Wetland Quality

June 2019

Status and trends of wetlands in Minnesota: Depressional Wetland Quality Assessment (2007 – 2017)

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

MINNESOTA POLLUTION CONTROL AGENCY

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Foreword

This report is the third 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 2017, this report compares the latest data set to results from two previous depressional wetland surveys conducted in 2007 (report released in 2012) and 2012 (report released in 2015). The Minnesota Pollution Control Agency (MPCA) is exploring opportunities to integrate this survey with the broader Minnesota Wetland Condition Assessment—also conducted by the MPCA—for seamless coverage of all wetland types in the state.

Contents

Figures

Tables

Executive summary

From the prairie potholes in the southwest to the vast expanses of peatlands in the north, the diversity of Minnesota's wetlands is arguably unmatched by any other state. Although roughly half of Minnesota's original wetlands have been lost to draining or filling, public perception began to shift in the 1970s with recognition of the many ecological and societal benefits that wetlands provide. In Minnesota this trend lead 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, however, that a statewide wetland monitoring program was initiated to assess status and trends of both wetland quantity and quality, providing a way to evaluate 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. The focus of this report is on the 2017 results of the Depressional Wetland Quality Assessment (DWQA), evaluating the ecological condition of depressional marshes and ponds throughout the state and whether it has changed since the last assessment was completed in 2012.

The DWQA uses a survey approach to produce unbiased condition estimates for the entire population of depressional wetlands and ponds in the study regions based on results obtained from a sample of randomly selected sites. Unlike the initial baseline survey, the 2012 and 2017 surveys are limited to just the Temperate Prairies (TP) and Mixed Wood Plains (MWP) ecoregions in the state. Plant and macroinvertebrate indicators developed and calibrated by community type or ecoregion, respectively, were the primary indicators of wetland condition used in the DWQA. Criteria for categorizing the condition of each sample site as exceptional, good, fair, poor, or absent were established for each indicator relative to least- or minimally disturbed reference sites within each ecoregion (macroinvertebrates) or by community type statewide (plants). In the 2017 survey, a transition occurred for the plant community condition indicator, switching from the index of biological integrity (IBI) to the Floristic Quality Assessment (FQA). This was done to better align with the Minnesota Wetland Condition Assessment (MWCA), a comprehensive survey that encompasses all wetland community types found in Minnesota. In addition to biological indicators, several water quality parameters were measured at each study site to better characterize wetland condition.

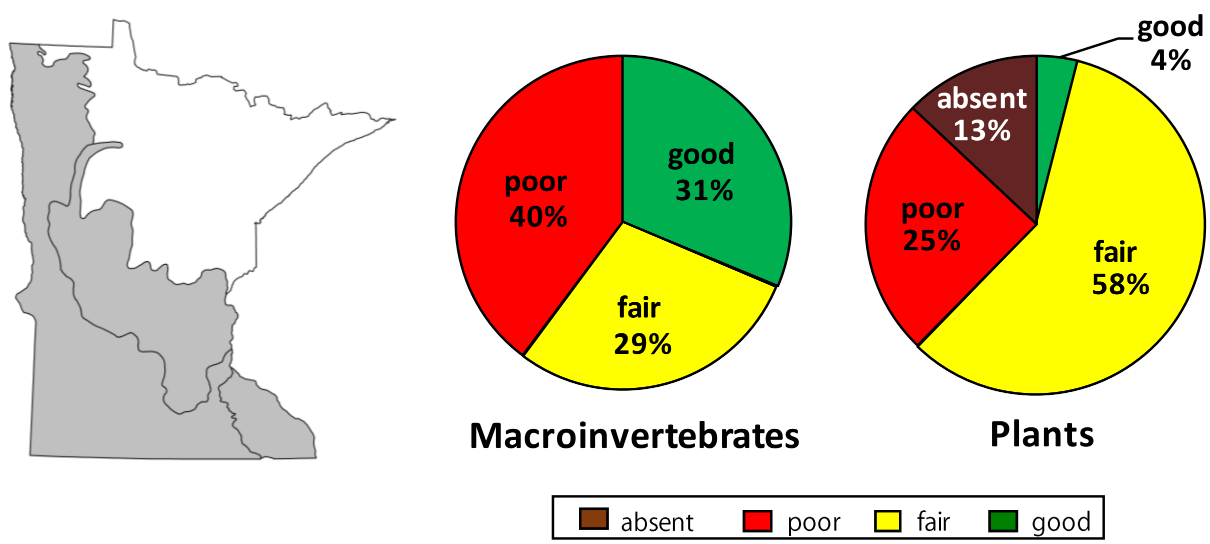
An estimated 102,185 depressional marshes and ponds—representing 577,174 acres—occur in the TP and MWP ecoregions of Minnesota. These estimates are not significantly different from those obtained in previous DWQA surveys, indicating relative stability in the quantity of this particular type of wetland. About two thirds of depressional marshes and ponds occur within the MWP ecoregion and in both ecoregions the vast majority of basins are on private property. In 2017, the estimated numbers of naturally formed and man-made wetlands were roughly equivalent due to the continuing decline of natural wetlands over the three surveys. Natural wetlands still hold a sizeable advantage in terms of their acreage with an estimated acreage 13 times that of man-made wetlands, demonstrating that on average man-made basins are much smaller than natural ones.

One hundred sites were monitored for the 2017 DWQA survey with 50 of these being ones that were also sampled in the 2012 survey (i.e., revisits). Aquatic macroinvertebrate communities (aquatic insects, snails, leeches, and crustaceans) are in good condition at 31% of depressional basins while 40% are in poor condition across the study area (Figure 1). While these estimates represent a slight decrease in condition compared to previous surveys, the changes are not statistically significant given the margins of error for this survey. An estimated 4% of depressional wetland basins have plant communities that are in good condition while 58% are fair, 25% are poor, and 13% receive an absent community rating (Figure 1). These estimates represent a dramatic and statistically significant decline in plant community condition compared to the previous two surveys. This decline is likely due to the shift in methodology that occurred in 2017, transitioning to the FQA indicator and individual plant community evaluations. Therefore, it is impossible at this time to ascribe the observed changes in wetland vegetation condition to actual changes on the landscape.

In terms of wetland area, approximately 50% of depressional wetland acres harbor aquatic macroinvertebrate communities that are in good condition. In contrast, only 9% of the depressional wetland acreage have good plant community condition in the MWP and TP ecoregions. The overall plant community condition ratings were highly influenced by the condition of two commonly occurring wetland plant communities: shallow open water and shallow marsh. Shallow marsh communities were in predominantly (71%) poor condition while shallow open water communities were primarily (70%) in fair condition when estimated by area. Changes in condition, based on wetland acreage, could not be conducted in this round of the DWQA due to methodological differences between survey cycles.

Condition and stressor results were compared among various wetland categories, such as natural vs. man-made or on public vs. private property, acknowledged in the DWQA. Based on those comparisons, the following is recommended: 1) Natural wetlands—particularly those > 12 acres and located on private property—have the highest biological condition and represent a top conservation priority; 2) Restoration or improvement efforts should focus on smaller wetlands (< 12 acres), particularly those that are man-made and located on public land; 3) Policies and practices that address invasive plant species as well as excess chloride, phosphorus, and sediment inputs will be most effective for restoring depressional wetland condition; and 4) Individual wetland restoration or protection activities should always conduct a thorough evaluation of possible stressors or potential threats to each site, considering that it may identify ones not directly measured in this survey (e.g., pesticides, herbicides, altered water levels, invasive fish).

Figure 1. Aquatic macroinvertebrate and plant community condition estimates for Minnesota's depressional wetlands and ponds in 2017. Shaded area of map indicates spatial extent of survey.



Change analyses were run for individual wetland categories to evaluate whether condition is changing for certain types of wetlands that otherwise would have gone undetected had analyses only been run with all categories pooled together. In the MWP ecoregion both man-made wetlands and wetlands occurring on public property exhibited declines in macroinvertebrate community condition as well as increases in stressors such as chloride, Kjeldahl nitrogen, and total phosphorus. Concurrently, natural wetlands in the TP ecoregion experienced increased chloride and Kjeldahl nitrogen concentrations in 2017 relative to the previous surveys. Similar changes existed among the categories when the ecoregions were pooled in the analyses; however, they were not as distinct as those observed within the ecoregions.

Overall, given the transition in methodology for evaluating wetland vegetation condition, it is difficult to evaluate whether “no-net-loss” goals are being achieved for wetland quality and biological diversity based on the 2017 survey results. Relative stability in aquatic macroinvertebrate community condition over the three surveys (i.e., no significant change) would suggest that at present, the quality of depressional wetlands and ponds is being maintained. However, it won’t be possible to evaluate this more comprehensively until multiple survey cycles are completed with the new FQA method. The next DWQA is scheduled to occur in 2022. In the meantime, the 2017 survey provides useful information on areas and types of depressional wetlands that should be prioritized for protection or condition restoration as well as a list—though not exhaustive—of the primary stressors that should be addressed.

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. Public recognition of the value of wetlands culminated in Minnesota with the passage of the WCA in 1991.

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, 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 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). Initial change analyses of the WSTMP have indicated a possible reversal of this historical wetland loss trend by demonstrating small (but significant) net gains in wetland acreage across the first three cycles of the survey: 2006 – 2014 (Kloiber and Norris 2017). While these results are encouraging, it is important to point out that the largest gains come from un-vegetated ponds and that gains in emergent wetlands are largely offset by conversions to ponds and farmed wetlands. Given historical losses of wetlands in Minnesota (~50%) and concerns regarding the types of wetlands currently increasing in quantity, monitoring wetland quality is essential for understanding whether the numerous ecosystem services that wetlands provide are maintained at local, watershed, and regional scales.

The MPCA is currently conducting two somewhat independent wetland quality surveys: the Depressional Wetland Quality Assessment (DWQA) and the Minnesota Wetland Condition Assessment (MWCA; see Table 1 for comparison). The findings of these surveys are intended to compliment the results of the WSTMP, together providing a comprehensive assessment of WCA's no net loss goal.

The MPCA is exploring opportunities to integrate the two wetland quality surveys for seamless coverage of all wetland types in the state. The focus of this report, however, is on results from the third round of the DWQA and whether conditions have changed relative to previous iterations of the survey.

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

Characteristics	Depressional Wetland Quality Assessment	Minnesota Wetland Condition Assessment
Wetland community types	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)
Condition indicators	macroinvertebrates, vegetation	vegetation
Stressor indicators	surface water chemistry, human disturbance assessment	human disturbance assessment
Reporting units	basin or acres	acres
Spatial extent	central forested and prairie regions	statewide
Cycles completed	3	2
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.

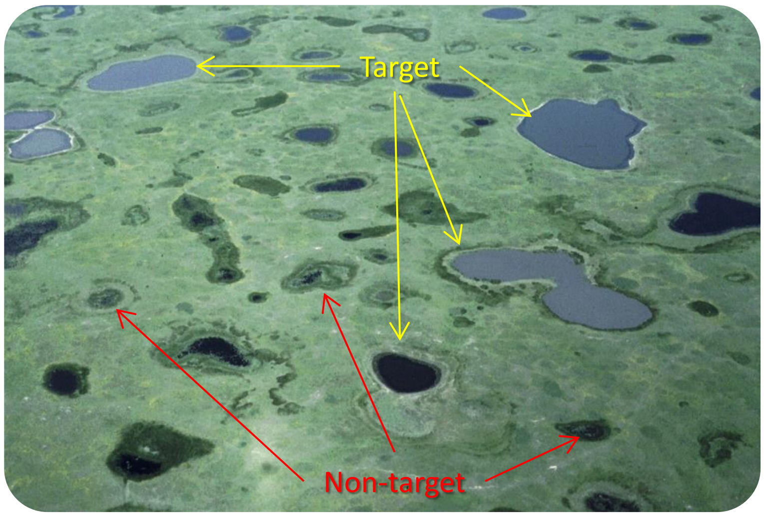
The MPCA began developing IBIs for wetlands in the early 1990s, focusing on depressional marshes and ponds. During this work, attributes of the aquatic plant and macroinvertebrate (aquatic insects, snails, leeches, and crustaceans) communities were investigated to determine their response pattern along a gradient of human disturbance. These efforts resulted in the development and validation of ecoregion-specific, wetland IBIs (Appendix A). The first two DWQA surveys utilized plant and macroinvertebrate IBIs as indicators of wetland condition. At the start of 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 (see box 'What's new in the DWQA?').

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

Methods

The focus of the DWQA is depressional wetlands that are semi-permanently to permanently flooded and comprised primarily of herbaceous vegetation around the margin with open water in the interior. These wetlands occupy areas of low relief or depressions in the landscape, and are commonly referred to as potholes in the prairie region (Figure 2). Disturbance, natural or otherwise, can result in a lack of submergent and/or emergent vegetation in these wetlands, making them indistinguishable from man-made ponds in some cases. Rather than attempting to distinguish between disturbed wetlands and man-made ponds, which often requires knowing the history of a site, both vegetated and un-vegetated basins were included in the survey. Furthermore, the authors felt that this decision was appropriate since wetland quantity surveys (e.g., Dahl 2011, Kloiber and Norris 2013, Dahl 2014, Kloiber and Norris 2017) include open water wetlands and ponds in their wetland acreage estimates and evaluations of no-net-loss. For further details regarding the target population and wetland classification see Genet (2012).

Figure 2. Prairie potholes are an example of the type of wetland included in the DWQA. Temporary and seasonally flooded wetlands were not included in this survey (= non-target). Aerial photo courtesy of USFWS.



The DWQA utilizes Level II ecoregions (Omernik 1987, White and Omernik 2007) as a geographical framework that aims to improve the ability of indicators to distinguish human disturbance from natural variability. Three major ecoregions converge in Minnesota with the Temperate Prairies (TP) occupying the western and southern portions, the Mixed Wood Plains (MWP) occupying the central and southeastern portions, and the Mixed Wood Shield (MWS) occupying the north central and northeastern portion of the state (Figure 3). The baseline DWQA included all three ecoregions. However, due to the relative scarcity of target depressional wetland types 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 quality 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.

What's new in the DWQA?

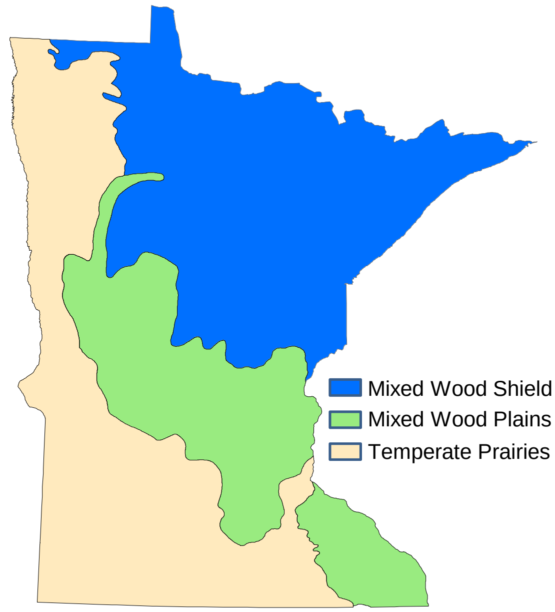
- **Vegetation condition now measured using Floristic Quality Assessment.** The approach for measuring the condition of plant communities was changed from the IBI to the Floristic Quality Assessment or FQA (Bourdaghs 2012). This method focuses on assessing the condition of individual plant communities within the wetland rather than the entire basin and relies upon a more refined assessment framework called the Biological Condition Gradient. The FQA sampling approach also more fully represents the vegetation occurring in wetland sites compared to the single sample plot used in the IBI method.
- **Repeat sites included in analyses of wetland extent (i.e., # basins or acres).** In 2012, sites sampled in two consecutive rounds of the survey (revisit sites) were excluded from analyses that estimated the total extent of depressional wetlands or subpopulations thereof. At that time it was believed that these sites, due to their relatively high inclusion probability, carried a disproportionate amount of weight in these analyses. Therefore, rather than just using the new sites (or ~50% of the total sample size), all sample sites were utilized in the estimation of wetland extent. The 2012 data set has been re-analyzed to reflect this approach and thus extent results presented here will not match those in the 2012 report.
- **Reporting results by wetland area re-emphasized.** Due the discrete nature of depressional wetlands and ponds, reporting condition results based on the percentage of basins—or the number of individual waterbodies—is still the primary focus of this report. However, to better accommodate comparisons with the MWCA and the WSTMP, condition results will also be presented based on the percentage of wetland acres in numerous places throughout the report. Results may differ dramatically based on the reporting unit due to the large weight that small wetlands have in analyses by basin versus the small weight they have in analyses by acre, while large wetlands have the opposite relationship.
- **Added HDA as a measure of stress.** The Human Disturbance Assessment (HDA) was added to the DWQA in 2017 as a way to further characterize the degree of human-related impacts to depressional wetlands and ponds. A comprehensive measure of stress, the tool combines an evaluation of the landscape surrounding a wetland with field observations of impacts occurring within the wetland.
- **Changed how invasive plant species characterized.** In past surveys, abundance of invasive plants was estimated within representative plots and used to characterize each study site as being low (< 20%), medium (20 – 50%), or high (> 50%). The FQA requires a different approach to estimate invasive species cover across the entire site, considering how individual plant communities are evaluated and community-specific cover estimates are produced. An overall qualitative rating of invasive species abundance is also provided within the HDA framework based on observations made by the plant survey crew. Both of these new approaches require further investigation into how they can be used to inform the DWQA.

Survey design

Similar to how an opinion poll gauges public interest on a topic or candidate running for office, the DWQA utilizes survey techniques that allow estimates (± margin of error) to be generated for an entire group of wetlands by measuring a comparatively small sample of sites. Wetlands are randomly selected to ensure that derived estimates are unbiased. In addition, the selection process is spatially stratified (Stevens and Olsen 2004) to increase the likelihood that the sample represents all regions of the state. To maximize participation in the survey, all landowners are contacted in the weeks prior to sampling to obtain permission and/or the appropriate permits.

The DWQA uses updated wetland spatial data from the permanent plots (1 mi^2) of Minnesota's WSTMP (Kloiber and Norris 2013) to randomly select a sample of depressional wetlands and ponds each cycle of the survey. Since the plots also represent a sample of the entire state (~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 and 2017 DWQA were limited to the MWP and TP ecoregions (i.e., the DWQA's new definition of statewide) and sampling in both ecoregions was completed in the same year, eliminating any confounding effects of 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 3. Level II ecoregions in Minnesota.



The target number of 2017 DWCA sample sites was 100—split evenly by ecoregion with a 50% revisit rate of 2012 sample 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: < 2.5 acres (< 1 ha), 2.5-12.4 acres (1-5 ha), and > 12.4 acres (> 5 ha). Site selection was provided by the U.S. Environmental Protection Agency (EPA) National Health and Environmental Effects Research Laboratory, Corvallis, Oregon.

Figure 4. Stormwater retention pond in Scott County.

Based on an evaluation of site characteristics as well as aerial imagery, sampled survey sites were classified as either 'natural' or 'man-made'. Examples of man-made basins include stormwater retention ponds (e.g., Figure 4), golf course water hazards, livestock ponds, and ornamental ponds. If a survey site has been created for mitigation purposes, it was classified as 'natural' since it is intended to replace a wetland that has been drained, filled, or physically altered. Waterbodies that require continuous pumping or lining to maintain their hydrology were not included in this survey.

Field methods

Prior to sampling, each of the potential study sites was investigated using GIS applications to determine ownership and obtain access permission. If permission was granted, sites were visited in May to evaluate whether they met specifications of the survey (i.e., semi-permanently flooded, depressional wetland or pond) and to determine their origin (man-made vs. natural). If sites had to be dropped from the survey for any reason (e.g., landowner access denial, non-target), replacement sites were added in sequential order from the random selection until the desired sample size of 50 sites/ecoregion was reached.

