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Review papers

Advancing environmental flows approaches to streamflow depletion management

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ABSTRACT

Groundwater use can reduce streamflow by reducing groundwater flows into streams and/or increasing losses from the stream into the aquifer ('streamflow depletion'). Streamflow depletion can impact aquatic ecosystems through changes in the availability and temperature of surface water. Regions with a combination of groundwater withdrawals and groundwater-dependent resources therefore require management strategies that respond to the needs of both humans and aquatic ecosystems. Here, we review and evaluate opportunities and challenges for applying an environmental flows approach to streamflow depletion management based on functional flows and the Ecological Limits of Hydrological Alteration (ELOHA) frameworks. We highlight the need for explicit recognition of temperature in streamflow depletion science, especially given the realities of climate change. Using a demonstrative analysis on Wisconsin streams, we show that both the magnitude and variability of streamflow and stream temperatures are likely to be impacted by groundwater withdrawal, with particular impacts on low flows during the baseflow period. Then, we evaluate potential challenges to integrating existing groundwater withdrawal management and environmental flows approaches and provide a pathway to address inherent tensions between these two frameworks. In particular, we find that uncertainty associated with the first two ELOHA steps (setting a baseline and classifying streams) can lead to substantially different estimates of ecological impacts in streamflow depletion contexts. Navigating these tensions requires stakeholder engagement throughout the process of setting acceptable management thresholds to move towards practical, management-focused integration of environmental flows and streamflow depletion science.

1. Introduction

Groundwater withdrawals have increased substantially since the mid-20th century worldwide, with accelerated growth since 1990 (Wada et al., 2014). Increasing groundwater withdrawals can be detrimental both to human users, when over-use leads to groundwater depletion (Wada et al., 2014), and rivers and streams that rely on discharge from groundwater sources (e.g., (Perkin et al., 2017)). *Streamflow depletion* occurs when groundwater withdrawals capture water that would otherwise discharge to a stream or when altered groundwater flow pathways enhance or induce infiltration from the stream into the streambed (Barlow and Leake, 2012). Most pumped groundwater is captured from streamflow as streamflow depletion (Konikow and Leake, 2014; Gleeson and Richter, 2018), and therefore represents a pervasive challenge even in settings where groundwater levels are relative stable. See Barlow and Leake (2012) for a detailed introduction to streamflow depletion. Streamflow depletion is already a significant global problem. de Graaf et al. (2019) found that streamflow in 15–20% of watersheds with groundwater withdrawals has already dropped below the flow needs of aquatic ecosystems (environmental flows), and up to 80% of watersheds will fail to maintain environmental flows by 2050 under a business-as-usual development and climate change scenario. The impacts of streamflow depletion extend beyond ecosystem collapse (Perkin et al., 2017) to impact surface water availability for human users as well.

Given these challenges, effective integrated management of interconnected surface water and groundwater resources is essential for maintaining water availability for human and ecosystem needs moving

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into the future. Due to potentially long time lags between groundwater withdrawals and measurable impacts (including both streamflow depletion and water quality changes; (Currell, 2016; Currell et al., 2012))–especially when wells are further from streams or buffered from streams by low-transmissivity aquifer materials (Barlow and Leake, 2012)–and the challenges in removing water rights once they have been allocated, groundwater management is a long-term process that requires a multi-generational perspective (Gleeson et al., 2012). However, managing surface water-groundwater interactions is particularly challenging because it is difficult to attribute changes in streamflow to specific stressors without detailed modeling or analysis. Beyond pumping, changes in streamflow can also be driven by climate variability, changes in groundwater levels, climate change, land use change, and other human actions, with impacts that vary regionally and locally (Carlisle et al., 2019; Cuthbert et al., 2019; Craig et al., 2017). If these uncertainties are built into a modeling framework, scientists and water managers can still robustly make predictions of potential future streamflow depletion impacts and develop appropriate management plans (Doherty and Moore, 2020). However, more commonly, complex uncertainties result in groundwater management goals that often lack explicit, quantifiable connections to the ecological communities they are meant to protect (Saito et al., 2021).

Environmental flows (Poff and Zimmerman, 2010) are one approach to setting science-informed water management goals by quantifying the hydrological regime necessary to support aquatic ecosystems. Recognizing that in-stream habitat requires more than a simple minimum level of flow (e.g., (Poff et al., 1997; Bunn and Arthington, 2002)), environmental flows incorporate a more comprehensive view of the magnitude, timing, variability, and quality of streamflow (Poff and Zimmerman, 2010). Given that groundwater level-based management methods like drawdown triggers often fail to capture ecologically relevant changes in streamflow (Currell, 2016), it is important to invest in improving and adapting environmental flows approaches. There are many approaches to estimate environmental flows. For example, the functional flows framework (Yarnell et al., 2020) relates qualitative aspects of a hydrograph that have important effects on ecological communities (i.e., functional flow components) to a quantifiable set of ecologically important flow characteristics that can be used for management. Once quantitative flow characteristics are set, the Ecological Limits of Hydrological Alteration (ELOHA) framework (Poff and Zimmerman, 2010) provides a process for decision-making on ecohydrological resources. The ELOHA framework consists of four main technical steps: (1) build a hydrological foundation for analysis, (2) classify streams, (3) quantify hydrological alteration, and (4) relate hydrological alteration to ecological impacts (Poff and Zimmerman, 2010). Though the ELOHA framework has been applied with success (e.g., (Kendy et al., 2012)), integrating streamflow depletion remains a stubborn challenge and few existing groundwater management plans include quantifiable streamflow depletion targets or thresholds (e.g., jurisdictions in the European Union, India, and most jurisdictions in the US lack quantifiable targets; (Gage and Milman, 2021; Gleeson and Richter, 2018; Kallis and Butler, 2001; Srinivasan and Kulkarni, 2014; Harsha, 2016)).

In this review, we focus on the specific management challenge of designing a decision process for permitting groundwater withdrawals in a manner that adequately accounts for potential impacts on aquatic ecosystems and respects environmental flows. To provide a scientific foundation for addressing this challenge, this synthesis paper focuses on the following objectives:

1. Discuss how existing environmental flows approaches (such as using functional flows in the ELOHA approach) can provide an ecological framework for streamflow depletion management (Section 2).
2. Review and identify challenges in integrating environmental flows into groundwater management (Section 3).

This review is targeted primarily towards the groundwater and ecohydrological research communities. While we primarily focus on management examples in the United States of America, the social and technical challenges highlighted in this review are relevant for streamflow depletion management globally. The intent of this review is to provide a pathway for overcoming existing scientific limitations at the streamflow depletion-ecology nexus to guide future development of decision support systems for streamflow depletion management that explicitly account for potential impacts on aquatic ecosystems. In particular, we emphasize a need for more research on detecting and predicting hydrological changes from groundwater withdrawal beyond flow reduction (i.e., changes in temperature, variability, and timing) and the associated ecological impacts on ecosystems, including impacts on macroinvertebrates and fish. Such research is essential for providing reliable guidance on management thresholds.

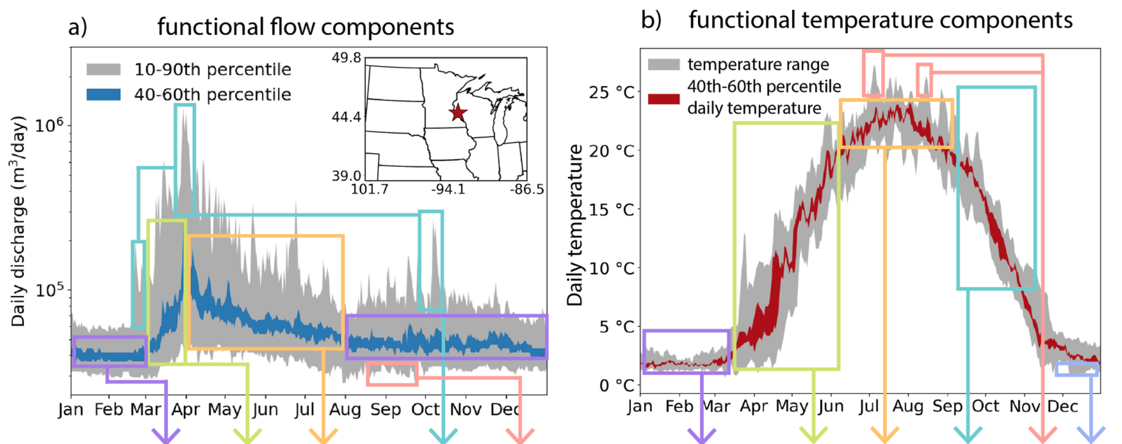
2. Ecological impacts of streamflow depletion

It is well-recognized that groundwater withdrawal can result in streamflow depletion (e.g., (Barlow and Leake, 2012)), demonstrated by the strong focus on the impacts of pumping on flow magnitudes in both management (e.g., (Hamilton and Seelbach, 2011; Diebel et al., 2015)) and scientific contexts (e.g., (Zipper et al., 2019; Wen and Chen, 2006; Burt et al., 2002)). However, the choice of the specific hydrological targets used for management can have a large impact on allowable pumping limits (Granato and Barlow, 2005). In this section, we use a functional flows framework (Yarnell et al., 2020) and introduce a complementary 'functional temperature' framework to identify ecologically important aspects of a stream hydrograph (functional flow components) and thermograph (functional temperature components), using the Upper Midwest as an example (Section 2.1). We then identify which functional components are likely to be impacted by groundwater withdrawal (Section 2.2). Finally, we link these functional flows and temperatures components to water management using the ELOHA framework (Section 2.3).

2.1. *A functional flows and temperatures framework for the Upper Midwest*

Growing awareness of the importance of the entire streamflow regime (beyond just mean streamflow) to aquatic ecosystems and communities (e.g., (Poff and Zimmerman, 2010; Nuhfer et al., 2017)) has led to efforts to quantitatively classify the flow regime in both the scientific (e.g., (Yarnell et al., 2020)) and management (e.g., Susquehanna River Basin; (DePhilip and Moberg, 2010)) communities. The functional flows framework starts with a qualitative description of ecologically relevant aspects of the flow regime (such as baseflow, recession, peak flow, or fall pulse) called functional components (Yarnell et al., 2020). Each component can then be broken down into flow characteristics (magnitude, timing, duration, frequency, rate of change), each of which is measured using a hydrological signature, which is a metric derived from the hydrograph that quantify different aspects of hydrological behavior (McMillan, 2020). The outcome of the functional flows framework is a set of hydrological signatures that describe the ecologically relevant aspects of the flow regime. While functional flows focus on the streamflow hydrograph, water temperature is also essential to high-quality habitat (Olden and Naiman, 2010). Here, we adapt the functional flows framework to the streamflow thermograph to demonstrate functional temperatures as a potential tool to understand the ecological impacts of pumping-induced water temperature changes relative to the natural thermal regime, a description of a thermograph under reference conditions (Arismendi et al., 2013).

For each component, the characteristics we use in this study are: magnitude, timing, duration, frequency, rate of change, and variability (Poff et al., 1997). *Magnitude* refers to the value of flow/temperature. *Timing*, *duration*, and *frequency* describe when certain flow/temperature magnitudes occur, how long they last, and how often they occur



	functional flow components				
flow characteristics	baseflow	freshet	summer recession	peak flow	low flow
magnitude	x	x	x	x	x
timing	x	x	x	x	x
duration				x	x
frequency			x	x	x
rate of change	x	x	x		
variability	x	x	x		

	functional temperature components					
temperature characteristics	winter average	spring warming	summer heat	fall cooling	peak temp.	low temp.
magnitude	x	x	x	x	x	x
timing	x				x	x
duration			x	x	x	x
frequency	x				x	x
rate of change			x	x		
variability	x		x		x	

Fig. 1. (a) Functional flow and (b) temperature components for an example stream (Eau Galle River) in Wisconsin. Inset map in panel (a) shows location of the river in the Upper Midwest USA with a red star. The colored boxes identify different functional flow/functional temperature components of the hydrograph/thermograph, which are summarized in the tables below each plot. Each flow/temperature component can then be quantitatively described in terms of the characteristics of magnitude, timing, duration, frequency, rate of change, and variability. 'X' marks in the table indicate functional flow characteristics with potential ecological importance in this region (see A).

throughout a year or season, respectively. It is common to use only "rate of change" or "variability" as a functional characteristic (e.g., (Poff and Zimmerman, 2010)), but we argue that these are distinct characteristics with diverse potential impacts on aquatic ecosystems. Therefore, we separately define *variability* as the overall range of flow or temperature and the coefficient of variation for different time periods (e.g., (Arismendi et al., 2013; Xu et al., 2010)), whereas the *rate of change* describes the overall change per unit time over a given time period, i.e., rate of decrease in flow or temperature over the recession period (e.g., (Nicola et al., 2009; Cattanéo et al., 2002)).

