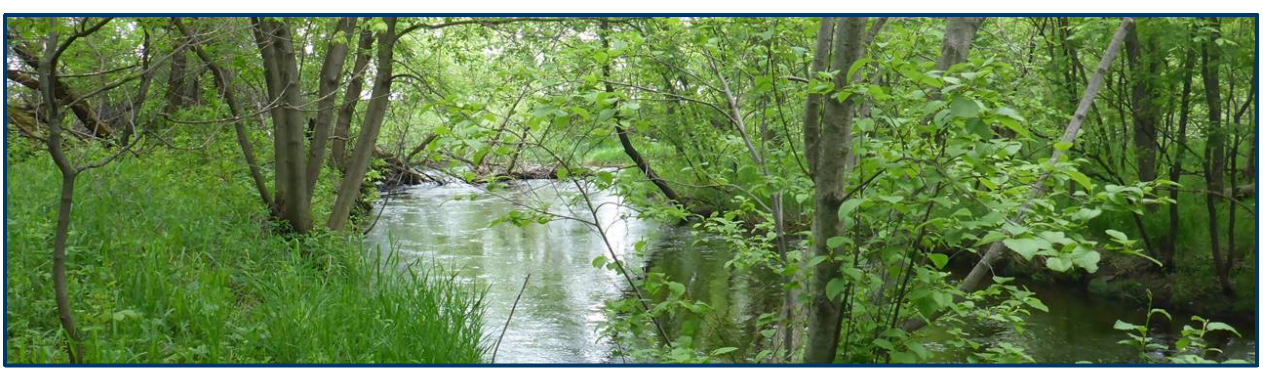
# Instream Flow Incremental Methodology (IFIM) Study for Little Rock Creek, Mississippi River – Sartell Watershed, Minnesota

River Ecology Unit
Inventory, Monitoring, and Analysis Section
Ecological and Water Resources Division, MN DNR
1/7/2021



# Table of Contents

# List of Figures

# List of Tables

# Executive Summary

This study and analysis of the aquatic habitats and potential impacts of groundwater pumping is part of a comprehensive effort to examine water use and associated resource impacts in the Little Rock Creek area with the aim of ensuring sustainable water management. Streamflow has been described as the 'master variable' driving the response of many critical physical, chemical, and biological characteristics of rivers. Groundwater and surface water are closely connected, constituting one resource; impacts to one will impact the other. Headwater streams like Little Rock Creek perform ecological functions that are critical for sustaining fish, fisheries, and ecosystem services throughout their drainage basins.

We used the Instream Flow Incremental Methodology (IFIM), which is a modular decision support system for assessing potential flow management schemes. The IFIM is designed to assist natural resource and water management agencies in comparing the relative impacts of proposed instream flow management schemes. The unique feature of the IFIM is the simultaneous analysis of habitat variability over time and space. This effort relied on the multidisciplinary expertise within our agency to accomplish the analysis in this report: streamflow gaging on Little Rock Creek, MODFLOW groundwater modeling, and estimation of streamflow depletion from groundwater use are all necessary elements of the investigation of surface water impacts on this stream.

Using the IFIM, we assessed the impact of the estimated hydrologic change caused by groundwater pumping on the habitat and ecology of Little Rock Creek. We generated a series of alternative scenarios of streamflow depletion levels using incremental percentages of the August median base flow (ABF); (e.g., 5% ABF, 10% ABF, 15% ABF, and 20% ABF). Analyses of alternative scenarios were used to assess the impacts of groundwater depletion relative to the threshold of ecological harm (>20% change in habitat) and identify viable management levels.

Our key findings are:

- Low flows are significantly reduced by the currently authorized groundwater pumping. The reduction in flows below 75% exceedance is significant and increases as flows decrease. The percent change between reference (without groundwater pumping) and baseline (with groundwater pumping) conditions was greater than 35% in low flow magnitude, frequency, duration, and recurrence.
- The impact of these low flow reductions in Little Rock Creek corresponds to a significant loss of fish habitat.
    - Five of six habitat guilds lose significant (>20%) habitat as flows decrease under the current groundwater pumping levels. Only shallow pool species are relatively unaffected by the decrease in low flows.
    - Habitat decreases greater than 20% occur as flows decrease below the 75% exceedance flow.
- Based on our habitat modeling and review of scientific literature, the current magnitude of habitat loss during August across the majority of habitat types (5/6), equates to ecological harm.
    - Habitat loss and pollution are the primary causes of extirpation of aquatic biota.
- Reductions in streamflow depletion are needed to avoid ecosystem harm.
    - Alternative scenario analyses shows that impacts to all fish habitat types (i.e., ecology) are below the 20% threshold if streamflow depletion remains at or below 10% of the ABF (August median base flow).
    - Impacts exceed the 20% threshold for one species, representing one habitat type (i.e., slow riffles; 21% habitat loss), if diversions are at or below 15% of the ABF.

- As more flow is diverted, equivalent to 20% of the ABF, habitat loss for five of the modeled species, representing four habitat types, exceeds 20%.

On the basis of these findings, the aquatic science literature, and the goal of minimizing ecological harm and working towards water management sustainability, we recommend a Sustainable Diversion Limit (SDL) not to exceed 15% of the August median base flow. A sustainable diversion limit of 10% of the August median base flow would avoid all known risk of ecosystem harm due to streamflow depletion.

The analysis in this report relied on information collected on Little Rock Creek during the 12-year period of record (spanning 2006-2018). Based on the Palmer Drought Severity Index, the measured hydrology of Little Rock Creek during this period can be characterized as wetter than average. It is unknown if this trend will persist in the future. However, the percentage of flow approach to establish an SDL should effectively maintain stream ecology, but may be risky if inter-annual rainfall patterns or change in climate substantially reduces the amount of water in the Little Rock Creek system. Therefore, we recommend continuing to monitor streamflow in Little Rock Creek as a way of ensuring that the findings here remain applicable.

# 1. Introduction

The Minnesota Department of Natural Resources (DNR) is charged with managing water resources to assure an adequate and sustainable supply for multiple uses, including for future generations, and avoidance of ecological harm to aquatic systems. Minnesota has a modified riparian water law system in which landowners have the right to make reasonable use of the abutting surface waters or the groundwater beneath their land, as defined in Statute and Rule and regulated by the DNR's water appropriation permitting program. Water is a public trust resource, and the state grants the right to water beyond personal use – above 10,000 gallons per day or one million gallons per year – through water appropriation permits (MN DNR 2016).

The variability of Minnesota's climate and geography mean that rainfall is not always available in the quantities we need at the times when it is most needed. In recent years, it has become increasingly clear that Minnesota's water resources, while abundant in many areas, are not unlimited (**Figure 1**). In some areas, increasing water withdrawals are using more groundwater than is naturally being recharged (MN DNR 2016). In other areas, groundwater supplies are limited due to the underlying geology. Groundwater contamination is also a limiting factor in many areas.

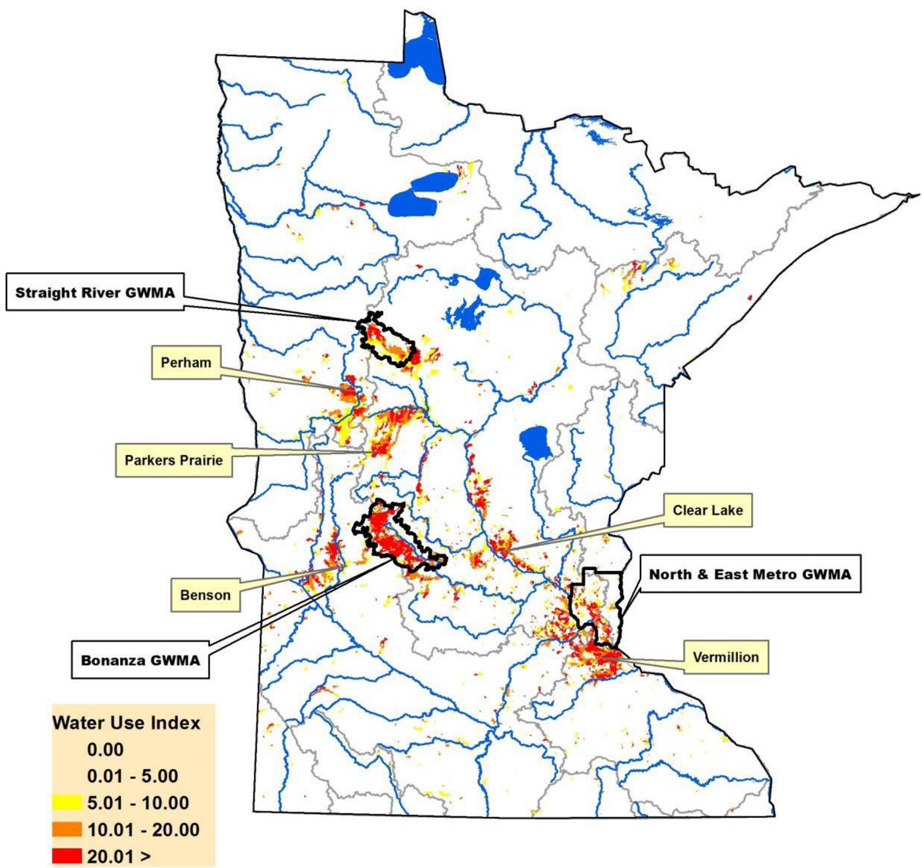



***Figure 1. Catchments showing intensive reported use compared to available water resources.***
*The Water Use Index is a ratio of reported water use to the mean annual surface water discharge from each catchment. A ratio of 20.01 means that more than 20% of the mean annual discharge is being withdrawn. The analysis was completed at the watershed scale of DNR Level 09 catchments (average catchment size = 491 Ac). The index is the average for the years 2007-2011. Note that this ratio is not the same as the suggested stream threshold of 10% of the mean August flow. For further information on how this index is calculated, see DNR's Watershed Health Assessment Framework webpage.*

Increasing demands on both surface water and groundwater supplies can cause negative impacts to the ecosystems and riparian uses of streams, lakes, and wetlands. While water levels fluctuate naturally throughout the year and across multiple years, water appropriations can push low levels lower, significantly reducing streamflows and more frequently putting fish, wildlife, plant communities and riparian uses at risk.

## 1.1 Purpose

The purpose of the analysis described in this report is to help the DNR meet its management responsibilities to ensure sustainable water use for present and future generations and avoid ecological harm. This report summarizes investigations into the habitat and ecological impacts of groundwater withdrawals on Little Rock Creek. This investigation is part of a larger Departmental effort collectively examining streamflows, groundwater use, related depletion of Little Rock Creek flows, and its impacts on the aquatic habitat and ecology of Little Rock Creek.

## 1.2 Need

In 2015, the Little Rock Creek area (LRCA) was identified by the DNR as an area where groundwater use is at increased risk of overuse and contamination (MN DNR 2020a). Groundwater is also the source for nearby public water supplies, private domestic supplies, non-crop agricultural uses, and nearly all agricultural irrigation in the area. Agricultural irrigation is the largest category of water use in the LRCA. Groundwater use for crop irrigation began in the area in the late 1960s, and the number of water appropriation permits in the area has increased steadily for four decades.

There are a number of Minnesota Statutes and Rules that establish the responsibilities of the DNR to manage water use for present and future generations, as well as for fish and wildlife habitat and the avoidance of ecological harm. A partial list is presented **in Table 1**. To meet our responsibilities to future generations and avoid ecological harm to Minnesota's freshwater systems, the amount of water withdrawn from a hydrologic system must be sustainable.

Groundwater is commonly an important source of water for use and is connected to surface waters (Winter et al. 1998, Alley et al. 1999). While the relative contribution of groundwater is variable from one stream to another, hydrologists estimate that average contribution is somewhere between 40 and 50 % in small and medium-sized streams (MN DNR 2020a). Groundwater also is a major source of water to lakes and wetlands. In terms of total freshwater available on the planet, 68.7% is estimated to be stored in polar ice and glaciers and 30.1% is estimated to be stored in groundwater, with the remaining 1.2% in surface and other freshwater (data from Shiklomanov 1993).

Little Rock Creek and its tributary Bunker Hill Creek are designated trout streams that are dependent on a steady supply of groundwater. Groundwater withdrawals divert groundwater discharge to streams (Konikow and Leake 2014). While some diversion of stream discharge is an expected consequence of consumptive groundwater use, the net or cumulative rate of these diversions must remain below sustainable limits. Water resource sustainability involves the use of scientific analysis to balance the ecologic, economic, and social use of water.

Minnesota Statute 103G.287, Subd. 5 states that permitted water use must be sustainable and not "harm ecosystems, degrade water, or reduce water levels beyond the reach of public water supply and domestic wells." Determining the threshold for sustainable use and developing management prescriptions that maintain conditions below this threshold (where use becomes unsustainable and harms the ecosystem) is part of the DNR's responsibility and the purpose of this investigation.

*Table 1. List of Statutes that are critical to water management.*

| MN Statutes | Headnotes |
|---|---|
| 116B.01 and .02. | Environmental rights; no adverse impairment (1) |
| 103G.265. Subdivision 1. | Assurance of supply; to meet long range seasonal requirements (2) |
| 103G.271. Subdivision 3. | No restriction of amount authorized in a permit for agricultural land |
| 103G.285. Subdivision 2. | Surface water appropriations; limits appropriations during periods of specified low flows (3) |
| 103G.287. Subdivision 2. | Groundwater appropriations; relationship to surface water resources (4) |
| 103G.287. Subdivision 3. | Protection of groundwater supplies (4) |
| 103G.287. Subdivision 5. | Sustainability standard |

*Shown are a select list of Statutes that are critical to water management and when taken as a whole, help define ecological and sustainability thresholds. These, and other, statutes and rules identify the need to balance the immediate consumptive water use with long-term needs of Minnesotans and the environment. The statutes listed: 1) assert 'no adverse impact' to the environment, 2) acknowledge that stream resources, including fish and wildlife, have long-range seasonal requirements, 3) protect these systems, including their biology, from adverse impacts during low flow periods, and, 4) link groundwater and surface water resources. Visit the Minnesota Statutes website for more details.*

## 1.3 Approach and Rationale

This study and analysis of the surface water impacts from groundwater pumping is part of an overall DNR effort to examine water use and associated resource impacts in the Little Rock Creek area with the aim of ensuring sustainable water management. We employed the Instream Flow Incremental Methodology (IFIM), which is a modular decision support system for assessing potential flow management schemes (Stalnaker et al. 1995, Bovee et al. 1998). The IFIM is designed to assist natural resource and water management agencies in comparing the relative impacts of proposed instream flow management schemes with reference conditions. The unique feature of the IFIM is the simultaneous analysis of habitat variability over time and space. We also relied on the multidisciplinary expertise in our agency to accomplish the analysis in this report: streamflow gaging on Little Rock Creek, MODFLOW groundwater modeling, and estimation of streamflow depletion from groundwater use are all necessary elements of the investigation of surface water impacts on this stream (see MN DNR 2020a).

The maintenance of environmental flows capable of sustaining healthy river ecosystems should be viewed as both a goal and a primary measure of sustainability in water resources management (Richter 2010). Essentially, if we fail to manage water use and preserve the ecology of this stream, we also fail to achieve our goal of sustainability. From a water management perspective, groundwater is overwhelmingly important to both surface water resources and human uses of freshwater. The natural flow regime in surface waters is connected to and interacts with any associated groundwater aquifers

(Alley et al. 1999, Gleeson et al. 2012). As a result, depletion of the surface waters from groundwater pumping potentially impacts the ecology of surface waters by altering the structure and function of these systems.

The dominant influence of hydrology in natural aquatic ecosystems and the singular nature of the water system (surface water and groundwater are a single resource) imply that sustainable water management should be designed to maintain a natural flow regime even in the presence of high levels of off-stream and in-stream water use. To ensure water use does not harm ecosystem health and sustainability, a threshold for harm must be identified. Based on that threshold, management options can be developed that encompass the principles of ecologically sustainable water use.

### 1.3.1 Thresholds for Preserving Stream Systems

Identifying the threshold for ecological health and sustainability of surface waters then is a key task. A document outlining this task, 'Definitions and thresholds for negative impacts to surface waters' was prepared for the Minnesota State Legislature (MN DNR 2016), following a stakeholder engagement process. The process for identifying this threshold is described below.

In stream systems, the basic challenge is the difficulty of determining how much alteration from natural flows can be tolerated without compromising ecological integrity, and the subsequent ecosystem services these systems provide (Richter et al. 2012). When considering the integrity of ecological systems, we must account for three important principles: 1) the biota span a variety of spatial and temporal scales, 2) the living stream system includes the biota (organisms or elements of biodiversity) plus the processes that generate and maintain them; and, 3) living stream systems are embedded within evolutionary and biogeographical contexts (Karr and Chu 1999). Ecological systems, and society's use of them, depend on both parts and processes. As increasing human activity (e.g., use of water) changes stream systems, they move along a continuum of ecosystem health, ultimately to a state where little life is left. Identifying the thresholds along that continuum is useful in making management decisions. They may signal when an ecosystem or valued ecological attribute has been shifted to the limits of resiliency and when collapse or a shift to an alternative and often undesirable state is likely to occur (Folke et al. 2004). The "threshold" is essentially the point at which negative impacts occur to ecological integrity and health. Looked at conceptually (**Figure 2**), the threshold is the point at which, based on specific criteria, conditions or activities are no longer healthy or sustainable in terms of supporting the stream ecosystem. When water is removed from streams, the structure and function of their ecology is altered. This changes the river's habitat and biota, causing it to diverge from integrity and health. At some point, i.e., the threshold, increasing alteration of natural flows degrades the health of the stream and harms the stream ecosystem.

An approach to determine a threshold is to systematically examine increments of change in the driving variable and track its effect on the system. Evidence that a 10% flow alteration is likely to have a negligible effect on most taxa, stream types, and hydrologic conditions is generally agreed on by experts (Acreman and Ferguson 2010). Therefore, we assume a high degree of ecological protection will be provided when daily flow alterations are no greater than 10%; a high level of protection means that the structure and function of the riverine ecosystem will be maintained with minimal changes (Richter et al. 2012). Alternately, water appropriations of 20% or greater will likely result in moderate to major changes in natural structure and function of ecosystems (Acreman et al. 2008, Carlisle et al. 2010; **Figure 3**). As a result, we recommend a 10% limit in most circumstances, but recognize a diversion limit of up to 15% may be appropriate in some locations where off-stream water uses are less dependent on a consistent supply. However, with a 15% diversion limit, a low flow threshold, the annual Q90, would be needed to limit changes in the low flow extremes and to provide a buffer against water quality concerns

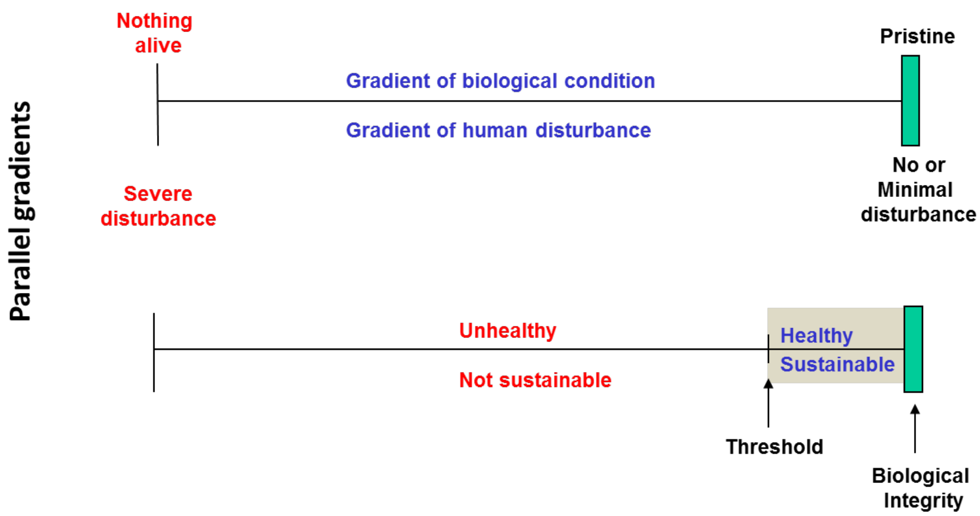



***Figure 2. Conceptual representation of the threshold for ecological health and sustainability.*** *This illustrates the relationship of health and integrity and the threshold for ecological harm. At one end of a continuum of human influence on biological condition, severe disturbance eliminates all life (top gradient line). At the other end of the gradient are "pristine" or minimally disturbed living systems; these systems possess biological integrity. A parallel gradient (bottom) from integrity toward nothing alive passes through healthy, or sustainable conditions or activities. Below a threshold defined by specific criteria, the conditions or activities are no longer healthy or sustainable in terms of supporting living systems. That point constitutes the target threshold (modified from Karr and Chu 1999).*

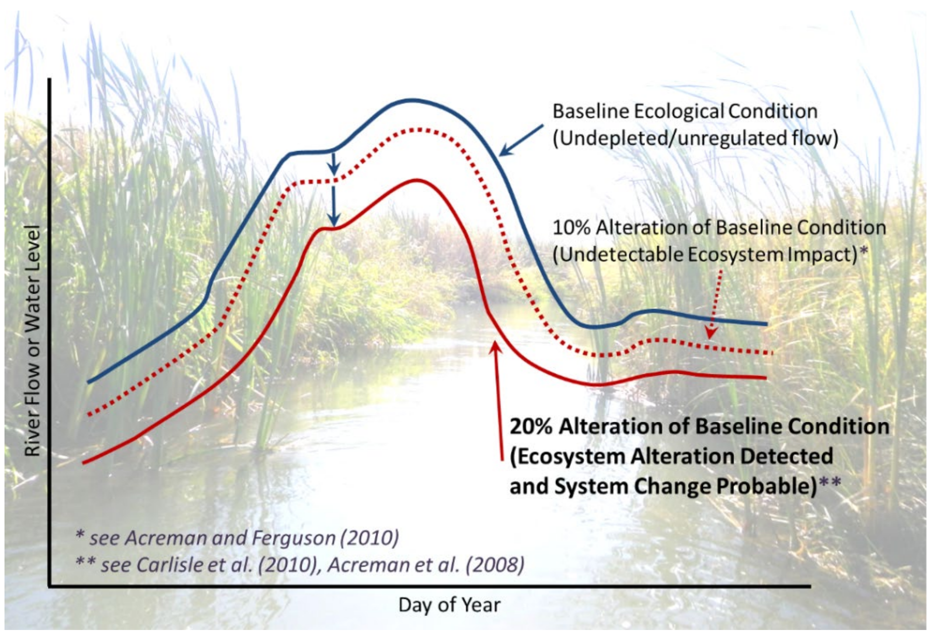



***Figure 3. An illustration of the sustainable ecosystem boundary and thresholds for flow depletion limits.*** *The red dotted line represents a 10% depletion of the baseline condition. Note here that there is an undetectable ecosystem impact. Evidence that a 10% flow depletion is likely to have a negligible effect on most taxa, stream types, and hydrologic conditions is widely accepted. As a result, a high degree of ecological protection will be provided when daily flow depletions are no greater than 10%. Alternately, water appropriations of 20% or greater, shown by the solid red line here, will likely result in moderate to major changes in natural structure and function of ecosystems (modified from Postel and Richter 2003; Richter et al. 2012).*

under drought conditions (MN DNR 2016). The “annual Q90” is the stream discharge that statistically was exceeded 90% of the time during the period of record analyzed. This follows the recommendations of Gleeson and Richter (2018), who state that high levels of ecological protection will be provided if groundwater pumping decreases monthly natural base flow by less than 10% through time.

### 1.3.2 Management Recommendations

How we translate a threshold, targeted to provide for sustainable use and to maintain health and avoid harm, into provisions we can manage and adjust as necessary is a critical part of managing water and ecosystems. The threshold limits outlined above ultimately have to account for the five interacting components that are essential to the structure and function of river ecosystems: hydrology, biology, geomorphology, water quality, and connectivity (Annear et al. 2004; see **Figure 4**). The natural hydrograph of streams and rivers influences the biology of rivers through several inter-related mechanisms (**Figure 5**), such that each part of the hydrograph is important for the ecological health of the river.

The importance of a river’s flow regime for sustaining biodiversity and ecological integrity is well-founded (Poff et al. 1997; Hart and Finelli 1999; Bunn and Arthington 2002). Streamflow is viewed as a ‘master variable’ (Power et al. 1995; Hart and Finelli 1999) or ecosystem ‘driver’ (Annear et al. 2004) that shapes many fundamental ecological characteristics of riverine ecosystems. From a basic ecological

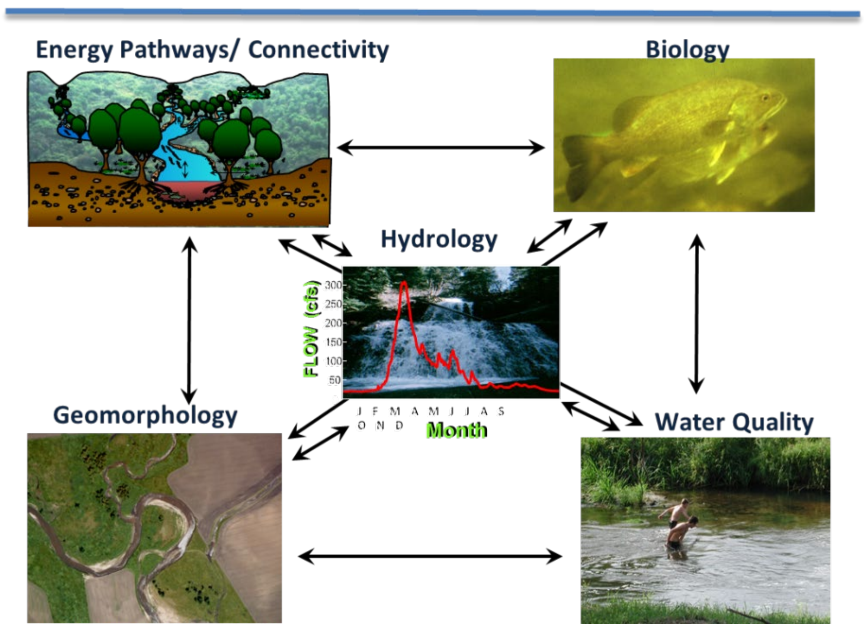



***Figure 4. Five components of river ecosystems.***

*The components are defined as: Biology encompasses the plant and animal species present in the stream, riparian lands, and contributing watershed. Connectivity represents the maintenance of pathways that move organisms, energy, information (e.g., DNA), and matter throughout the watershed. Geomorphology is the topographic and bathymetric features of the stream, its floodplain and riparian lands, and contributing watershed and the processes that continue to shape them. Hydrology is the inter-relationships and interactions between water and its environment in the “hydrologic cycle." Water Quality refers to the chemical, biological, and physical characteristics of both surface water and interconnected groundwater. See the DNR’s Watershed Health Assessment Framework website and Annear et al. (2004) for more information on the five components of river ecosystems.*

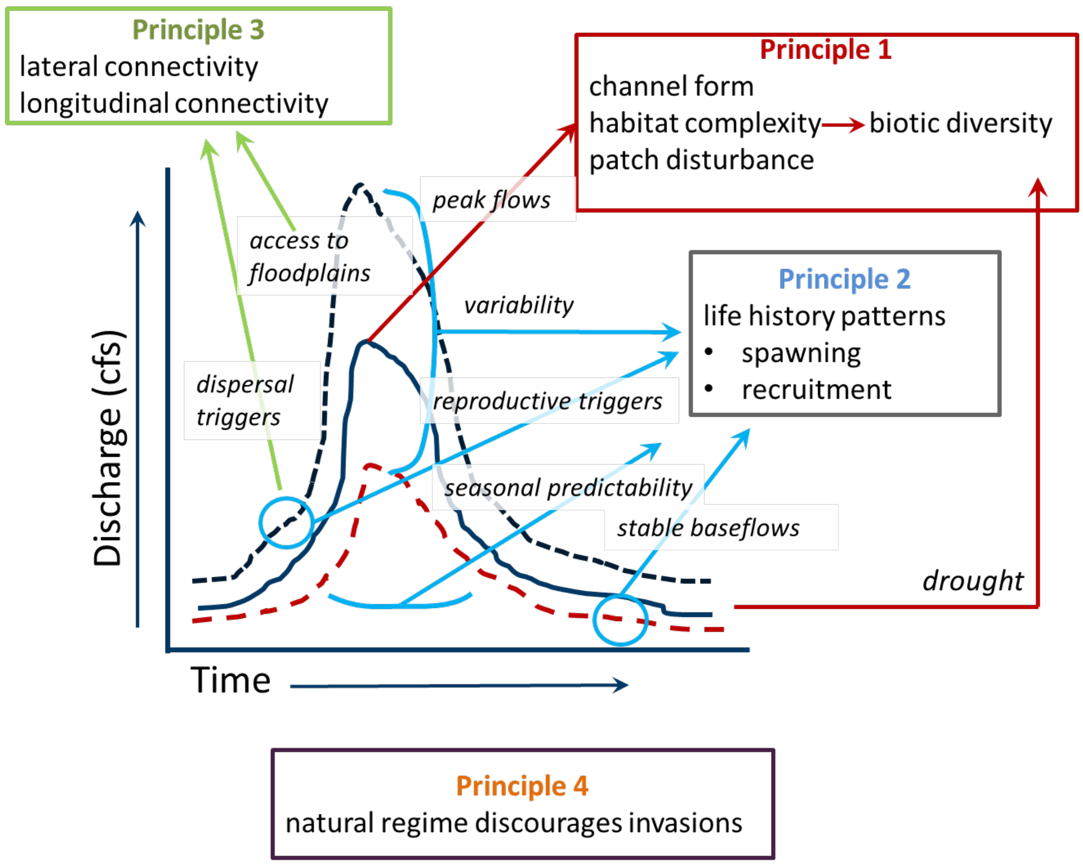



***Figure 5. Aquatic biodiversity and natural flow regimes.***

*The natural flow regime of a river influences aquatic biodiversity through several interrelated mechanisms that operate over different spatial and temporal scales. The relationship between biodiversity and the physical nature of the aquatic habitat is likely to be driven primarily by large events that influence channel form and shape (graph principle 1). However, droughts and low-flow events are also likely to play a role by limiting overall habitat availability. Many features of the flow regime influence life history patterns, especially seasonality and predictability of the overall pattern, but also the timing of particular flow events (graph principle 2). Some flow events trigger longitudinal dispersal of migratory aquatic organisms and other large events allow access to otherwise disconnected floodplain habitats (graph principle 3). Catchment land use change and associated water resource development can often lead to changes in one or more aspects of the flow regime resulting in declines in aquatic biodiversity via these mechanisms. Invasions by introduced or exotic species are more likely to succeed at the expense of native biota if the former are adapted to the modified flow regime (graph principle 4) (from Bunn and Arthington 2002).*

perspective, extreme events such as high flows and low flows exert selective pressure on populations to dictate the relative success of different species, and patterns of variation in 'sub-lethal' flows can influence the relative success of different species and regulate ecosystem process rates (Resh et al. 1988; Hart and Finelli 1999). The range and variation of flows over recent historical time, referred to as the natural flow regime (Richter et al. 1996; Poff et al. 1997), sets a template for contemporary ecological processes (Resh et al. 1988; Doyle et al. 2005), evolutionary adaptations (Lytle and Poff 2004) and native biodiversity maintenance (Bunn and Arthington 2002). Simply stated, the native biota have evolved in response to the overall flow regime. Statutory language calling for sufficient flows to maintain fish in good condition inherently addresses all riverine components because a healthy fishery depends on all of them.

**Sustainable Diversion Limit (SDL)**

A management system that protects the critical components of the hydrograph (i.e., duration, magnitude, timing, frequency, and rate of change), and thereby the other inter-related components of river ecosystems, must accomplish two key tasks: 1) establish a sustainable limit on the cumulative amount of water that can be withdrawn in a watershed which protects natural variability of flows over time, i.e., hydrograph shape, and 2) fully consider extreme events (i.e., drought duration and frequency), such that additional streamflow depletion does not cause ecological harm. In the past, protected streamflows were set based on fixed percentages of hydrologic variables and represented "minimum flows," – i.e., essentially, "what is the minimum flow required for the species of concern to survive?" The recommended minimum flow value was set for the entire year. However, under this approach, as demand for water increases, the result is simply a "flat line"—i.e., a static flow across the year at the minimum flow level. This management approach does not preserve key elements of riverine health, including the channel-forming high flows and variable flows needed to maintain habitat and ecological integrity, and has been demonstrated to degrade the stream's ecosystem over time (Baron et al. 2003, Annear et al. 2004).

Since the Sustainable Diversion Limit (SDL), as a fixed percentage of flow that can be withdrawn, remains constant along the seasonal hydrograph, it protects the stream's natural flow variability. The SDL is defined in relation to water appropriations as a maximum amount of water that can be removed directly or indirectly from a surface water body in a defined geographic area on an annual basis without causing a negative impact to the surface water body (MN DNR 2016). A Sustainable Diversion Limit is established as a means of keeping water flows and elevations above the threshold. To avoid ecological harm, the SDL is assessed on a monthly basis (i.e., for August) and established with consideration for the type and seasonality of use, including the pumping rate and volume. The SDL, as a percentage of flow, is not a new method. Adopting a percentage of flow approach for managing stream and river diversions is similar to that being applied by other water management agencies nationally and internationally, as in **Table 2.**

This approach can be effective in preserving water to maintain stream ecology, but may be risky if the allocation plan overestimates the amount of water available or where inter-annual rainfall patterns or climate change reduce the amount in the system (Arthington 2012). Often it is the aquatic environment that is sacrificed first in times of water shortage or drought (Bond et al. 2008). To identify a SDL, we must determine when cumulative withdrawals will likely cause an adverse resource impact exceeding the threshold limit we have identified (i.e., 10-20%). Assessing the impact requires building one or more relationships between the potential stressor (e.g., discharge as a result of streamflow depletion) and a variable of ecological importance (e.g., aquatic habitat) and examining the frequency and duration of habitat minima.

In contrast to the minimum flow method, the percentage of flow approach identifies the allowable depletion limit – i.e., the negative impact threshold – and sets the diversion limit to avoid the negative impact, over the long term. As discussed above, water diversions of 10% have a minimal impact on most species, stream types, and hydrologic conditions, while diversions above a 20% threshold produce moderate to severe ecosystem changes.

***Table 2. Percent of flow approach used for water management.***

| Location | Ecological Goal | Cumulative allowable depletion | Considerations | Decision process |
|---|---|---|---|---|
| **Florida (SWFWMD)** | Avoid significant ecological harm (maximum 15% habitat loss) | 8-19% of daily flows | Seasonally variable extraction limit; ‘hands-off’ flow (no withdrawals below) | Scientific peer review of site-specific studies |
| **Michigan** | Maintain baseline or existing condition | 6-15% of August median flow | Single extraction limit for all flow levels | Stakeholders with scientific support |
| **Maine** | Protect class AA: ‘outstanding natural resources’ | 10% of daily flow | Single extraction limit for all flow levels above a ‘hands-off’ flow level | Expert derived |
| **Massachusetts** | Sustainable management of water resources that balance human and ecological needs | Basin safe yield: 55% of annualized Q90.<br>For sub-basins, maximum level of August median streamflow alteration ranges from 3-10% for Categories 1 and 2 for each season. | Seasonal extraction limit based on category | Expert, scientific support |
| **Rhode Island** | Maintain habitat conditions essential to a healthy aquatic ecosystem | 6 Bio-periods and 5 classes<br><br>Summer Period Class 1-3 streams can deplete 10, 20, and 30% of the 7Q10, respectively | Allocation limited by cumulative streamflow depletion.<br>Identify allowable depletion limit even during dry conditions | Scientific support, stakeholders, public process |
| **European Union** | Maintain good ecological condition | 7.5-20% of daily flow<br><br>20-35% of daily flow | Lower flow; warmer months; ‘hands-off’ flow.<br>Higher flow; cooler months | Expert derived |

*Examples of the percent of flow approach actively being used for water management. The percent-of-flow-based management prescription has been increasing recognized and used to guide management prescriptions by governments across the country and world. The table shows various examples of the percent of flow approach being used for water management, including both groundwater and surface water allocations (from Richter et al. 2012, with additions).*

**Index Flow for the Sustainable Diversion Limit**

A key step in establishing a percentage of flow is to decide 'what flow' the percentage will be applied to. Withdrawals from streams ("natural and altered natural watercourses") must be limited so that "consumptive appropriations are not made from the watercourses during periods of specified low flows. The purpose of the limit is to safeguard water availability for in-stream uses and for downstream higher priority uses located reasonably near the site of appropriation" (Minnesota Statutes 103G.285). August is a low flow month during the summer growing season in Minnesota (**Figure 6**). Base flows are those flows contributed by groundwater to the stream; groundwater pumping will impact base flows. Summer low flows are considered critical to populations of aquatic species (Poff et al. 1997; Power et al. 1999; Bradford and Heinonen 2008). As such, August is an important month for fish and wildlife, in terms of their annual growth and condition heading into the winter months. It is also part of the annual growing season for agriculture and a time of peak water use. By setting total allocation limits based on the most sensitive and intensive water use season there will likely be less risk of unintended negative impacts on other life stages occurring during the year or on other ecologic functions or processes. Finally, August discharge is used as the index flow in the Michigan water withdrawal assessment tool (Hamilton and Seelbach 2010), providing a practical management example of this approach. Median values represent

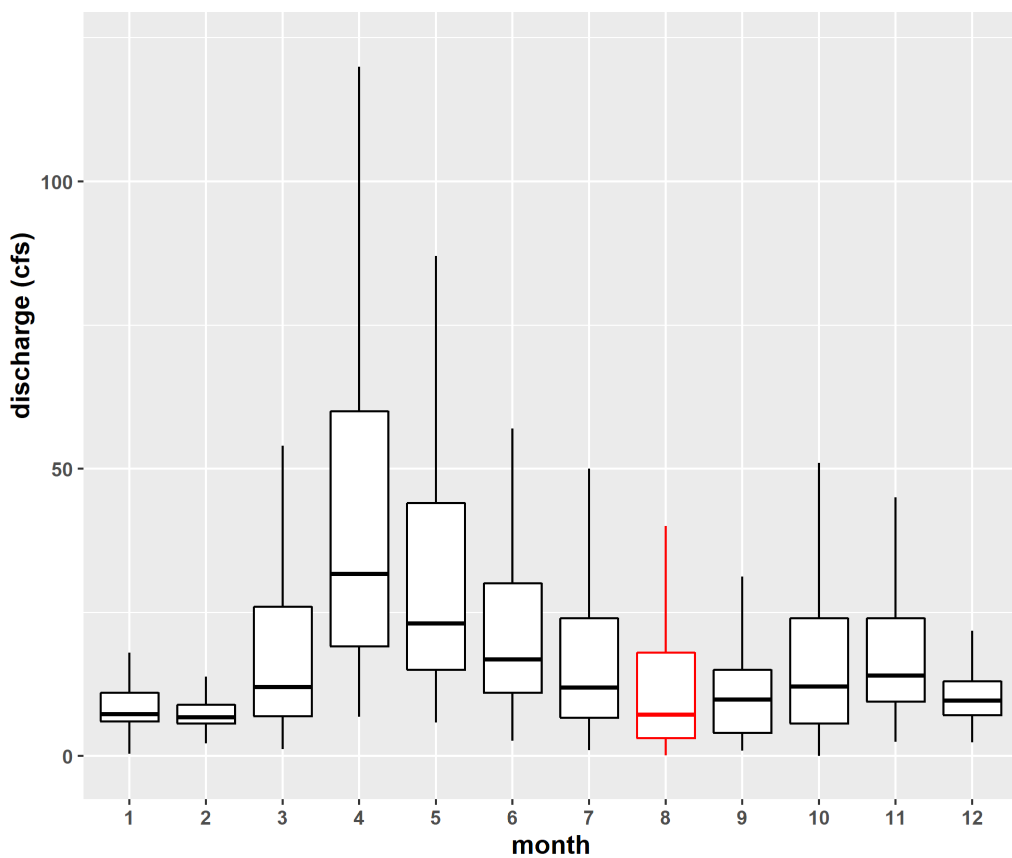



***Figure 6. Monthly box-plots of daily streamflow in Little Rock Creek.***

*Daily streamflow is shown for each month with the position of August (red box plot) relative to other months in the growing season. Note the median value, shown as the horizontal line in the box for each month, is lower in August, even though the variability, denoted by the size of the box, is greater than that in September. Data downloaded from the Minnesota Cooperative Stream Gaging site for the Little Rock Creek near Rice gage (site id=15029001). Daily streamflow data for the years 1998 through 2020 are included, however, not all years are complete.*

the midpoint of the data – half of the values are higher, half are lower. Median values are not sensitive to extreme events or skewed data, which may raise or lower the average. Given these considerations, the median August base flow (ABF) was chosen as the index flow from which to assess the degree of ecological harm from water diversions. Comparisons of the impact of depletion for various percentages of the August median base flow on habitat are included in the Results section, below.

### 1.3.3 Justification for Use of Habitat

Protection of stream ecosystems from impact caused by water withdrawal requires a reliable, flow-sensitive indicator. Aquatic habitat is such an indicator, well-suited to water management issues and challenges. Change in the availability of physical habitat is the most fundamental response to altered flow magnitude (Rolls et al. 2012). Hydraulic habitat is fundamental to an aquatic organism's existence and is directly related to flow (Annear et al. 2004). There is considerable evidence to suggest that both the quality and quantity of available habitat affect the structure and composition of resident biological communities; hence the importance of physical habitat is clear (Hynes 1970; Ward and Stanford 1979; Meffe and Sheldon 1988; Calow and Petts 1994). Selection for or persistence in a particular habitat is directly related to survival and reproduction, which determines fitness (Hutchinson 1957; Southwood 1977). Habitat is a critical factor in determining a species distribution and abundance (Hanski 1982; Kolasa and Strayer 1988; Tokeshi 1993; Venier and Fahrig 1996; Gaston et al. 2000; VanDerWal et al. 2009). Moreover, strong non-linear responses of channel width, depth and velocity are basic to fluvial geomorphology (Leopold et al 1964), and have been characterized as power functions of discharge where the rate of loss of width, depth, or velocity is steepest at low flows (Armstrong and Nislow 2012; Caissie et al. 2015). For organisms adapted to fast water with well-defined minimum velocity thresholds, habitat loss may at a rate greater than the loss of total wetted area because the area and depth of riffles tends to shrink more quickly at low flows than pools (Rosenfeld et al. 2011; Rolls et al. 2012).

The threshold report (MN DNR 2016) states that site-specific conditions for watercourses should be considered in establishing thresholds. Despite the threshold being defined in hydrologic terms, it is imperative that the ecological consequence is examined for site-specific conditions, when possible (Arthington et al. 2006).

As a result we assess current level of streamflow depletion on hydrology (do the current cumulative depletions exceed the threshold for harm) and how the streamflow depletion impacts aquatic habitat conditions. Explicitly stated, the negative impact and ecosystem harm is defined in terms of biology. Other resource management agencies also make this connection, for example Florida limits depletions such that habitat losses do not exceed 15% and Michigan limits depletions such that their fish community metric is not decreased more than 10%. For this assessment of Little Rock Creek, we equate hydrologic change and biological change (measured using physical habitat), using the rationale that a 10% change will have minimal impacts while a 20% change will result in ecosystem harm.

### 1.3.4 Fish and their Habitat as a Surrogate for Ecology

The most direct and effective measure of a waterbody's integrity, and of its place in the water cycle, is the status of life in the water (Karr and Chu 1999). "Ecosystem integrity is primarily a biological concern" (Reynoldson et al. 1995). Living communities reflect watershed conditions better than any chemical or physical measure because they respond to the entire range of biogeochemical factors in the environment (Karr and Chu 1999, 2000).

Fish are known to be good indicators of ecological status of aquatic ecosystems because they live permanently in water, occupy a wide range of ecological niches, and operate over a variety of spatial scales (Fausch et al. 1990, Ibanez et al. 2010). Temporal variability in streamflow provides a range of ecosystem processes and habitat needs that sustain high native diversity (see Poff and Ward 1989; Puckridge et al. 1998; Resh et al. 1988; Richter et al. 1996; Stanford et al. 1996; Walker et al. 1995). There is a significant correlation between habitat diversity and fish species diversity (Schlosser 1982). Fish communities in large rivers are characterized by high diversity, which reflects the structural diversity and habitat richness of the channel and connected floodplains (Schiemer 1998). Biological diversity is critical to ecosystem health and sustainability (Rapport et al. 1998; Loreau et al. 2001; Dudgeon et al. 2006).

Flow is the ultimate driver of river size, shape and physical habitat, that in-turn is a major determinant of the fish occurrence, abundance, and diversity (Leonard and Orth 1988; Bunn and Arthington 2002; Xenopoulos et al. 2005). The high longevity of some fish species enables the detection of disturbances over a long time frame. Fish are relatively easy to identify and their taxonomy, ecological requirements and life history traits are generally better known than those of other species groups, making their assessment easier, cheaper, and more accurate (Ibanez et al. 2010). Life history traits of freshwater fishes are well studied and well suited as a platform to test general relationships between the flow regime and biological communities; they also have direct implications for ecosystem function (Winemiller 2005). In addition, fish usually occupy high trophic levels, integrating disturbances affecting lower trophic levels. Because of all of these factors, since 1987, Minnesota DNR has sampled 188 sites on 53 rivers across the state, with 129 fish species and 345 fish species-life stages, for a total of 11,177 individual samples of fish microhabitat association data, and over 263K individual fish.

### 1.3.5 Fish Habitat Guild Approach

Alterations in flow are associated with changes in availability of flow-dependent physical habitat for stream fishes. Species exploiting similar resources should be affected similarly by the alteration of those resources (Roberts and O'Neil 1985). Flow regimes and catchment area have often been cited as the most important environmental drivers of guild composition in river systems (Bunn and Arthington 2002, Ibarra et al. 2003, Welcomme et al. 2006, Rolls and Arthington 2014, Taylor et al. 2014). Guilds are the 'basic building blocks' of communities (Simberloff and Dayan 1991). The habitat guild concept denotes that the structure of the fish community is determined by the functional structure of the aquatic habitat, in terms of habitat available and the prevalent hydrological processes (Noble et al. 2007). Therefore, any disturbance in the functionality or structure of the riverine habitat will be reflected by responses in the functional structure of the fish community. The guild approach provides an operational unit between individual species and the community as a whole (Root 1967), giving the potential to overcome zoogeographic problems of species distributions when considering fish communities across basins or over large geographic scales (Ibanez et al. 2010). Because flow-dependent physical characteristics of a stream (i.e., habitat – e.g., depth, velocity, substrate, cover) influence fish community structure, target species selected for modeling impacts should have a wide range of habitat needs (Leonard and Orth 1988, Aadland 1993).

A community-based approach emphasizing fish community diversity and varying sensitivity to flow by fish species-life stage and habitat preference was therefore used. A range of habitat types occupied by different functional guilds were modeled, as selection of appropriate target species is crucial for robust habitat modeling results. Aadland (1993) defined six habitat-preference guild criteria for use in fish community analyses using PHABSIM. The representative species-life stages for each guild are updated based on fish sampling data collected by DNR *Ecological and Water Resources Division's* – River Ecology Unit for use in development of habitat suitability criteria (Aadland and Kuitunen 2006). The habitat-preference guild definitions are based on depth and velocity preferences (**Table 3**), and the habitat type

*Table 3. Six fish habitat guild categories.*

| Guilds | Depth (ft), Velocity (ft/s) | Depth (cm), Velocity (cm/s) |
|---|---|---|
| Slow riffle | <1.97, 0.98-1.96 | <60, 30-59 |
| Fast riffle | <1.97, ≥1.97 | <60, ≥60 |
| Raceway | 1.97-4.91, ≥0.98 | 60-149, ≥30 |
| Shallow pool | <1.97, <0.98 | <60, <30 |
| Medium pool | 1.97-4.91, <0.98 | 60-149, < 30 |
| Deep pool | ≥4.92 | ≥150 |

*Guilds are identified by characteristic depth and velocity ranges (see Aadland 1993). The guild classes include slow riffle, fast riffle, raceway, shallow pool, medium pool, and deep pool.*

at which that species-life stage is found in the highest density from Statewide-based data. More detailed methods are documented by Aadland (1993). In fish community habitat analyses, it is not feasible to model conditions for every species-life stage present in the system. Species-life stages are selected from each habitat-preference guild with the assumption that the representative species share the same habitat-flow relationships as other species in the same guild (Leonard and Orth 1988). This approach acknowledges that some habitat types such as raceways and riffles are more sensitive to flow changes than other habitats such as pools (Aadland 1993). The representative species-life stages selected for Little Rock Creek modeling and simulation are discussed in Section 2.3.3.4.

# 2. Methods

## 2.1 Modeling Approach

### 2.1.1 Instream Flow Incremental Methodology (IFIM) – River2D

The IFIM is a decision-support system designed to help natural resource managers and their constituencies determine the benefits or consequences of different water management alternatives (Bovee et al. 1998). The IFIM is supported by an integrated habitat simulation and analysis system that was developed to assist users in application of the methodology. The IFIM is used for this study because it: 1) integrates our responsibilities to provide for sustainable use while maintaining ecological health), 2) addresses the primary stressor - summer streamflow depletion, and 3) is incremental, allowing for assessment of management alternatives.

The dynamic interactions among flow, sediment, and topographic features (fluvial geomorphology) play a key role in determining current habitat conditions within a river. Models like the physical habitat simulation model (PHABSIM) and River2D are the most widely used habitat simulation modeling approaches for predicting biological responses to altered flow regimes (Tharme 2003; Souchon et al. 2008) and are applied worldwide. To describe habitat conditions accurately, the PHABSIM continues to evolve as more sophisticated hydraulic models are developed and applied, e.g., River2D modeling (Waddle et al. 2000, Gard and Ballard 2003). The River2D model produces more accurate representations of spatial flow patterns in rivers. Bovee (1996) suggested that 2D models used in environmental flow studies had the potential to accurately and explicitly quantify spatial variations and combinations of flow patterns important to stream flora and fauna. Ghanem et al (1996) compared 1-D and 2-D hydraulic model velocity outputs along a reach on the Waterton River, Alberta, Canada and found the 2-D velocity values had significantly less error and correctly predicted the spatial trends. Crowder and Diplas (2002) demonstrated the utility of applying 2-D modeling for spatially varying flows in channels having complex topography, developing and using spatial metrics to better quantify stream habitat, and examining the role that channel morphology and hydrologic regime have on stream hydraulics.

While the process of developing a 2-D hydraulic model to represent a specific stream reach remains demanding, the advantages of applying these hydraulic models are significant. Therefore, to address the instream flow needs and measure habitat, a two-dimensional depth averaged model, River2D (Version 0.95a), was used to simulate in-stream hydraulics and summarize habitat suitability at each flow using weighted usable area (Bovee 1996; Steffler and Blackburn 2002). The River2D fish habitat component uses the PHABSIM approach of weighted usable area (WUA). Weighted Usable Area is an index of habitat quantity and quality and is defined as the wetted area of a stream weighted by its suitability for use by aquatic organisms; its units are in square feet or square meters per specified length of stream (Stalnaker et al. 1995; Annear et al. 2004).

### 2.1.2 Overview of process

1) **Study segment selection** – determined location and extent of study area, which included all habitat types present in the stream system.
2) **Study site survey setup** – selected upstream and downstream data collection boundaries, set up survey benchmarks and control points, and located appropriate cross-sections for discharge profile measurements.

3) **Hydraulic data collection** – collected water surface elevations (WSELs) throughout study reach and discharge data at designated cross-section transects to develop stage-discharge relationships for modeling site based on observed measurements. Tied this data in with gage data to extrapolate relationship for modeling additional non-observed flows. These data were also used for calibrating and validating the flow models.
4) **Terrain and bathymetry data collection** – surveyed stream-channel bed and riparian topography while inventorying substrate and cover types.
5) **Habitat suitability criteria data collection** – collected depth, velocity, substrate and cover data for all fish species-life stages caught.
6) **Habitat-Preference relationships for species-life stage representatives** – used the fish habitat guild approach to select species-life stage representatives for the fish assemblage in Little Rock Creek. Developed habitat suitability curves for each species-life stage selected as representatives for habitat simulation.
7) **Two Dimensional Hydraulic Model Construction and Calibration (River2D)** – simulated depths and velocities throughout the study site at a range of flows (four observed, 10 estimated). Calibrated flow models by comparing observed and simulated WSELs at the model inflow by adjusting dynamic model inputs (e.g. bed roughness, groundwater depth and transmissivity). Additional measured WSELs, depths, and velocities throughout the modeled stream reach were used to validate the flow models for the four observed flows once stable models were produced.
8) Habitat simulation (River2D) – simulated and summarized habitat suitability at each modeling flow using weighted usable area (WUA) as an index of habitat.