The aquatic macroinvertebrate community of each site was sampled in June using a D-frame dip net with a 500 µm mesh size. Macroinvertebrates were primarily collected from the emergent vegetation zone in depths ranging from 0.3 – 1 m. If emergent vegetation was not present within the wetland the following zones (listed in decreasing order of preference) were sampled at similar depths: floating-leaved aquatic vegetation, submergent aquatic vegetation, and open water (< 25% vegetation cover). Samples were collected by sweeping the 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 25 m radius) were collected and placed on hardware cloth screen (1.3 x 1.3 cm mesh) overlaying two plastic pans to separate the macroinvertebrates from the vegetation that invariably gets swept into the net. Over a period of ten minutes, vegetation was spread apart on the hardware cloth to allow macroinvertebrates to drop or crawl 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 spreading process was repeated, after which the vegetation 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 separate macroinvertebrate samples. Samples were sent to a taxonomy lab for identification of macroinvertebrates. More information on the dip net method is included in 'Macroinvertebrate Community Sampling Protocol for Depressional Wetland Monitoring Sites' available at: http://www.pca.state.mn.us/index.php/view-document.html?gid=6101.

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

Chemical and physical properties of the water column were measured during the June macroinvertebrate sampling visit. A multi-meter (Hach HQ40d18) was used to measure water temperature (^oC), dissolved oxygen (mg/L), specific conductance (µS/cm), and pH. Water samples were collected from the near shore zone of each site just below the water surface and packed in ice until delivery 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), total organic carbon (mg/L), chloride (mg/L), sulfate (mg/L), and alkalinity (mg/L) was determined in each sample using standard protocols (Appendix B). Water column transparency or clarity was measured using a 100 cm Secchi tube. Details of the water chemistry sampling procedure can be found in 'Water Chemistry Assessment Protocol for Wetland Monitoring Sites' available at: http://www.pca.state.mn.us/index.php/view-document.html?gid=10251.

Vegetation species composition and abundance were characterized according to wetland plant community types at each survey site between June and mid-September. Plant communities present within the site were determined and their extent was mapped on printed aerial photos. The Eggers and Reed (2011) classification of wetland plant communities of Minnesota and Wisconsin was followed (Table 2). A timed meander sampling approach was used to collect vegetation data—where the observer walked through the study site and recorded all vascular plant taxa by community type as they were encountered (Bourdaghs 2014). Taxa were identified to the lowest taxonomic division possible in the field, typically species or subspecies. When taxa could not be identified to the species level—specimens were collected, pressed, and dried for lab identification. Areal cover for each taxa by community type was then estimated according to cover classes (Table 3). In this way, the entire site was essentially treated as a large sampling plot divided into plant communities.

Table 2. Eggers & Reed (2011) plant community classes that represent the target population of the DWQA. Two classes have been slightly modified from the original classification (Bourdaghs 2012). Fresh Meadow combines both the Eggers and Reed Sedge Meadow and Fresh (Wet) Meadow classes into a single class.

Community class	Community class description
Shallow open water	Open water aquatic communities with submergent and floating leaved aquatic species
Deep marsh	Emergent vegetation rooted within the substrate that is typically inundated with > 6" of water. Submergent and floating leaved aquatic species typically a major component of community
Shallow marsh	Emergent vegetation on saturated soils or inundated with typically < 6" of water. May consist of a floating mat. Submergent and floating leaved aquatic species typically a minor component
Fresh meadow	Graminoid dominated, soils typically saturated
Rich fen	Graminoid dominated communities on circumneutral or slightly acidic peat soils. Often occurs as a floating mat and *Carex lasiocarpa* (wiregrass sedge) is often a dominant
Shrub-carr	Tall shrub community typically dominated by Willows (*Salix* spp.). Typical understory species composition similar to Fresh Meadow

Table 3. Cover classes, percent cover ranges, and midpoints.

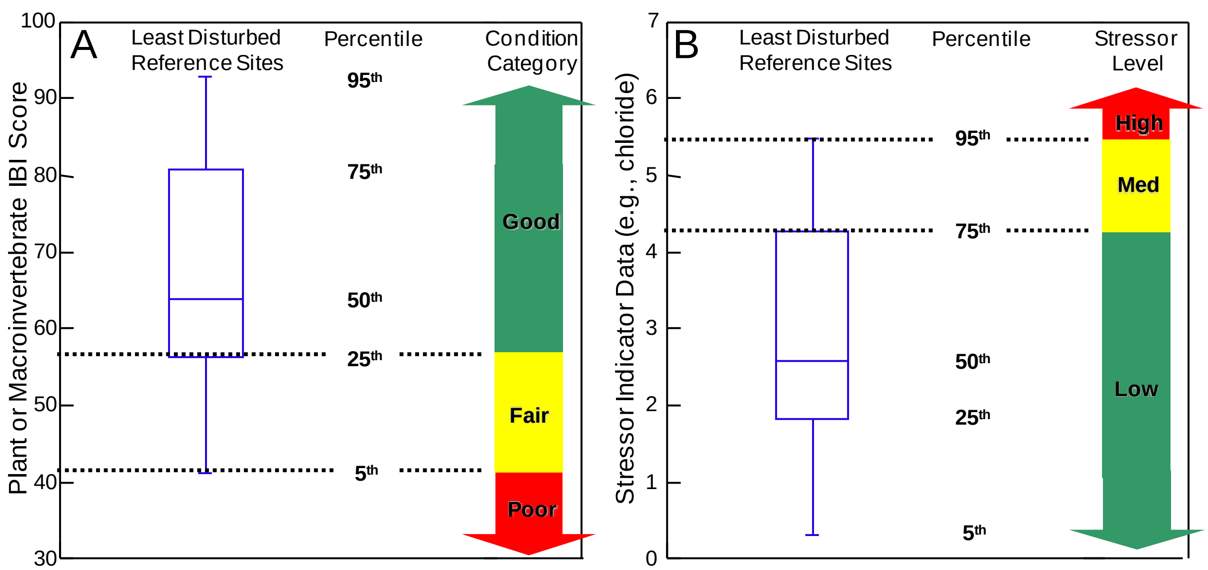
Cover class	Cover class range	Midpoint
7	> 95 - 100%	97.5%
6	> 75 - 95%	85%
5	> 50 - 75%	62.5%
4	> 25 - 50%	37.5
3	> 5 -25%	15%
2	> 1 - 5%	3%
1	> 0 - 1%	0.5%

Data analysis

Data from both macroinvertebrate samples collected at each wetland 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 A 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 stressor indicator data collected from reference sites were used to represent the range of expected values for least-impacted conditions within each ecoregion. The distribution of each indicator data set was used to establish thresholds between good/fair/poor condition categories or high/medium/low stress categories (Figure 6A and B). The 25th percentile of the reference distribution was used to separate the good and fair condition categories. Study sites with indicator values above this threshold are considered to be comparable to least-impacted reference sites and in good condition (Figure 6A). The 5th percentile of the distribution of least-impacted scores was used to separate the fair and poor categories, meaning that wetlands in the poor category 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 C. Reference site selection criteria can be found in the baseline DWQA report (Genet 2012) as well as Genet et al. (2004).

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 sample site. Sites were categorized independently based on each indicator.



Floristic Quality Assessment

Following field sampling, plant community mapping was completed using GIS based on field GPS data, the hand drawn maps, and aerial photo interpretation. A categorical Human Disturbance Assessment (HDA; Bourdaghs 2012) was completed at each site to describe the exposure of wetlands to human impacts (i.e., stressors). The HDA incorporates six well-documented factors that have been associated with degraded wetland vegetation condition:

- Surrounding landscape alteration (500 m buffer)
- Immediate upland alteration (50 m buffer)

- Within wetland physical alteration (e.g., plowing, logging, etc.)
- Hydrologic alteration (e.g., partial drainage, directed inputs, etc.)
- Chemical pollution (e.g., excess sediment or nutrients, human sources present)
- Non-native invasive species

Each HDA factor was rated separately as minimal/low/moderate/severe using best professional judgment according to standard narrative criteria. Ratings were based on aerial photo interpretation and field observations. An overall HDA rating of minimally/moderately/severely impacted was then determined based on combinations of the individual factor ratings. Complete HDA documentation is provided in MPCA (2015).

The primary FQA metric used to quantify vegetation condition from the community data was the weighted Coefficient of Conservatism (*wC*), which is the sum of each species' proportional abundance (*p*) multiplied by its *C*-value:

$$wC = \sum pC$$

In this case, the abundance measure used to calculate *p* was the midpoint percent cover estimate from the observed cover classes (Table 3). *wC* incorporates both species composition and abundance, is not affected by sampling area, and has been found to be a more responsive indicator of wetland condition than FQA metrics that rely on species composition alone (Bourdaghs 2012).

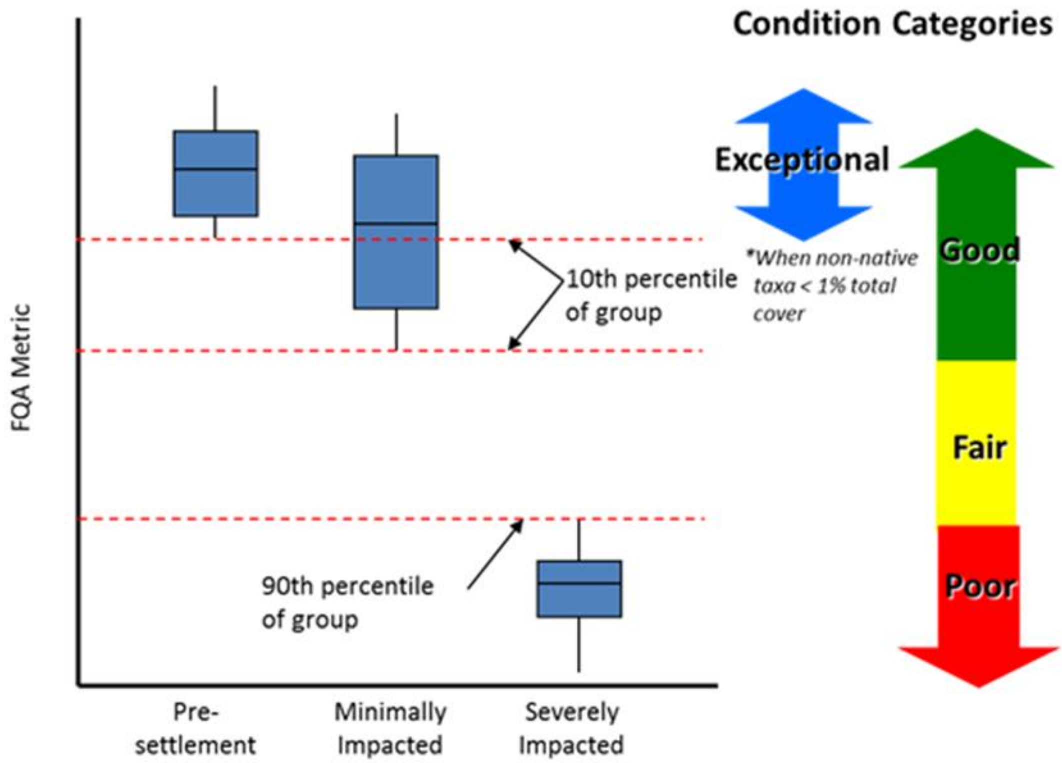
The FQA assessment framework for Minnesota wetlands used to translate quantitative *wC* scores into meaningful results was built around a general model of biological response to anthropogenic impacts called the Biological Condition Gradient (BCG; US EPA 2005). The BCG describes biological condition according to categories that range from conditions that are equivalent to those thought to be found prior to European settlement to conditions that are found at sites known to be severely impacted by human activities. A five-level BCG model specific to wetlands has been developed to serve as criteria for evaluating vegetation condition (Table 4). Numeric *wC* assessment criteria have been established by calibrating *wC* scores to the BCG using a large dataset (Bourdaghs 2012). This was done by assigning targeted data to three analysis groups (pre-settlement, minimally impacted, and severely impacted) based on HDA and Minnesota Biological Survey condition ratings (DNR 2009), and establishing thresholds at the 10th percentile values for the pre-settlement and minimally impacted groups and the 90th percentile value of the severely impacted group (Figure 7). *wC* assessment criteria were developed for each plant community (Appendix C) as both the expected natural and impact response ranges differ by type (Bourdaghs 2012).

Table 4. Wetland vegetation condition category descriptions.

Condition category	Description
Exceptional (1)	Community composition and structure as they exist (or likely existed) in the absence of measurable effects of anthropogenic stressors representing pre-European settlement conditions. Non-native taxa may be present at very low abundance and not causing displacement of native taxa.
Good (2)	Community structure similar to natural community. Some additional taxa present and/or there are minor changes in the abundance distribution from the expected natural range. Extent of expected native composition for the community type remains largely intact.
Fair (3)	Moderate changes in community structure. Sensitive taxa are replaced as the abundance distribution shifts towards more tolerant taxa. Extent of expected native composition for the community type diminished.

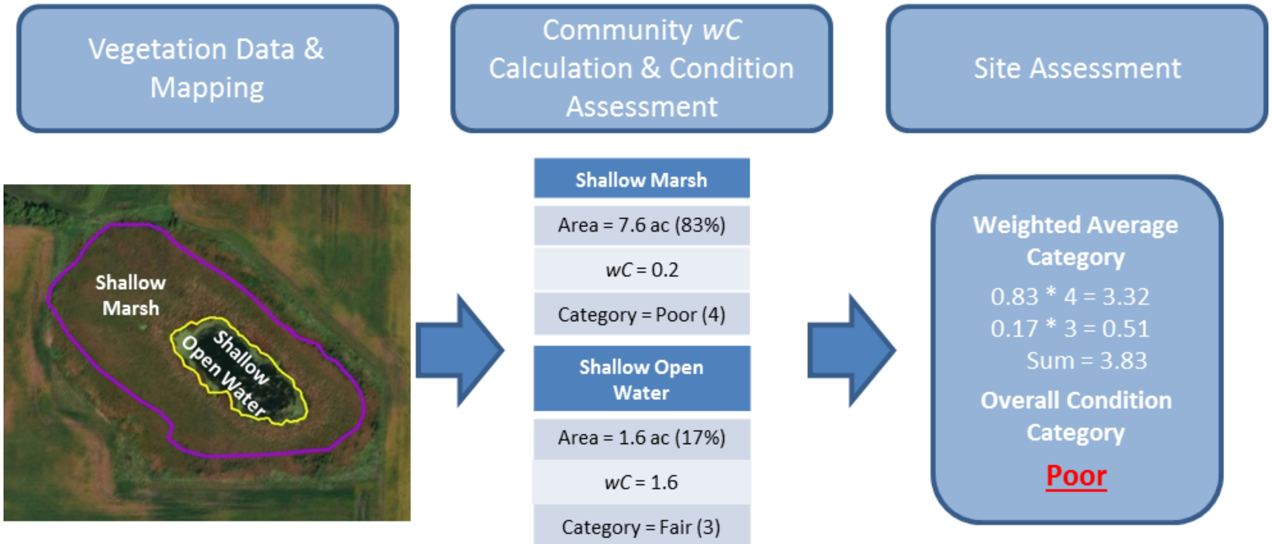
Condition category	Description
Poor (4)	Large to extreme changes in community structure resulting from large abundance distribution shifts towards more tolerant taxa. Extent of expected native composition for the community type reduced to isolated pockets and/or wholesale changes in composition.
Absent (5)	Plant life only marginally supported or soil/substrate largely devoid of hydrophytic vegetation due to ongoing severe anthropogenic impacts.

Figure 7. Diagram of FQA assessment criteria threshold development. Community samples were assigned to data analysis groups based on the degree of exposure to human impacts (pre Settlement, minimally impacted, or severely impacted). Thresholds were determined at designated percentiles of the FQA metric distribution for each data analysis group that correspond to the condition categories (Table 8). Separating the exceptional and good threshold required an additional narrative criterion (< 1% non-native taxa cover) to be met.



Because data were gathered by (and assessment criteria were specific to) plant community type, completing a final assessment for each study site was a multi-step process (Figure 8). *wC* scores were first calculated for each community present in a wetland based on the vegetation data and *C*-values. The condition category for each type was then determined according to the established community assessment thresholds (Appendix C). Community condition results were then aggregated to the site scale by calculating the weighted average condition category based on the relative extent of each community present derived from the community mapping (Figure 8).

Figure 8. Process to complete a depressional site level vegetation assessment: 1) vegetation data are gathered by plant community; 2) *wC* is calculated and the condition category of each community is determined; and 3) community results are aggregated by a weighted average based on the relative extent of each community type and rounded to the nearest whole number which corresponds to the condition category.



Estimation procedures

Condition and stressor categorization criteria were used to rate indicator results individually for each study site. Initial design weights were adjusted based on the exclusion of sites that were confirmed as non-target during site evaluation. Sites where access permission was denied were assumed to be target based on the desktop evaluation 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 3.5.0 (R Core Team 2018) using the spatial survey design and analysis package (spsurvey 3.3; Kincaid and Olsen 2016). An analysis script was written to estimate 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. 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. To compare results of subpopulations (e.g., man-made vs. natural wetlands), cumulative distribution function (CDF) tests were performed on quantitative indicator data using spsurvey 3.3 (Kincaid and Olsen 2016). However, analyses of combined *wC* results across all plant communities present within a site would not be appropriate due to each community having independently calibrated condition scales. Rather, comparisons of vegetation condition were based on community-specific *wC* scores when the community was present in at least 10 sites of each subpopulation. Unlike the previous DWQA that reported only on numbers of basins, results in this report are provided on both numbers of basins as well as wetland area (acres).

Statewide results and discussion

Data collected in 2017 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 since the previous survey cycle. In this section, wetland quantity status and change results are presented first, followed by wetland condition status, and then changes in wetland condition for the combined MWP and TP ecoregions.

