COMPARISON OF DIVERSIONS AND IMPOUNDMENTS ON DOWNSTREAM WATER QUALITY AND ECOSYSTEM PROCESSES

by

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The final copy of this thesis has been examined by the signatories, and we find that both the content and the form meet acceptable presentation standards of scholarly work in the above mentioned discipline.

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Comparison of Dams and Diversions on Downstream Water Quality and Ecosystem Processes

Thesis directed by Professor Michael N. Gooseff

ABSTRACT

Water resources infrastructure, such as impoundments and diversions, is expected to expand in order to supply water for an increasing population living in a drier west. River impoundments and diversions, termed discontinuities, are known to disrupt natural gradients and processes, causing a shift in the balance between water rescources supply and riverine ecosystem services. Deep-release reservoirs and their effects on river processes have been well-studied, but few studies quantify the effects of diversions and surface-release impoundments on water quality and ecosystem processes. Due to the effects of seasonal and climatic variations on river hydrographs, it is important to understand the impacts of discontinuities within the same basin and temporals scale. I deployed sensors upstream and downstream of two deep-release impoundments, one surface- release impoundment, one hydroelectric diversion (power diversion), and one irrigation diversion to understand the relative effects of these projects on river dissolved oxygen (DO), DO saturation, temperature, and specific conductivity (SpC) as well as these water quality relationships with discharge and temperature. Impoundments had an

overall greater effect on downstream water quality than diversions. Additionally, water-quality-discharge relationships became weaker downstream of both deep-release impoundments while these relationships did not shift as significantly downstream of the surface-release impoundment and diversions. The irrigation diversion had a larger effect on temperature, DO, and metabolism at lower flows. Combined with model results of downstream temperature changes, this suggests that my results do not capture the largest effect of the irrigation downstream water quality. This combined with a correlation between temperature and DO saturation at low flow suggest that the irrigation diversion has the potential to stimulate ecosystem activity at low flows, especially if they occurred during the predicted warmer temperatures in the future. Adding additional infrastructure to provide water sustainability will exacerbate the compounding effects of these discontinuities within the river network require strategic monitoring and adaptive management in order to maintain ecosystem services provided by rivers.

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

INTRODUCTION

Water resources sustainability, particularly in arid regions, has been accomplished through the modification of river corridors via storage reservoirs and diversions. Such modifications have been shown to disrupt hydrology (Poff et al., 1997, Kondof & Batalla, 2005) , sediment transport (Angelaki et al., 1995, Baker et al., 2011), and thermal regimes (Caissie 2006, Olden et al., 2010) in rivers. These regime shifts are physical changes that cause bottom-up effects in river ecosystems. Water resources infrastructure will likely grow due to the combined pressures on water resources resulting from an increasing population and a hotter, drier future climate (Gleick, 2000). While water sustainability is necessary, river ecosystems play an important role in maintaining water quality, human recreation, and the economy, and intensifying river modification and water stress will adversely affect these aspects of the river system (Böck et al., 2018).

Important ecosystem services provided by rivers include fish habitat and nutrient cycling. Discharge has been termed the “master variable” due to its influence on river temperature and the resulting effects on biogeochemistry and ecology (Poff et al., 1997). Water temperature is tightly coupled with dissolved oxygen (DO) dynamics, and both DO and water temperature are important for fish survival (Chapra, 2008, Matthews & Berg, 1997).

Rivers do not act as pipes, transferring water to the ocean as previously thought; rather they cycle nutrients through various forms, contributing to water quality along the way (Bencala, 1993). One rough measure for this is specific conductivity (SpC) which is a measure for the amount of ions dissolved in the water and can be affected by human activities or geology (Spahr et al., 2000). Primary producers, including aquatic plants and algae, interact with the water quality by taking up nutrients necessary for their energetic functions (photosynthesis and respiration). Heterotrophic organisms, such as microbes and fungi, breakdown the organic matter from primary producers and detritus in order to satisfy their energetic needs (community respiration). Metabolism is the sum of gross primary productivity (GPP) and community respiration (CR). It quantifies organic matter and energy fluxes within a river ecosystem and thus the river's capacity to cycle nutrients. It is also an indicator of ecosystem function because primary producers and heterotrophic organisms are affected by water quality and ecosystem changes. Monitoring changes in metabolism integrates these changes over the stream community (Grace and Imberger, 2006). While people benefit from these ecosystem services provided, balancing them with water resources demand remains a challenge.

Two commonly used hydrologic modifications to sustain water resources are run-of-river diversions and impoundments. Run-of-river diversions are broadly defined, but consist of dams and weirs within a channel that do not create reservoirs. They either divert water from the channel for hydroelectric purposes and restore the channel's flow at one location downstream or divert the water for irrigation and human consumption, allowing water to return to the channel as point or non-point sources downstream. Impoundments create reservoirs upstream of the dam for storage purposes and can be further divided into the categories of surface-release (releasing water to the river downstream from the reservoir surface) or deep-release (releasing water to the river downstream from the reservoir's depths) (Anderson et al, 2015). I will refer to impoundments and diversions together as discontinuities. Few studies have examined the effects of surface-release impoundments and diversions on metabolism (Sabater et *al*., 2018), but the impacts of deep-release impoundments on energy and organic matter fluxes within the river continuum have been well-studied (Vannote et *al*., 1980, Ward and Stanford, 1983).

Ward and Stanford (1983) proposed numerous testable hypotheses for how deep-release impoundments impact rivers at different river orders. Among these, they predicted that deep-release impoundments would decrease water temperature and increase dissolved oxygen concentrations due to the release of water from the cool depths of the reservoir in the summer. In the winter, the deep-release impoundments would increase downstream temperatures due to decreased atmospheric temperatures and this buffered temperatures at the bottom of the reservoirs. They also predicted that temperature variability would be reduced. Stanford and Ward (2001) found that these predictions and hypotheses have generally been supported.

Although large deep-release impoundments play a large role in water resources management, future projects are likely to focus on smaller projects, such as run-of-river diversions and small surface-release reservoirs, since economicallyfeasible, larger water projects and impoundments have already been developed. Since it has been thought that run-of-river diversions are less impactful than impoundments, there are fewer studies on their impacts (Anderson et al, 2015). Run-of-river diversions have been shown to limit fine sediment transport, degrade habitat (Baker et al., 2011), and increase erosion downstream, but few studies have focused on downstream ecological effects (Anderson et al, 2015). Many surface-release impoundment studies have focused on the downstream increased temperatures and their impact on macroinvertebrate and fish communities (Lessard & Hayes, 2003). Sabater et al. (2018) performed a meta-analysis on river regulation and water stress studies and found that flow regulation had the strongest impact on ecosystem processes and that flow removal had the second strongest effect on these processes.

Although studies such as the cascading reservoir continuum concept (Barbosa et al, 1999), have found that multiple deep-release impoundments have compounding effects on the river continuum, to my knowledge, none have examined or compared multiple types of discontinuities within a basin. Since future projects are likely to include multiple, smaller discontinuities as opposed to one larger discontinuity (Anderson et al., 2015), it is important to understand the effects that these have on a river ecosystem.

The Colorado River is an important water source in the west. In order to ensure legal access to this important resource, water rights are enforced. In Colorado and many other western states, “Prior Appropriation” is the law, meaning that the owners of the oldest rights have priority and are guaranteed their water before owners with newer rights. As part of these water rights, the Colorado River is split into the Upper and Lower Basin. The Upper Basin is where the Colorado River forms as a headwater stream. Headwaters are important for water quality because there are more opportunities for nutrient cycling due to increased contact between the water and the streambed compared to the higher discharge reaches downstream (Alexander et al., 2007). In this region, discharge is driven by snowmelt, with flows peaking in April, May, and June and receding throughout the rest of the year. In addition to seasonal trends, the region follows annual climatic variations. These temporal variations highlight the need to compare the effects of diversions and impoundments on water quality within the same time period.

This leads us to ask the overarching question: What are the effects of impoundments and diversions on water quality and ecosystem processes in the Upper Colorado River? Ihypothesized that both residence time within the discontinuity and total change in flow would determine the parameter response- mean and variability- downstream. To test this hypothesis, Iplaced in situ dissolved oxygen, conductivity, and atmospheric pressure sensors upstream and downstream of 5 different discontinuities in the Upper Colorado River: 2 deep-release impoundments, 1 surface release impoundment, and 2 run-of-river diversions. Iexpected that discontinuities would affect specific conductivity, dissolved oxygen concentration, percent oxygen saturation, water temperature, and metabolism differently depending on each discontinuity's differing residence times and flow and temperature regimes (Figures 1-4).

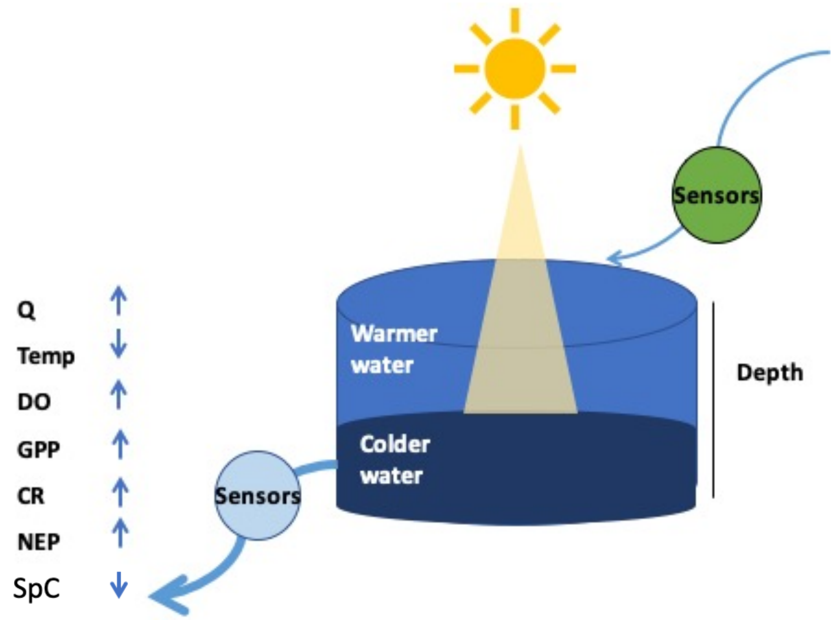


Figure 1: Conceptual schematic of my **predictions** for the studied deep-release reservoirs. The reservoir is deep enough so that the water stratifies due to to attenuated light penetration and resulting thermal gradients (Wetzel, 2001). The water released from the bottom of the reservoir is cold, causing an increase in overall DO concentrations. The reservoir also acts as a settling tank and the increased residence time of the water allows for chemical reactions to occur, decreasing SpC concentrations and increasing water clarity. This results in increased GPP and overall shift towards autrophy (increased NEP).

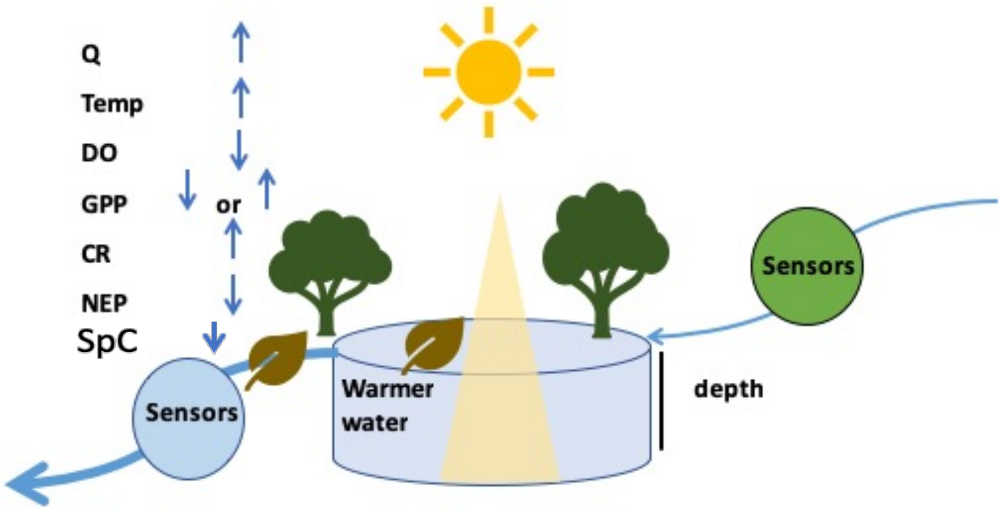


Figure 2: Conceptual schematic of my **predictions** for the studied surface-release reservoir. The residence time of the river is increased so that incoming solar radiation heats the water, and the depth of the reservoir does not allow for thermal stratification for a cooler lower buffer. The water release from the surface of the reservoir then increases the downstream temperature and decreases the overall DO concentration. . The reservoir also acts as a settling tank and the increased residence time of the water allows for chemical reactions to occur, decreasing SpC concentrations and increasing water clarity. This results in increased GPP. However, the smaller size of this reservoir results in a smaller impact than the deep-release reservoir.

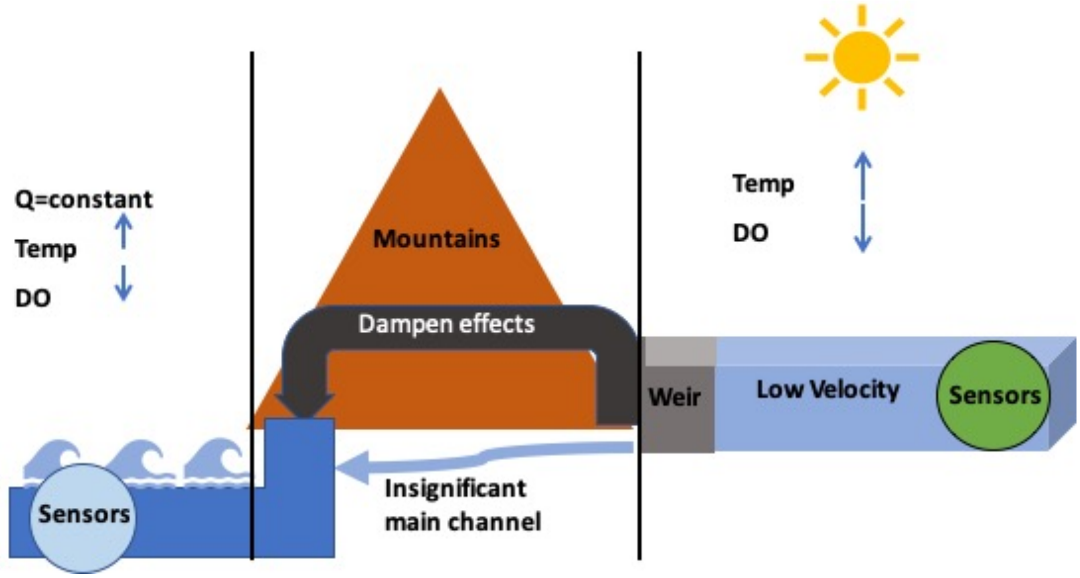


Figure 3: Conceptual schematic of my **predictions** for the studied power diversion. I predict that the effects on the downstream water quality will be affected by three parts of the diversion:upstream of the weir, diversion into tunnels, and re- convergence of the diverted water with the main channel. The weir impounds water within the upstream channel, slowing water velocity,increasing temperature, and decreasing DO. The majority of the river water is diverted into underground tunnels at the weir while a small amount of water remains in the main channel. I predict that the tunnels will dampen the effects of the upstream section on the water and that the reduced volume of water in the main channel would increase in temperature. The diverted water converges with the main channel at the end of the tunnels, causing large turbulence and reaeration. I predicted that the combination of the tunnels with the upstream channel would cause a muted increase in downstream temperature. Although this increase will be small, the high reaeration will result in fast equilibration of DO with saturation concentration, a high correlation with temperature, and an overall decrease in DO.

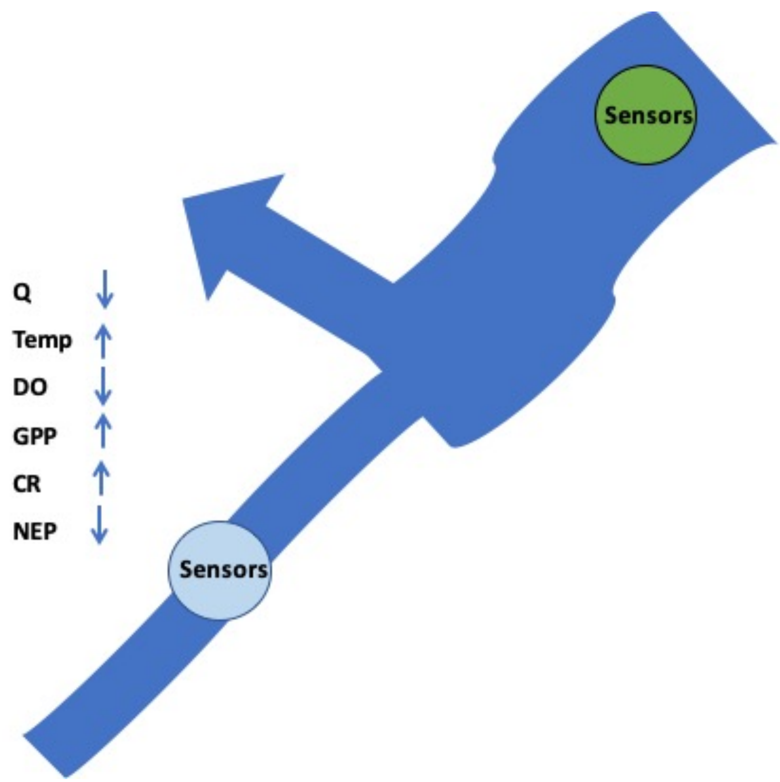


Figure 4: Conceptual schematic of my **predictions** for the studied irrigation diversion. The majority of the river is diverted between the sensor sites. This discharge decrease reduces the thermal buffering of the river, resulting in a temperature increase and overall DO decrease. The discharge decrease also reduces depth, increasing light penetration to the benthos, and enhancing GPP. I also predicted that the increase in ecological activity and slower water velocities would result in higher organic matter deposition on the benthos and thus in higher CR. This combination of GPP and CR changes could result in NEP shift toward autotrophy or heterotrophy.

In addition to affecting the measured parameters, I predicted that the discontinuities would alter the discharge-temperature and discharge- oxygen relationships. To explore this further, I asked the question: how do discontinuities affect water quality relationships with flow and temperature? I predicted discharge would control the temperature regime which would control oxygen dynamics as well as biological activity and that discontinuities with a higher residence time would have a greater impact on these relationships. As biological activity increases, I predict that flow and temperature will have less of an effect on oxygen dynamics. Ialso predict that discharge-amount and variability-will control specific conductivity. Understanding the effects of different discontinuities on water quality and ecosystem processes, as well as the controls on these parameters, within a semi-arid basin will provide management guidance for water operators and regulators in order to balance the needs of water demand and ecosystem health.

METHODS AND SITE DESCRIPTION

In order to understand the effects of impoundments and diversions on water quality and ecosystem processes, I focused this study within a portion of the Upper Colorado River watershed (area=9,800 mi^2) from Grand Lake, CO to Grand Junction, CO. This section of the river contains four different discontinuities including two deep-release impoundments (Granby and Williams Fork), one surface-release impoundment (Windy Gap), and two run-of- river diversions (power and irrigation diversion) (Figure 5).

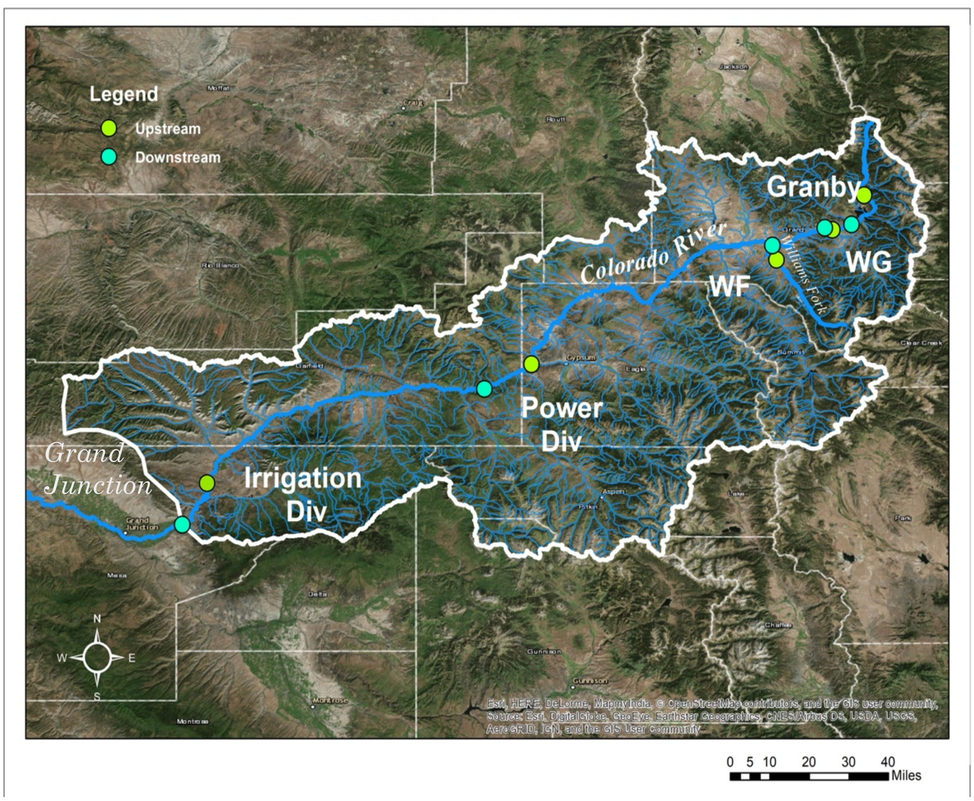


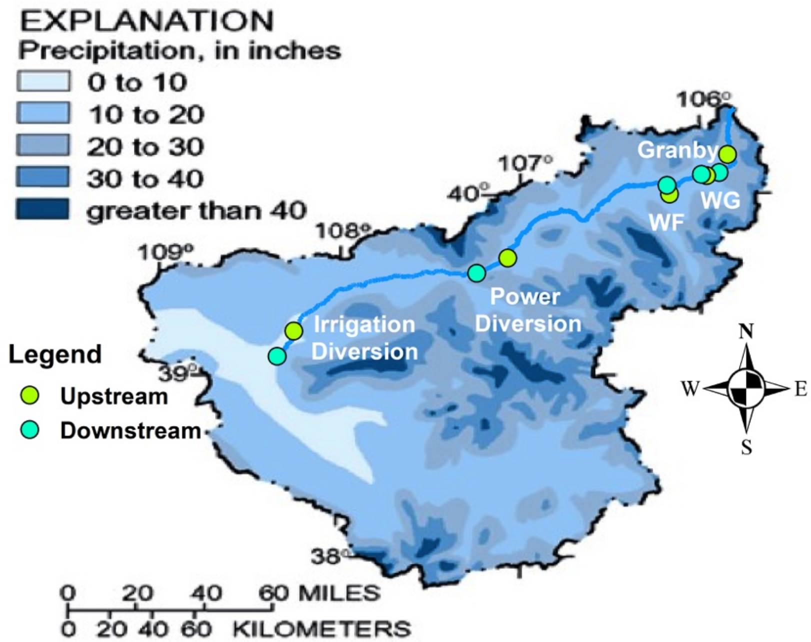
Figure 5: Map of Colorado River watershed and study sites above the most downstream site.Study sites include up and downstream of the Lake Granby complex (Granby), Williams Fork Reservoir (WF), Windy Gap Reservoir (WG), the Shoshone Power Plant (power diversion), and the Grand Valley Diversion (irrigation diversion) Flow travels from northeast to southwest.

These discontinuities have been developed over the last century to meet water demands on both the west and east side of the continental divide.