## 2.2 Site Selection

### 2.2.1 Study Location

When choosing a study site for 2-D hydraulic modeling and habitat availability analysis, we looked for a reach of Little Rock Creek that contained multiple series of riffle, run, and pool habitats. The selected reach also contained no major inflows or outflows into the main channel, was close to a stream gage, and was accessible with multiple types of survey equipment. The reach location chosen (**Figure 7**) circled below is located east of Royalton MN, about 800 feet (ft) downstream of County Road 26 (Nature Road).


***Figure 7. Location of River2D hydraulic and habitat modeling site.***

*The surveyed stream reach on Little Rock Creek was 1041 feet long, with 1019 feet of that modeled in the River2D software program.*

### 2.2.2 Reach Description

A 1,041-foot stream segment was selected for study. It is dominated by sand and gravel substrate, but also contained a mixture of streambed materials classified into the following standard substrate groupings: detritus, silt, sand, gravel, cobble, rubble, small boulder, and large boulder (**Table 4**; Aadland 1993). The only substrate class not present was bedrock. The reach contained three small intermittent backwater areas with no major secondary inflows or outflows to the main channel along the entire study reach.

***Table 4. Steam substrate classification types, size ranges, and Little Rock Creek substrate composition.***

| Substrate | Diameter (inches) | Percent of Surveyed Stream Substrate |
|---|---|---|
| Detritus | organic matter (no size restrictions) | 0.4% |
| Silt | <0.0024 | 6.9% |
| Sand | 0.0024 - 0.125 | 41.7% |
| Gravel | 0.125 - 2.5 | 37.0% |
| Cobble | 2.5 - 5 | 5.4% |
| Rubble | 5 - 10 | 0.6% |
| Small Boulder | 10 - 20 | 7.2% |
| Large Boulder | 20 - 40 | 0.8% |
| Bedrock | >40 | 0.0% |

### 2.2.3 Study Site Survey Setup

A closed loop traverse survey was completed to define the study area (**Appendix A**). The upstream and downstream extents of the survey site were defined, and four cross-section locations selected for discharge profile measurements. Initial setup of cross-sections at riffle locations was competed using laser levels and submerged rebar as benchmarks. Additional benchmarks and control points were added using a total station and tied into true vertical elevation (z) and x, y positioning using a Trimble R10 GNSS System and base station setup (**Figure 8(a))**. The base station was set up at MNDOT Geodetic Control Station "BUNKER" with a repeater used for the signal to reach the study site. A geodetic monument on bridge number 7343 over Little Rock Creek (**Figure 8(b)**) along County Road 26 was used to tie into the study site. **Figure 8(a)** shows a diagram of the benchmark, control point, and survey station setup used for terrain and bathymetry survey measurements on Little Rock Creek.

## 2.3 Data Collection

The River2D model required input of a point based map of streambed elevations and substrate as well as a stage-discharge relationship at the downstream boundary of the study side (the outflow of the model). This required a detailed distributed terrain survey of elevation and associated substrate at each point detailing important stream features and elevation changes over the study area.

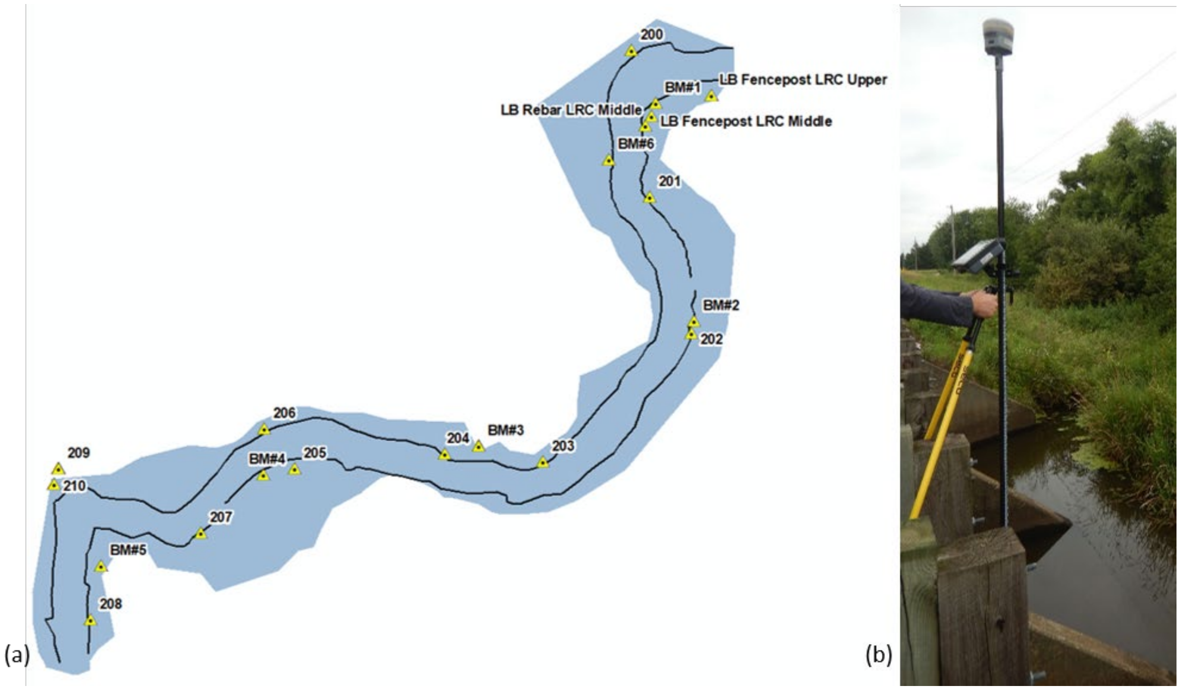



***Figure 8. Survey stations, benchmarks, and geodetic monument used on Little Rock Creek.***
***(a)*** *Benchmark locations LB Fencepost LRC Upper, LB Rebar LRC Middle, station 205 and station 208 are cross-section locations where discharge data was collected. A cross-section at station 208 is where the model outflow rating curve was based on.* ***(b)*** *Taking a GPS point at MNDOT bridge 7343 geodetic monument to obtain real world elevations and positioning data to tie in the survey of the site to real world locations.*

### 2.3.1 Hydrology and Hydraulic Data Collection

#### 2.3.1.1 Stream Gages, Discharge Data, and Water Surface Elevations

Development of flow models and instream flow prescriptions is dependent on an understanding of the overall hydrology of Little Rock Creek, and more specifically the study reach. DNR stream gage number 15029003 near County Road 26 (Nature Road) was the closest stream gage to the study site, but has limited years of data collected. For this study, it was necessary to use data from both the gage just upstream of the study site (gage 15029003) and the gage downstream at the minimum maintenance road, gage number 15029001.

Discharge measurements observed in the field were used to create a rating curve for the outflow of the study area in combination with measured WSELs. Milhous et al. (1981) states that measured flows can be extrapolated to simulate flows from 40% lower to 250% higher than the measured flows since it is not practical to collect observed data at each flow. Discharge data was collected at four cross-section locations shown in **Figure 8**. The cross sections were named LB Fencepost LRC Upper, LB Rebar LRC Middle, station 205 and station 208. The furthest downstream cross-section (station 208, Figure 8) is where an outflow discharge to WSEL rating curve was developed to estimate additional flows for modeling simulations.

Discharge profiles and water surface elevation measurements were taken on multiple sampling dates, some of which had to be disregarded after quality control checks. Collected discharge data was also compared to gage data for validation before proceeding. A SonTek FlowTracker Handheld ADV with top set wading rod and a HACH FH950 portable velocity flow meter were both used with standard USGS methods for discharge profile collection and total discharge calculations (Turnipseed and Sauer 2010). The HACH FH950 flow meter was used along with a four foot USGS top set wading rod. A detailed log of which device was used for each sample collection was recorded, along with the device serial number. Water surface elevations were collected on the same dates at approximately the same time with both a laser level and total station setup.

### 2.3.2 Terrain Model

To obtain a map of streambed elevation and substrate in the study reach for input into the River2D depth-averaged model, a detailed terrain elevation point map was surveyed (**Appendix B**). The terrain survey covering a 1,041-foot reach of Little Rock Creek was completed in three days after the initial survey setup was completed as described in an earlier section. Surveying of the streambed and substrate was completed on July 25, August 2 and August 3, 2017. The substrate classifications were the same ranges used in the fish habitat sampling (**Table 4**). A survey code list (**Appendix B**) was developed for use in the field survey and to create a channel index file of substrate required for the River2D habitat modeling. Log and boulder cover were the only cover types recorded as part of this study, and were not used in building the model.

At each distributed bed elevation point surveyed across the study site, concentrated around the stream channel, x, y, and z coordinates were recorded using a total station. X and y positioning was recorded as northing and easting, collected in the UTM_Zone_15N coordinate system, while the linear unit was set to US Survey Feet. The terrain and bathymetry survey data was imported into ArcGIS for display, editing, accuracy checks, and preparation for the River2D modeling system. The survey data was converted into meters for use in River2D, which requires the International System of Units (SI). Therefore, a combination United States Customary units and SI units are displayed throughout this report as appropriate. U.S. Customary units were used where possible to be consistent with the Little Rock Creek groundwater modeling report's unit of choice (MN DNR 2020a).

### 2.3.3 Biology

#### 2.3.3.1 Fish Habitat Sampling

Habitat preference criteria were developed by sampling a section of Little Rock Creek for fish and determining the habitat type used by the individuals captured. The methods used to collect fish habitat preference data followed Aadland and Kuitunen (2006). The fish sampling gear used was a prepositioned area shocker (PAS). The PAS used was a modified version of the one described by Bain et al. (1985) which samples an area of 6 x 25 feet (150 feet2).

The placement of the PAS in relation to the distance from the left bank was determined using a random numbers table. Once a sample was taken, the PAS was moved upstream a minimum of 5 feet and its location from the left bank was again randomly determined. This process is repeated until 20 PAS had been sampled.

Once set, the PAS was left undisturbed for a minimum of 11 minutes, as recommended by Bain et al. (1985). A 5,000-watt generator was used for the power source, and 120/210 –voltage, alternating current was used. A 6 x 4 foot catch net was held immediately downstream of the PAS to collect the stunned fish (**Figure 9(a)**). The current was activated for 20 seconds, after which time the area was thoroughly searched with dip nets for fish that did not drift into the catch net (**Figure 9(b)**). The collected fish were immediately placed into a container of water (**Figure 9(c)**).

Captured fish were identified to species, total length measured in millimeters, and life stages were assigned based on length. When many fish of a particular species-life stage were captured, only the first 10 fish were measured. The length range (minimum and maximum), and the number of remaining fish of that species-life stage were determined and recorded. The fish were immediately returned to the river once identification and measurements were completed.

Microhabitat data were recorded at each sampling location, regardless of whether any fish were captured (**Figure 9(d)**). Variables recorded with each sample included: river, site location, date, weather conditions, water and air temperature, conductivity, dissolved oxygen, sample location, gear type, three water depths, three mean column velocities, substrate types, and cover types (Bovee 1986). The depths and velocities were measured using a HACH FH950 portable velocity flow meter along with a four-foot USGS top set wading rod.


***Figure 9. Fish Habitat Sampling Sequence***

*The process for PAS sampling is shown here in a sequence of four images involving MN DNR staff and interns. (a) Holding a catch net at the downstream end of PAS; (b) Searching for and netting stunned fish in the sampled 150 feet$^2$ area; (c) Placing stunned fish into holding container, (d) Collecting microhabitat data of depth, velocity, and cover in each PAS.*

Depth and velocity were measured at three points within each sample location: at the upstream and downstream boundaries of the sample PAS, and in the middle. These measurements were averaged to obtain one depth and one velocity value per sample (Aadland et al. 1991). Water depth was measured with a top setting wading rod. Mean column water velocity was measured at 0.6 of the depth in water <2.5 feet deep and at 0.2 and 0.8 of the depth in water ≥2.5 feet deep (Turnipseed and Sauer 2010). Water velocity was measured with a commercially available HACH FH950 flow meter attached to a top-setting wading rod. Meters were tested and calibrated prior to use. Substrate was described according to the criteria in **Table 4**; cover criteria can be found in Aadland and Kuitunen (2006) but was not used for habitat modeling in this study. The percentage of a cell area covered by a particular substrate category was visually estimated to the nearest 10% in each cell. All cover types present within a cell were recorded.

### 2.3.3.2 Species List for Little Rock Creek

A total of 483 fish were caught using the habitat focused PAS sampling method at two sites. One site was located in the modeling site downstream of County Road 26, the other was located directly above the minimum maintenance road/15th Ave NW (north of County Road 40). The species-life stages and number observed are listed in **Table 5**. This includes a total of 21 different species-life stages from 12 fish species. **Table 6** shows the 1987-2009 Little Rock Creek fish assemblage, comprised of 32 fish species in total above the Sartell Wildlife Management Area impoundment. The historical species list includes 32 species sampled by the MN DNR Division of Fish and Wildlife over five assessments conducted in years ranging from 1987-2009. The two most recent fish community assessments sampled 16 and 17 species in 2006 and 2009, respectively (MN DNR 2010).

***Table 5. Fish species-life stage and number observed.***

| Species – common name | Species – scientific name | Life Stage | Number Caught |
|---|---|---|---|
| blacknose dace | Rhinichthys atratulus | adult | 122 |
| blacknose dace | Rhinichthys atratulus | young-of-year | 1 |
| brook stickleback | Culaea inconstans | adult | 2 |
| brook stickleback | Culaea inconstans | young-of-year | 3 |
| brown trout | Salmo trutta | adult | 8 |
| brown trout | Salmo trutta | juvenile | 40 |
| brown trout | Salmo trutta | young-of-year | 74 |
| central mudminnow | Umbra limi | adult | 14 |
| central mudminnow | Umbra limi | young-of-year | 3 |
| central stoneroller | Campostoma anomalum | adult | 1 |
| common shiner | Luxilus cornutus | adult | 1 |
| creek chub | Semotilus atromaculatus | adult | 51 |
| creek chub | Semotilus atromaculatus | juvenile | 11 |
| creek chub | Semotilus atromaculatus | young-of-year | 15 |
| fathead minnow | Pimephales promelas | adult | 1 |
| Johnny darter | Etheostoma nigrum | adult | 2 |
| Johnny darter | Etheostoma nigrum | young-of-year | 46 |
| longnose dace | Rhinichthys cataractae | adult | 5 |
| northern pike | Esox lucius | adult | 1 |
| white sucker | Catostomus commersonii | juvenile | 54 |
| white sucker | Catostomus commersonii | young-of-year | 28 |

*Fish captured on Little Rock Creek from two sites with PAS sampling from 2017 to 2019.*

***Table 6. Little Rock Creek historic fish assemblage.***

| Species – common name | Species – scientific name | Caught with PAS |
|---|---|---|
| bigmouth shiner | *Notropis dorsalis* | |
| black bullhead | *Ameiurus melas* | |
| black crappie | *Pomoxis nigromaculatus* | |
| blacknose dace | *Rhinichthys atratulus* | x |
| bluegill | *Lepomis macrochirus* | |
| brassy minnow | *Hybognathus hankinsoni* | |
| brook stickleback | *Culaea inconstans* | x |
| brook trout | *Salvelinus fontinalis* | |
| brown trout | *Salmo trutta* | x |
| burbot | *Lota* | |
| central mudminnow | *Umbra limi* | x |
| central stoneroller | *Campostoma anomalum* | x |
| common carp | *Cyprinus carpio* | |
| common shiner | *Luxilus cornutus* | x |
| creek chub | *Semotilus atromaculatus* | x |
| fathead minnow | *Pimephales promelas* | x |
| finescale dace | *Chrosomus neogaeus* | |
| goldfish | *Carassius auratus* | |
| hornyhead chub | *Nocomis biguttatus* | |
| Johnny darter | *Etheostoma nigrum* | x |
| largemouth bass | *Micropterus salmoides* | |
| logperch | *Percina caprodes* | |
| longnose dace | *Rhinichthys cataractae* | x |
| mimic shiner | *Notropis volucellus* | |
| northern pike | *Esox lucius* | x |
| pumpkinseed | *Lepomis gibbosus* | |
| sand shiner | *Notropis stramineus* | |
| spotfin shiner | *Cyprinella spiloptera* | |
| walleye | *Sander vitreus* | |
| white crappie | *Pomoxis annularis* | |
| white sucker | *Catostomus commersonii* | x |
| yellow perch | *Perca flavescens* | |

*Twelve of 32 fish species (sampled 1987-2009 by MN DNR Fisheries) making up the Little Rock Creek fish assemblage were observed in the PAS fish habitat preference sampling conducted in 2017-2019, and are indicated in the third column. The table contains species common name, scientific name, and record of if it was caught using the prepositioned area shocker (PAS) sampling method.*

#### 2.3.3.3 Habitat Preference Relationships

The fish habitat preference curves used in this study are density based, rather than presence/absence. This provides an area component to observations. Habitat preference values for most species-life stages were calculated for depth, velocity, and substrate. Preference values were calculated as follows:

1) Each habitat variable was divided into intervals (e.g., depth intervals were 0 – 0.16 ft, 0.17 – 0.49 ft, 0.50 – 0.82 ft, etc.).
2) The number of fish collected within each interval was summed, yielding habitat use.
3) The number of samples taken within each interval was summed, yielding available habitat.
4) Habitat preference values were calculated by dividing the habitat-use for each interval by the available habitat for that interval.
5) Preference values were expressed on a normalized scale from 0.0 to 1.0 by dividing each preference value by the maximum preference value.

A preference value of 0.0 represents the least preferred or least suitable habitat, while a value of 1.0 indicates the most preferred or most suitable habitat. Preference values were calculated for each site sampled, or for each flow if the flow changed significantly during a sampling period. A composite preference curve was then calculated by weighting the preference data for each site/flow by the number of observations at that flow and fitting a curve to the composite preference values.

Preference curves were constructed for each species-life stage and represent the optimum range of microhabitat variables of depth and velocity. Several techniques were used to construct the habitat preference curves from preference values, including histogram analysis and nonlinear regression. Preferences curves were developed for depth and velocity, while histograms were used to depict preferences between substrate. The habitat preference curves used for the habitat modeling in this study were taken from Aadland and Kuitunen 2006, updated in 2009, and updated again in 2020 as needed based on the fish habitat data collected in Little Rock Creek between 2017 and 2019 (**Appendix C**).

#### 2.3.3.4 Species-Life Stage Representatives for Habitat Types

To execute a community-based approach for IFIM analysis, habitat-preference guilds were used by selecting representative species present in Little Rock Creek for habitat modeling from each habitat guild type. Aadland et al. (1989) recommends six guilds based on cluster analyses of habitat parameter means for sampled species-life stages. The six guilds are shallow pool (< 1.97 ft deep and velocities < 0.98 ft/s), medium pool (1.97 – 4.91 ft deep and velocities < 0.98 ft/s), deep pool (>=4.92 ft deep), raceway (1.97 – 4.91 ft deep and velocities >= 0.98 ft/s), slow riffle (<1.97 ft deep and velocities 0.98 – 1.96 ft/s), and fast riffle (<1.97 ft deep and velocities >= 1.97 ft/s).

Eleven species-life stages shown in **Table 7** were selected as representatives from the six habitat guild types from Little Rock Creek. They were selected from the 22 total species-life stages sampled in Little Rock Creek using the PAS sampling methods. In one case the exact species-life stage was not caught in sampling, but the species at another life stage was. This was required to cover a diversity of habitat guilds. Two species-life stages from each habitat guild were selected, with the exception of one for deep pool due to limited species with that habitat preference. The only species caught, which prefers deep pools (at the juvenile life stage), was the common shiner. Species-life stage habitat guild assignments were determined using Little Rock Creek and statewide PAS sampling data (Section 1.3.5).

*Table 7. Species-life stage representatives for each of the six habitat guilds.*

| SPECIES – common name | SPECIES – scientific name | Life Stage | Species and Life Stage Code | Habitat Guild |
|---|---|---|---|---|
| blacknose dace | *Rhinichthys atratulus* | Adult | BNDA | Fast Riffle (FR) |
| brown trout | *Salmo trutta* | Adult | BNTA | Raceway (RW) |
| brown trout | *Salmo trutta* | Juvenile | BNTJ | Raceway (RW) |
| creek chub | *Semotilus atromaculatus* | Young-Of-Year | CRCY | Shallow Pool (SP) |
| common shiner | *Luxilus cornutus* | Adult | CSHA | Medium Pool (MP) |
| common shiner | *Luxilus cornutus* | Juvenile | CSHJ | Deep Pool (DP) |
| Johnny darter | *Etheostoma nigrum* | Adult | JNDA | Slow Riffle (SR) |
| Johnny darter | *Etheostoma nigrum* | Young-Of-Year | JNDY | Shallow Pool (SP) |
| longnose dace | *Rhinichthys cataractae* | Adult | LNDA | Fast Riffle (FR) |
| white sucker | *Catostomus commersonii* | Juvenile | WTSJ | Medium Pool (MP) |
| white sucker | *Catostomus commersonii* | Young-Of-Year | WTSY | Slow Riffle (SR) |

*Two species-life stages were selected from the species caught in Little Rock Creek fish sampling to represent each habitat guild. The exception was for deep pool – only one species caught was part of that guild type.*

## 2.4 Hydraulic Modeling

### 2.4.1 River2D Model Construction

The River2D Modeling system requires inputs of channel bed topography, bed roughness, transverse eddy viscosity distributions, boundary conditions, and initial flow conditions. Transverse eddy viscosity was not applicable in this study due to no tributary inflows or other concentrated inputs. A triangulated irregular network (TIN) or discrete mesh is also required to accurately capture flow variations in the modeled stream reach (Steffler and Blackburn 2002). The River2D software has separate programs for developing bed and mesh input files for use in the River2D system. The R2D_Bed and R2D_Mesh programs were used to prepare data for input into River2D.

#### 2.4.1.1 Bed

After field surveying was complete, creating a streambed topography file was the first step towards building the River2D flow models. The R2D_Bed program is a graphical user interface we used to edit point data and prepare the streambed topography file for River2D. The R2D_Bed program requires input of a text file with positioning and elevation data (x, y, z) as well as channel roughness for each point assigned. It also allows one optional descriptor column (Steffler 2002). The text file was prepared in ArcGIS ArcMap and imported into R2D_Bed. Once in the bed program, point data was triangulated and a system boundary for the modeling extent was defined. This included defining the inflow and outflow boundary segments of the hydraulic model. Breaklines were digitized at important features such as toe (bottom) of streambank and top of bank (**Appendix D**). Another main function of the R2D_Bed program is to identify and fix unrealistic shapes and bad triangles produced during the triangulation process. Bad areas were identified, fixed, and re-triangulated.

The channel roughness value started off as a substrate code recorded in the field survey (**Appendix B**) and assigned to corresponding channel index value. A channel index file (with a substrate value assigned to each point) is required for the River2D fish habitat module. At that point, a channel index file was saved for later use (**Appendix D**). Next, the channel roughness value was set for the study reach, and replaced the channel index substrate value for building the flow models. A bed roughness of 0.8 was assigned to the entire reach as a starting point and was later adjusted in the flow model calibration steps as appropriate. The bed file was then ready to import into the R2D_Mesh program.

#### 2.4.1.2 Mesh

The final bed file of pointwise elevations and channel roughness height was opened in R2D_Mesh (**Appendix D**). This is an interactive mesh generation program for 2-D depth-averaged finite element hydrodynamic modeling used with River2D (Waddle and Steffler 2002). The first step in mesh building is to define mesh breaklines along the final bed breaklines. Additional breaklines were then added between the initial ones – following the flow of the stream, and around a large sand wedge in the downstream most bend. Boundary nodes were then added, and the inflow and outflow boundaries were set. A uniform fill of points was applied, and the mesh was created by triangulating between those points. Nodes were added or removed as needed to improve the quality of the mesh measured by the R2D_Mesh program using a Quality Index (QI). The QI represents the lowest triangle quality value of any triangle generated in the mesh and how much it deviates from an equilateral triangle. As nodes were changed, triangulation was repeated and smoothed multiple times to increase the QI by improving the aspect ratio of the triangles. Triangles with large elevation differences between the mesh and bed topography were also identified. Nodes were added to improve interpolation in those areas, usually with large variations in bed elevation. As the mesh was refined, including adding at least 8-10 elements across the inflow and outflow boundaries, the QI started to increase. When breaklines are used the acceptable range for the QI is between 0.15- 0.5 (Steffler and Blackburn 2002). When a good QI was obtained, the mesh file **(Appendix D)** was saved as a River2D Characteristic Dissipative Galerkin (CDG) input file.

#### 2.4.1.3 River2D Flow Model Construction and Calibration

Once the River2D CDG input file was developed, it was brought into the River2D software program where separate models were built and calibrated for each flow being simulated. This included specifying different inflow and outflow boundary condition for each flow, as well as checking and adjusting several other model parameters as needed to obtain acceptable ranges of values for a reliable flow model. Fourteen separate CDG flow models were built (one for each flow condition) combining the computational bed topography mesh with the following model inputs: WSEL at the outflow of the modeling reach, the stream discharge entering the study reach, and the estimated WSEL at the inflow (Cowan et al. 2017; Steffler and Blackburn 2002). The estimated WSEL entered into each flow model was based on the WSEL to discharge rating curve for Little Rock Creek plus 0.05 m to be sure the starting inflow WSEL was higher than the simulated steady state solution. The simulations computed depths, velocities, and WSELs throughout the study site at each flow (Steffler and Blackburn 2002).

A 2-D hydraulic model was built first for the highest flow simulated, before attempting lower flows. Less adjustment is needed to the models for each flow if the highest flow simulation is calibrated first, and the other flow models built off that (Steffler and Blackburn 2002). The flow model was run to a steady state solution and checked to see if it met several criteria. If issues were found, the bed roughness in the bed topography file or computation mesh for each model had to be further refined to correct any immediate issues identified in the steady state flow simulation. This is an iterative process and had to be repeated numerous times in some cases to reach a steady state solution that met all of the requirements.

Several parameters were checked to see if model adjustments were needed. Steffler and Blackburn (2002) state that a steady state solution change of less than approximately 0.00001 is recommended to be considered converged. This usually coincided with the model solution reaching convergence and the inflow and outflow discharges nearly matching. The difference between the discharge at the inflow and outflow of the model is referred to as the Net Q, and was required to be within about a 0.003 cubic meters per second (cms; equivalent to 0.1 cubic feet per second (cfs)) threshold for the model solution to be considered converged at a steady state. If the solution change was not initially small enough, that typically indicated there was a small area in the mesh where the flow was oscillating back and forth between wet and dry conditions, sometimes with an eddy present near the wetted edge. This was fixed by editing the mesh nodes in the location of repeated oscillation – making the mesh more or less dense there.

**Appendix E** presents a detailed table of the model inputs and parameters used for each modeling flow, the most relevant output values examined, and checks that were used to validate each model simulation. Values labeled as "model check" were assessed as to whether they fell within an acceptable range. The first model check was a comparison of the simulated inflow WSEL to the actual (estimated or measured) WSEL from the rating curve. If the difference in values was less than 0.03 m (0.1 ft) the model was accepted in relation to this variable, a standard threshold used by other studies (California Department of Fish and Wildlife 2016). If the simulated WSEL was more than 0.03 m different from the estimated value, the bed roughness had to be adjusted in the R2D_Bed file. If the simulated WSEL needed to raised, the roughness was increased. If the WSEL needed to be lowered, the roughness was decreased. The roughness value was sometimes adjusted for the entire study area, or in other cases just a certain region depending on the flow. Five different bed topography files with different roughness values for certain regions of the simulated area were used to build the flow models. Many of the models used the same roughness bed files as other flows, but the higher flow models needed more adjustment than others to calibrate the WSEL simulations.

As described earlier, the Net Q was examined, and was required to be within 0.003cms (0.1 cfs) as was standard in other studies (California Department of Fish and Wildlife 2016; Cowan et al. 2017). If this threshold was not met by any of the fourteen flow simulations, this indicated a problem with the model. The mesh was either refined or model parameters were changed from defaults for parameters not yet discussed. When calibration of WSEL was not achieved by bed roughness adjustments alone, or the discharge did not converge, other variables were adjusted.

Addition parameters adjusted by the user included the minimum depth to groundwater and groundwater transmissivity. The minimum depth for groundwater flow was increased as the simulated flow decreased to increase the stability of the model, and provide flow connection through riffles at the two lowest simulated flows (Cowan et al. 2017). This changes the depth at which the River2D model switches from using surface water flow equations to groundwater flow equations. In shallow areas (such as riffles in low flow), there can be computational difficulties without adjusting the groundwater parameters (Steffler and Blackburn 2002). Groundwater transmissivity ranged from 0.01 to the default value of 0.1. Using a small value is recommended by Steffler and Blackburn (2002) to make sure groundwater discharge is negligible in relation to surface flow.

Depth averaged 2D models like River2D are best at simulating subcritical (laminar) stream conditions (Froude number less than 1), but can also model supercritical turbulent conditions and transient flow where present (Froude number greater than 1). Low gradient stream systems are usually assumed to have subcritical flow, where the Froude number should be less than one throughout the modeled area

(California Department of Fish and Wildlife 2016). After the model for each flow was run, the maximum Froude number was examined, and tiny areas with Froude numbers over one were refined to make sure the mesh was representative of the bed topography. Occasionally areas with a boulder produced a high Froude number (greater than 1), which diminished when more nodes were added to better simulate flow around the boulder substrate.

Visual assessment of velocity magnitude and vectors produced by each flow simulation was also completed. The mesh was edited by identifying small, isolated areas with erroneous high velocity. The mesh was also refined to fix areas where the velocity magnitude oscillated drastically when the model was run or the velocity vector arrows were very large and aimed upstream or into the banks.

### 2.4.2 River2D WSEL, Depth, and Velocity Simulation Validation

Once each of the flow models were run, refined, and met the thresholds discussed above – the final step was to validate the model simulation results. Water surface elevations simulated in River 2D for the four observed calibration flows were compared to WSELs measured in the field throughout the study reach. This was a much more intensive comparison than the inflow WSELs that were examined previously. **Appendix F** details the number of observed WSELs measured at each flow on how well it compared to the simulated values. The WSEL for each computational node in River2D was exported to ArcGIS ArcPro and compared to the nearest measured value (using a threshold of plus or minus 0.03 m for acceptance of the simulated value). Depth and velocity validation were the final step in accepting the River2D models. At the approximate location of terrain survey Station 205, a cross-section was extracted from the depth and velocity simulations. This was compared to the discharge profile depth and velocities measured at the same flow in the field (**Appendix F**). This process was repeated for the other three observed flows.

## 2.5 Fish Habitat Modeling

### 2.5.1 River2D Habitat Calculations

Eleven species-life stages were selected as guild representatives for modeling in River2D using methods defined previously. **Table 7** (Section 2.3.3.4) lists those species-life stages and the habitat guild they belong to. The River2D habitat module allows weighted usable area to be computed for a fish species-life stage at one hydrodynamic flow solution at a time. The required inputs were a channel index file, which contained the same bed topography nodes as the R2D_Bed file, but instead of channel roughness contained substrate values at each node. Habitat preference curves for the species-life stage being modeled were also required, and were obtained using the methods described in Section 2.3.3.3. The habitat module used discrete interpolation to calculate channel index at each computational node in the mesh from the channel index file. WUA is calculated by multiplying combined suitability (for the species-life stage) by the area in square meters. Combined suitability was calculated in River2D using the product method. This multiplied depth suitability, velocity suitability, and channel index suitability. River2D output the overall WUA for the species-life stage at the specified flow, as well as the depth suitability index, velocity suitability index, channel index (substrate) suitability, combined suitability and WUA at each node. This process was then repeated for each of the 11 species-life stages for each hydrodynamic flow model. Combined suitability index simulations for the four observed flows is presented for two species-life stages in **Appendix G**.

### 2.5.2 Habitat Time Series

The habitat time series is the fundamental tool for quantifying the effect of alternative flow regimes on a species-life stage (Stalnaker et al. 1996, Bovee et al. 1998). Creating the habitat time series involves merging the discharge – habitat relationship (i.e., curve) with a hydrologic time series typically based on mean daily values. The habitat time series can be examined for qualitative information, but additional transformations of the data are need to quantitatively assess change between habitat time series alternatives. The transformation involves converting the habitat time series to a habitat duration curve, then summarizing various frequency of habitat conditions across the alternatives. Three separate habitat metrics are calculated for each species-life stage examined. The metrics identify the amount of habitat available under the reference conditions and the baseline (i.e., observed) condition. A change of less than 10% in a metric is considered undetectable while a decrease in habitat of greater than 20% is considered an ecological impact and results in unsustainable streamflow depletions. The use of 10% and 20% as a threshold of harm is discussed in Section 1.3.1.

Because of the nature of the hydrologic alteration in Little Rock Creek, quantification of habitat change is focused on summer or low flow magnitude and duration impacts on habitat. While the entire natural flow regime is critical to all aquatic ecosystems, the impact of streamflow depletion is focused on the summer low flow. Additionally, the effective habitat metric often used downstream of hydroelectric plants is not pertinent, since flows are not rapidly ramped up and down. The metrics are described below.

### 2.5.3 Habitat Summary Analysis – Habitat Metrics

The habitat time series provides a qualitative view of the change in habitat with time similar to viewing a flow time series or hydrograph. Presenting the habitat as a duration curve provides additional information on the duration of each event and combining the two curves helps to identify large groupings of events; such as, low, moderate, or high flow or habitat events. Caution should be taken in reviewing the habitat duration curve within the context of the flow duration curve because habitat can be nonlinear resulting in multiple flows potentially having the same habitat value (Bovee et al 1998). For each habitat duration curve, we developed three metrics of habitat availability, representing different perspectives of the habitat time series. The percent change in each metric is used to determine whether the 20% change threshold is exceeded and the final description of ecological harm or not.

Multiple metrics were chosen to capture different signals in the habitat time series that result from differential patterns in the hydrology of the stream (e.g., average conditions, minimum habitat conditions, low-habitat events occurring over a three-day period; Bovee et al. 1994). Use of multiple metrics has a rich history in assessing stream health (Karr and Chu 1999). Following Karr and Chu (1999), the metrics were identified with five factors in mind, 1) classifying homogeneous environments, 2) selecting measurable attributes that provide a signal about the biological effects resulting from an environmental change, 3) consistent, accurate and precise measurement, 4) a defined analytical procedure to extract the patterns in the data, and 5) communication of results.

The use of habitat area for multiple species representing multiple habitats addresses each of the five concerns identified in Karr and Chu (1999). First, using hydraulic habitat captures the conditions that most aquatic species use to define their distribution in a stream; these include, fish (Aadland 1993), macroinvertebrates (Gore et al. 2001) including mussels (Parasiewicz et al. 2012) and macrophytes (Riis and Biggs 2001). Physical attributes help to define a species niche and in turn sets limits on the species abundance. Secondly, the stream is divided into six homogeneous habitat guilds and the habitat area of at least 2 (where possible) representative species of each habitat guild are assessed. Using the habitat of multiple species for each delineated habitat helps to ensure that all biotic components of the stream are considered, thereby managing for the sustainability of the stream rather than managing for a single species (Gore et al. 2001).

To evaluate impacts to the habitat time series thoroughly, average and low flow conditions need to be examined. Three (3) metrics were designed to capture potentially important and different signals of the habitat regime as described in Bovee et al. (1994) and Bovee et al. (1998). One metric (1) assesses change occurring under normal conditions and two metrics (2 and 3) assess the habitat under potentially restricted habitat conditions often occurring under low flows. By using a well-defined approach and tools (instream flow incremental method and river2d) and previously identified habitat metrics, a consistent and accurate assessment of the streamflow regime alternatives can be made and communicated. Each of the three (3) habitat metrics are described below.

Metric 1 is the mean of the habitat values with exceedance values between the 10% and 90% exceedance (Bovee et al. 1998). Metric 1 quantifies the habitat during normal conditions but with the influence of extreme values removed. Our expectations were that Metric 1 would show little change in habitat because streamflow depletion is most impactful at the lower flows.

Metrics 2 and 3 are designed to assess a change in the magnitude and frequency of potential habitat bottlenecks. Habitat bottlenecks, for example significant restrictions in usable area (Wiens 1977), often occur at hydrologic extremes (Stalnaker et al. 1996) particularly when space is restricted during low flows (Nehring and Anderson 1993, Bovee et al. 1994). Growing season bottlenecks have been identified using weighted usable area in warm water (Bovee et al. 1994) and cold water streams (Jowett 1990, 1992; Nehring and Anderson 1993).

Metric 2 is the mean of the habitat values with an exceedance value of 80% to 100%. This metric focuses on the magnitude of the habitat conditions during low flows and is described in Bovee et al. 1998. Metric 3 is the 3-day minimum habitat value with a 10-year return interval (3dH10), which is analogous to low flow frequency 3-dayQ10. The period of record available for the Little Rock Creek habitat study area was 12 years of August data, Curran and Olsen (2009) suggest a minimum of 10 years. This is calculated by creating a running 3-day average of habitat amount and finding the minimum value each year of the period of record. The Pearson Type 3 distribution was fit to the observed annual (of only August values) minimums and the 10-year return interval estimated. The sample L-moments were estimated using the *lmoms* function and the 10 year return interval estimated using the *parpe3* function both available in the R package lmomco (Asquith 2020). Subsequently, the return interval for a given value was estimated using the *cdfpe3* function also available in the lmomco package.

# 3. Results

## 3.1 Introduction

An assessment of groundwater pumping effects on the streamflow of Little Rock Creek is described in the report Groundwater Flow and Groundwater / Stream Interaction in the Little Rock Creek Area (MN DNR 2020a). From the results of MN DNR (2020a), two alternative scenarios are examined: 1) the reference condition hydrology (i.e., the hypothetical condition of no groundwater pumping) and 2) the baseline condition (i.e., the observed streamflow for Little Rock Creek). The habitat assessment was conducted immediately downstream of the Little Rock Creek near Rice, Number 15029003 stream gage. Both reference and baseline hydrology were provided by MN DNR (2020), with additional gage record extension based on the downstream gage (Number 15029001) and using the maintenance of variance technique (Hirsch 1982).

## 3.2 Hydrology

Hydrologic conditions in the Mississippi River – Sartell watershed, which includes Little Rock Creek, have been relatively wet since 2008. The Palmer Drought Severity Index (PDSI) uses readily available temperature and precipitation data to estimate relative dryness. Since 2000, the Palmer Drought Severity Index (PDSI) for the month of August in this watershed shows that all but 4 years have a PDSI value above 0 (normal conditions), and 11 of the past 20 years have a PDSI value above 3, which is considered very moist (**Figure 10**). Scores of +4.0 and above are considered extremely moist and have occurred in five of the last 7 years.

However, local hydrologic conditions experienced by irrigators in Little Rock Creek may not be adequately characterized by the watershed-scale PDSI. When annual August streamflow, for observed and with estimated pumping impacts, is compared to August PDSI for the same period of record (2006 - 2018), three of the four years that pumping caused noticeable depletion were not classified as drought (2008, 2012, 2013; see **Figure 11**). It is clear from this that irrigation pumping activity in Little Rock Creek is not well predicted by the PDSI for its watershed. If August water use is fairly constant during 2006 to 2018 and mirrors annual reported use, then it may be that depletion impacts on Little Rock Creek are simply negligible during wetter years.

To examine the nature and extent of estimated change occurring in August between the observed streamflow (baseline = flow with groundwater pumping occurring) and estimated streamflow (reference=flow with groundwater depletion removed), the August time series was plotted as a duration curve (**Figure 12**). Two aspects are evident: there is virtually no impact on median flows or greater (exceedance values ≤50%), and pumping impacts intensify after approximately the 75% exceedance flow range.

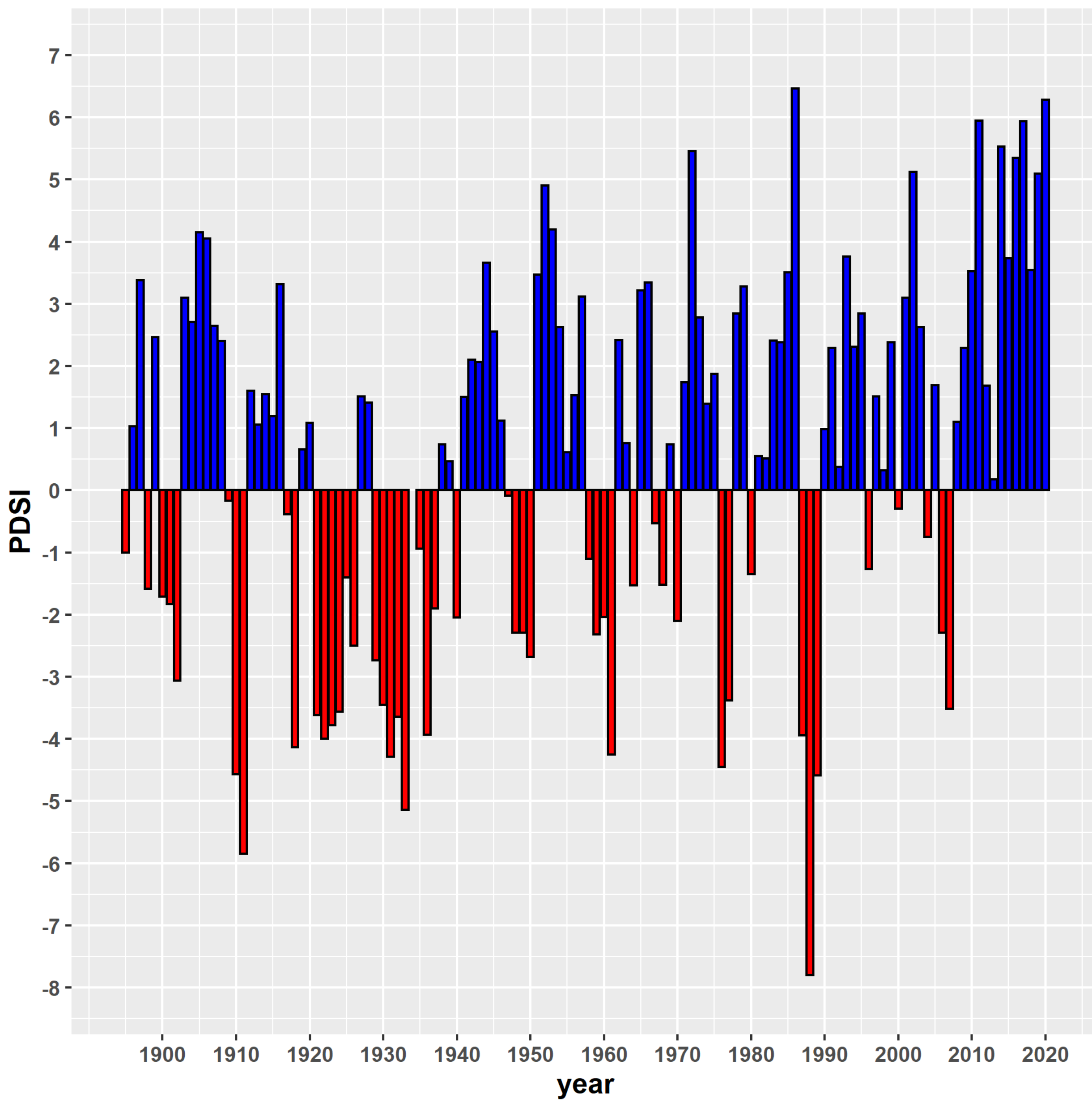



***Figure 10. August palmer drought severity index (PDSI) values for the Mississippi River- Sartell watershed.*** *Data was downloaded from [Minnesota Climate Trends website](). Values less than zero indicate a dry period. Values from -2 to -2.99 are considered moderate droughts, with values from -3 to -3.99 as severe drought, and -4 or less as an extreme drought.*

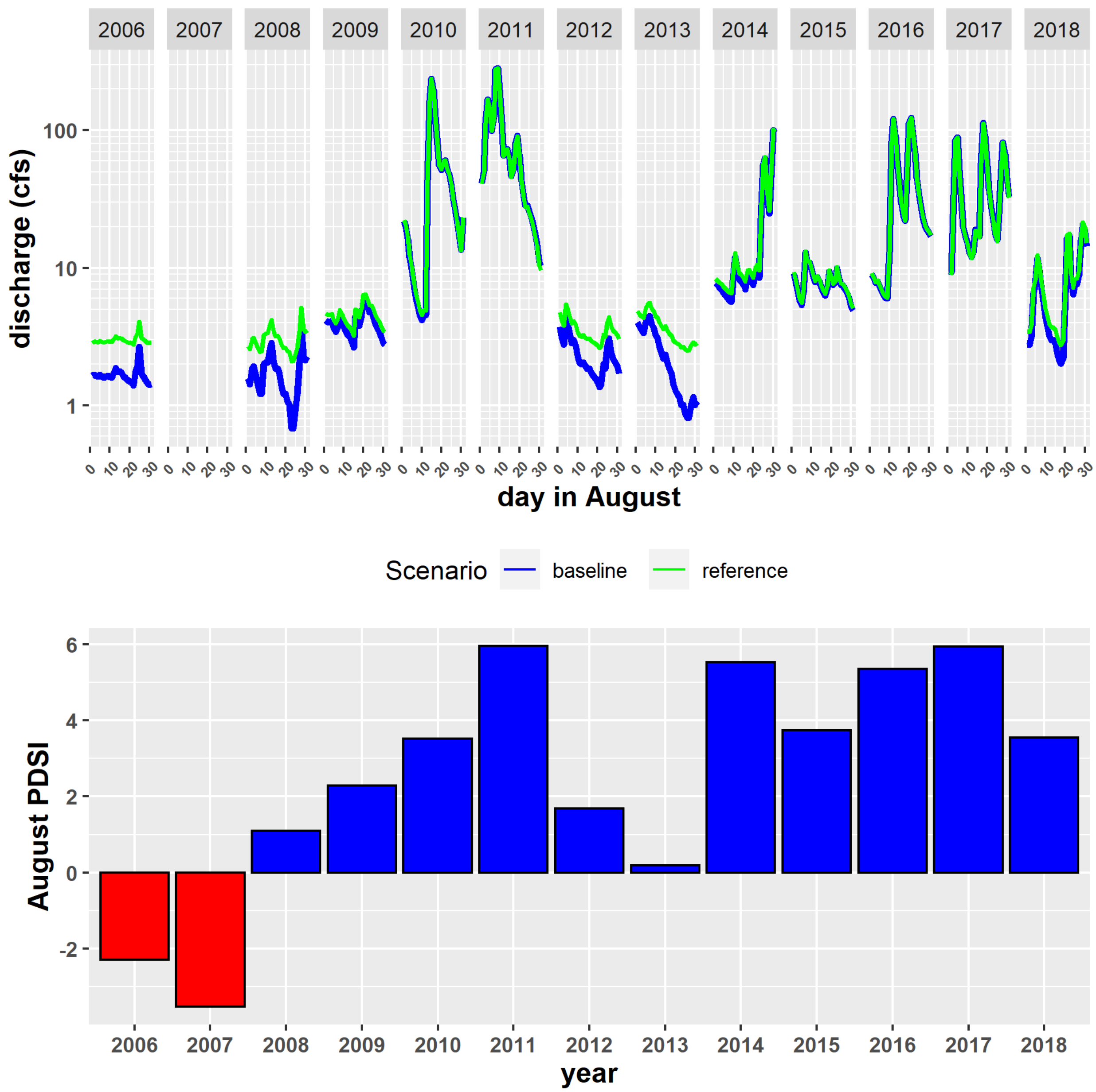



***Figure 11. Comparison of streamflow estimates.***
*Streamflow estimates are shown with and without pumping impacts and the Palmer Drought Severity Index (PDSI) for Little Rock Creek. The top graph represent the streamflow time series (August only) for Little Rock Creek at gage 15029003. The blue line represents the observed gage data referred to as the baseline condition. The green line represents the hypothetical streamflow at gage 15029003 if no streamflow depletion occurred. The hypothetical streamflow data were provided as part of the MN DNR EWR Groundwater report "Groundwater Flow and Groundwater / Stream Interaction in the Little Rock Creek Area." The streamflow data includes some data that were extended using the downstream gage (15029001). The bottom graph displays the PDSI August values for the stream gage period of record.*

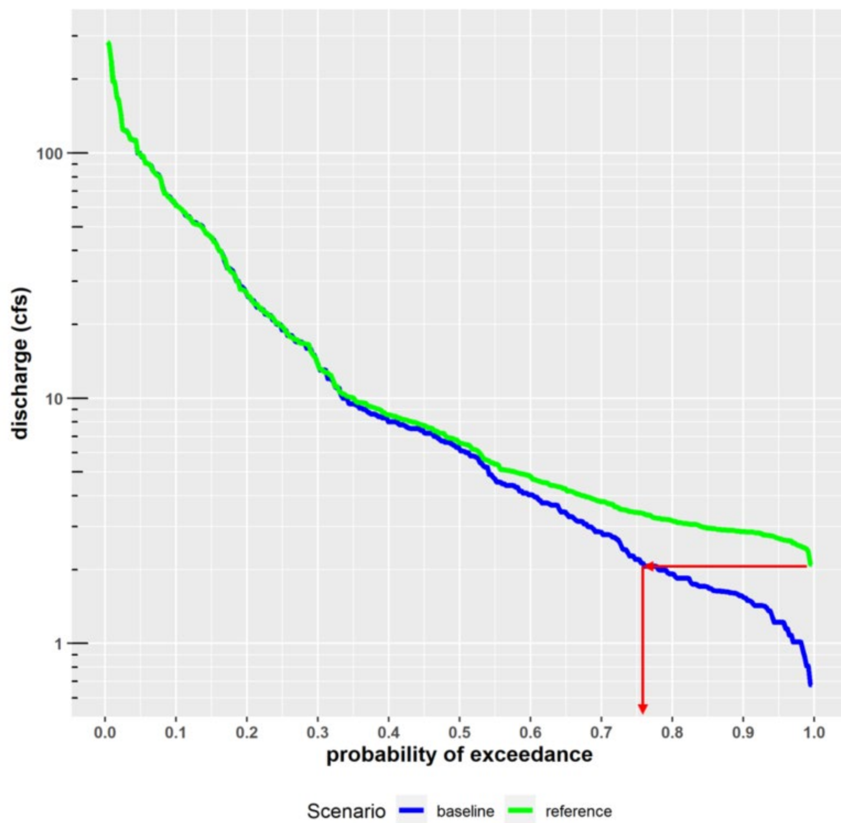



***Figure 12. August time series for reference and baseline conditions.***
*The August time series for both the reference (estimated flow without groundwater depletion) and baseline (observed flow with groundwater pumping occurring) condition, are displayed as a flow duration curve. The red lines show that the lowest flow, (2.1 cfs, occurring 0% of the time), for the reference condition now occurs 24% of the time (exceedance value = 0.76 for the baseline condition). The median streamflow for the reference condition is 6.6 cfs and 6.2 cfs for the baseline condition.*

Mean flow and median flows show only a slight change between reference and baseline condition, decreasing by 2.8% and 5.8%, respectively (**Table 8**). Base flows are decreased from the reference condition by approximately 10% (**Table 8**). At lower flows (e.g., >75% exceedance), groundwater pumping is estimated to decrease the flows in Little Rock Creek significantly, reducing flows from the reference condition by approximately 40% or more (**Figure 12**). The lowest flows are impacted further; the minimum reference flow for the period of record, 2.1 cfs, is now under 0.7 cfs for the baseline condition, a decrease of approximately 67% (**Table 8**). The number of days at low flows has increased substantially: the percent increase in number of days at the 80% exceedance flow, from reference to the current baseline condition, is roughly 66%. At the 90% exceedance flow, the frequency of these low flows increases by greater that 200%.

Recurrence of the 3-day 10 year low flow dropped nearly 72%, from 10 years to every 2.84 years. Change in recurrence for the 7-day 10 year low flow was similar, dropping about 71% - from recurring every 10 years to every 2.93 years. Since many freshwater stream fish species have lifespans at or around 3 years, this change in recurrence of low flows potentially has significant implications. **Table 8** provides a range of normal to low flow statistics that have been identified as hydrologically and biologically relevant (Richter et al. 1996, Poff et al. 1997, Richter et al. 1998, Smakhtin 2001, Richter et al. 2006). The statistics include an analog to the habitat metrics but using hydrologic data. For example, trimMeanQ is the trimmed (Q10 to Q90) mean discharge similar in calculation to Habitat Metric 1; trimMeanLowQ is the trimmed (Q80 to lowest Q) mean discharge similar in calculation to Habitat Metric 2; and, Q3dayQ10 is the lowest 3-day average discharge with a 10-year recurrence interval similar in calculation to Metric 3. The base flow for the reference condition (i.e., 5.5 cfs) is the value referred to in following discussions as the index flow or the ABF (i.e., August median base flow). The base flow values were obtained from MN DNR (2020).

Although the median estimated streamflow depletion from groundwater pumping was 0.64 cfs and less than 12% of the median August base flow (i.e., the index flow), it varied depending on the water year. Groundwater pumping occurs in response to climate conditions and so will be variable from year to year. The variability in use is mismatched with the availability of water creating some years with high use and others with no noticeable use. Calculated streamflow depletion exceeded the 20% of the index flow (August median base flow) four of 12 years (**Table 9**).