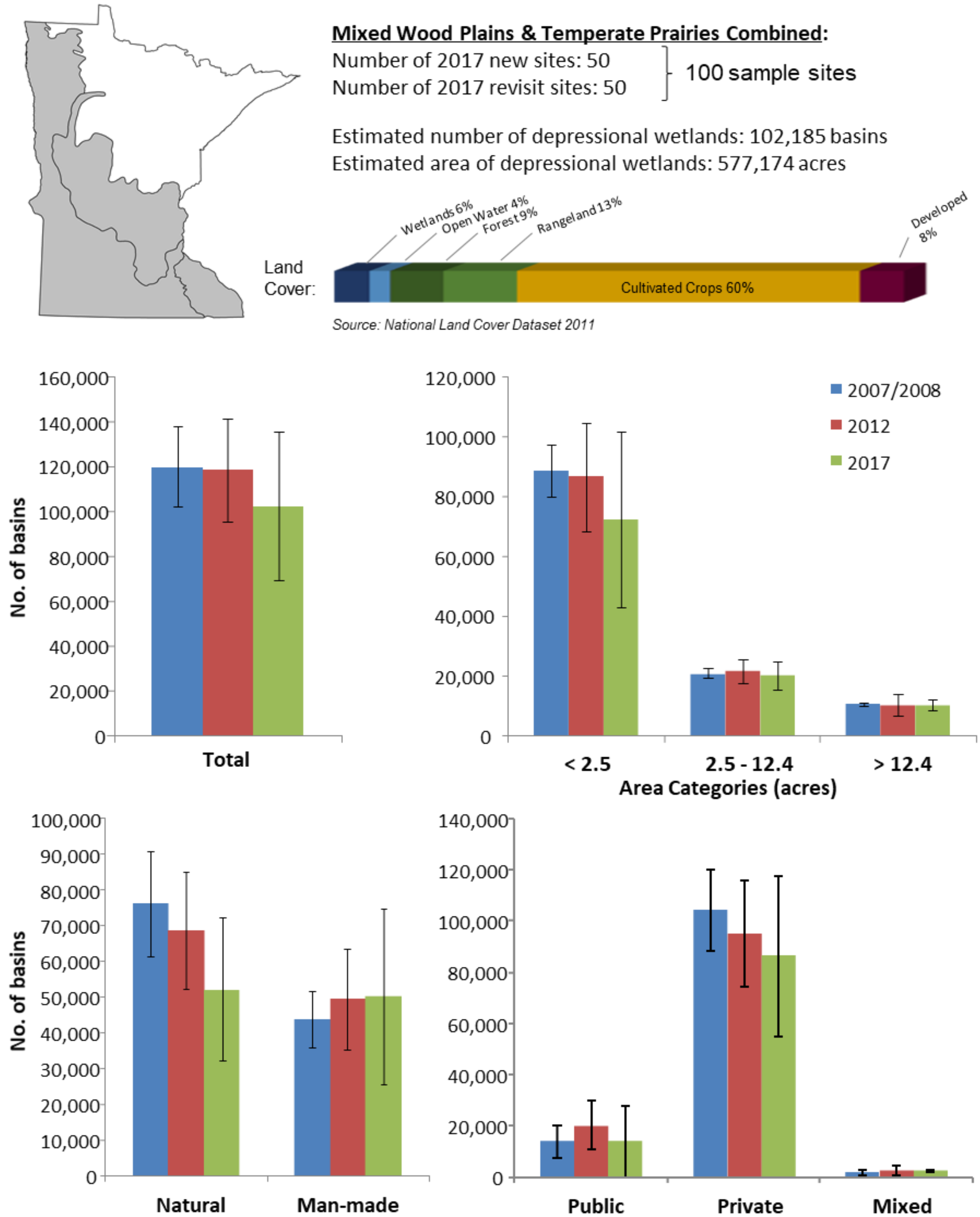
Depressional wetland quantity (status and change)

Based on the 2017 survey results there are an estimated 102,185 depressional wetlands and ponds occurring in the combined area of the MWP and TP ecoregions. Recall that this estimate only pertains to semi-permanent to permanent wetlands and ponds, and does not include ephemeral, temporary, or seasonal wetland basins. As noted previously, revisit sites—those that were sampled in the previous survey as well—were included in the extent estimation procedure. Considering this new approach, the 2012 basin estimates needed to be revised and has increased from the previous estimate of 111,335 to 117,963 basins for the combined ecoregions. Across the first three survey cycles, the number of depressional wetlands and ponds appears to be declining (Figure 9). It is likely that some of this decline, however, can be attributed to the ongoing evaluation of potential study sites that occurs with each survey. As non-target wetlands are identified and removed from the pool of potential sites (i.e., sample frame), estimates for total extent will decrease. This correction process should diminish over time, resulting in change estimates that are primarily driven by the restoration, draining, or filling of depressional wetlands on the landscape as identified by WSTMP updates.

Declines in the number of wetland basins were evident amongst natural wetlands, privately owned wetlands, and wetlands in the smallest wetland area category (Figure 9). However, considering all comparisons between 2017 and the previous two cycles of the survey, only the decline in natural wetland basins between 2007/2008 and 2017 was statistically significant ($Z = -1.87$, $p = 0.031$). The number of man-made wetlands remained relatively stable in comparison to the 2012 estimate, as did medium (2.5-12.4 acres) and large (> 12.4 acres) wetlands. It is notable, the shift in the distribution between natural and man-made wetlands; whereas there were many more natural wetlands relative to man-made wetlands in 2007/2008, they are virtually even in 2017. Given the large margins of error associated with these estimates, the observed changes are not yet cause for concern. Rather, it highlights the importance of continued monitoring to determine whether these initial patterns in the data amount to actual trends.

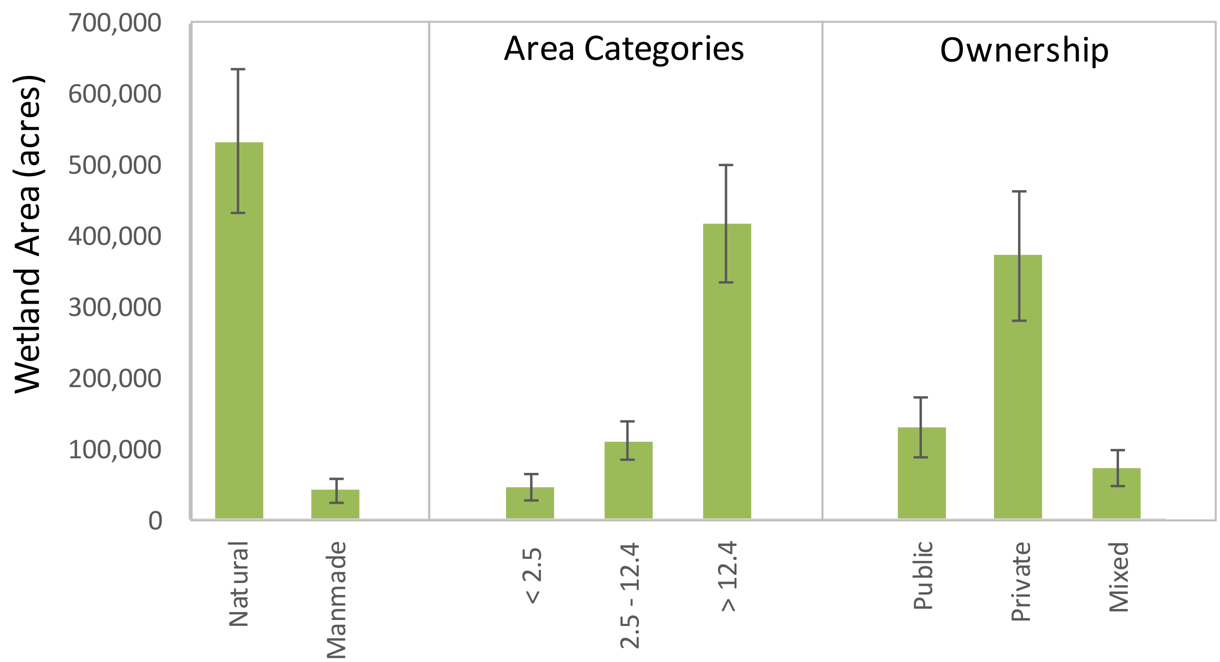
The estimated area of depressional wetlands and ponds in the combined MWP and TP ecoregions was 577,174 acres. Comparing this result to the estimated number of basins yields an average basin area of 5.6 acres. According to MWCA, the estimated total area of DWQA target community types (i.e., Table 2) is 2.6 million acres (± 191,000), indicating that depressional wetlands and ponds—as defined in this survey—account for about 22% of this total. Total wetland acreage could not be estimated for past surveys in a manner consistent with the approach taken to produce the 2017 area results. This was due to the shift to the FQA and its level of precision for mapping plant communities within the wetland. Detailed community mapping resulted in the adjustment of the original area (i.e., recognized in the survey design) and associated area category for numerous study sites, affecting estimation procedures of total area extent. Area estimates from past surveys could not be revised based on the new FQA methodology.

Figure 9. Estimates of the total number of depressional wetlands in the Mixed Wood Plains and Temperate Prairies ecoregions over the three cycles of the DWQA survey. Bracketed lines represent the 95% confidence interval associated with each estimate.



In contrast to the approximately equal number of natural and man-made basins (Figure 9), natural basins account for the majority of depressional wetland area in 2017 (Figure 10). The distribution of depressional wetland area also exhibits an opposite pattern than what is observed based on the number of basins for the three wetland size categories; large wetlands (> 12.4 acres) accounts for the majority of depressional wetland area (Figure 10). This is important because it explains why condition results reported as numbers of basins often contrast with results reported as area: small (< 2.5 acre) wetlands have substantially more weight in the analyses by basin while large wetlands have more weight in the analyses by area. The pattern amongst the three ownership categories was the same regardless of whether evaluating by basin or area –the majority of depressional wetlands and ponds occurred on private property (Figures 9 and 10).

Figure 10. Comparisons of 2017 wetland acreage estimates among the various subpopulations of the DWQA survey design. Bracketed lines represent the 95% confidence interval associated with each estimate.

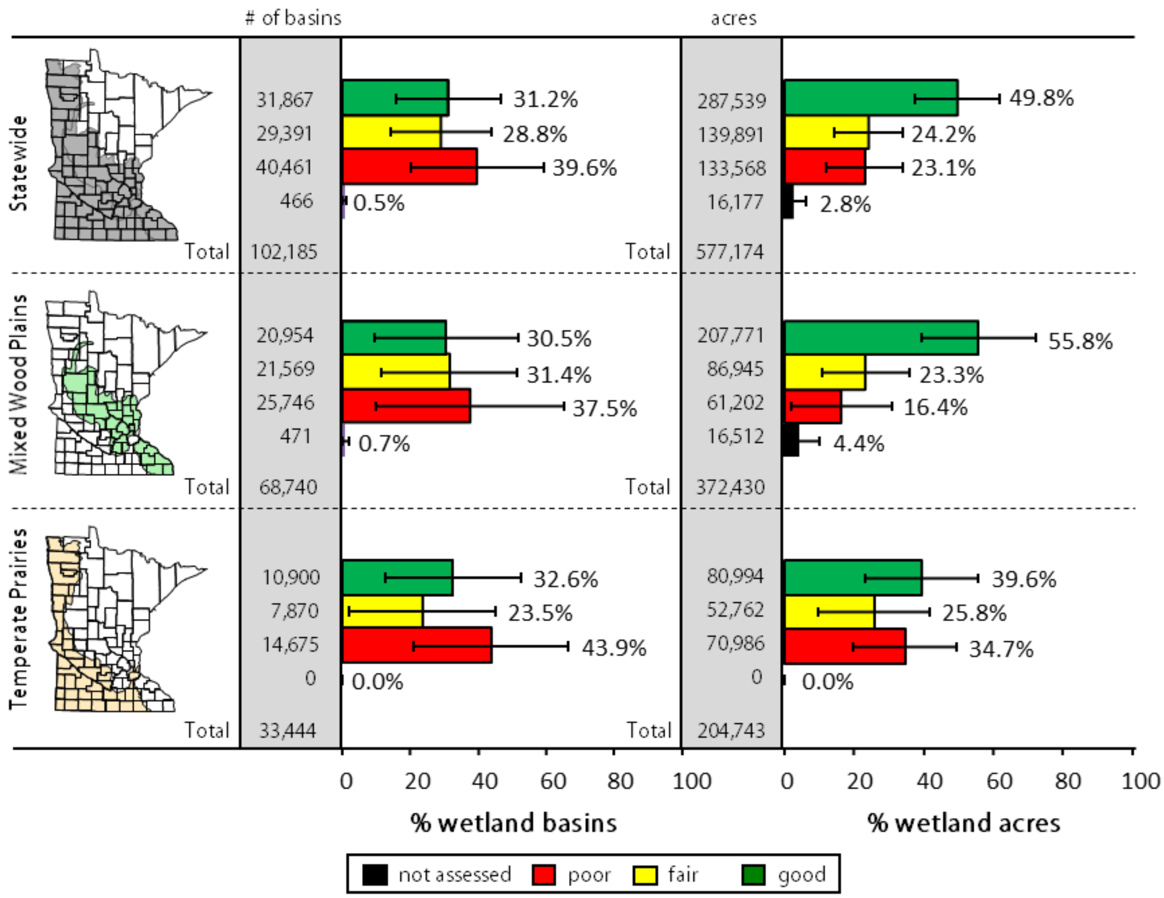


Depressional wetland condition

Aquatic macroinvertebrate communities in depressional wetlands and ponds have a relatively even distribution amongst the three condition categories in terms of the number of basins (Figure 11). The TP ecoregion exhibits the largest departure from an even split, with 44% poor. In terms of wetland area, by contrast, the percentage of acres in the good category is twice (or more) the percentage of acres in either the fair or poor categories for the statewide and MWP analyses. The difference between the basin and area results is due to the influence of large wetlands on the area analysis and their relatively healthy macroinvertebrate communities. Overall, for the two ecoregions combined, approximately 40,000 depressional wetland and pond basins are in poor condition, which is the equivalent of approximately 134,000 acres. Comparisons in the sections below provide insight on the factors that affect aquatic macroinvertebrate community condition.

The condition of vegetation in depressional wetlands and ponds is substantially worse than the condition of aquatic macroinvertebrates. Regardless of whether reporting by basin or acreage, only a small percentage of wetland plant communities are in good condition in any of the regions (Figure 12). The transition to the FQA resulted in two additional condition categories: exceptional and absent. However, at the site level—as opposed to individual plant communities—no depressional wetlands or ponds with an exceptional vegetation rating were observed in 2017. An absent vegetation condition was most prevalent in the MWP ecoregion (Figure 12) and was exclusive to the shallow open water community. The majority of depressional wetland basins and acres are in fair condition. Overall, for the two ecoregions combined, approximately 38,000 basins have a vegetation community rating less than fair (i.e., poor or absent), which is equivalent to 185,000 acres.

Figure 11. Aquatic macroinvertebrate community condition of Minnesota's depressional wetlands and ponds in 2017, estimated in terms of the number of basins as well as their corresponding 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.



The lower percentages of basins or acres rated as good for vegetation are likely due to the predominance of non-native, invasive plant species in depressional wetlands and ponds (see Wetland Stressors). While these species have direct impacts on the plant community, their role as a stressor to the aquatic macroinvertebrate community is more indirect in nature and currently there appears to be no equivalent invasive macroinvertebrate species that would cause similarly severe impacts to this community in depressional wetlands. Invasive fish species such as fathead minnow (*Pimephales promelas*) and common carp (*Cyprinus carpio*) are certainly candidates for having strong negative effects on the macroinvertebrate community—through their disruptions to the food web and physical habitat (Hanson and Riggs 1995, Anteau and Afton 2008, Sundberg et al. 2016)—but were not quantified in this survey.

Figure 12. Plant community condition of Minnesota's depressional wetlands and ponds in 2017, estimated in terms of the number of basins as well as their corresponding 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.

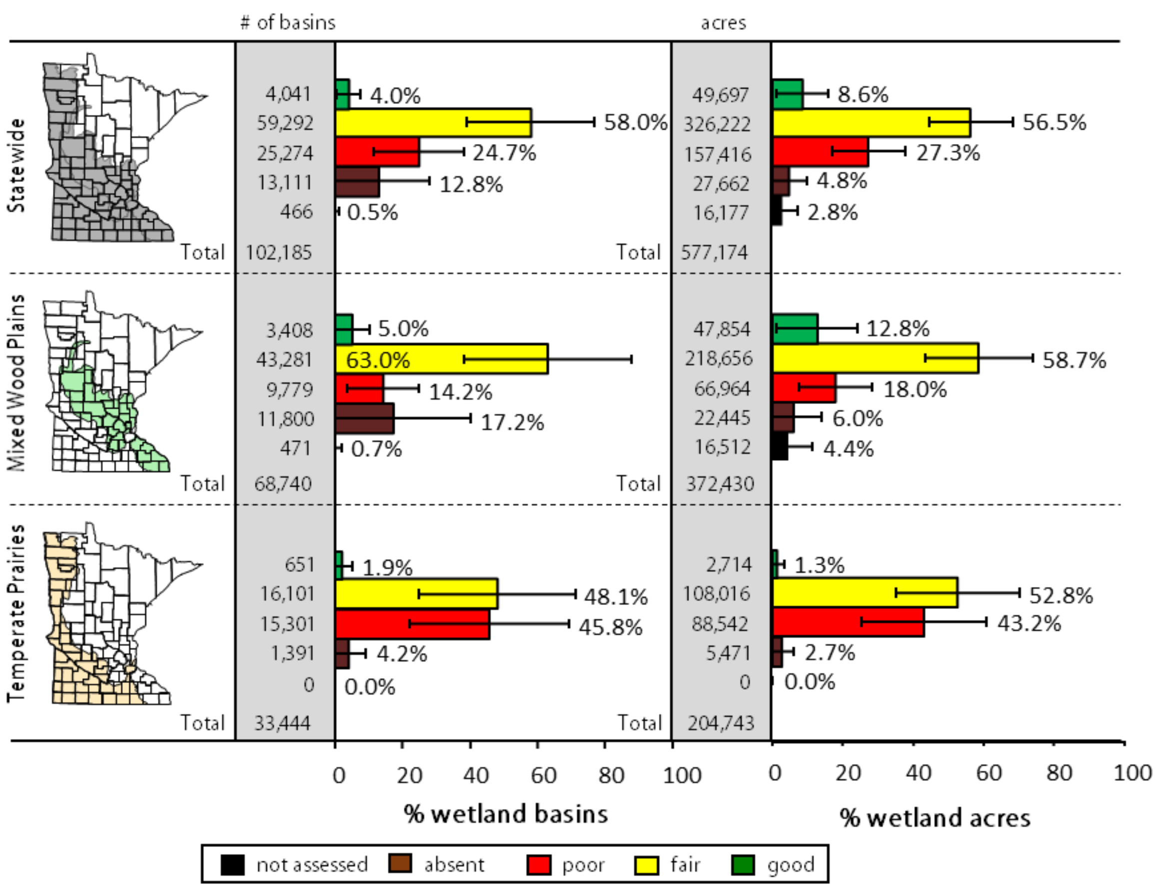


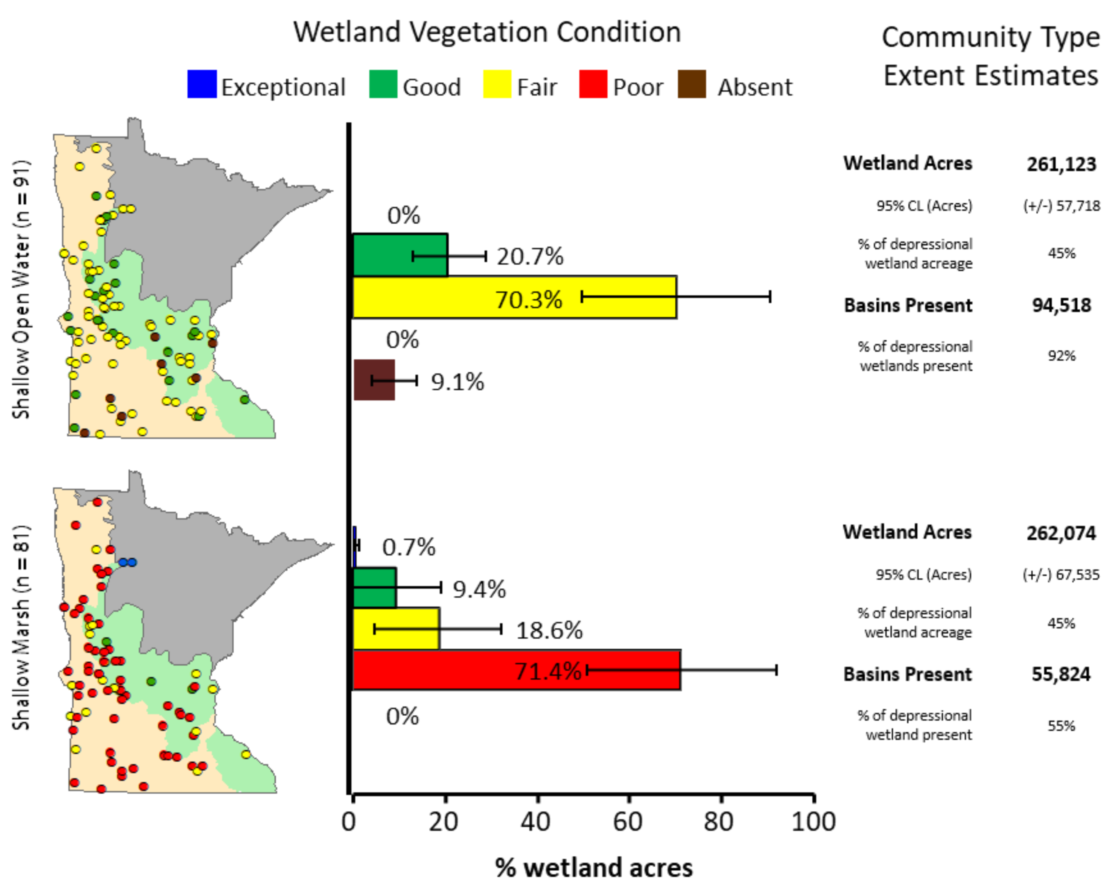
Table 5. Frequency of occurrence and area estimates for depressional wetland plant communities. Frequency estimates do not sum to 100% because basins may include multiple community types.

