Wetland Quality

May 2025

Status and trends of wetlands in Minnesota: Depressional Wetland Quality Assessment (2007 – 2023)

An evaluation of the ecological health of marshes and ponds in the central forested and prairie regions of Minnesota.

MINNESOTA POLLUTION CONTROL AGENCY

Authors

John Genet
Michael Bourdaghs
Mark Gernes

Contributors/acknowledgements

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Editing and graphic design

Paul Andre
Lori McLain

Minnesota Pollution Control Agency

520 Lafayette Road North | Saint Paul, MN 55155-4194 |

651-296-6300 | 800-657-3864 | Or use your preferred relay service. | Info.pca@state.mn.us

This report is available in alternative formats upon request, and online at www.pca.state.mn.us.

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Foreword

This report represents the fourth installment in an ongoing series of status and trend reports on the ecological condition of marshes and ponds in the central forested and prairie regions of the state. Based on data collected during the summer of 2023, this report compares the latest data set to results from previous wetland surveys conducted in 2007/2008 (report released in 2012), 2012 (report released in 2015), and 2017 (report released in 2019). The latest survey represents a significant transition away from a whole-basin approach to one that is focused specifically on the shallow open water community within a wetland basin, requiring past survey results to be re-analyzed in light of this new approach. Additionally, the 2023 survey represents a temporary disruption to the 5-year schedule of the DWQA due to COVID-19 restrictions that suspended monitoring activities during the summer of 2020 and the resulting schedule shift of wetland monitoring projects that occurred (i.e., data collection efforts all shifted +1 year).

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Executive summary

From the prairie potholes in the southwest to the vast expanses of peatlands in the north, the diversity of Minnesota's wetland habitats is extraordinary. Roughly half of Minnesota's original wetlands, however, have been drained or filled following European settlement of the region. The public's perception of wetlands took a dramatic shift in the 1970s with recognition of the many ecological and societal benefits that they provide. In Minnesota this shift led to the passage of the Wetland Conservation Act (WCA) in 1991, which aims to "achieve no-net-loss in the quantity, quality, and biological diversity of Minnesota's existing wetlands" and eventually accomplish gains in these areas. It wasn't until 2006, that a statewide wetland monitoring program was initiated to assess status and trends of both wetland quantity and quality, providing an evaluation of whether the WCA was meeting its stated goals.

The Minnesota Department of Natural Resources (DNR) is primarily responsible for the implementation of the wetland quantity monitoring program, while the Minnesota Pollution Control Agency (MPCA) conducts the state's wetland quality monitoring program. Currently wetland quality is being tracked via two separate surveys, the Minnesota Wetland Condition Assessment (MWCA) and the Depressional Wetland Quality Assessment (DWQA). The focus of this report is on the latest results of the DWQA, evaluating the current ecological condition of open water wetlands and ponds in Minnesota's central forested and prairie regions as well as determining whether conditions have changed relative to previous surveys conducted in 2007/2008, 2012, and 2017.

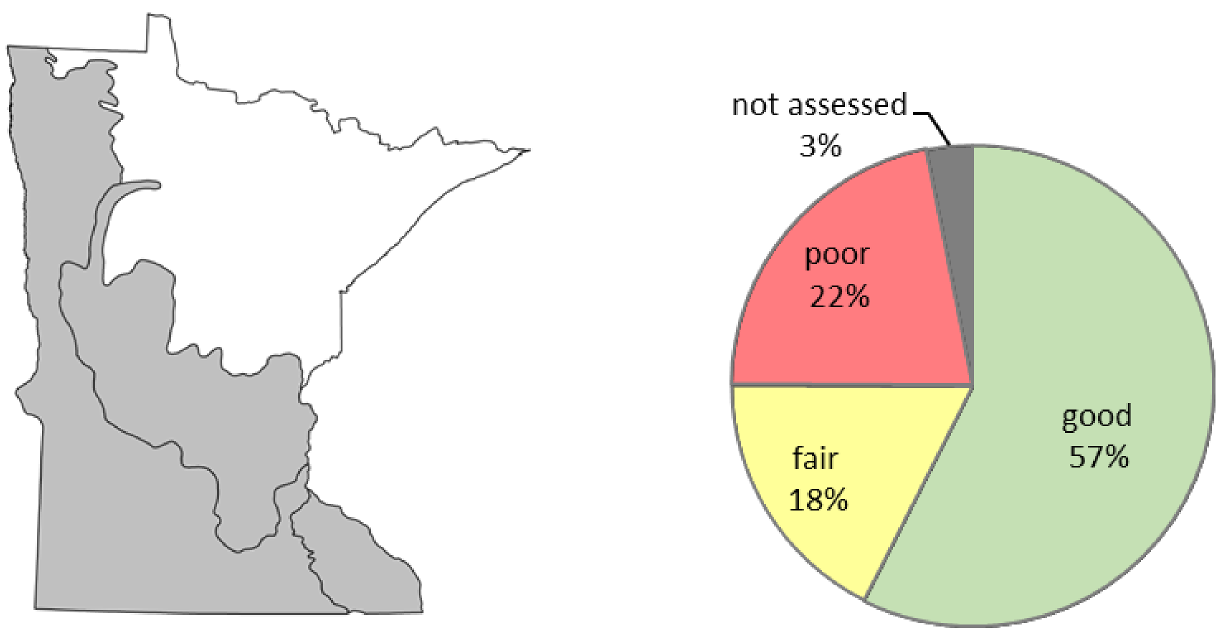
The DWQA uses a survey approach to produce unbiased condition estimates for the population of depressional wetlands and ponds in the study regions based on results obtained from a sample of randomly selected sites. Plant and macroinvertebrate indicators developed and calibrated by community type or ecoregion, respectively, are the primary indicators of wetland condition used in the DWQA. Additionally, several physical and chemical characteristics of the water column were measured in each survey site to gauge its exposure to various types of pollution. In the latest survey, several refinements were implemented that shifted the focus away from the entirety of depressional wetland basins—assessing all plant community types therein—to just assessing the condition of the shallow open water community within each wetland basin. While adjustments to the analysis of past survey results (e.g., modifications to extrapolation weights, Appendix A) allowed continuity and comparability of macroinvertebrate indicator results across all surveys, comparable results among surveys could not be achieved for the wetland plant community indicator. Therefore, the latest DWQA does not include results for wetland vegetation while sampling methods, indicator refinement, and condition criteria continue to be investigated by MPCA plant ecologists. It is anticipated that wetland vegetation will be re-integrated into the next DWQA survey, currently scheduled to occur in 2027.

Aquatic macroinvertebrates are small animals that can be seen with the naked eye and have no backbone, such as aquatic insects (adult or larval stages), crayfish, snails, and leeches. These organisms reside either permanently (e.g., snails) or temporarily (e.g., aquatic insects) in the shallow open waters of depressional wetlands and ponds. Monitoring the condition of the aquatic macroinvertebrate community is therefore a reliable indication of ecological condition for the shallow open water community overall. Criteria for categorizing the condition of survey sites as good, fair, or poor based on the macroinvertebrate indicator were established relative to least-disturbed reference sites within each region (i.e., central forests and prairie). A similar approach was used for categorizing water quality measurements as high, medium, or low to characterize the amount of human disturbance impacting each site.

The latest DWQA results are based on site evaluations and monitoring that was conducted in May and June of 2023. Currently there are an estimated 342,769 acres of semi-permanently to permanently flooded shallow open water wetlands and ponds in the study region, which represents an estimated 137,962 distinct areas of this community type. Across all surveys (2007-2023) there appears to be an increasing trend in shallow open water area; however, there are no statistically significant differences between any of the survey periods at this time. Shallow open water communities associated with naturally formed wetlands represent a much larger proportion of the total acreage compared to shallow open water associated with human-created ponds, a pattern that is evident across all surveys.

The majority of shallow open water communities are in good condition across the study region according to macroinvertebrate indicators (Figure 1). The ecological condition of this community type does, however, vary between the central forested and prairie regions. In particular, a much larger percentage of shallow open water area in the prairie region is in poor condition (45%) compared to the central forested region (11%), with a similar difference in the percentage of those in good condition. While estimates have varied slightly across the four survey periods, percentages of shallow open water communities in good or poor conditions have remained stable with no statistically significant differences observed between surveys.

Figure 1. 2023 aquatic macroinvertebrate community condition estimates (in % acres) for the shallow open water portion of depressional wetlands and ponds. Shaded area of map indicates spatial extent of the DWQA, focused on the central forested and prairie regions of the state, excluding the northeastern region.



Of the water quality variables measured in the survey, chloride, phosphorus, and sulfate concentrations were most frequently elevated (i.e., high) compared to least-disturbed reference wetlands. In terms of their impact on aquatic macroinvertebrate communities, elevated phosphorus and sulfate concentrations appear to be the most harmful. Both variables consistently posed an elevated risk to aquatic macroinvertebrates in each DWQA survey. While it's too early to determine whether either of these pollutants is trending in one direction or another, sulfate may be increasing in concentration within shallow open water communities across the study region. However, decreasing sulfate concentrations over time were observed among MPCA's long-term monitoring wetland sites. Given sulfate's impact on aquatic macroinvertebrate community health, determining whether a broad-scale trend exists for this pollutant will be a priority for future DWQA surveys.

Overall, based on aquatic macroinvertebrate indicator results to date, it appears that the WCA "no-net-loss" goal for wetland quality is being achieved for shallow open water communities associated with depressional wetlands and ponds in the central forested and prairie regions of Minnesota. It should be noted that wetland quality is at a relatively low level in the prairie region where roughly 40-45% of shallow open water communities are in poor condition, depending on the survey. The next DWQA is scheduled to occur in 2027 and in addition to sub-decadal change analyses, evaluating the presence of long-term trends across the survey periods will occur. Therefore, in the very near future the DWQA will have more sophisticated methods for ascertaining "no-net-loss" or wetland quality gains for this subset of Minnesota's wetlands.

Introduction

Healthy ecosystems rely on a diversity of wetland community types to provide habitat for native vegetation and wildlife, reduce erosion during peak flow events, maintain stream flow during drier periods, recharge aquifers, and assimilate pollutants derived from upland sources. Globally, wetlands are gaining attention for their ability to trap and store carbon, and thus may be a key component in the strategy to reduce the effects of climate change. Wetlands have also been woven into the fabric of Minnesota's culture, beginning with the customs of Native Americans who harvested wild rice and traditional medicinal plants from wetland habitats. These traditions continue today and have been supplemented by other uses such as waterfowl hunting, bird watching, and outdoor recreation. Wetlands that become degraded as a result of physical alteration, pollution, hydrologic modification, or invasive species may not be able to provide some or all of these benefits. Recognition of the importance of water quality, including in wetlands, continues to increase. A milestone in public recognition of the value of wetlands occurred in 1991 with the passage of Minnesota's WCA.

The purpose of the WCA is to "achieve no net loss in the quantity, quality, and biological diversity of Minnesota's existing wetlands" (Minn. R. Ch. 8420.0100). Furthermore, the act seeks to increase wetland quantity, quality, and biological diversity in the state by restoring or enhancing diminished or drained wetlands. Full implementation of the WCA began in 1994 and reporting of wetland gains and losses, focused primarily on quantity (acres or projects), began soon thereafter (BWSR 1996, 1998, 2000, 2001, 2005). However, this reporting system did not account for wetlands lost or degraded by unregulated actions (e.g., WCA exemptions, illegal activities, nonpoint source pollution), deviations from actions proposed in permit applications, temporary losses (i.e., the period before a replacement wetland is mature and fully functioning), mitigation credits for the establishment of upland buffers or wetland preservation, restoration projects that involve multiple organizations, and private restorations (Gernes and Norris 2006). In 2006, a Comprehensive Wetland Assessment, Monitoring, and Mapping Strategy was developed by state and federal agencies responsible for wetland protection and regulation in Minnesota to address these information gaps.

A primary outcome of the comprehensive strategy was the development of statewide random surveys to assess status and trends of wetland quantity and quality in Minnesota. Minnesota's Wetland Status and Trends Monitoring Program (WSTMP), the state's wetland quantity survey, was modeled after the U.S. Fish and Wildlife Service wetland status and trends program (e.g., Dahl 2006, 2011) and is conducted by the Minnesota Department of Natural Resources (MN DNR). Prior to European settlement in the region, Minnesota had an estimated 18.6 million acres of wetlands (Anderson and Craig 1984) that accounted for about 34% of the state's area. The WSTMP estimates that currently ~10.6 million acres of wetlands remain in the state (Kloiber 2010, Kloiber and Norris 2013, Kendig et al. 2024) but also indicates that the historical trend of (net) wetland loss may be reversing. Change analyses across four cycles of the WSTMP (2006 – 2020) have demonstrated small—but statistically significant—net gains in wetland acreage statewide (Kloiber and Norris 2017, Kendig et al. 2024). While early cycles of the WSTMP show that much of the wetland gains were due to increases in un-vegetated ponds (i.e., unconsolidated bottom), recent cycles indicate that gains are now primarily driven by emergent wetlands.

Assessing wetland quality status and trends is conducted by the MPCA via the Depressional Wetland Quality Assessment (DWQA) and the Minnesota Wetland Condition Assessment (MWCA; see Table 1 for comparison). The findings of these two independent surveys are intended to compliment the results of the WSTMP, together providing a comprehensive assessment of WCA's no net loss goal. Following publication of the 2019 DWQA report, the MPCA solicited feedback from various wetland professionals on ways to improve the DWQA so that results were more useful and complimentary to the MWCA. For example, prior to the 2023 survey the DWQA included six plant community types that were also included in the MWCA (Table 1). Recognizing the strength of the DWQA in evaluating the condition of shallow open water (SOW) communities—with indicators based on biology and water chemistry—the MPCA decided to focus the DWQA on this community type rather than continue to utilize a whole-basin approach where all community types present within the depressional wetland or pond (i.e., to the upland edge) are evaluated. It is important to note however that the MWCA will continue to include SOW communities as the DWQA does not include the Mixed Wood Shield ecoregion, nor does it examine SOW communities in hydrogeomorphic settings other than depressions. See Appendix A for more information regarding the transition of the DWQA to the SOW community type.

The focus of this report is on results from the fourth round of the DWQA and whether conditions have changed relative to previous iterations of the survey, evaluated from the new perspective of shallow open water communities.

Table 1. Comparison of MPCA's wetland quality surveys.

Characteristics	Depressional Wetland Quality Assessment (DWQA)	Minnesota Wetland Condition Assessment (MWCA)
Wetland plant community types (pre-2023)	shallow marsh, deep marsh, shallow open water, shrub-carr, rich fen, fresh meadow (restricted to depressional landscape setting)	shallow marsh, deep marsh, shallow open water, rich fen, wet prairie, fresh meadow, calcareous fen, shrub-carr, alder thicket, open bog, coniferous bog, coniferous swamp, hardwood swamp, floodplain forest (restricted to < 1m in depth)
Wetland plant community types (post-2023)	shallow open water	
Condition indicators	macroinvertebrates, vegetation	vegetation
Stressor indicators	surface water chemistry	human disturbance assessment
Reporting units	numbers or acres	acres
Spatial extent	central forested and prairie regions	statewide
Cycles completed	4	3
Year initiated	2007	2011

Measuring wetland quality

Biological monitoring and assessment is one of the most commonly used approaches for measuring the ecological condition of aquatic ecosystems (Karr and Chu 1999). Aquatic organisms, constantly exposed to their environment, are able to integrate the effects of multiple stressors over time and space. A successful biological assessment approach requires the adoption of a classification scheme to reduce natural variability, establishment of regional reference conditions, utilization of standard data collection procedures, and identification of non-redundant community attributes (i.e., metrics) that reliably respond to human disturbance (Karr and Chu 1999, Whittier et al. 2007). The index of biological integrity (IBI), a multi-metric indicator originally developed to assess the condition of rivers and streams (Karr 1981), has been successfully adapted to a variety of aquatic and terrestrial habitats, including wetlands.

The MPCA began developing IBIs for wetlands in the early 1990s, focusing on depressional marshes and ponds. During this work, attributes of the aquatic plant and macroinvertebrate (aquatic insects, snails, leeches, and crustaceans) communities were investigated to determine their response pattern along a gradient of human disturbance. These efforts resulted in the development and validation of ecoregion-specific, wetland IBIs (Appendix B). The first two DWQA surveys utilized plant and macroinvertebrate IBIs as indicators of wetland condition. Beginning with the 2017 survey, after identifying deficiencies with the plant IBI, the decision was made to switch to the Floristic Quality Assessment (FQA)—the primary indicator of the MWCA—for assessing condition of depressional wetland plant communities. However, following the decision to focus the DWQA on SOW communities and acknowledging some limitations of the FQA method in this community type, a hiatus on plant community monitoring was taken in the 2023 DWQA survey. This time was instead used to refine sampling methodology and improve performance of the FQA method in the SOW community for use in future DWQA surveys.

Similar to a medical professional evaluating human health by measuring body temperature, blood sugar, cholesterol and other parameters, the DWQA includes measurements of some key parameters to help

What's new in the DWQA?

- **Survey focused on shallow open water community rather than entire wetland basin.** Previous surveys utilized a whole-basin approach where all plant community types within the extent of the depressional wetland (i.e., to the upland edge) were evaluated, which frequently created a discrepancy between the two biological indicators. For example, macroinvertebrate indicator results are primarily applicable to the shallow open water and deep marsh community types, whereas plant indicators are applicable to these community types as well as other community types also present within the basin. When other community types are present there is the potential for discrepant indicator results that stem from this difference in assessment area. Shifting the focus of both indicators to the shallow open water community eliminates this issue and reduces redundancy with the MWCA (see Appendix A for details).
- **Reporting units based on wetland area (acres), no longer reporting based on number of wetlands.** The reason for this transition is twofold. First, with the focus now on the shallow open water community it is no longer appropriate to think of study sites as discrete water bodies, as there can be multiple 'pockets' of open water within a wetland basin (Figure 2C). Focusing on area-based results reported in acres also facilitates comparisons among Minnesota's three wetland surveys: DWQA, MWCA, and WSTMP.
- **2023 survey limited to macroinvertebrate and water chemistry results.** Following the 2017 DWQA, significant limitations were identified in FQA methodology for evaluating the ecological integrity of shallow open water communities. Initial plans were to address these concerns between DWQA surveys through additional data collection and analysis, however, the disruption to the field monitoring schedule caused by COVID-19 prevented this work from being completed in time for the 2023 DWQA survey. It is anticipated that the FQA method will be revised and ready to implement in time for the next iteration of the DWQA survey.
- **Area categories changed to < 1, 1-5, and > 5 <u>acres</u>.** Previous DWQA surveys based on the whole-basin approach utilized area categories of < 1, 1-5, and > 5 <u>hectares</u> (1 hectare = 2.47 acres). Changing the focus of the survey to SOW communities resulted in a decrease in mean site area and required a reevaluation of the existing area categories. Simply switching units from hectares to acres and keeping the same numerical cut-offs yielded a distribution of sites/area category that approximated those of past surveys. Thus, it is not appropriate to make comparisons by area category across surveys as they apply to different things: basin area vs SOW area. Reporting by area category is limited to the current 2023 SOW results.

diagnose why some wetlands in the survey are in poor condition. Several water quality parameters were selected based on their potential to impact wetland community integrity. By monitoring these 'stressors', their relationship with the biological communities could be explored through a relative risk analysis. A relative risk analysis provides an estimate of the likelihood that a biological community will be in poor condition when elevated levels of a stressor are present. For instance, a relative risk estimate of two indicates that the probability of having a poor biological community is twice as likely when stressor levels are elevated compared to when stressor levels are low. Having an estimate of how often a stressor is elevated, in addition to its impact on biological communities, provides a better understanding of its relative importance within the population.

Dragonflies begin their lifecycle as nymphs in aquatic environments where they are predators of other aquatic macroinvertebrates.

