Status and trends of wetlands in Minnesota: Depressional Wetland Quality Assessment (2007 – 2012)

Minnesota Pollution Control Agency

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Abbreviations, acronyms, and symbols

BWSR	Minnesota Board of Water and Soil Resources
cm	centimeter
Ch.	Chapter
CL	Confidence Limit
DNR	Minnesota Department of Natural Resources
DWQA	Depressional Wetland Quality Assessment
EPA	U.S. Environmental Protection Agency
FQA	Floristic Quality Assessment
GIS	geographic information system
ha	hectare
IBI	Index of Biological Integrity
m	meter
µg/L	microgram per liter or part per billion
µm	micrometer
mg/L	milligram per liter or part per million
MPCA	Minnesota Pollution Control Agency
MnRAM	Minnesota Routine Assessment Method
MWCA	Minnesota Wetland Condition Assessment
MWP	Mixed Wood Plains
MWS	Mixed Wood Shield
WSTMP	Minnesota Wetland Status and Trends Monitoring Program
N	Nitrogen
P	Phosphorus
R.	Rule
SE	Standard Error
TP	Temperate Prairies
USFWS	U.S. Fish and Wildlife Service
WCA	Wetlands Conservation Act

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 resulted in the passage of the Wetlands 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. Until recently, existing wetland monitoring programs were unable to accurately evaluate whether the WCA was meeting its stated goals.

In 2006, a statewide wetland monitoring program was initiated to assess status and trends of both wetland quantity and quality. 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 round two of the Depressional Wetland Quality Assessment (DWQA), evaluating the ecological condition of depressional marshes and ponds throughout the state and whether this has changed since the initial assessment was completed in 2009.

The DWQA uses a survey approach to produce condition estimates for the entire population of depressional wetlands and ponds based on results obtained from a sample of randomly selected sites. Unlike the initial survey, the 2012 assessment was limited to just the Temperate Prairies (TP) and Mixed Wood Plains (MWP) ecoregions with all data collected in the same year. Plant and macroinvertebrate indices of biological integrity (IBIs) developed and calibrated for each ecoregion, were the primary indicators of wetland condition used in the DWQA. Criteria for categorizing the condition of each sample site as good, fair, or poor were established for each indicator relative to least-disturbed reference sites within each ecoregion. In addition to the biological indicators, several water quality parameters were measured at each study site to better understand their effect on wetland condition.

In order to compare the results of this assessment to the baseline DWQA, some analyses of the baseline survey needed to be adjusted and re-run. For instance, the Mixed Wood Shield (MWS) ecoregion was excluded from the statewide analyses of the baseline survey so that comparisons could be made to 2012 'statewide' estimates that only included the TP and MWP ecoregions. Biological indices as well as the criteria used to categorize all indicators were left unchanged between the two survey periods.

An estimated 111,335 depressional marshes and ponds occur in the TP and MWP ecoregions of Minnesota. This figure is not significantly different from the estimate obtained in the baseline DWQA of 119,779 basins. Almost 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. Natural wetlands still outnumber manmade wetlands and ponds but the gap appears to be shrinking compared to the baseline DWQA results—natural wetlands decreased from 64% to 57% of the population between the two surveys. Change detection analyses revealed that only the number of large wetlands (>12.4 acres) changed between the two survey periods, exhibiting a statistically significant decrease.

One hundred sites were sampled for the 2012 DWQA with 30 of these being revisit sites from the baseline DWQA (Figure 1). Aquatic macroinvertebrate communities (aquatic insects, snails, leeches, and crustaceans) are in good condition at 43% of depressional wetlands while 29% are in poor condition across the study area. An estimated 17% of depressional wetlands have plant communities that are in good condition while 56% are in poor condition. According to analyses of both biological indicators, wetland condition has not changed between the two survey periods. Man-made wetlands appear to be in worse biological condition when compared to natural wetlands, although this comparison could not

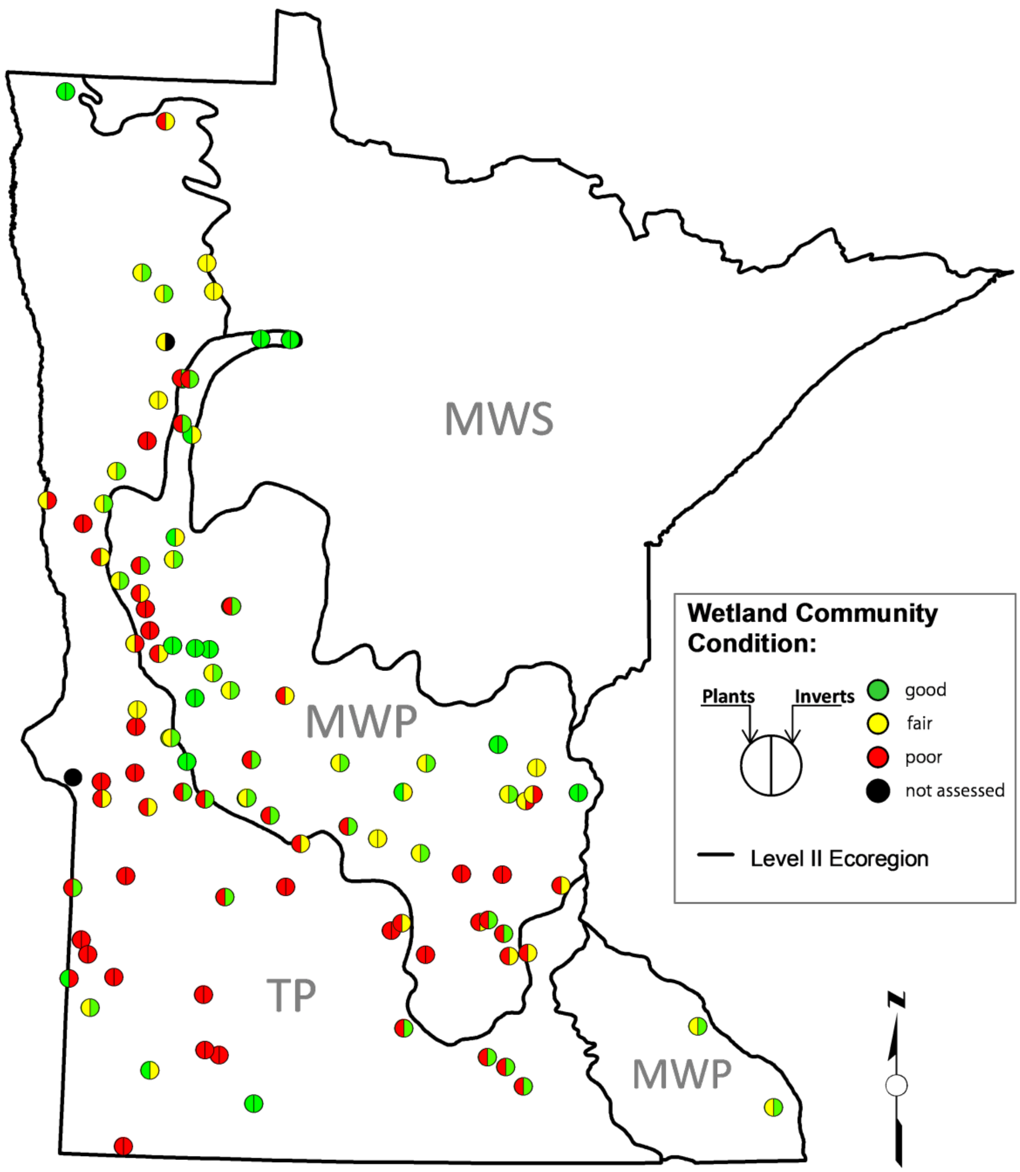


Figure 1. Wetland biological condition at the 2012 DWQA survey sites. Level II ecoregions: MWP – Mixed Wood Plains; MWS – Mixed Wood Shield; TP – Temperate Prairies.

be tested for statistical significance at the statewide scale due to cross-regional differences in the IBIs. Considering the current number (and potential increasing trend) of man-made wetlands and ponds in Minnesota, this will likely have ramifications on meeting no-net-loss goals for wetland quality. Poor plant community quality within the TP ecoregion as well as for man-made and large wetlands appears to be partially due to a greater abundance of non-native invasive plants.

Pollutants such as chloride, phosphorus, and nitrogen may pose an elevated risk to plant and macroinvertebrate communities in depressional wetlands and ponds. However, comparisons of results between surveys and ecoregions reveal inconsistency in these relationships. For example, in the baseline DWQA elevated chloride concentrations were associated with poorer quality plant and macroinvertebrate communities according to relative risk analyses. Results from the 2012 survey indicate that chloride is associated with an elevated risk to wetland plants but not macroinvertebrates (statewide analyses). Of the measured pollutants in this survey, the relationship between increased total phosphorus concentrations and the biological communities was the most consistent. Total phosphorus posed a significant risk to plants and macroinvertebrates in all analyses (across both surveys) with the exception of macroinvertebrates in the TP ecoregion. Similar to the biological results, chemical stressor levels did not exhibit any significant changes between the baseline and 2012 DWQA when examined at the statewide scale.

Based on results of the 2012 DWQA, it appears that the goal of no-net-loss of wetland quality was met for depressional marshes and ponds during the period between the surveys. While a statistically significant change in wetland quality was not detected between the two time periods, the current degraded status of depressional wetland quality reflects a long history of wetland quality loss prior to the initiation of this monitoring program. Determination of whether the state is meeting the no-net-loss goal considering all wetland types will not be possible until the Minnesota Wetland Condition Assessment (MWCA) has completed at least two rounds of data collection. The baseline MWCA survey was conducted in 2011-2012 and the second round is scheduled for 2016. Neither survey considers what may have been lost or gained in terms of wetland condition over the period since the WCA was adopted up until the surveys were first implemented—a period of approximately 20 years. The results obtained in the 2007-2008 DWQA serve as a baseline against which results can be compared as future iterations of the survey are completed. Data collection for the next DWQA is scheduled for summer 2017.

Introduction

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 quantity component of Minnesota's Wetland Status and Trends Monitoring Program (WSTMP) estimates that there are currently 10.6 million acres of wetlands in the state (Kloiber 2010). The WSTMP also demonstrated, as have other studies before it, that wetland area is not distributed evenly across the state with a large percentage of pre-settlement wetlands remaining in the northern forested region and a small percentage remaining in the historically prairie region of the state. Given the observed losses in wetland quantity, it is important to monitor the quality of wetlands that remain as well as those that serve to replace wetlands that have been drained and/or filled (e.g., mitigation wetlands) in order to ensure that the ecosystem services wetlands provide are maintained at the local, watershed, and ecoregion scale.

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. In Minnesota, public recognition of this resulted in the passage of the WCA in 1991.

The overall goal 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 does 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 existing information gaps.

Minnesota Wetland Condition Assessment

A second wetland quality survey was initiated in 2011. The Minnesota Wetland Condition Assessment (MWCA) broadens the scope of wetland types being monitored across the state. The results of this survey which focuses on the condition of wetland plant communities will be presented in a separate report to be published simultaneously with this report.

One of the primary outcomes of this strategy was the development of statewide random surveys to begin assessing the status and trends of wetland quantity and quality in Minnesota (i.e., WSTMP). The wetland quantity survey, modeled after the U.S. Fish and Wildlife Service wetland status and trends program (e.g., Dahl 2006, 2011), is being implemented by the DNR. The second iteration of this survey, looking at change between 2006 and 2011, estimated a net gain of ~2,000 acres statewide over this period with man-made ponds accounting for most of this gain (Kloiber and Norris 2013). The MPCA is responsible for conducting the wetland quality survey. The baseline wetland quality survey was limited to depressional wetlands (Genet 2012). The results of the second round of this survey, the DWQA, are the focus of this report.

Measuring wetland quality

Biological monitoring and assessment is one of the most commonly used approaches for measuring the ecological condition of aquatic ecosystems. Aquatic organisms are constantly exposed to their environment and, as a result, are able to integrate the effects of multiple stressors occurring 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 community attributes (i.e., metrics) that reliably respond to human disturbance (Karr and Chu 1999). 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 culminated with the development and validation of ecoregion-specific, wetland plant and macroinvertebrate IBIs (Appendix A). Currently, depressional marshes and ponds are the only types of wetlands that the MPCA has developed IBIs and assessment criteria for. The DWQA utilizes these plant and macroinvertebrate IBIs as indicators of wetland condition.

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 stress 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 many 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 unvegetated 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) 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).

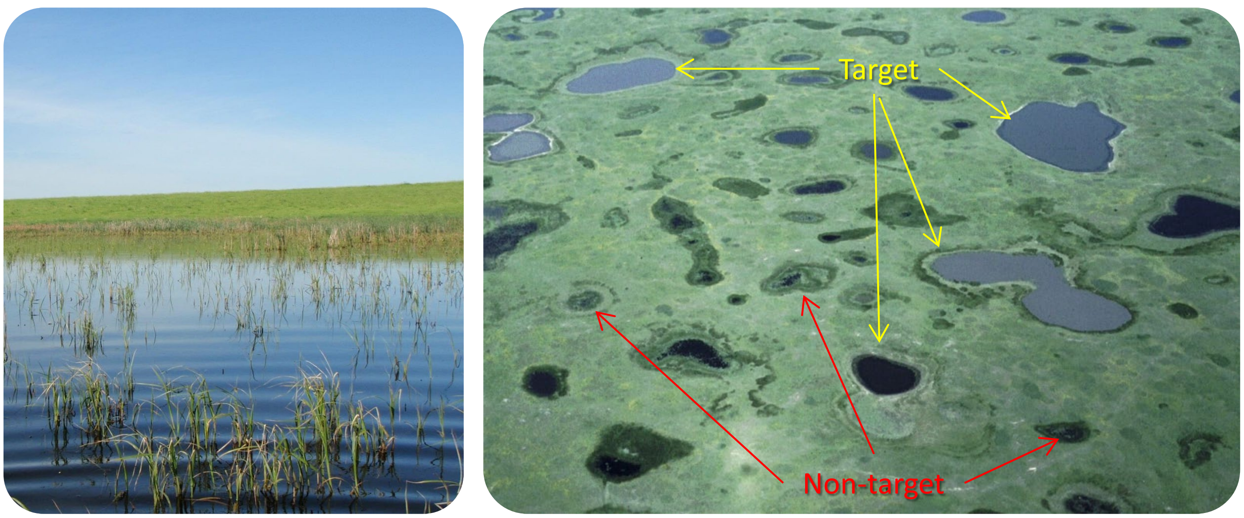


Figure 2. Prairie potholes are an example of the type of wetland included in the DWQA. Temporary and seasonally flooded wetlands were not included in this survey (= non-target above). Aerial photo courtesy of USFWS.

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 and regional 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 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 will be 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?

- **Limited to the Mixed Wood Plains and Temperate Prairies Ecoregions.** Target wetland types of the DWQA represent a small portion of the overall wetland resource in the MWS ecoregion. Also, the companion survey to the DWQA, the Minnesota Wetland Condition Assessment or MWCA, will include depressional wetlands in the MWS. Thus, the decision was made to exclude the MWS ecoregion from the DWQA.
- **Data collection limited to one year.** The baseline DWQA collected data over three years on a rotating ecoregion basis, requiring a subset of sites measured each of those years to **estimate** interannual variability. The 2012 survey limits data collection to one year, eliminating the need for such annual sites and their analysis.
- **MnRAM functional assessments excluded.** Resource constraints resulted in this suite of functional indicators being excluded from the 2012 survey. It is possible that it could be added back into future iterations of the DWQA.
- **Sulfate added as an indicator of stress.** Due to issues with lab analyses of water samples in 2009, sulfate data from the MWS ecoregion could not be included in the baseline survey report (see Box 1). Data from both surveys is available for the MWP and TP ecoregions and results are included in this report.
- **Water transparency now measured using Secchi tube.** A change in the methodology for water transparency was adopted in the 2012 DWQA to increase the precision and accuracy of these measurements (see Box 2).
- **Change Analysis.** Time 2 data allows for an analysis of change in the quantity and condition of depressional wetlands and ponds over the five year period since the baseline DWQA survey was conducted.
- **Assumptions regarding landowner denied sites.** In the baseline survey, landowner denied sites were assumed to be 'non-target' and thus did not factor into the population estimates. This approach was changed in the 2012 survey to remove the influence that landowner sentiment (i.e., willingness to allow site access) has on population estimates from one survey to the next. Therefore, 2007-2008 population totals presented in this report reflect this change and differ from those presented in the baseline DWQA report.
- **Results presented as numbers of basins.** Similar to lakes, depressional wetlands and ponds are discrete objects. Rather than present findings in terms of both numbers and acreage—as was done in the baseline survey—the 2012 DWQA focuses on the number of basins when reporting population estimates.

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 the entire population of wetlands by measuring a comparatively small sample of wetland sites. Wetlands were randomly selected to ensure that derived estimates are unbiased. In addition, the selection process was spatially stratified (Stevens and Olsen 2004) to increase the likelihood that the sample represented all regions of the state. To maximize participation in the survey, all landowners were contacted in the weeks prior to sampling to obtain permission and/or the appropriate permits.

The DWQA utilizes 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 year of the survey. Since the plots themselves represent only a sample of the entire state (~5,000 plots randomly selected throughout the state), the DWQA survey design represents 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. The 2012 DWQA used a different approach and sampled the MWP and TP ecoregions (i.e., the DWQA's new definition of statewide) 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).

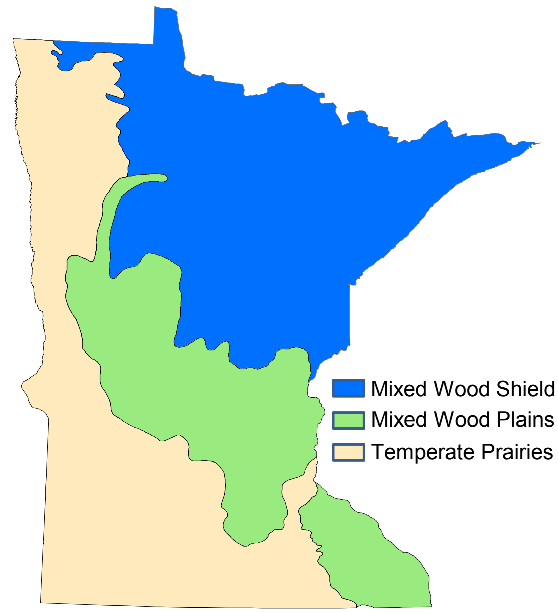


Figure 3. Level II ecoregions in Minnesota.

The desired sample size for each ecoregion was 50 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). To increase the power of the survey to detect trends a subset of sites from the original DWQA were randomly selected for sampling in this iteration of the DWQA. The site distribution was 35 new sites and 15 repeat sites (i.e., a 30% re-sample rate) within each ecoregion, resulting in a total of 100 sites for the 2012 survey.

Site selections for both the wetland quantity and quality surveys were provided by the U.S. Environmental Protection Agency (EPA) National Health and Environmental Effects Research Laboratory, Corvallis, Oregon. For more details on the design of this survey and its relationship to Minnesota's wetland quantity survey see design summary paper (http://www.pca.state.mn.us/index.php/view-document.html?gid=6095).

Based on an evaluation of site characteristics as well as aerial imagery, a post-stratification process was used to classify sampled survey sites 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 residential (ornamental) ponds. Waterbodies that require continuous pumping or lining (e.g., geo-textile fabric) to maintain their hydrology were not included in this survey.

Figure 4. Stormwater retention pond in Scott County.

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-permanent, 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). Several 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 laboratory for identification of macroinvertebrates. More information on the dip net method is 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 macroinvertebrate sampling visit. A multi-meter (Hach HQ40d18) was used to measure water temperature (^{o}C), 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),

Box 1. Sulfate (SO_4) added to DWQA as an indicator of stress

Sulfate is a chemical compound that naturally occurs in lakes, streams, and wetlands. In areas of these water bodies that lack oxygen (e.g., substrates) sulfate may be reduced to hydrogen sulfide (H_2S) through microbial decomposition of organic matter. The natural background concentration of sulfate may be increased when surface waters receive discharge from industrial facilities and wastewater treatment plants or runoff from agricultural land. When this occurs H_2S may accumulate and reach levels toxic to aquatic organisms. Increased sulfate concentrations, through its effect on iron availability, can also promote the release of phosphorus from bottom sediments, which may lead to excess algal growth (Wetzel 2001).