	Estimated frequency		Estimated area	
Plant community	**No. basins**	**%**	**Acres**	**%**
Shallow open water	94,518	92	261,123	45
Shallow marsh	55,824	55	262,074	45
Fresh meadow	16,689	16	29,665	5
Deep marsh	13,748	13	9,185	2
Shrub-carr	13,700	13	5,193	1
Rich fen	1,004	1	9,935	2

Depressional wetlands and ponds are not typically comprised of a single plant community, rather they are often a mosaic of communities that intergrade both spatially and over time depending on factors such as water depth and permanence. Shallow open water was the most frequently encountered community type in the 2017 survey at the combined ecoregion scale, estimated to be present within 94,500 (92%) basins and accounting for 261,123 acres (45%) of depressional wetland area (Table 5).

This is not surprising, however, given that the presence of permanent standing water is one of the key characteristics of the DWQA target wetland population. The second most prevalent plant community estimated to occur in depressional wetlands and ponds was shallow marsh, followed by fresh meadow, deep marsh, shrub-carr, and rich fen. Among the survey sites, the most common combination was shallow marsh/open water community types and the most types present in any given site was three. Of the community types encountered in the DWQA, only shallow marsh and shallow open water had sample sizes sufficient for the estimation of community-specific condition. The vast majority (70%) of shallow open water communities were in fair condition in the two ecoregions combined, while shallow marsh communities were overwhelmingly (71%) in poor condition (Figure 13). Shallow open water condition categories were evenly dispersed throughout the two ecoregions. Currently, this community type only has criteria in place to distinguish between good, fair, and absent condition categories. Shallow open water plant communities—due to the < 1 m depth sampling requirement—are underrepresented in the MWCA. By including this community type within a depressional landscape setting regardless of depth, the DWQA results supplement the broader MWCA. The limited number of good or exceptional shallow marsh communities were restricted to the MWP ecoregion (Figure 13).

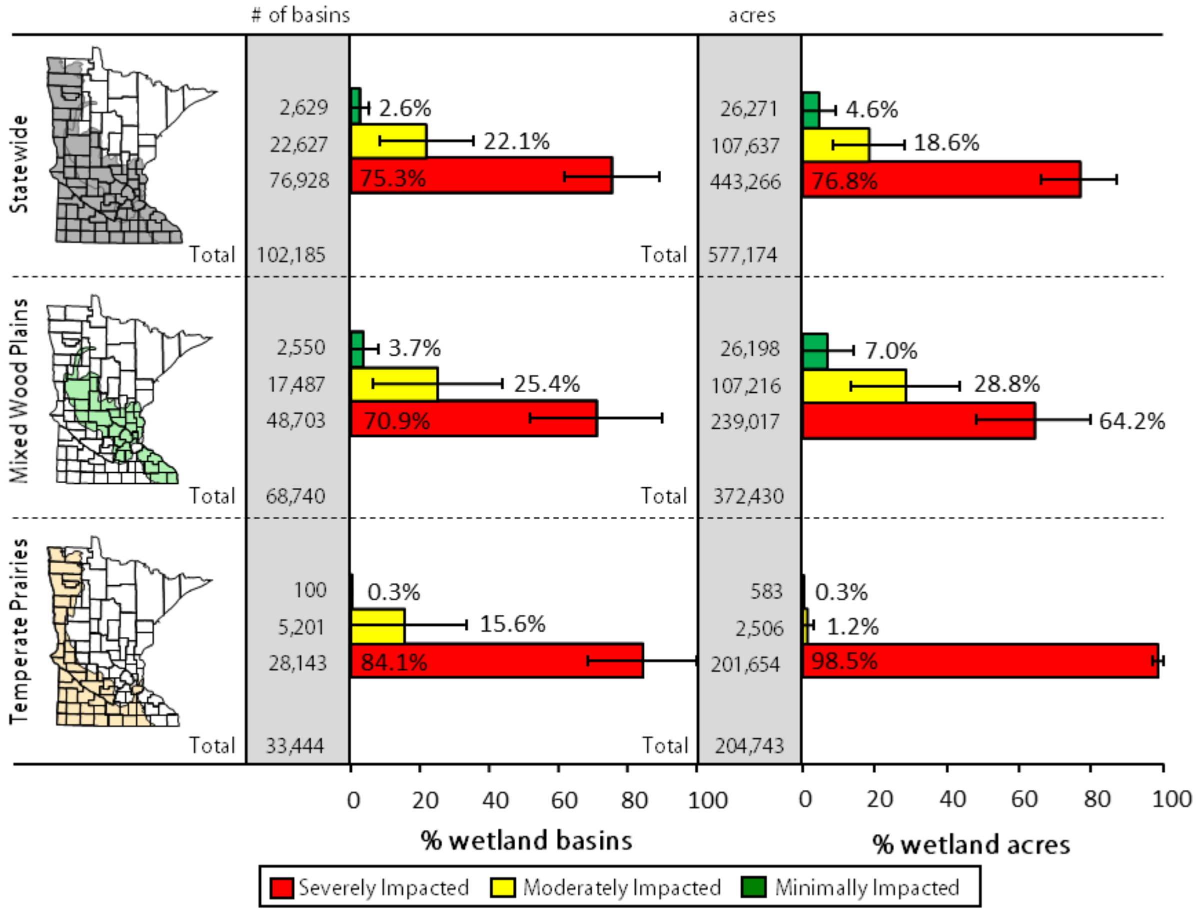
Figure 13. Site location maps, condition category proportion estimates, and population extent estimates by predominant community types present in target depressional wetlands.



Wetland stressors

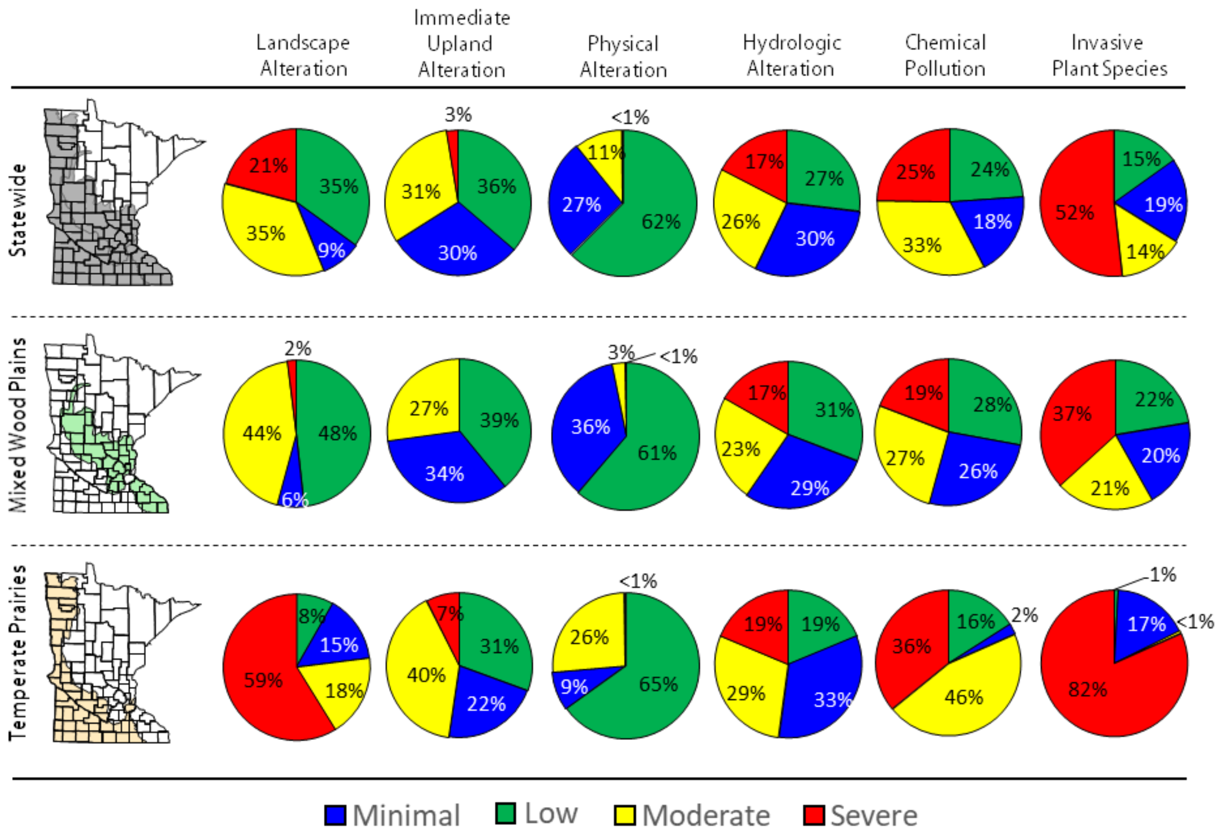
The HDA predicts that the majority of depressional wetlands and ponds are exposed to high levels of human disturbance with the most severe impacts in the TP ecoregion (Figure 14). Based on the similarity of the percentage results between basins and acres, it appears that small and large wetlands are roughly exposed to the same amount of stressors. Both the macroinvertebrate and plant communities exhibit better condition than the HDA would predict, suggesting that there are additional factors that lessen the impacts of the stressors included in the HDA. Even so, the main purpose of including the HDA was to provide information on the potential causes of poor biological integrity, supplementing the measures of stress provided by the water chemistry parameters. Of the factors measured, landscape alteration, hydrologic alteration, invasive plant species, and chemical pollution all exhibited significant levels of moderate to severe impacts to the biological integrity of depressional wetlands and ponds (Figure 15).

Figure 14. Human Disturbance Assessment ratings of Minnesota's depressional wetlands and ponds in 2017, estimated in terms of the number of basins as well as their corresponding 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.



Sites with moderate to severe landscape alteration have urban, agricultural, or industrial land use exceeding 50% within a 500m radius of the upland boundary. The degree of stress created by these land use types however depends on the intensity of each (e.g., till vs. no till cropland, grazing intensity, housing density, etc.), the presence and size of an immediate vegetated buffer, as well as the number and volume of conduits from the disturbed land use to the site (e.g., ag drain tile, storm sewer, ditches, gullies, etc.). Thus, water chemistry results provide some insight into the degree that surrounding land use stresses the biological communities inhabiting depressional wetlands and ponds. According to the HDA chemical pollution rating, it appears that a large percentage of depressional wetlands—between 46% and 82% (Figure 15; moderate + severe)—are in fact experiencing elevated levels of pollutants from their surrounding landscape due to surface run-off, direct discharge, and in some cases possibly groundwater input (e.g., nitrate). An individual evaluation of several pollutants measured in the survey is provided below.

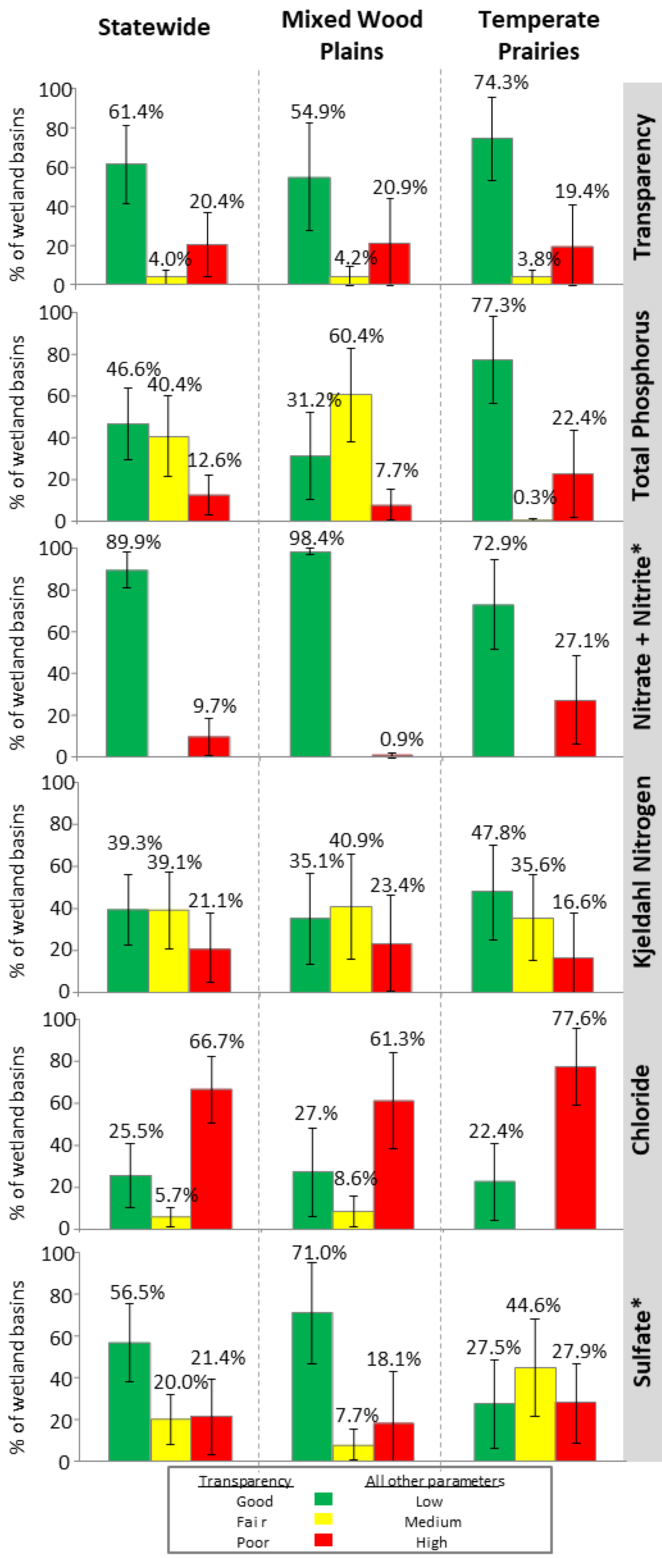
Figure 15. HDA factor ratings of Minnesota's depressional wetlands and ponds in 2017, estimated in terms of the percentage of basins.



Hydrologic alteration includes any activity in or near depressional wetlands and ponds that change its depth, frequency, duration, and source of water. Examples of hydrologic alteration include ditching within or adjacent to the wetland, drain tile or stormwater input, artificial outlets, road berms that restrict surface or shallow groundwater flow, and pumping water from or into the wetland. The extent of hydrologic alteration was relatively consistent among the three regions examined, ranging between 40% and 50% of depressional wetlands with moderate to severe ratings (Figure 15).

Invasive plant species appear to be an important factor influencing the biological integrity of depressional wetlands and ponds. An overwhelming number of basins in the TP ecoregion have plant communities dominated by invasive species, replacing most of the native plant species (Figure 15; severe). Common invasive plant species encountered in the survey include Narrow leaved cattail (*Typha angustifolia* L.), Hybrid cattail (*Typha x. glauca* Godr.), and Reed canary grass (*Phalaris arundinacea* L.). While present in a variety of community types, invasive *Typha* typically reached problematic densities in the shallow marsh community while Reed canary grass was primarily a problem in fresh meadows. Invasive plant species can colonize habitats following anthropogenic or natural disturbance events, tolerate a broad range of disturbance, produce enormous amounts of recalcitrant (persistent) litter that can smother native species, and out-compete other native plants for limited resources (e.g., light and nutrients) through aggressive clonal reproduction (Zedler and Kercher 2004, Larkin et al. 2012). Depending on the circumstances of colonization and proliferation within a wetland, invasive plant species may represent a stressor to biological integrity, a pronounced response to human impacts, or a simultaneous combination of both. Regardless of their origin, given their detrimental effects, invasive plant species abundance is an important indicator for wetland condition assessments.

Figure 16. Stressor levels in Minnesota's depressional wetlands and ponds. Bracketed lines represent the width of the 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).



Non-native invasive plants occurring at problematic levels appeared to increase from 2012 to 2017. The 2012 survey estimated that invasive plant species were high (> 50% cover) in 59% of depressional wetland basins in the TP ecoregion, compared to an estimated 82% that were considered severe (invasive species dominant and evidence of significant replacement of native species) according to the HDA in 2017. There are some key differences, however, between the surveys in how invasive species abundance was estimated that likely explain the observed change. The 2012 survey utilized cover estimates from the plot sampling technique that only evaluated a small portion of the wetland's total area. In many circumstances, this plot straddled two—possibly three—community types. Therefore, accurate abundance estimates for invasive plants at the site scale depended on how well the plot represented each community as a whole in the wetland. The transition to the FQA approach—which provides data for rating the invasive species HDA factor—resulted in a more comprehensive evaluation of the entire wetland and each plant community present. While it may be more difficult for the evaluator to estimate cover across the entire community or wetland, an inexact estimate at this scale is likely still more representative than cover estimates derived from a plot and extrapolated to the entire wetland. Also, the FQA method evaluated a larger extent of fresh meadow and shrub-carr community types, which may have only been incidentally included in the 2012 sampling plots. It is likely that these methodological differences contributed to the observed increase in invasive plant species within depressional wetlands and ponds. Either way, the high percentage of basins with severe invasive plant species ratings is cause for concern and warrants further investigation.