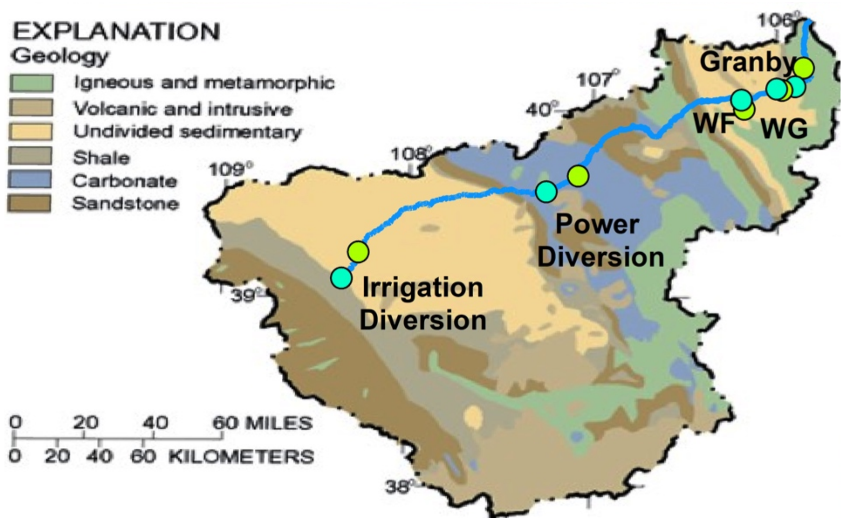
Differing geologic and hydrologic characteristics contribute to water quality and vegetative gradients along this along this section of the Colorado River. The geological composition transitions from primarily igneous and metamorphic, sedimentary, and sandstone to a section that consists of largely carbonate, which is more erodible and contributes greatly to specific conductivity levels in the river. A greater proportion of sedimentary rock in the western basin also contributes to increased specific conductivity as the river flows west. Additionally, as the river flows from ~4,270 to ~1370 m, the climate becomes more arid. A vegetative gradient from evergreen to more sparse vegetation follows this elevation drop and aridification (Spahr et al., 2000) (**Error! Reference source not found.**).

Dams and Diversions

I studied five discontinuities representing four different types of discontinuity. I treated each discontinuity as a control volume around the entire water project, including channel modifications, to see their impact as a whole on the Colorado River. The impoundment characteristics are summarized in (Table 1) and site distances from each discontinuity in (Table 2).



a)



b)

Figure 6: Precipitation (a) and geologic (b) maps of the study area. (Spahr et al.,2000)

Table 1: Table 1: Characteristics of impoundments (“Water Projects”,n.d.[a], “Williams Fork Reservoir”,n.d. [b])

Characteristic	**Granby**	**Windy Gap**	**Williams Fork**
Type	Deep Release[a]	Surface Release[a]	Deep Release[b]
Surface Area (ac)	8597[a]	106[a]	1860[b]
Maximum Capacity (acre-ft)	557,112[a]	445[a]	96,822[b]
Max Depth (ft)	221[a], dam height=298[a]	25[a]	>52[b], dam height=273[b]
River Order	3[a]	5[a]	4[2]

Table 2: Distances from sensor sites to discontinuity (measured on ArcGIS)

Site	**Distance from discontinuity to site (km)**
DR1 Up	0.56
DR1 Down	6.7
SR Up	0.93
SR Down	2.65
DR2 Up	0.16
DR2 Down	4.83
Power Diversion Up	13.91
Power Diversion Down	3.11
Irrigation Diversion Up	9.8
Irrigation Diversion Down	1.89

Granby (Deep-Release)

The Granby complex includes Shadow Mountain Reservoir and Lake Granby (**Error! Reference source not found.**). These water bodies are part of the Colorado Big Thompson Project, and water has been stored in Lake Granby since 1949. Lake Granby stores water that is eventually pumped to Shadow Mountain Reservoir and is transported via the trans-mountain diversion, the Alva B. Adams diversion, to the eastern side of the Continental Divide. The Colorado River enters upstream of Shadow Mountain Reservoir and flows out from the bottom of Granby Dam (~65-200 ft). (“Lake Granby”, n.d.)

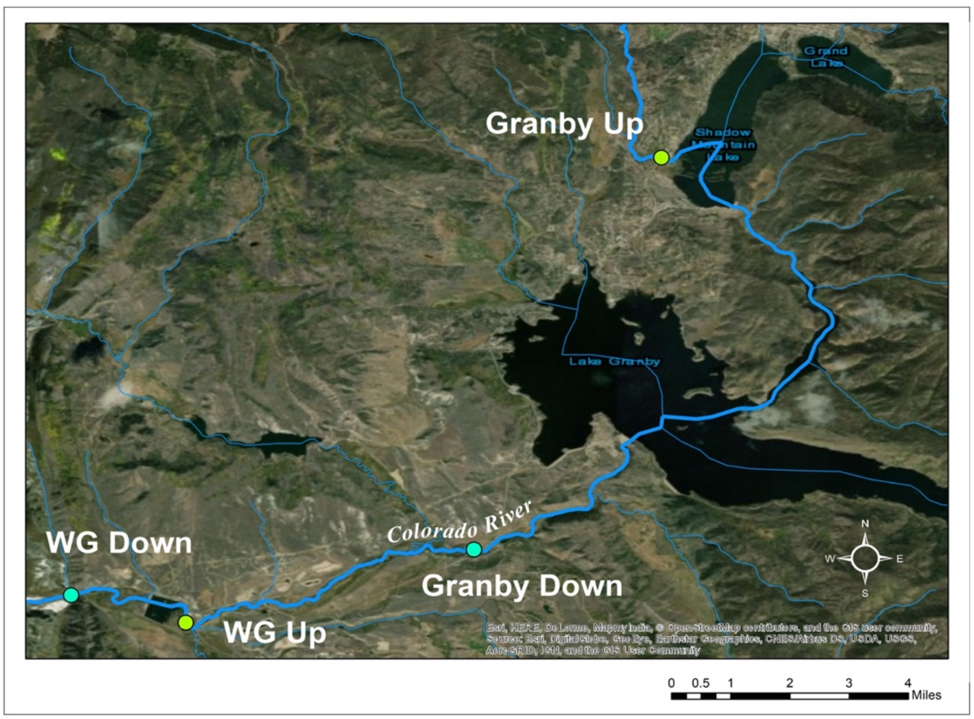


Figure 7: Map of Granby and WG. Flow travels from northeast to west. The upstream site is 0.15 km downstream from the confluence with the Fraser River.

Windy Gap (Surface-Release)

Windy Gap Reservoir was completed in 1985 and is also part of the Colorado Big Thompson Project. The project consists of a diversion and a small storage reservoir. It diverts water from the river and pumps it to Lake Granby, but has junior water rights to other rights in the watershed, so does not always pump water during dry years ("Northern Water", n.d.). The water is then either diverted to the eastern side of the Continental Divide or travels back to the Colorado River. The water that is not diverted to Lake Granby flows over the surface of the dam into the Colorado River. For this reason, it is characterized as a surface release impoundment (depth=25 ft) ("How Windy Gap Works", n.d.).

Williams Fork (deep-release)

Williams Fork Reservoir is on the Williams Fork River (Figure 8). The Williams Fork River flows out of the bottom of the Williams Fork hydroelectric dam and into the Colorado River downstream of Windy Gap. Williams Fork was built in 1959 and releases water and generates electricity for the western slope when Denver Water diverts water from an upstream site located at Steelman Creek ("Williams Fork Reservoir",n.d.).

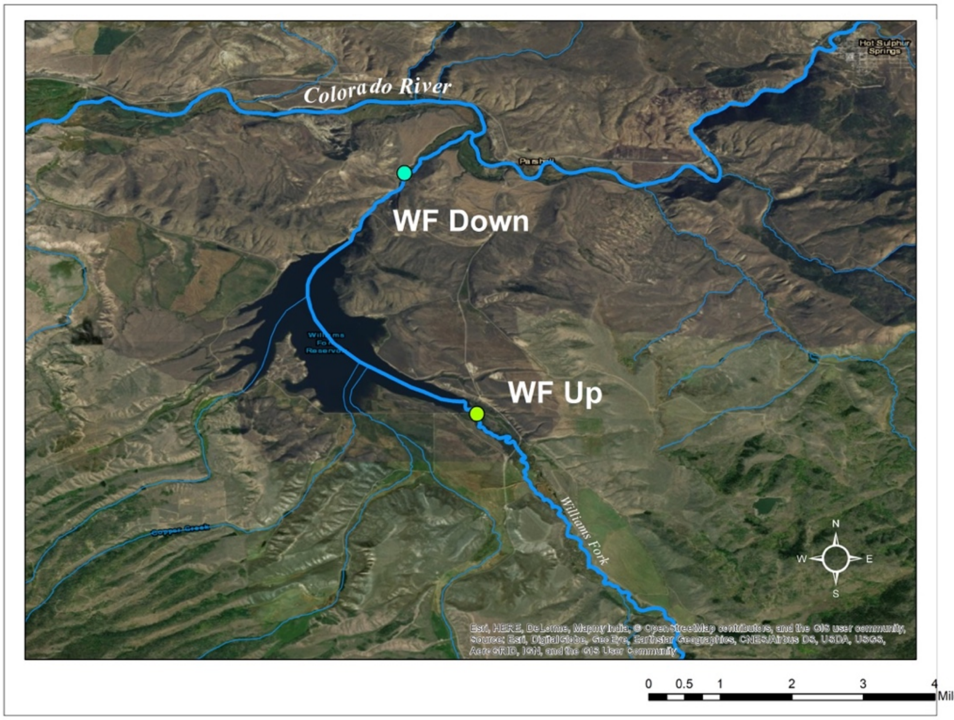


Figure 8: Map of Williams Fork and sensor sites. Flow travels from south to north.

Power Diversion

The power diversion is the Shoshone Powerplant which began operation in 1909 and has the oldest, largest, non-consumptive water right (1,250 cfs) in this part of the Upper Colorado Basin (**Error! Reference source not found.**).

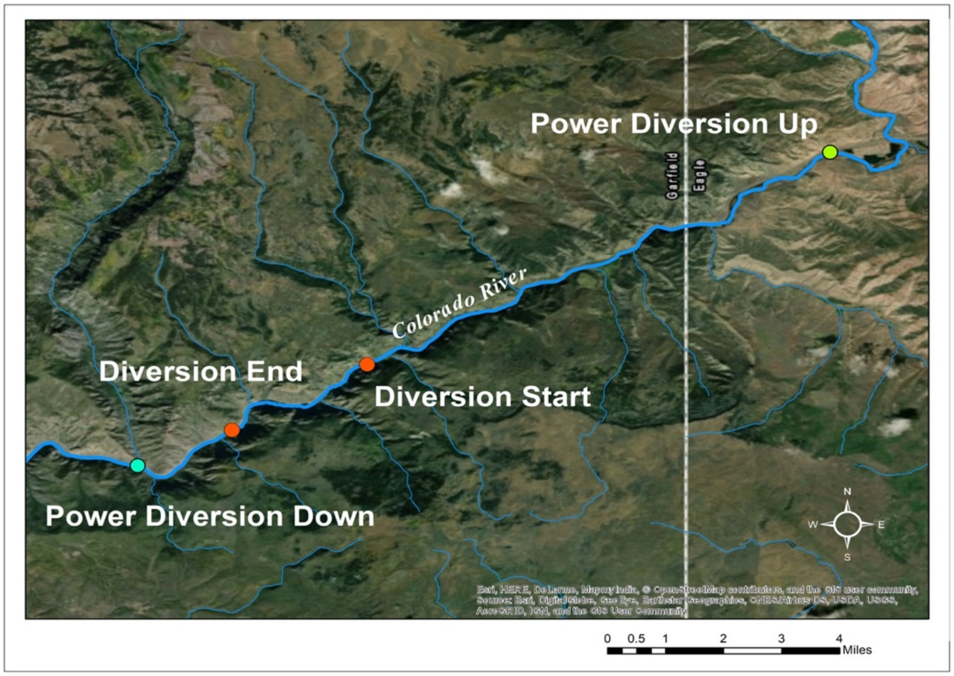


Figure 9: Map of power diversion. Flow travels from east to west. The upstream site is 2.30 km downstream of the confluence with the Eagle River.

The Shoshone Power Plant was one of the first hydroelectric plants to get its electricity from the flow of a river and not from a reservoir (Colorado's Decision Support Systems, 2007). Xcel energy owns the powerplant, and it generates 15 MW. A weir impounds the river without overflowing the main channel, slowing flow velocity approximately from the weir to the upstream site and diverts river water into tunnels. The tunnels exit through electricity-generating turbines at the diversion end and enter back into the Colorado River, creating a turbulent reach known as the Shoshone Rapids. When flow is sufficient, excess water remains in the channel, creating a small river until it rejoins the majority of flow at Shoshone Rapids. When discharge is less than the 1,250 cfs water right, the plant is able to divert the entire river. Although the plant has first right to this water, Denver Water has negotiated with Xcel Energy for some of the water and compensated for lost revenue during dry years, resulting in reduction of power plant use of the water (Sloan, 2004).

Irrigation Diversion

The Irrigation Diversion consists of 2 different main structures: the roller dam, part of the Grand Valley Project, at the upstream end of the diversion and the Grand Valley Irrigation Canal at the downstream end (**Error!**

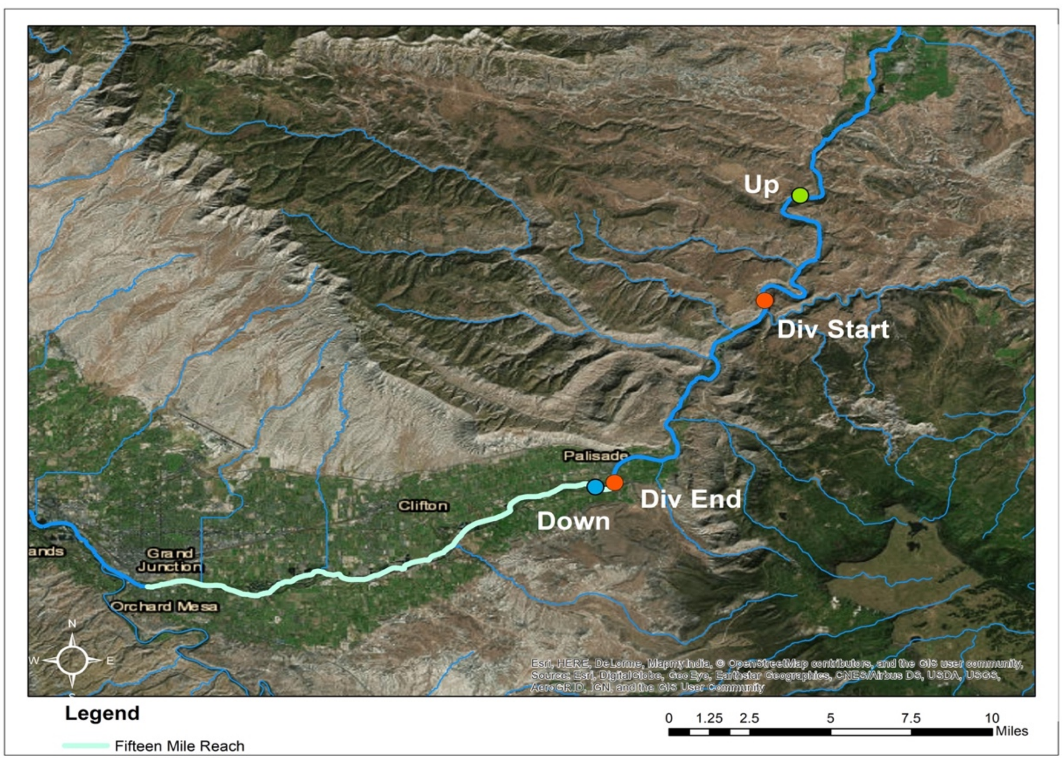


Figure 10: Map of the irrigation diversion. The Colorado River flows from east to west.

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The water is used for irrigation, hydroelectric power, and cooling water, and the total amount of water rights at the upstream end is about 2,260 cfs. Since these water rights are senior to most in the basin, this flow is assured at the upstream end of the diversion when users demand it. Most of this water is used most years, some of it is non-consumptive (hydroelectric water), and some of it returns to the river as run-off from irrigated fields. The irrigation diversion is above an important reach for fish habitat and connectivity within the Upper Colorado Basin called the 15 Mile Reach which has the potential to dry up (Colorado's Decision Support Systems, 2007).

Eulerian Sensor Measurements

Eulerian measurements quantify a parameter at one location over time and provide an aggregate measurement of the reach upstream of the sensors. In order to compare the differences in DO, DO saturation, temperature, and SpC in the reaches above and below each discontinuity, I collected continuous data upstream and downstream of each discontinuity. I deployed Onset DO, conductivity, and pressure sensors at each site during May, June, and July 2018. I selected sensor locations based on relationship to discontinuities (upstream or downstream), proximity to USGS stream gauges, and ability to access streambed. To quantify upstream and downstream differences in discharge and understand water quality relationships with discharge, I downloaded discharge data from these USGS sites. The Granby upstream and power diversion downstream sites were greater than a mile from the USGS gauges, so I used stage loggers at the sensor sites to calculate the lag time and adjust the timing of discharge at each site.

To quantify residence time, I used different approaches for impoundments and diversions. For impoundments, I assumed that the reservoir was at steady state and calculated residence time as the volume of the reservoir divided by the downstream discharge (Dingman, 2015). Since I did not have time-series data of reservoir volume for the Williams Fork site, I used the design capacity volume for all sites. Since it was a drier year, I used the approximate capacity of Williams Fork to estimate how full each reservoir was (66%) and reduce the design capacity volume. The residence time of the power diversion was the lag time between the upstream and downstream sites. The irrigation diversion residence time was assumed to be insignificant due to no significant observed pooling upstream of the first dam.

To ensure quality data collection, I protected the sensors and cleaned the sites bi-weekly to monthly. To protect the sensors, I encased them within a 4-inch diameter, 18-inch long PVC tube and secured it to the streambed parallel to the direction of flow at each site. This housing filled up with sediment or became bio-fouled at some sites, which resulted in data not representative of the water column. During my site visits, I recorded the state of the housing and sensors and cleaned them. To ensure that the data quantified the reach, I removed data from periods that represented the microenvironment of the PVC or when the sensors were out of the water.

Dissolved oxygen measurements were sensitive to biofouling and sedimentation, displaying erratic patterns or anoxia. Low conductivity data was indicative of the sensors being out of the water and would drop to values close to zero. I compared this data with field notes and removed data that was not representative of the water column. This, along with differing deployment and collection dates, resulted in different date ranges for each discontinuity (**Error! Reference source not found.**. **Error! Reference source not found.**).

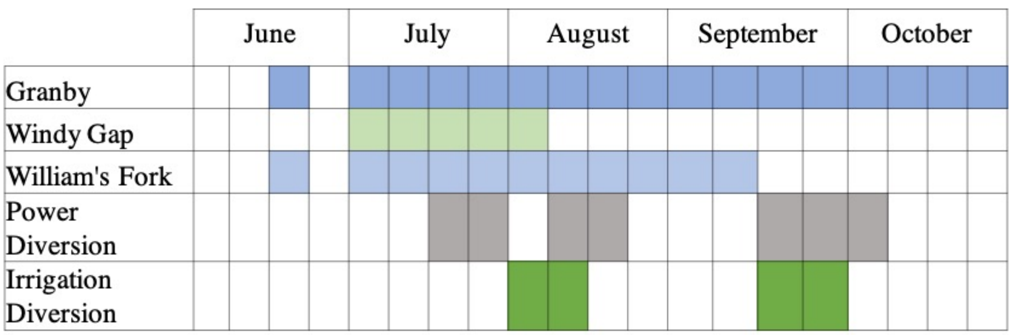


Figure 11: Visualization of data used in analysis after raw data was filtered.

Table 3: Actual dates used in analysis for each discontinuity.

Site	**Deployment Dates**
Granby	6/22/18 17:45-6/25/18 23:45, 7/3/18 16:15-11/7/18 20:00
Windy Gap	7/3/18 18:15-8/12/18 11:15
Williams Fork	6/22/18 15:45-6/30/18 15:30, 7/03/18 17:30-9/23/18 13:45
Power Diversion	7/23/18 18:45-7/29/18 22:45, 8/10/18 15:15-8/22/18 19:30, 9/21/18 21:30 - 10/13/18 13:30
Irrigation Diversion	7/29/18 13:45-8/14/18 21:00, 9/21/18 19:15-10/04/18 1:30

To compare trends across similar time periods, each discontinuity was separated into “seasons”. The power and irrigation diversions were separated into late summer and fall seasons. Windy Gap only had a summer season. I also classified Williams Fork as only having a summer season since it ended at the start of the fall season for the power and irrigation diversions. Granby had the longest deployment.

Granby spanned three distinct seasons (Figure 12): the trailing end of snowmelt (6/23/18-8/2/18, melt season), summer baseflow (8/3/18-9/21/18, summer), and fall (9/22/18-11/02/18). Fall was selected as 9/22/18 to correspond with the power and irrigation diversion seasons. Upon water temperature analysis, I observed that the upstream and downstream temperatures began equilibrating on 9/20/18 thus further justifying 9/22/18 as a good start time for the fall season.

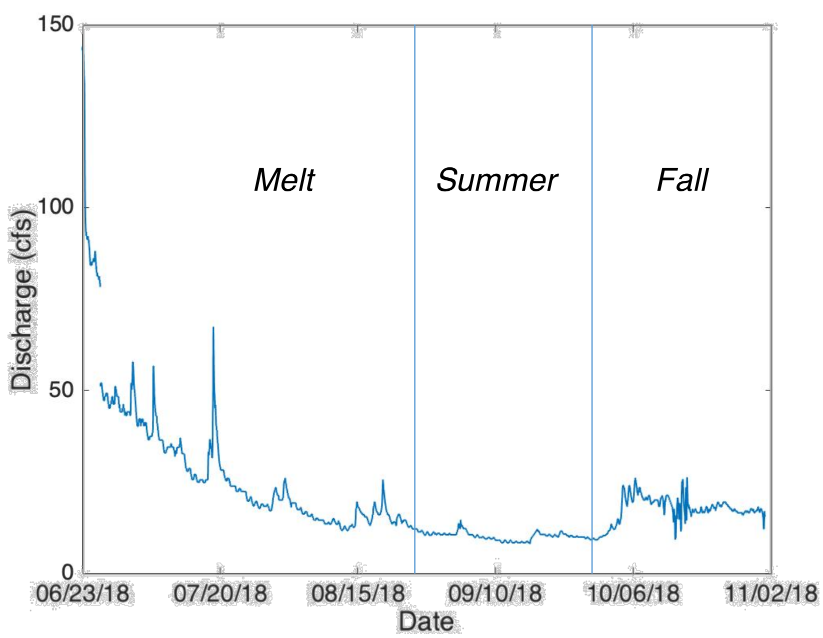


Figure 12: Granby hydrograph during sensor deployment and resulting seasonal divisions.

Reach Temperature Change Modeling

For sites where the temperature increased downstream, I modelled steady state temperature at the downstream end of the reach using the Stream Segment Temperature Model (SSTEMP) (Bartholow, 2002) to determine if the temperature increase was due to the discontinuity or due to the site being downstream. Following Bartholow (2000b), Iused measured values from the field as well as values from regional sources to represent each stream segment. Iran the model three times for each reach representing the observed daily minimum, mean, and maximum upstream to downstream temperature changes for the entire deployment (three model runs per stream). I then compared the daily means and ranges of each run to the measured daily amplitudes for each reach. Since I couldn't use the observed downstream discharge at the irrigation diversion for these model runs, I assumed that downstream discharge was equal to the upstream discharge. Additionally, I modelled downstream temperature for a case (case 1) in which the downstream discharge was what equal to observed discharge and also a case (case 2) for which the downstream discharge was equal to the observed discharge plus 50% of the diverted water.

Metabolism Modeling

I used a single station model in streamMetabolizer (Appling et al, 2017) to model the impacts of discontinuities on stream metabolism. streamMetabolizer is based on the open diel oxygen method (Odum, 1956) and uses bayesian modeling to calculate

GPP, CR, and the reaeration constant based on DO changes at every timestep. I used the DO and atmospheric pressure probes at all sites to measure DO changes. I calculated light using the *calc_light* function in streamMetabolizer. This function uses the site coordinates and a maximum PAR value to model irradiance at every timestep. I used a maximum value of 2600 $\mu m^{-2} s^{-1}$ which I collected from a HOBO Lux sensor deployed near the irrigation diversion during September 2018. I calculated depth using the *calc_depth* function in streamMetabolizer using discharge as an input.

I ran the models until they reached convergence. I used streamMetabolizer's observation error $\hat{R}$, process error $\hat{R}$, and log posterior density $\hat{R}$, to assess convergence following Appling et al. (2018). The model returns daily mean and standard deviations (SD) for GPP, CR, and reaeration.

