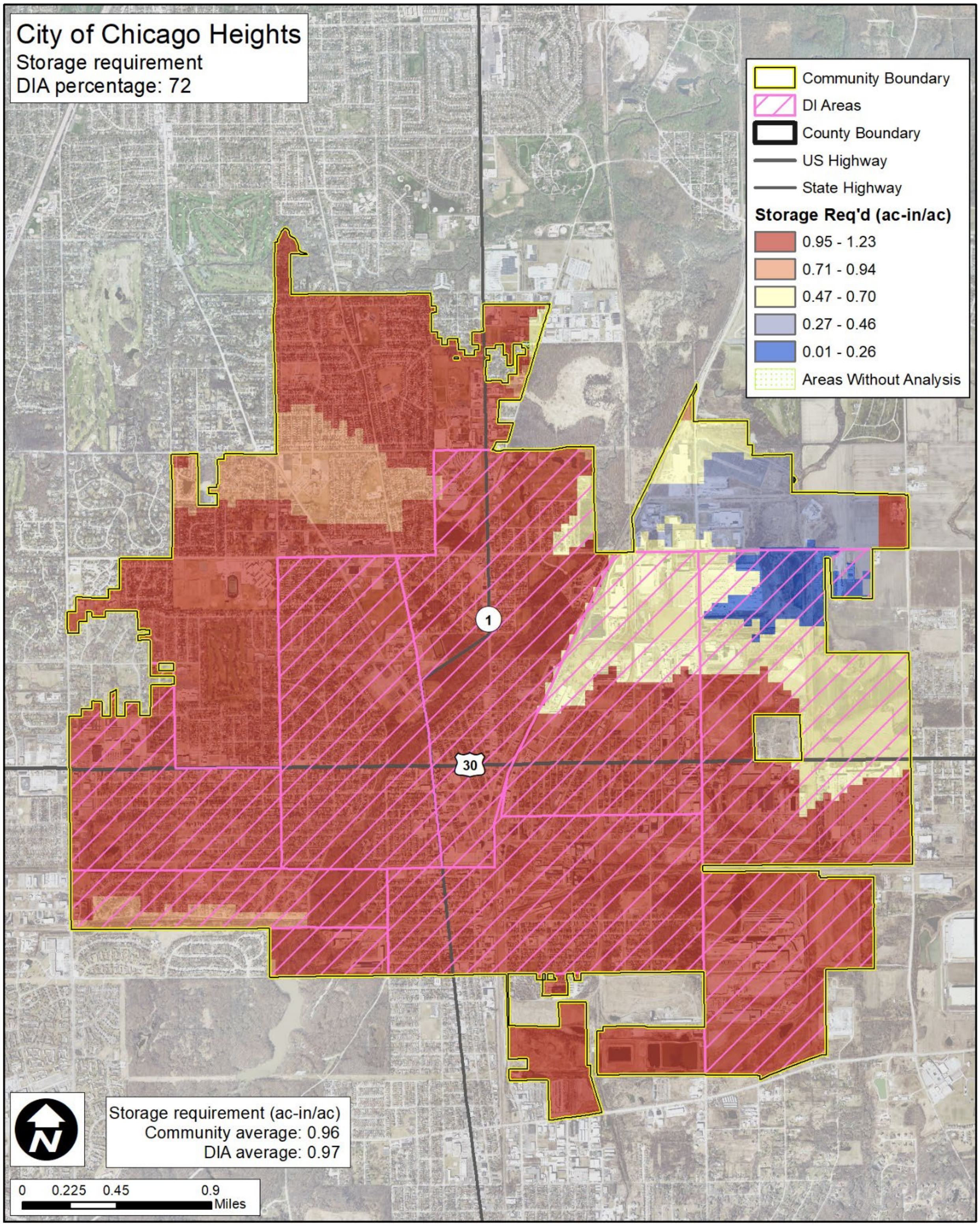
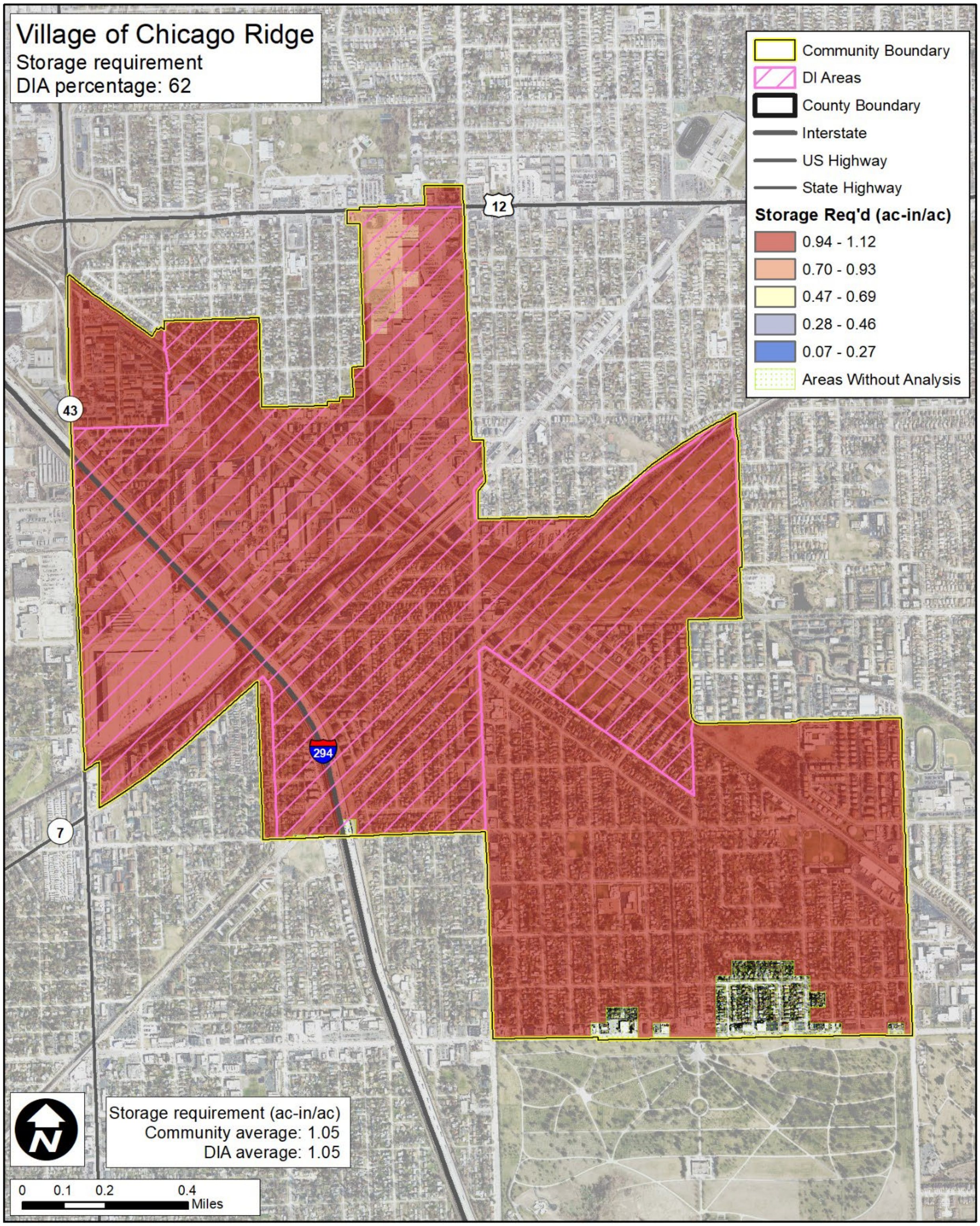

City of Calumet City
Storage requirement
DIA percentage: 89
83
94
6
Lake County, IN
Community Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.95 - 1.23
0.71 - 0.94
0.47 - 0.70
0.27 - 0.46
0.01 - 0.26
Areas Without Analysis
N
Storage requirement (ac-in/ac)
Community average: 0.76
DIA average: 0.78
0 0.225 0.45 0.9
Miles

Figure A8



City of Chicago Heights
Storage requirement
DIA percentage: 72
Community Boundary
DI Areas
County Boundary
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.95 - 1.23
0.71 - 0.94
0.47 - 0.70
0.27 - 0.46
0.01 - 0.26
Areas Without Analysis
1
30
Storage requirement (ac-in/ac)
Community average: 0.96
DIA average: 0.97
N
0 0.225 0.45 0.9
Miles

Figure A9



Village of Chicago Ridge
Storage requirement
DIA percentage: 62
Community Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.94 - 1.12
0.70 - 0.93
0.47 - 0.69
0.28 - 0.46
0.07 - 0.27
Areas Without Analysis
12
43
294
7
Storage requirement (ac-in/ac)
Community average: 1.05
DIA average: 1.05
0 0.1 0.2 0.4
Miles

Figure A10

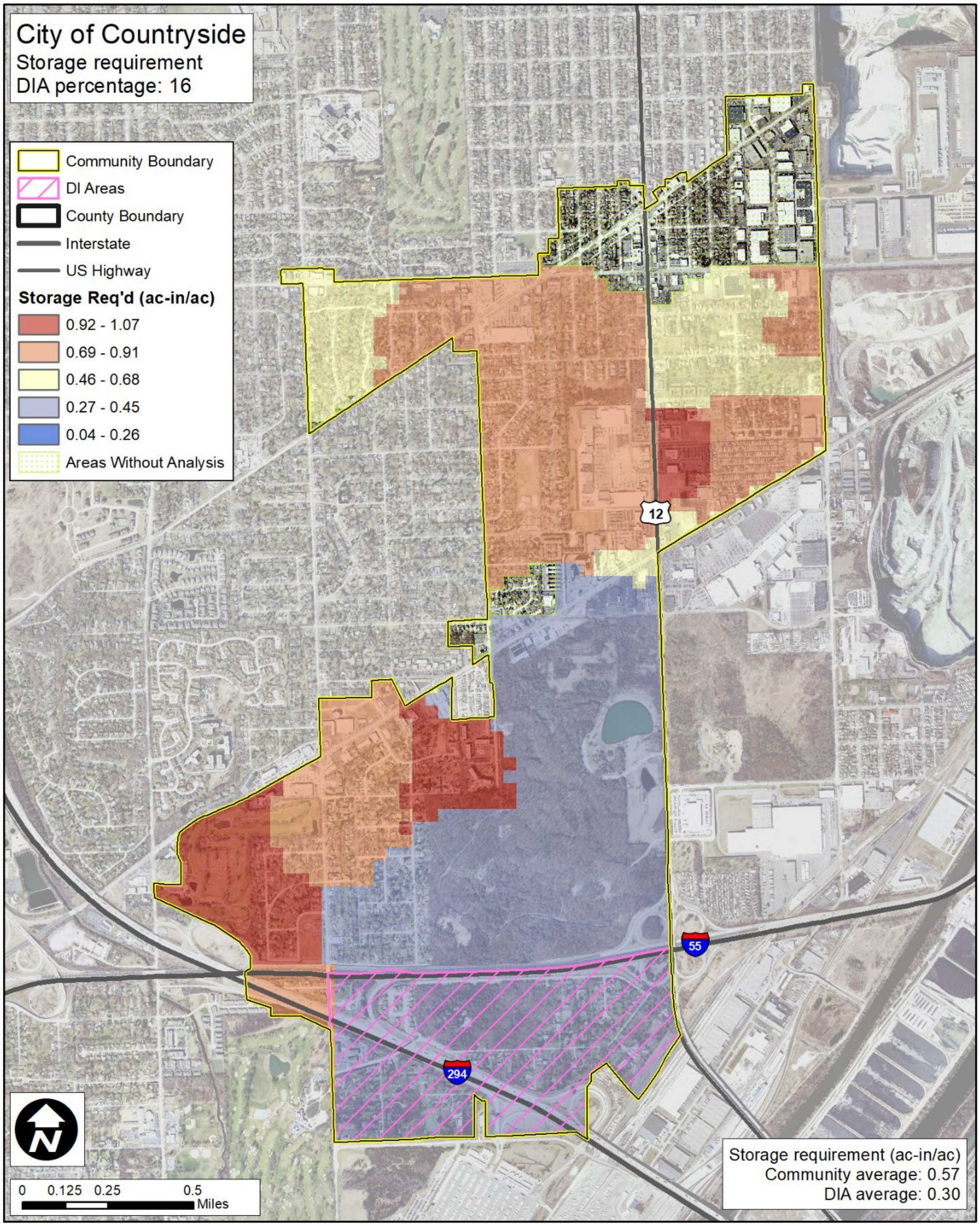

City of Countryside
Storage requirement
DIA percentage: 16
Community Boundary
DI Areas
County Boundary
Interstate
US Highway
Storage Req'd (ac-in/ac)
0.92 - 1.07
0.69 - 0.91
0.46 - 0.68
0.27 - 0.45
0.04 - 0.26
Areas Without Analysis
12
55
294
N
0 0.125 0.25 0.5
Miles
Storage requirement (ac-in/ac)
Community average: 0.57
DIA average: 0.30

Figure A11

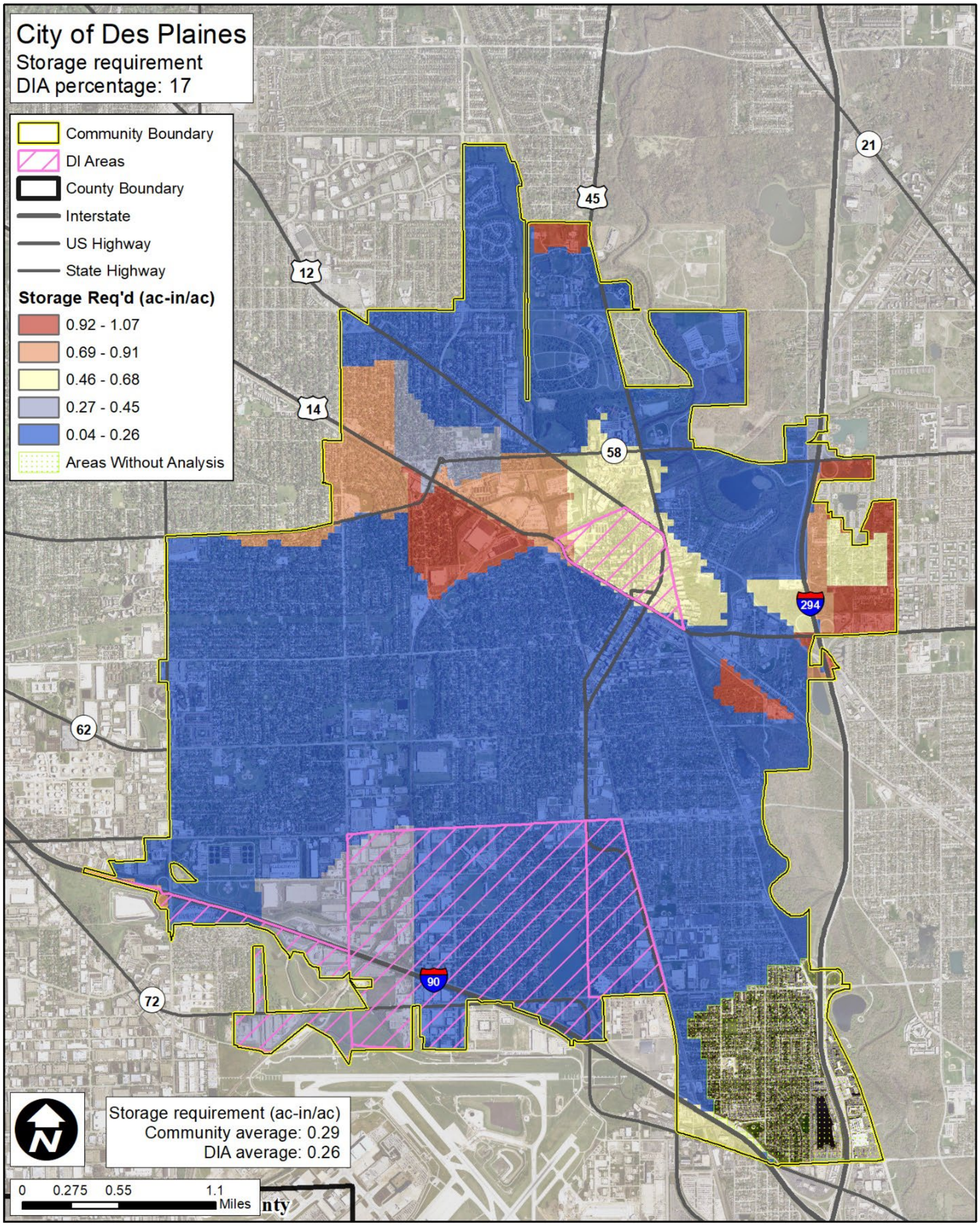

City of Des Plaines
Storage requirement
DIA percentage: 17
Community Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.92 - 1.07
0.69 - 0.91
0.46 - 0.68
0.27 - 0.45
0.04 - 0.26
Areas Without Analysis
21
45
12
14
58
294
62
90
72
Storage requirement (ac-in/ac)
Community average: 0.29
DIA average: 0.26
0 0.275 0.55 1.1
Miles

Figure A12

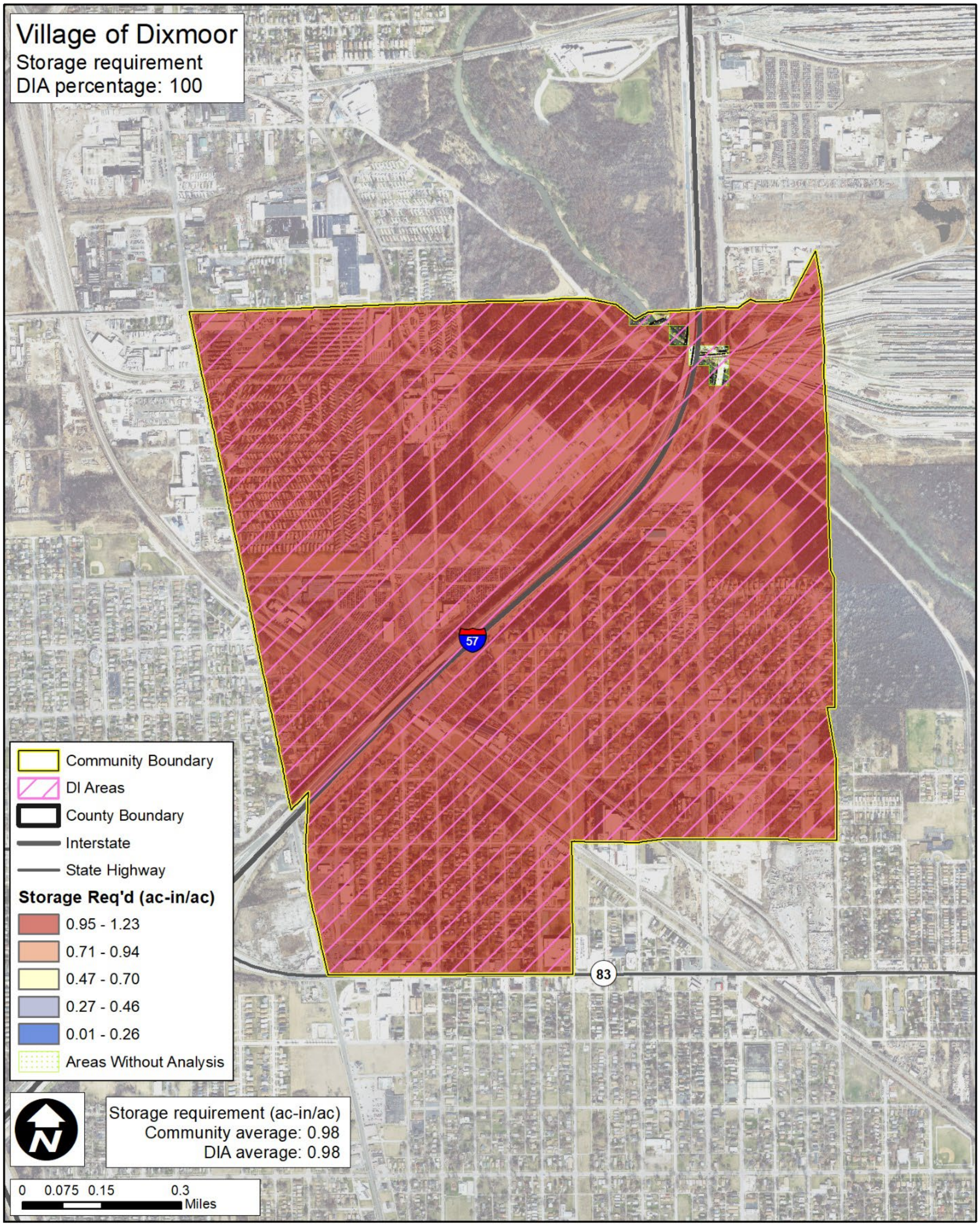

Village of Dixmoor
Storage requirement
DIA percentage: 100
Community Boundary
DI Areas
County Boundary
Interstate
State Highway
Storage Req'd (ac-in/ac)
0.95 - 1.23
0.71 - 0.94
0.47 - 0.70
0.27 - 0.46
0.01 - 0.26
Areas Without Analysis
Storage requirement (ac-in/ac)
Community average: 0.98
DIA average: 0.98
0 0.075 0.15 0.3
Miles
57
83

Figure A13

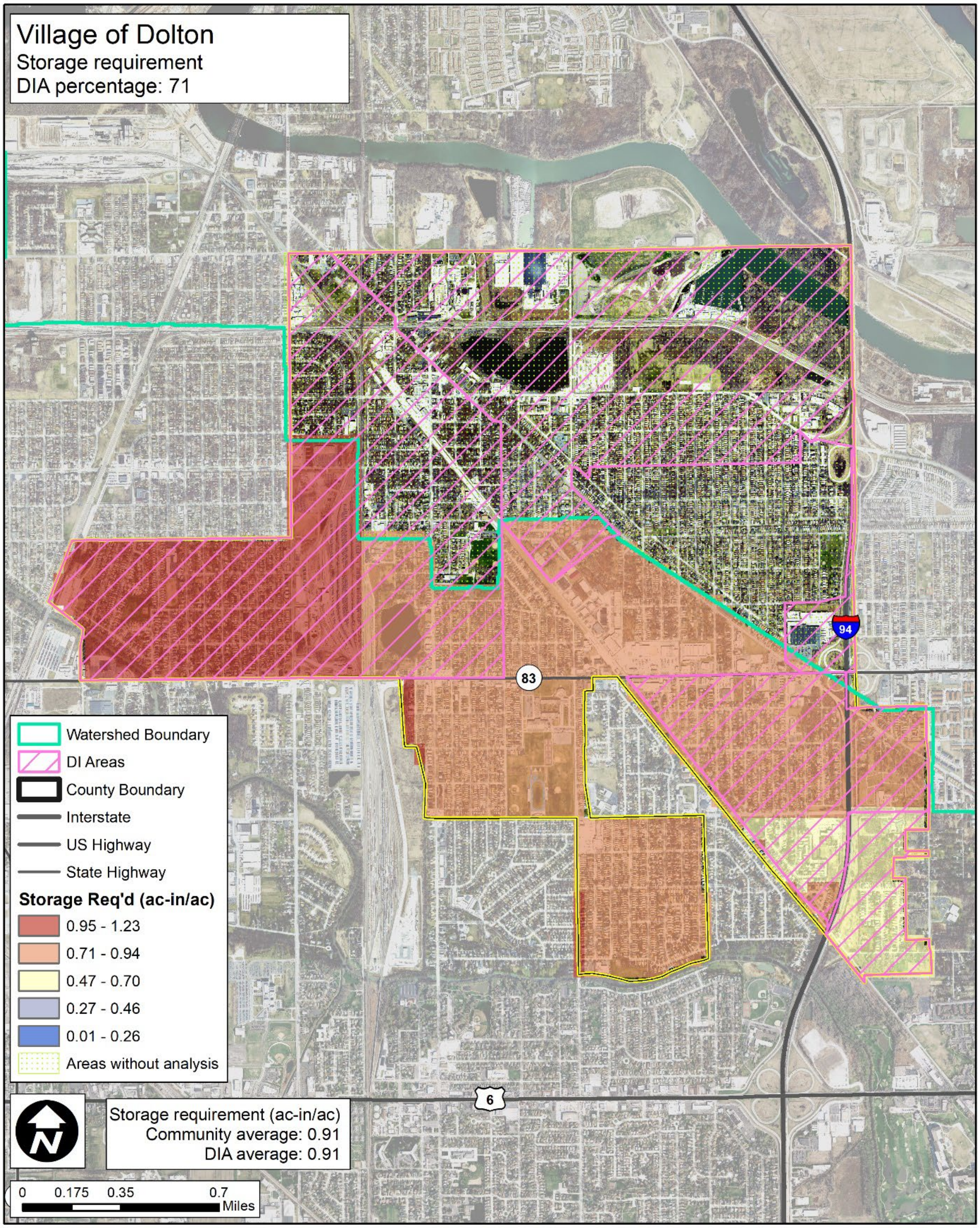

Village of Dolton
Storage requirement
DIA percentage: 71
83
94
6
Watershed Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.95 - 1.23
0.71 - 0.94
0.47 - 0.70
0.27 - 0.46
0.01 - 0.26
Areas without analysis
Storage requirement (ac-in/ac)
Community average: 0.91
DIA average: 0.91
0 0.175 0.35 0.7
Miles

Figure A14

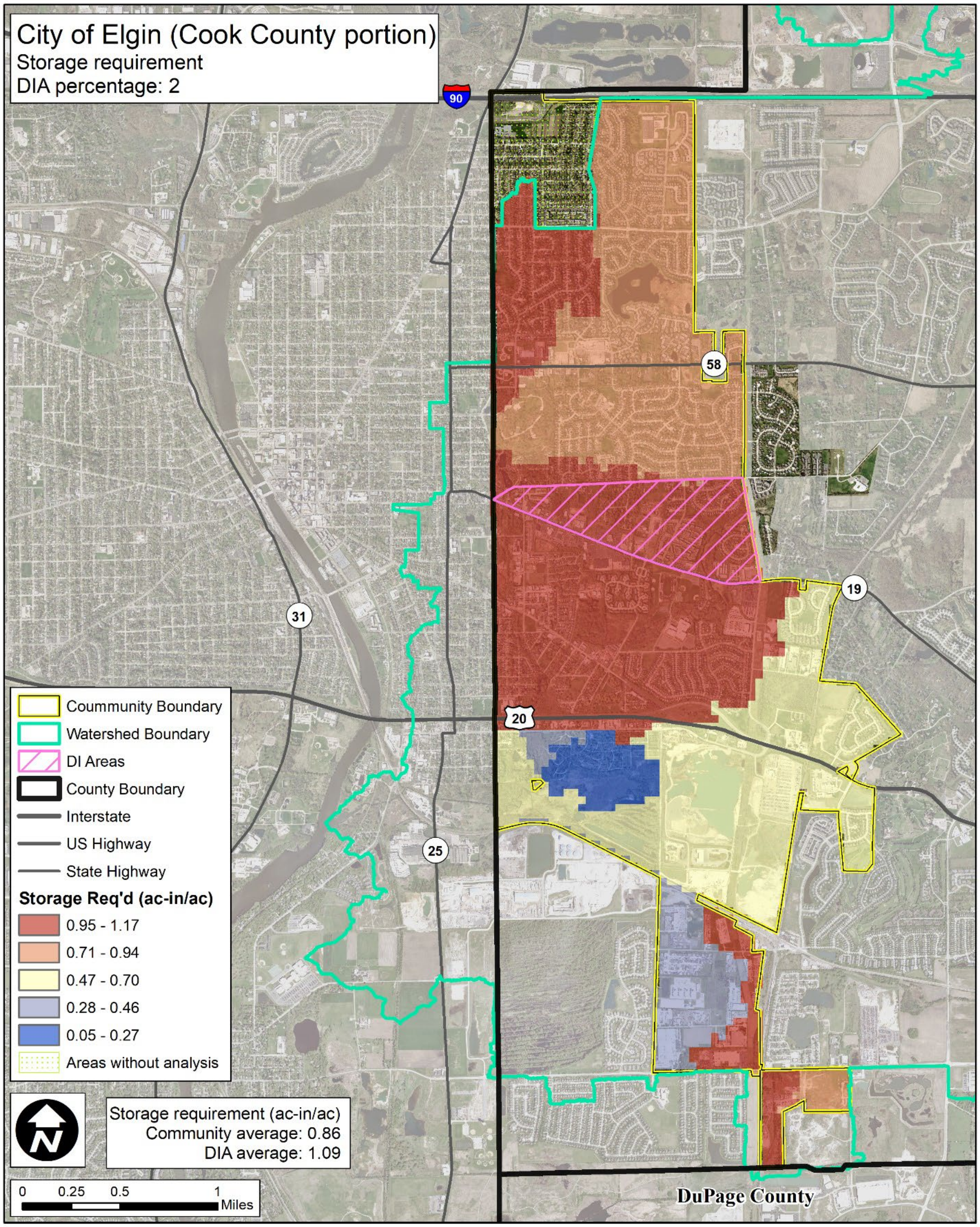
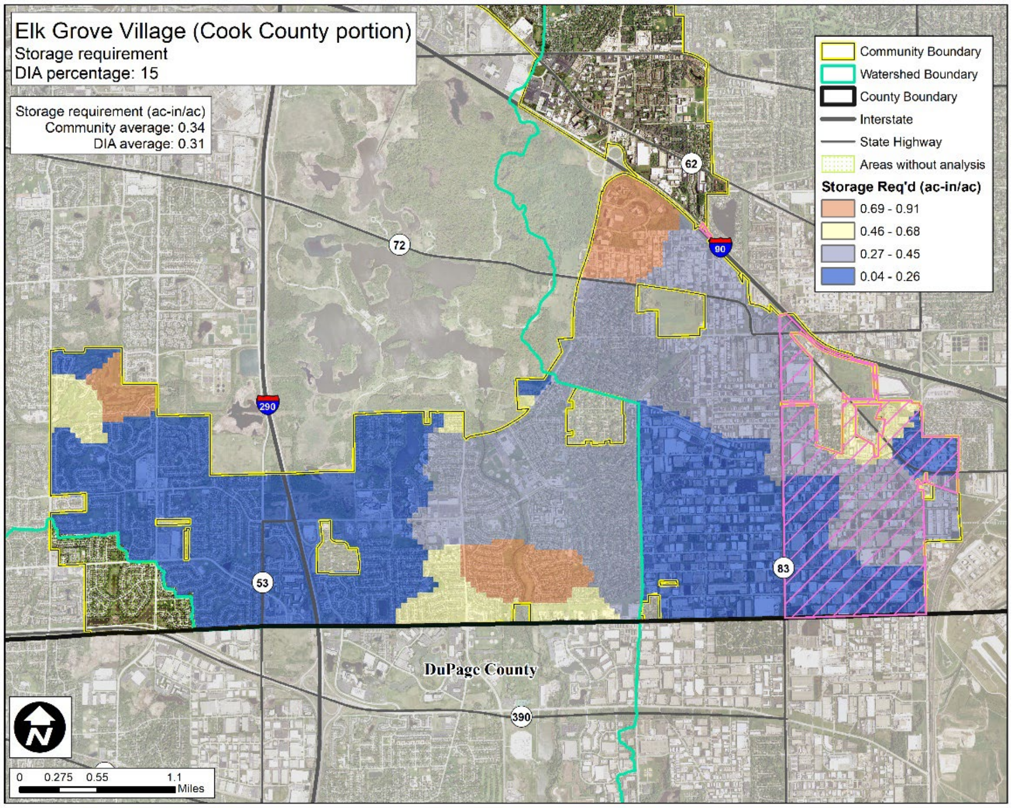