It was originally intended to include sulfate as a water quality indicator in the baseline (2007-2009) DWQA. However, analyses of sulfate concentrations for water samples collected in 2009 (MWS ecoregion) yielded inaccurate results. Thus, without any sulfate data from the MWS ecoregion, it was decided to exclude sulfate results from the baseline DWQA. With the DWQA now limited to the MWP and TP ecoregions, it was possible to re-introduce sulfate back into the survey, including the data that was collected in 2007 and 2008. While results from the baseline DWQA are not directly presented in this report, sulfate data are included in the change analyses that compare the 2012 results to the baseline survey.

Ecoregion expectations for high, medium, and low sulfate stressor categories were established using the same reference site approach that was used for other indicators in the survey. Due to surface geology being the primary determinant of its natural occurrence in surface water, sulfate concentrations tend to be higher in water bodies located in the western and southern area of the state, corresponding to the Temperate Prairies ecoregion. As such, stressor category criteria values are notably higher for the TP ecoregion compared to the MWP (see Appendix C).

and sulfate (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; a modification of the method used in the baseline DWQA (see Box 2). Details of the water chemistry sampling procedure can be found at: http://www.pca.state.mn.us/index.php/view-document.html?gid=10251.

Wetland plant community sampling at each study site was conducted in July using a releve sampling method. Releve sampling is a technique where the biologist selects a plot location that is representative of the overall targeted plant community. Plot placement focused on the emergent community of each wetland, though the final sampling area typically straddled the emergent/ submerged aquatic vegetation interface and the shape of the plot (square or rectangular) depended on the width of the emergent fringe. In both ecoregions a single 100 m^2 plot was used as a representative sample of the wetland plant community. An inventory of plant species was generated for each plot and the percentage of plot area occupied by each species was estimated using cover classes (Table 1). More information on this method is available at: http://www.pca.state.mn.us/index.php/view-document.html?gid=6111.

Table 1. Cover classes and corresponding ranges of percent cover.

Cover class	% Cover range
8	95 – 100%
7	75 – 94%
6	50 – 74%
5	25 – 49%
4	10 – 24%
3	5 – 9%
2	2 – 4%
1	1%
0.5	0.1 – 0.9%
0.1	Single/few

Data analysis

Biological and stressor indicator data collected from reference sites were used to represent the range of expected values for least-disturbed 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 stressor categories (Figure 6A and B). For example, the 25th percentile of the reference distribution was used to separate the good and fair condition categories. In other words, study sites with indicator values above this threshold are considered to be in good condition; i.e., comparable to the condition of least-disturbed reference sites (Figure 6A). The 5th percentile 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-disturbed 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).

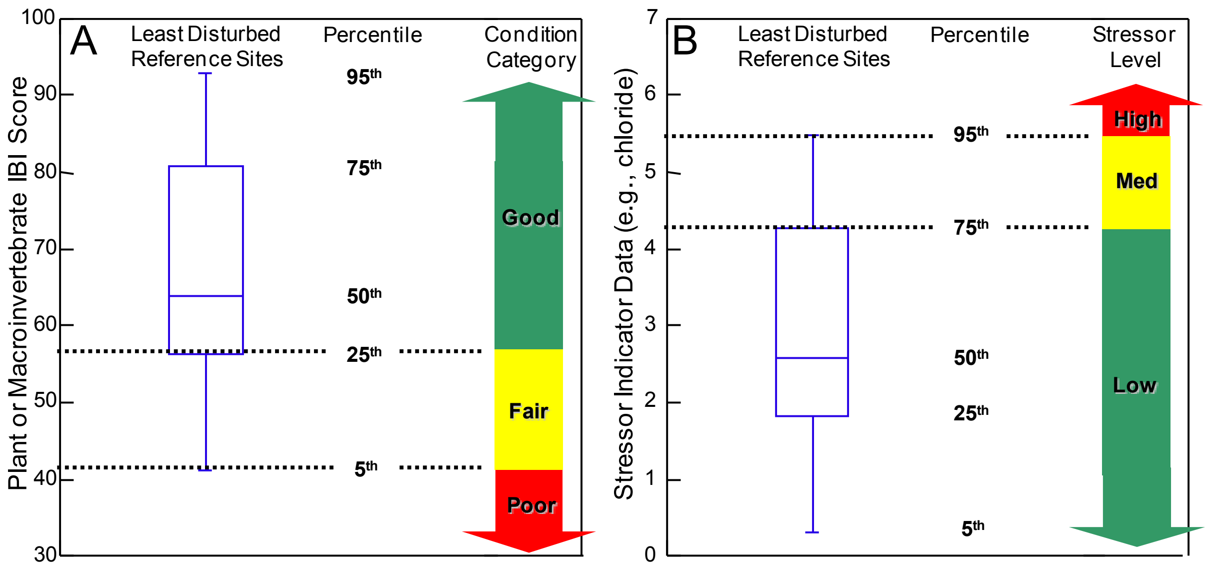


Figure 6. Generalized depiction of the distribution (represented as a boxplots) of indicator values at reference sites and the process for using this information to categorize the (A) condition and (B) stressor levels of each sample site. Sites were categorized independently based on each indicator.

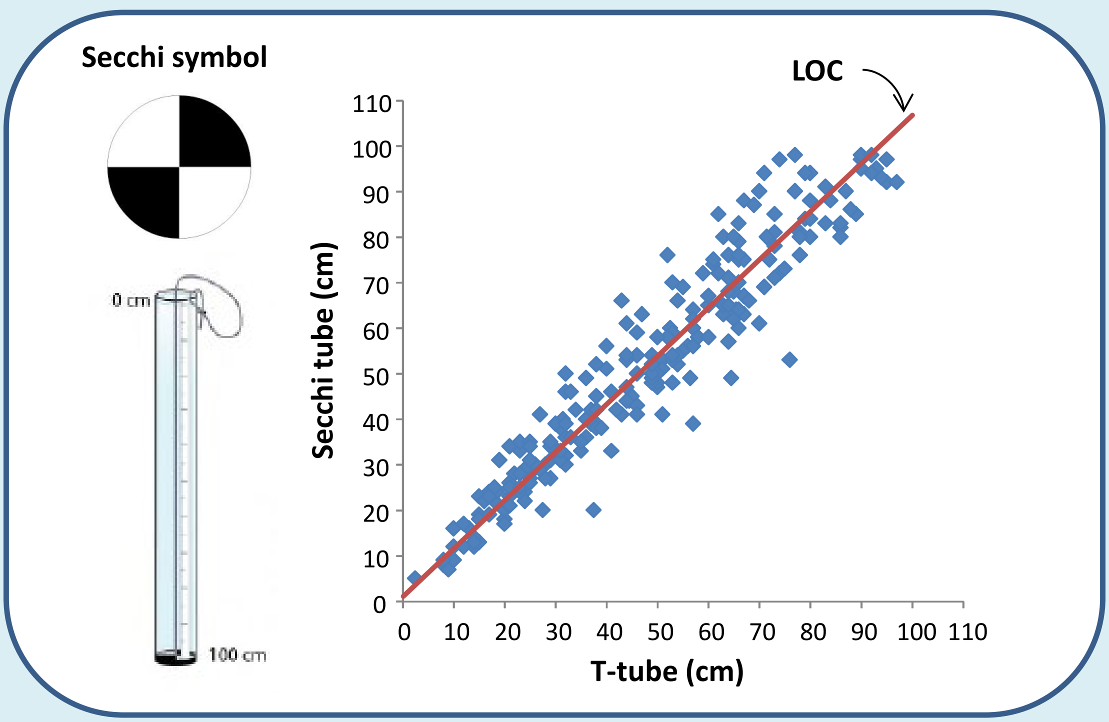
Condition and stressor categorization criteria were used to rate indicator results individually for each study site. The results from this random sample of sites were used in conjunction with the design weights incorporated into the site selection process to estimate the proportion of the population in each category. All analyses were performed in R version 3.0.2 (R Core Team 2013) using the spatial survey design and analysis package (spsurvey 2.6; Kincaid and Olsen 2013). An analysis script was developed 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 the initial survey. Relative risk was estimated using the ratio of the probability of poor condition/high stressor levels (numerator) to the probability of poor condition/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 (see Box 3) using spsurvey 2.6 (Kincaid and Olsen 2013). Unlike the baseline DWQA, results here were only reported based on the number or percentage of wetland basins, the ability to report findings based on wetland area was not a component of the analyses.

Box 2. A new approach to measuring water transparency.

The baseline DWQA survey utilized transparency tubes or T-tubes to measure the clarity of the water column in depressional wetlands and ponds. In 2012 the MPCA switched to using a slightly different piece of equipment to measure water clarity, the Secchi tube. Both methods rely on the key principle of determining when the Secchi symbol becomes distinguishable to the observer in a tube (100 cm or 60 cm in length) filled with sample water. The T-tube's approach for making this measurement is to drain sample water out of the bottom of the tube until the symbol becomes visible, record this depth, continue draining until the center screw becomes visible, record this depth, and then average the two readings. For the Secchi tube, rather than releasing water from the tube an observer moves a mini-Secchi disk within the tube, manipulating its depth to find the point at which the symbol becomes distinguishable. This typically involves slowly moving the disk up and down to see the exact depth at which the symbol disappears and reappears. A standard 100 cm Secchi tube is used and only one depth is recorded, thereby increasing the precision of the method compared to the T-tube.

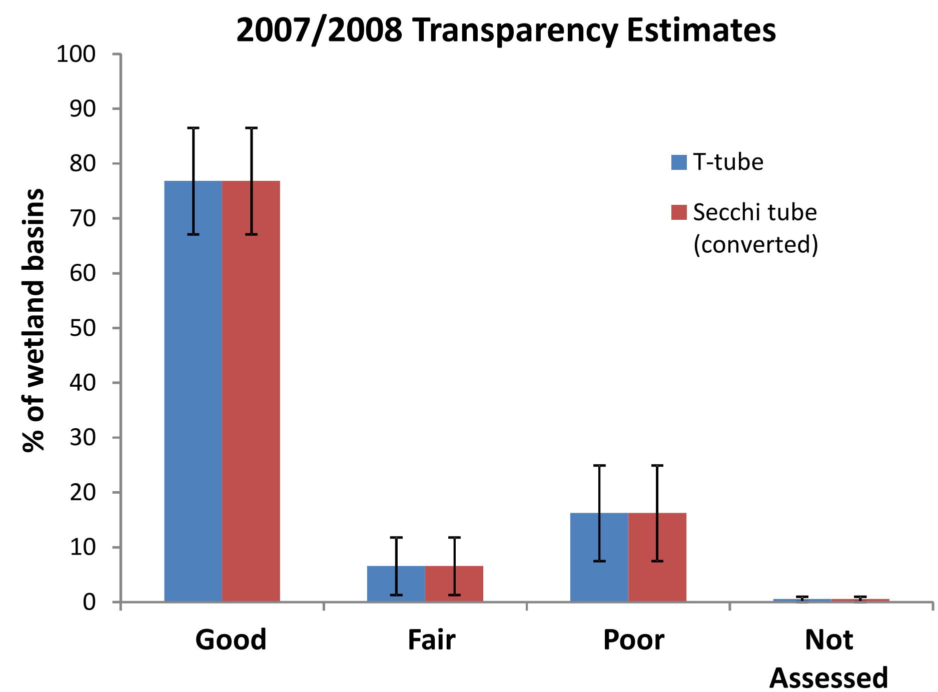
In order to compare Secchi tube data collected in the 2012 survey to T-tube data collected in the baseline survey, the relationship between the two measurements needed to be understood so that T-tube measurements could be converted into their equivalent Secchi tube measurements. Therefore, paired Secchi tube and T-tube measurements were made at each of the 2012 wetland survey sites. This data set was combined with paired measurements collected in rivers and streams throughout the state in preparation for the MPCA's transition to Secchi tube measurements. A Line of Organic Correlation (LOC) was fit to the data set and used as the basis of the conversion where:

$$\text{Secchi Tube} = 1.203 + (1.056 \times \text{T-tube})$$



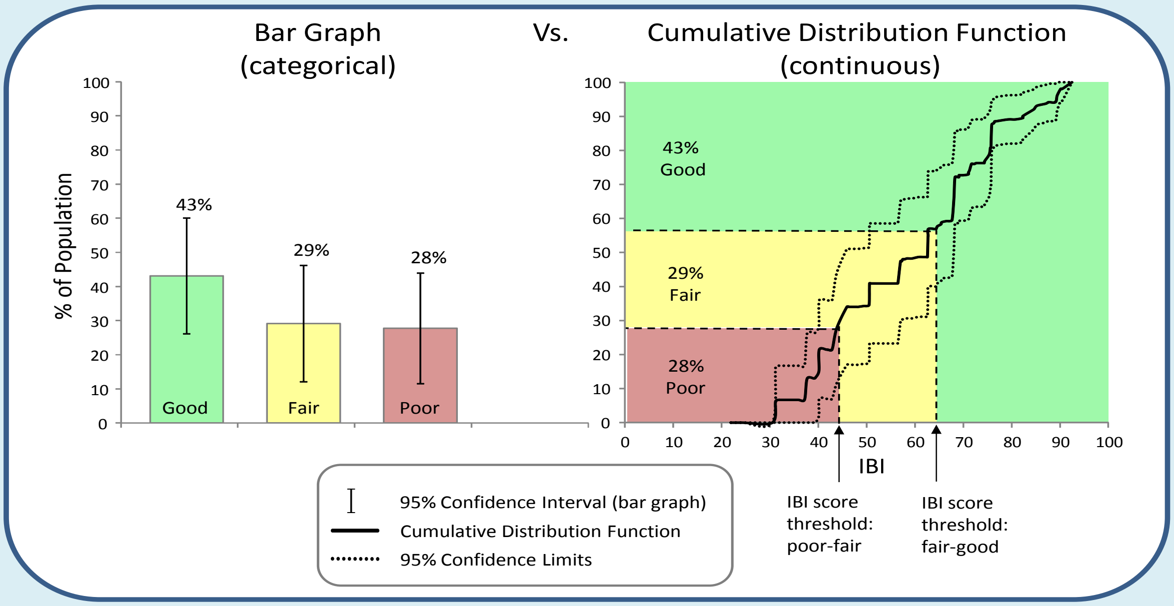
Box 2. A new approach to measuring water transparency (continued).

Using the above equation, 2007 and 2008 T-tube measurements were converted to Secchi tube measurements. The criteria used to categorize T-tube data as Good/Fair/Poor were also converted to their equivalent Secchi tube readings. Using these criteria, Secchi tube data from the baseline survey (i.e., converted 2007 and 2008 T-tube data) were analyzed to insure that results were similar to the original results obtained using T-tube data (see graph below). This provided a clear indication that comparing 2007-2008 converted Secchi tube data to the 2012 Secchi tube measurements for the change analysis would be valid.



Box 3. Presenting the results of the DWQA.

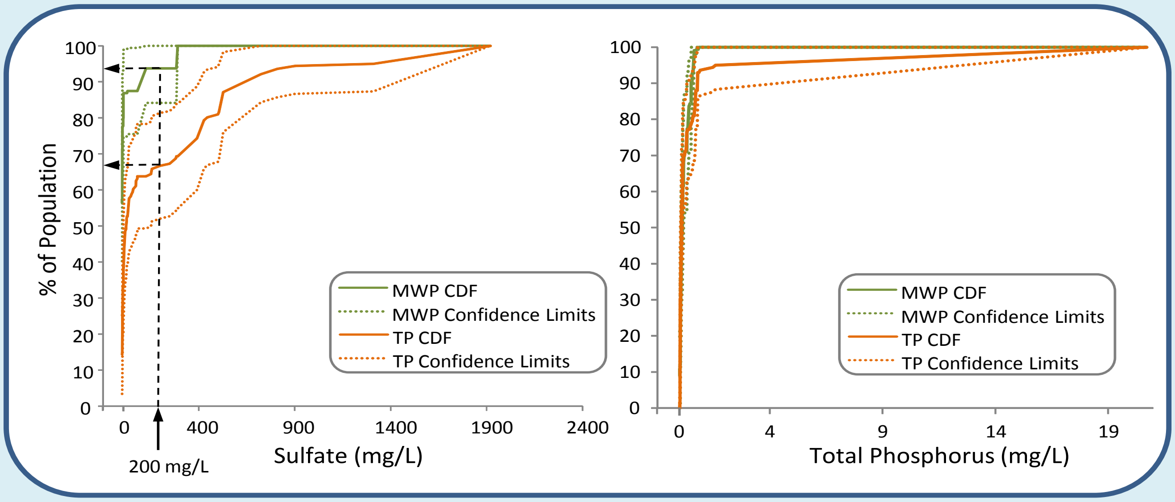
There are a variety of options for analysis of probabilistic survey data and presentation of those results. Choosing which results to include and how to present them depends on the objectives of the survey and the intended audience of the report. The intended audience of the DWQA reports is broad, including well-informed citizens, wetland professionals, and policy-makers. As such, the report includes figures intended to present estimates of wetland condition and stress in a clear, concise manner (e.g., pie charts). Other figures, such as bar charts, include the margin of error (i.e., 95% confidence limits) associated with each estimate, which is useful when different groups or subsets of the data set are being compared. Alternatively, rather than presenting the data categorically (e.g., good/fair/poor), data can be examined and analyzed as a continuous variable (e.g., IBI scores). Cumulative distribution functions or CDFs summarize a probabilistic survey data set across the entire range of values, providing the estimated proportion of the population that equals or is less than any given value of an indicator (e.g., 28% of the population has an IBI score of 44 or less in the CDF below). CDFs are generally not as easy to interpret as bar or pie charts, although additional information (e.g., categorical criteria) can be added to aid with their interpretation as was done below. Given the number of results and comparisons this report required, the relative simplicity of pie and bar charts was the basis for taking a categorical approach to the presentation of population estimates. The graphs below show the same data set (MWP macroinvertebrate IBI scores) presented both categorically as a bar graph and continually as a CDF to illustrate the relationship between these two representations of the results.



CDFs offer a robust method for comparing subsets of the data to look for relationships (e.g., regional, wetland area, etc.) in the observed results. Even though comparisons of population estimates (e.g., man-made vs. natural) were presented as either bar or pie graphs in this report, any statistical testing of these different groupings was done using the CDF test in spsurvey 2.6 (Kincaid and Olsen 2013). This analysis utilized the Wald Test (F Distribution) to test for differences between CDFs of various subpopulations in a pairwise manner. Throughout this

Box 3. Presenting the results of the DWQA (continued).

report, other than the change detection analyses, statistical test results for examining differences between subsets of the population (i.e., 2012 status results) are based on the Wald Test (see example below).



The graph on the left depicts a comparison of CDFs where the two subpopulations, in this case ecoregions, are significantly different according to the Wald Test ($F = 12.13$, $df_1 = 2$, $df_2 = 94$, $p < 0.001$). Depressional wetlands in the TP ecoregion have significantly higher sulfate concentrations than wetlands in the MWP. Using 200 mg/L as an arbitrary value to illustrate this point, a vertical line drawn from this value on the x-axis intersects the TP CDF at 67% of the population (i.e., 67% have SO_4 concentrations at or below 200 mg/L) and the MWP CDF at 94% of the population. In other words, the MWP only has an estimated 6% of its depressional wetlands that exceed 200 mg/L while the TP has an estimated 33% exceeding this concentration. The graph on the right shows a comparison of CDFs that do not exhibit a statistically significant difference ($\alpha = 0.05$).

Throughout this report, results of CDF tests accompany the associated bar graphs that show the comparison of categorical distributions (e.g., high/medium/low) of each subpopulation. On occasion, these two approaches to handling the data (continuous vs. categorical) may appear to be inconsistent. For instance, there may be cases where the distribution amongst categories high/medium/low for an indicator are virtually identical between ecoregions while the CDF test yields a statistically significant difference in their continuous distributions. Such situations arise when criteria for determining categories, which take into account the natural, regional variability of an indicator, vary considerably between ecoregions (see Appendix C). While the bar graphs account for this regional variability in the comparison of subpopulations (e.g., a concentration that is considered high in one ecoregion, may be considered medium in the other), the CDF tests do not and instead examine whether a difference exists between the raw data distributions of each subpopulation. All comparisons that incorporate data from both ecoregions (see Statewide Results and Discussion) are affected by the scenario described above; ecoregion comparisons being the most affected with other subpopulation comparisons affected to a lesser degree. Any comparisons within ecoregions (see Results and Discussion sections for each ecoregion), although afflicted with smaller sample sizes, do not have this discrepancy between categorical and continuous treatments of the data because the same set of criteria is used to categorize all indicator values.

Statewide results and discussion

Data collected in 2012 provides a snapshot of the current number and condition of depressional wetlands and ponds in Minnesota. In addition to this status update, the new data set also allows an analysis of change (since 2007-2008 survey) to be conducted. In this section, wetland quantity status and change results are presented first, followed by wetland condition status, and then changes in wetland condition.