To examine metabolism from another perspective, I used the metrics as found in Mulholland et al. (2005). This included maximum daily DO deficit for CR and the amplitude of the daily DO deficit for GPP. Since reaeration estimation is so difficult, I also examined the discharge-reaeration relationship to see how well reaeration was related with discharge

Lagrangian Measurements

To understand the longitudinal effects of the irrigation diversions on the downstream thermal regime at a higher spatial resolution, we collected Lagrangian profiles as in Hensley et al. (2014). On 07/12/18, we floated two reaches downstream of the irrigation diversion: the 15 Mile Reach and a section above the Gunnison River confluence. Since the 15 Mile Reach was shallow, we used smaller “duckie” boats and a larger cataraft for the section downstream of the Gunnison. To characterize spatial temperature profiles, we attached a CS-109 temperature probe (Campbell Scientific, Logan Utah) collecting 15-second interval data to the boat while we travelled down the river and collected concurrent GPS measurements with a GPS16-HVS (Campbell Scientific, Logan Utah). During stretches where the water was too shallow and we risked damaging the sensors, we took them out and noted the times of removal. During data processing, we removed these times from the data stream.

Irrigation Diversion Conceptual Model

To understand how the irrigation diversion affected the thermal buffering capacity of the river and how this affected the thermal regime of the sensitive 15 Mile Reach, I combined mydata and USGS stream gauge data into a mass and energy balance model (Figure 13). This was done to 1) estimate temperatures at the end of the reach before the Gunnison restores flow to the Colorado River (point BG) and 2) model the importance of the diffuse heat flux to the heat flux at point BG for the observed flow conditions.

The first step for both of these objectives was to quantify the diffuse inflow coming into the fifteen mile, Q_{15}, reach using a discharge mass balance:

$$Q_{15} = Q_{SL} - Q_G - Q_D$$

where Q_{SL} was the discharge measured at the State Line stream gauge, Q_G was the discharge measured at the Gunnison stream gauge, and Q_D was the discharge

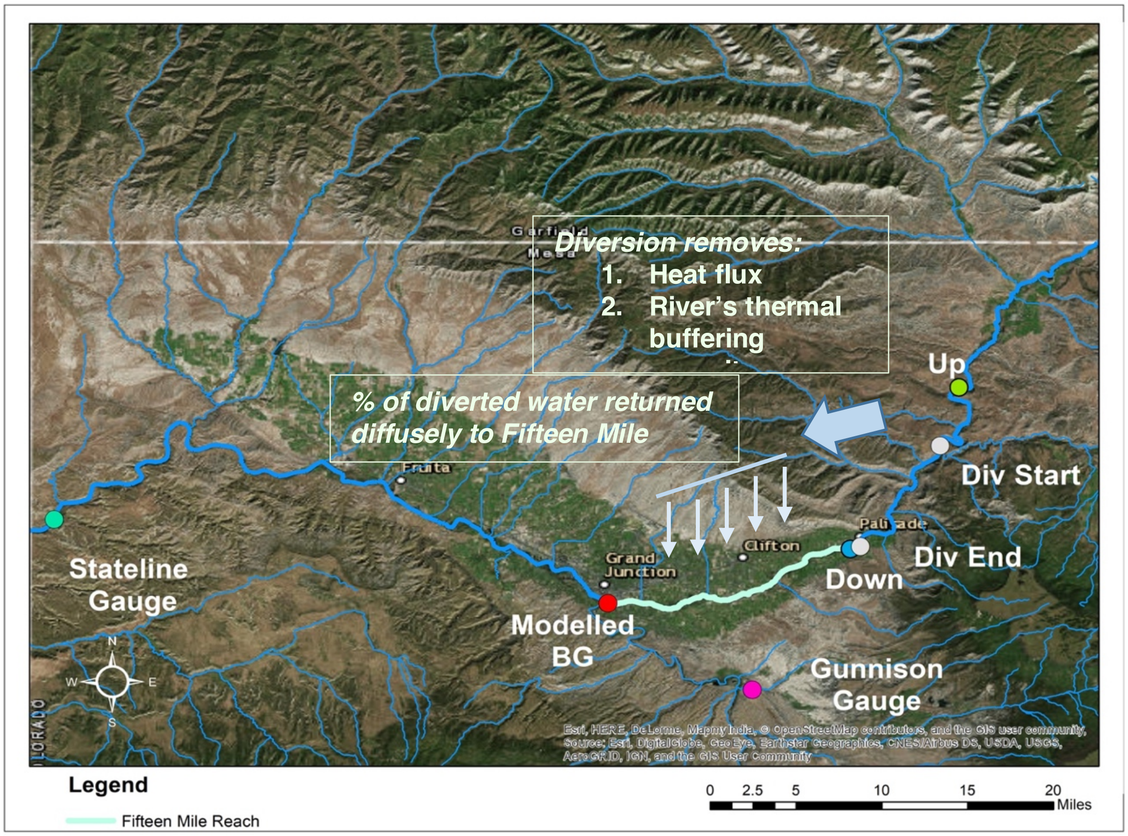


Figure 13: Map of Irrigation Diversion, conceptual model of longitudinal effects of diversion on thermal regime of Colorado River, and gauges used for model. Colorado River flows from East to West. USGS discharge gauges are located concurrently with my up and downstream sensor locations.

measured at the downstream discharge gauge. Since it was late summer and early fall and the Colorado River is a snowmelt dominated hydrograph, I assumed that incoming tributary streamflow between these points was insignificant to the mass balance.

To model the temperature at point BG, I used an energy balance at a point just after the point of mixing of the Gunnison and the Colorado River. Since the State Line gauge is located far away from this point and has warmer temperatures than would be experienced at that point, I corrected the State Line temperatures using data I had collected from our Lagrangian profiling. During our Lagrangian profiling, I collected temperature at the point of mixing. Since our data was timestamped, I was able to reference this data with the state line temperature data and use a proportion of 0.95 to adjust the State Line temperature. The temperature at BG, T_{BG}, was then calculated with the following equation:

$$T_{BG} = \frac{0.95T_{SL}Q_{SL} - Q_G T_G}{Q_{BG}}$$

Where T_{SL} was the temperature at the State Line stream gauge, T_G was the temperature at the Gunnison stream gauge, and Q_{BG} was the discharge at point BG and was calculated by the following equation:

$$Q_{BG} = Q_{15} + Q_D$$

Heat flux was calculated by multiplying the discharge at each site by the temperature. I calculated this representative heat flux, H_{15}, as proportional to the percent of diffuse inflow over discharge diverted as follows:

$$H_{15} = \frac{Q_{15}}{Q_U - Q_D} \cdot (T_U Q_U - T_D Q_D)$$

Where Q_U and T_U were the upstream discharge and temperature, respectively. I then analyzed the contribution of the diffuse inflow heat flux to the heat flux at BG, H_{BG}, as H_{15}/H_{BG}.

Water Quality-Discharge and DO-Saturation-Temperature Relationships

To understand how discontinuities affected the relationships between flow and water quality relationships and the relationship between temperature and DO saturation, I quantified the relationship between average daily discharge and average daily temperature, DO, DO saturation, and SpC. I used the same seasonal distinctions as for the water quality concentration comparisons, but split Williams Fork into two categories due to a large diversion that shifted water quality-discharge relationships (see below).

On 6/29/18, the water level dropped at the upstream WF site, and mysensors were no longer submerged in the river until I checked on them on 7/4/18. This resulted in a data gap. This water level drop was a result of a diversion upstream in the headwaters at Steelman Creek on 6/29/18 (Figure 14) and resulted in different relationships between variables and discharge before and after the diversion. I therefore split the WF data into before and after diversion to compare water quality relationships with both temperature and discharge. Since this time period was only eight days of the deployment before the shift, I did not split it for the overall concentration comparisons.

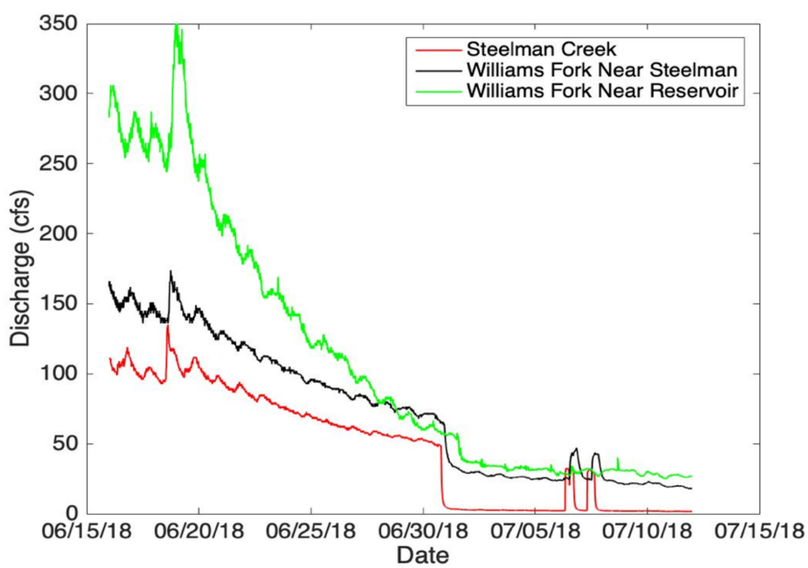


Figure 14: Discharge at Williams Fork upstream site and two sites upstream demonstrating the large diversion on 06/29/18

Statistics

Sign tests were used to determine significance of upstream and downstream water quality differences over the seasonal deployments. The inter quartile range (IQR) was used to determine the variability of the data. Sen-Theil regressions were run to determine the linear relationships between flow and all water quality parameters as well as temperature and DO saturation. Spearman rank coefficients were used to correlate upstream discharge with downstream discharge. I used a significance level of 0.05 for all tests. Statistics were performed in R (R Core Team, 2014)

RESULTS

In order to examine the effects of discontinuities, I compared measurements of the upstream reach to the impacted downstream reach. I did this by comparing aggregate measurements over each season and examining daily variation with temporal plots.

Discharge

Each discontinuity had a distinctive downstream discharge shift (Figure 15). Downstream discharge trends did not change seasonally (Figure 16).

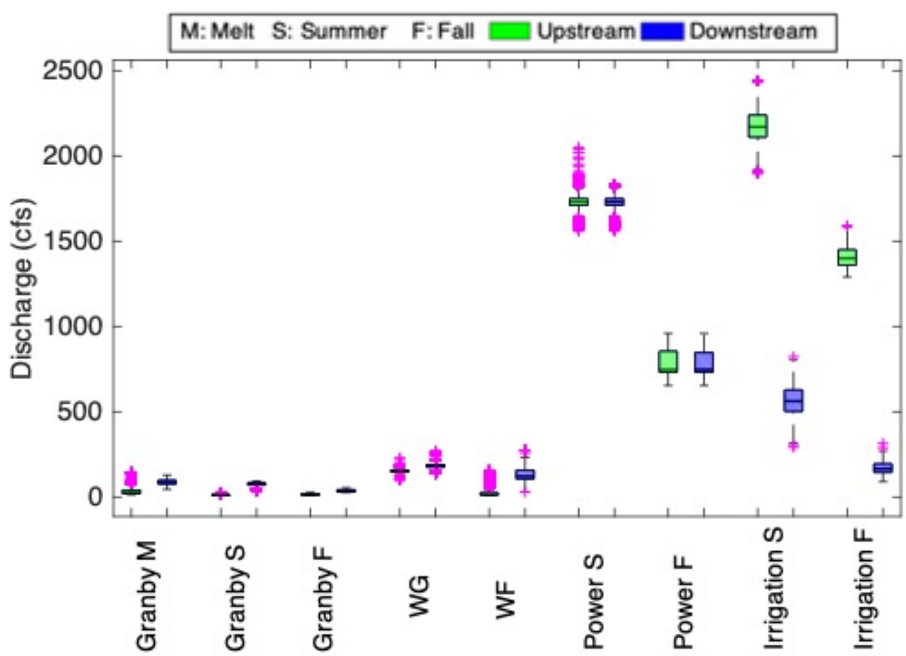


Figure 15: Temporal boxplots of discharge upstream and downstream of each discontinuity.

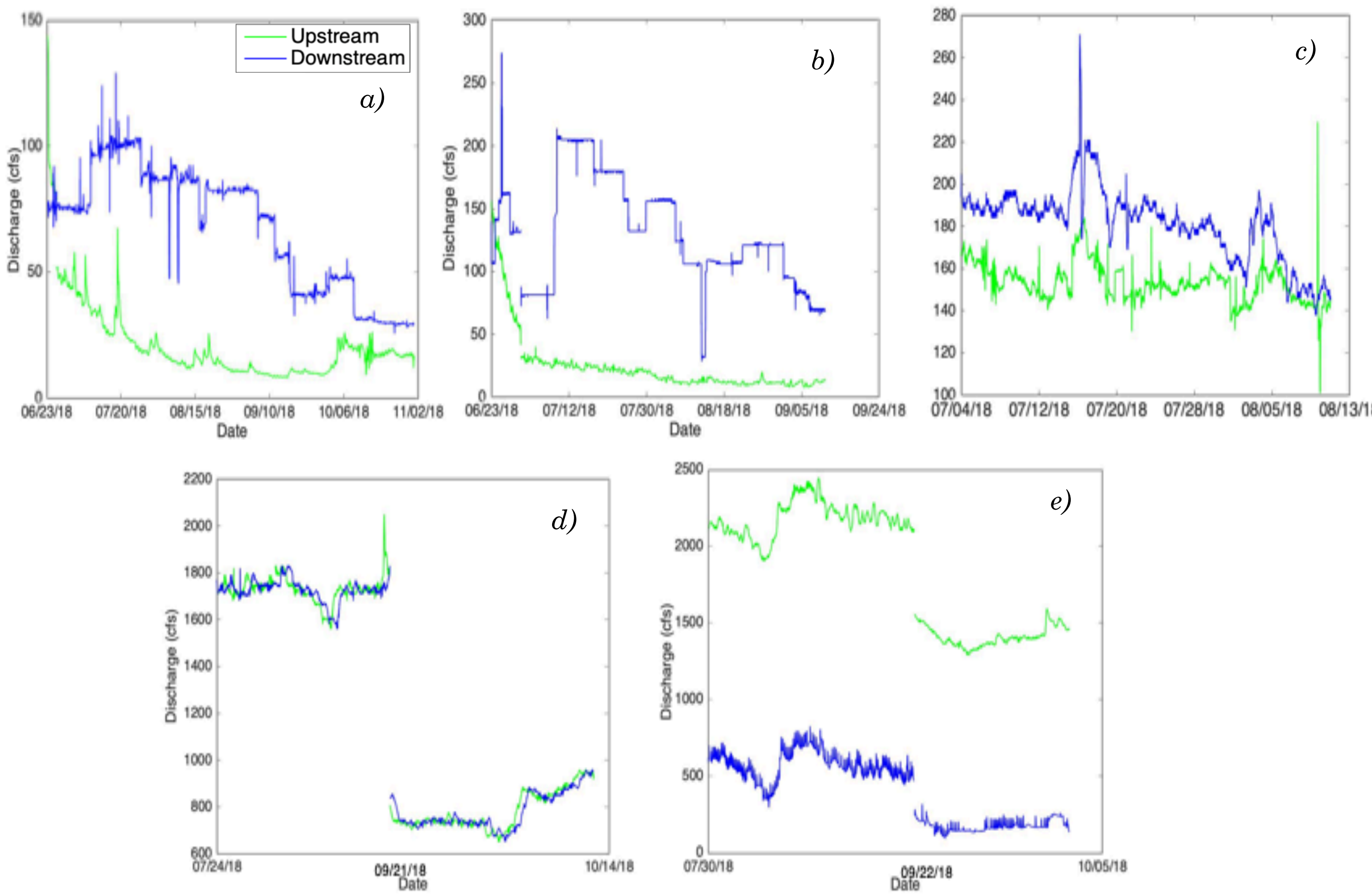


Figure 16: Temporal discharge measurements for a) Granby, b) Williams Fork, c) Windy Gap, d) Power Diversion, and e) Irrigation Diversion

Granby (deep-release)

Downstream median discharge increased, though to different degrees, while the interquartile range (IQR) changes differed in direction during all three hydrological seasons. Median discharge increased from 26.3 to 86.5 cfs, a 229% increase ($p<0.001$), during the melt season, from 11 to 82.3 cfs, a 648% increase ($p<0,001$), in the summer, and from 17 to 32.6 cfs, a 92% increase ($p<0.001$), in the fall. The IQR increased from 22 to 23 cfs, a 5% increase, in the melt season, from 3.65 to 11.9 cfs, a 226% increase, in the summer, and increased from 8.2 to 12.6 cfs, a 54% increase, in the fall. The hydrograph at the upstream site is not affected by any upstream impoundments and is therefore representative of a snowmelt pattern for this region, whereas the downstream hydrograph clearly displays step-changes due to release management decisions. Downstream discharge did not correlate with upstream discharge ($\rho=0.29$).

Williams Fork (deep-release)

Median discharge below WF increased from 16.7 to 121 cfs, a 625% increase ($p<0.001$), and the IQR increased from 13.3 to 51 cfs, a 283% increase. The hydrograph above WF had a similar pattern to that above Granby indicative of a typical snowmelt hydrograph, whereas the downstream hydrograph (also similar to Granby) displayed several step changes. At this site, the downstream discharge timeseries did not correlate strongly to the upstream discharge timeseries ($\rho=0.39$).

Windy Gap (surface-release)

Median discharge below Windy Gap increased from 153 to 185 cfs, a 21% increase ($p<0.001$), and the IQR increased from 11to 15 cfs, a 32% increase. The downstream discharge tracked upstream discharge more closely than above and below Granby and WF ($\rho=0.55$).

Power Diversion

I calculated a 16 hour lag time between hydrograph peaks between upstream discharge and the downstream stage logger. Since flow is restored before the downstream site, discharge should equal upstream and downstream over a time period larger than this lag time. I did not calculate statistics for this because it was a diversion design assumption.

Irrigation Diversion

Median discharge and the IQR decreased downstream during both seasons, but the IQR decreased more in the fall (90 to 50.8 cfs) than in the summer (130 to 126 cfs). The median discharge decreased downstream from 2170 to 562 cfs, a 286% decrease ($p<0.001$), in the summer and from 1400 to 165 cfs, an 2000% decrease ($p<0.001$), in the fall. Downstream discharge correlated strongly with the upstream discharge ($\rho=0.91$).

Summary of Discharge for all Discontinuities

Discharge increased below all impoundments and decreased below the irrigation diversion. The power diversion discharge lagged the upstream discharge. Both deep-release reservoirs displayed changes from natural hydrographs at the upstream site to resembling an altered hydrograph with several step-changes. The discharge measurements below the diversions and surface-release impoundment resembled and correlated with the upstream discharge measurements. It appears that the low correlation between upstream and downstream discharge for the deep-release impoundments is partly due to a high residence time (but more likely caused by reservoir operations) and that the high correlation between upstream and downstream discharge for the irrigation diversion is due to a low residence time.

Residence Time

The residence times for Granby, Williams Fork, and Windy Gap were on the order of 100-1000, 100-1000, and 1 day(s), respectively. The residence time for the power diversion was the lag time, 16 hours, and I assumed that the irrigation diversion residence time was sufficiently short (i.e., <0.5 day).

Temperature

Temperature increased longitudinally from Grand Lake to Grand Junction (Figure 17).

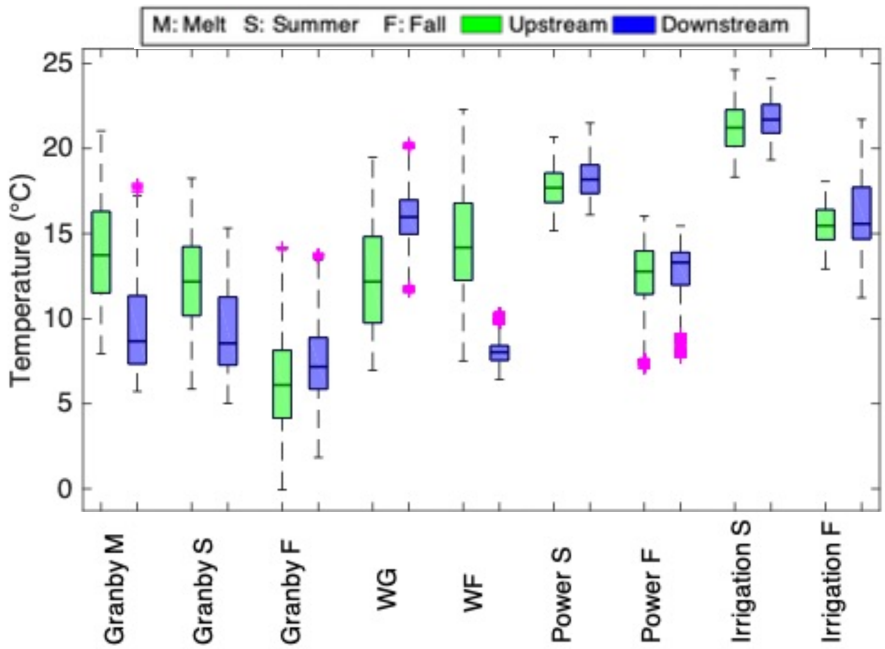


Figure 17: Temporal boxplots of temperature upstream and downstream of each discontinuity.

Despite the presence of the discontinuities, water temperature varied diurnally. Each discontinuity had an effect on water temperature, though effects at each site varied with type of discontinuity and season.

Granby (deep-release)

From above to below Granby, median temperature decreased from 13.73 to 8.67 °C, a 58% decrease ($p<0.001$), during the melt season and from 12.17 to 8.55 °C, a 42% decrease ($p<0.001$), in the summer and increased from 6.1 to 7.17 °C, an 18% increase ($p<0.001$), during the fall. The IQR decreased from 4.79 to 4 °C, a 20% decrease and stayed stable in the summer, 4.05 to 4.00 °C, a 1% decrease, and decreased in the fall, 3.98 to 3.02 °C, a 32% decrease (**Error! Reference source not found.**). The downstream temperature was lower than the upstream temperature during the melt season and summer, but during the fall, the upstream temperature began to decrease, equilibrate with the downstream temperature for a time period, and then drop below the downstream temperature. This temperature trend occurred while the downstream discharge was decreasing (Figure 18)

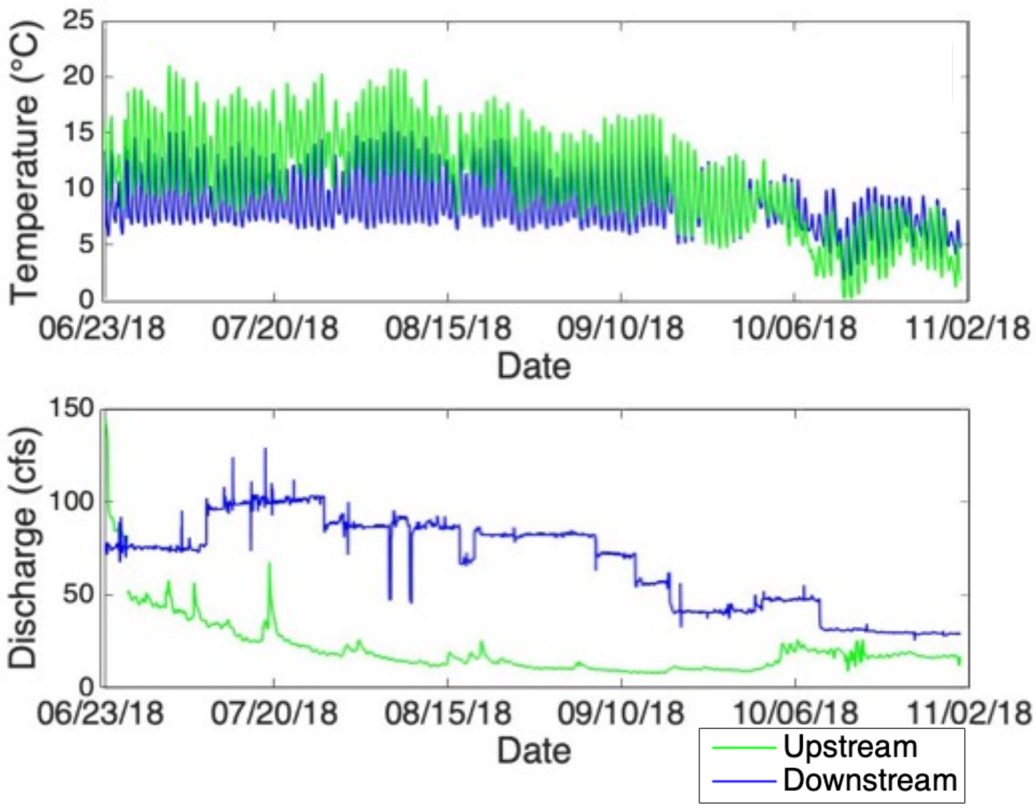


Figure 18: Temporal measurements of temperature and discharge for Granby

Williams Fork (deep-release)

The downstream temperature remained more stable, both in daily range and daily variation, throughout the deployment. From above to below Williams Fork, median temperature decreased from 14.18 to 8.03 °C, a 77% decrease, and the IQR decreased from 4.54 to 0.86 °C, an 428% decrease (Figure 19).