Methods

Since its inception the DWQA has focused on semi-permanently to permanently flooded depressional wetlands; wetlands that occupy areas of low relief or depressions on the landscape. In the current round of the DWQA, the focus has been sharpened to just the open water portion of depressional wetlands (see Figure 2 examples). Considering this change, past survey data were re-analyzed to produce results in terms of the acreage of SOW communities (see Appendix A for details). Re-analysis of past results was not possible for plant community data due to ongoing method revisions for evaluating SOW condition.

Ponds are included in the DWQA's definition of depressional wetlands. Despite not having a universal definition, ponds are generally regarded as small, shallow waterbodies. A recent summary of the scientific literature by Richardson et al. (2022) has resulted in an evidence-based definition as follows:

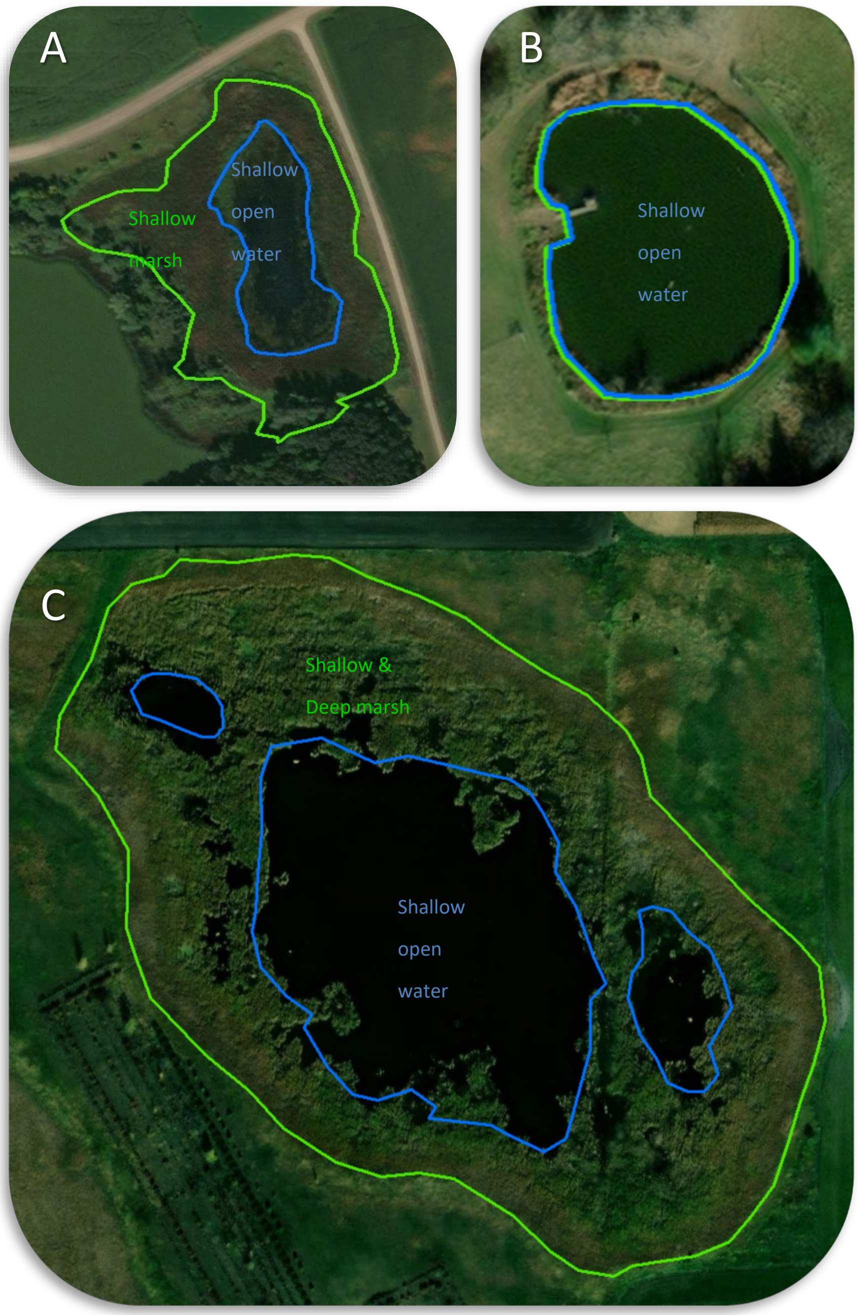
Ponds are small and shallow waterbodies with a maximum surface area of 5 ha, a maximum depth of 5 m, and < 30% coverage of emergent vegetation. Ponds will have light penetration to the sediments if water clarity permits and can be permanent or temporary and natural or human-made.

For the purposes of the DWQA this definition now provides guidance for distinguishing ponds, which are included in the survey, from shallow lakes, which are not. For generating the map from which survey sites are randomly selected, this definition was recognized by including areas mapped according to the National Wetland Inventory (NWI) as lacustrine-littoral aquatic bed only when those areas were not in association with the deepwater lacustrine-limnetic class. While this recognizes the depth/light penetration criteria of the definition, it can result in waterbodies greater than 12 acres being included on the site selection map. Potential survey sites (i.e., those randomly selected for monitoring) were scrutinized a step further to screen-out shallow lakes, examining MN DNR's lake database to determine depth and fish community characteristics (if available) as well as conducting a site visit to gain insight on whether the waterbody supports recreational activities (e.g., presence of docks or boat ramps).

Based on an evaluation of site characteristics as well as aerial imagery, potential survey sites were classified as either natural or human-made. Examples of human-made basins include stormwater retention ponds (e.g., Figure 3), golf course water hazards, livestock ponds, and ornamental ponds. If a survey site represented a restored wetland or one created for mitigation purposes, it was classified as natural since it's intended to replace wetland habitat that had been drained, filled, or physically altered.

Waterbodies that require pumping or have an artificial liner to maintain their hydrology (e.g., waste stabilization ponds) were not included in this survey.

Figure 2. Examples illustrating the difference in site boundaries between whole-basin approach (green line) of previous surveys and shallow open water approach (blue line) of current and future surveys: A) depiction of site area reduction due to exclusion of shallow marsh; B) depiction of pond showing no change in site boundary; and C) depiction of site area reduction and delineation of three independent sites due to presence of multiple pockets of shallow open water within the previous basin.



The DWQA utilizes Level II ecoregions (Omernik 1987, White and Omernik 2007) as a geographical framework to improve the ability of indicators to distinguish human disturbance from natural variability. Three major ecoregions converge in Minnesota. The Temperate Prairies (TP) occupies the western and southern portions, the Mixed Wood Plains (MWP) occupies the central and southeastern portions, and the Mixed Wood Shield (MWS) occupies the north central and northeastern portion of the state (Figure 4). The first DWQA included all three ecoregions (Genet 2012). However, due to the relative scarcity of depressional wetlands—as defined by the DWQA—in the MWS compared to the overall wetland resource in this ecoregion as well as wetland classification issues presented by bogs and fens in this region, the MWS ecoregion has been excluded from the DWQA. Wetland condition estimates for the MWS ecoregion are included in the MWCA. Throughout the remainder of this report, combined results from the MWP and TP ecoregions will be referred to as 'statewide' even though the MWS ecoregion is excluded.

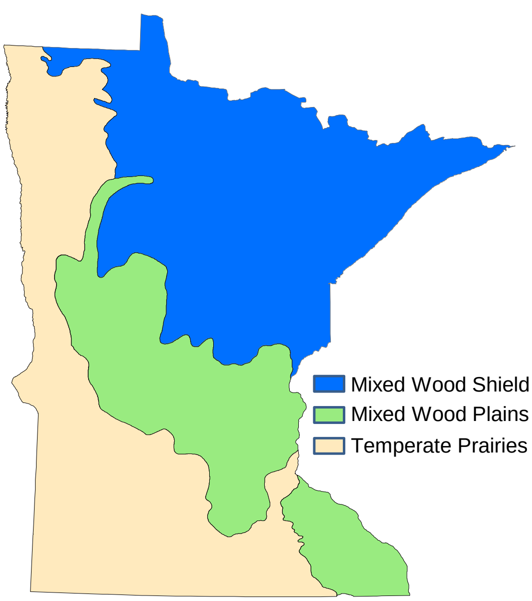
Figure 3. Stormwater retention pond in Scott County.

Survey design

Similar to how an opinion poll gauges public interest on a topic or candidate running for office, the DWQA utilized survey techniques that allow estimates (± margin of error) to be generated for an entire population of wetlands by monitoring a comparatively small sample of wetlands. Wetland monitoring locations were randomly selected to ensure that derived estimates were unbiased. In addition, the selection process was spatially stratified (Stevens and Olsen 2004) to increase the likelihood that the sample represents all regions of the state.

Monitoring locations were randomly selected from the photo-interpreted wetland "maps" in the 1 mi^2 plots of Minnesota's WSTMP (Kloiber and Norris 2013). Since the plots also represent a sample (i.e., ~5,000 plots randomly selected throughout the state), the DWQA survey design is considered a two-phase sampling approach. The baseline DWQA followed a rotating ecoregion schedule: 2007-Mixed Wood Plains, 2008-Temperate Prairies, 2009-Mixed Wood Shield. This approach required three years to obtain complete statewide coverage. Alternatively, the 2012, 2017, and 2023 DWQA were limited to the MWP and TP ecoregions (i.e., the DWQA's new definition of statewide) where both ecoregions were monitored in the same year, eliminating any confounding effects from interannual variability (i.e., wet vs. dry years) and the need for annual sampling at a subset of sites ('annual sites' in the baseline report).

Figure 4. Level II ecoregions in Minnesota.



The target number of 2023 DWQA monitoring sites was 100—split evenly by ecoregion with a 50% revisit rate of 2017 survey sites (i.e., 50 new sites/50 revisit 2017 sites). Unequal probability weighting was used in the random selection process to increase the likelihood of obtaining an equal number of sites in each of three wetland area categories: < 1 acres, 1 - 5 acres, and > 5 acres. Note: these categories have been revised in the 2023 DWQA to reflect the relatively smaller SOW area (i.e., compared to the area of the basins in which they occur; Figure 2); past DWQA surveys utilized the same numerical breakpoints but were based on hectares rather than acres (1 hectare = 2.47 acres). The site selection process was conducted by the U.S. Environmental Protection Agency (US EPA) National Health and Environmental Effects Research Laboratory located in Corvallis, Oregon.

Field methods

Prior to monitoring, each potential survey site was investigated using geographic information system (GIS) software to determine ownership and map out access routes. To maximize participation in the survey, landowners were contacted weeks in advance to obtain permission and/or the necessary permits. The sequence of contacting landowners was as follows: 1) contact by phone; 2) if unable to reach via phone, visit residence during site reconnaissance; 3) if not home during visit, mail letter requesting permission. If permission was granted, sites were visited in May to evaluate whether they met specifications of the DWQA (i.e., SOW community in depressional wetland) as well as their origin (human-made vs. natural). If a site was dropped from the survey for any reason (e.g., landowner access denial, absence of SOW community), a replacement site was added in sequential order from a list of randomly selected sites until the desired sample size of 50 sites/ecoregion was achieved. Site replacement considered ecoregion and whether a site was new or a revisit, but not area category. For example, if access permission was denied for a small (< 1 acre) revisit site in the MWP, a replacement site came from the MWP revisit overdraw list and could belong to any area category. More information on MPCA's site evaluation procedure can be found here: Procedure for evaluating wetland biological monitoring sample sites (state.mn.us)

Aquatic macroinvertebrates were sampled at each survey site in June using a D-frame dip net with a 500 µm mesh size. Macroinvertebrate samples were primarily collected at the boundary between the shallow open water and deep marsh communities, in depths ranging from 0.3 – 1 m. At this location macroinvertebrate habitat typically includes submerged vegetation, submerged portions of emergent and floating-leaved vegetation, and plant litter in various stages of decomposition. Samples were collected by sweeping the dip net through the water column over a horizontal distance of approximately 1 m (Figure 5A). Three to five sweeps at various locations within the wetland (typically within a 15 m radius) were collected and placed on hardware cloth screen (1.3 x 1.3 cm mesh) overlaying two plastic pans to separate macroinvertebrates from the vegetation that invariably gets caught in the net. Over a period of ten minutes, macroinvertebrates were 'picked' using forceps and placed directly into the pans while also spreading out the vegetation/detritus on the hardware cloth to facilitate macroinvertebrates falling into the pans below (Figure 5B). After ten minutes the vegetation was removed from the hardware cloth and a second series of dip net sweeps were collected and placed on the cleared screen. The ten-minute 'picking' process was repeated, after which the vegetation/detritus was discarded and the contents of the plastic pans were consolidated into one 16-ounce plastic jar and preserved with 95% ethanol. This dip net method was performed by both members of the sampling crew resulting in the collection of two macroinvertebrate samples/site. After all survey sites were monitored, samples were shipped to a lab for taxonomic identification. More information on MPCA's dip net method can be found here: Macroinvertebrate community sampling protocol for depressional wetland monitoring sites (state.mn.us)

Chemical and physical properties of the water column were measured during the macroinvertebrate monitoring visit. A multi-parameter sonde (YSI ProDSS) was used to measure temperature (^{o}C), dissolved oxygen concentration (mg/L), specific conductance (µS/cm), and pH just below the water surface. Water samples were collected at each site just below the water surface and packed in ice until they could be delivered to the Minnesota Department of Health Environmental Laboratory for analysis. The concentration of total phosphorus (mg/L), Kjeldahl nitrogen (mg/L), nitrate + nitrite (mg/L), chloride (mg/L), sulfate (mg/L), and alkalinity (mg/L) was determined for each sample using standard protocols (Appendix C). Water column transparency or clarity was measured using a 100 cm Secchi tube. Details of the water chemistry sampling procedure can be found here: [Water quality chemistry assessment protocol for wetland monitoring sites (state.mn.us)]

Figure 5. Aquatic macroinvertebrate sample collection is a two-step process involving (A) dip nets to collect organisms and vegetative material, and (B) hardware cloth with pans underneath to separate collected macroinvertebrates from the vegetation and detritus.

Data from both macroinvertebrate samples collected at each site were combined to determine metric values. Each metric value was standardized to a 0-10 score based on the range of metric values (see Appendix B for equations). Distinct macroinvertebrate IBIs were used to characterize condition for wetlands in the MWP and TP ecoregions with ten and eight metrics, respectively. IBIs were standardized to a 0-100 scale where a score of 100 represents minimally impacted conditions.

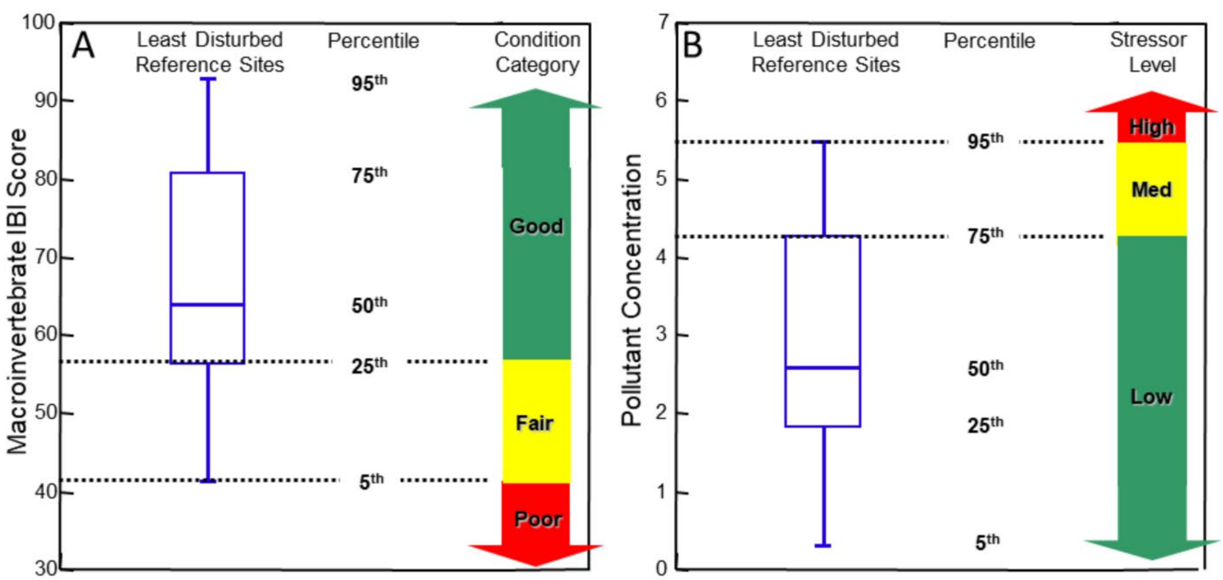
Macroinvertebrate condition and water quality data collected from reference sites—selected independently from the DWQA survey—were used to represent the range of expected values for least-impacted conditions within each ecoregion. The distribution of each data set was used to establish criteria for categorizing condition as either good, fair, or poor, or stress (i.e., pollutant concentrations) as either high, medium, or low (Figure 6A & B). The 25th percentile of the reference distribution was used to distinguish good from fair condition categories. Sites with IBI values above this threshold are comparable to least-impacted reference sites (Figure 6A). The 5th percentile of the distribution of least-impacted scores was used to distinguish fair from poor condition categories, meaning that sites categorized as poor are in worse condition than 95% of the least-impacted reference sites. Specific values for each of the thresholds used for categorizing condition and stressor levels can be found in Appendix D. Reference site selection criteria can be found in the baseline DWQA report (Genet 2012) and in Genet et al. (2004).

Estimation procedures

Condition and stressor categorization criteria were used to rate indicator results individually for each survey site. Initial design weights were adjusted based on the exclusion of sites that were confirmed as 'non-target' (i.e., did not meet definition of SOW community) during site evaluation. Sites where access permission was denied were assumed to represent SOW communities and were included in analyses of wetland extent (i.e., numbers or acres). Indicator results from the random sample of sites were used in conjunction with the adjusted design weights to estimate the proportion of the population in each category.

All analyses were performed in R version 4.4.1 (R Core Team 2024) with the assistance of RStudio (2023.12.0) using the Spatial Sampling Design and Analysis package (spsurvey 5.5.0; Dumelle et al. 2023) and other associated packages. Analysis scripts were developed for estimating the overall extent of the population, the proportion within each condition category and stressor level, the relative risk posed by each of the measured stressors, and the amount of change that has occurred within the condition categories and stressor levels since previous surveys. These analyses were also performed for each subpopulation differentiated by ecoregion, wetland origin, wetland ownership, or wetland area. Relative risk was estimated using the ratio of the probability of poor condition at high stressor levels (numerator) to the probability of poor condition at low stressor levels (denominator) occurring in the population (Van Sickle and Paulsen 2008). A relative risk estimate statistically greater than one indicates that there is an increased likelihood of poor biological condition when a stressor level is high. Comparisons of condition/stressor levels between subpopulations (e.g., human-made vs. natural) were based on cumulative distribution function (CDF) tests (Dumelle et al. 2023) of the quantitative data for each indicator.

Figure 6. Generalized depiction of the distribution (represented as boxplots) of indicator values at reference sites and the process for using this information to categorize (A) macroinvertebrate condition and (B) stressor levels of each survey site. Sites were categorized independently based on each indicator.



Statewide results and discussion

Data collected in 2023 provide a snapshot of the current extent and condition of depressional wetlands and ponds in Minnesota. In addition to this status update, the new data set makes it possible to evaluate whether any changes have occurred relative to previous surveys. In this section, quantity status and change results are presented first followed by quality status and change for the shallow open water community of depressional wetlands in the combined MWP and TP ecoregions.