Depressional wetland quantity (status and change)

The 2012 survey estimated a total of 111,335 depressional wetlands and ponds occurring within the Temperate Prairies and Mixed Wood Plains ecoregions combined. This estimate represents a slight decrease in the total number of basins when compared to the 2007-2008 estimate of 119,779 for the two ecoregions combined (Figure 7). However, this does not represent a statistically significant change between the two time periods given the margin of error (95% CL) associated with each estimate. Unlike the basin quantity estimates presented in the baseline DWQA report, results reported here for 2007-2008 and 2012 include the landowner denied portion of the population. This revised approach assumes that sites occurring on private property where access was denied during the site evaluation process (i.e., landowner denied) belong to the target population and eliminates the influence of landowner sentiment on quantity estimates between surveys. An examination of aerial imagery for the landowner denied sites from the baseline survey suggests that this is a valid assumption in most cases. All condition analyses have been adjusted so that population totals include landowner denied sites when results are presented in terms of the number of basins.

Compared to the 2007-2008 results, the proportion of man-made depressional wetlands, which tend to be open water ponds, appears to be on the rise (Figure 7); however, the observed increase in the number of man-made wetlands between the two surveys was not statistically significant. If real, this increase would be consistent with the findings of wetland quantity status and trend surveys that distinguish natural vs. man-made wetland types (e.g., Dahl et al. 2011, Kloiber and Norris 2013). It will take several more iterations of the DWQA survey to determine whether such a trend exists for man-made depressional wetlands in Minnesota.

Of all the subpopulations examined in the survey, only the large (>12.4 acre) wetland size category showed a statistically significant change in quantity between the two time periods ($Z = -3.12$, $p = 0.002$). There were an estimated 10,366 large wetland basins in the MWP and TP ecoregions in 2007-2008 (Figure 7). This estimate declined to 7,713 large wetland basins in 2012. It would be premature to draw conclusions at this point regarding this result as changes between any two time periods are not necessarily cause for concern. Rather, it is expected that some estimates will demonstrate a statistically significant change in one survey cycle, while showing no change in the next. Such results may hint at the need for a review of existing policies and regulations. However, a consistent pattern of increases or decreases in the data over multiple time periods (i.e., trend)—or lack thereof—is a more meaningful result, and only then can a confident evaluation of the effectiveness of policies and regulations be achieved.

Between the 2007-2008 and 2012 surveys, the number of depressional wetlands in the MWP and TP ecoregions remained steady. Preliminary indications provided by the 2012 estimates suggest that man-made wetlands may be increasing while the number of large depressional wetlands may be in decline.

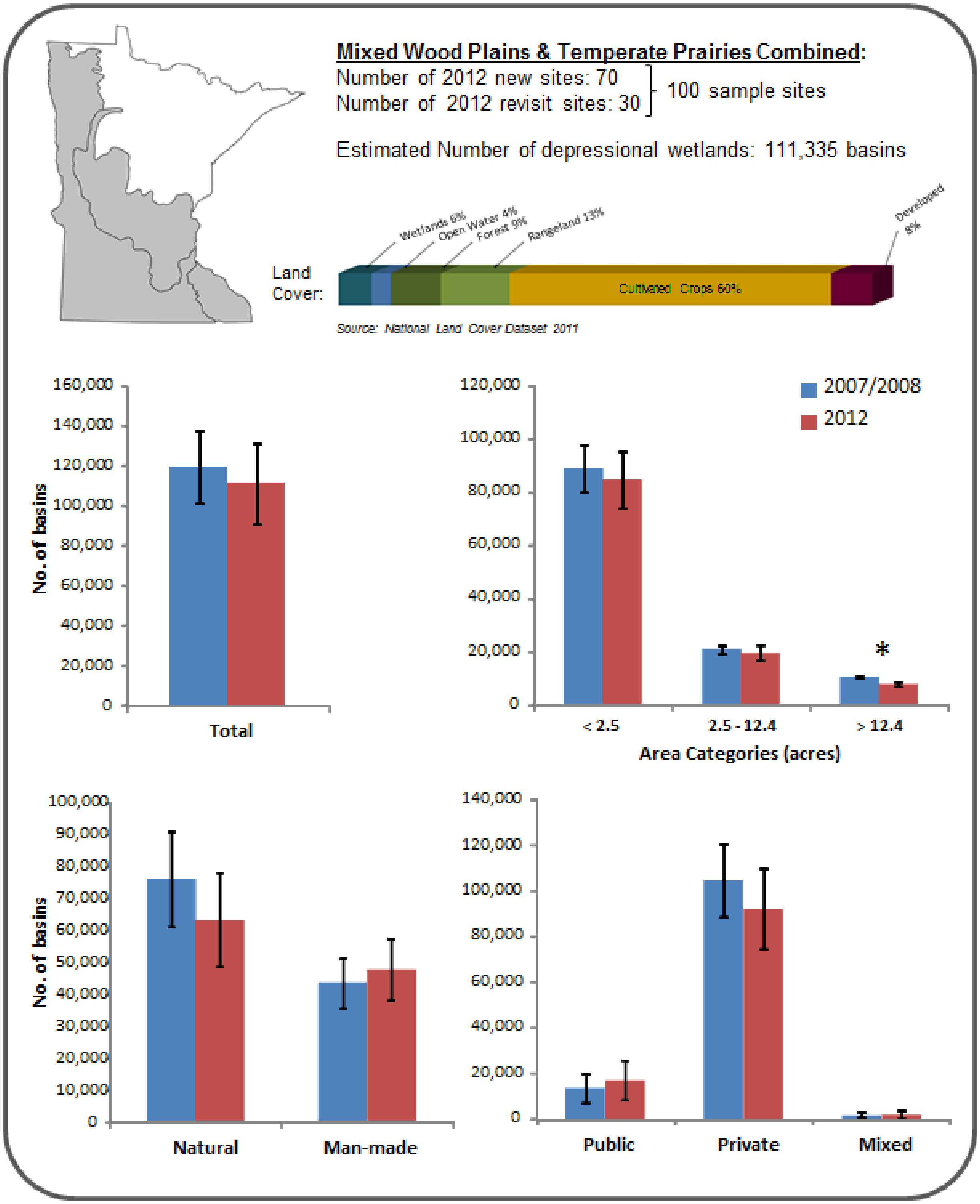


Figure 7. Estimates of the total number of depressional wetlands in the Mixed Wood Plains and Temperate Prairies ecoregions comparing time 1 of the survey (2007-2008) to time 2 (2012). Bracketed lines represent the 95% confidence interval associated with each estimate. Asterisk indicates a statistically significant change between time periods.

Both results should be interpreted with caution until data have been collected over multiple time periods to validate these initial findings. The 2012 estimates reiterate the 2007-2008 findings and show that the majority (~76%) of depressional wetlands in the state are less than 2.5 acres in size and occur most frequently (~83% of the time) on private property.

Depressional wetland condition

Macroinvertebrate communities are in good condition in 43% of depressional wetlands and ponds throughout Minnesota while 29% are in poor condition (Figure 8). The condition of macroinvertebrate communities is remarkably similar between the two ecoregions. However, a CDF test could not be used to compare the biological condition of the two ecoregions because even though both IBIs range from 0 to 100, each was developed and calibrated independently using least-disturbed reference conditions (e.g., Figure 6A). This means that an IBI score of 50 in the MWP doesn't necessarily mean the same thing as a score of 50 in the TP in terms of biological condition. Furthermore, each ecoregion IBI has different criteria (Appendix C). For example, a wetland scoring below 56 in the TP is considered to be in poor condition while in the MWP the criterion for a poor condition rating is a macroinvertebrate IBI score of 44 or less. It is for this reason that CDF comparisons which included macroinvertebrate or plant IBI scores from both ecoregions (i.e., comparisons of subpopulations at the statewide scale) were not valid and testing of the statistical significance of observed differences was not performed.

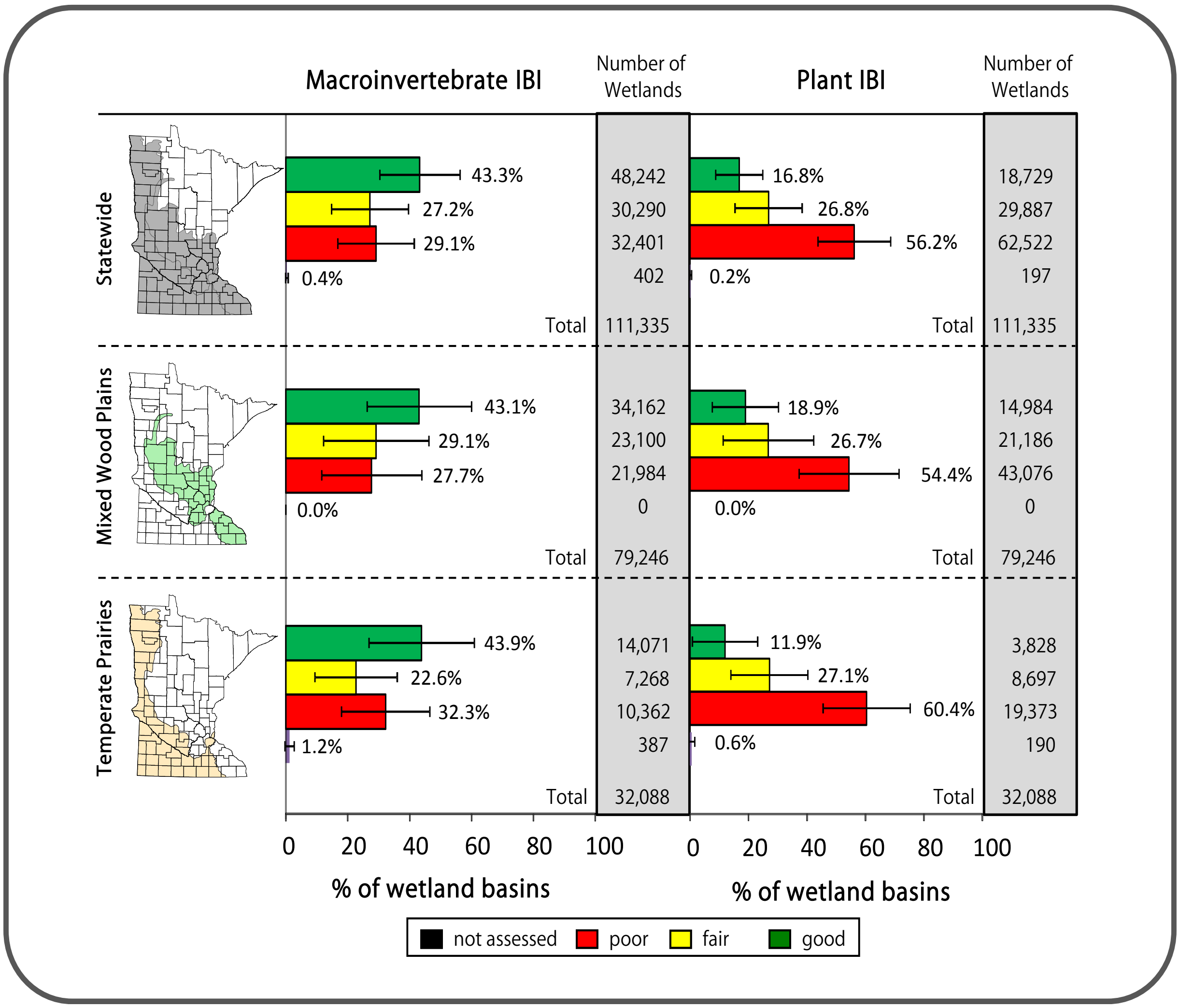


Figure 8. Biological condition of Minnesota's depressional wetlands and ponds in 2012 according to macroinvertebrate and plant IBIs, including the estimated number of wetlands within each condition category. Bracketed lines represent the width of the 95% confidence interval associated with each estimate. Percentages may not add up to 100% due to rounding.

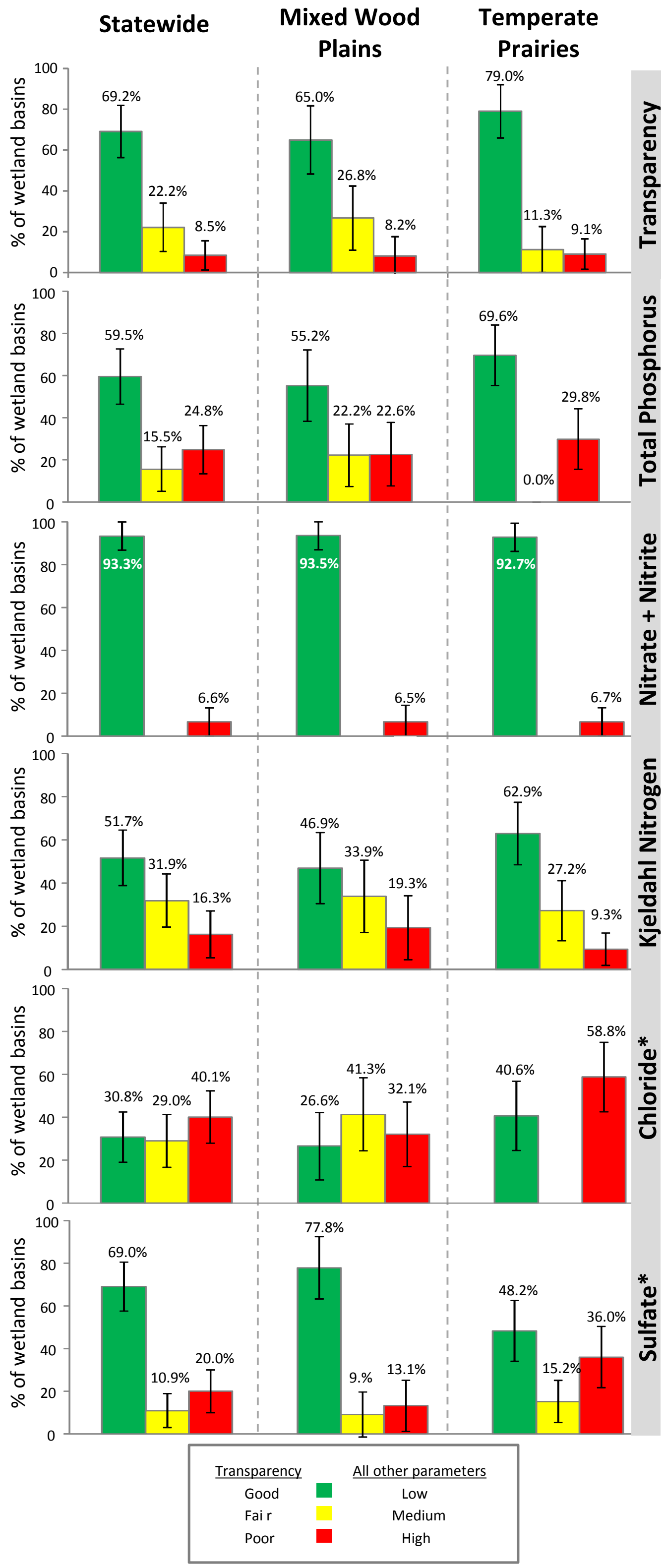


Figure 9. Stressor levels in Minnesota's depressional wetlands and ponds. Bracketed lines represent the width of the 95%confidence interval associated with each estimate. Asterisks indicate significant differences in stressor levels between ecoregions according to CDF test.

According to the IBI results, approximately 17% of depressional wetlands and ponds have plant communities that are in good condition while 56% are in poor condition (Figure 8). As observed with the macroinvertebrates, plant community condition is very similar between the MWP and TP ecoregions. In both ecoregions, depressional wetland plant communities appear to be in worse condition than macroinvertebrate communities. This result is not surprising and actually highlights the rationale for using two biological communities to assess ecological condition: communities respond differently to various chemical, physical, and biological stressors. In this particular situation, invasive plant species, when viewed as a biological stressor, impact wetland plant communities to a greater degree than they do macroinvertebrate communities (see Figure 11).

Based on the percentage of wetlands that were categorized as having high concentrations of a pollutant, phosphorus, sulfate, and chloride appear to be common chemical stressors to the condition of depressional wetland communities (Figure 9). Relative to least-disturbed reference conditions, chloride concentrations are high in 40% of depressional wetlands and ponds statewide— the highest amongst the measured chemical stressors. High (i.e., detectable) nitrate + nitrite concentrations occurred in approximately 7% of the population (Figure 9). These two pollutants are ephemeral in wetland habitats (Whitmire and Hamilton 2005) and most often detected following precipitation, particularly when subsurface drain tile run-off flows into a wetland. A large portion of the TP ecoregion saw limited rainfall during the month of June, which likely contributed to the low occurrence of nitrate + nitrite detections in the water samples (see Appendix D for details).

Of the measured water chemistry parameters, only sulfate and chloride are significantly different between the two ecoregions. According to the CDF test (see Box 3 for details on this analysis) chloride concentrations are significantly higher in the TP ecoregion ($F = 4.05$, $df_1 = 2$, $df_2 = 94$, $p < 0.05$) as is the case for sulfate ($F = 12.13$, $df_1 = 2$, $df_2 = 94$, $p < 0.001$). Given the orientation of these two ecoregions, this pattern is not surprising. Previous work by Moyle (1956) demonstrated a regional gradient in the surface water concentration of these two ions in Minnesota, increasing northeast to southwest. This regional difference was also observed in the least-disturbed reference wetland dataset, which resulted in different high/medium/low categorization criteria for chloride and sulfate in the two ecoregions (see Appendix C). It is important to take into consideration natural, spatial variation in biological and chemical indicators when evaluating whether or not a resource is degraded. This survey accounted for this variation and, even so, demonstrated that a portion of the population exceeded these regionally calibrated thresholds (Figure 9), indicating potentially stressful conditions for the aquatic life that inhabit these wetlands.

The abundance of non-native invasive wetland plants such as Purple loosestrife (*Lythrum salicaria* L.), Narrowleaf cattail (*Typha angustifolia* L.), and Reed canary grass (*Phalaris arundinacea* L.) in the emergent vegetation zone was examined to determine its effects on depressional wetland quality. Non-native invasive plant species can colonize habitats following anthropogenic or natural disturbance; tolerate a broader range of further impacts (e.g., nutrient enrichment, hydrologic alteration); and aggressively spread by out-competing other native plants for limited resources (e.g., light and nutrients). Depending on the circumstances of colonization and proliferation within a wetland, invasive plant species may be considered a stressor to the ecological condition of depressional wetlands or a response to human impacts. In most cases it is impossible to determine the conditions, natural or anthropogenic, that led to the establishment and spread of invasive species. Regardless, given their detrimental effect on biodiversity, invasive plant species abundance is a vital indicator for wetland condition assessments.

It is a common sight in Minnesota to see depressional wetlands in the southern and western parts of the state that are predominantly narrowleaf and hybrid cattail, two invasive wetland plants.

Across the two ecoregions, the percent cover of non-native invasive plant species at the emergent/aquatic interface is high (>50%) in 45% of depressional wetlands and ponds (Figure 10). This estimate increases to 59% in the TP ecoregion where non-native invasive plant species abundance is significantly greater compared to the MWP ecoregion ($F = 3.84$, $df_1 = 2$, $df_2 = 94$, $p < 0.05$). With only 14% of depressional wetlands in the TP ecoregion having low (<20%) abundance, non-native invasive plant species are a serious threat to the ecological integrity of wetlands in this ecoregion. Mixed and monospecific stands of Invasive Cattail (*Typha X glauca* Godr. and *T. angustifolia*) were the most widespread invasive plants inhabiting the emergent zone of depressional wetlands and ponds, accounting for greater than 50% cover of the sample plots in 37 out of 99 study sites. No other non-native invasive plant species was observed at >50% cover (cover classes 6-8) by itself. This can be attributed to the definition of the target population including a fringe of marsh community and the placement of sampling plots, which typically straddled the emergent marsh and submerged aquatic vegetation zones of each wetland. The optimal habitat for Invasive Cattail is found in marsh habitats—where soils are saturated, or frequently inundated by water up to 6 inches (Eggers and Reed 2011). Given that, it was not surprising that Invasive Cattail was the most frequent non-native invasive group occurring at high abundance in the DWQA. The optimal habitat for Purple Loosestrife is also marsh (Eggers and Reed 2011); however, while Purple Loosestrife is certainly a threat (Galatowitsch et al. 1999), not observing it at high abundance suggests that it is not currently having much of an impact on depressional wetland condition in the MWP and TP ecoregions. Reed canary grass, on the other hand, tends to proliferate in drier fresh meadow wetland habitats (Eggers and Reed 2011)—which were not part of the target wetland definition and typically excluded from vegetation sampling in the DWQA. Thus, not observing high abundance of Reed canary grass was likely a product of the target population definition and sampling protocol as opposed to Reed canary grass not being prevalent in the broader wetland population of the MWP and TP ecoregions.