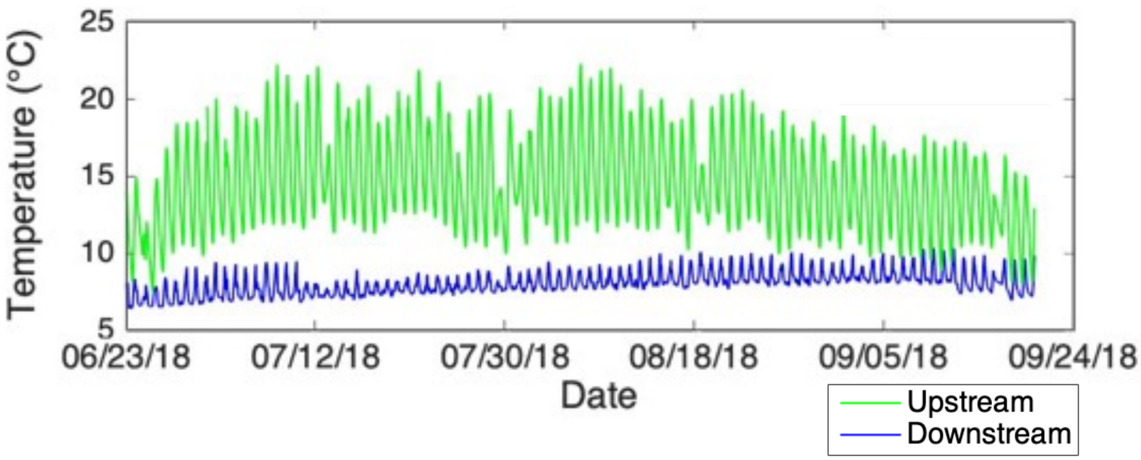


Figure 19: Temporal measurements of temperature at Williams Fork.

Windy Gap (surface-release)

From above Windy Gap to below, median temperature increased from 12.17 to 15.97 °C, a 31% increase, and the IQR decreased from 5.07 to 2.02 °C, a 150% decrease (Figure 20).

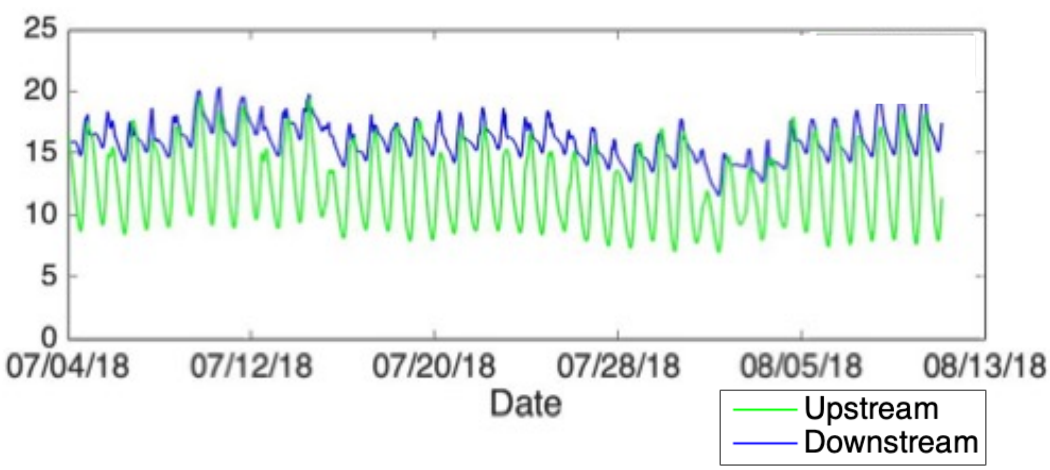


Figure 20: Temporal measurements of temperature at Windy Gap

On a daily average, the downstream average maximum temperature was 10% higher than the upstream maximum temperature, and the downstream minimum was 70% higher than the upstream average daily minimum, indicating that the increase in median water temperature was largely from an increase in lower temperatures rather than an increase in peak temperatures.

Power Diversion

From upstream to downstream of the power diversion, median water temperature increased in the summer from 17.69 to 18.18, a 3% increase ($p<0.001$), and from 12.77 to 13.30, a 4% increase ($p<0.001$), in the fall. The IQR decreased from 2.18 to 1.68, a 30% decrease, in the summer and from 2.54 to 1.91, a 33% decrease, in the fall. The water temperature peaked earlier in the day at the upstream site than the downstream site. In the summer, the downstream daily temperature trends resembled the upstream trends, but in the fall, at lower discharge, the downstream site did not track the upstream site as closely (Figure 21). On average, the downstream daily peak temperatures equaled the upstream daily peak temperatures, but the upstream temperatures were 7% lower than the downstream temperatures. This trend fluctuated during the first part of the fall at low discharge when the downstream maximums decreased.

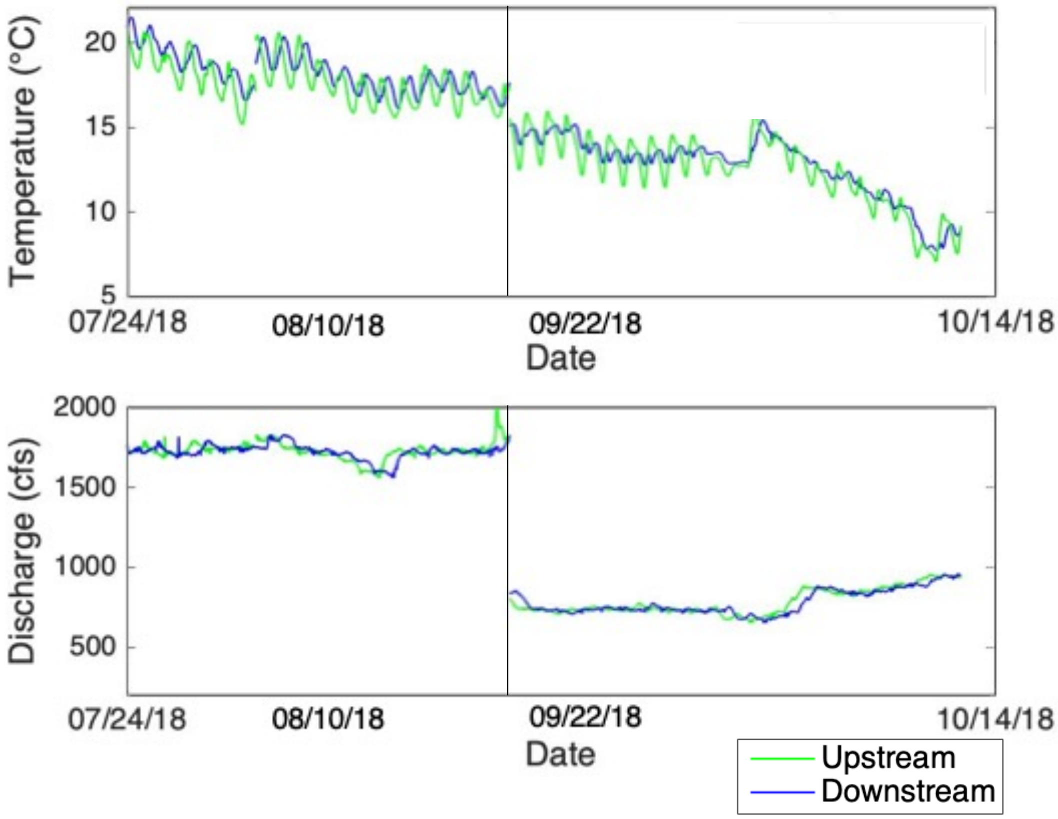


Figure 21: Temporal measurements of temperature and discharge for the power diversion

Irrigation Diversion

From above to below the irrigation diversion, median water temperature increased from 21.22 to 21.69 °C, a 2% increase ($p<0.001$), in the summer and did not change significantly in the fall, 15.46 to 15.57 °C, a 1% increase ($p=0.66$). The IQR decreased from 2.17 to 1.68 °C, a 30% decrease, in the summer and increased from 1.76 to 3.05 °C, a 73% decrease, in the fall. The average daily maximum and minimum upstream temperatures were 3% and 5% lower, respectively, than that of the downstream temperature in the summer. In the fall, it was 17% lower and 4% higher, respectively (Figure 22).

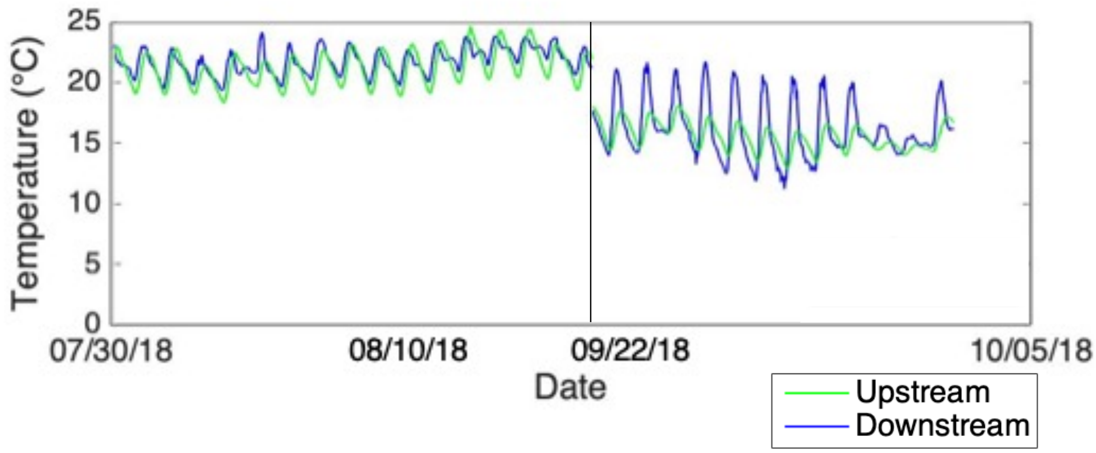


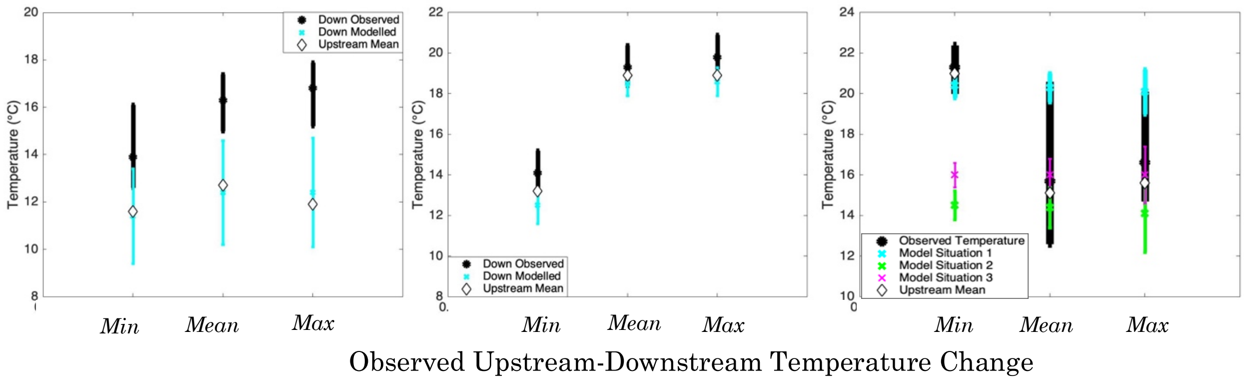
Figure 22: Temporal measurements of temperature at the irrigation diversion.

Summary of Temperature Results for all Discontinuities

Overall, temperature decreased below the deep-release impoundments and increased below the surface-release impoundment and diversions, but there were some seasonal exceptions. The magnitudes of the temperature changes were greater below all impoundments than the diversions.

Comparison of Observed Downstream Temperature with SSTEMP Model

Since downstream temperature increased at Windy Gap, the power diversion, and irrigation diversion, I modelled downstream temperature for these reaches assuming no discontinuity existed. In contrast to the observed downstream temperatures, the mean modelled temperature decreased from the observed upstream temperature for most scenarios and discontinuities (Figure 23).



a) b) c)

Figure 23: Modelled and observed temperatures for Windy Gap (a), the power diversion (b), and the irrigation diversion (c). Modelled and observed temperature values were compared for three scenarios: minimum, mean, and maximum observed daily temperature changes between the upstream and downstream sites. The irrigation diversion model was run for three different downstream discharges: Model cases 1,2 and 3. Downstream discharge equaled upstream discharge (case 1) (no diversion), downstream discharge equaled observed downstream discharge plus 50% diverted water (case 2), and downstream discharge equaled observed discharge (case 3).

°C (a 3.71% increase) for the observed minimum, mean, and maximum temperature changes, respectively. Observed downstream temperature changes were larger than modelled temperatures with increases of 19.62, 28.45, and 40.62 % for the observed minimum, mean, and maximum temperature changes, respectively.

For the power diversion, modelled temperature decreased, 13.17 to 12.50°C (a 5% decrease), 18.91 to 18.51°C (a 2% decrease), and 18.86 to 18.64°C (a 1% decrease) for the observed minimum, mean, and maximum temperature changes, respectively. With the exception of the minimum case and an order of magnitude difference, observed downstream temperature changes for the power diversion were of similar magnitude to the modelled values with increases of 0.4, 2, and 5% for the observed minimum, mean, and maximum temperature changes, respectively.

For the irrigation diversion case 1 (upstream Q = downstream Q), modelled temperature decreased, 21.02 to 20.39°C (a 3% decrease) and 15.13 to 14.53°C (a 4% decrease), and increased from 15.55 to 15.97°C (a 3% increase) for the observed minimum, mean, and maximum temperature changes, respectively. For case 2 (downstream Q increased by 50% of diverted Q), modelled temperature decreased, 21.02 to 20.31°C (a 3% decrease) and 15.13 to 14.43°C (a 5% decrease), and increased from 15.55 to 16.03 °C (a 3.07% increase) for the observed minimum, mean, and maximum temperature changes, respectively. For case 3 (observed downstream Q), modelled temperature decreased, 21.02 to 20.12 °C (a 4 % decrease) and 15.13 to 14.11 °C (a 7% decrease), and increased from 15.55 to 16.23°C (a 4.36% increase) for the observed minimum, mean, and maximum temperature changes, respectively. Observed downstream temperature changes for the irrigation diversion were also of similar magnitude to the modelled values with increases of 1.22, 9.92, and 6.95% for the observed minimum, mean, and maximum temperature changes, respectively.

The modelled daily temperature ranges were larger than the observed temperature ranges for Windy Gap, similar for the powerdiversion, and smaller both the irrigation diversion. The modelled daily temperatures at Windy Gap ranged from 9.39 to 13.48°C, 10.21 to 14.54°C, and 10.08 to 14.67°C for the observed minimum, mean, and maximum temperature changes, respectively. The observed daily temperatures at Windy Gap ranged from 12.6 to 16.1°C , 15.0 to 17.4°C , and

15.2 to 17.9°C for the observed minimum, mean, and maximum temperature changes, respectively.

The modelled daily temperatures at the power diversion ranged from 11.59 to 13.41°C, 17.91 to 19.10°C, and 17.92 to 19.37°C for the observed minimum, mean, and maximum temperature changes, respectively. The observed daily temperature ranged from 13.4 to 15.2°C, 18.4 to 20.4°C, and 19.2 to 20.5°C for the observed minimum, mean, and maximum temperature changes, respectively.

The modelled daily ranges were smaller than the observed daily ranges, but they increased with decreasing downstream discharge. For case 1 at the Irrigation Diversion, the modelled daily temperatures ranged from 19.8 to 21.0 °C, 13.8 to 15.3°C, and 15.4 to 16.6°C for the observed minimum, mean, and maximum temperature changes, respectively. For case 2 at the Irrigation Diversion, the modelled daily temperatures ranged from 19.6 to 21.0°C, 13.48 to 15.39 °C, and 15.26 to 16.8°C for the observed minimum, mean, and maximum temperature changes, respectively. For case 3 at the Irrigation Diversion, the modelled daily temperatures ranged from 19.1 to 21.2°C, 12.23 to 16.41, and 14.83 to 17.63°C for the observed minimum, mean, and maximum temperature changes, respectively. The observed daily temperatures ranged from 19.7 to 22.7°C, 12.7°C to 20.6°C, and 14.7 to 20.2°C for the observed minimum, mean, and maximum temperature changes, respectively.

Dissolved Oxygen

Discontinuities had varying effects downstream DO conditions and showed seasonal trends(Figure 24).

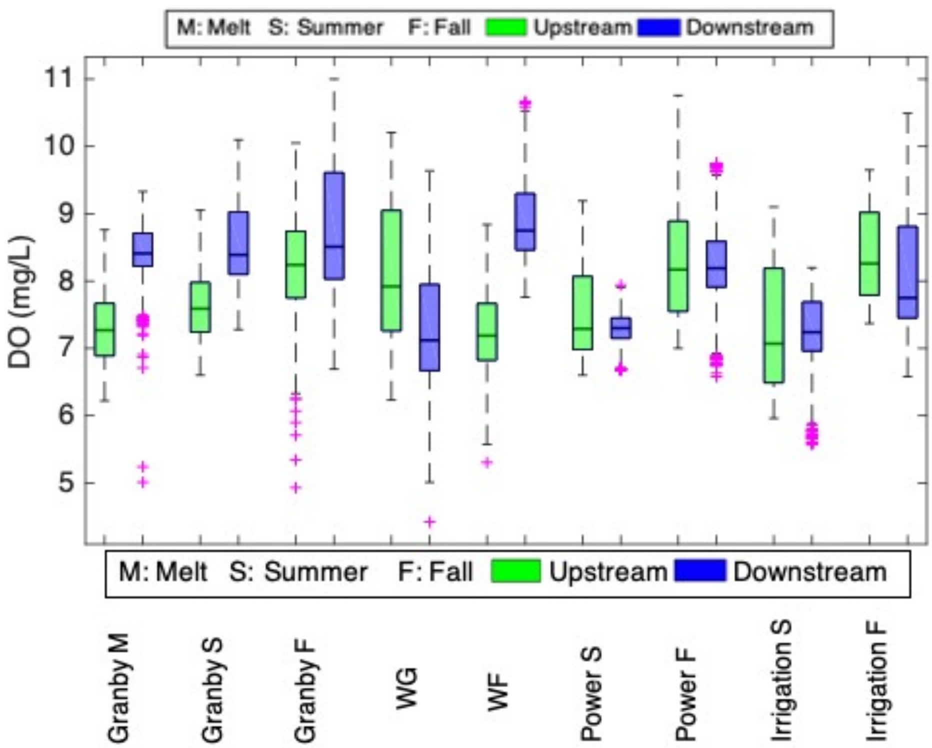


Figure 24:Temporal boxplots of DO upstream and downstream of each discontinuity.

Granby (deep-release)

From above to below Granby, median DO increased from 7.27 to 8.41 mg/L, a 16% increase ($p<0.001$), in the melt season, from 7.59 to 8.39 mg/L, a 26% increase ($p<0.001$), in the summer, and 8.24 to 8.51 mg/L, a 3% increase ($p<0.001$), in the fall. The IQR decreased 0.78 to 0.49 mg/L, a 59% decrease, in the melt season, and increased from 0.74 to 0.93 mg/L, a 26% increase, in the summer, and from 0.99 to 1.58 mg/L, a 60% increase, in the fall (**Error! Reference source not found.**).

Williams Fork (deep-release)

From above to below Wiliams Fork, median DO increased from 7.19 to 8.75 mg/L, a 22% increase ($p<0.001$), and the IQR stayed constant (0.84) (**Error! Reference source not found.**).

Windy Gap (surface-release)

From above to below Windy Gap, median DO decreased from 7.92 to 7.12 mg/L, a 11% decrease ($p<0.001$), and the IQR decreased from 1.79 to 1.28 mg/L, a 40% decrease (**Error! Reference source not found.**).

Power Diversion

The power diversion median DO did not significantly change downstream in the summer and fall (7.29 to 7.30, 8.17 to 8.19 mg/L, $p=0.21,0.30$) and the IQR decreased in both seasons, (1.09 to 0.3 mg/L, a 445% decrease, and 1.34 to 0.68 mg/L, a 97% decrease, respectively) (**Error! Reference source not found.**).

Irrigation Diversion

From upstream to downstream of the irrigation diversion, median DO increased from 7.07 to 7.24 mg/L, a 2% increase ($p<0.001$), in the summer and decreased from 8.26 to 7.75 mg/L, a 7% decrease ($p<0.001$), in the fall. The IQR decreased from 1.7 to 0.73 mg/L, a 133% decrease, and increased from 1.23 to 1.36 mg/L, an 11% increase (**Error! Reference source not found.**).

Summary of DO for all Discontinuities

Dissolved Oxygen (DO) responded differently to each type of discontinuity. Median DO increased below the deep-release impoundments, decreased below the surface-release impoundments, did not change significantly below the power diversion, and increased in the summer and decreased in the fall below the irrigation diversion. The magnitudes of the DO changes (like temperature) were greatest below all impoundments. Downstream variability changes were seasonal, but the power diversion showed the largest downstream change, reducing the IQR in both seasons.

DO Saturation

Discontinuities had varying effects on DO saturation medians and variability (Figure 25).

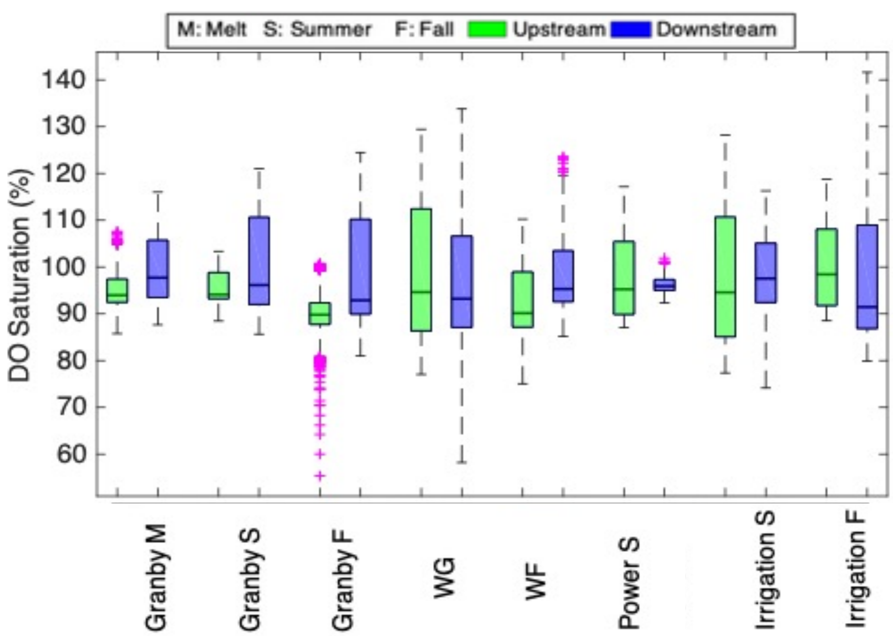


Figure 25:Temporal boxplots of DO Saturation upstream and downstream of each discontinuity.