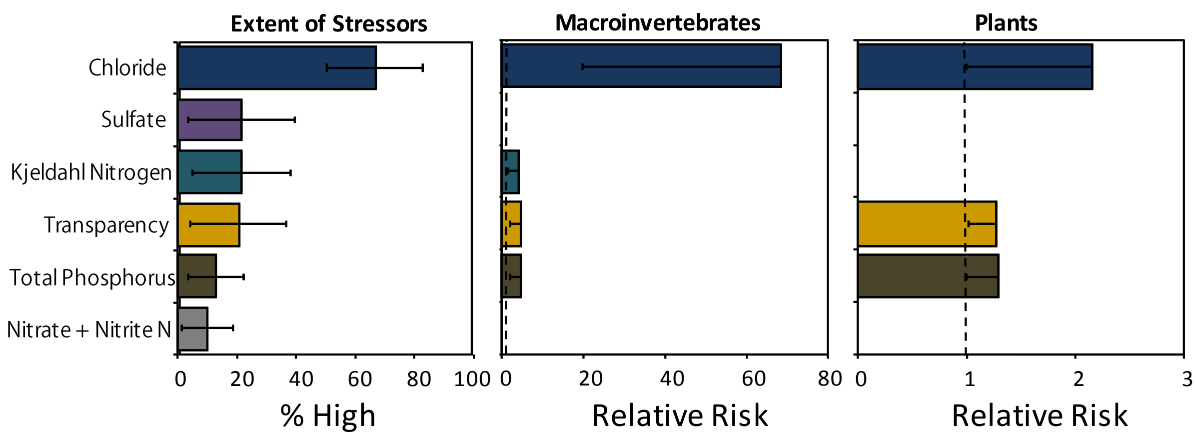
Water chemistry sampling confirms that human disturbance in and around study sites—as documented by the HDA—are affecting the water quality of depressional wetlands and ponds in both ecoregions. Elevated chloride concentrations (i.e., high) were widespread in 2017, similar to 2012 results (Figure 16). 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 depressional wetlands. Additional anthropogenic sources include livestock waste, water softening discharge, fertilizer (e.g., KCl), and municipal sewage effluent (Kelly et al. 2012). Apart from stormwater retention ponds, sample sites within or adjacent to large animal enclosures had some of the highest chloride concentrations. In addition to its prevalence, elevated chloride concentration is concerning due to the risk it poses to plant and macroinvertebrate communities (Figure 17). The relative risk analysis indicates that when chloride is elevated, plants are about twice as likely to be in poor condition compared to when chloride is low (i.e., relative risk ≈ 2) while aquatic macroinvertebrates are over 60 times more likely to be in poor condition. Chloride relative risk estimates for macroinvertebrates, however, have not exhibited much consistency across the three surveys with an estimate of 3.7 in the 2007/2008 survey and < 1 (not significant) in 2012. Therefore, the extremely large relative risk value estimated by the 2017 survey should be weighed against previous estimates, making it difficult at this time to draw conclusions regarding the risks posed by chloride. Plant communities, on the other hand, have consistently had chloride relative risk estimates of ~2 across the three surveys, suggesting that chloride is a key stressor to this community.

Unlike the previous survey when it was approximately 7% in all regions, the combined concentration of nitrate and nitrite is not similar across the three reporting regions in 2017. Nitrate plus nitrite was detected (=high) more frequently in the TP ecoregion in 2017 according to the CDF test (Figure 16; $F = 5.78$, $df_1 = 1$, $df_2 = 91$, $p < 0.05$). The amount of nitrate and nitrite entering a depressional wetland or pond is 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 100 study sites, 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 do with a single water sample. Examination of weekly rainfall maps for June 2017—when water sample data was collected—indicates that the TP ecoregion had slightly higher rainfall amounts throughout most of the month (https://www.dnr.state.mn.us/climate/weekmap/weekmap.html), suggesting that precipitation at least contributed to the observed difference between ecoregions. The higher density of row crop agriculture—specifically corn production—and subsurface drain tiles in the TP ecoregion is likely the main driver of this difference. Nitrate plus nitrite detection rates were significantly higher in the TP ecoregion (28%) compared to the MWP ecoregion (8%) in the 2007/2008 survey as well (Genet 2012), suggesting that the 2012 survey results may reflect the difficulties of accurately characterizing the exposure of aquatic communities to this ephemeral pollutant in wetlands.

Sulfate concentrations were also significantly higher in the TP ecoregion (Figure 16; $F = 8.43$, $df_1 = 1$, $df_2 = 91$, $p < 0.05$), a pattern that was observed in the 2012 survey as well. The observed difference in sulfate concentrations between the two ecoregions is in part due to a natural gradient in surficial geology, with the highest concentrations located in the west-central portion of the TP ecoregion (Moyle 1956). High sulfate concentrations did not pose an elevated risk to aquatic macroinvertebrates or vegetation in depressional wetlands in 2017 (Figure 17). However, in the previous survey sulfate did pose an elevated risk to both communities. Thus, it is difficult at this time to state with much confidence that high sulfate concentrations represent a major stressor to aquatic life in depressional wetlands and ponds of the TP and MWP ecoregions.

Total phosphorus concentrations and transparency of the water column, two factors that are frequently correlated in surface waters, did not vary between ecoregions (Figure 16). Both stressors (i.e., high phosphorus and poor transparency) have consistently been associated with degraded macroinvertebrates and plants across all three surveys. There are two 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. Therefore, the consistency of the elevated relative risk posed by these two parameters across the surveys it is not surprising.

Figure 17. Extent of stressors and their relative risk to plant and 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 graphs indicates that it did not pose an elevated risk to that community.



In addition to ecoregions, comparisons were made among other categories of interest. The DWQA documents the origin—natural vs. man-made—of each study site based on physical characteristics of the site, its current use (e.g., stormwater retention, livestock watering, wildlife habitat), and examination of aerial imagery. Naturally formed wetlands and ponds harbored healthier aquatic macroinvertebrate communities than did man-made sites (Figure 18; $F = 6.83$, $df_1 = 2$, $df_2 = 92$, $p < 0.05$). Similarly, in wetlands where this community type was present (estimated to occur in ~55% of depressional wetlands and ponds), shallow marsh vegetation was in better condition in natural wetlands ($F = 15.64$, $df_1 = 2$, $df_2 = 74$, $p < 0.05$). Shallow marsh vegetation was present—in sufficient quantity to characterize—more often in natural wetlands; estimated to occur in 64% and 45% of natural and man-made wetlands, respectively. In terms of stressors, man-made wetlands had significantly higher chloride, total phosphorus, and sulfate concentrations. These results support the notion put forth by wetland quantity surveys (e.g., Dahl et al. 2011, Kloiber and Norris 2017) that man-made ponds, while considered wetland habitat, are not equivalent to natural wetlands in terms of their condition or ability to provide multiple environmental services. Given the demonstrated shifts in vegetated wetlands to un-vegetated ponds across Minnesota (Kloiber and Norris 2017) and the nation (Dahl et al. 2011), the diminished condition of ponds has broad implications for water quality, wildlife habitat, and biological diversity.

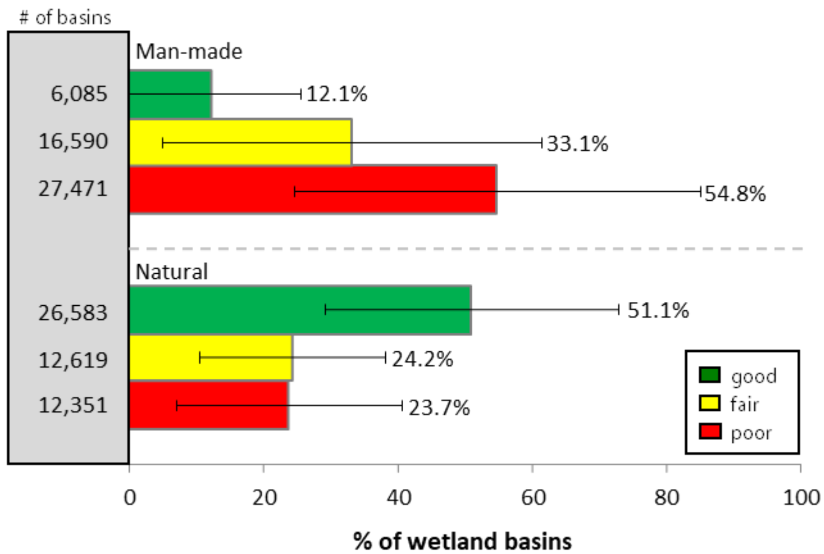


Figure 18. Aquatic macro-invertebrate community condition comparison based on origin, estimated in terms of the number of basins. Bracketed lines represent the 95% confidence interval associated with each estimate. Values may not add up to 100% due to rounding and/or exclusion of not assessed category (> 1%) from figure.

Three wetland area categories were incorporated into the DWQA survey design to obtain—through unequal probability weighting—a good distribution of wetland sizes in the random sample. Comparing these categories revealed that aquatic macroinvertebrates were in better condition in large (> 12.4 acre) wetlands compared to small (< 2.5 acre) ones ($F = 12.32$, $df_1 = 2$, $df_2 = 63$, $p < 0.05$). This was the only statistically significant difference for macroinvertebrates among the three possible comparisons. Shallow open water and shallow marsh—the only communities with enough observations to analyze—did not differ in condition among the wetland area categories. Few differences existed among stressor levels in each size category. Large wetlands had chloride and total phosphorus concentrations that were significantly lower than small and medium wetlands, respectively. Overall, the pattern of condition among the wetland area categories has remained constant across the DWQA surveys thus far and is believed to be at least partially attributed to the disproportionate number of man-made wetlands in the small category.

Comparisons amongst landowner categories (public, private, mixed) provided somewhat surprising results. Depressional wetlands and ponds located on private property supported healthier aquatic macroinvertebrate communities than did those on public land ($F = 6.21$, $df_1 = 2$, $df_2 = 86$, $p < 0.05$). Shallow open water plant communities were also of higher quality on private property ($p < 0.05$). Where present, however, the condition of the shallow marsh community was marginally better on public property ($F = 5.12$, $df_1 = 2$, $df_2 = 71$, $p < 0.05$). The term 'marginally' is used here because both landowner categories had severely degraded shallow marsh communities with an estimated 80% of wetlands on public and private property having *wC* values less than 1.7 and 1.1, respectively, which are both below the poor condition threshold. Similar to man-made wetlands, the shallow marsh community was absent from an estimated 56% of ponds and wetlands located on private property compared to a 1% absence rate on public property. Chloride and sulfate concentrations were significantly lower in private wetlands compared to those on public property. Kjeldahl nitrogen, on the other hand, was significantly lower in public wetlands ($F = 4.14$, $df_1 = 2$, $df_2 = 86$, $p < 0.05$). A small sample size in the mixed ownership category ($n = 7$) precluded comparisons with this category. Considering that most people think of public lands as protected (e.g., state parks) or managed (e.g., national wildlife refuges, wildlife management areas, state forests), the relatively higher quality of wetlands on private property demonstrated here was unexpected. While there were study sites on public land that represent the examples listed above, there were also numerous stormwater retention ponds located on city property that likely contributed to the decreased condition/higher stress of this category.

Based on the above findings, the following are recommended to maintain and restore the condition of depressional wetlands and ponds in Minnesota: 1) Natural wetlands—particularly those > 12 acres in size—located on private property have the highest biological condition and should represent a top conservation priority; 2) Restoration or enhancement efforts should focus on smaller wetlands (< 12 acres)—particularly those that are man-made—located on public land; 3) Policies and practices that address invasive plant species as well as excess chloride, phosphorus, and sediment inputs will be most effective for restoring depressional wetland condition; and 4) Individual wetland restoration or protection activities should first conduct a thorough evaluation of the stressors or potential threats to each site, considering that it may identify ones not measured in this survey (e.g., pesticides, herbicides, altered water levels, invasive fish).

Changes in depressional wetland condition

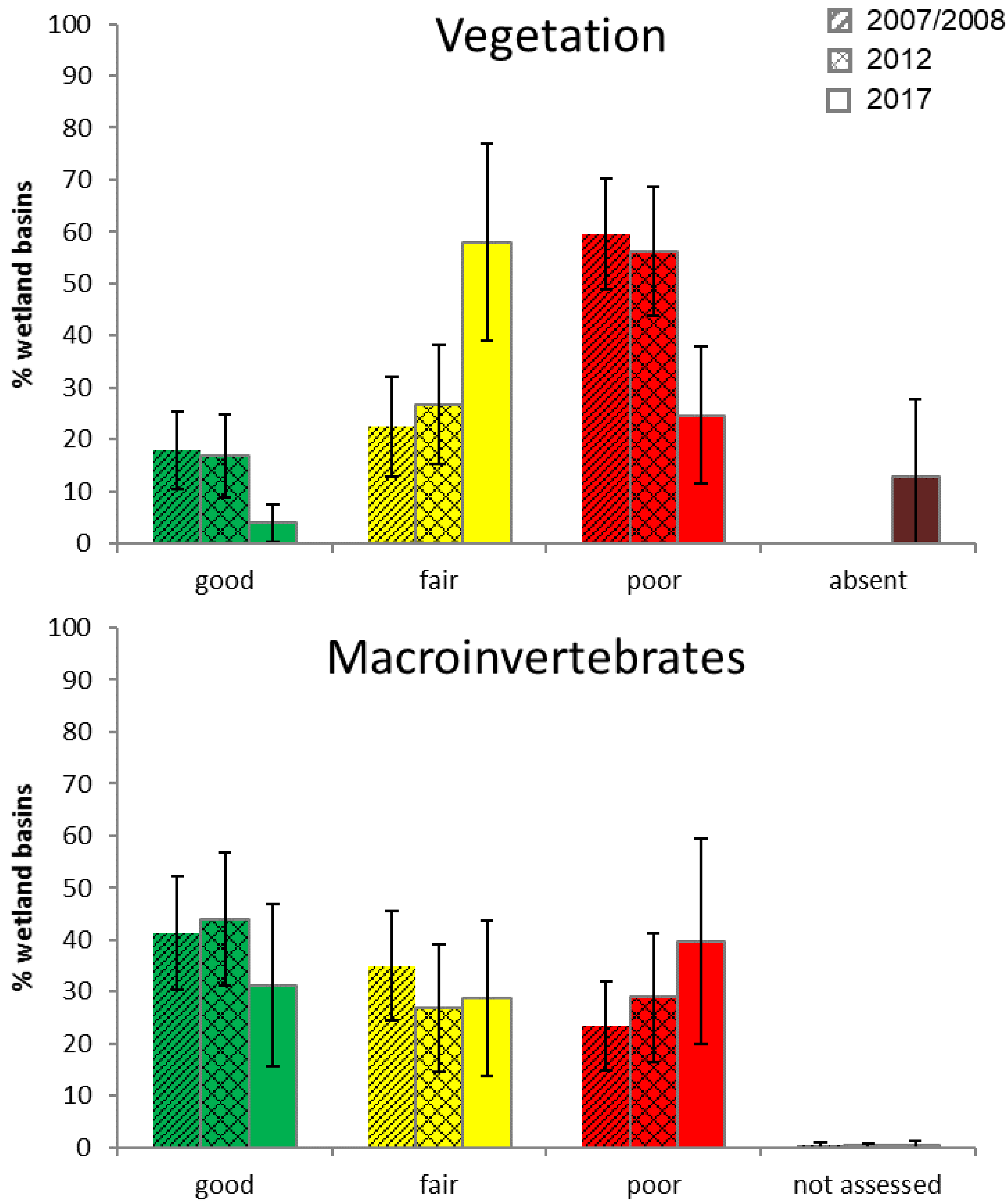
Potential differences in macroinvertebrate and vegetation condition as well as the stressors between survey periods 2007/2008, 2012, and 2017 were analyzed. Several more iterations of the survey will be required before trends can be evaluated, a primary goal of this status and trends survey. Appendix D includes the categorical change detection results for the combined ecoregions and each subpopulation therein, excluding mixed landowner sites due to a small sample size. A summary of those results is provided here.

Statewide vegetation condition appears to be decreasing in the latest cycle of the survey (Appendix D). However, the abrupt shift amongst condition categories that occurs in 2017, compared to the subtle differences between the first two surveys (Figure 19), suggests that the transition to the new FQA method is largely responsible for the observed changes. Results from the next survey—again using the FQA method—will help resolve whether vegetation condition in depressional wetlands is truly decreasing statewide. Macroinvertebrate community condition has been evaluated using the same IBI across all three surveys and does not exhibit any statistically significant changes in the percentage of good or poor condition categories over time (Appendix D). While not statistically significant, the decrease in % good coupled with an increase in % poor over time does suggest declining macroinvertebrate community condition (Figure 19). The CDF test—a more quantitative analysis of change—does reveal a statistically significant decrease in the macroinvertebrate IBI score in 2017 compared to the first two surveys. Future surveys will elucidate whether this pattern is consistent with a long term trend or merely represents annual variability.

The macroinvertebrate community exhibited a significant decrease in good condition between 2012 and 2017 in man-made wetlands and ponds (Appendix D). This decline in biological condition was accompanied by increases in chloride, Kjeldahl nitrogen, and total phosphorus concentrations. Natural wetlands experienced a decrease in water clarity in 2012 and 2017.

Compared to the previous surveys, macroinvertebrate community condition decreased significantly in 2017 amongst wetlands and ponds located on public property (Appendix D). In 2017, public wetlands had elevated chloride and sulfate concentrations relative to 2012 results and elevated Kjeldahl nitrogen and sulfate concentrations coupled with decreased transparency relative to 2007/2008 results. Notably, there were no significant changes between the first two surveys for wetlands and ponds located on public property. The 2017 macroinvertebrate condition results for public wetlands may be atypical and, as previously mentioned, is likely due to an increased influence of man-made wetlands on the condition estimates for this landowner category. Man-made wetlands contributed significantly more weight to the 2017 estimation of public wetland condition, accounting for 14%, 39%, and 74% of the weights in 2007/2008, 2012, and 2017, respectively.

Figure 19. Comparison of biological condition results amongst the three DWQA survey periods. Bracketed lines represent the width of the 95% confidence interval associated with each estimate.

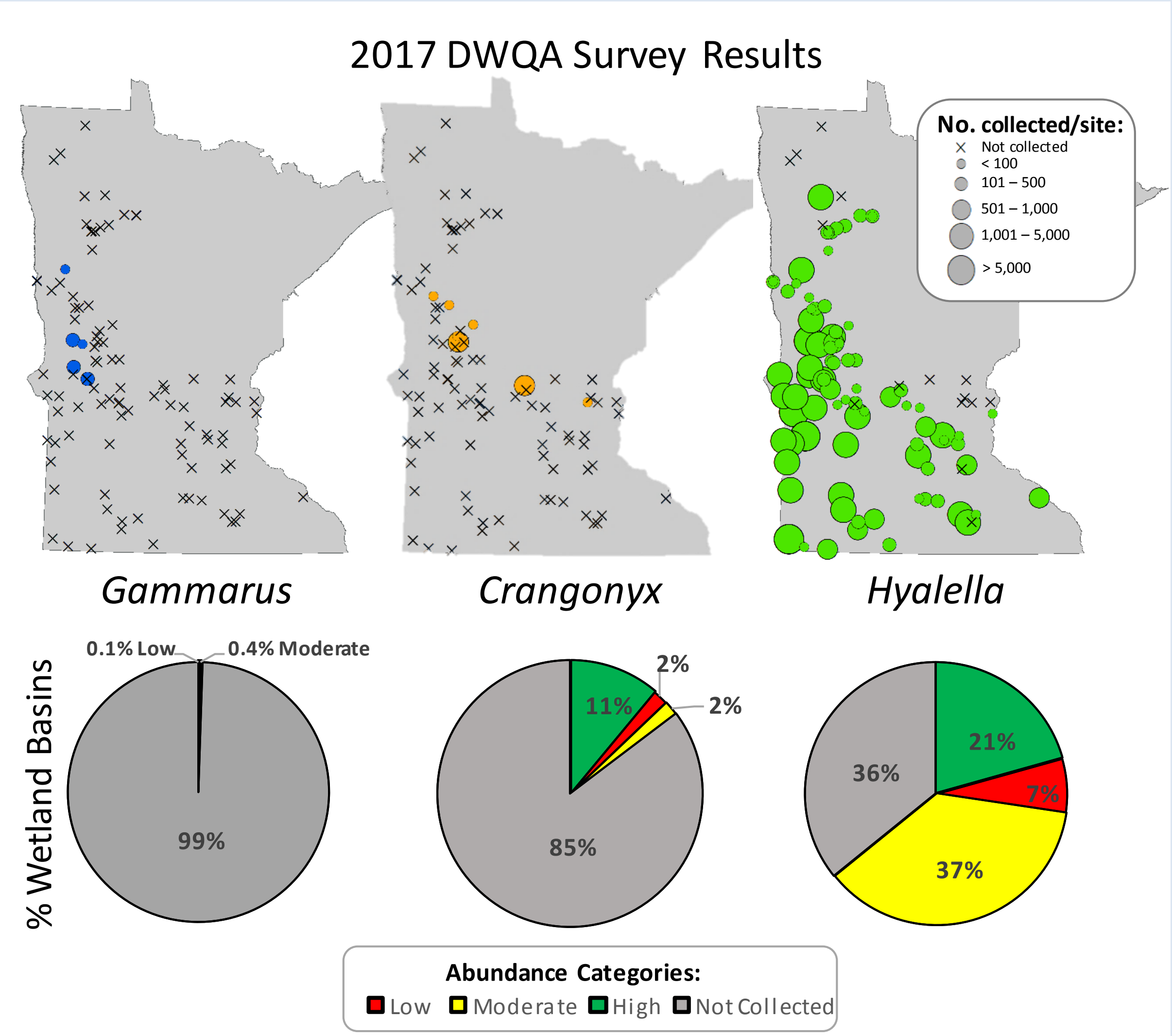


A closer look at freshwater shrimp in depressional wetlands and ponds

Amphipods (a.k.a. freshwater shrimp, scuds, side-swimmers) are important inhabitants of depressional wetlands and ponds throughout Minnesota, serving as a significant food source for waterfowl, salamanders, wading birds, and fish. Growing up to 2 centimeters in length, these aquatic macroinvertebrates can easily be seen with the naked eye. Identification to species, however, requires a microscope to distinguish them based on antennae characteristics. There are three amphipod genera common to wetlands and ponds in Minnesota: *Crangonyx*, *Gammarus*, *Hyalella*. These crustaceans have to rely on other organisms (e.g., waterfowl and aquatic mammals) for transport to water bodies that lack surface water connections. Upon colonization, the persistence of amphipods in an isolated water body is determined by the number of individuals immigrating, frequency of immigration, presence of predators, water quality, and characteristics of the basin. For instance, some species cannot survive the winter if the basin is too shallow to allow for some water to remain unfrozen. A current concern is whether amphipod abundance is decreasing throughout the Prairie Pothole Region, an area of particular significance for migrating and breeding waterfowl, and if so, what is causing the decline?