City of Elgin (Cook County portion)
Storage requirement
DIA percentage: 2
90
58
19
31
20
25
Coummunity Boundary
Watershed Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.95 - 1.17
0.71 - 0.94
0.47 - 0.70
0.28 - 0.46
0.05 - 0.27
Areas without analysis
Storage requirement (ac-in/ac)
Community average: 0.86
DIA average: 1.09
N
0 0.25 0.5 1
Miles
DuPage County

Figure A15



Elk Grove Village (Cook County portion)
Storage requirement
DIA percentage: 15
Storage requirement (ac-in/ac)
Community average: 0.34
DIA average: 0.31
Community Boundary
Watershed Boundary
County Boundary
Interstate
State Highway
Areas without analysis
Storage Req'd (ac-in/ac)
0.69 - 0.91
0.46 - 0.68
0.27 - 0.45
0.04 - 0.26
DuPage County
0 0.275 0.55 1.1
Miles

Figure A16

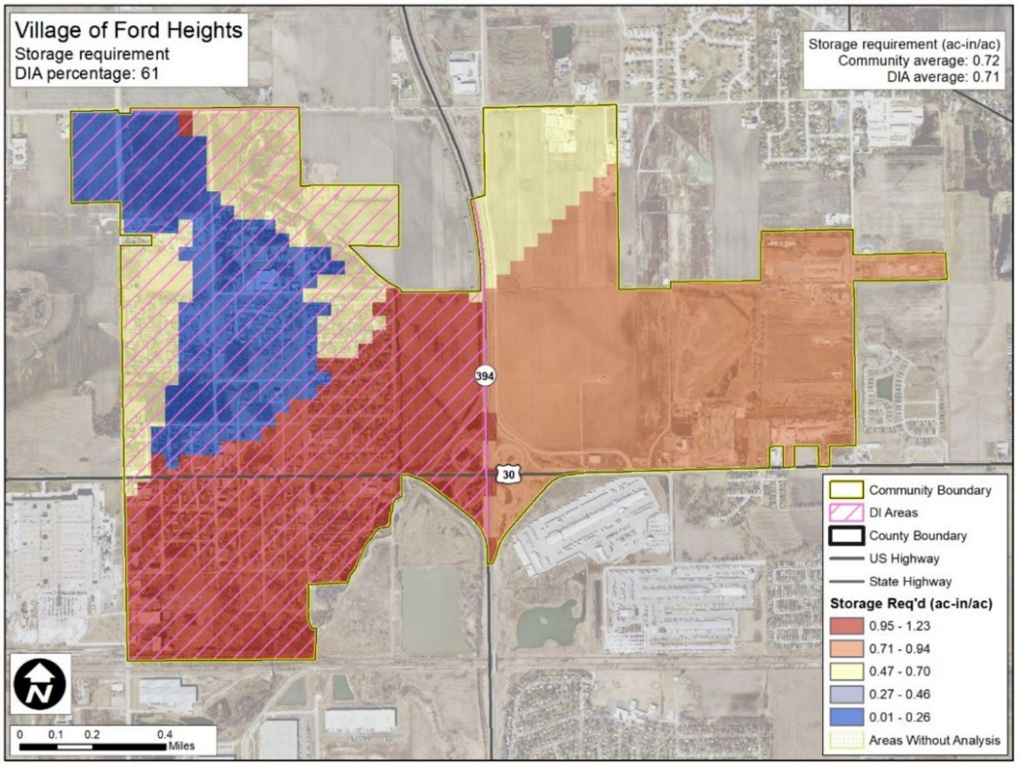


Figure A17

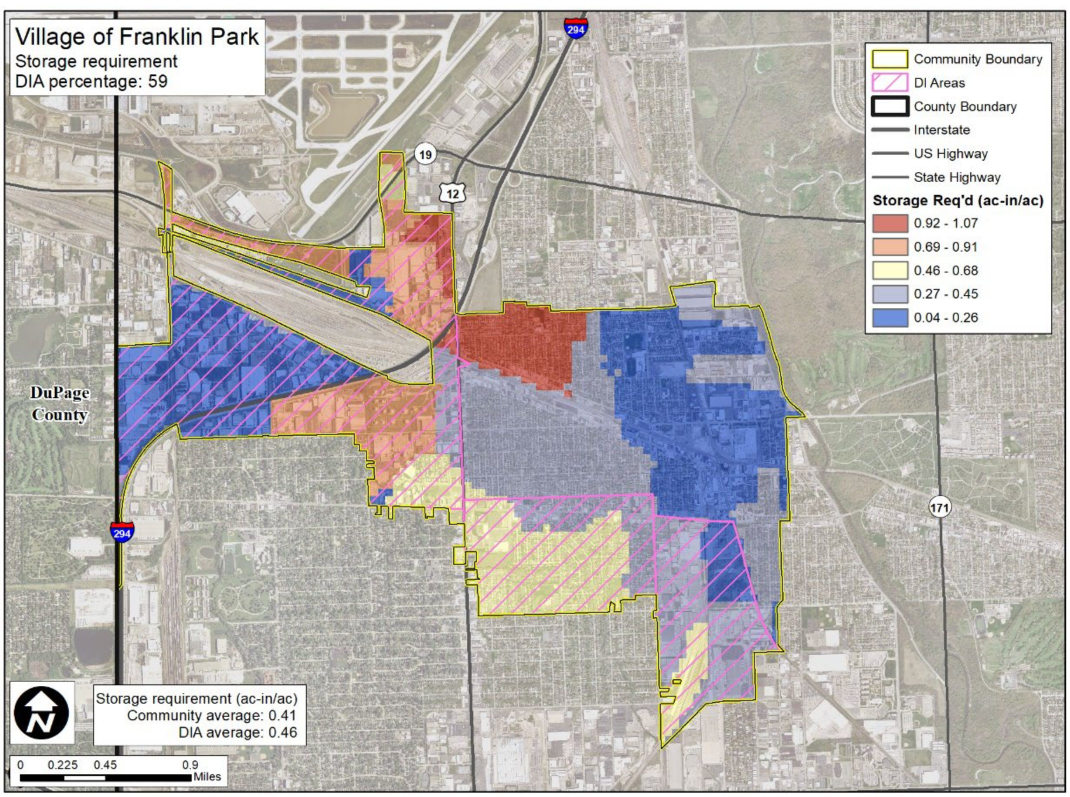


Figure A18

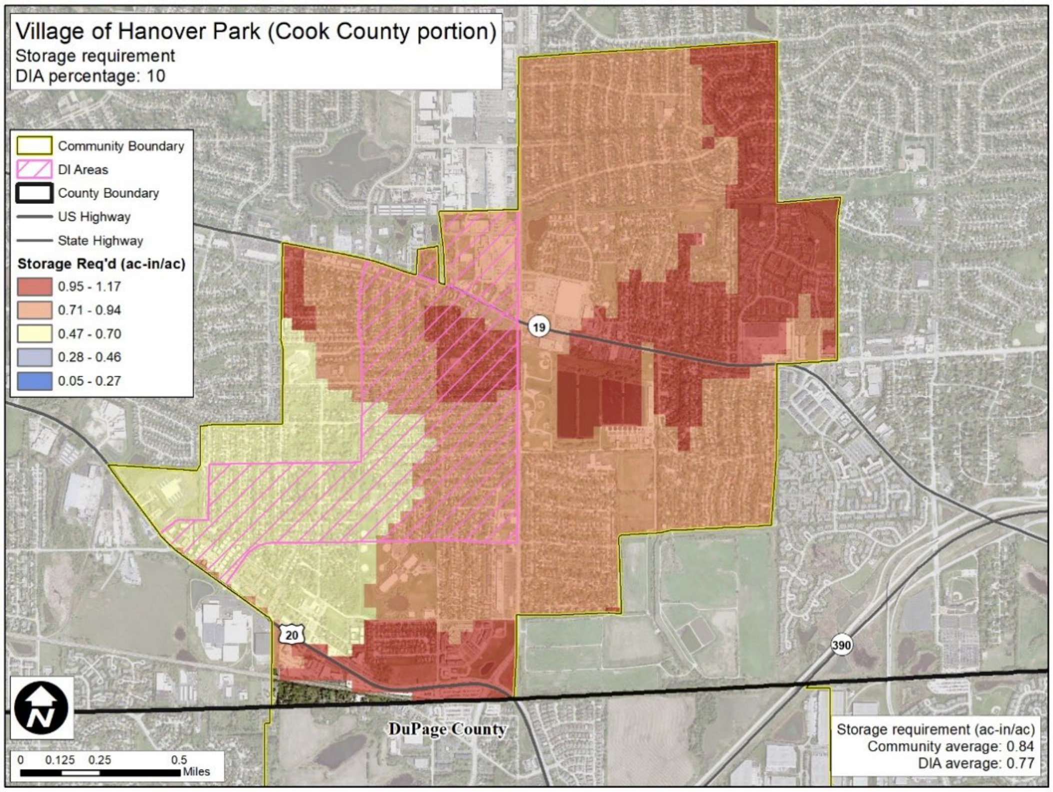

Village of Hanover Park (Cook County portion)
Storage requirement
DIA percentage: 10
Community Boundary
DI Areas
County Boundary
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.95 - 1.17
0.71 - 0.94
0.47 - 0.70
0.28 - 0.46
0.05 - 0.27
19
20
390
DuPage County
Storage requirement (ac-in/ac)
Community average: 0.84
DIA average: 0.77
0 0.125 0.25 0.5 Miles

Figure A19

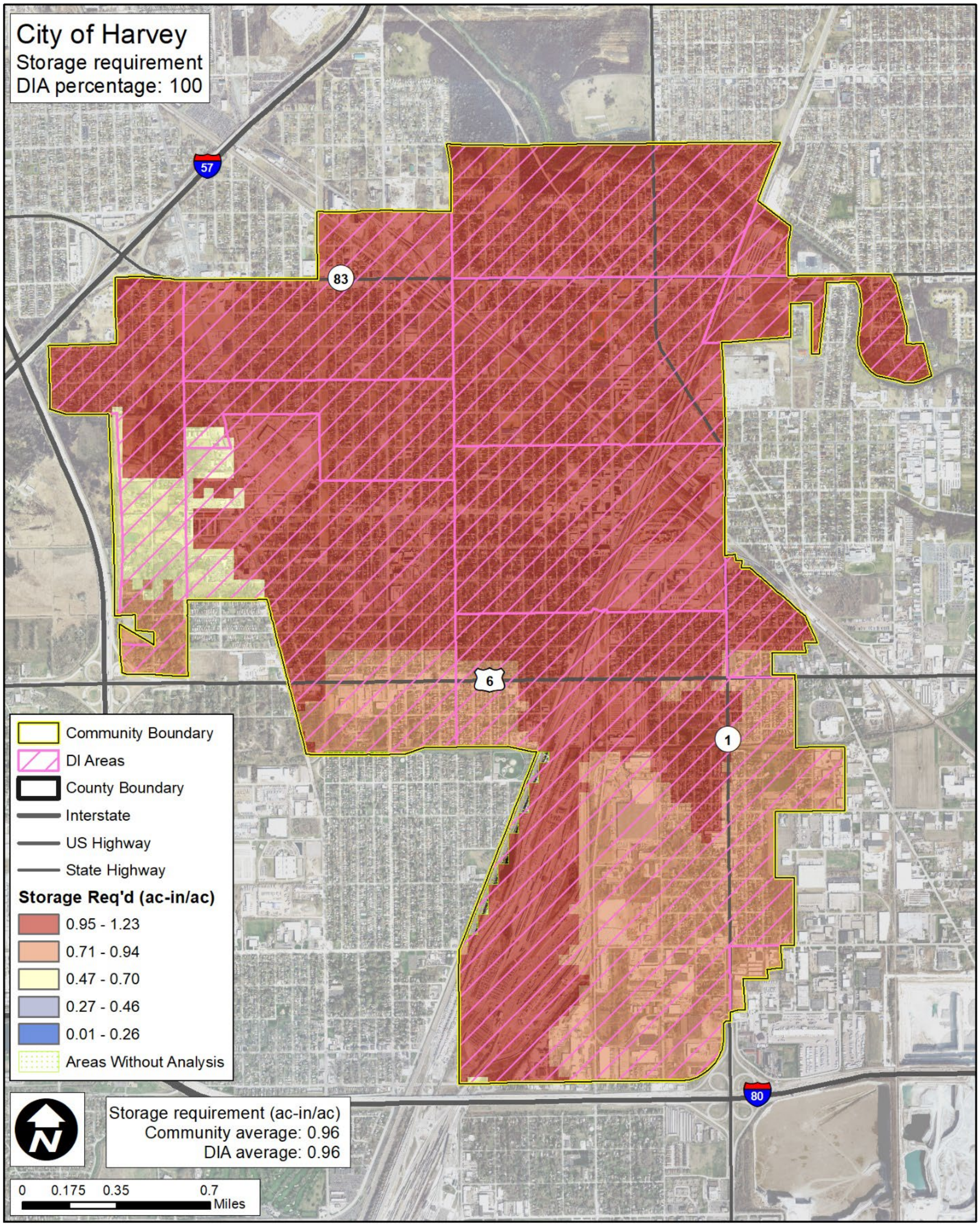

City of Harvey
Storage requirement
DIA percentage: 100
57
83
6
1
80
Community Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.95 - 1.23
0.71 - 0.94
0.47 - 0.70
0.27 - 0.46
0.01 - 0.26
Areas Without Analysis
N
Storage requirement (ac-in/ac)
Community average: 0.96
DIA average: 0.96
0 0.175 0.35 0.7
Miles

Figure A20

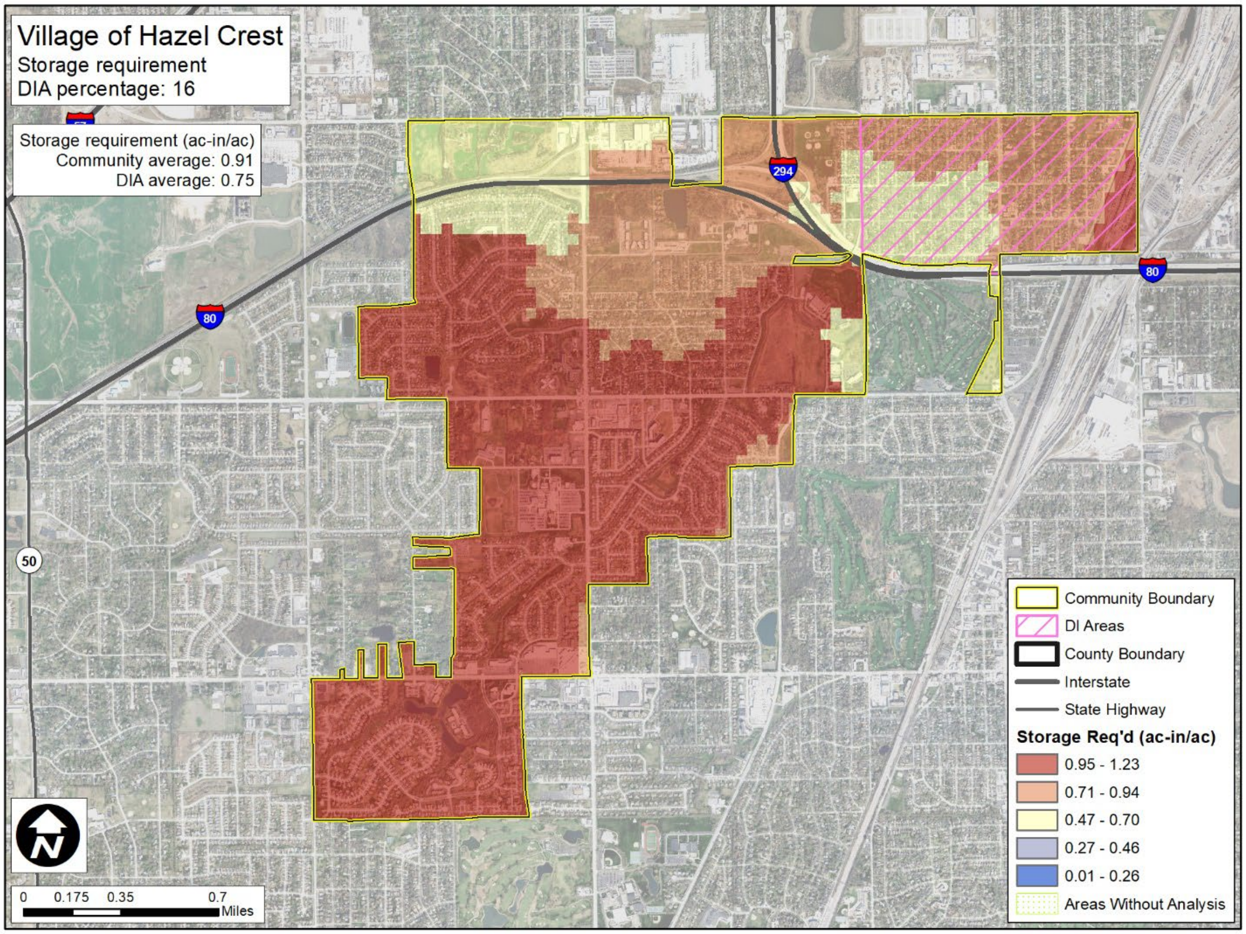

Village of Hazel Crest
Storage requirement
DIA percentage: 16
Storage requirement (ac-in/ac)
Community average: 0.91
DIA average: 0.75
294
80
80
50
Community Boundary
DI Areas
County Boundary
Interstate
State Highway
Storage Req'd (ac-in/ac)
0.95 - 1.23
0.71 - 0.94
0.47 - 0.70
0.27 - 0.46
0.01 - 0.26
Areas Without Analysis
0 0.175 0.35 0.7
Miles

Figure A21

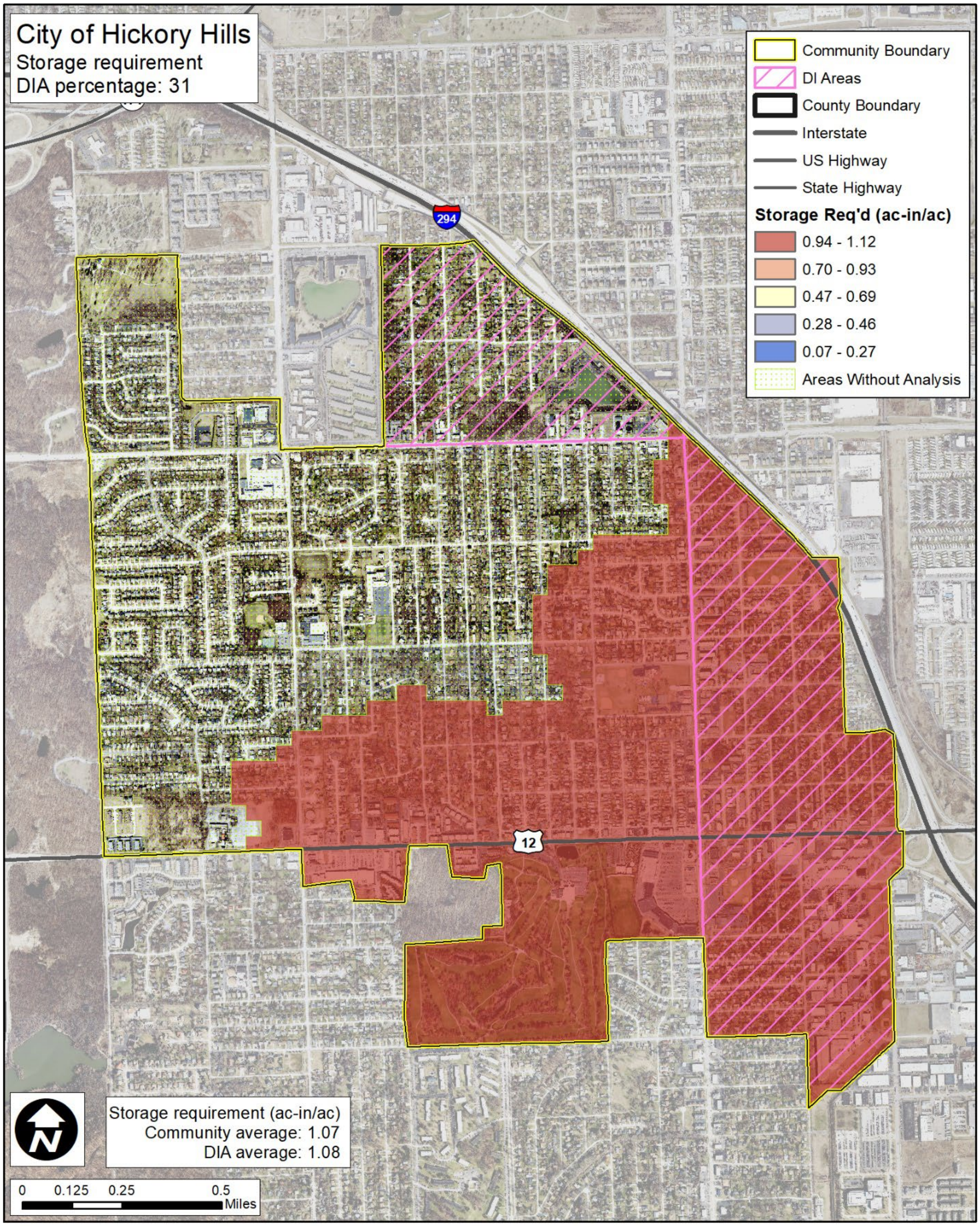
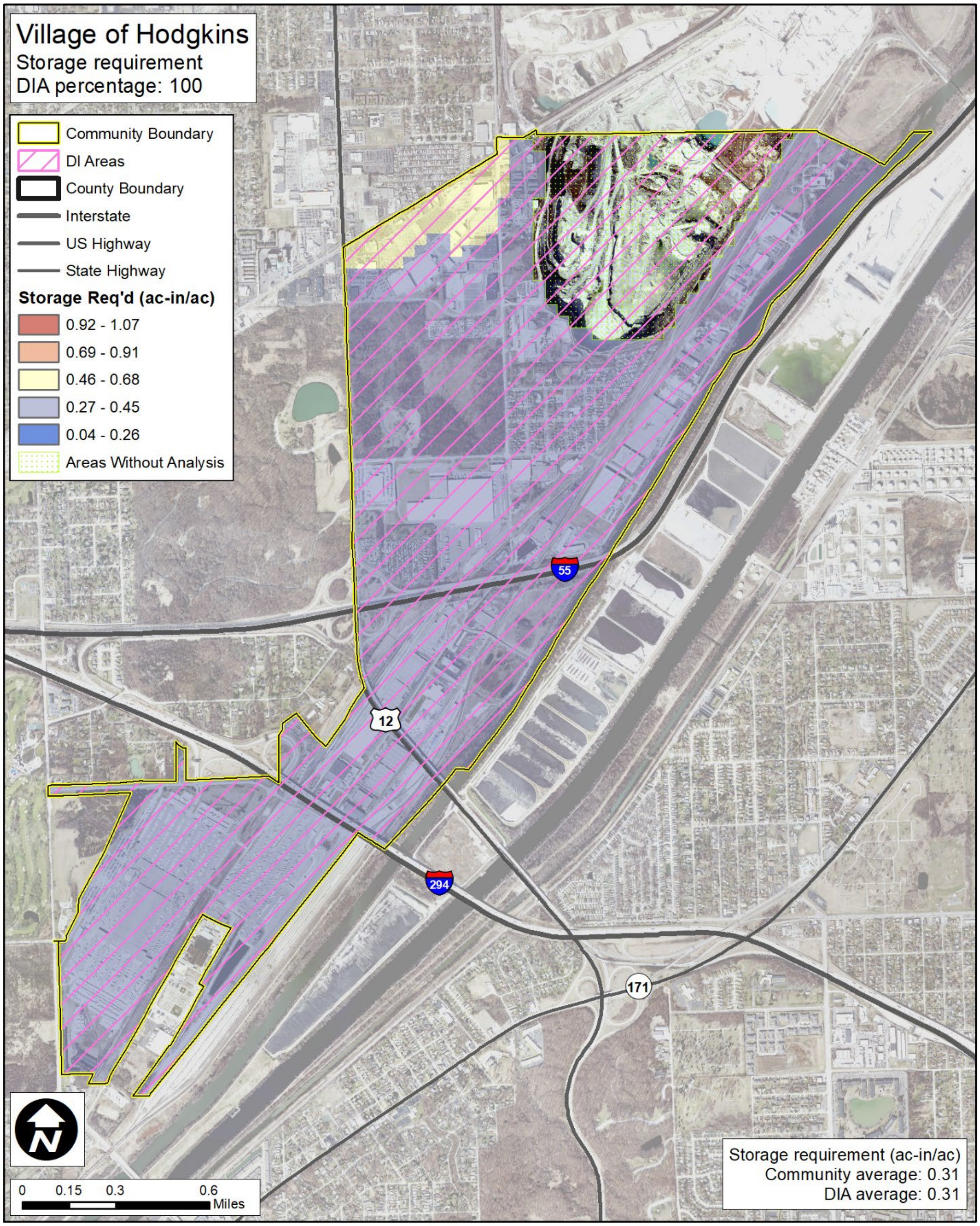
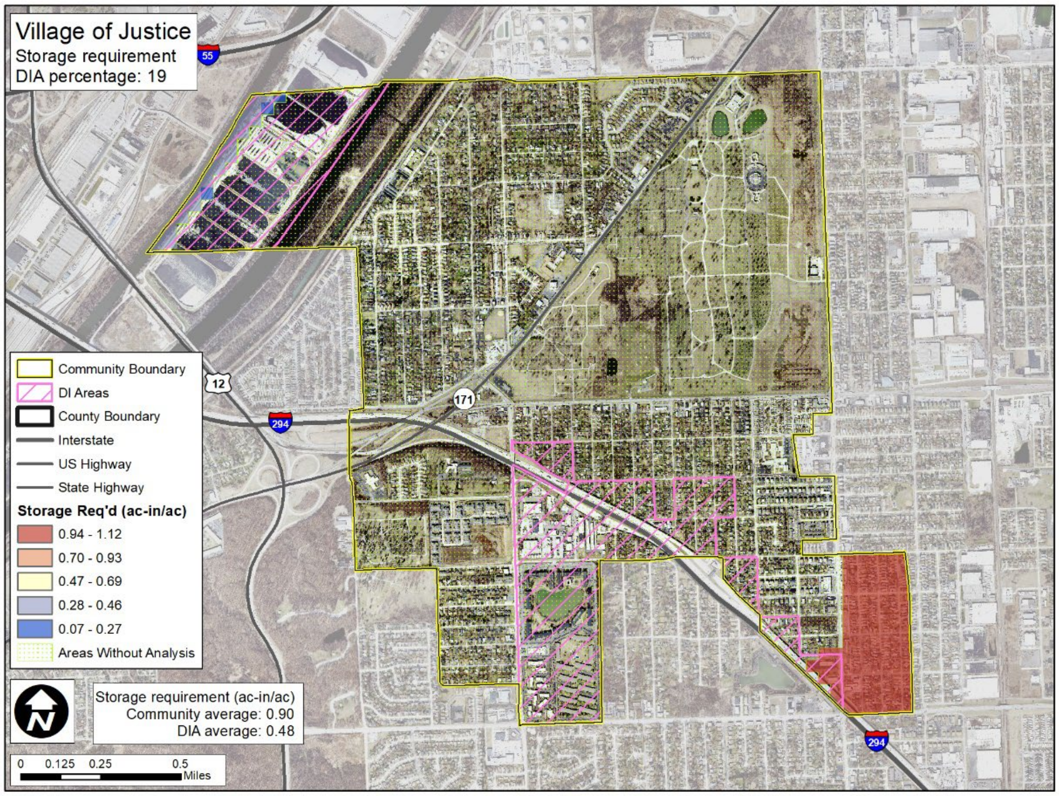

City of Hickory Hills
Storage requirement
DIA percentage: 31
Community Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.94 - 1.12
0.70 - 0.93
0.47 - 0.69
0.28 - 0.46
0.07 - 0.27
Areas Without Analysis
294
12
Storage requirement (ac-in/ac)
Community average: 1.07
DIA average: 1.08
0 0.125 0.25 0.5
Miles

Figure A22



Village of Hodgkins
Storage requirement
DIA percentage: 100
Community Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.92 - 1.07
0.69 - 0.91
0.46 - 0.68
0.27 - 0.45
0.04 - 0.26
Areas Without Analysis
55
12
294
171
N
0 0.15 0.3 0.6
Miles
Storage requirement (ac-in/ac)
Community average: 0.31
DIA average: 0.31

Figure A23



Village of Justice
Storage requirement
DIA percentage: 19
Community Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.94 - 1.12
0.70 - 0.93
0.47 - 0.69
0.28 - 0.46
0.07 - 0.27
Areas Without Analysis
Storage requirement (ac-in/ac)
Community average: 0.90
DIA average: 0.48
0 0.125 0.25 0.5 Miles
55
12
294
171