***Table 8. Hydrologic metrics for the Baseline and the Reference conditions.***

| Flow Level | Metric | Baseline | Reference | Percent Change [i] |
|---|---|---|---|---|
| | No. Observed | 372 | 372 | NA |
| | Median Diversion | 0.64 | NA | NA |
| Low | Min Q | 0.68 | 2.10 | **-67.62** |
| Low | Q90 | 1.57 | 2.86 | **-45.10** |
| Low | NdaysQ90Ref [a] | 115 | 38 | **202.63** |
| Low | Q80 | 1.92 | 3.18 | **-39.62** |
| Low | NdaysQ80Ref [b] | 125 | 75 | **66.67** |
| Low | Mean Low Q [d] | 1.47 | 2.82 | **-47.87** |
| Low | Q3dayQ10 [e] | 0.86 | 2.24 | **-61.61** |
| Low | Q7dayQ10 [f] | 0.99 | 2.36 | **-58.05** |
| Low | RI3day10Ref [g] | 2.84 | 10 | **-71.60** |
| Low | RI7day10Ref [h] | 2.93 | 10 | **-70.70** |
| Low | Base Flow | 4.90 | 5.50 | -10.91 |
| Moderate | Q50 | 6.20 | 6.58 | -5.78 |
| Moderate | Q10 | 60.73 | 60.82 | -0.15 |
| Moderate | Mean Q | 21.02 | 21.62 | -2.78 |
| Moderate | Mean Q[c] | 11.44 | 12.06 | -5.14 |

[a] number of days the Reference Q90 is not exceeded
[b] number of days the Reference Q80 is not exceeded
[c] mean of the daily flow values between the 10 and 90% exceedance
[d] mean of the daily flow values less than the 20% exceedance
[e] lowest 3 day average flow that occurs on average once every 10 years
[f] lowest 7 day average flow that occurs on average once every 10 years
[g] return interval (years) for the reference condition lowest 3 day Q10
[h] return interval (years) for the reference conditions lowest 7 day Q10
[i] (baseline-reference)/(reference)*100

*The Baseline is observed, and the Reference is the no use condition. Flow conditions are displayed in cfs and return intervals in years. Percent change values greater than 20% are bolded.*

***Table 9. Percent of August median base flow diverted by groundwater pumping.***

| Year | Percent ABF |
|---|---|
| 2006 | **24.3** |
| 2008 | **23.9** |
| 2009 | 10.6 |
| 2010 | 4.5 |
| 2011 | -11.2 |
| 2012 | **21.9** |
| 2013 | **25.4** |
| 2014 | 17.1 |
| 2015 | 3.1 |
| 2016 | 3.1 |
| 2017 | -5.6 |
| 2018 | 13.0 |

*The Percent of the August median base flow (ABF=5.5 cfs) estimated to be diverted by groundwater pumping, based on daily data. A negative value indicates that return flows are adding water to the stream. Calculated as: mean ((Reference-Baseline)/ABF*100). Values greater than 20% are bolded.*

## 3.3 Hydraulic Modeling

### 3.3.1 River2D Flow Model Results

A total of 14 flow simulations were run through 14 individually calibrated River2D models of the Little Rock Creek study site. The flows ranged from 1.22 to 56.37 cubic feet per second and are listed in **Table 10** with the discharge to water surface elevation relationship used for modeling. A rating curve from the observed measurements predicted the estimated values.

A stable simulation was reached for each of the 14 modeled flows with the model inputs, parameters, and simulation results for each within an acceptable range as specified in the methods (specific values can be seen in **Appendix E** for each model). The difference between the simulated and estimated or measured inflow WSEL for each model were within 0.03 m (0.1 ft). Then Net Q (difference between the discharge coming into and out of the models) were all within 0.003 cms (0.1cfs) which indicated the model steady state solutions all converged. There were no unusually high maximum velocities, and all Froude numbers were under 1, indicating subcritical flow was simulated as expected. The solution change for each model was under 0.00001, indicating the flow model was converged and stable. Finally, the mesh quality index for each flow model was within the acceptable range of 0.15-0.5.

Validation of the models through comparison of simulations WSELs throughout the modeled stream reach to observed measurements show very good agreement. Of the four observed sets of measured WSELs, the differences between the simulated and measured WSELs were within 0.01 standard deviations, well within the 0.03 m threshold established in the methods. **Appendix F** Details the number of WSEL observations at each flow that were compared for validation, the average and maximum difference between values, and the standard deviation of the differences. Simulated depth and velocity at the observed flows were plotted against measured values from the discharge profiles and were visually inspected for acceptance. **Appendix F** includes a plot of measured and simulated depth and velocity compared for the highest measured flow. Some variability was expected between simulated and observed depths and velocities because the extraction of the simulated cross-sections were

***Table 10. Fourteen modeling flows simulated in River2D.***

| Type of measurement | Date of Measurement | Q at inflow (cfs) | Q at inflow (cms) | WSEL at outflow (feet) | Water surface elevation at inflow (feet) |
|---|---|---|---|---|---|
| estimated | | 56.37 | 1.597 | 1073.445 | 1075.709 |
| estimated | | 25.97 | 0.735 | 1073.314 | 1074.954 |
| observed | 15-Aug-2017 | 22.81 | 0.646 | 1073.281 | 1074.889 |
| estimated | | 13.39 | 0.379 | 1073.051 | 1074.626 |
| observed | 14-Sep-2017 | 9.59 | 0.272 | 1072.920 | 1074.528 |
| estimated | | 8.00 | 0.227 | 1072.822 | 1074.495 |
| estimated | | 6.20 | 0.176 | 1072.723 | 1074.462 |
| observed | 1-Aug-2019 | 4.90 | 0.139 | 1072.625 | 1074.462 |
| estimated | | 4.03 | 0.114 | 1072.592 | 1074.396 |
| estimated | | 2.85 | 0.081 | 1072.494 | 1074.364 |
| observed | 9-Aug-2019 | 2.75 | 0.078 | 1072.494 | 1074.331 |
| estimated | | 1.92 | 0.054 | 1072.395 | 1074.364 |
| estimated | | 1.56 | 0.044 | 1072.362 | 1074.331 |
| estimated | | 1.22 | 0.035 | 1072.329 | 1074.331 |

*A discharge to water surface relationship was used to simulate fourteen flow conditions in River2D. Four flows (Q) and water surface elevations (WSELs) were measured in the field, while 10 were estimated from a Q:WSEL rating curve built from the four observed flows. Modeling flows range from 1.22 to 56.37 cfs. This is the standard range of flows that can be estimated from observed flows. Milhous et al. (1981) states that flows between 0.4 times the lowest measured discharge, to 2.5 times the highest can be modeled using PHABSIM.*

approximate. The discharge profile locations across the stream channel were not explicitly positioned with GPS in the field. Small differences in positioning can explain a lot of variability in depth and velocities throughout a stream, but comparison of measured and simulated valued at the cross sections showed good association.

Depth and velocity were simulated for the 14 flows listed in **Table 10**. **Figures 13 and 14** show depth and velocity simulations for the Little Rock Creek modeling site for the lowest observed flow of 2.75 cfs. Depth and velocity simulations for all fourteen flows can be found in **Appendix G** along with simulated combined habitat suitability for two species-life stages modeled.

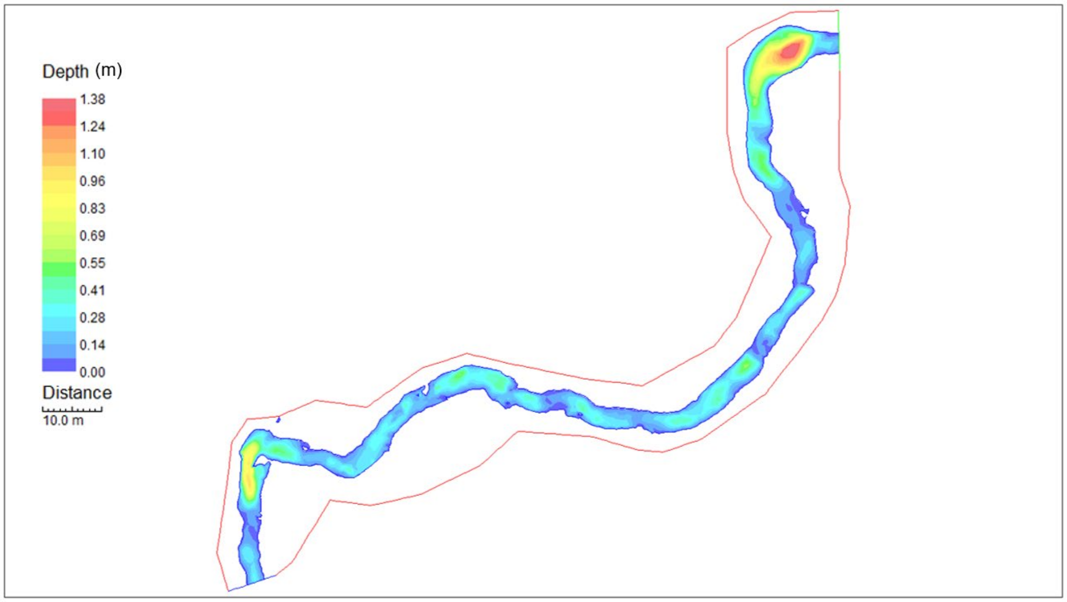



***Figure 13. River2D simulation of depths.***
*Simulation of depths in meters throughout the Little Rock Creek Study area at the lowest observed flow, 2.75 cfs.*

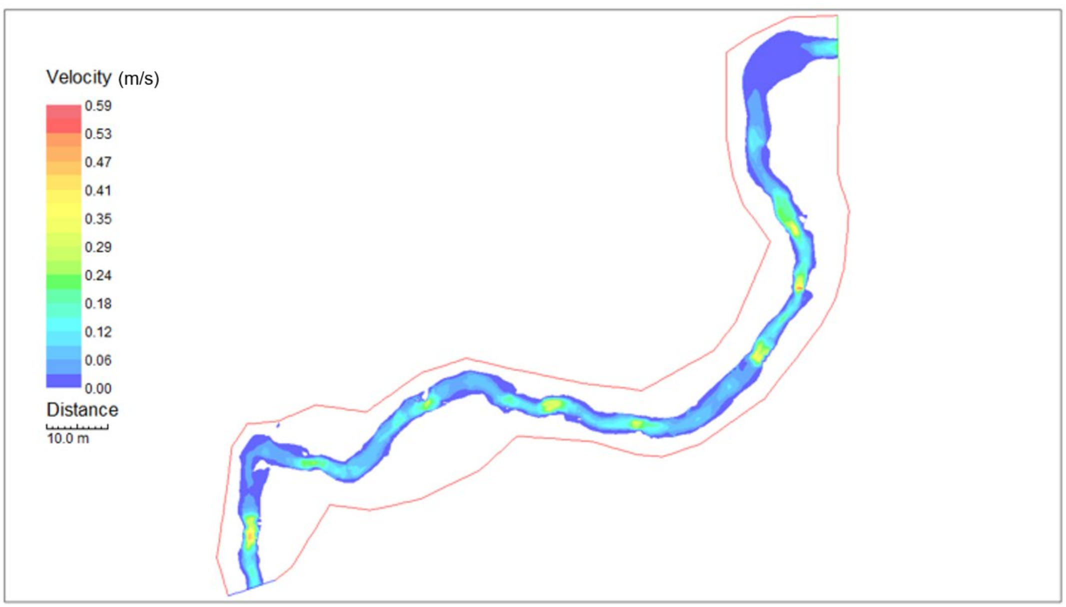



***Figure 14. River2D simulation of velocities.***
*Simulation of velocities in meters per second throughout the Little Rock Creek Study area at the lowest observed flow, 2.75 cfs.*

Hydraulic data at each of the 14 study flows in Little Rock Creek were combined with the depth and velocity requirements for each of the six habitat guilds to display the distribution of each habitat type across the range of flows (**Figure 15**). **Figure 15** provides a qualitative view of the diversity of habitat conditions. At the highest flows (bottom row of graph), habitat exists for all guilds, including deep pool species. As flows decrease below 23 cfs to 8 cfs, (second row up from bottom) deep pool habitat and race way habitat is no longer present. At the same time, occurrences of habitat in the lowest depths and velocities, represented by shallow pool guild, is increasing, as shown by the increasing presence of the yellow hexagons in this row of plots. Habitat distribution for flows between approximately 6 and 3 cfs is shown by the next row of plots ($3^{rd}$ row up from the bottom). Recall that August median flow is 6.2 cfs, and August median base flow is 4.9 cfs for the baseline condition in Little Rock Creek at the study site (stream gage 15029003). As flows decrease to 2.85 cfs, fast riffle habitat is lost and the occurrence of medium pool habitat becomes scarcer. More and more habitat occurs as shallow pool, indicated by the intensity of yellow hexagons in the lower left corner of each plot. When flows drop below 3 cfs (top row of plots), habitat decreases at the same time, until at approximately 1.5 cfs, all habitat is shallow or medium pool. The red hexagon positioned in the lowest depths and velocities of the shallow pool category denotes that nearly all habitat occurs in this hydraulic range when flows go below 1.5 cfs. The habitat is almost exclusively shallow pool and within that guild is dominated by velocities less than 0.1 m/s (i.e., 0.3 ft/s) and depths less than 0.3 m (i.e., 1 ft).

The wetted area of the stream reveals a similar relationship to flow when flows decrease below 10 cfs (**Figure 16**). Wetted area decreases sharply at flows less than the inflection point, at approximately 10 cfs, and again at 3.2 cfs - the 80% exceedance flow and the steepest section of the line (see inset of **Figure 16**). Because of the steepness of the area – discharge curve, a small magnitude change, for example, from 2.86 to 1.57 cfs (the reference and baseline Q 90, respectively) results in a 60% decrease in wetted area, from 3729 $ft^2$ to 1500 $ft^2$. This rapid loss in wetted area and change in hydraulic habitat as flows drop below 3 cfs indicates a vulnerability and potential ecological bottleneck.

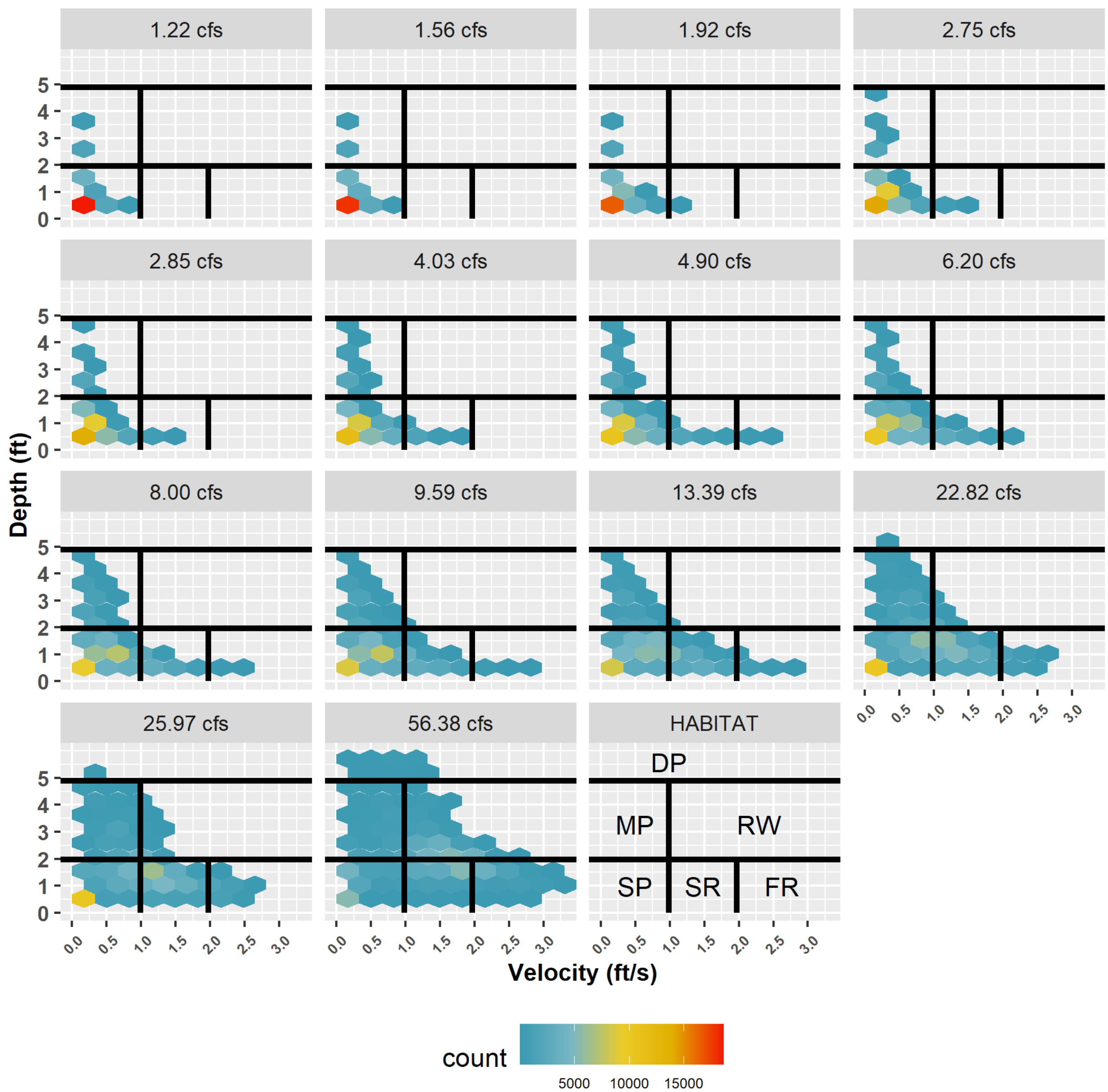



***Figure 15. Distribution and frequency of hydraulic conditions.***

*Depth and velocity distribution and frequency are shown for each modeled flow discharge (cfs) indicated in the title bar of each box. Habitat guilds are delineated by the solid black lines and defined in the last subplot (bottom row, third plot from left). Each dot (hexagon) is color coded to represent a number of cells that have the hydraulic conditions indicated on the x and y-axes. For production of the graphic the modeled area was subdivided into cells each representing 0.03 m$^2$ (0.32 ft$^2$) of stream.*

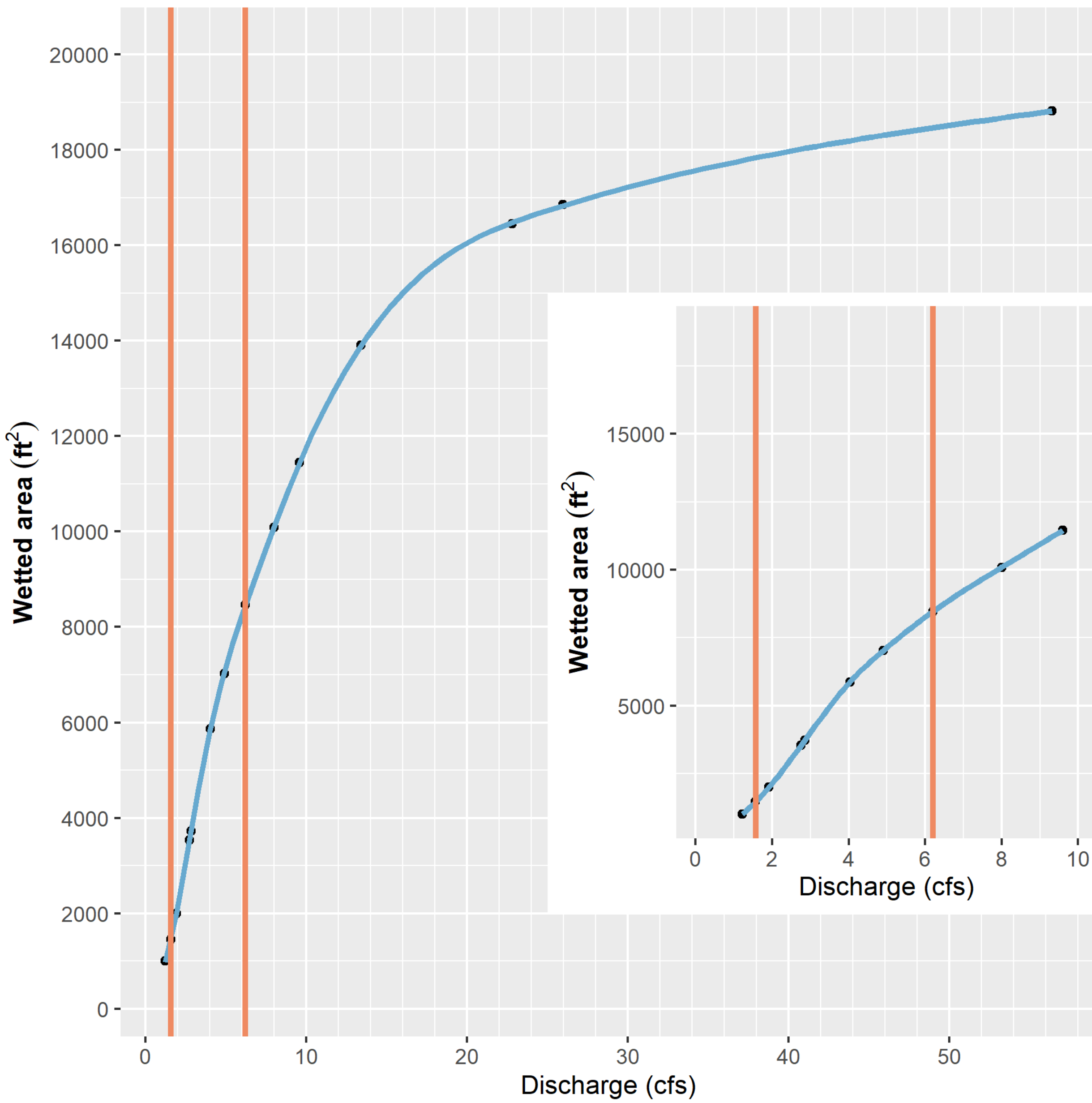



***Figure 16. Area - discharge relationship for the Little Rock Creek study area.***

*Based on modeled flows and area defined as depth greater than zero and a velocity of 0.1 ft/s (0.03 m/s), which is the velocity of one tenth of the defining starting point for a slow riffle. The vertical lines are drawn at the baseline condition Q50 (6.2 cfs) and Q90 (1.6 cfs). The inset graph displays the steep slope range of the graph (i.e., discharge less than 10 cfs).*

## 3.4 Fish Habitat Modeling

A key piece of the analysis of ecosystem sustainability in Little Rock Creek is the results of the habitat modeling. The relationship of fish habitat to discharge during August in Little Rock Creek is examined for each habitat guild species and provides a site-specific basis for establishing the impact of flow changes on the ecology of this stream. **Figure 17** presents the relationship of flow to habitat for the 11 species-life stages that represent the range of habitat conditions for aquatic organisms. Each species-life stage is represented by a species and life stage code defined in **Table 7**. Taken collectively, these relationships represent the range of habitat diversity and therefore the ecology of this stream. The species-life stages modeled show a range of responses to flow (**Figure 17**). For shallow pool organisms, represented by creek chub young-of-year, and Johnny darter young-of-year, peak habitat is under low flow conditions. As discharge increases, shallow pool habitat declines and their flow/habitat curve reflects that decrease. Medium pool organisms, represented by white sucker juveniles and common shiner adults, have habitat increase as flow increases, up to moderate discharge levels, and then begin to drop off as flows increase further. Deep pool organisms, represented by common shiner juvenile in Little Rock Creek, show a similar pattern of increasing habitat with increasing flow to moderate discharge levels (10-25 cfs), but do not lose habitat as rapidly when flows increase past that point. Raceway and fast riffle organisms typically prefer fast water. As discharge increases, raceway organisms, represented by brown trout adults and brown trout juveniles, show corresponding habitat increases, and maximum habitat at the maximum flow. This same pattern is revealed for fast riffle organisms, as represented by longnose dace and blacknose dace. For fast riffle organisms, habitat increases as flow increases, to the maximum flow modeled. White sucker young-of-year and Johnny darter adults represent slow riffle organisms and show a positive habitat response as flows increase to moderate levels (approximately 10-25 cfs) and begin to decline as flow increase past that point.

A plot of the relationship of habitat area, for each of the six (6) habitat types, to flow is matched with a plot of flow duration curves (bottom graph) for the reference and baseline conditions in **Figure 18**. Together, they illustrate each habitat type's pattern of response, and the collective fate of habitat types in Little Rock Creek, as flows decrease. Greatest habitat diversity exists at the study site when discharge is above 25 cfs. Four of the six habitat types are present at flows between approximately 4 and 15 cfs. Deep pool habitat (light blue line, top plot) constitutes the lowest amount of habitat at the study site, existing at higher to medium flows, and disappearing by the time flows decrease below 20 cfs. At the highest modeled flow (56 cfs), shallow pool, raceway, and slow riffle habitats make up the top three (3), respectively, in terms of area. Raceway and fast riffle habitat decline rapidly as flows decrease from that point. Depletion of flows from groundwater pumping further decreases habitat frequency relative to the reference condition. Raceway habitat (dark green line) is lost at flows below 15 cfs, and fast riffle habitat (dark blue line) disappears as flows decrease below 4 cfs. For example, the 4 cfs under the reference condition has an approximate 67% exceedance value (see bottom graph, **Figure 18**). Under the baseline condition (i.e., with pumping) it occurs about 0.6 or 60% of the time. So some amount of fast riffle was present 67% of the time under reference conditions and only 60% of the time under the baseline condition.

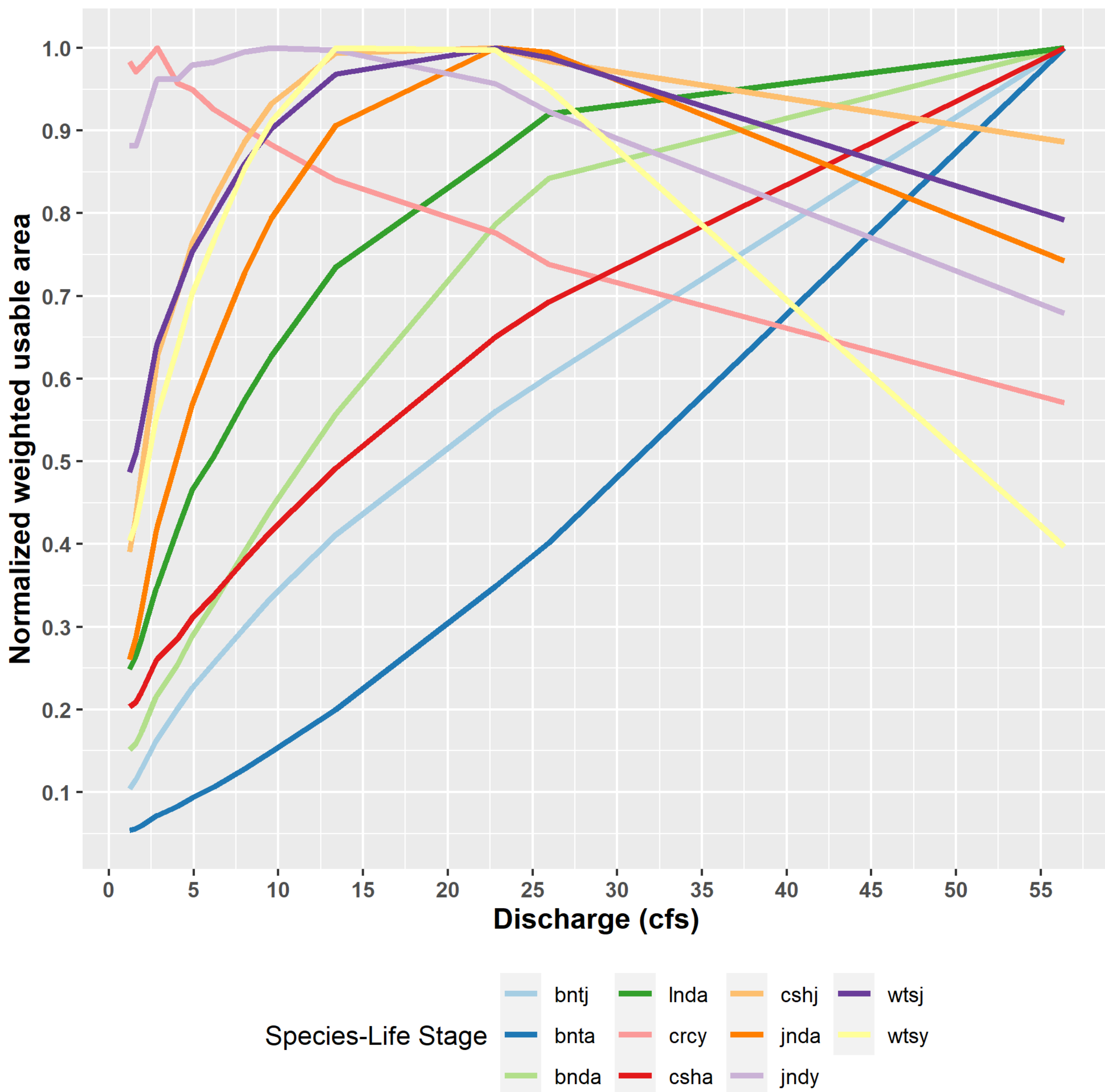



***Figure 17. The flow - ecology relationship in Little Rock Creek.***
*The normalized WUA for each of the 11 modeled species-life stages are shown. Species-life stage codes are defined in Table 7; column 4. Habitat values are normalized to their maximum habitat area.*

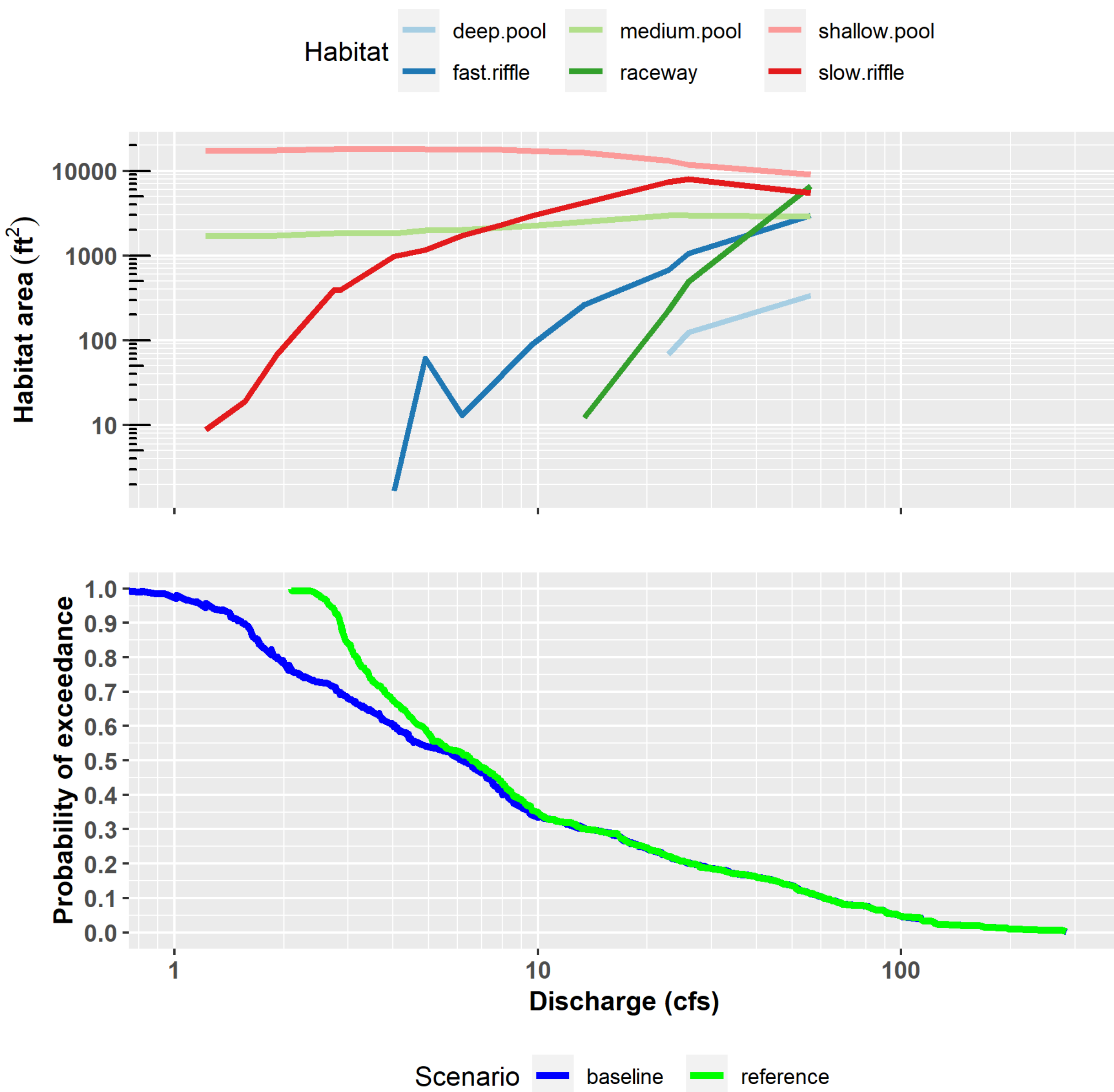



***Figure 18. Flow-ecology habitat guild relationship and flow duration curve.***
*Top graph is a flow - ecology relationship for each habitat guild. Habitat guilds are defined using specific combinations of depth and velocity. The bottom graph is a flow duration curve with exceedance plotted on the y-axis to show the effect of flow on habitat for each guild. The discharge values for the two graphs correspond vertically. For example, 2 cfs has an exceedance of 0.77 in the baseline conditions and approximately 7 square meters of slow riffle habitat.*

Shallow pool and medium pool habitat remain relatively unchanged as flow changes, throughout the range of modeled flows. Shallow pool habitat actually increases, slightly as flows decrease below 25 cfs. Slow riffle habitat decreases rapidly below approximately 4 cfs, but persists, at low levels, under the lowest flow conditions. Slow riffle habitat at 3 cfs, approximately 30 $m^2$ in the study area, occurs 85% of the time under reference conditions, and approximately 67% of the time under baseline conditions, a reduction in frequency of 18%.

### 3.4.1 Habitat Time Series

**Figures 19-24** display time series of weighted usable area followed by a habitat duration curve for three species; blacknose dace (fast riffle guild), brown trout juvenile (raceway guild), and Johnny darter adult (slow riffle guild). All other species-life stages modeled are displayed in **Appendix H**. All habitat graphs display only the month of August habitat conditions. The black line on the habitat duration curves delineates a 20% decrease in habitat at each value for the reference condition. Habitat time series plots are useful for qualitatively reviewing habitat changes between alternative scenarios. Despite the nonlinear response of habitat area to discharge (**Figure 17**), the pattern in each habitat time series generally matches flow magnitude patterns and the relationship between scenarios displayed in the hydrograph (**Figure 11**). The change in habitat between the reference and baseline conditions are most pronounced in the years with the lowest discharge; 2006, 2008, 2012, and 2013. The shape and inter-relationship between scenarios in the habitat duration curves is similar to the patterns in the flow duration curves. For each habitat duration curve (**Figures 20, 21, 22**), the baseline habitat conditions cross the 20% loss line at between the 70% and 75% exceedance. In addition to the reduced area of habitat in the baseline condition, the frequency of the low habitat conditions increased. The baseline frequency of the reference minimum for eight of the 11 species increased to 24% of the days, the two shallow pool species (crcy and jndy) increased to 11 and 12% of the days, respectively and one slow riffle species (wtsy) increased to 15%.

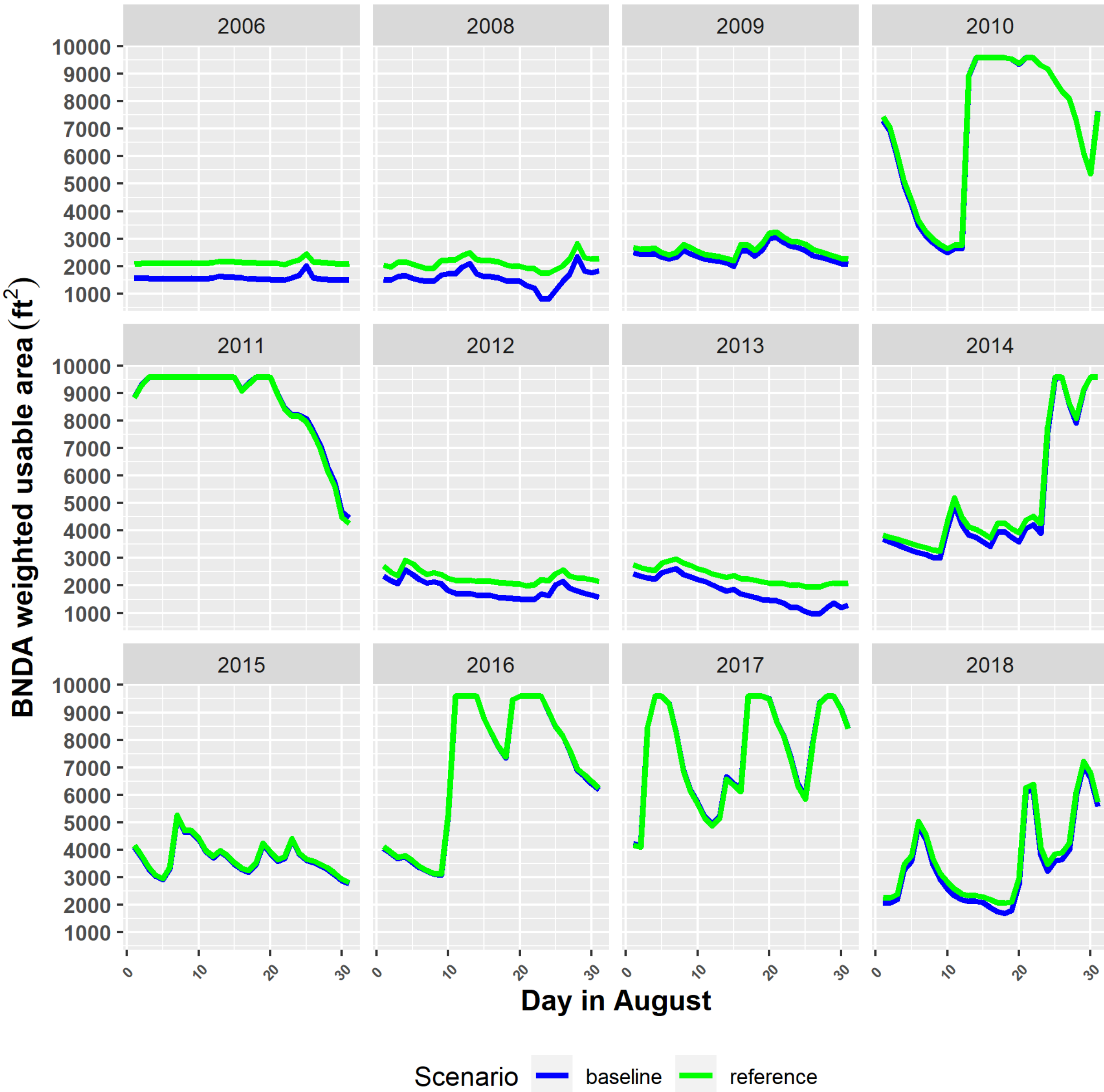



***Figure 19. August habitat time series for blacknose dace adult, a fast riffle habitat guild representative.***

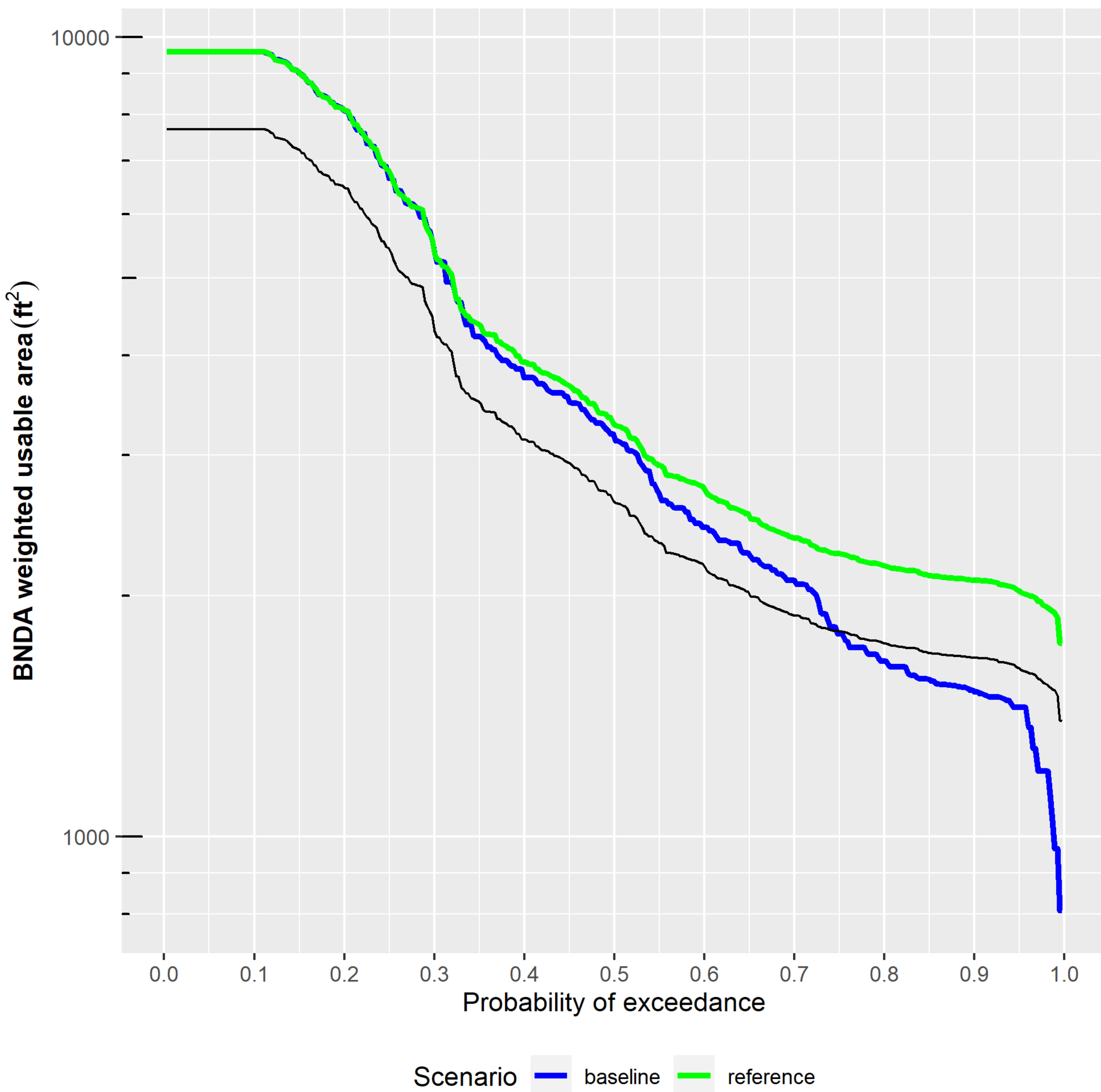



***Figure 20. August habitat duration curve for blacknose dace adult, a fast riffle habitat guild representative.*** *The black line represents a 20% decrease in habitat area.*

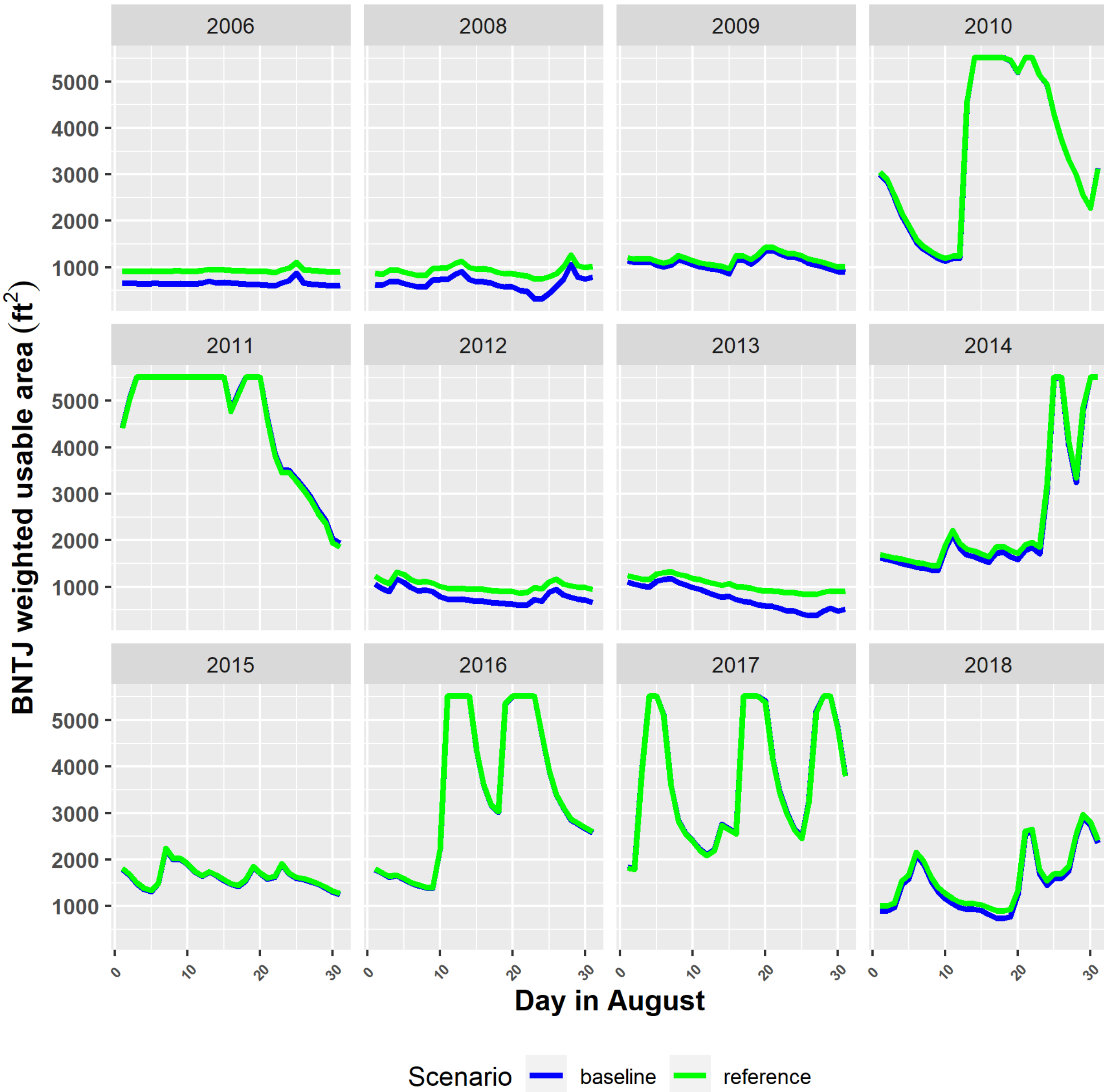



*Figure 21. August habitat time series for brown trout juvenile, a raceway habitat guild representative.*

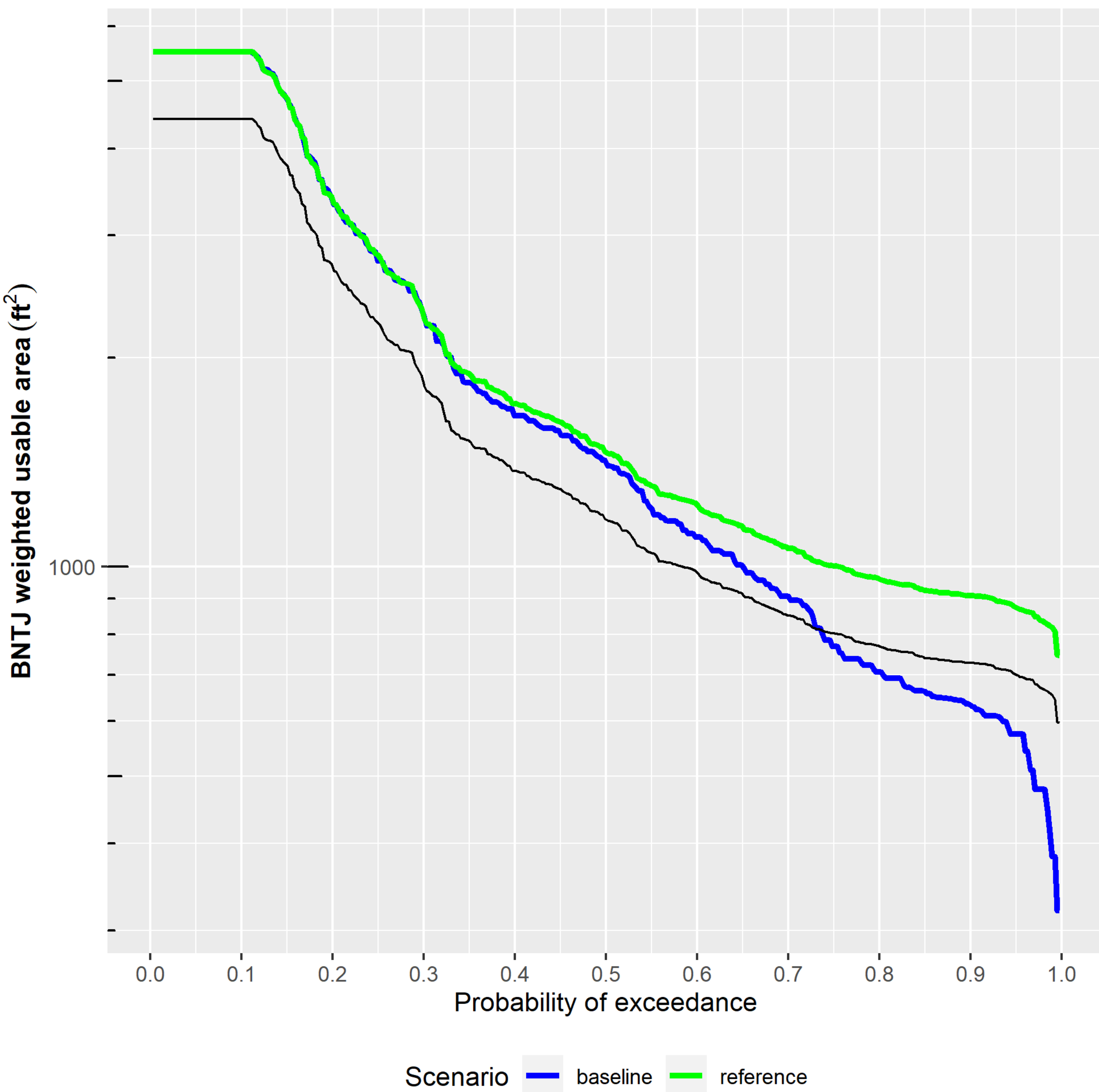



***Figure 22. August habitat duration curve for brown trout juvenile, a raceway habitat guild representative.*** *The black line represents a 20% decrease in habitat area.*

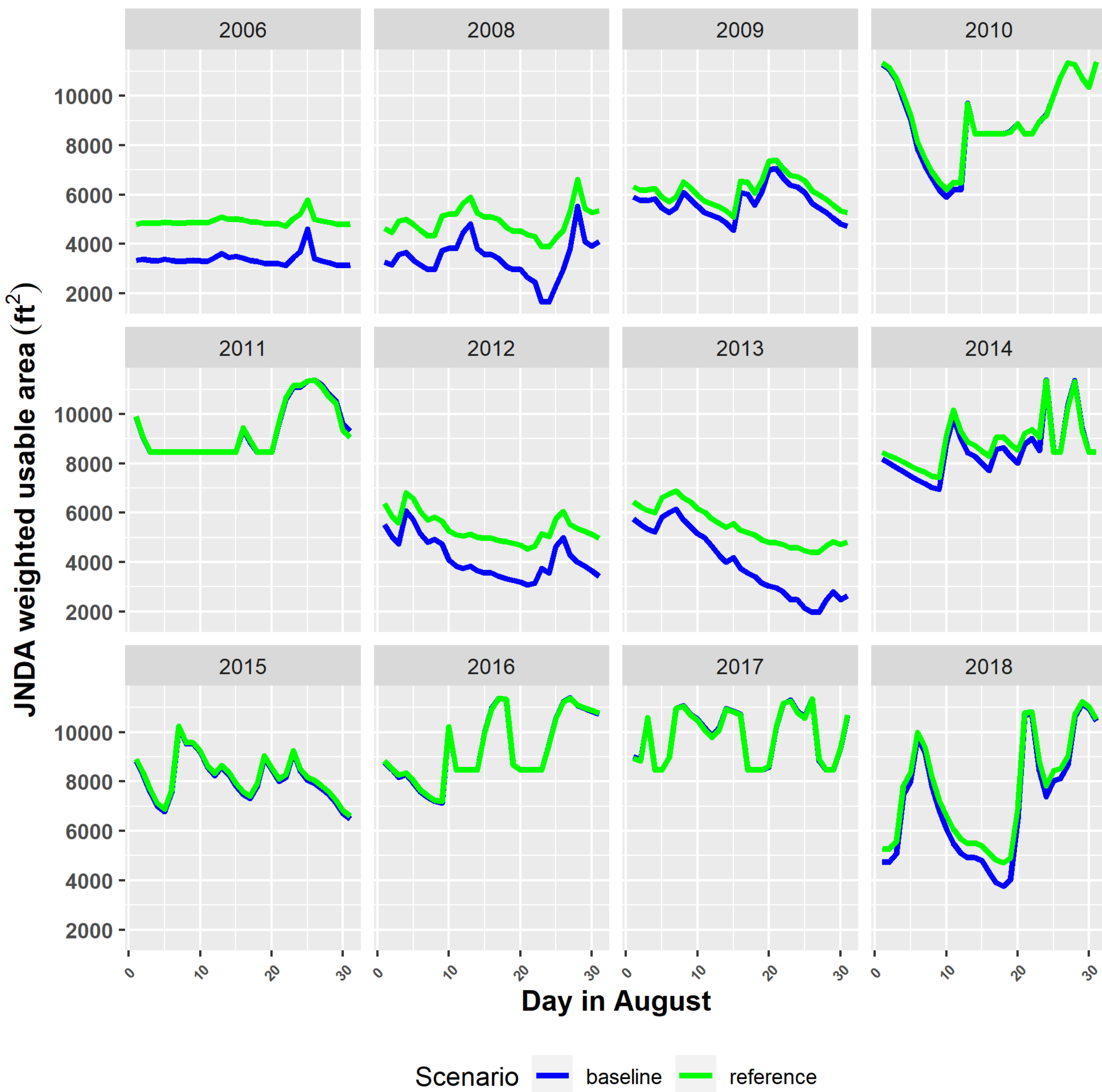



*Figure 23. August habitat time series for Johnny darter adult, a slow riffle habitat guild representative.*

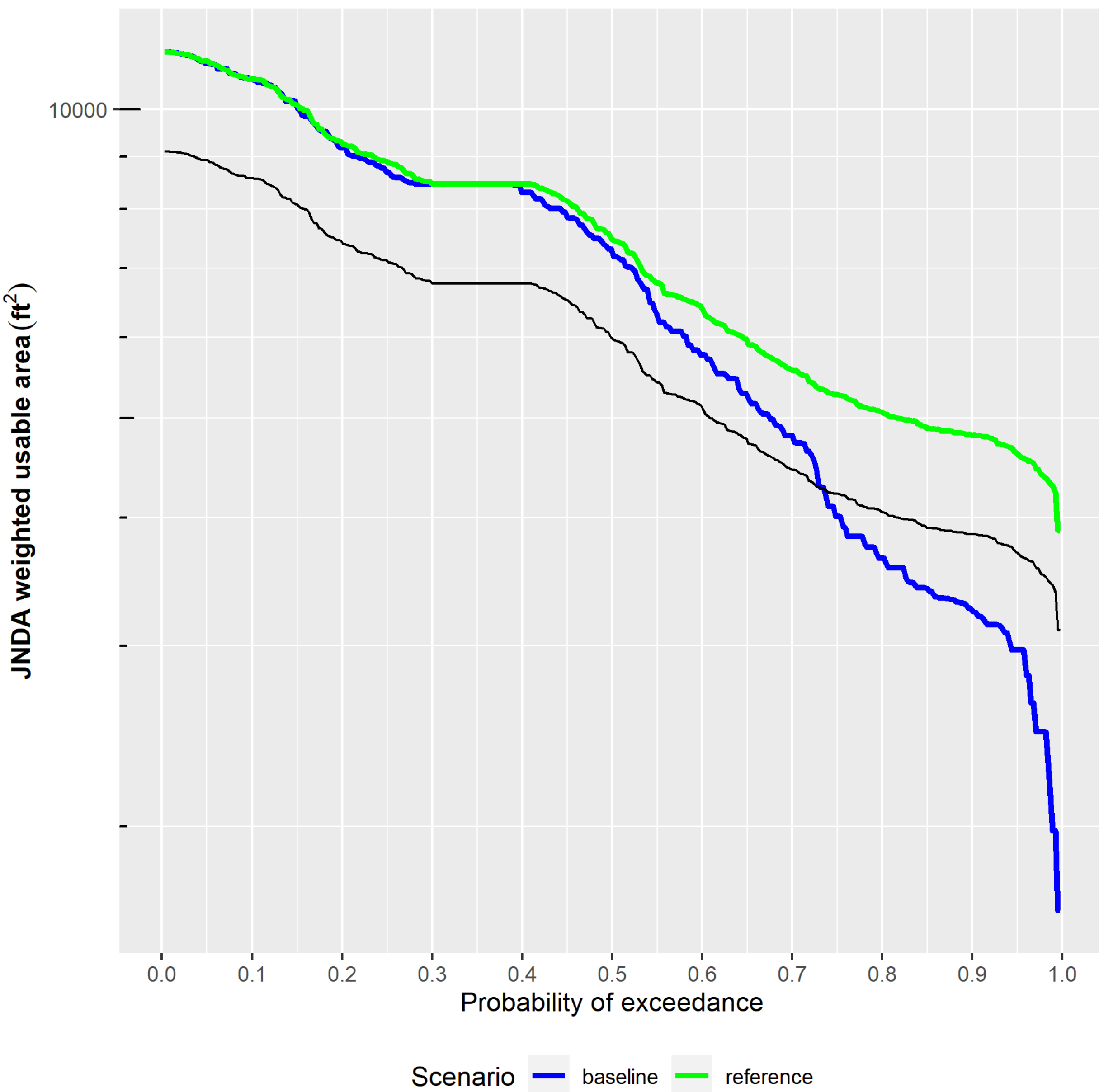



***Figure 24. August habitat duration curve for Johnny darter adult, a slow riffle habitat guild representative.*** *The black line represents a 20% decrease in habitat area.*

An additional quantitative assessment of the habitat time series reveals that five of the six habitat guilds experienced greater than a 20% decline in habitat area annually for 2-4 years when comparing the reference condition to the baseline condition (**Table 11**). In three of the habitat guilds (i.e., fast riffle, raceway, and slow riffle), both representative species-life stages experienced a greater than 20% reduction in at least 3-4 years. In the comparison of reference to baseline conditions, three species (CRCY, JNDY, CSHA) did not experience an average annual decrease of 20% in habitat area.