Granby (deep-release)

The DO Saturation median and amplitude increased downstream of Granby. (Figure 26). From above Granby to below, the median DO saturation increased from 93.97 to 97.71%, a 4% increase ($p<0.001$), in the melt season and from 94.10 to 96.14%, a 2% increase ($p<0.001$), in the summer, and from 89.80 to 92.88%, a 3% increase ($p<0.001$), in the fall. The Granby IQR increased from 4.96 to 12.27%, a 147% increase, during the melt season, from 5.60 to 18.64%, a 233% increase, during the summer, and from 4.60 to 20.25%, a 340% increase (**Error! Reference**

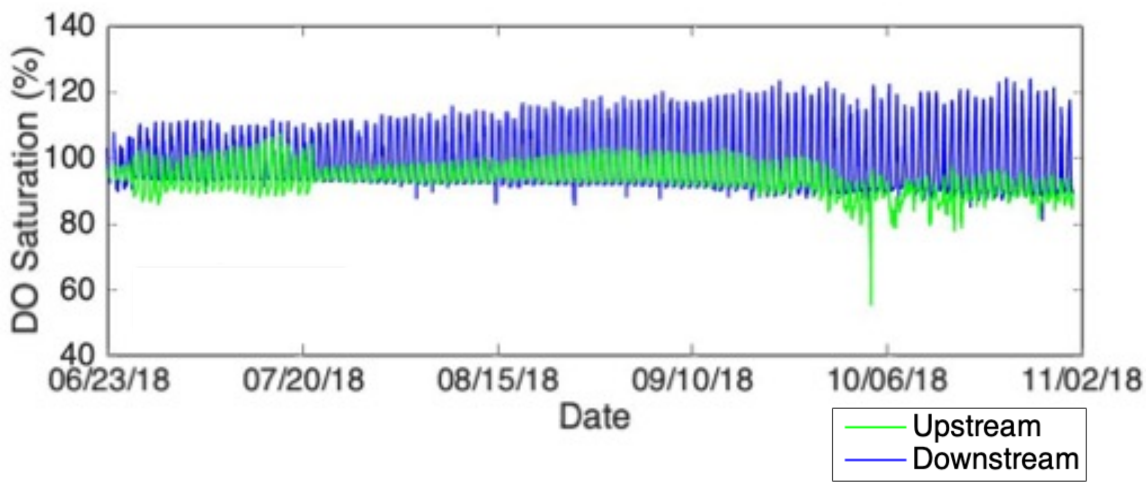


Figure 26: Temporal measurements of DO Saturation for Granby

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Williams Fork (deep-release)

From above Williams Fork to below, DO saturation increased from 90.13 to 95.31%, a 6% increase ($p<0.001$), and the IQR decreased from 11.81 to 10.88, an 9% decrease (**Error! Reference source not found.**). The difference between upstream and downstream DO saturation varied throughout the deployment (Figure 27).

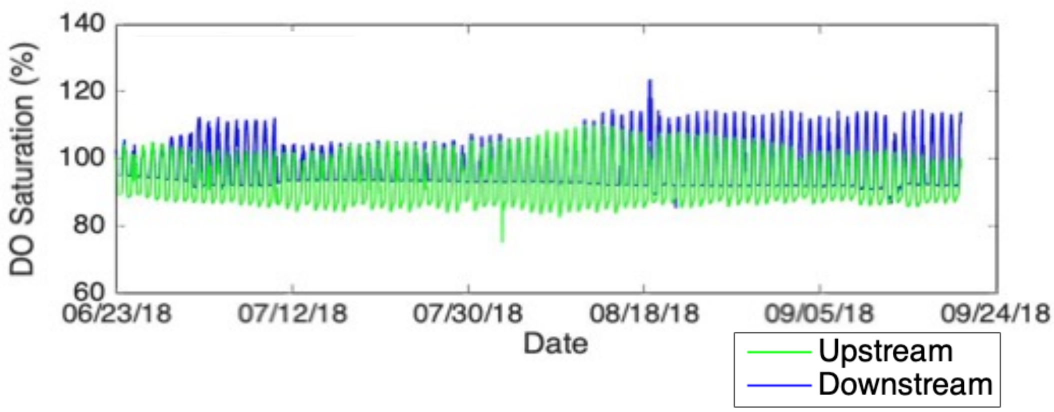
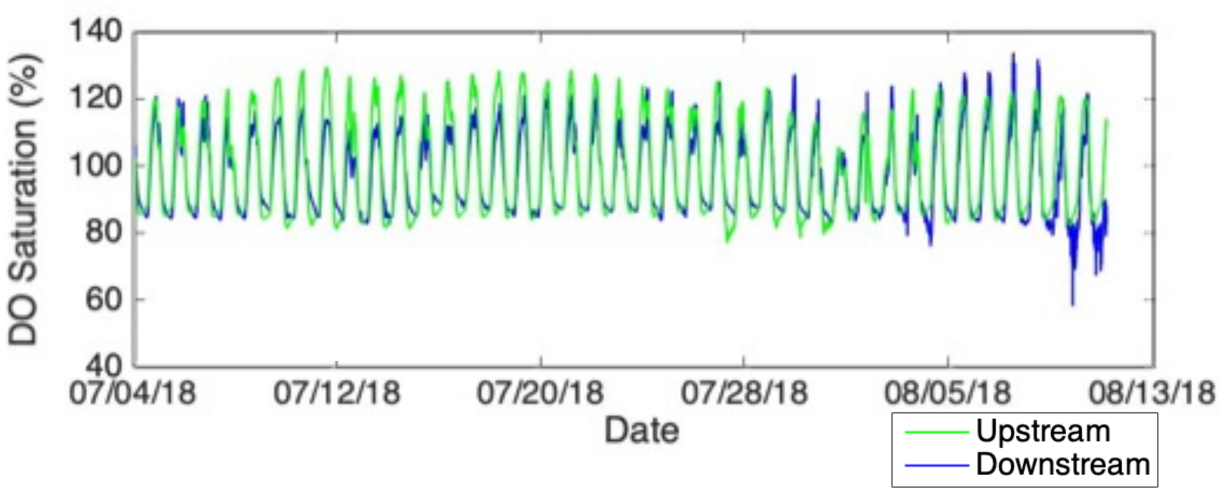


Figure 27: Temporal measurements of DO Saturation for Williams Fork

Windy Gap (surface-release)

From above Windy Gap to below, median DO saturation decreased from 94.62 to 93.22%, a 2% decrease ($p=0.007$), and the IQR decreased from 26.05 to 19.54 %, a 33% decrease (**Error! Reference source not found.**). The difference between



upstream and downstream DO Saturation did not vary during the deployment with the exception of the last few days (Figure 28).

Power Diversion

Due to atmospheric pressure sensor failure, I was only able to calculate DO saturation for the power diversion in the summer. The power diversion median DO saturation increased from 95.24 to 95.91%, a 1% increase (p<0.001), and the IQR

Figure 28:Temporal measurements of DO Saturation for Windy Gap.

decreased from 13.38 to 2.30%, an 500% decrease (Figure 29). Downstream DO saturation values remained constant while the upstream site varied.

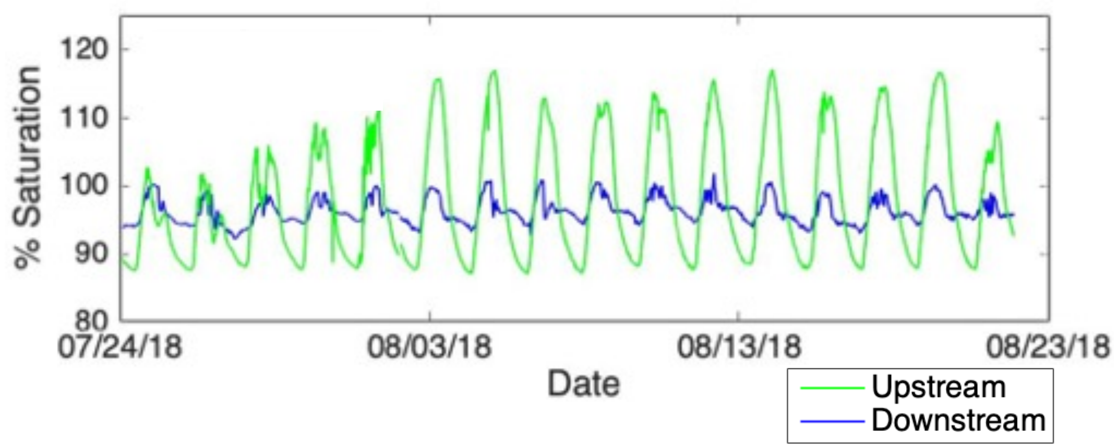


Figure 29: Temporal measurements of DO saturation for the power diversion

From above to below the irrigation diversion, opposite trends were observed for DO saturation in the summer and fall. Median saturation increased from 94.58 to 97.55%, a 3% increase (p<0.001), and the IQR decreased from 25.58 to 12.71, a 101% decrease, in the summer while DO saturation decreased from 94.51 to 91.45%

(p<0.001), a 3% decrease, and the IQR increased from 19.30 to 22.11%, a 15% increase, in the fall (Figure 30).

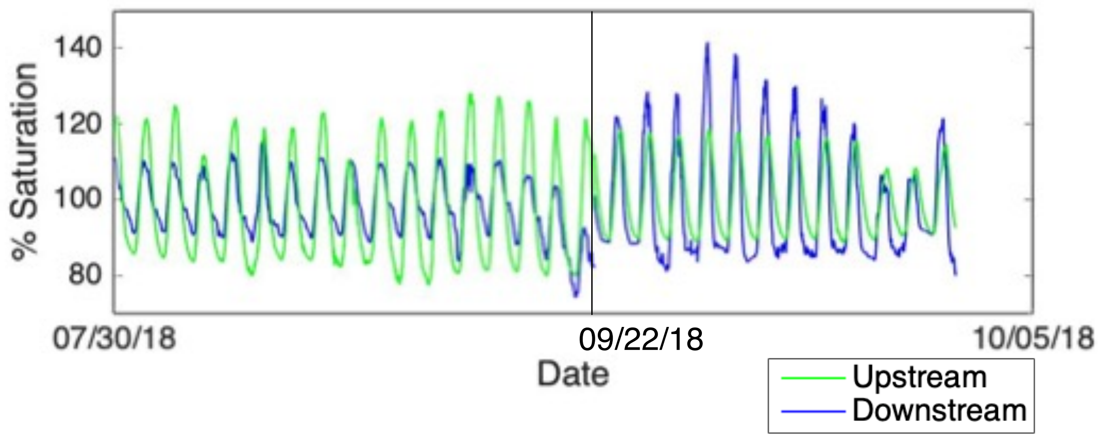


Figure 30: Temporal measurements of DO saturation for the irrigation diversion

Summary of DO Saturation for all Discontinuities

Downstream median DO Saturation responses were similar to median DO responses. Median DO saturation increased below the deep-release impoundments, decreased below the surface-release impoundment, increased below the power diversion, and increased in the summer and decreased in the fall below the irrigation diversion. Discontinuities had different effects on downstream variability of DO saturation. The IQR increased below Granby (deep-release), remained constant below Williams Fork (deep-release), decreased below the surface-release impoundment, decreased the most below the power diversion, and decreased in the summer and increased in the fall below the irrigation diversion.

Specific Conductivity

Specific conductivity (SpC) increased over the measured length of the river. Each discontinuity had differing effects on downstream SpC concentrations with the power diversion having the largest effect (Figure 31).

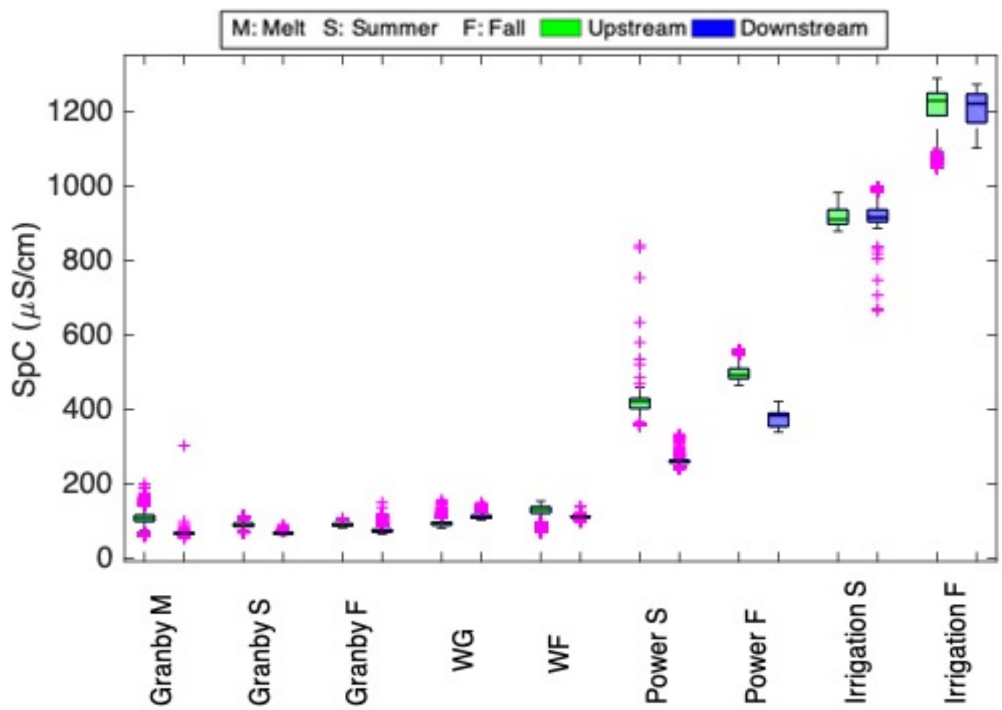


Figure 31: Temporal boxplots of SpC upstream and downstream of each discontinuity.

Granby (deep-release)

From above to below Granby, median SpC decreased from 107.84 to 67.30 µS/cm, a 60% decrease ($p<0.001$), in the melt season, from 89.16 to 67.79 µS/cm, a 32% decrease ($p<0.001$), in the summer, and from 90.16 to 73.68 µS/cm, an 22% decrease ($p<0.001$), in the fall. The IQR decreased from 16.42 to 1.98 µS/cm, an 730% decrease, in the melt season, from 8.38 to 3.09 µS/cm, a 171% decrease, in the summer, but increased from 5.79 to 6.28 µS/cm, a 9% increase, in the fall (**Error!**

Reference source not found.). These trends do not vary on a smaller temporal scale than the seasonal separations (Figure 32).

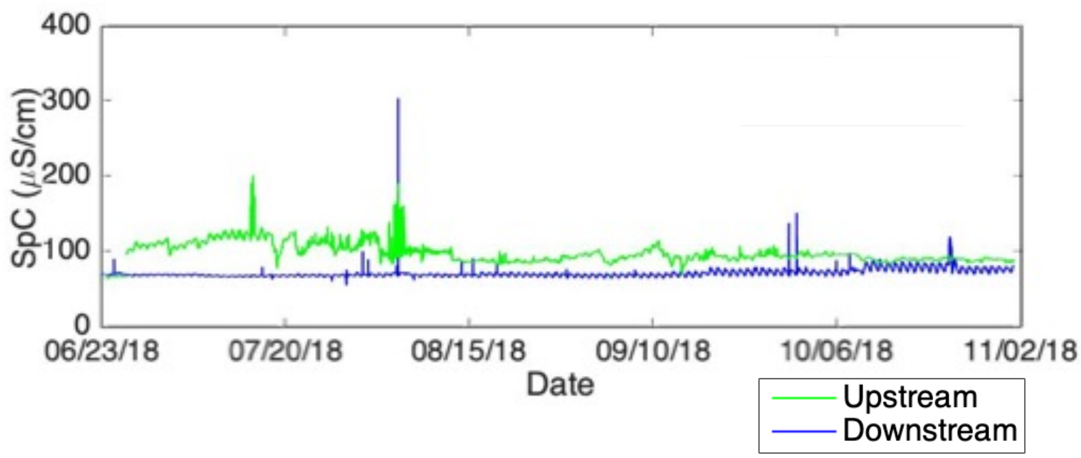


Figure 32: Temporal measurements of SpC for Granby

Williams Fork (deep-release)

From above to below Williams Fork, median SpC decreased from 132.12 to 111.80 μS/cm, a 18% decrease ($p<0.001$), and the IQR decreased from 16.79 to 1.90 μS/cm, an 784% decrease (**Error! Reference source not found.**). The upstream site initially had a lower SpC signature, but increased over the deployment. Downstream SpC stayed constant over the deployment with the exception of sporadic, intense changes in SpC that returned to the median level (Figure 33).

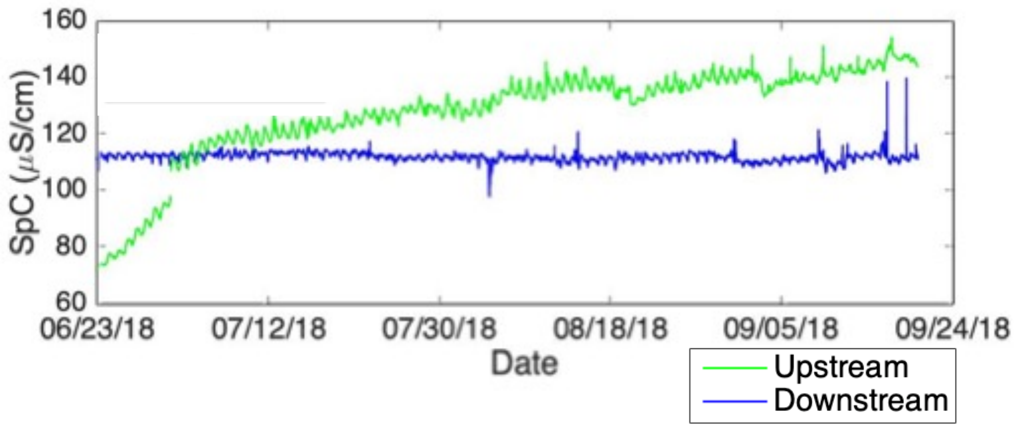


Figure 33: Temporal measurements of SpC for Williams Fork

Windy Gap (surface-release)

From above to below Windy Gap (surface-release), median SpC increased from 93.4 to 109.9 µS/cm, an 18% increase($p<0.001$), while the IQR stayed constant (7.6 µS/cm) (**Error! Reference source not found.**). Apart from upstream noise in the SpC signature, downstream SpC was consistently higher (Figure 34).

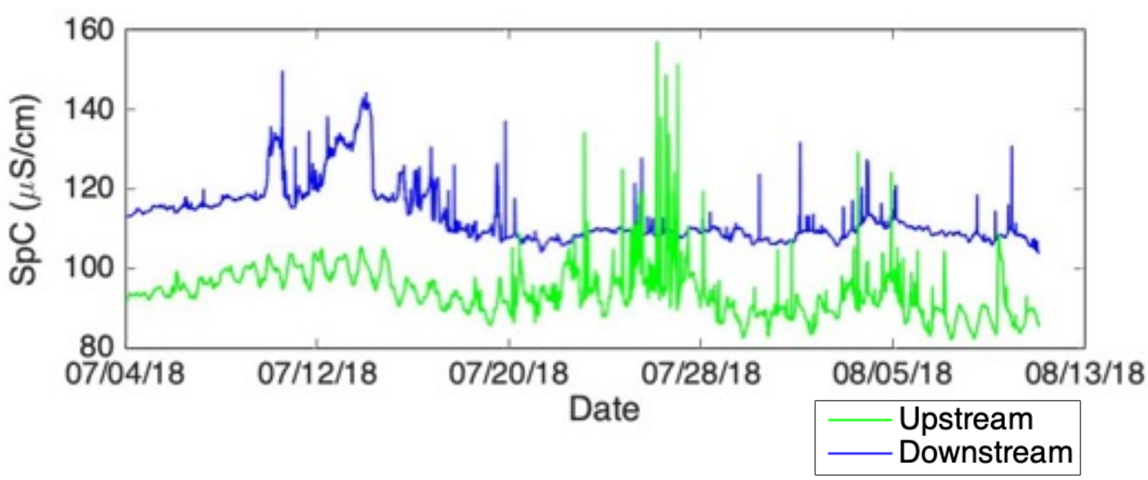


Figure 34:Temporal measurements of SpC for Windy Gap

Power Diversion

From above to below the power diversion, median SpC decreased from 421.1 to 260.9 μS/cm, a 61% decrease ($p<0.001$), in the summer and from 493.2 to 384.0 μS/cm, a 28% decrease($p<0.001$), in the fall. The IQR decreased from 26.1 to 5.8 μS/cm, a 350% decrease, in the summer and increased from 26.4 to 35.7 μS/cm, a 5% increase, in the fall. The downstream SpC decrease was consistent throughout the deployment (Figure 35).

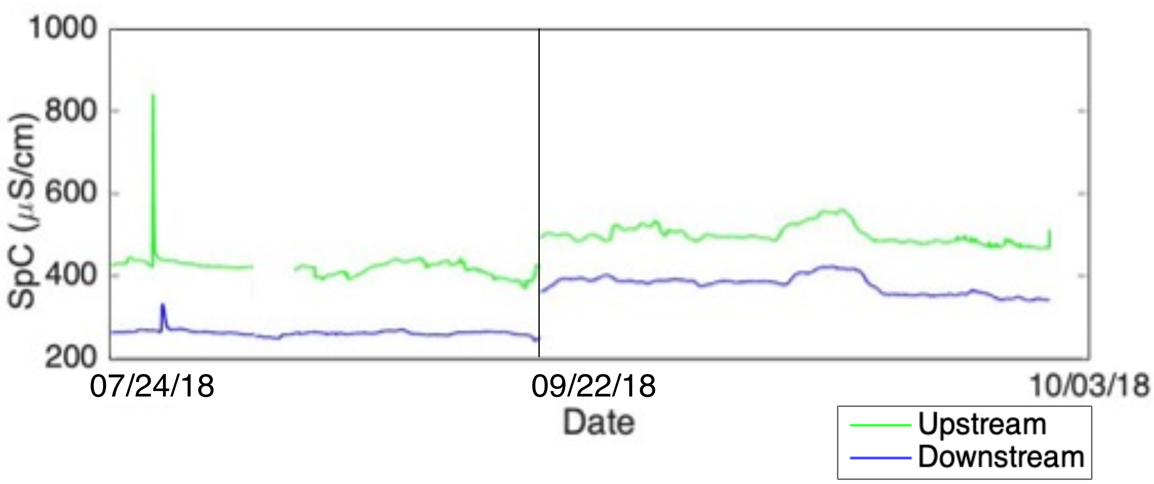


Figure 35: Temporal measurements of SpC for the power diversion

Irrigation Diversion

From above to below the irrigation diversion, median SpC increased from 911.0 to 916.1 μS/cm, a 1% increase ($p<0.001$), in the summer and decreased from 1230.0 to 1222.0 μS/cm, a 1% decrease ($p=0.0052$), in the fall. The IQR decreased from 39.0 to 32.4 μS/cm, a 17% decrease, in the summer and increased from 60.00 to 76.81 μS/cm, a 78% decrease, in the fall. The downstream change was not consistent throughout the deployment (Figure 36).

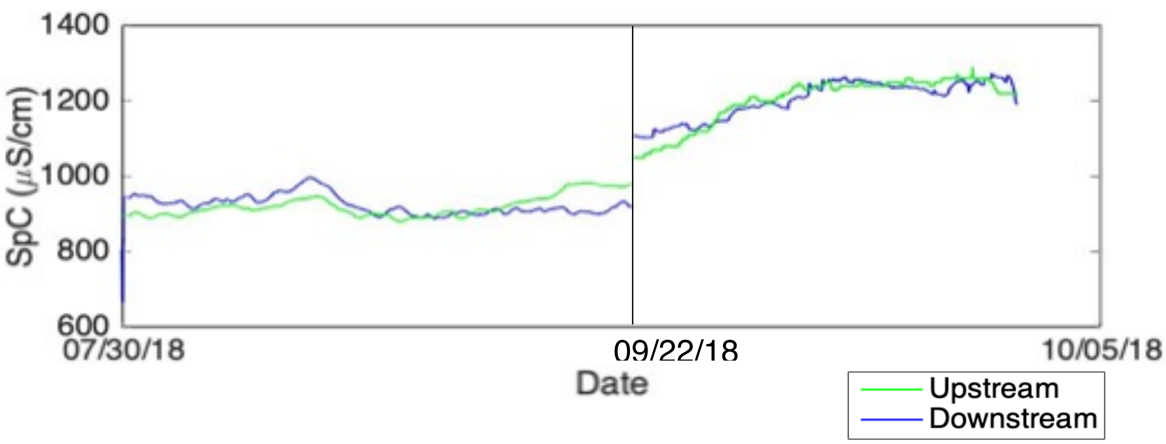


Figure 36: Temporal measurements of SpC for the irrigation diversion

Summary of SpC Results for all Discontinuities

Median SpC decreased below both deep-release reservoirs, increased below the surface-release reservoir, decreased below the power diversion, and changed by 1% below the irrigation diversion. The power diversion had the largest effect on the magnitude of SpC change.