Yarnell et al. (2020) demonstrate functional flows for a mixed rain-snowmelt system common in California. In Fig. 1, we adapt this approach to demonstrate how functional flows and functional temperatures may appear for a snowmelt and groundwater-dominated system (Eau Galle River, WI, location marked in Fig. 1a) that is representative of many streams in mid- to high-latitudes around the world. The particularities of the hydrograph shape may vary substantially among regions, but this shape is common among all 34 streamflow stations in Wisconsin examined in this study (for plots of all annual hydrograph shapes, see the data supplement: (Lapides et al., 2021)). The annual shape of the hydrograph in the top left shows a baseflow period throughout most of the year with a pronounced spring freshet and summer recession. Peak flows occur throughout the year following snowmelt or rain events, while extreme low flows occur during the baseflow period at the end of summer and beginning of fall.

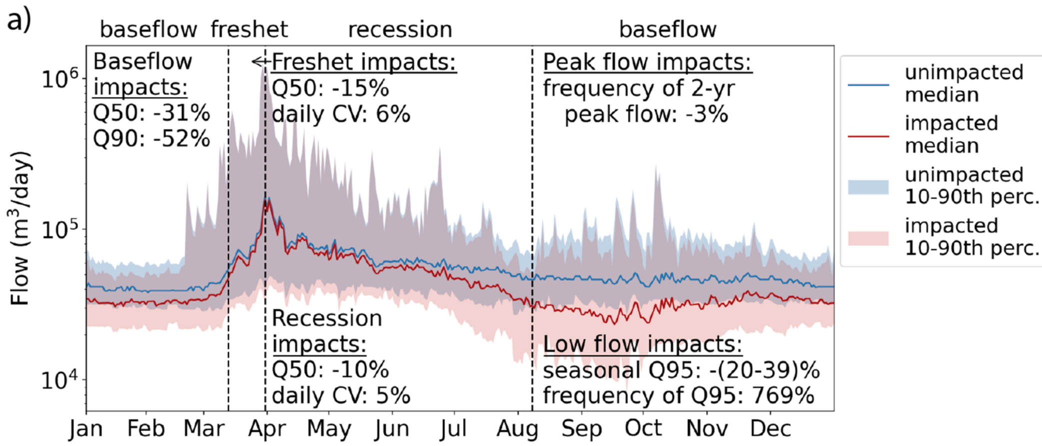
Because climate in the Midwest differs substantially from that of California, we performed a literature review on the seasonal importance of streamflow for aquatic organisms in this region (i.e., autumn, winter, spring, and summer—which roughly correspond to the hydroperiods in Fig. 1). Where possible, ecological literature from the U.S. Upper Midwest was used in this review, but literature from other mid- to high-latitude studies with similar hydrograph functions were used to expand the scope of information where studies were not available in the Upper Midwest. At the bottom left of Fig. 1, we identify functional characteristics associated with each functional flow component with an 'x' based on our literature review and analogy to Yarnell et al. (2020). See Appendix A for a detailed summary of the regional functional flows review.

We also performed a literature review on the importance of stream temperature during autumn, winter, spring and summer for aquatic organisms in the Upper Midwest. Olden and Naiman (2010) describe a 'natural thermal regime' based on factors important for stream organisms, which they use to explore the impact of dams on thermal stream characteristics, and Poff (2018) extend the concept to consider a flexible thermal regime that supports aquatic ecosystems. Here, we build on this in our development of a functional temperatures description of Upper Midwest streams. In the right half of Fig. 1, we use temperature data from the USGS (United States Geologic Survey, 2021) to illustrate functional temperature components for the Eau Galle River in Wisconsin. Stream temperature tracks seasonal temperature changes so that streams are typically around freezing during the winter months and 20–25°C warmer during the summer. Each functional temperature component identified as important for stream communities in the Upper Midwest is marked with an 'x' in Fig. 1. See Appendix B for a detailed summary of the regional functional temperatures review.

2.2. Hydrological signatures sensitive to groundwater withdrawal

The functional flows/temperatures approach outlines all of the aspects of the hydrograph/thermograph that are important for aquatic ecosystems, but not all of the functional characteristics or components may be equally impacted by streamflow depletion. Since it can be difficult to obtain all of the prerequisite data to calculate an entire suite of signatures, we seek to identify a smaller set of signatures that capture the ecologically relevant aspects of stream conditions that are most sensitive to groundwater withdrawal.

To identify which flow components are sensitive to groundwater withdrawal, we selected a set of hydrological signatures that summarize



b)

	functional flow components				
flow characteristics	baseflow	freshet	summer recession	peak flow	low flow
magnitude	x	x	x	x	x
timing	x	x	x	x	x
duration				x	x
frequency			x	x	x
rate of change	x	x	x		
variability	x	x	x		

x more than 10% median impact from groundwater pumping
x more than 10% interquartile impact from groundwater pumping
x expect impact from groundwater pumping

Fig. 2. (a) Median annual hydrograph for the Eau Galle River, WI (blue) with pumping impacts (red) from a seasonal pumping schedule simulated using streamDepletr (Zipper, 2020). All impacts are given as a percent difference. (b) A summary of hydrograph impacts across all sites for each flow characteristic identified as having ecohydrological relevance. See A for the complete list of signatures calculated and how they relate to ecosystem services. Boxes indicate whether any single signature that represents the flow characteristic (red) shows more than a 10% median impact across all 34 sites, (gold) shows more than a 10% median impact at the 25th or 75th percentile among all sites or (grey dashed) would be expected to be impacted but did not show large enough impacts to get a .red or gold box.

the findings of the literature review (see Tables A.1 and B.2) and applied these signatures to hydrographs from 34 Wisconsin streams with long-term USGS streamflow records (United States Geologic Survey, 2021) and mean annual streamflow below 1.13 m^3/s (40 ft^3/s). Limiting this analysis to smaller streams reduces the importance of precipitation-driven events in the hydrographs, allowing the results to apply more specifically to groundwater-dominated stream systems. We simulated altered hydrographs by assuming one well for each stream at a distance of 1,000 m using the Glover and Balmer analytical solution (Glover and Balmer, 1954) implemented in streamDepletr (Zipper, 2020). For simplicity, the aquifer properties are held constant across all sites with hydraulic conductivity of $K = 10$ m/d, aquifer thickness of $b = 100$ m, and specific yield of $S_y = 0.2$, which is typical for an unconfined, unconsolidated sedimentary aquifer (Freeze and Cherry, 1979). Since pumping is often seasonal (i.e., for irrigation), the well is pumped from June 1 to September 1 at a rate of 1.25 times the median baseflow, implemented via the analytical superposition technique of Jenkins (1968). Since it is important to examine the impacts of groundwater pumping on streamflow over multiple years (Bradbury et al., 2017), pumping was simulated throughout the duration of each stream gauge record.

Associated thermographs for the original and altered hydrographs were calculated using a simple groundwater-surface water end-member mixing model. Groundwater is assumed to have a constant temperature year-round of 9.9°C, which is the median of mean annual temperatures from all Wisconsin streams with temperature data. Surface flow temperatures were simulated using air temperature in Madison, WI (NationalCenters, 2021). Temperatures were calculated daily, so sub-daily signatures are not calculated in this study, although temperature changes at sub-daily temporal resolution could be impacted by groundwater withdrawal and have ecological importance. We adopted this simplified approach as an illustrative exploration of the flow and temperature components most sensitive to streamflow depletion rather than a detailed site-specific characterization of actual impacts, which is beyond the scope of this review paper but an important future research direction.

As an example, this analysis is shown for the Eau Galle River, WI in Fig. 2. This site was chosen because it has particularly well-defined hydrograph seasonality. Fig. 2a shows the original hydrograph (blue) and the calculated altered hydrograph (red) with selected change statistics for the Eau Galle River. Across all studied sites, streamflow depletion leads to a substantial decrease in baseflow that extends beyond the summer pumping season, with decreases up to 38% of median unimpacted baseflows and an increase in variability, as shown by the CV statistics. While less pronounced than during the baseflow period, flows are reduced by streamflow depletion during freshet and recession periods, flows at all times of year exhibit a greater variability when considering streamflow depletion impacts (larger interquartile

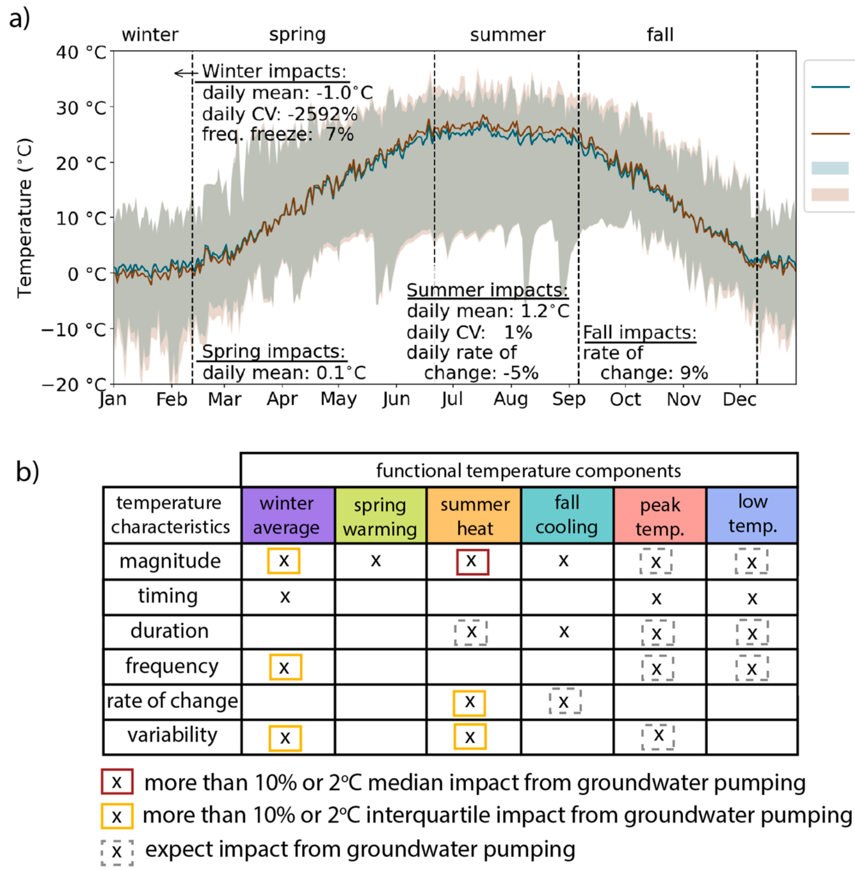


Fig. 3. (a) Median annual thermograph for the Eau Galle River, WI (teal) with pumping impacts (brown) from an end-member mixing model and (b) a summary of thermograph impacts across all sites. All impacts are measured as a percent difference except temperature magnitude changes, all of which are measured in absolute difference in °C. See B for a review of temperature characteristics and a full list of signatures calculated. Boxes indicate whether any single signature that represents the flow characteristic (red) shows more than a 10% median impact across all 34 sites, (gold) shows more than a 10% median impact at the 25th or 75th percentile among all sites or (grey dashed) would be expected to be impacted but did not show large enough impacts to get a .red or gold box.

range), and (since baseflows are lower) there is a more rapid recession following the freshet.

We found that the strongest impacts from groundwater withdrawal on Wisconsin streams are on the magnitude characteristics, frequency of extreme flows, and flow variability (Fig. 2b). These results are largely unsurprising since the most apparent and recognized impact from groundwater withdrawal on a hydrograph is to reduce streamflow magnitude (Barlow and Leake, 2012). By reducing magnitude, variability (as measured by coefficient of variation, the ratio between standard deviation and mean) during different components of the hydrograph also increases since the range of flows does not change significantly, but the mean drops. The frequency of peak flows and low flows can be impacted by a decrease in magnitude since the number of flows throughout the year that exceed a given high flow decreases when all flow magnitudes decrease–this can even be the case for an *n*-yr return interval flow. The same is true for low flows, where flow more frequently drops below a low-flow threshold when overall hydrograph magnitude drops due to streamflow depletion. We hypothesize that the duration of the longest low-flow period could be longer and the timing change for the same reason.