***Table 11. The average percent decrease in habitat area in years.***

| Guild | Species-Life Stage | Average Percent Decrease | Years >20% Decrease |
|---|---|---|---|
| Fast Riffle | BNDA | 25.82 | 2006, 2008, 2013 |
| Fast Riffle | LNDA | 24.41 | 2006, 2008, 2013 |
| Raceway | BNTA | 22.29 | 2006, 2008, 2013 |
| Raceway | BNTJ | 27.19 | 2006, 2008, 2012, 2013 |
| Slow Riffle | JNDA | 28.06 | 2006, 2008, 2012, 2013 |
| Slow Riffle | WTSY | 23.05 | 2006, 2008, 2013 |
| Medium Pool | WTSJ | 20.37 | 2006, 2008 |
| Deep Pool | CSHJ | 27.82 | 2006, 2008, 2013 |

*The years listed are those in which the habitat decreased on average by more than 20% from the reference condition to the baseline condition by species-life stage.*

### 3.4.2 Habitat Metrics

The habitat metrics were analyzed by comparing the amount of habitat under the reference condition to the baseline condition. The metrics represent a summary of habitat based on the habitat time series and the habitat duration curve. Each metric is calculated for each species-life stage representing each of the six habitat guilds. The species-life stages representing the guild also represents all of the organisms that require the depth, velocity and substrate conditions in the habitat guild. As a result, a 20% or greater decrease in a habitat metric identifies significant harm to the stream ecosystem.

Metric 1 (i.e., normal habitat conditions), showed little impact of streamflow depletion from the reference condition to the baseline condition for all species-life stages (**Figure 25** and **Table 12 and 13**). The largest change from the reference to the baseline condition was an approximately 7% drop for the slow riffle species.

Metric 2 (i.e., low habitat conditions) identified impact levels exceeding significant harm (i.e., greater than 20% reduction) in five of the six habitat guilds and eight of the 11 species-life stages (**Figure 25** and **Table 12 and 13**). No representative of the shallow pool habitat guild experienced significant harm. The reduction in Metric 2 exceeded 30% in three species-life stages representing three habitat guilds (raceway, slow riffle, and deep pool).

Metric 3, like Metric 2, assesses impacts under low habitat conditions, but measures the frequency and duration of the low habitat events (**Table 14**). Metric 3 both measures the magnitude of change of the 3-day 10 year event and presents the new frequency (return interval) for the reference condition 3-day 10 year under the baseline condition. Metric 3 results exceed a 20% reduction for all habitats and species-life stages with a maximum of a 52% reduction in the 3-day 10 year habitat condition for a raceway representative, brown trout juvenile. Additionally, the return interval for the reference condition 3-day 10 year return interval is now 3 years or less for all representatives of fast riffle, raceway, shallow pool, and deep pool habitats.

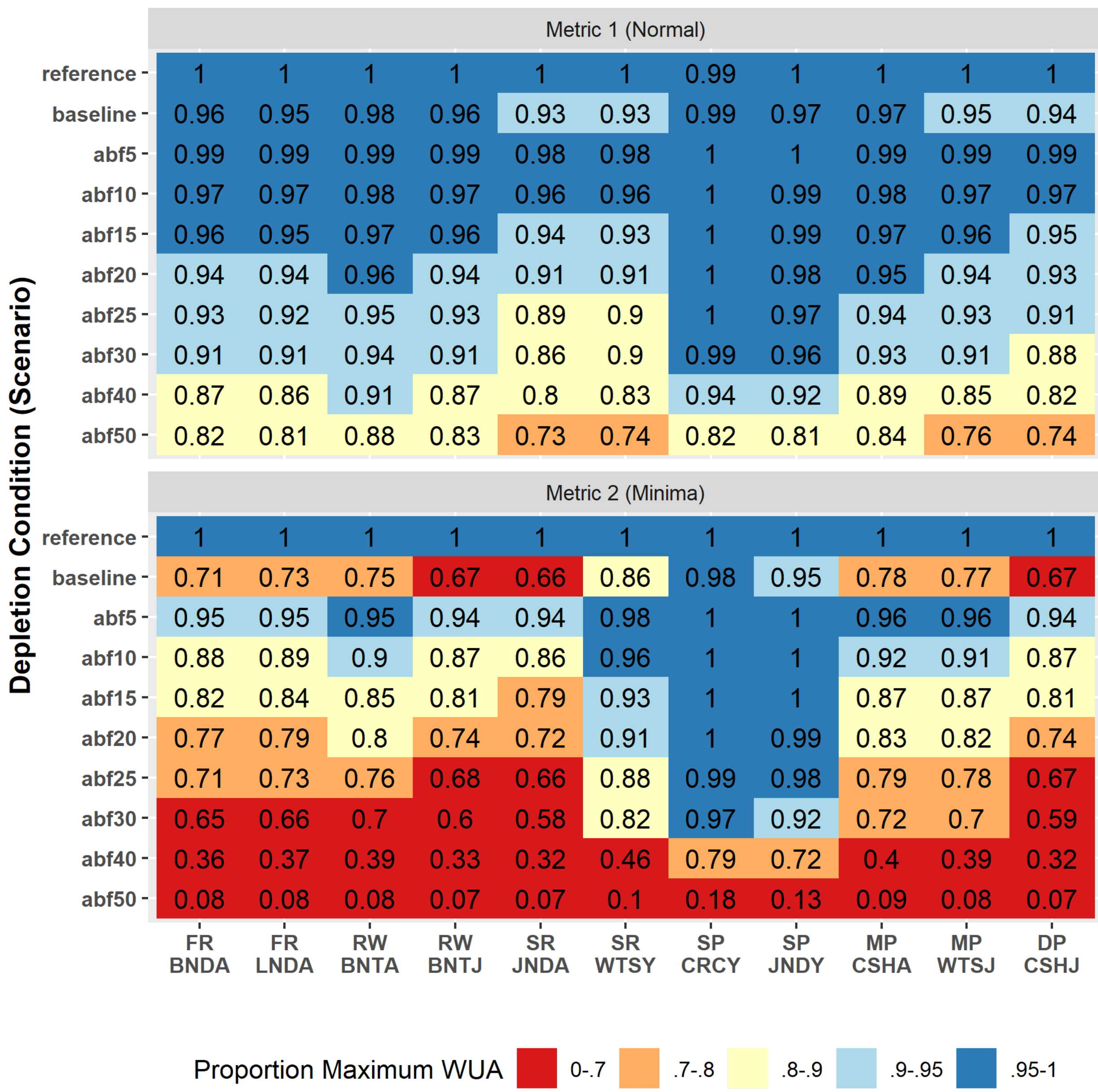



***Figure 25. Matrix displaying change in habitat for each modeled species-life stage and flow condition during August.***

*The top matrix displays the change in normal habitat conditions (Metric 1) and the bottom matrix the change in habitat minima (Metric 2). Habitat conditions are normalized to each species-life stage maximum. The abbreviation abf refers to August median base flow and the number equates to the percentage of the ABF that the daily streamflows are reduced. The x-axis labels display the guild and species-life stage. FR=fast riffle, RW=raceway, SR=slow riffle, SP=shallow pool, MP=medium pool, and DP=deep pool.*

*Table 12. Habitat (WUA: weighted usable area) change between reference and baseline conditions for Metric 1 (Normal).*

| Guild | Species-Life Stage | Reference WUA ($ft^2$) | Baseline WUA ($ft^2$) | Percent Change |
|---|---|---|---|---|
| Fast Riffle | BNDA | 4814 | 4616 | -4.1 |
| Fast Riffle | LNDA | 1489 | 1420 | -4.7 |
| Raceway | BNTA | 1539 | 1505 | -2.2 |
| Raceway | BNTJ | 2316 | 2222 | -4.1 |
| Slow Riffle | JNDA | 7302 | 6774 | -7.2 |
| Slow Riffle | WTSY | 4082 | 3804 | -6.8 |
| Shallow Pool | CRCY | 7949 | 7897 | -0.7 |
| Shallow Pool | JNDY | 11583 | 11273 | -2.7 |
| Medium Pool | CSHA | 6077 | 5880 | -3.2 |
| Medium Pool | WTSJ | 7734 | 7343 | -5.0 |
| Deep Pool | CSHJ | 9026 | 8451 | -6.4 |

*Table 13. Habitat (WUA: weighted usable area) change between reference and baseline conditions for Metric 2 (Minima).*

| Guild | Species-Life Stage | Reference WUA ($ft^2$) | Baseline WUA ($ft^2$) | Percent Change |
|---|---|---|---|---|
| Fast Riffle | BNDA | 2070 | 1465 | -29.2 |
| Fast Riffle | LNDA | 822 | 598 | -27.2 |
| Raceway | BNTA | 371 | 279 | -24.8 |
| Raceway | BNTJ | 899 | 605 | -32.7 |
| Slow Riffle | JNDA | 4749 | 3119 | -34.3 |
| Slow Riffle | WTSY | 2798 | 2403 | -14.1 |
| Shallow Pool | CRCY | 5689 | 5621 | -1.2 |
| Shallow Pool | JNDY | 9273 | 8799 | -5.1 |
| Medium Pool | CSHA | 3200 | 2485 | -22.3 |
| Medium Pool | WTSJ | 6312 | 4838 | -23.3 |
| Deep Pool | CSHJ | 6918 | 4632 | -33.0 |

*Changes greater than 20% are bolded.*

***Table 14. Habitat (WUA: weighted usable area) change between the reference and baseline conditions for Metric 3 (3dHab10).***

| Guild | Species-Life Stage | Reference WUA (ft$^2$) | Baseline WUA (ft$^2$) | Percent Change | New Return Interval |
|---|---|---|---|---|---|
| Fast Riffle | BNDA | 1805 | 947 | -47.5 | 2.7 |
| Fast Riffle | LNDA | 722 | 396 | -45.1 | 3.0 |
| Raceway | BNTA | 332 | 188 | -43.1 | 2.6 |
| Raceway | BNTJ | 767 | 368 | -52.1 | 2.9 |
| Slow Riffle | JNDA | 4041 | 1967 | -51.3 | 3.2 |
| Slow Riffle | WTSY | 2404 | 1873 | -22.1 | 2.4 |
| Shallow Pool | CRCY | 1805 | 947 | -47.5 | 2.7 |
| Shallow Pool | JNDY | 9379 | 7370 | -21.4 | 2.1 |
| Medium Pool | CSHA | 2927 | 1765 | -39.7 | 2.7 |
| Medium Pool | WTSJ | 5831 | 3584 | -38.5 | 3.1 |
| Deep Pool | CSHJ | 6128 | 3131 | -48.9 | 3.0 |

*Metric 3 is the change in the 3-day minimum habitat with a 10-year return interval. New Return Interval is the return interval of the reference 3 day minimum 10 year for the baseline condition. For example, 167.7 is the 3-day minimum 10-year return interval habitat for BNDA under the reference condition, under the baseline condition 947 is the 3-day minimum habitat with a 2.7-year return interval. Changes greater than 20% are bolded.*

# 4. Discussion

## 4.1 Hydrologic Change

When assessing impacts in a stream the context of the hydrologic conditions are critical (Walters 2016). The conditions in the period of record (2006-2018, excluding 2007) encompass drought (one year according to the PDSI; see Figure 10) and extreme wet conditions, however, the last five years in the period of record have been wet. Given variable climatic conditions combined with the small number of years in the period of record, 12 years, the long-term hydrology may not be adequately represented by this data set.

Typically, to examine flow regime change a full suite of hydrologic measures are examined, such that each of the five major components of a flow regime are assessed, including magnitude, frequency, duration, timing, and rate of change (Poff et al. 1997). In Little Rock Creek, the assessment is focused on agricultural groundwater pumping. Groundwater pumping reduces streamflow by capturing groundwater that would have discharged to the stream (Bredehoeft et al. 1982). Generally, streamflow depletion caused by nearby pumping wells will quickly reach equilibrium and influence the stream within the pumping season while wells at a greater distance will likely not influence streamflow until after the pumping season (Kendy and Bredehoeft 2006). This pattern typifies Little Rock Creek, leading to greater concern for streamflow depletion during the pumping season and in particular during August. As such, the hydrologic focus was on August discharge and includes assessing magnitude, duration and frequency changes.

Groundwater pumping has led to a decrease in the magnitude and an increase in the frequency and duration of low flows in the Little Rock Creek. The low flows, defined as the discharge values less than or equal to the 75% exceedance (DePhilip and Moberg 2010), are critical to ecosystem health (Woodward et al. 2016). The biological component of streams have been found to be sensitive to changes in this part of the flow regime (Freeman and Marcinek 2006; Bradford and Heinonen 2008; Walters and Post 2011; Walters 2016). In Little Rock Creek, the 75% exceedance flow decreases from 3.4 cfs under reference conditions, to 2.2 cfs – a decline of 35 percent. The decline steepens as flows decrease: the difference between reference and baseline for the 80% exceedance flow was 39%, and drops to 45% for the 90% exceedance values (**Table 8**). The number of days at lower flows increases substantially under baseline conditions. Ten-year recurrence intervals for the 3-day and 7-day mean low flows for the reference condition are reduced to a 3-year recurrence interval for the baseline condition (**Table 8**). Congruent with the increase in frequency of the low flow event, the duration (i.e., number of days in August) of 80% and 90% exceedance flows for the reference condition increases by 67% and over 200% in the baseline condition, respectively (**Table 8**)**.**

## 4.2 Habitat Changes

Physical habitat for stream dwelling species is defined using depth, velocity, and substrate. As a result of the direct physical connection between discharge and hydraulic conditions in the stream, physical changes occur as flows are reduced, such as, reduced depth, velocity, wetted area, and the capacity to transport silt and fine sediment. Each of these physical properties decrease as discharge decreases. The change in depth and velocity throughout the stream is shown in **Figure 15**. Changes in depth and velocity are synonymous with changes in habitat for aquatic biota. We can see qualitatively in **Figure 15** that the depth and velocity is reduced, and the conditions become less diverse. At the lowest modeled flows (down to 1.2 cfs), conditions for only two habitat guilds (shallow pool, medium pool) were present with an area greater than 10 $ft^2$, while at the highest modeled flows (to 56.7 cfs), six habitat guilds were present in quantity greater than 300 $ft^2$ (**Figure 18**). The physical conditions set the quantity and quality of habitat for fish and macroinvertebrate species and the diversity of depth, velocity and substrate defines the species diversity (Schlosser 1982). There is also a clear nonlinear relationship between wetted perimeter or area and discharge (Gippel and Stewardson 1998); the Little Rock Creek relationship is displayed in **Figure 16**. Some previous instream flow methodologies exploited the wetted area or perimeter discharge relationship and used the break point on the curve to define a single protected flow (Jowett 1997; Tharme 2003). A key objective of setting a low flow limit using the wetted perimeter methodology is to protect the more productive riffle habitat (Gippel and Stewardson 1998). As depth and velocity are reduced, the stream has less shear stress and decreased power, resulting in the accumulation of fine sediment (Buendia et al. 2014). Such increases in fine sediment affect macroinvertebrates and fish (Waters 1995; Kemp et al. 2011), and when significant constitute an ecological impact.

Each species-life stage habitat discharge response curve (**Figure 17**) reveals how sensitive each is to changes in flow. The pattern in habitat loss across the six habitat guilds can be specific to stream and flow regime; however, fluvial specialists (i.e., species requiring flowing water) and their corresponding riffle habitat are particularly sensitive to flow change (Freeman and Marcinek 2006). Our analysis for Little Rock Creek indicates that the species that lost habitat the fastest at low flows (i.e., less than 10 cfs on **Figure 17**) generally included riffle and raceway species but also included the deep pool representative **(Table 13**). The observed relationships (i.e., curves) were typical of other reported stream habitat discharge relationships (Leonard and Orth 1988; Jowett et al. 2005; Krstolic et al. 2006; Bovee et al. 2007; Jowett and Davey 2007). Leonard and Orth (1988) identified four general categories of habitat discharge response curves with Type 1 and 2 both having a mid-discharge peak but rising with discharge at the lower end at a different rate. In Type 3 and 4, the habitat decreases at different rates with a peak at the lowest flows. The 11 habitat discharge relationships developed for Little Rock Creek span the range of categories described in Leonard and Orth (1988). The fact that we see each of these patterns in the discharge habitat relationship (**Figure 17**) suggests that we modeled species representative of a large range of habitat conditions. Assessing the habitat discharge response for a wide range of conditions is critical when the objective is to assess stream ecosystems and not focus on single species management (Gore et al. 2001).

Formally, the use of multiple species representing a wide range of habitats is referred to as the habitat guild approach (Leonard and Orth 1988). Use of the habitat guild approach has implications beyond ensuring that wide ranges of fish species are modeled. Depth, velocity, and substrate are three environmental variables that define the space that many stream organisms, such as fish (Aadland 1993), macroinvertebrates (Gore et al. 2001) including mussels (Parasiewicz et al. 2012), and macrophytes (Riis and Biggs 2001) can occupy (sensu Hutchinson 1957). The list is not complete in defining living space for aquatic organisms (Jackson et al. 2001), but accurately describes the hydraulic variables that both define the space and are directly influenced by the flow regime.

Converting the individual species-life stage habitat discharge relationships into a time series and summarizing into the three habitat metrics allows us to define when and whether a management threshold was exceeded. For example, in comparing the reference and baseline scenarios using habitat Metric 2, habitat area decreases between 2% (creek chub young-of-year) and 34% (Johnny darter adults). These results can be expected after reviewing the habitat discharge response curves at the low discharge end (**Figure 17**). Eight of the eleven species-life stages exceeded the 20% threshold. The only species-life stages not significantly impacted were the two shallow pool and one of the two slow riffle representatives. Metric 3 address low habitat frequency and duration, with all habitat guilds and representative species-life stages experiencing significant harm as measured by a decrease of 20% or more in the magnitude of the 3 day low habitat (**Table 14**). The frequency of the 10-year low habitat event also increased to generally a 3-year event for each habitat guild (**Table 14**).

The MN DNR (2016) Thresholds Report established that a 20% change in hydrology represents the threshold for ecological harm. It also states, that where data are available, we will use it to refine the threshold. Ecological harm occurs when there is a 20% change in the ecosystem. Biology integrates all agents of system change. The analysis in this report more directly connects hydrology (flows) to ecology, using biology (habitat). Two of the three habitat metrics identified significant ecological harm. Metric 1 identified only minor changes to the habitat conditions during normal conditions; normal habitat conditions occur during normal flow conditions for Little Rock Creek. The small change in Metric 1 was anticipated because streamflow depletion as a result of agricultural groundwater pumping most affects the low flow portion of the hydrograph (Kendy and Bredehoeft 2006), as was observed in Little Rock Creek. Metrics 2 and 3 measure the change in habitat during low habitat conditions, which coincide with low flow periods in Little Rock Creek. The reduction of habitat in the majority of the species-life stages of Metric 2 and 3 are greater than the 20% loss threshold for significant ecological harm, and this threshold is exceeded in multiple habitat guilds (**Table 12, 13,** and **14**). The ecological implications beyond fish habitat are discussed in the following section.

Low flows and reduction of low flows are significant to fish (Webb et al. 2013), and restrictions of habitat during the low flow period has been identified as a bottleneck or a limiting factor for fish survival and abundance (Bovee et al. 1994; Lamouroux et al. 1999; Freeman et al. 2001; Hakala and Hartman 2004; Jowett et al. 2005; Milhous and Bartholow 2006; Armstrong et al. 2011; Lamouroux and Olivier 2015).

## 4.3 Implications to Ecology

Pumping groundwater decreases river flows, which can in turn impact ecosystems, especially at ecologically sensitive times, such as during the summer low flows (Gleeson and Richter 2018). A significant decrease in virtually all habitat types under baseline conditions was documented on Little Rock Creek. The loss of habitat quantity and diversity at low flows and its detrimental effects to stream ecology is well documented in the literature. In other stream ecosystem analyses, hydrologic metrics, particularly low flow measures, have been directly related to biological measures of health (Carlisle et al. 2010), richness and diversity (Freeman and Marcinek 2006; Yang et al. 2008; McCargo and Peterson 2010; Armstrong et al. 2011), growth and survival (Hakala and Hartman 2004; Harvey et al. 2006; Avery-Gromm et al. 2014; Richard et al. 2015), and abundance (Zorn et al. 2012). Each of the above studies found a significant relationship to low flow measures. Carlisle et al. (2010), in a geographically widespread analysis, found that the primary predictor of a fish index of biological integrity was low flow.

Studies of fish communities in streams with diminished low flows reveal shifts on the species assemblage, with an increase in species preferring low velocities and fine sediments. Armstrong et al. (2011) observed that as the August median discharge was depleted, the abundance and richness of fluvial species decreased; this included brook trout and blacknose dace. Blacknose dace abundance decreased 40% with a 20% decrease in the August median discharge. In addition to changes in abundance, richness decreased by one species with August median depletion changing from zero to 14% (Armstrong et al. 2011). Freeman and Marcinek (2006) document a loss of one fluvial specialist fish species with water withdrawals increasing from the equivalent of 0 to 80% of the 7-day Q10 and an additional three species with increase up to 13 times the 7-day Q10. In addition to abundance and richness impacts, fish growth and mortality are related to mean summer flow, with greater growth and lower mortality in years with high mean summer discharge (Hakala and Hartman 2004; Harvey et al. 2006; Avery-Gromm et al. 2014; Richard et al. 2015).

The magnitude of the impact of flow alteration is context dependent, subject to attributes of the low flow event, the habitat, and the species present and assessed (Walters 2016). Attributes of low flow events will lead to varying degrees of impact, including the magnitude of the reduction and the frequency and duration of the low flows; more severe conditions lead to more severe impacts. Physical habitats are differentially affected by decreasing water levels. Riffle habitat tends to be areas of high macroinvertebrate production (Allan and Castillo 2007) and important ecological processes such as nitrogen uptake (Risse-Buhl et al. 2020) and is often the first to be depleted, resulting in a shift from fluvial specialist species to generalist species (Bradford and Heinonen 2008; Rolls et al. 2012). In addition to preferred habitat, other attributes of fish, such as size, physiological tolerance and reproductive traits make some species more or less susceptible to low flow changes (Walters 2016). Each of these context-defining attributes point to the importance of assessing a range of biological and physical components.

Generally, the three (3) habitat guild metrics provide the primary evidence for ecological harm at the current level of water use. The metrics use individual species-life stages that encompass the ecological components in a stream. Within each metric, if more than one habitat guild experiences ecological harm, it suggests larger scale degradation. Metric 1, which assesses 'normal' flow conditions, did not indicate loss of habitat for any of the modeled species. However, the results of Metric 2 and Metric 3 did indicate impacts that are important for the Little Rock Creek ecosystem. First, low flows are known to strongly influence fish and macroinvertebrates, including individual measures such as growth, mortality (Hakala and Hartman 2004; Harvey 2006; Armstrong et al. 2011; Avery-Gomm et al. 2014; Richard et al. 2015; Walters 2016) and abundance or biomass (Haxton and Findlay 2008; Armstrong et al. 2011; Walters and Post 2011; Lamouroux and Olivier 2015) and at the community level, such as, measures of fish and invertebrate IBI and species richness (Freeman and Marcinek 2006; Konrad et al. 2008; Carlisle et al. 2010; McCargo and Peterson 2010; Armstrong et al. 2011). Second, the impact of reduced flow is wide spread in the ecosystem reaching more than individual species (Rolls et al. 2012; Woodward et al. 2016). Drought triggers loss of food web links (i.e., loss of species) and a reduction in consumer and predator biomass (Ledger et al. 2013). The smaller food webs have been directly linked to restriction of habitat during low flows (McHugh et al. 2015; Woodward et al. 2016).

Corroborating hydrologic, water quality, and hydraulic evidence quantified at Little Rock Creek also points to ecological harm from current levels of groundwater use. August median base flow was reduced by more than 20% in four (2006, 2008, 2012, and 2013) of the 12 years evaluated; only one of those years was classified as drought (2006). Most measures of low flow have been reduced by more than 20%. Low flows are also known to effect growth, survival, and abundance of individual species, and reduce measures of aquatic health such as fish and invertebrate IBI. Generally, the more severe the magnitude decrease, the greater the impact. Compounding the low streamflow reductions, low discharge values that did not exist under reference conditions and now (under baseline conditions) occur with a frequency of greater than 20%. A decrease in low flows in Little Rock Creek (see the changes in low flow measures in **Table 8**) was implicated in a reduction in water quality (Benton SWCD 2009; MPCA 2015). Little Rock Creek is listed as impaired for aquatic life and drinking water with the pollutant or stressor listed as a lack of a coldwater assemblage and low dissolved oxygen for aquatic life and nitrates as the pollutant impacting drinking water (MPCA 2015). To meet the standards for the Little Rock Creek impairments, the MPCA (2015) implementation plan scenarios emphasizes the need to increase groundwater flow. In addition to the hydrological and biological measures, the in-stream hydraulics also point to ecological harm. While many ecological measurements lack obvious thresholds (Acreman et al. 2014), the wetted area versus discharge relationship displays a clear threshold (**Figure 16**). The lower section of the curve drops rapidly, indicating the stream losing area at a higher rate at low flows as compared to high flows. The baseline low flows are on the extreme low end of the curve, creating higher risk of desiccation of riffles and the organisms living there.

Current stream ecology literature expands on the importance and relationships between biology and low flow conditions. Studies have found both fish and macroinvertebrate growth, survival, abundance, and community structure are all negatively influenced by low flow conditions (Dewson et al. 2007; Walters and Post 2010; Piniewski et al. 2016; Woodward et al. 2016). The literature suggest that certain groups of species will experience the greatest harm, these included species and their life stages that depend on fast flowing habitat and stream margins (Bradford and Heinonen 2008; Rolls et al. 2012). Our findings fit this pattern.

## 4.4 Alternative Analysis

Current August streamflow depletion as a percentage of the ABF resulting from groundwater pumping ranged from less than zero to greater than 25% (**Table 9**). Groundwater depletions over the period of record average 12%. In addition to the reference and baseline condition comparison, eight other streamflow depletion scenarios were examined to identify the relationship between depletion and degree of impact, and guide management recommendations. Each depletion scenario was based on removing a percentage of the August median base flow (ABF) also referred to as the management index flow. The ABF was determined to be 5.5 cfs for the reference condition. The eight incremental percentages removed included 5, 10, 15, 20, 25, 30, 40, and 50% of the ABF. For example, the streamflow depletion was determined to be 0.55 cfs (i.e., 5.5*0.10) for the 10% scenario. Subsequently, the 0.55 cfs was removed from each daily streamflow for the reference condition. This is clearly a simplified version of how groundwater pumping is actually used in an agricultural setting and how streamflow depletion occurs, but is necessary for practical analysis. Groundwater pumping and streamflow depletion is a response to ongoing short-term climatic conditions and varies on a daily basis. These simplified depletion scenarios help set management guide posts that can prevent ecological harm and identify the maximum daily amount that can be depleted before crossing the 20% ecological harm threshold.

Changes for each species-life stage (by habitat guild) are provided for Metric 1 and Metric 2 in **Figure 25**. The baseline streamflow depletions are similar in habitat impacts to the removal of 25% of the ABF, despite an average streamflow depletion compared to the ABF of 12%. Limiting streamflow depletions to 10% of the ABF avoids significant ecological harm to all species modeled. Removing 15% of the ABF results in significant harm (21% loss in habitat) to one habitat guild species life stage (representing slow riffle organisms). Removing 20% of the ABF results in significant harm for five species-life stages spanning three habitat guilds.

# 5. Conclusions and Recommendations

Streamflow has been described as the 'master variable,' driving the response of many critical physical and chemical characteristics of rivers, including water temperature and dissolved oxygen, geomorphology, and riverine habitat diversity (Poff et al. 1997, Annear et al. 2004). Groundwater and surface water are closely connected, constituting one resource (Winter et al. 1998; Alley at al. 1999); changes in one impact the other. Headwater streams like Little Rock Creek perform ecological functions (i.e., biological, geochemical, and physical processes that occur within an ecosystem) that are critical for sustaining fish, fisheries, and ecosystem services throughout their drainage basins (Colvin et al. 2019). This IFIM study on Little Rock Creek integrates the results of the **Groundwater Flow and Groundwater / Stream Interaction in the Little Rock Creek Area** (MN DNR 2020a), and the **Assessment of Instream Temperatures in Little Rock Creek near Sartell Wildlife Management Area** (MN DNR 2020b) to investigate ecological impacts. Using the IFIM, we assessed the impact of the estimated hydrologic change caused by groundwater pumping on the habitat and ecology of Little Rock Creek. We generated a series of alternative scenarios of depletion level using incremental percentages of the August median base flow (ABF; e.g., 5% ABF, 10% ABF, 15% ABF . . . 50% ABF). Analyses of alternative scenarios were used to assess the impacts of groundwater depletion relative to the threshold of ecological harm (>20% change in habitat) and identify viable management levels.

Our key findings are:

1. Low flows are reduced by currently authorized groundwater pumping.
    a. The reduction in flows below 75% exceedance is significant and increases as flows decrease.
        i. The percent change between reference (without groundwater pumping) and baseline (with groundwater pumping) conditions was greater than 35% in low flow magnitude, frequency, duration, and recurrence.
2. The impact of this reduction in Little Rock Creek corresponds to a significant loss of fish habitat.
    a. Five of six habitat guilds lose significant (>20%) habitat as flows decrease under the current groundwater pumping levels. Only shallow pool species are relatively unaffected by the decrease in low flows.
    b. Habitat decreases greater than 20% occur as flows decrease below the 75% exceedance flow.
3. Based on our habitat modeling and review of scientific literature, the current magnitude of habitat loss during August across the majority of habitat types (5/6), equates to ecological harm.
    a. Habitat loss and pollution are the primary causes of extirpation of aquatic biota.
4. Reductions in streamflow depletion are needed to avoid ecological harm.
    a. Alternative scenario analyses shows that impacts to all fish habitat types (i.e., ecology) are below the 20% threshold if groundwater depletions are at or below 10% of the ABF (August median base flow).
    b. Impacts exceed the 20% threshold for one species, representing one habitat type (i.e., slow riffles; 21% habitat loss), if diversions are at or below 15% of the ABF.
    c. As more flow is diverted, equivalent to 20% ABF, habitat loss for five of the modeled species, representing four habitat types, exceeds 20%.

## 5.2 Sustainable Diversion Limit

A Sustainable Diversion Limit is established as a means of keeping water flows above the threshold of ecological harm, defined previously in the 'Report to the Minnesota State Legislature: Definitions and Thresholds for Negative Impacts to Surface Waters' (MN DNR 2016) as 20% of the August Median Base flow. The SDL is expressed in relation to water appropriations as a maximum amount of water that can be removed directly or indirectly from a surface water body in a defined geographic area on an annual basis without causing a negative impact to the surface water body (MN DNR 2016). To avoid ecological harm, the SDL is assessed on a monthly basis (i.e., for August) and established with consideration for the type and seasonality of use, including the pumping rate and volume.

Results of the fish habitat analysis reveal that current pumping levels (baseline conditions) during lower flows decrease fish habitat by more than 20% for five of the six habitat types. This study also shows that impacts to all fish habitat types (i.e., ecology) are below the 20% threshold if diversions are at or below 10% of the ABF (August median base flow). If diversions are at or below 15% of the ABF, habitat loss greater than 20% occurs for one species (21% loss), representing one (i.e., slow riffle) habitat type. As more flow is diverted, equivalent to 20% ABF, habitat loss for five of the modeled species, representing four habitat types, exceeds 20%. Given the results for the Little Rock Creek study, the literature related to this science, and working towards water management sustainability, we recommend a SDL not to exceed 15% of the August median base flow. A sustainable diversion limit of 10% of the August median base flow would avoid all known risk of ecosystem harm due to streamflow depletion.

## 5.3 Additional Management Considerations

Potential climate change impacts underscore the importance of adopting an SDL which keeps water depletion at 15% or less of the August Median base flow. In Minnesota, there is moderately high confidence drought will increase in severity, coverage, and duration, beyond 2025 (source: Kenny Blumenfeld, climatologist; March 30, 2017, DNR Science Chat: 'Keys to understanding Minnesota's changing climate'). Our recommended percentage of flow approach for water management can be effective in preserving water to maintain stream ecology, but may be risky if the allocation plan overestimates the amount of water available, or where inter-annual rainfall patterns or climate change reduce the amount in the system (Arthington 2012). The analysis in this report relied on information collected on Little Rock Creek during the 12-year period of record. Overall, the measured hydrology on Little Rock Creek can be currently characterized as wetter than average. It is unknown if the recent period of record will represent future conditions. Under wetter conditions, there will be less need for groundwater pumping, and therefore less surface water depletion and related impacts to the stream ecosystem. For example, if subsequent years are warmer than the period of record used in this study or there is less precipitation, an increased demand for groundwater irrigation is likely, and the impacts associated with depletion of streamflow and documented here become more likely. Given the low flow analysis (Metrics 2 and 3) completed by this study, the subsequent recommendation for a SDL of 10% - 15% ABF should account for such conditions. However, the 15% ABF is essentially approaching the threshold for ecological harm. Therefore, we also recommend continuing to monitor streamflow in Little Rock Creek as a way of ensuring that the findings here remain applicable.

Little Rock Creek, as a small headwater stream, cannot be viewed in isolation of its drainage or outside the context of water management in Minnesota as a whole. In the larger view, aquatic systems form both a mosaic and a continuum of habitats (Arthington et al. 2016). Isolated and fragmented habitats typically present the most challenging environments for small, specialized freshwater fishes (Arthington et al. 2016).

As a small headwater stream, the need for proposed management changes for Little Rock Creek may be underestimated. However, headwater streams provide numerous services that are essential to ecosystems (Peterson et al. 2001; Meyer et al. 2003), including sustaining aquifers and supplying clean water for more than one-third of the U.S. population (USEPA 2009). Overall, they are an extensive part of the landscape. Headwaters, if defined as first order streams, comprise 48% of total stream length in the United States, and 73% of total stream length is first and second order streams (Leopold et al. 1964). In Minnesota, the composition is similar: 51% of the total stream length are first order streams, and 73% are first and second order combined.

At regional scales, headwaters are critical for sustaining aquatic biodiversity (Meyer et al. 2007; Clarke et al. 2008) and for providing vital spawning and rearing habitat for migratory fishes, including commercially fished species (Quinn 2005; Schindler et al. 2010; McClenachan et al. 2015). Headwaters provide dispersal corridors and habitat for fishes and other aquatic and semi-aquatic organisms (e.g., invertebrates, amphibians, and birds), including many endemic and rare species (Steward et al. 2012; Jaeger et al. 2014; Sullivan et al. 2015).

## Literature Cited

Aadland, L. P. 1993. Stream habitat types: their fish assemblages and relationship to flow. North American Journal of Fisheries Management 13 (4): 790-805.

Aadland, L. P., C. M. Cook, M. T. Negus, H. G. Drewes, and C. S. Anderson. 1991. Microhabitat preferences of selected stream fishes and a community-oriented approach to instream flow assessments. Minnesota Department of Natural Resources, Section of Fisheries, Investigation report 406, St. Paul.

Aadland, L., and A. Kuitunen. 2006. Habitat suitability criteria for stream fishes and mussels of Minnesota. Minnesota Department of Natural Resources, Special Publication 162, St. Paul.

Aadland, L., C. Waltner, M.T. Negus, H.G. Drewes, and C.S. Anderson. 1989. Microhabitat criteria for selected stream fishes and methodological considerations for instream flow studies in Minnesota. Minnesota Department of Natural Resources, Section of Fisheries Technical Report 1987-1989: report to the Legislative Commission on Minnesota Resources, St. Paul.

Acreman, M.C., and A.J. D. Ferguson. 2010. Environmental flows and the European Water Framework Directive. Freshwater Biology 55 (1): 32-48. doi: 10.1111/j.1365-2427.2009.02181.x.

Acreman, M., J. Hannaford, O. Mountford, P. Wood, N. Holmes, I. Cowx, R. Noble, C. Extence, J. Aldrick, J. King, A. Black, and D. Crookall. 2008. Developing Environmental Standards for Abstractions From UK Rivers to Implement the EU Water Framework Directive. Hydrological Sciences Journal 53:1105-1120.

Allan, J.D. and Castillo, M.M., 2007. *Stream ecology: structure and function of running waters*. Springer Science & Business Media.

Alley, W.M., T. E. Reilly, and O. L. Franke. 1999. Sustainability of ground-water resources. U.S. Geological Survey circular: 1186. United States Geological Survey. Denver, Colorado, USA.

Annear, T., I. Chisholm, H. Beecher, A. Locke, P. Aarrestad, C. Coomer, C. Estes, J. Hunt, T. Jacobsen, G. Jöbsis, J. Kauffman, J. Marshall, K. Mayes, G. Smith, R. Wentworth, and C. Stalnaker. 2004. Instream Flow for Riverine Resource Stewardship, revised edition. Instream Flow Council, Cheyenne, Wyoming, USA.

Armstrong, J. D., and K.H. Nislow. 2012. Modelling approaches for relating effects of change in river flow to populations of Atlantic Salmon and brown trout. Fisheries Management and Ecology, 19, 527–536.

Armstrong, D.S., T.A. Richards, and S.B. Levin. 2011, Factors influencing riverine fish assemblages in Massachusetts: U.S. Geological Survey Scientific Investigations Report 2011–5193, 59 p. (Also available at http://pubs.usgs.gov/sir/2011/5193).

Arthington, A.H. 2012. Environmental flows: saving rivers in the Third Millennium. University of California Press. Berkeley, California, USA. 406pp.

Arthington, A. H., N. K. Dulvy, W. Gladstone, and I. J. Winfield. 2016. Fish conservation in freshwater and marine realms: status, threats and management. Aquatic Conservation: Marine and Freshwater Ecosystems 26:838–857.

Asquith, W.H. 2020. Lmomco---L-moments, censored L-moments, trimmed L-moments, L-comoments, and many distributions. R package version 2.3.6.

Avery-Gromm, S., J.S. Rosenfeld, J.S. Richardson, and M. Pearson. 2014. Hydrological drought and the role of refugia in an endangered riffle-dwelling fish, Nooksack dace (Rhinichthys cataractae ssp.). Canadian Journal of Fisheries and Aquatic Sciences 71:1625-1634.

Bain, M.B., J. T. Finn, and H. E. Booke. 1985. A quantitative method for sampling riverine microhabitats by electrofishing. North American Journal of Fish Management 5:489-493.

Benton Soil and Water Conservation District (SWCD). 2009. Little Rock Creek Stressor Identification Report. http://www.pca.state.mn.us/sites/default/files/wq-iw8-09m.pdf.

Bond, N. R., P.S. Lake, and A.H. Arthington. 2008. The impacts of drought on freshwater ecosystems: an Australian perspective. Hydrobiologia 600, 3–16. doi:10.1007/S10750-008-9326-Z.

Bovee, K.D. 1986. Development and evaluation of habitat suitability criteria for use in the Instream Flow Incremental Methodology. Instream Flow Information Paper No. 21. U.S. Fish and Wildlife Service, Biological Report 86 (7), Washington, D.C., USA.

Bovee, K.D., 1996. Perspectives on two-Dimensional River Habitat Models: The PHABSIM Experience. Proceedings of the Second International Symposium on Habitat Hydraulics, INRS-Eau, Quebec, Canada, pp. B149-B162.

Bovee, K.D., T.J. Newcomb, and T.G. Coon. 1994. Relations between habitat variability and population dynamics of bass in the Huron River, Michigan. National Biological Survey Biological Report 21. 63 pp.

Bovee, K.D., B.L. Lamb, J.M. Bartholow, C.B. Stalnaker, J. Taylor, and J. Hendricksen. 1998. Stream habitat analysis using the instream flow incremental methodology. U.S. Geological Survey. Biological Resources Division Information and Technology Report USGS/BRD-1998-0004. 131pp.

Bovee, K.D., Waddle, T.J., Bartholow, J. and Burris, L., 2007. A decision support framework for water management in the upper Delaware River.

Bradford, M.J., and J.S. Heinonen. 2008. Low flows, instream flow needs and fish ecology. Canadian Water Resources Journal 32: 165-180.

Bredehoeft, J. D., S. S. Papadopulos, and H. H. Cooper Jr. 1982. Groundwater: The water-budget myth, in Studies in Geophysics: Scientific Basis of Water-Resource Management, pp. 51 – 57, Natl. Acad. Press, Washington, D. C.

Buendia, C., C.N. Gibbins, D. Vericat, and R.J. Batalla. 2014. Effects of flow and fine sediment dynamics on the turnover of stream invertebrate assemblages. Ecohydrology, 7:1105-1123.

Bunn, S. E., and A. H. Arthington. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environmental Management 30, 492–507. doi:10.1007/S00267-002-2737-0.

Caissie, J., D. Caissie, and N. El-Jabi. 2015. Hydrologically based environmental flow methods applied to rivers in the Maritime provinces (Canada). River Research and Applications, 31, 651–662.

California Department of Fish and Wildlife. 2016. Instream Flow Evaluation: Upstream Spring-Run Chinook Salmon Passage in Butte Creek, California. Stream Evaluation Report 16-1, February 2016. Sacramento, California.

Calow, P., and G. E. Petts. 1994. The rivers handbook: hydrological and ecological principles, Volume 2. Blackwell Scientific Publications, Boston, Massachusetts, USA.

Carlisle, D.M., J. Falcone, D.M. Wolock, M.R. Meador, and R.H. Norris. 2010. Predicting the natural flow regime: models for assessing hydrological alteration in streams. River Research and Applications 26: 118-136. doi: 10.1002/rra.1247.

Clarke, K.D., T.C. Pratt, R.G. Randall, D.A. Scruton, and K.E. Smokorowski. 2008. Validation of the flow management pathway: effects of flow alteration on fish habitat and fishes downstream from a hydropower dam. Canadian Technical Report for Fisheries and Aquatic Sciences 2784: vi+111 p.

Colvin, S.A.R., M.P. Sullivan, P.D. Shirey, R.W. Colvin, K.O. Winemiller, R.M. Hughes, K.D. Fausch, D.M. Infante, J.D. Olden, K.R. Bestgen, R.J. Danehy, and L Eby. 2019. Headwater streams and wetlands are critical for sustaining fish, fisheries, and ecosystem services. Fisheries 44(2):73-91.

Cowan, W. R., Rankin, D. E., and Gard, M. 2017. Evaluation of Central Valley Spring-Run Chinook Salmon Passage Through Lower Butte Creek Using Hydraulic Modelling Techniques. River Research and Applications 33: 328– 340. doi: 10.1002/rra.3098.

Cox, M., G. Arnold, and S. Villamayor Tomas. 2010. A Review of Design Principles for Community-based Natural Resource Management. Ecology and Society 15(4): 8. [online] URL: http://www.ecologyandsociety.org/vol15/iss4/art38/.

Crowder, D.W., and P. Diplas. 2000. Using two-dimensional hydrodynamic models at scales of ecological importance. Journal of Hydrology 230:172-191.

Curran, C.A. and Olsen, T.D., 2009. Estimating Low-Flow Frequency Statistics and Hydrologic Analysis of Selected Streamflow-Gaging Stations, Nooksack River Basin, Northwestern Washington and Canada. US Geological Survey.

DePhilip, M., and T. Moberg. 2010. Ecosystem flow recommendations for the Susquehanna River basin. The Nature Conservancy, Harrisburg, PA. Available at: http://www.nature.org/media/pa/tnc-final-susquehanna-river-ecosystem-flows-study-report.pdf.

Dewson, Z.S., A.B. James, and R.G. Death, R.G., 2007. A review of the consequences of decreased flow for instream habitat and macroinvertebrates. Journal of the North American Benthological Society, 26:401-415.

Doyle, M.W., E.H. Stanley, D.L. Strayer, R.B. Jacobson, and J.C. Schmidt. 2005. Effective discharge analysis of ecological processes in streams. Water Resources Research 41: 1-16. doi: 10.1029/2005WR004222.

Fausch, K. D., J. Lyons, J. R. Karr, and P. L. Angermeier. 1990. Fish communities as indicators of environmental degradation. *In* Biological indicators of stress in fish. American Fisheries Society Symposium 8:123-144. Bethesda, Maryland, USA.

Folke, C., S. Carpenter, B. Walker, M. Scheffer, T. Elmqvist, L. Gunderson, and C.S. Holling. 2004. Regime Shifts, Resilience, and Biodiversity in Ecosystem Management. Annual Review of Ecology, Evolution, and Systematics, Vol. 35 (2004), pp. 557-581.

Freeman, M.C., Z.H. Bowen, K.D. Bovee, and E.R. Irwin. 2001. Flow and habitat effects on juvenile fish abundance in natural and altered flow regimes. Ecological Applications 11 (1): 179-190.

Freeman, M.C., and P.A. Marcinek. 2006. Fish assemblage responses to water withdrawals and water supply reservoirs in piedmont streams. Environmental Management 38: 435-450.

Gard, M., and E. Ballard. 2003. Applications of new technologies to instream flow studies in large rivers. North American Journal of Fisheries Management 23:1114–1125.

Gaston, K.J., T.M. Blackburn, J.D. Greenwood, R.D. Gregory, R.M. Quinn, and J. H. Lawton. 2000. Abundance–occupancy relationships. Journal of Applied Ecology 37(1):39-59.

Ghanem, A., P. Steffler, F. Hicks, and C. Katopodis. 1996. Two-dimensional hydraulic simulation of physical habitat conditions in flowing streams. Regulated Rivers Research and Management 12:185–200.

Gippel, C.J., and M.J. Stewardson. 1998. Use of wetted perimeter in defining minimum environmental flows. Regulated Rivers: Research and Management 14: 53-67.

Gleeson, T. and B. Richter. 2018. How much groundwater can we pump and protect environmental flows through time? Presumptive standards for conjunctive management of aquifers and rivers. River Research and Applications 34:83-92.

Gleeson, T., W.M. Alley, D.M. Allen, M.A. Sophocleous, Y. Zhou, M. Taniguchi, and J. VanderSteen. 2012. Towards sustainable groundwater use: setting long-term goals, backcasting, and managing adaptively. Ground Water 50(1):19-26.

Gore, J.A., J.B. Layzer, and J. Mead. 2001. Macroinvertebrate instream flow studies after 20 years: a role in stream management and restoration. Regulated Rivers: Research and Management 17: 527-542.

Hakala, J.P., and K.J. Hartman. 2004. Drought effects on stream morphology and brook trout (Salvelinus fontinalis) populations in forested headwater streams. Hydrobiologia 515: 203-213.

Hamilton, D.A., and P.W. Seelbach. 2010. Determining environmental limits to streamflow depletion across Michigan, volume 42. Pages 534-537 in: A.S. Wall, editor. The book of the states, 2010 edition. The Council of State Governments, Lexington, Kentucky.