Figure A24

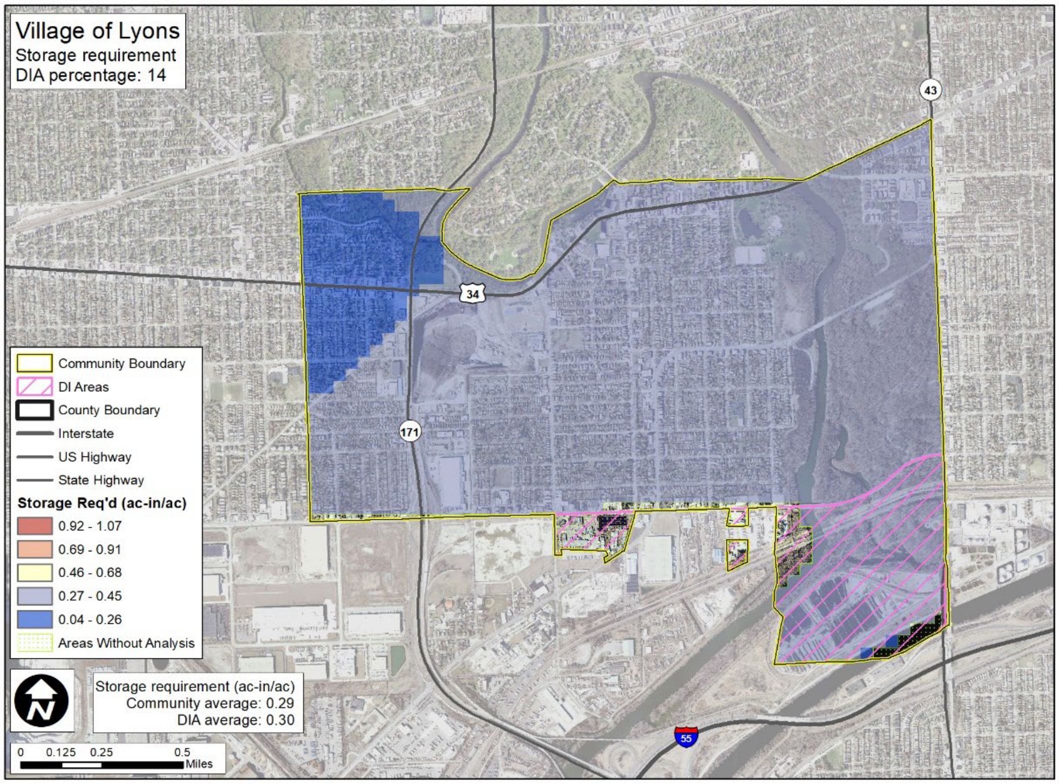


Figure A25

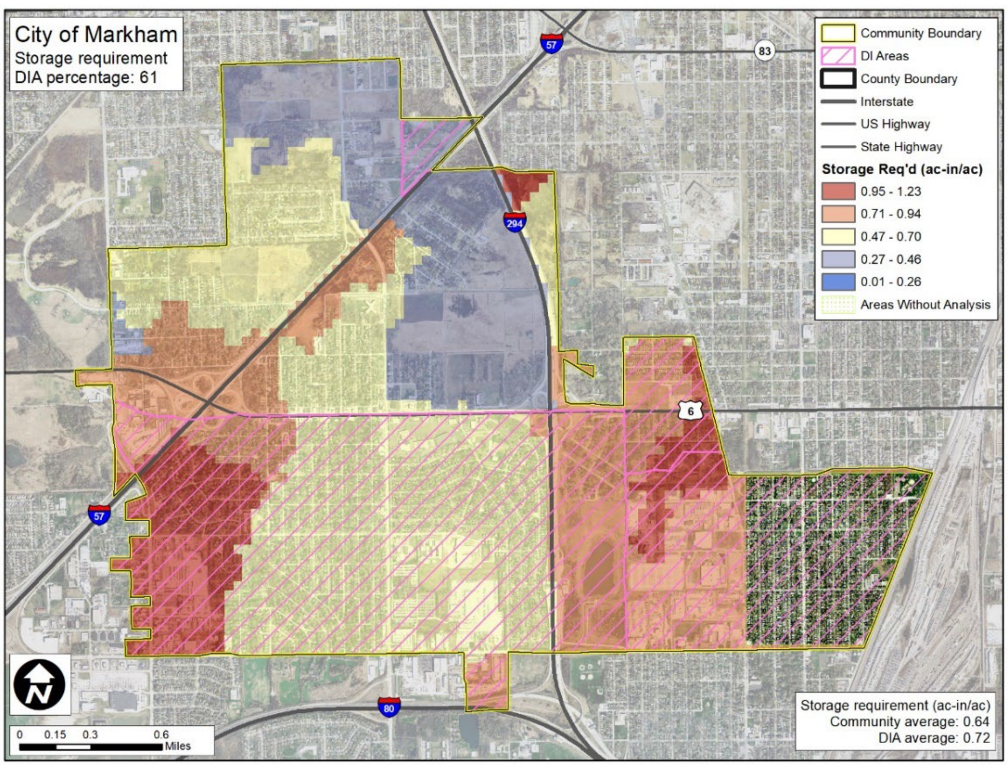


Figure A26

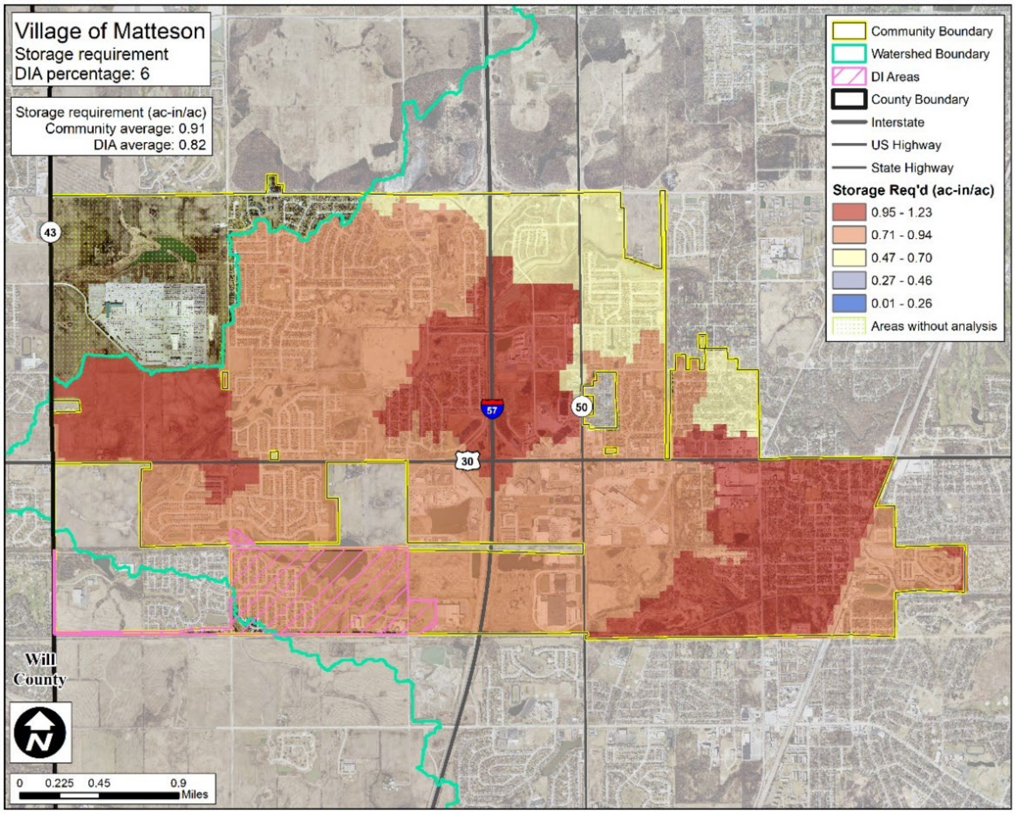

Village of Matteson
Storage requirement
DIA percentage: 6
Storage requirement (ac-in/ac)
Community average: 0.91
DIA average: 0.82
Community Boundary
Watershed Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.95 - 1.23
0.71 - 0.94
0.47 - 0.70
0.27 - 0.46
0.01 - 0.26
Areas without analysis
43
57
50
30
Will County
0 0.225 0.45 0.9 Miles

Figure A27

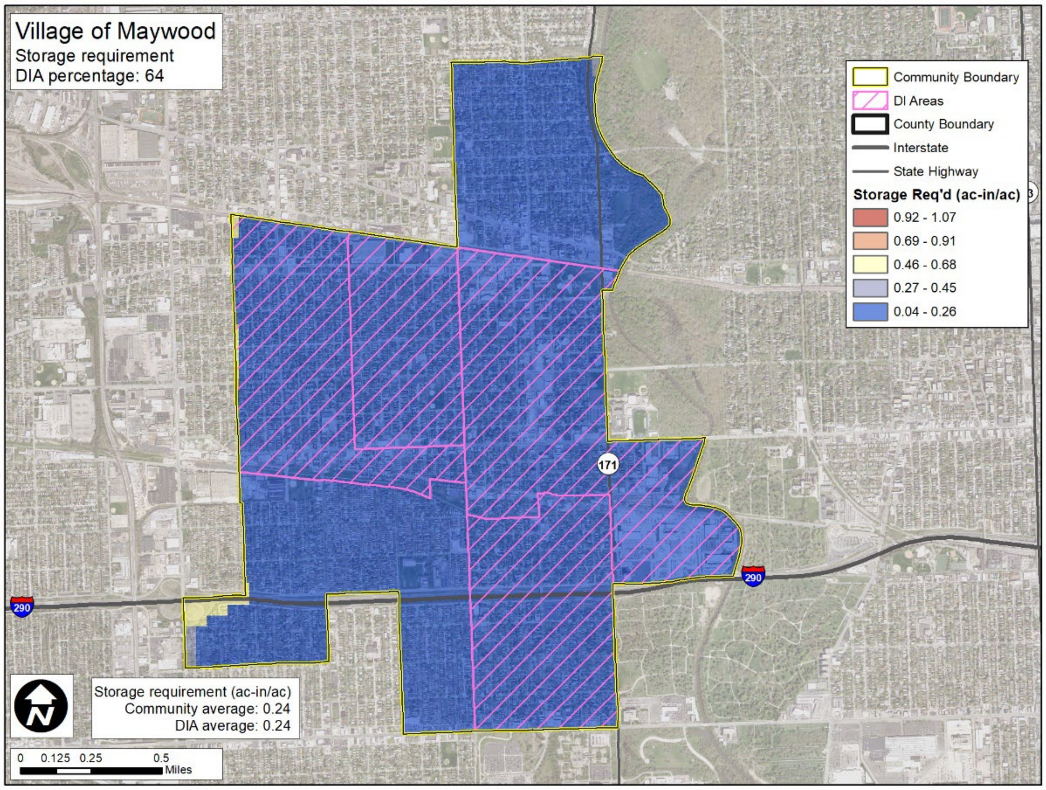

Village of Maywood
Storage requirement
DIA percentage: 64
Community Boundary
DI Areas
County Boundary
Interstate
State Highway
Storage Req'd (ac-in/ac)
0.92 - 1.07
0.69 - 0.91
0.46 - 0.68
0.27 - 0.45
0.04 - 0.26
171
290
290
Storage requirement (ac-in/ac)
Community average: 0.24
DIA average: 0.24
0 0.125 0.25 0.5
Miles

Figure A28

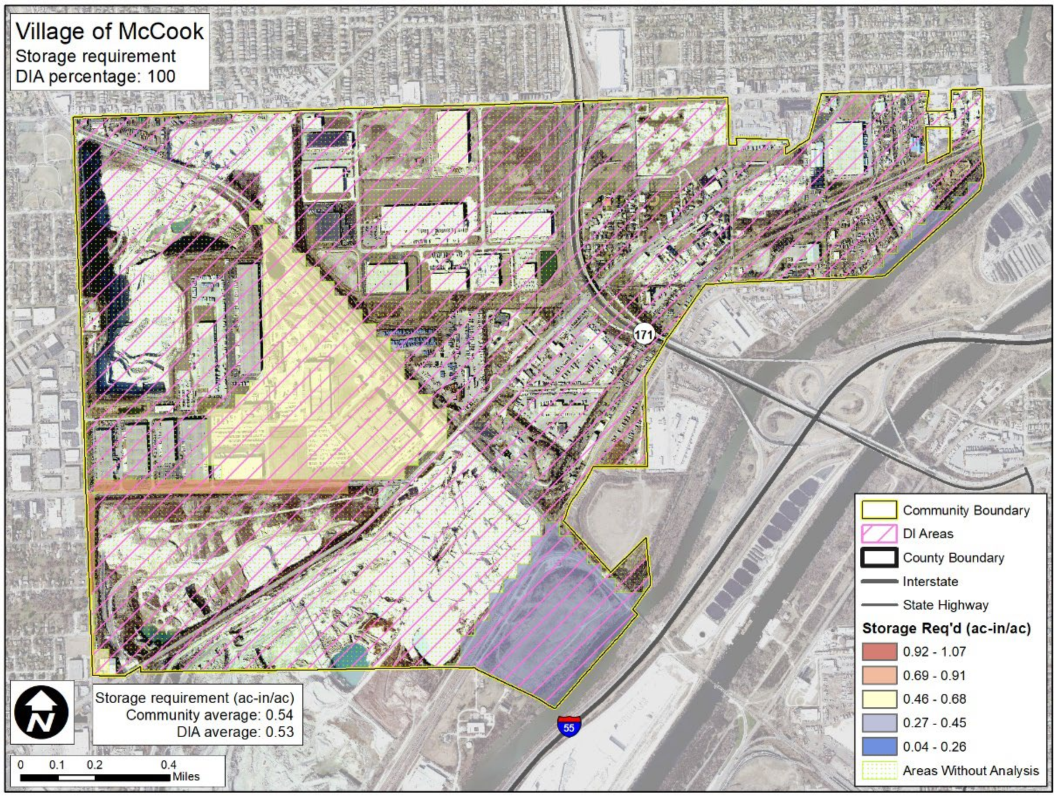
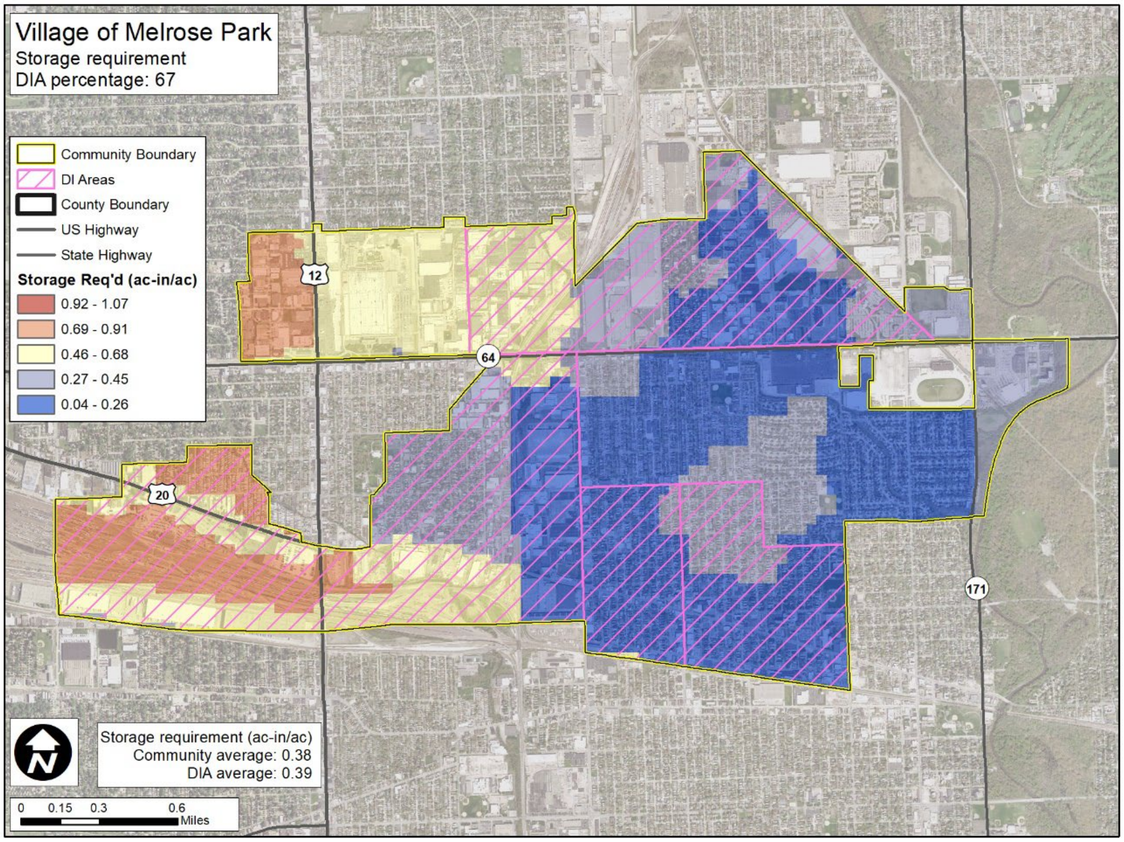
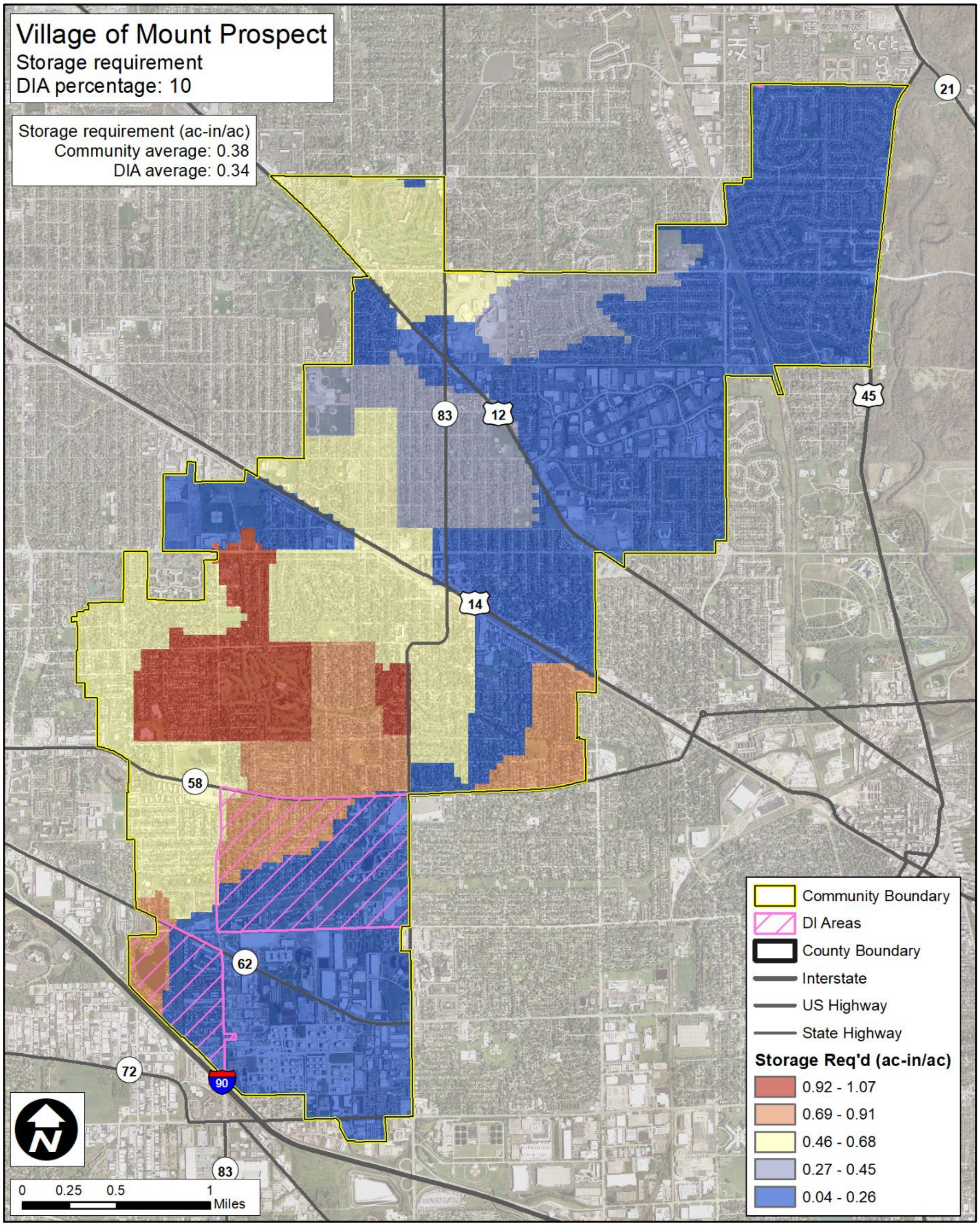

Village of McCook
Storage requirement
DIA percentage: 100
171
55
Community Boundary
DI Areas
County Boundary
Interstate
State Highway
Storage Req'd (ac-in/ac)
0.92 - 1.07
0.69 - 0.91
0.46 - 0.68
0.27 - 0.45
0.04 - 0.26
Areas Without Analysis
Storage requirement (ac-in/ac)
Community average: 0.54
DIA average: 0.53
N
0 0.1 0.2 0.4
Miles

Figure A29



Village of Melrose Park
Storage requirement
DIA percentage: 67
Community Boundary
DI Areas
County Boundary
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.92 - 1.07
0.69 - 0.91
0.46 - 0.68
0.27 - 0.45
0.04 - 0.26
12
64
20
171
Storage requirement (ac-in/ac)
Community average: 0.38
DIA average: 0.39
0 0.15 0.3 0.6
Miles

Figure A30



Village of Mount Prospect
Storage requirement
DIA percentage: 10
Storage requirement (ac-in/ac)
Community average: 0.38
DIA average: 0.34
Community Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.92 - 1.07
0.69 - 0.91
0.46 - 0.68
0.27 - 0.45
0.04 - 0.26
0 0.25 0.5 1 Miles
21
45
83
12
14
58
62
72
90
83

Figure A31

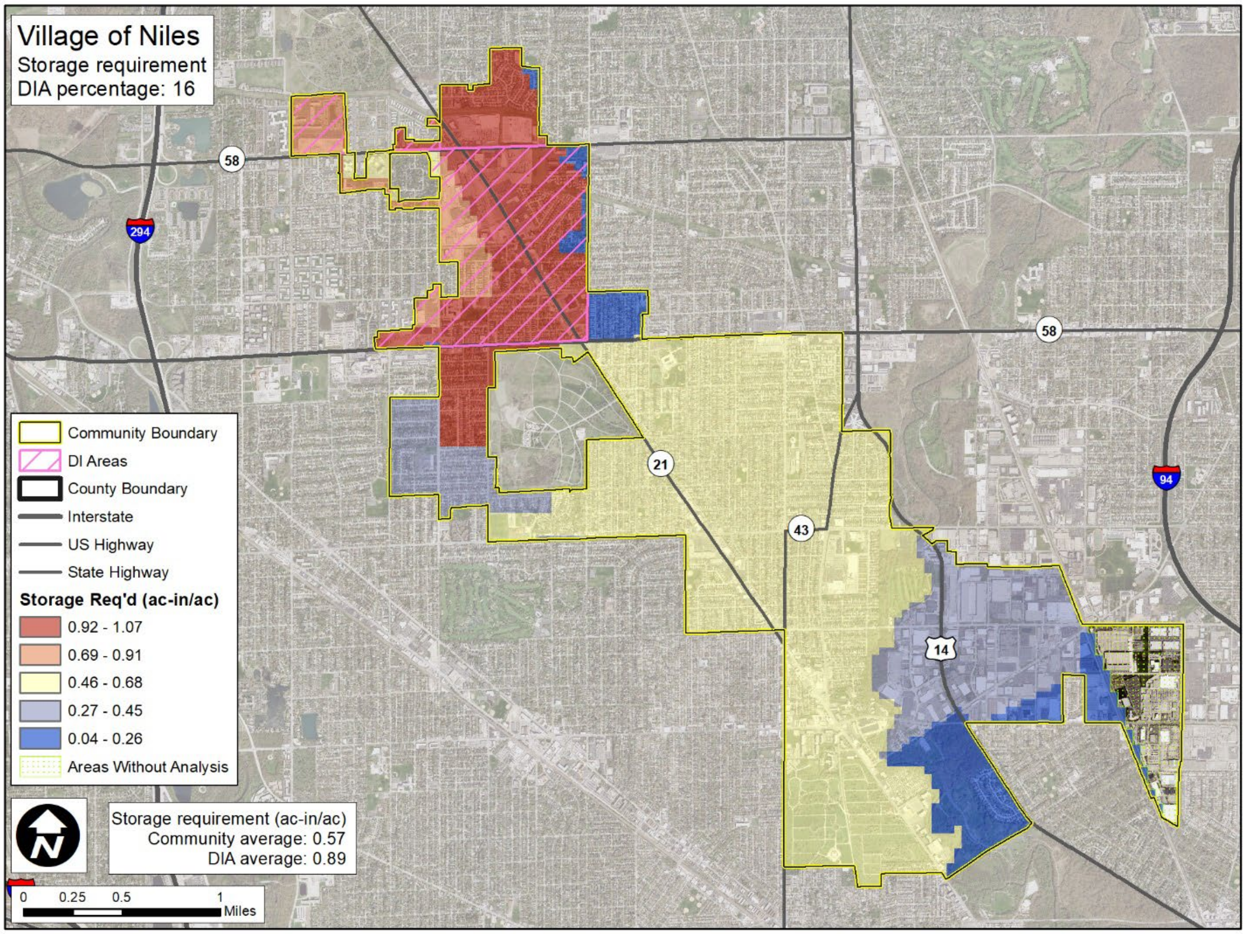
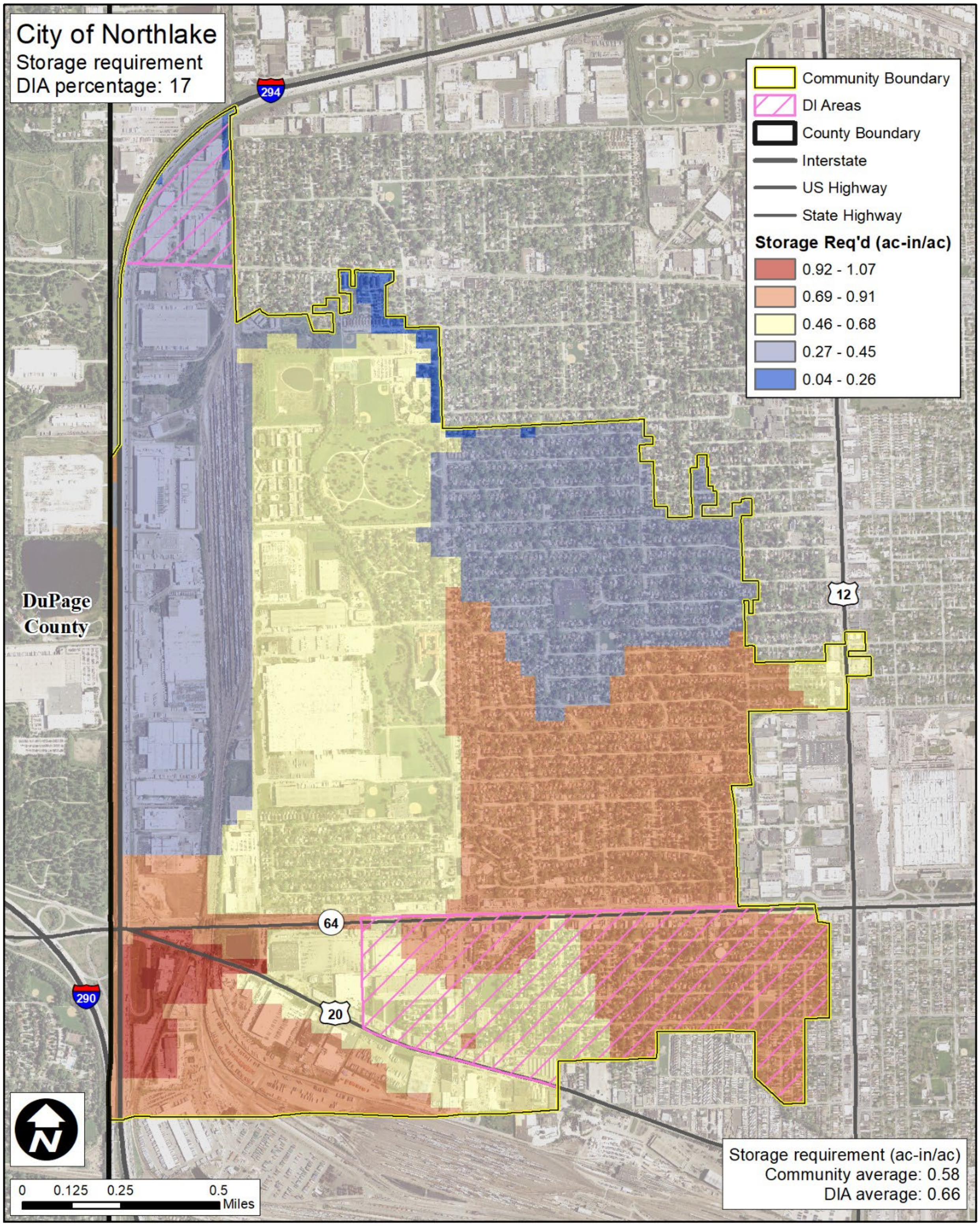

Village of Niles
Storage requirement
DIA percentage: 16
Community Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.92 - 1.07
0.69 - 0.91
0.46 - 0.68
0.27 - 0.45
0.04 - 0.26
Areas Without Analysis
Storage requirement (ac-in/ac)
Community average: 0.57
DIA average: 0.89
0 0.25 0.5 1
Miles
58
294
21
43
14
94

Figure A32



City of Northlake
Storage requirement
DIA percentage: 17
Community Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.92 - 1.07
0.69 - 0.91
0.46 - 0.68
0.27 - 0.45
0.04 - 0.26
294
290
64
20
12
DuPage County
N
0 0.125 0.25 0.5 Miles
Storage requirement (ac-in/ac)
Community average: 0.58
DIA average: 0.66

Figure A33

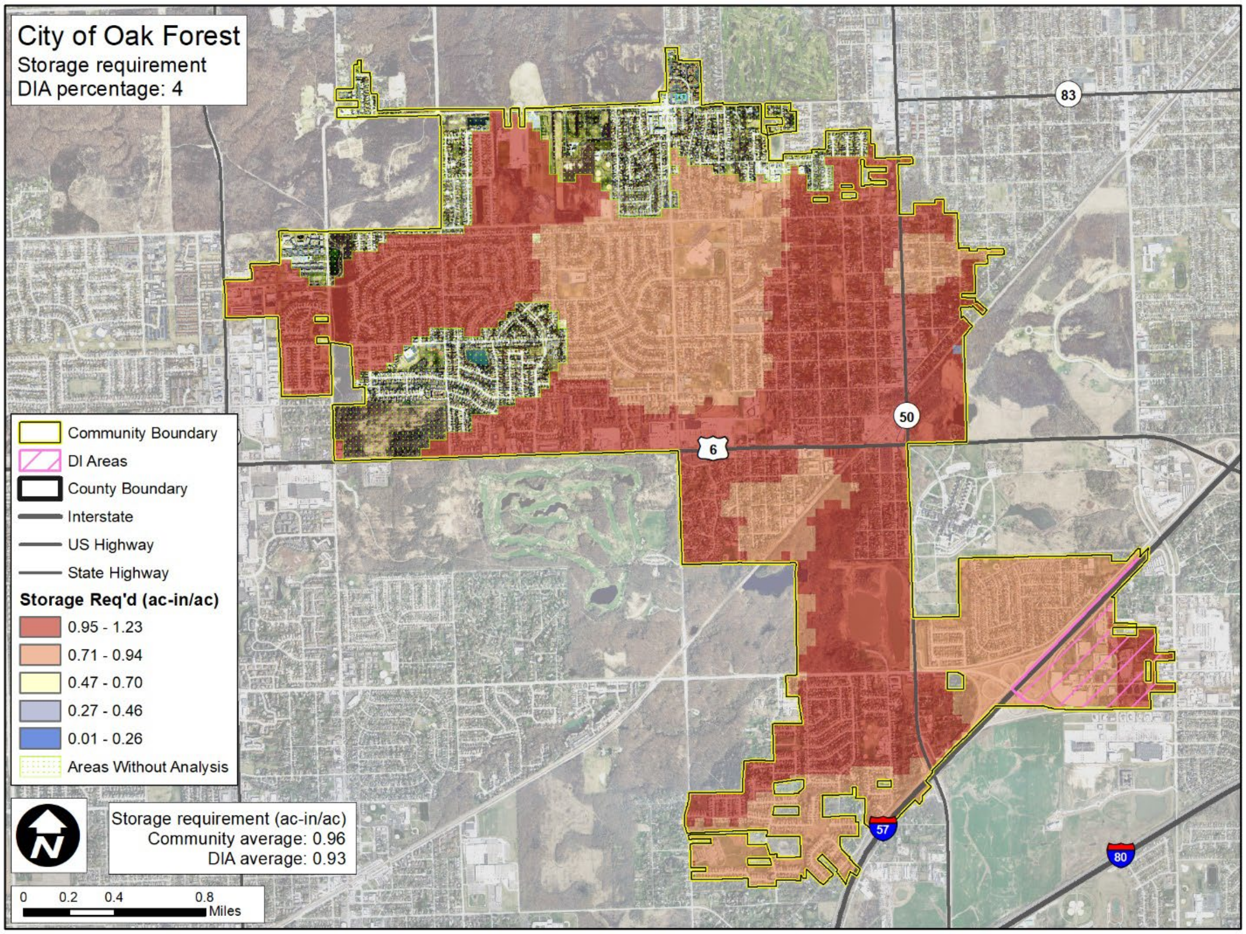

City of Oak Forest
Storage requirement
DIA percentage: 4
Community Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.95 - 1.23
0.71 - 0.94
0.47 - 0.70
0.27 - 0.46
0.01 - 0.26
Areas Without Analysis
Storage requirement (ac-in/ac)
Community average: 0.96
DIA average: 0.93
0 0.2 0.4 0.8
Miles
83
50
6
57
80

