Biological stressors

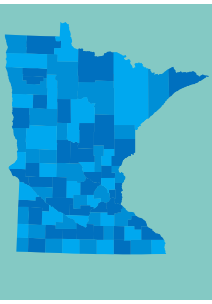
November 2022

Stressors Candidate Causes

Stressors to biological communities in Minnesota’s river and streams

.

MINNESOTA POLLUTION
CONTROL AGENCY



Authors

Stressor ID Lateral Team

Minnesota Pollution Control Agency

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

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

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

Document number: wq-ws1-27

Contents

Tables

Figures

Candidate causes of biological stress and applicable water quality standards in Minnesota

This information provides an overview of the pathway and effects of each candidate stressor considered in the biological stressor identification process. Data and standards specific to Minnesota are also included. The U.S. Environmental Protection Agency (EPA) has additional information, conceptual diagrams of sources and causal pathways, and publication references for numerous stressors on its CADDIS website. The Causal Analysis/Diagnosis Decision Information System, or CADDIS, is a website developed to help scientists and engineers in the regions, states, and tribes conduct causal assessments in aquatic systems.

Temperature

The factors that control stream water temperature and the biological effects of elevated temperature are complex. Stream temperature naturally varies due to:

- Air temperature
- Geological setting
- Amount of shading
- Water inputs from tributaries and springs

Human activities can increase stream temperatures through:

- Altering riparian vegetation (loss of shading)
- Urban runoff from warm impervious surfaces such as parking lots
- Agricultural runoff
- Loss of landscape water storage and thus periods of reduced stream water volume
- Direct discharges of warm wastewater to the stream

Different organisms are adapted to and prefer different temperature ranges, and will thrive or decline based on the temperature ranges found in a stream.

Figure 1. Cold water from a tributary spring

Warmer water holds less dissolved oxygen (DO). Water temperature also affects the toxicity of many chemicals in the aquatic environment. Algal blooms are often associated with temperature increases (EPA, 1986). Water temperature affects metabolism, and thus food and oxygen needs, and regulates the ability of organisms to survive and reproduce (EPA, 1986).

Increases in temperature due to altered landscapes can lead directly to extirpation of coldwater assemblages. Warmer water impacts organisms indirectly due to the inverse relationship with dissolved oxygen and directly through changes in growth and reproduction, egg mortality, disease rates, and direct mortality. Macroinvertebrate species have well-known tolerances to thermal changes, and community composition of macroinvertebrates is useful in tracking the effects of increasing temperature.

Fish assemblages also change with temperature. Coldwater adapted species either leave, are unable to reproduce, or die in warmer regimes. For example, when temperatures rise near 21°C (69.6 °F), other fish can have a competitive advantage over trout for the food supply (Behnke, 1992). The temperature at which fish continue to feed and gain weight is considered their functional feeding temperatures. The limits for brown trout growth are 4-19.5°C (39.2-67.1°F) (Elliott & J.A., 1995); however, for egg development, brown trout need temperatures between 0 and 15 °C (0-59°F). According to Bell (2006), brown trout may be physiologically stressed in the thermal window of 19-22°C (66.2-71.6°F). These temperatures are near the upper metabolic limit for trout and may affect the ability to maintain normal physical function and ability to gain weight.

Brook trout functional feeding temperatures are between 12.7°C and 18.3°C (54.9-64.9°F) (Raleigh, 1986). They can briefly tolerate temperatures near 22.2°C (71.9°F), but temperatures of 23.8°C (74.8°F) for a few hours are generally lethal (Flick, 1991). Juvenile brook trout density is negatively correlated with July mean water temperatures (Hinz, 1997). Growth and distribution of juvenile brook trout is highly dependent on temperature (McCormick et al., (1972). For more information on the causes and effects of elevated temperature, see EPA's CADDIS temperature webpage.

Water quality standards

The standard for Class 2B (warmwater) waters of the state is not to exceed 5°F above natural, based on a monthly average of maximum daily water temperature. In addition, this temperature metric cannot exceed the daily average of 86°F (30°C).

The state standard for temperature in Class 2A streams is “no material increase” (7050.0222 subp.2).

Types of temperature data

Both grab sample (instantaneous) and/or continuous temperature data have been collected and is available in many locations depending on the watershed. Continuous data are measured at 15-minute intervals.

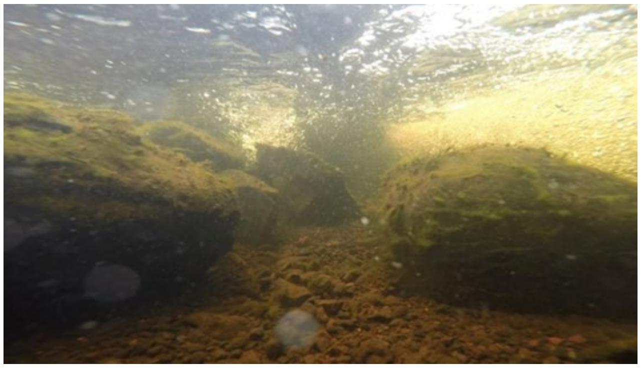
Sources and causal pathways model for temperature

The causes and potential sources of excess temperature are modeled at EPA's CADDIS Temperature webpage.

Dissolved oxygen

DO refers to the concentration of oxygen gas within the water column. Adequate DO is important to growth and reproduction of aquatic life. Oxygen diffuses into water from the atmosphere (turbulent flow enhances this diffusion) and from the release of oxygen by aquatic plants during photosynthesis. DO concentrations in streams are driven by several factors.

Figure 2. Oxygen diffusion within the water column



Large-scale factors include:

- Climate
- Topography
- Hydrologic pathways

These in turn influence smaller scale factors:

- Water chemistry
- Temperature
- Biological productivity

DO concentrations change hourly, daily, and seasonally in response to these driving factors. As water temperature increases, its capability to hold oxygen decreases. Low DO can be an issue in streams with slow currents, excessive temperatures, high biological oxygen demand, and/or high groundwater seepage (Hansen, 1975). In most streams and rivers, the critical seasonal conditions for stream DO usually occur during late summer when water temperatures are at or near the annual high while stream flow volumes and rates are near base flow. The critical daily period for DO is early morning, when the daily DO flux is at its minimum.

Human activities can alter many of these driving factors and change the DO concentrations of water resources. Increased nutrient content of surface waters is a common human influence, which can result in excess aquatic plant growth. This situation often leads to a decline in daily minimum oxygen concentrations and an increase in the magnitude of daily DO concentration fluctuations due to the decay of the excess organic material, increased usage of oxygen by plants at night, and their greater oxygen production during the daytime. Humans may directly add organic material by municipal or industrial effluents. These forms of pollution increase the risk of eutrophication, which can also lead to low DO.

Aquatic organisms require oxygen for respiration. Inadequate oxygen levels can alter fish behavior, such as moving to the surface to breathe air, or moving to another location in the stream. These behaviors can put fish at risk of predation, or may hinder their ability to obtain necessary food resources (Kramer, 1987). Additionally, low DO levels can significantly affect fish growth rates (Doudoroff, 1965). Fish species differ in their preferred temperature range, so alterations in water temperature (and DO) will alter the composition of fish communities.

Low DO, or highly fluctuating concentrations of DO can have detrimental effects on many fish and macroinvertebrate species (Davis, 1975) (Nebeker, 1991). Increased water temperature raises the metabolism of organisms, and thus their oxygen needs, while at the same time, the higher-temperature water holds less oxygen. Some aquatic insect species have anatomical features that allow them to access atmospheric air, though many draw their oxygen from the water column. Macroinvertebrate groups (Orders) that are particularly stressed by low DO levels include mayflies (with a few exceptions), stoneflies, and caddisflies.