Metabolism

I was unable to model metabolism at the upstream Granby site and downstream power diversion site due to high reaeration and small deviation from 100% saturation, so I present metabolism results for Williams Fork, Windy Gap, and the Irrigation Diversion. Due to the proximity of the WG upstream site with the confluence of the Colorado and Fraser Rivers, I also present results of the Granby downstream site (upstream of the Fraser River) to compare with the WG downstream site.

All sites had process error equal to 1.00, but WG Up, WG Up (7/4/18-7/14/18),WF Up, and WF Down had observation error and log posterior greater than 1.05 (Table 4*)* indicating that the models hadn't converged. Additionally, all models had low Bayesian Fraction of Missing Information in at least one chain.

Table 4: Model convergence values for each site. Values close to or less than 1.05 indicate MCMC convergence.

Site	Observation Error	Process Error R	Log Posterior Density Error R
Granby Down	1.00	1.00	1.00
WG Up	1.27	1.00	1.35
WG Up (7/4/18-7/14/18)	1.13	1.00	1.15
WG Down	1.00	1.00	1.00
WF Up	1.30	1.00	1.36
WF Down	1.50	1.00	1.69
Irrigation Up	1.00	1.00	1.00
Irrigation Down	1.00	1.00	1.00

The correlations between the Bayesian modelled metabolism values and the metabolism metrics I used were correlated most highly at the sites with the lowest reaeration (Table 5).

Table 5: Correlations between daily DO saturation amplitude and GPP, DO deficit and CR, and discharge and reaeration for all sites.

Site	DO Deficit Amplitude-GPP Correlation	DO Deficit-CR Correlation	Discharge-Reaeration Correlation
Granby Down	-0.23	-0.06	-0.11
WG Up	0.65	-0.03	0.02
WG Down	0.05	-0.22	0.46
WF Up	0.69	-0.23	-0.13
WF Down	0.02	0.03	0.40
Irrigation Up	0.90	0.49	0.70
Irrigation Down	0.73	0.36	-0.57

The discontinuities had varying effects on NEP, GPP, CR, and reaeration (Figure 37).

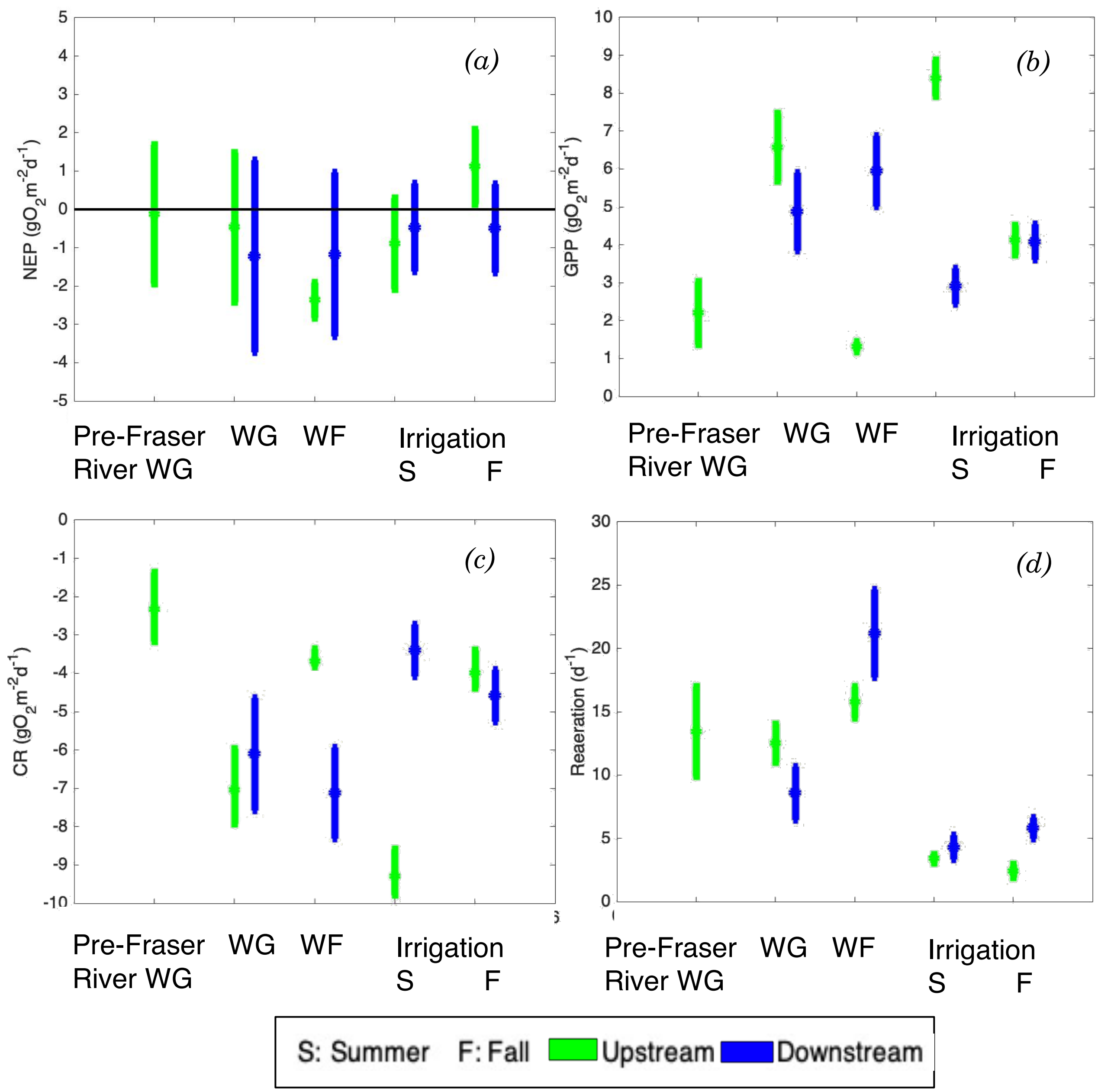


Figure 37: Mean, lower, and upper ranges of NEP (a), GPP (b), CR (c), and Reaeration (d) over deployment dates for each site.

Williams Fork (deep-release)

For all sites, CR is a negative value and GPP is a positive value. A negative NEP indicates a heterotrophic environment, and a positive NEP indicates an autotrophic environment. From upstream to downstream, average GPP increased from 1.32 to 5.95 $gO_2m^{-2}d^{-1}$, average CR increased in magnitude from -3.68 to -7.12 $gO_2m^{-2}d^{-1}$, average NEP increased from -2.36 to -1.17 $gO_2m^{-2}d^{-1}$ (a 50% shift towards autotrophy), and the average reaeration constant increased from 15.77 to 21.20 d^{-1} (Figure 38). The standard deviation (SD) increased downstream for all metabolism parameters. From upstream to downstream, the GPP SD increased from 0.14 to 0.94 $gO_2m^{-2}d^{-1}$, the CR SD increased from 0.32 to 1.20 $gO_2m^{-2}d^{-1}$, the NEP SD increased from 0.46 to 2.13 $gO_2m^{-2}d^{-1}$, and the reaeration SD increased from 1.24 to 3.48 d^{-1}.

Windy Gap (surface-release)

In addition to convergence issues with WG Up, the confluence of the Fraser River with the Colorado River 0.15 km upstream of the upstream site posed challenges to the assumption that the upstream reach is well-mixed, so I compared WG down to Granby Down (above the Fraser River) in addition to WG Up. From WG Up to WG Down, average GPP decreased from 6.58 to 4.88 $gO_2m^{-2}d^{-1}$, average CR decreased in magnitude from -7.03 to -6.10 $gO_2m^{-2}d^{-1}$, average NEP decreased from -0.46 to -1.22 $gO_2m^{-2}d^{-1}$ (a 165% shift towards heterotrophy), and the average reaeration constant decreased from 12.53 to 8.59 d^{-1} (Figure 39). Uncertainty increased downstream for all metabolism parameters. From upstream to downstream, the GPP SD increased from 0.89 to 1.04 $gO_2m^{-2}d^{-1}$, the CR SD increased from 1.07 to 1.47 $gO_2m^{-2}d^{-1}$, the NEP SD increased from 1.96 to 2.51 $gO_2m^{-2}d^{-1}$, and the reaeration SD increased from 1.52 to 2.10 d^{-1}.

From Granby Down to WG Down, average GPP increased from 2.21 to 4.88 $gO_2m^{-2}d^{-1}$, average CR increased in magnitude from -2.33 to -6.10 $gO_2m^{-2}d^{-1}$, average NEP decreased from -0.12 to -1.22 $gO_2m^{-2}d^{-1}$ (a 917% shift towards heterotrophy), and the average reaeration constant decreased from 13.46 to 8.59 d^{-1} (Figure 40). Uncertainty increased downstream for all metabolism parameters except for reaeration. From upstream to downstream, the GPP SD increased from 0.83 to 1.04 $gO_2m^{-2}d^{-1}$, the CR SD increased from 0.98 to 1.47 $gO_2m^{-2}d^{-1}$, the NEP SD increased from 1.81 to 2.51 $gO_2m^{-2}d^{-1}$, and the reaeration SD decreased from 3.53 to 2.10 d^{-1}.

Irrigation Diversion

The irrigation diversion had different effects on metabolism in the summer than in the fall. In the summer, average GPP decreased from 8.40 to 2.92 $gO_2m^{-2}d^{-1}$, average CR decreased in magnitude from –9.29 to -3.39 $gO_2m^{-2}d^{-1}$, average NEP increased from -0.89 to -0.47 $gO_2m^{-2}d^{-1}$ (a 47% shift towards autotrophy), and the average reaeration constant increased from 3.42 to 4.30 d^{-1}, from upstream to downstream (Figure 41). Uncertainty stayed constant downstream of the diversion with the exception of reaeration. The GPP SD was 0.48 $gO_2m^{-2}d^{-1}$at the upstream site and to 0.47 $gO_2m^{-2}d^{-1}$ at the downstream site, the CR SD was 0.71 $gO_2m^{-2}d^{-1}$to 0.68 $gO_2m^{-2}d^{-1}$ at the downstream site, the NEP SD was 1.19 $gO_2m^{-2}d^{-1}$ at the upstream site and 1.15 $gO_2m^{-2}d^{-1}$ at the downstream site, and the reaeration SD increased from from 0.34 to 0.96 d^{-1}.

GPP and CR were lower at the upstream site and higher at the downstream site in the fall. Average GPP was similar at both sites and was 4.13 $gO_2m^{-2}d^{-1}$ at the upstream site and 4.08 $gO_2m^{-2}d^{-1}$ at the downstream site. From upstream to downstream, average CR increased in magnitude from 3.98 to -4.57 $gO_2m^{-2}d^{-1}$, average NEP decreased from 1.12 to -0.49 $gO_2m^{-2}d^{-1}$ (a 144% shift towards heterotrophy), and the average reaeration constant increased from 2.43 to 5.82 d^{-1}, from upstream to downstream. Uncertainty was still relatively constant downstream of the diversion, but increased more in the fall than in the summer for all parameters except for reaeration. The GPP SD was 0.38 $gO_2m^{-2}d^{-1}$at the upstream site and to 0.47 $gO_2m^{-2}d^{-1}$ at the downstream site, the CR SD was 0.60 $gO_2m^{-2}d^{-1}$to 0.69 $gO_2m^{-2}d^{-1}$ at the downstream site, the NEP SD was 1.30 $gO_2m^{-2}d^{-1}$ at the upstream site and 1.16 $gO_2m^{-2}d^{-1}$ at the downstream site, and the reaeration SD increased from from 0.52 to 0.86 d^{-1}.

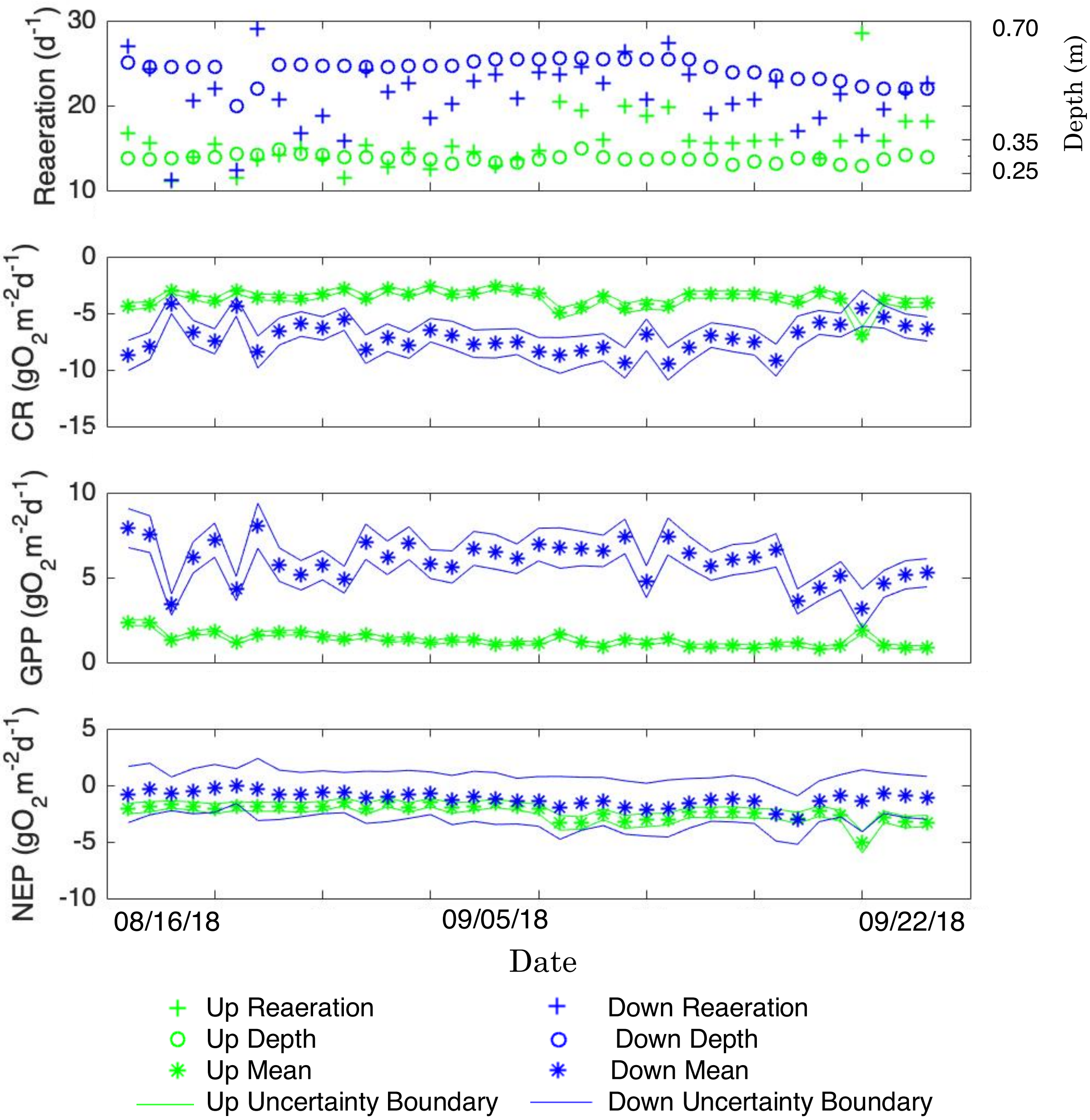


Figure 38: Time series of daily metabolism values for Williams Fork.

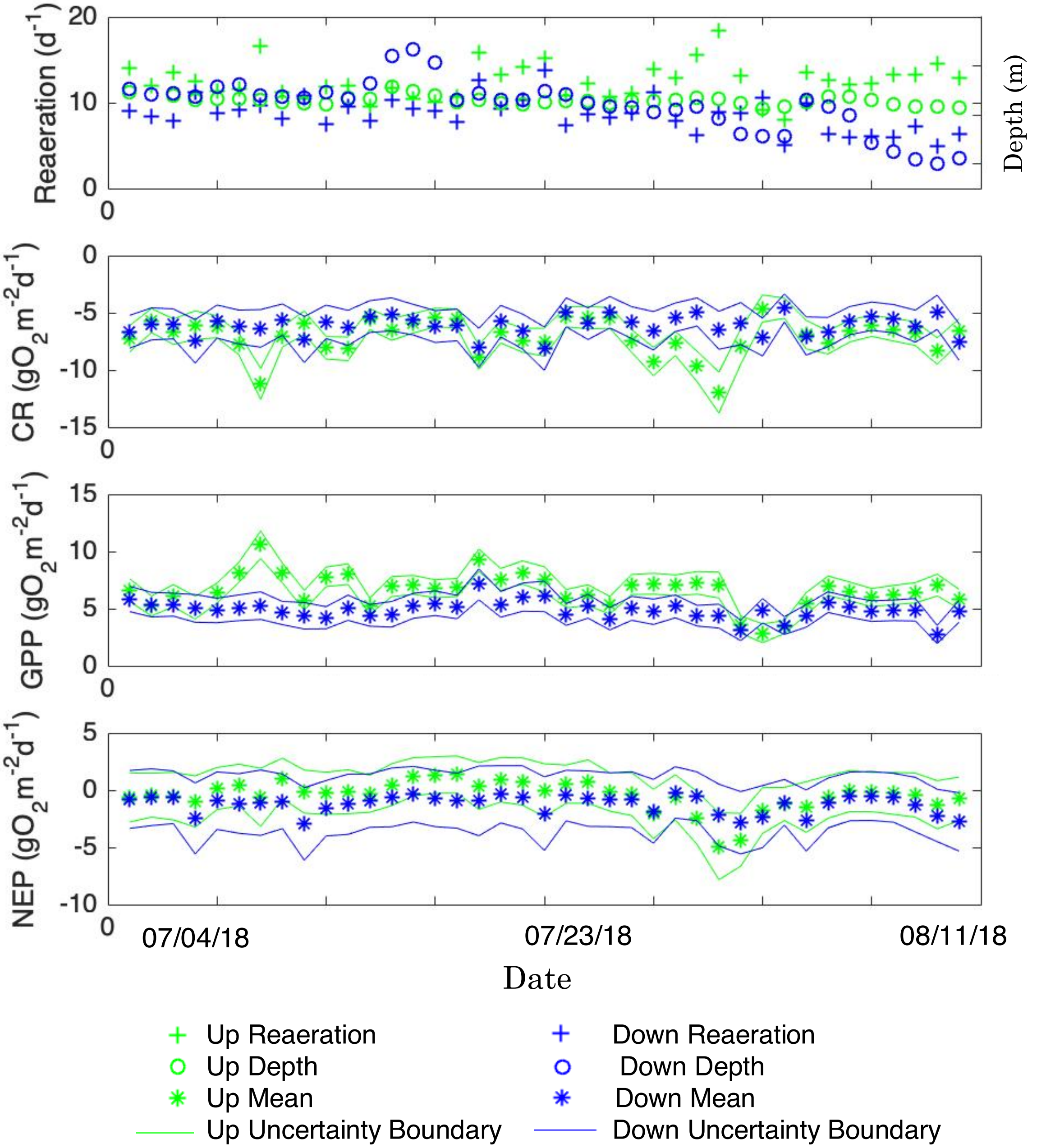


Figure 39:Time series of daily metabolism values for Windy Gap.

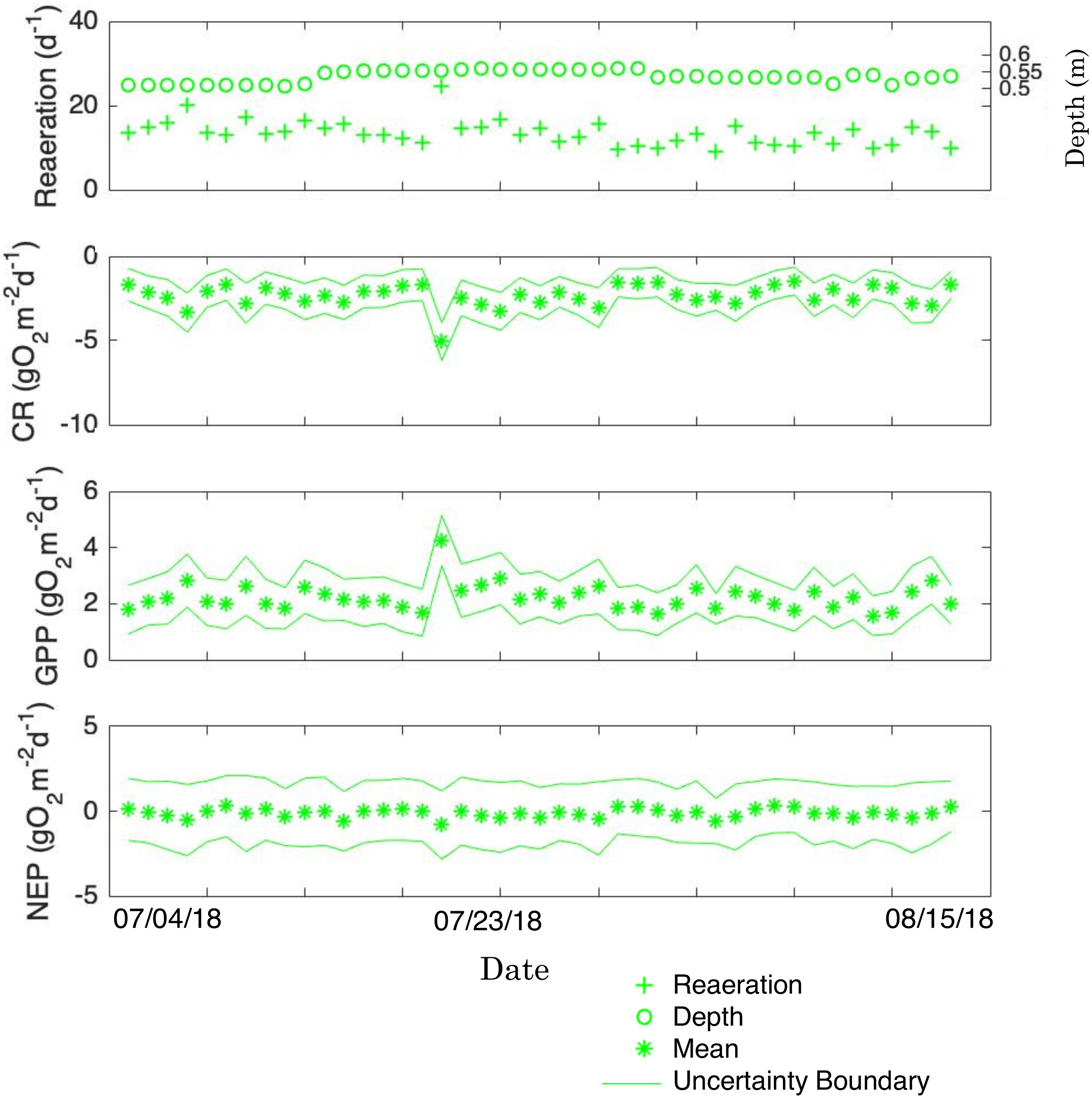


Figure 40: Time series of daily metabolism values for Granby Down used as an alternative control site for Windy Gap Up. The Fraser River joins the Colorado River between Granby Down and Windy Gap Up.