An assessment of the relative risk of each stressor measured in this survey revealed that the two most pervasive stressors, invasive plant species and elevated chloride concentrations, posed an elevated risk to wetland plant communities but not to aquatic macroinvertebrates (Figure 11). Compared to 2007-2009 statewide results, which included the Mixed Wood Shield ecoregion, invasive plant species went from third to first in ranking the stressors by relative extent (i.e., % High) but exhibited the same pattern of no risk to macroinvertebrates and an elevated risk to plants. In the baseline survey, chloride was the top stressor ranked by extent and represented an elevated risk to both plants and macroinvertebrates. Between the two surveys, total phosphorus was the most consistent stressor based on its impacts to the biological communities, representing a significant elevated risk in all analyses with the exception of macroinvertebrates in the TP ecoregion. Sulfate also posed an elevated risk to both plants and macroinvertebrates, and fell in the middle when stressors were ranked by extent (Figure 11). As mentioned previously (see Box 1), this water quality indicator was excluded from the initial baseline survey due to laboratory analysis issues. Results from the current survey suggest that sulfate is an important parameter to monitor in the assessment of depressional wetland quality. In general, the relative risks posed by the measured water quality contaminants to macroinvertebrates were large, but variable compared to plants.

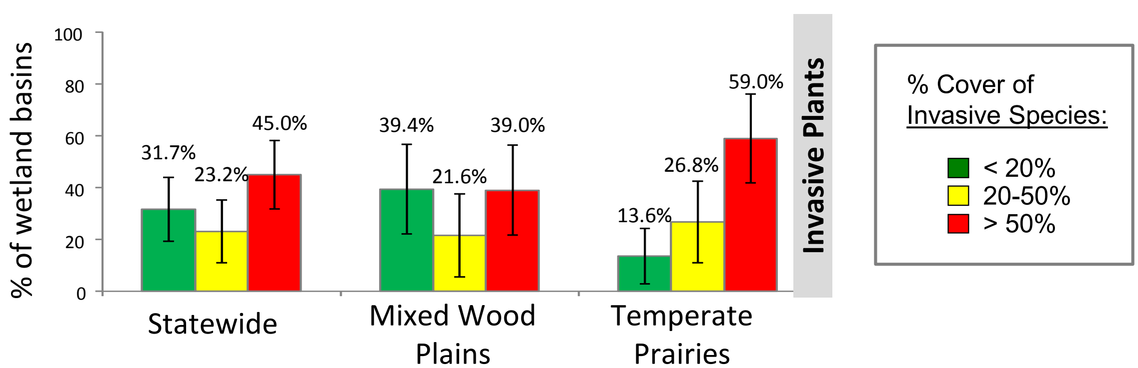


Figure 10. Abundance of invasive plant species in the emergent zone of Minnesota depressional wetlands and ponds. Bracketed lines represent the width of the 95% confidence interval associated with each estimate. Percent cover of invasive species was significantly higher in the TP ecoregion according to the CDF test.

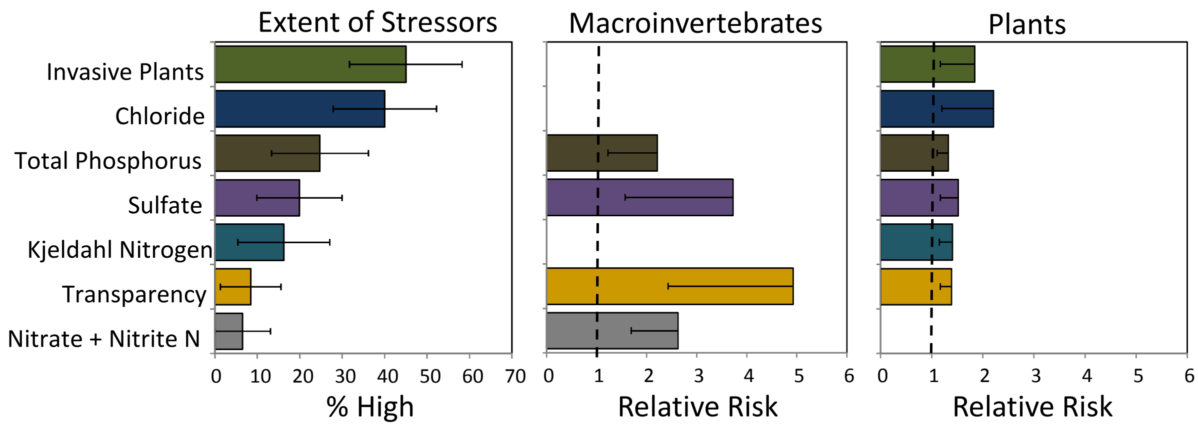


Figure 11. Extent of stressors and their relative risk to plant and macroinvertebrate communities in Minnesota depressional wetlands and ponds. Bracketed lines represent 95% confidence intervals (for percentage estimates) or lower confidence limits (for relative risk estimates). A stressor without an associated bar on the relative risk graphs indicates that it did not pose an elevated risk to that community.

Natural vs. man-made

Plant and macroinvertebrate communities are in poor condition more often in basins of man-made origin than they are in natural wetlands (Figure 12). This same pattern was observed in the 2007-2009 baseline survey macroinvertebrate results, but not in the plant community results where the poor category percentage was similar between man-made and natural. The elevated plant IBI scores in the baseline survey were likely due to a sampling artifact—where (due to the narrow marsh habitat commonly present in man-made depressional wetlands) the sampling plots were more likely to cross into multiple wetland plant community types, artificially boosting IBI scores.

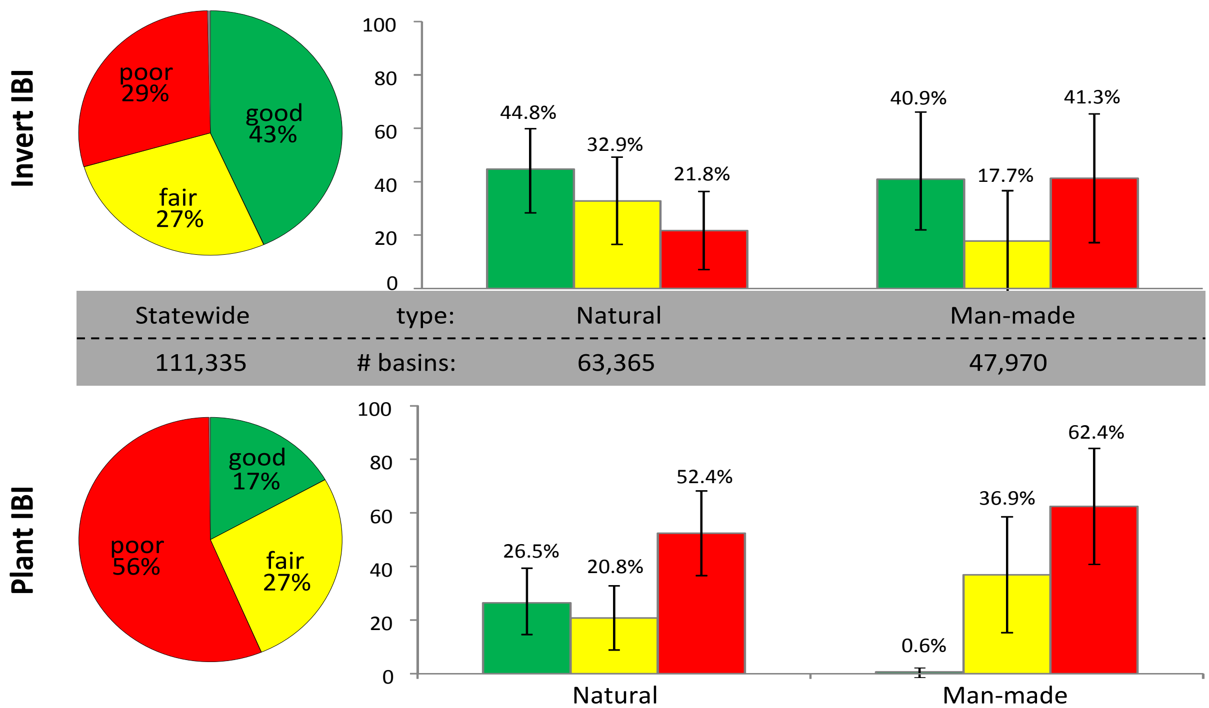


Figure 12. Comparison of the biological condition of natural vs. man-made depressional wetlands. Less than 1% of the population was 'not assessed' by each IBI; this information is not included on graphs.

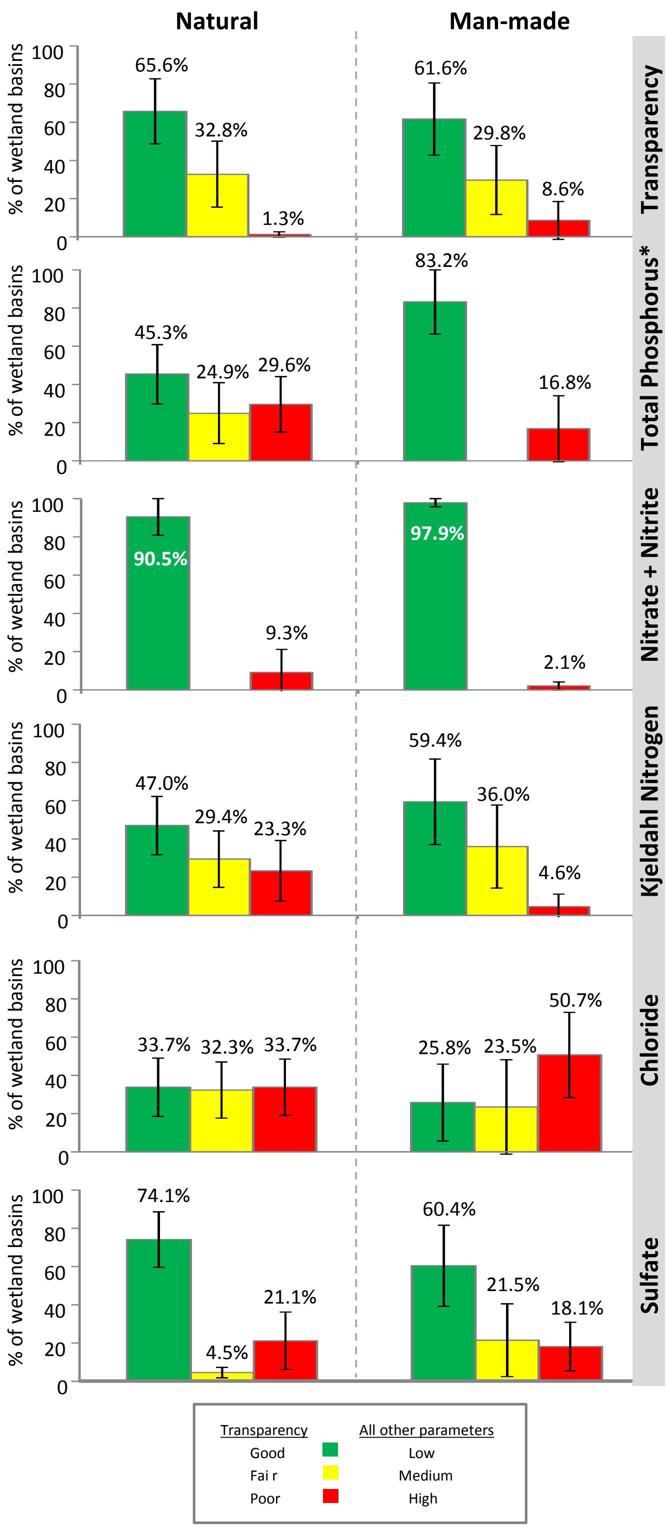


Figure 13. Comparison of stressor levels between natural and man-made wetlands and ponds. Bracketed lines represent the width of the 95% confidence interval associated with each estimate. Asterisks indicate significant differences in stressor levels between basin types according to CDF test.

Sample plot locations were adjusted accordingly in 2012 to account for these narrow fringes, minimizing the inclusion of non-target plant communities. Compared to natural wetlands, man-made basins may be demonstrating degraded biological conditions due to the presumably limited habitat conditions associated with their construction and/or continued physical alteration (e.g., dredging, stormwater inundation, artificial shoreline substrate, animal trampling).

Examination of the measured water quality parameters does not reveal any obvious causes for the increased percentage of poor biological communities in man-made wetlands and ponds. CDF tests comparing the data distributions of man-made and natural wetlands did not reveal statistically significant differences for transparency, nitrate + nitrite, Kjeldahl nitrogen, chloride, and sulfate concentrations (Figure 13). Only total phosphorus concentrations differed between the two wetland types ($F = 3.57$, $df_1 = 2$, $df_2 = 94$, $p < 0.05$), but concentrations were higher in natural basins. However, the abundance of invasive plant species was significantly higher in the emergent zone of man-made wetlands (Figure 14; $F = 11.38$, $df_1 = 2$, $df_2 = 94$, $p < 0.001$) which likely contributed to their degraded plant community.

Wetland area categories

The condition of macroinvertebrate communities is relatively degraded in small (<2.5 acre) wetlands compared to the two larger wetland area categories (Figure 15). Wetland plant communities are in slightly better condition in the medium (2.5 – 12.4 acre) wetland size category. Both communities exhibited similar patterns amongst the wetland area categories in the 2007-2009 baseline survey. As in the baseline survey, the relatively poor condition of small wetlands may be largely driven by so many being man-made waterbodies. Approximately half of the survey sites in the small wetland area class were man-made, whereas 19% and 0% were of man-made origin in the medium and large size classes, respectively. The frequency of man-made sites amongst the area classes does not entirely explain the plant community condition results considering large (>12.4 acre) wetland condition was similar to that of the small wetland area class. Of the measured stressors, only the predominance of invasive plant species offers any evidence to explain these observed results. The large wetland class had significantly greater abundance of invasive plant species than either of the other two smaller area classes (Figure 16; $p \approx 0.001$ for both CDF tests). Large depressional wetlands tend to be more hydrologically connected to the watershed via drainage features (e.g., open ditches and drain tile) as well as natural streams. Altered hydrology in these larger basins may result in larger pollutant loads and greater water level fluctuations which could partially account for the higher abundance of non-native invasive plant species.

Total phosphorus concentrations were significantly higher in the small wetland class compared to the large class ($F = 4.18$, $df_1 = 2$, $df_2 = 69$, $p < 0.05$). This result likely reflects the natural tendency of permanent, deeper waterbodies to have lower water column concentrations of phosphorus as it is a pattern observed in least-disturbed wetlands as well (*unpublished data*). In deeper waterbodies (including ponds), an aerobic surface water column separated from the anaerobic interstitial water in the sediment may lead to phosphorus accumulation in the bottom sediments (Neely and Baker 1989). In contrast, shallower wetlands with dense emergent vegetation often have anaerobic surface water which typically enhances phosphorus release from the substrate to the water column. Coupled with the findings of the previous section (i.e., higher P conc. in natural basins), the difference in P concentrations observed among the wetland area classes suggests that P concentrations are greatest in small—typically shallow—natural basins. Shallower waterbodies do not have as great a distinction between the aerobic water column and anaerobic sediment layer (Neely and Baker 1989) and thus are not able to retain a disproportionate amount of their P load in the substrate. Phosphorus water column concentrations are further increased in basins that periodically go dry (LeBaugh et al. 1987), an event that presumably occurs more often in shallow basins.

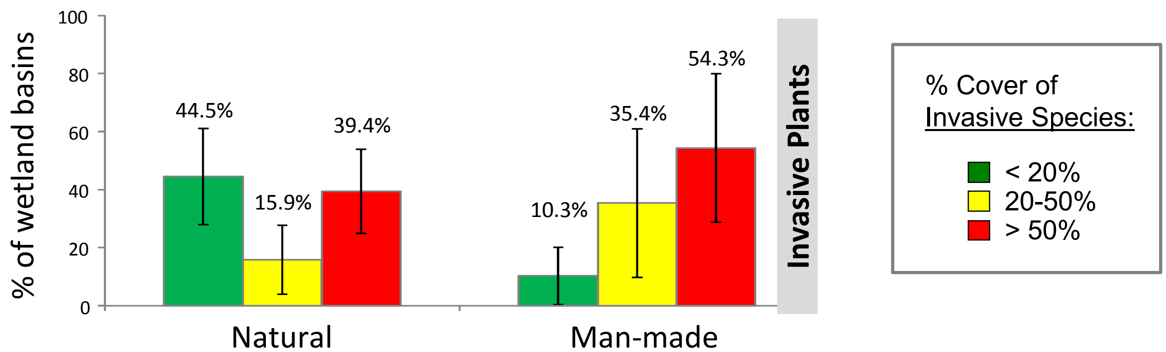


Figure 14. Comparison of invasive plant species abundance between natural and man-made wetlands and ponds. Bracketed lines represent the width of the 95% confidence interval associated with each estimate. Percent cover of invasive species was significantly higher in man-made wetlands and ponds according to the CDF test.

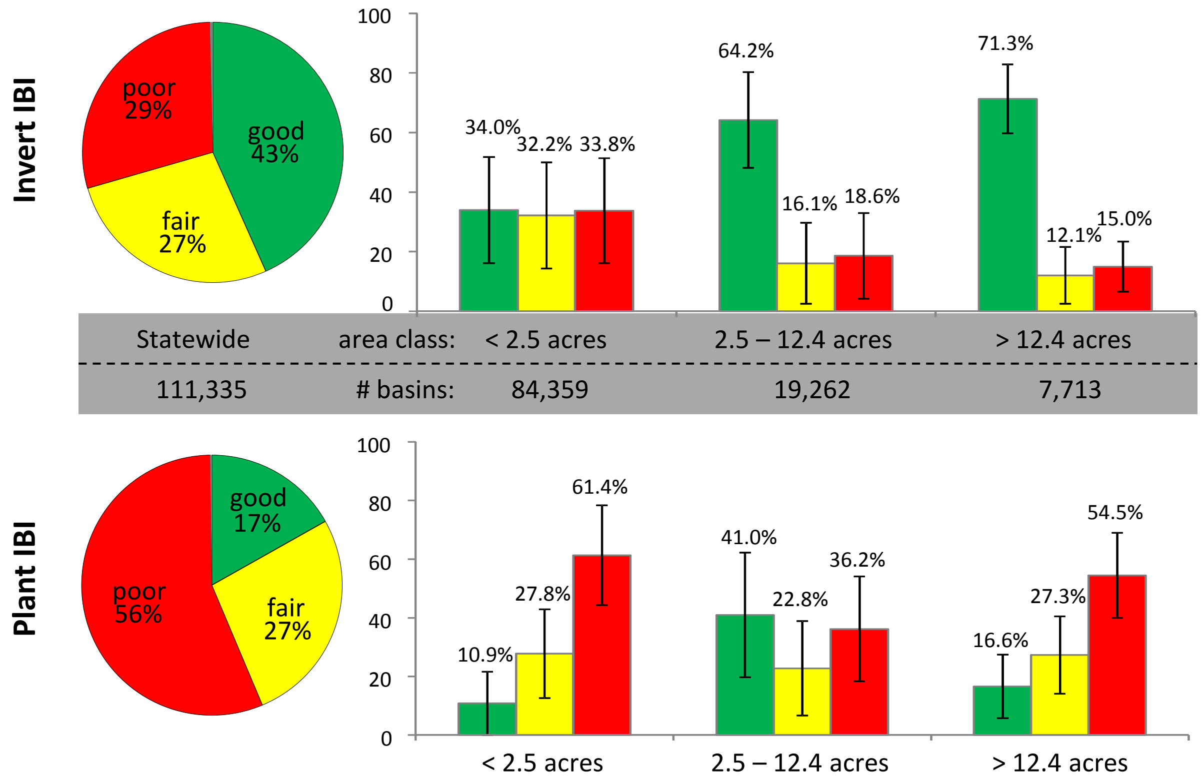


Figure 15. Comparison of the biological condition of wetland area classes used in the survey design. Less than 1% of the population was 'not assessed' by each IBI; this information is not included on graphs.