All the streamflow characteristics with observed impacts from groundwater withdrawal in our simple analysis are impacted primarily by a decrease in flow magnitude. Lacking data necessary to define the ecological response to each of these flow characteristics, we used a threshold of 10% for demonstrative purposes, as this large of a change could have impacts on ecological function. Since this analysis is limited by design, we also mark with a grey dashed box the characteristics that we expect could be impacted by groundwater withdrawal, although we did not observe large impacts in our simple analysis. However, different or more complex groundwater withdrawal schedules or different aquifer properties could result in more important transient effects on hydrographs, resulting in potentially earlier onset of the baseflow season (baseflow timing), which could impact the timing and duration of low flows as well. The strong impact of streamflow depletion on low flows is particularly problematic, because pumping requirements (i.e., irrigation) are likely to be greatest during dry periods when low flows are naturally already at their lowest.

Fig. 3a shows the estimated unimpacted (teal) and altered (brown) thermographs for the Eau Galle River, WI. Decreased groundwater contributions caused by streamflow depletion led to warmer summer temperatures (up to 3.9°C increase in mean daily across all sites) in the summer and slightly colder temperatures in the winter (up to 1.8°C in mean daily across all sites) than the unimpacted thermograph.

Fig. 3b summarizes temperature impacts across all 34 Wisconsin streams. For temperature magnitude signatures, a 2°C change is used as a threshold for impacts rather than a 10% change, to better reflect the influence of temperature on fish bioenergetics (Hartman and Cox, 2008; Hayes et al., 2000). The largest impacts from pumping on the thermograph are on winter and summer temperatures. Decreases in flow magnitude increase the annual temperature range, leading to hotter summer temperatures and colder winter temperatures. Ultimately, stream temperatures are more responsive to air temperature as groundwater inputs to the stream decrease, resulting in more temperature variability, faster rates of change in temperature, and more extreme temperatures. For these reasons, we also expect that peak temperatures and low temperatures could be more impacted by flow reduction, including the duration of heat during the summer. We tended to observe the lowest thermal impacts during the summer and fall when air temperature is closest to the groundwater temperature and therefore the relative contributions of groundwater and surface water do not have a substantial thermal impact, though these two water sources may still have ecologically important chemical distinctions.

As with the functional flows analysis, this was meant to be an illustrative exploration of potential temperature sensitivity to streamflow depletion as part of this review paper rather than a detailed site-specific characterization, so there are several potential limitations to the thermal analysis. In reality, surface water temperature is likely to be lower than air temperature, and groundwater has a muted seasonal temperature signal, so the real difference in temperature between surface water and groundwater is likely to be smaller than we estimated using our end-member mixing model. This model also assumes a nearby direct contribution of groundwater to streamflow by giving the baseflow the temperature profile of groundwater–a reasonable assumption in gaining streams but not for losing streams. Losing streams will likely be more responsive to air temperature variation with a relatively small influence of groundwater temperature since water flows from the stream into the groundwater system (Barlow and Leake, 2012), and areas where streamflow depletion has caused streams to shift from gaining to losing may exhibit threshold-type changes in the temperature regime rather than the linear mixing-based changes estimated here. It is difficult to identify over broad areas which streams are gaining or losing (Jasechko et al., 2021), but these assumptions indicate that our analysis should be considered a baseline understanding and could be used to prioritize future research on affected aspects of the thermograph. A full description of the methods used for the analyses in this section can be found in C. The full set of hydrological signatures can be found in A, and the full set of temperature signatures can be found in B.

2.3. Using ELOHA to relate hydrological signatures to ecological impacts

The previous section illustrates how the functional flows framework can be used to identify a set of temperature and hydrological signatures that describe attributes of ecological importance that are sensitive to groundwater withdrawal. The Ecological Limits of Hydrological Alteration (ELOHA) framework (Poff and Zimmerman, 2010) describes one method for designing a management procedure to respect environmental flow needs as described by these signatures. The ELOHA framework assumes that as flow becomes more altered from a reference condition, impacts on ecosystems increase. The ELOHA framework follows four general steps: (1) Setting a hydrological reference condition from which alteration can be assessed; (2) classifying streams to generalize results and plan monitoring strategies; (3) quantifying hydrological alteration; and (4) developing flow-function curves that relate hydrological alteration to ecological impacts. Based on these steps, a management threshold can be set using the flow-function curves as a guide.

In practice, flow-function curves can be developed for individual species or whole communities (e.g., (Zorn et al., 2012; Armstrong et al., 2011; Diebel et al., 2015; Wilding and Poff, 2008)). These curves are developed by monitoring species presence, population size, and/or community composition and relating change in these ecological indicators to different hydrological signatures. For example, both Wisconsin (Diebel et al., 2015) and Michigan (Zorn et al., 2012) use standardized, long-term monitoring data from stream surveys to generate flow-function relationships between (Wisconsin) fish presence and (Michigan) fish abundance data and flow/temperature signatures. For statewide environmental flows protection in Ohio, (Kendy et al., 2012) related mean daily flow in September to sensitive fish species using quantile regression on data from an existing habitat-flow model. To address statewide water planning in Colorado, Kendy et al. (2012) developed flow-function curves for fish, riparian vegetation, invertebrates, and whitewater kayak/raft conditions using regression or categorical relationships from literature with expert input. And in Rhode Island, Kendy et al. (2012) took advantage of literature from Georgia on flow-fish and flow-invertebrate relationships to examine impacts from changes in 7Q10 (the lowest 7-day flow that has a probability of occurring once every 10 years).

One particularly difficult aspect of using flow-function curves is identifying an appropriate threshold of hydrological change which corresponds to unacceptable ecological impacts. Thresholds can be species-specific or a composite threshold for impacts on all species of interest in a stream, and they depend on the choice of hydrological signature. While it is sometimes visually apparent where ecological condition starts to deteriorate, purely ecological thresholds can be rare (Hillebrand et al., 2020), and a choice of threshold typically constitutes a compromise between needs of stakeholders and ecosystems to identify thresholds and flow standards that are acceptable to all parties (see D for examples of decision processes).

Throughout the remainder of this review, we will use the functional flows and ELOHA frameworks as a starting point to consider opportunities to build on current research and practice to develop more effective management protocols to manage groundwater withdrawals for ecological health, but many of the opportunities and tensions we discuss are broadly generalizable across different environmental flows assessment approaches.

3. Challenges to integrating environmental flows into groundwater withdrawal permitting

The environmental flows framework from the previous section provides a way to evaluate how well existing approaches to groundwater withdrawal management support the needs of aquatic ecosystems. We identified a suite of signatures or types of signatures that are sensitive to groundwater withdrawal, many of which are already embedded in management of groundwater withdrawals. For instance, in Wisconsin (Diebel et al., 2015), Ohio (Kendy et al., 2012), and Michigan (Zorn et al., 2012) statewide management focuses on the mean/median flow for the low-flow month in each state. To capture more of the holistic hydrograph shape, Colorado (Sanderson et al., 2012), Maine (Weinberg, 2013), Massachusetts (Weinberg, 2013), and the Susquehanna River Basin (DePhilip and Moberg, 2010) evaluate hydrological signatures on a monthly or seasonal basis, including mean/median flows in all cases and high/low flows for the Susquehanna River Basin. Flow variability is also evaluated in the Colorado (Sanderson et al., 2012) and Susquehanna River Basin (DePhilip and Moberg, 2010).

In contrast to streamflow data, temperature data are less commonly available, which may possibly explain why temperature is often not included in management processes. Notable exceptions are the Wisconsin (Diebel et al., 2015) and Michigan (Zorn et al., 2012) statewide processes, which only focus on mean water temperature during the hottest low flow month, July. Even by focusing on a small number of signatures, it may be possible to capture most of the major impacts from groundwater withdrawal, though, since hydrograph impacts are primarily attributed to decreasing flow levels, and thermograph impacts are primarily attributable to changes to winter and summer temperatures (see Section 2).

Adherence to environmental flows paradigms moves beyond selecting appropriate signatures. Uncertainty is a general problem in groundwater management, and minimizing uncertainty may be the largest technical challenge in effective management of streamflow depletion (Doherty and Moore, 2020). As described above, the ELOHA framework has four main technical steps: (1) setting a reference condition, (2) stream classification, (3) quantifying alteration, and (4) relating alteration to ecological impacts. Uncertainty from each step propagates through to the final management decision. Thus, it is important to both quantify and minimize the uncertainty in the outcome of each step, thinking of each step as a separate model that needs to be optimized for the specific outcomes required by the next step. To manage for

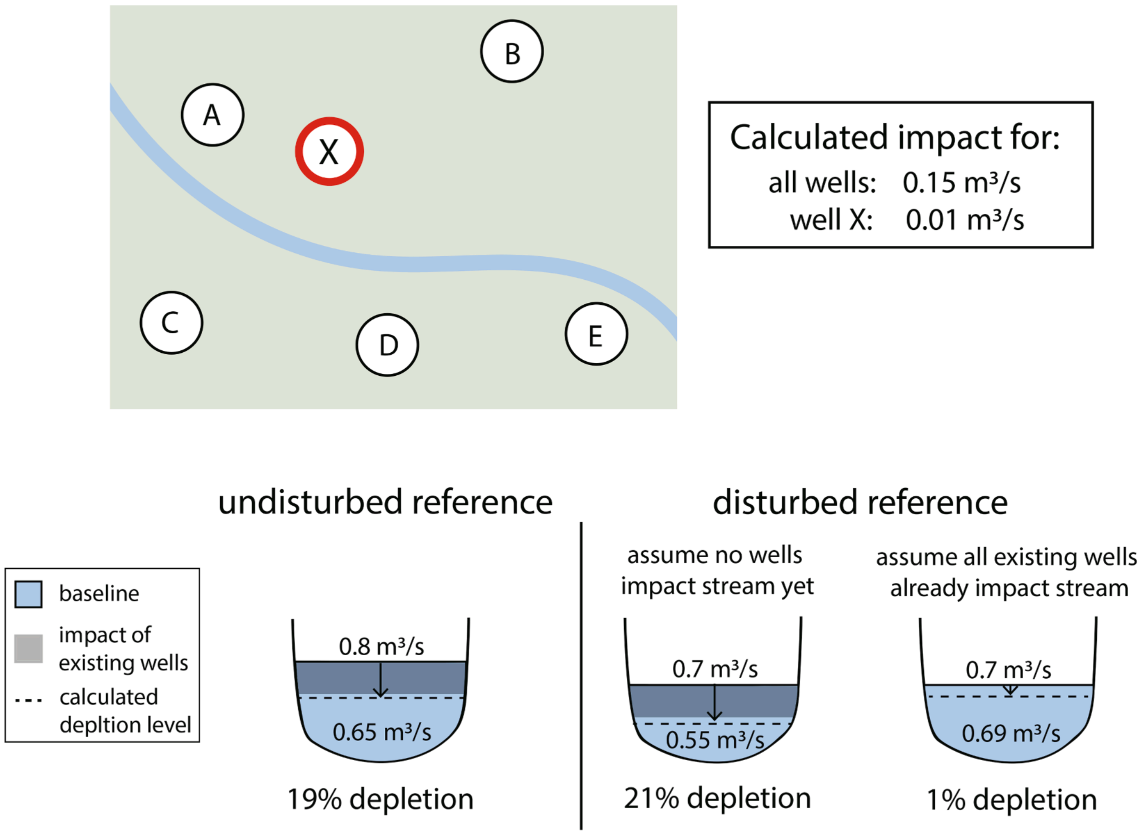


Fig. 4. Choices and assumptions that define a hydrological reference condition have a large impact on streamflow depletion calculations. This demonstrative example shows more than an order of magnitude difference in depletion calculations given different reference conditions: an undisturbed reference and a disturbed reference for which two different assumptions are made about current impacts on the stream.

environmental flows, it is important that the reference conditions (Section 3.1), stream classification (Section 3.2), and quantifying alteration from streamflow depletion (Sections 3.3 and 3.4) used for calculating the signatures are accurately calculated.