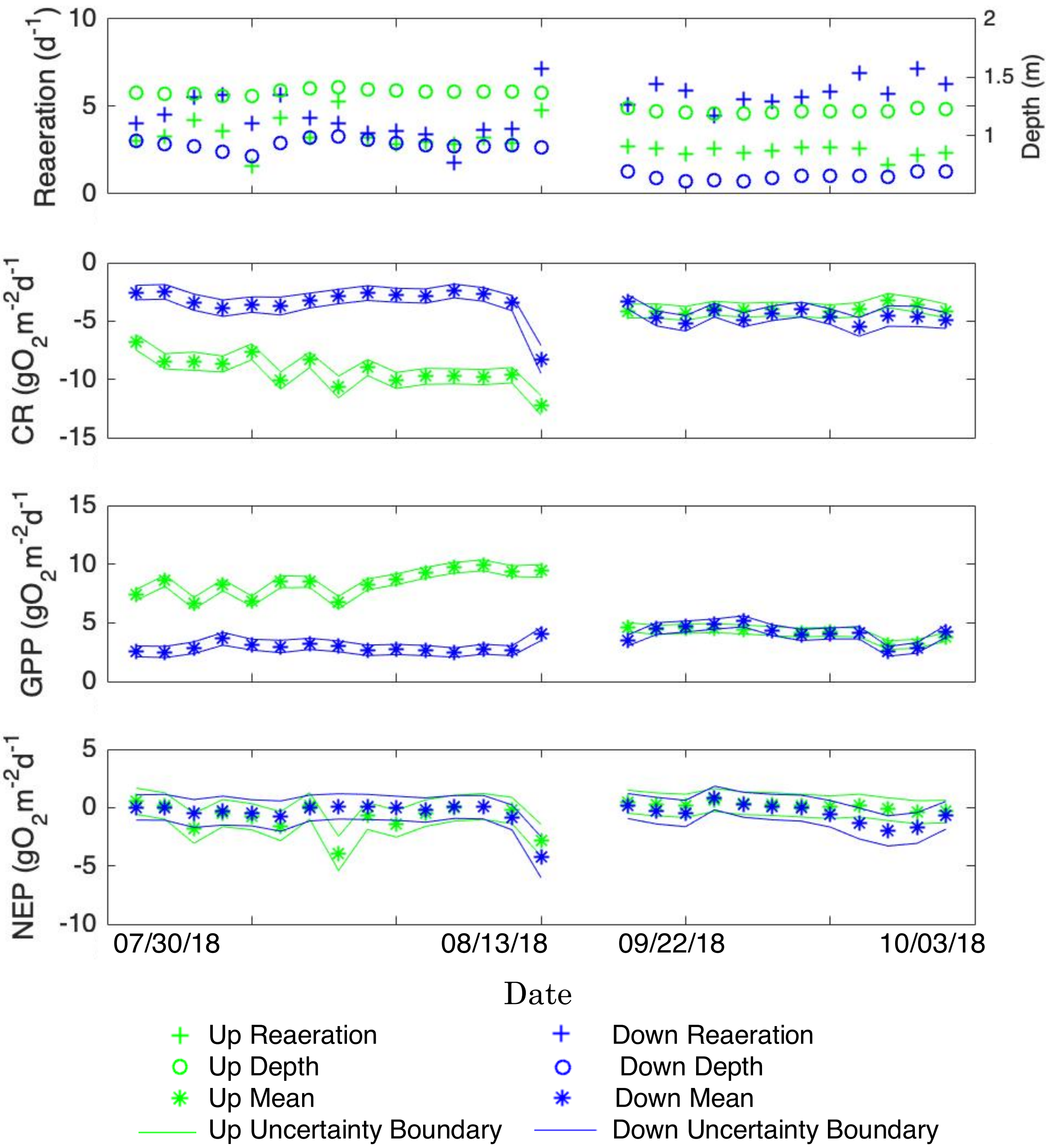


Figure 41:Time series of daily metabolism values for the irrigation diversion.

Metabolism Summary

Williams Fork shifted towards autotrophy, WG shifted towards heterotrophy, and the irrigation diversion shifted towards autotrophy in the summer and heterotrophy in the fall. Williams Fork and WG had more uncertainty involved in their metabolism estimation than the irrigation diversion.

Impacts of Discontinuities on Water Quality-Discharge Relationships and DO Saturation-Temperature Relationships

Granby (deep-release)

At Granby, strong seasonal differences existed between discharge and water quality upstream of the impoundment. This shift occurred in both the melt season to the summer and from the summer to the fall (Figure 42a and b). From the melt season to the summer season, the upstream temperature-discharge relationship changed from -0.036 °C/cfs ($p<0.001$) to 0.23°C/cfs ($p<0.001$), a 538% increase, and to -0.32 °C/cfs ($p<0.001$), in the fall, a 39% increase. For the upstream DO saturation-discharge relationship, the upstream slope changed from -0.017 %/cfs to 0.029 %/cfs ($p<0.001$), a 71% increase, from the melt season to the summer and changed to -0.24 %/cfs ($p<0.001$), a 728% increase, in the fall. The upstream DO-discharge relationship changed from 0.0047 mg/L/cfs ($p<0.001$) to -0.034 mg/L/cfs ($p<0.001$), a 623% increase, and became insignificant in the fall. The upstream relationship between SpC and discharge remained negative from the melt season to the summer, but increased in magnitude from -0.010 μS/cm/cfs ($p<0.001$) to -1.13 μS/cm/cfs ($p<0.001$), an 11,200% increase, and became insignificant in the fall.

In contrast to the upstream site, the relationships between discharge and SpC or DO were weaker, except in the melt season. In the melt season, the temperature-discharge and DO-discharge relationships followed the same trend as the upstream site, but were weaker. For the melt season, the downstream temperature-discharge slope equaled -0.014 °C/cfs ($p<0.001$), a 61% decrease from the upstream site, and the downstream DO-discharge slope equaled 0.004 mg/L/cfs ($p<0.001$), a 15% decrease from the upstream site. The downstream SpC-discharge slope equaled -0.05 μS/cm/cfs ($p<0.001$), a 400% increase. In the summer, the only downstream significant relationship with flow was SpC (slope=-0.05 μS/cm/cfs, $p<0.001$), a 95% decrease, and became weaker than the upstream site. In the fall, the DO-discharge relationship became significant downstream (-0.026 mg/L/cfs, $p<0.001$), and the temperature-discharge relationship changed to 0.10 °C/cfs ($p<0.001$), a 69% decrease from the upstream site.

Williams Fork (deep-release)

The upstream site water quality variables had different relationships with discharge before and after 06/29/18 (Figure 42,c). Before 06/29/18, DO saturation did not have a significant relationship with discharge, but temperature and SpC had a negative relationship with discharge and DO had a positive relationship with discharge. After 06/29/18, all parameters besides SpC changed the direction of their relationship with discharge. The temperature-discharge relationship changed from -0.047 °C/cfs ($p<0.001$) to 0.1 °C/cfs ($p<0.001$), a 113% increase. The DO-discharge relationship 0.0098 mg/L/cfs ($p<0.001$) to -0.016 mg/L/cfs ($p<0.001$), a 63% increase. The DO saturation relationship became significant, -0.016 %/cfs ($p<0.001$). The SpC-discharge relationship increased from -0.24 μS/cm/cfs ($p<0.001$) to -1.28 μS/cm/cfs ($p<0.001$), a 433% increase.

Downstream of Williams Fork, discharge and water quality had different relationships before and after 6/29/19, and were weaker than upstream relationships. All relationships with discharge were insignificant before 6/29/18. After 6/29/18, all downstream relationships were significant except for the DO-discharge relationship. The temperature-discharge relationship decreased and changed direction from 0.1 °C/cfs ($p<0.001$) upstream to -0.0063 °C/cfs ($p<0.001$) downstream, a 94% decrease. The DO saturation-discharge relationship decreased from -0.03 %/cfs ($p<0.001$) to -0.01 %/cfs ($p<0.001$), a 67% decrease. The SpC-discharge relationship decreased and changed direction from -1.28 μS/cm/cfs ($p<0.001$) to 0.006 μS/cm/cfs ($p<0.001$), a 100% decrease.

The DO saturation-temperature relationship was insignificant above and below Williams Fork before 06/29/18, and became significant after 6/29/18. In contrast to the discharge relationships, the DO saturation-temperature relationship increased downstream from 0.24 %/°C ($p<0.001$) to 0.84 %/°C ($p<0.001$), a 250% increase.

Windy Gap (surface-release)

Water quality-discharge relationships did not change significantly downstream of Windy Gap, except for the SpC-discharge relationship (Figure 43,a). The temperature-discharge and DO saturation-discharge relationships changed little downstream. The temperature-discharge relationship was 0.028 °C/cfs ($p<0.001$) upstream and equaled 0.026 °C/cfs ($p<0.001$) downstream, a 7% decrease. The DO saturation-discharge relationship was 0.057 %/cfs ($p=0.003$) upstream and 0.052 %/cfs ($p<0.001$) downstream, a 9% decrease. DO did not have a significant relationship with discharge. SpC changed from being insignificant upstream to significant and positive downstream (slope=0.16 µS/cm/cfs, $p<0.001$). The DO saturation-temperature relationship decreased and changed direction from 1.10 %/°C ($p<0.001$) to -0.12 %/°C ($p=0.03$).

Power Diversion

Overall, water quality-discharge relationships at the power diversion did not change downstream, but discharge- temperature and discharge-DO relationships varied seasonally (Figure 43). In the summer, temperature had a positive relationship with discharge while DO had a negative relationship with discharge (slopes=0.012 °C/cfs, 0.016°C/cfs, and -0.003 mg/L/cfs, -0.003 mg/L/cfs, $p=0.005$, $p<0.001$, $p=0.002$, $p<0.001$). In the fall, these relationships changed direction (slopes=-0.018°C/cfs, 0.019°C/cfs and 0.003 mg/L/cfs, 0.0065 mg/L/cfs, $p<0.001$ for all). The SpC-discharge relationship was insignificant in the summer, but became significant in the fall and increased from -0.15 µS/cm/cfs ($p<0.001$) to -0.29 µS/cm/cfs ($p<0.001$), a 93% increase. The DO saturation-discharge relationship was insignificant both upstream and downstream. The DO saturation-temperature relationship decreased downstream from -1.19 %/°C ($p<0.001$) to -0.18 %/°C ($p=0.007$), an 85% decrease.

Irrigation Diversion

The irrigation diversion had few significant relationships with discharge (Figure 43). It only showed significant relationships with flow and saturation, downstream in the summer (slope=0.004, $p=0.008$) and upstream in the fall (slope=-0.006, $p=0.003$), and SpC downstream in the fall (slope=1.13, $p<0.001$). The DO saturation-temperature relationship changed from being insignificant at both sites in the summer to significant for both sites in the fall (slopes=0.64 %/°C, 1.12 %/°C, $p<0.001$, 0.042, respectively), a 75% increase in slope downstream (Figure 44).

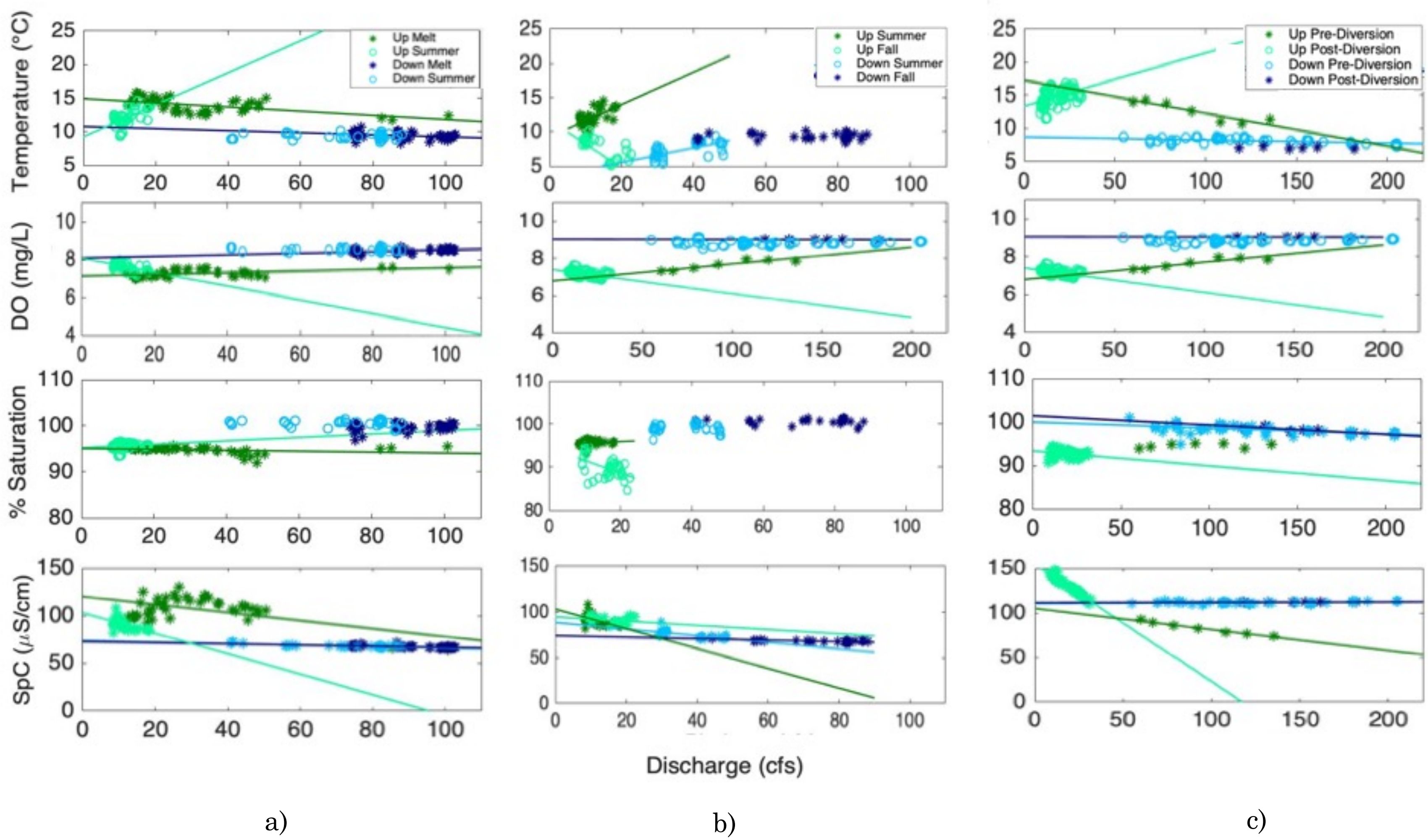


Figure 42: Water quality-discharge relationships for Granby (a and b) and Williams Fork (c).

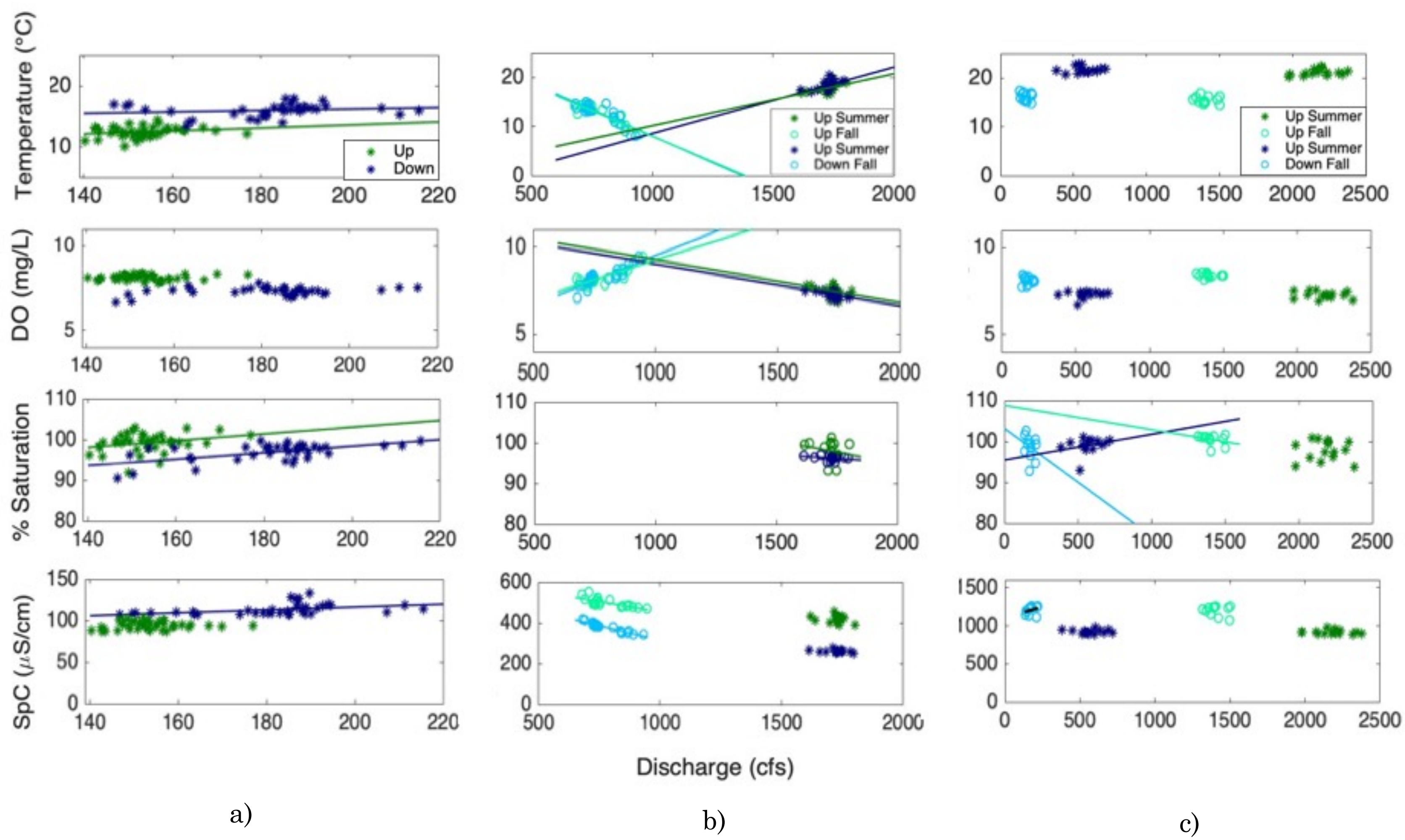


Figure 43: Water-quality discharge relationships for Windy Gap (a), the power diversion (b), and the irrigation diversion (c).

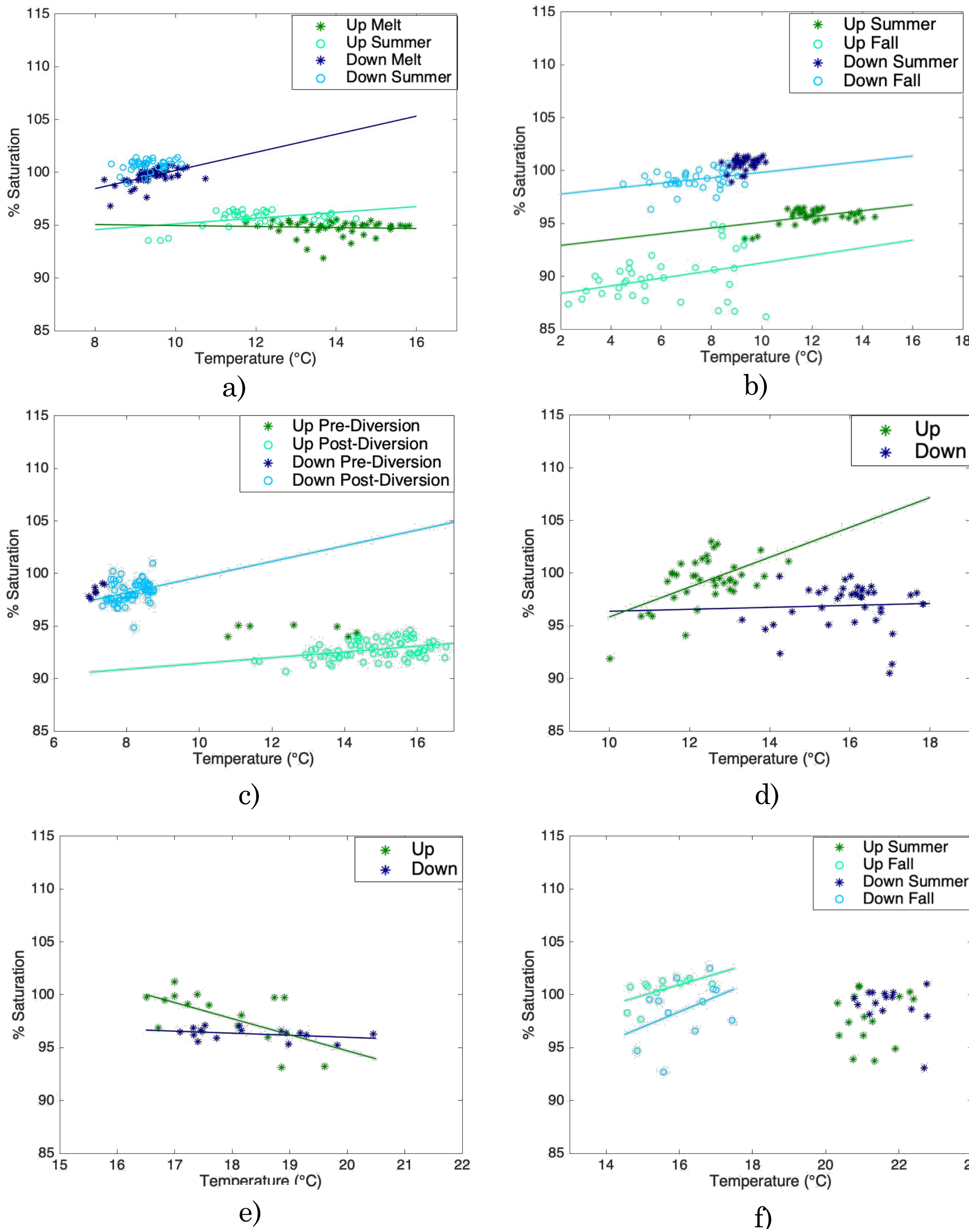


Figure 44: Water quality-temperature relationships for Granby in the melt season and summer (a), Granby in the summer and fall (b), Williams Fork (c), Windy Gap (d), the power diversion (e), and the irrigation diversion (f).

Lagrangian Temperature Profiles

Temperature profiles for both reaches resembled each other in their average slopes (Figure 45). The After Gunnison profile (higher flow) was longer than the 15 Mile Reach profile due to higher velocity in the cataraft than the duckies. The average slope of the 15 Mile Reach (low flow) profile was 58.8 °C km^{-1} and the average slope in the Colorado River, below inflow of the Gunnison River (high flow), was 51.5 °C km^{-1}. Strong temperature drops are evident in both profiles, but there are more in the 15 Mile Reach than the reach below the Gunnison confluence.