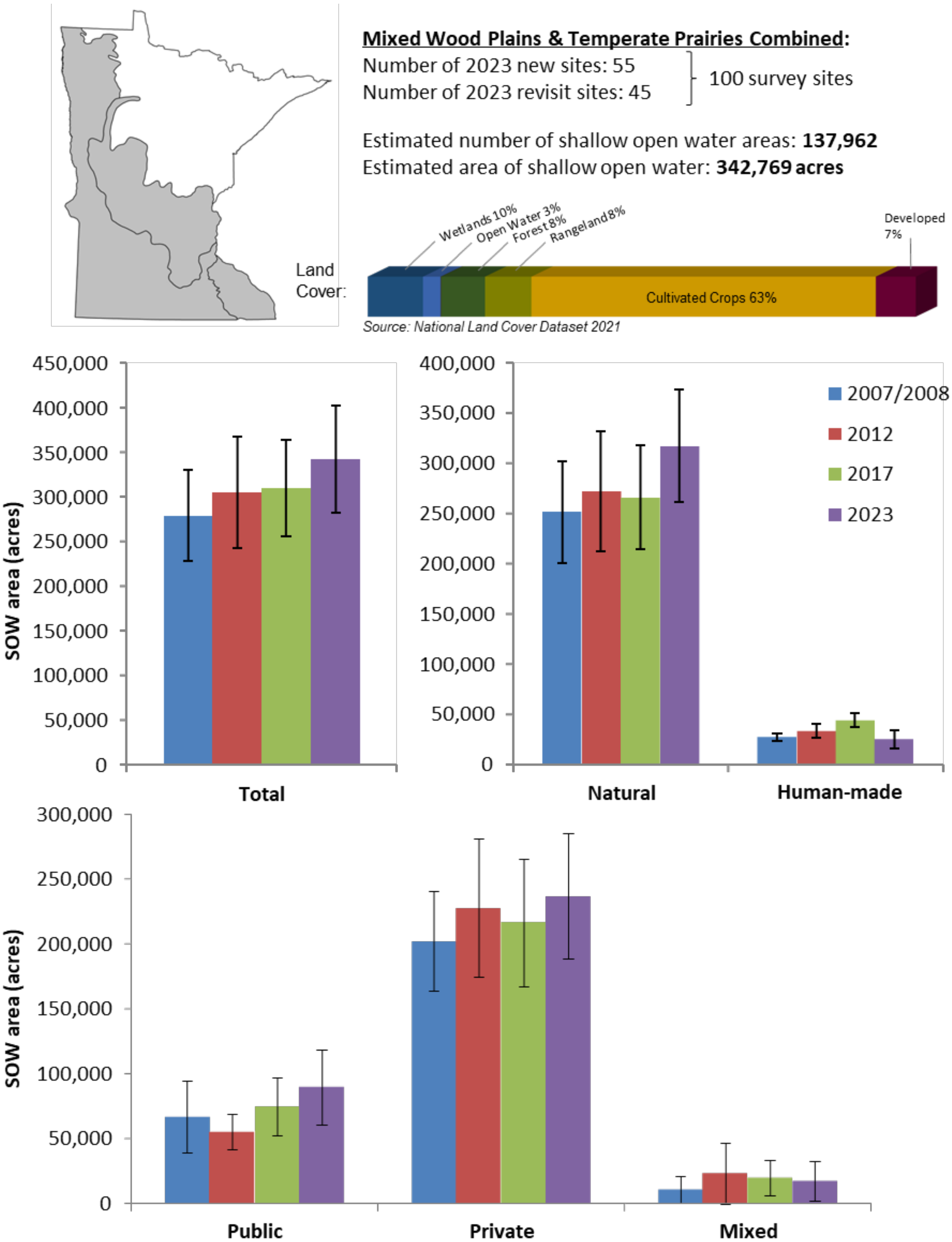
Depressional wetland quantity (status and change)

Based on the 2023 survey results there are an estimated 342,769 acres of SOW community associated with depressional wetlands and ponds occurring in the MWP and TP ecoregions. This estimate only pertains to semi-permanently to permanently flooded wetlands and ponds, and does not include SOW communities associated with temporarily or seasonally flooded wetland basins. Across the first four survey cycles, the total acreage of SOW appears to be increasing over time (Figure 7); however, there were no statistically significant differences between any of the survey cycles. Minnesota's wetland quantity survey (i.e., WTSMP; Kendig et al. 2024) has demonstrated net gains in unconsolidated bottom wetlands (approximately equivalent, along with aquatic bed wetlands, to the SOW community) across all iterations of that survey (2006 – 2020). The latest US FWS Wetland Status and Trends report indicates that this pattern is not unique to Minnesota. From 2009 to 2019, the area of palustrine ponds as well as non-vegetated freshwater wetlands in general increased across the conterminous US, consistent with previous Status and Trend surveys (Lang et al. 2024).

Not surprisingly, the area of naturally formed SOW communities is greater than the extent of those created by humans (e.g., ponds) across all years of the survey (Figure 7). Similarly, the area of SOW communities is consistently greater on private property compared to the area occurring on public land or on a combination of private and public property ('Mixed'). While similar patterns of increasing extent over time are apparent among the various sub-groups of SOW communities, it is premature to determine whether these patterns represent significant and meaningful trends.

The estimated number of distinct SOW areas in the combined MWP and TP ecoregions was 137,962. Dividing the estimated total SOW area by the estimated total number yields an average community area of 2.5 acres. In 2017, the average estimated area was 5.6 acres when the survey encompassed the entire wetland basin and included other community types (Genet et al. 2019). Shallow open water communities originating from human activity were smaller than those formed by natural processes, averaging 0.5 and 3.5 acres, respectively. Though ~5x less numerous, SOW communities on public land were larger (on average) than those located on private property, 4.4 and 2.1 acres respectively.

Figure 7. Estimates of the total area of shallow open water community in the Mixed Wood Plains and Temperate Prairies ecoregions over four cycles of the DWQA survey as well as area broken down by origin and property ownership. Bracketed lines represent the 95% confidence interval associated with each estimate.



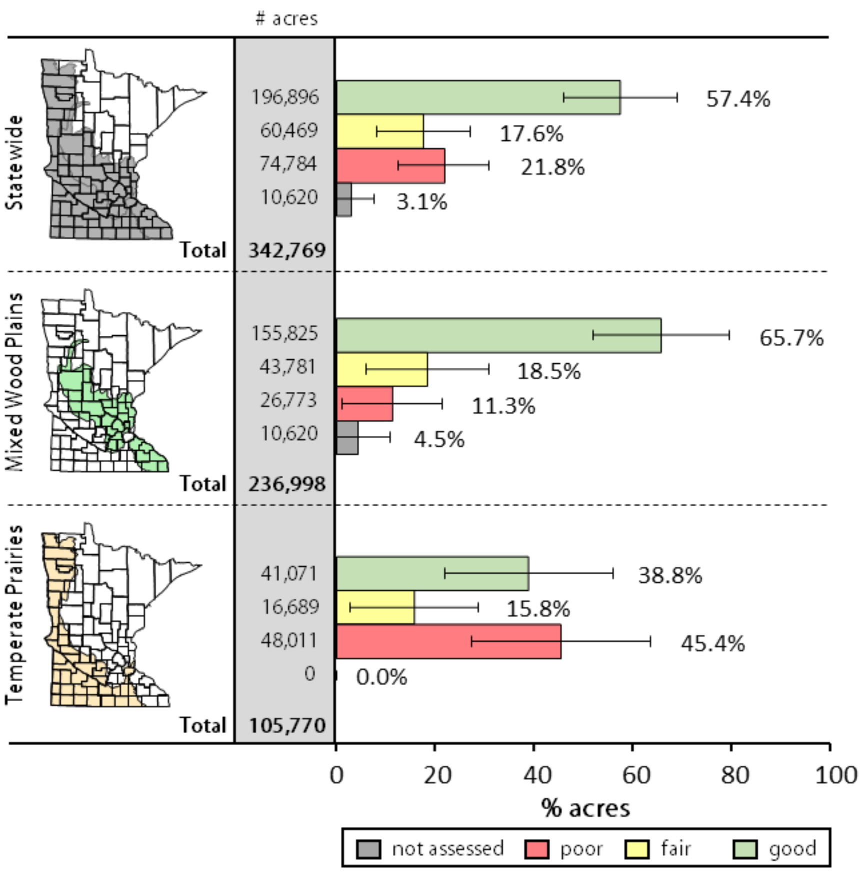
Depressional wetland condition

For the two ecoregions combined, aquatic macroinvertebrates are in primarily good condition within the shallow open water portion of depressional wetlands and ponds (Figure 8). This outcome is largely driven by results in the MWP ecoregion where approximately 66% of the SOW acres in that region are supporting a healthy macroinvertebrate community. On the other hand, the TP ecoregion exhibits a slightly higher percentage of macroinvertebrate communities in poor condition compared to those in good condition (Figure 8). A small number of survey sites in the MWP could not be safely sampled for macroinvertebrates due to the interaction of a fringing/floating mat of emergent vegetation and water depth in the SOW community, accounting for the 4.5% of SOW area that was not assessed.

Despite the focus of the 2023 DWQA on SOW communities—as opposed to all communities present within the wetland basin—the 2023 aquatic macroinvertebrate community results are similar to the results reported by acres in the 2017 DWQA report (i.e., Genet et al. 2019). For example, the breakdown of Good/Fair/Poor for the MWP was 66%/19%/11% and 56%/23%/16% in the 2023 and 2017 surveys, respectively. In the TP ecoregion, the results are also comparable with a 39%/16%/45% split in 2023 and a 40%/26%/35% in 2017. The discrepancies between the two approaches (SOW vs basin) represented by each survey period can likely be attributed to the handful of sites included in the 2017 results that no longer meet survey site criteria (i.e., they lack SOW) and were thus excluded from the 2023 results (i.e., changed to non-target). Henceforth, all subsequent comparisons between survey cycles in this report utilize revised estimates for past surveys where the current SOW-based approach has been implemented in the analyses (Appendix A).

Macroinvertebrate community condition did not vary significantly ($P > 0.05$) between natural and human-made depressional wetlands and ponds in 2023. Revised estimates of past survey results—based on SOW area—show that macroinvertebrate communities were healthier in natural wetlands during the 2007/2008 and 2012 surveys, but not in the 2017 survey. Comparisons based on property ownership (i.e., public vs private) reveals that macroinvertebrate condition is similar among these categories across all years of the DWQA.

Figure 8. Aquatic macroinvertebrate community condition in the shallow open water portion of depressional wetlands and ponds in 2023, estimated in terms of acreage. Bracketed lines represent the 95% confidence interval associated with each estimate. Values may not add up to 100% or corresponding totals due to rounding.



Wetland stressors

Water chemistry monitoring results from the 2023 DWQA indicate that concentrations of several parameters are elevated in comparison to relatively undisturbed, reference wetlands. Of the pollutants measured in this survey, chloride was most frequently found to be elevated (i.e., high) in the SOW community (Figure 9). Given the larger percentage of high concentrations in the TP ecoregion—a region with less urban land use and fewer major roadways—it is likely that road salt is not the only source of contamination for SOW communities. Other anthropogenic sources include livestock waste, water softening discharge, fertilizer (e.g., KCl), and municipal sewage effluent (Kelly et al. 2012). In addition to stormwater retention ponds, sample sites within or adjacent to large animal enclosures had some of the highest chloride concentrations. Chloride concentrations in surface waters also vary naturally due to surficial geology with the highest concentrations occurring in southwest Minnesota (Moyle 1956). In terms of chloride's impact on the health of aquatic macroinvertebrates, the evidence is not overwhelming. When the relative risk analysis is based on SOW area, there does not appear to be an elevated risk posed by chloride to macroinvertebrate community condition (Figure 10). In fact, revised estimates from past surveys for the two ecoregions combined reveals that only in 2017 does chloride pose a statistically significant risk to the macroinvertebrates. Chloride also poses a significant risk when data across all surveys are combined into one relative risk analysis, but it has the lowest estimated risk

(1.9) among the stressors monitored in this survey. In summary, while chloride was most frequently measured above concentrations typical of least-disturbed reference wetlands, it does not appear that those concentrations (> 7.9 mg/L MWP, > 8.6 mg/L TP) are particularly detrimental to aquatic macroinvertebrates.

For reference, the water quality standard for the protection of aquatic life in surface waters (including wetlands) is a chloride concentration of 230 mg/L. In 2023, only two survey sites—one in each ecoregion—had concentrations that exceeded Minnesota's water quality standard, resulting in an estimated <1% of SOW area that exceed this concentration. In the MWP site, the aquatic macroinvertebrate community had a condition rating of "fair" while the TP site was rated as "poor", evidence that these concentrations may be detrimental to certain macroinvertebrates. It should be noted however, that a single water sample may not accurately characterize a water body's prevailing conditions. Statewide, chloride concentrations were significantly higher in human-made SOW communities compared to those of natural origin based on the CDF test ($F = 5.29$, $df_1 = 2$, $df_2 = 97$, $p = 0.007$). This finding was also observed in the 2017 data set, but not in the first two cycles of the DWQA.

Sulfate (SO_4) concentrations were significantly higher in the TP ecoregion (Figure 9; $F = 13.69$, $df_1 = 2$, $df_2 = 97$, $p < 0.001$), a pattern observed in all previous DWQA surveys as well. The observed difference in sulfate concentrations between the two ecoregions is in part due to natural gradients in surficial geology and rates of evaporation. Sulfate concentrations are typically highest in southwest Minnesota where waters occur in glacial drift rich in gypsum (calcium sulfate dihydrate) underlain by Cretaceous sedimentary rocks rich in iron sulfide and

Figure 9. Stressor levels in Minnesota's depressional wetlands and ponds. Bracketed lines represent width of 95% confidence interval associated with each estimate. Asterisks indicate significant differences in stressor levels between ecoregions according to CDF test. Not all percentages add to 100% due to exclusion of 'not assessed' category on graphs (e.g., Transparency).

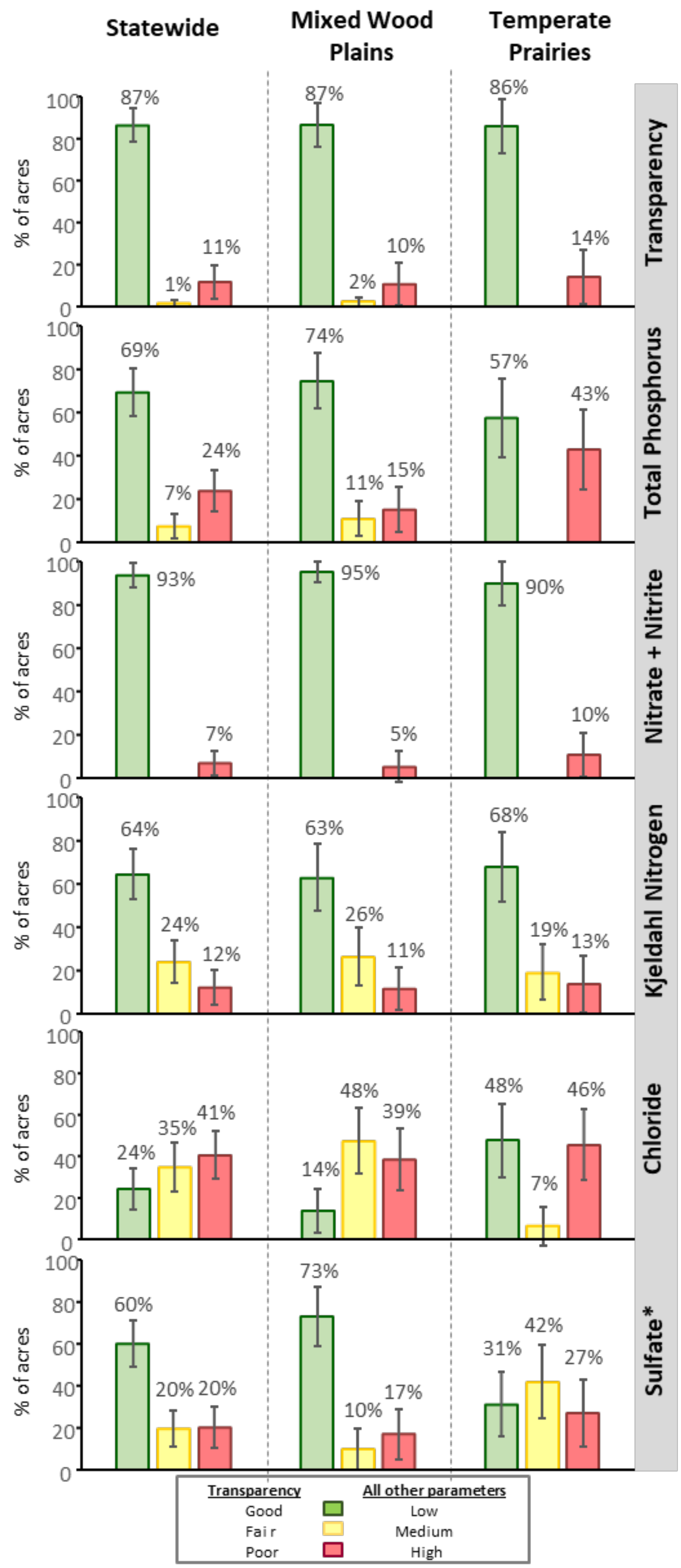
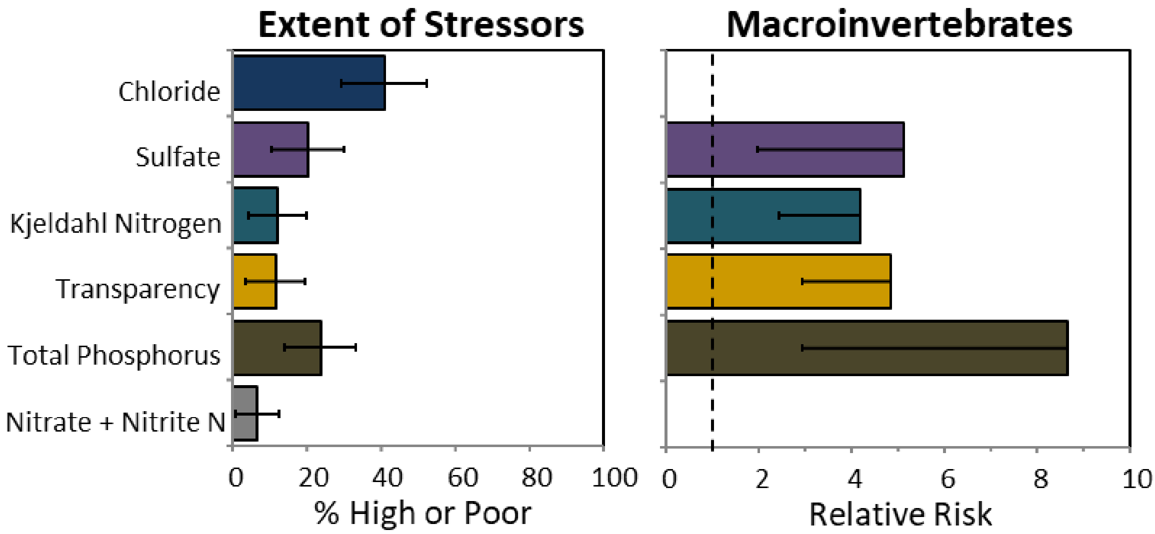


Figure 10. Extent of stressors and their relative risk to macroinvertebrate communities in Minnesota depressional wetlands and ponds. Bracketed lines represent 95% confidence intervals (for % estimates) or lower confidence limits (for relative risk estimates). A stressor without an associated bar on the relative risk graph indicates that it did not pose an elevated risk (i.e., not significantly greater than 1).



where evaporation from open water surfaces is high, concentrating any salts that are present (Moyle 1956, Gorham et al. 1983). High sulfate concentrations posed an elevated risk to aquatic macroinvertebrate communities in 2023 (Figure 10). In fact, re-analyses of past survey results based on SOW area reveals that sulfate is the most impactful stressor being monitored by the DWQA, representing a significant risk to macroinvertebrates in every iteration of the survey when the two ecoregions are combined as well as analyses of ecoregions individually. This is particularly interesting when considering the large difference between the two ecoregions in the criteria used for rating a site as having a high sulfate concentration: > 12.5 mg/L in the MWP and > 127.4 mg/L in the TP. This suggests that concentrations much lower than 127.4 mg/L may be detrimental to the macroinvertebrate community. A possible explanation may be related to lower calcium and magnesium concentrations in surface waters of the MWP—compared to the TP ecoregion—as demonstrated at the county level by Gorham et al. (1983). Sulfate toxicity has been shown to generally increase as Ca^{2+} and Mg^{2+} concentrations decrease (Elphick et al. 2011, Erickson et al. 2022). However, Karjalainen et al. (2023) calculated an HC5 (i.e., concentration hazardous for 5% of species) of 117-194 mg/L for sulfate in soft waters (i.e., low Ca/Mg conc.), a range of values predominantly higher than criteria used to rate sulfate concentrations as 'high' in the DWQA regardless of ecoregion (Appendix D). Though, in addition to aquatic macroinvertebrates, Karjalainen et al. (2023) included fish, algae, rotifers, and aquatic plants in their analysis of sulfate toxicity.