3.1. Setting a reference condition

To detect alteration in the ELOHA framework, hydrological signatures are compared under reference and altered conditions. A hydrological reference is meant to characterize undisturbed stream conditions (Poff and Zimmerman, 2010) and therefore is best represented as the historical (pre-disturbance) conditions. In practice, this is not done at large scales because few streams have adequate historical data to characterize a pre-disturbance state. Instead, hydrological reference conditions are commonly set using present-day conditions (e.g., driest month flows in Wisconsin, Ohio, and Michigan statewide processes; (Diebel et al., 2014; Kendy et al., 2012; Zorn et al., 2012)), reference streams (e.g., the Susquehanna River Basin, Massachusetts statewide process; (DePhilip and Moberg, 2010; Archfield and Vogel, 2008)), or by removing modeled impacts on current flows (e.g., Rhode Island streamflow depletion Methodology, Colorado statewide assessment; (Richardson, 2005; Sanderson et al., 2012)). Statistical approaches can then be used to extend streamflow reference estimates from gauged to ungauged sites (e.g., Wisconsin, Ohio, Michigan, Susquehanna River Basin, Massachusetts, Rhode Island, Colorado, California (Diebel et al., 2014; Kendy and Bredehoeft, 2006; Zorn et al., 2012; DePhilip and Moberg, 2010; Archfield and Vogel, 2008; Richardson, 2005; Sanderson et al., 2012; Zimmerman et al., 2018)). To decrease uncertainty in calculating the reference condition, many agencies only calculate specific signatures like the mean flow during the driest month (Michigan, Wisonsin, Ohio; (Diebel et al., 2015; Kendy et al., 2012)) or 7Q10 (Rhode Island, (Richardson, 2005)). While practical due to data limitations, focusing on the mean flow during part of the year necessarily excludes many potentially important functional flow and functional temperature components (Section 2).

The underlying intention of setting the reference condition is to characterize ideal conditions for the stream ecosystem, and it is typically assumed that an 'undisturbed' natural condition is the appropriate reference condition (e.g., (Archfield and Vogel, 2008; Richardson, 2005; Sanderson et al., 2012; DePhilip and Moberg, 2010)). However, the meaning of 'undisturbed' is complicated, given a long history of land management (Wagner et al., 2000) that often predates existing records. Regardless of how the 'undisturbed' condition is defined, stream ecosystems that have adapted to a modified flow regime and/or contain introduced species may be better conserved using a well-chosen reference representing disturbed conditions. For example, brown trout are a socially and politically important species in the Central Sands region of Wisconsin, but they are also an introduced species. In this case, a return to a historical or 'undisturbed' reference condition may actually be detrimental to the current ecosystem, although we are not aware of any significant documented examples of this phenomenon related to groundwater withdrawal. However, regardless of what conditions the current community is adapted to, the true reference condition for aquatic ecosystems may not be captured in available datasets (Pauly, 1995). Thus it may be preferable, or simply the only feasible option, to use a disturbed state for the reference based on the conditions in which species of interest have been observed at a socially acceptable level rather than an 'undisturbed' or historical state.

Ultimately, choosing a hydrological reference condition is a social decision about which aspects of an ecosystem are valuable, and the choice of reference has a significant impact on management outcomes. To demonstrate this impact, we consider an example stream in Fig. 4

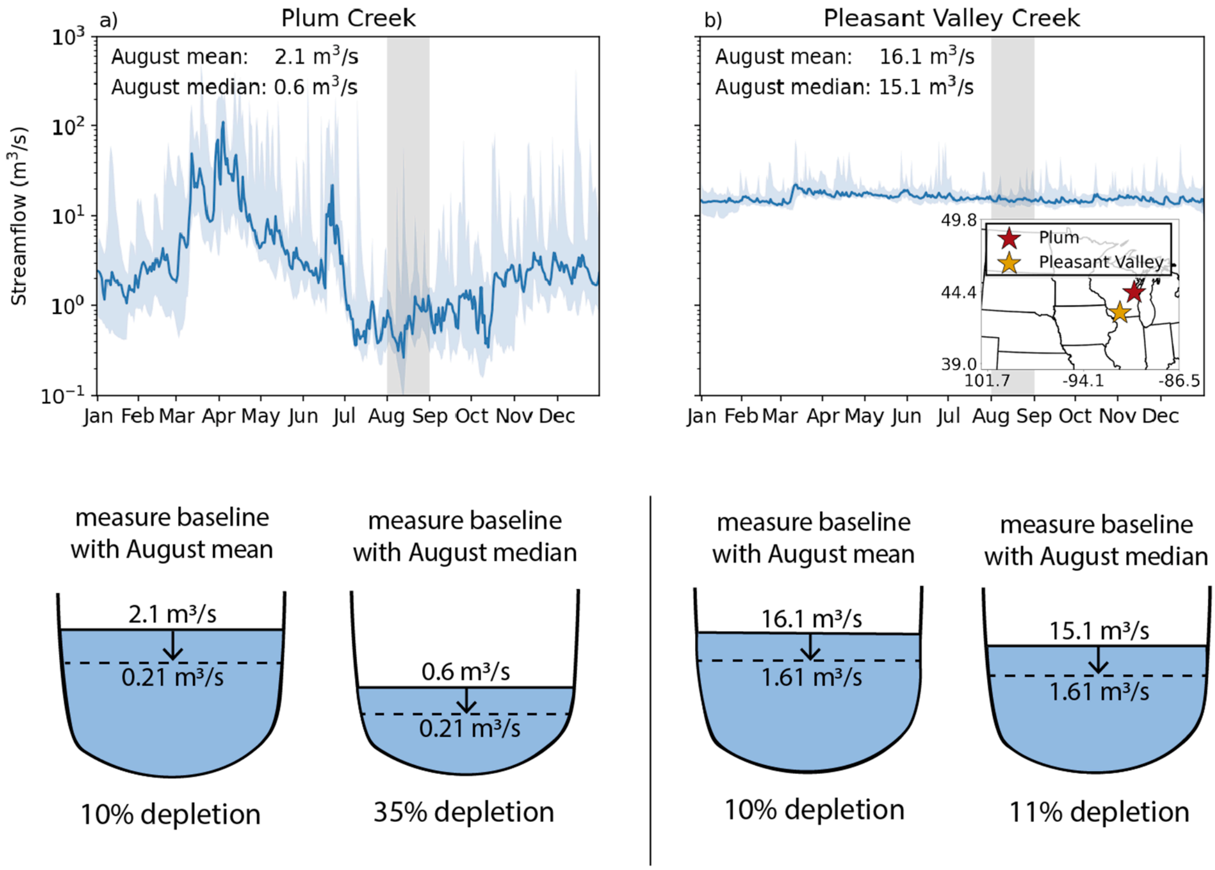


Fig. 5. Plum Creek has a highly variable annual hydrograph (a), whereas Pleasant Valley Creek is very stable (b). In this example, we assume that groundwater pumping at both sites results in a 10% reduction of streamflow from the mean August streamflow. This same streamflow reduction could be measured as a 35% depletion from the August median for Plum Creek or 11% for Pleasant Valley Creek.

with a pre-development flow of 0.8 m^3/s and a current flow of 0.7 m^3/s caused by 0.1 m^3/s of depletion from existing wells. There has been request for groundwater extraction (well X) nearby that will result in streamflow depletion of an additional 0.01 m^3/s. We explore three different scenarios for setting a reference condition: (1) the reference is set at an undisturbed reference condition when no wells impact the stream (0.8 m^3/s); (2) the reference is set under the incorrect assumption that no existing wells impact the stream yet (0.7 m^3/s); and (3) the reference is set under the incorrect assumption that the full impacts of all existing wells have already impacted the stream. From the undisturbed reference of 0.8 m^3/s, the depletion includes the estimated depletion from all wells (0.15 m^3/s, 19%). For the disturbed reference, if it is assumed that none of the existing wells are impacting the stream, then the reference is set at 0.7 m^3/s and the estimated cumulative future depletion of 0.15 m^3/s is 21% change relative to the reference. Finally, if all existing wells are assumed to impact the stream at the reference condition, then only the proposed well X contributes to the depletion calculation (0.01 m^3/s, 1%). Depending on the reference condition, the estimated cumulative streamflow depletion ranges from 1% to 21%. If allowable depletion levels are less than 21%, then this uncertainty range includes the allowable depletion range, and assumptions about the reference directly impact whether well X will be permitted or approved by a regulatory agency.

In this example, very little streamflow depletion is likely to result from approving well X. When considered alone, this well is unlikely to harm the stream and would be likely to be approved by a groundwater manager. However, the cumulative impacts of all of the existing wells on the stream are more substantial. This leads directly to a social issue in groundwater permitting: should well X be denied due to groundwater pumping by other water users with wells A-E? If yes, then it is very difficult for newer property owners to access groundwater rights, even when their needs are modest. If not, then groundwater managers will need to find other creative ways of maintaining streamflow, for instance by reducing existing groundwater rights or other programs that encourage compromises in water use among water users in a water use district. As many regions reach their ecological limits for groundwater withdrawals, managers will be faced with overcoming this type of social dilemma.

Beyond the decision about which impacts to include in a reference condition, the metric used to define the reference condition can have a large impact on streamflow depletion calculated. In Fig. 5, we explore how using the August mean or median streamflow could impact streamflow depletion calculations. Plum Creek (Fig. 5a) has a very variable hydrograph, whereas Pleasant Valley Creek (Fig. 5b) is very stable. As a result, the August mean and median streamflow at Pleasant Valley are nearly the same, but the August median is significantly less than the mean at Plum Creek. For both streams, we assume that groundwater pumping has resulted in a streamflow reduction equal to 10% of the mean flow. Since the mean and median are nearly the same at Pleasant Valley Creek, this level of depletion is measured as an 11% depletion from the August median streamflow. However, this 10% depletion from the mean is measured as a 35% depletion from the median in Plum Creek.

Both of these case study examples also demonstrate that a large amount of uncertainty is embedded in the decisions made about reference conditions and existing disturbance. In setting a reference condition, uncertainty can come from two different sources. The best time period or type of reference may be uncertain (Fig. 4), but there also may be uncertainty in the calculated streamflow reference for each scenario. Uncertainty can be more important in some scenarios than others. If

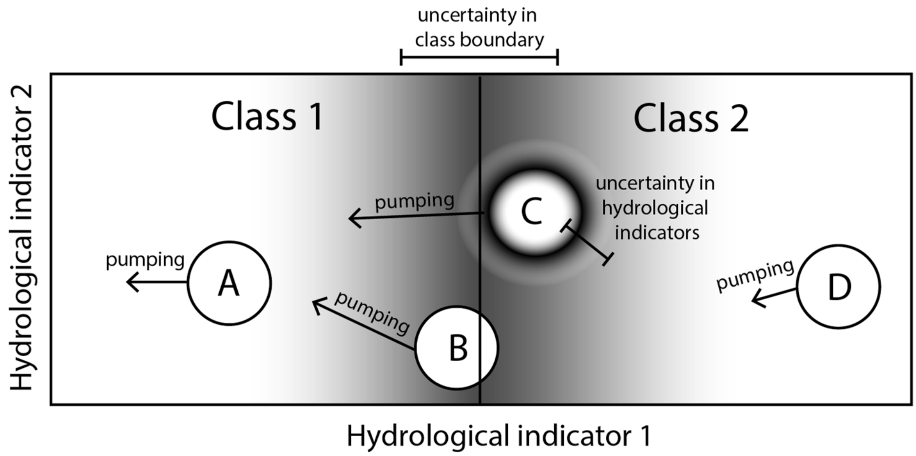


Fig. 6. Uncertainty in class boundaries and stream quantities can impact stream classification. Stream A is clearly in Class 1. Stream B appears to be in Class 1 but could be in Class 2 if the class boundary moves within the uncertainty bound. Stream C appears to be in Class 2, but uncertainty in the stream quantity means that it could be in Class 1. Arrows show potential trajectories for streams given changes that could occur with streamflow depletion.

streamflow depletion is small, then uncertainty in the reference condition may be greater than the expected level of depletion making it impossible to estimate hydrological alteration caused by groundwater pumping. Comparing across the three scenarios in Fig. 4, if there is a 0.1 m^3/s uncertainty in the reference condition, then there is no quantifiable difference between the three reference conditions. This issue may be particularly acute in small headwater streams that tend to have limited monitoring (Nadeau and Rains, 2007), which increases uncertainty, and low flows, which means that small magnitudes of uncertainty can be a large percentage of flow. Headwater streams are both highly vulnerable to streamflow depletion and important ecological habitat (Bishop et al., 2008; van Meerveld et al., 2020), so enhanced monitoring and improved modeling of headwater systems is essential for streamflow depletion management.