Wetland ownership

Macroinvertebrate community condition was similar amongst the three property ownership categories and did not vary significantly from the statewide condition rates (Figure 17). Likewise, vegetation condition was also similar between private and public ownership categories and did not vary significantly from statewide condition estimates. Wetlands that occurred on both public and private property (i.e., Mixed), however, seemed to have relatively degraded plant communities compared to wetlands that occurred exclusively on public or private property. This is largely due to all mixed ownership wetlands having >50% invasive plant species cover estimates, significantly greater than either privately ($F = 13.68$, $df_1 = 2$, $df_2 = 75$, $p < 0.001$) or publicly ($F = 7.24$, $df_1 = 2$, $df_2 = 21$, $p < 0.05$) owned wetlands (Figure 18). However, it should be noted that the sample size for the mixed category was particularly low at n = 5 and the observed results are probably more a reflection of wetland area than of property ownership; i.e., four of the five sites were large (>12.4 acres). As stated earlier, hydrologic modification of the landscape has likely increased the pollutant load and water level fluctuations in large wetland basins, conditions that tend to favor non-native invasive plant species.

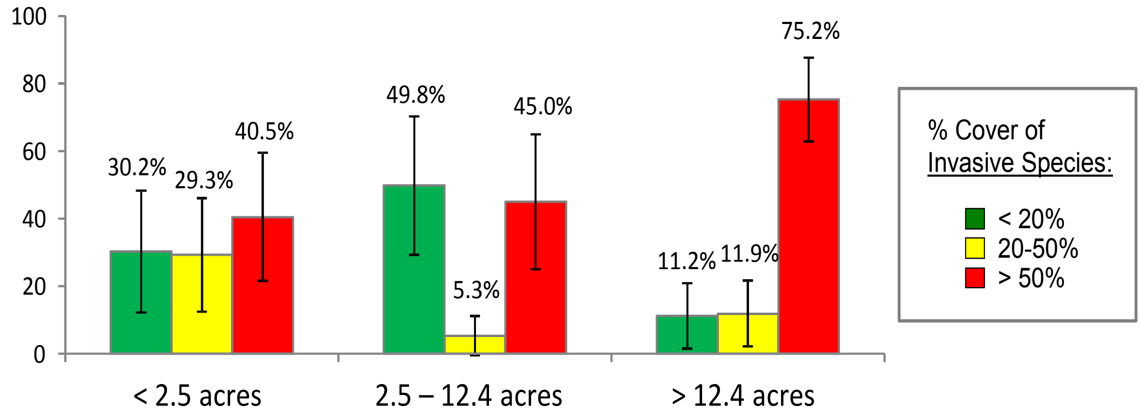


Figure 16. Comparison of abundance estimates of invasive plant species in the emergent zone amongst the wetland area classes used in the survey design. Approximately 2% of the population was 'not assessed' in the >12.4 acre class; this information is not included on the graph. Percent cover of invasive species was significantly higher in the large area class of wetlands and ponds according to the CDF test.

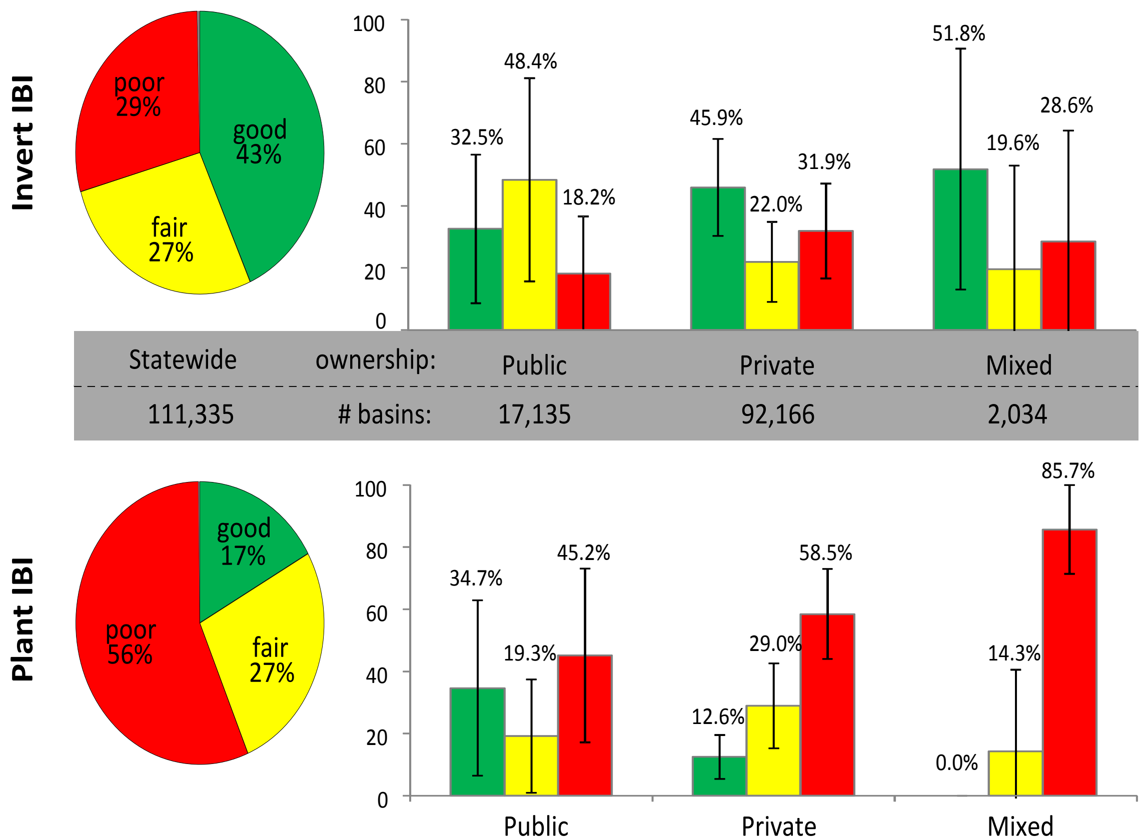


Figure 17. Comparison of the biological condition of wetland ownership categories. Less than 1% of the population was 'not assessed' by each IBI; this information is not included on graphs.

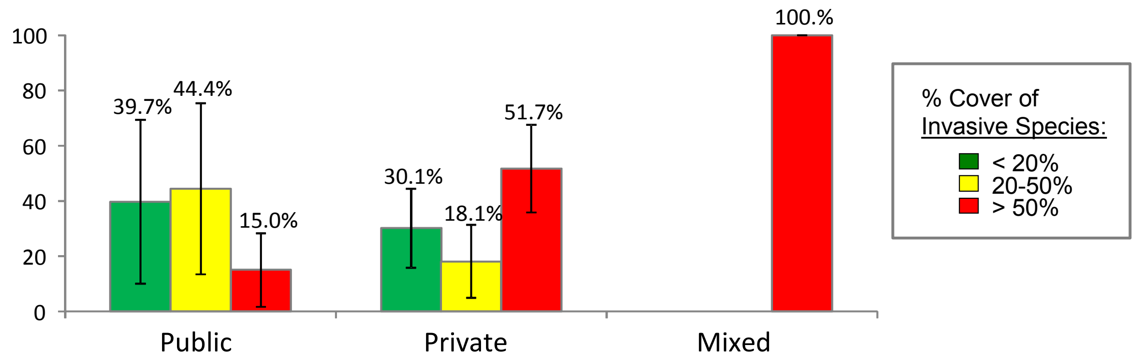


Figure 18. Comparison of abundance estimates of invasive plant species in the emergent zone amongst the wetland ownership categories. Less than 1% of the population was 'not assessed' in the public category; this information is not included on the graph. Percent cover of invasive species was significantly higher in the mixed ownership class compared to publicly and privately owned wetlands and ponds.

Changes in depressional wetland condition

A change detection analysis was performed to determine whether any significant differences in the condition and stressor estimates exist between the two survey periods—2007-2008 and 2012. Several more iterations of the survey will be required before trends can be evaluated, a primary goal of this status and trends survey. The results presented below represent change or lack thereof for the population of depressional wetlands and ponds in the state of Minnesota, not individual site changes.

Depressional wetland plant and macroinvertebrate community condition did not change significantly between the two survey periods (Figure 19). None of the stressor indicators exhibited a significant change between the two periods comparing statewide data sets (Table 2). Examining the results of the subpopulation analyses shows that overall conditions have not changed significantly between the two time periods; however, it is premature to place too much confidence in these results. An analysis of trends after several more cycles of the survey will provide a more robust evaluation of whether depressional wetland condition is improving or degrading in Minnesota. Focusing on the differences that were observed between the two time periods, man-made wetland plant community condition decreased (apparently due to correcting a sampling procedural error) while nutrient concentrations decreased (Table 2). The condition of large (>12.4 acre) wetlands improved according to the macroinvertebrate IBI and total phosphorus concentrations but non-native invasive plant species increased in abundance in this wetland size class.

Table 2. Change detection analysis results for the comparison of 2012 indicator data to 2007-2008 data. nc = no significant change detected. Category abbreviations: G = good, P = poor, L = low, H = high. Green text indicates condition improvement, red indicates degrading conditions.

		Wetland origin		Wetland area			Ownership	
Indicators	**Statewide**	**Natural**	**Man-made**	**<2.5**	**2.5-12.4**	**>12.4**	**Public**	**Private**
Plant IBI	nc	nc	-21% G	nc	nc	nc	nc	nc
Invert IBI	nc	nc	nc	nc	nc	+21% G	nc	nc
Transparency	nc	-21% G	nc	nc	nc	nc	nc	nc
Total Phosphorus	nc	nc	+45% L	nc	nc	+23% L	nc	nc
Nitrate + Nitrite	nc	nc	-27% H	nc	-8% H	nc	nc	nc
Kjeldahl Nitrogen	nc	nc	nc	nc	nc	nc	nc	nc
Chloride	nc	nc	nc	nc	-23% H	nc	nc	nc
Sulfate	nc	nc	nc	nc	nc	nc	nc	nc
Invasive Species	nc	nc	-59% L	nc	nc	+21% H	-44% H	nc

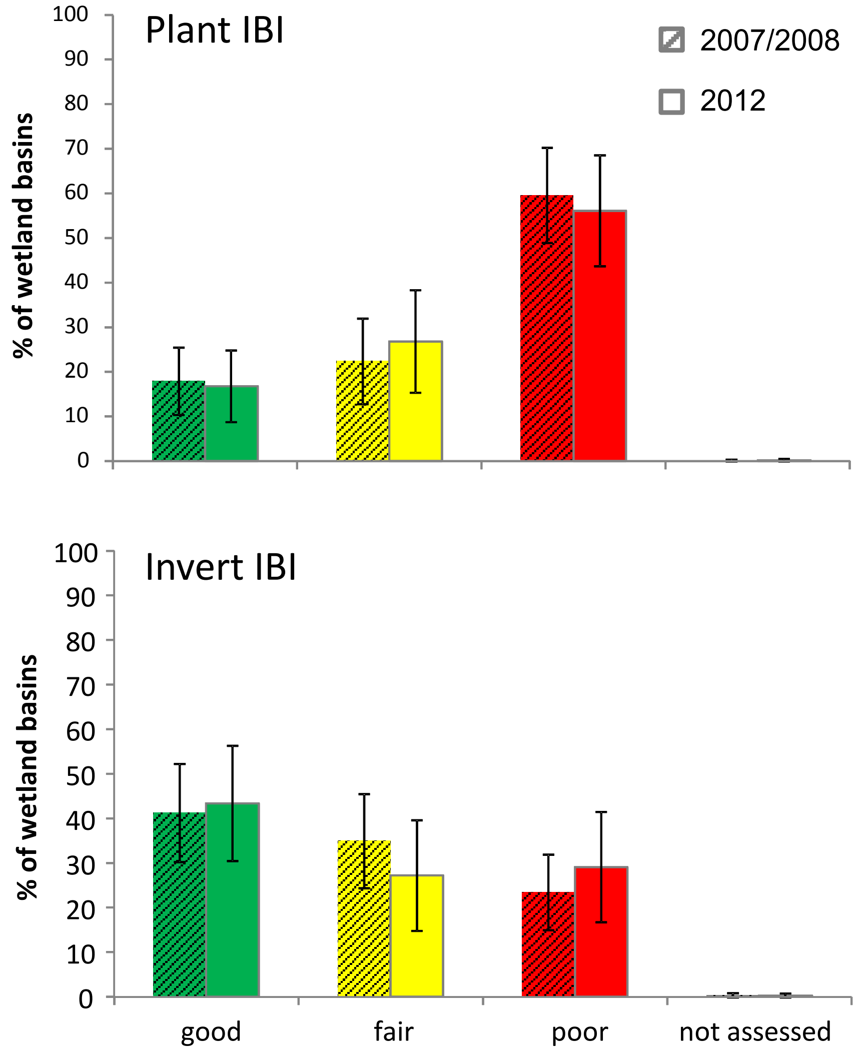


Figure 19. Comparison of baseline DWQA biological condition results to biological condition in 2012. Bracketed lines represent the width of the 95% confidence interval associated with each estimate.

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 that was 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, and Alexandria. 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 79,247 depressional wetlands and ponds occur within the MWP ecoregion according to 2012 survey results. This estimate represents a decrease in the total number of basins compared to the 2007 estimate of 87,479 for the ecoregion (Figure 20); however, this difference is not statistically significant at the 95% confidence level. Similar to the statewide results, the proportion of man-made depressional wetlands in this ecoregion appears to be on the rise (Figure 20).

Of the subpopulations examined in this ecoregion, only the large (>12.4 acre) wetland size category showed a statistically significant change in quantity between the two time periods ($Z = -2.06$, $p = 0.039$). There were an estimated 6,696 large wetland basins in the MWP ecoregion in 2007, dropping to 5,454 in 2012 (Figure 20). This result may be partially driven by inconsistences between the two surveys in the distinction between shallow lakes and large depressional wetlands. Distinguishing between these water body types is difficult and the MPCA's approach for doing so has been evolving over the past several years. Thus, it is possible that more shallow lakes were included in the 2007 survey, erring on the side of caution because depth profiles and other pertinent information often wasn't available to assist with these determinations. Benefitting from a few more years of experience making such determinations, the 2012 survey may have tended to treat more shallow lakes as 'non-target' during the site evaluation phase, which would decrease the population estimates for this size category.

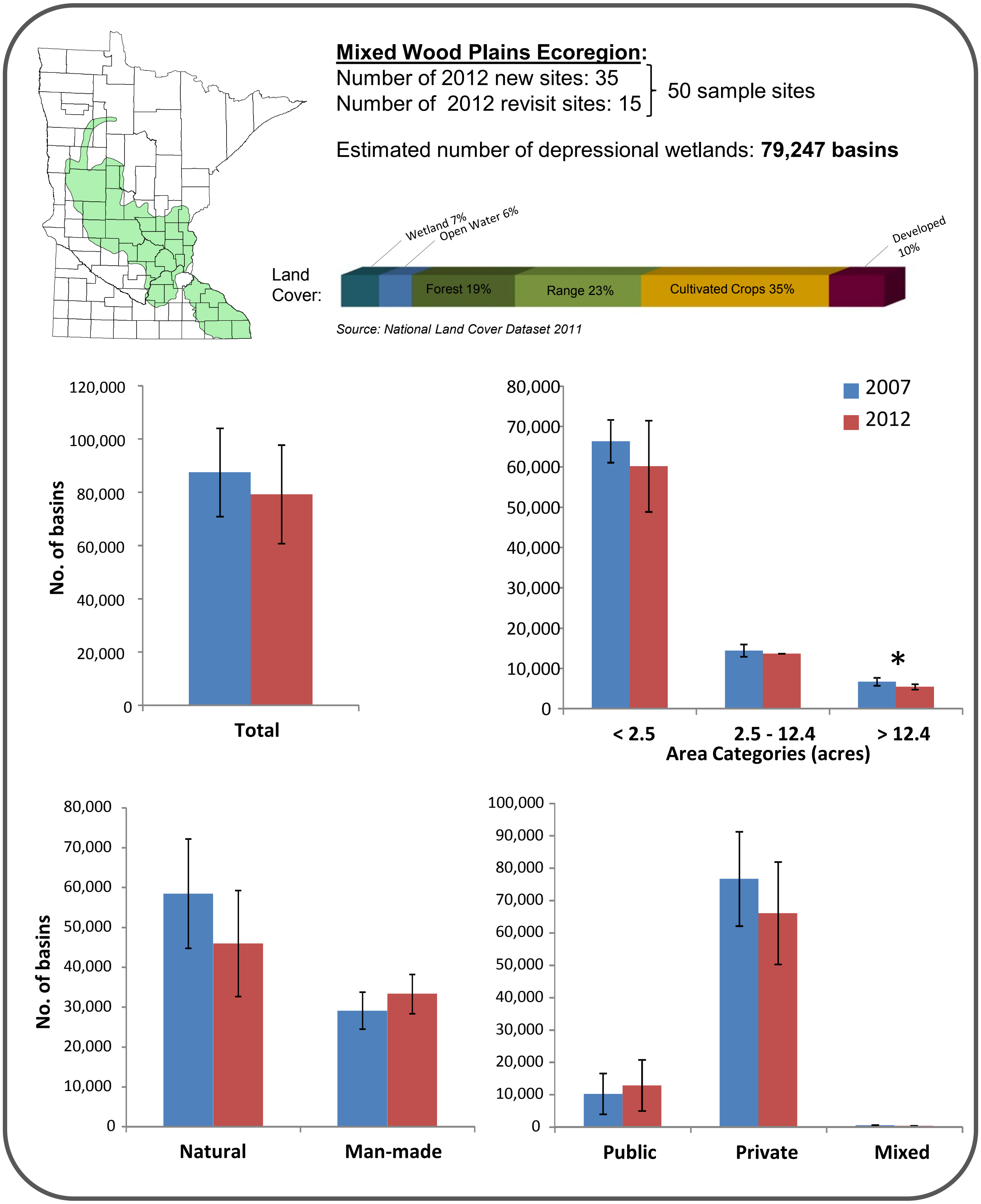


Figure 20. Estimates of the total number of depressional wetlands in the Mixed Wood Plains ecoregion comparing time 1 of the survey (2007) to time 2 (2012). Bracketed lines represent the 95% confidence interval associated with each estimate. Asterisk indicates a statistically significant change between time periods.

Overall, the majority of depressional wetlands and ponds in the MWP ecoregion are small (<2.5 acres) and are located on private property. Most are naturally formed wetlands; however the gap between natural and man-made wetlands and ponds may be decreasing (Figure 20).

Depressional wetland condition

- Similar to the statewide results, plant communities are in good condition in 19% of MWP depressional wetlands and ponds while 54% are in poor condition (Figure 21). Plant community condition did not vary significantly between natural and man-made wetland types or among the wetland area and ownership categories according to CDF tests.
- Macroinvertebrate IBI results indicate that an estimated 43% of MWP depressional wetlands and ponds are in good condition while 28% are in poor condition (Figure 21). Aquatic macroinvertebrate community condition did not vary significantly between natural and man-made basins or by wetland ownership categories. Community condition did vary significantly among the wetland area classes. CDF tests indicated that both large and medium wetland classes were in significantly better condition than small wetlands ($p = 0.002$ for both tests) (Figure 22).
- An estimated 65% of MWP depressional wetlands and ponds have good water clarity while only 8% exhibit turbid or poor conditions (Figure 21). CDF tests comparing transparency results among the various subpopulations could not be performed due to insufficient variability in transparency values (i.e., a large number of >100 cm readings).
- Total phosphorus concentrations are low in 55% of MWP depressional wetlands and ponds while 23% are high (Figure 21). Concentrations did not vary significantly among wetland ownership categories or between natural and man-made wetland types. The only statistically significant CDF test was the comparison of small and large wetland area classes; large wetlands exhibited lower total phosphorus concentrations than the small wetland class ($F = 5.60$, $df_1 = 2$, $df_2 = 32$, $p = 0.008$). As mentioned previously, this result is likely a reflection of the intrinsically lower phosphorus concentrations that are characteristic of large, permanent (and typically deeper) water bodies.
- Nitrate + nitrite concentrations are below detection (= Low) in 94% of MWP depressional wetlands and ponds (Figure 21). CDF tests comparing the results among the various subpopulations could not be performed due to insufficient variability in the values (i.e., majority below detection).
- Approximately 47% of MWP depressional wetlands and ponds have low concentrations of Kjeldahl nitrogen while 19% exhibit high concentrations (Figure 21). Kjeldahl-N concentrations were significantly lower in the large wetland class compared to small wetlands ($F = 4.28$, $df_1 = 2$, $df_2 = 32$, $p = 0.023$) and higher in privately owned wetlands compared to those on public property ($F = 7.13$, $df_1 = 2$, $df_2 = 44$, $p = 0.002$) according to the CDF test. Concentrations did not vary significantly between natural and man-made wetland types.
- Chloride concentrations are low in 27% of MWP depressional wetlands and ponds while 32% are high (Figure 21). Concentrations did not vary significantly among wetland ownership and area categories. Man-made wetlands had significantly higher chloride concentrations than did natural wetlands ($F = 7.63$, $df_1 = 2$, $df_2 = 45$, $p = 0.001$).
- An estimated 78% of MWP depressional wetlands and ponds have low sulfate concentrations while 13% have high concentrations (Figure 21). Concentrations did not vary significantly among wetland ownership and area categories. Man-made wetlands have significantly lower sulfate concentrations than natural wetlands ($F = 3.53$, $df_1 = 2$, $df_2 = 45$, $p = 0.038$).
- The abundance of non-native invasive plant species is low (<20% cover) in 39% of MWP depressional wetlands and ponds and high (>50% cover) in 39% (Figure 21). Invasive plant species are significantly more abundant in man-made wetlands and ponds compared to natural wetlands ($F = 8.37$, $df_1 = 2$, $df_2 = 45$, $p < 0.001$). Medium-sized (2.5–12.4 acre) wetlands have significantly lower invasive species abundance than small and large wetland area classes ($p = 0.046$ and $p < 0.001$, respectively) (Figure 23). In addition, wetlands on public property have significantly less invasive species plant cover than privately owned wetlands ($F = 7.28$, $df_1 = 2$, $df_2 = 44$, $p = 0.002$).