Hanski, I. 1982. Dynamics of regional distribution: the core and satellite species hypothesis. Oikos:210-221.

Hart, D.D., and C.M. Finelli. 1999. Physical-biological coupling in streams: the pervasive effects of flow on benthic organisms. Annual Review of Ecology and Systematics 30: 363-395. doi: 10.1146/annurev.ecolsys.30.1.363.

Harvey, B.C., R.J. Nakamoto, and J.L. White. 2006. Reduced streamflow lowers dry-season growth of rainbow trout in a small stream. Transactions of the American Fisheries Society 135 (4): 998-1005. Doi:10.1577/T05-233.1.

Haxton, T.J., and C.S. Findlay. 2008. Meta-analysis of the impacts of water management on aquatic communities. Canadian Journal of Fisheries and Aquatic Sciences 65 (3): 437-447.

Hutchinson, G. E. 1957. Concluding remarks. Cold Spring Harbour Symposia on Quantitative Biology 22:415-427.

Hynes, H.B.N. 1970. The ecology of running waters. Liverpool University Press. Liverpool, United Kingdom.

Ibanez, C., N. Caiola, P. Sharpe, and R. Trobajo. 2010. Ecological indicators to assess the health of river systems. *In* Jorgensen, S.E., F. Xu, and R. Costanza (editors). Handbook of Ecological Indicators for the Assessment of Ecosystem Health. CRC Press; Taylor and Francis Group. Boca Raton, Florida, USA.

Ibarra, A. A., M, Gevrey, Y.-S. Park, P. Lim, and S. Lek. 2003. Modelling the factors that influence fish guilds composition using a back-propagation network: Assessment of metrics for indices of biotic integrity. Ecological Modelling 160: 281-290.

Jackson, D.A., P.R. Peres-Neto, and J.D. Olden. 2001. What controls who is where in freshwater fish communities: the roles of biotic, abiotic and spatial factors? Canadian Journal of Fisheries and Aquatic Sciences 58: 157-170. doi:10.1139/cjfas-58-1-157.

Jaeger, K.L., J.D. Olden, and N.A. Pelland. 2014. Climate change poised to threaten hydrologic connectivity and endemic fishes in dryland streams. P. Nat. Acad. Sci. USA 111: 13894-13899.

Jennings S., F. Melin, J.L. Blanchard, R.M. Forster, N.K. Dulvy, and R.W. Wilson. 2008. Global-scale predictions of community and ecosystem properties from simple ecological theory. Proceedings of the Royal Society B: Biological Sciences 275:1375–1383.

Jowett, I.G. 1990. Factors related to the distribution and abundance of brown and rainbow trout in New Zealand clear-water rivers. New Zealand Journal of Marine and Freshwater Research 24: 429-440. Doi: 10.1080/00288330.1990.9516434.

Jowett, I.G. 1997. Environmental effects of extreme flows. Pages 103-116 in: M.P. Mosely and C.P. Pearson, editors. Floods and droughts: the New Zealand experience. New Zealand Hydrological Society, Wellington.

Jowett, I.G., and A.J.H. Davey. 2007. A comparison of composite habitat suitability indices and generalized additive models of invertebrate abundance and fish presence-habitat availability. Transactions of the American Fisheries Society 136 (2): 428-444.

Jowett, I.G., J. Richardson, and M.L. Bonnett. 2005. Relationship between flow regime and fish abundances in a gravel-bed river, New Zealand. Journal of Fish Biology 66: 1419-1436. Doi: 10.1111/j.1095-8649.2005.00693.x.

Karr, J.R., and E.W. Chu. 1999. Restoring life in running waters: better biological monitoring. Island Press. Washington D.C., USA.

Karr, J.R., and E.W. Chu. 2000. Sustaining living rivers. *In:* M. Jungwirth, S. Muhar, S. Schmutz (editors). Assessing the Ecological Integrity of Running Waters. Reprinted from Hydrobiologia 422/423: 1-14. Kluwer Academic Publishers. Norwell, Massachusetts, USA.

Kemp, P., D. Sear, A. Collins, P. Naden, and I. Jones. 2011. The impacts of fine sediment on riverine fish. Hydrological Processes. 25 (11): 1800-1821. doi: 10.1002/hyp.7940.

Kendy, E., and J.D. Bredehoeft. 2006. Transient effects of ground-water pumping and surface-water irrigation returns on streamflow. Water Resources Research 42 W08415, doi:10.1029/2005WR004792.

Kolasa, J., and D. Strayer. 1988. Patterns of the abundance of species: a comparison of two hierarchical models. Oikos 53:235-241.

Konikow, L.F., and S.A. Leake. 2014. Depletion and capture: revisiting “the source of water derived from wells”. Groundwater 52:100-111.

Konrad, C.P., A.M.D. Brasher, and J.T. May. 2008. Assessing streamflow characteristics as limiting factors on benthic invertebrate assemblages in streams across the western United States. Freshwater Biology 53: 1983-1998. doi: 10.1111/j.1365-2427.2008.02024.x.

Krstolic, J.L., D.C. Hayes, and P.M. Ruhle. 2006. Physical habitat classification and instream flow modeling to determine habitat availability during low-flow periods, North Fork Shenandoah River, Virginia. U.S. Geological Survey Scientific Investigations Report 2006-5025, 63 pp.

Lamouroux, N., & Olivier, J. M. 2015. Testing predictions of changes in fish abundance and community structure after flow restoration in four reaches of a large river (French Rhône). Freshwater Biology, 60(6), 1118-1130.

Lamouroux, N., J.M. Olivier, H. Persat, M. Pouilly, Y. Souchon, and B. Statzner. 1999. Predicting community characteristics from habitat conditions: fluvial fish and hydraulics. Freshwater Biology 42: 275-299.

Ledger, M., L. Brown, F. Edwards, A.M. Milner, and G. Woodward. 2013. Drought alters the structure and functioning of complex food webs. Nature Climate Change 3: 223-227.

Leonard, P.M., and D.J. Orth. 1988. Use of habitat guilds to determine instream flow requirements. North American Journal of Fisheries Management 8 (4): 399-409.

Leopold, L. B., M.G. Wolman, and J.P. Miller. 1964. Fluvial processes in geomorphology. New York: Dover Publications.

Loreau, M., S. Naeem, P. Inchausti, J. Bengtsson, J. P. Grime, A. Hector, D. U. Hooper, M. A. Huston, D. Raffaelli, B. Schmid, D. Tilman, and D. A. Wardle. 2001. Biodiversity and ecosystem functioning: current knowledge and future challenges. Science 294:804-808.

Lytle, D.A., and N.L. Poff. 2004. Adaptation to natural flow regimes. Trends in Ecology and Evolution 19: 94-100. Doi: 10.1016/j.tree.2003.10.002. PMID:16701235.

McCargo, J.W., and J.T. Peterson. 2010. An evaluation of the influence of seasonal base flow and geomorphic stream characteristics on coastal plain stream fish assemblages. Transactions of the American Fisheries Society 139 (1): 29-48.

McClenachan, L., S. Lowell, and C. Keaveney. 2015. Social benefits of restoring historical ecosystems and fisheries: alewives in Maine. Ecology and Society 20:31.

McHugh, P.A., R.M. Thompson, H.S. Greig, H.J. Warburton, and A.R. McIntosh. 2015. Habitat size influences food web structure in drying streams. Ecography 38:700-712.

Meffe, G.K., and A.L. Sheldon. 1988. The influence of habitat structure of fish assemblage composition in southeastern blackwater streams. American Midland Naturalist 120: 225-240.

Meyer, J. L., L. A. Kaplan, J. D. Newbold, D. L. Strayer, C. J. Woltemade, J. B. Zedler, R. Beilfuss, Q. Carpenter, R. Semlitsch, M. C. Watzin, and P. H. Zedler. 2003. Where rivers are born: the scientific imperative for defending small streams and wetlands. Sierra Club and American Rivers, Washington, D.C., USA.

Meyer, J.L., D.L. Stayer, J.B. Wallace, S.L. Eggert, G.S. Helfman, and N.E. Leonard. 2007. The contribution of headwater streams to biodiversity in river networks. Journal of the American Water Resources Association 43 (1): 86-103.

Milhous, R.T., J.M. Bartholow. 2006. Two analytical approaches for quantifying physical habitat as a limit to aquatic ecosystems. International Journal of River Basin Management 4: 191-199.

Milhous, R. T., D. L. Wegner, and T. Waddle. 1981. User's guide to the physical habitat simulation system. Instream flow information paper no. 11. US Fish and Wildlife Service FWS/OBS-81/43. Ft. Collins, Colorado.

Miller, R. R., J. D. Williams, and J. E. Williams. 1989. Extinctions of North American fishes during the past century. Fisheries 14(6):22–38.

MN DNR. 2010. Little Rock Creek Basin Coldwater Fisheries IBI Assessment. Division of Fisheries and Wildlife, Minnesota Department of Natural Resources (MN DNR), Little Falls, MN, USA.

MN DNR. 2016. Report to the Minnesota State Legislature: Definitions and Thresholds for Negative Impacts to Surface Waters. Minnesota Department of Natural Resources (MN DNR), St. Paul, MN, USA.

MN DNR. 2020a. Groundwater Flow and Groundwater/Stream Interaction in the Little Rock Creek Area. Minnesota Department of Natural Resources (MN DNR) Report, St. Paul, MN, USA.

MN DNR. 2020b. Assessment of Instream Temperatures in Little Rock Creek near Sartell Wildlife Management Area. Minnesota Department of Natural Resources (MN DNR) Report, St. Paul, MN, USA.

Minnesota Pollution Control Agency (MPCA). 2015. Little Rock Creek watershed total maximum daily load report: dissolved oxygen, nitrate, temperature and fish bioassessment impairments. http://www.pca.state.mn.us/sites/default/files/wq-iw8-09e.pdf.

Nehring, R.B., and R.M. Anderson. 1993. Determination of population-limiting critical salmonid habitats in Colorado streams using the Physical Habitat Simulation system. Rivers 4(1): 1-19.

Noble, R., I. Cowx, D. Goffaux, and P. Kestemont. 2007. Assessing the health of European rivers using functional ecological guilds of fish communities: standardizing species classification and approaches to metric selection. Fisheries Management and Ecology 14:381-392.

Parasiewicz, P., E. Castelli, J.N. Rogers, and E. Plunkett. 2012. Multiplex modeling of physical habitat for endangered freshwater mussels. Ecological Modeling 228:66-75.

Peterson, B. J., W. M. Wollheim, P. J. Mulholland, J. R. Webster, J. L. Meyer, J. L. Tank, E. Martí, W. B. Bowden, H. M. Valett, A. E. Hershey, W. H. McDowell, W. K. Dodds, S. K. Hamilton, S. Gregory, and D. D. Morrall. 2001. Control of nitrogen export from watersheds by headwater streams. Science 292:86–90.

Piniewski, M., C. Prudhomme, M.C. Acreman, L. Tylec, P. Oglęcki, and T. Okruszko. Responses of fish and invertebrates to floods and droughts in Europe. Ecohydrology 10:e1793.

Poff, N. L., J.D. Allan, M.B. Bain, J.R. Karr, K.L. Prestegaard, B.D. Richter, R.E. Sparks, and J.C. Stromberg. 1997. The natural flow regime: A paradigm for river conservation and restoration. Bioscience 47: 769–784. doi:10.2307/1313099

Poff, N.L., and J.V. Ward. 1989. Implications of streamflow variability and predictability for lotic community structure: a regional analysis of streamflow patterns. Canadian Journal of Fisheries and Aquatic Sciences 46 (10): 1805 1818. Doi: 10.1139/f89-228.

Postel, S.L., and B.D. Richter. 2003. Rivers for life: managing water for people and nature. Island Press. Washington, D.C., USA.

Power, M.E., A. Sun, G. Parker, W.E. Dietrich, and J.T. Wootton. 1995. Hydraulic food-chain models: an approach to the study of food-web dynamics in large rivers. BioScience 45: 159-167. doi: 10.2307/1312555.

Power, G., R. Brown, and J. Imhof. 1999. Groundwater and fish—insights from northern North America. Hydrological Processes 13(3):401-422.

Puckridge, J.T., F. Sheldon, K.F. Walker, and A.J. Boulton. 1998. Flow variability and the ecology of large rivers. Australian Journal of Marine and Freshwater Research 49: 55-72. doi: 10.1071/MF94161.

Quinn, T.P. 2005. The behavior and ecology of Pacific salmon and trout. American Fisheries Society and the University of Washington Press. Seattle, Washington, USA.

Rapport, D., R. Costanza, and A. McMichael. 1998. Assessing ecosystem health. Trends in Ecology and Evolution 13(10):397-402.

Resh, V.H., A.V. Brown, A.P. Covich, M.E. Gurtz, H.W. Li, G.W. Minshall, S.R. Reice, A.L. Sheldon, J.B. Wallace and R.C. Wissmar. 1988. The Role of Disturbance in Stream Ecology. Journal of the North American Benthological Society 7(4):433-455. https://www.jstor.org/stable/1467300 JSTOR.

Reynoldson, T.B., R.C. Bailey, K.E. Day, and R.H. Norris. 1995. Biological guidelines for freshwater sediment based on **Be**nthic **A**ssessment of **S**edimen**t** (the BEAST) using a multivariate approach for predicting biological state. Australian Journal of Ecology 20:198-219.

Richard, A., F. Cattanéo, and J.F. Rubin. 2015. Biotic and abiotic regulation of a low-density stream-dwelling brown trout (Salmo trutta L.) population: effects on juvenile survival and growth. Ecology of Freshwater Fish 24:1-14.

Richter, B.D. 2010. Re-thinking environmental flows: from allocations and reserves to sustainability boundaries. River Research and Applications 26: 1052-1063.

Richter, B.D., J.V. Baumgartner, J. Powell, and D.P. Braun. 1996. A method for assessing hydrologic alteration within a river network. Conservation Biology 10: 1163-1174. Doi: 10.1046/j.1523-1739.1996.10041163.x.

Richter, B.D., J.V. Baumgartner, D.P. Braun, and J. Powell. 1998. A spatial assessment of hydrologic alteration within a river network. Regulated Rivers 14: 329-340.

Richter, B. D., A. T. Warner, J.I. Meyer, and K. Lutz. 2006. A collaborative and adaptive process for developing environmental flow recommendations. River Research and Applications 22: 297–318. doi:10.1002/RRA.892.

Richter, B.D., M.M. Davis, C. Apse, and C. Konrad. 2012. A presumptive standard for environmental flow protection. River Research and Applications 28:1312–1321.

Riis, T. and B.J. Biggs. 2003. Hydrologic and hydraulic control of macrophyte establishment and performance in streams. Limnology and oceanography 48:1488-1497.

Risse-Buhl, U., C. Anlanger, C. Noss, A. Lorke, D. von Schiller, and M. Weitere. 2020. Hydromorphologic sorting of in-stream nitrogen uptake across spatial scales. *Ecosystems (Dec) pp. 1-9.* https://doi.org/10.1007/s10021-020-00576-7.

Roberts, T. H., and O'Neil, L. J. 1985. "Species Selection for Habitat Assessments," Miscellaneous Paper EL-85-8, US Army Engineer Waterways Experiment Station, Vicksburg, Mississippi, USA.

Rolls, R.J., and A.H. Arthington. 2014. How do low magnitudes of hydrologic alteration impact riverine fish populations and assemblage characteristics? Ecological Indicators 39: 179–188.

Rolls, R. J., A.J. Boulton, I. Growns, S.E. Maxwell, D.S. Ryder, and D.P. Westhorpe. 2012. Effects of an experimental environmental flow release on the diet of fish in a regulated coastal river. Hydrobiologia 686, 195–212. doi:10.1007/S10750-012-1012-5.

Root, R.B. 1967. The niche exploitation pattern of the blue-gray gnatcatcher. Ecological Monographs 37: 317-350.

Rosenfeld, J.S., K. Campbell, E.S. Leung, J. Bernhardt, and J. Post. 2011. Habitat effects on depth and velocity frequency distributions: Implications for modeling hydraulic variation and fish habitat suitability in streams. Geomorphology 130:127-135.

Schiemer, F. 1998. Fish as indicators for the assessment of the ecological integrity of large rivers. In: M. Jungwirth, S. Muhar, and S. Schmutz (editors). Assessing the Ecological Integrity of Running Waters. Reprinted from Hydrobiologia 422/423: 271-289. Kluwer Academic Publishers. Norwell, Massachusetts, USA.

Schindler, D. E., R. Hilborn, B. Chasco, C. P. Boatright, T. P. Quinn, L. A. Rogers, and M. S. Webster. 2010. Population diversity and the portfolio effect in an exploited species. Nature 465:609–612.

Schlosser, I. J. 1982a. Fish community structure and function along two habitat gradients in a headwater stream. Ecological Monographs 52:395-414.

Shiklomanov, I.A. 1993. World fresh water resources. Pages 13-24 in P.H. Gleick, editor. Water in crisis: a guide to the world's fresh water resources. Oxford University Press, New York.

Simberloff, D. and T. Dayan.1991. The guild concept and the structure of ecological communities. Annual Review of Ecology and Systematics 22:115-143. http://www.jstor.com/stable/2097257.

Smakhtin, V.U. 2001. Low flow hydrology: a review. Journal of Hydrology 240:147-186.

Souchon, Y., C. Sabaton, R. Deibel, D. Reiser, J. Kershner, M. Gard, C. Katopodis, P. Leonard, N. L. Poff, J. Miller, and B. Lee. 2008. Detecting biological responses to flow management: missed opportunities; future directions. River Research and Applications 24:506–518.

Southwood, T. R. E. 1977. Habitat, the templet for ecological strategies? The Journal of Animal Ecology 46:337-365.

Stalnaker, C.B., B.L. Lamb, J. Henriksen, K. Bovee, and J. Bartholow. 1995. The instream flow incremental methodology: a primer for IFIM. National Biological Service, Biological Report 29. 45 pp.

Stalnaker, C.B., K.D. Bovee, T.J. Waddle. 1996. Importance of the temporal aspects of habitat hydraulics to fish population studies. Regulated Rivers: Research & Management, 12:145-153.

Stanford, J.A., J.V. Ward, W.J. Liss, C.A. Frissell, R.N. Williams, J.A. Lichatowich, and C.C. Coutant. 1996. A general protocol for restoration of regulated rivers. Regulated Rivers: Research and Management 12: 391-414.

Steffler, P. 2002. River2D_Bed. Bed Topography File Editor. User's manual. University of Alberta, Edmonton, Alberta. 32 pp. http://www.River2D.ualberta.ca/download.htm.

Steffler, P. and J. Blackburn. 2002. River2D, Two-dimensional Depth Averaged Model of River Hydrodynamics and Fish Habitat, Introduction to Depth Averaged Modeling and User's Manual. University of Alberta, Edmonton, Alberta. 120 pp. http://www.River2D.ualberta.ca/download.htm.

Steward, A.L., D. von Schiller, K. Tockner, J.C. Marshall, and S.E. Bunn. 2012. When the river runs dry: human and ecological values of dry riverbeds. Frontiers in Ecology and the Environment 10 (4): 202-209. doi: 10.1890/110136.

Sullivan, S. M. P., K. Hossler, and C. M. Cianfrani. 2015. Ecosystem structure emerges as a strong determinant of food-chain length in linked stream-riparian ecosystems. Ecosystems 18:1356–1372. Taylor, J.M., T.S. Seilheimer, and W.I. Fisher. 2014. Downstream fish assemblage response to river impoundment varies with degree of hydrologic alteration. Hydrobiologia 728: 23-39.

Tharme, R. E. 2003. A global perspective on environmental flow assessment: emerging trends in the development and application of environmental flow methodologies for rivers. River Research and Applications 19:397–441.

Tokeshi, M. 1993. Species abundance patterns and community structure. Advances in Ecological Research 24:111-186.

Turnipseed, D. P. and V. B. Sauer. 2010. Discharge measurements at gaging stations. Techniques and Methods (No. 3-A8). US Geological Survey. Reston, VA. http://pubs.usgs.gov/tm/tm3-a8/.

USEPA (U.S. Environmental Protection Agency). 2009. Section 404 of the Clean Water Act. Geographic information systems analysis of the surface drinking water provided by intermittent, ephemeral, and headwater streams in the U.S. U.S. Environmental Protection Agency. Washington, D.C., USA.

VanDerWal, J., L.P. Shoo, C.N. Johnson, and S.E. Williams. 2009. Abundance and the environment environmental niche: environmental suitability estimated from niche models predicts the upper limit of local abundance. The American Naturalist 174 (2): 282-291.

Venier, L., and L. Fahrig. 1996. Habitat availability causes the species abundance-distribution relationship. Oikos 76:564-570.

Waddle, T. and P. Steffler. 2002. R2D_Mesh - Mesh Generation Program for River2D Two Dimensional Depth Averaged Finite Element. Introduction to Mesh Generation and User's manual. U.S. Geological Survey, Fort Collins, CO. 32 pp. http://www.River2D.ualberta.ca/download.htm.

Walker, K.F., F. Sheldon, and J.T. Puckridge. 1995. An ecological perspective on dryland rivers. Regulated Rivers: Research and Management 11: 85-104.

Walters, A.W. 2016. The importance of context dependence for understanding the effects of low-flow events on fish. Freshwater Science 35 (1): 216-228.

Walters, A.W., and D.M. Post. 2011. How long can you go? Impacts of a low-flow disturbance on aquatic insect communities. Ecological Applications 21: 163-174.

Ward, J. V., and J. A. Stanford. 1979. Ecological factors controlling stream zoobenthos with emphasis on thermal modification of regulated streams. Pages 35-55 *In* J. V. Ward, and J. A. Stanford (editors). The Ecology of Regulated Streams. Plenum Press, New York, New York, USA.

Waters, T.F. 1995. Sediment in streams: sources, biological effects, and control. American Fisheries Society Monograph 7. American Fisheries Society. Bethesda, Maryland, USA.

Webb, J.A., K.A. Miller, E.L. King, S.C. deLittle, M.J. Stewardson, J.K.H. Zimmerman, and N.L. Poff. 2013. Squeezing the most out of existing literature: a systematic re-analysis of published evidence on ecological responses to altered flow. Freshwater Biology. 58(12): 2439-2451.

Welcomme, R.L., K.O. Winemiller, and I.J. Cowx. 2006. Fish environmental guilds as a tool for assessment of ecological condition of rivers. River Research and Applications 22: 377-396.

Wiens, J.A., 1977. On Competition and Variable Environments: Populations may experience" ecological crunches" in variable climates, nullifying the assumptions of competition theory and limiting the usefulness of short-term studies of population patterns. American Scientist *65:*590-597.

Winemiller, K.O. 2005. Life history strategies, population regulation, and implications for fisheries management. Canadian Journal of Fisheries and Aquatic Sciences 52: 875.

Winter, T.C., J.W. Harvey, O.L. Franke, and W.M. Alley. 1998. Ground water and surface water -- A single resource: U.S. Geological Survey Circular 1139, 79 p.

Woodward, G., N. Bonada, L.E. Brown, R.G. Death, I. Durance, C. Gray, S. Hladyz, M.E. Ledger, A.M. Milner, S.J. Ormerod, and R.M. Thompson. 2016. The effects of climatic fluctuations and extreme events on running water ecosystems. Philosophical Transactions of the Royal Society B: Biological Sciences, 371:20150274.

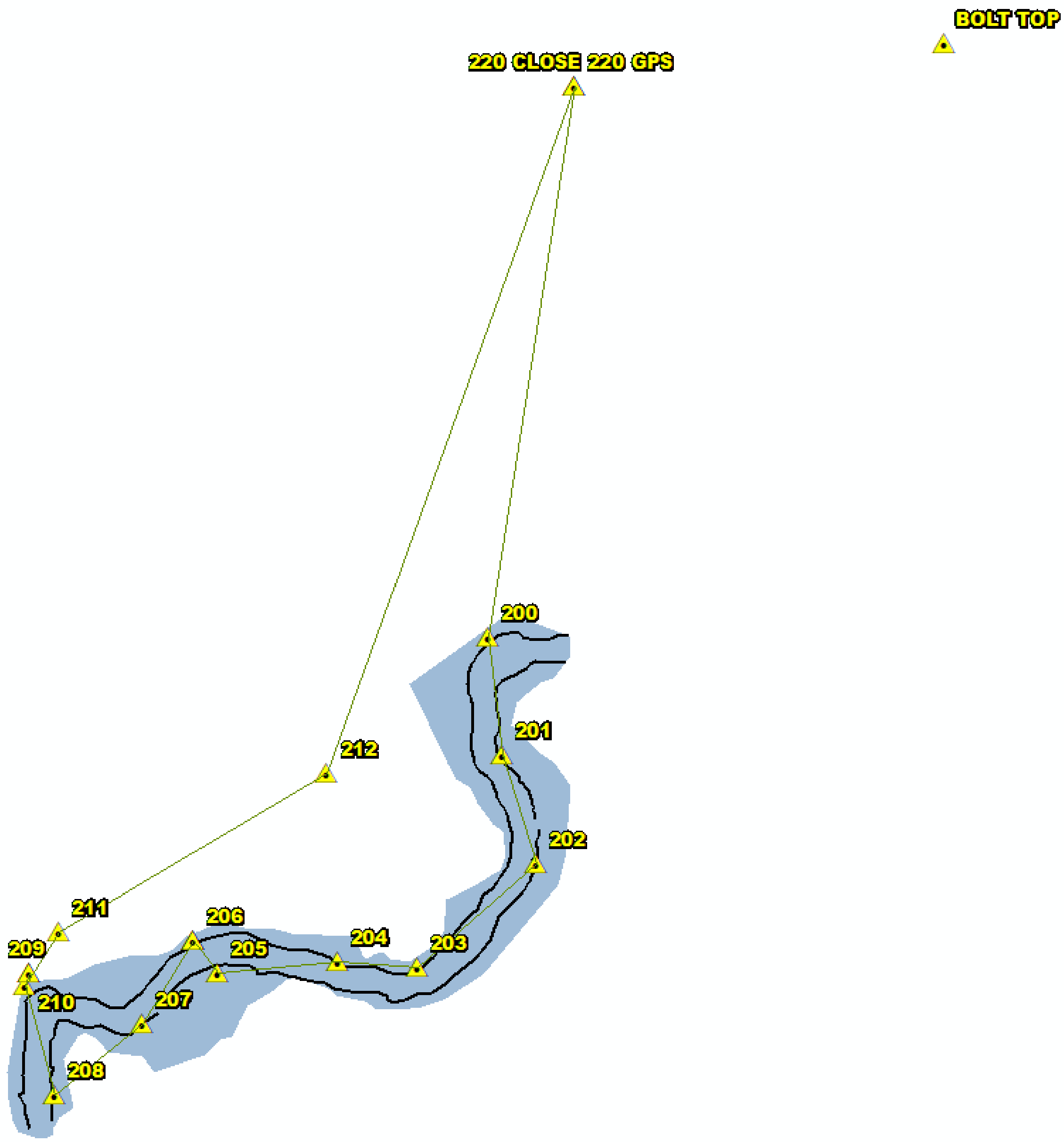
Xenopoulos, M.A., D.M. Lodge, J. Alcamo, M. Marker, K. Schulze, and D.P. van Vuuren. 2005. Scenarios of freshwater fish extinctions from climate change and water withdrawal. Global Change Biology 11 (10): 1557-1564.

Zorn, T.G., P.W. Seelbach, and E.S. Rutherford. 2012. A regional-scale habitat suitability model to assess the effects of flow reduction on fish assemblages in Michigan streams. Journal of the American Water Resources Association 48: 871-895.

## Appendices

### Appendix A. Study Site Survey Setup

Survey setup of the Little Rock Creek modeling site with control points and benchmarks used for closed loop traverse shown.

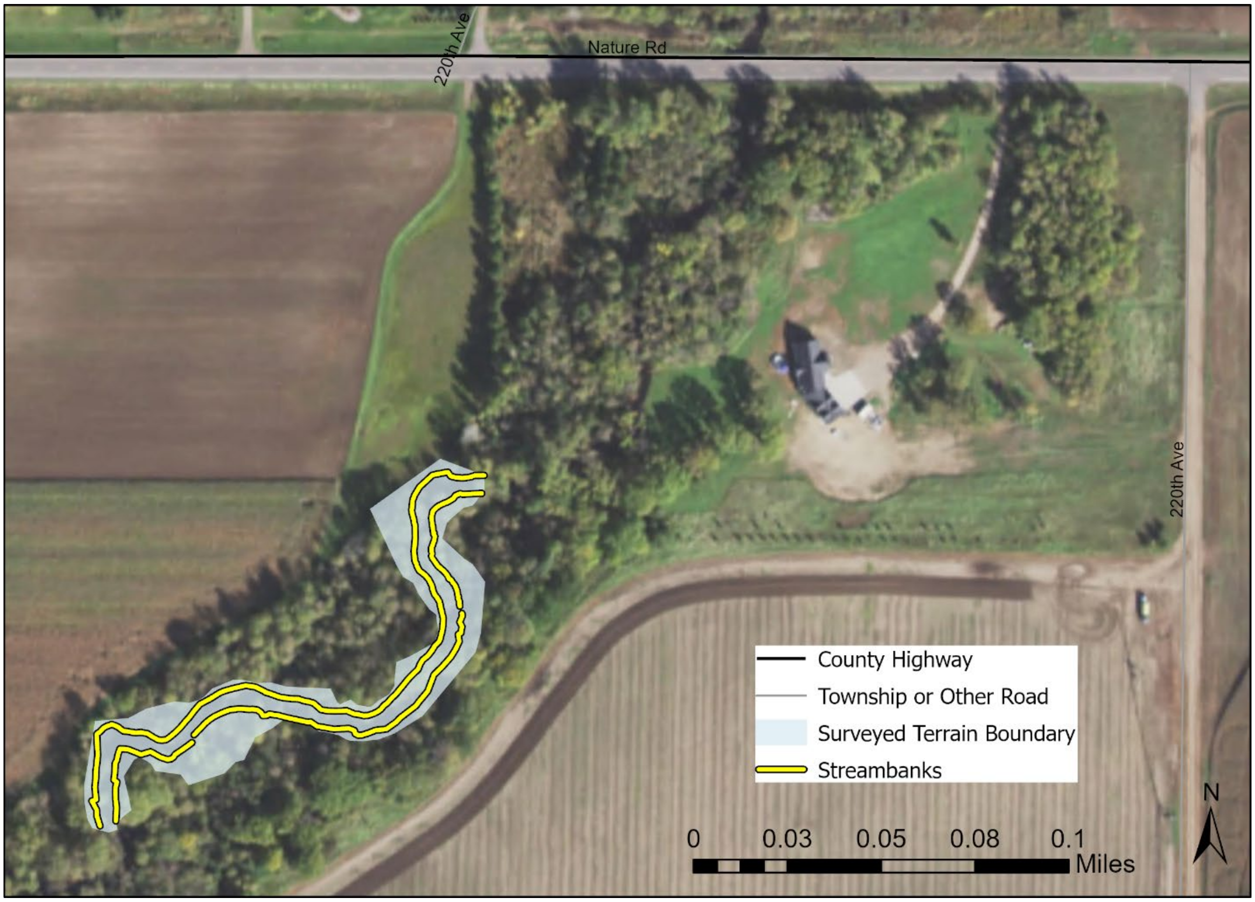



***Benchmark and control point setup used for terrain and bathymetry survey measurements on Little Rock Creek.***

## Appendix B. Terrain Model Surveying

Location of survey and modeling site, survey codes used to identify feature and substrate, surveyed points, and elevation map interpolation in R2D_Bed from surveyed data.

**Location and survey extent**

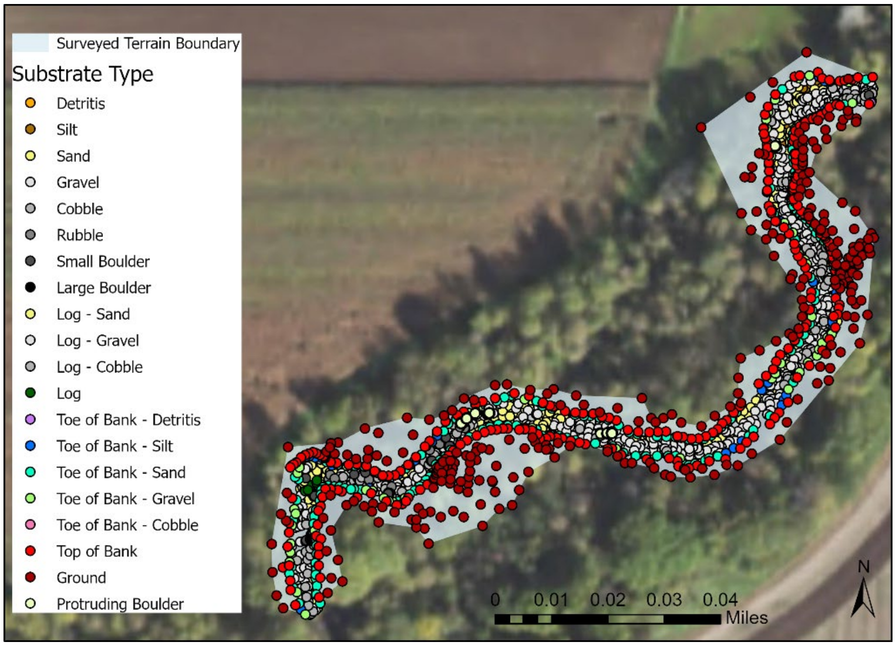


**Point Codes**

Codes used in field survey along with channel index values used in River2D habitat modeling

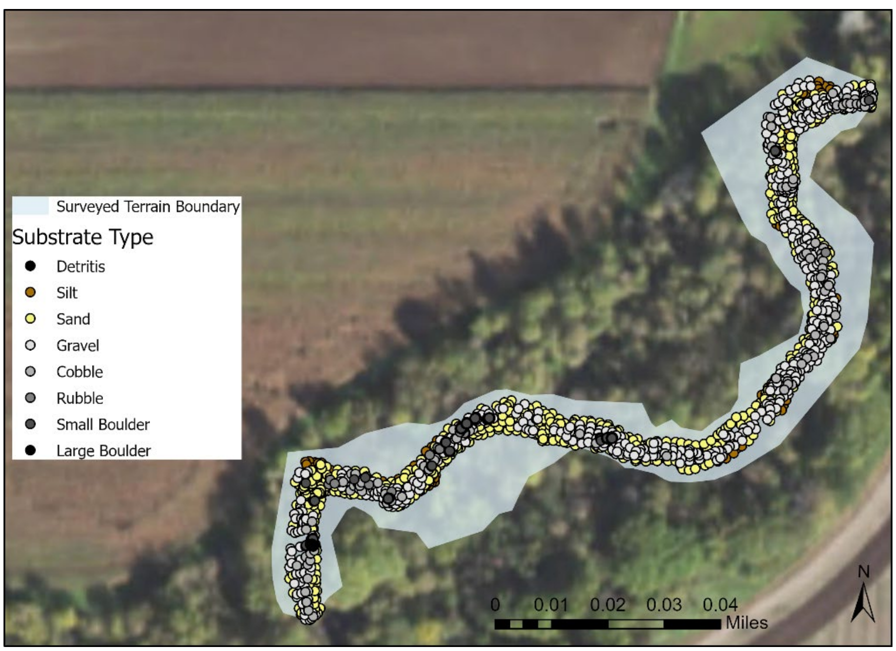
| Point Code | Description – Substrate Type | Channel Index Value – River2D |
|---|---|---|
| 1 | detritus | 1 |
| 2 | silt | 2 |
| 3 | sand | 3 |
| 4 | gravel | 4 |
| 5 | cobble | 5 |
| 6 | rubble | 6 |
| 7 | small boulder | 7 |
| 8 | large boulder | 8 |
| 9.x | log.(substrate suffix) | |
| 9.9 | bottom of log <10in off bottom | |
| 9.1 | bottom of the log > 10in off bottom | |
| 10.x | toe of bank (substrate suffix) | |
| 11 | top of bank | |
| 12 | ground shot | |
| 13 | bedrock | |
| 14 | water surface | |
| 15 | protruding boulder | |
| 103 | horizontal control point | |
| 104 | found monument | |
| 105 | bench mark | |
| 106 | check azimuth | |

**Terrain – all points**




**Terrain – reclassified streambed points**

Points regrouped by substrate type to match channel index values used in habitat modeling. Ground and top of bank points removed for better visualization of substrate in stream channel.



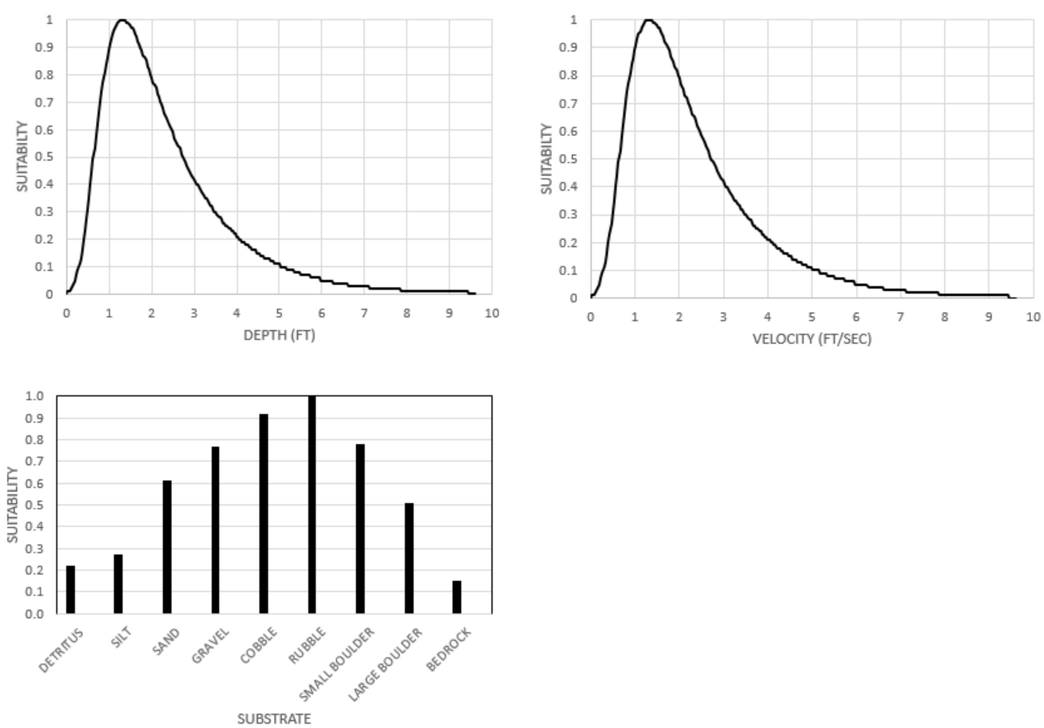

Little Rock Creek -- Bed Elevation (meters)
Bed Elevation
328.95
328.66
328.37
328.08
327.79
327.50
327.20
326.91
326.62
326.33
326.04
Distance
10.0 m

## Appendix C. Fish Habitat Preference Curves used in River2D model

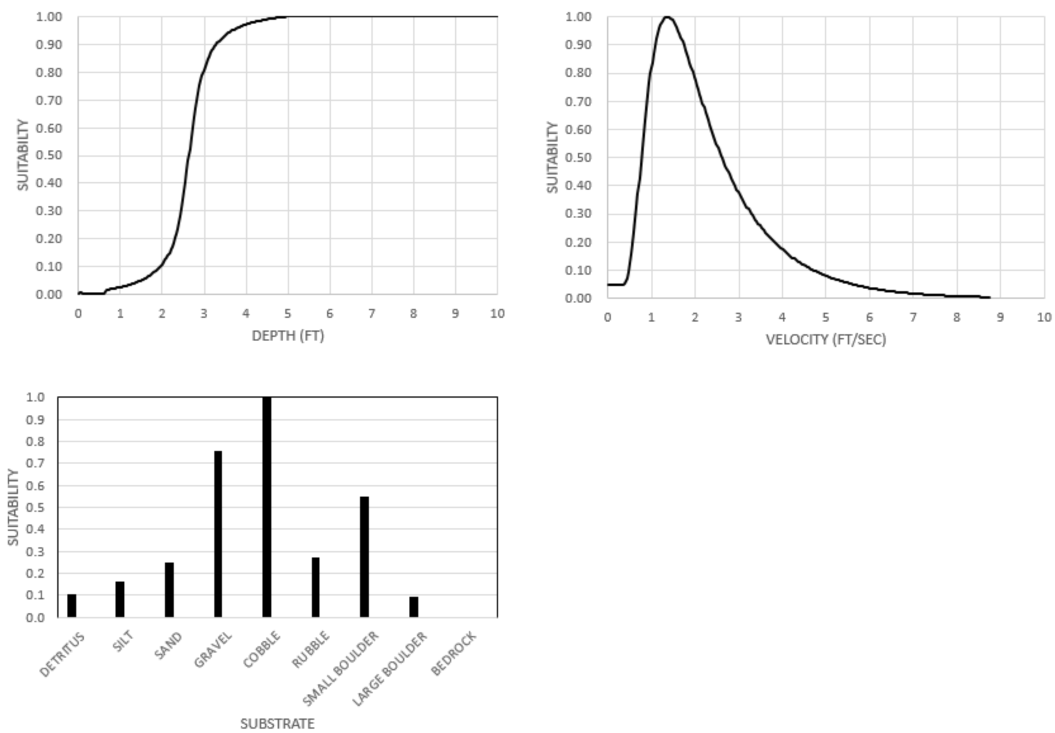
Habitat preference curves and preference histograms for each species-life stage used in the modeling.

- Blacknose dace, adult
- Brown trout, adult
- Brown trout, juvenile
- Creek chub, young-of year
- Common shiner, adult
- Common shiner, juvenile
- Johnny darter, adult
- Johnny darter, young-of year
- Longnose dace, adult
- White sucker, juvenile
- White sucker, young-of year