3.2. Stream classification

In stream classification, uncertainty can arise from how class boundaries are defined. In practice, stream classifications have distinct boundaries, but for streams on the borderline between classes, the limits chosen for the classification scheme can have a large impact on how the stream is managed. To our knowledge, no sensitivity analyses have been performed to identify the impact of classification boundaries on resulting management decisions. Fig. 6 shows a demonstrative example with only two classes. In this example, streams A and D are clearly members of Classes 1 and 2, respectively, but streams B and C are both borderline and falling within the grey uncertainty area separating the two classes. Uncertainty in classification can be caused by either uncertainty in the precise location of the boundaries between classes (i.e., Stream B) and/or uncertainty in the stream's position in parameter space (i.e., Stream C).

A further consideration is that streamflow depletion may cause a stream to move within classification space if pumping affects the hydrological signatures used for classification. The arrows in Fig. 6 indicate possible trajectories for streams with increased groundwater withdrawal. These arrows show how streams can change classification over time, as emphasized by Rypel et al. (2019) for lakes in Wisconsin. If the class boundary corresponds to an ecologically relevant threshold, for example a threshold on a flow-function curve (Section 3), then maintaining class membership could be essential for ecological health. In that case, stream C would require more strict management than stream D to maintain class membership in Class 2. However, this also brings up concerns about why stream C should be managed differently from stream D if they are in the same region of the classification space. This type of thinking could motivate the inclusion of fuzzy classification and 'transitional' classes along class boundaries, as in Rypel et al. (2019) or Zorn et al. (2012). Further exploration into the prevalence of borderline streams under different classification systems and the impacts on aggregate management actions would support thoughtful management.

3.3. Quantifying streamflow depletion

Quantifying streamflow depletion consists of two steps: first, establishing the conceptual model for the problem and second, applying a quantitative method to calculate streamflow depletion. One particularly difficult aspect of establishing a conceptual model for streamflow depletion is determining the appropriate geographical extent for analysis. If the geographical extent is too small, then local impacts on a stream may be exaggerated in importance since aquatic organisms are able to migrate throughout a stream channel to find suitable habitat. Conversely, if the geographical extent is too large, then it may be difficult to identify the importance or validity of local changes or even to account for spatially variable phenomena (Noorduijn et al., 2019). These considerations in model set-up are also tied closely to the inclusion of nearby wells and cumulative impacts in the model, as discussed in Section 3.1. With a small geographical extent, cumulative impacts are more likely to be neglected since nearby wells fall outside of the model domain, whereas the impact of a new well may be hard to tease out of a larger model with already substantial impacts to the stream from existing wells. The correct geographical extent for each management scenario depends on the geological setting. After identifying an appropriate conceptual model for the management scenario and defining the geographical extent, streamflow depletion is calculated using a quantitative model.

Streamflow depletion is the difference between actual streamflow and what streamflow would have been in the absence of groundwater withdrawal (Barlow et al., 2018). As a result, streamflow is impossible to directly measure because it requires comparing observed streamflow to a hypothetical scenario, and is challenging to estimate because groundwater withdrawal impacts are masked by other causes of hydrological variability (Barlow and Leake, 2012). At the scale of an individual stream reach, detailed field observations directly estimate streamflow depletion (i.e., (Hunt et al., 2001; Sophocleous et al., 1988; Kollet and Zlotnik, 2003; Flores et al., 2020; Weeks et al., 1965)). However, estimating streamflow depletion from observational data alone is not practical at larger scales such as a watershed, aquifer, or region. For these spatial scales, streamflow depletion can be estimated using analytical, numerical, and/or statistical methods. Zipper et al. (2021) review and compare methods for calculating streamflow depletion; Huang et al. (2018),Reeves et al. (2009), and Hunt (2014)

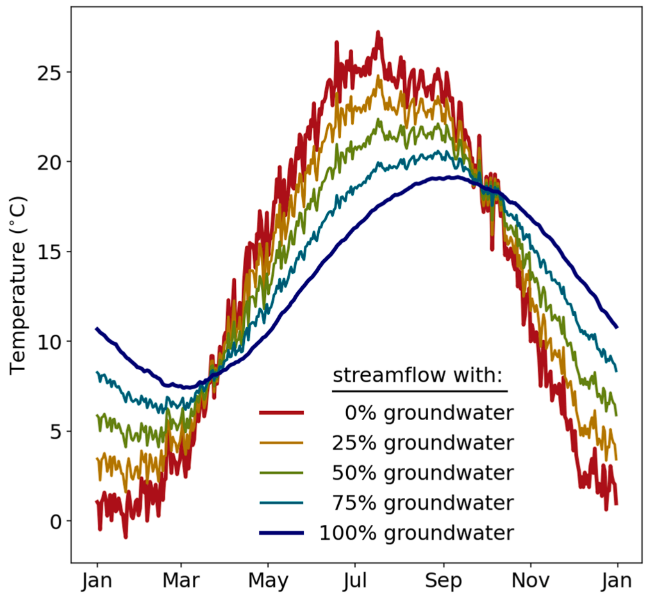


Fig. 7. Schematic demonstrating the difference in thermograph for streamflow that consists of 100% groundwater (navy) and 0% groundwater (red). As the groundwater contribution to streamflow increases, as shown with the examples for 25%, 50%, and 75% groundwater, the streamflow thermograph shifts closer to the 100% groundwater thermograph.

comprehensively review analytical methods; Griebling and Neupauer (2013) describe standard numerical techniques. Here, we briefly discuss performance and capabilities of different streamflow depletion estimation methods with specific emphasis on their use for groundwater withdrawal permitting.

Analytical methods were the first approach used to estimate streamflow depletion (e.g., (Glover and Balmer, 1954; Theis, 1941; Hantush, 1965; Jenkins, 1968; Hunt, 1999)). They are the simplest methods available, requiring relatively little input data and training to use, and are the most common choice for decision support systems for groundwater withdrawal permitting (e.g., Wisconsin, Ohio, Massachusetts, and Michigan; (Diebel et al., 2014; Kendy et al., 2012; Reeves et al., 2009)). Analytical methods are powerful because of their modest data requirements and fast computation, but they are limited by simplifying assumptions, commonly including homogeneity of stream and aquifer properties in space and time (Huang et al., 2018). Analytical solutions also typically apply only to one well and one stream, with cumulative impacts of wells assumed to be linearly additive (Reilly et al., 1987), which is not a valid assumption when streams dry completely (Ahlfeld et al., 2016). Analytical depletion functions were recently developed to distribute the impacts of a well across multiple stream reaches, with good agreement with process-based numerical models (Zipper et al., 2021; Zipper et al., 2019; Reeves et al., 2009; Li et al., 2020; Zipper et al., 2018). Even though data requirements for analytical methods are modest, data limitations are still common. Streambed hydraulic properties, for instance, can have a significant impact on estimated depletion (Neupauer et al., 2020; Lackey et al., 2015; Christensen, 2000) but are rarely known at adequate spatiotemporal resolution.

Numerical models such as MODFLOW/GSFLOW, ParFlow, and HydroGeoSphere typically use process-based governing equations to route water through a two- or three-dimensional representation of a landscape. Numerical models can explicitly represent stream-aquifer exchange at the scale of each model grid cell and therefore provide fine spatial resolution estimates of streamflow depletion and other related ecohydrological processes, and some numerical models can simulate water temperature and other water quality impacts in addition to flows of water. However, this spatial fidelity requires significantly more data, time, and expertise to build and calibrate. Numerical models are the most powerful tool we have for projecting changes in streams and identifying mechanisms, and can be subjected to robust uncertainty assessments to bound depletion estimates with confidence intervals (Foster et al., 2021; Doherty and Moore, 2020). New tools are making it easier to develop and analyze numerical models (e.g., MODFLOW-setup, https://github.com/aleaf/modflow-setup; (Bakker et al., 2016; Fienen et al., 2021; White et al., 2021)) and are well-suited for detailed reviews where managers have site-specific expertise and sufficient data and resources available to represent key model processes. Numerical models have been used extensively for evaluation of groundwater withdrawal (Zipper et al., 2021).

Analytic element models have intermediate complexity between analytical solutions and numerical models (Strack, 2003). They are typically two-dimensional steady-state simulations with a great deal of flexibility in terms of landscape heterogeneity and problem geometry (Haitjema, 1995). While the two-dimensional setup makes them less accurate than multi-layer numerical models where vertical flow is important (Haitjema, 1987), they can provide spatial resolution difficult to achieve with large numerical models and problem specificity that cannot be achieved with standard analytical solutions. While recent research has helped develop Analytic Element models which can simulate transient conditions, multi-layer flow, or both (e.g., TTim, TimML, AnAqSim; (Bakker, 2013; Bakker and Strack, 2003; Fitts, 2010)), Analytic Element models are still not as widely-used as other approaches, such as finite-difference or finite-element models (Hunt, 2006). They have been applied in Wisconsin to model groundwater systems (e.g., (Juckem and Coalition, 2009; Juckem and Dunning, 2015)), but the relative advantages of these different approaches in terms of quantitative performance have yet to be well-defined.

Statistical methods have also been used to infer and estimate streamflow depletion, although they have seen less use than analytical or numerical approaches in streamflow depletion assessment (Zipper et al., 2021). Statistical assessments can be used to identify trends and relationships among data and may be best-suited for settings where data essential to numerical methods, such as subsurface and streambed data, are unavailable. While statistical approaches are typically *informed* by process understanding, they—unlike numerical and analytical approaches—are not underpinned by physical laws. To incorporate more physical knowledge, emerging process-guided machine learning approaches can use physical laws to penalize unreasonable model behavior (e.g., (Read et al., 2019)). These approaches have not yet been tested for streamflow depletion assessment. Data-driven methods have been shown to be more accurate than numerical models in some hydrological modeling scenarios (Gauch et al., 2019; Booker and Woods, 2014) and can be a powerful tool to tease out the importance of different landscape factors and land use in impacting low flows in streams (Hammond and Fleming, 2021).

Given the numerous options for estimating streamflow depletion, the best tool for use in a management framework depends on the complexity of the problem, the level of accuracy needed, and the time constraints for getting results. In many cases, fast review turn-around for groundwater withdrawal applications is essential for maintaining support from water users, and is even legislated to short timeframes in some areas, which would preclude the development of a site-specific calibrated numerical model. While some model intercomparisons have compared within and among different estimation methods (Flores et al., 2020; Sophocleous et al., 1995; Li et al., 2020; Knowling et al., 2019), more thorough benchmarking and comparison among a wide suite of different methods using common input data is a critical research need to better understand the capabilities and limitations of simpler quantitative tools.

3.4. Quantifying temperature changes due to groundwater withdrawal

Stream temperature is a complex quantity dependent on climatic variables, physical attributes (shade from plants and wood, hyporheic exchange, water depth, and water velocity), and relative contributions of surface water and groundwater to streamflow (Stark et al., 1994). Since groundwater is buffered from air by soil and rock, groundwater temperatures are typically cooler than surface water temperatures during the summer, warmer than surface water temperatures during the winter, and change much more slowly than surface water temperatures (Bonan, 2015). Thus, groundwater inputs to streams serve to moderate extreme temperatures and smooth out variability. As a simple example, Fig. 7 shows how groundwater contributions to streamflow moderate streamflow temperatures and reduce variability (effect becomes more pronounced as groundwater contribution increases from 0% groundwater (red) to 100% groundwater contribution (blue)). A reduction in groundwater contribution (for instance from 50% to 25%) approximates the impact of streamflow depletion on a thermograph since streamflow depletion reduces groundwater inputs to streams.

While temperature is ecologically important, literature assessing the impacts of groundwater withdrawal on temperature in nearby streams is limited. Streamflow depletion field experiments generally use surface water diversion to simulate streamflow depletion (e.g., (Nuhfer et al., 2017; SStelzer and Pillsburytelzer and Pillsbury, 2020)), do not report temperature data (e.g., (Hunt et al., 2001, 1966, 2002, 1988, 2004)), or are not controlled enough to allow for evaluating the impacts of pumping on stream temperature (Kwon et al., 2020). One exception is Flores et al. (2020), who examined the impact of pumping on stream temperature in a field study and identified an immediate stream temperature change of a few degrees when pumping started or stopped, indicating the potential importance of this understudied aspect of streamflow depletion.

Several studies have investigated potential temperature impacts of streamflow depletion. Risley et al. (2010) evaluated the impacts of pumping on stream temperature in archetypal numerical models. They found that while pumping generally increased stream temperature in the summer and reduced stream temperature in the winter, the change was generally less than 0.5°C. Conversely, Stark et al. (1994) found that pumping for irrigation could result in a 0.5–1.5° increase in summer streamflow temperatures, and Foglia et al. (2013) found that a 50% increase in minimum summer flows could significantly reduce the proportion of the stream network where temperatures exceed 25°C. Andrews (2018) did a comparison among a suite of temperature models to identify which variables and processes were most important to include in order to capture observed thermal properties of streams. They found that it was important to consider three thermal water pools (groundwater, surface water, and overland flow). A model that considers only water depth (such as those used for assessment in Michigan and Wisconsin; (Zorn et al., 2012; Diebel et al., 2015)) was not sufficient to capture observed behavior. Using the best fit models, changes in stream temperature induced by pumping were generally less than 0.5°C.