For more detailed information on DO as a stressor, see the EPA’s CADDIS DO webpage.

Water quality standards

The DO standard (as a daily minimum) is 5 mg/L for class 2B (warmwater) streams and 7 mg/L for Class 2A (coldwater). Additional stipulations are included in this standard, detailed in the Guidance Manual for Assessing the Quality of Minnesota Surface Waters (MPCA, 2009a).

Types of dissolved oxygen data

1. **Point measurements**
   Instantaneous (one moment in time) DO data are collected and used as an initial screening for low DO reaches. Because DO concentrations can vary significantly with changes in flow conditions and time of sampling, conclusions using instantaneous measurements need to be made with caution and are not completely representative of the DO regime at a given site.
2. **Longitudinal (synoptic)**
   This sampling method involves collecting readings of DO from several locations along a significant length of the stream path in a short period of time. It is best to perform this sampling in the early morning in order to capture the daily minimum DO readings.
3. **Diurnal (continuous)**
   Continuous sampling using water quality sondes (a submerged electronic sampling device) provides a large number of measurements to reveal the magnitude and pattern of diurnal DO flux at a site. This sampling captures the daily minimum DO concentration, and when deployed during the peak summer water temperature period allows an assessment of the annual low DO levels in a stream system.

Sources and causal pathways model for low dissolved oxygen

DO concentrations in streams are driven by a combination of natural and anthropogenic factors. Natural background characteristics of a watershed, such as topography, hydrology, climate, and biological productivity influence the DO regime of a waterbody. Wetlands and groundwater influence can be natural sources of low DO water to a stream. Agricultural and urban land uses, impoundments (dams), and point source discharges are just some of the anthropogenic factors that can cause unnaturally high, low, or fluctuating DO concentrations. The conceptual model for low DO as a stressor is found on the EPA's CADDIS DO website.

Eutrophication

Phosphorus (P), an important plant nutrient, is typically in short supply in natural systems, but human activity on the landscape often exports phosphorus to waterways, which can impact stream organisms. Nutrient sources can include:

- Agricultural runoff
- Animal waste
- Fertilizer
- Industrial and municipal wastewater facility discharges
- Non-compliant septic system effluents
- Urban stormwater runoff

Figure 3. Excessive production of algae in a stream

Phosphorus exists in several forms, with the soluble form, orthophosphorus, readily available for plant and algal uptake. While phosphorus itself is not toxic to aquatic organisms, it can have detrimental effects via other associated chemistry when levels are elevated above natural concentrations. Increased nutrients can cause excessive aquatic plant and algal growth (eutrophication), which alters physical habitat, food resources, and oxygen levels in streams. Excess plant growth increases DO during daylight hours and saps oxygen from the water during the nighttime. As plant material dies, bacterial decomposition lowers DO through absorption.

Streams dominated with submerged macrophytes experience the largest swings in DO and pH (Wilcox R. a., 2001). Suspended algae in the water column (often measured as chlorophyll-a) also produce these effects. In some cases, oxygen production leads to extremely high levels of oxygen in the water (supersaturation), which can cause gas bubble disease in fish. The wide daily fluctuations in DO caused by excess plant growth and algae are also correlated to degradation of aquatic communities (Heiskary, 2013). Increasing primary production due to elevated nutrients can change plant species composition and cause proximate impacts to stream biology by altering food resources, altering habitat structure, or by algal toxins, in addition to higher risk for low DO situations. More information on the effects of phosphorus and related eutrophication issues can be found on EPA's CADDIS nutrients webpage.

Water quality standards

The Minnesota Pollution Control Agency (MPCA) has developed standards for river eutrophication designed to protect aquatic life (Heiskary, 2013). River eutrophication criteria were developed for three geographic regions (Table 1). The standard is a combination of a maximum total phosphorus (TP) concentration and at least one of three related stressors above its threshold.

Table 1. River eutrophication criteria ranges by river nutrient region for Minnesota

Region	TP µg/L	Related stressor		
		Chl-a µg/L	DO flux mg/L	BOD_5 mg/L
North	≤ 50	≤ 7	≤ 3.0	≤ 1.5
Central	≤ 100	≤ 18	≤ 3.5	≤ 2.0
South	≤ 150	≤ 35	≤ 4.5	≤ 3.0

Types of eutrophication data

Water samples are collected from streams and rivers throughout the state. The most common data are for TP, though orthophosphorus samples are collected in some cases. Related stressor parameters – chl-a, DO flux, five-day Biochemical Oxygen Demand (BOD_5) – are analyzed in conjunction with TP to understand potential impacts and connections.

Sources and causal pathways for eutrophication

Phosphorus is delivered to streams by wastewater treatment facilities, urban stormwater, agricultural runoff, upstream eutrophic lakes, and direct discharges of sewage. Phosphorus bound to sediments in the river channel can contribute to concentrations. Orthophosphorus is the form of phosphorus that is readily available for plant and algal uptake, and can influence excess algae suspended in the water column and submerged aquatic macrophyte growth. While orthophosphates occur naturally in the environment, river and stream concentrations may become elevated with additional inputs from wastewater treatment plants, noncompliant septic systems, and fertilizers in urban and agricultural runoff. The causes and potential sources for excess phosphorus are modeled at EPA's CADDIS Phosphorus webpage.

Flow alteration

Flow alteration is the change of a stream's flow volume and/or flow pattern caused by anthropogenic activities, including:

- Channel alteration
- Water withdrawals
- Land cover alteration
- Wetland drainage
- Agricultural tile drainage
- Impoundments

Changes in landscape vegetation, pavement, and drainage can increase how fast rainfall runoff reaches stream channels. This creates a stronger pulse of flow, followed later by decreased baseflow levels. According to the authors of a review on flow effects (Poff, 1997), "Streamflow quantity and timing are critical components of water supply, water quality, and the ecological integrity of river systems. Indeed, streamflow, which is strongly correlated with many critical physicochemical characteristics of rivers, such as water temperature, channel geomorphology, and habitat diversity, can be considered a 'master variable'..."

Reduced flow or baseflow reduction

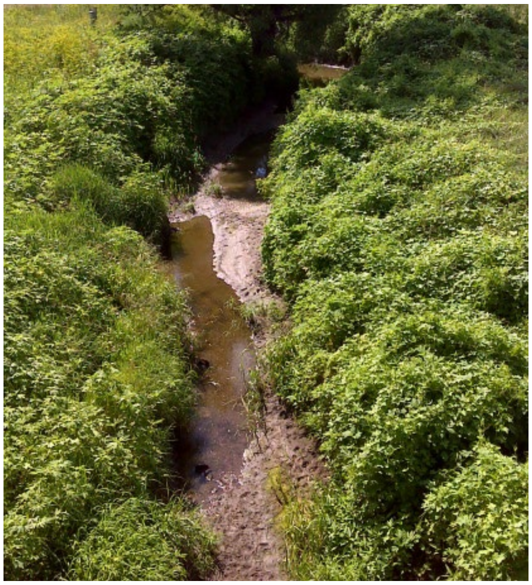
Fish and macroinvertebrate species have many habits and traits that can be either helpful or detrimental in different flow conditions. Across the conterminous United States, (Carlisle et al, 2011) found a strong correlation between diminished streamflow and impaired biological communities. Habitat availability can be scarce when flows are interrupted, low for a prolonged duration, or extremely low, leading to decreased wetted width, cross sectional area, and water depth. Flows that are reduced beyond normal baseflow decrease living space for aquatic organisms and competition for resources increases.