Ions, such as sulfate and chloride, are toxic to aquatic organisms due to the disruption they cause in osmoregulation or the organism's ability to maintain a balance of water and electrolytes relative to its aqueous environment. However, relatively little is known regarding sulfate uptake in freshwater insects as well as in aquatic organisms in general. Recent studies have demonstrated that ionic composition may be just as important as overall ionic strength when it comes to toxicity. Scheibener et al. (2017) found that increasing sodium (Na) concentrations decreased the lethal impacts of sulfate. Similarly, Kunz et al. (2013) demonstrated that water with a high SO_4:Na ratio was toxic to mayflies while water with a low SO_4:Na ratio was not at a comparable conductivity. Furthermore, a field study of Ohio streams examining the impacts of chloride and sulfate on aquatic macroinvertebrates demonstrated a response pattern suggesting that sulfate was more broadly toxic to taxa while the response to chloride was mixed, with some taxa responding positively to increasing Cl concentrations and others responding negatively (Miltner 2021). The relative risk analyses over the past four cycles of the DWQA align with these results, with sulfate consistently exhibiting an elevated risk to macroinvertebrates while chloride only does so infrequently.

Sulfate (SO_4) naturally occurs in lakes, streams, and wetlands, and in areas of these water bodies that lack oxygen, sulfate may be reduced to hydrogen sulfide (H_2S) through microbial decomposition. Natural concentrations of sulfate in water bodies may be increased via discharge from industrial facilities, mining operations, and wastewater treatment plants, fossil fuel combustion via atmospheric deposition, as well as contaminated runoff from surrounding land. Given that wetlands frequently lack oxygen—either temporally or spatially—it cannot be ascertained based on the data collected in this survey whether sulfate, hydrogen sulfide, or both are impacting the aquatic macroinvertebrates in SOW communities.

Predaceous diving beetles (Family Dytiscidae) are somewhat unique inhabitants of shallow open water communities in that both the adult (on left) and larval (on right) life stages can be found there together.

Total phosphorus concentrations were high in 24% of SOW acres across the two ecoregions (Figure 9) and represented a significant risk to aquatic macroinvertebrates in the 2023 survey (Figure 10). Similar to sulfate, it was also determined to pose an elevated risk to macroinvertebrates in each of the past three surveys when that analysis is based on area. Total phosphorus concentrations and transparency of the water column are frequently correlated in surface waters. In the 2023 survey, however, the occurrence of poor water clarity is consistently lower than the percentage of SOW acres with high phosphorus (Figure 9), and poor transparency only represents an elevated risk to macroinvertebrates in 2017 and 2023. There are two main pathways in which total phosphorus and water clarity are related in surface water. First, water clarity may be reduced through sediment input (e.g., erosion) into a wetland, which increases total phosphorus concentrations due to its adsorption to sediment particles (Neely and Baker 1989). In the second pathway, increased phosphorus concentrations can reduce water clarity through the stimulation of phytoplankton growth. Phytoplankton can flourish to an even greater extent when fish (e.g., fathead minnows) are present in the basin and consume large quantities of zooplankton—a primary consumer of phytoplankton (Zimmer et al. 2001). In either case, the reduction in water clarity can lead to decreases in aquatic plant density and diversity that also impacts aquatic macroinvertebrates through reduced habitat complexity and numerous other pathways.

Statewide, total phosphorus concentrations were significantly higher in SOW communities located on public land compared to those on private property ($F = 4.11$, $df_1 = 2$, $df_2 = 97$, $p = 0.019$). Large (> 5 ac) SOW communities had significantly lower total phosphorus concentrations compared to the 1-5 ac and < 1 ac area categories ($p < 0.05$). This result is consistent with other studies that have demonstrated a decreasing trend in total phosphorus as surface area and/or depth of a water body increases (e.g., Richardson et al. 2022). The concentration of total phosphorus did not differ between natural SOW communities and those of human origin ($p > 0.05$).

The combined concentration of nitrate (NO_3) and nitrite (NO_2) was predominantly below method detection limits (i.e., > 0.05 mg/L = 'Low') in the SOW portion of depressional wetlands and ponds across the two ecoregions (Figure 9). Given this low frequency of detection, it is not surprising that it did not pose an elevated risk to the aquatic macroinvertebrate community in 2023 (Figure 10). Across the two ecoregions, nitrate and nitrite posed an elevated risk to macroinvertebrates in the first and second iterations of the DWQA when the analysis was revised on SOW area. Nitrate and nitrite contamination of surface waters occurs primarily as a result of runoff from cropland with wastewater treatment plants, septic and urban runoff, forests, and atmospheric deposition representing smaller sources. Groundwater discharge may also contribute to elevated nitrate and nitrite concentrations in wetlands located in predominantly agricultural landscapes. Hence, the amount of nitrate/nitrite entering a depressional wetland or pond is largely dependent on the following factors: 1) nitrate/nitrite concentrations in the surrounding upland soil; 2) the number and volume of surface water or drain tile inlets; and 3) precipitation. However, nitrate and nitrite are readily removed from wetlands through denitrification, where microbes utilize nitrogen for respiration in an environment that lacks oxygen (Neely and Baker 1989). Therefore, concentrations in wetlands fluctuate widely depending on the timing and magnitude of rainfall events and fertilizer application.

Across all of the survey's monitoring locations, this time-sensitivity to such factors introduces a large amount of variability into measurements that are intended to provide an indication of stress caused by surrounding land use practices. Obviously, in the case of nitrate plus nitrite, this is difficult to accomplish based on a single water sample. Examination of weekly rainfall maps for June 2023—when water samples were collected—indicates that across the two ecoregions relatively little precipitation occurred (https://www.dnr.state.mn.us/climate/weekmap/weekmap.html), resulting in most of the region being rated as abnormally dry or in some stage of drought by June 20th (https://droughtmonitor.unl.edu/data/png/20230620/20230620_mn_trd.png). Thus, the relatively low rates of nitrate/nitrite detection that were observed in depressional wetlands and ponds across the two ecoregions in 2023 it not surprising.

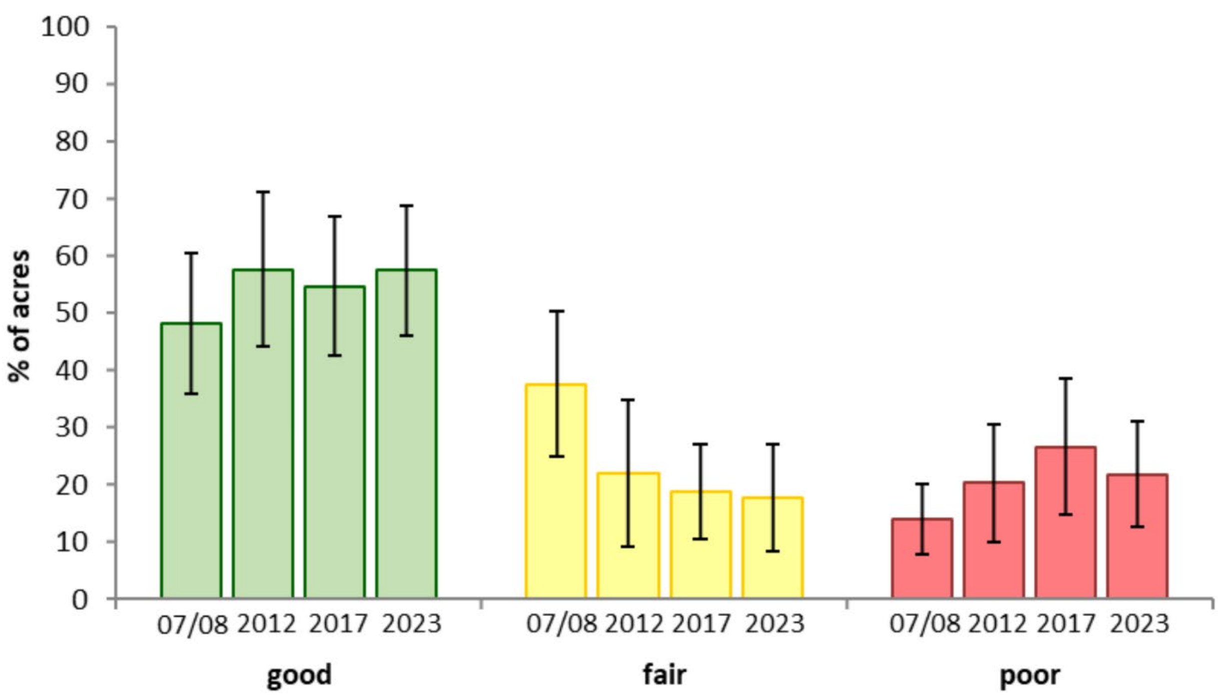
Total Kjeldahl Nitrogen or TKN is the sum of nitrogen bound in organic substances and ammonia/ammonium, which comprises total nitrogen when nitrate and nitrite concentrations are included. Elevated TKN concentrations in surfaces waters can be due to input from wastewater treatment plants, failing septic systems, runoff from animal manure and storage areas, and certain industrial dischargers. Concentrations of TKN were elevated in 12% of depressional wetlands and ponds in 2023 (Figure 9) and posed a significant risk to macroinvertebrate communities across the two ecoregions (Figure 10). The re-analysis of past surveys based on SOW area revealed that TKN posed an elevated risk to macroinvertebrates in the last three survey cycles. In 2023, several sites were located within pastures and had some of the highest TKN concentrations, evidence of manure contamination within the shallow open water community. Among the various comparisons, the only significant difference in TKN was observed among the area categories where < 1 ac SOW communities had a higher concentration than did > 5 ac SOW communities.

Changes in depressional wetland condition

Macroinvertebrate community condition and stressor levels were evaluated to determine whether they are changing over time. A few more iterations of the DWQA will be required before trends can be confidently evaluated, a primary goal of this status and trends survey. Appendix E includes the change analysis results for the combined ecoregions and each subpopulation therein, excluding mixed landowner sites due to small sample sizes in that category. A summary of those results is provided here.

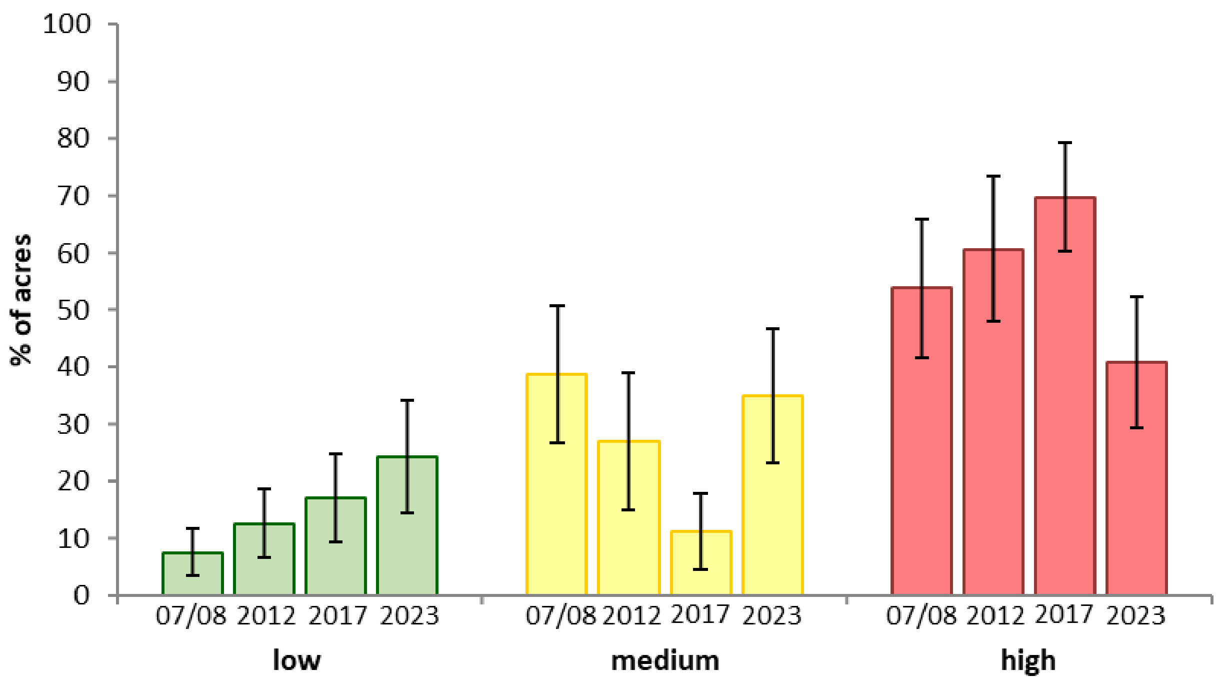
Macroinvertebrate community condition does not exhibit any statistically significant changes in the percentage of good or poor categories over time (Figure 11 & Appendix E). While not statistically significant, the overall pattern of decreasing % fair coupled with a pattern of increasing % poor over time is concerning and suggests that macroinvertebrate community condition may be degrading. Future surveys will elucidate whether this pattern represents a significant trend or merely reflects annual variability. Focusing on natural wetlands, macroinvertebrate health did vary between survey cycles with the 2007/2008 survey exhibiting a significant lower % poor than either the 2017 and 2023 surveys (Appendix E). Among SOW communities of human origin, the only significant change in macroinvertebrate condition occurred between the first and third DWQA surveys where the poor category declined by 32%.

Figure 11. Comparison of macroinvertebrate community condition over four cycles of the DWQA survey. Bracketed lines represent the width of the 95% confidence interval associated with each estimate. A small percentage (<5%) was not evaluated for macroinvertebrate condition in 2007/08 and 2023 (results not shown).



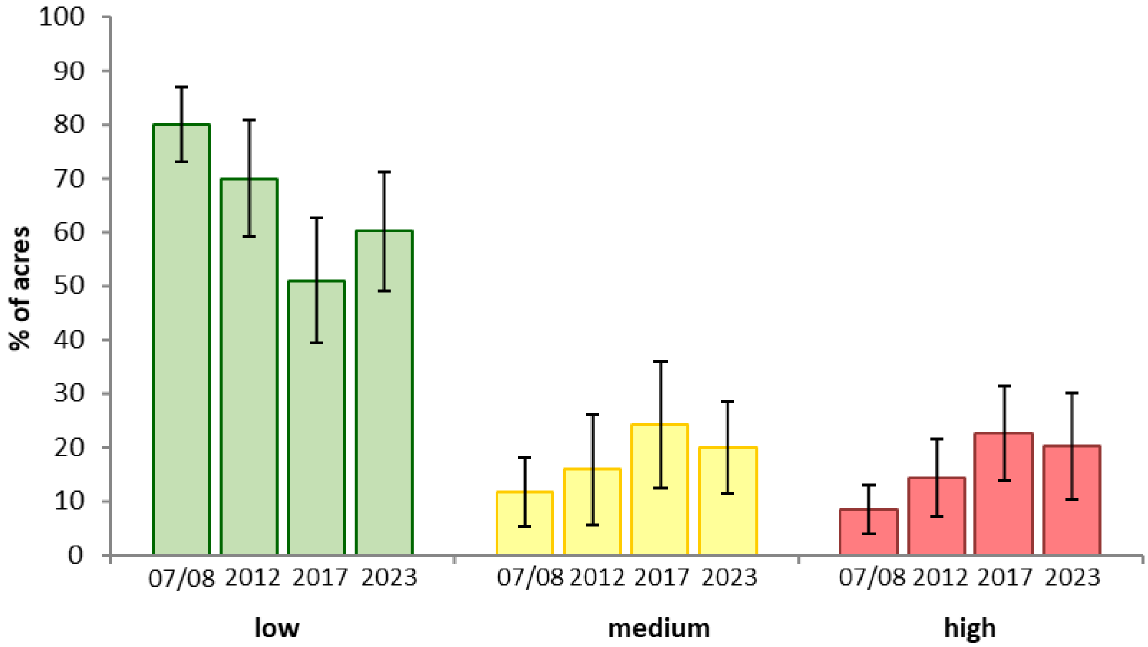
Among the water chemistry variables measured in the DWQA, only chloride and sulfate exhibited any relatively consistent patterns of change over time. Chloride concentrations were significantly lower across the two ecoregions in 2023 compared to the previous three surveys (Figure 12), largely mirroring the pattern observed among natural wetlands and those occurring on private property (Appendix E).

Figure 12. Comparison of chloride concentration categories over four cycles of the DWQA survey. Bracketed lines represent the width of the 95% confidence interval associated with each estimate. A small percentage (<5%) was not evaluated for sulfate concentrations in 2017 (results not shown).



Sulfate concentrations exhibited the opposite pattern with significantly higher concentrations in 2017 and 2023 compared to the 2007/2008 survey (Figure 13), reflective of the pattern observed among natural wetlands regardless of ownership category (Appendix E). These results are consistent with recent trends observed across the Midwest. Hinckley and Driscoll (2022) demonstrate net increases in sulfur (S) across the Midwest beginning around 2005 that are driven by an increasing trend in the application of S-containing fertilizers outpacing declines in atmospheric S deposition. Air quality regulations such as the Clean Air Act have reduced the free supply of S in the US and elsewhere, requiring increased application of S fertilizers to meet plant growth requirements. SOW communities adjacent to or near row crops such as corn and soybeans may therefore be more likely to exhibit increasing sulfate concentrations compared to those surrounded by other land use types.

Figure 13. Comparison of sulfate concentration categories over four cycles of the DWQA survey. Bracketed lines represent the width of the 95% confidence interval associated with each estimate. A small percentage (<5%) was not evaluated for sulfate concentrations in 2017 (results not shown).