These initial explorations of the impacts of groundwater withdrawal on stream temperature, combined with our preliminary analysis of functional temperature components (Section 2.2), indicate that there may be a substantial thermal change induced by pumping in some cases. There are numerous tools available that are capable of calculating stream temperatures (Andrews, 2018), and some calibrated groundwater models with temperature already exist (e.g., (Woolfenden et al., 2011; Chunn et al., 2019)). However, some of these existing temperature models are not capable of accurately predicting pumping impacts on streamflow and field data to evaluate models for the thermal impacts of pumping are still very limited (see above). Because a warming climate contributes directly to increases in stream temperatures (Ducharne et al., 2007) and may combine with agricultural expansion to increase streamflow depletion through increasing human demand for water resources (Wada et al., 2013), it is critical to consider the combined impacts of climatic warming, land use change, and streamflow depletion on stream temperatures (Deitchman and Loheide, 2012). Enhanced research focus on the thermal impacts of groundwater withdrawal situated in the broader context of changing thermal regimes, including diverse field studies and exploration of widely-applicable model structures, is necessary to disentangle diverse drivers of temperature change and further develop a functional temperatures approach to streamflow depletion management.

3.5. Connecting stream alteration to ecological impacts

As emphasized in Section 2, environmental flows management requires evaluating more than streamflow quantity. Many of the methods described above for temperature and streamflow modeling can be used to calculate multiple flow/temperature characteristics. To manage for future conditions, though, it is essential to incorporate future impacts on streamflow and temperature that derive from climate change or land use change as well (Craig et al., 2017). Understanding these feedbacks is essential for making decisions about acceptable thresholds of change (described in Section 2).

For instance, climate change, land use change, and groundwater withdrawal can all increase variability in stream conditions above variability ecosystems are adapted to. Greater variation in streamflow results in even greater variation in wetted channel extent in headwater streams, with implications for seasonal stream habitat dynamics that aquatic ecosystems rely on (Lapides et al., 2021). When variability becomes more extreme than the ecosystems are adapted for, more variation can result in, for example, more extreme dry years when flows become low enough to cause catastrophic population declines. In particular, shifts in streamflow patterns from perennial to non-perennial can have devastating impacts on aquatic ecosystems (Bogan and Lytle, 2011). A general shift to non-perennial flow has become increasing common across much of the southern United States and Europe (Zipper et al., 2021; Tramblay et al., 2021) and has been linked in some regions to streamflow depletion (Perkin et al., 2017; Zimmer et al., 2020). Ecosystems can recover after dry years in certain situations due to density dependent recruitment, but it is important to understand long-term population dynamics to decide whether it is acceptable to plan for occasional community collapse (Wang et al., 2003; Wang et al., 2001; Zorn et al., 2012) or whether community collapse is an unacceptable impact.

Lack of information about ecosystem resilience to the impacts of groundwater withdrawal can complicate discussions around socially acceptable impacts. As an example, Bradbury et al. (2017) found that there were significant impacts to stream ecosystems in the Little Plover River, Wisconsin as a result of groundwater withdrawal. However, years of high precipitation in Wisconsin resulted in replenished streamflow with rebounding fish communities by the time the results were published. This drove water users to challenge the importance of ecological impacts because of visible ecosystem resilience. Part of quantifying the risks of streamflow depletion is a greater understanding of how ecosystems respond to and rebound from stressors over long timescales so that there is greater ability to identify types of stresses that will cause reversible vs. irreversible damage to ecosystems.

4. Towards ecologically-informed groundwater withdrawal decision-making

Developing a well-rounded description of environmental flow needs and the timing and magnitude of streamflow depletion is the foundation of a groundwater withdrawal decision-making protocol designed to balance the needs of society and aquatic ecosystems. Incorporating aquatic ecosystem water needs into groundwater withdrawal permitting is challenging due to uncertainty in the quantification of streamflow depletion impacts, environmental flow requirements, and implementation challenges related to stakeholder buy-in and limited resource availability. The functional flows framework provides a structure with which to describe a holistic streamflow and stream temperature state, a description that can guide use of an environmental flows management framework such as ELOHA, which—as we review here—has been used for groundwater management in some form in several jurisdictions. Given the difficulties inherent to describing streams and ecoystems quantitatively at large scales, it is important that uncertainty be incorporated into all technical steps of a management process, from calculating reference conditions and classifying streams to quantifying alteration. Particular attention should be paid to how a reference condition is defined and calculated to ensure that it is appropriate for the management goals.

Even with general agreement among the scientific community on the necessary technical steps for successful management of streamflow depletion, substantial challenges remain in terms of functional implementation. Numerous studies have identified jurisdictional complexities to groundwater management (e.g., (Molle and Closas, 2020; Gage and Milman, 2021)), structural barriers to improving monitoring and scientific programs (Saito et al., 2021), and difficulties in balancing the needs of water users and ecosystems (e.g., (DNR Water Use Section, 2021; Closas and Molle, 2018; Molle and Closas, 2020)). While a complete review of the sociopolitical challenges in managing streamflow depletion is beyond the scope of this study, prior research identifies the importance of stakeholder buy-in for groundwater permitting programs (e.g., (Tsvetanov and Earnhart, 2020; Deines et al., 2019; Drysdale and Hendricks, 2018; Baldwin et al., 2012; Kendy et al., 2012)). Considering the needs of all stakeholders (including water users, people who enjoy outdoor recreation in waterways, indigenous peoples, and ecosystems) in groundwater management allows for the development of management protocols that work for everyone. Additionally, streamflow depletion management needs to be placed in the larger context of impacts to streams, taking into account future land use and climate changes that could amplify or cancel out impacts of streamflow depletion. Since ecosystems cannot show up to advocate for themselves, the strategies discussed in this review can help to bring ecosystem needs to the table in a quantitative way.

As other jurisdictions move towards ecologically-informed groundwater withdrawal decision-making, they can prioritize the successful approaches we review here, most importantly stakeholder engagement throughout the process of identifying ecosystem needs and setting appropriate streamflow depletion thresholds. Many of the challenges raised in this review are more generally applicable to a larger range of streamflow management problems such as defining Waters of the United States (WOTUS; (Department of the Army et al., 2020)) as a basis for regulation. To support these management needs, the scientific community should prioritize research on user-friendly methods to estimate pumping impacts (and associated uncertainty) on streamflow, stream temperature, and aquatic ecosystems to empower local managers and stakeholders to make informed decisions.

Data Availability

Code and data used for analyses in this study are available at https://github.com/lapidesd/Eco_Stream_Depletion_Review. (Lapides et al., 2021).

CRediT authorship contribution statement

Dana A. Lapides: Conceptualization, Methodology, Investigation, Software, Data curation, Formal analysis, Writing - original draft, Writing - review & editing, Visualization. **Bryan M. Maitland:** Conceptualization, Methodology, Investigation, Writing - original draft, Writing - review & editing. **Samuel C. Zipper:** Conceptualization, Methodology, Investigation, Writing - original draft, Writing - review & editing. **Alexander W. Latzka:** Writing - review & editing, Supervision. **Aaron Pruitt:** Writing - review & editing. **Rachel Greve:** Conceptualization, Writing - review & editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Identifying hydrological signatures for the upper Midwest

References for Table A.1:

1. Blum et al. (2018)–VA, USA
2. Kanno et al. (2016)–VA, USA
3. Xu et al. (2010)–MA, USA
4. Xu et al. (2010)–MA, USA
5. Kanno et al. (2015)–VA, USA
6. Nuhfer et al. (2017)–MI, USA
7. Kanno et al. (2017)–TN, USA
8. Jensen and Johnsen (1999)–Norway
9. Roghair et al. (2002)–VA, USA
10. George et al. (2015)–NY, USA
11. Letcher et al. (2015)–MA, USA
12. Deitchman and Loheide (2012)–WI, USA
13. Bassar et al. (2016)–MA, USA
14. Wang et al. (2003)–WI, USA

15. Wang et al. (2001)–WI, USA
16. Zorn et al. (2012)–MI, USA
17. Cattanéo et al. (2002)–France
18. Nicola et al. (2009)–Spain
19. Hakala and Hartman (2004)–WV, USA
20. Kanno et al. (2015)–CT, USA
21. Kennen et al. (2010)–NE, USA
22. Kennedy et al. (2016)–CO, USA
23. Lobón-Cerviá (2004)–Spain
24. Lobón-Cerviá et al. (2018)–Spain
25. Lobón-Cerviá and Rincón and Rincón (2004)–Spain
26. Warren et al. (2009)–NY, USA
27. Wenger et al. (2011)–western USA
28. Wills et al. (2006)–MI, USA
29. Morley and Karr (2002)–PNW, USA
30. Dare et al. (2002)–WY, USA

Table A.1
Summary of citations for Wisconsin functional hydrological signature selection.

Flow Component	Flow Characteristic	Flow Metric	Fish	Inverts
Baseflow	Magnitude	Average flows (Q50) Winter/Fall precipitation Winter peak precipitation	1, 13, 14, 15, 17, 18 2, 5, 7 7, 8, 9	
	Timing			
	Duration			
	Frequency			
	Variability	CV, Variability index	17	
	Rate of Change	(+) and (-) changes in flow bw days	17, 18	
Spring freshet	Magnitude	Average flows (Q50) Spring precipitation Spring peak precipitation	17, 18, 23, 24, 25, 26 2, 5, 7 7, 8, 9	21, 22
	Timing			
	Duration			
	Frequency			
	Variability	CV, Variability index	17	
	Rate of Change	(+) and (-) changes in flow bw days	17, 18	22
Summer recession	Magnitude	Summer precipitation Summer mean daily flow	2 3, 4, 6, 11, 12, 17, 19, 20	28, 29, 31
	Timing			
	Duration			
	Frequency	% year daily flow > annual mean flow		30, 31
	Variability	CV daily flow	3	
	Rate of Change	change in discharge/d		
Peak flows	Magnitude	High flows (95 ptile, Q10/Max/Max7) annually and by hydroperiod	1, 10, 17, 18, 26, 27	
	Timing/ Seasonality	Date of peak flow annually and by hydroperiod	18, 26	22
	Duration	Duration of peak flow annually and by hydroperiod Percent time where flow > 2-yr peak annually and by hydroperiod	17, 18 17	
	Frequency	No. extreme events per 5 years during winter baseflow season No. daily flows > 2-yr peak annually and by hydroperiod	5 17, 18	22
Low flows	Magnitude	Low flows (5th percentile, Q90/Min) annually and by hydroperiod	2, 3, 5, 11, 17, 18, 19, 30, 35	28, 29
	Timing/ seasonality	Season/date of low flow event	30, 35, 36, 37	
	Duration	Duration of low flow annually and by hydroperiod	18, 34	32, 33*
	Frequency			
	Rate of change	change in discharge/d	36, 41, 42	

Yarnell et al. (2020) demonstrate functional flows for a mixed rain-snowmelt system common in California. We demonstrate functional flows for a snowmelt and groundwater-dominated system (Eau Galle River, WI) in the left half of Fig. 1 that is representative of many streams typical of Wisconsin and mid- to high-latitudes around the world. The annual shape of the hydrograph in the top left shows a baseflow period throughout most of the year with a pronounced spring freshet and summer recession. Peak flows occur throughout the year, while extreme low flows occur during the baseflow period at the end of summer and beginning of fall.

Because climate regimes in the Midwest differ substantially from those of California, we performed a literature review on the seasonal importance of streamflow for aquatic organisms in this region (i.e., Autumn, winter, spring, and summer—which roughly correspond to the hydroperiods in Fig. 1). Where possible, ecological literature from the U.S. Upper Midwest was used in this review, but literature from other mid- to high-latitude studies with similar hydrograph functions were used to expand the scope of information where studies were not available in the Upper Midwest (see Table A.1 for full set of references). At the bottom left of Fig. 1, we identify functional characteristics associated with each functional flow component with an ‘x’ based on our literature review and analogy to Yarnell et al. (2020). These results are summarized below.