Pollutant concentrations can increase when flows are lower than normal, increasing the exposure dosage to organisms. Tolerant organisms can out-compete others in such limiting situations and will thrive. Low flows of prolonged duration lead to macroinvertebrate and fish communities comprised of generalist species or that have preference for standing water (U.S.EPA, 2012).

Changes in fish community composition are affected by species' differences in spawning behavior (Becker, 1983), flow velocity preference (Carlisle et al, 2011), and body shape (Blake, 1983). When baseflow is reduced, nest-guarding fish species increase and simple nesters, which leave eggs unattended, are reduced (Carlisle et al, 2011). Nest guarding increases reproductive success by protecting eggs from predators and providing "continuous movement of water over the eggs, and to keep the nest free from sediment" (Becker, 1983). Active swimmers, such as the green sunfish, contend better under low velocity conditions (Carlisle et al, 2011).

Dewson et al. (2007) found the low-flow effects on macroinvertebrates were complex, and not easy to generalize. More often, the behavior called drift (using the current to be transported to a new location) increased. Many studies reported that species composition changed, and taxonomic richness generally decreased in streams experiencing prolonged low flows. Those invertebrates that filter food particles from the water column have shown negative responses to low flows. The EPA's CADDIS website (U.S.EPA, 2012) lists the responses of reduced flow as lower total stream productivity, elimination of large fish, changes in taxonomic composition of fish communities, fewer migratory species, fewer fish per unit area, and more-concentrated aquatic organisms, potentially benefiting predators.

Figure 4. Stream with lack of adequate flow



Increased flow or channelization

Increasing surface water runoff and seasonal variability in stream flow have the potential for both indirect and direct effects on fish populations (Schlosser, 1990).

Direct effects include:

- Decreased survival of early life stages
- Potentially lethal temperature and oxygen stress on adult fish (Bell, 2006)

Indirect effects include:

- Alteration in habitat suitability
- Nutrient cycling
- Production processes
- Food availability
- Increases in erosion

When flows increase, bank and bottom scouring can occur, which deepens the channel so that higher flows are now contained and no longer spill out into the floodplain. This increases the flow volume and erosive energy of water, leading to consequence such as excess sediment smothering habitat. High flows and the associated increased flow velocities can displace fish and macroinvertebrates downstream, and move habitat features like woody debris out of the stream. Woody debris and other habitat features are important as flow refugia for fish and living surfaces for clinging invertebrates. Macroinvertebrate types may shift from those species having long life cycles to shorter ones, because these species can complete their life cycle within the bounds of the recurrence interval of the elevated flow conditions (U.S.EPA, 2012). Fish species that have streamlined body forms experience less drag under high velocities and will have advantage over non-streamlined fish species (Blake, 1983).

Increased flows may directly impair the biological community or may contribute to additional stressors. Increased channel shear stresses, associated with increased flows, often cause increased scouring and bank destabilization. With these stresses added to the stream, the fish and macroinvertebrate community may be influenced by the negative changes in habitat and sediment dynamics. To learn more about flow alteration as a stressor go to the EPA CADDIS webpage here.

Water quality standards

There currently is no applicable standard for flow alteration. The standard for minimum streamflow, according to Minn. Stat. § 7050.0210, subp. 7 is:

> Point and nonpoint sources of water pollution shall be controlled so that the water quality standards will be maintained at all stream flows that are equal to or greater than the $7Q_{10}$ [the lowest streamflow for 7 consecutive days that occurs on average once every 10 years] for the critical month or months, unless another flow condition is specifically stated as applicable in this chapter.

Types of flow alteration data

Stream gaging stations are located throughout Minnesota. The outlet of each major watershed has a permanent gage that collects important long-term flow information. There are also stations at smaller scales within each watershed. The stations have differing lengths of monitoring history. If there is sufficient monitoring data, detailed hydrologic analysis can be used to help analyze for flow alteration. In addition, hydrologic models can be used to predict flows in a watershed or subwatershed when measured data are not available. An indirect determination of flow alteration can be found via geomorphological measurements, as channel form and dimensions are related to flow volumes. Information regarding the extent of tile drainage, altered watercourses, wetland storage, and water usage (surface and groundwater) are also used to analyze for flow alteration impacts in a watershed.

Sources and causal pathways model for flow alteration

The conceptual model for flow alteration can be found on the EPA webpage. The causes and potential sources for altered flow are modeled at EPA’s CADDIS Flow Alteration webpage.

Total suspended solids

Sediment and turbidity are among the leading pollutant issues affecting stream biological impairment in the United States (U.S.EPA, 2012). Recent studies in Minnesota have demonstrated that human activities on the landscape have dramatically increased the sediment entering streams and rivers since European settlement (Triplet, 2009) and (Engstrom, 2009). Sediment can come from land surfaces such as exposed soil or from unstable streambanks. The soil may be unprotected for a variety of reasons, such as construction, mining, agriculture, or insufficiently vegetated pastures. Human actions on the landscape, such as channelization of waterways, riparian land cover alteration, and increased impervious surface area can cause stream bank instability leading to sediment input from bank sloughing. Although sediment delivery and transport are an important natural process for all stream systems, sediment imbalance (either excess suspended sediment or lack of sediment) can be detrimental to aquatic organisms.

Figure 5. Streambank failure contributing sediment

As described in a review by Waters (1995), excess suspended sediments harm aquatic life through two major pathways:

- Direct physical effects on biota such as abrasion of gills, suppression of photosynthesis, and avoidance behaviors
- Indirect effects such as loss of visibility and increase in sediment oxygen demand

Elevated total suspended solids (TSS) concentrations can reduce the penetration of sunlight and can thwart photosynthetic activity and limit primary production (Munawar et al, 1991); (Murphy et al, 1981). Sediment can also increase water temperature, as darker (turbid) water will absorb more solar radiation.

Organic particles, including algae, can also contribute to TSS. Determining the type of suspended material (mineral vs organic) is important for proper conclusions about the stressor and source (erosion vs. nutrient enrichment vs. a wastewater discharge). Elevated Total Suspended Volatile Solids TSVS concentrations can impact aquatic life in a similar manner as suspended sediment, with the suspended particles reducing water clarity. Unusually high concentrations of TSVS can indicate excess nutrients (causing algal growth) and an unstable DO regime. More information on sediment impacts can be found on EPA’s CADDIS sediment webpage.

Water quality standards

The new TSS standard in Minnesota is stratified by geographic region and stream class due to differences in natural background conditions resulting from varied geology and biological sensitivity. There is currently no standard for TSVS in Minnesota.

Table 2. TSS standard concentrations by region

Region	TSS mg/L	Secchi tube surrogate*
North	15	40 cm
Central	30	25 cm
South	65	10 cm
Coldwater	10	55 cm

*shown here for comparison to the TSS standard

Types of suspended sediment data

Particles suspended in the water column can be either organic or mineral. Generally, both are present to some degree and measured as TSS. TSS is determined by collecting a stream water sample, filtering it, and weighing it to determine the concentration of particulate matter in the sample. To determine the mineral component of the suspended particles, a second test is run using the same procedure except the organic material is burned off in an oven before weighing the remains, which are only mineral material.

Secchi tubes can also be used to help understand suspended sediment concentrations in streams. The secchi surrogate values for TSS are shown in Table 2.