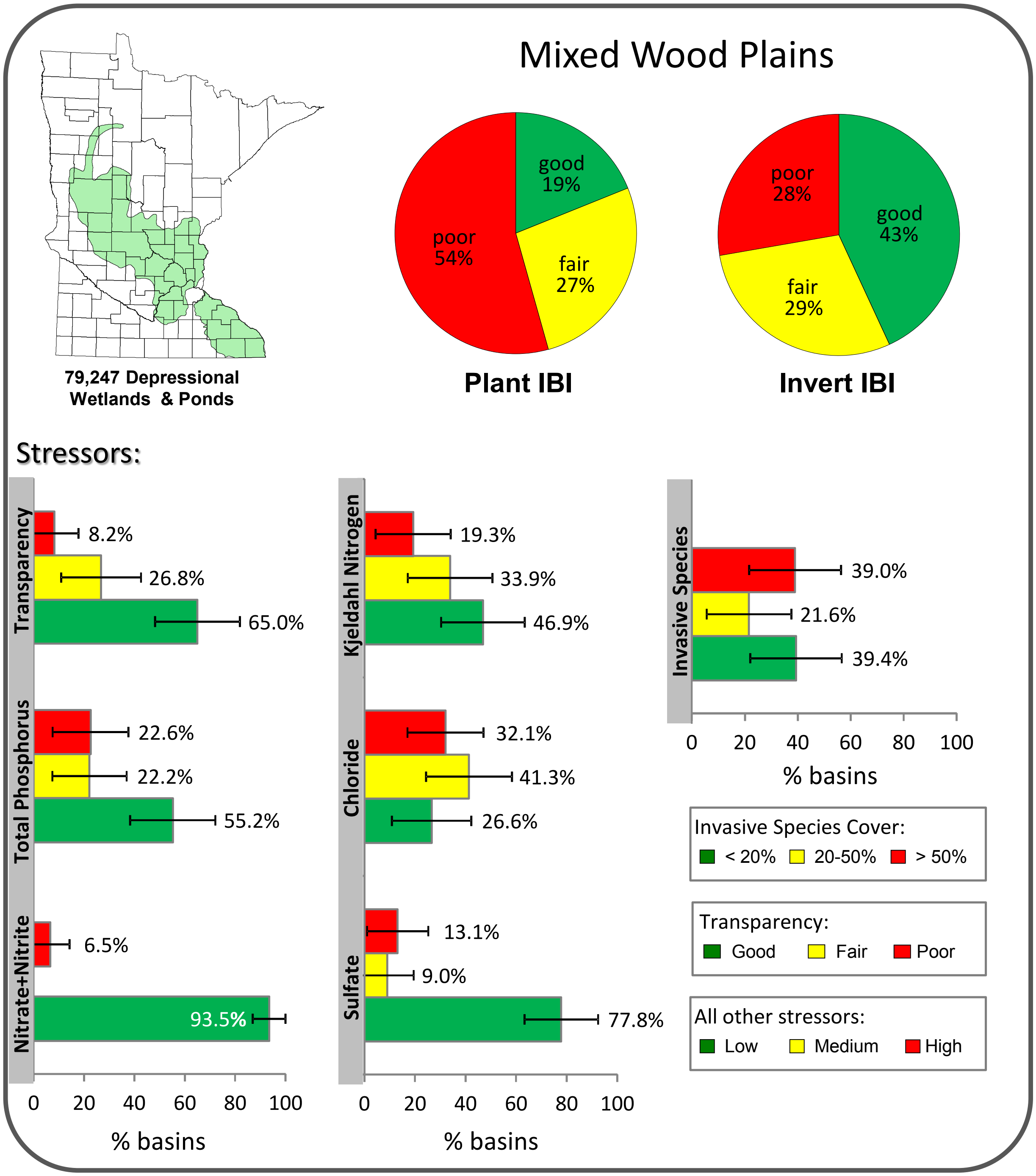


Figure 21. Biological condition and stressor level estimates for Mixed Wood Plain depressional wetlands and ponds. Bracketed lines represent the width of the 95% confidence interval associated with each estimate.

The relative ranking of the measured stressors by the extent of high levels (i.e., percent high) in the MWP was very similar to the ranking observed at the statewide level (Figure 24). The only difference was that sulfate and Kjeldahl nitrogen traded positions in this ranking (note: Figure 24 depicts the list of stressors in the ranked order observed at the statewide scale). The relative risk to macroinvertebrates within MWP depressional wetlands and ponds was identical to the statewide results; total phosphorus, sulfate, transparency, and nitrate + nitrite all posed elevated risks to this community. Compared to the statewide results, chloride dropped out as a significant risk and nitrate + nitrite was added as a significant risk to wetland plant communities in the MWP ecoregion.

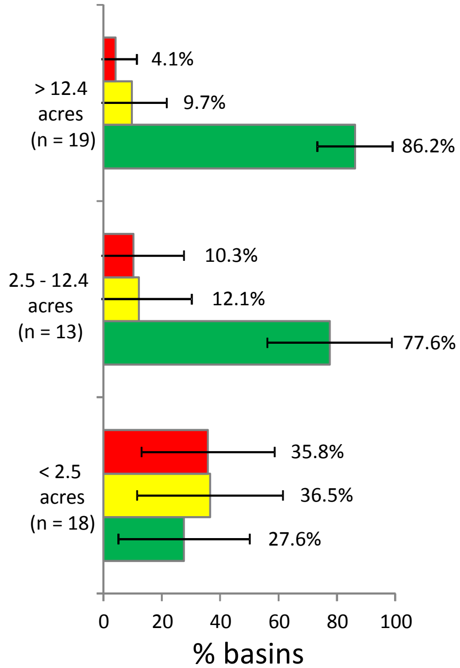


Figure 22. Macroinvertebrate community condition among the wetland area classes.

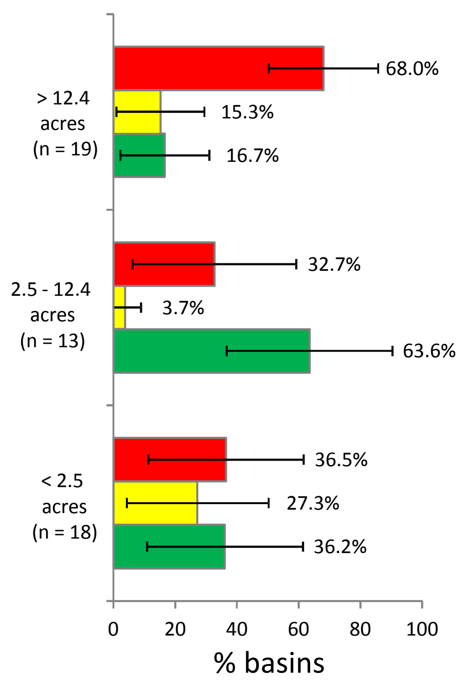


Figure 23. Invasive plant species abundance among the wetland area classes.

High nutrient levels represented an increased risk to both wetland plant and macroinvertebrate communities in this ecoregion (Figure 24). This was the pattern observed in the baseline survey as well. Elevated nutrients can affect biological condition through a variety of different pathways. High nutrient concentrations can lead to increased algal and macrophyte (e.g., duckweed) growth and widely fluctuating dissolved oxygen concentrations due to the daily photosynthesis/respiration cycle of living plants as well as decomposition (i.e., microbial respiration) of dead vegetation. This can result in extremely low dissolved oxygen concentrations (<1 mg/L), detrimental to less tolerant wetland invertebrates. A competitive edge may be given to invasive species such as Invasive cattail that can more effectively utilize excess nutrients, leading to their dominance of the plant community (Woo and Zedler 2002). Another example of nutrient impacts is a shift in the trophic status of open water wetlands, from macrophyte-dominated, clear-water conditions to phytoplankton-dominated, turbid conditions. This shift has obvious impacts to the open water plant community, decreasing the abundance and diversity of submerged aquatic vegetation, and reduces habitat complexity for aquatic macroinvertebrates.

Turbid conditions were not prevalent in the MWP ecoregion, occurring in less than 10% of depressional wetlands and ponds. However, when water transparency was low or turbid it posed an elevated risk to both plants and macroinvertebrates (Figure 24). Turbid conditions may occur due to increased sediment loading from upland erosion or increased phytoplankton abundance in response to excess nutrient loading. Unfortunately the DWQA did not collect chlorophyll-a, total suspended solids, and total volatile suspended solids data—information that is necessary to distinguish between these two types of turbidity. Given the mix of urban and agricultural land use within this ecoregion, both scenarios are likely accounting for the turbid conditions observed in some MWP depressional wetlands and ponds. In addition to the trophic shift mentioned in the paragraph above, increased turbidity can decrease plant growth and seedling emergence through decreased light penetration and increased sedimentation (Jurik et al. 1994, Wang et al. 1994, Mahaney et al. 2004). At the same time, increased sedimentation may promote the germination and emergence of more tolerant plant species, leading to an overall decrease in plant community composition (Mahaney et al. 2004). Decreased transparency can also reduce the feeding efficiency of macroinvertebrate filter-feeders (i.e., if primarily due to suspended sediment) and predators, and may ultimately lead to decreased abundance (Martin and Neely 2001) and diversity within these groups if turbid conditions persist.

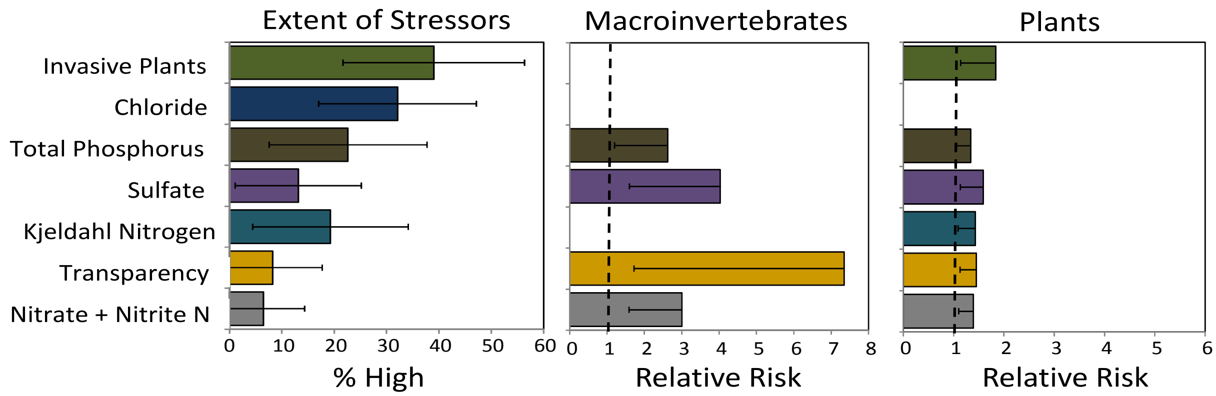


Figure 24. Extent of stressors and their relative risk to plant and macroinvertebrate communities in Mixed Wood Plains depressional wetlands and ponds. Bracketed lines represent 95% confidence intervals (for percent estimates) or lower confidence limits (for relative risk estimates). A stressor without 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

In the MWP ecoregion, depressional wetland plant and macroinvertebrate community condition did not change significantly between 2007 and 2012 (Figure 25). None of the stressor indicators exhibited a significant change between to the two periods when comparing the entire MWP data sets (Table 3). Results were varied when comparisons were made of the subpopulations between 2007 and 2012. For example, large (>12.4 ac) depressional wetlands and ponds showed improving conditions according to the macroinvertebrate IBI, total phosphorus, and chloride, yet also demonstrated an increase in invasive plant species abundance (Table 3). This discrepancy may be a reflection of improvements in water quality not being immediately manifested in the plant community due to persistence of changes when non-native invasive plant species dominate composition. Publically-owned wetlands and ponds exhibited an increase in plant community condition, a decrease in invasive plant species as well as an increase in sulfate concentrations. Invasive plant species abundance exhibited the most change among the various comparisons. Natural and publically-owned wetlands showed decreases in invasive plant species abundance while man-made and large wetlands showed increases in abundance (Table 3). As mentioned previously, it is premature to place too much emphasis on these initial findings of this long-term monitoring survey. An analysis of trends will provide a much better indication of whether the condition of depressional wetlands and ponds is improving or degrading.

Table 3. Change detection analysis results for the Mixed Wood Plains ecoregion, comparison of 2012 indicator data to 2007 data. nc = no significant change detected. Category abbreviations: G = good, P = poor, L = low, H = high. Green text indicates condition improvement, red indicates degrading conditions.

	MWP	Wetland origin		Wetland area			Ownership	
Indicators	**Ecoregion**	**Natural**	**Man-made**	**<2.5**	**2.5-12.4**	**>12.4**	**Public**	**Private**
Plant IBI	nc	nc	nc	nc	nc	nc	+40% G	nc
Invert IBI	nc	nc	nc	nc	nc	+26% G	nc	nc
Transparency	nc	-29% G	nc	nc	nc	nc	nc	nc
Total Phosphorus	nc	nc	+54% L	nc	nc	+36% L	nc	nc
Nitrate + Nitrite	nc	nc	nc	nc	nc	nc	nc	nc
Kjeldahl Nitrogen	nc	nc	nc	nc	nc	nc	nc	nc
Chloride	nc	nc	nc	nc	-31% H	-24% H	nc	nc
Sulfate	nc	nc	nc	nc	nc	nc	-37% L	nc
Invasive Species	nc	+26% L	-66% L	nc	nc	+28% H	+45% L	nc

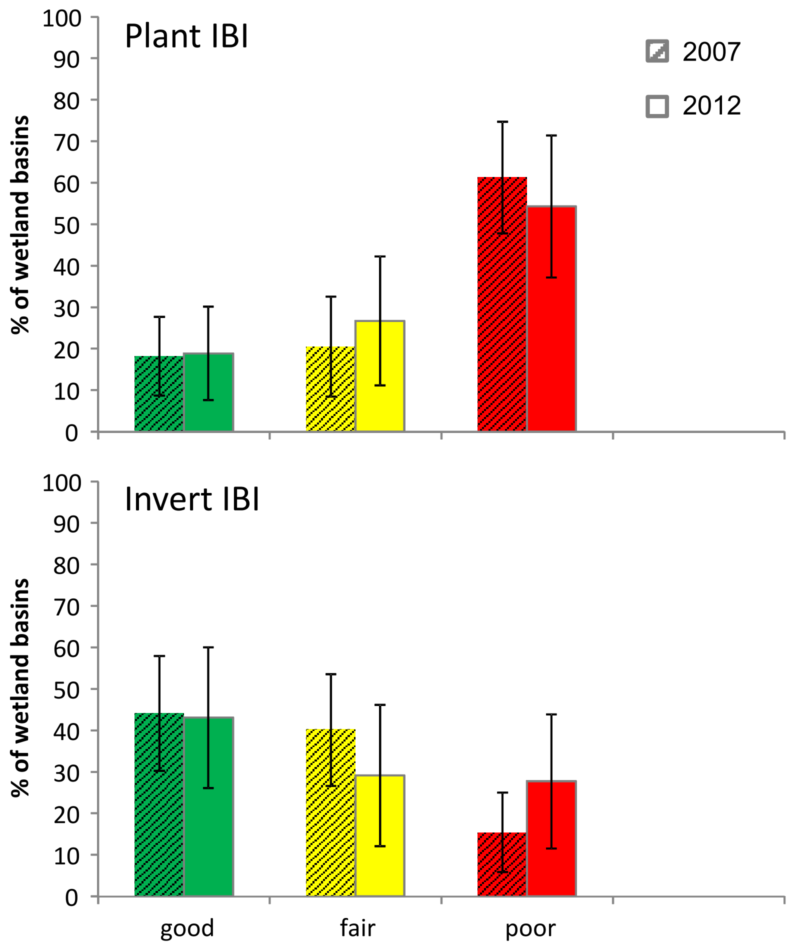


Figure 25. Comparison of baseline Mixed Wood Plain wetland biological condition results to biological condition in 2012. Bracketed lines represent the width of the 95% confidence interval associated with each estimate.

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 (PPR), 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 2012 survey estimates a total of 32,088 depressional wetlands and ponds in the Temperate Prairies ecoregion. This estimate is remarkably similar to the 2008 estimate of 32,300 basins in the ecoregion (Figure 26). Thus, early indications suggest that no-net-loss in the quantity of this particular wetland type is being achieved in the TP ecoregion. Unlike the MWP ecoregion, the proportion (and number) of man-made depressional wetlands in the TP ecoregion did not change between the two surveys (Figure 26).

Once again, large wetlands (>12.4 acres) exhibited a statistically significant decrease in quantity between the two surveys ($Z = -4.97$, $p < 0.001$). There were an estimated 3,670 large wetland basins in the TP ecoregion in 2008, decreasing to 2,259 in 2012 (Figure 26). As mentioned in the MWP depressional wetland quantity section, this decrease may be somewhat of an artifact of how large depressional wetlands and shallow lakes were distinguished during the site evaluation phases of the two surveys.

Overall, the majority of depressional wetlands and ponds in the TP ecoregion are small (<2.5 acres) and are located on private property. Estimates indicate that naturally-formed wetlands outnumber the man-made ones in the TP ecoregion, though this gap is much narrower than that observed in the MWP ecoregion. The estimated number of basins in the TP ecoregion is similar to estimates obtained within the PPR of Minnesota which roughly corresponds to the extent of the TP ecoregion in the state. Dahl (2014) estimated a total of 12,971 semi-permanent, emergent wetland basins and 13,227 ponds in the PPR of Minnesota (26,198 combined total). These figures are similar to the 17,420 semi-permanent, natural basins and 14,668 man-made ponds estimated by the DWQA; 32,088 combined total (Figure 26).