Streamflow is a master variable that shapes the ecological condition of flowing water ecosystems and thus the life history traits of aquatic biota (Power et al., 1995). Streamflow characteristics vary over a year and relate to different bioperiods of freshwater species. Autumn (September-November) is a key bioperiod for brook and brown trout in the Upper Midwest as it coincides with adult spawning activity. Field studies have found a positive relationship between Autumn streamflows and trout growth, recruitment, and survival. For example, Xu et al. (2010) found that high Autumn streamflow increased brook trout growth in a Massachusetts stream, and numerous studies have found that low Autumn flows or precipitation are associated with decreased population sizes in subsequent years (Kanno et al., 2015; Kanno et al., 2016; Bassar et al., 2016; Blum et al., 2018). Indeed, declines in late summer and autumn base flows have been implicated in the degradation of both cold- and warm-water fish assemblages in southern Wisconsin (Wang et al., 2003; Wang et al., 2001).

Streamflow patterns during winter (December-February) have historically been considered less consequential for animal population dynamics in temperate freshwater ecosystems; however recent research has challenged this view. High winter precipitation and streamflow events during the egg incubation and fry rearing period have been shown to be related to low age 0 + fish abundances, suggesting that winter high flows result in bed-scouring that negatively impacts survival of eggs and newly hatched individuals (Blum et al., 2018; Kanno et al., 2016; Xu et al., 2010; Kanno et al., 2015; Kanno et al., 2017; Jensen and Johnsen, 1999; Roghair et al., 2002). This single-year effect translates into decreasing occurrence probability for fall-spawning trout with increasing winter high flows (Wenger et al., 2011).

Streamflow during or just after trout emergence is a primary factor in determining recruitment and subsequent population dynamics across geographic regions. In the upper Midwest, this time period corresponds with Spring (March-May). During spring, the timing, magnitude and duration of extreme flow conditions and the rate of change in discharge directly impact autumn-spawning trout, survival, growth, and recruitment (Xu et al., 2010; Blum et al., 2018; Kanno et al., 2015; Kanno et al., 2017). Spring streamflow can also show non-linear relationships with trout population dynamics; in Spain, brown trout recruitment increases with increasing spring streamflows until a high flow threshold, beyond which recruitment declines (Lobón-Cerviá, 2004; Lobón-Cerviá et al., 2018; Lobón-Cerviá and Rincón and Rincón, 2004). While extreme high flows during spring or winter can be detrimental to trout populations, if displacing floods occur infrequently (i.e., a few years apart), fall-spawned fish populations can be resilient to floods (Warren et al., 2009; Kanno et al., 2015).

Streamflow conditions during the summer recession and base flow periods (June-August) are described by low-flow hydrology, which is strongly driven by variability in the magnitude, timing/seasonality, frequency, duration, and rate of change flow characteristics (Rolls et al., 2012). Changes in flow magnitude can negatively affect fish populations (Poff and Allan, 1995; Letcher et al., 2015; Kanno et al., 2016) and macroinvertebrate communities (Kennedy et al., 2016; Kennen et al., 2010; Wills et al., 2006; Morley and Karr, 2002) due to changes in physical habitat (Rolls et al., 2012), such as sediment or pool size (SStelzer and Pillsburytelzer and Pillsbury, 2020), or reduction in access to food or refugia (Hakala and Hartman, 2004; Deitchman and Loheide, 2012; Kanno et al., 2015). The importance of these changes in flow variation to biota in tributary and mainstem reaches can be very different, though (Xu et al., 2010; Letcher et al., 2015; McCargo and Peterson, 2010). For instance, Letcher et al. (2015) found that low flows in tributaries were more harmful to trout populations than in mainstem reaches.

In Wisconsin, the summer season corresponds to the growth and adult pre-spawning bioperiod of brook and brown trout, and the magnitude, duration and timing of summer low flows appear to be a critical factor for survival of young trout (Nicola et al., 2009; Dare et al., 2002; Harvey et al., 2006; Sotiropoulos et al., 2006). Generally, higher summer streamflow improves fish population health. Xu et al. (2010) found that high summer flows increased brook trout growth, and Kanno et al. (2016) found that high summer precipitation, and thus high streamflows, were associated with high young of year abundances the following year. Conversely, increasing duration of low-flow events during the summer has been associated with increases in macrophyte biomass (Suren and Riis, 2010), decreases in macroinvertebrate richness (Datry, 2012), and decreases in fish growth and abundance (Nuhfer et al., 2017; Jowett et al., 2005). Zorn et al. (2012),Letcher et al. (2015), and Nuhfer et al. (2017) all found that low summer flows and streamflow depletion had strong negative effects on trout survival and growth.

In addition, the rate of change in summer flow recession can rapidly alter abiotic conditions and influences access to refugia by mobile biota (Rolls et al., 2012). For example, rapid dewatering of stream channels can result in mortality of fish that were not able to emigrate and thus were stranded in shallow pools or on dry ground (Walker et al., 2018), and thus has been associated with decreased survival in riverine fish (Harvey et al., 2006; Bradford, 1997). The frequency of low-flow events during summer and baseflow periods in regions that rarely experience ecologically critical low-flow magnitudes—like Wisconsin—can also have negative effects on biota (Rolls et al., 2012). But few studies have investigated these effects, despite the fact that groundwater extraction has been associated with increased frequency (and duration) of low flows (Kustu et al., 2010). Finally, high variability in summer streamflows can also reduce fish population survival; (Xu et al., 2010) found that adult brook trout survival declined as the coefficient of variation (CV) of daily streamflow increased.

Appendix B. Identifying temperature signatures for the upper Midwest

We also performed a literature review on the importance of stream temperature during Autumn, winter, spring and summer for aquatic organisms in the Upper Midwest. See Table B.2 for a summary of the literature review with a full reference list. In the right half of Fig. 1, we use temperature data from the USGS (United States Geologic Survey, 2021) to illustrate functional temperature components for the Eau Galle River in Wisconsin. Stream temperature tracks seasonal temperature changes in Wisconsin so that streams are around freezing during the winter months and about 20–25°C warmer during the summer. Each functional temperature component identified as important for stream communities in the Upper Midwest is marked with an 'x' in Fig. 1.

Observed differences in stream communities are often attributable to variation in stream thermal regimes (Wehrly et al., 2003; Wehrly et al., 2007; Lyons et al., 2009). Because decreasing autumn temperature is an environmental cue for fall-spawning fishes (Warren et al., 2012), changes to a natural thermal regime, such as extreme high fall temperatures (Xu et al., 2010), or delays in the onset/timing of fall cooling, can delay or reduce spawning activity, with implications for annual recruitment and subsequent population dynamics (Letcher et al., 2015). Stream trout can also experience a metabolic deficit during acclimation to rapidly declining water temperatures in the fall (Cunjak and Power, 1987), which has implications for overwinter survival. In addition to temporal variation, spatial variation of temperature can impact fish populations. Spatial variation in water temperatures can enable better growth if fish have access to warmwater forage during the winter (Armstrong et al., 2021).

During winter, the physiological ecology of stream-dwelling salmonids suggests a positive relationship between population dynamics and temperature (Huusko et al., 2007). High winter temperatures have a positive effect on incubating eggs since trout eggs hatch faster and are more successful in warmer water (Baxter and McPhail, 1999). Additionally, earlier hatching results in a longer growing season for newly hatched fry, further improving their chances of survival. Similarly, high winter water temperatures are correlated with greater survival for overwintering young of year fish (Hunt, 1969), higher growth rates (Xu et al., 2010), and increasing young of year abundances in subsequent summers (Kanno et al., 2016; Kanno et al., 2015; Kanno et al., 2017). Conversely, extreme low temperatures (e.g., $<= 1^{\circ}C$) can physiologically stress trout and lead to lower overwinter survival (Letcher et al., 2015) or result in direct mortality from severe ice or snow conditions. As the young of year cohort is extremely important for long-term population stability (Kanno et al., 2016), such effects can lead to population declines (Personal communication, Kirk Olson, WDNR Fisheries Biologist) or induce downstream movement when frequent freezing and thawing lead to variable surface ice cover and frequent supercooling ($< 0^{\circ}C$; (Jakober et al., 1998)).

The spring warming period is related to the fry rearing bioperiod of Autumn-spawning salmonids. Increases in average spring water and air temperatures are consistently and positively associated with young-of-year (Kanno et al., 2015) and adult trout survival (Xu et al., 2010), growth (Xu et al., 2010), and the following summer's young-of-year abundance (Kanno et al., 2016), although Tsang et al. (2016) found a slight negative association between brook and brown trout populations and maximum daily mean temperatures in spring. We also expect that the variability and rate of change in magnitudes, and predictability of thermal patterns in spring over time, are important for recruitment, growth, and survival of stream trout, and have the potential to drive non-linear population effects within the season (Arismendi et al., 2013). For example, Blum et al. (2018) found that warmer maximum daily spring temperatures were associated with increased young-of-year abundance up to about 1.5 standard deviations, above which abundance declined.

Stream temperatures during the summer heat period have important implications for species adapted to cold-water, which, if possible, actively avoid high temperatures (Petty et al., 2012). For example, when stressful summer water temperatures occurred in a mainstem river of Michigan's Upper Peninsula, Hayes et al. (1998) documented brook trout migrating several kilometers upstream to a tributary to avoid the unfavorable thermal condition. When trout cannot avoid stressful habitats and stream temperatures reach or exceed critical levels, direct mortality events, such as fish kills, can occur (Till et al., 2019). Prolonged periods of elevated summer temperatures can have negative physiological effects on individuals that reduce population biomass (Kratzer and Warren, 2013), growth of young-of-year (Bassar et al., 2016) and adult fish (Xu et al., 2010; Robinson et al., 2010; Nuhfer et al., 2017), reduce annual survival (Xu et al., 2010; Letcher et al., 2015), and delay spawning and reduce redd construction (Warren et al., 2012).

Given the substantial and unequivocal support for the negative effects of warming temperatures on cold-water adapted animals, combined with growing evidence showing how flow reductions can increase stream temperatures (Gaffield et al., 2005; Nuhfer et al., 2017), it follows that changes in stream thermal regimes during the summer warm period may pose the most serious threats to stream trout. For example, trout population inhabiting cold-transitional streams in Michigan declined in abundance when a mere 10% in flow reduction occurred because these streams had reference summer temperatures near the critical thermal tolerance levels for trout (Zorn et al., 2012). Similarly, stream temperatures in Wisconsin may reach critical maximum thresholds for stream trout mortality if both air temperature increases and baseflow declines (Deitchman and Loheide, 2012; Selbig, 2015), with Mitro et al. (2019) projecting a 68% and 32% decline in brook and brown trout thermal habitat by mid-century in response to warming summer air temperatures.

It is important to also note that streamflow and temperature can have strong and complex interactive effects (Xu et al., 2010; Nuhfer et al., 2017). But a general takeaway is that population declines result from low streamflows and high temperatures in summer, and high streamflows and low stream temperatures in winter (Letcher et al., 2015).

References for Table B.2:

Table B.2
Summary of citations for Wisconsin functional temperature signature selection.