Sources and causal pathways model for suspended sediment

High TSS occurs when heavy rains fall on unprotected soils, dislodging the soil particles, which are transported by surface runoff into the rivers and streams (MPCA and MSUM, 2009b). The soil may be unprotected for a variety of reasons, such as construction, mining, agriculture, or insufficiently vegetated land. Decreases in bank stability may also lead to sediment loss from the streambanks, often caused by perturbations in the landscape such as channelization of waterways, riparian land cover alteration, increases in impervious surfaces, and tile drainage. Part of the increased TSS at higher flows is due to resuspension of deposited sediment, which will be worse when streams have received excess sediment from banks and uplands.

Rangeland and pasture are also common landscape features in Minnesota. In some areas, the riparian corridor has been cleared for pasture and is heavily grazed, resulting in a riparian zone that lacks deep-rooted vegetation necessary to protect streambanks and provide shading. Exposures of these areas to weathering, trampling, and shear stress (water friction) from high flow events can increase the quantity and severity of bank erosion. The same effects can occur when residential yards are maintained to the edge of the stream channel. Additional causes and potential sources for increases in sediment are modeled at EPA's CADDIS Sediments webpage.

Physical habitat

Habitat is a broad term encompassing all aspects of the physical, chemical, and biological conditions needed to support a biological community. This section will focus on the physical habitat structure including geomorphic characteristics and vegetative features (Griffith, Rashleigh, & Schofield, 2010). Physical habitat is often interrelated to other stressors such as sediment, flow and dissolved oxygen.

Excess fine sediment deposition on benthic habitat has been proven to adversely impact fish and macroinvertebrate species that depend on clean, coarse stream substrates for feeding, refuge, and/or reproduction (Newcombe & MacDonald, 1991)

Aquatic macroinvertebrates are generally affected in several ways:

- Loss of certain taxa due to changes in substrate composition (Erman, 1988)
- Increase in "drift" (an avoidance behavior) due to sediment deposition or substrate instability (Rosenberg & Wiens, 1978)
- Reductions in the quality and abundance of food sources such as periphyton and other prey items (Peckarsky, 1984)

Fish communities are typically influenced through:

- Reduction in spawning habitat or egg survival (Chapman, 1988)
- Reduction in prey items as a result of decreases in primary production and benthic productivity (Bruton, 1985); (Gray & Ward, 1982)

Figure 6. Brook trout and clean, course substrate

Specific habitats needed by a healthy biotic community can be minimized or altered by practices on the landscape by way of resource extraction, agriculture, forestry, urbanization, and industry. These landscape alterations can lead to reduced habitat availability, such as decreased riffle habitat, or reduced habitat quality, such as embedded gravel substrates. Biotic population changes can result from decreases in availability or quality of habitat by way of altered behavior, increased mortality, or decreased reproductive success (Griffith, Rashleigh, & Schofield, 2010). Fish species that are simple lithophilic spawners require clean, coarse substrate for reproduction. These fish do not construct nests for depositing eggs, but rather broadcast them over the substrate. Eggs often find their way into interstitial spaces among gravel and other coarse particles in the streambed. Increased sedimentation can reduce reproductive success for simple lithophilic spawning fish because eggs become smothered by sediment and become oxygen deprived. Habitat can also be affected through direct stream projects like removing large woody debris from stream channels, which used to be a common practice. Large woody debris is important in creating habitat by causing scour pools, providing cover for fish, creating pockets of protection from faster currents, and providing a living surface for macroinvertebrates that cling to hard objects (Gurnell, 1995); (Cordova, 2006); (Magilligan F.J., 2008).

Degraded physical habitat is a leading cause of impairment in streams on 303(d) lists. According to the EPA CADDIS website, a stream provides six main features of physical habitat structure:

1. Stream size and channel dimensions
2. Channel gradient
3. Channel substrate size and type
4. Habitat complexity and cover
5. Vegetation cover and structure in the riparian zone
6. Channel-riparian interactions

Just like for terrestrial settings and those animals, aquatic population and community changes can result from decreases in availability or quality of habitat by way of altered behavior, increased mortality, or decreased reproductive success (U.S.EPA, 2012). To learn more about physical habitat go to the EPA CADDIS webpage here.

Water quality standards

Since habitat is a physical measurement, there is no water quality standard. Other measures are used to understand physical habitat limitations.

Types of physical habitat data

MPCA biological survey crews conduct a qualitative habitat assessment using the MPCA Stream Habitat Assessment (MSHA) protocol for stream monitoring sites. The MSHA protocol can be found here. MSHA scores can be used to review habitat conditions such as channel development, depth variability, and substrate types and conditions. These habitat metrics can then be compared for similar streams.

MPCA and Minnesota Department of Natural Resources (DNR) partners are collecting stream channel dimension, pattern and profile data at select stream locations of various sizes and biological condition. These data can be used to compare channel departure from a reference condition. Habitat features can be analyzed to determine if a stream is lacking pool depth, pool spacing, adequate cross sectional area to convey discharge, and various other physical habitat features. The applied river morphology method created by (Rosgen, 1996) is the accepted method of data collection by the MPCA and DNR.

Deposited sediment is visually estimated by measuring the degree to which fine material surrounds rock or woody substrate within the channel (embeddedness). Deposited sediment is also analyzed by randomly measuring numerous substrate particles (pebble count) and calculating the 50th percentile particle size (D_{50}).

Sources and causal pathways for physical habitat

Alterations of physical habitat, defined here as changes in the structural geomorphic or vegetative features of stream channels, can adversely affect aquatic organisms. Many human activities and land uses can lead to myriad changes in in-stream physical habitat. Mining and resource extraction, agriculture, forestry, urbanization, and industry can contribute to increased sedimentation, via increased erosion for example, and changes in discharge patterns, such as increased stormwater runoff and point effluent discharges. These land use activities can also lead to decreases in streambank habitat and instream cover, including large woody debris. See the Sediment and Flow modules for more information on sediment- and flow-related stressors.

Direct alteration of streams channels also can influence physical habitat, by changing discharge patterns, changing hydraulic conditions (water velocities and depths), creating barriers to movement, and decreasing riparian habitat. These changes can alter the structure of stream geomorphological units, such as increasing the prevalence of run habitats, decreasing riffle habitats, and increasing or decreasing pool habitats.

Typically, physical habitat degradation results from reduced habitat availability, such as decreased snag and riffle habitats, or reduced habitat quality, such as increased fine sediment cover. Bedded sediments are closely related to suspended sediments. Decreases in bank stability lead to sediment loss from the streambanks, causing sediment loads in the water column, and deposition on the streambed. Bank instability is often caused by perturbations in the landscape such as channelization of waterways, riparian land cover alteration, and increases in impervious surfaces. Decreases in habitat availability or habitat quality may contribute to decreased condition, altered behavior, increased mortality, or decreased reproductive success of aquatic organisms. Ultimately, these effects may result in changes in population and community structure and ecosystem function. The narrative and conceptual model can be found on the EPA CADDIS webpage here.

Connectivity

Connectivity in river ecosystems generally refers to how water features are linked to each other on the landscape or how locations within a stream are connected. Connectivity also pertains to locations adjacent to a stream, such as a stream's connectivity to its floodplain. These different types of connectivity affect biology differently, do not often produce the same effects, and often times are linked to other stressors like habitat.