Figure A34

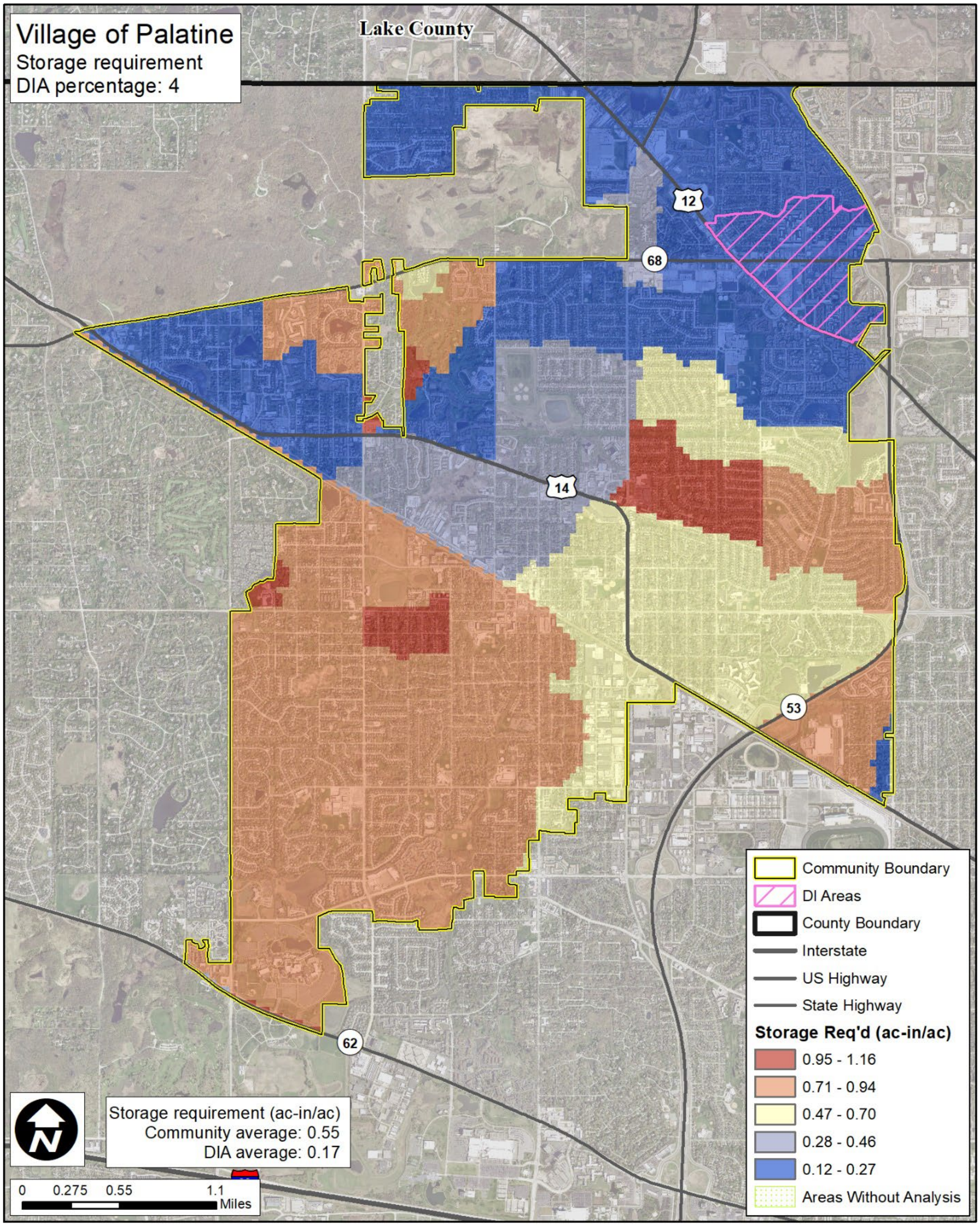

Village of Palatine
Storage requirement
DIA percentage: 4
Lake County
12
68
14
53
62
Community Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.95 - 1.16
0.71 - 0.94
0.47 - 0.70
0.28 - 0.46
0.12 - 0.27
Areas Without Analysis
Storage requirement (ac-in/ac)
Community average: 0.55
DIA average: 0.17
0 0.275 0.55 1.1
Miles

Figure A35

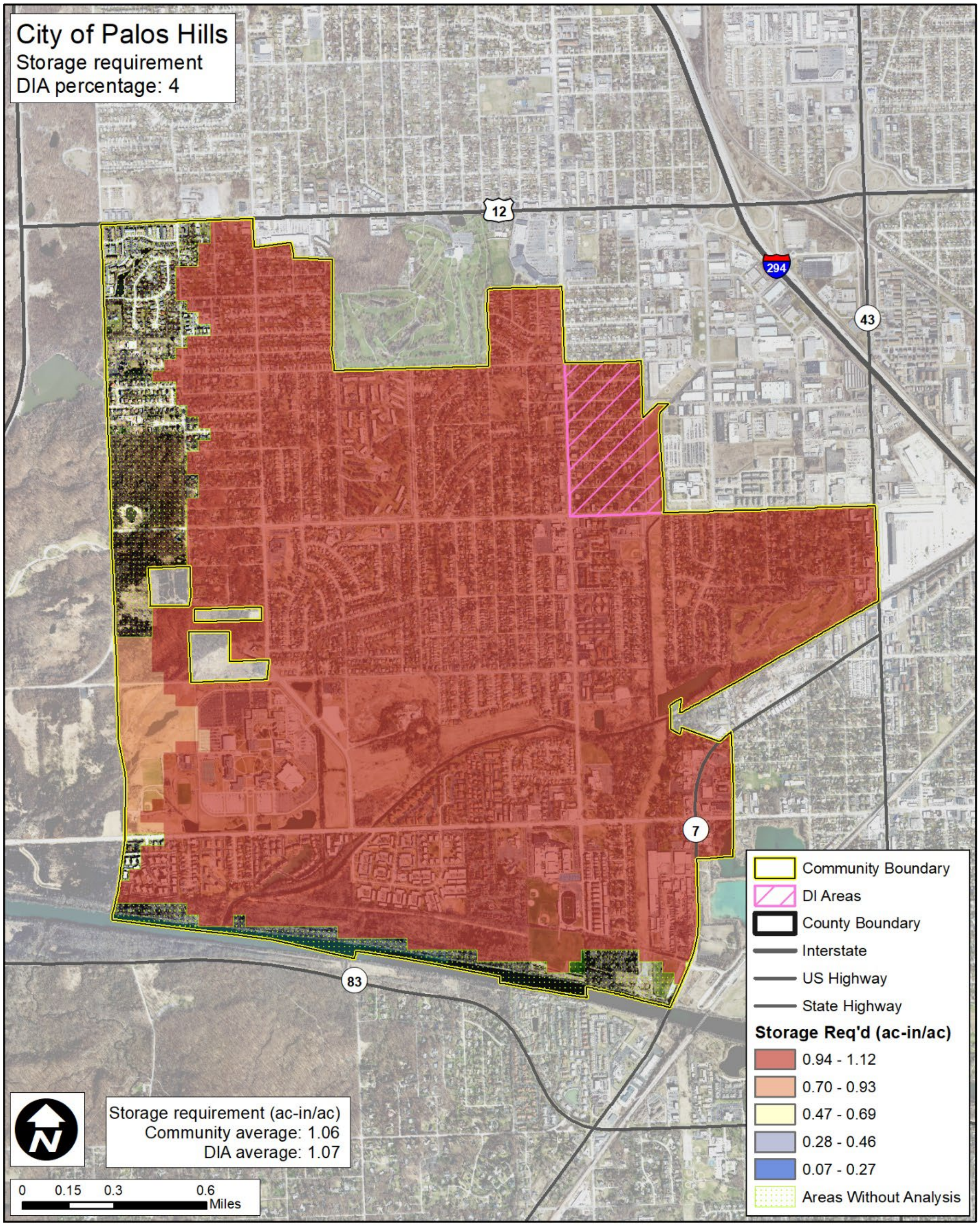

City of Palos Hills
Storage requirement
DIA percentage: 4
12
294
43
7
83
Community Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.94 - 1.12
0.70 - 0.93
0.47 - 0.69
0.28 - 0.46
0.07 - 0.27
Areas Without Analysis
Storage requirement (ac-in/ac)
Community average: 1.06
DIA average: 1.07
0 0.15 0.3 0.6
Miles

Figure A36

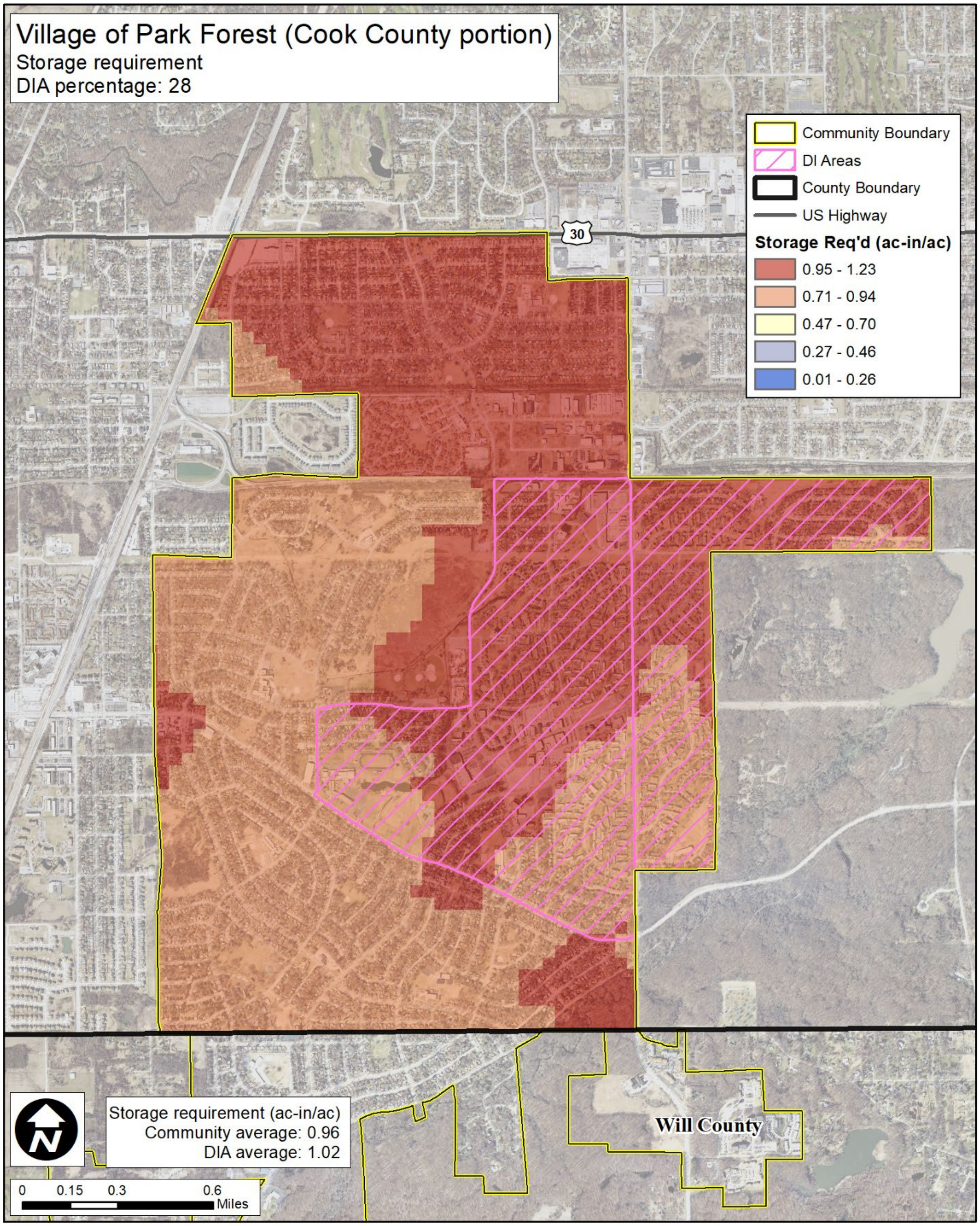

Village of Park Forest (Cook County portion)
Storage requirement
DIA percentage: 28
Community Boundary
DI Areas
County Boundary
US Highway
Storage Req'd (ac-in/ac)
0.95 - 1.23
0.71 - 0.94
0.47 - 0.70
0.27 - 0.46
0.01 - 0.26
30
Will County
Storage requirement (ac-in/ac)
Community average: 0.96
DIA average: 1.02
0
0.15
0.3
0.6
Miles

Figure A37

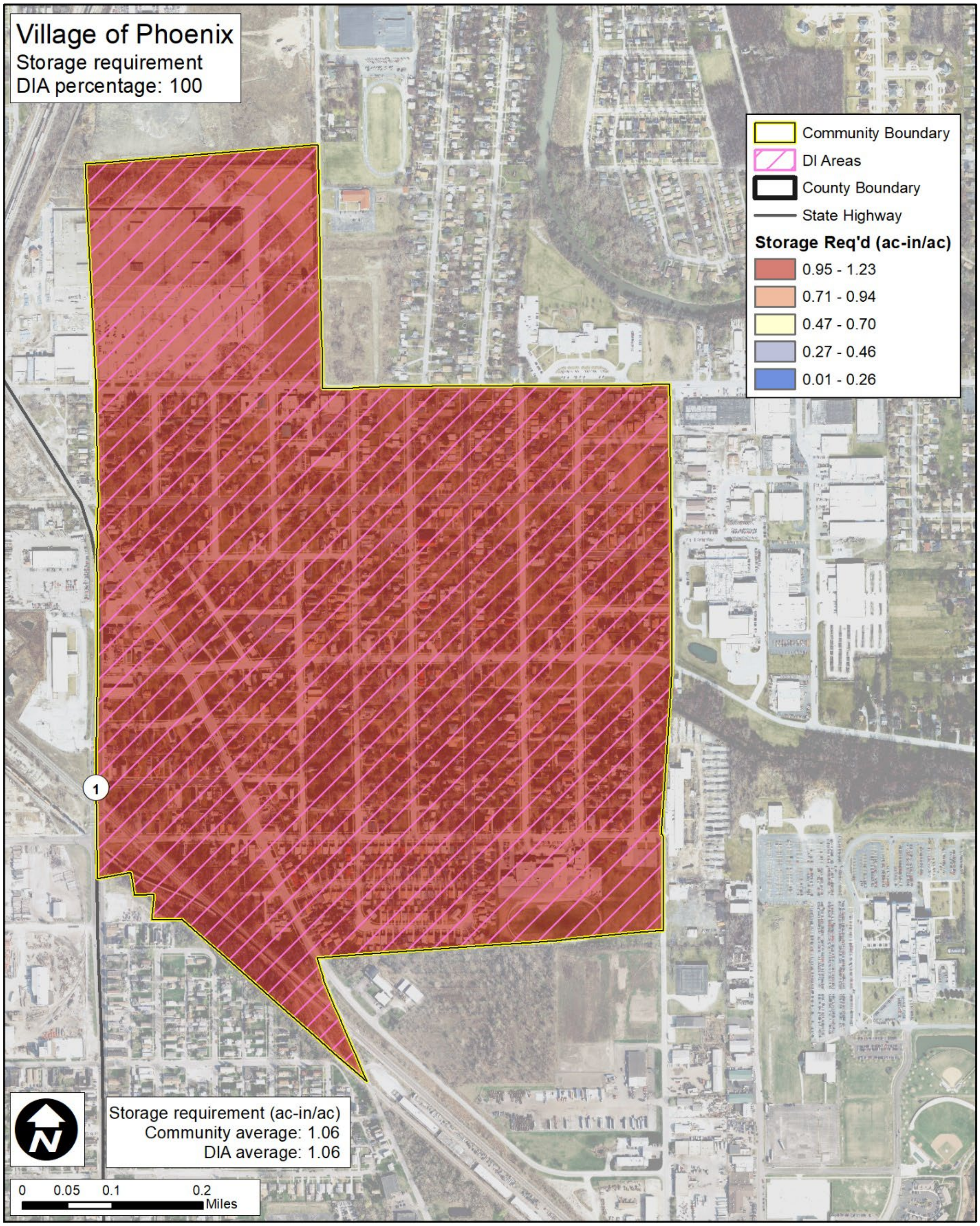
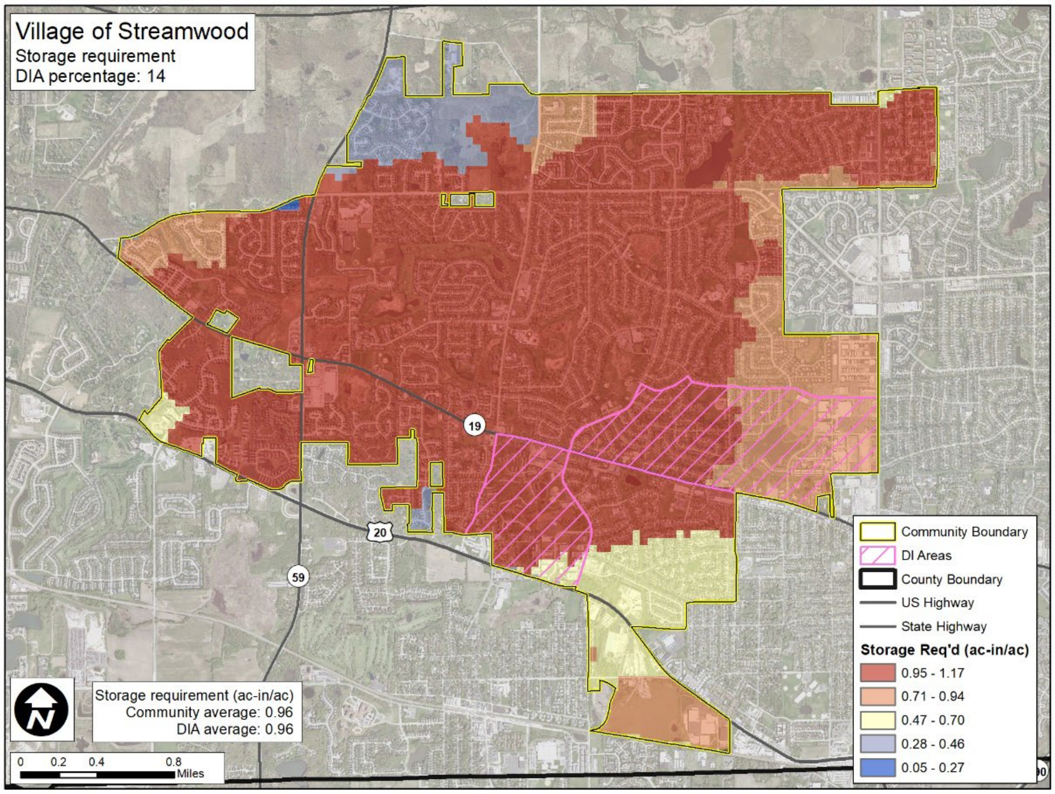

Village of Phoenix
Storage requirement
DIA percentage: 100
Community Boundary
DI Areas
County Boundary
State Highway
Storage Req'd (ac-in/ac)
0.95 - 1.23
0.71 - 0.94
0.47 - 0.70
0.27 - 0.46
0.01 - 0.26
1
Storage requirement (ac-in/ac)
Community average: 1.06
DIA average: 1.06
0 0.05 0.1 0.2
Miles

Figure A38



Village of Streamwood
Storage requirement
DIA percentage: 14
19
20
59
Community Boundary
DI Areas
County Boundary
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.95 - 1.17
0.71 - 0.94
0.47 - 0.70
0.28 - 0.46
0.05 - 0.27
Storage requirement (ac-in/ac)
Community average: 0.96
DIA average: 0.96
0 0.2 0.4 0.8
Miles

Figure A39

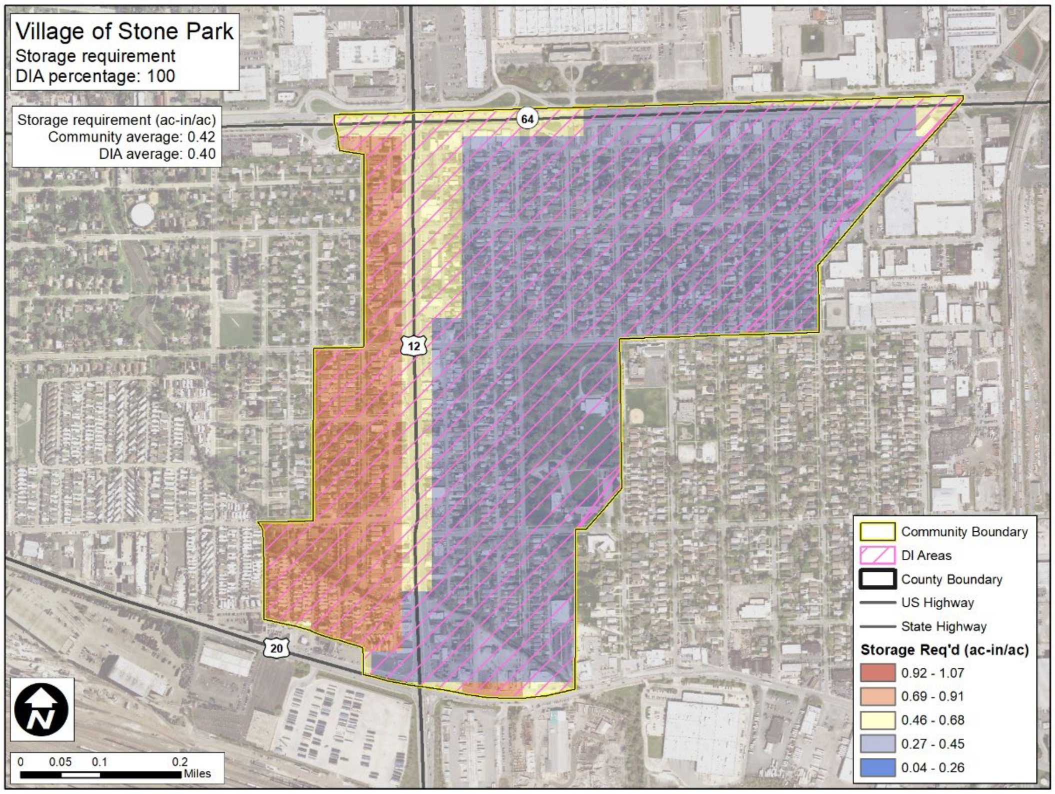

Village of Stone Park
Storage requirement
DIA percentage: 100
Storage requirement (ac-in/ac)
Community average: 0.42
DIA average: 0.40
64
12
20
Community Boundary
DI Areas
County Boundary
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.92 - 1.07
0.69 - 0.91
0.46 - 0.68
0.27 - 0.45
0.04 - 0.26
0 0.05 0.1 0.2
Miles

Figure A40

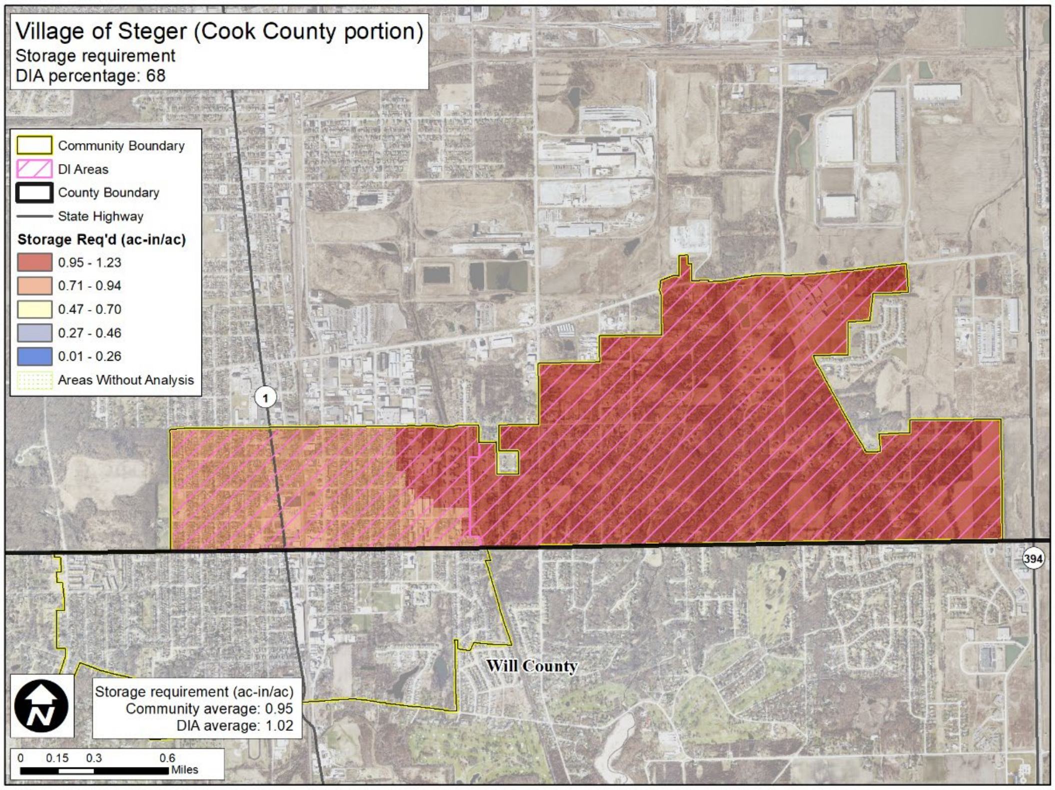

Village of Steger (Cook County portion)
Storage requirement
DIA percentage: 68
Community Boundary
DI Areas
County Boundary
State Highway
Storage Req'd (ac-in/ac)
0.95 - 1.23
0.71 - 0.94
0.47 - 0.70
0.27 - 0.46
0.01 - 0.26
Areas Without Analysis
1
394
Will County
Storage requirement (ac-in/ac)
Community average: 0.95
DIA average: 1.02
0 0.15 0.3 0.6
Miles

Figure A41

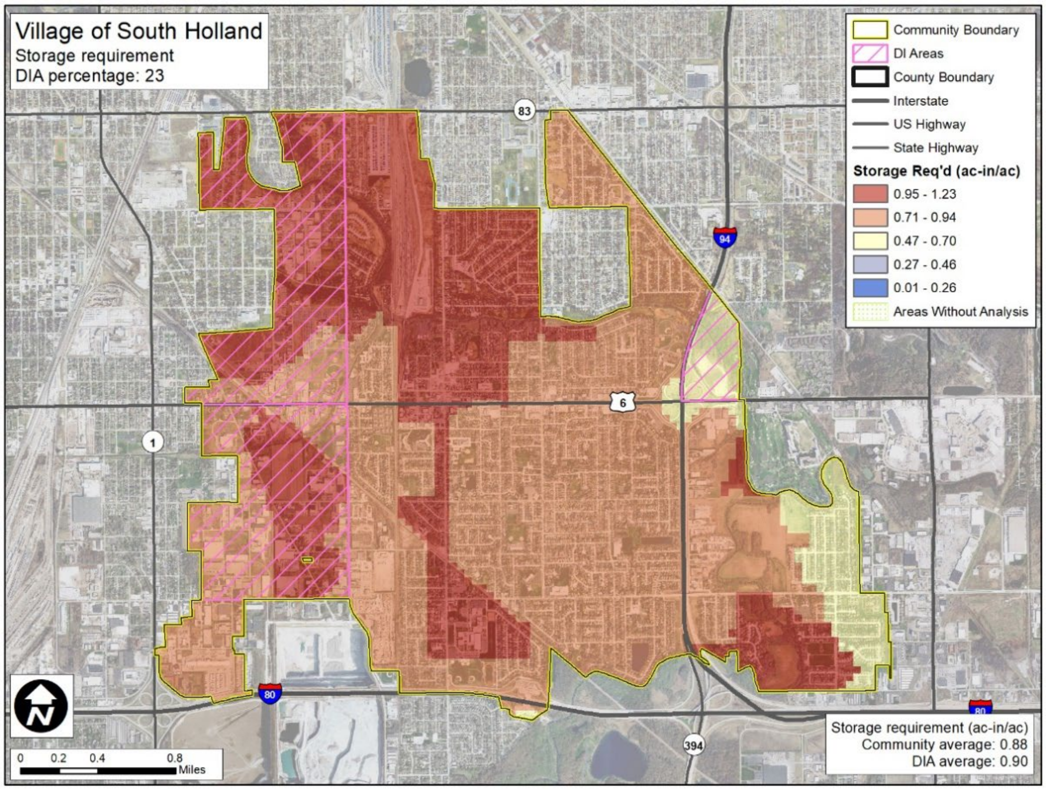

Village of South Holland
Storage requirement
DIA percentage: 23
Community Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.95 - 1.23
0.71 - 0.94
0.47 - 0.70
0.27 - 0.46
0.01 - 0.26
Areas Without Analysis
Storage requirement (ac-in/ac)
Community average: 0.88
DIA average: 0.90
0 0.2 0.4 0.8 Miles
83
94
6
1
80
394

Figure A42

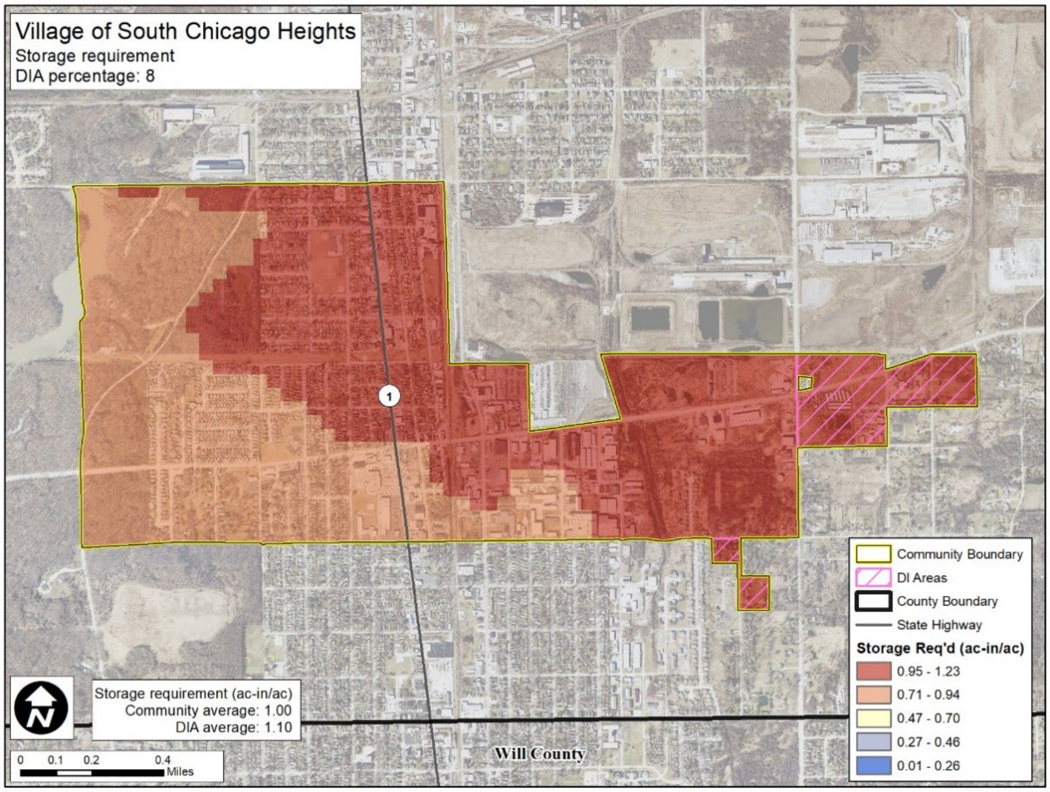

Village of South Chicago Heights
Storage requirement
DIA percentage: 8
1
Community Boundary
DI Areas
County Boundary
State Highway
Storage Req'd (ac-in/ac)
0.95 - 1.23
0.71 - 0.94
0.47 - 0.70
0.27 - 0.46
0.01 - 0.26
Storage requirement (ac-in/ac)
Community average: 1.00
DIA average: 1.10
0 0.1 0.2 0.4
Miles
Will County