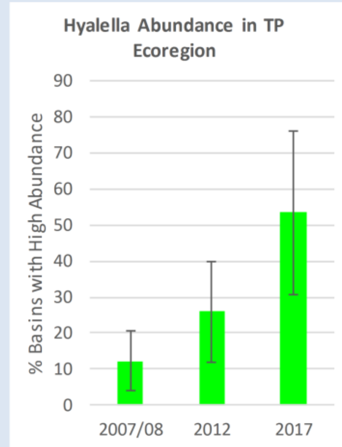
To help address these questions, the DWQA's probabilistic design was utilized to investigate whether amphipod abundance is increasing or decreasing over the 10 years of the survey and whether differences exist between genera. Amphipods were collected from the wadeable margins of each study site using dip nets and a timed field-pick of organisms from the vegetation and detritus. Therefore, abundance data should be considered semi-quantitative at best and failure to collect a species at a site ('Not Collected') should not be interpreted as certainty of its absence from that location. In the lab, organisms were identified to genus and the entirety of the picked sample was enumerated if the total count was less than 900. Subsampling, using randomly selected cells on a gridded tray to dictate the order of sample material processing, was employed if the total number of macroinvertebrate organisms in each sample exceeded 900. The abundance of each genera as well as the combined abundance of all amphipods at a site were used in the extrapolation procedures to estimate the percentage of basins in the MWP and TP ecoregions where they are present and that support low, moderate, or high abundances of each genera. Abundance categories were derived based on the distribution of total amphipod abundances within least-impacted reference sites as follows: below median value (40) = 'low'; $\geq$ median and below 75th percentile value (454) = 'moderate'; and $\geq$ 75th percentile value = 'high'. Abundance values of zero were categorized as 'not collected'. CDF tests based on the percentage of basins was used to test for differences in abundance between subpopulations (e.g., ecoregions).

Hyalella is the most abundant amphipod genera occurring within depressional wetlands and ponds, estimated to be present in 64% (~65,000) and in high abundance in 21% (~21,000) of TP and MWP basins. This is not surprising given its reputation as being ubiquitous in permanent water bodies of North America and tolerant to organic pollution in streams (Hilsenhoff 1987) as well as increased conductivity (Gibbons and Mackie 1991, Muck and Newman 1992). The highest observed abundance of *Hyalella* among the 2017 survey sites was ~8,000, which roughly translates to a density of 8,000 individuals/m^3 (assuming eight 1 meter sweeps per sample, two samples/site). This genus, which is most likely represented here by one species (*H. azteca*), was significantly more abundant in the TP ecoregion and on private property. The greater abundance of *Hyalella* in privately owned wetlands was largely attributed to its higher densities in small (private) wetlands, which have more weight in analyses by basin, compared to public wetlands that had higher densities in large wetlands. Over the three cycles of the DWQA survey, *Hyalella* abundance was significantly higher in the latest cycle in TP basins and those located on private property compared to both previous cycles. Abundance estimates in the first two cycles were very similar, suggesting that 2017 may represent a "bumper" year as opposed to being indicative of a trend. With just three data points it is impossible to determine at this point.



Gammarus and *Crangonyx* were encountered less frequently among the 2017 survey sites resulting in much lower estimates for the percentage of basins where these genera occur, < 1% (~500) and 15% (~15,000), respectively. Holsinger (1972) estimated the range of *G. lacustris*, which is likely the sole species represented here, as limited to the northern 4/5th of Minnesota and is thought to prefer cold water habitats. MPCA wetland biological monitoring data are consistent with this range estimate, having rarely collected *Gammarus* south of the Minnesota River over the program's history. Similarly, *Crangonyx* has only been collected by the MPCA in depressional wetlands and ponds in the central region of the state. In 2017, *Gammarus* was exclusively collected from large (>12.4 acres), natural wetlands located on private property, while *Crangonyx* was only collected in the MWP ecoregion and had significantly higher densities in natural wetlands relative to man-made ponds. These apparent distribution patterns should be interpreted with caution however, considering the low occurrence rate of each genera in the 2017 survey (see figure above). Compared to previous surveys, *Gammarus* occurrence was significantly lower in 2017. *Crangonyx* exhibited a different pattern with occurrence rates significantly higher in 2012 and 2017 compared to the first survey cycle.

Overall, total amphipod densities appear to be stable or possibly increasing since 2007, owing almost exclusively to the proliferation of the relatively tolerant *Hyalella*. Further monitoring and analyses are required to understand whether the relatively low occurrence/ abundance of *Gammarus* and *Crangonyx* are similar to what would have been observed pre-European settlement of the region or if they're a product of population declines within those genera. Considering that the study region has experienced significant historical wetland loss, it is unlikely that the current status of these two genera mirrors that which existed during pre-settlement times. This analysis highlights the importance of examining individual genera (if not species) when looking at population trends and illuminates another research question: Are amphipod species equal in quality as a food resource for wildlife?



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Hilsenhoff, W.L. 1987. An improved biotic index of organic stream pollution. The Great Lakes Entomologist 20:31-39.

Holsinger, J.R. 1972. The freshwater amphipod crustaceans (Gammaridae) of North America. U.S. Environmental Protection Agency, ELD 04/72. Cincinnati, OH.

Muck, J.A. and R.M. Newman. 1992. The distribution of Amphipods in southeastern Minnesota and their relation to water quality and land use. Journal of Iowa Academy of Science 99:34-39.

Mixed Wood Plains results and discussion

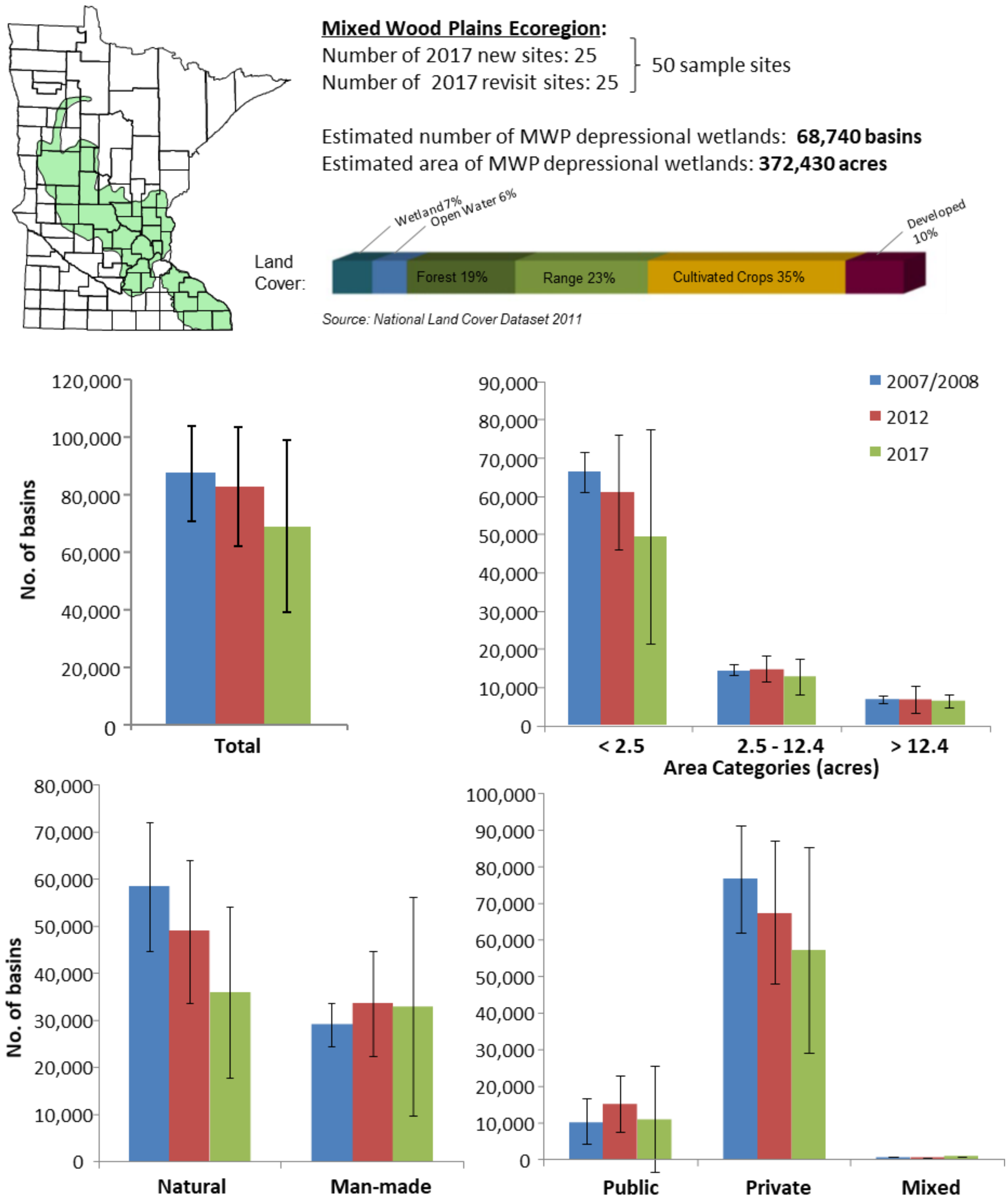
The MWP ecoregion (Omernik 1987, White and Omernik 2007) represents a transitional zone between the Great Plains and Northern Laurentian Forests. In Minnesota, the MWP ecoregion occupies the central part of the state in a southeast to northwest orientation. The southeast portion of this ecoregion is known as 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. 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. Wetlands in the southeast driftless area are primarily located within floodplain, riverine, and slope geomorphic settings. 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 36 inches in the southeast (State Climatology Office, 2012).

Depressional wetland quantity (status and change)

An estimated 68,740 depressional wetlands and ponds (67% of combined ecoregion total) occur within the MWP ecoregion according to 2017 survey results. This estimate represents a decline in the total number of basins compared to previous survey estimates (Figure 20); however, over this limited time frame it is not yet possible to evaluate for trends. Change analyses did not reveal any statistically significant differences between 2017 extent estimates and either of the previous two surveys, including comparisons made within the subpopulations. Decreases in the number of small (< 2.5 acres), naturally formed wetlands located on private property seem to be responsible for the overall declining pattern that is evident within the MWP ecoregion and statewide (Figure 20). As mentioned previously, the observed decrease in basins is partially due to the on-going improvement of the maps used to select sites; removing non-target waterbodies from those maps as they are discovered during site evaluation. Overall, the majority of depressional wetlands and ponds in the MWP ecoregion are small (< 2.5 acres), naturally formed, and located on private property; however the gap between these categories and their counterparts appears to be decreasing (Figure 20).

The estimated area of depressional wetlands and ponds in the MWP ecoregion is 372,430 acres, approximately 65% of the combined ecoregion total acreage. Comparing this result to the estimated number of basins yields an average basin area of 5.4 acres for the ecoregion.

Figure 20. Estimates of the total number of depressional wetlands in the Mixed Wood Plains ecoregions over the three cycles of the DWQA survey. Bracketed lines represent the 95% confidence interval associated with each estimate.



Depressional wetland condition

- Wetland vegetation condition in the MWP ecoregion approximates the statewide results with most basins exhibiting a fair overall condition when all community types present at a site are combined (Figure 21). No depressional wetlands or ponds are considered to have an overall exceptional plant community rating and only 5% are in good condition.
- Considering individual plant community types, an estimated 0.3% of depressional wetlands in the MWP has exceptional shallow marsh community condition. These represent the only instances of exceptional community ratings in the 2017 DWQA. In contrast, shallow marsh communities were not present (i.e., basin lacked the required depth range and water regime to support this community; ≠ absent condition category) in 59% of MWP depressional wetlands and ponds.
- The condition of aquatic macroinvertebrate communities in MWP depressional wetlands and ponds was also very similar to statewide results. Estimated percentages of basins in each condition category were virtually equal, representing about a third each (Figure 21).
- Similar to past surveys, elevated chloride concentrations represent the most prevalent stressor—among those measured—for depressional wetlands and ponds in the MWP ecoregion. However, with 61% of the basins estimated to have high concentrations (Figure 21), elevated chloride was more common in 2017 than in past surveys.
- Nitrate + nitrite nitrogen concentrations were below detection in virtually all MWP depressional wetlands and ponds in 2017 (Figure 21). 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.
- Total phosphorus concentrations in the MWP were predominantly medium in 2017 and only 8% of the basins in the ecoregion were estimated to have high concentrations (Figure 21).

Ranking the stressors measured in this survey by the estimated extent of high concentrations (or poor quality) in the MWP reveals a similar order to that observed when the two ecoregions are combined (=statewide), with chloride at the top of the list (Figure 22). Unlike the statewide results, chloride does not appear to pose an elevated risk to wetland plants and macroinvertebrates in the MWP despite its prevalence of high concentrations. For macroinvertebrates, however, the lack of an elevated risk was likely caused by zero occurrences of high stressor/good condition among the survey sites (i.e., the analysis cannot handle zeroes in the contingency table), as the overall pattern in the data show a strong association between high stressor/poor condition and low stressor/good condition. Both Kjeldahl nitrogen and total phosphorus exhibit dramatically high relative risk estimates for macroinvertebrates, suggesting a strong negative impact on this community (Figure 22). These results should be interpreted with caution due to low sample sizes in one key area of the analysis. In short, a high relative risk value results from strong associations of high stressor/poor condition as well as low stressor/good condition. For both Kjeldahl nitrogen and total phosphorus, while there is a strong association between low concentration and good macroinvertebrate condition, there are very few instances of high concentration and poor condition (n = 3), calling into question the validity of the relative risk estimates. While it is conceivable that an elevated risk does exist for both of these stressors, it is doubtful that it is as strong as Figure 22 suggests. As mentioned previously, total phosphorus consistently exhibits an elevated risk to both plants and macroinvertebrates across ecoregions and survey years.

Figure 21. 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.

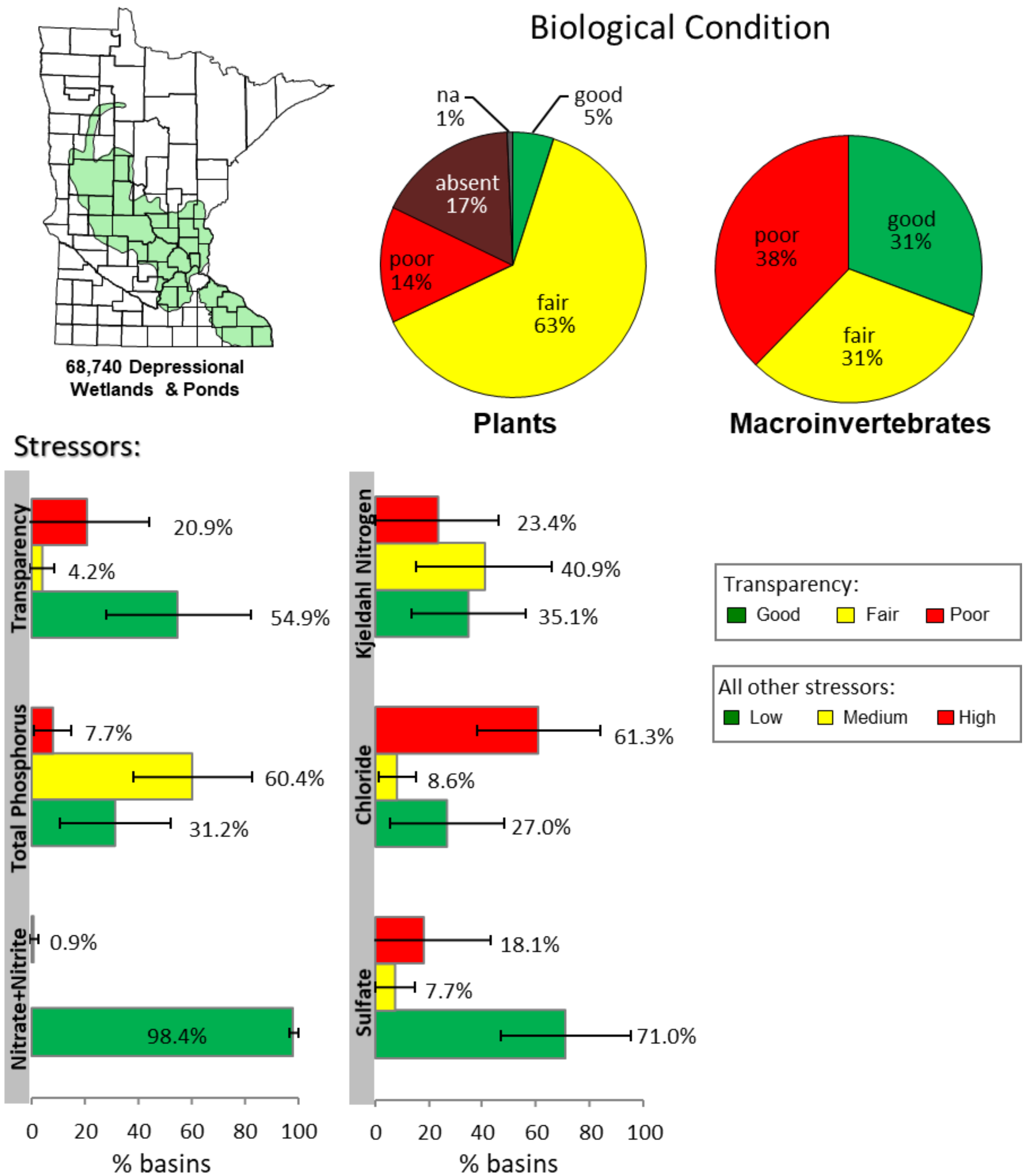
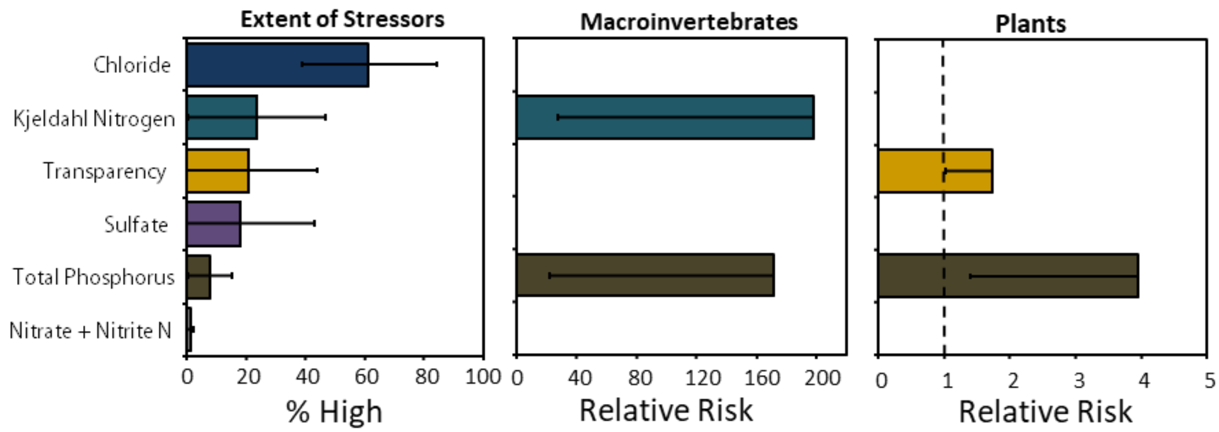


Figure 22. Extent of stressors and their relative risk to plant and macroinvertebrate communities in MWP depressional wetlands and ponds. Bracketed lines represent 95% confidence intervals (for % estimates) or lower confidence limits (for relative risk estimates). A stressor without an associated bar on the relative risk graphs indicates that it did not pose an elevated risk to that community.