Temperature Component	Temperature Characteristic	Temperature Metric	Reference
Winter cold	Magnitude	Mean of daily mean temp Cumulative DD	4, 8, 14, 15, 16, 19, 23 14
	Timing	Date of 5-75th percentile cumulative temperature distribution (CTD)	14
	Duration		
	Frequency	Freq. freeze thaw events	26
	Variability	Winter mean of daily range	14
Spring warming	Magnitude	Mean of daily mean	2, 4, 8, 15, 16, 19
	Timing		
	Duration		
	Variability		
Summer warm	Magnitude	Mean of daily (n-day) mean Daily (n-day) mean Cumulative DD	2, 3, 7, 8, 10, 11, 12, 16, 17, 18, 27, 28 19, 20, 21 14
	Timing		
	Duration	Time > threshold temp	12, 13, 14, 23
	Frequency		
	Variability	July weekly range ("fluctuation") CV mean daily temp Mean of daily range	1 7 2, 14
	Rate of change	Hourly/daily rate of change	10
Fall cooling	Magnitude	Mean of daily mean temp	4, 8, 15, 16, 19
	Timing		
	Duration		22
	Variability		
	Rate of Change		22
Peak temperature	Magnitude	Mean of max (n-day) daily temp Max of mean/max daily (n-day) temp	1, 5, 6, 9, 12 2, 3, 5, 10, 11, 12, 17, 20, 21, 24
	Timing	Date of max of daily max temp.	2, 14
	Duration	Time > 4.5 C in winter	12, 13, 14, 23
	Frequency	No. 'events' > threshold temp in summer	12, 14
	Variability	Max of daily range	5
Low temperature	Magnitude	Mean of daily min temp	21
	Timing	Date of min of daily min temp	14
	Duration	Time < threshold temp. in winter	14
	Frequency	No. 'events' < threshold temp. in winter	14

1. Wehrly et al. (2003)–MI, USA
2. Tsang et al. (2016)–MI/WI/MN, USA
3. Lyons et al. (2009)–MI/WI, USA
4. Kanno et al. (2016)–VA, USA
5. Wehrly et al. (2007)–MI/WI, USA
6. Blum et al. (2018)–VA, USA
7. Xu et al. (2010)–MA, USA
8. Xu et al. (2010)–MA, USA
9. Kanno et al. (2015)–VA, USA
10. Nuhfer et al. (2017)–MI, USA
11. Mitro et al. (2019)–WI, USA
12. Petty et al. (2012)–NY, USA
13. Kratzer and Warren (2013)–VT, USA
14. Arismendi et al. (2013)–OR, USA
15. Kanno et al. (2017)–TN, USA
16. Letcher et al. (2015)–MA, USA
17. Deitchman and Loheide (2012)–WI, USA
18. Selbig (2015)–WI, USA
19. Bassar et al. (2016)–MA, USA
20. Gaffield et al. (2005)–WI, USA
21. Wang et al. (2003)–WI, USA
22. Cunjak and Power (1987)–Waterloo, Canada
23. Hunt (1969)–WI, USA
24. Warren et al. (2012)–NY, USA
25. Robinson et al. (2010)–NY, USA
26. Jakober et al. (1998)–MT, USA
27. Hayes et al. (1998)–MI, USA
28. Zorn et al. (2012)–MI, USA

Appendix C. Characterizing hydrological signatures sensitive to groundwater withdrawal in Wisconsin

As an illustrative example, we calculated the signatures in Tables A.1 and B.2 for 34 Wisconsin streams under existing conditions and with simulated pumping from one nearby well. Hydrographs were obtained from USGS Surface Water for the Nation (United States Geologic Survey, 2021), and pumping was simulated using the Glover solution (Glover and Balmer, 1954) implemented in StreamDepletr (https://cran.r-project.org/web/packages/streamDepletr/index.html). For simplicity, the aquifer properties are held constant across all sites with hydraulic conduction of $K = 10$ m/d, aquifer thickness of $b = 100$ m, and specific yield of $S_y = 0.2$. The well has a seasonal pumping schedule with constant pumping from June 1 until September 1 every year.

For thermographs, we used an end-member mixing model on the same 34 Wisconsin streams used for the hydrograph analysis to simulate temperatures with two water sources: groundwater and surface flow. Flows were modeled with the same temperature profiles in all cases for simplicity. Groundwater was assumed to be 9.9°C, the median of mean annual streamflow temperatures across all temperature sites, and surface flow was assumed to match daily air temperature in Madison, WI from NOAA Climate Data Online (NationalCenters, 2021). Groundwater and surface flow components of the hydrograph were separated using the USGS HYSEP fixed interval method (Sloto and Crouse, 1996) implemented in Python at https://github.com/dadelforge/baseflow-separation/blob/master/physep/hysep.py using a window size of 52 days. We identified 40 USGS sites in Wisconsin with both temperature and streamflow data. For these sites, the median R^2 between the actual temperature measurements and the end-member mixing model is 0.42. Hydrograph separation window was chosen to maximize the median R^2. The low R^2 can be accounted for by the simplicity of the model and the fact that the same groundwater temperature is used in all cases. When groundwater temperature is site-specific, median R^2 increases to 0.55, but site-specific estimates are not available statewide, so the single value is used in all cases. Changes in temperature signatures are calculated by comparing the results of the end-member mixing model for the original and altered hydrograph timeseries for all streams so that (i) the analysis could be applied at sites without temperature data and (ii) results do not account for inaccuracies in the model.

To calculate seasonal flow and temperature signatures, we algorithmically identified boundaries between hydroperiods/thermoperiods. We calculated a best fit to a broken stick function (two lines that hinge at a point) through a range of time that encompasses the expected timing of the seasonal boundary. The seasonal boundary is defined as the hinge point for the broken stick function. Timeframes used for identifying seasonal breaks for each season are included in Table C.3. An algorithmic method was required in order to assess changes in seasonal timing objectively. About 30 sites were discarded from available USGS data since the seasonal fitting algorithm was not adequate to capture visual season boundaries. To determine overall impacted flow and temperature signatures, percent difference between the original and well-impacted hydro- and thermographs were calculated for each site. Timing signatures do not make sense as a percent difference so are reported as a difference in number of days. We considered signatures to be generally impacted if the median percent difference is greater than 10%. Signatures are often impacted if the 25th-75th percentiles of percent

Table C.3
Timeframes used to calculate breaks between hydro- and thermoperiods.

Season boundary	Timeframe for fit
hydroperiods	
baseflow to freshet	November 1-April 1
freshet to recession	March 1-July 15
recession to baseflow	June 1-October 1
thermoperiods	
winter to spring	January 1-April 1
spring to summer	May 1-September 1
summer to fall	July 1-November 15
fall to winter	October 1-December 31

different for the signature extend beyond 10%. For timing signatures, a difference of more than 5 days was considered significant. No hydrograph showed a difference in timing for any period of larger than 2 days. All median and interquartile impacts are shown in Table C.3

Appendix D. Stakeholder engagement

Maintaining stakeholder buy-in is important for the success of a management strategy because water users are far more likely to agree to water rights reductions or water rights reductions if they are involved in the process of setting guidelines and deciding on implementation based on local knowledge (Baldwin et al., 2012). There are many potential stakeholders for management decisions related to streamflow depletion, including those who pump groundwater across the domestic, industrial, and municipal sectors; institutions within these sectors such as groundwater management boards; the communities in which these stakeholders are embedded; and NGOs such as environmental groups, to name a few.

Stakeholders have played an important role in building many existing groundwater management systems. For example, the advisory council for the Michigan Water Withdrawal Assessment Process consisted of diverse stakeholders, who collaborated to decide acceptable ecological thresholds, and determined that decline past a 10% reduction in characteristic fish abundance would constitute a significant impact (Zorn et al., 2012). In a similar working group process with stakeholders, the Rhode Island SDM used a stream classification based on existing alteration to prioritize stream protections (Kendy et al., 2012). There, a board representing 68 organizations with varying interests formed a working group involved in setting limits. The group agreed that a reduction of greater than 50% 7Q10 during the summer would cause a significant loss of fish and stream invertebrates. To help meet the needs of water users, they decided to allow more water withdrawal during non-summer seasons when water resources are less strained, giving water users security against limited water supply during the summer months. Additional examples where stakeholders were involved in setting ecological limits to streamflow depletion can be found in Tables C.4, C.5, C.6.

Different projects have included a range of levels of stakeholder engagement. Higher levels of stakeholder engagement can be difficult to coordinate, but the rewards can be great. Including diverse stakeholders can ensure that the management decisions represent the values of all interested parties, emphasizing the inclusion both of water users and environmental groups. Most projects include stakeholders as advisors to the technical process to either set goals or agree on interpretation of results. This role means that effective engagement with stakeholders relies on the ability of the technical team to adequately communicate findings. For instance, Colorado produced a series of easily interpretable maps to guide discussions with stakeholders. Choosing appropriate aspects of the project for inclusion of stakeholders is important to maintain stakeholder interest, limit the number of decision-makers involved in certain decisions, and ensure that stakeholder input is valuable.

Table C.4
Calculated differences in hydrological signatures from simulated groundwater withdrawal across 34 streams in Wisconsin.

signature	median change	interquartile change
baseflow Q50	−31%	(-29)-(-33)%
baseflow Q90	−88%	(-67)-(-162)%
baseflow Q5	−4%	(-2)-(-11)%
baseflow Q95	−116%	(-91)-(-251)%
freshet Q5	−1%	(-1)-(-2)%
freshet Q95	−55%	(-40)-(-76)%
freshet Q50	−10%	(-8)-(-15)%
freshet rate of change	0%	(0)-(0)%
freshet CV	5%	(4)-(7)%
recession rate of change	0%	(0)-(1)%
recession CV	4%	(2)-(10)%
recession Q50	−9%	(-5)-(-11)%
recession Q5	−1%	(-1)-(-5)%
recession Q95	−22%	(-16)-(-34)%
annual Q5	−3%	(-2)-(-8)%
2-yr flow frequency	−4%	(-3)-(-5)%
5-yr flow frequency	0%	(0)-(1)%
10-yr flow frequency	0%	(0)-(1)%
annual Q95	−101%	(-44)-(-205)%
2-yr flow	−1%	(0)-(-1)%
5-yr flow	0%	(0)-(-1)%
10-yr flow	0%	(0)-(-1)%
annual Q95 frequency	367%	(297)-(434)%
annual Q5 frequency	−3%	(-2)-(-12)%
baseflow Q5 frequency	−4%	(-2)-(-17)%
freshet Q5 frequency	0%	(0)-(-4)%
recession Q5 frequency	−2%	(-1)-(-11)%
baseflow Q95 frequency	379%	(311)-(489)%
freshet Q95 frequency	127%	(80)-(267)%
recession Q95 frequency	198%	(100)-(425)%
freshet start date	0 days	(0)-(0) days
recession start date	0 days	(0)-(0) days
baseflow start date	0 days	(0)-(0) days
baseflow daily rate of change	0%	(-14)-(34)%
% year daily flow > mean	2%	(1)-(6)%
% year daily flow < mean	−1%	(0)-(-3)%

Table C.5
Calculated differences in stream temperature signatures from simulated groundwater withdrawal across 34 streams in Wisconsin.

signature	median change	interquartile change
winter mean daily temp.	−0.2°C	(0)-(-0.4)°C
winter 5th temp. percentile date	0 days	(0)-(0) days
winter 75th temp. percentile date	0 days	(0)-(0) days
winter temp. CV	0%	(-70)-(30)%
winter DD2.5	5%	(1)-(19)%
winter min. date	0 days	(0)-(0) days
winter min. daily temp.	−0.5°C	(0.0)-(-1.0)°C
winter days below 2°C	2%	(0)-(6)%
winter frequency below 2°C	0%	(-1)-(1)%
winter frequency freeze	1%	(0)-(8)%
winter days above 4.5°C	−2%	(0)-(-13)%
spring mean daily temp.	0.3°C	(0.0)-(0.1)°C
spring temp. CV	2%	(0)-(6)%
summer mean daily temp.	0.3°C	(0.0)-(0.8)°C
summer max. daily temp.	0.4°C	(0.0)-(0.6)°C
summer max. temp. date	0 days	(0)-(0) days
summer DD25	21%	(7)-(30)%
summer weekly temp. range	2%	(0)-(6)%
summer daily temp. rate of change	4%	(-1)-(20)%
summer 5-day mean temp.	0.4°C	(0.06)-(0.8)°C
summer temp. CV	0%	(0)-(11)%
fall mean daily temp.	0.2°C	(0.0)-(0.4)°C
fall temp. rate of change	3%	(0)-(8)%

Table C.6
[1] Zorn et al. (2012), Kendy et al. (2012), [3] DePhilip and Moberg (2010).

Location	Decision process	management thresholds
Michigan[1]	Advisory council with diverse stakeholders	Defined ecological risk stages and determined that decline past the first two risk stages (10% reduction in characteristic fish abundance) constitutes a significant impact.
Rhode Island[2]	Board of 68 organizations to set management thresholds	A reduction of greater than 50% 7Q10 during the summer was determined to cause a significant loss of fish and bugs. To help meet the needs of water users, they decided to allow more water withdrawal during non-summer seasons when water resources are less strained, giving water users security against limited water supply during the summer months.
Colorado[2]	Technical team produced maps to inform stakeholder discussions.	Defined risk classes based on expert opinion for each ecological variable and produced risk maps that will be used in discussions with stakeholders to set restoration goals.
Susquehanna River Basin[3]	Stakeholders selected ecological indicators, and experts identified full flow needs and recommendations.	Median flow for each of six defined seasons must be preserved within the 45th-55th percentiles, no more than a 20% change is allowed in flow range for May-October, and different low flow recommendations are applied to headwaters (no change in monthly Q75 and monthly low flow range) and larger streams (no change to monthly Q95 and <10% change to low flow range) so that headwater streams have greater protection.
Ohio[2]	Expert and data-guided	Defined significant impacts for different stream types, ranging from a 2% loss of fish in high-quality streams to a 50% loss for streams with no species of interest.

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