Longitudinal connectivity or fish passage

Humans can alter the degree of connectivity within stream systems. In Minnesota, there are more than 800 dams on streams and rivers for a variety of purposes, including flood control, maintenance of lake levels, wildlife habitat, and hydroelectric power generation. Dams change stream habitat by altering streamflow, water temperature, and sediment transport (Cummins M.J., 1979), (Waters, 1995). Dams also directly block seasonal fish migration for reproduction and overwintering. Disrupted migration not only alters reproduction of fish; it also impacts mussel species that utilize fish movement to disperse their offspring. Structures, such as dams, have been shown to reduce species richness of systems, while also increasing the abundance of tolerant or undesirable species (Winston, 1991), (Santucci V.A., 2005).

Figure 7. A perched culvert disrupting flow and fish passage in a stream

DNR has conducted numerous dam removal projects in recent years which have demonstrated benefits to fish populations. A more detailed presentation of the effects of dams on water quality and biological communities can be found in the DNR publication "Reconnecting Rivers: Natural Channel Design in Dam Removals and Fish Passage" (Aadland, 2010).

Culverts at road crossings can also be significant barriers to fish passage if they are installed or sized incorrectly. Culverts can be perched above the downstream water level, have too high an angle, resulting in high velocity flow which many species cannot traverse, or be undersized for the stream size, which also results in high velocity within the culvert. An excellent review of studies regarding culvert impacts to fish migration, including information specifically from Minnesota, has been conducted by the Minnesota Department of Transportation (MNDOT, 2013).

Lateral and floodplain connectivity

Lateral connectivity represents the connection between a river and its floodplain. The degree to which lateral connectivity exists is both a time-dependent phenomenon (Tockner, 1999) and dependent upon the physical structure of the channel. Rivers are hydrologically dynamic systems where their floodplain inundation relates to prevailing hydrologic conditions throughout the seasons. Riverine species have evolved life history characteristics that exploit flood pulses for migration and reproduction based on those seasonally predictable hydrologic conditions that allow streams to access their floodplains (Weclomme 1979, McKeown 1984, Scheimer 2000). When a stream system degrades to a point where it can no longer access its floodplain, the system's capacity to dissipate energy is lost. Without dissipation of energy through floodplain access, sheer stress on streambanks builds within the channel causing channel widening. Channel widening reduces channel stability and causes loss of integral habitat that in turn reduces biotic integrity of the system until the stream can reach a state of equilibrium once again. These changes can be connected to other stressors, such as suspended sediment and habitat.

Water quality standards

There is no applicable water quality standard for connectivity impacts, though new design guidelines for culverts have been developed by the MNDOT for fish passage.

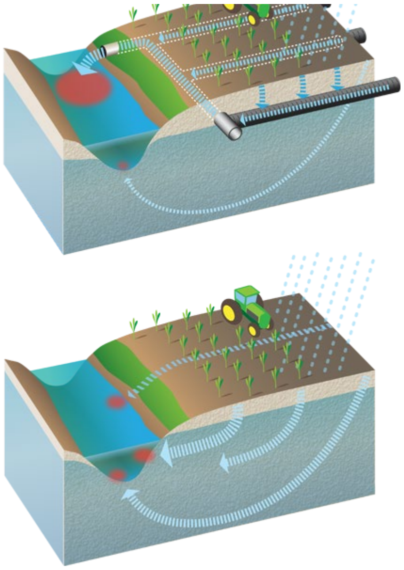
Types of physical connectivity data

Locations for dams are available from the DNR GIS coverage. Culvert surveys are conducted by the MPCA and DNR to determine their passage capability. Stream survey data can also indicate the degree of incision which shows the degree of floodplain connectivity. With high degree of incision there is an associated high rate of bank failure due to increased sheer stress.

Nitrate-nitrogen

Nitrate (NO_3) and nitrite (NO_2) forms of nitrogen are components of the natural nitrogen cycle in aquatic ecosystems. NO_2 anions are naturally present in soil and water, and are readily converted to NO_3 by microorganisms as part of the nitrification process of the nitrogen cycle. As a result, nitrate is far more abundant than nitrite. Although the water test commonly used measures both nitrate and nitrite, because a large percentage is nitrate, this report will refer to these data as being nitrate. Nitrogen is commonly applied as a crop fertilizer. Nitrogen transport pathways can be different depending on geology and hydrology of the watershed. When water moves quickly through the soil profile, as in the case of watersheds with karst geology or in heavily tiled watersheds, nitrate transport can become significant.

Figure 8. Examples of how nitrate moves in a watershed



Apart from its function as a biological nutrient, some levels of nitrate can become toxic to organisms. Nitrate toxicity depends on concentration and exposure time, as well as the sensitivity of the individual organisms. The intake of nitrate by aquatic organisms converts oxygen-carrying pigments into forms that are unable to carry oxygen, thus inducing a toxic effect on fish and macroinvertebrates (Grabda et al, 1974). Certain species of caddisflies, amphipods, and salmonid fishes seem to be the most sensitive to nitrate toxicity according to Camargo and Alonso (2006), who cited a maximum level of 2.0 mg/L nitrate N as appropriate for protecting the most sensitive freshwater species and nitrate-N concentrations under 10.0 mg/L to protect several other sensitive fish and aquatic invertebrate taxa. For more information of nutrients like nitrate, see EPA’s nutrients webpage.

Water quality standards

Water quality standards for nitrate (aquatic life) are still in development. The draft proposed nitrate criteria for protection of aquatic life include an acute value (maximum standard) of 60 mg/L N:NO3 for a one-day duration concentration for all Class 2 waters. Additionally, the draft chronic values are 8 mg/L N:NO3 mg/L for Class 2B (warmwater) and 5 mg/L N:NO3 for Class 2A (coldwater) for concentrations based on a four-day duration. For more details see: Aquatic Life Water Quality Standards Technical Support Document for Nitrate (state.mn.us)

Minnesota’s Class 1 waters, designated for domestic consumption (drinking water), have a nitrate water quality standard of 10.0 mg/L (Minn. Stat. 7050.0222 subp. 3).

Types of nitrate-nitrogen data

Stream and river water samples are collected at various locations throughout the watershed and often include nitrogen analysis. Samples are sent to a state certified laboratory and analyzed for a number of water quality parameters including nutrients. Laboratory analytical data is then stored in the EQuIS database and can be accessed via the MPCA webpage.

Sources and causal pathways for nitrate-nitrogen

Nitrogen is commonly applied as a crop fertilizer, predominantly for corn. A statewide nitrogen study found that cropland commercial fertilizers make up 47% of nitrogen added to the landscape, 21% occurs through cropland legume fixation, 16% from manure application, and 15% from atmospheric deposition (MPCA, 2013). These land applications can reach waterways through surface runoff, tile drainage, and leaching to groundwater, with tile drainage being the largest pathway (MPCA, 2013). Other nitrogen sources are non-compliant septic systems and municipal wastewater discharges. For more information on the sources and effects of nitrate, see the EPA's CADDIS nutrient webpage.

Ammonia (NH_3)

Figure 9. Manure spreading on a field

Ammonia is found in an ionized form (ammonium, NH_4^+) and the un-ionized form (ammonia, NH_3), with NH_4^+ being the prevalent form in natural waters. Ammonia is converted to nitrate in the natural nitrogen cycle. An increase in water temperature and/or pH increases the unionized ammonia (NH_3) concentration, which is toxic to aquatic organisms at certain concentrations. The fraction of NH_3 is not directly measured, but instead is calculated using measures of total ammonia, pH, and temperature. Many human activities can contribute to elevated ammonia concentrations in streams. Sources of ammonia (NH_3) include human and animal waste, fertilizers, and natural chemical processes. Channel alteration can result in decreased natural conversion of ammonia to nitrate, and alteration or removal of riparian vegetation can reduce the interception of nitrogen compounds in runoff from the surrounding landscape. Channel alteration and water withdrawals can reduce ammonia volatilization by reducing the turbulence of the water. For more information on the causes and effects of ammonia, see EPA's CADDIS ammonia website.