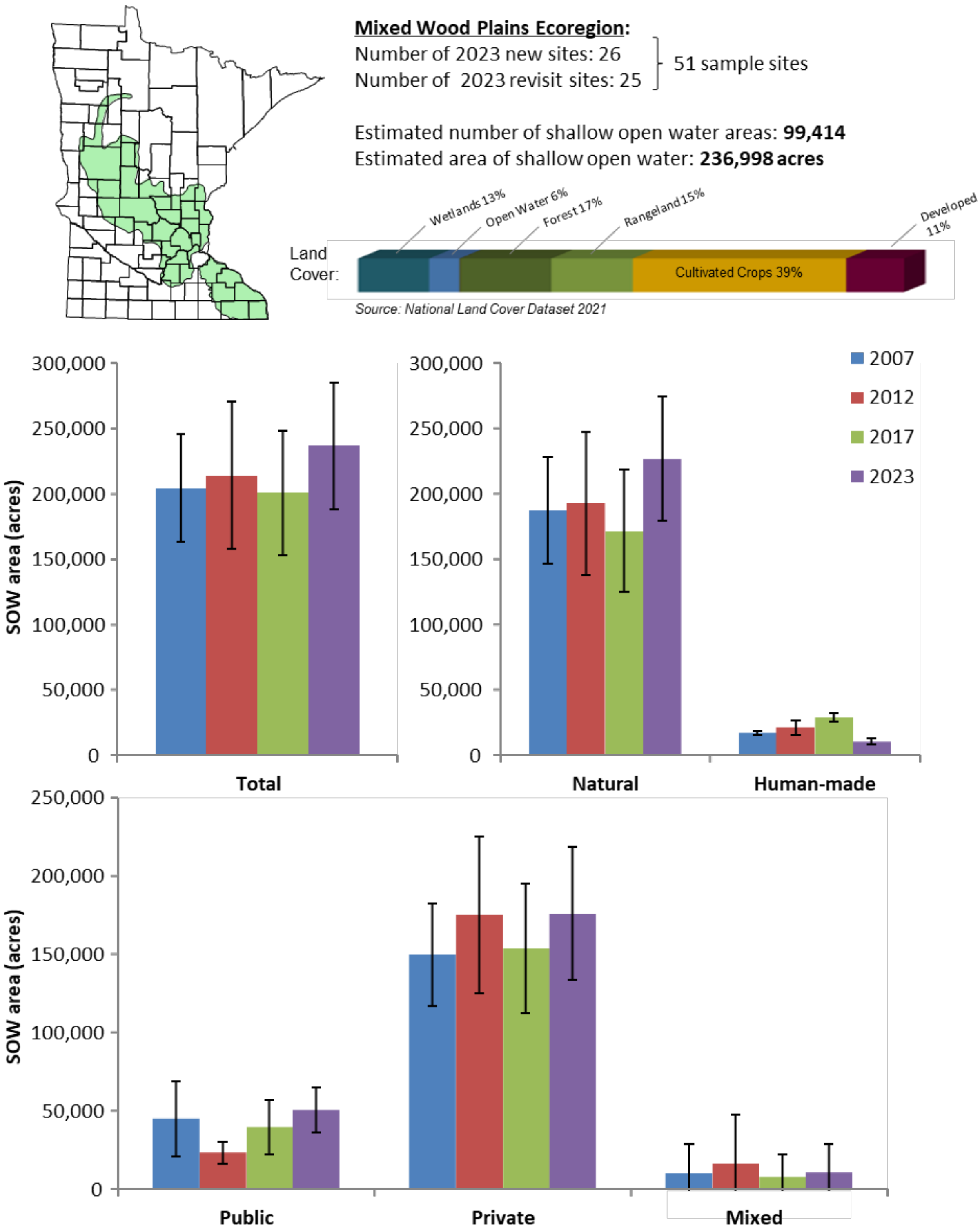
Mixed Wood Plains results and discussion

The MWP ecoregion (Omernik 1987, White and Omernik 2007) represents a transitional zone between the Great Plains and Northern Laurentian Forests. In Minnesota, the MWP ecoregion occupies the central part of the state in a southeast to northwest orientation. The southeast portion of this ecoregion overlaps Minnesota's portion of the Driftless Area—a region not covered by the last glacial advance—that has a steeply dissected, stream-dominated topography with numerous valleys and bluffs. In the southeast, Oak and Maple-Basswood forests are primarily restricted to steep valley walls while agriculture (row crops and cattle) is prevalent on more level terrain. Wetlands in the southeast driftless area are primarily located within floodplain, riverine, and slope geomorphic settings which is a stark contrast to the glaciated northern and western parts of the ecoregion. The remainder of this ecoregion, the area to the north and west of the Twin Cities Metropolitan Area, has a gentler topography consisting of nearly level to rolling glacial till plains as well as hilly moraines and beach ridges. Pre-settlement vegetation in this region consisted of maple-basswood forest, oak savanna, and tall-grass prairie. Numerous lakes and depressional wetlands dot the landscape in the western portion of this ecoregion but are virtually nonexistent in the southeast. Current land use is a combination of agriculture (row crops, cattle, orchards, sod), natural vegetation (forests, grasslands, wetlands), and urban development. In fact, much of Minnesota's population is concentrated within this ecoregion in cities such as Minneapolis, St. Paul, Rochester, St. Cloud, Alexandria, and Fergus Falls. Precipitation ranges from an average annual of 24 inches in the west to 38 inches in the southeast (State Climatology Office 2021).

Depressional wetland quantity (status and change)

An estimated 236,998 acres of SOW community associated with depressional wetlands and ponds occur within the MWP ecoregion according to 2023 survey results (Figure 14). Total acreage of SOW in the ecoregion has remained steady over time; no statistically significant differences exist between any of

Figure 14. Estimates of the total area of shallow open water community in the Mixed Wood Plains ecoregion over four cycles of the DWQA survey as well as area broken down by origin and property ownership. Bracketed lines represent the 95% confidence interval associated with each estimate.



the survey cycles to date. Examining SOW habitat created by humans, acreage of that category was significantly lower in 2023 compared to all three previous surveys. While not statistically significant, the area of natural wetlands was noticeably higher in 2023 compared to other years. This apparent shift in the extent estimates in the MWP may partially be the result of the imperfect weight adjustments for converting whole-basin results to SOW-based results in past surveys (Appendix A). For example, the average weight used in the total acreage estimation for the MWP was 3,586 in 2017 and 4,232 in 2023 with large (>5 acres), primarily natural SOWs having a substantially greater weight in the 2023 survey. In comparison, for the SOWs created by human activity, the average weight was ~2X higher in 2017 compared to 2023. Differences in extent estimates influenced by shifts in survey design are not expected in comparisons of future surveys; however, the transition between the 2017 and 2023 surveys will need to be considered in any change or trend analyses of SOW community extent in the MWP ecoregion which include the first three surveys.

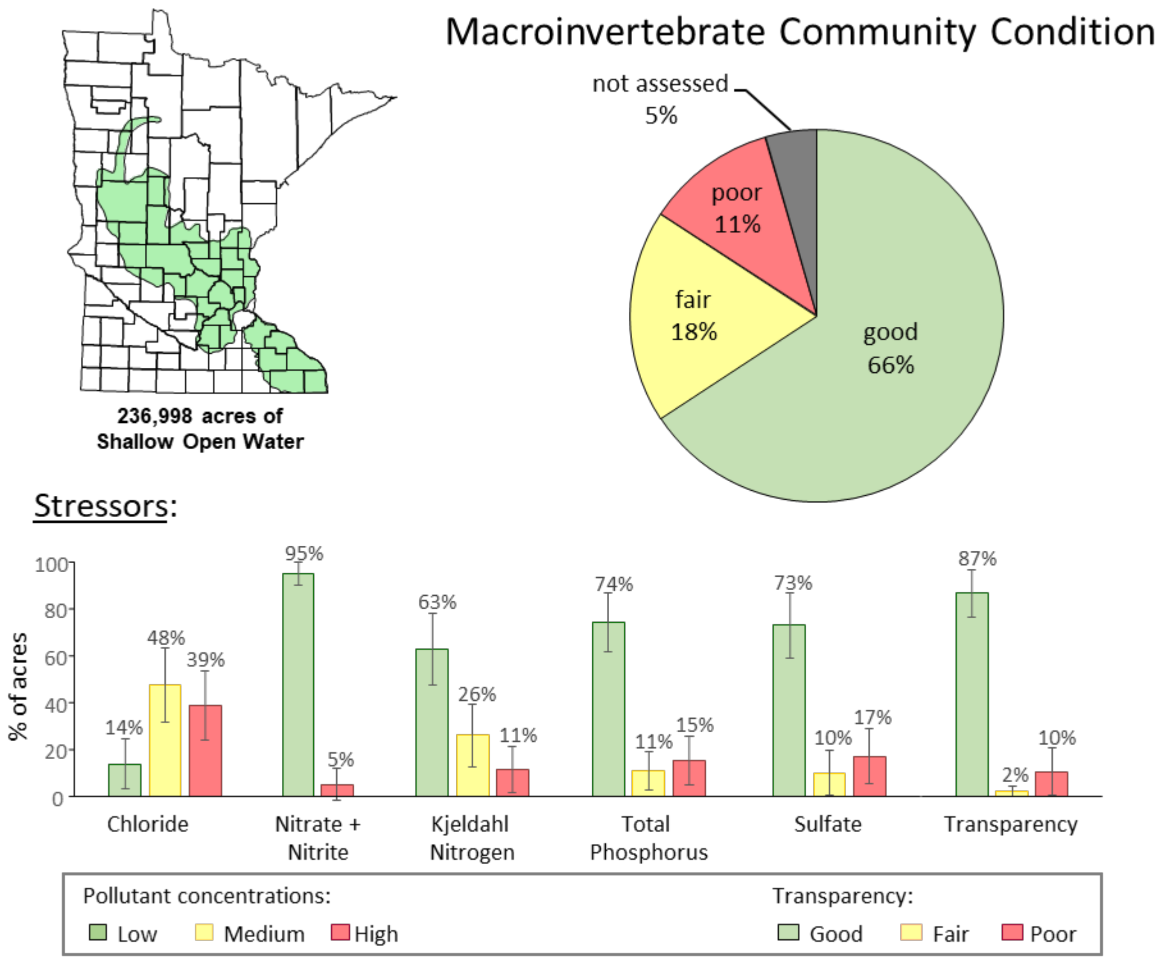
In terms of the wetland ownership, there were no notable patterns in SOW area over time within any of the categories (Figure 14). Overall, the majority of SOW wetlands and ponds in the MWP ecoregion are naturally formed and located on private property, a pattern that is relatively consistent across survey years. The overall ecological condition of these shallow open water habitats in this ecoregion is therefore largely dependent on the actions of private citizens as well as the efficacy of regulations and policies that protect wetlands in Minnesota.

Depressional wetland condition

- The latest aquatic macroinvertebrate results indicate that depressional wetlands and ponds in the MWP ecoregion are relatively healthy with 84% of SOW acres attaining a macroinvertebrate condition rating of fair or good (Figure 15). This finding suggests that SOW in this ecoregion is providing adequate habitat for this subset of the aquatic community and that pollutant concentrations are not detrimental to the survival and reproduction of these organisms.
- Similar to past surveys, elevated chloride concentrations represent the most prevalent stressor—among those measured—for depressional wetlands and ponds in the MWP ecoregion. This result is not surprising given the concentration of major metropolitan areas in this ecoregion and the associated de-icing applications that are required to keep the network of roadways in those areas safe during Minnesota's winter season.
- Nitrate + nitrite nitrogen concentrations were below detection (= 'Low') in the overwhelming majority of MWP depressional wetlands and ponds in 2023 (Figure 15). As discussed in past reports, detection of this pollutant in wetlands is highly dependent on the timing of sample collection relative to rain events as denitrification in wetlands quickly transforms it to gaseous forms of nitrogen, removing it from the aquatic system. Nitrate can also be depleted from the water column through plant uptake as it is an important macronutrient.
- The relative extent of stressors (i.e., % High) in the MWP ecoregion and the associated relative risk that they pose to the macroinvertebrate community largely mirrors the results obtained for the two ecoregions combined (Figure 16). However, with the exception of total phosphorus, relative risk estimates were greater in the MWP compared to the statewide results, indicating stronger impacts to aquatic macroinvertebrates and/or more ecologically meaningful criteria for characterizing stressor-response relationships (i.e., high stress/poor condition) in this ecoregion.
- Across all years of the DWQA, sulfate was the only stressor that consistently represented an elevated risk to the macroinvertebrate community in the MWP ecoregion. Total phosphorus, nitrate + nitrite, and Kjeldahl nitrogen posed elevated risks in three out of four surveys to date.
- Despite chloride typically representing the most prevalent stressor—among those measured—in the MWP ecoregion, it does not pose an elevated risk to macroinvertebrates in any of the DWQA surveys when the analysis is based on area (rather than number of SOW areas). This may be due to the criterion used in the DWQA for rating a site in this ecoregion as 'high' (7.9 mg/l)

not coinciding with a chloride level that is detrimental to most aquatic macroinvertebrates. As mentioned previously, the chronic standard for protecting aquatic life in Minnesota is 230 mg/l.

Figure 15. Biological condition and stressor level estimates for Mixed Wood Plain depressional wetlands and ponds. Bracketed lines represent the width of the 95% confidence interval associated with each estimate.

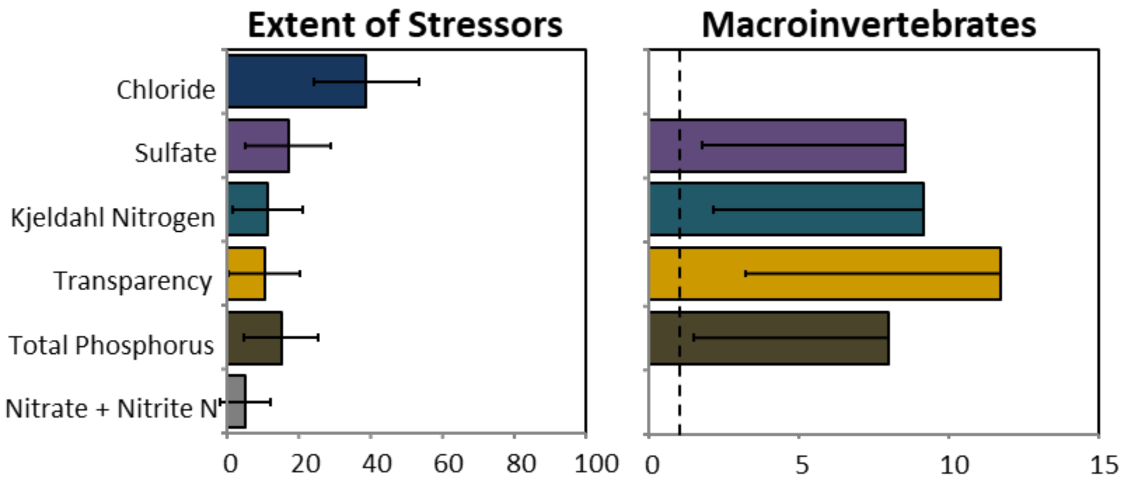


Changes in depressional wetland condition

Macroinvertebrate condition as well as the stressor levels among the four survey periods were evaluated to determine whether any changes were occurring. A few more iterations of the DWQA will be required before trends can be confidently evaluated, a primary goal of this status and trends survey. Appendix E includes the change detection results for the Mixed Wood Plains ecoregion data set. A summary of those results is provided here.

The condition of aquatic macroinvertebrate communities has remained stable over the four survey periods, remarkably so after 2012 (Figure 17); no significant changes were detected between any of the survey periods. These results are encouraging considering the occurrence of some historic wet (e.g., 2016 and 2019) and dry periods (e.g., 2021 and 2022) that have impacted significant portions of the MWP ecoregion during this timeframe. It is also noteworthy that the average macroinvertebrate IBI score was greatest in 2023, a period somewhat still impacted by drought conditions that began in the

Figure 16. Extent of stressors and their relative risk to macroinvertebrate communities in Mixed Wood Plains depressional wetlands and ponds. Bracketed lines represent 95% confidence intervals (for % estimates) or lower confidence limits (for relative risk estimates). A stressor without an associated bar on the relative risk graph indicates that it did not pose an elevated risk (i.e., not significantly greater than 1).



summer of 2021. A combination of low water levels and high temperatures, conditions typical of a drought, can create harsh conditions for aquatic macroinvertebrates inhabiting depressional wetlands and ponds, such as reduced habitat availability and highly fluctuating dissolved oxygen concentrations, including extended periods of anoxia (Salimi et al. 2021). Additionally, pollutant concentrations may increase during drought conditions as water levels within SOW communities recede.

Over the first three cycles of the DWQA, sulfate concentrations increased in MWP depressional wetlands and ponds (Figure 18). In the latest survey, sulfate concentrations appear to have leveled out

Figure 17. Comparison of macroinvertebrate community condition in the MWP ecoregion over four cycles of the DWQA survey. Bracketed lines represent the width of the 95% confidence interval associated with each estimate. A small percentage (<5%) was not evaluated for macroinvertebrate condition in 2007/08 and 2023 (results not shown).

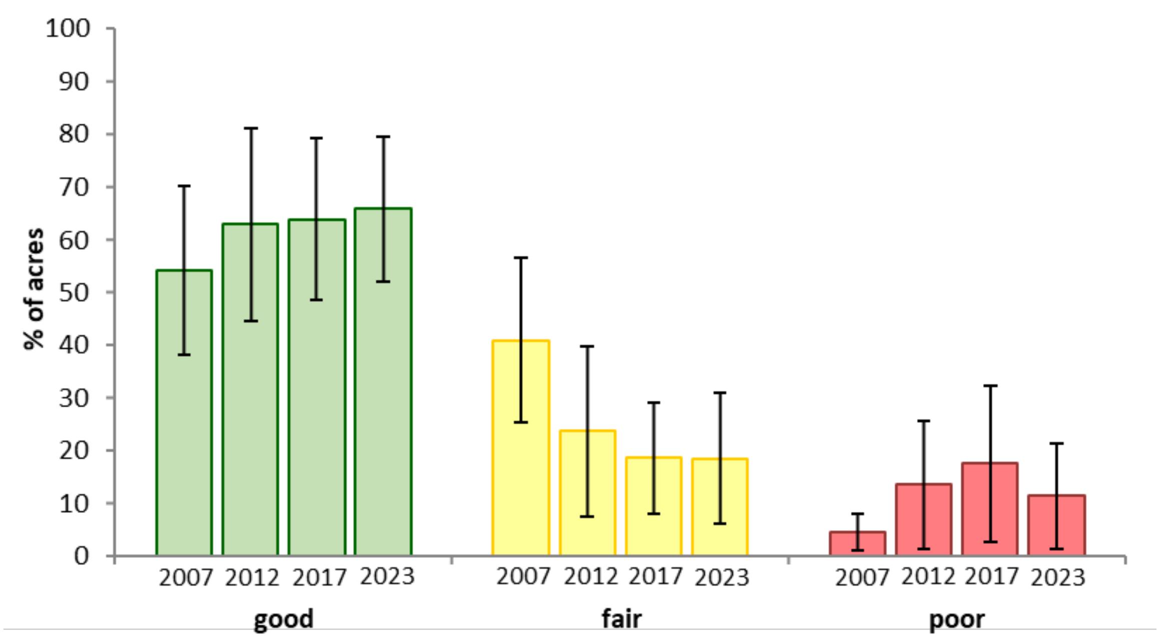
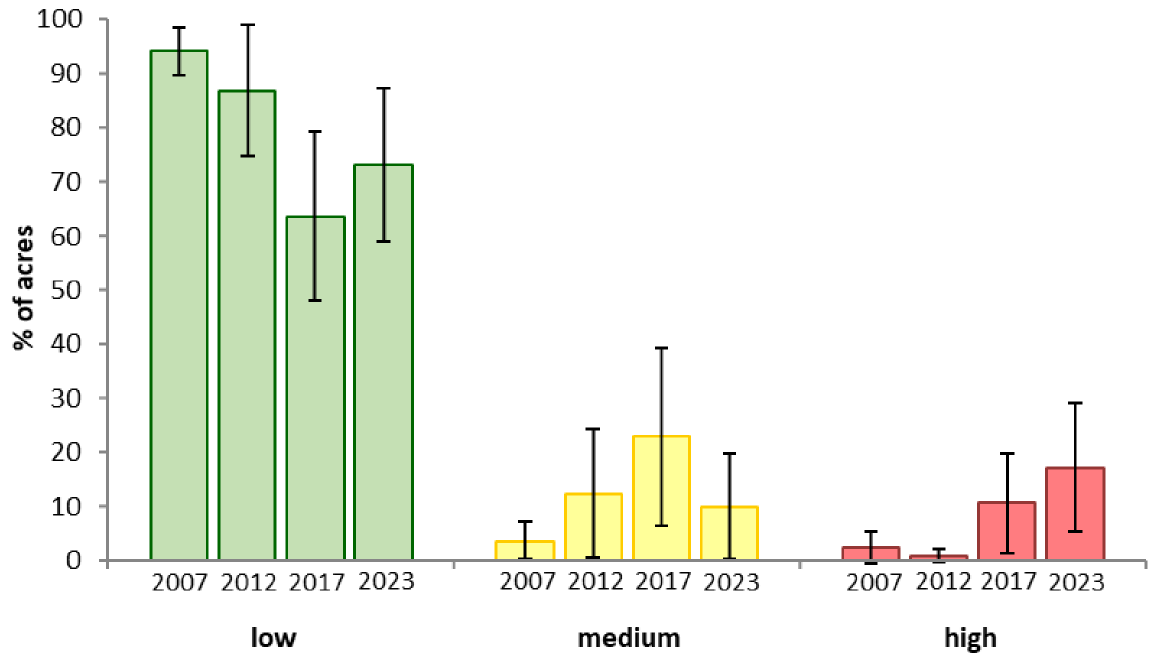


Figure 18. Comparison of sulfate concentration categories in the MWP ecoregion over four cycles of the DWQA survey. Bracketed lines represent the width of the 95% confidence interval associated with each estimate. A small percentage (<5%) was not evaluated for sulfate concentrations in 2017 (results not shown).



with no significant differences between 2017 and 2023 for any of the statistical tests. Mean sulfate concentrations have increased 7.5 mg/L since 2007 in the MWP ecoregion. Future surveys will help to determine whether these results are part of a long-term trend as suggested by Hinckley and Driscoll (2022) or merely reflect short-term responses to cyclical phenomena such as wet/dry cycles.