Changes in depressional wetland condition

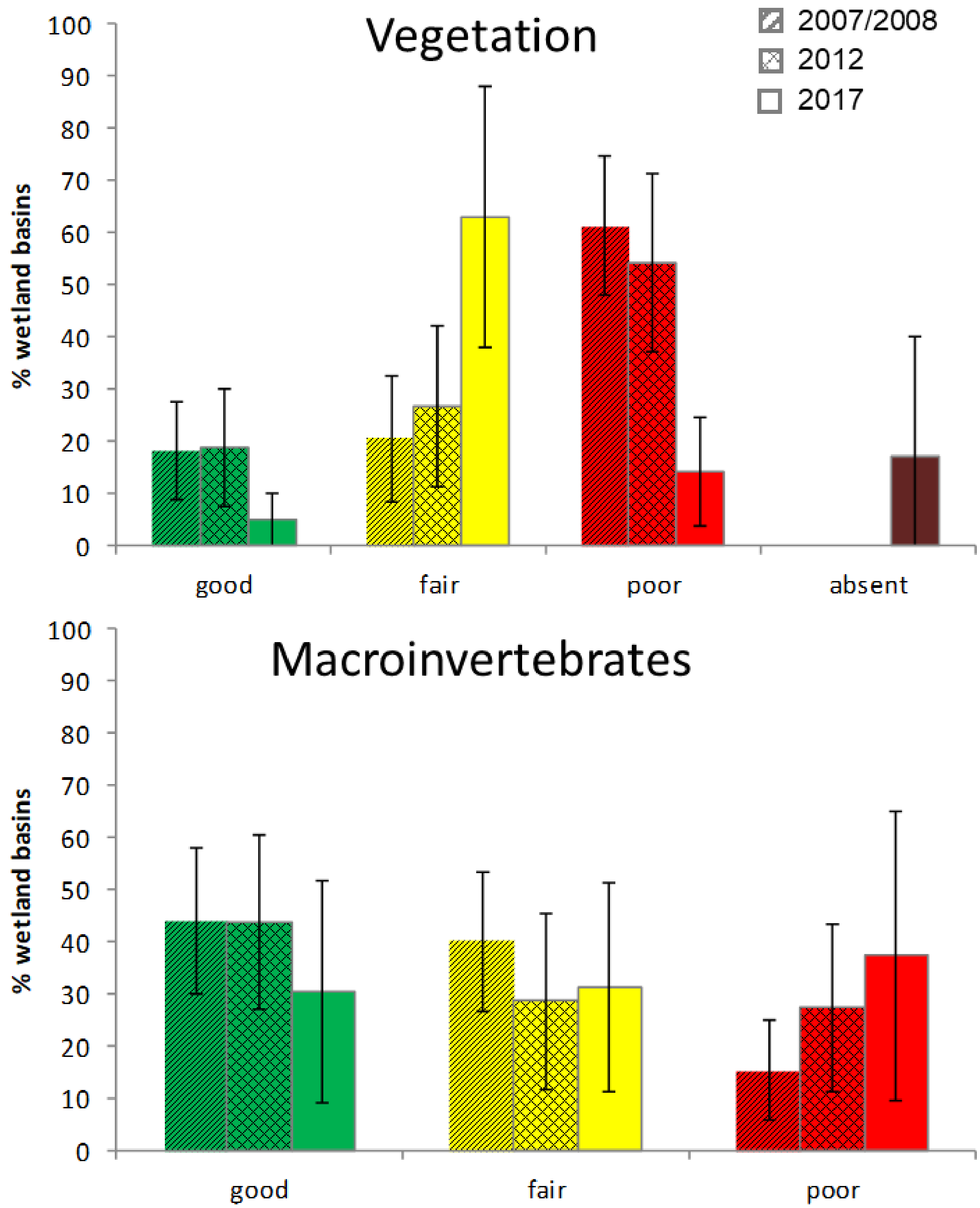
Potential differences in macroinvertebrate and vegetation condition as well as the stressors between survey periods 2007, 2012, and 2017 were analyzed. Several more iterations of the survey will be required before trends can be evaluated, a primary goal of this status and trends survey. Appendix D includes the categorical change detection results (e.g., Is % poor increasing over time?) for the Mixed Wood Plains ecoregion data set as well as for each subpopulation therein, excluding mixed landowner sites due to a small sample size. A summary of those results is provided here.

Similar to the statewide results, vegetation exhibits a decrease in condition for the MWP ecoregion in 2017 compared to the previous two surveys (Figure 23 and Appendix D). As discussed previously, it is likely that the transition to the new FQA method is largely responsible for these observed changes. Otherwise, there is no overall pattern of declining or improving condition for depressional wetlands and ponds in the MWP ecoregion. Analyses for individual subpopulations within the ecoregion did reveal significant changes occurring.

In 2017, man-made wetlands and ponds in the MWP exhibited significant declines in macroinvertebrate condition as well as increases in chloride and Kjeldahl nitrogen concentrations compared to both previous surveys (Appendix D). Total phosphorus concentrations also significantly increased in 2017 compared to 2012, but were lower than they were in the first survey. If the proportion of man-made wetlands is increasing—as suggested by Figures 20 and 24—their reduced condition will represent a major obstacle to maintaining biological diversity in depressional wetlands and ponds across the state.

Similar to the statewide results, macroinvertebrate community condition in the MWP decreased significantly in 2017 within wetlands and ponds located on public property (Appendix D). This decline in biological condition was accompanied by increases in chloride, Kjeldahl nitrogen, sulfate, and total phosphorus concentrations. Of the various categories examined, this subpopulation has the most consistent pattern among condition and stressor indicators—all indicating decline—and appears to be driving the results observed for public wetlands and ponds at the statewide scale. As previously mentioned, the poor quality of this landowner category in 2017 is likely due to an increase in the influence (i.e., number and weight of sites) of man-made wetlands on its condition estimates. As such, the 2017 results probably represent a short-term dip as opposed to the general slope of a long-term trend for this category of depressional wetlands.

Figure 23. Comparison of biological condition in MWP depressional wetlands and ponds amongst the three DWQA survey periods. Bracketed lines represent the width of the 95% confidence interval associated with each estimate.



Temperate Prairies results and discussion

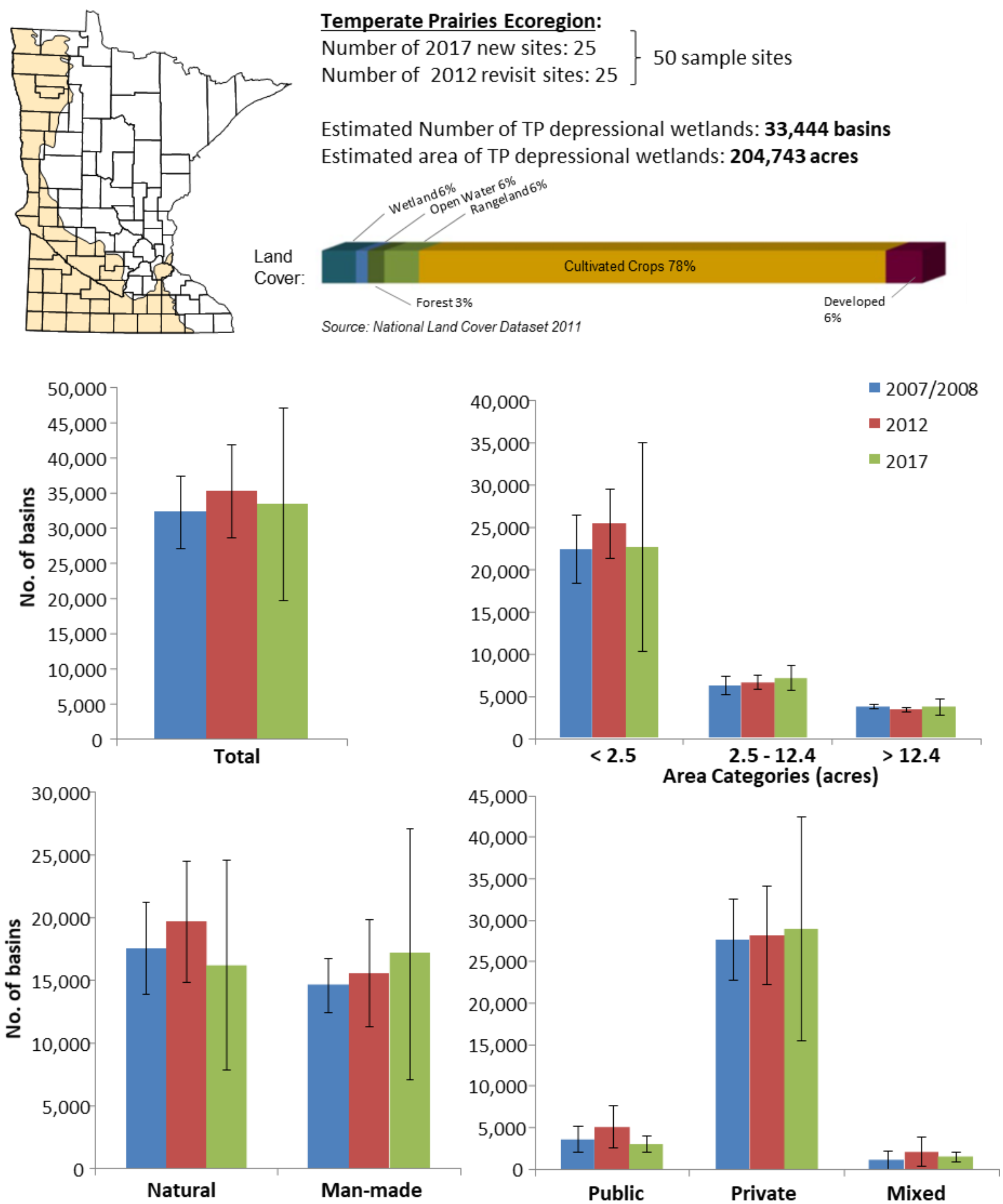
The topography of the TP ecoregion (Omernik 1987, White and Omernik 2007) ranges from the gently rolling glacial till plains of the southern part of the state to the nearly level basin of ancient Glacial Lake Agassiz in the northwest. Prior to European settlement the vegetation within this region was 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 34 inches in the southeast (State Climatology Office, 2012).

Depressional wetland quantity (status and change)

The 2017 survey estimates a total of 33,444 depressional wetlands and ponds (33% of combined ecoregion total) in the TP ecoregion. Unlike the MWP ecoregion, the number of depressional wetlands and ponds in the TP ecoregion seems to be stable (Figure 24), suggesting that no-net-loss in the quantity of this particular wetland type is being achieved and/or the on-going removal of non-target sites from site selection maps—resulting in decreased extent estimates—is largely restricted to the MWP. A more comprehensive evaluation will be possible once wetland acreage has been measured consistently between cycles of the survey. Changes to the plant community assessment procedure in 2017, resulted in area adjustments for individual survey sites, precluding meaningful wetland acreage comparisons with past survey results. Change analyses did not reveal any statistically significant differences between 2017 basin quantity estimates and either of the previous two surveys, including comparisons made within the subpopulations (e.g., natural). Overall, the majority of depressional wetlands and ponds in the TP ecoregion are small (< 2.5 acres) and located on private property (Figure 24). Estimates from the current survey indicate that naturally formed wetlands roughly equal the number of man-made ones in the TP ecoregion.

The estimated area of depressional wetlands and ponds in the TP ecoregion is 204,743 acres, approximately 35% of the combined ecoregion total acreage. Comparing this result to the estimated number of basins yields an average basin area of 6.1 acres for the ecoregion.

Figure 24. Estimates of the total number of depressional wetlands in the Temperate Prairies ecoregion comparing survey periods. Bracketed lines represent the 95% confidence interval associated with each estimate. Asterisk indicates a statistically significant change between time periods.

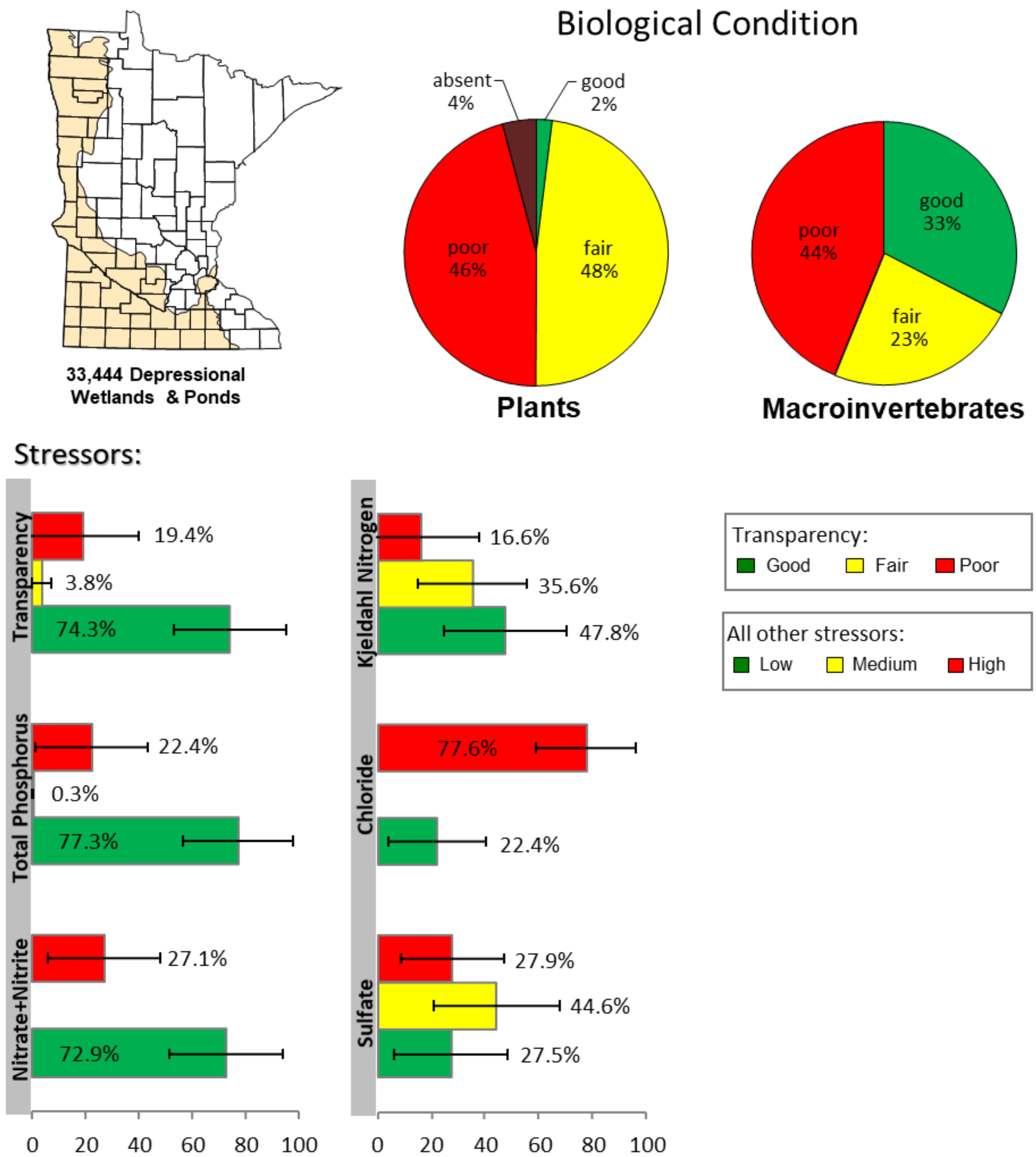


Depressional wetland condition

- According to 2017 estimates, wetland vegetation condition is substantially lower in the TP ecoregion compared to statewide and MWP results with 50% of basins in poor or absent condition when community types are combined (Figure 25). No depressional wetlands or ponds are considered to have an exceptional plant community rating and only 2% are estimated to be in good condition.
- Shallow open water was the most prevalent plant community type in the TP ecoregion, occurring in an estimated 84% of depressional wetlands and ponds. Approximately 42% of basins had shallow open water communities in fair condition. Shallow marsh community types occur in an estimated 81% of TP depressional wetlands with the overwhelming majority (77%) in poor condition.
- Aquatic macroinvertebrate community condition in TP depressional wetlands and ponds was similar to statewide results. Estimated percentages of basins weren't quite evenly distributed however, with a slightly higher percentage of poor and lower percentage of fair compared to the MWP ecoregion (Figure 25).
- Elevated chloride concentrations is an issue for an estimated 78% of TP depressional wetlands and ponds according to 2017 survey results (Figure 25). This represents the largest '% high chloride' estimate obtained for an ecoregion over the three survey periods. The increased proportion of man-made wetlands and ponds in the 2017 survey (Figure 24) likely contributed to the elevated chloride concentrations. However, the role of fertilizer application should not be underestimated in agricultural regions as a study in the Prairie Pothole Region of Iowa found a significant positive correlation between chloride concentration and the amount of cropland surrounding permanently flooded wetlands (Sundberg et al. 2016).
- Unlike the MWP ecoregion, Nitrate + nitrite nitrogen was detected in approximately 27% of TP depressional wetlands and ponds in 2017 (Figure 25). This is not surprising given that this pollutant is most often associated with sub-surface drainage of agricultural landscapes. However, higher precipitation amounts in the TP during the month of June (i.e., water chemistry sampling period) likely contributed to the observed difference in detection rate between the two ecoregions (Figure 16).
- Total phosphorus and transparency, two water quality parameters that are often associated, had similar results in the TP ecoregion with the majority of basins exhibiting low concentrations of phosphorus and good water clarity (Figure 25). This pattern was also observed in the TP ecoregion in 2012 and suggests that the inundated portion of most basins are well-buffered from surface run-off, the primary transport pathway for sediment—and phosphorus adsorbed to these particles—into aquatic systems.