Water quality standards

The unionized ammonia-N (NH_3) standard for Class 2A (coldwater) and Class 2B (warmwater) streams is 0.016 mg/L and 0.040 mg/L respectively. Most of NH_3+NH_4 are in the NH_4 form, but a shift toward increased NH_3 happens with higher temperatures and pH concentrations.

Types of ammonia data

The MPCA collects grab samples for ammonium and has them analyzed at state-certified labs. The concentration of ammonia in water is measured as total ammonia and is reported in mg/L. The fraction of unionized ammonia (NH_3) is not directly measured, but instead is calculated using measures of total ammonia, pH, temperature, and specific conductivity. The EPA CADDIS webpage has a calculator available for calculating unionized ammonia; it is located here.

Sources and causal pathways for ammonia

Many human activities and associated sources can contribute to high ammonia concentrations in aquatic systems, which can lead to lethal and sub-lethal effects on aquatic organisms. Channel alteration can result in decreased nitrogen uptake within the stream, while decreases in riparian and watershed vegetation associated with agriculture and urbanization can reduce nitrogen uptake in the surrounding landscape. Channel alteration and water withdrawals can reduce ammonia volatilization due to changes in water velocities and depths. Sources associated with agriculture, urbanization, industry and aquaculture also can directly increase ammonia inputs to aquatic systems via four main transport pathways (or transport-defined sources): stormwater runoff, leakage or leachate into groundwater sources, atmospheric emissions and deposition, or direct effluent discharges (U.S.EPA, 2012). For a more detailed explanation of ammonia, sources and pathways visit EPA's CADDIS Ammonia webpage.

Specific conductance and ionic strength

Figure 10. Road salt application

Specific conductance refers to the collective amount of ions in the water. In general, the higher the level of dissolved minerals in water, the more electrical current can be conducted through that water. Dissolved salts and minerals occur naturally in surface waters, and biota are adapted to a natural range of ionic strengths. Aquatic organisms maintain a careful water and ion balance, and can become stressed by an increase in ion concentrations (SETAC (Society of Environmental Toxicology and Chemistry), 2004). Ions of many elements, such as calcium, sodium, and magnesium are necessary for aquatic health, but imbalances can be toxic. One ion that is primarily a human contribution to surface waters is chloride. The negative effects of elevated chloride concentrations on aquatic life have been well documented. The use of road salt and de-icing products has increased considerably in the United States since 1950, putting more urban streams at risk for this stressor. Studies around the country have found that as salt levels increase, streams begin to lose their most sensitive species. Caddisfly, stoneflies, and mayflies are among the first things to disappear. Amphibians and fish follow shortly after. For more information on the causes and effects of ionic strength, see EPA's Ionic Strength Webpage.

Water quality standards

A standard of 1,000 µmhos/cm at 25 °C exists for Class 4 waters of the state (Minn. Stat. 7050.0224 subp. 2) that is protective of agricultural and irrigation uses, but is not an aquatic life standard.

The chronic standard for chloride in Minnesota is 230 mg/L. The EPA recommended chronic criterion for aquatic life is a four-day average chloride concentration of 230 mg/L with an occurrence interval of once every three years, and the recommended acute criterion concentration for chloride is 860 mg/L.

Types of specific conductance or ionic strength data

Like other water quality parameters, specific conductance readings can be collected by deployed devices at defined time intervals, or a single, instantaneous reading taken during a site visit. Elevated conductivity can serve as a surrogate or indicator for ions in the water, such as chloride or sulfate.

Stream and river water samples are collected at various locations throughout the watershed and can include chloride analysis. Specific conductance can also be used as a surrogate measure for chloride or other ions. Laboratory analytical data is then stored in the EQuIS database and can be accessed via the MPCA webpage here.

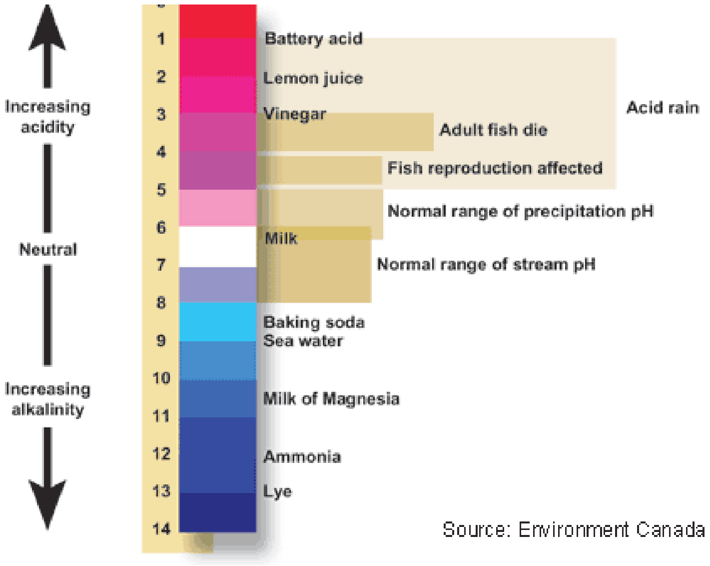
Sources and causal pathways for specific conductance and ionic strength

Industry runoff and discharges, road salt, urban stormwater drainage, agricultural drainage, wastewater treatment plant effluent, and other point sources can increase ions in downstream waters. The causes and potential sources for ionic strength are modeled at EPA's CADDIS Ionic strength webpage.

pH

Acidity is measured on a scale called pH, ranging from 0 to 14. A pH of 7 is considered neutral; less than 7 is acidic, and greater than 7 is basic. Some geological material produces naturally high hydrogen ions that can leach into surface water. Photosynthesis from unnaturally abundant plants or algae removes carbon dioxide from the water, causing a rise in pH. As pH increases, unionized ammonia (the toxic form of ammonia) increases, and can reach toxic concentrations (U.S.EPA, 2012). High or low pH effects on biology include decreased growth and reproduction, decreased biodiversity, and damage to skin, gills, eyes, and organs. Values of pH outside the range of 6.5 – 9 or highly fluctuating values are stressful to aquatic life (U.S.EPA, 2012). For additional information on pH as a stressor, see EPA's CADDIS pH webpage.

Figure 11. pH ranges of common liquids



Source: Environment Canada

Water quality standards

The pH standard for Class 2B (warmwater) streams is within the range of 6.5 as a daily minimum and 9.0 as a daily maximum (MN Statute 7050.0222 subp. 4).

The pH standard for Class 2A (coldwater) streams is within the range of 6.5 as a daily minimum and 8.5 as a daily maximum (Minn. Stat. 7050.0222 subp. 4).

Types of pH data

Like DO, pH readings can be collected by deployed devices at defined time intervals, or a single, instantaneous reading taken during a site visit.

Sources and causal pathways for pH

Human effects on pH values can result from agricultural runoff, urbanization, and industrial discharges. Some geology has naturally high hydrogen ions that can leach into surface water, but it would be rare for this to be the only cause. Photosynthesis of overabundant macrophytes and algae can remove carbon dioxide from the water, causing a higher pH. Effects on biology include decreased growth and reproduction, decreased biodiversity, and damage to skin, gills, eyes, and organs. Concentrations of nutrients (especially nitrogen) also play a significant part in pH dynamics, as nitrification and respiration both produce hydrogen ions (U.S.EPA, 2013). The conceptual model for pH as a candidate stressor is modeled at EPA's CADDIS pH webpage.