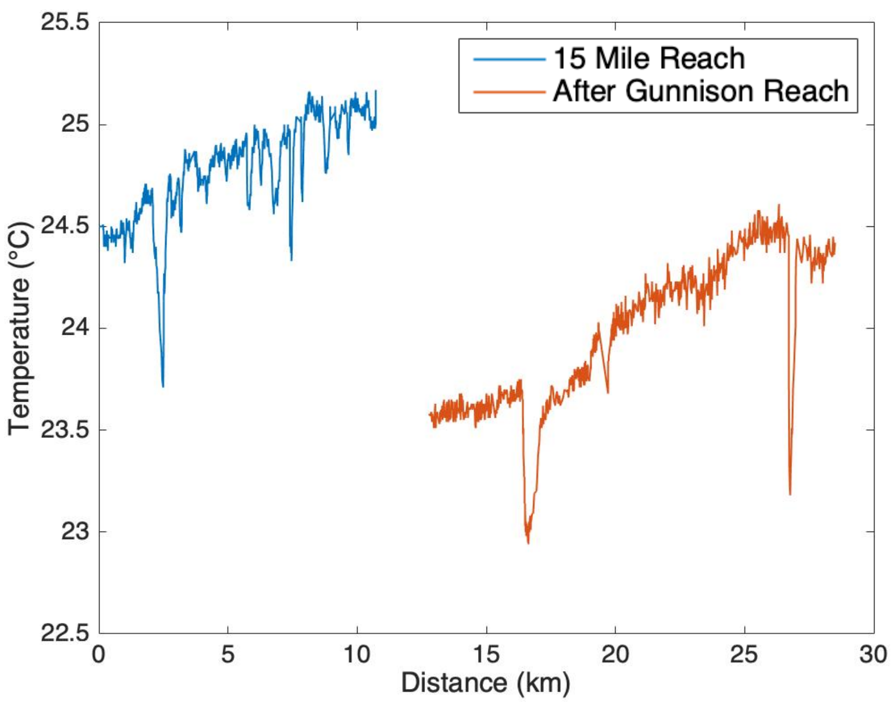


Figure 45: Longitudinal Profiles taken on the same day and time for the 15 Mile Reach and the After Gunnison Reach

Irrigation Diversion Temperature Analysis

The modelled temperature at point BG increased from the downstream site's temperature during both seasons (Figure 46). Median temperature at the downstream site was 21.69°C and significantly increased ($p<0.001$) to 22.01°C at modelled point BG. In the fall, median temperature at the downstream site was 15.60°C and 16.21°C, which was also a significant increase ($p<0.001$). The amplitude of the modelled temperature was not as large as the downstream site. The percent of heat flux contributing of the diffuse inflow contributing to the heat flux at point BG increased in the fall when discharge at the downstream site was lower (Figure 47

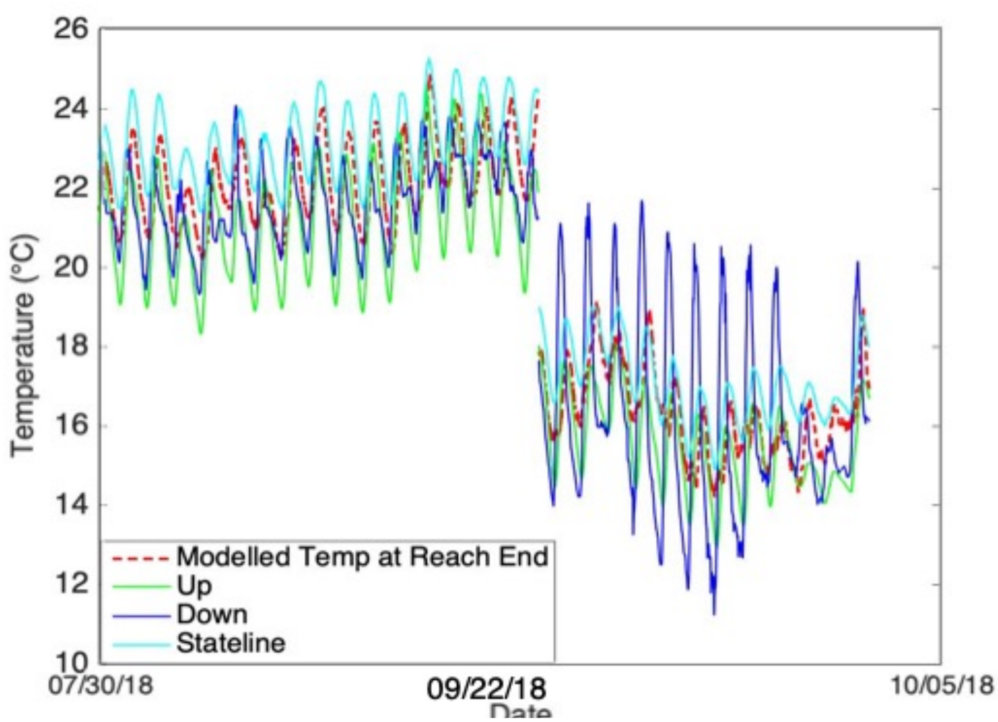


Figure 46: Modelled and observed temperatures for the irrigation diversion

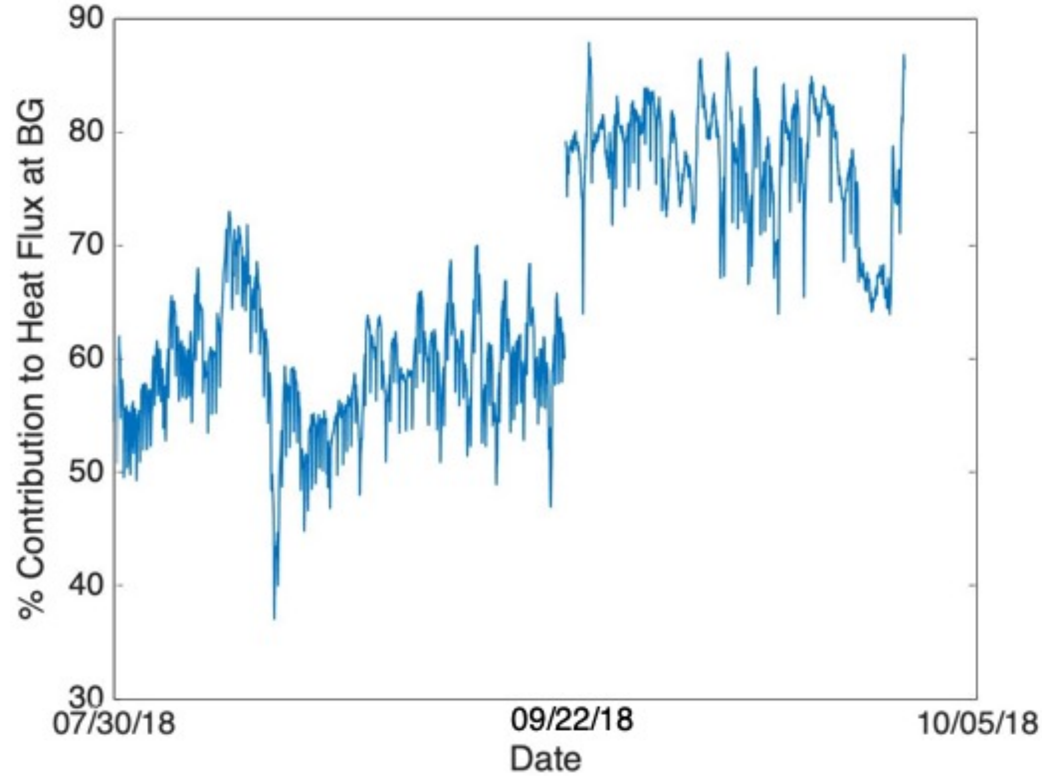


Figure 47: Heat flux of diffuse inflow relative to the heat flux at point BG through the deployment

DISCUSSION

Metabolism Modelling Limitations

The diel open channel method assumes that discharge remains constant every day. This assumption was violated at some point at most sites and may interfere with the accuracy of the models and may explain why the Bayesian Fraction of Missing Information was low. The inability of some models to converge on the observation error metric indicate that mylight model may not be vary accurate for those sites. Constraining reaeration as well as using a different light measurement and modelling method may reduce the observed and log posterior density $\hat{R}$ for the sites that did not converge. Although myconfidence is low in the WG Up, WF Up, and WF Down models, I can still glean some insight about ecosystem processes by comparing them with DO saturation and DO values.

Temperature Modelling Limitations

Because rivers generally increase in temperature downstream, I did not expect modelled downstream temperatures to decrease. I believe this is due to uncertainty in variables that I chose that the model is sensitive to, causing a larger heat flux out of the system than into it. These include air temperature for Windy Gap and the power diversion, and solar radiation for all discontinuities. Additionally, I did not include vegetation or topographic shading, but this would only further decrease the downstream temperature. I believe the model still shows changes that are on a similar order of magnitude to changes that would be present in an un-impacted reach and gives us useful comparisons for how the river temperature would change if the discontuities were not present.

Impacts of Deep-Release Impoundments on Downstream River Ecosystem

While the impacts of Williams Fork and Granby varied among the parameters evaluated, the direction and relative magnitude of the effects of both deep-release impoundments were similar. For the summer, the deep-release downstream reaches experience decreased water temperatures, increased discharge, increased DO concentrations and DO saturation, and decreased SpC. Additionally, Granby experienced turnover in the fall resulting in a downstream temperature increase. I expected to see these temperature and DO changes due to the release from the bottom of the reservoir and the predictions from Ward and Stanford (1983).

The magnitude in these changes was larger at Williams Fork than Granby, and the discharge difference was also greater. The water quality-discharge relationships, however, responded similarly downstream, becoming weaker. This weakening relationship with discharge suggests that, at discharges similar to discharges observed in this study, the water traveling downstream of the reservoirs will have similar water quality regardless of discharge. Since the water quality-discharge relationships became weaker downstream of the impoundments, this magnitude in difference between water-quality discharge relationships is likely due to within-reservoir morphology and processes affecting the water being released

(Wetzel, 2001) rather than the difference in discharge experienced by both deep-release reservoirs. The effects on downstream variability differ which may have to do with the different watersheds, stream order, and upstream reach environments.

The models did not converge for both WF sites, and downstream boundaries encased the upstream boundaries, making inference about downstream trends difficult. Mean NEP shifted towards autotrophy, and the downstream increase in DO Saturation support this result.

Impacts of Surface-Release Impoundment on Downstream River Ecosystem

The surface-release impoundment impacted the downstream reach similar to a deep-release impoundment in some ways and in other ways similar to diversions. The magnitudes of the changes were similar to the deep-release impoundments, but the directions resembled those of the diversion. Windy Gap caused a DO decrease, a small decrease in DO Saturation, and an increase in SpC from upstream to downstream. The DO decrease was due to the large temperature increase and resulting DO solubility decrease. The decrease in DO Saturation may be due to the change in DO Saturation-temperature relationship from positive upstream to negative downstream. I expected Windy Gap to act as a sedimentation basin and therefore did not expect the SpC increase. There are a few tributaries that enter between the sites which may increase SpC. I also observed iron seeps near the upstream site which may have contributed to the increase in SpC. The relationship between SpC and discharge becomes significant downstream which I believe is due to increased mixing time afforded to the river in the reservoir after the confluence with the Fraser River. Since the upstream site was only 0.15 km from the confluence with the Fraser River, myupstream site may have been in the mixing length, resulting in measurements that did not always represent the Colorado River just before it entered the reservoir.

Although the direction of the DO and SpC changes are unique to Windy Gap, their magnitudes were similar to those of the deep-release impoundments. The magnitude of the temperature change is also similar to that of the impoundments. Windy Gap's residence time was much closer to the power diversion's residence time, but it had a much higher effect on the downstream temperature. However, the direction of the temperature change resembles the diversions, and the water quality-discharge relationships do not change downstream. These results suggest that only a small residence time is needed to heat the water, but that Windy Gap's residence time is not large enough to reset the downstream reach. The result is that the surface-release impoundment affects water quality on a spectrum between impoundment and diversion. Since climatic variations and regional water decisions will impact the amount of water flowing through Windy Gap and ultimately its residence time, Windy Gap may impact the downstream reach more like a reservoir or more like a diversion depending on incoming discharge.

When comparing the impacted reach to the modelled un-impacted reach, the smaller modelled downstream temperature change combined with the larger modelled daily temperature range demonstrates the large effect of the surface-release reservoir on the downstream reach. The increased volume of water combined with the slow water velocity in the reservoir receiving sunlight acts as a buffer on the river by decreasing the temperature range and increasing downstream temperature.

Although the upstream model did not converge, the surface-release reservoir shifts towards heterotrophy when compared with both the upstream site and the Granby downstream site. However, the uncertainty intervals for all three sites overlap, suggesting that this change is not large and may not be a constant effect of the surface-release reservoir. This can be seen in the daily time series where some days the downstream site is more autrophic than the upstream site. This overall shift towards heterotrophy is confirmed by the slight decrease in DO Saturation medians.

Impacts of the Power Diversion on Downstream River Ecosystem

The power diversion's effects on water quality were due partially to the physical impacts on the river channel and partially to the specific water rights and decisions being made by the plant operators. The physical effects on water quality can be broken into three parts: the decreased velocity in the channel above the weir, the tunnels, and turbulent reach where the diverted water returns to the main channel.

I expected to see the channel upstream of the weir act as a sedimentation basin, but I did not expect to observe the large decrease in SpC that I measured. I assume that the residence time of the upstream channel is less than the 16 hour lag time. Although the Granby residence time is likely overestimated due to the exclusion of the Alva B. Adams tunnel, it is still much higher than the 16 hour lag time of the power diversion. The Williams Fork lag time (231 days) is also much higher. Since the residence time was not greater than that of the deep-release impoundments, I expected that the SpC decrease would be minimal. However, it was the largest change I observed. This may be due to the low concentrations entering both deep-release reservoirs and the fact that there may not have been much to settle out whereas SpC concentrations were higher entering the power diversion and therefore had more potential to settle out.

I did expect to see temperatures increase below the power diversion, but did not expect that the increase would be so minimal or to see the patterns that I did. I thought that the water would heat up in the reach above the weir and that these effects would remain relatively stable in the tunnels before rejoining the channel. However, the peak temperatures did not increase on average below the diversion. I believe that there are two explanations for this. The first is that the river, flowing east to west, enters a canyon below the upstream site, causing reduced solar influx to the river and dampening the increased temperature effects of the low velocity reach. I expect this is the reason for the consistent daily lag in peak temperature between upstream and downstream temperatures, particularly in the summer. The second is that the time spent in the tunnels (<16 hours) away from solar influx is significant enough to reduce the temperature effects. The result of both of these explanations is that part of the temperature effect I see is due to the diversion type and part is due to location downstream as well as within the canyon.

MySSTEMP model results also suggest that location downstream is contributes significantly to the observed temperature increase . The magnitude and amplitude of changes are very close for the model and observed temperature changes and daily ranges. Thus, the temperature change observed at the power diversion is most likely due to location downstream rather than an overall effect that the diversion has on the river. The observed temperature change is only larger than the model temperature change for the maximum temperature change case, indicating that the diversion may have some effect on river temperature in some instances, but that the location downstream plays a significant part in this temperature increase. A power diversion in another location may impart greater temperature effects due to higher incoming solar radiation.

During the summer, the downstream temperature resembles the upstream temperature, but with higher minimums. In contrast, at low flow in the fall (the first part of the fall deployment), the entire river is likely being diverted into the tunnels due to discharge being less than the plant's water rights (<1,250 cfs). The increase in temperature and decrease in the temperature variability suggests that the tunnels dampen the temperature variability. This also suggests that, although the majority of the river is diverted into the tunnels at higher flow, the small river in the main channel is still important for the thermal regime of the downstream reach. During the second part of the fall deployment when discharge begins to decrease, the downstream temperature measurements lose the daily shape and both sites begin to decrease. I observed a storm across the watershed during this increase as well as a release from the upstream Green Mountain Reservoir, suggesting that the change in relationship between flow and temperature from summer to fall may have occurred concurrently with a decrease in air temperature as well as a change in upstream source water.

Although I could not model metabolism for the power diversion, I can speculate about its effect on ecological activity. The morphological change from the confluence of the tunnels and resulting turbulence causes a change in downstream DO saturation trends. I see this in the downstream decrease in variability of DO saturation as well as in the decrease in relationship between DO saturation and temperature. The daily variation from 100% DO saturation at the upstream site contrasts with the stable DO saturation values near 100% at the downstream site. This is due to the high reaeration caused by turbulence downstream of the diversion and the fact that the DO concentrations in the water can equilibrate quickly with the atmosphere (Churchill et al., 1964). Since metabolic activity increases with temperature (Demars et al., 2011), and metabolic activity causes saturation to vary from 100% at the upstream site, this explains why DO saturation has a stronger relationship with temperature at the upstream site. In contrast, the build-up of DO caused by metabolic activity cannot be measured because the oxygen concentrations equilibrate quickly with the atmosphere. I cannot say that the power diversion affects ecological activity due to a change in DO saturation because its effects are eclipsed by high reaeration, but the deep, scouring flows of the downstream reach (with no sediment transfer from upstream) create a harsher environment for many primary producers (Biggs & Close, 1989) than the low-velocity upstream reach, likely resulting in a shift in metabolic activity.

Impacts of the Irrigation Diversion on the Downstream Ecosystem

The irrigation diversion's effects on water quality varied seasonally, largely due to lower discharge in the fall. I expected to see temperatures increase, but to a greater degree than I measured. I believe that this occurred for two reasons. First, the downstream flow was only low enough in the fall for changes in the thermal buffering capacity of the reach to be observed, and second, mysensor location was not far enough from the diversion to capture the maximum effect of the diversion on water quality.

The first explanation is supported by the trend changes between summer and fall. Not only are there seasonal and associated atmospheric temperature changes, but the downstream discharge also decreased from 562 to 165 cfs. During this shift, the downstream temperature amplitude (3.00 °C) doubled (6.46 °C), increasing from an amplitude similar to the upstream summer amplitude (3.64 °C) while the upstream fall amplitude (2.56 °C) decreased. This suggests that the river passes through a threshold discharge where it becomes more sensitive to the surface energy balance and ecological activity becomes more active. DO saturation changes from having a significant linear relationship with discharge in the summer to having a significant linear relationship with temperature in the fall. The resulting relationship between temperature and DO saturation under a threshold discharge shows that temperature is a significant control on metabolic activity at low flows in this system, but not at higher flows.

Since the summer downstream amplitudes resembled the upstream amplitudes, I do not believe that this threshold discharge was reached in the summer, though it may have been close, due to the higher temperatures in the summer as opposed to the fall. If the 15 Mile Reach did experience a transition across this threshold discharge in the summer, there is a potential for metabolic activity to increase substantially due to the combined effects of agricultural runoff, high temperatures, and this relationship between temperature and DO saturation.

It takes time for the effects of the reduced thermal capacity of the river to have their maximum effect. When the water is first diverted from the main channel, the remaining water will not immediately experience a substantial change. Over time, as it travels through the channel with reduced thermal capacity and begins to warm from external sources, it will not be able to buffer the effects as it did before and will warm more quickly. This can also be seen in the increase of relative contribution of heat flux of the diffuse inflow to the downstream heatflux during the fall (low flow) as opposed to the summer (high flow).

I observe this in the model of the 15 Mile Reach temperature increase in which the channel experiences a larger increase in temperature from the downstream of the diversion site to the modelled BG point in the fall at low discharge despite lower average temperatures. The Lagrangian profiles did not show a substantial difference in their temperature slopes. This may be due to the fact that the summer flows were not below the threshold discharge that saw a substantial change in temperature amplitudes. This may also be due to time of sampling. These profiles were taken between 9:00 AM and 12:00 PM, a time in which stream temperature is increasing due to incoming solar radiation. The 15 Mile Reach inflows cause temperature decreases when they enter the Colorado River. Since inflow and river temperatures are largely dependent on solar radiation, they are dynamic and the inflows may increase river temperature at other times of the day. Since there are more inflows in the 15 Mile Reach than the reach after the Gunnison, this may have dampened the temperature increase in the 15 Mile Reach.

It is important to consider where the maximum impact of the diversion is likely to be at these low discharges so that monitoring can capture the whole impact on the reach. I do not believe that this study captured the entire impact of the diversion dam due to the placement of the sensors being only 1.89 km from the downstream end of the diversion.

Similar to the power diversion, the irrigation diversion observed temperature change for the minimum case is smaller than the modelled change, but the observed downstream changes for the mean and maximum cases are twice as high as the modelled changes, suggesting that location downstream contributes some to the temperature change, but not as much as the power diversion. The modelled temperature decrease with the downstream discharge decrease was the opposite of what I expected because I thought that less water would mean more solar warming. This result can be explained by the fact that SSTEMP is a steady state model that does not take into account the transient warming occurring after the water is diverted, only the steady state heat flux balance. Taking out water results in a removal of heat flux and if there is not a substantial heat source to make up for that, then the temperature will go down. The modelled increasing amplitude with decreasing discharge matches myexpectations that less water in the channel leaves less buffer from the surface energy balance. This supports myconclusion that the increasing temperature amplitude in the fall is due to the decreased flow.

I am the most confident in the metabolism models at this site since both sites have lower reaeration and they correlate well with the metabolism metrics I used for comparison. The diversion caused a shift towards autotrophy in the summer and heterotrophy in fall. This shift mirrors the seasonal downstream changes in DO and DO saturation in which DO and DO saturation increase in the summer and decrease in the fall. This could be due to 1) changes in the physical environment due to low flow in the fall, and 2) this allows for the diversion to have more of an effect on the river before the sensor measures it due to lower discharge and lower velocity. I believe it is a mix of these two things, but that the combination of metabolism results and DO and DO saturation trends provide evidence for strong ecological impact on DO concentrations in the 15 Mile Reach. Future studies with more strategic placement of the downstream sensors could provide better understand of DO dynamics in the 15 Mile Reach.

Diversions Show Opposite Temperature Response to Low Flow

The diversions are used for different purposes and have two hydrological impacts on the river. The power diversion restores flow at one point, and the irrigation diversion returns some of the flow diffusely to the channel from non-point sources. Both diversions increased the temperature of the channel slightly, but their IQR's responded in opposite directions during the low flow deployment. The power diversion temperature amplitude decreased while the irrigation temperature amplitude increased. Although I can not confirm the powerplant operator's decisions during this time, the fact that the discharge was less than 1,250 cfs means that the power diversion was able to divert the entire river into its tunnels, eliminating solar input and dampening temperature variability. The irrigation diversion change in amplitude was due to a change in thermal sensitivity, a physical change. This means that these two diversions not only differ in how they affect temperature, but also in that the power diversion's temperature effects are dependent on water rights and plant operator decisions whereas the irrigation diversion experiences its temperature change due to environmental effects.

Diversions and Impoundments and their Effects on the River Network

In mystudy, impoundments had an overall larger effect on water quality and flow relationships than diversions. However, the maximum effect of impoundments occurs directly below impoundments, and measuring their impact on the downstream reach is more straightforward due to linear recovery of the parameter.

In contrast, for a diversion where a large amount of flow is removed, the maximum effect of the diversion is seen at some location downstream of the diversion and then recovery proceeds as flow is restored by lateral inflows or tributaries.

The effects of both diversions and impoundments have the potential to compound within river networks, especially with water quality parameters that have long recovery gradients on the order of hundreds of kilometers such as temperature (Ellis and Jones, 2013). Since impoundments have such a large effect on water quality, their effects are likely to propogate further down the river network than diversions. However, since residence time is so important, a diversion that travels a longer distance before its flow is restored may have a larger impact on the river and have greater effects downstream.

Trends Captured Do Not Tell the Annual Story

This study shows seasonal effects of impoundments and diversions on the Colorado River specific to the summer and fall. The combination of the snowmelt-dominated hydrograph and the fact that data collection occurred during the summer and fall of a low flow year result in highly seasonal results that would likely change at different times of the year. The collected data captured the tail end of the snowmelt season at the upstream William's Fork site in which SpC was lower than the downstream SpC. If the study had been done in the spring, I would've likely observed different effects on SpC by Williams Fork (deep-release). Seasonal effects are also apparent at Granby (deep-release). During the first few days of deployment, upstream discharge is higher than downstream discharge due to being at the end of the snowmelt hydrograph and then it is lower for the remainder of the deployment. Additionally, the downstream decrease in temperature during the melt season and summer to an switched direction to an increase in temperature during the fall. This would likely have been seen in Williams Fork if the sensors had been deployed there longer (Ward and Stanford, 1983).

Mitigating Future Discontinuities

Mystudy demonstrates the the impacts of discontinuities on river water quality and metabolism, particularly for the surface-release impoundment and irrigation diversion. It also demonstrates the importance of strategic data collection location and the utility of continuous measurements for capturing not only the full impact of a discontinuity but also its seasonal trends and relationships with discharge. These trends provide insight into how discontinuities impact the river and how humans can mitigate them. With the predicted increase in water resources infrastructure, strategic monitoring programs that establish monitoring locations beyond the maximum impact of the discontinuity and use continuous measurements as guidance can help sustain the balance between water resources sustainability and ecosystem services provided by rivers.

CONCLUSIONS

Impoundments and diversions impacted the Colorado River water quality during the 2018 summer. Impoundments had the greatest impact on the magnitude of temperature and dissolved oxygen. Deep-release impoundments also acted as a buffer, weakening water-quality-discharge relationships. The surface-release impoundment increased temperature and decreased dissolved oxygen to the same degree as deep-release impoundments, but did not substantially change water-quality discharge relationships.

Diversions impacted downstream reaches less in magnitude at the sensor site, highlighting the importance of discontinuity residence time for affecting downstream concentrations. The irrigation diversion showed an increase in thermal sensitivity past a threshold flow and the heat flux of the incoming runoff increased in importance relative to downstream temperatures once it was past this threshold. Diversions, while acting as point discontinuities in discharge and sediment dynamics, impact water quality over a longitudinal gradient and their more severe effects, particularly on metabolism and temperature, take more time and space to develop.

This study highlights the individual effects of impoundments and diversions on downstream water quality and ecosystem processes and their potential for creating compounding effects down the river network. Balancing the water demand of an increasing population with the ecosystem health and services of rivers will require strategic monitoring to understand and mitigate these impacts.

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