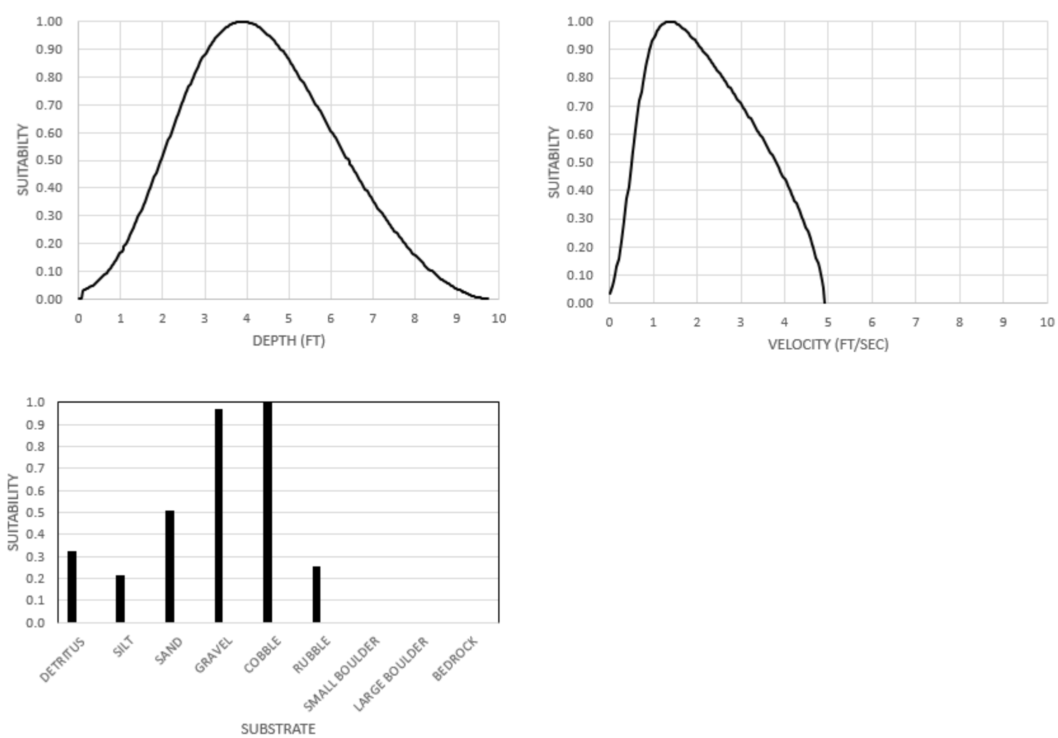
BLACKNOSE DACE ADULT



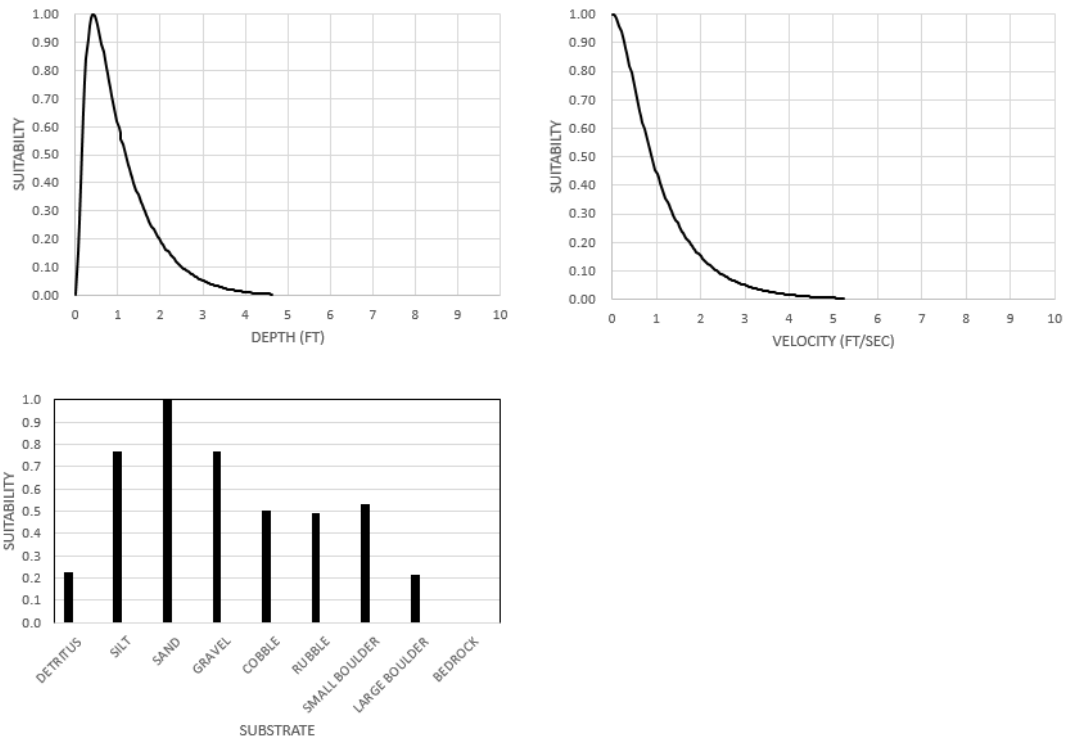
BROWN TROUT ADULT



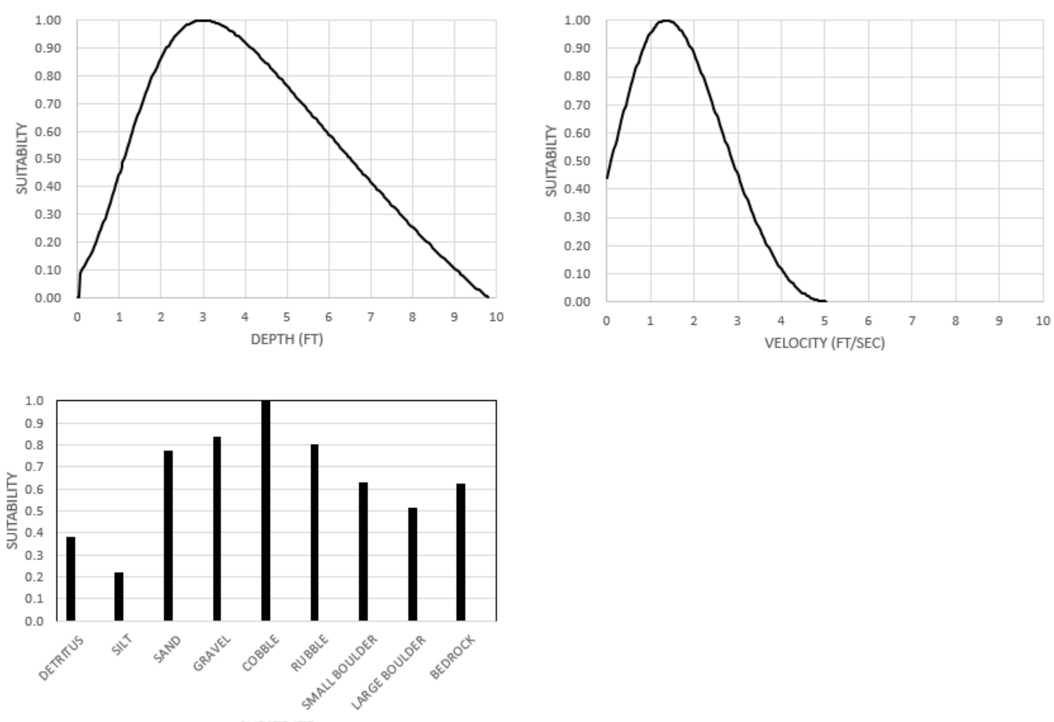

BROWN TROUT JUVENILE



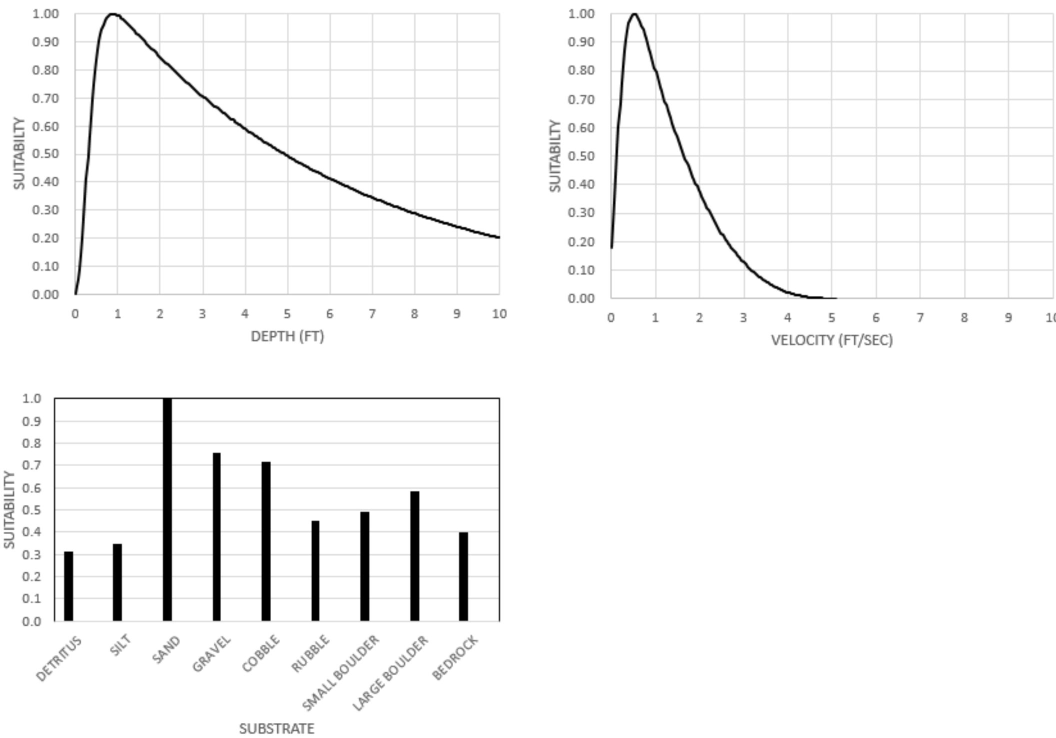
CREEK CHUB YOUNG-OF-YEAR



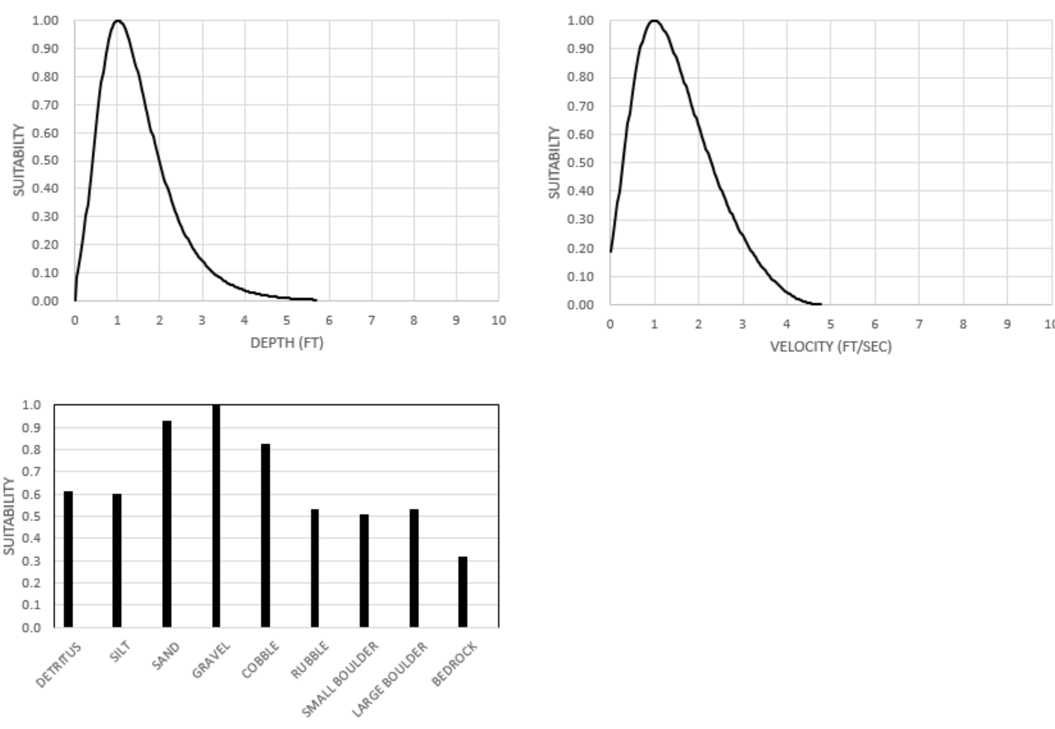

COMMON SHINER ADULT



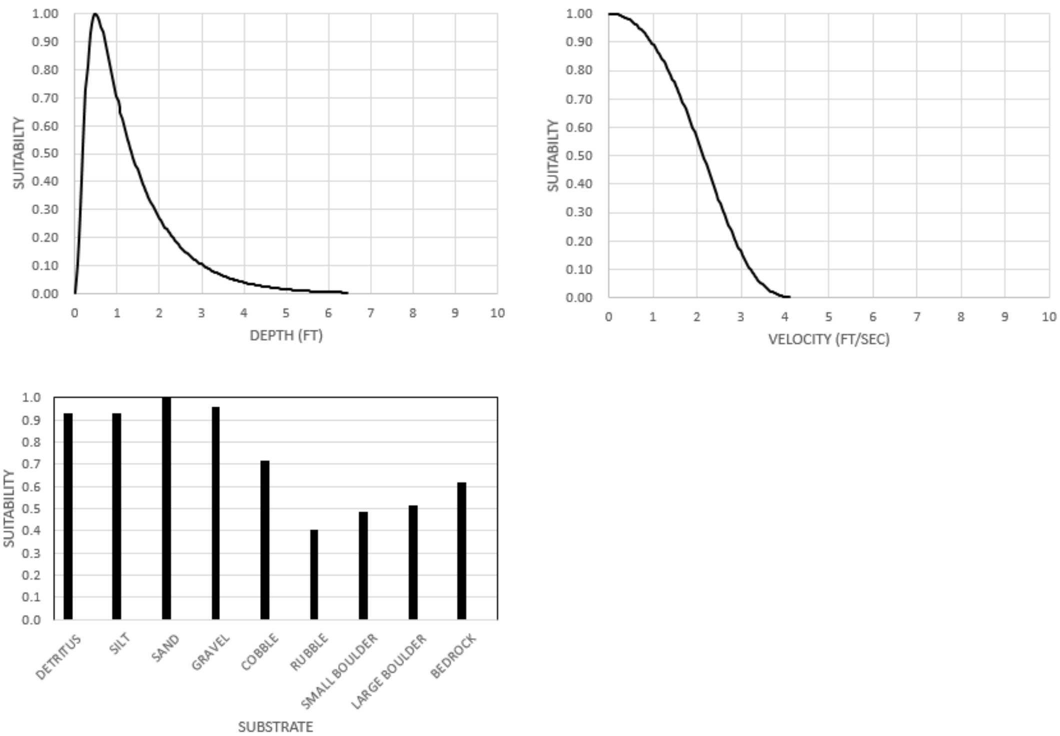
## COMMON SHINER JUVENILE



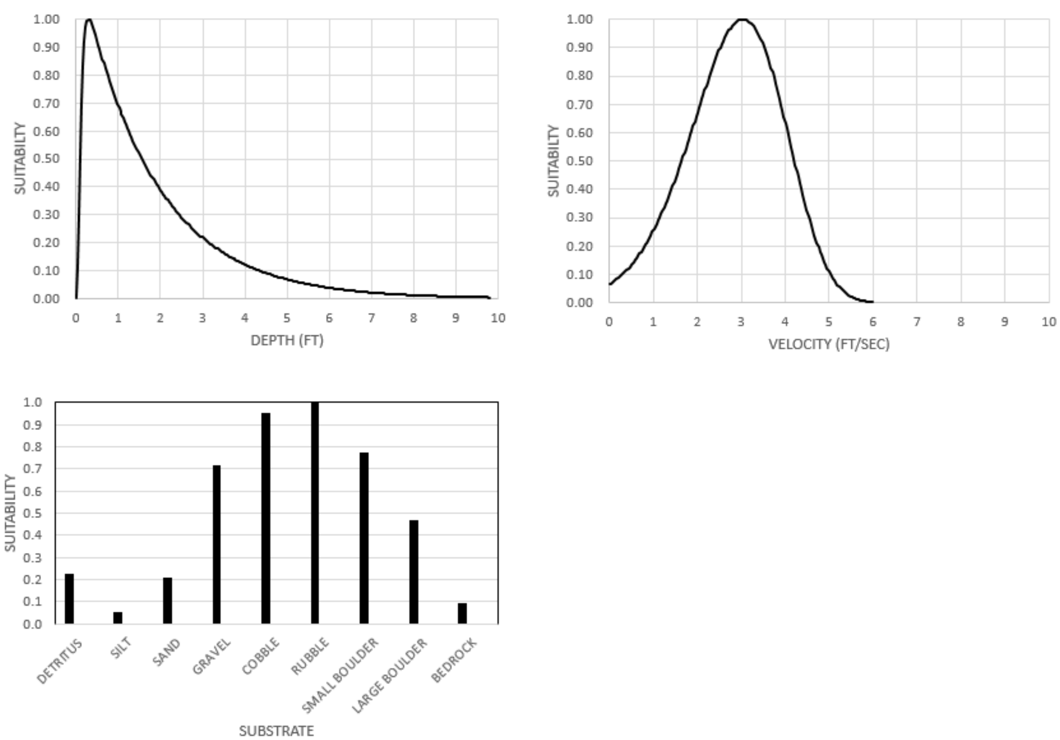

## JOHNNY DARTER ADULT



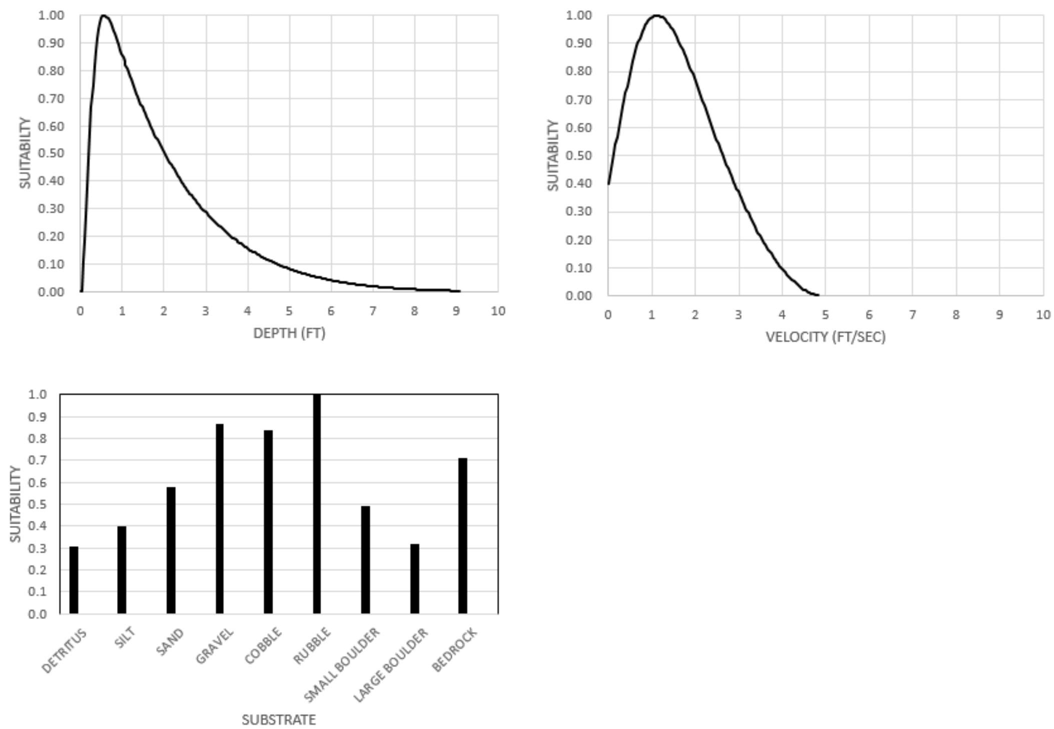
## JOHNNY DARTER YOUNG-OF-YEAR



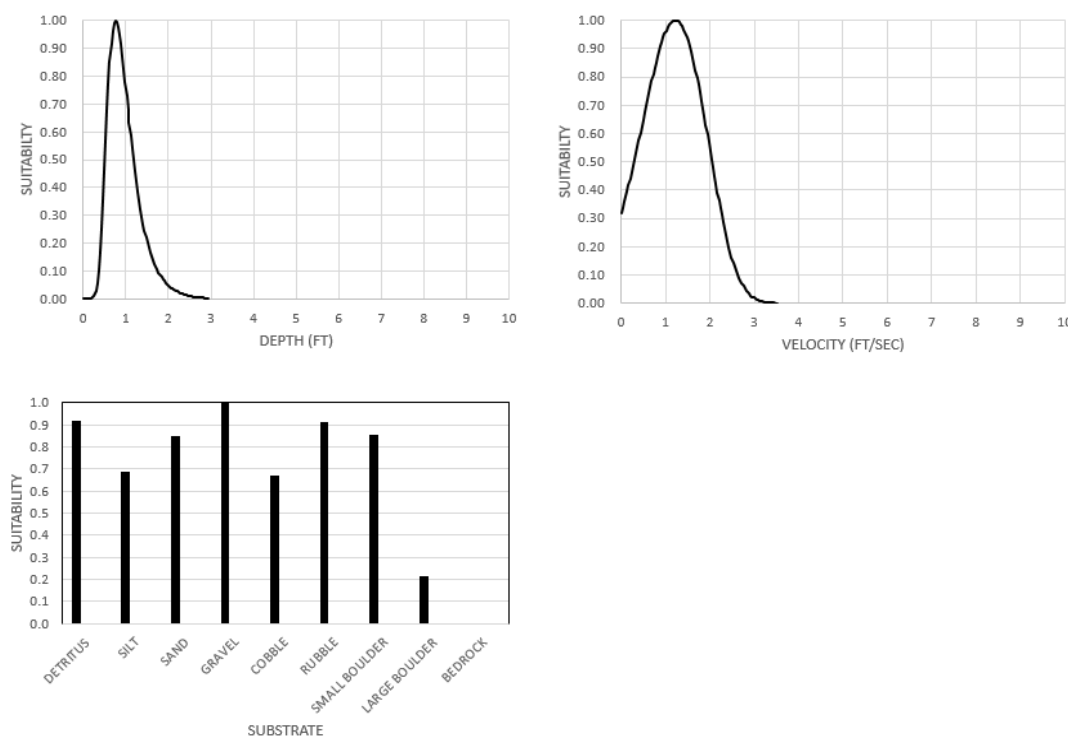

## LONGNOSE DACE ADULT



## WHITE SUCKER JUVENILE



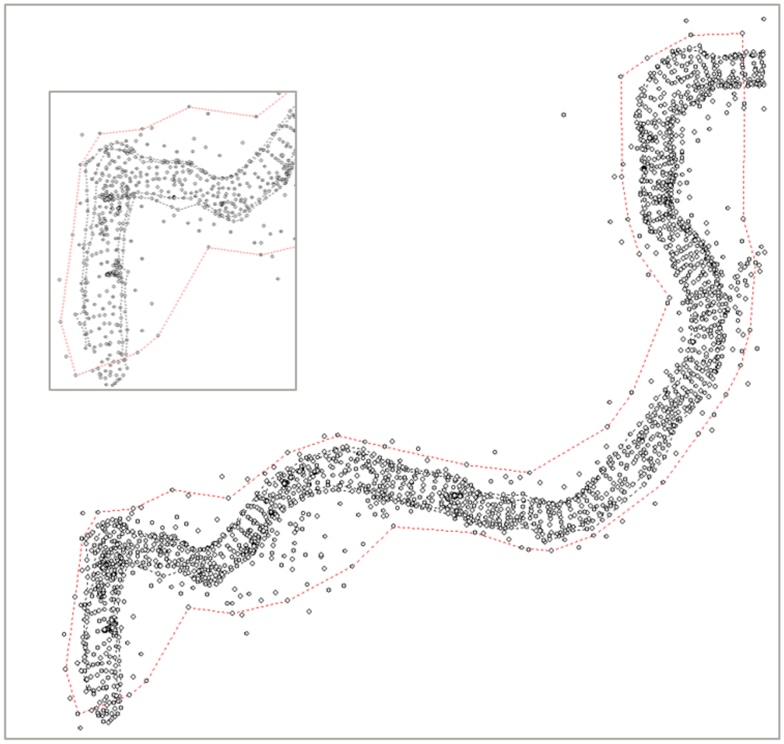

## WHITE SUCKER YOUNG-OF-YEAR



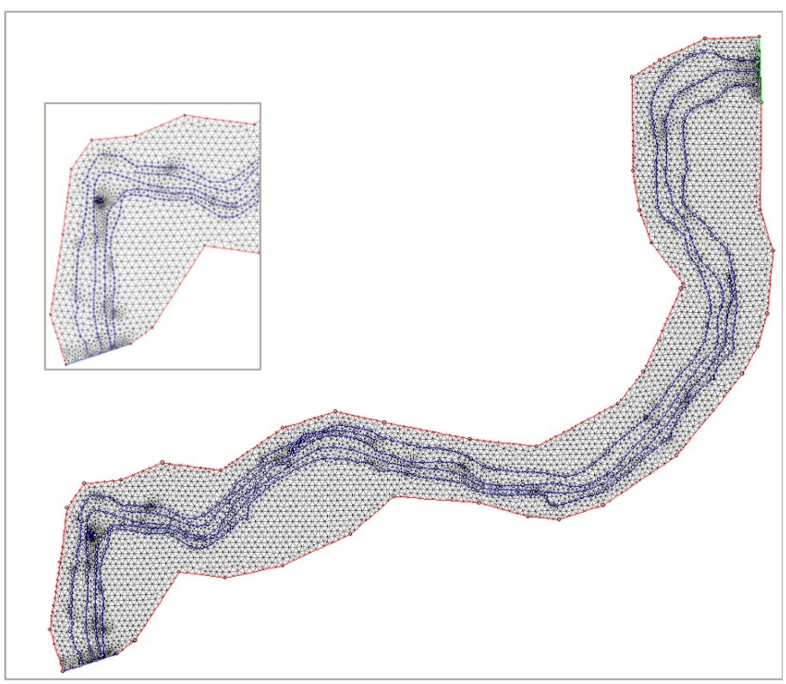
## Appendix D. River2D Bed, Mesh, Channel Index

Primary inputs for River2D flow and habitat modeling are shown here and were developed with the River2D software components R2D_Bed and R2D_Mesh from surveyed bed elevation data with substrate identification connected to each point.

**Bed – elevation and roughness assigned to each point. Break-lines inserted at important features such as toe of bank.**

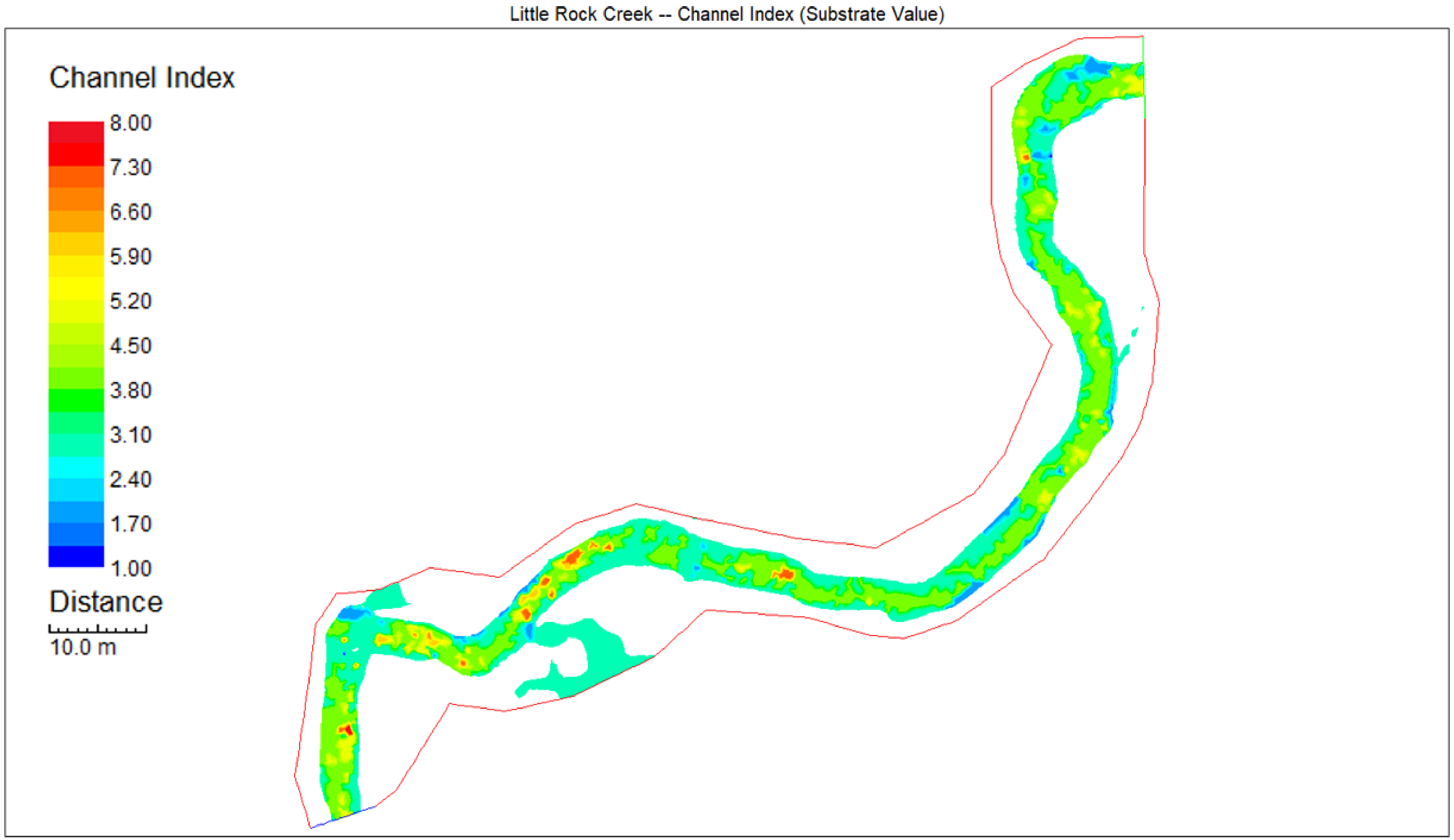


**Mesh – triangulated irregular network**



**Channel Index file of substrate types and definitions**

| Substrate Type | Channel Index Value |
|---|---|
| large boulder | 8 |
| small boulder | 7 |
| rubble | 6 |
| cobble | 5 |
| gravel | 4 |
| sand | 3 |
| silt | 2 |
| detritus | 1 |



## Appendix E. River2D Simulation Statistics

***Modeling flows used to develop River2D flow model, including parameters entered and quality checks. The values in the WSEL In (m) column uses the measured (observed in the field) WSEL which is why the measured values are sometime higher than the estimated inflow WSEL for flows higher than the measured flow. This is because the estimated values were extracted from the discharge to WSEL rating curve using regression analysis. Bolded values indicate field measurements.***

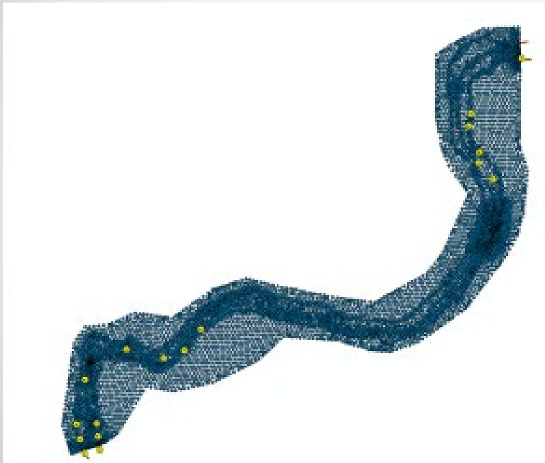
| **Type** | **Q Inflow (cms)** | **Q Inflow (cfs)** | **WSEL Outflow (m)** | **WSEL Inflow (m)** | **estimated WSEL Inflow + 0.05m** | **Model output WSEL Inflow (m)** | **WSE Inflow difference (<0.03 m acceptable)** | **Total Outflow Q (cms)** | **Net Q – Difference between Q In and Out (≤ 0.003 cms acceptable)** | **Max Velocity Magnitude** | **Max Froude (< 1)** | **Minimum depth for groundwater flow** | **Groundwater transmissivity** | **Solution Change (<0.00001)** | **Mesh QI (>0.15)** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | *Model Input* | | *Model Input* | | *Model Input* | *Output* | *Model Check* | *Output* | *Model Check* | *Simulation value* | *Simulation value* | *Input parameter* | *Input parameter* | *Model Check* | *Model Check* |
| estimated | 1.5965 | 56.38 | 327.186 | 327.883 | 327.933 | 327.870 | 0.01 | 1.5935 | 0.003 | 1.01 | 0.79 | 0.01 | 0.10 | Y | 0.31 |
| estimated | 0.7354 | 25.97 | 327.150 | 327.650 | 327.700 | 327.645 | 0.00 | 0.7334 | 0.002 | 0.81 | 0.57 | 0.03 | 0.10 | Y | 0.21 |
| measured | 0.6461 | 22.82 | 327.138 | 327.626 | 327.676 | 327.627 | 0.00 | 0.6442 | 0.002 | 0.78 | 0.56 | 0.03 | 0.10 | Y | 0.21 |
| estimated | 0.3793 | 13.39 | 327.073 | 327.553 | 327.603 | 327.556 | 0.00 | 0.3776 | 0.002 | 0.86 | 0.82 | 0.03 | 0.10 | Y | 0.23 |
| measured | 0.2716 | 9.59 | 327.029 | 327.520 | 327.570 | 327.528 | -0.01 | 0.2699 | 0.002 | 0.88 | 0.89 | 0.03 | 0.10 | Y | 0.24 |
| estimated | 0.2265 | 8.00 | 327.003 | 327.512 | 327.562 | 327.516 | 0.00 | 0.2245 | 0.002 | 0.77 | 0.89 | 0.03 | 0.10 | Y | 0.23 |
| estimated | 0.1756 | 6.20 | 326.971 | 327.498 | 327.548 | 327.498 | 0.00 | 0.1734 | 0.002 | 0.70 | 0.88 | 0.03 | 0.10 | Y | 0.23 |
| measured | 0.1387 | 4.90 | 326.941 | 327.500 | 327.550 | 327.496 | 0.00 | 0.1385 | 0.000 | 0.74 | 0.93 | 0.03 | 0.01 | Y | 0.23 |
| estimated | 0.1141 | 4.03 | 326.925 | 327.482 | 327.532 | 327.483 | 0.00 | 0.1130 | 0.001 | 0.63 | 0.86 | 0.03 | 0.05 | Y | 0.23 |
| estimated | 0.0807 | 2.85 | 326.897 | 327.473 | 327.523 | 327.479 | -0.01 | 0.0804 | 0.000 | 0.58 | 0.75 | 0.03 | 0.01 | Y | 0.23 |
| measured | 0.0780 | 2.75 | 326.896 | 327.464 | 327.514 | 327.477 | -0.01 | 0.0777 | 0.000 | 0.59 | 0.88 | 0.03 | 0.01 | Y | 0.26 |
| estimated | 0.0543 | 1.92 | 326.872 | 327.465 | 327.515 | 327.453 | 0.01 | 0.0516 | 0.003 | 0.47 | 0.65 | 0.03 | 0.10 | Y | 0.23 |
| estimated | 0.0443 | 1.56 | 326.862 | 327.463 | 327.513 | 327.444 | 0.02 | 0.0427 | 0.002 | 0.37 | 0.60 | 0.03 | 0.10 | Y | 0.23 |
| estimated | 0.0345 | 1.22 | 326.852 | 327.460 | 327.510 | 327.443 | 0.02 | 0.0337 | 0.001 | 0.43 | 0.59 | 0.04 | 0.08 | Y | 0.23 |

## Appendix F. River2D Model Validation

Comparison of values measured in the field (observed) to simulation outputs from River2D hydraulic models for WSEL, depth, and velocity.

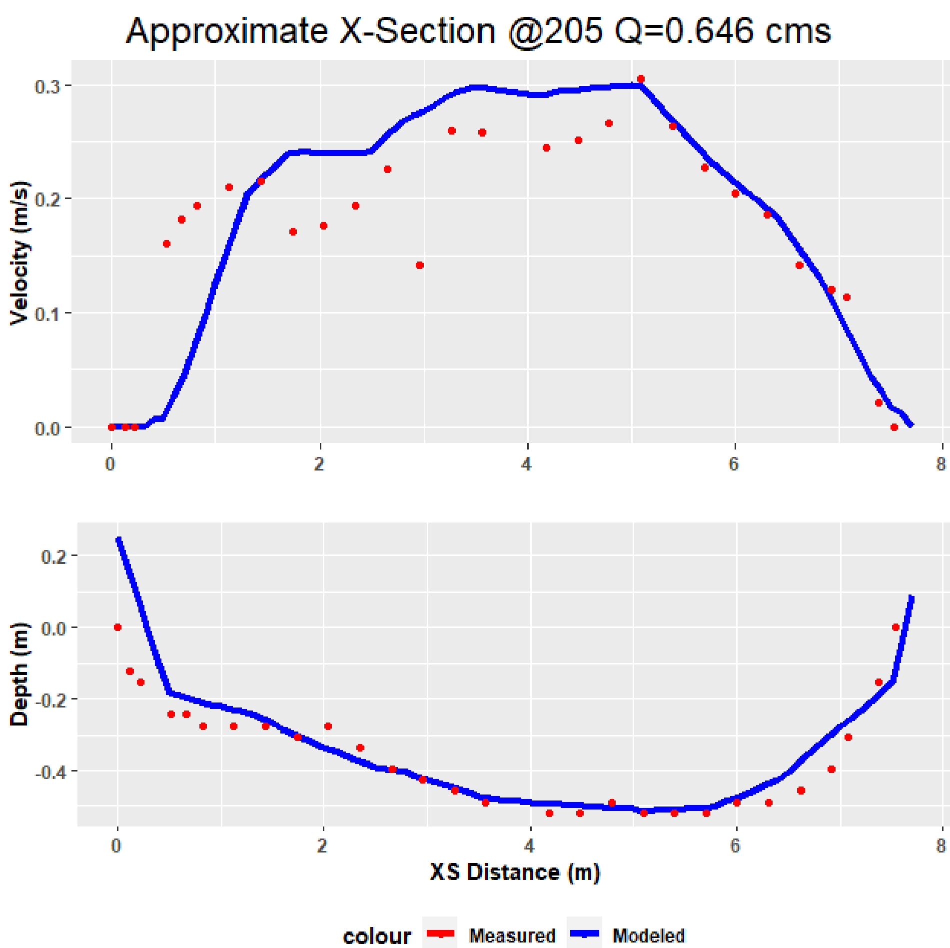
| Type of Flow Data | Discharge (Q) at Inflow (cfs) | Estimated vs Simulated WSEL difference at Inflow *(<0.03 acceptable)* | Total N Observations<br><br>Measured WSELs throughout Study Reach Compared to Simulated | Average Difference<br><br>Measured and Simulated WSEL (<0.03 acceptable) | Max Difference<br><br>Measured and Simulated WSEL (<0.03 acceptable) | SD of Difference<br><br>Measured and Simulated WSEL (<0.03 acceptable) |
|---|---|---|---|---|---|---|
| estimated | 56.37 | 0.01 | | | | |
| estimated | 25.97 | 0.00 | | | | |
| observed | 22.81 | 0.00 | 11 | 0.00 | 0.01 | 0.01 |
| estimated | 13.39 | 0.00 | | | | |
| observed | 9.59 | -0.01 | 17 | 0.00 | 0.01 | 0.01 |
| estimated | 8.00 | 0.00 | | | | |
| estimated | 6.20 | 0.00 | | | | |
| observed | 4.90 | 0.00 | 20 | 0.01 | 0.03 | 0.01 |
| estimated | 4.03 | 0.00 | | | | |
| estimated | 2.85 | -0.01 | | | | |
| observed | 2.75 | -0.01 | 31 | 0.00 | 0.02 | 0.01 |
| estimated | 1.92 | 0.01 | | | | |
| estimated | 1.56 | 0.02 | | | | |
| estimated | 1.22 | 0.02 | | | | |

## Simulated Water Surface Elevations compared to Surveyed at 9.59 cfs



| Field WSEL (meters) | River2D exported WSEL (closest point) | Difference |
|---|---|---|
| 327.5204 | 327.5259 | -0.01 |
| 327.5086 | 327.5254 | -0.02 |
| 327.5116 | 327.5200 | -0.01 |
| 327.5095 | 327.5179 | -0.01 |
| 327.4991 | 327.5073 | -0.01 |
| 327.4534 | 327.4560 | 0.00 |
| 327.1986 | 327.1904 | 0.01 |
| 327.1943 | 327.1887 | 0.01 |
| 327.1955 | 327.1806 | 0.01 |
| 327.1583 | 327.1537 | 0.00 |
| 327.1230 | 327.1464 | -0.02 |
| 327.0315 | 327.0352 | 0.00 |
| 327.0309 | 327.0340 | 0.00 |
| 327.0247 | 327.0306 | -0.01 |
| 327.0279 | 327.0311 | 0.00 |
| 327.0111 | 327.0298 | -0.02 |
| 327.0279 | 327.0290 | 0.00 |

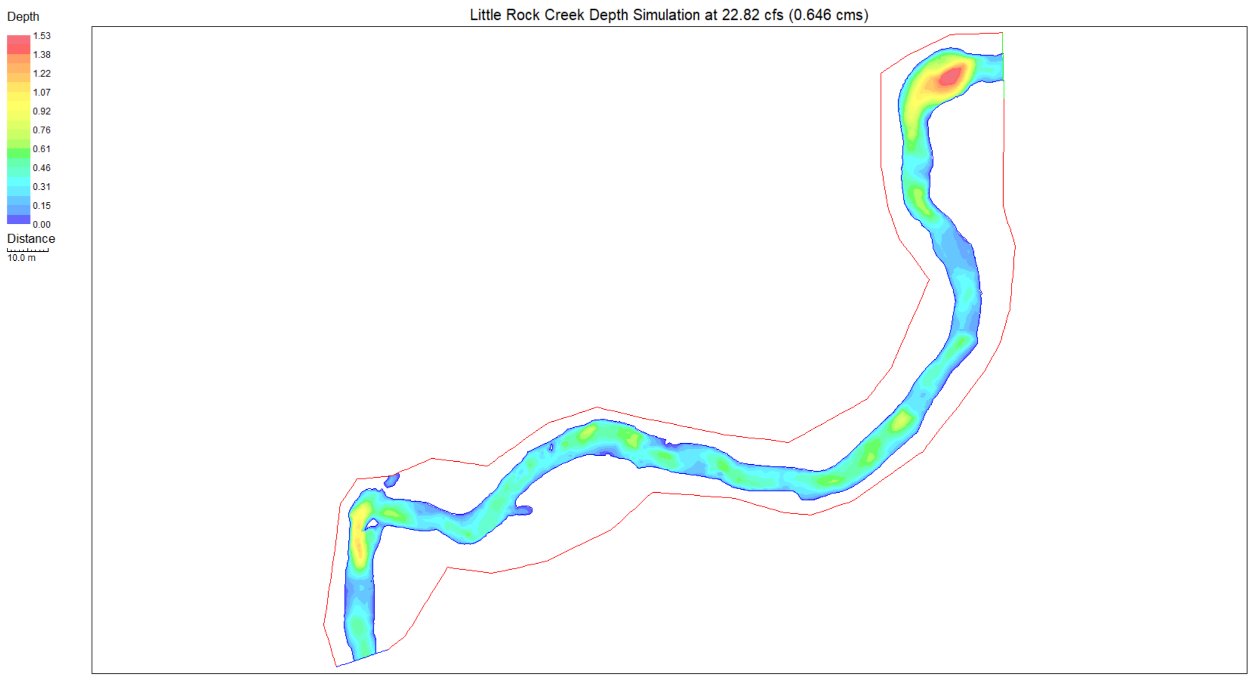
Threshold for acceptable difference between field measured and simulated water surface elevations: +/- 0.03 meter (1/10 foot)

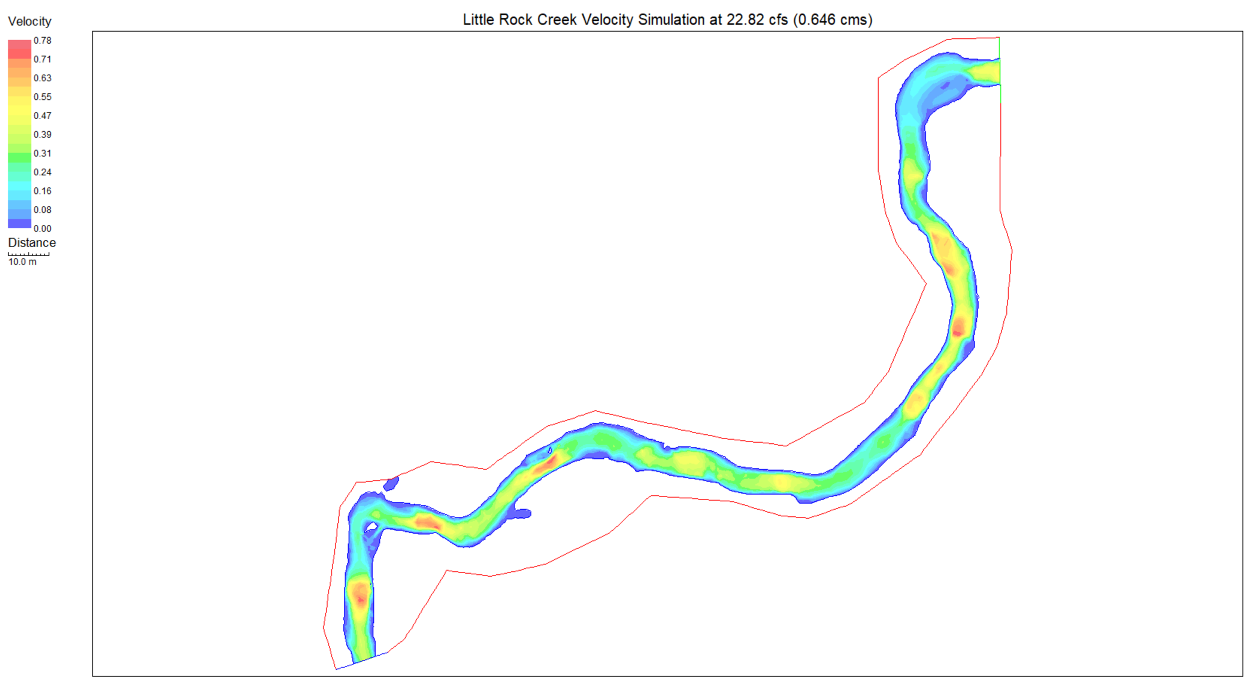


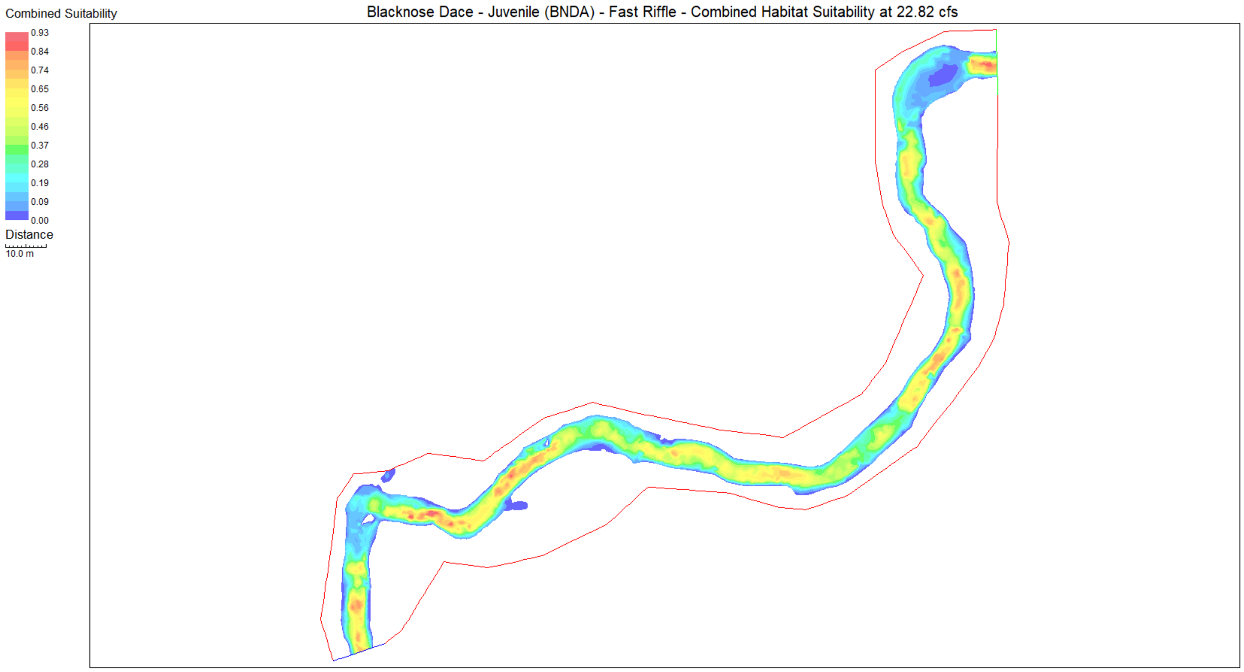
## Appendix G. River2D Simulations at Observed Flows

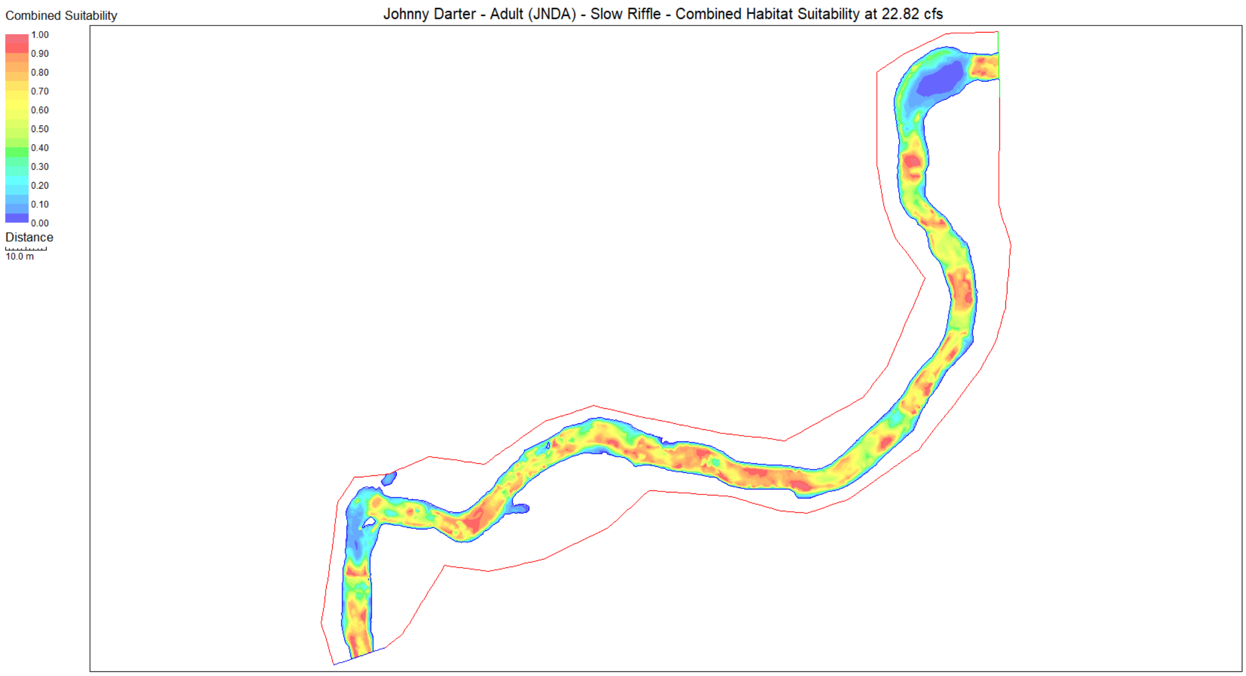
Depth and velocity hydraulic simulations, and combined habitat suitability index simulations for two species-life stages at the four observed flows. Blacknose dace adult, a fast riffle species-life stage and Johnny darter adult in the slow riffle guild, combined suitability results are shown here.

### River2D Simulations for 22.82 cubic feet per second

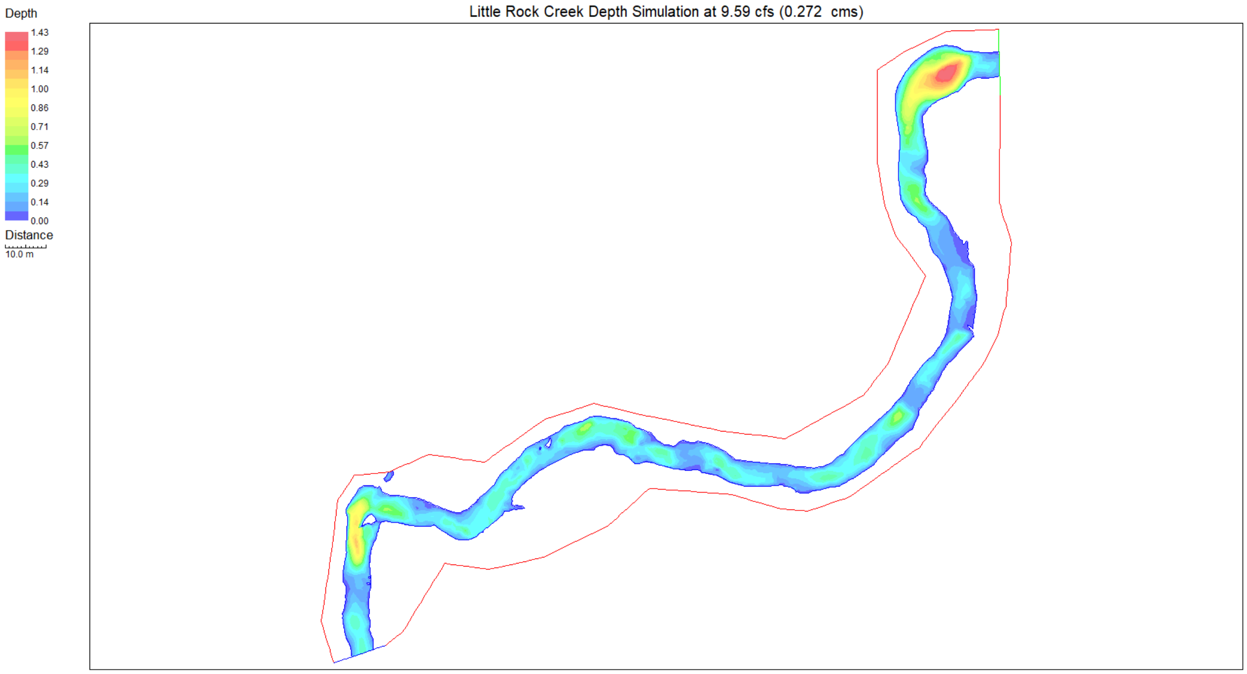


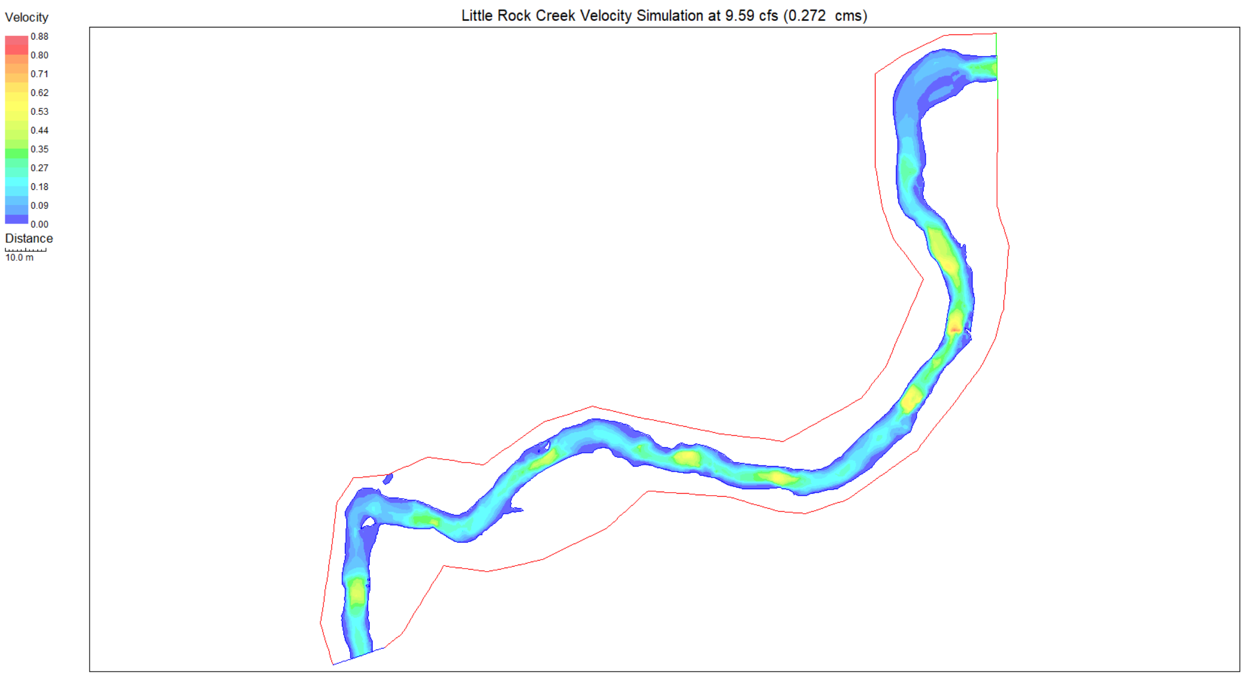



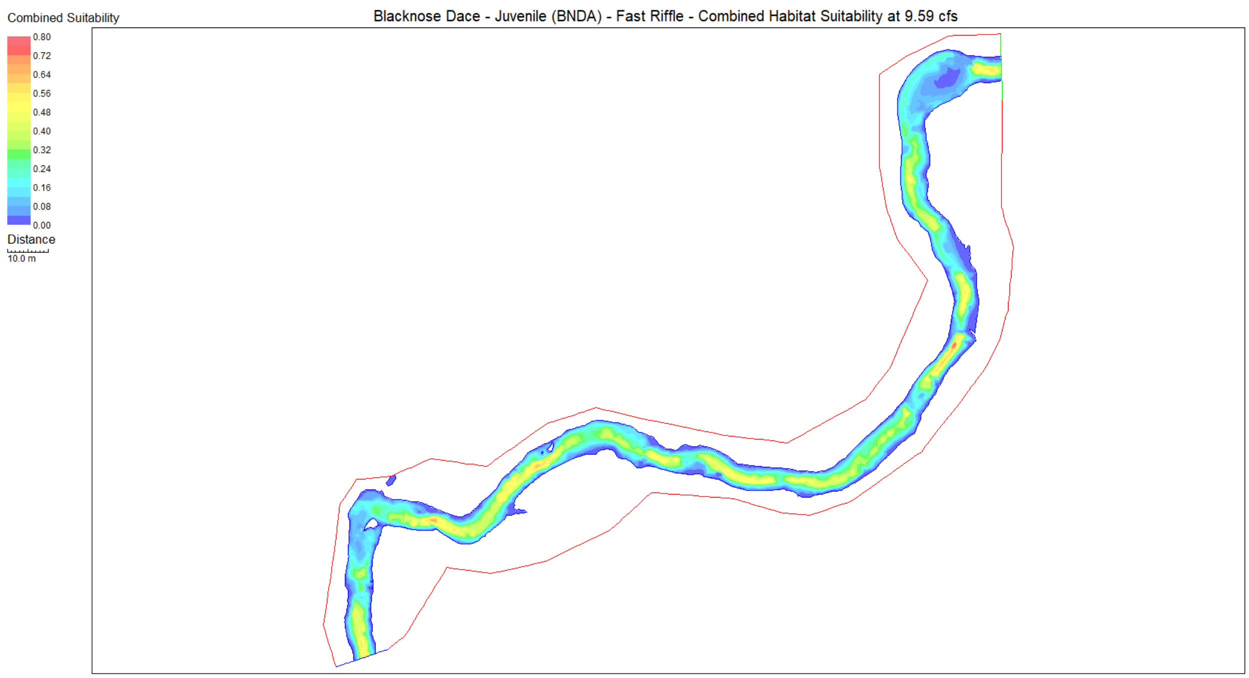

Combined Suitability
Blacknose Dace - Juvenile (BNDA) - Fast Riffle - Combined Habitat Suitability at 22.82 cfs
0.93
0.84
0.74
0.65
0.56
0.46
0.37
0.28
0.19
0.09
0.00
Distance
10.0 m

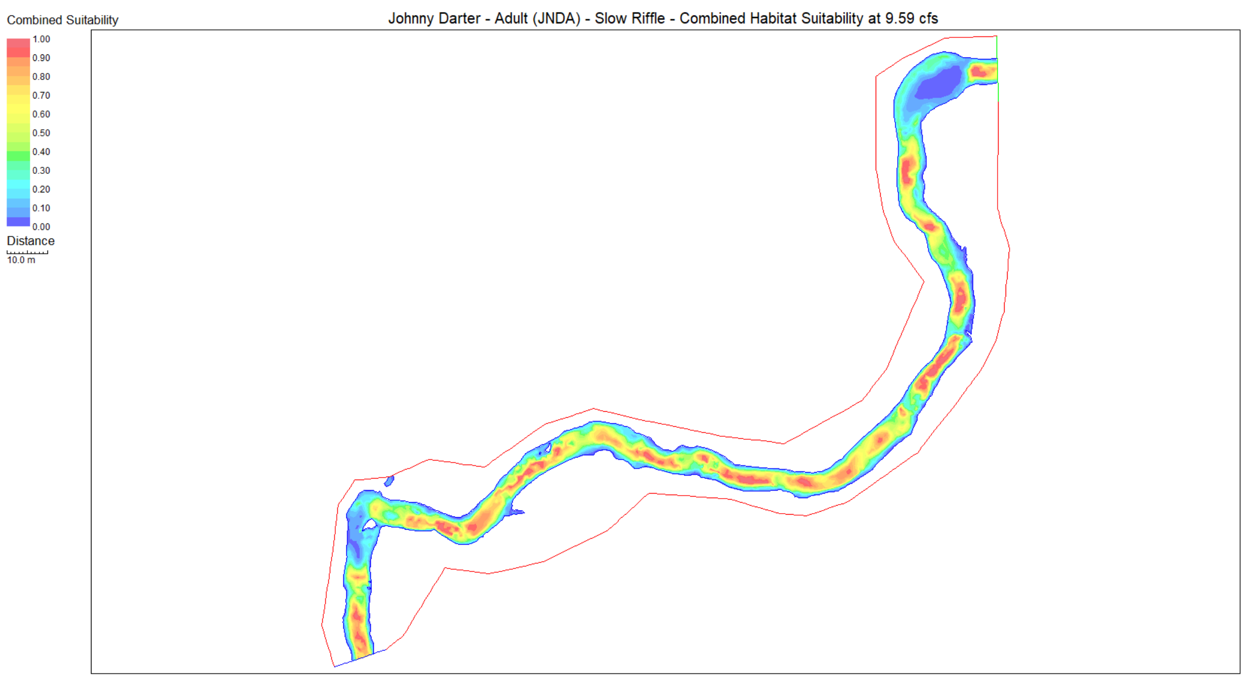

Combined Suitability
Johnny Darter - Adult (JNDA) - Slow Riffle - Combined Habitat Suitability at 22.82 cfs
1.00
0.90
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
Distance
10.0 m