Pesticides

A pesticide is defined by the EPA as, “any substance intended for preventing, destroying, repelling or mitigating any pest.” In this document, pesticides refer to fungicides, insecticides, and herbicides used to control various pests.

Figure 12. Pesticide application in a farm field.

Courtesy of EPA Region 9

Herbicides are chemicals used to control undesirable vegetation. The most widespread application of herbicides occurs in row-crop farming, usually during an early growth stage (often in June) to reduce the competition for water and nutrients from weeds. They may also be applied before crop emergence, a second time during the growing season, and pre-harvest. In suburban and urban areas, herbicides are applied to lawns, parks, golf courses, and other areas.

Herbicides are also applied to water bodies to control aquatic weeds that impede irrigation withdrawals or interfere with recreational and industrial uses of water (Folmar, Samders, & Julin, 1979). To learn more about herbicides, their applications, along with associated biological problems, refer to the EPA’s CADDIS herbicide website.

Insecticides are chemicals used to control insects. Many insecticides act upon the nervous system of the insect, such as Cholinesterase inhibition, while others act as growth regulators. Insecticides are commonly used in agricultural, public health, and industrial applications, as well as household and commercial uses (e.g. control of roaches and termites). In 2001, the U.S. Department of Agriculture reported that insecticides accounted for 12% of total pesticides applied to the surveyed crops. Corn and cotton account for the largest shares of insecticide use in the United States. To learn about insecticides and their applications, along with associated biological problems, refer to the EPA’s CADDIS insecticide website.

Water quality standards

The MPCA has developed toxicity-based aquatic life standards for four herbicides and one insecticide; the chronic and maximum standards for these pesticides are shown in Table 3.

Table 3. Summary of MPCA surface water standards for pesticides (all units are µg/L)

Pesticide	Chronic Class 2A[1]	Chronic Class 2B	Maximum Standard 2A and 2B
Acetochlor	3.6	3.6	86
Alachlor	3.8	4.2	800
Atrazine	3.4	3.4	323
Chlorpyrifos	0.041	0.041	0.083
Metolachlor	23	23	271

[1] Chronic standards for aquatic organisms are protective for exposure duration of four days

Types of pesticide data

Since 1985, the Minnesota Department of Agriculture (MDA) and Minnesota Department of Health have been monitoring the concentrations of common pesticides in groundwater near areas of intensive agricultural land use. In 1991, these monitoring efforts were expanded to include surface water monitoring sites on select lakes and streams. The MDA annually collects samples from various surface water bodies throughout the state and analyzes those samples for the presence of pesticides and any degradates. The MDA attempts to capture the influence of different land uses on surface water resources. Out of the 100-plus pesticides this program routinely analyzes for, three have been named a "surface water pesticide of concern" in Minnesota – acetochlor, atrazine, and chlorpyrifos. When pesticides are detected at problematic levels, the MDA intensifies its monitoring in that area to locate the source and extent of the problem, so that it can be corrected. To learn more about the MDA pesticide monitoring plan and results, see the MDA monitoring web page.

Sources and causal pathways for pesticides

Background and conceptual models are available on the EPA's CADDIS webpage for herbicides and insecticides.

References

Aadland, L. (2010). *Reconnecting Rivers: Natural Channel Design in Dam Removals and Fish Passage.* Fergus Falls: Minnesota Department of Natural Resources, Stream Habitat Program.

Allan, J. (1995). *Stream Ecology: structure and function of running waters.* Dordrecht, Netherlands: Kluwer Academic Publishers.

Becker, G. C. (1983). *Fishes of Wisconsin1052 pp.* Madison, WI: Univ. Wisconsin Press.

Behnke, R. (1992). *Native Trout of Western North America.* Bethseda, Maryland: American Fisheries Society Monograph 6.

Bell, J. M. (2006, September). The Assessment of Thermal Impacts on Habitat Selection, Growth, Reproduction and Mortality in Brown Trout (Salmo trutta L): A Review of the Literature. *Applied Ecological Services Inc.*, p. 23 pp.

Blake, R. W. (1983). *Fish Locomotion.* London: Cambridge University Press.

Brooker, M. (1981). The impacts of impoundments on the downstream fisheries and general ecology of rivers. *Advances in Applied Biology 6*, 91-152.

Bruton, M. N. (1985). The effects of suspensoids on fish. *Hydrobiologica 125*, 221-242.

Camargo, J., & Alonso, A. (2006). Ecological and toxicological effects of inorganic nitrogen pollution in aquatic ecosystems: a global assessment. *Environmental International 32*, 831-849.

Carlisle, D., Wolcock, D. M., & Meador, M. R. (2011). Alteration of streamflow magnitudes and potential ecological consequences: a multiregional assessment. *Front Ecol Environ (10)*, 264-270.

Chapman, D. (1988). Critical review of variables used to define effects of fines in reds of large salmonids. *Transactions of the American Fisheries Society 117*, 1-24.

Cordova, J. E.-M. (2006). Quantity, Controls and Functions of Large Woody Debris in Midwestern USA Streams. *River Research and Applications 23(1)*, 21-33.

Cummins M.J., a. K. (1979). Feeding ecology of stream invertebrates. *Annual Review of Ecology and Systematics 10*, 147-172.

Davis, J. (1975). Minimal Dissolved Oxygen Requirements of Aquatic Life with Emphasis on Canadian Species: A Review. *Journal of the Fisheries Reasearch Board of Canada*, 2295-2331.

Dewson, Z. a. (2007). A review of the consequences of decreased flow for instream habitat and macroinvertebrates. *Journal of the North American Benthological Society 26(3)*, 401-415.

Doudoroff, P. a. (1965). Dissolved oxygen requirements of fishes. *Biological Problems in Water Pollution: Transactions of the 1962 Seminar. 999-WP-25.* Cincinatti, Ohio: Taft Sanitary Engineering Center, U.S. Public Health Service, Health Service Publication.

Dowling, D. a. (1986). *The effects of dissolved oxygen, temperature, and low stream flow on fishes: A literature review.* Illinois Natural History Survey, Champagne, IL.: Aquatic Biology Section Technical Report.

Elliott, J., & J.A., E. (1995). The effect of the rate of temperature increase on the critical thermal maximum for parr of Atlantic salmon and brown trout. *Journal of Fisheries Biology*, 47,917.

Engstrom, D. J. (2009). Historical changes in sediment and phosphorus loading to the upper Mississippi River:Mass-balance reconstructions from the sediments of Lake Pepin. *Journal of Paleolimnology 41(4)*, 563-588.

EPA. (1986). *Quality Criteria for Water 1986.* Washington, D.C.: Office of Water Regulations and Standards.

EPA, U. (1988). Ambient Water Quality Criteria for Chloride.

Erman, D. a. (1988). Effects of discharge fluctuation and the addition of fine sediment on stream fish and macroinvertebrates below a water filtration facility. *Environmental Management*, 85-97.

Flick, W. (1991). Brook Trout. In J. S. Schnell, *The wildlife series: Trout* (pp. 196-207). Harrisburg, PA: Stackpole Books.

Folmar, L. C., Samders, H. O., & Julin, A. M. (1979). Toxicity of the herbicide glyphosate and several of its formulations to fish and aquatic invertebrates. *Archives of Environmental Contamination and Toxicology*, 8:269-278.