Figure A43

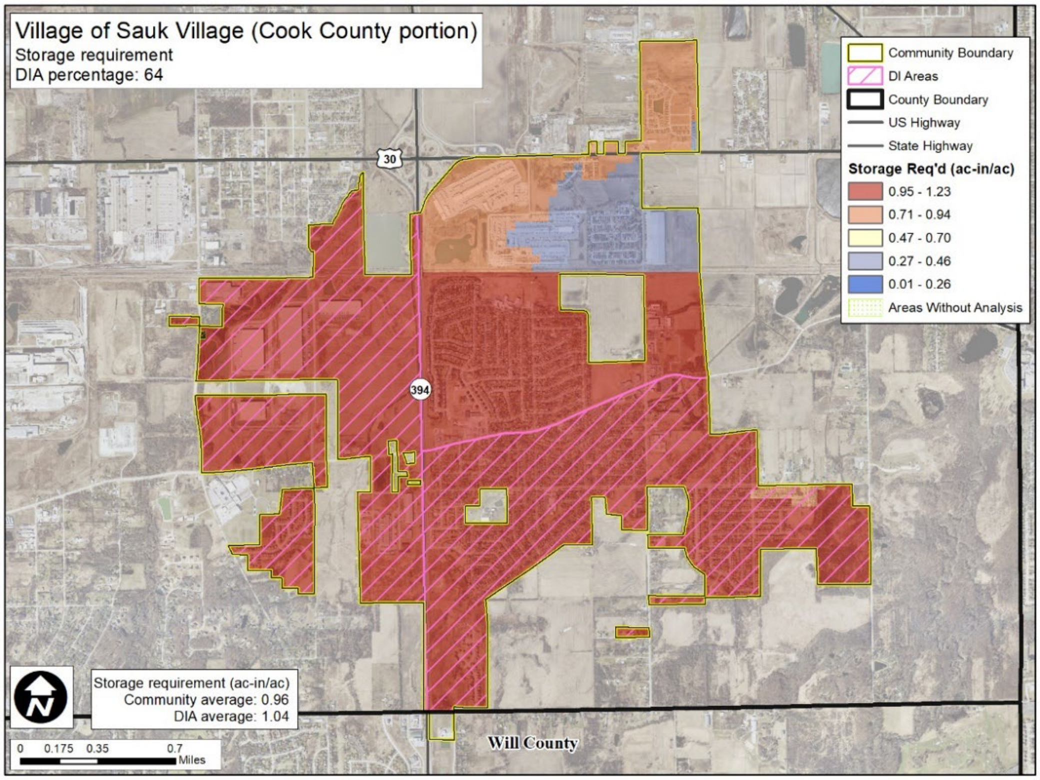
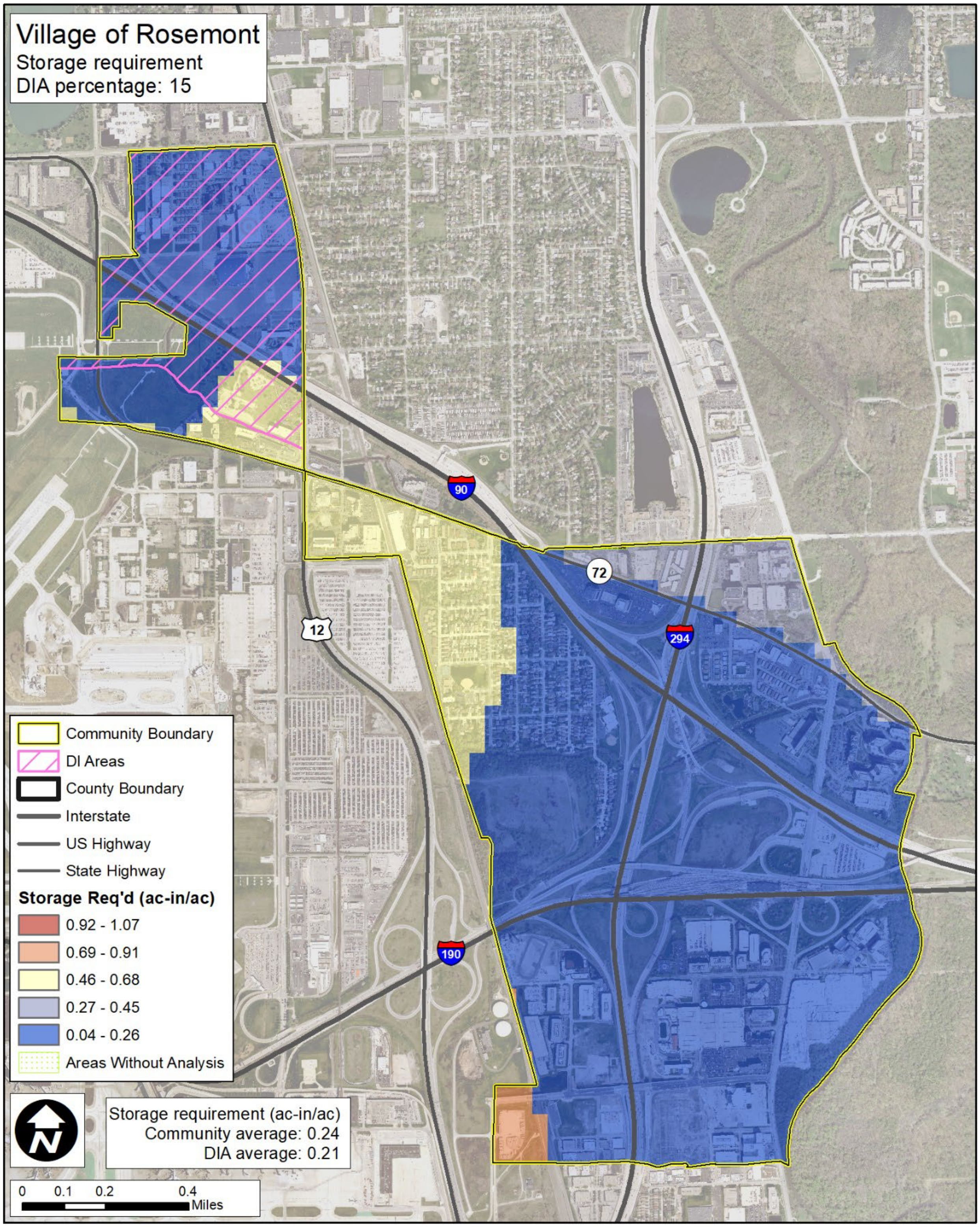
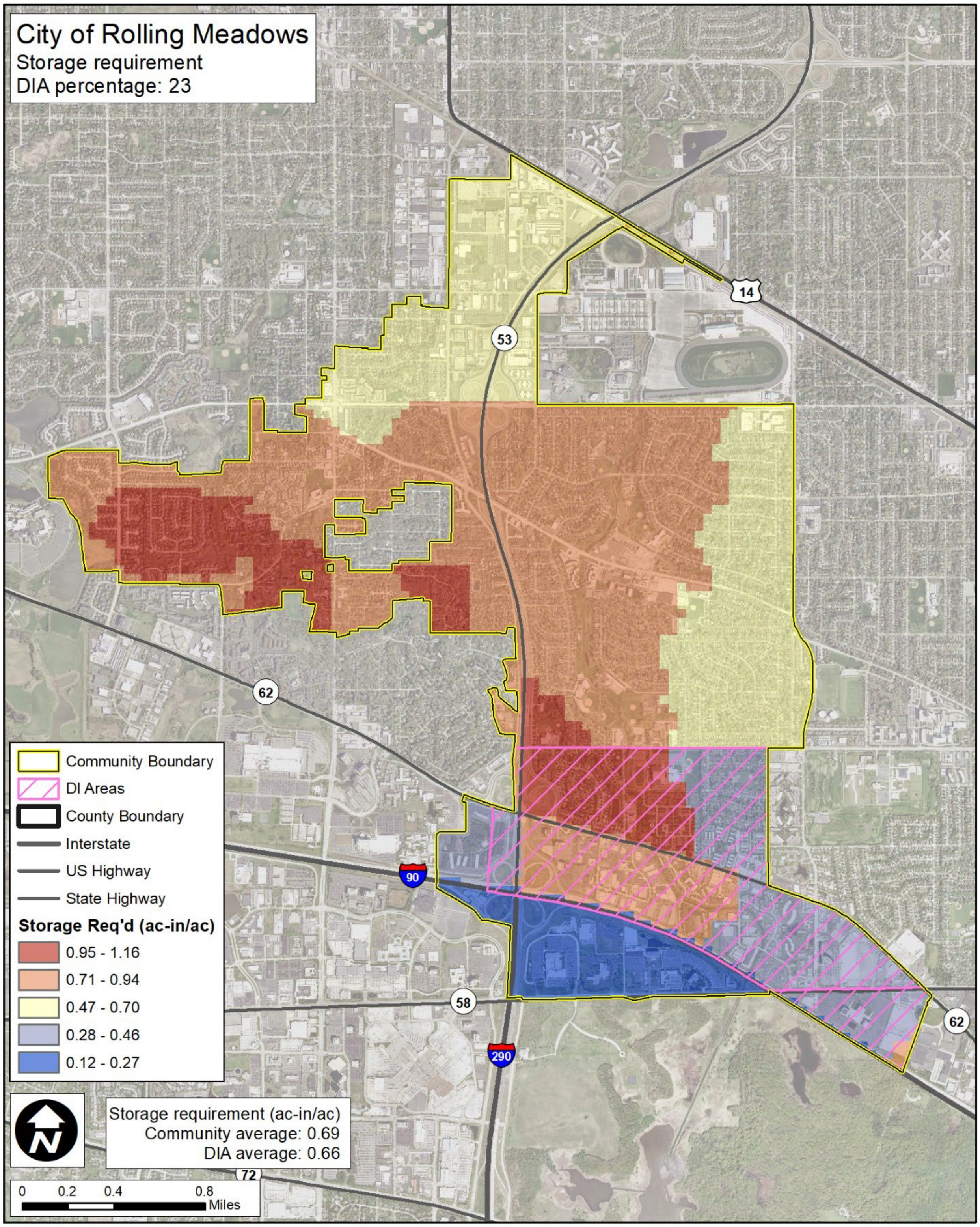

Village of Sauk Village (Cook County portion)
Storage requirement
DIA percentage: 64
Community Boundary
DI Areas
County Boundary
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.95 - 1.23
0.71 - 0.94
0.47 - 0.70
0.27 - 0.46
0.01 - 0.26
Areas Without Analysis
30
394
Storage requirement (ac-in/ac)
Community average: 0.96
DIA average: 1.04
Will County
0 0.175 0.35 0.7
Miles

Figure A44



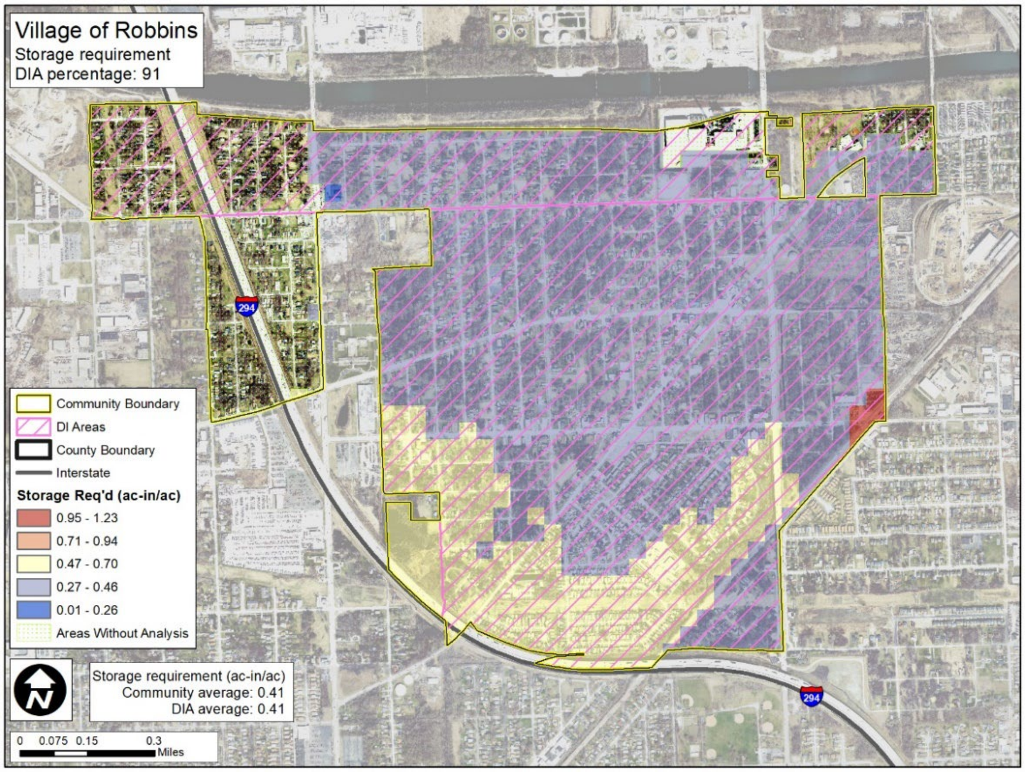
Village of Rosemont
Storage requirement
DIA percentage: 15
Community Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.92 - 1.07
0.69 - 0.91
0.46 - 0.68
0.27 - 0.45
0.04 - 0.26
Areas Without Analysis
Storage requirement (ac-in/ac)
Community average: 0.24
DIA average: 0.21
0 0.1 0.2 0.4
Miles
90
72
294
12
190

Figure A45



City of Rolling Meadows
Storage requirement
DIA percentage: 23
Community Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.95 - 1.16
0.71 - 0.94
0.47 - 0.70
0.28 - 0.46
0.12 - 0.27
Storage requirement (ac-in/ac)
Community average: 0.69
DIA average: 0.66
0 0.2 0.4 0.8
Miles
53
14
62
90
58
290
72

Figure A46



Village of Robbins
Storage requirement
DIA percentage: 91
Community Boundary
DI Areas
County Boundary
Interstate
Storage Req'd (ac-in/ac)
0.95 - 1.23
0.71 - 0.94
0.47 - 0.70
0.27 - 0.46
0.01 - 0.26
Areas Without Analysis
Storage requirement (ac-in/ac)
Community average: 0.41
DIA average: 0.41
0 0.075 0.15 0.3
Miles
294
294

Figure A47

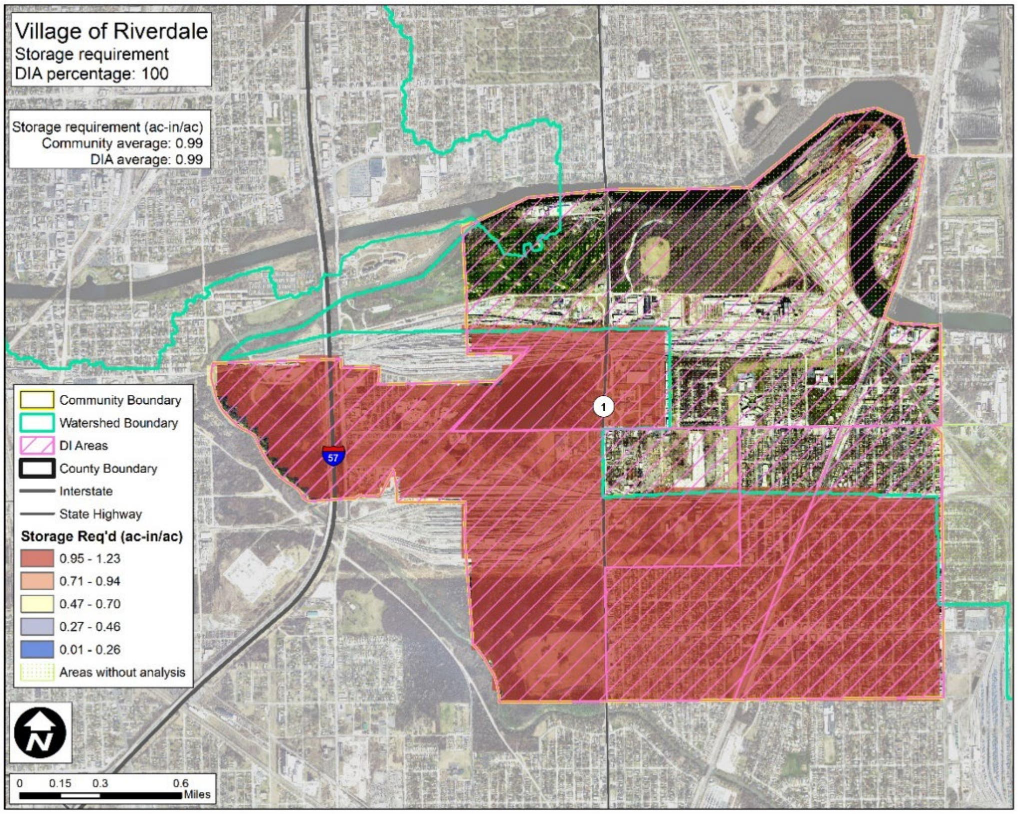

Village of Riverdale
Storage requirement
DIA percentage: 100
Storage requirement (ac-in/ac)
Community average: 0.99
DIA average: 0.99
Community Boundary
Watershed Boundary
DI Areas
County Boundary
Interstate
State Highway
Storage Req'd (ac-in/ac)
0.95 - 1.23
0.71 - 0.94
0.47 - 0.70
0.27 - 0.46
0.01 - 0.26
Areas without analysis
0 0.15 0.3 0.6
Miles

Figure A48

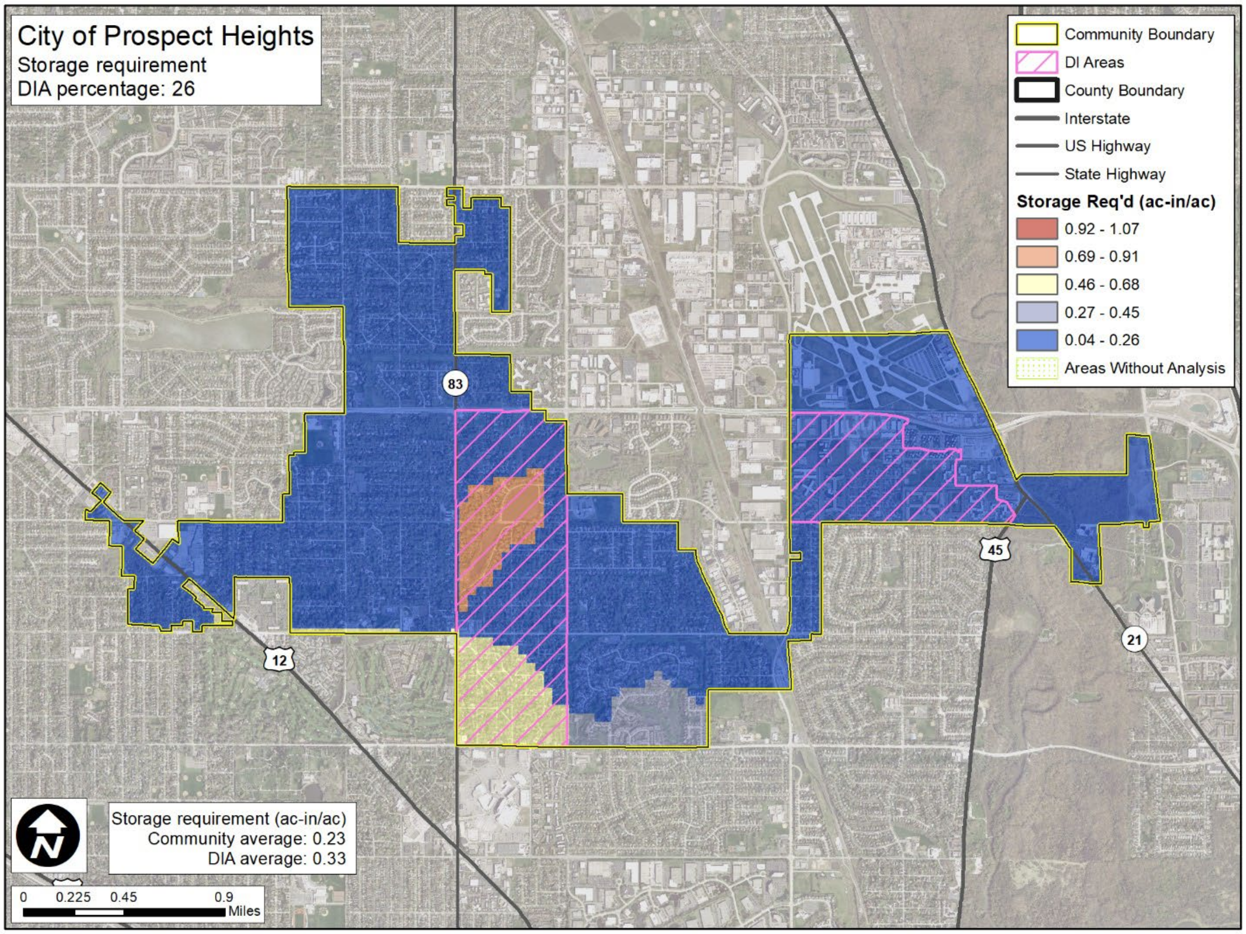
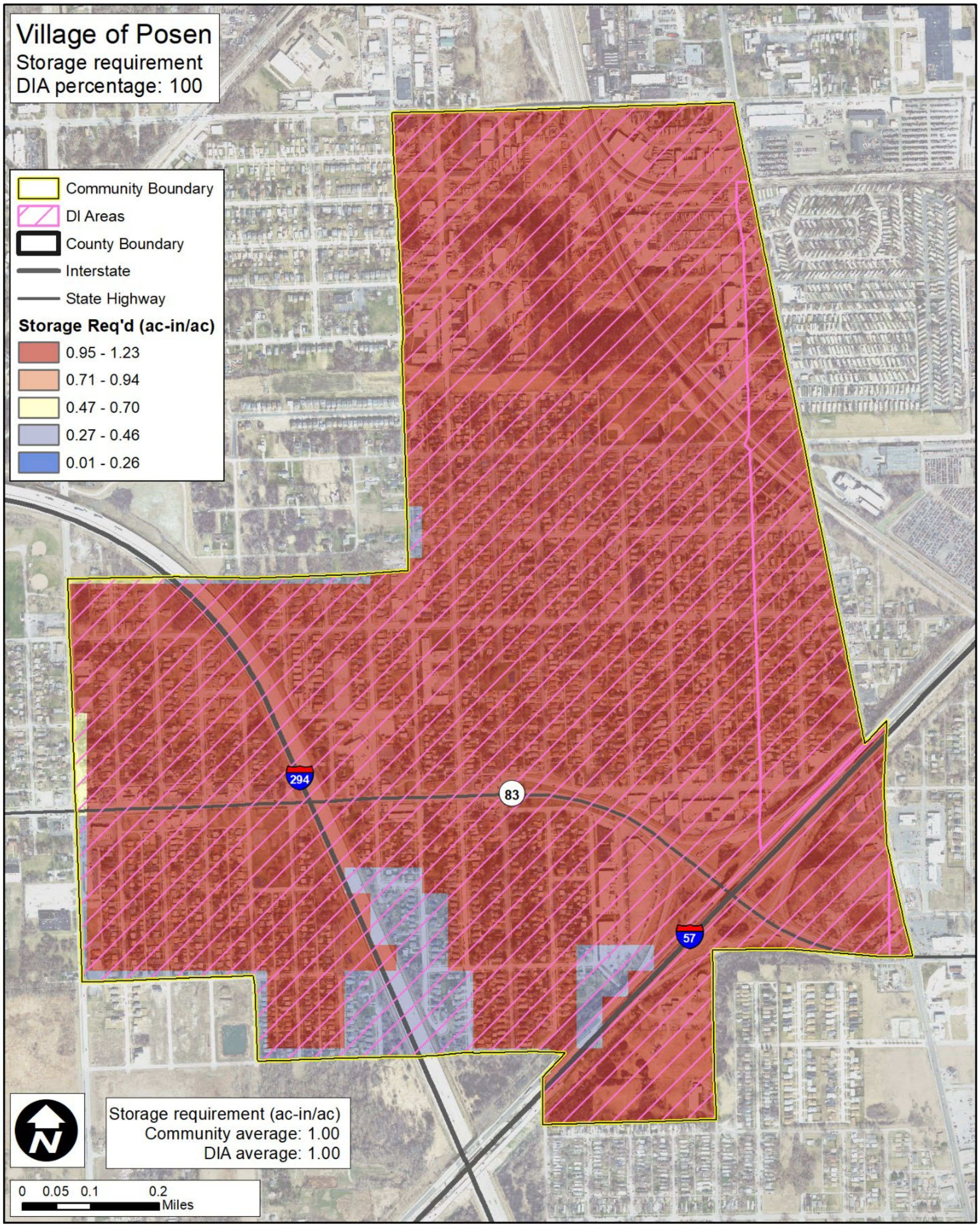

City of Prospect Heights
Storage requirement
DIA percentage: 26
Community Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.92 - 1.07
0.69 - 0.91
0.46 - 0.68
0.27 - 0.45
0.04 - 0.26
Areas Without Analysis
83
12
45
21
Storage requirement (ac-in/ac)
Community average: 0.23
DIA average: 0.33
0 0.225 0.45 0.9
Miles

Figure A49

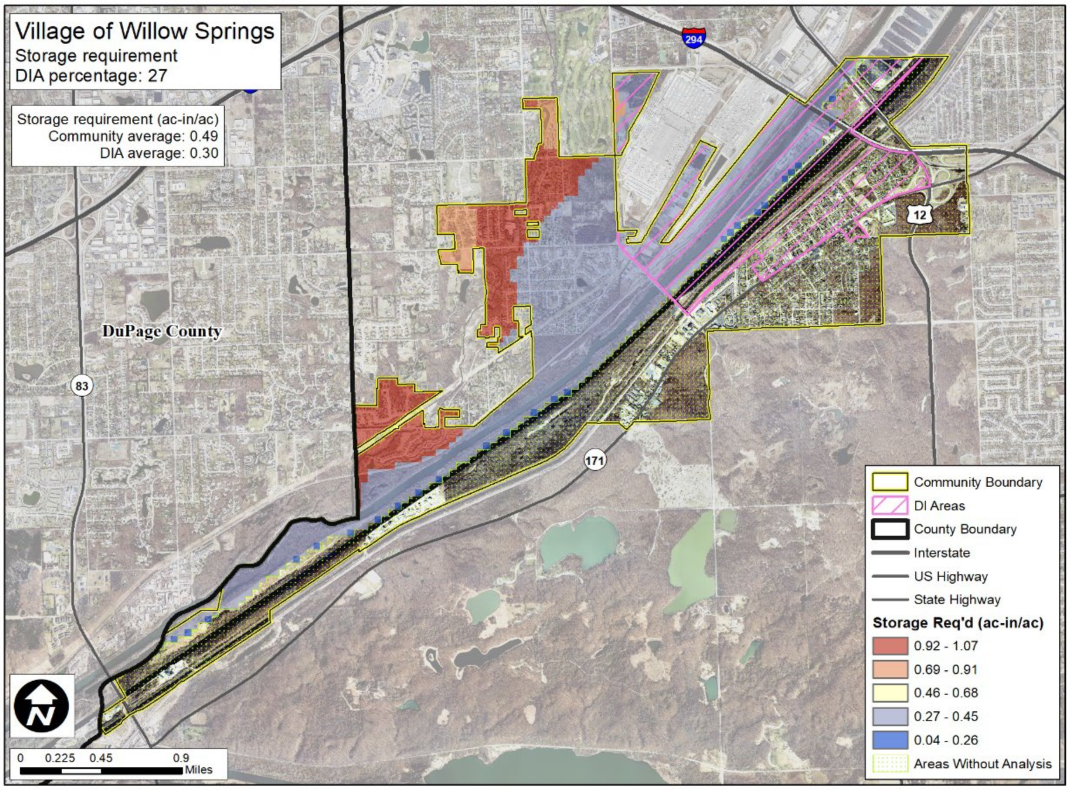


Village of Posen
Storage requirement
DIA percentage: 100
Community Boundary
DI Areas
County Boundary
Interstate
State Highway
Storage Req'd (ac-in/ac)
0.95 - 1.23
0.71 - 0.94
0.47 - 0.70
0.27 - 0.46
0.01 - 0.26
294
83
57
N
Storage requirement (ac-in/ac)
Community average: 1.00
DIA average: 1.00
0 0.05 0.1 0.2
Miles

Figure A50

Village of Summit
Storage requirement
DIA percentage: 74
Community Boundary
DI Areas
County Boundary
Interstate
State Highway
Storage Req'd (ac-in/ac)
0.92 - 1.07
0.69 - 0.91
0.46 - 0.68
0.27 - 0.45
0.04 - 0.26
Areas Without Analysis
Storage requirement (ac-in/ac)
Community average: 0.29
DIA average: 0.29
0 0.125 0.25 0.5 Miles
55
171
43

Figure A51



Village of Willow Springs
Storage requirement
DIA percentage: 27
Storage requirement (ac-in/ac)
Community average: 0.49
DIA average: 0.30
DuPage County
294
83
12
171
Community Boundary
DI Areas
County Boundary
Interstate
US Highway
State Highway
Storage Req'd (ac-in/ac)
0.92 - 1.07
0.69 - 0.91
0.46 - 0.68
0.27 - 0.45
0.04 - 0.26
Areas Without Analysis
0 0.225 0.45 0.9
Miles

Figure A52

Appendix B. Map exhibits of flood mitigation levels in various study area communities