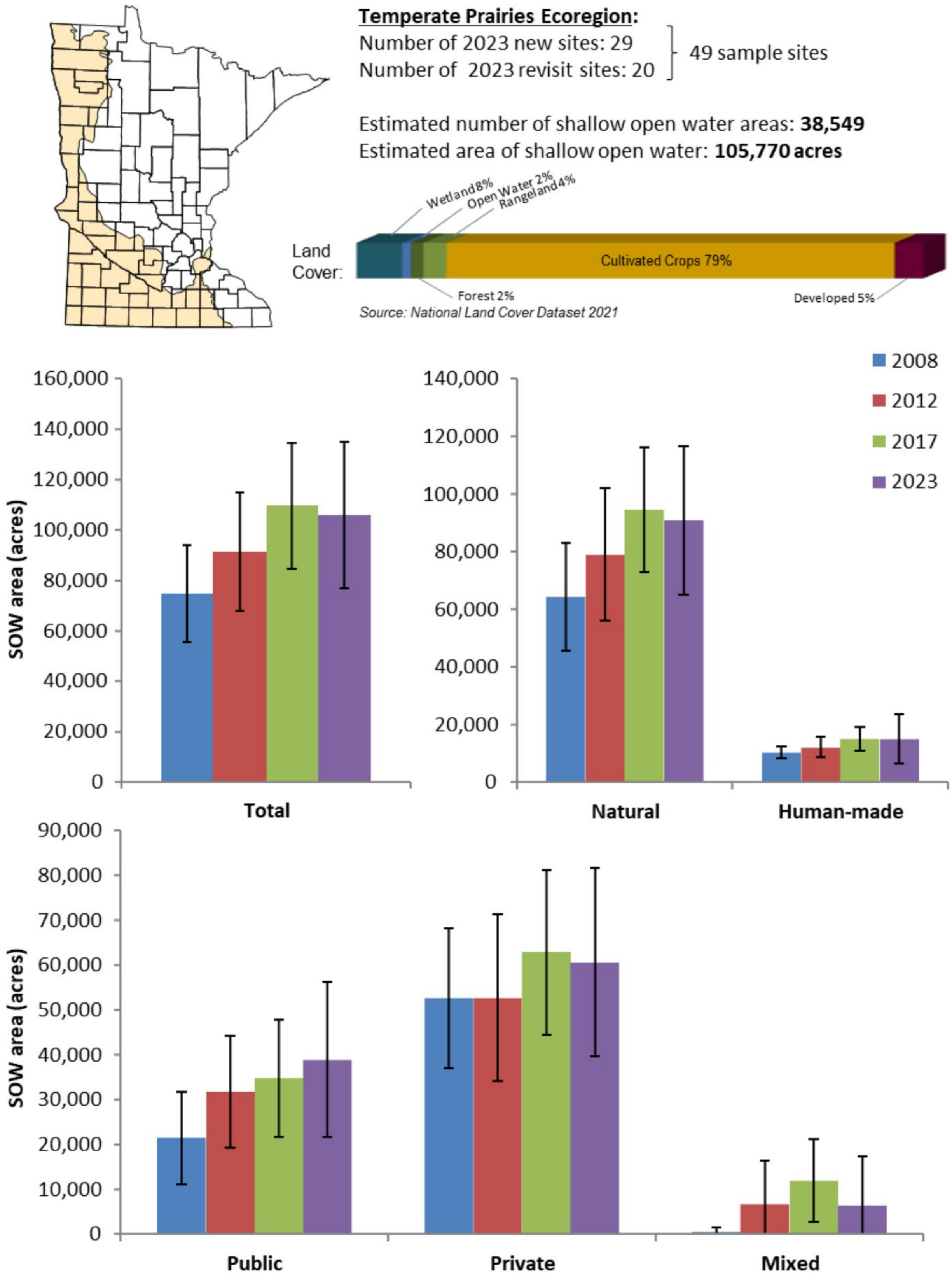
Temperate Prairies results and discussion

The topography of the TP ecoregion (Omernik 1987, White and Omernik 2007) ranges from the gently rolling glacial till plains of the southern part of the state to the nearly level basin of ancient Glacial Lake Agassiz in the northwest. Prior to European settlement the vegetation within this region was primarily tall-grass prairie interspersed with often expansive wet prairie communities. A large portion of this ecoregion coincides with the Prairie Pothole Region, an area characterized by its high density of seasonally to permanently inundated depressional wetlands. Today the dominant land use within the ecoregion is agriculture with both row crop farming (corn, soybeans, grains, sugar beets) and livestock production (cattle, swine, poultry) being prevalent. Large cities in this ecoregion include Albert Lea, Austin, Crookston, Mankato, Marshall, Moorhead, and Willmar. Annual precipitation ranges from 20 inches in the northwest to 36 inches in the southeast (State Climatology Office 2021).

Depressional wetland quantity (status and change)

An estimated 105,770 acres of SOW community associated with depressional wetlands and ponds occur within the TP ecoregion according to 2023 survey results (Figure 19). The total area of SOW in the ecoregion appears to be increasing over time with statistically significant differences existing between survey cycles. Total area of SOW in 2017 was significantly greater than SOW in 2008 ($p = 0.04$) and in 2023 total SOW area was also greater than in 2008, though that result was only marginally significant ($p = 0.08$). There were no other differences observed in SOW area between survey cycles for any SOW sub-categories despite some obvious patterns in the results (e.g., Public; Figure 19). Overall, the majority of SOW wetlands and ponds in the TP ecoregion are naturally formed and located on private property, although the difference between private and public ownership is less than that observed in the MWP.

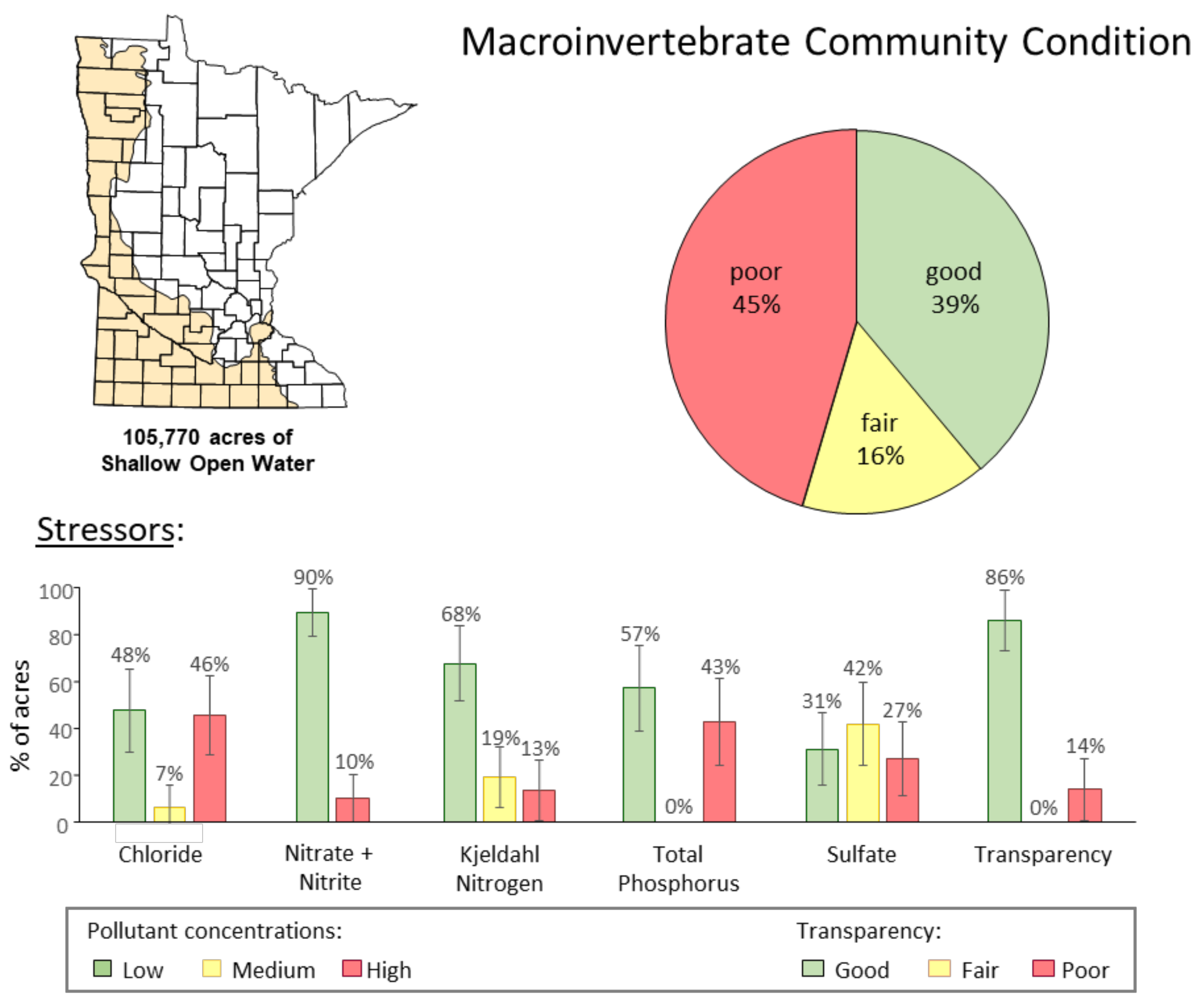
Figure 19. Estimates of the total area of shallow open water community in the Temperate Prairies ecoregion over four cycles of the DWQA survey as well as area broken down by origin and property ownership. Bracketed lines represent the 95% confidence interval associated with each estimate.



Depressional wetland condition

- The condition of aquatic macroinvertebrate communities inhabiting the SOW areas of TP depressional wetlands and ponds is mixed. In 2023, 45% of the SOW area in this ecoregion had macroinvertebrate communities that were in poor condition, while 39% were in good condition (Figure 20). This finding indicates that roughly half of the SOW area in this ecoregion is not providing adequate habitat for macroinvertebrates, and likely other components of the aquatic community as well.

Figure 20. Biological condition and stressor level estimates for Temperate Prairie depressional wetlands and ponds. Bracketed lines represent the width of the 95% confidence interval associated with each estimate.

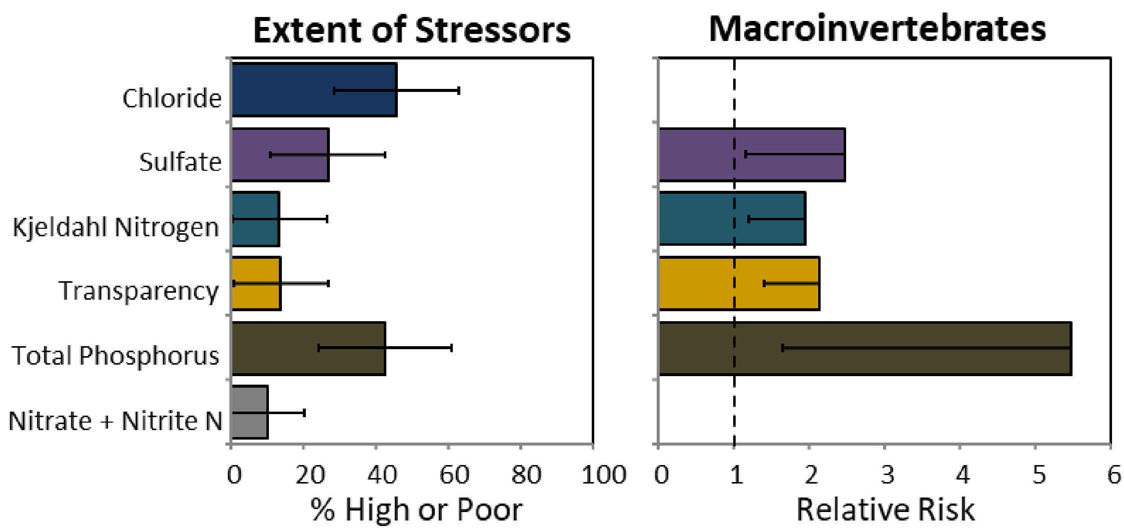


- Increased chloride concentration represents one of the most prevalent stressors—among those measured—for depressional wetlands and ponds in the TP ecoregion (Figure 20). Sources of excess chloride in this ecoregion are of natural and anthropogenic origin, including surficial geology (i.e., in west-central MN), de-icing treatments, water softener discharge, fertilizer application (e.g., KCl), and dust control treatments (e.g., $CaCl_2$, $MgCl_2$) on gravel roads. High concentrations of chloride in the water column of SOW areas did not, however, pose a significant risk to the aquatic macroinvertebrate communities that live there (Figure 21).
- High total phosphorus concentrations were also relatively widespread among SOW areas in the TP ecoregion (Figure 20) and is noteworthy considering it had the largest relative risk estimate

for macroinvertebrates in the 2023 survey (Figure 21). Another important finding is that total phosphorus has posed an elevated risk to macroinvertebrate communities in this ecoregion across all four surveys. Elevated phosphorus in SOW areas may be due to runoff from the surrounding landscape that includes sediment (i.e., adsorbed P), fertilizer, or animal waste, atmospheric deposition of soil particles, as well as the release of phosphorus from wetland sediments during anoxic conditions.

- Sulfate is another important stressor to aquatic macroinvertebrate communities inhabiting SOW areas in the TP ecoregion. High sulfate concentrations have frequently (i.e., 25-46%) been encountered in this ecoregion and posed an elevated risk to macroinvertebrates in all four surveys.
- Kjeldahl nitrogen is another noteworthy stressor to macroinvertebrates in the TP ecoregion. It has represented a significant relative risk to macroinvertebrates in all four surveys but has less frequently been found at high concentrations compared to sulfate or total phosphorus.

Figure 21. Extent of stressors and their relative risk to macroinvertebrate communities in Temperate Prairie depressional wetlands and ponds. Bracketed lines represent 95% confidence intervals (for % estimates) or lower confidence limits (for relative risk estimates). A stressor without an associated bar on the relative risk graph indicates that it did not pose an elevated risk (i.e., not significantly greater than 1).



Changes in depressional wetland condition

Similar to the MWP ecoregion, aquatic macroinvertebrate community condition has remained stable over the four survey periods in the TP ecoregion (Figure 22); no significant changes were detected between any of the survey periods. Again, these results are encouraging considering the relatively extreme climate variability that has occurred during this period as well as the abnormally dry conditions that were present in this region when monitoring occurred for the 2023 survey (https://droughtmonitor.unl.edu/data/png/20230620/20230620_mn_trd.png). The macroinvertebrate results also suggest that no-net-loss of wetland quality in the TP ecoregion—for SOW wetlands—was achieved between 2008 and 2023, albeit at a relatively degraded condition (~40% poor).

In 2023, chloride concentrations in SOW communities of the TP ecoregion dropped significantly compared to all three previous surveys (Figure 23, Appendix E). This may be due to the relatively drier conditions in the days and weeks prior to monitoring in 2023, as previously mentioned. In fact, most of the TP ecoregion was either abnormally dry or experiencing moderate drought conditions from January through the 2023 DWQA monitoring season (June) according to the US Drought Monitor Map Archive for Minnesota. Reduced precipitation and snowmelt likely resulted in less runoff into surface waters in this region, which transported less pollutants such as chloride, nitrate, and sediment into these waters. Nitrate and nitrite also exhibited significantly lower concentrations in the 2023 survey compared to all of the other surveys (Appendix E).

The pattern of sulfate concentrations across the four surveys was slightly different than that of chloride and nitrate/nitrite. Mean sulfate concentrations in the TP ecoregion were significantly elevated in 2012 and 2017 compared to the 2008 and 2023 surveys (Appendix E). Thus, sulfate concentrations were low in 2023, similar to chloride and nitrate/nitrite, but were also low in the initial baseline survey of 2008. Sulfate concentrations did not exhibit a significant decreasing trend among MPCA's long-term monitoring wetlands located within the TP ecoregion (see next section), though the current long-term monitoring data set suggests that sulfate is decreasing across the two ecoregions combined.

Figure 22. Comparison of macroinvertebrate community condition in the TP ecoregion over four cycles of the DWQA survey. Bracketed lines represent the width of the 95% confidence interval associated with each estimate.