Grabda, E., Einszporn-Orecka, T., Felinska, C., & Zbanysek, R. (1974). Experimental methemoglobinemia in trout. *Acta Ichtyol.*, 4,43.

Gray, L. J., & Ward, J. V. (1982). Effects of sediment releases from a reservoir on stream macroinvertebrates. *Hydrobiologica 96*, 177-184.

Griffith, M. B., Rashleigh, B., & Schofield, K. (2010). *Physical Habitat.In USEPA Causal Analysis/ Diagnosis Decision Information System (CADDIS)*. Retrieved 02 10, 2014, from https://www.epa.gov/caddis-vol2/caddis-volume-2-sources-stressors-responses-physical-habitat

Gurnell, A. K. (1995). The role of coarse woody debris in forest aquatic habitats: Implications for management. *Aquatic Conservation:marine and Freshwater Ecosystems 5(2)*, 143-166.

Hansen, E. A. (1975). Some effects of groundwater on brook trout redds. *Trans. Am. Fish. Soc., 104*(1), 100-110.

Heiskary, e. (2013). *Minnesota Nutrient Criteria Development for Rivers.* St. Paul: Minnesota Pollution Control Agency.

Hinz, L. J. (1997). *Growth and reproduction of juvenile trout in Michigan streams: influence of temperature.* Michigan Department of Natural Resources, Fisheries Research Report No. 2041.

Kauffman, J. &. (1984). Livestock Impacts on riparian ecosystems and streamside management implications: a review. *Journal of Range Management, 37*, 430-438.

Kramer, D. (1987). Dissolved oxygen and fish behavior. *Environmental Biology of Fishes 18(2)*, 81-92.

Magilligan F.J., K. N. (2008). the geomorphic function and charactersitics of large woody debris in low gradient rivers, coastal Maine, USA. *Geomorphology 97*, 467-482.

McCormick, J., Hokanson, K., & Jones, B. (1972). Effects of temperature on growth and survival of young brook trout, Salvelinus fontinalis. *Journal of the Fisheries Research Board of Canada*, 29,1107.

MNDNR. (2014). *Missouri River Watershed Hydrology, Connectivity, and Geomorphology Assessment Report.* Minnesota Department of Natural Resources, Division of Ecological and Water Resources.

MNDOT. (2013). *Culvert Designs for Aquatic Organism Passage: Culvert Design Practices Incorporating Sediment Transport, TRS1302.* St Paul: Minnesota Department of Transportation, Office of Policy Analysis, Research & Innovation, research Services Section.

MPCA. (2009a). *Guidance Manual for assessing the Quality of Minnesota Surface Waters for Determination of Impairment 305(b) Report and 303(d) List.* St. Paul, MN: Minnesota Pollution Control Agency.

MPCA. (2013). *Nitrogen in Minnesota Surface Waters.* St. Paul: Minnesota Pollution Control Agency.

MPCA and MSUM. (2009b). *State of the Minnesota River, Summary of Surface Water Quality Monitoring 2000-2008.* Retrieved from http://mrbdc.wrc.mnsu.edu/reports/basin/state_08/2008_fullreport1109.pdf

Munawar, M. N. (1991). A method for evaluating the impacts of navigationally induced suspended sediments from the Upper Great Lakes connecting channels on the primary productivity. *Hydrobiologia, 219*, 325-332.

Murphy, M. L. (1981). Effects of canopy modification and accumulated sediment on stream communities. *Trans. Am. Fish. Soc., 110*, 469–478.

Nebeker, A. D. (1991). Effects of low dissolved oxygen on survival, growth and reproduction of Daphnia, Hyallella and Gammarus. *Environmental Toxicology and Chemistry*, 373-379.

Newcombe, C. P., & MacDonald, D. D. (1991). Effects of suspended sediments on aquatic ecosystems. *North American Journal of Fisheries Management 11*, 72-82.

Peckarsky, B. (1984). *Predator-prey interactions among aquatic insects, in The Ecology of Aquatic Insects pp 196-254.* NY: Praeger Scientific.

Pekarsky, B. L. (1984). Predator-prey interactions among aquatic insects. In V. H. Resch, & D. M. Rosenberg, *The Ecology of Aquatic Insects* (pp. 196-254). NY: Praeger Scientific.

Poff, N. a. (1997). The Natural Flow Regime: A paradigm for river conservation and restoration. *BioScience 47(11)*, 769-784.

Pringle, C. (2003). What is Hydrologic Connectivity and Why is it Ecologically Important? *Hydrological Processes*, 17:2685-2689.

Raleigh, R. L. (1986). *Habitat suitability index models and instream flow suitability curves:brown trout. Biological report 82.* U.S. Fish and Wildlife Service.

Rosenberg, D., & Wiens, A. (1978). Effect of sediment addition on macrobenthic invertebrates in a northern Canadian river. *Water Research 12*, 753-763.

Rosgen, D. (1996). *Applied River Morphology.* Pagosa Springs, Colorado: Wildland Hydrology.

Santucci V.A., e. (2005). Effects of Multiple Low-Head Dams on Fish, Macroinvertebrates, Habitat, and Water Quality in the Fox River, Illinois. *North American Journal of Fisheries Management*, 25:975-992.

Scheimer, F. (2000). Fish as indicators for the assessment of the ecological integrity of large rivers. *Hydrobiologia*, 422/423,271-278.

Schlosser, I. (1990). Environmental variation, life history attributes, and community structure in stream fishes: implications for environmental management and assessment. *Environmental Management 14*, 621-628.

SETAC (Society of Environmental Toxicology and Chemistry). (2004). *Whole effluent toxicity testing: Ion imbalance.* Pensacola, FL, USA: Technical issue paper.

Tiemann, J., Gillette, D., Wildhaber, M., & Edds, D. (2004). Effects of lowhead dams on riffle-dwelling fishes and macroinvertebrates in a midwestern river. *Transactions of the American Fisheries Society*, 133;705-717.

Tockner, K. a. (1999). Biodiversity along riparian corridors. *Archiv fur Hydrobiologie. Supplementband. Large Rivers*, 11:293-310.

Triplet, L. D. (2009). A whole-basin stratigraphic record of sediment and phosphorus loading to the St. Croix River, USA. *Journal of Paleolimnology 41(4)*, 659-677.

U.S.EPA. (2012a). *CADDIS Volume 2 Sources, Stressors & Responses*. Retrieved 02 11, 2014, from CADDIS Volume 2 Sources, Stressors & Responses: https://www.epa.gov/caddis-vol2 U.S.EPA. (2013). *CADDIS:Sources, Stressors & Responses.* U.S. EPA.

Waters, T. (1995). *Sediment in Streams: Sources, Biological Effects and Control.* Bethseda, MD: American Fisheries Society.

Welcomme, R. (1979). *Fisheries ecology of floodplain rivers.* London, England: Longman.

Wilcox, R. a. (2001). Effects of aquatic macrophytes on physico-chemical conditions of three contrasting lowland streams: a conseauence of diffuse pollution from agriculture. *Water Science and Technology 43(5)*, 163-168.

Wilcox, R. J., & Nagels, J. W. (2001). Effects of aquatic macrophytes on physico-chemical conditions of three contrasting lowland streams: a consequence of diffuse pollution from agriculture? *Water Science and Technology 43(5)*, 163-168.

Winston, M. C. (1991). Upstream exterpation of four minnow species due to damming of a prairie stream. *Transactions of the American Fisheries Society*, 120:98-105.