## River2D Simulations for 9.59 cubic feet per second

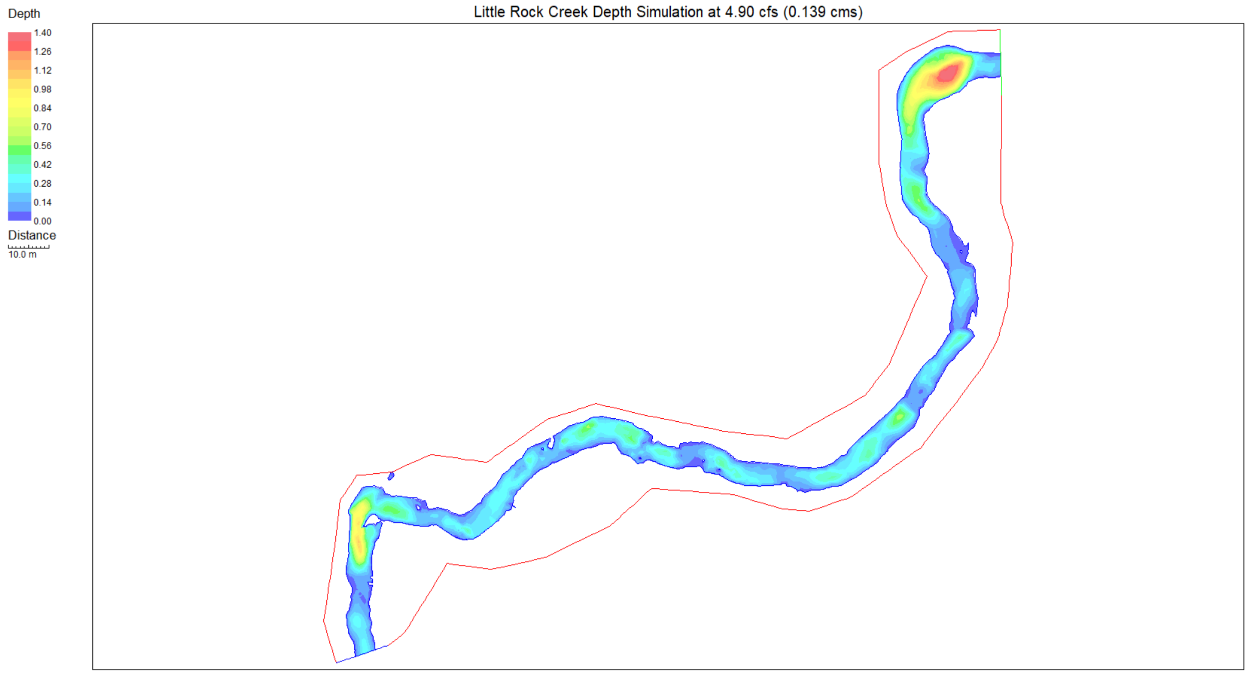


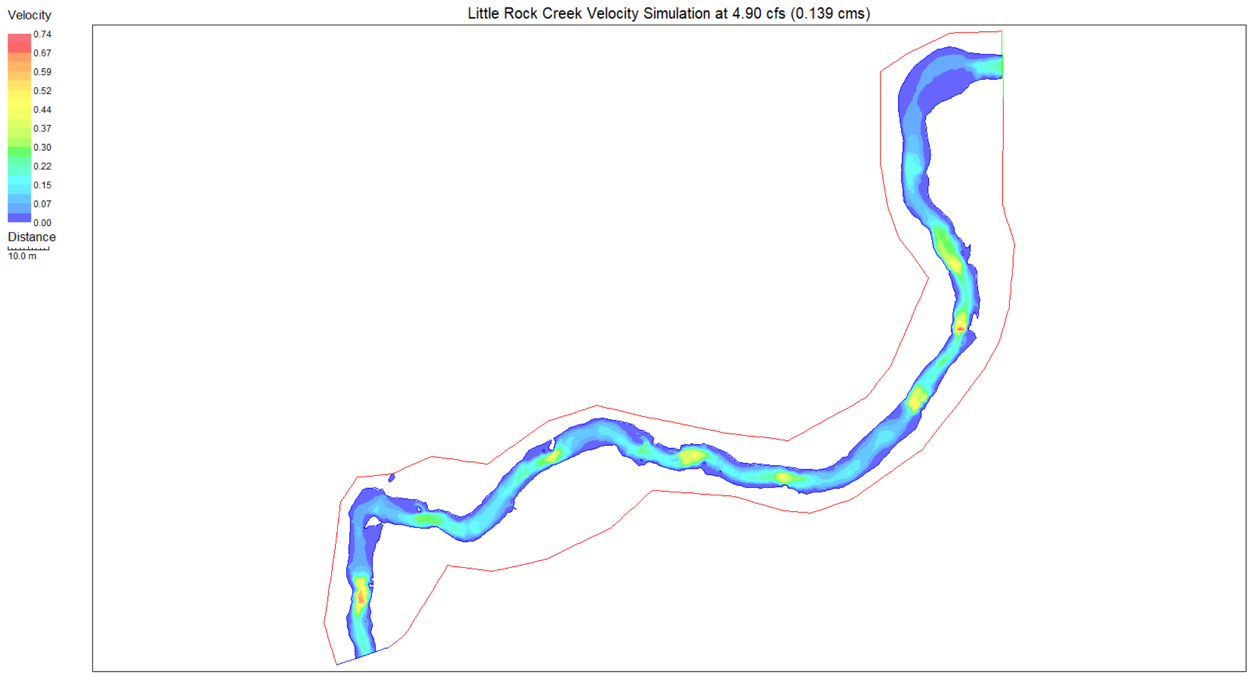



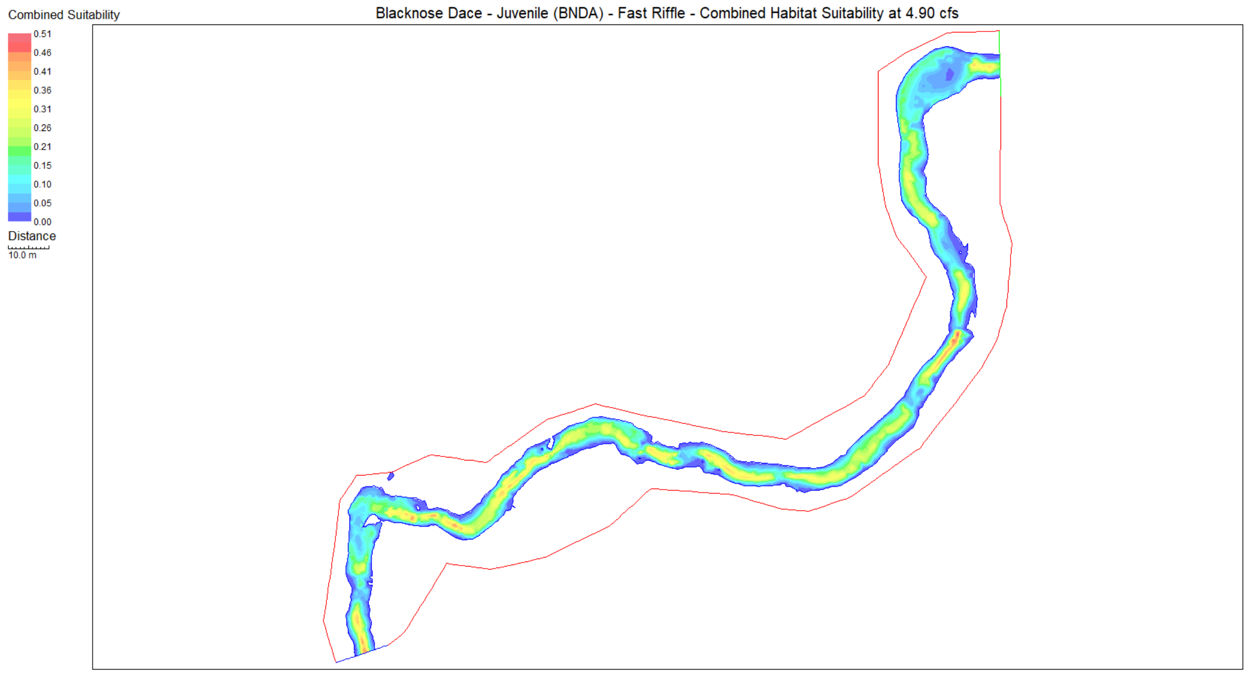

Combined Suitability
Blacknose Dace - Juvenile (BNDA) - Fast Riffle - Combined Habitat Suitability at 9.59 cfs
0.80
0.72
0.64
0.56
0.48
0.40
0.32
0.24
0.16
0.08
0.00
Distance
10.0 m

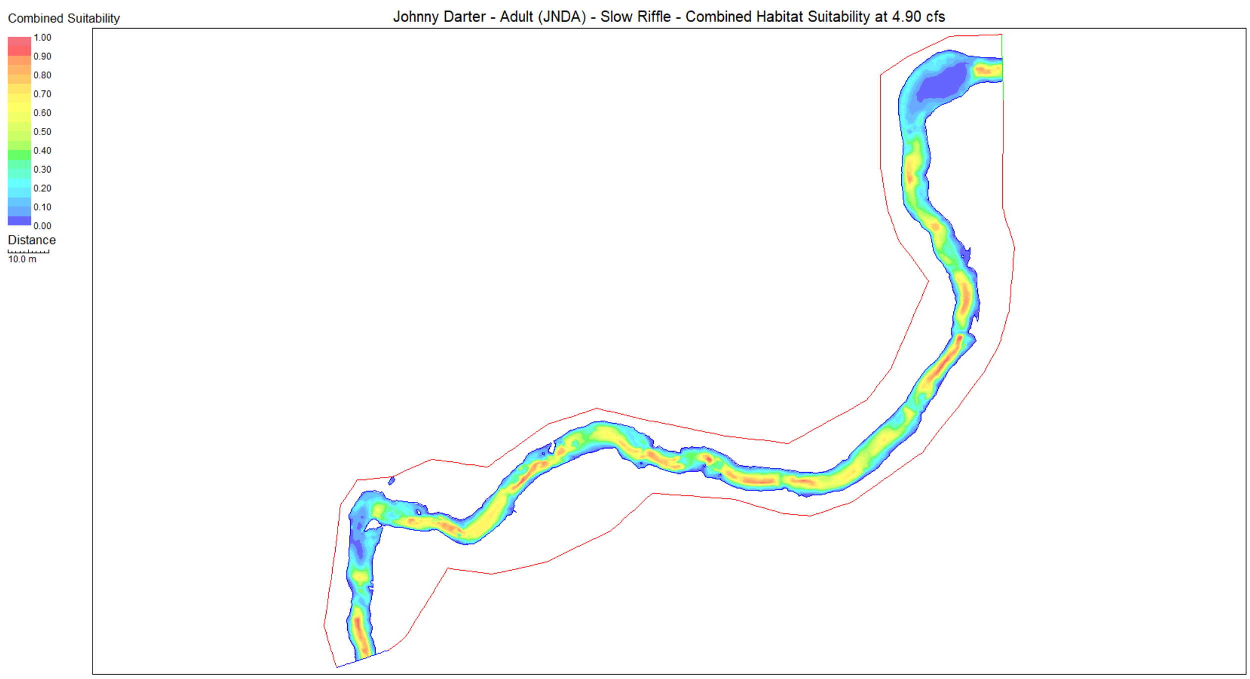

Combined Suitability
Johnny Darter - Adult (JNDA) - Slow Riffle - Combined Habitat Suitability at 9.59 cfs
1.00
0.90
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
Distance
10.0 m

## River2D Simulations for 4.90 cubic feet per second

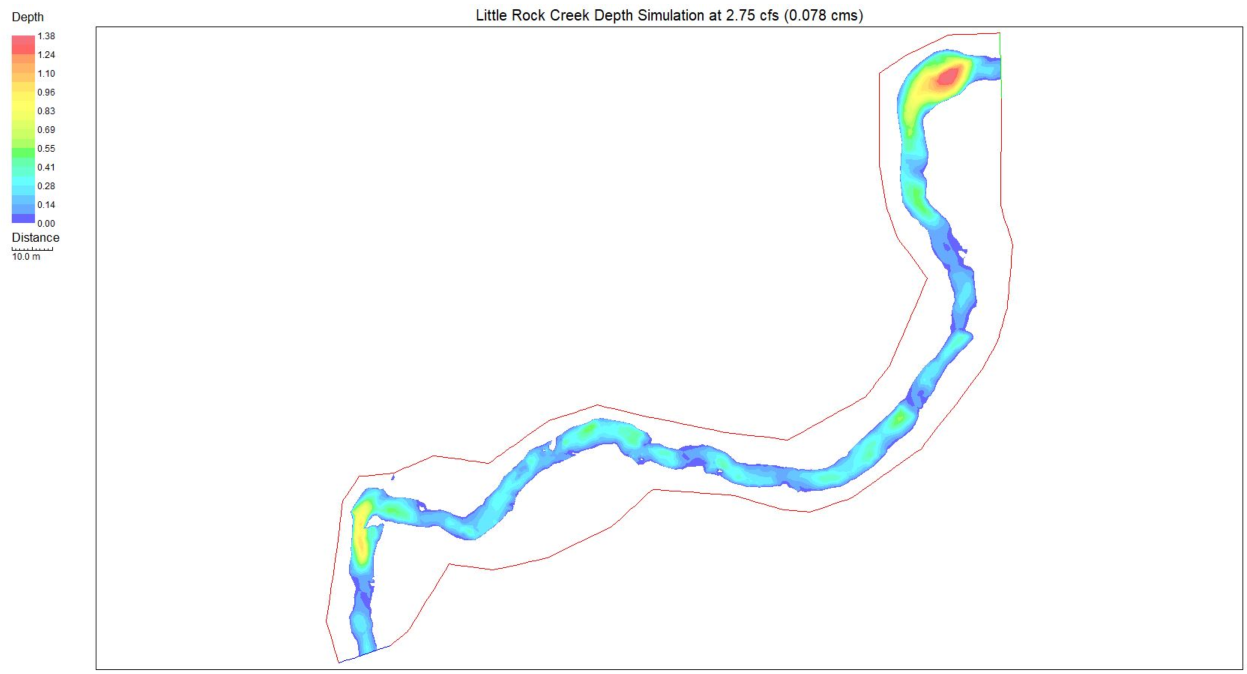


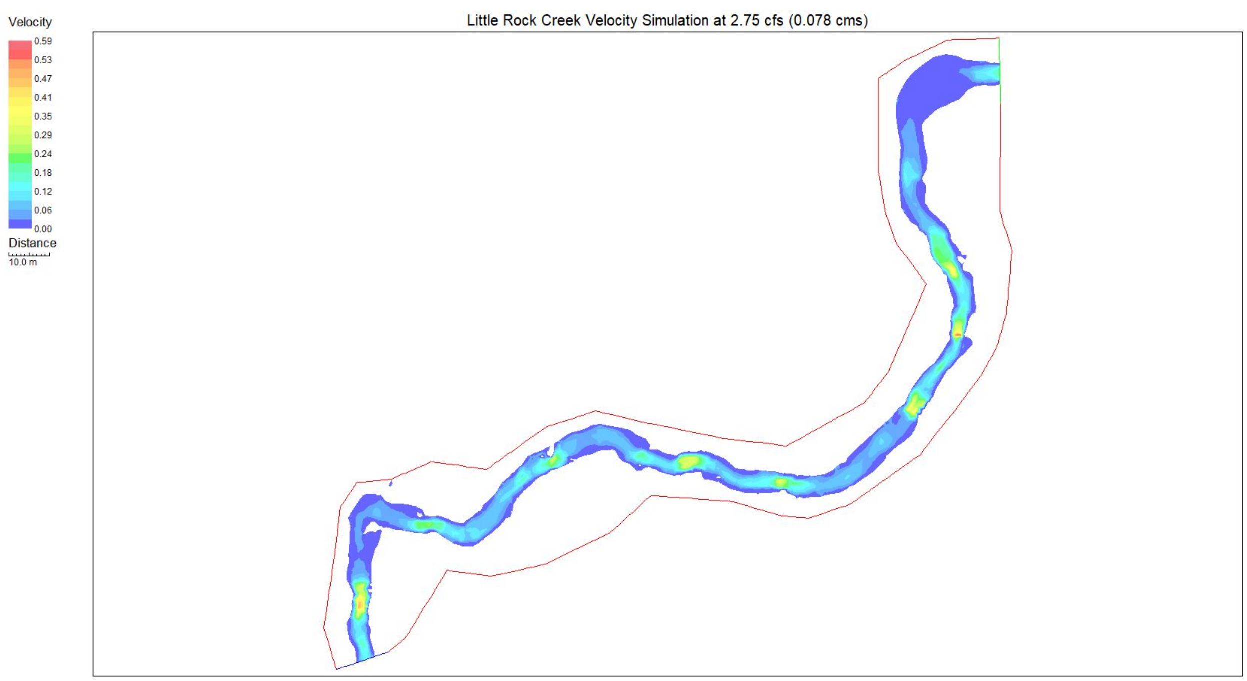



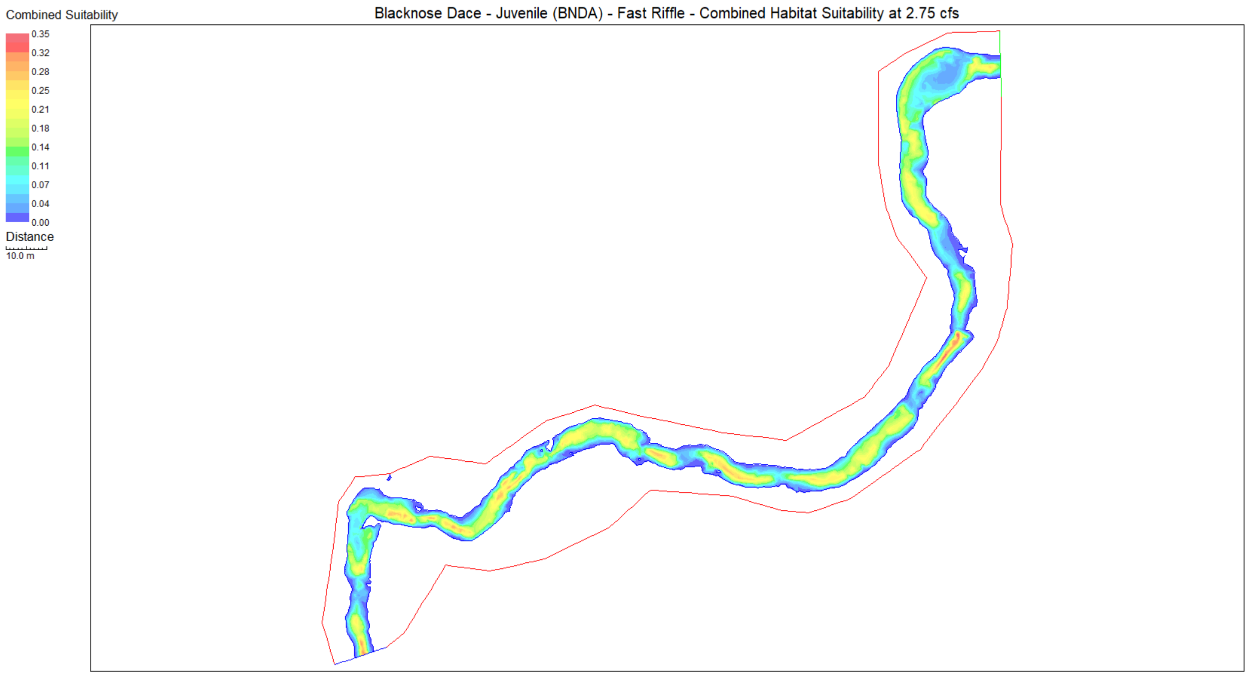

Combined Suitability
Blacknose Dace - Juvenile (BNDA) - Fast Riffle - Combined Habitat Suitability at 4.90 cfs
0.51
0.46
0.41
0.36
0.31
0.26
0.21
0.15
0.10
0.05
0.00
Distance
10.0 m

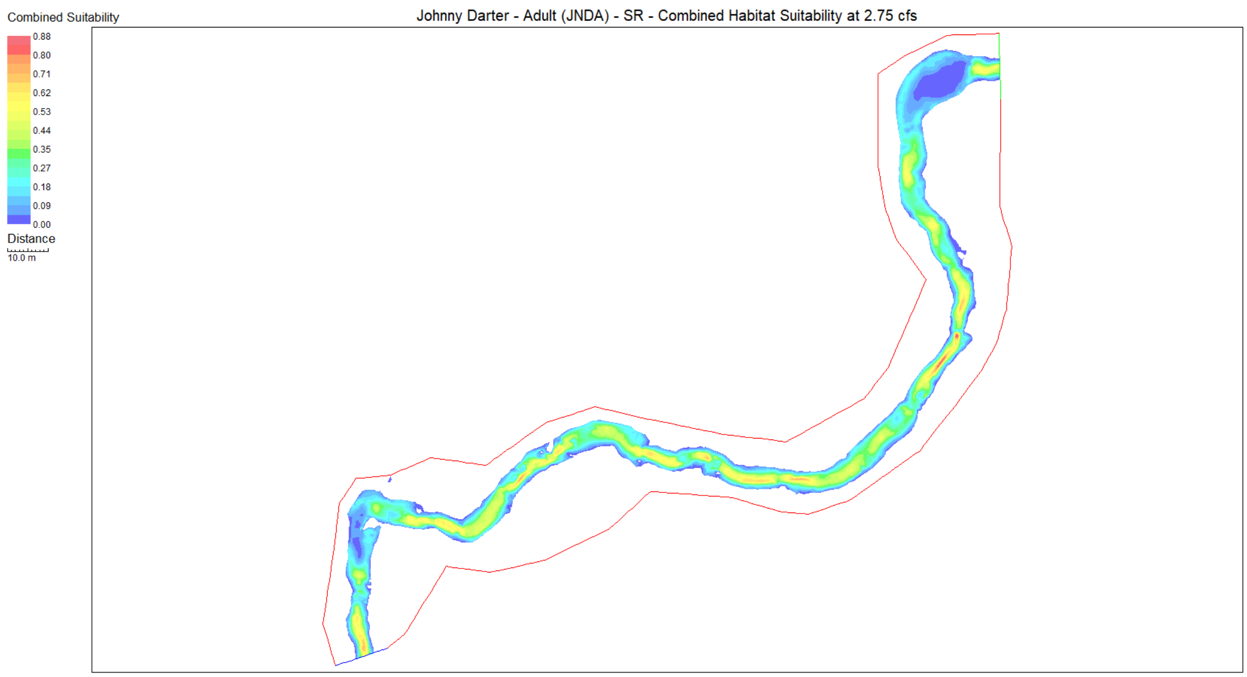

Combined Suitability
Johnny Darter - Adult (JNDA) - Slow Riffle - Combined Habitat Suitability at 4.90 cfs
1.00
0.90
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
Distance
10.0 m

## River2D Simulations for 2.75 cubic feet per second



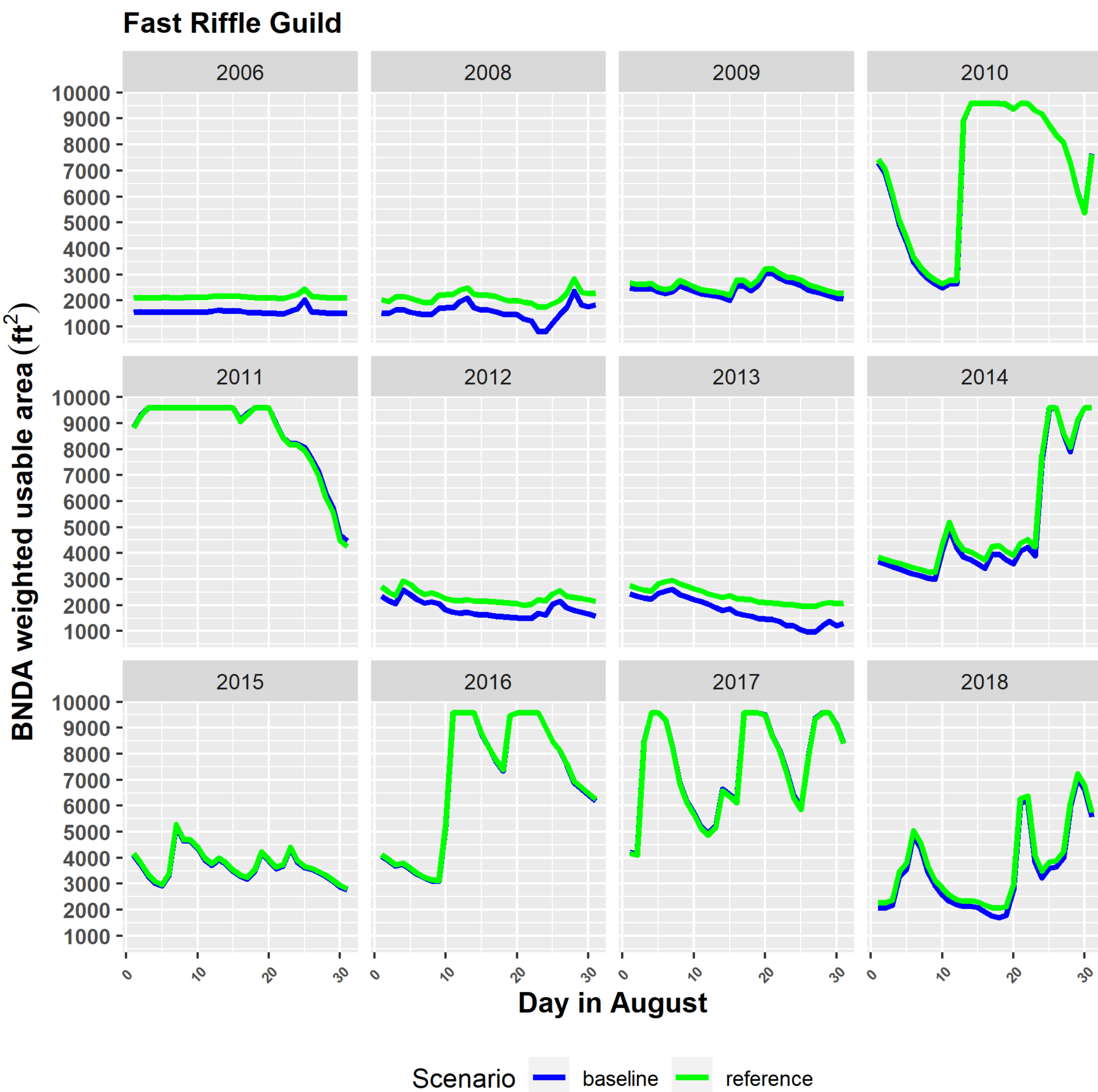



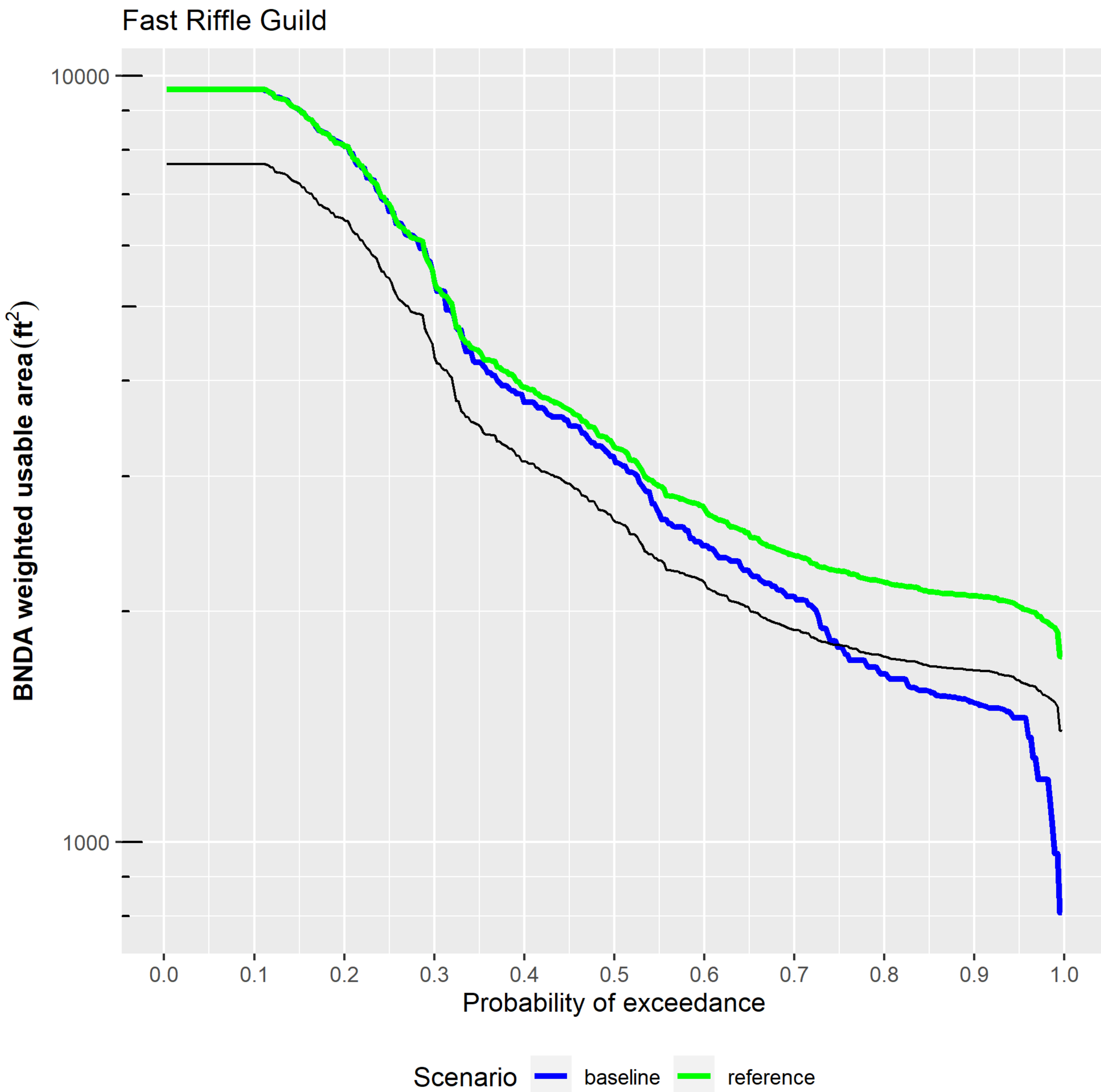

Combined Suitability
Blacknose Dace - Juvenile (BNDA) - Fast Riffle - Combined Habitat Suitability at 2.75 cfs
0.35
0.32
0.28
0.25
0.21
0.18
0.14
0.11
0.07
0.04
0.00
Distance
10.0 m

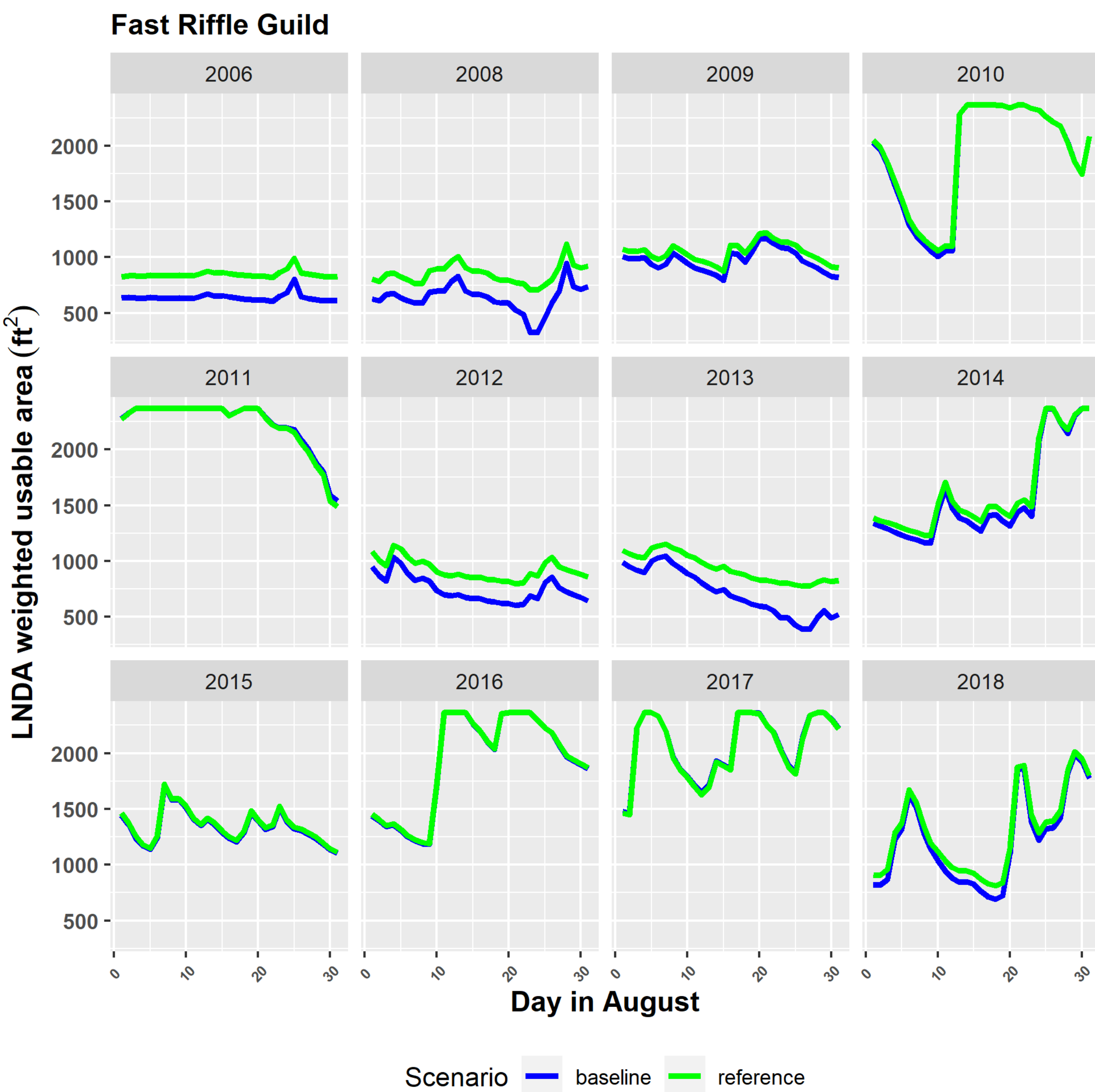

Combined Suitability
Johnny Darter - Adult (JNDA) - SR - Combined Habitat Suitability at 2.75 cfs
0.88
0.80
0.71
0.62
0.53
0.44
0.35
0.27
0.18
0.09
0.00
Distance
10.0 m

## Appendix H. Habitat Time Series and Duration Curve for All Species-Life Stages Modeled

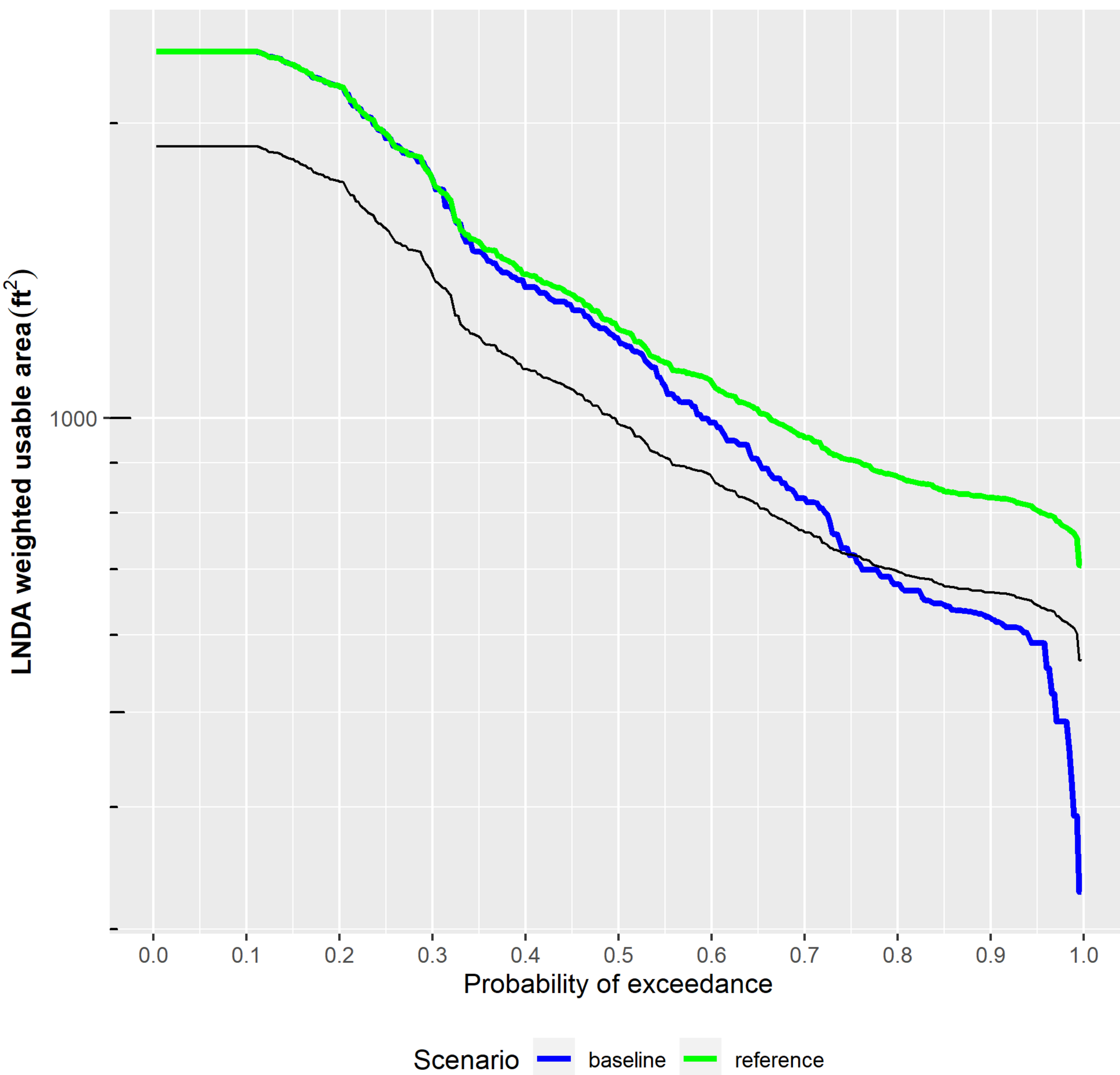
The guild is identified at the top of each graph and the species-life stage identified on the y-axis. For each species-life stage the habitat time series is displayed first followed by the habitat duration curve.

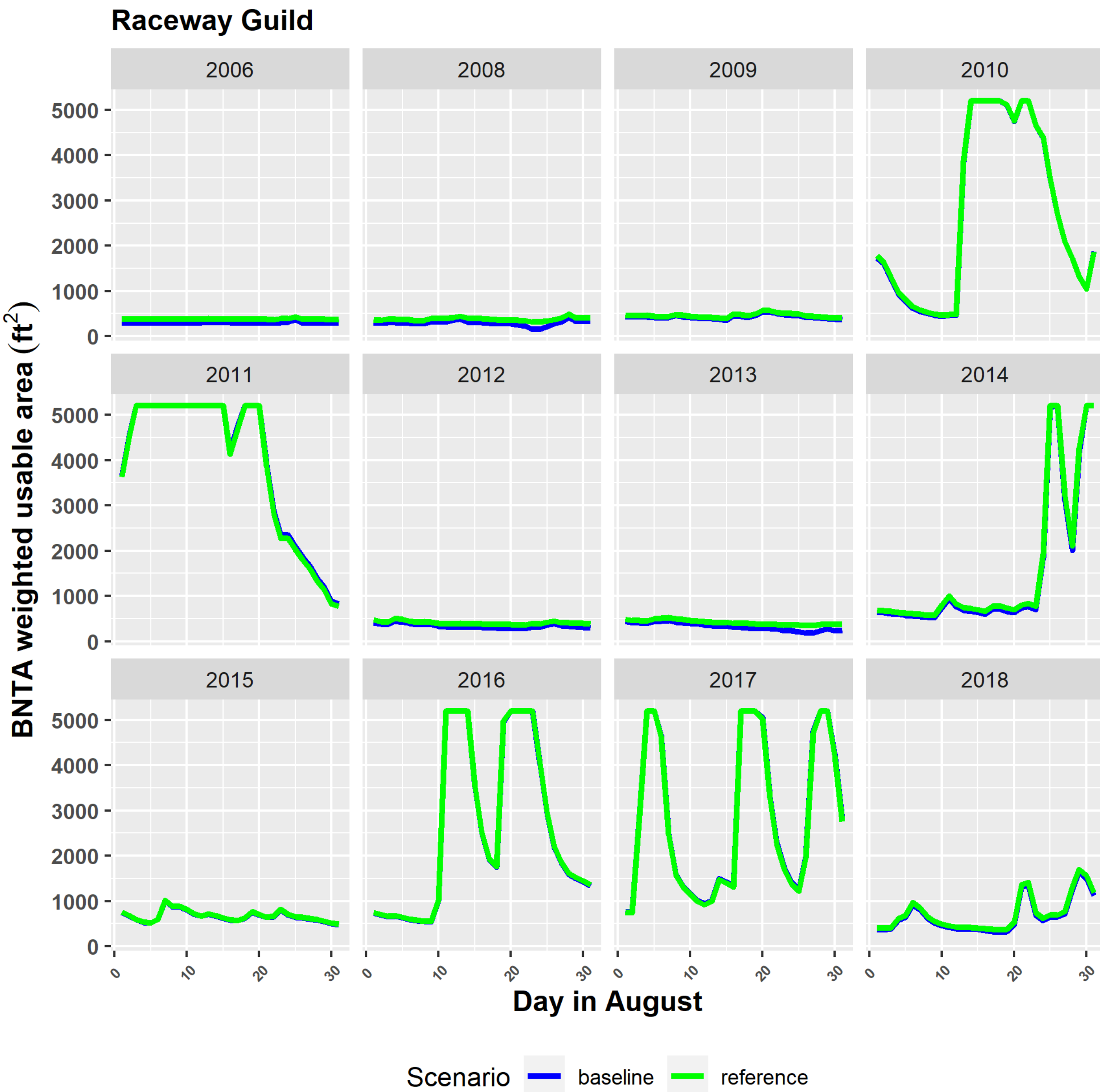


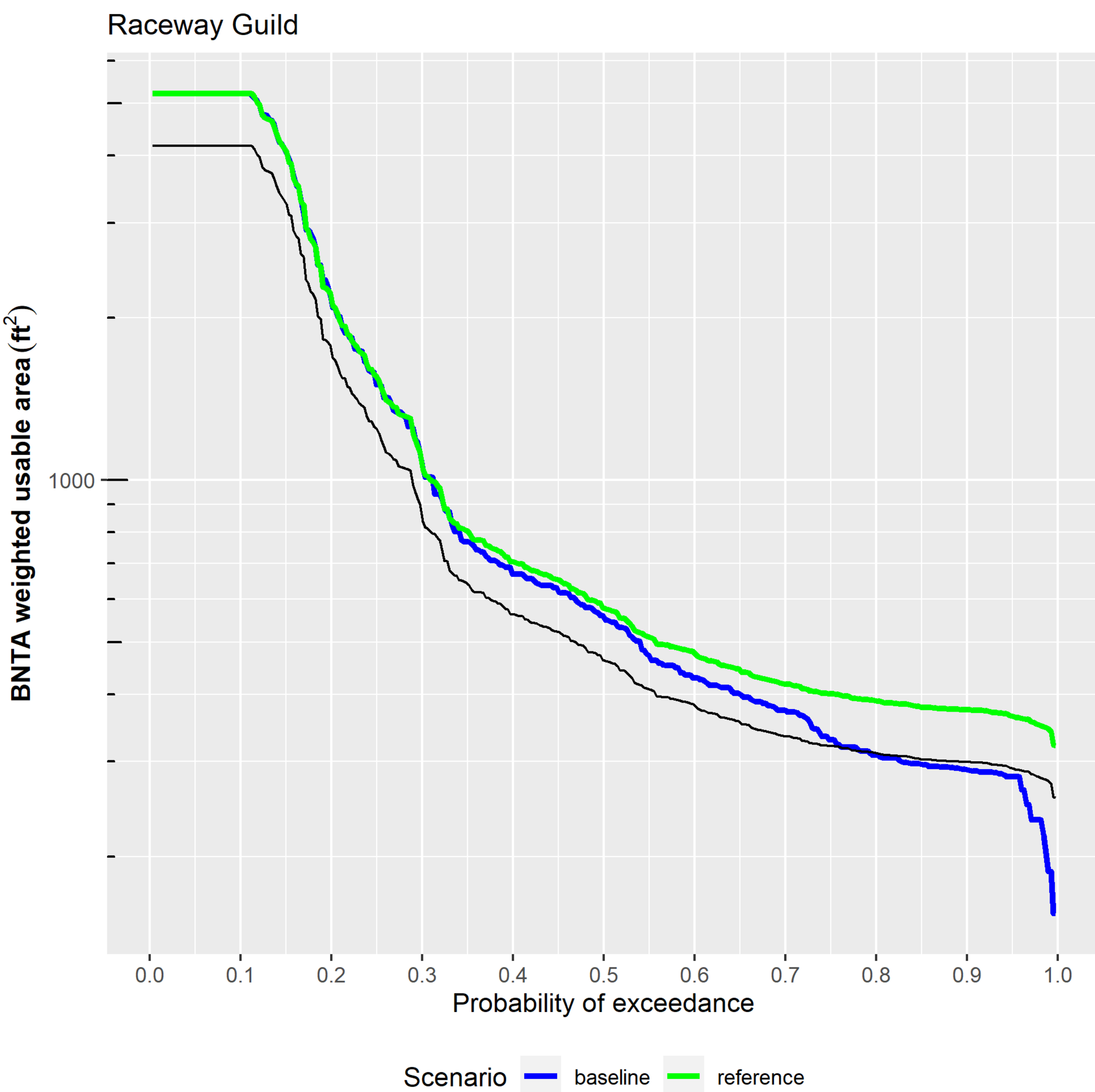

Fast Riffle Guild
10000
1000
BNDA weighted usable area (ft²)
0.0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0.9
1.0
Probability of exceedance
Scenario
baseline
reference

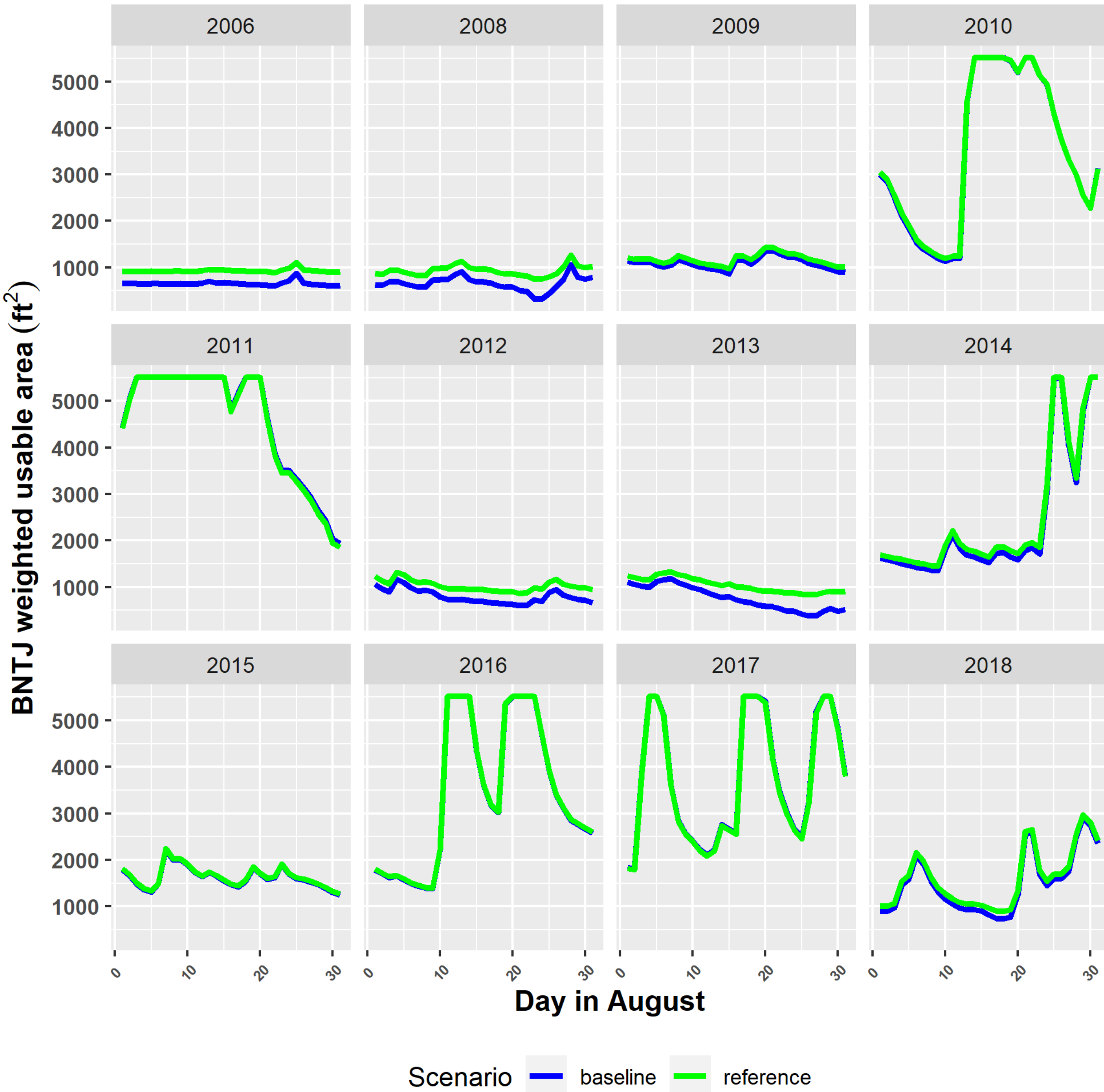

Fast Riffle Guild
2006
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
2018
LNDA weighted usable area (ft²)
2000
1500
1000
500
0
10
20
30
Day in August
Scenario
baseline
reference

Fast Riffle Guild

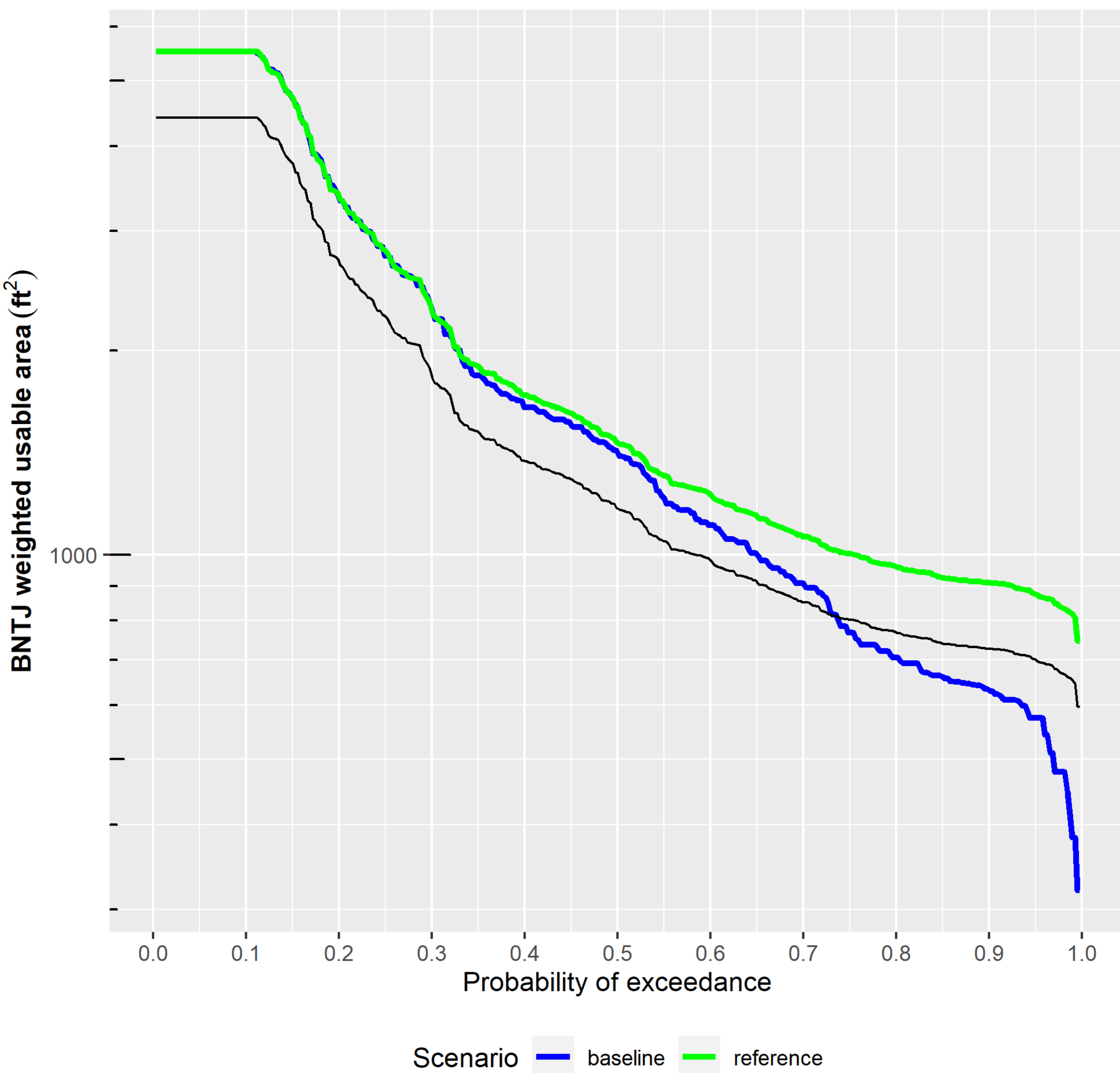



Raceway Guild
2006
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
2018
5000
4000
3000
2000
1000
0
BNTA weighted usable area (ft2)
0
10
20
30
Day in August
Scenario
baseline
reference