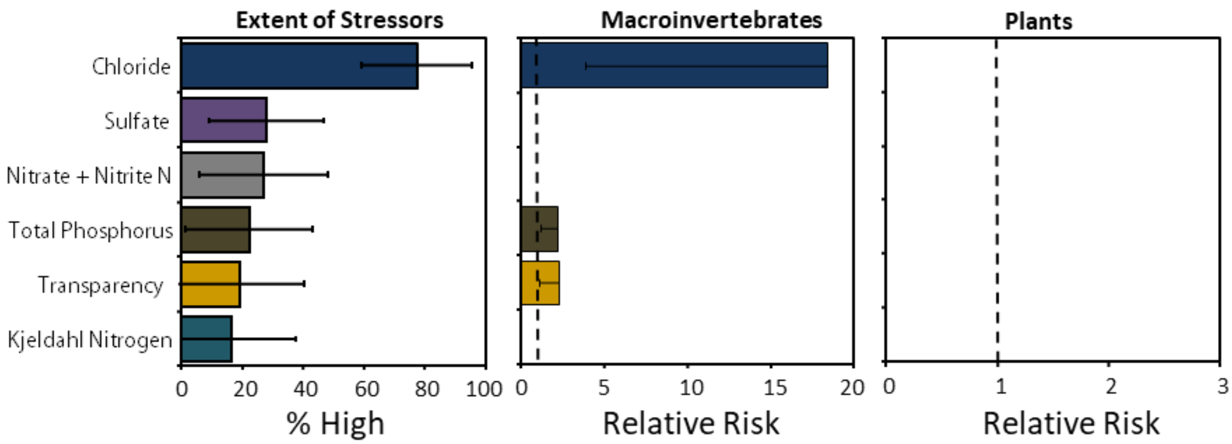
The stressors measured in this survey ranked by the estimated extent of high concentrations (or poor quality) in the TP had the same top two stressors (chloride and sulfate) as did the two ecoregions combined (Figure 26). In the TP ecoregion, however, nitrate + nitrite N and Kjeldahl nitrogen switched positions in the rankings as did total phosphorus and transparency. Given the predominantly agricultural landscape of the TP ecoregion, the increased prevalence of elevated phosphorus and nitrate concentrations—relative to the MWP—is expected. Of these two, only total phosphorus poses an elevated risk to aquatic macroinvertebrates in this ecoregion (Figure 26). Chloride poses the largest risk to macroinvertebrates in TP depressional wetlands and ponds, consistent with 2017 statewide results and likely MWP results as well (i.e., it could not be calculated in the MWP). It is worth noting that when TP and MWP data were combined, stressor-response relationships were bolstered, resulting in a larger relative risk estimate for chloride on macroinvertebrate condition (Figure 17) compared to the TP ecoregion alone. This outcome further supports the notion that chloride also poses as elevated risk to macroinvertebrate communities in the MWP ecoregion even though it was not possible to calculate an estimate there due to constraints of the analysis.

Figure 25. 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.



Somewhat surprisingly, none of the stressors measured in the 2017 survey pose an elevated risk to the condition of plant communities in TP depressional wetlands and ponds (Figure 26). In previous surveys, total phosphorus, Kjeldahl nitrogen, and transparency have all posed elevated risks to wetland plant communities in the TP ecoregion. It is likely that the low occurrence of % good plant community condition—a requirement to calculate relative risk—is partly responsible for the lack of any stressors exhibiting an elevated risk to this community in 2017. If this is due to the transition to the FQA methodology, and estimates of exceptional or good plant communities remain low (< 5% of basins), then it is unlikely that relative risk estimation will be possible for TP plant communities in future surveys.

Figure 26. Extent of stressors and their relative risk to plant and macroinvertebrate communities in TP 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 graphs indicates that it did not pose an elevated risk to that community.



A recent study conducted in the Prairie Pothole Region of Iowa supports many of the DWQA's findings as well as highlights potential stressors not monitored by the DWQA. Consistent with the relative risk estimates for the TP ecoregion, Sundberg et al. (2016) found that macroinvertebrate taxa richness decreased as chloride concentrations and turbidity increased. Components of the Iowa study that are not represented in the DWQA include fish abundance and herbicide concentrations. Multiple regression analysis revealed that turbidity increased with fish biomass and that plant cover declined as turbidity and herbicide concentrations increased (Sundberg et al. 2016). These impacts to the plant community cascade down to other organisms as they demonstrated that macroinvertebrate taxa richness decreased as the total cover of submerged, emergent, and floating-leaved plants decreased. Of all the variables measured in the study, total fish biomass had the greatest influence on macroinvertebrate taxa richness. And finally, herbicides were detected in 96% of the wetlands investigated in the Iowa study, a frequency that likely characterizes depressional wetlands and ponds in the TP ecoregion of Minnesota. This study demonstrated the interrelatedness and complexity of stressors and responses—amongst a limited number of variables—highlighting the importance of research in the area of stressor identification/causal pathways and how it can facilitate the remediation of wetland condition after it has been degraded.

Further highlighting the likelihood of multiple cumulative impacts to wetland aquatic communities, Williams and Sweetman (2019) examined the occurrence of neonicotinoid insecticides in seasonally and semi-permanently flooded wetlands in west-central Minnesota. Despite most study sites being surrounded by upland perennial vegetation, neonicotinoids were detected in 50% of them between April and June of 2017 and concentrations were positively correlated with the amount of cultivated crops within a distance of 500 m from each site. Other research has demonstrated even higher occurrences and concentrations of neonicotinoids in wetlands that lack vegetative buffers (Main et al. 2014, Evelsizer and Skopec 2016).

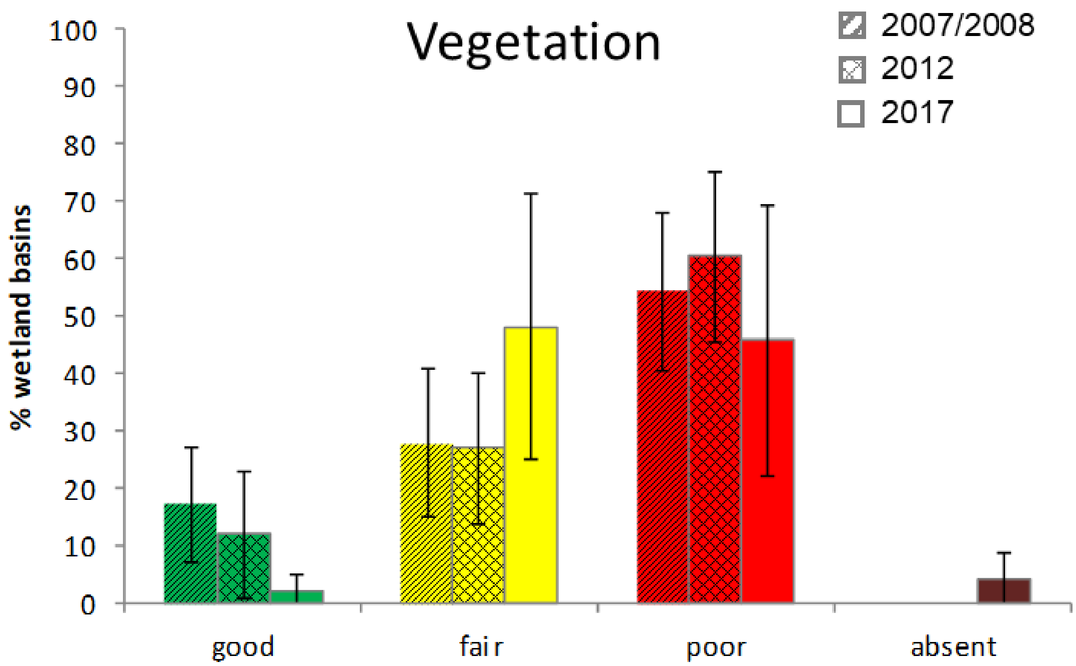
Changes in depressional wetland condition

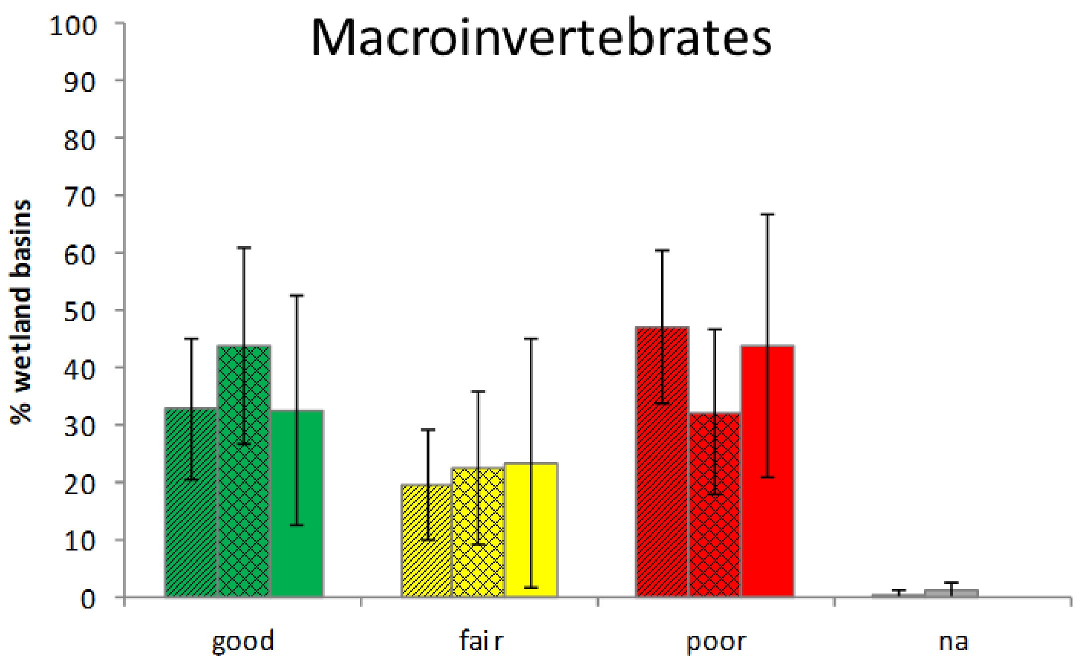
Potential differences in macroinvertebrate and vegetation condition as well as the stressors between survey periods 2008, 2012, and 2017 were analyzed. Several more iterations of the survey will be required before trends can be evaluated, a primary goal of this status and trends survey. Appendix D includes the categorical change detection results (e.g., Is % good increasing over time?) for the Temperate Prairies ecoregion data set as well as for each subpopulation therein, excluding mixed landowner sites due to a small sample size. A summary of those results is provided here.

The condition of aquatic macroinvertebrate communities did not exhibit any significant changes in 2017 across the TP ecoregion or within any of its subpopulations (Figure 27 and Appendix D). Vegetative condition did change in 2017, but only when compared to the baseline 2007/2008 survey. Plant community condition degraded for the ecoregion as a whole as well as in the natural and public ownership categories (Figure 27 and Appendix D). In contrast, plant community condition improved in large (> 12.4 acre) wetlands between the baseline and 2017 surveys. As stated in earlier sections, it is impossible at this time to determine whether these results stem from true changes in condition or the transition in methodology for evaluating the condition of wetland plant communities.

Overall, significant change estimates between survey periods are mixed, without much consistency of indicators showing improving or degrading depressional wetland condition. An exception to this inconsistency was observed within natural wetlands. In 2017, plant community condition degraded in naturally formed wetlands compared to 2007/2008 results (Appendix D). This decline in biological condition was accompanied by increases in chloride and Kjeldahl nitrogen concentrations compared to both previous surveys. In addition, water clarity significantly decreased in 2017 compared to 2012 survey results. Since macroinvertebrate community condition did not show any significant declines and the observed decline in vegetation condition is questionable—due to the method change—it is premature to draw any conclusions regarding the significance of these results at this time.

Figure 27. Comparison of biological condition in TP depressional wetlands and ponds amongst the three DWQA survey periods. Bracketed lines represent the width of the 95% confidence interval associated with each estimate.





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Appendix A

Metrics and scoring criteria for the macroinvertebrate IBIs used to assess the condition of depressional marshes 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
Abundance of non-insect individuals divided by total abundance of sample	Increase		
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 B

Water quality parameters analyzed by Minnesota Department of Health (MDH) 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-1997
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 C

Criteria used to determine wetland condition and stressor levels relative to least disturbed, regional reference sites (macroinvertebrates and stressor indicators) or BCG categories (*wC*).

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

wC condition category assessment criteria for all community types. An additional narrative criterion (< 1% non-native taxa cover) is required to meet the Exceptional condition category (i.e., a community must score above the numeric threshold and meet the narrative requirement to be assessed as Exceptional).

	Community						
Condition category	**Shallow open water**	**Deep marsh**	**Shallow marsh**	**Fresh meadow**	**Wet prairie**	**Calcareous fen**	**Rich fen**
Exceptional			> 4.9*	> 4.2*	> 4.8*	> 7.0*	> 6.4*
Good	> 5.0	> 4.1	> 4.2	> 4.2	> 4.1	> 6.4	> 5.9
Fair	< 5.0	< 4.1	1.9 - 4.2	1.4 - 4.2	1.4 - 4.1	5.9 - 6.4	1.8 - 5.9
Poor			< 1.9	< 1.4	< 1.4	< 5.2	< 1.8
	Community						
Condition category	**Open bog**	**Coniferous bog**	**Shrub-carr**	**Alder thicket**	**Hardwood swamp**	**Coniferous swamp**	**Floodplain forest**
Exceptional	> 7.4*	> 7.3*	> 4.5*	> 4.2*	> 4.6*	> 5.8*	> 4.2*
Good	> 7.0	> 7.1	> 4.5	> 3.9	> 4.2	> 5.6	> 2.7
Fair	5.4 - 7.0	5.9 - 7.1	3.2 - 4.5	2.3 - 3.9	2.5 - 4.2	5.6 - 3.8	2.1 - 2.7
Poor	< 5.4	< 5.9	< 3.2	< 2.3	< 2.5	< 3.8	< 2.1

* Total non-native species cover < 1%

Appendix D

Statistically significant ($p < 0.05$) changes in the percentage of basins represented by condition (G = good/P = poor) or stressor level (L = low/H = high) categories between cycles of the DWQA survey. Green text indicates condition improvement (e.g., +20% G or -20% H), while red text indicates degradation (e.g., -20% G or +20% H). Results presented for statewide data set first followed by MWP and TP ecoregions.

				Wetland Origin						Ownership					
	Statewide			Natural			Man-made			Private			Public		
Indicator	T1 vs T2	T1 vs T3	T2 vs T3	T1 vs T2	T1 vs T3	T2 vs T3	T1 vs T2	T1 vs T3	T2 vs T3	T1 vs T2	T1 vs T3	T2 vs T3	T1 vs T2	T1 vs T3	T2 vs T3
Invertebrate									-29% G					+64% P	+62% P
Vegetation		-14% G	-12% G			-17% G	-21% G	-21% G			-14% G			-62% P	-33% P
Chloride			+28% H					+35% H	+39% H			+24% H			+48% H
Nitrate+Nitrite							-27% H								
Kjeldahl Nitrogen									-35% L					-46% L	
Sulfate														+68% H	+60% H
Total Phosphorus		-20% H					+45% L	-45% H	-49% L		-23% H				
Transparency				-21% G		+18% P								-71% G	

				Area Category					
	< 2.5 ac.			2.5 - 12.4			> 12.4 ac.		
Indicator	T1 vs T2	T1 vs T3	T2 vs T3	T1 vs T2	T1 vs T3	T2 vs T3	T1 vs T2	T1 vs T3	T2 vs T3
Invertebrate						-31% G	+21% G		
Vegetation		-33% P	-29% P		-32% G	-29% G		-25% P	
Chloride			+32% H	-23% H	+20% L				
Nitrate+Nitrite				-8% H					
Kjeldahl Nitrogen									
Sulfate					-25% L			-23% L	-20% L
Total Phosphorus		-27% H	-20% H			-25% L	+23% L		
Transparency								-16% G	

T1 = 2007/2008 survey
T2 = 2012 survey
T3 = 2017 survey

				Wetland Origin						Ownership					
	Mixed Wood Plains			Natural			Man-made			Private			Public		
Indicator	T1 vs T2	T1 vs T3	T2 vs T3	T1 vs T2	T1 vs T3	T2 vs T3	T1 vs T2	T1 vs T3	T2 vs T3	T1 vs T2	T1 vs T3	T2 vs T3	T1 vs T2	T1 vs T3	T2 vs T3
Invertebrate								-26% G	-37% G					-53% G	+77% P
Vegetation		-13% G	-13% G		-30% P						-14% G		+40% G	-82% P	-36% G
Chloride			+31% H		+37% L			+48% H	+56% H						+50% H
Nitrate+Nitrite															
Kjeldahl Nitrogen					+35% L			-38% L	-55% L					-53% L	-46% L
Sulfate												-15% H	-37% L	+86% H	-55% L
Total Phosphorus		-21% H					+54% L	-49% H	-80% L		-22% H			-44% L	-49% L
Transparency				-29% G										-88% G	

				Area Category					
	< 2.5 ac.			2.5 - 12.4			> 12.4 ac.		
Indicator	T1 vs T2	T1 vs T3	T2 vs T3	T1 vs T2	T1 vs T3	T2 vs T3	T1 vs T2	T1 vs T3	T2 vs T3
Invertebrate						-52% G	+26% G		
Vegetation		-45% P	-38% P		-39% G	-39% G			
Chloride				-31% H			-24% H		
Nitrate+Nitrite									
Kjeldahl Nitrogen									
Sulfate					-32% L			-28% L	-31% L
Total Phosphorus		-30% H	-27% H			-48% L	+36% L		
Transparency								-25% G	

T1 = 2007/2008 survey
T2 = 2012 survey
T3 = 2017 survey

	Wetland Origin									Ownership					
	Temperate Prairies			Natural			Man-made			Private			Public		
Indicator	T1 vs T2	T1 vs T3	T2 vs T3	T1 vs T2	T1 vs T3	T2 vs T3	T1 vs T2	T1 vs T3	T2 vs T3	T1 vs T2	T1 vs T3	T2 vs T3	T1 vs T2	T1 vs T3	T2 vs T3
Invertebrate															
Vegetation		-15% G			-15% G									-12% G	
Chloride					+29% H	+31% H									+43% H
Nitrate+Nitrite	-21% H						-34% H			-25% H					
Kjeldahl Nitrogen					-27% L	-35% L									
Sulfate								-36% L	-32% L						
Total Phosphorus		+25% L						+37% L			+34% L				
Transparency						-35% G		+24% G							

	Area Category								
	< 2.5 ac.			2.5 - 12.4			> 12.4 ac.		
Indicator	T1 vs T2	T1 vs T3	T2 vs T3	T1 vs T2	T1 vs T3	T2 vs T3	T1 vs T2	T1 vs T3	T2 vs T3
Invertebrate				-21% G					
Vegetation								-30% P	
Chloride						+24% H		+29% H	
Nitrate+Nitrite	-23% H			-28% H					
Kjeldahl Nitrogen							-28% L		
Sulfate									
Total Phosphorus					+40% L				
Transparency									

T1 = 2007/2008 survey
T2 = 2012 survey
T3 = 2017 survey