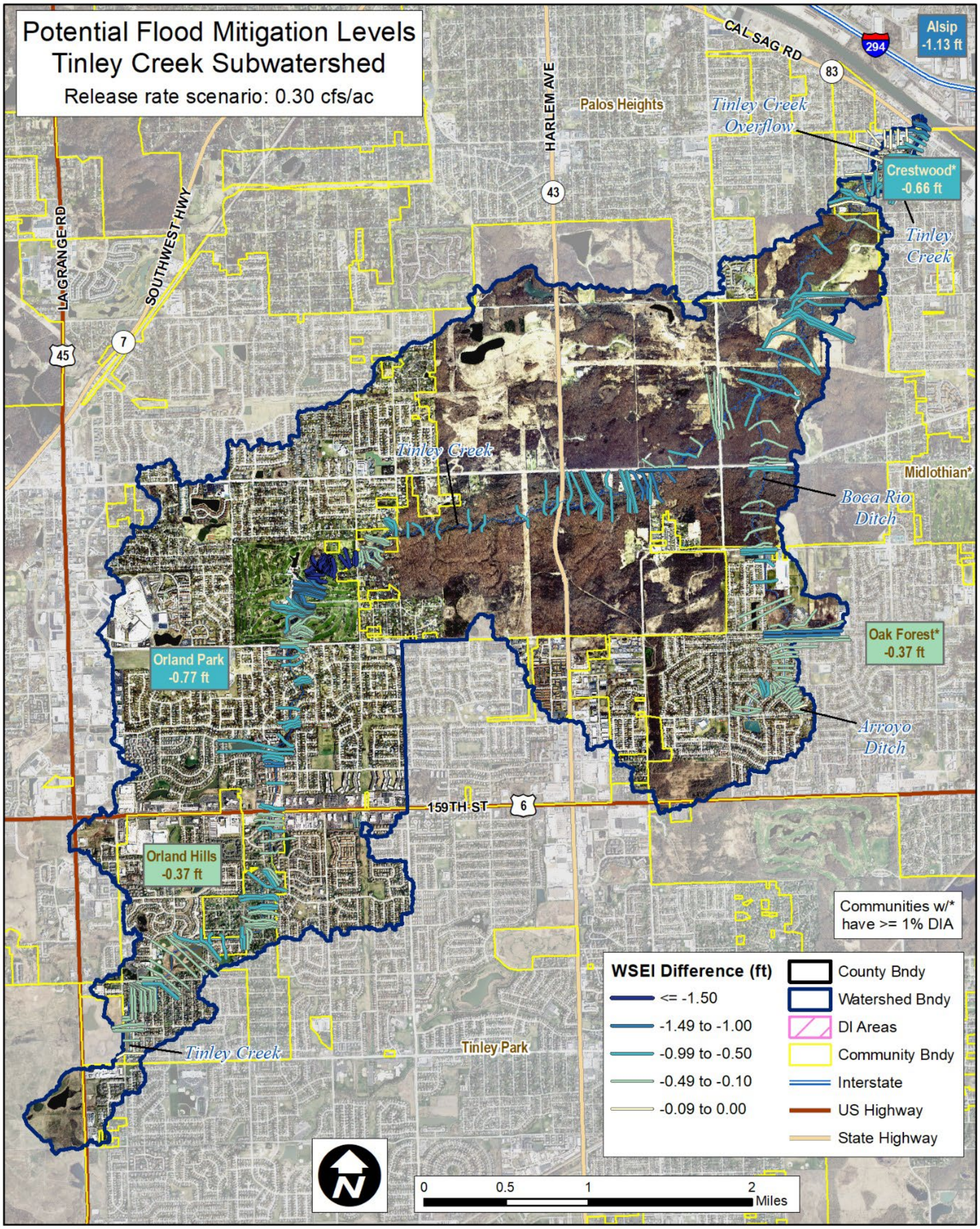


Figure B1: Flood mitigation levels in Tinley Creek subwatershed (Cal Sag watershed) communities at WMO specified release rate, i.e., 0.30 cfs/ac

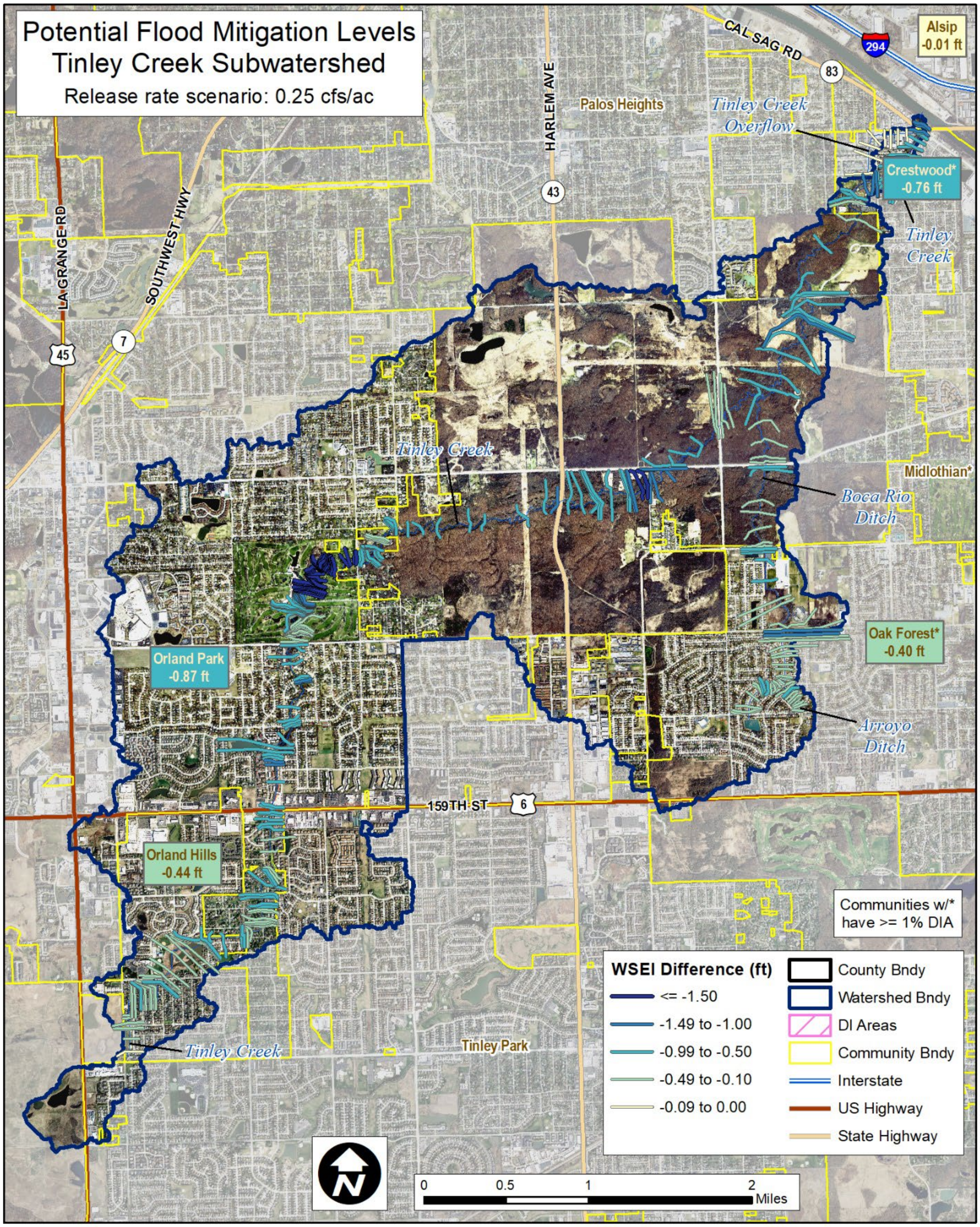


Figure B2: Flood mitigation levels in Tinley Creek subwatershed (Cal Sag watershed) communities at release rate = 0.25 cfs/ac

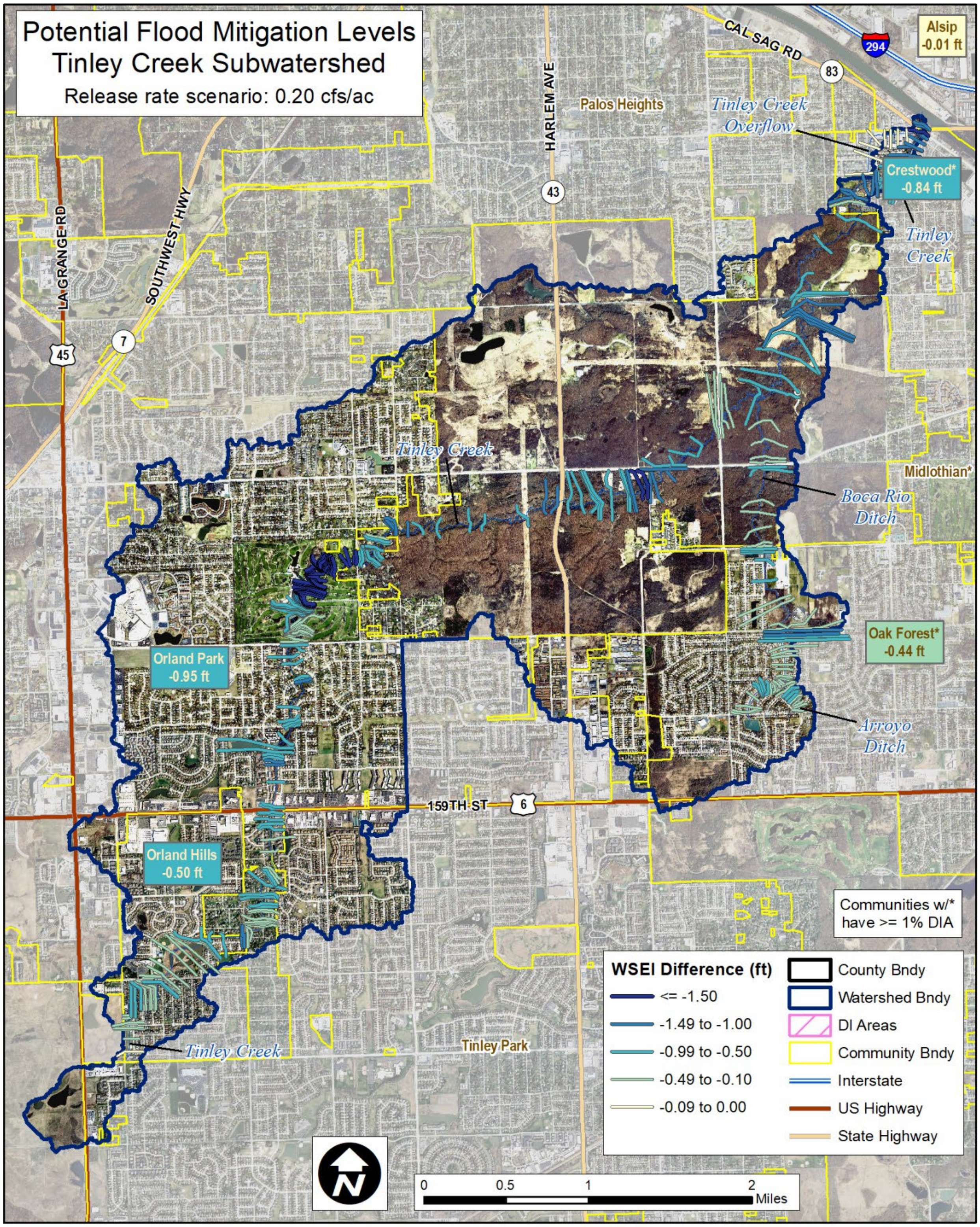


Figure B3: Flood mitigation levels in Tinley Creek subwatershed (Cal Sag watershed) communities at release rate = 0.20 cfs/ac

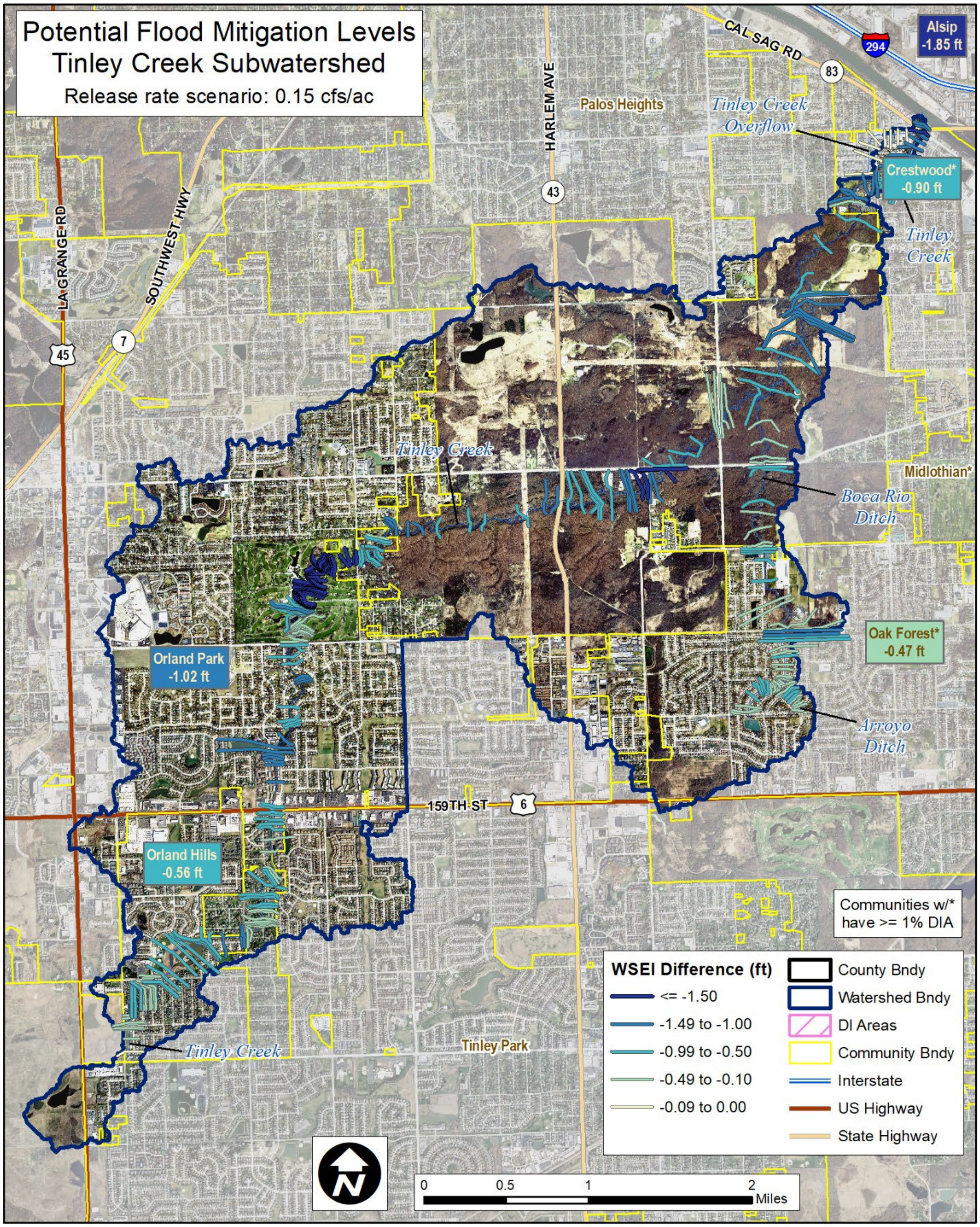


Figure B4: Flood mitigation levels in Tinley Creek subwatershed (Cal Sag watershed) communities at release rate = 0.15 cfs/ac

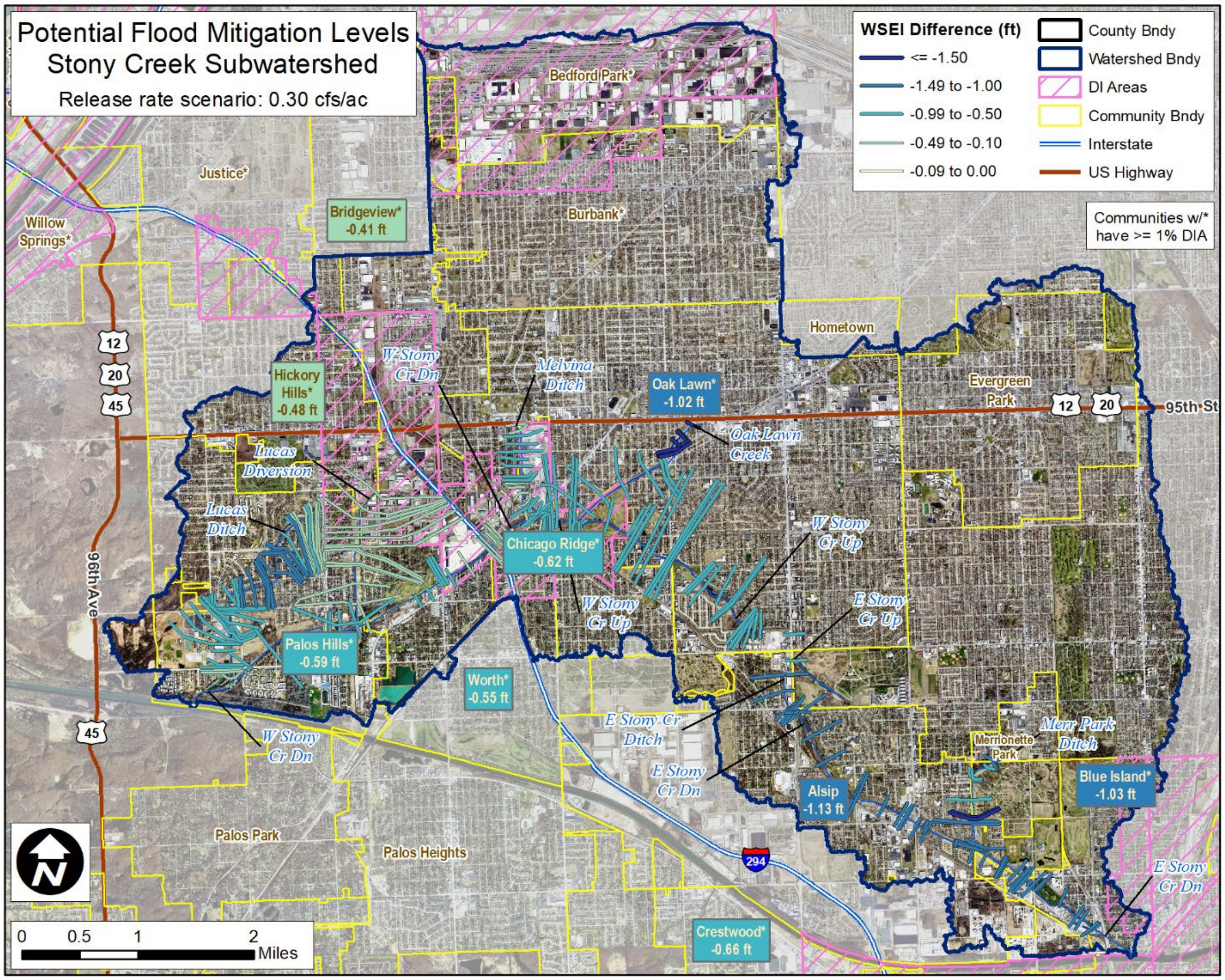


Figure B5: Flood mitigation levels in Stony Creek subwatershed (Cal Sag watershed) communities at WMO specified release rate, i.e., 0.30 cfs/ac

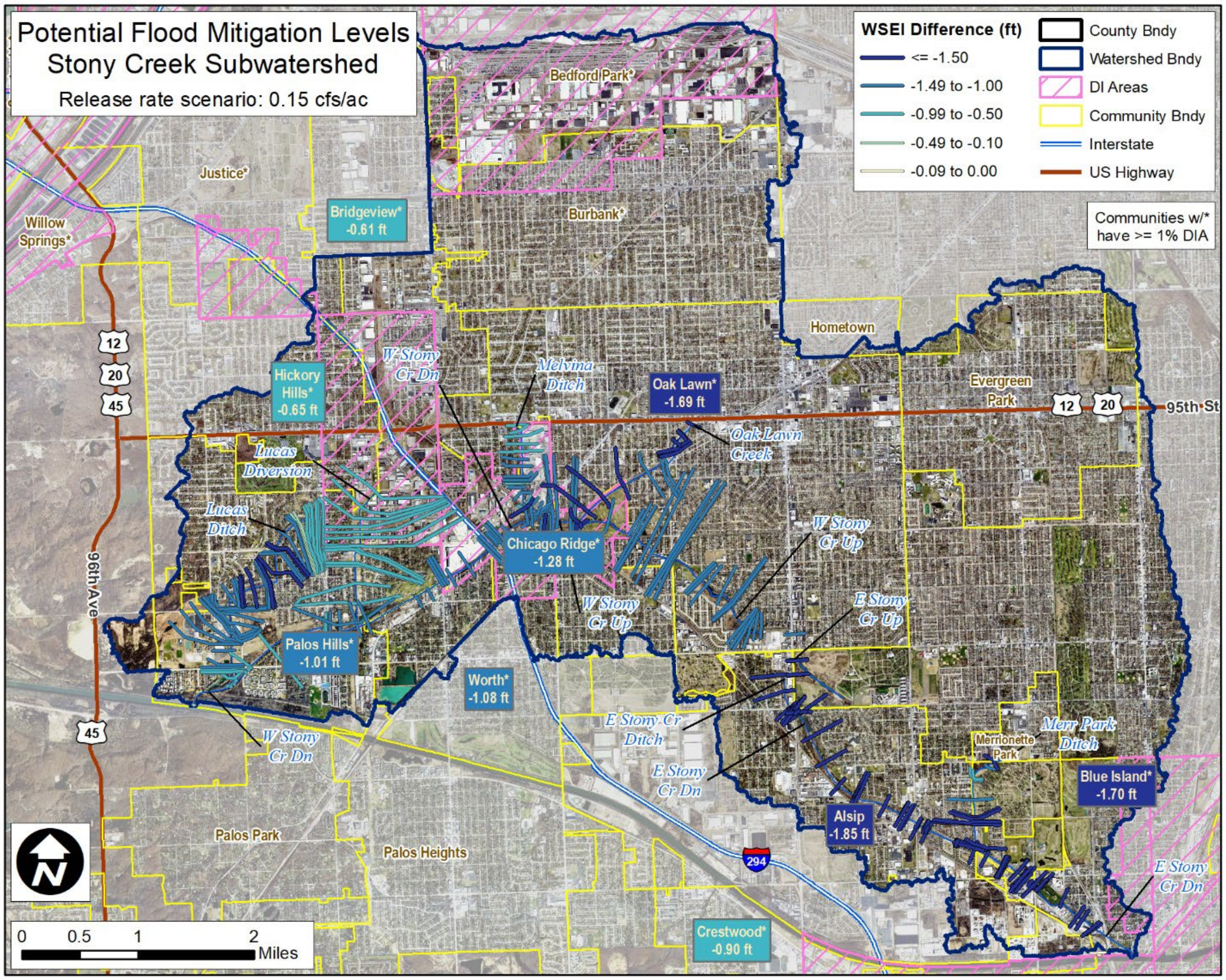


Figure B6: Flood mitigation levels in Stony Creek subwatershed (Cal Sag watershed) communities at release rate = 0.15 cfs/ac

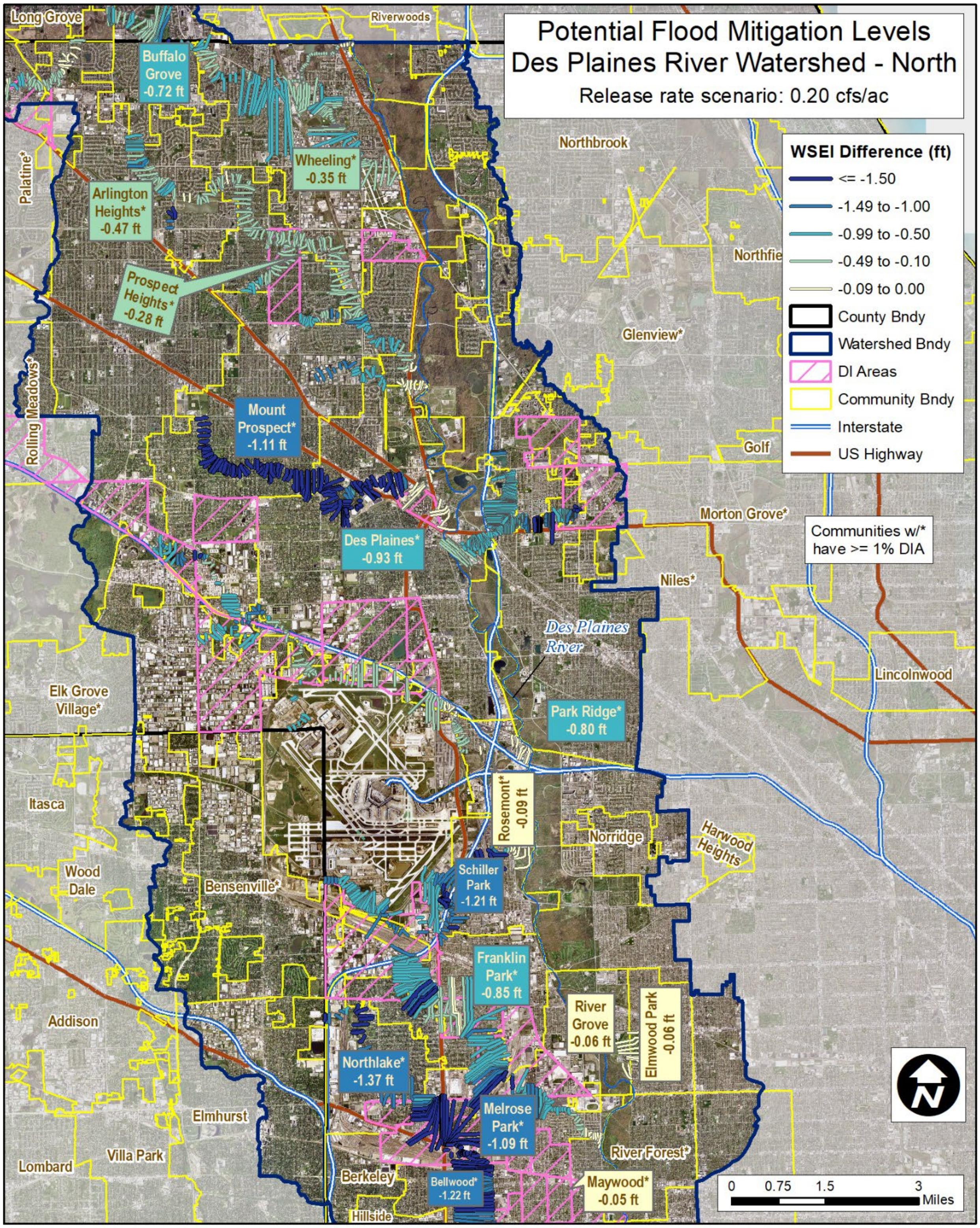


Figure B7: Flood mitigation levels in Des Plaines River watershed (northern half) communities at WMO specified release rate, i.e., 0.20 cfs/ac

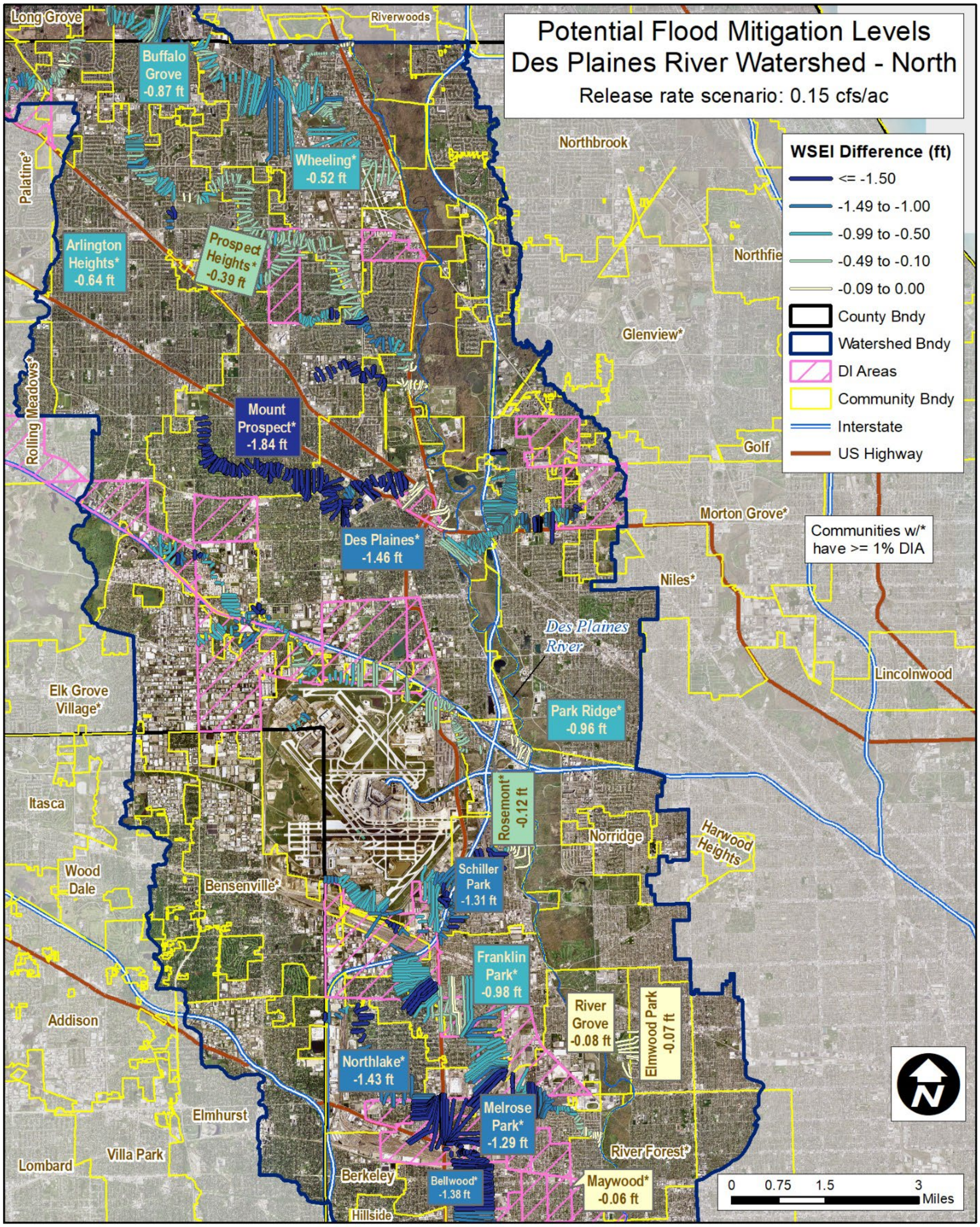


Figure B8: Flood mitigation levels in Des Plaines River watershed (northern half) communities at release rate = 0.15 cfs/ac

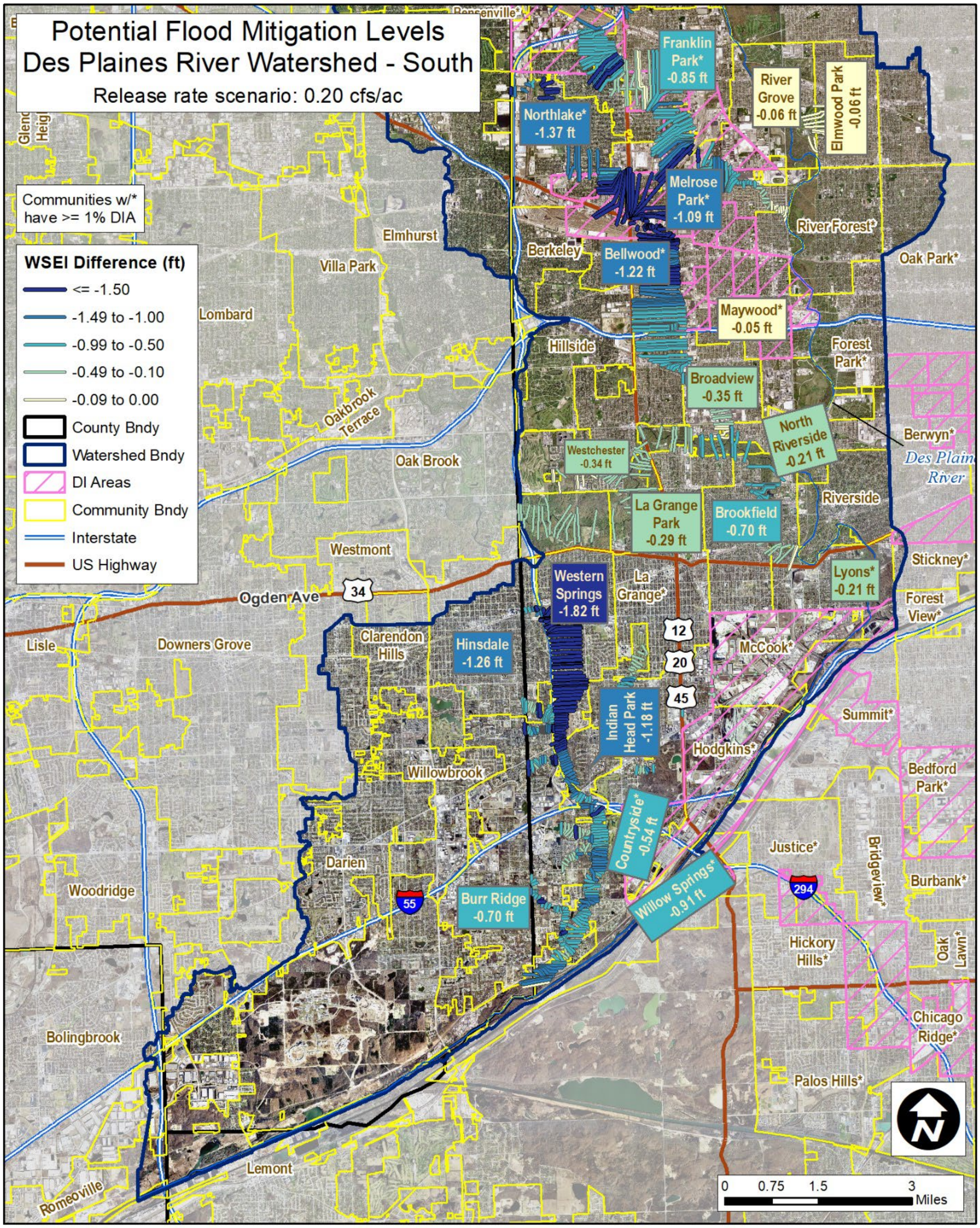


Figure B9: Flood mitigation levels in Des Plaines River watershed (southern half) communities at WMO specified release rate, i.e., 0.20 cfs/ac