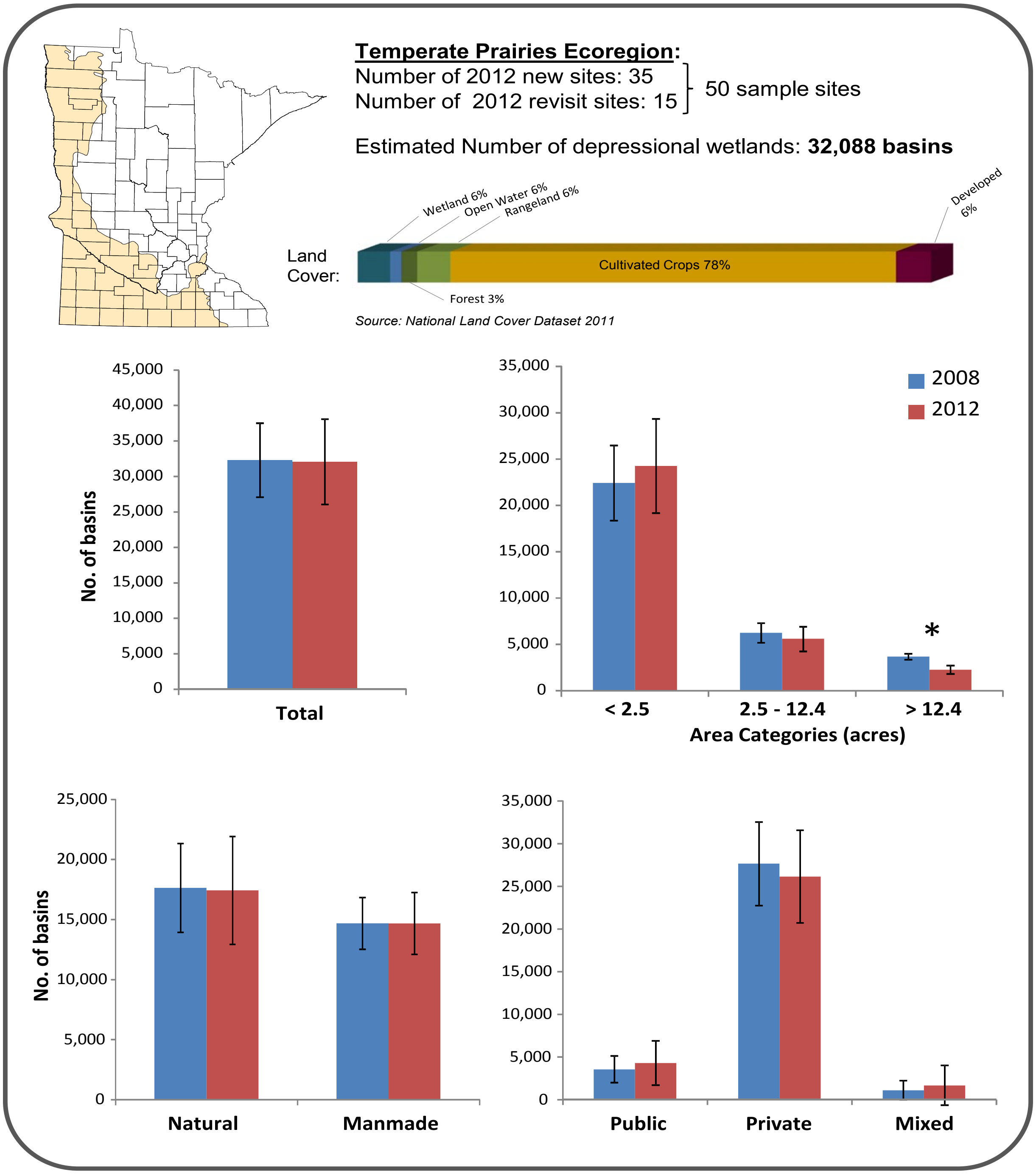


Figure 26. Estimates of the total number of depressional wetlands in the Temperate Prairies ecoregion comparing time 1 of the survey (2008) to time 2 (2012). Bracketed lines represent the 95% confidence interval associated with each estimate. Asterisk indicates a statistically significant change between time periods.

The difference between the two estimates may stem from how wetland basins were classified in each survey. Dahl (2014) used a spatial dominance approach where the most predominant water regime (e.g., temporary, seasonal, semi-permanent) was used to categorize the basin. The DWQA categorized basins based on its most permanent water regime regardless of its spatial extent. This approach would categorize any basin with semi-permanently inundated emergent vegetation as 'semi-permanent emergent' whereas the method used in Dahl (2014) would not if another wetland type was more extensive within the basin. Based on the difference in methodology alone, the DWQA survey would be expected to yield a higher estimate of semi-permanent basins. Overall, the PPR status and trends survey estimated a total of 128,330 basins (2009 status) in the prairie region of Minnesota, but this figure included all types of wetlands as well as lakes and ponds. Based on these results, semi-permanent wetlands and ponds together account for approximately 20% of the basins in the PPR of Minnesota.

Depressional wetland condition

- Plant IBI results indicate that an estimated 12% of TP depressional wetlands and ponds are in good condition while 60% are in poor condition (Figure 27). Plant community condition did not vary significantly among the wetland area and ownership categories according to CDF tests. Plant communities within natural wetlands are in significantly better condition than communities within man-made wetlands and ponds ($F = 10.98$, $df_1 = 2$, $df_2 = 45$, $p < 0.001$).
- An estimated 44% of TP depressional wetlands and ponds have healthy macroinvertebrate communities while 32% are in poor condition (Figure 27). Macroinvertebrate community condition did not vary significantly among the wetland area classes or between natural and man-made wetland types. However, wetlands on private property are in significantly better condition than those located on public land according to the macroinvertebrate IBI ($F = 3.48$, $df_1 = 2$, $df_2 = 40$, $p < 0.041$).
- Water clarity is rated as good in 79% of TP depressional wetlands and ponds while only 9% exhibit turbid or poor conditions (Figure 27). CDF tests comparing transparency results among the various subpopulations could not be performed due to insufficient variability in transparency values (i.e., a large number of >100 cm readings).
- Total phosphorus concentrations are low in 70% of TP depressional wetlands and ponds while 30% are high (Figure 27). Man-made wetlands had significantly lower phosphorus concentrations than did natural wetlands ($F = 3.24$, $df_1 = 2$, $df_2 = 45$, $p = 0.048$). Concentrations did not vary significantly among wetland ownership and area categories.
- Similar to the MWP ecoregion, nitrate + nitrite concentrations are below detection (= Low) in 93% of TP depressional wetlands and ponds (Figure 27). CDF tests comparing the results among the various subpopulations could not be performed due to insufficient variability in the values (i.e., majority below detection).
- An estimated 63% of TP depressional wetlands and ponds have low concentrations of Kjeldahl nitrogen while 9% exhibit high concentrations (Figure 27). Kjeldahl-N concentrations were significantly lower in wetlands on private property compared to those with mixed (public/private) ownership ($F = 6.95$, $df_1 = 2$, $df_2 = 34$, $p = 0.003$). Concentrations did not vary significantly among the wetland area classes or between natural and man-made wetland types.
- Chloride concentrations are low in an estimated 41% of TP depressional wetlands and ponds while 59% are high relative to least-disturbed reference sites (Figure 27). Chloride was significantly higher in wetlands on private property compared to those that had mixed ownership ($F = 11.55$, $df_1 = 2$, $df_2 = 34$, $p < 0.001$). Concentrations did not vary significantly among the wetland area classes or between natural and man-made wetland types.

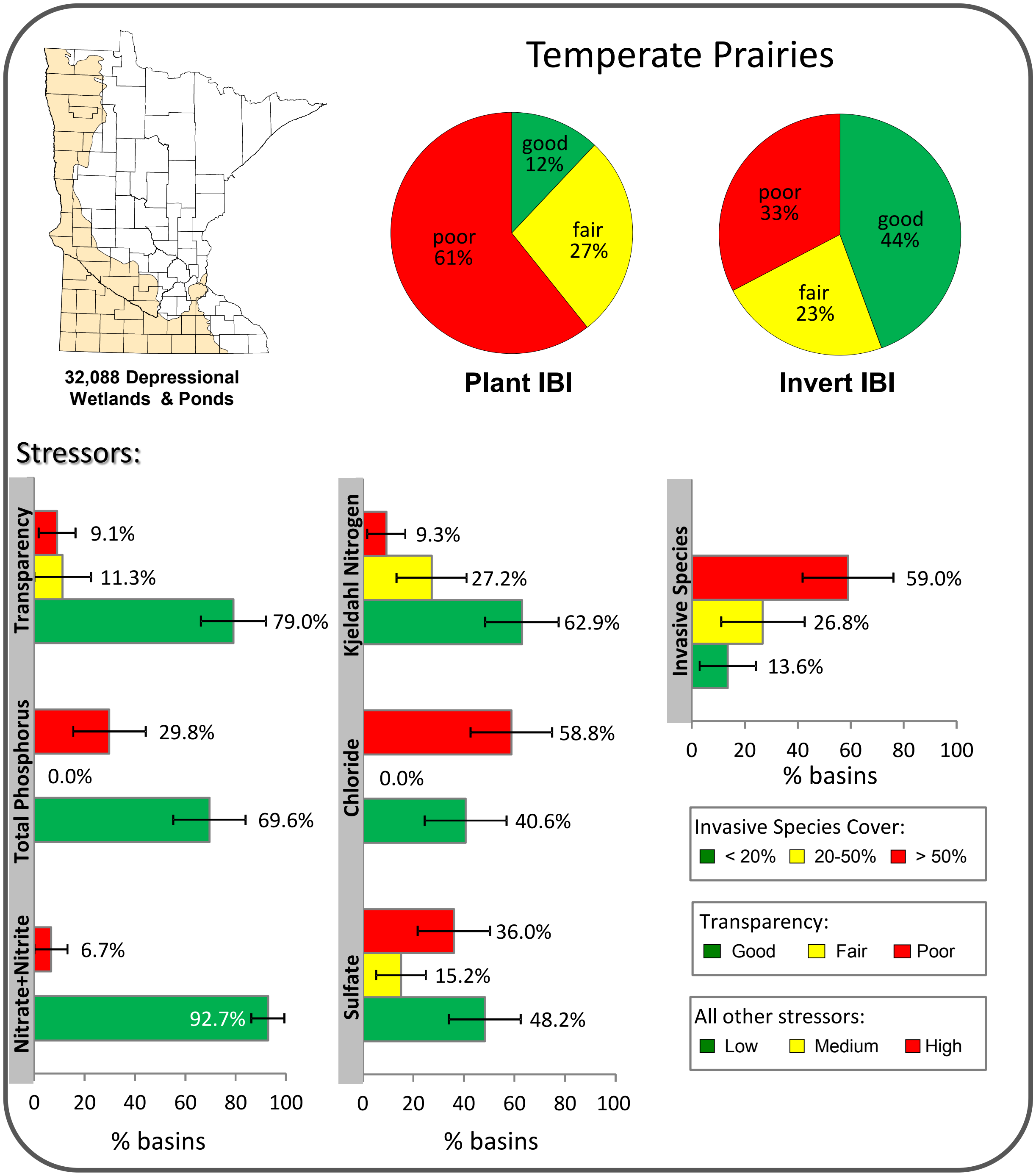


Figure 27. Biological condition and stressor level estimates for Temperate Prairie depressional wetlands and ponds. Bracketed lines represent the width of the 95% confidence interval associated with each estimate.

- An estimated 48% of TP depressional wetlands and ponds have low sulfate concentrations while 36% have high concentrations (Figure 27). Concentrations did not vary significantly among wetland ownership and area categories. Unlike what was found in the MWP ecoregion, man-made wetlands in the TP ecoregion have significantly higher sulfate concentrations than natural wetlands ($F = 3.39$, $df_1 = 2$, $df_2 = 45$, $p = 0.043$).

- The abundance of invasive plant species is low (<20% cover) in 14% of TP depressional wetlands and ponds and high (>50% cover) in 59% (Figure 27). Small (<2.5 acres) wetlands have significantly lower invasive species abundance than the medium and large wetland area classes ($p = 0.016$ and $p < 0.001$, respectively) (Figure 28A). In addition, all three wetland ownership categories are significantly different from one another in terms of their invasive species abundance according to CDF tests. One hundred percent of wetlands on mixed ownership property have high invasive species cover (note: this group suffers from a low sample size; n = 4), followed by 64% high on private property, and 30% high on public land (Figure 28B).

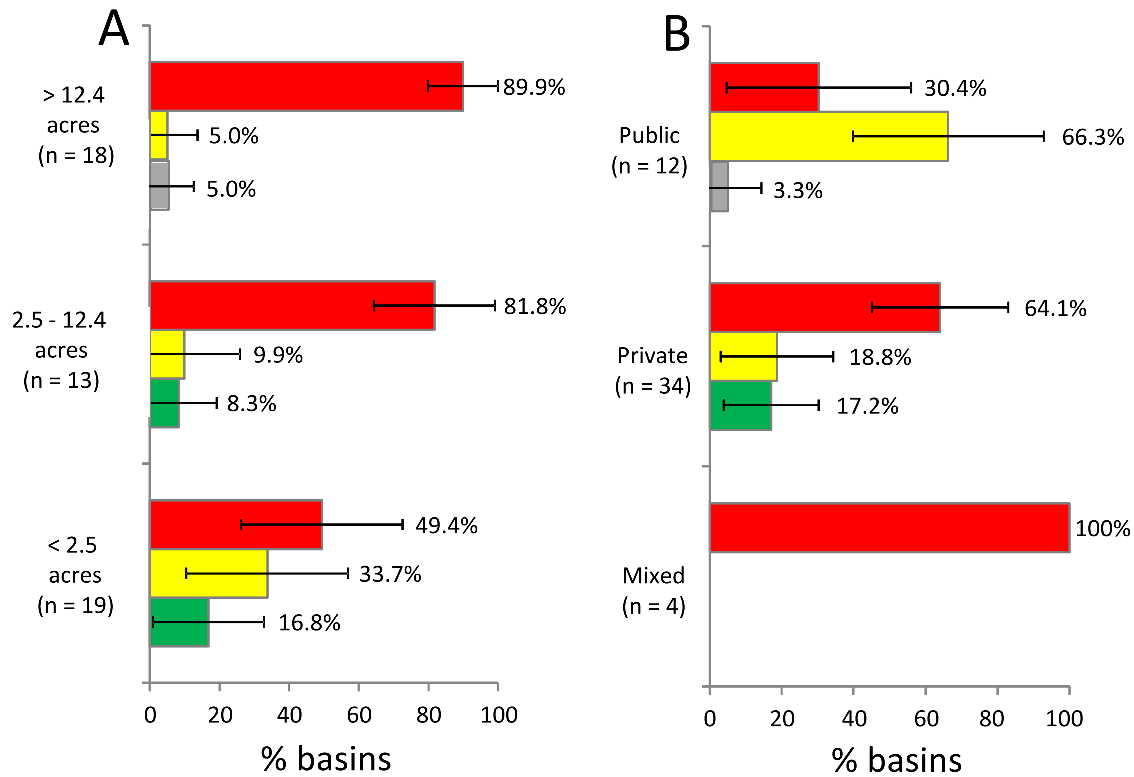


Figure 28. Invasive plant species abundance among (A) wetland area classes and (B) ownership categories. Gray bar = not assessed.

The extent of the measured stressors, as ranked by '% high' in the TP, was very similar to the ranking observed at the statewide level (Figure 29). The only difference was that total phosphorus and sulfate switched positions compared to the statewide stressor rankings (note: Figure 29 depicts the list of stressors in the ranked order observed at the statewide scale). Unlike results at the statewide scale, chloride, sulfate, Kjeldahl nitrogen, and reduced transparency all posed an elevated risk to wetland macroinvertebrate communities. Compared to the statewide results, invasive plant species and sulfate dropped out as significant risks to wetland plant communities in the TP ecoregion. Unfortunately, relative risk estimates are sensitive to the occurrence of sites in the data set exhibiting low levels of a stressor and poor condition (i.e., denominator of estimate). This was the case for the relative risk estimate of non-native invasive plants on wetland plant communities in the TP ecoregion; there were plenty of high stress/poor condition sites in the data set (24 out of 25) but the proportion of low stress/poor condition sites (3 out of 5) diminished this signal. Combining the 2012 and 2008 TP data sets in an unweighted analysis (i.e., doesn't factor in design weights) yields a similarly low relative risk estimate for invasive plant species. Once again this was due to the relatively high proportion (~50%) of low (invasive species) stress/poor condition sites, lending support to the findings presented in Figure 29 that other stressors significantly impact plant community condition in this ecoregion as well.

Similar to the baseline DWQA results, chloride represents the largest threat, among those measured in the survey, to the biological condition of TP wetlands and ponds. This is because it is found at high concentrations in approximately 60% of the wetlands and ponds in this ecoregion (tied with invasive plant species) and it poses a significant threat to both communities (Figure 29). The highest chloride concentration measured in this ecoregion (211 mg/l) was from a man-made stormwater retention pond. However, similar to 2008 results, elevated chloride concentrations were also found in a number of wetlands and ponds located in rural areas, indicating that de-icing compounds may not be the only significant source of chloride in this ecoregion. Unlike the 2008 results, chloride posed an elevated risk to plant community condition in this round of the DWQA. High chloride concentrations can disrupt the ability of some plants to regulate water absorption, which may lead to dehydration and the eventual spread of salt-tolerant, invasive species such as narrow-leaved cattail (*Typha angustifolia*) and purple loosestrife (*Lythrum salicaria*) (Wilcox 1986, Isabella et al. 1987). Chloride can have similar effects on the macroinvertebrate community by disrupting the osmoregulatory abilities (i.e., maintaining a proper balance of salt and water internally) of certain species (Sutcliffe 1961), which may lead to the proliferation of more tolerant species.

As was observed in the MWP, transparency was poor in less than 10% of the population yet posed an elevated risk to both plants and macroinvertebrates (Figure 29). Detecting a significant risk despite having a relatively low occurrence of poor or turbid conditions suggests that a tight relationship between water clarity and wetland biological condition exists. It should be noted, however, that the majority of sites exhibiting poor transparency were man-made ponds and likely also experience a wide variety of other impacts.

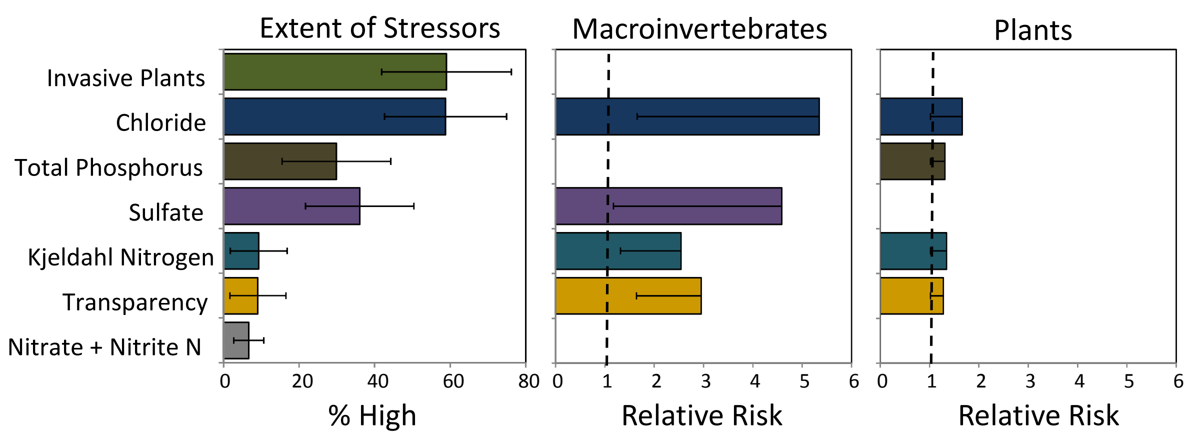


Figure 29. Extent of stressors and their relative risk to plant and macroinvertebrate communities in Temperate Prairie depressional wetlands and ponds. Bracketed lines represent 95% confidence intervals (for % estimates) or lower confidence limits (for relative risk estimates). A stressor without an associated bar on the relative risk graphs indicates that it did not pose an elevated risk to that community.

Changes in depressional wetland condition

Plant and macroinvertebrate community condition in TP depressional wetlands and ponds did not change significantly between 2008 and 2012 (Figure 30, Table 4). The concentration of nitrate + nitrite exhibited significant improvement (i.e., decreased concentrations) between the two time periods at both the ecoregion scale as well as within the subpopulations (Table 4). While these results are promising, the effect of precipitation patterns during these two time periods cannot be discounted. As discussed in the Statewide Results and Discussion section, a large area of the TP ecoregion saw limited rainfall in June 2012, which likely contributed to the low occurrence of nitrate + nitrite detections

(= high) in water samples taken that year (see Appendix D). Invasive plant species in the TP ecoregion showed a marked increase between 2008 and 2012 at both the ecoregion scale as well as within the subpopulations (Table 4). However, these changes may largely be a reflection of the vegetation sampling procedure adjustment (see Natural vs. Man-made section in Statewide Results and Discussion) as suggested by the man-made category exhibiting the greatest (degrading) change. Publically-owned depressional wetlands and ponds was the only category that exhibited a decrease in the abundance of invasive plants.

Table 4. Change detection analysis results for the Temperate Prairies ecoregion, comparison of 2012 indicator data to 2008 data. nc = no significant change detected. Category abbreviations: G = good, P = poor, L = low, H = high. Green text indicates condition improvement, red indicates degrading conditions.