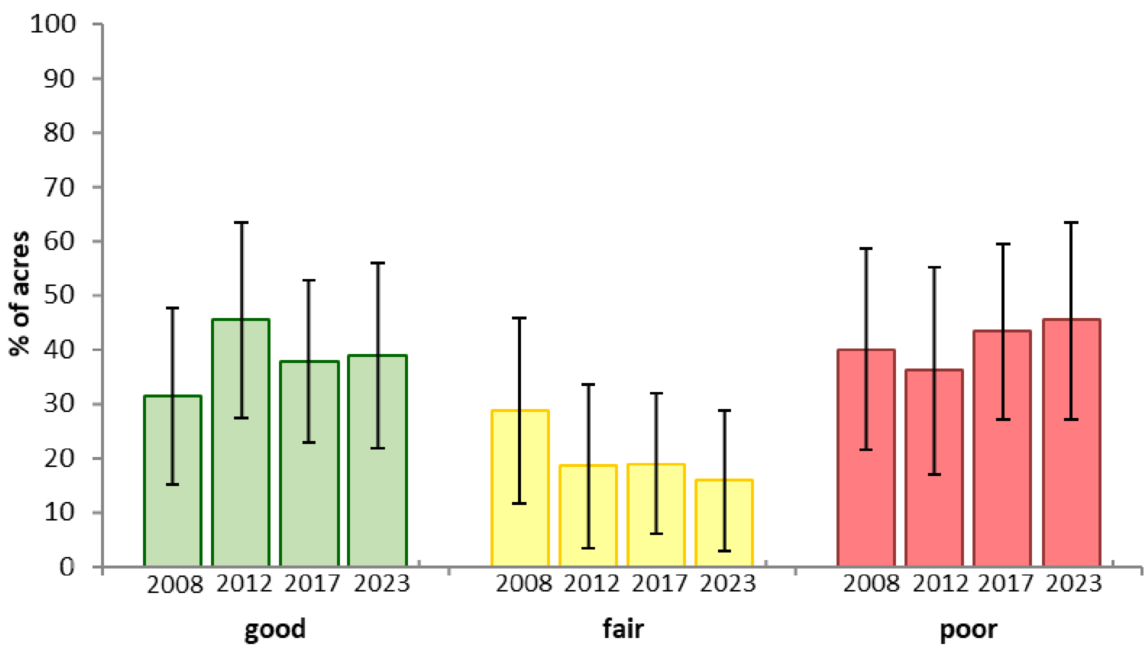
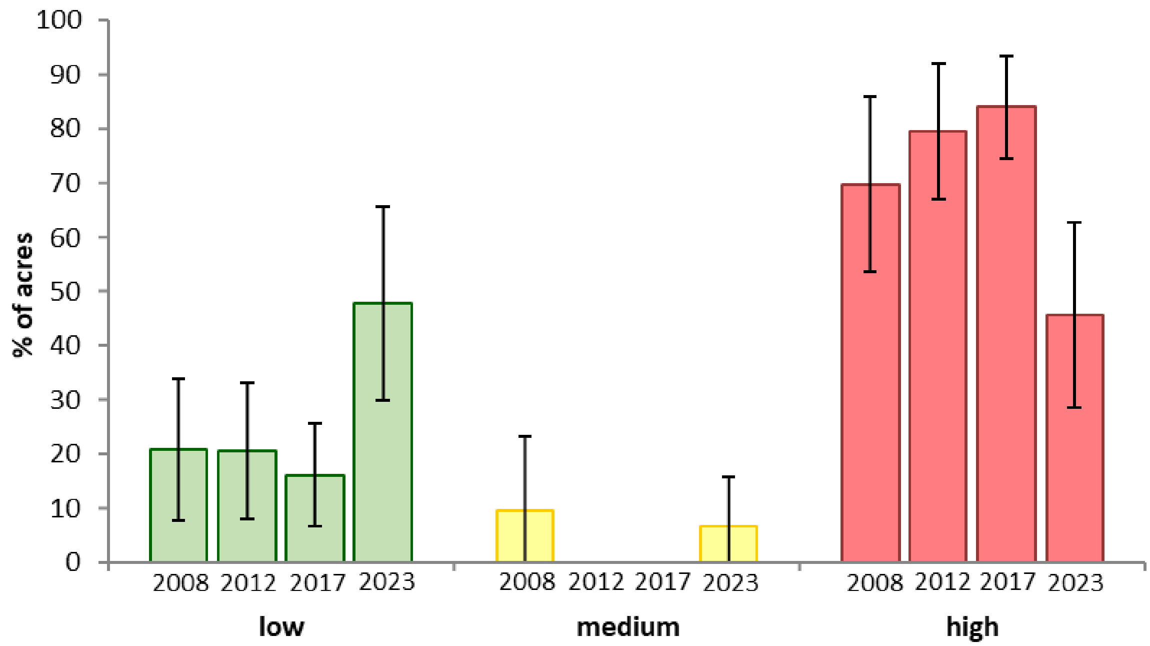


Figure 23. Comparison of chloride concentration categories in the TP ecoregion over four cycles of the DWQA survey. Bracketed lines represent the width of the 95% confidence interval associated with each estimate.



Long-term depressional wetland monitoring

In 2012 the MPCA began monitoring a set of 12 depressional wetlands in synchrony with the monitoring schedule of the DWQA (i.e., every five years). The reason for monitoring these sites is to provide independent evidence for evaluating patterns and trends observed in the probabilistic survey data set. In addition, repeated monitoring of these sites over time will help to elucidate any broad impacts to wetland condition such as climate change, which is essential for understanding the effectiveness of policies, regulations, and management activities meant to protect wetlands in Minnesota. Furthermore, supplementing randomized broad-scale surveys with more in-depth site investigations provides additional evidence for evaluating change over time (Carleton and Washington 2021).

Unlike the randomly selected sites of the DWQA, these long-term monitoring (LTM) sites were targeted to represent wetlands occurring within three land use settings (natural, agricultural, and urban) in each of the two ecoregions (MWP and TP) included in the survey (Table 2 & Figure 24). The TP ecoregion however, is largely agricultural with a much smaller extent of urban land use relative to the MWP ecoregion, and thus does not currently include any LTM sites classified as urban. Additional considerations in LTM site selection included: 1) length of historical data record; 2) ownership (i.e., sites on public land provide a higher likelihood of long-term access); and 3) land use stability (i.e., sites were selected that have a surrounding land use that is unlikely to change for the foreseeable future).

Table 2. MPCA's depressional wetland long-term monitoring sites.

Site Name	Ecoregion	Land Use	Monitoring History
Breen	MWP	Agricultural	1999 to present
New Prairie	MWP	Agricultural	1999 to present
Glacial	MWP	Natural	1995 to present
Prairie	MWP	Natural	1995 to present
Legion	MWP	Urban	1995 to present
Wood	MWP	Urban	1995 to present
03MURR066	TP	Agricultural	2003 to present
Rohliks	TP	Agricultural	2002 to 2017
Tyler	TP	Agricultural	2002 to present
Willow Lake	TP	Agricultural	2002 to present
Lake Charlotte	TP	Natural	2002 to present
Prairie Marsh	TP	Natural	2002 to present

Biological and water chemistry monitoring within the shallow open water community of LTM sites follows the same protocols used at DWQA survey sites as outlined in the Methods chapter of this report. LTM data collected prior to the onset of the DWQA may have been accomplished using variations of these current methods (field and laboratory), however these methods were primarily established in the mid- to late 1990s with minor refinements adopted over time. Thus, while methodological sources of variability cannot be dismissed entirely, their impact on temporal trends is likely to be minor. Prior to 2012, LTM sites were monitored in conjunction with various projects investigating depressional wetland condition. Across years/projects, monitoring did not follow a pre-determined schedule and therefore monitoring frequency varies among LTM sites pre-2012 (Table 3). To minimize seasonal variability, data was limited to the month of June, corresponding to the period of wetland macroinvertebrate monitoring.

Linear mixed models provide an appropriate framework for evaluating temporal trends when the same group of stations are monitored repeatedly over time (Starcevich et al. 2018). This is primarily due to their ability to model the correlation among observations collected within the same year and/or site. Therefore, macro-invertebrate IBI scores and water chemistry variables (chloride, sulfate, total phosphorus, nitrate + nitrite nitrogen, Kjeldahl nitrogen, pH, water temperature & specific conductivity) were evaluated using linear mixed-effects models (lme4 package; Bates et al. 2015) in R version 4.4.1 (R Core Team 2024) to determine if they are changing over time. Specifically, three primary research questions were investigated for each variable: 1) Is the variable changing over time across both ecoregions?; 2) Is the variable changing over time within each ecoregion and does the direction of the relationship differ between ecoregions?; 3) Is the variable changing over time within each land use category and does the direction of the relationship differ between categories?

To fit each model, each dependent variable was related to the fixed effect of time (i.e., monitoring year where 1995 = year 0) as well as random effects of year and site name, following lme4 model structure:

$$\textbf{Dependent variable} \sim \textbf{Time} + \textbf{(1|Year)} + \textbf{(1|SiteName)}$$

This model was fit for each variable by restricted maximum likelihood (REML) using the following data sets: all sites, MWP sites, TP sites, agricultural (Ag) sites, natural (Nat) sites, and urban sites. Statistical

Figure 24. Map of southern Minnesota depicting locations of depressional wetland LTM sites.

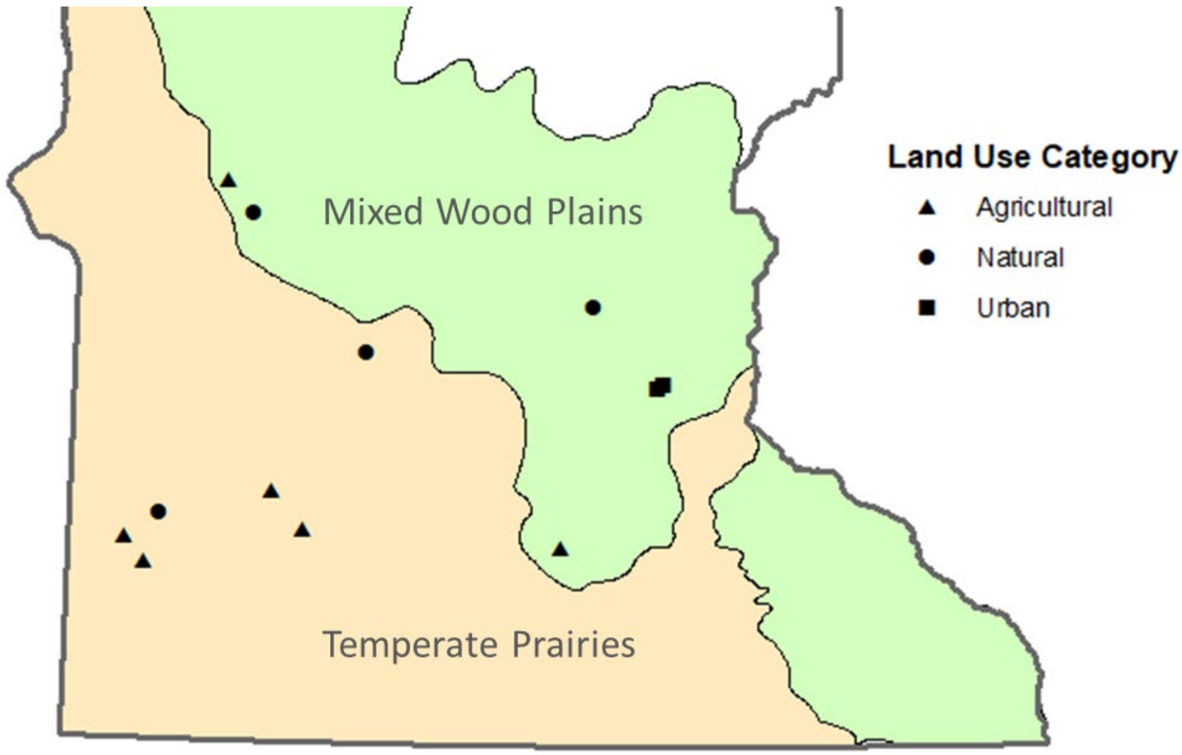


Table 3. Monitoring history of depressional wetland LTM sites; X = data collected in June of that year.

Site Name	Eco	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022	2023
Breen	MWP					X		X	X	X			X				X		X		X	X	X	X						X
Glacial	MWP	X				X			X	X			X				X		X		X	X	X							X
Legion	MWP	X				X			X	X			X		X	X	X		X					X						X
New Prairie	MWP					X		X	X				X				X		X	X	X	X		X						X
Prairie	MWP	X				X		X	X	X			X		X	X	X	X	X	X				X						X
Wood	MWP	X				X		X	X	X			X		X	X	X		X					X						X
03MURR066	TP									X	X				X				X		X	X		X						X
Lake Charlotte	TP								X		X				X		X		X											X
Prairie Marsh	TP								X	X	X	X			X		X	X	X	X				X						X
Rohliks	TP								X	X	X				X		X		X					X						
Tyler	TP								X	X	X				X				X	X	X	X		X						X
Willow Lake	TP								X	X	X				X				X					X						X

significance of the estimated slope (i.e., slope ≠ 0) for each model was evaluated using t-tests via Satterthwaites's degrees of freedom method (lmerTest package; Kuznetsova et al. 2017). Each dependent variable was log transformed to better approximate a normal distribution and to facilitate the estimation of percent annual change for each model. Percent annual change was calculated as follows: 100 * [exp(slope)-1]

Results & Discussion

Linear mixed models were successfully fit for all variables with the exception of nitrate + nitrite nitrogen which had a data set largely comprised of non-detects (i.e., < 0.05 mg/L). Sulfate also posed a problem in this regard as it also had a large number of concentrations that were below the detection limit of the analysis and the detection limit has steadily decreased over time. Currently, it is not clear whether a reliable method exists for accommodating left-censored data (i.e., non-detects) in the context of linear mixed-effects models. Therefore, sulfate models were fit using only the measured concentrations, which significantly reduced the MWP data set while not impacting the TP data set at all.

Macroinvertebrate community condition—as measured by the IBI—among the LTM sites did not exhibit any significant trends over the current history of the program (1995 – 2023). While not statistically significant, it is encouraging that all annual percent change estimates were positive at this time (Table 4). Results among the LTM sites are consistent with those of the probabilistic survey where very few changes were observed between survey periods for the macroinvertebrate IBI (Appendix E).

Table 4. Annual percent change estimates for the macroinvertebrate IBI and water chemistry variables measured at LTM stations. Bold values indicate statistical significance at alpha = 0.05.

		Ecoregion		Land Use Category		
	All Sites	MWP	TP	Ag	Nat	Urban
Parameter	*(n = 12)*	*(n = 6)*	*(n = 6)*	*(n = 6)*	*(n = 4)*	*(n = 2)*
M-IBI Score	0.3%	0.2%	0.6%	0.5%	0.2%	0.1%
Chloride (mg/L)	-0.2%	0.7%	**-2.2%**	-1.4%	-1.4%	4.3%
Sulfate (mg/L)*	**-4.2%**	**-7.6%**	-1.6%	**-6.0%**	-0.3%	-7.8%
Total Phosphorus (mg/L)	**-3.8%**	**-3.5%**	**-4.1%**	**-3.9%**	**-4.2%**	**-2.7%**
Kjeldahl Nitrogen (mg/L)	**-1.2%**	**-1.1%**	**-1.5%**	**-1.5%**	-1.3%	-0.9%
pH	0.1%	0.1%	0.0%	0.0%	0.1%	0.3%
Specific Conductivity (µS/cm)	0.2%	0.6%	**-1.2%**	-0.8%	-0.2%	3.4%
Water Temperature (^{o}C)	-0.1%	-0.5%	**1.4%**	0.2%	0.1%	-0.2%

* All Sites, MWP, Natural, and Urban models excluded a large number of non-detects.

Among the entire network of LTM sites, the only water chemistry parameters that exhibited significant trends were sulfate, total phosphorus, and Kjeldahl nitrogen, all of which were decreasing in concentration (Table 4). For sulfate, these results contrast with those of the probabilistic survey where concentrations have largely been increasing over time (Appendix E). Similarly, when data from the MWP were modelled, sulfate, total phosphorus, and Kjeldahl nitrogen were the only three parameters that exhibited a significant trend; again decreasing in concentration. Relative to probabilistic survey results, the decreasing sulfate trend observed among MWP LTM sites contrasts with the significant increases in sulfate concentrations seen between surveys within the MWP ecoregion (Figure 18, Appendix E). The declining trend in total phosphorus for MWP LTM sites is, however, consistent with the significant decreases in total phosphorus concentrations observed between survey periods within the MWP (Appendix E).

The decreasing sulfate trend among LTM wetlands in the agricultural (Ag) land use category (Table 4) is unexpected considering recent patterns in sulfur availability and application across the Midwest. Recent analyses have demonstrated an increasing trend in the application of fertilizers containing sulfate which is the result of declining atmospheric sulfate deposition (Hinckley and Driscoll 2022). Overall, most years between 2005 and 2017 have seen a net increase of sulfur in the Midwest. The decreasing sulfate trends observed among LTM wetlands suggest that declining atmospheric deposition may be more influential—compared to increasing fertilizer application—on sulfate concentrations within SOW communities. Given the demonstrated impact of sulfate on aquatic macroinvertebrates across the DWQA surveys, and in the literature (e.g., Elphick et al. 2011, Erickson et al. 2022, Karjalainen et al. 2023, Miltner 2021), determining the net impact of these two opposing trends should be a priority for all freshwater habitats in the Midwest.

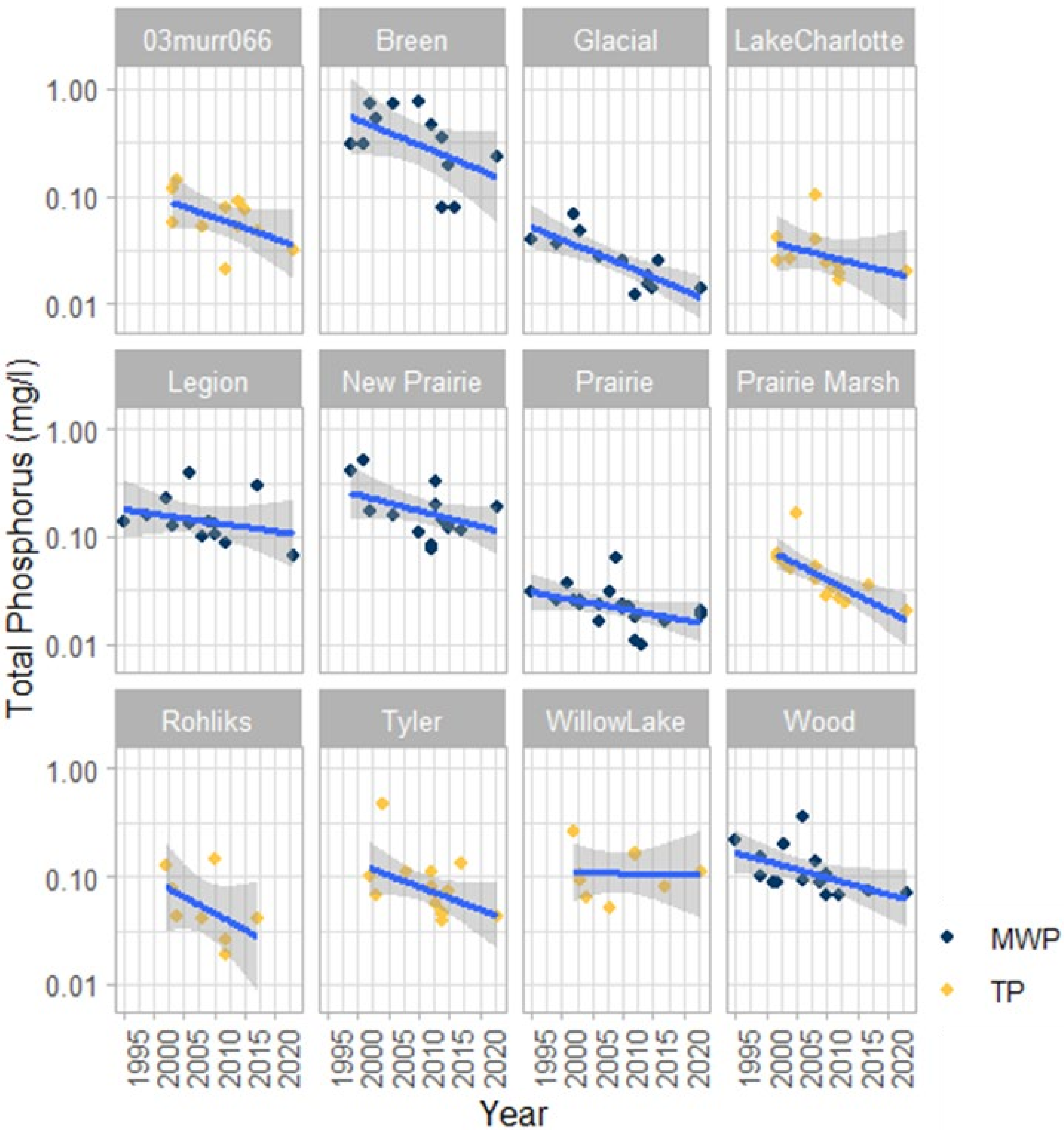
The TP ecoregion exhibited the most significant trends with four decreasing parameters (chloride, total phosphorus, Kjeldahl nitrogen, & specific conductivity) and one increasing (water temperature) (Table 4). In terms of how these trends compare to the change analyses of the probabilistic survey, the results are mixed. For example, the decreasing Cl trend among the LTM sites is consistent with the lower Cl concentrations observed primarily during the 2023 survey (Figure 23, Appendix E). On the other hand, the decreasing total phosphorus LTM trend is not evident in the change analyses for the probabilistic survey (i.e., some evidence that concentrations are increasing). Although interesting to note, the positive trend in water temperature should be regarded with skepticism at this time as measurements were taken at a wide variety of times during the day as well as during various days throughout the month of June, factors which can greatly influence surface water temperatures in these relatively shallow water bodies. A more robust data set (e.g., continuous hourly measurements) coupled with a focus on specific metrics such as monthly average or monthly average high would be required for a more reliable evaluation of water temperature trends, which is currently beyond the scope of this study. Comparing the two ecoregions, there were no variables that exhibited significant trends in opposing directions.