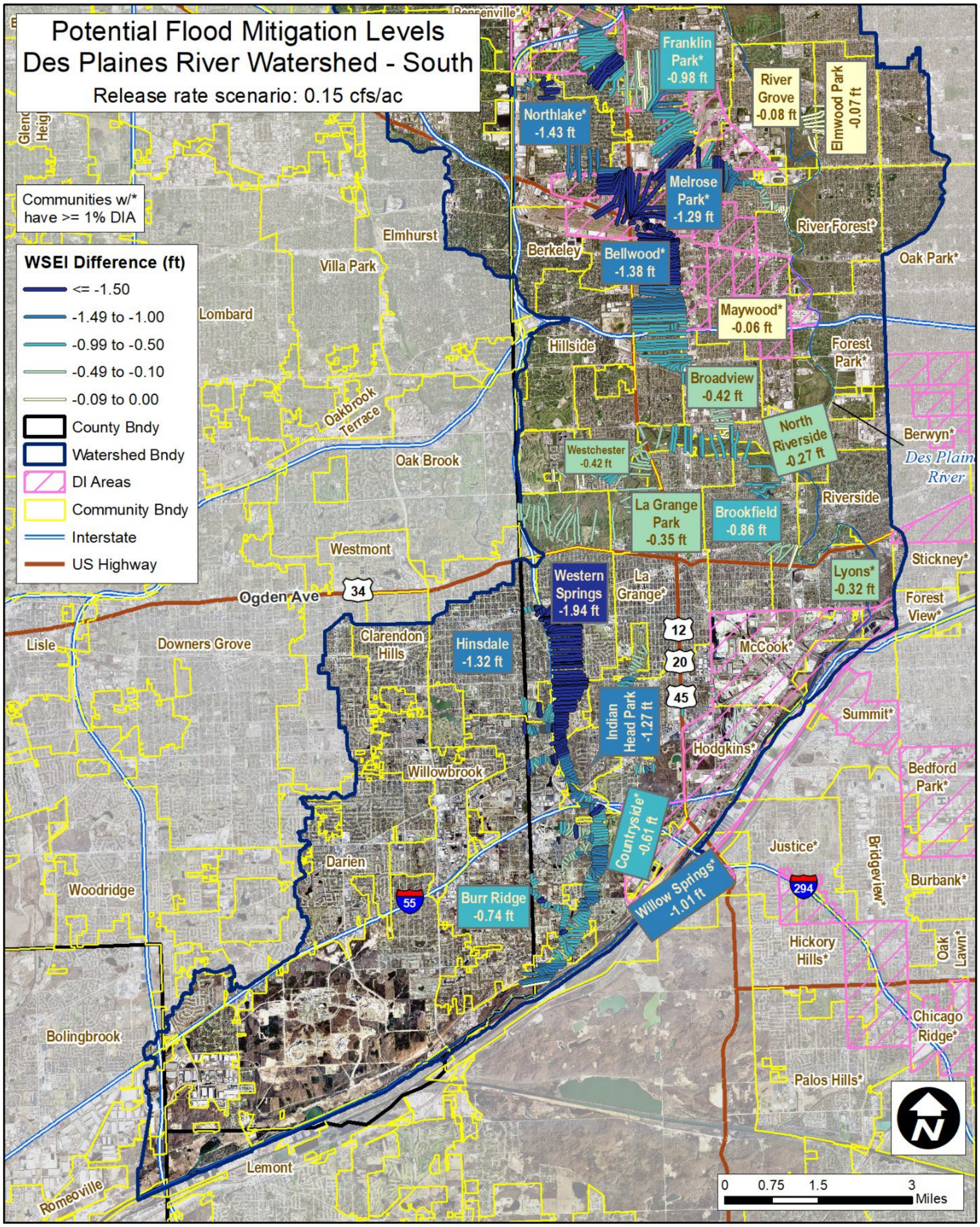


Figure B10: Flood mitigation levels in Des Plaines River watershed (southern half) communities at release rate = 0.15 cfs/ac

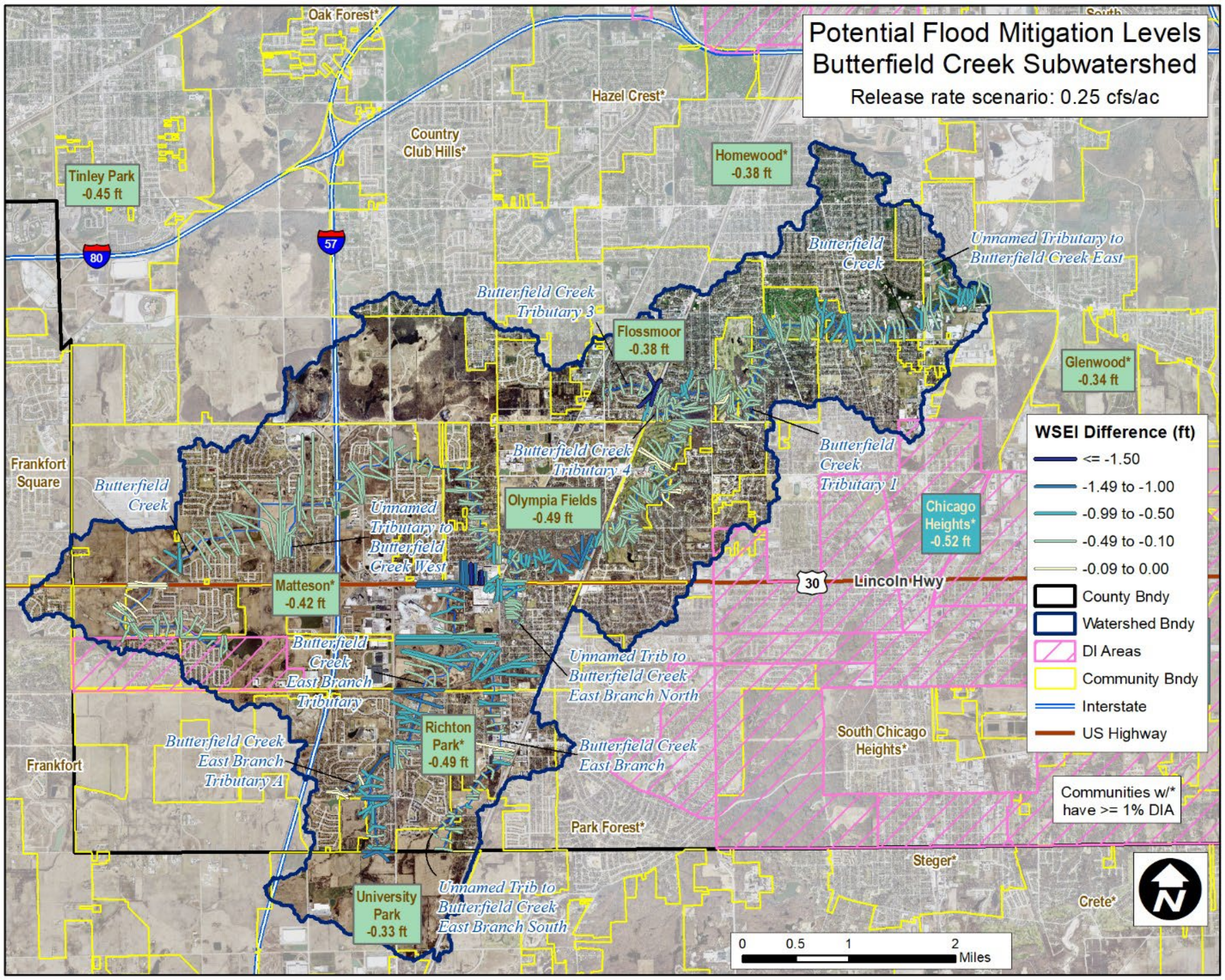


Figure B11: Flood mitigation levels in Butterfield Creek subwatershed (Little Calumet watershed) communities at WMO specified release rate, i.e., 0.25 cfs/ac

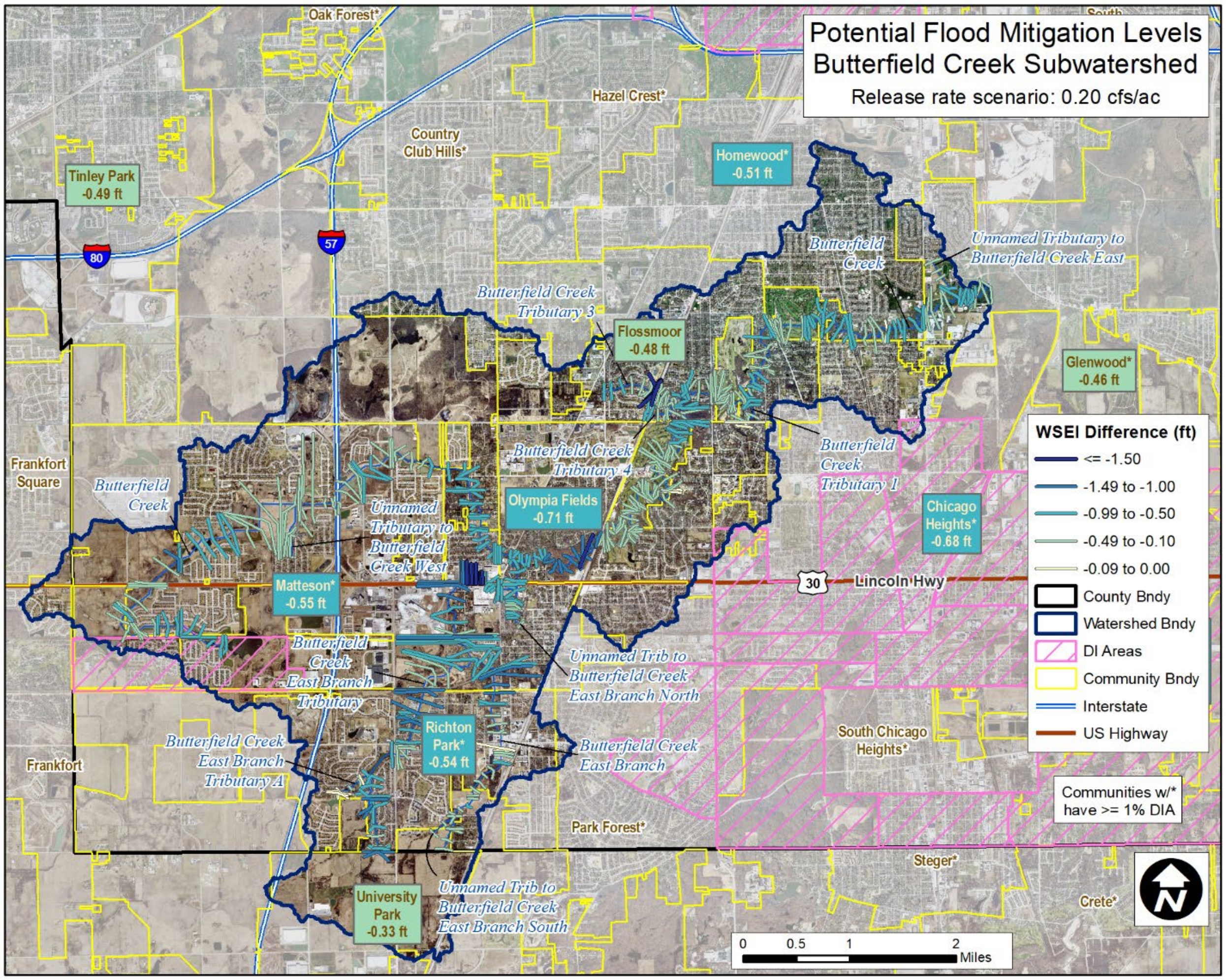


Figure B12: Flood mitigation levels in Butterfield Creek subwatershed (Little Calumet watershed) communities at release rate = 0.20 cfs/ac

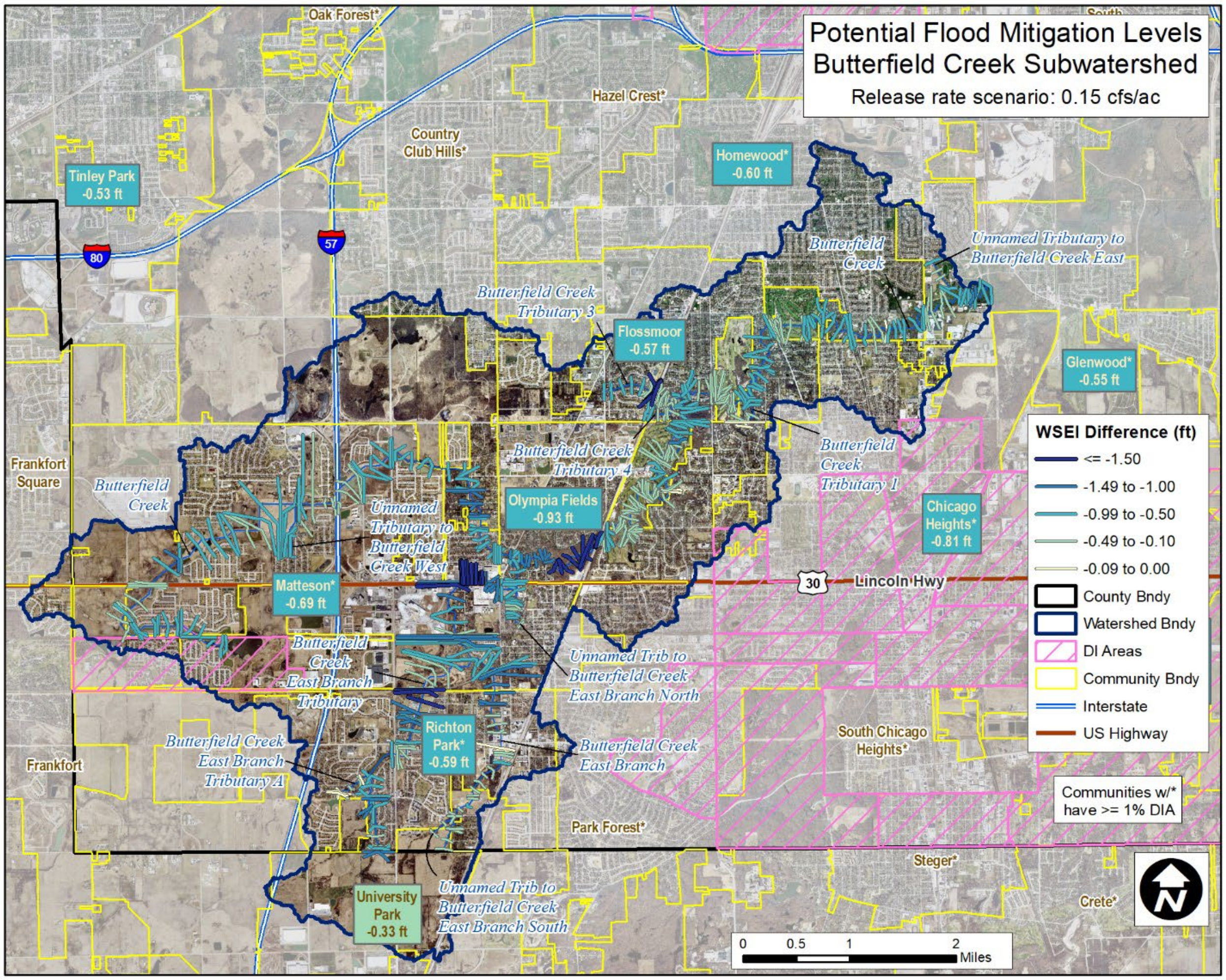


Figure B13: Flood mitigation levels in Butterfield Creek subwatershed (Little Calumet watershed) communities at release rate = 0.15 cfs/ac

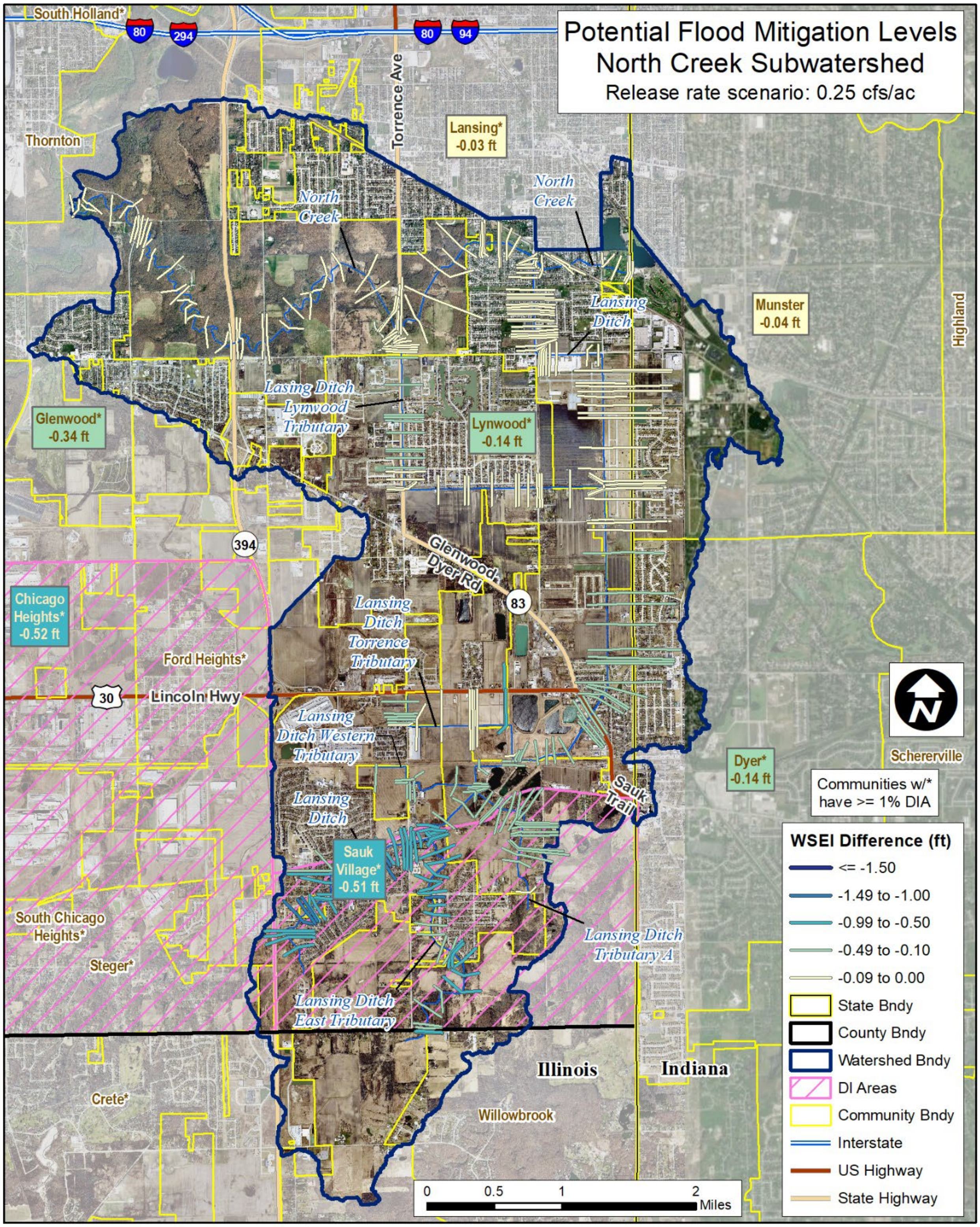


Figure B14: Flood mitigation levels in North Creek subwatershed (Little Calumet watershed) communities at WMO specified release rate, i.e., 0.25 cfs/ac

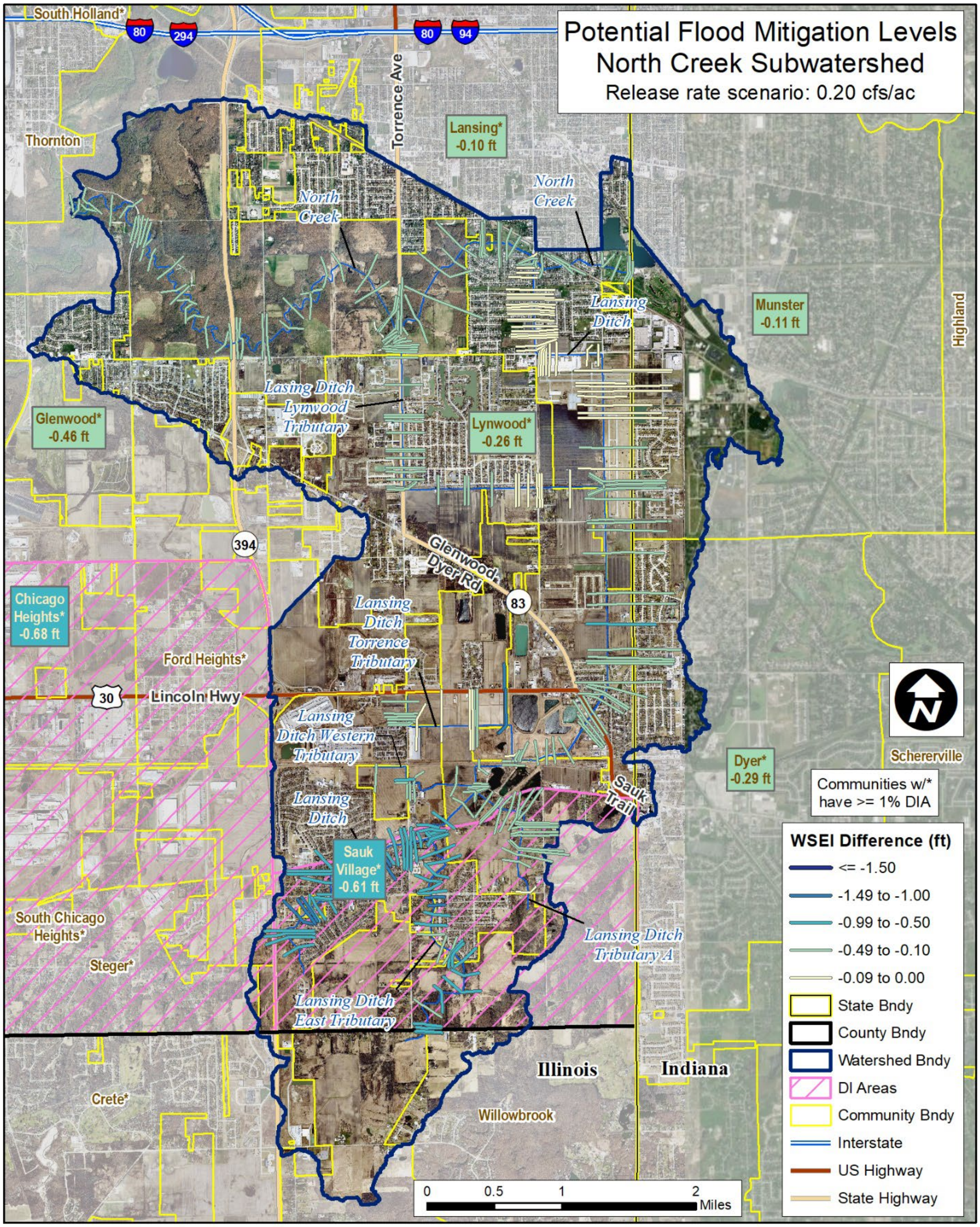


Figure B15: Flood mitigation levels in North Creek subwatershed (Little Calumet watershed) communities at release rate = 0.20 cfs/ac

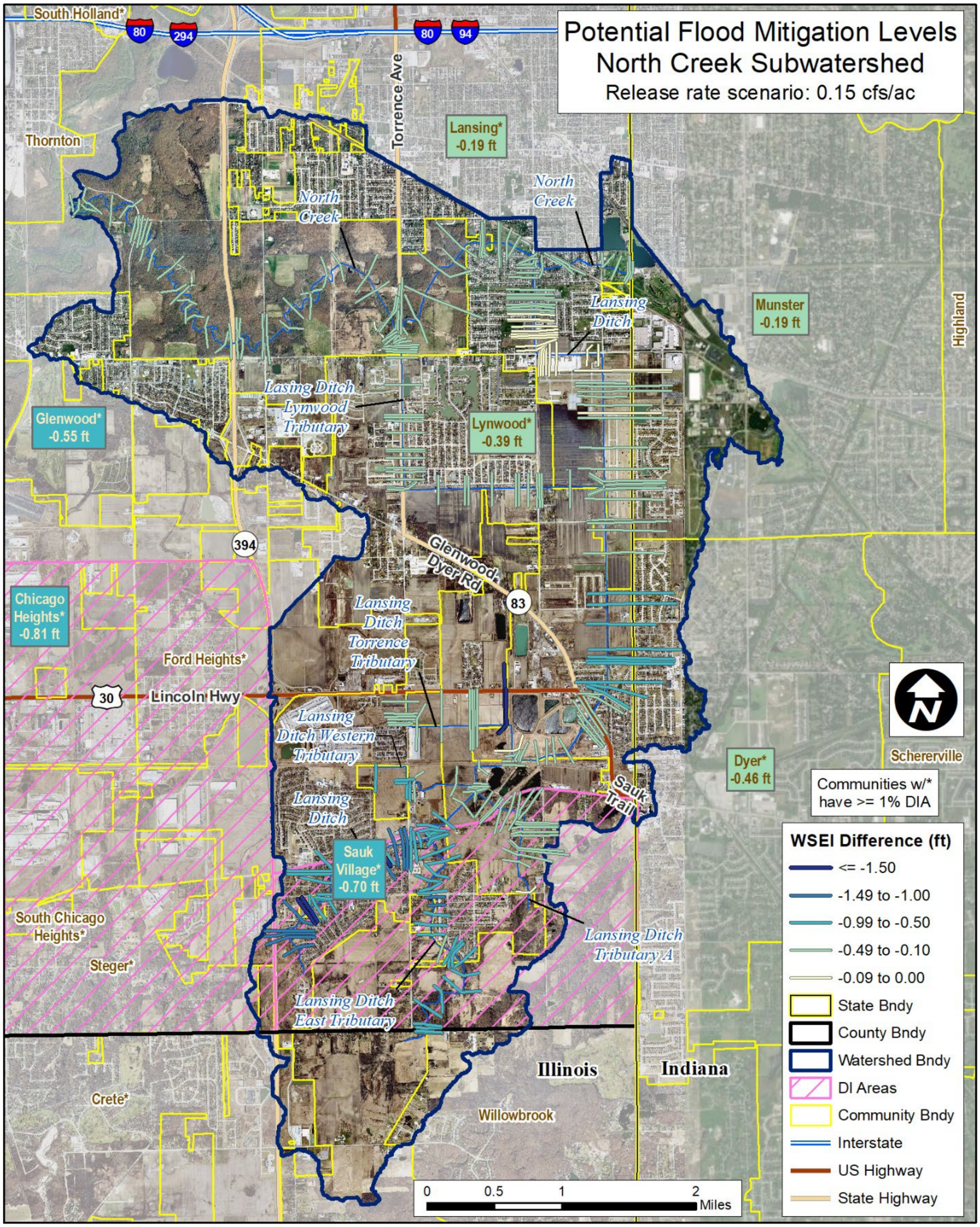


Figure B16: Flood mitigation levels in North Creek subwatershed (Little Calumet watershed) communities at release rate = 0.15 cfs/ac

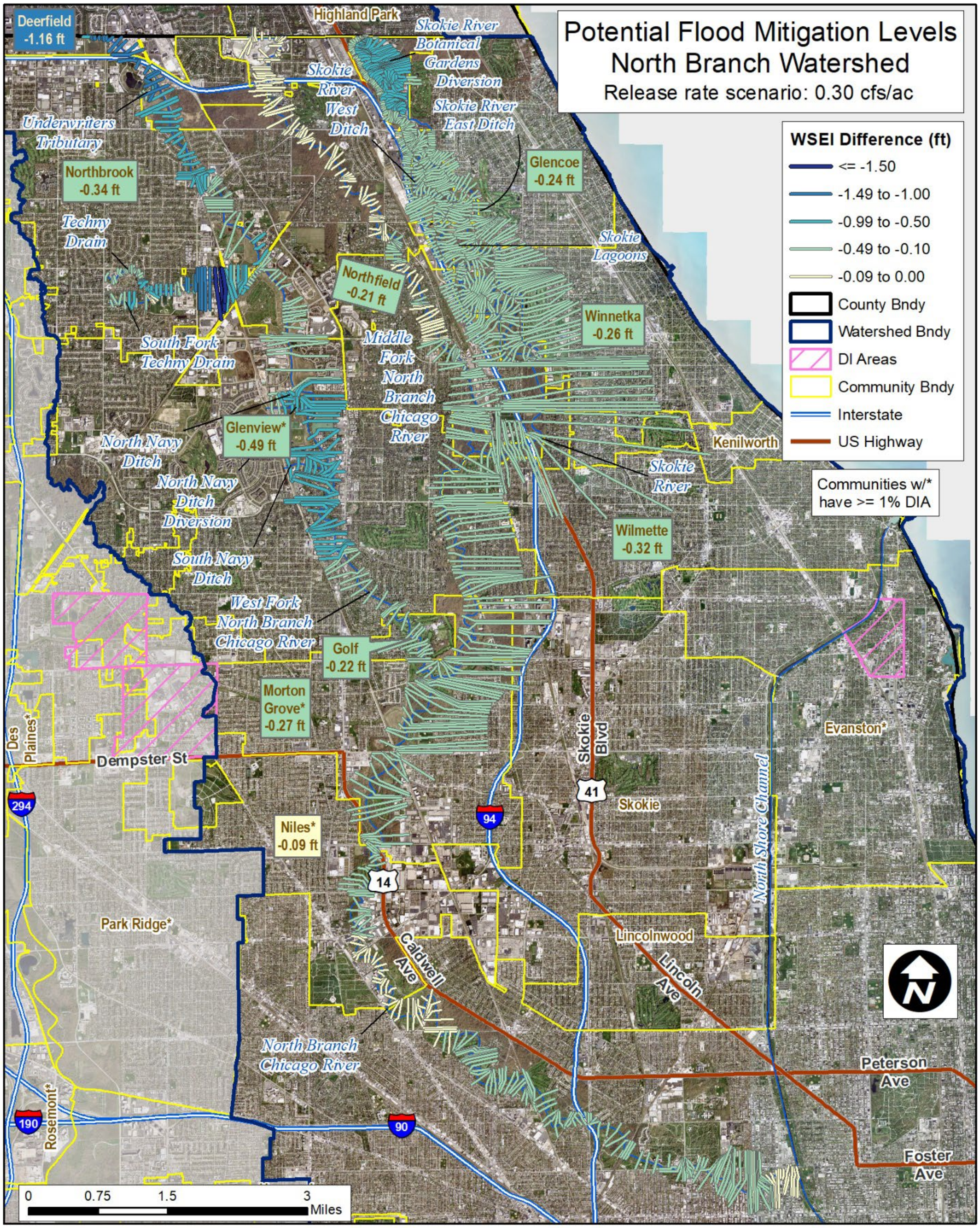


Figure B17: Flood mitigation levels in North Branch watershed communities at WMO specified release rate, i.e., 0.30 cfs/ac