	TP	Wetland origin		Wetland area			Ownership	
Indicators	**Ecoregion**	**Natural**	**Man-made**	**<2.5**	**2.5-12.4**	**>12.4**	**Public**	**Private**
Plant IBI	nc	nc	nc	nc	-21% G	nc	nc	nc
Invert IBI	nc	nc	nc	nc	nc	nc	nc	nc
Transparency	nc	nc	nc	nc	nc	nc	nc	nc
Total Phosphorus	nc	nc	nc	nc	nc	nc	nc	nc
Nitrate + Nitrite	-21% H	nc	-34% H	-23% H	-28% H	nc	nc	-25% H
Kjeldahl Nitrogen	nc	nc	nc	nc	nc	-28% L	nc	nc
Chloride	nc	nc	nc	nc	nc	nc	nc	nc
Sulfate	nc	nc	nc	nc	nc	nc	nc	nc
Invasive Species	-25% L	nc	-46% L	nc	-42% L	nc	-58% H	-28% L

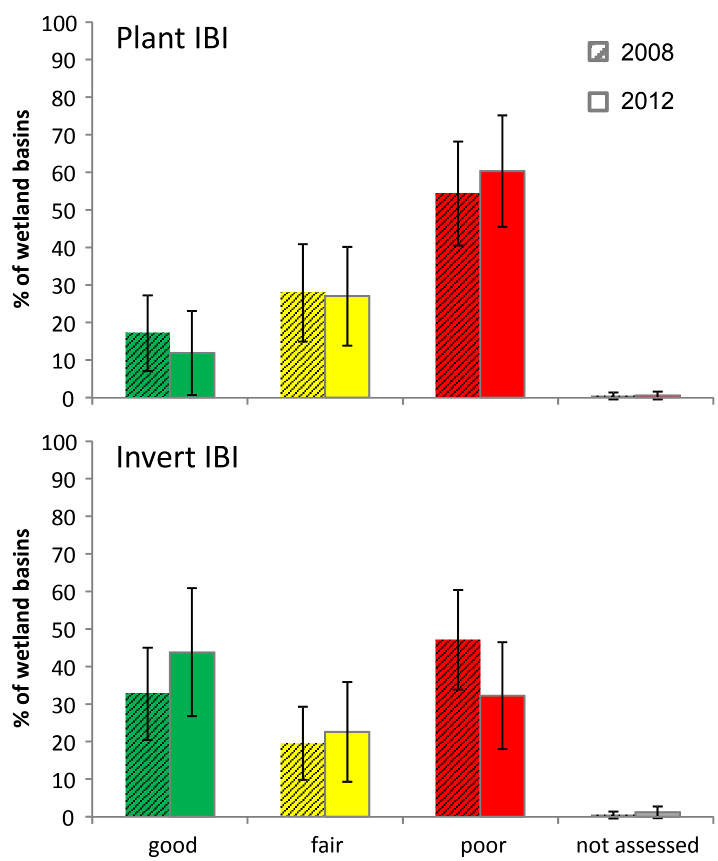


Figure 30. Comparison of baseline Temperate Prairie wetland biological condition results to biological condition in 2012. Bracketed lines represent the width of the 95% confidence interval associated with each estimate.

Next steps

The next cycle of data collection for the DWQA is scheduled for summer 2017. Current plans include examining the feasibility and effectiveness of using the Floristic Quality Assessment (FQA) as a replacement for the depressional wetland plant IBI. The FQA is a more advanced approach that has more detailed condition categories and can describe depressional wetland communities at the plant community scale (Bourdaghs 2012). Similar to the transition from the Transparency tube to the Secchi tube, relationships will need to be developed between the two assessment methodologies that will allow plant IBI results to be converted into equivalent FQA results and carried forward in future trend analyses.

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Appendix A

Metrics of the plant and macroinvertebrate IBIs used to assess the condition of depressional wetlands and ponds throughout the state. Tables indicate the ecoregions where each metric applies and can be used to construct the six individual IBIs.

Plant IBI Metrics	Response to Disturbance	Ecoregion[1]		
		MWP	TP	MWS
Number of native aquatic plant species.	Decrease	X	X	
Number of native wetland graminoid species.	Decrease	X	X	
Number of native wetland perennial species.	Decrease	X	X	X
Number of vascular genera.	Decrease	X	X	
Number of nonvascular taxa	Decrease	X		
Number of sensitive species	Decrease	X		
Number of disturbance tolerant taxa divided by the total taxa richness	Increase	X		
Number of distinct plant guilds.	Decrease		X	
Number of taxa sensitive to disturbance, defined by Coefficient of Conservatism values ≥ 7	Decrease		X	X
Number of disturbance tolerant taxa divided by the total taxa richness. Tolerant taxa defined by Coefficient of Conservatism values ≤ 3 or is introduced.	Increase		X	
Number of native Cyperaceae (Sedges, bulrushes, etc) species divided by the total emergent taxa richness	Decrease			X
Cover of *Carex* spp.	Decrease	X		
Cover of invasive *Typha* spp. and small floating aquatics (*Lemna*, *Spirodela*, *Wolfia*, *Riccia*, *Ricciocarpos* spp.)	Increase		X	
Cover of small floating aquatics (*Lemna*, *Spirodela*, *Wolfia*, *Riccia*, *Ricciocarpos* spp.) divided by total aquatic cover	Increase			X
Cover of the 3 most dominant species divided by total sample cover	Increase	X		
Cover of the 2 most dominant emergent species divided by total emergent cover	Increase			X
Cover of taxa with persistent litter divided by total sample cover	Increase	X		
Cover of disturbance tolerant taxa. Tolerant taxa defined by Coefficient of Conservatism values ≤ 3 or is introduced.	Increase			X
Shannon Diversity index based only on native species	Decrease			X
Total number of metrics in IBI:		**10**	**8**	**7**

Macroinvertebrate IBI Metrics	Response to Disturbance	Ecoregion[1] MWP	TP	MWS
Number of Ephemeroptera, Trichoptera, and Odonata genera	Decrease	X	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		X
Number of Diptera genera	Decrease		X	
Number of collector-gatherer genera	Decrease	X		
Number of collector (collector-gatherer & collector-filterer) 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 most dominant genus divided by total abundance of sample	Increase			X
Abundance of the 3 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			X
Total number of metrics in IBI:		**10**	**8**	**5**

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

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

Appendix B

Water quality parameters analyzed by Minnesota Department of Health and years each method was utilized in the first three years of the survey.

Analyte	Fraction	Report Limits	Units	Method Reference	MDH Code
Chloride	Total	1.00	mg/L	EPA 300.1	297
Nitrate + Nitrite Nitrogen	Total	0.05	mg/L as N	EPA 353.2	69
Kjeldahl Nitrogen	Total	0.10	mg/L as N	EPA 351.2	68
Organic Carbon	Total	1.0	mg/L	SM 5310 C	98
Phosphorus	Total	0.010	mg/L as P	SM 4500 P-I*	59
Sulfate	Total	1.00	mg/L	EPA 300.1	293

* Total phosphorus was analyzed using EPA 365.1 method in baseline DWQA. Methodological issues associated with the transition to SM 4500P-I may have resulted in low readings of total phosphorus, particularly in low nutrient waters.

Appendix C

Criteria used to determine depressional wetland condition and stressor levels relative to regional reference sites (MWP = Mixed Wood Plains; TP = Temperate Prairies).

Parameter/ Ecoregion	Wetland Condition Categories		
	Good	Fair	Poor
Plant IBI score (0-100):			
MWP	> 56	< 56, > 42	< 42
TP	> 78	< 78, > 61	< 61
Macroinvertebrate IBI score (0-100):			
MWP	> 64	< 64, > 44	< 44
TP	> 66	< 66, > 56	< 56

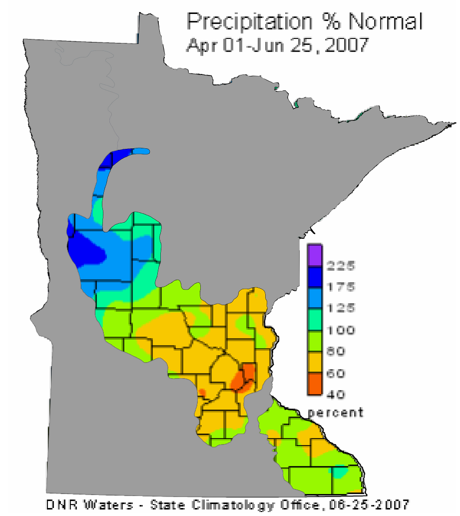
Parameter/ Ecoregion	Transparency Categories		
	High	Medium	Low
Secchi Tube Reading (cm):			
MWP	> 66	< 66, > 38	< 38
TP	> 65	< 65, > 45	< 45

Parameter/ Ecoregion	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
TP	< 1.60	> 1.60, < 2.97	> 2.97
Total Phosphorus (mg/L):			
MWP	< 0.148	> 0.148, < 0.384	> 0.384
TP	< 0.180	> 0.180, < 0.202	> 0.202
Chloride (mg/L):			
MWP	< 1.4	> 1.4, < 7.9	> 7.9
TP	< 7.6	> 7.6, < 8.6	> 8.6
Sulfate (mg/L):			
MWP	< 5.9	> 5.9, < 12.5	> 12.5
TP	< 18.7	> 18.7, < 127.4	> 127.4
Invasive Plant Species (% cover):			
MWP & TP	< 20	> 20, < 50	> 50

Appendix D

Characterization of climatic conditions during 2007-2008 and 2012 DWQA survey periods.

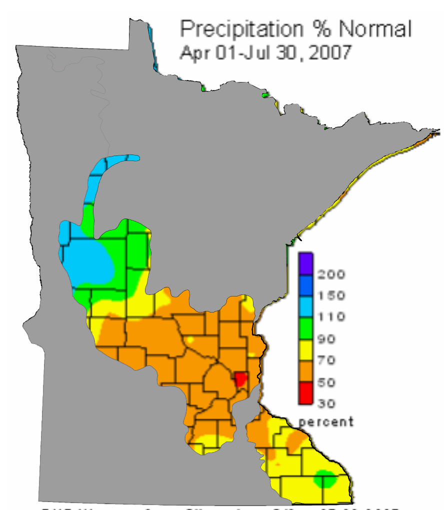
Climate affects the chemical, physical, hydrological, and biological constituents that together make up a wetland ecosystem. Thus, characterizing and comparing climatic conditions during each round of the depressional wetland survey is vital information that provides context for the observed results. This appendix provides a synopsis of precipitation and temperatureduring the summer months of 2007, 2008, and 2012, the seasonal index period of the biological indicators, and to some extent the months leading up to this period. For reference, macroinvertebrate and water chemistry data is collected in the month of June, while plant community data is primarily collected in July. Maps below were obtained (and modified to show ecoregion boundaries) from the DNR Division of Ecological Services, Weekly Precipitation, Departure, and Ranking Maps (http://climate.umn.edu/doc/weekmap.asp) and the U.S. Drought Monitor (http://droughtmonitor.unl.edu/).



2007 Mixed Wood Plains Ecoregion Survey

Precipitation (June): The MWP ecoregion received heavy rainfall during the first week of June, but other than that the region received very little rainfall until the end of the month when the southeast portion of the ecoregion had 1 to 2 inches of rain. Overall, considering the amount of precipitation since April 1st, the northwest part of the ecoregion was wetter than normal (i.e., the 30-year average), the central part was drier than normal, and the southeast part was close to normal by the end of June. The U.S. Drought Monitor) categorized the majority of the ecoregion as 'abnormally dry' with portions of the east central region listed under a 'moderate drought' by the end of the month.

Precipitation (July): With the exception of the first week, the ecoregion received very little rainfall during the month of July. Only the southeast, driftless region of the MWP received any substantial rainfall towards the end of the month. However, very few survey sites were located in this part of the ecoregion. By month's end the majority of the ecoregion was much drier than normal. The U.S. Drought Monitor characterized these conditions as a 'moderate' to 'severe drought' across most of the ecoregion as well as the state.

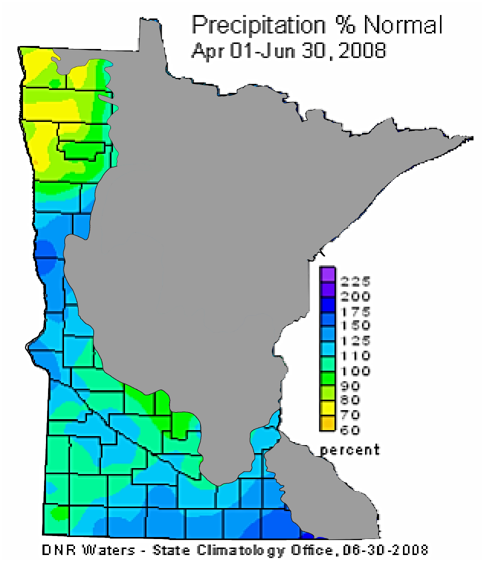


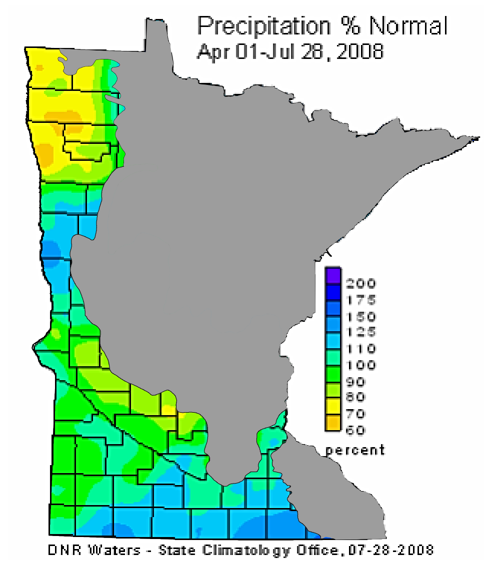
Temperature: Using St. Cloud, Minnesota as a proxy location for the entire ecoregion, and the period of January 1 to August 1, 2007 to characterize conditions preceding and during the monitoring index period, the degree day value above 50° F was 1785 hrs, representing a value 391 hrs above normal. In other words, it was warmer than average during this eight month period.

2008 Temperate Prairie Ecoregion Survey

Precipitation (June): The TP ecoregion received significant rainfall amounts throughout the month of June. This precipitation as well as month's prior led to wetter than normal conditions throughout most of the ecoregion. According to the U.S. Drought Monitor, the northwest portion of the ecoregion was 'abnormally dry', but other parts of the ecoregion were not in any state of drought.

Precipitation (July): Throughout July all parts of the TP ecoregion received significant rainfall at some point in time. The northwest portion received precipitation during the second week of the month, while the southern portion received rain during the last two weeks. Overall, by the end of July the region was slightly above normal or normal compared to the 30-year running average. According to the U.S. Drought Monitor, the northwest and west central portions of the ecoregion were characterized as 'abnormally dry' on July 29, 2008.





Temperature: Using Morris, Minnesota as a proxy location for the entire ecoregion, and the period of January 1st to August 1, 2008 to characterize conditions preceding and during the monitoring index period, the degree day value above 50^{o} F was 1280 hrs, a value that is 262 hrs less than normal. Over this eight month period temperatures were cooler than average.

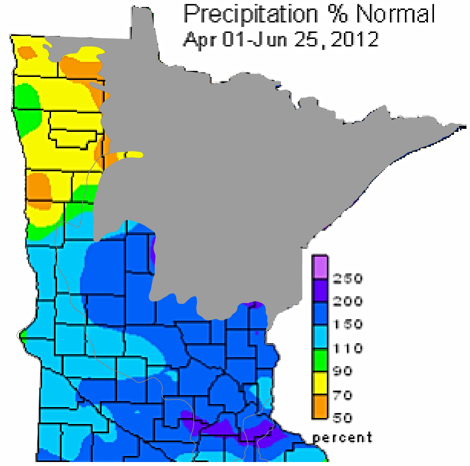
2012 Depressional Wetland Survey (MWP and TP ecoregions)

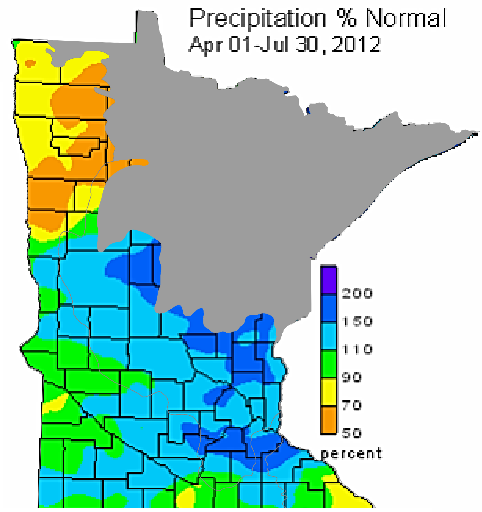
Precipitation (June): For the MWP ecoregion, the first several days of June had virtually no rainfall. The weeks that followed saw consistent precipitation with one storm event that occurred mid-month dropping 10+ inches of rain in certain areas. Overall, the ecoregion was wetter than normal by month's end with only small portions of the northwest and southeast rated as 'abnormally dry.'

Portions of the TP ecoregion received very little precipitation in June, while other portions received a tremendous amount during the large storm event that occurred in the middle of the month. The northwest and southwest regions of the ecoregion were relatively dry in June, while the west central and southeast regions were wetter than normal. Considering the amount of rainfall since April 1, the majority of the ecoregion was wetter than normal with only the northwest region being drier than normal. By the end of June the U.S. Drought Monitor rated the northwest part of the ecoregion as being in a 'moderate drought' and portions of the west central and southeast as 'abnormally dry.'

Precipitation (July): It is difficult to summarize the precipitation pattern of the MWP ecoregion in July of 2012. Various 1-3" precipitation events affected most portions of the ecoregion at one time or another in July. Only the west central and southeast portions of the ecoregion saw any extended dry spells that lasted two weeks or more. The overall impact of July precipitation was that conditions were approaching normal across the ecoregion. With the exception of the extreme northwest and southeast portions of the ecoregion, the MWP was not impacted by drought in July 2012.

The TP ecoregion saw very little rainfall during the month of July. Only the extreme northwest and west central portions saw any significant precipitation (1+ inches) as well as a few localized areas in the southern part of the ecoregion. At the end of July, considering the cumulative precipitation since April 1, the northwest portion of the ecoregion was drier than normal with the remainder at or approaching normal conditions. However, considering a broader time scale, much of the ecoregion was experiencing moderate to severe drought conditions by the end of July according to the U.S. Drought Monitor.





U.S. Drought Monitor
Minnesota

July 31, 2012
(Released Thursday, Aug. 2, 2012)
Valid 7 a.m. EST

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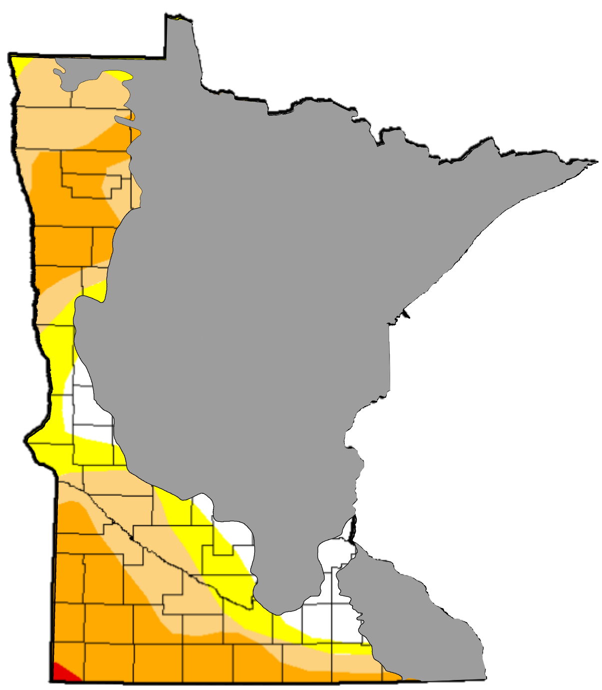

Drought Conditions (Percent Area)

	None	D0-D4	D1-D4	D2-D4	D3-D4	D4
Current	52.11	47.89	35.02	16.25	0.18	0.00
Last Week 7/24/2012	37.88	62.12	37.37	13.05	0.00	0.00
3 Months Ago 5/1/2012	1.78	98.22	60.07	18.65	0.00	0.00
Start of Calendar Year 1/3/2012	0.79	99.21	57.45	24.08	0.00	0.00
Start of Water Year 9/27/2011	48.42	51.58	19.26	4.58	0.00	0.00
One Year Ago 8/2/2011	91.99	8.01	0.00	0.00	0.00	0.00

Intensity:
D0 Abnormally Dry
D1 Moderate Drought
D2 Severe Drought
D3 Extreme Drought
D4 Exceptional Drought

The Drought Monitor focuses on broad-scale conditions. Local conditions may vary. See accompanying text summary for forecast statements.

Author:
Mark Svoboda
National Drought Mitigation Center

USDA

http://droughtmonitor.unl.edu/

Temperature: Using St. Cloud, Minnesota as a proxy location for the MWP ecoregion, and the period of January 1st to August 1, 2012 to characterize conditions preceding and during the monitoring index period, the degree day value above 50^{o} F was 1918 hrs or 524 hrs above normal. A total of 495 hrs above 50^{o} F occurred between March 1 and June 1, unusually warm temperatures for this time of the year in Minnesota.

Using Morris, Minnesota as a proxy location for the TP ecoregion, and the period of January 1 to August 1, 2012 to characterize conditions preceding and during the monitoring index period, the degree day value above 50^{o} F was 1779 hrs or 237 hrs above normal.