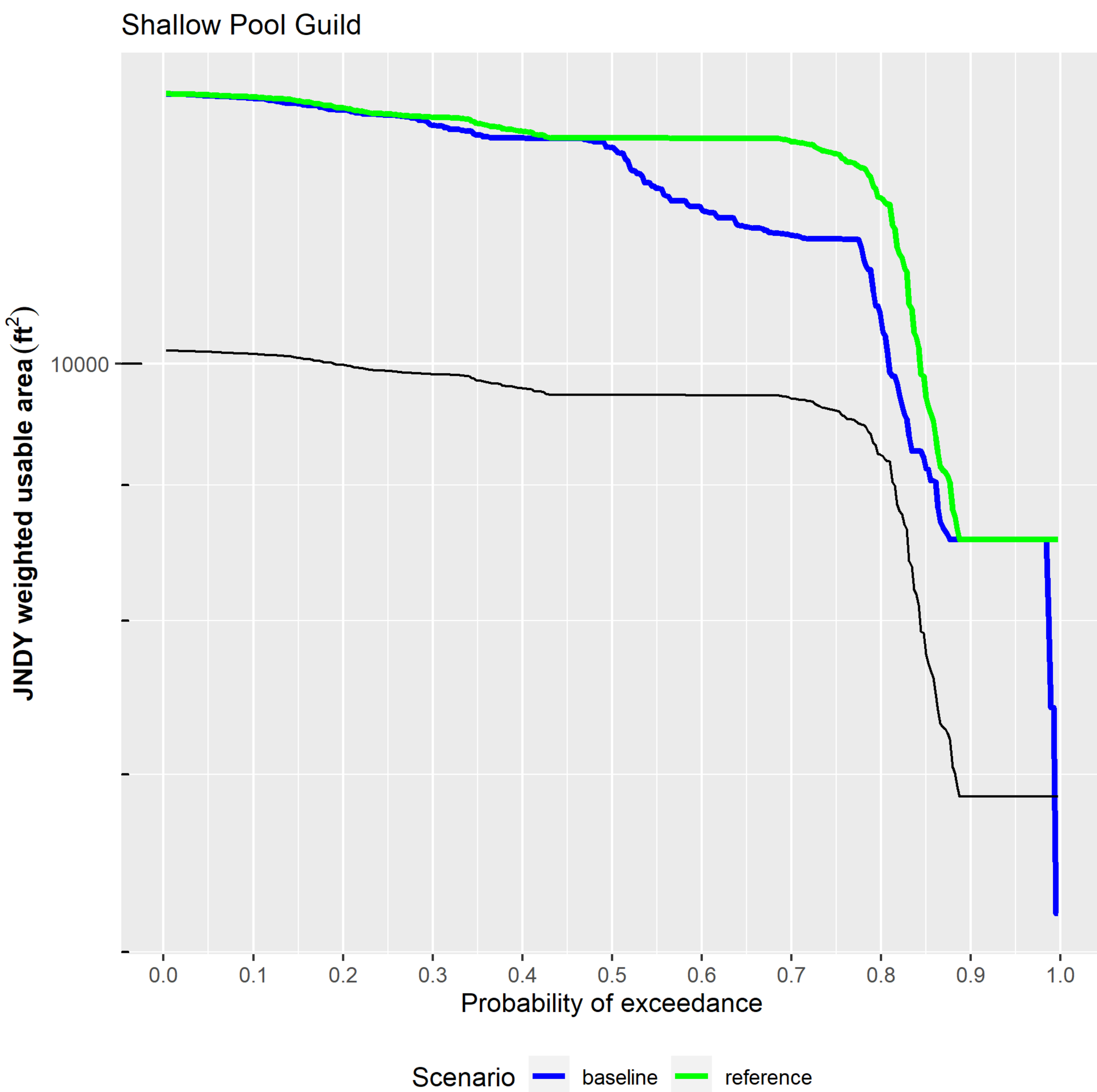

Raceway Guild
BNTA weighted usable area (ft$^2$)
1000
0.0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0.9
1.0
Probability of exceedance
Scenario
baseline
reference

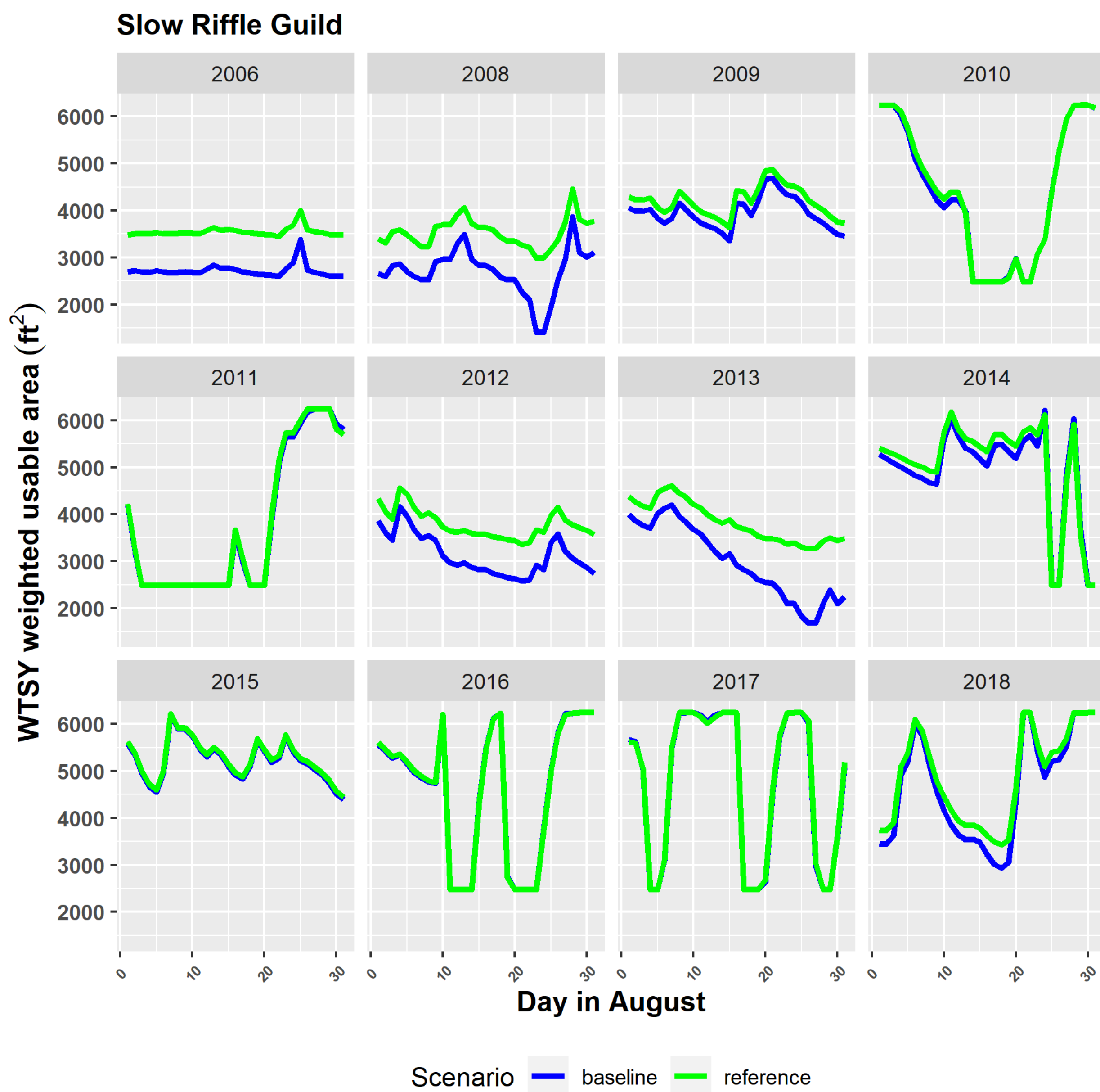

2006
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
2018
BNTJ weighted usable area (ft²)
5000
4000
3000
2000
1000
0
10
20
30
Day in August
Scenario
baseline
reference

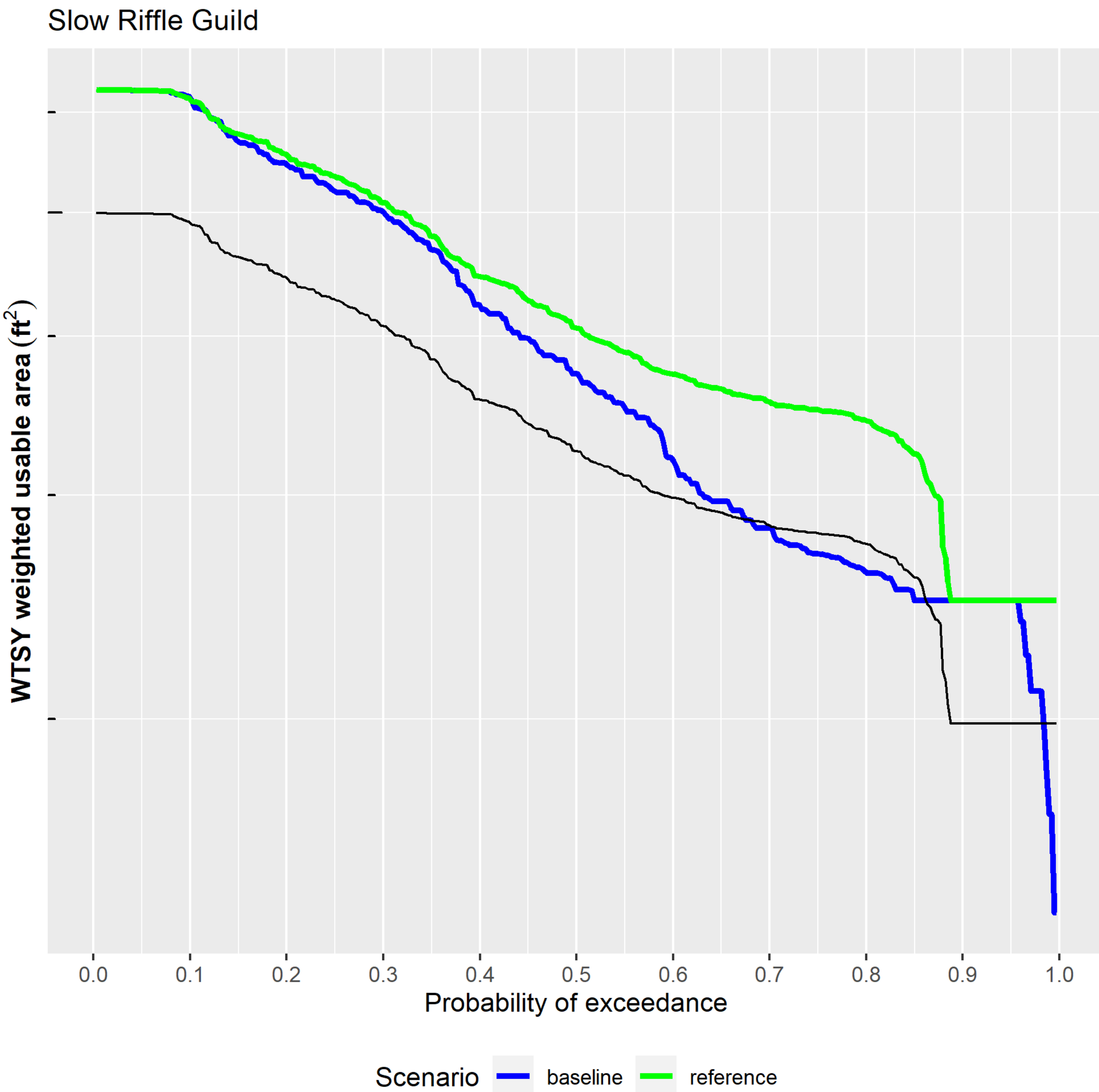
Raceway Guild

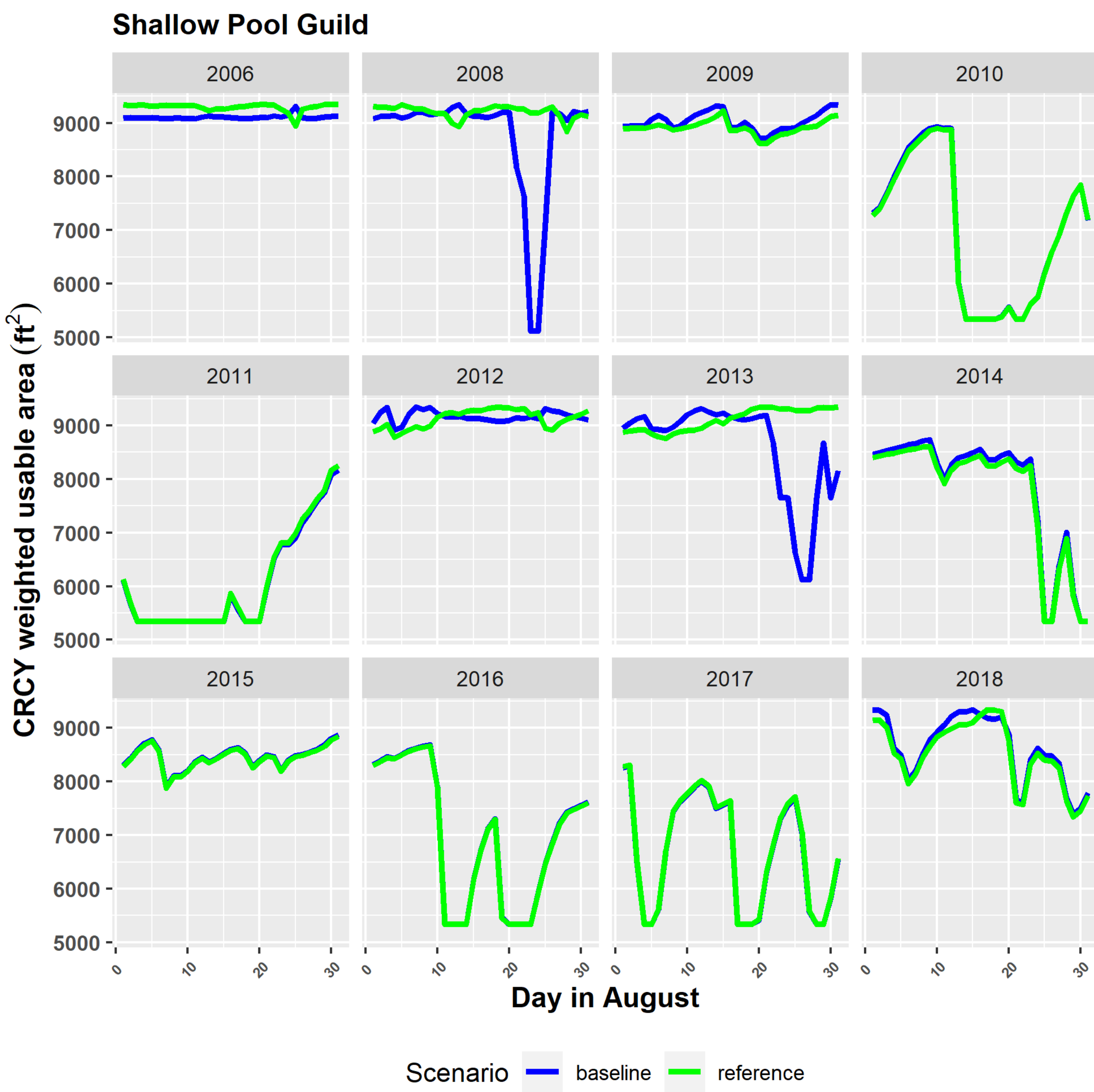


Slow Riffle Guild
2006
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
2018
10000
8000
6000
4000
2000
0
10
20
30
JNDA weighted usable area ($ft^2$)
Day in August
Scenario
baseline
reference


Shallow Pool Guild
JNDY weighted usable area (ft²)
10000
0.0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0.9
1.0
Probability of exceedance
Scenario
baseline
reference

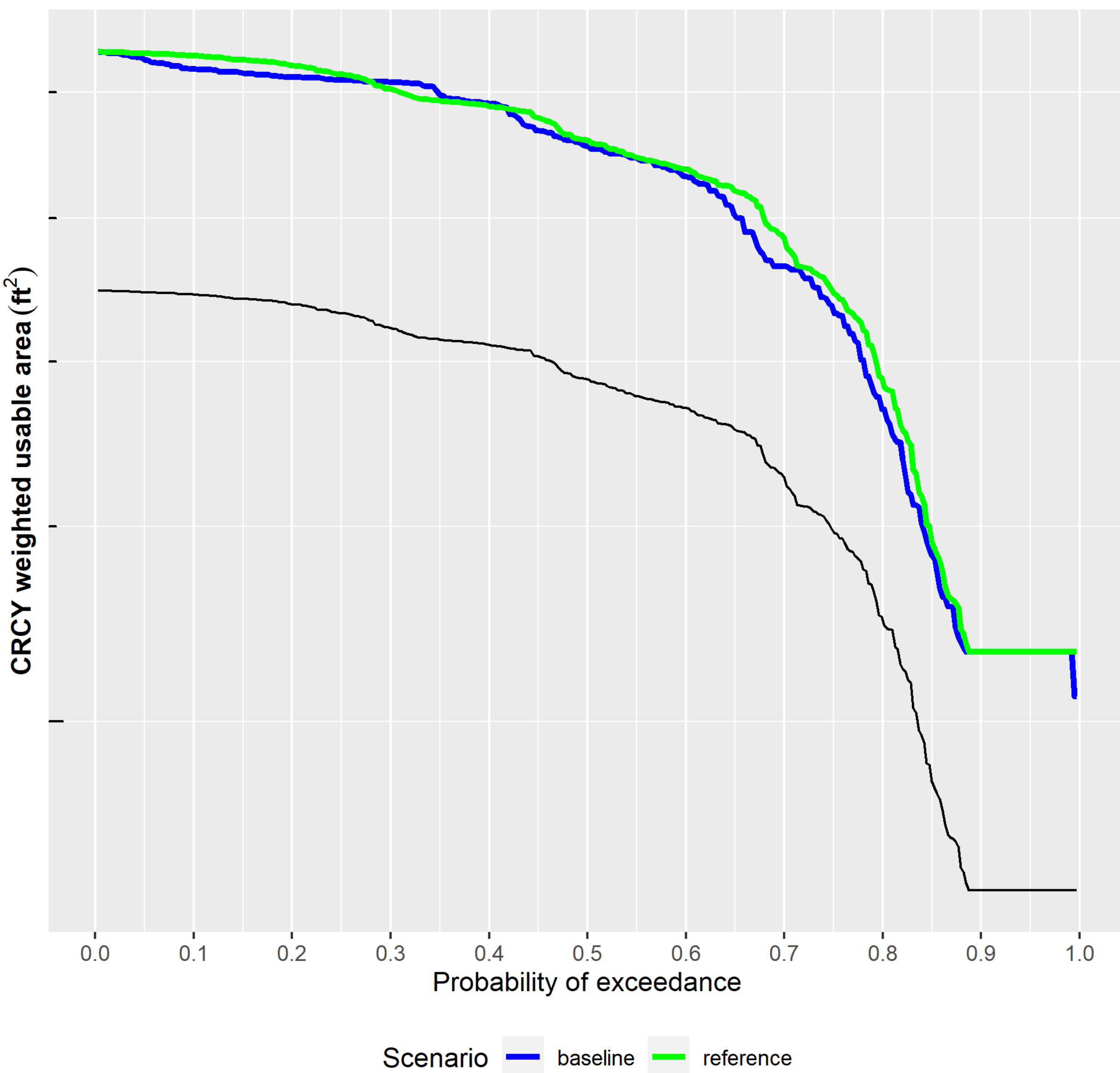

Slow Riffle Guild
2006
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
2018
WTSY weighted usable area (ft2)
6000
5000
4000
3000
2000
0
10
20
30
Day in August
Scenario
baseline
reference

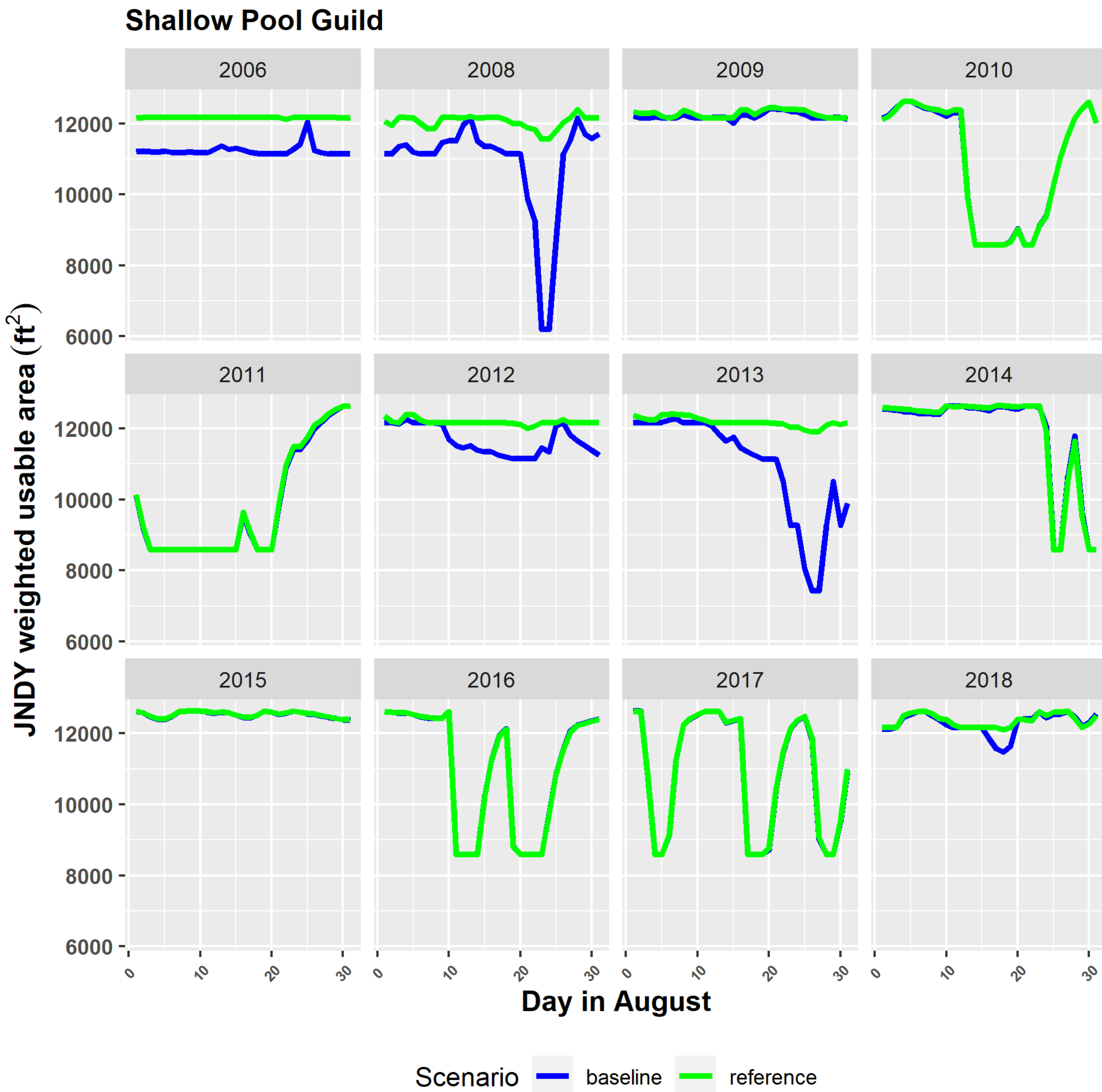

Slow Riffle Guild
WTSY weighted usable area (ft²)
0.0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0.9
1.0
Probability of exceedance
Scenario
baseline
reference

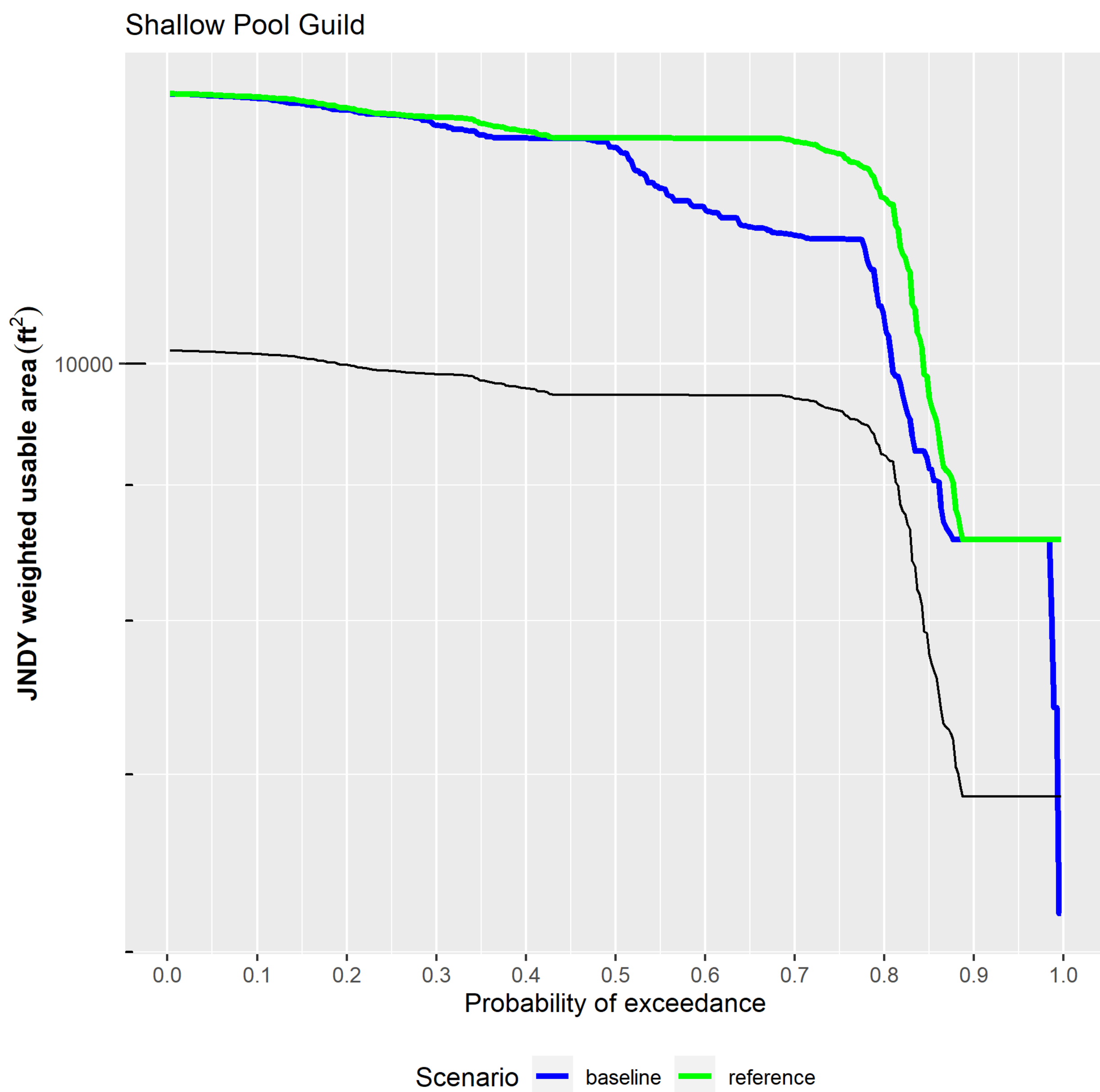


Shallow Pool Guild
2006
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
2018
CRCY weighted usable area (ft²)
9000
8000
7000
6000
5000
0
10
20
30
Day in August
Scenario
baseline
reference

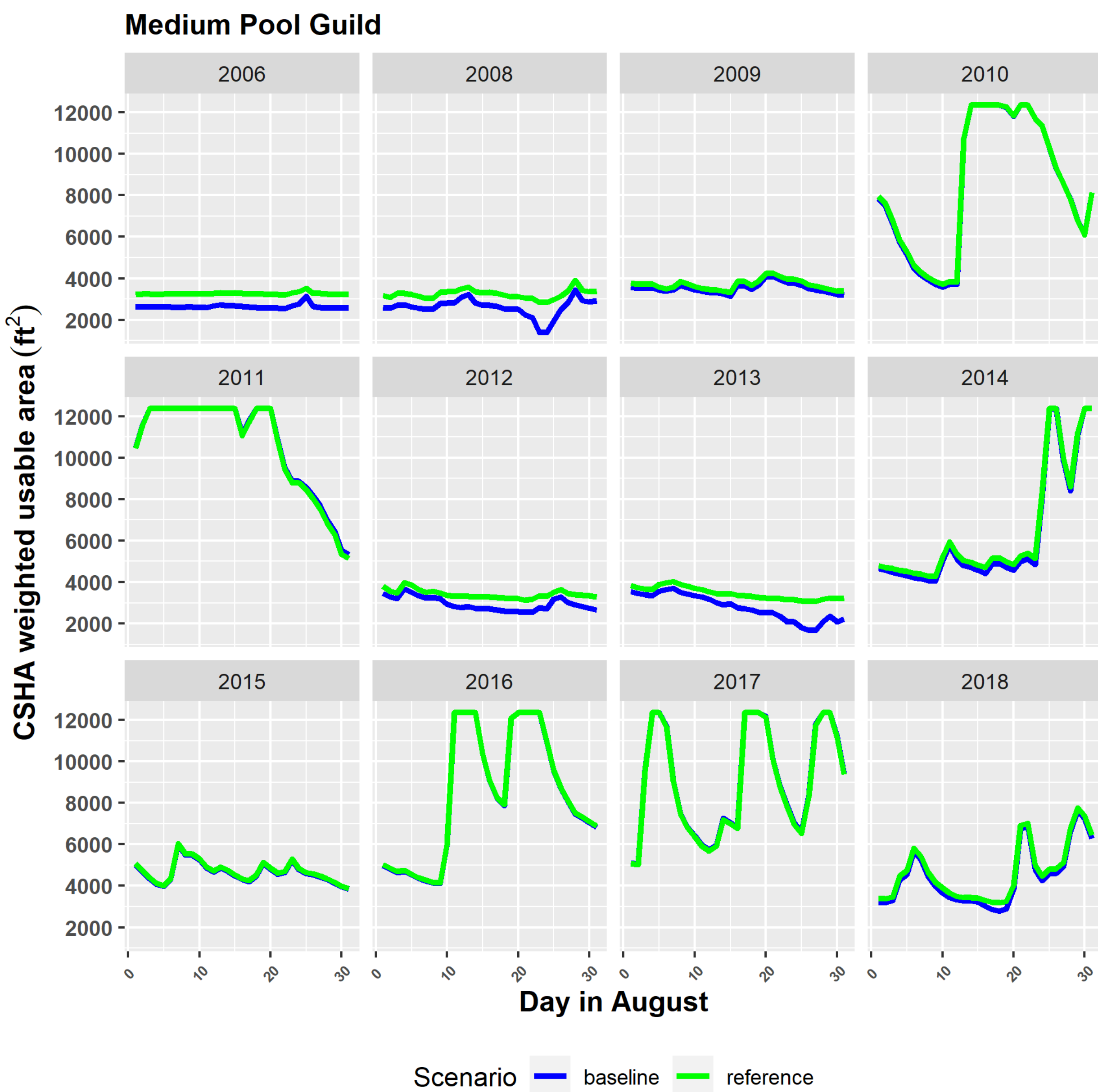
Shallow Pool Guild

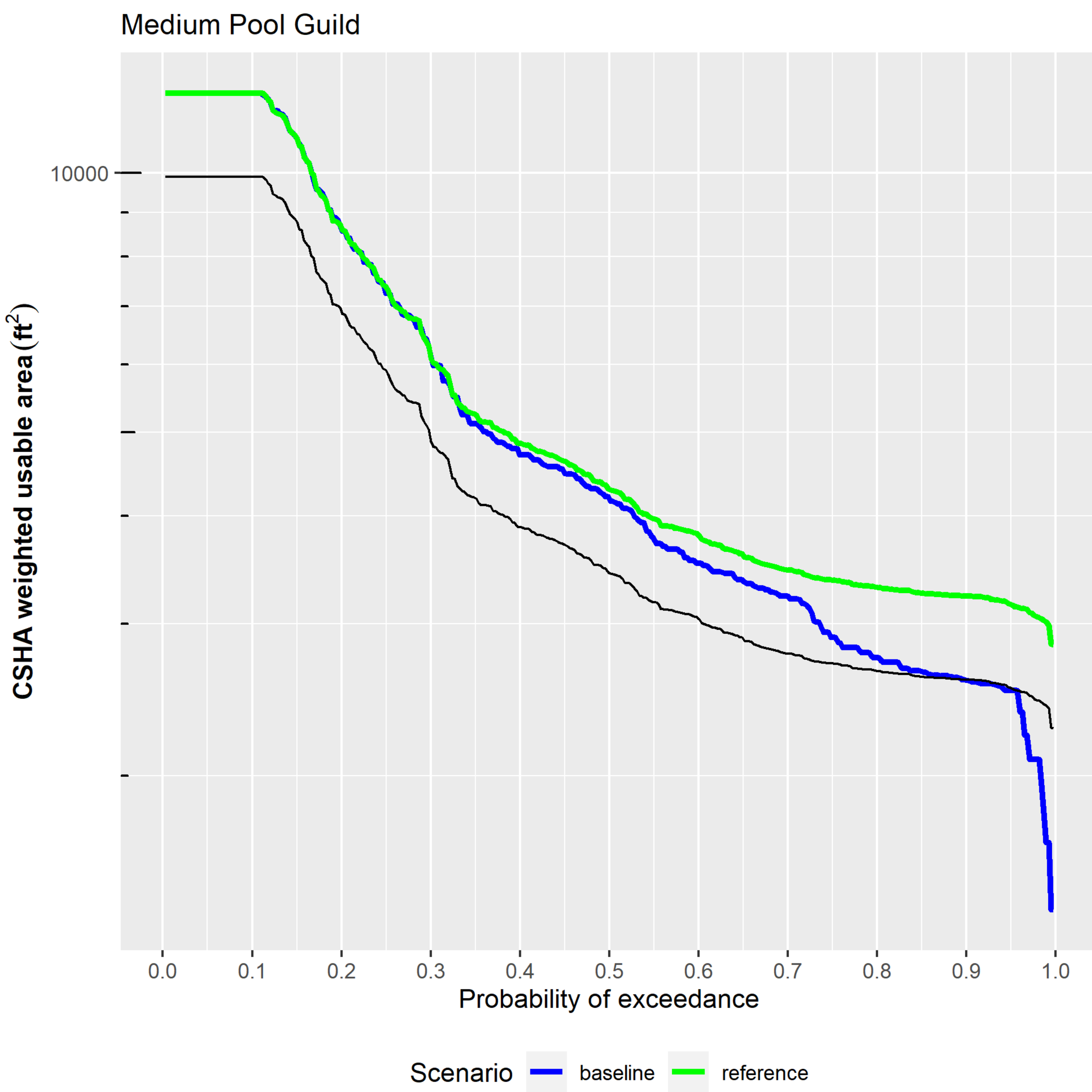


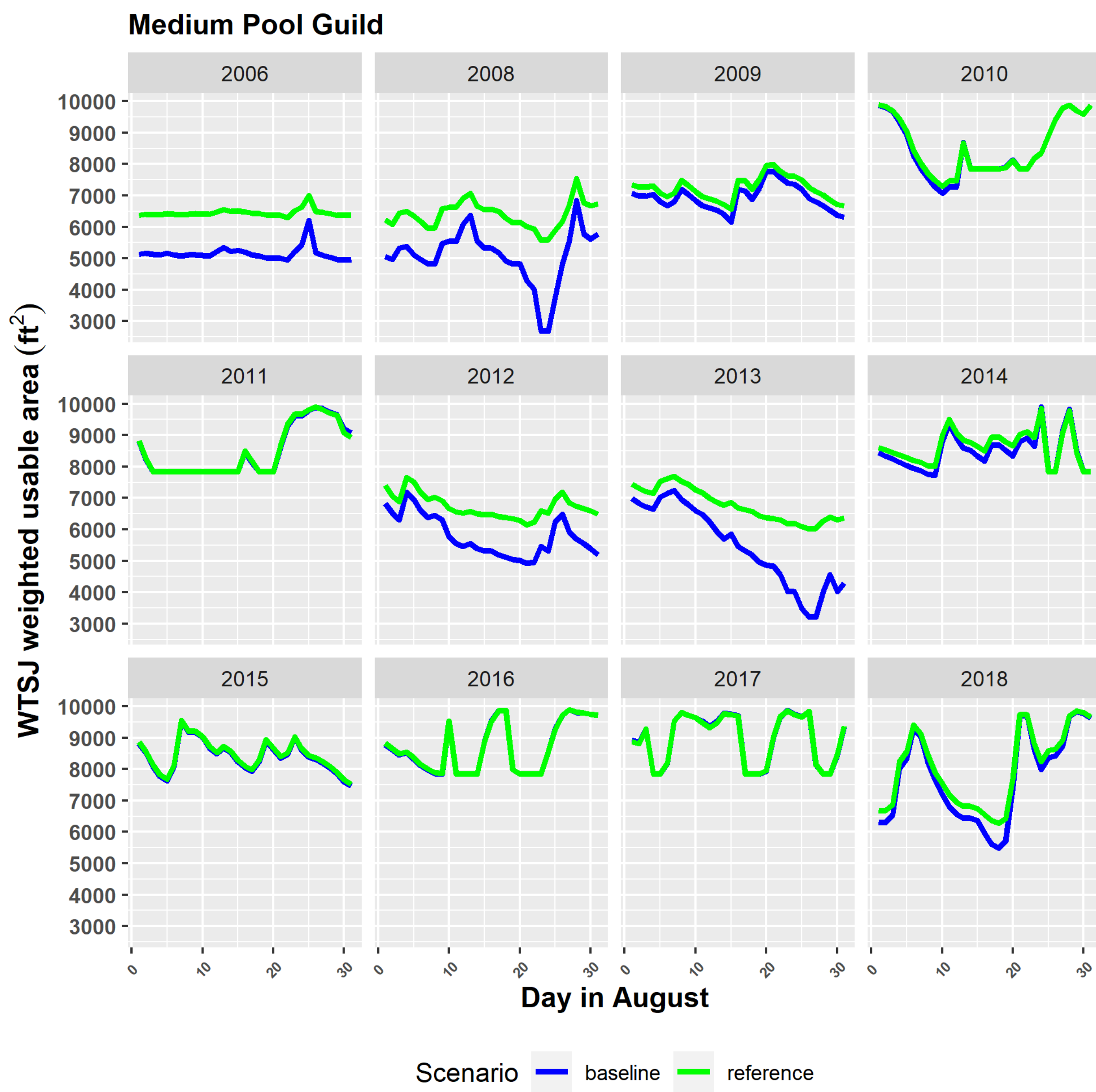

Shallow Pool Guild
2006
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
2018
12000
10000
8000
6000
0
10
20
30
JNDY weighted usable area (ft$^2$)
Day in August
Scenario
baseline
reference

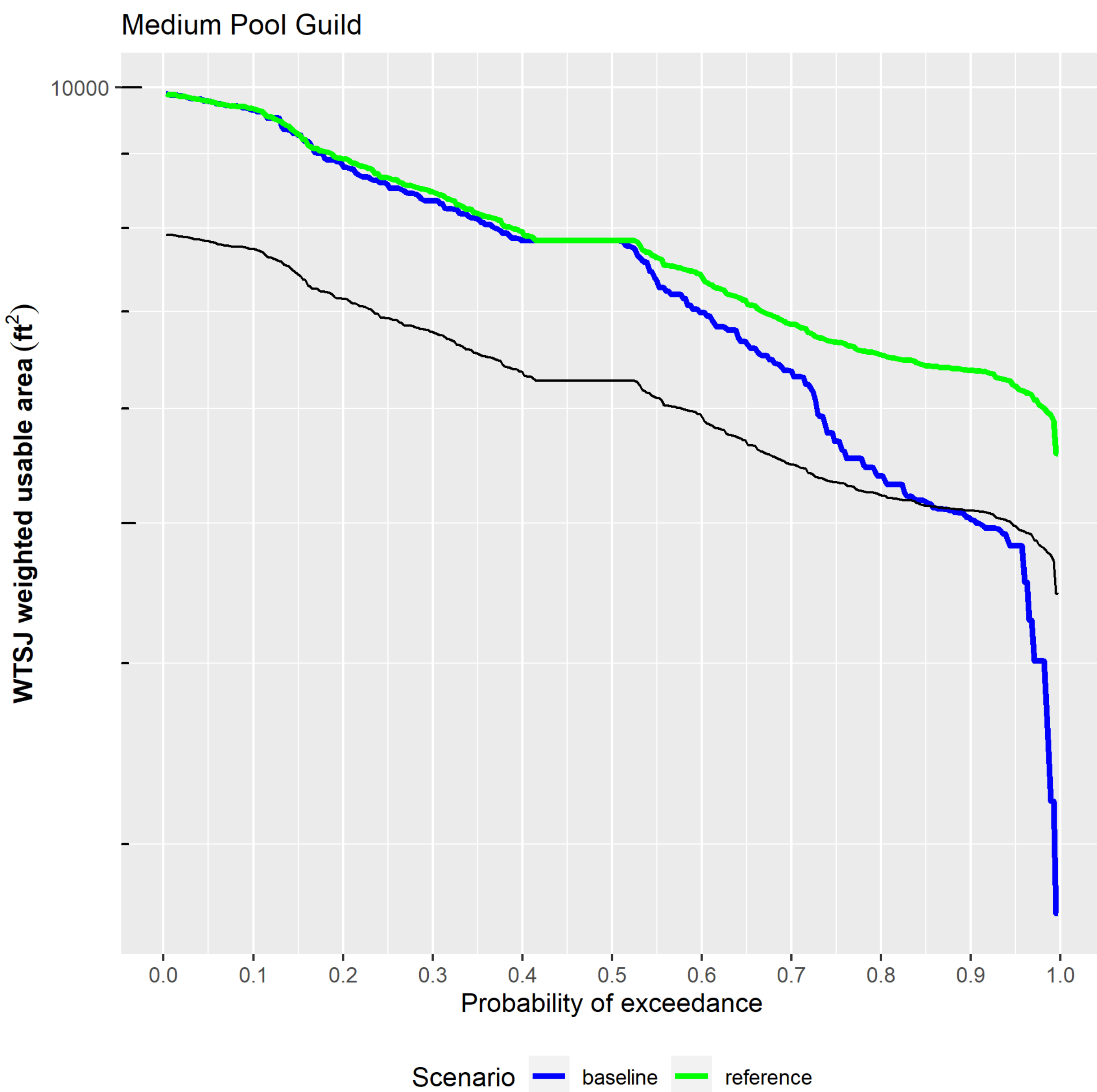


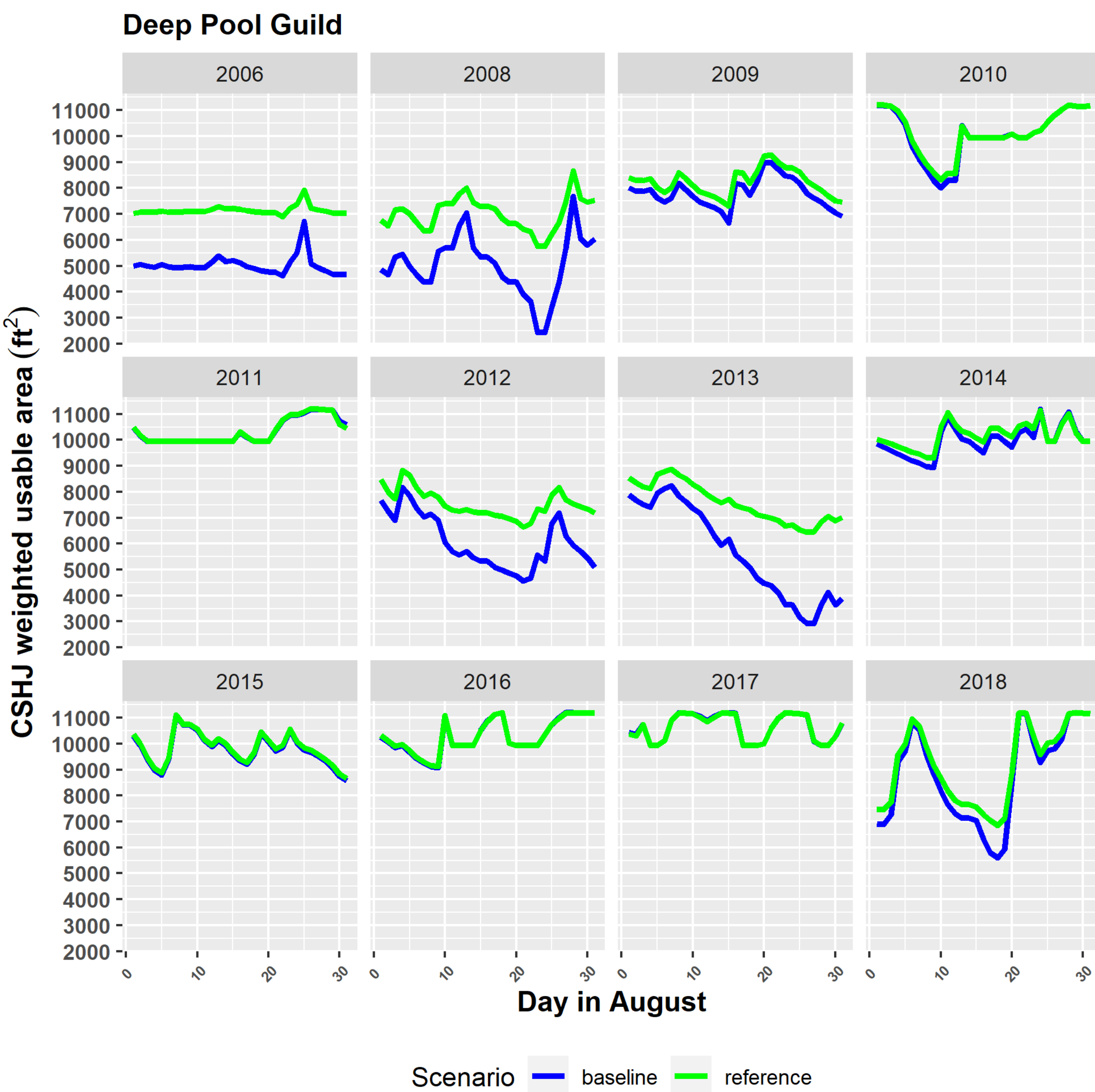
Shallow Pool Guild
JNDY weighted usable area (ft²)
10000
0.0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0.9
1.0
Probability of exceedance
Scenario
baseline
reference

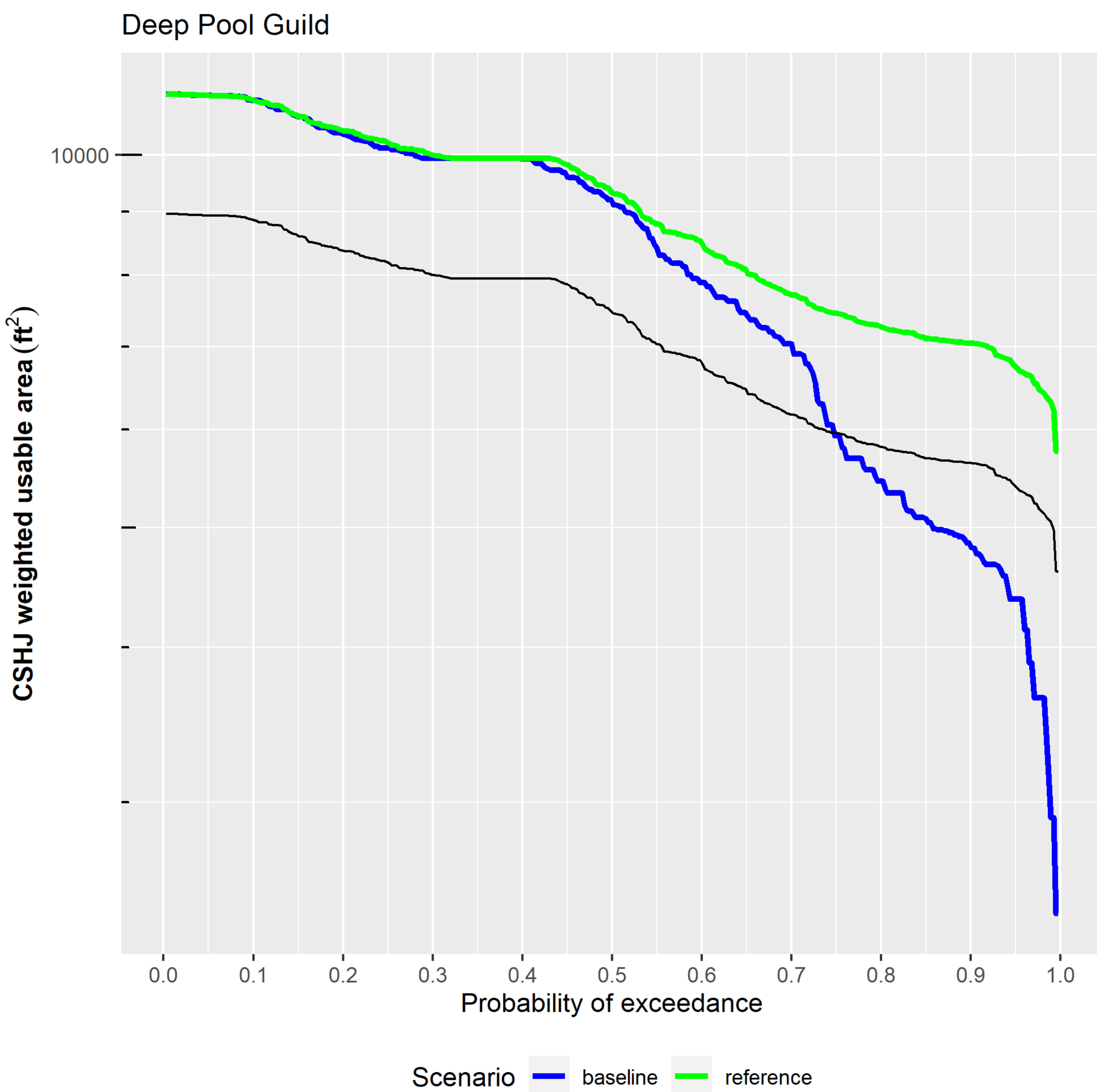

Medium Pool Guild
2006
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
2018
12000
10000
8000
6000
4000
2000
0
10
20
30
CSHA weighted usable area (ft2)
Day in August
Scenario
baseline
reference


Medium Pool Guild
CSHA weighted usable area (ft$^2$)
10000
0.0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0.9
1.0
Probability of exceedance
Scenario
baseline
reference


Medium Pool Guild
2006
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
2018
WTSJ weighted usable area (ft2)
10000
9000
8000
7000
6000
5000
4000
3000
0
10
20
30
Day in August
Scenario
baseline
reference


Medium Pool Guild
10000
WTSJ weighted usable area (ft2)
0.0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0.9
1.0
Probability of exceedance
Scenario
baseline
reference


Deep Pool Guild
2006
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
2018
CSHJ weighted usable area (ft²)
11000
10000
9000
8000
7000
6000
5000
4000
3000
2000
0
10
20
30
Day in August
Scenario
baseline
reference



Deep Pool Guild
10000
CSHJ weighted usable area (ft²)
0.0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0.9
1.0
Probability of exceedance
Scenario
baseline
reference