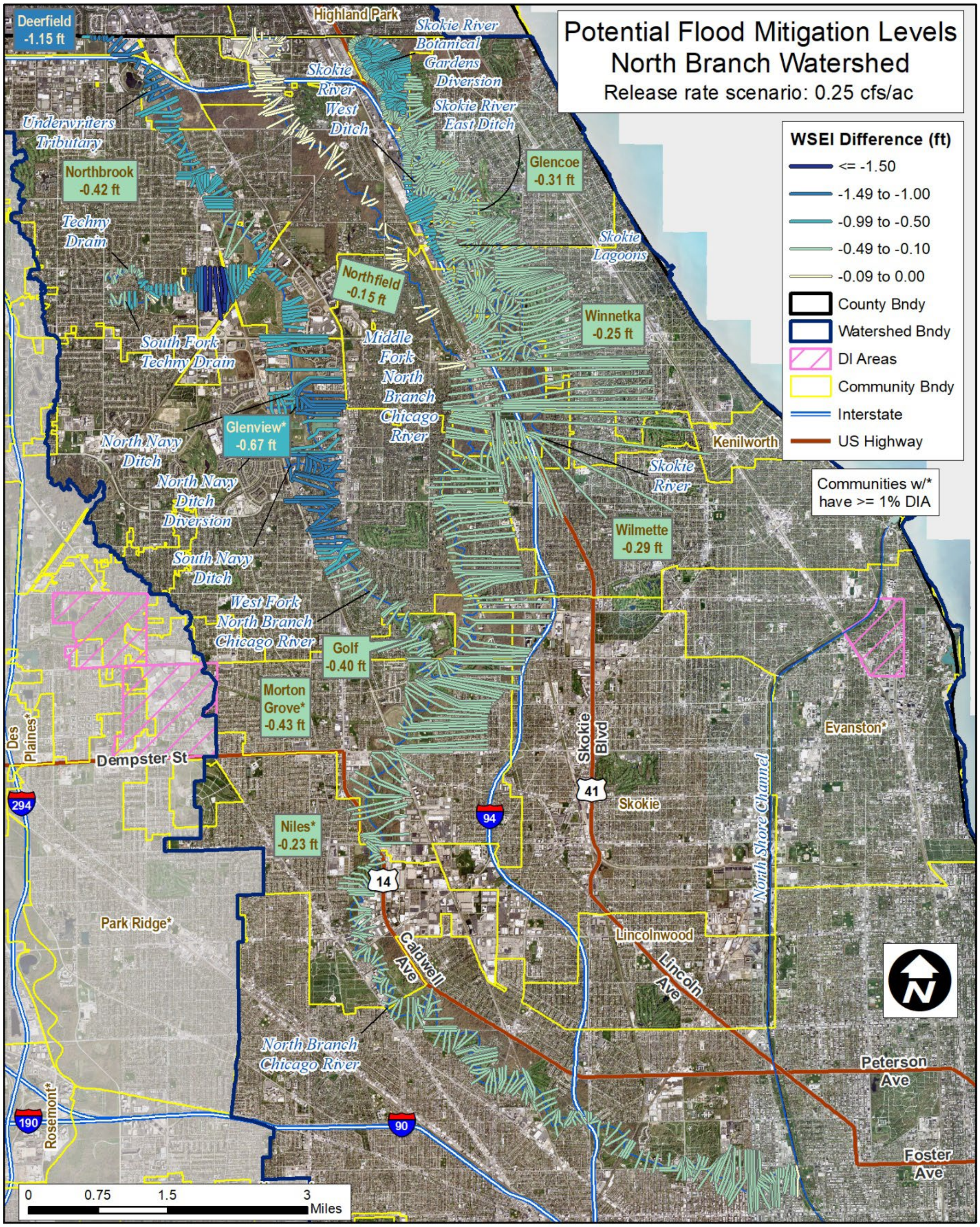


Figure B18: Flood mitigation levels in North Branch watershed communities at release rate = 0.25 cfs/ac

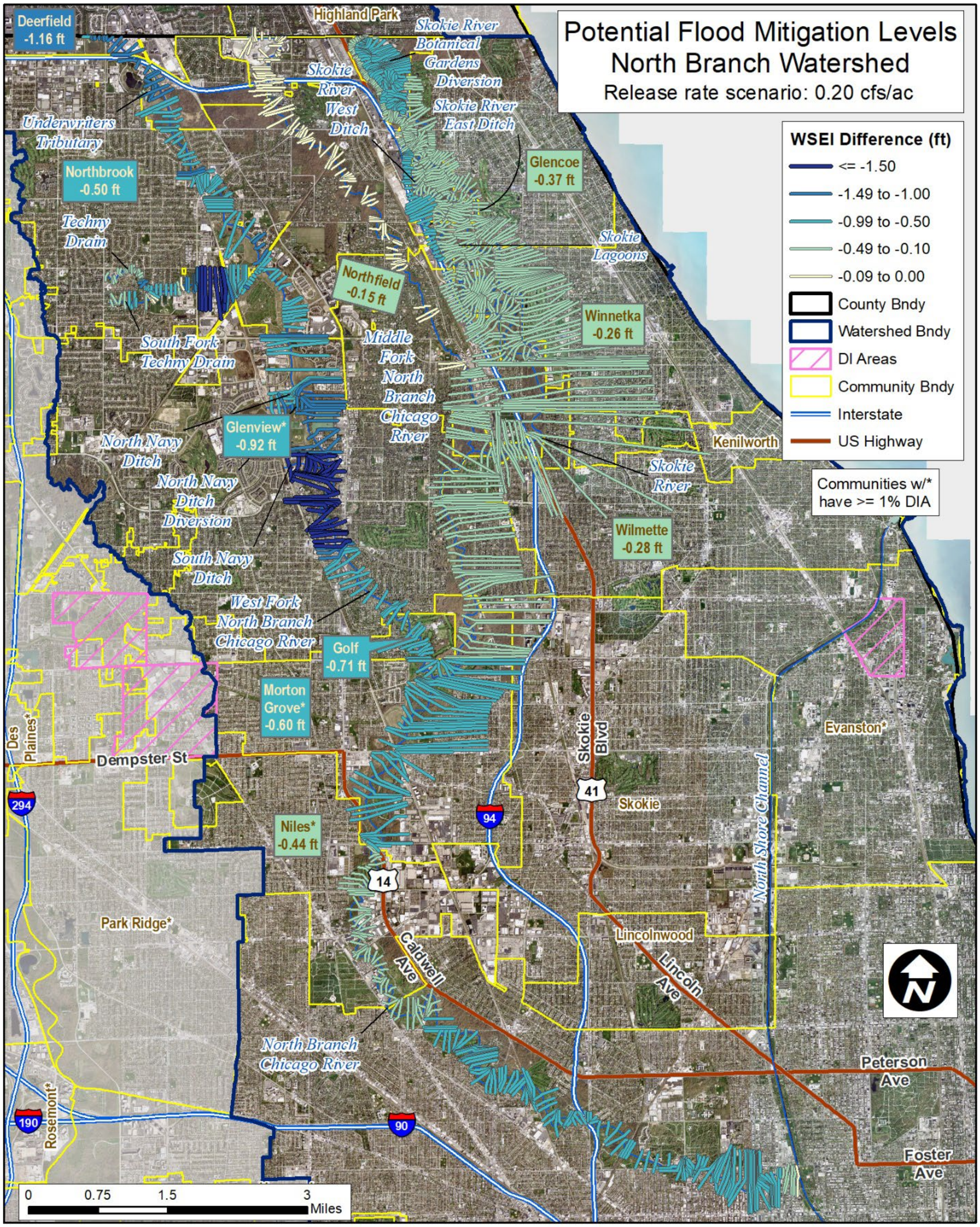


Figure B19: Flood mitigation levels in North Branch watershed communities at release rate = 0.20 cfs/ac

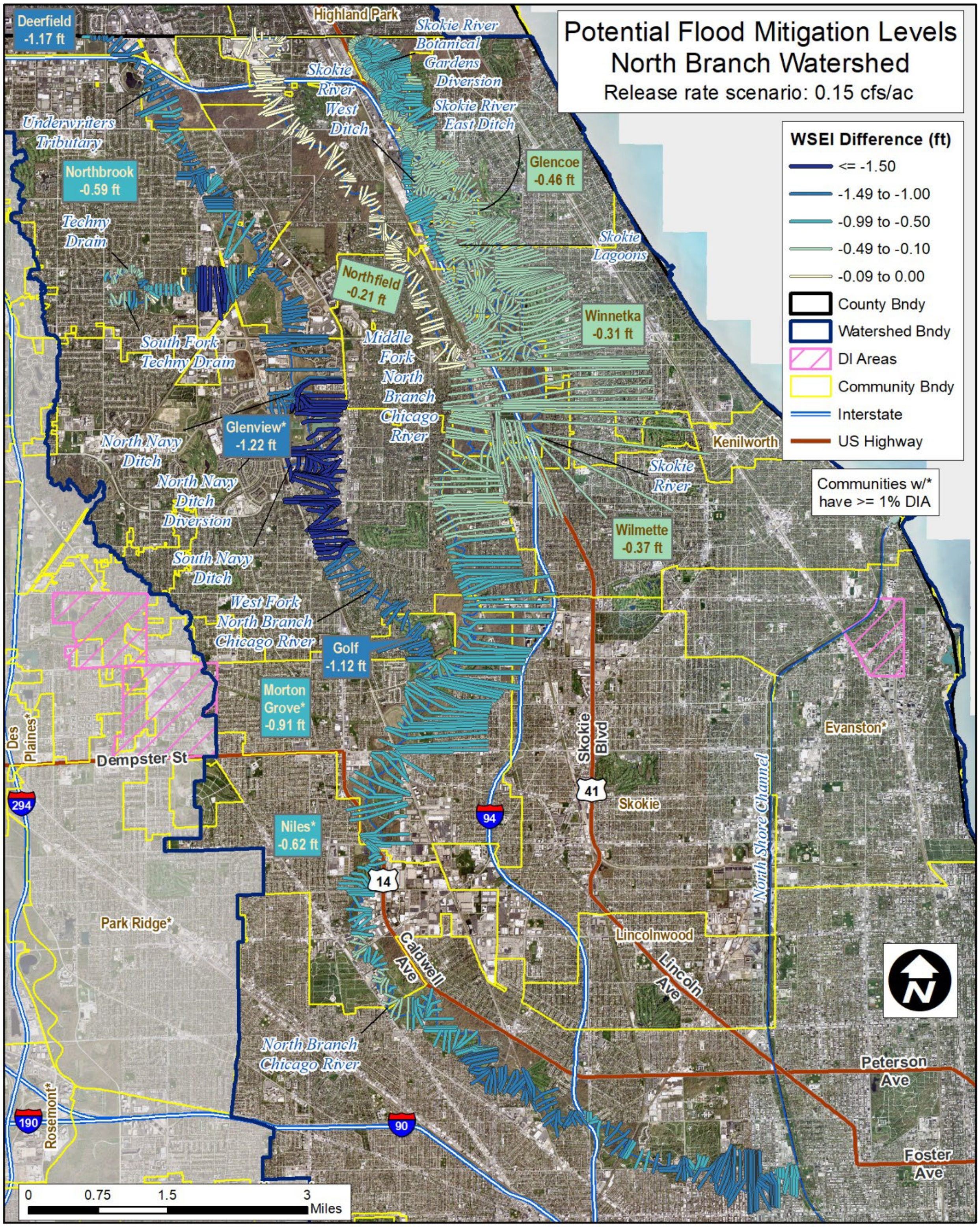


Figure B20: Flood mitigation levels in North Branch watershed communities at release rate = 0.15 cfs/ac

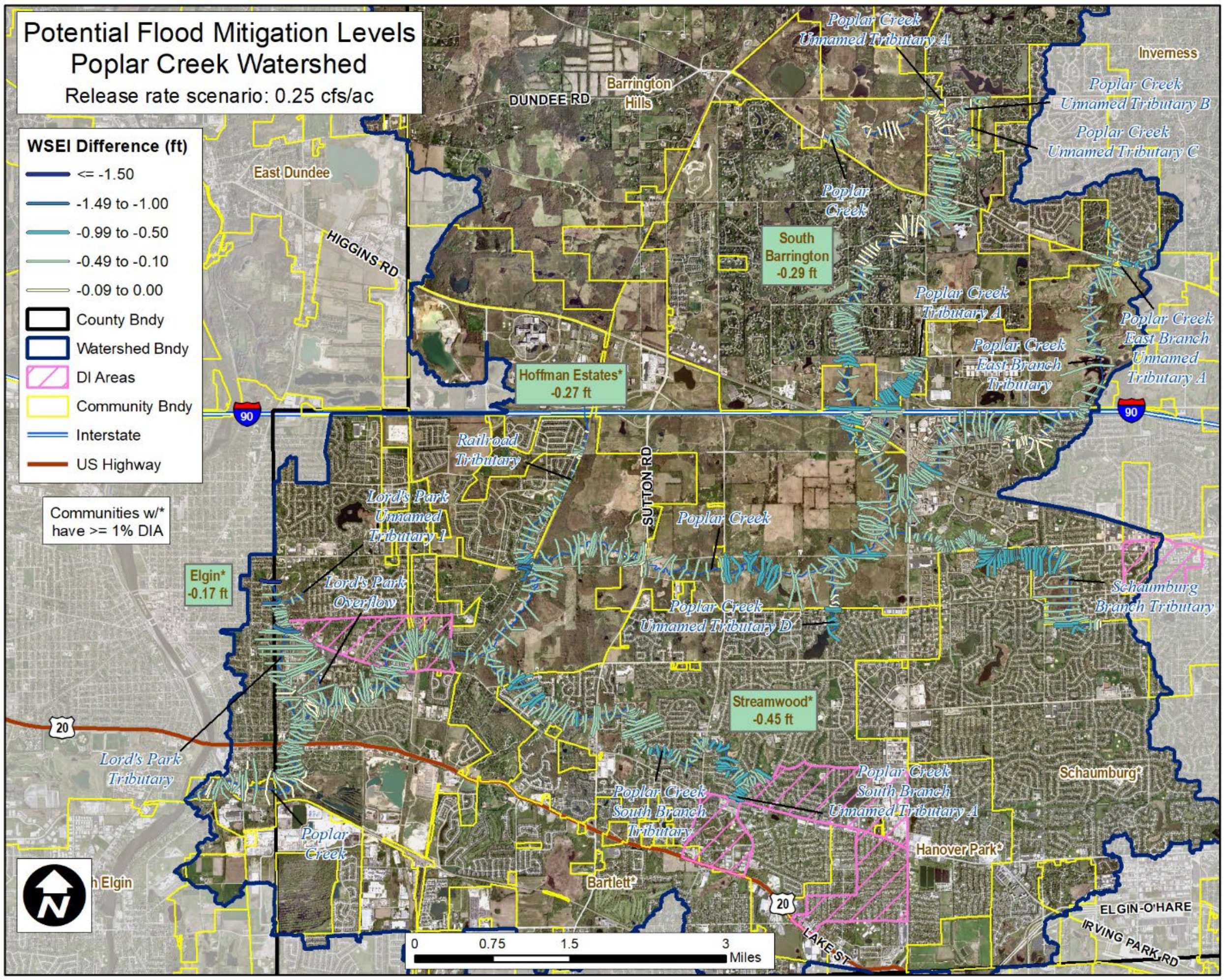


Figure B21: Flood mitigation levels in Poplar Creek watershed communities at WMO specified release rate, i.e., 0.25 cfs/ac

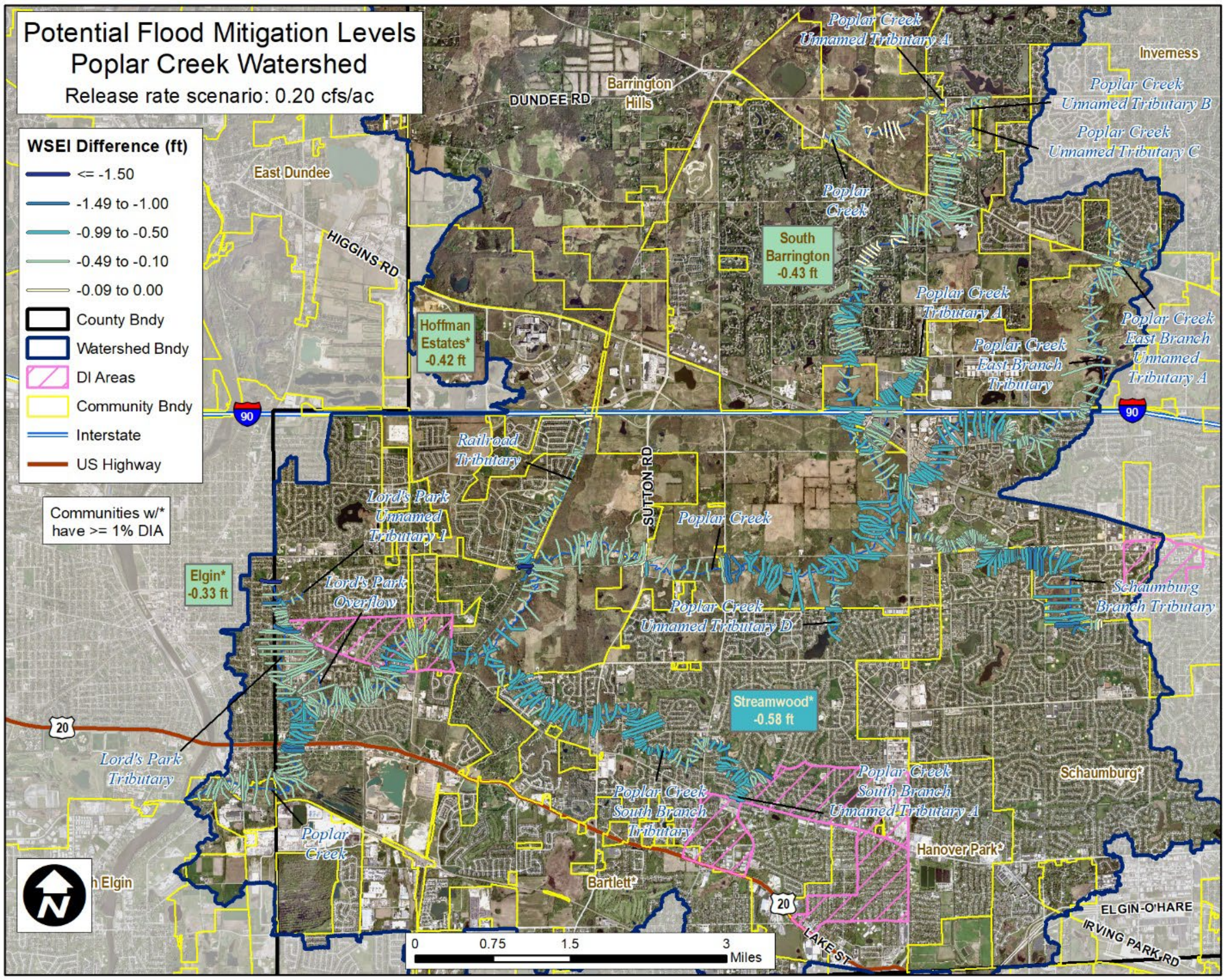


Figure B22: Flood mitigation levels in Poplar Creek watershed communities at release rate = 0.20 cfs/ac

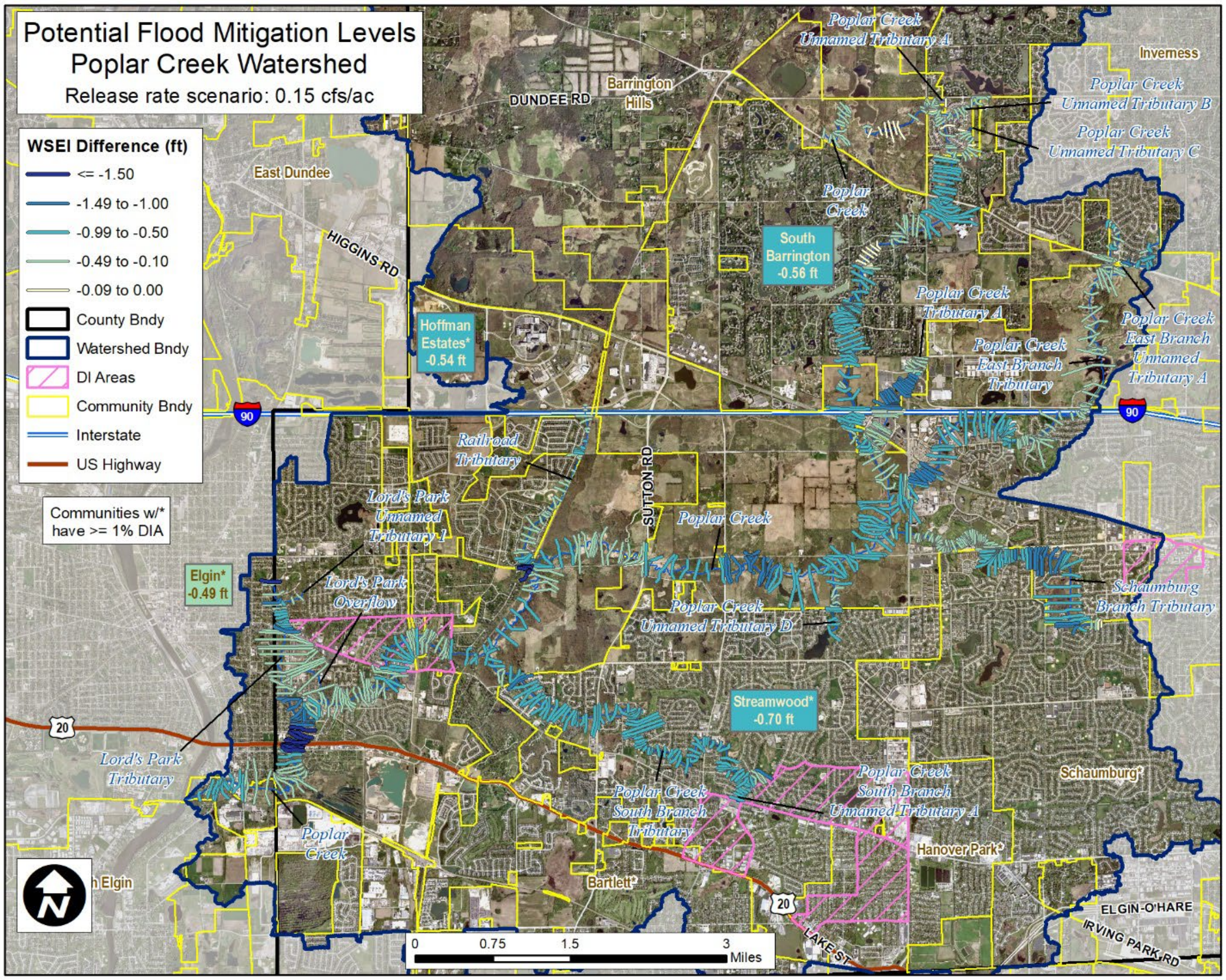


Figure B23: Flood mitigation levels in Poplar Creek watershed communities at release rate = 0.15 cfs/ac

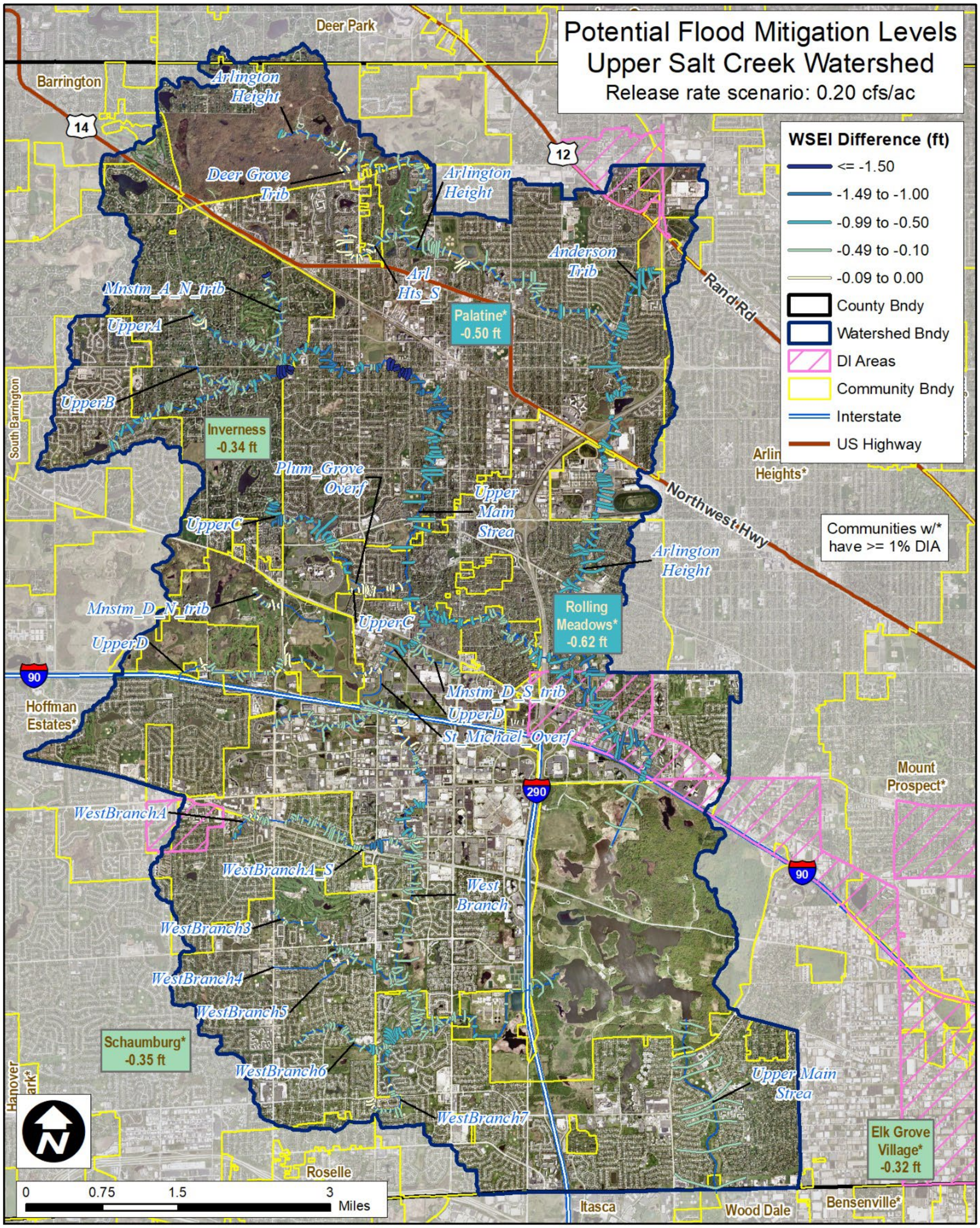


Figure B24: Flood mitigation levels in Upper Salt Creek watershed communities at WMO specified release rate, i.e., 0.20 cfs/ac

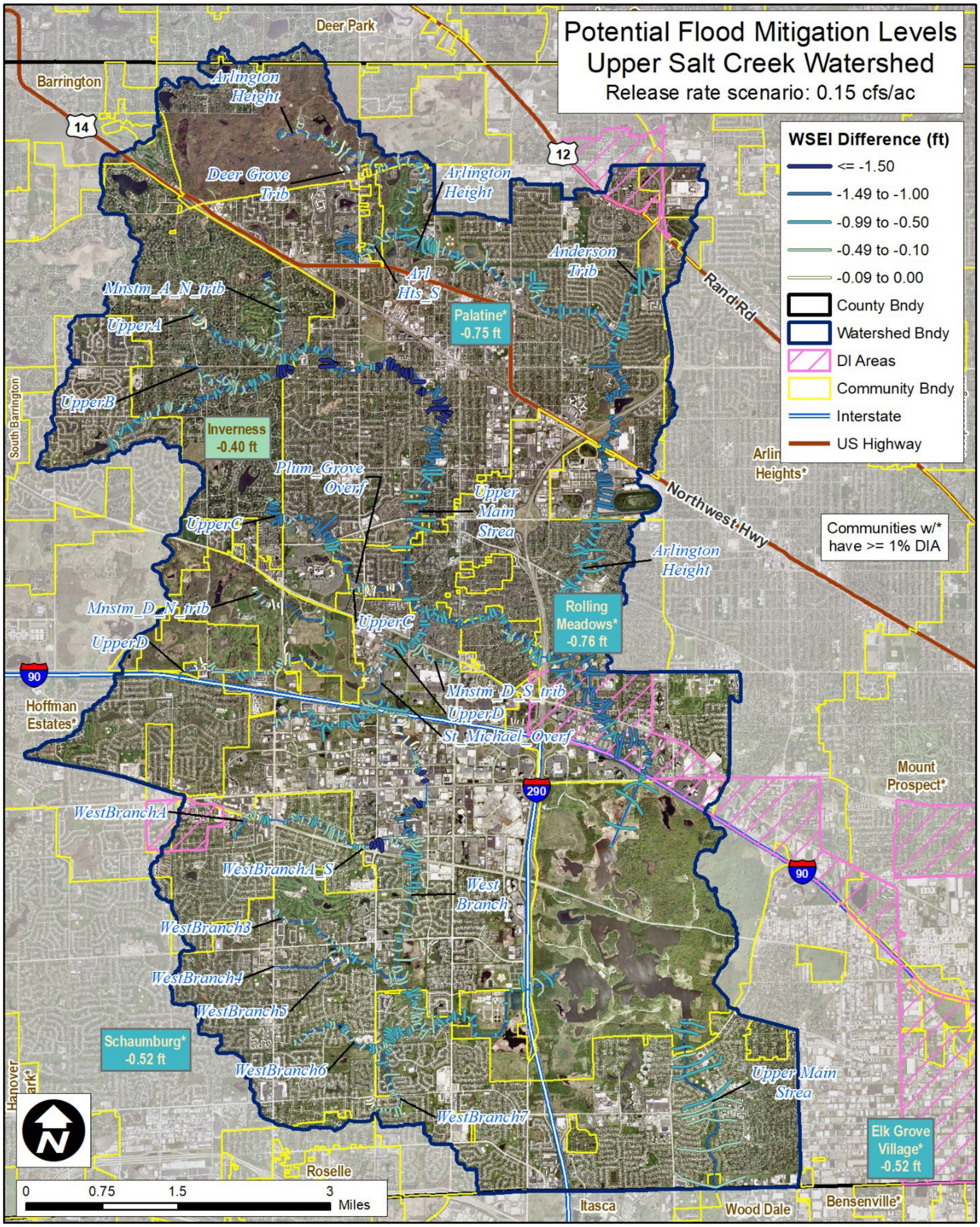


Figure B25: Flood mitigation levels in Upper Salt Creek watershed communities at WMO specified release rate, i.e., 0.15 cfs/ac