Among the land use categories used to classify the LTM wetlands, those surrounded predominantly by agriculture exhibited the greatest number of significant trends, all of which were decreasing (Table 4). These trends mirrored those observed in the MWP ecoregion. It is worth pointing out that none of the parameters exhibited statistically significant trends in opposing directions between land use categories. Chloride came the closest with an estimated annual increase of 4.3% within the urban category, while the agricultural and natural categories had estimates of -1.4%; however, none of these slope estimates were statistically significant.

The only parameter exhibiting a significant trend among the LTM wetlands largely surrounded by natural land use (i.e., least-disturbed reference wetlands) was total phosphorus (Table 4). This suggests that the declining phosphorus concentrations may be the result of broader/regional factors since these sites are relatively buffered from direct impacts in their surrounding landscape. However, none of these sites are completely isolated from human actions (degrading or otherwise) and thus it is premature to draw any conclusions regarding the driving forces behind this apparent trend. Nonetheless, it is encouraging that total phosphorus exhibits a declining trend across all of the LTM data sets examined (Figure 25), indicating that water quality conditions in depressional wetlands may be improving. Future monitoring data and analyses will be required in order to determine whether the current pattern merely represents a declining period of a short-term cycle or reflects a long-term declining trend.

National aquatic resource surveys (NARS) conducted periodically by the US EPA have documented increasing total phosphorus concentrations in lakes and streams (Stoddard et al. 2016). Subsequent analyses by Carleton and Washington (2021), using NARS data as well as additional large-scale data sets where sites were monitored more frequently than NARS sites, found that increasing phosphorus concentrations were not ubiquitous across the continental US as suggested by Stoddard et al. (2016). Perhaps the most interesting aspect of the Carleton and Washington paper is the recommendation to enhance broad-scale probabilistic surveys with a set of sites monitored at a higher frequency in order to more thoroughly investigate temporal trends. MPCA's LTM wetland program attempts to serve this purpose but would benefit from the following refinements geared towards generating a more robust (spatially and temporally) data set: 1) LTM sites should be monitored more frequently than the current 5-year DWQA cycle; 2) Two urban land use category LTM sites should be added to the TP ecoregion, either supplementing the current sites in that ecoregion or replacing two existing agricultural sites; 3) Supplement the number of LTM sites surrounded by natural land use with wetlands located in the northwestern and/or southeastern portion of each ecoregion; and 4) Characterize the immediate surrounding landscape as well as the catchment area of each LTM wetland over time to better understand the nature and/or scale of observed trends.

Figure 25. Concentration of total phosphorus (log10 transformed) over the history of monitoring LTM wetlands in the Mixed Wood Plains and Temperate Prairies ecoregions. Blue line = fitted linear regression line for each data set; gray bands = 95% confidence interval of the fitted line.



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Appendix A. Transitioning the DWQA to a focus on shallow open water communities

From 2007 to 2017 the DWQA survey design utilized a whole-basin approach, meaning that entire wetland basins (or depressions) were randomly selected and evaluated for their ecological condition. The diversity of plant community types present within these basins ranged from only shallow open water (e.g., ponds) to a complex of shallow open water, deep marsh, shallow marsh, wet meadow, and shrub-carr. Wetlands, however, often do not occur as discrete basins in the landscape, making it difficult to generate the map used for selecting survey sites. And while the transition from the plant IBI to the FQA method made it possible to evaluate the diversity of plant community types present within a basin, the macroinvertebrate IBI's assessment capabilities were primarily limited to shallow open water and deep marsh community types that have semipermanent to permanent standing water. This limitation occasionally created a discrepancy between the two biological indicators when the basin included a diversity of plant community types and the condition of the shallow open water community differed from that of the other wetland types present. For example, a wetland could have a shallow open water community that is largely isolated from water quality impacts such as excess nutrients and sediment and is in 'good' condition while other plant communities within the same basin are in poor condition due to physical alteration and invasive species. A solution was needed that addressed this discrepancy and reduced the amount of overlap between MPCA's two wetland surveys.

The Minnesota Wetland Condition Assessment (MWCA) evaluates the ecological condition of all wetland plant communities that are less than 1 meter in depth and are not in active cultivation (Bourdaghs 2019). Shallow open water communities, especially when they are created by excavation (i.e., human-made ponds), are often deeper than 1 meter and thus are not fully represented in MWCA results. By focusing on the shallow open water community, the DWQA addresses this gap and more thoroughly evaluates open water wetlands using three distinct indicators: macroinvertebrates, vegetation (still in development), and water quality. The other community types that are no longer included in the DWQA will continue to be included in the MWCA, though not limited to a depressional hydro-geomorphic setting.

Shifting the focus of the survey required a re-visit of past DWQA surveys to adjust components of their survey designs and generate new results specific to the re-defined target population: shallow open water communities within a depressional hydrogeomorphic setting. To do so, each site evaluated in past DWQA surveys—including sites where access was denied—was examined via aerial imagery to determine: 1) if it contained shallow open water (if not, it was changed to 'non-target'); 2) the number of distinct shallow open water areas present within the basin; and 3) the area of the shallow open water pocket where monitoring occurred. A minimum mapping unit of 200 m^2 was utilized for all three determinations, shallow open water areas smaller than this were considered 'non-target'. This information was then used to convert the 'basin' weight (i.e., the extrapolation multiplier for each site) of past surveys into a 'shallow open water' weight; one that reflects the extent of the shallow open water community within the basin. Since the DWQA presents results as either the number of shallow open water pockets or the area of shallow open water, two separate weight adjustments were required:

- Weight Adj. for Reporting by Number: Basin # Wgt x (# SOW pockets in Basin) = SOW # Wgt
- Weight Adj. for Reporting by Area: Basin Area Wgt x (SOW area/Basin area) = SOW Area Wgt

where # = number, SOW = shallow open water, and wgt = weight. If a wetland basin was entirely shallow open water, as is the case for most ponds, then basin weights and SOW weights were the same. The following illustration provides an example of how procedure was used to adjust the weights for site 17YELL282.

Whole-basin approach of previous surveys **New shallow open water approach**

The 2017 design weights (based on basin approach) for 17YELL282 were 4.4 and 180.6 for number and area, respectively. Considering the new approach, there are three distinct areas of shallow open water within the former wetland basin, accounting for 9 acres or 43% of the basin area. Therefore, to derive the weight for generating results in terms of the number of shallow open water pockets, the basin weight of 4.4 is multiplied by 3 to obtain a SOW weight of 13.3. For the area-based weight, the basin area weight 180.6 is multiplied by 0.43 to obtain a SOW weight of 77.7.

Unfortunately, as the above example illustrates, when multiple shallow open water pockets exist within a basin, the weight adjustment process used here assumes that the monitored SOW area (indicated by the star in the above illustration) represents all of the SOW areas present within the basin. This assumption originated with the whole-basin approach of past surveys and thus cannot be dismissed when interpreting results from the 2007 – 2017 surveys. Current and future DWQA surveys, however, will be based on the assumption that the monitored location only represents the shallow open water area within which the monitoring occurred (e.g., the small SOW area indicated by the star). Thus, while the weight adjustment process makes it possible to compare current and future survey results to those of past surveys (i.e., results applicable to shallow open water), the aforementioned assumption associated with past surveys prevents results from being entirely equivalent across all iterations of the DWQA survey.

To prevent the same weight adjustment (i.e., basin → SOW) from being a requirement in the 2023 DWQA as well as future surveys, a map (a.k.a. sample frame) was needed for the site selection process that was limited to the shallow open water areas of depressional wetlands and ponds. This new sample frame was created through a combination of updating the map that was used in the 2017 DWQA as well as supplementing this map with additional 1 mi^2 plots from the WSTMP. Updating the 2017 sample frame to reflect shallow open water communities required selecting areas within the wetland basins that were coded according to the National Wetland Inventory (NWI) as: palustrine aquatic bed (PAB), palustrine unconsolidated bottom (PUB), lacustrine-littoral aquatic bed (L2AB), and lacustrine-littoral unconsolidated bottom (L2UB). Lacustrine wetland types were only retained in the sample frame when they were not in association with lacustrine-limnetic (i.e., deepwater) habitat. In addition to the 954 1mi^2 plots used to create the updated sample frame of past surveys, 758 plots (i.e., Panel 2 plots being retained by the WSTMP) were examined via aerial imagery (as recent as summer 2021) and Minnesota's update to the NWI (Kloiber et al. 2019) to supplement the extent of shallow open water areas from which randomly selected DWQA survey sites could be drawn from.

Appendix B. Macroinvertebrate IBIs

Metrics and scoring criteria for the macroinvertebrate IBIs used to assess condition of shallow open water wetlands and ponds throughout the state. Tables indicate the ecoregions where each metric applies and can be used to construct the two distinct IBIs.

Macroinvertebrate IBI metrics	Response to disturbance	Ecoregion[1]	
		MWP	**TP**
Number of Ephemeroptera, Trichoptera, and Odonata genera	Decrease	X	X
Number of intolerant genera[2]	Decrease	X	X
Number of macroinvertebrate taxa (most groups identified to genus, snails and leeches identified to species)	Decrease	X	X
Number of Chironomidae genera	Decrease	X	
Number of Diptera genera	Decrease		X
Number of collector-gatherer genera	Decrease	X	
Number of scraper genera	Decrease	X	
Abundance of Corixidae divided by total abundance of Hemiptera and Coleoptera	Increase	X	X
Abundance of tolerant taxa divided by total abundance of sample[2]	Increase	X	X
Abundance of Ephemeroptera, Trichoptera, and Odonata divided by total abundance of sample	Decrease	X	
Abundance of the three most dominant genera divided by total abundance of sample	Increase	X	
Abundance of Chironomidae divided by total abundance of sample	Increase		X
Abundance of Pleidae divided by abundance of Hemiptera	Decrease		X
	Total number of metrics in IBI:	**10**	**8**

[1] Ecoregion abbreviations: MWP - Mixed Wood Plains; TP - Temperate Prairies

[2] Tolerant/intolerant macroinvertebrate taxa designations determined empirically (see Genet and Bourdaghs 2006).

Formula for converting metric values to scores:

Metrics that Decrease with Increasing Disturbance:

$$\text{Score} = \left(\frac{\text{metric value - minimum value}}{\text{95th percentile value- minimum value}} \right) \times 10$$

Metrics that Increase with Increasing Disturbance:

$$\text{Score} = 10 - \left(\left(\frac{\text{metric value - 5th percentile value}}{\text{maximum value- 5th percentile value}} \right) \times 10 \right)$$

Appendix C. Water quality parameters analyzed by Minnesota Department of Health Environmental Laboratory

Analyte	Fraction	Report limits	Units	Method reference
Chloride	Total	0.50	mg/L	EPA 300.1
Nitrate + Nitrite Nitrogen	Total	0.05	mg/L as N	EPA 353.2
Kjeldahl Nitrogen	Total	0.20	mg/L as N	EPA 351.2
Alkalinity	Total	10.0	mg/L	SM 2320 B-2011
Phosphorus	Total	0.003	mg/L as P	EPA 365.1*
Sulfate	Total	0.50	mg/L	EPA 300.1

* Total phosphorus was analyzed using SM 4500P-I method in 2012 DWQA. Methodological issues associated with this method may have resulted in low readings of total phosphorus, particularly in low nutrient waters, for the 2012 samples.

Appendix D. Categorizing condition and stressors

Criteria used to determine wetland condition and stressor levels relative to least disturbed, regional reference sites.

Indicator	Ecoregion	Condition categories		
		Good	**Fair**	**Poor**
Macroinvertebrate IBI	MWP	> 64	< 64, > 44	< 44
Macroinvertebrate IBI	TP	> 66	< 66, > 56	< 56
		Transparency categories		
		High	**Medium**	**Low**
Secchi Tube Reading (cm)	MWP	> 66	< 66, > 38	< 38
Secchi Tube Reading (cm)	TP	> 65	< 65, > 45	< 45
		Stressor level categories		
		Low	**Medium**	**High**
Nitrate + Nitrite Nitrogen (mg/L)	MWP & TP	no detect	n/a	detect
Kjeldahl Nitrogen (mg/L)	MWP	< 1.49	> 1.49, < 3.10	> 3.10
Kjeldahl Nitrogen (mg/L)	TP	< 1.60	> 1.60, < 2.97	> 2.97
Total Phosphorus (mg/L)	MWP	< 0.148	> 0.148, < 0.384	> 0.384
Total Phosphorus (mg/L)	TP	< 0.180	> 0.180, < 0.202	> 0.202
Chloride (mg/L)	MWP	< 1.4	> 1.4, < 7.9	> 7.9
Chloride (mg/L)	TP	< 7.6	> 7.6, < 8.6	> 8.6
Sulfate (mg/L)	MWP	< 5.9	> 5.9, < 12.5	> 12.5
Sulfate (mg/L)	TP	< 18.7	> 18.7, < 127.4	> 127.4

Appendix E. Summary of change analyses

Diff %Good/Poor: Statistically significant ($p < 0.05$) changes in the percentage of SOW community area represented by condition (G = good/P = poor) or stressor level (L = low/H = high) categories between cycles of the DWQA survey.

Diff Mean: Statistically significant ($p < 0.05$) differences in mean indicator values (units) between survey iterations with arrows indicating directionality of the change.

CDF test: Results of CDF test evaluating whether the distribution (i.e., cumulative distribution function) of indicator values has changed between surveys. “sig diff” = statistically significant ($p < 0.05$) difference in the distribution of indicator values between surveys.

Green text indicates condition improvement, while red text indicates degradation. Surveys: T1 = 2007/2008, T2 = 2012, T3 = 2017, T4 = 2023

Table E-1.

								Wetland Origin											
	Data Set:	Statewide						Natural						Human					
Indicator	Test	T1 vs T2	T1 vs T3	T1 vs T4	T2 vs T3	T2 vs T4	T3 vs T4	T1 vs T2	T1 vs T3	T1 vs T4	T2 vs T3	T2 vs T4	T3 vs T4	T1 vs T2	T1 vs T3	T1 vs T4	T2 vs T3	T2 vs T4	T3 vs T4
Invert IBI	Diff %Good/Poor								+18% P	+12% P					-32% P				
Invert IBI	Diff Mean (pts.)															↑ 10.8			
Invert IBI	CDF test																		
Chloride	Diff %High/Low		+16% H	+17% L		-20% H	-29% H		+18% H	+15% L		-21% H	-30% H	+21% G		+39% G			
Chloride	Diff Mean (mg/L)						↓ 3.8			↓ 3.6		↓ 5.4	↓ 3.0						
Chloride	CDF test					sig diff	sig diff					sig diff	sig diff						
NOx	Diff %High/Low						+10% L							+24% L		+29% L			
NOx	Diff Mean (mg/L)					↓ 0.2						↓ 0.2							
NOx	CDF test																		
Kjeld N	Diff %High/Low																		
Kjeld N	Diff Mean (mg/L)																		
Kjeld N	CDF test																		
Sulfate	Diff %High/Low		-29% L	-20% L	-19% L				-31% L	-23% L	-20% L								
Sulfate	Diff Mean (mg/L)	↑ 61.5	↑ 61.0				↓ 53.3	↑ 63.9	↑ 68.2				↓ 57.5						
Sulfate	CDF test		sig diff						sig diff										
Total P	Diff %High/Low	+18% L												+32% L	-30% H		-29% L		
Total P	Diff Mean (mg/L)																		
Total P	CDF test													sig diff					
Transparency	Diff %Good/Poor	-10% P			-19% G		+18% G	-11% P			-21% G		+18% G		-24% P		-23% P		
Transparency	Diff Mean (cm)										↓ 12.9								
Transparency	CDF test																		

Table E-2.

		Ecoregion											
	Data Set:	Mixed Wood Plains						Temperate Prairies					
Indicator	Test	T1 vs T2	T1 vs T3	T1 vs T4	T2 vs T3	T2 vs T4	T3 vs T4	T1 vs T2	T1 vs T3	T1 vs T4	T2 vs T3	T2 vs T4	T3 vs T4
Invert IBI	Diff %Good/Poor												
Invert IBI	Diff Mean (pts.)												
Invert IBI	CDF test												
Chloride	Diff %High/Low		+15% L				-23% H			+27% L		-34% H	-38% H
Chloride	Diff Mean (mg/L)												↓ 7.0
Chloride	CDF test									sig diff		sig diff	sig diff
NOx	Diff %High/Low				-8% L					-23% H		-27% H	-19% H
NOx	Diff Mean (mg/L)											↓ 0.6	
NOx	CDF test												
Kjeld N	Diff %High/Low											+31% L	
Kjeld N	Diff Mean (mg/L)												
Kjeld N	CDF test												
Sulfate	Diff %High/Low		-30% L	-21% L	-23% L	+16% H		+21% H					
Sulfate	Diff Mean (mg/L)		↑ 5.8	↑ 7.5	↑ 5.0	↑ 6.7		↑ 192.2	↑ 129.9			↓ 198.9	↓ 136.7
Sulfate	CDF test					sig diff							
Total P	Diff %High/Low	+20% L		+21% L									+23% H
Total P	Diff Mean (mg/L)	↓0.08									↑ 0.08		
Total P	CDF test		sig diff										
Transparency	Diff %Good/Poor				-24% G		+21% G						
Transparency	Diff Mean (cm)												
Transparency	CDF test												

Table E-3.

		Land Owner											
	Data Set:	Private						Public					
Indicator	Test	T1 vs T2	T1 vs T3	T1 vs T4	T2 vs T3	T2 vs T4	T3 vs T4	T1 vs T2	T1 vs T3	T1 vs T4	T2 vs T3	T2 vs T4	T3 vs T4
Invert IBI	Diff %Good/Poor												
Invert IBI	Diff Mean (pts.)												
Invert IBI	CDF test				sig diff								
Chloride	Diff %High/Low		+21% H	+15% L	+17% H	+14% L	-30% H	+21% L	+21% L			-31% H	
Chloride	Diff Mean (mg/L)					↓ 7.5	↓ 4.3						
Chloride	CDF test						sig diff					sig diff	sig diff
NOx	Diff %High/Low											+25% L	
NOx	Diff Mean (mg/L)						↓ 0.1	↑ 0.88				↓ 0.86	
NOx	CDF test												
Kjeld N	Diff %High/Low												
Kjeld N	Diff Mean (mg/L)												
Kjeld N	CDF test			sig diff		sig diff	sig diff						
Sulfate	Diff %High/Low		-21% L	-17% L	+7% H			-31% L	+30% H	+25% H			
Sulfate	Diff Mean (mg/L)								↑ 93.9				
Sulfate	CDF test								sig diff				
Total P	Diff %High/Low	+25% L		+24% L									
Total P	Diff Mean (mg/L)												
Total P	CDF test	sig diff	sig diff	sig diff									
Transparency	Diff %Good/Poor				-20% G		+21% G						
Transparency	Diff Mean (cm)				↓ 14.2		↑ 16.1						
Transparency	CDF test												